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The role of valuation and bargaining in optimising transboundary watercourse treaty regimes

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Abstract In the face of water scarcity, growing water demands, population increase, ecosystem degradation, or climate change, transboundary watercourse states inevitably have to make difficult decisions on how finite quantities of water are distributed. Such waters, and their associated ecosystem services, offer multiple benefits. Valuation and bargaining can play a key role in the sharing of these ecosystems services and their associated benefits across sovereign borders. Ecosystem services in transboundary watercourses essentially constitute a portfolio of assets. While challenging, their commodification, which creates property rights, supports trading. Such trading offers a means by which to resolve conflicts over competing uses and allows states to optimise their ‘portfolios’. However, despite this potential, adoption of appropriate treaty frameworks that might facilitate a market-based approach to the discovery and allocation of water-related ecosystem services at the transboundary level remains both a challenge and a topic worthy of further study. Drawing upon concepts in law and economics, this paper therefore seeks to advance the study of how treaty frameworks might be developed in a way that supports such a market-based approach to ecosystem services and transboundary waters.

Keywords Transboundary watercourses · Treaty regimes · Law and economics · Portfolio optimisation · Asset management · Ecosystem services · Market-based approaches

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1 Introduction

Transboundary watercourses, and their associated ecosystem services, offer multiple benefits. The disciplines of law and economics have advanced our understanding of how such benefits might be discovered and distributed among beneficiaries, including between states. In the law literature, there has been a concerted effort to analyse the normative content of rules and principles of international law on transboundary waters, ascertain their status as customary international law obligations, and identify areas where legal norms or treaty¹ arrangements might be strengthened (McCaffrey 2007; McIntyre 2007; Wouters 2011). Much of this literature has focused on the principle of equity, and its capacity to allocate benefits between states, as well as the procedural and institutional structures needed to implement any equitable arrangement (e.g. McIntyre 2017). In the economics literature, game theory has been advanced to explain why riparians may or may not cooperate over their transboundary waters (Barrett 1994; Rogers 1997). There have also been significant efforts to estimate the costs and benefits of transboundary water cooperation (Sadoff and Grey 2002; UNECE 2015) and to explore the potential role of markets at the transboundary level (Dinar and Wolf 1994; Fisher 1995; Zeitouni et al. 1994). While the aforementioned literature demonstrates that there is much to be gained by states cooperating over transboundary waters, it should also be appreciated that such benefits are far from guaranteed. Particularly in power asymmetric settings, and where governance arrangements are weak, ensuring that benefits are equitably distributed between states, as well as their populations, remains a key challenge (Meijerink and Huitema 2014; Mirumachi 2015).

This paper aims to help address this key challenge through a combined study of law and economics. While there has been significant attention placed on transboundary water conflict or cooperation from the disciplines of law or economics, little of this work seems to rely on first principles of both law and economics. There is, however, a great potential to draw from the economic analysis of law, which entails the deployment of the tools of microeconomics to determine the effects of legal rules and their social desirability. The parties involved, such as consumers, producers, or arbitrageurs, optimising their respective objective functions, are assumed to behave rationally, and the social desirability of equilibrium outcomes is examined under the framework of welfare economics. In other words, the study of law and economics seeks to establish not only the effects of legal rules on the optimising behaviour of parties, but also the implications for societal well-being.

Not much research in law has drawn on economics to explain the correlation between treaty design, normativity, and compliance. From an economics perspective, Dombrosky (2007) applies different economic methods to analyse transboundary water ‘institutions’, which are broadly defined, and treaty design is only covered partially (see also Dinar 2008; Schmeier 2013). Makridis (2013) attempts to explore the role of markets within a transboundary water context, but recognises that, ‘... many of the advances in industrial organisation and applied microeconomics have not yet been integrated into the literature on transboundary governance and sustainability’. Makridis (2013) goes on to suggest that, ‘there

¹ For the purposes of this paper, the term ‘treaty’ will be defined as, ‘an international agreement concluded between States in written form and governed by international law, whether embodied in a single instrument or in two or more related instruments and whatever its particular designation’ (1969 Vienna Convention on the Law of Treaties). Conventions, agreements, and other legal instruments may therefore fall under the definition of a treaty if it can be demonstrated that those adopting them intended them to be governed by international law.

is sparse literature that applies market design elements of incentive compatibility among heterogeneous actors in the context of transboundary water governance’.

Through its endeavour to combine the study of law and economics, and to build upon the existing literature, this paper seeks to make the following contributions. Firstly, the paper seeks to revisit the study of market-based approaches within the context of transboundary cooperation. While such a study is not novel in itself, its contribution is to draw together a series of economic approaches and developments in a novel way, namely the notion of property rights and contracts, ecosystem services, valuation and natural capital, commodity trading, and portfolio optimisation. Secondly, the paper seeks to characterise, in the context of a transboundary watercourse, key considerations within treaty regime design that might help foster a market-based approach to transboundary cooperation. Additionally, the paper will consider both the constraints precluding states from securing those preferable treaty arrangements, and incentives that might be put in place to tackle them. Thirdly, the paper explores the advances in knowledge and understanding of the concept of ecosystem services and considers that knowledge and understanding within a transboundary water context. While there has been a significant amount of work related to ecosystem services and water, little of that work has considered the transboundary nature of rivers, lakes, and aquifers (Rieu-Clarke and Spray 2013).

Ultimately, the paper advances the notion that ecosystem services in transboundary watercourses essentially constitute a portfolio of assets and that their commodification, creating property rights, supports market exchange or trading. In so doing, the paper explores the risks and opportunities associated with incorporating ecosystem services into a system of portfolio optimisation and commodity trading at the transboundary level.

