Economics in integrated water management

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ABSTRACT

Integrated basin scale analysis that accurately accounts for the impacts of proposed policies on the environment and society’s economic welfare can comprehensively and consistently inform water resource policies. Cost benefit analysis (CBA) has considerable potential to support water decisions by consistently appraising proposals in terms of society’s total environmental and economic impact in monetary terms. However, the difficulty of correctly applying CBA to environmental programs with complex physical and economic interactions weakens policymakers’ confidence in comprehensive economic assessments at the basin scale. This paper describes and illustrates a method by which costs and benefits can be systematically integrated into an integrated physical, institutional and economic analysis for a river basin. A simple hydroeconomic model is presented. Its size is small enough to build, understand, and interpret with paper and pencil. But its structure is sufficiently flexible to permit expansion for comprehensive policy design that rests on a foundation of a basin’s hydrology, institutional constraints, and economic relations. The use of cost benefit analysis to support environmental policy will always be limited by ethical questions on the distribution of benefits and costs among sectors, income groups, locations, and generations. Nevertheless, hydroeconomic models offer a potential resource to efficiently and consistently integrate hydrologic, economic, and institutional impacts of policy proposals to support basin scale cost-benefit environmental assessments.

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1. Background

It has been recognized for centuries that the allocation of water to one use at one place in one time period affects other uses of water and other resources in other places and in other time periods. In responding to that challenge, the principle of the river basin as a unit for comprehensive planning and management has emerged. For example, in the 1870s, John Wesley Powell urged that the new governments of the American west be formed around river basin boundaries. While his advice was ignored in the political boundaries established, river basins have emerged more recently as a principle for organizing water resource management and policy. For a recent example, the 1992 International Conference on Water and the Environment formulated the Dublin Principles and focused on the river basin as a unit of analysis. The comprehensive planning approach was further advanced by the 1997 United Nations Convention on the Law of the Non-Navigational Uses of International Watercourses, and the 2000 report of the World Commission on Dams. In 2000 the European Union established its Water Framework Directive (WFD). The WFD established a common approach for protecting the water environment, setting environmental objectives for all waters of the European Union (EU), and providing a framework for designing and evaluating future EU water legislation.

In both research and policy circles there is continued emphasis on integrated river basin analysis in Europe, US, Australia, and numerous developing countries. A few examples include work on cost-benefit analysis of the WFD (Bateman et al., 2006), integrated economic–hydrologic–ecologic analysis of the Patuxent River basin in Maryland (Bockstael et al., 1995), a review of hydroeconomic models (Heinz et al., 2007), conference proceedings on hydroeconomic models (Copenhagen, 2004; Amsterdam, 2005; Valencia, 2006), ecological sustainability (Richter et al., 2003), analysis of climate change on various continental river basins (Nijssen et al., 2001), integration of atmospheric and hydrologic models at the basin scale (Benoit et al., 2000; Leung et al., 2004; Miles et al., 2000), river...
basin analysis in Chile’s Maipo river basin (Rosegrant et al., 2000, Cai et al., 2003), multiple-criteria evaluation for several European countries (Mysiak et al., 2005). Work has been conducted on an integrated hydrologic-agricultural-economic model to support improved irrigation efficiency (Gai et al., 2003), analysis in Colorado that tracks hydrologic and institutional constraints (Dai and Labadie, 2001), improved information at the basin scale to support the WFD (Van Ast and Boot, 2003), measures for supporting both the EU Nitrates Directive and the WFD (Fasso et al., 2005), nutrient management in the Rhine Basin (van der Veen and Lorenz, 2002), and climate change assessment at the basin and regional scales.

Several papers recently published in Environmental Modelling & Software have implications for integrated basin-scale water management. Examples include the analysis by Poch et al. (2004) on environmental decision support systems, and Newham et al. (2004) and Mysiak et al. (2005) on integrated basin scale management, Gilmour et al. (2005) and Letcher et al. (2007) developed a framework for improved integrated water resource assessments. Booty et al. (2001) described methods to improve environmental decision support systems.