2 Property rights and contracts as fundamental building blocks

One of the building blocks of a market approach to the allocation of ecosystem services among riparian states is the concept of property rights. Property law creates a bundle of rights which property owners are free to exercise and which protect them from interference, and, while subject to transaction costs, voluntary exchange allocates property to those who value it most (Cooter and Ulen 2014). Elaborating on the canonical 1960 article ‘The Problem of Social Cost’ by Ronald Coase, Libecap (2016) demonstrates in detail that, if property rights are not defined, the number of claimants could be unlimited, rents dissipated, and trade becomes impossible. The greater the benefits from establishing property rights or the greater the losses from their absence, the higher the transaction costs. Moreover, the more stationary, observable, or smaller the resource, or the more stable its quality distributions, the lower the costs of measurement, enforcement, or exchange. This is why land or its associated environmental resources have tremendous potential for Coasean trade. And the more homogeneous the parties, the more frequently they communicate, or the more similar their objectives for conservation or protection, the lower the costs of assigning property rights.

In the context of natural capital, however, the assignment of property rights may not always be feasible, or their exercise may not always be beneficial. Libecap (2016) asserts that the transaction costs of Coasean exchange tend to be high for ‘very broad-scale environmental externalities’. Given that mitigation values or costs vary within or across countries, long-term benefits or costs are different or uncertain under international environmental agreements. As a result, international environmental agreements protecting

highly migratory ocean fish stocks or reducing greenhouse gas emissions, for example, are extremely elusive. Turner (2015) even maintains that the exercise of property rights could damage ecosystem services. For example, the laws of many countries provide landowners with a high degree of autonomy over their land-use practices, which can lead to unsustainable practices (Turner 2015). In the context of transboundary waters, whether or not the assignment or exercise of property rights is feasible or beneficial is likely in part to be a function of the cooperative arrangements that are, or are not, in place (Dombrowsky 2007). Such arrangements, as discussed below, have the potential to limit transactions costs.

Closely associated with property rights is the role of contracts. The economic analysis of contracts has a long tradition. Cooter and Ulen (2014) explain that a perfect contract is efficient and complete. If it is impossible to revise a contract in order to make at least one party better off and other parties not worse off, then the contract is Pareto efficient. Under a perfect contract, the terms have exhausted the possibilities for cooperation, the obligations of parties under every possible future state of the world are specified, and no potential gain for trade is left unrealised. Given that all contingencies have been anticipated under a complete contract, re-negotiation is inefficient (Salanié 1999).

However, it is obvious that few markets achieve the ideal of perfect competition. Various market failures, such as irrationality, coercion, externalities, lack of information, high transaction costs, or maker power, may lead to an imperfect contract (Cooter and Ulen 2014). In the likely event of contract imperfection, institutions can play a role in making the contract as perfect or complete as possible (Cooter and Ulen 2014). Almost every contractual dispute before the courts concerns the incompleteness of, or imperfections in, a contractual arrangement (Hart 1989).

A treaty regime for transboundary waters, which could be seen as a contractual arrangement between states, is likewise unlikely to be perfect or complete. The aforementioned market failures that lead to an imperfect contract are also likely to be evident in a transboundary water treaty setting. For example, there is often a lack of information on the dynamic uses of the waters of a transboundary rivers, lakes, or aquifers, especially in the light of climate variability; power asymmetries between upstream and downstream states may lead to coercion; externalities, such as transboundary pollution, may go unchecked; or the transaction costs of agreeing a transboundary treaty arrangement, and then seeking to 'perfect' it, may be extremely high. Examples of 'imperfect' treaty arrangements abound, especially where power asymmetries are evident (see generally Zeitoun and Mirumachi 2008). For instance, the high transaction costs in reaching a basin-wide agreement on the Nile have been well documented (Brunnée and Toope 2002; Salman 2012). Another example is how lack of information poses a key challenge in the debates over the development of hydropower projects on the mainstream of the Mekong and the application of the 1995 Mekong Agreement (Rieu-Clarke 2015). This has resulted in Cambodia, Vietnam, and Thailand all calling for a 10-year moratorium on hydropower development in the lower Mekong in order to establish baseline assessments of the River's ecosystem and to better understand the impacts of those potential developments (Orr et al. 2012).

The adoption of an 'imperfect treaty' may, however, be justified if, due to bounded rationality, the cost of accounting for improbable contingencies outweighs the benefits of writing specific provisions (Salanié 1999). In such a scenario, certain strategies can be adopted in order to 'perfect' the treaty. For instance, increasing precision or reducing ambiguity in treaty terms supports the effort of perfecting or completing a treaty. Precision, measuring how the agreement clearly and unambiguously defines what is required for compliance, not only increases the probability of detection, but also facilitates the resolution of conflicts of interpretation and the application of sanctions (Hafner-Burton et al. 2012).

Ambiguity or a lack of ‘textual determinacy’ is one of the main causes of poor treaty compliance (Fuller 1964; Franck 1998). If the stakes are high or the parties are extremely risk averse, concerns about mistaken signals or imperfect enforcement lead to the avoidance of vague agreements (Sand 1992). However, defining precise rights and obligations for complex problems at the outset may prove elusive (Franck 1998). Incomplete contracts may therefore be unavoidable when interests diverge or uncertainty is high (Hafner-Burton et al. 2012). Countries may put different weights on economic or environmental factors. A unified weight satisfying all parties may be difficult to achieve (Zhua and Li 2014). In such cases, institutions designed to address ambiguities, and methodologies by which to ascertain and harmonise values, become critical.

The use of money damages, as in the case of private contracts, may offer an additional means by which to enhance the prospects of treaty compliance (Guzman 2005). Cooter and Ulen (2014) explain that there are situations in which breach is more efficient than performance and that expectations damages can restore the position of a breach victim as if the other party had performed. It is possible, but rare, to have provisions for money damages in international agreements. A state violating an international commitment suffers a loss of reputation and perhaps a direct sanction, but sanctions, which bring a loss to the violator without an offsetting gain to the victim, represent a net loss to all parties. If it were possible to establish damages in the form of a zero-sum transfer from one party to another, more efficient forms of cooperation could be encouraged. In other words, while maintaining the reputational or direct sanctions, transfers enhance compliance with international agreements without adding to the disincentives arising from existing sanctions, especially if the damage is largely economic. Indeed, Cooter and Ulen (2014) confirm that, under contract law, if payoffs could be changed, games with inefficient solutions can indeed be converted into games with efficient solutions and that changing payoffs could occur through side payments. As noted above, an effective system of valuation becomes critical to the effectiveness of any compliance method based on monetary damages. Valuation will therefore be considered in the following section, followed by an analysis of legal and institutional arrangements.

3 Can ecosystem services, portfolio optimisation, and commodity trading foster transboundary water cooperation?

Commodity trading is the business of transforming commodities in time (storage), space (logistics), or form (processing) (Pirrong 2014). Value is created through the optimisation of commodity transformations along a chain of commercial activities (Pirrong (2014). It is typical for an investor to select asset classes, prepare estimates of returns and their variability, and identify alternative allocations corresponding to different levels of risk tolerance (Sharpe 2007). It is also typical for a commodity trader to buy or sell across different grades, volumes, pricing structures, transaction or delivery dates, or transaction or delivery locations (Schofield 2007).

The application of portfolio management principles beyond the commercial sector has proven to be a useful tool by which to analyse the behaviour of different entities. For example, Bertelli and John (2013) study the behaviour of democratic governments managing a portfolio of policy priorities for the public. Riparian states can be viewed as seeking to optimise the value of their portfolio of ‘transboundary watercourse assets’. A riparian, pondering how best to benefit from a transboundary watercourse, works with an initial

portfolio of watercourse assets, determines their valuations, considers candidate allocations consistent with its risk profile, and adjusts, through trading, the components of its portfolio. Subject to a riparian's risk appetite, watercourse assets can be commoditised and traded in order to capture their value. Designed and implemented properly, this approach reduces the imprecision or ambiguity of treaty regimes. However, adopting an asset management approach to transboundary waters is not without its challenges, which relate mainly to the nature of watercourse assets, difficulties in their valuation, and issues of risk mitigation and perception.

3.1 Nature of transboundary watercourse assets

A commodity is itself consumed, as opposed to, for example, a security in stock markets (Pirrong 2014). Just as energy commodities, such as gasoline, natural gas, or electricity, are consumed for the services they provide, including mobility, space or water heating, or lighting (Sweeney 2001), the services from (transboundary) watercourse assets can be consumed. Significant advances in environmental economics over recent decades can assist in helping to view transboundary watercourses as assets.

An ecosystem, such as a rainforest, desert, or coral reef, is a collection of plants, animals, or micro-organisms interacting not only with each other but also with their non-living environment. Such ecosystems provide significant 'natural capital'—a term that has become well established in environmental economics and is gaining increasing policy relevance (EC 2013; Jansson et al. 1994). Inherent in the concept of natural capital is the notion of 'assets which yield the goods and services that humans produce and/or use' (UK National Ecosystem Assessment 2014). In other words, the services derived from those ecosystems are benefits which humans obtain from nature and can be viewed as assets (Ranganathan et al. 2008). Such assets can encompass a wider set of services offered by freshwater ecosystems (Alward et al. 2005; Grizzetti et al. 2016; see Table).

Alward et al. (2005)

Provisioning services	Regulatory services	Cultural services
Water (quantity and quality) for consumptive use (for drinking, domestic use, and agriculture and industrial use)	Maintenance of water quality (natural filtration and water treatment)	Recreation (river rafting, kayaking hiking, and fishing as a sport)
Water for non-consumptive use (for generating power and transport/navigation)	Buffering of flood flows, erosion control through water/land interactions and flood control infrastructure	Tourism (river viewing)
Aquatic organisms for food and medicines		Existence values (personal satisfaction from free-flowing rivers)
<i>Supporting services</i>		
Role in nutrient cycling (role in maintenance of floodplain fertility), primary production		
Predator/prey relationships and ecosystem resilience		

Significant progress has been made in understanding ecosystems and their services as natural capital or assets, particularly since the adoption of the Millennium Ecosystem

Assessment in 2005 (Hassan et al. 2005; TEEB 2010; Burkhard et al. 2013; Smith et al. 2017; Willemen et al. 2014). However, there has been little work that has considered the transboundary nature of many river basins (Rieu-Clarke and Spray 2013). This is a huge gap in the literature. Transboundary river basins cover nearly half of the Earth's land surface and account for 60% of global freshwater flow (UNEP-DHI and UNEP 2016).

An analysis of ecosystem services within a transboundary context can build upon scholarship that has sought to ascertain the benefits of cooperation over transboundary waters. Sadoff and Grey (2002), for instance, identify four types of benefits from cooperation: better management of ecosystems (benefits to the river), increased food or energy production (benefits from the river), reduced tensions among riparians (cost reduction because of the river), and greater cooperation between states (benefits beyond the river) (see also Hensengerth et al. 2012). Furthermore, considering these services as assets provides opportunities for trade across borders and therefore portfolio optimisation (see discussion below).

When considering the nature of transboundary watercourse as assets, the concept of storability is particularly relevant. Pirrong (2014) provides a detailed discussion of the crucial role of storability in commodity consumption or production decisions. Most commodities are storable. Some storable commodities are continuously produced and consumed, but others, although continuously produced, have seasonality in consumption. In the context of ecosystem services, natural or artificial storage is crucial to water resources management. The productivity of rain-fed agricultural systems, for example, tends to be highly variable due to fickle rainfall, and such systems are also often vulnerable to floods or droughts (Salman and Martinez 2015). Forest ecosystems, often acting as sponges, intercept rainfall and absorb water through their root systems (Turner 2015). Water is then slowly released into surface water and groundwater. Forest ecosystems therefore play a key role in recharging groundwater supplies, regulating stream flows, and lowering peak flows during times of heavy rainfall and flooding. Low water storage *per capita*, as witnessed in many parts of Africa, is seen as an obstacle to agriculture or energy generation (Salman and Martinez 2015). The concept of storability raises a number of issues when considering transboundary watercourse assets. Dams provide a means by which to artificially store water. One example where states cooperate over water storage is the arrangement between Canada and the USA under the 1964 Columbia River Treaty. In that instance, Canada committed to build three large storage dams, Duncan, Keenleyside, and Mica, not only for flood protection in the USA but also for power generation in both Canada and the USA (Hensengerth et al. 2012). Given that an upstream state's decisions on water storage or release may have positive or negative effects on a downstream state, a transboundary approach to dam site selection and regulation is critical. Cooperation between states, such as in the case of Canada and the USA, potentially increases the return on investment if storage volumes and river heads are high (Hensengerth et al. 2012). In other instances, where the irrigation potential is upstream of the hydropower potential, trade-offs between energy generation and irrigation potential may have to be made (Salman and Martinez 2015). However, transboundary basin planning requires significant levels of trust between states. Such trust may derive from a history of cooperation, such as the case of USA and Canada. In other basins, a lack of trust may make cooperation harder to achieve. Where there are low levels of trust, third parties can play an important role in fostering cooperation. This can be the case of the World Bank and the adoption of the 1960 Indus Treaty between India and Pakistan (Zawahri 2019). The Indus case also demonstrates the importance of effective legal and institutional arrangements, which if designed appropriately can offer certainty in scenarios where there are low levels of trust between states (see discussion below).

3.2 The valuation of transboundary watercourse assets

A major challenge in applying market-based approaches within a transboundary context relates to valuation. The UK National Ecosystem Assessment (2014) maintains that although nature, its biodiversity, or its ecosystems are crucial to human prosperity, they are ‘... consistently undervalued in conventional economic analyses and decision-making’. Two reasons for the tendency to undervalue ecosystems are a lack of data and information and deficiencies in existing valuation methodologies. In the UK, for example, there is no comprehensive analysis of the contribution of ecosystem services to the economy (UK National Ecosystem Assessment 2014). This is not just a challenge for the UK, but worldwide (TEEB 2010). Typically, environmental or social costs are not fully expressed in terms of market prices because their monetisation can be methodologically complex, time-consuming, or costly, and the implications for compensation much disputed (Hensengerth et al. 2012). The non-financial nature of many water-related ecosystem benefits, together with inadequate estimates of their economic value, may cause them to be ignored (Schaafsma et al. 2015; Wam et al. 2016).

A key consideration in valuation methodologies is therefore how to deal with incomplete or missing markets. Varian (2010) describes the link between missing markets and negative externalities. If a transaction imposes a cost on a third party, the failure to account for the negative externality could lead to an inefficient outcome. In the case of pollution, there is a missing market, the market for the pollutant, and the polluter, in effect, faces a zero price for it. Libecap (2016) expounds on missing environmental markets. In principle, if property rights are defined, then traders are identified and addressing externalities through market exchange, or trading, becomes feasible. Property rights assign the benefits and costs to rights holders and provide incentives for conservation or investment. The fundamental approach to environmental valuation, Perman et al. (2011) narrates, is the ‘commodification’ of ecosystem services. Ecosystem services consumed by households and firms enter as arguments in utility and production functions, respectively, and the tools of consumer and producer theory are then used to estimate their monetary values (Perman et al. 2011).

In this regard, payment for ecosystem services (‘PES’) schemes have been widely considered in both developed and developing countries as a means by which to ‘commodify’ ecosystem services (Martin-Ortega et al. 2015). These schemes constitute a means by which to reward ecosystem managers for maintaining or enhancing services which beneficiaries value. Their implementation, however, can be problematic (Reid and Nsoh 2016; Schomers and Matzdorf 2013). Leisher (2015) demonstrates why such schemes are often problematic within the context of river basin management. She suggests that when commodity prices change, there is often a temptation by PES scheme members to withdraw large areas of land or deploy land-use practices which are detrimental to clean water provisioning. For instance, payments received for livestock not grazing on certain land may be rendered insufficient given an increase in the price of milk or beef. This in turn may lead to the spoilage of ecosystem services provided by other participants.

In other words, the conceptual or practical difficulties related to payments for ecosystem services tend to arise from incomplete or missing markets or poorly defined property rights. In the example above, a change in milk or beef prices shifts the opportunity cost of agricultural land, and there is a need to reorganise farm operations, including grazing activities, substitutes for which could be found elsewhere or at another time. In the absence of property rights and the signals they provide, however, it would be difficult to curtail

harmful land-use practices. Expanding the range of ecosystem services for valuation thus requires not only the identification of ‘pain points’, signalling enormous value, such as congestion preventing the delivery of a commodity to where or when it is needed, but also the addition to the portfolio of new commodities representing them.

In the field of finance, Blanchett and Straehl (2015) study the implications of adding human capital, housing wealth, and pensions, all of which tend to be excluded in the traditional analysis, to a retirement portfolio. In estimating the value of human capital, earnings are viewed as a dividend from the worker’s total human capital, and the return on human capital is estimated as the change in the total value of human capital over time. Similarly, in the context of ecosystem services, finding the value proposition which brings ‘new’ products to the portfolio is the starting point. Houdet et al. (2015) contend that ‘[m]onetary incentives (e.g. direct payments, premiums and state subsidies) and disincentives (e.g. environmental taxes, charges and penalties) can provide tangible reasons for corporate behaviour changes towards mainstreaming water ecosystem services stewardship in the private sector ... provided they are significant enough when compared to alternative undesirable behaviours and the associated costs and income streams’. In short, it is value, either received or foregone, which propels a product to be incorporated into mainstream decisions on consumption or production across locations, over time periods, or in form.

There are, of course, other types of valuations. The UK National Ecosystem Assessment (2014) shows that there are monetary and non-monetary values of cultural ecosystem services. In principle, if cost or benefit streams cannot easily be quantified, one alternative is to describe them in qualitative terms (Hensengerth et al. 2012). Moreover, multiple values at individual, community, or societal levels have to be considered alongside the economic analysis (UK National Ecosystem Assessment 2014). This requires effective systems to be in place to link non-state actors, i.e. individuals and communities, to decision-making processes that within a transboundary context are more often than not state-centric (see discussion below). The aggregation of different types of values, perceptions of them, and the ecosystem services they mainstream are crucial in the negotiation and design of cooperative arrangements. Shared values across groups, communities, or society, in both monetary and non-monetary terms, tend to differ from those arising from conventionally aggregated individual values (UK National Ecosystem Assessment 2014). Riparians have to perceive that a project is not only at least acceptable but also better than doing nothing or acting unilaterally (Hensengerth et al. 2012). They have an incentive to cooperate not only if the net benefits of cooperation are at least comparable to those accruing in its absence, but also if the distribution of benefits is perceived to be fair (Sadoff and Grey 2002).

3.3 Transboundary watercourses assets and risk mitigation

A further key area to address when considering transboundary watercourses as assets is risk mitigation. The fundamental principles of risk, uncertainty, or preferences will influence both the design and operation of any transboundary watercourse asset system. LeRoy and Singell (1987), elaborating on the canonical 1921 work of Frank Knight, review the underlying logic of risk and uncertainty. Risks are insurable hazards, but uncertainty pertains to uninsurable hazards. Under moral hazard, it is impossible to insure the outcome of entrepreneurship without inadvertently distorting the entrepreneur’s incentives. Under adverse selection, it is impossible to exclude entrepreneurial “lemons”, whose presence would raise the insurance premium to the point at which successful entrepreneurs would drop out. In other words, insurance markets collapse

because of moral hazard or adverse selection, and the key distinction between risk and uncertainty pertains to the presence or absence of markets.

Calling for prudent risk management and an adaptive approach, the UK National Ecosystem Assessment (2014) asserts that, given the ‘scientific uncertainty’ regarding the adverse effects of human development on ecosystems and their services, it makes sense to be cautious: ‘[w]e cannot wait for more complete information as this may result in services being further degraded’. Decisions should be made within a risk-based framework, emphasising flexibility and ‘learning by doing’ (UK National Ecosystem Assessment 2014). Depending on the extent of flexibility or the pace of learning, it seems practical to ask whether or not an ecosystem could be allowed to ‘fail’ in the same way as a business might. Consider the PES schemes mentioned above. Is it possible to insure the outcome of ecosystem management without inadvertently distorting the ecosystem manager’s incentives? An arbitrary level of payment, completely unrelated to the expected damage, could lead to over- or under-compensation. Is it possible to exclude ‘lazy’ ecosystem managers and offer payments only to industrious ones? The presence of ‘lazy’ ecosystem managers could raise the overall cost of the payment scheme or discourage industrious ones from participation. Is it possible to have an ecosystem service which is systematically important? Saving one or another ecosystem service from failure regardless of the ecological circumstances brings up the twin issues of perverse mechanisms or the design of efficient payment schemes.

The problem of missing markets is at the heart of risk management. Willems and Morbee (2010) analyse the missing markets problem and risk management. Markets are incomplete if perfect risk transfer is impossible, due perhaps to an insufficient set of hedge products. Not all potential gains from trade are exhausted in an incomplete market, and total welfare can be improved if a sufficient number of markets are added. In the context of ecosystem services, schemes seeking to replicate or recover the portfolio in case of a ‘bad’ event are essentially hedge products. Capon and Bunn (2015) provide several examples. Given the risks posed by climate change to the future provision of water ecosystem services, options for supply-side adaptation seek to secure their protection, restoration, enhancement, or replacement. If climate change risks exceed the capacity of networks to protect water ecosystem services, new reserves may be located in areas expected to be highly valuable in the future. Protecting stream flows from water extraction after long periods of drought introduces temporal and spatial flexibility into threat management strategies. It is practical to ask whether or not there is adequate liquidity or price discovery in the market for hedges. How likely would hedge markets collapse if the risks posed by climate change are not articulated at a level sufficiently granular for their valuation? A wide scope, and thus a potentially high cost, of supply-side adaptations to protect, restore, enhance, or replace water ecosystem services may require an exceedingly sophisticated market for hedging. How likely would a weather derivative, having a post-drought stream flow for its underlying commodity, achieve a perfect transfer of risk? Huge unknowns in the temporal or spatial behaviour of fluvial systems after a severe dry spell could raise hedge premiums or cause a misalignment between contracted protection and actual damage.

The concept of preferences is closely associated with the concepts of risk and uncertainty. A systematic approach to risk management, Lo (1999) explains, consists of price (how much to pay for hedging risk), probability (how to assess the likelihood of risk), and preference (how much risk to bear and how much to hedge). Interestingly, Lo (1999) asserts that, of the three, preference is likely the most important but the least understood. Coleman (2011) extensively analyses the crucial role of risk preferences. He asserts that, ‘[p]references are difficult to measure and vary from one investor to another’, and whether

or not a distribution is risky depends on the particular investor's preferences. There is no unique 'risk ranking' for all distributions or all investors.

In the context of transboundary watercourse asset management, riparians are likely to have different preferences, and the diversity of their preferences is necessary to produce a ranking of distributions. The risk mapping between ecosystem services and societal preferences for them requires close scrutiny. Although human well-being depends on biophysical underpinnings and society must conserve and protect aquatic natural capital, water policy should aim for '... good ecological status within the context of societal preferences for the water environment' (Blackstock et al. 2015). For cultural ecosystem services, additional research is needed to comprehend the preferences of diverse socio-economic or demographic groups (Church et al. 2015). Moreover, riparians, as discussed above, may apply different weights on various economic or environmental factors, which are obviously dynamic, and their perceptions of acceptability, benefits, or fairness are central not only to the valuation process but also to a shared characterisation of risk. The provision of ecosystem services may be 'vulnerable to regime shifts', and, if such a shift occurs, sizeable harms may be irrevocable or difficult to repair (Schaafsma et al. 2015). Indeed, '... the value of ecosystems is not fixed, and benefits depend on timing and location of ecosystem service delivery, on the relationship between water quality and quantity, on other ecosystem services related to water and finally on stakeholders' preferences' (Schaafsma et al. 2015).

It is practical to ask whether or not there is a strong potential for tail risk, indicating a highly skewed distribution. How likely would the restoration or repair of a severely damaged ecosystem service be rendered meaningless? An exorbitant hedge arising from an expectation of a massive shift in the temporal or locational regime of an ecosystem service may cause an upheaval in stakeholder preferences. The 'tipping point' could have been reached, and going back may cause unreasonably high costs. For example, Ranganathan et al. (2008) show that the collapse of a fishery could threaten livelihoods or cause conflict. In other words, there is a need to be sensitive to situations in which a natural sentiment for 'staying on' could dominate a logical inclination for 'moving on'.

4 Governance considerations in optimising transboundary watercourse treaty regimes

So far, this paper has explored how certain economic concepts and tools might advance transboundary water cooperation. This has included an analysis of property rights and the concept of perfect contracts. The paper has also explored ecosystem services and natural capital as a basis by which to view transboundary watercourses and their associated benefits as assets. This has led to a discussion of how such assets might be incorporated into a portfolio of assets that might be managed between states. Challenges in applying these economic concepts and tools have been discussed throughout the previous sections. However, one key challenge that is worthy of separate attention relates to the underpinning legal and institutional framework that might support any system of portfolio optimisation of transboundary watercourse assets. Conversely, ensuring that an adequate legal and institutional framework is in place is fundamental to addressing some of the key challenges that have previously been identified, such as lack of information, uncertainty, coercion, power asymmetry, valuations, and high transaction costs.

Prior to considering design features relating to legal and institutional frameworks for transboundary water cooperation, it is important to provide some context. The current environment for transboundary water cooperation can best be described as fragmented (Zawahri and Mitchell 2011). This poses a major challenge in ensuring that appropriate legal and institutional frameworks are in place to foster market-based approaches to transboundary cooperation. Traditionally, there has been a disconnection between geophysical and political watercourse boundaries. More than half of the world's transboundary watercourses lack a cooperative framework, and among those that have agreements, approximately two-thirds do not include all basin states and/or fail to establish a complete set of rules and principles for the equitable and sustainable management of transboundary waters (UN-Water 2008). Moreover, while stakeholder voices and the national discourse on cooperation are crucial to decision-making and affect the likelihood of cooperation, there is often a disconnection between national and transboundary decision-making processes (Subramanian et al. 2014). While an analysis of different national jurisdictions is beyond the scope of the paper, it is important to recognise here that transboundary water cooperation is also contingent on effective alignment between national and transboundary governance levels.

Regardless of these challenges, states sharing a transboundary watercourse inevitably have to contend with the highly sensitive issue of allocating a finite resource. In some cases, this has resulted in a 'tragedy of the commons' scenario whereby states unilaterally seek to get the most out of shared natural resources while minimising the costs of exploitation (Hardin 1968). However, others have maintained that such a tragedy of the commons is not inevitable (Ostrom 1990). Benvenisti (2002) maintains that 'with transboundary resources such as ecosystems, which are international common pool resources, there is a palpable potential for collective action between the co-owner states that will provide an optimal and sustainable use of the resource'.

The foundations for fostering collective action within international law already exist. The cornerstone principle addressing cooperation over international watercourse stipulates that states 'shall in their respective territories utilise an international watercourse in an equitable and reasonable manner' (Art. 5, UN Watercourses Convention). When determining what is equitable and reasonable, states are encouraged to take all relevant factors and circumstances into account, with a view to attaining 'optimal and sustainable utilisation thereof and benefits therefrom' (Art. 5 and 6, UN Watercourses Convention). Relevant factors include the physical characteristics of the watercourses (e.g. geographic, hydrographic, hydrological, climatic, and ecological); the social and economic needs of the watercourse states and their populations; the effects of existing or potential uses of the watercourse; the need to protect ecosystems; and the availability of alternative uses of comparable value to existing or planned uses.

While international law does not go as far as stipulating how these factors or circumstances should be weighed against each other, the principle of equitable and reasonable utilisation provides the legal foundations by which to support the portfolio optimisation of transboundary watercourse assets. Firstly, in taking into account all relevant factors and circumstances through an ecosystem services lens, it may be possible to more accurately identify the benefits derived from transboundary watercourse assets, as well as some of the services that have been traditionally undervalued. The relevant factors or circumstances are reflected or revealed in the process of optimising the portfolio of transboundary watercourse assets. Secondly, the aim of equitable and reasonable utilisation is to 'provide the maximum benefit to each basin state from the uses of the waters with the minimum detriment to each' (ILA 1996). Trading watercourse assets, as discussed above, offers one way

by which such benefits might be maximised. The trading or transformation of ecosystem products reveals their opportunity costs to riparians.

In principle, the portfolio optimisation of transboundary watercourse assets may provide a promising means by which states cooperate. However, some notable challenges must be taken into account. Firstly, the nature of transboundary ‘problems’ are likely to influence whether or not states enter into cooperative arrangements (Underdal 2002). There is a higher likelihood of cooperation if the states see some gain in such cooperation. Cooperation between Canada and the USA on the Columbia River system, as illustrated above, offers one such example. Other situations may not be so conducive to cooperation (Schmeier 2013). Where activities in one state cause harm another, it may be difficult to encourage states to establish a cooperative framework which fully accounts for such externalities. This may especially be the case where there are power asymmetries between states (Zeitoun and Warner 2006). However, one positive example of cooperation in such a scenario is the case between France and the Netherlands concerning the Mines de Postasse d’Alsace. In this case, the discharge of waste salts from a French state-owned potassium mine caused chloride pollution in the Netherlands. The Rhine states in 1986 were able to reach an agreement which called for the progressive reduction in chloride levels in the Rhine. Costs associated with pollution reduction were shared between France and Germany (30% each), the Netherlands (40%), and Switzerland (6%). It should be noted, however, that such a cost sharing arrangement is at variance with the polluter-pays principle and rather ‘reflects economic reality more than principle or law’ (McCaffrey 2003). One reason for this is that the polluter-pays principle ‘remains open to interpretation, particularly in relation to the nature and extent of the costs included and the circumstances in which the principle will, perhaps exceptionally, not apply’ (Sands and Peel 2012).

Strategies for fostering cooperation between states where transboundary externalities are evident might therefore need to rely on more than just customary international law principles. Side payments, either between riparian states themselves or through third-party intervention, have been identified as one way to induce such cooperation (Dinar et al. 2007). An example of such a cooperative arrangement can be seen in the Protection of the Orange–Senqu water sources (‘SPONGE’) project in Lesotho, where donor funding was used to improve land-use practices in upper part of the Orange–Senqu River (GOPA 2016). In so doing, the project had the objective of holistic protection and conservation of the ‘sponges’ in the Khubelu catchment with a view to ‘securing long-term availability and quality of water from the Upper Orange–Senqu catchment area’. The project, in turn, brings benefits to downstream states on the Orange–Senqu River, namely South Africa, Namibia, and Botswana. In other words, side payments, supporting compliance with international agreements without weakening existing sanctions, change the payoffs and help bring about efficient solutions.

While more fully accounting for the hidden or undervalued ecosystem services may be an advantageous way by which to ascertain the relevant factors and circumstances, asymmetric relationships between states may also come into play. States may feel that ecosystem services which upstream catchment areas provide, such as natural storage or flood prevention, are something for which downstream states should pay. If downstream states have traditionally benefitted from such services without paying, then changing that dynamic may prove challenging. This may especially be the case where downstream states are classified as ‘hydro-hegemony’ (Zeitoun and Warner 2006). As we discuss above, third-party side payments might be one option to foster cooperation in such a wider benefit which could be derived from such cooperation—e.g. mitigation of likely impacts from climate change—may also provide an incentive for states to cooperate. Moreover, such cooperation would be

underpinned by the customary international law obligation placed upon states to cooperate in good faith over international watercourses (Leb 2015).

Some efforts have been made to advance PES schemes at a transboundary level within existing governance settings. Under the UNECE Water Convention, Recommendations on Payments for Ecosystem Services in Integrated Water Resources Management were adopted in 2006. These recommendations provide guidance on the establishment of PES schemes, as well as examples of schemes which have been adopted within the UNECE region. While project-specific application remains limited, the UNECE example demonstrates the willingness of states to explore market-based approaches to transboundary water cooperation. This willingness is also reflected in the significant number of soft law instruments which call upon states and others to advance progress in the valuation of ecosystem services and explore innovative means for financing ecosystem protection.²

What remains to be done is to design treaties which optimally reflect the concepts of portfolio optimisation and commodity trading of transboundary watercourse assets. This is a huge challenge. While significant advances have been made, the process by which to identify and value ecosystem services is still subject to much debate over definitions, measurement techniques, boundaries, scales, societal values, trade-offs, non-market good, and uncertainties about the link between biological diversity itself and the services that flow from ecosystems (Cook and Spray 2012; Norgaard 2010). Moreover, within a transboundary context, it will be important that there are adequate mechanisms in place by which states regularly exchange data and information.

Many transboundary watercourse treaties include provision on the exchange and information (Allan 2012; Gerlak et al. 2011). While these treaties do not mention ecosystem services, they might offer the foundations by which to undertake any ecosystem services mapping and valuation exercise. Joint institutions must also play a role in the gathering of data and information, as well as in managing trade-offs between different users and interests. However, existing institutions tend to be too narrow in their focus. Irwin and Ranganathan (2007), for example, suggest that:

Today's institutions ... usually focus on a single sector such as forestry or finance at a single political level or geographical scale and often on a short timeframe. A community group or social network is concerned about local livelihoods, a national agency about planning national development, and a secretariat of an international convention about improving the state of specific types of resources such as biodiversity, migratory species, or wetlands globally. At every level, institutions are handicapped by their limited mandate, capacity or incentive to cooperate across geographical or political boundaries, or to consider the longer timeframes often needed to manage ecosystem services effectively. If ecosystem services are to be sustained to support human well-being, decision makers need to address drivers and effects of change that emerge at different levels and time scales. This will require working outside of traditional boundaries in multidisciplinary settings.

² See, for example, 4th Ministerial Conference on Protection of Forests in Europe, Vienna, 28–30 April 2003; Statement of the Ministerial Meeting on Forests, Rome, 14 March 2005; 13th Session on Water, Sanitation and Human Settlements, New York, 11–22 April 2015, Resolution IX 3–4 of the 9th Meeting of the Conference of the Contracting Parties to the Convention on Wetlands, Kampala, 8–15 November 2015, the 2006 International Tropical Timber Agreement, and the 6th session of the UN Forum on Forests, 13–24 February 2006; and Decisions VIII/9 and 17 of the 8th Ordinary Meeting of the Conference of the Parties to the convention on Biological Diversity, Curitiba, Brazil, 20–31 March 2006).

Adopting a holistic approach within the context of transboundary watercourses is not straightforward. The Millennium Assessment (Hassan et al. 2005), for instance, recognised that:

...the water cycle plays so many roles in the climate, chemistry, and biology of Earth, it is difficult to define it as a distinctly supporting, regulating, or provisioning service. Precipitation falling as rain or snow is the ultimate source of water supporting ecosystems. Ecosystems, in turn, control the character of renewable freshwater resources for human well-being by regulating how precipitation is partitioned into evaporative, recharge, and runoff processes. Together with energy and nutrients, water is arguably the centre piece of the delivery of ecosystem services to humankind.

Recognition of water's central role in sustaining ecosystem services has led to repeated calls for a river basin approach to water resources management (UN 1992). At the political level, a notable step forward is the adoption of the Sustainable Development Goals which, under Goal 6.5, requires the international community to implement integrated water resources management at all levels, including through transboundary cooperation as appropriate (UNDESA 2016).

Further recognition of the need to establish legal and institutional arrangements is evident from the adoption and eventual entry into force of the Convention on the Law of the Non-navigational Uses of International Watercourses in 2014 (UN 1997; Rieu-Clarke and Loures 2013), and the 'global opening' of the Convention on the Use and Protection of Transboundary Watercourses and International Lakes (UNECE 1992), which means that *all* States across the world may now become parties to the Convention. National governments are also increasingly recognising the importance of adopting legal and institutional frameworks to address challenges of water scarcity, climate variability, and security (Rieu-Clarke et al. 2012; US Senate Committee on Foreign Relations 2011).

A further challenge relates to the engagement of stakeholders at multiple levels in transboundary water cooperation. As noted previously, engagement of stakeholders beyond national governments is fundamental to the effective valuation of ecosystem services and any trade-offs between services and bundles of services. The Millennium Ecosystem Assessment (Hassan et al. 2005), for instance, observes that:

...a key focus for improving participatory processes is to help level the playing field through measures to increase the transparency of information; improve the representation of marginalised stakeholders; engage them upfront in the establishment of policy objectives and priorities for the allocation of freshwater services, with which specific project should be consistent; and create a space for deliberation and learning that accommodates multiple perspectives.

While the challenges of participation have been documented elsewhere (Rieu-Clarke 2010), it is important here to focus specifically on transboundary considerations. In this regard, Bruch et al. (2005) comment that 'people often have little or no opportunity to participate in watershed decisions that affect them, particularly when they live along international watercourses'. There has, however, been a growing recognition of the need to engage stakeholders within transboundary decision-making structures (Tignino and Sangbana 2015). Some treaty arrangements provide specific rights for non-state actors. The 1992 UNECE Water Convention, for example, requires that 'information concerning the conditions of transboundary waters, measures taken or planned to be taken to prevent, control and reduce transboundary impact, and the effectiveness of those measures, is made available to the public'. Under the EU Water Framework Directive, the public are to be given access

to data and information relating to the development of river basin management plans, and member states are encouraged to ensure the active involvement of all interested parties in the implementation of the Directive. A suite of law relating to human rights also supports the participation of non-state actors in transboundary decision-making processes (see, for example, the UNECE Aarhus Convention).

While the above discussion only provides a snapshot of the governance considerations that should be taken into account when seeking to optimise transboundary water treaty regimes, a number of important points can be observed. Firstly, the founding principle of international water law, equitable and reasonable utilisation, provides the foundation by which an ecosystem services approach to valuation might apply at the transboundary level. Secondly, there is growing momentum in support of valuing ecosystem services, including some embryonic initiatives at the transboundary level. Thirdly, both the challenges and importance of joint institutional arrangements have been highlighted. Lastly, the need to enhance the role of non-state actors in transboundary decision-making constitutes a key challenge and opportunity.

5 Conclusion

The first principles of law and economics support the design of legal or institutional arrangements facilitating a market-based approach to the discovery and allocation of ecosystem services among riparians. Potentially, a system of portfolio optimisation and commodity trading can be embedded within a transboundary treaty regime. There seems to be adequate foundations for fostering collective action within the law of international watercourses along such a basis, and advances in knowledge and understanding relating to ecosystem services also provide an important foundation for market-based approaches. Moreover, various soft law instruments reflect a political will to implement a system of portfolio optimisation and commodity trading for transboundary watercourse assets. While this paper has recognised that notable challenges exist, it has also shown that the potential is there and more should be done to study how market-based approaches might be applied to transboundary watercourse treaty regimes.

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