Despite these extensive achievements, little work has been done that illustrates simply and clearly how costs and benefits can be incorporated into an integrated physical, institutional and economic model to support water resources planning and policy. However, the difficulty of correctly applying CBA to environmental programs with numerous physical and economic interactions weakens policymakers’ confidence in comprehensive economic assessments at the basin scale.

Because of these gaps in the literature, this paper’s objective is to describe and illustrate one method by which costs and benefits can be systematically integrated into a physical, institutional and economic model for a river basin. A simple hydroeconomic model is presented. Its size is small enough to build, run, and interpret with paper and pencil. But its structure is sufficiently flexible to permit expansion for comprehensive water policy design that rests on a foundation of a basin’s hydrology, institutional constraints, and economic relations. Economic concepts are briefly presented. Then the model is described, critiqued, and interpreted as it could support design of water resource policies. Model results are shown. Last, the paper concludes.

2. Economic concepts for integrated analysis

Cost benefit analysis (CBA) can be used to provide information needed for three kinds of public environmental policy decisions: (1) a simple ranking of the comparative benefits of several possible actions; (2) the optimal size or scale of a project produced by a decision; and (3) the optimal timing or sequencing of several elements of a decision.

An early example of written public law that motivated development of CBA came from the US Federal Flood Control Act of 1936. This legislation permitted the US Army Corps of Engineers to build flood control projects only if the total benefits of a project to whomever they accrue exceeded its costs. So, the Corps of Engineers had to create systematic methods for measuring those benefits and costs (Hufschmidt, 2000). Recent years have seen CBA applied to a huge range of water resource policy questions in many countries and cultures. A few examples include crop irrigation (Al-Karkhi, 1998), surface water treatment regulations (Regli et al., 1999), drip irrigation (Tiwari et al., 1998), groundwater quality improvements (Yadav and Wall, 1998), health risks from drinking water (Odum et al., 1999), agricultural water pollution control (Qu, 2003), improvements of sewer systems (Schultz et al., 2004), groundwater recharge (Bozian et al., 1999), rainwater harvesting (Ngigi et al., 2005), river health (Bennett, 2002), and water re-allocations (Messner et al., 2006).

Water managers who rely on CBA to inform their decisions use a simple decision rule. If for some proposed action, the discounted net present value of benefits exceeds those of the costs by a larger amount than any other action with the same aim, the proposed action should be adopted where economic efficiency is the objective. Otherwise it should not.

This decision rule assumes that a marginal dollar lost or gained is worth the same to everyone, regardless of current income level. That is, an additional dollar of benefit produced by a public water program for a rich person is worth the same as a dollar of cost paid by a poor taxpayer who finances that benefit. Properly conducted, information provided by a CBA helps identify programs for which the value of a public program’s services exceeds the cost of resources used to implement that program. This ensures that the program supports economic efficiency gains in water management choices.

The basin’s economic data on price elasticities of demand for agricultural and urban water uses are based on a recent analysis of urban and agricultural use of water in North America’s Rio Grande Basin (Ward et al., 2006). The basin’s water supplies and water delivery obligations are illustrative only and represent the water supply facts of no known river basin.

The development and use of empirically-based computer models typically presents considerable advantages compared to theoretical paper-and-pencil mathematical equations. The advantage of a computer model lies in its capacity to handle enormously complex relations between hydrological, environmental and economic processes (e.g. Heinz et al., 2007). Nevertheless, the simple expandable mathematical prototype presented in this paper occupies a useful niche because of its clarity, simplicity, and focus, especially if can be expanded to a detailed empirical computer model.

Cost benefit analysis (CBA) is a comprehensive economic method of analysis to support decision-making, typically used by governments to evaluate the desirability of a proposed policy. Its aim is to gauge the economic efficiency of a proposed policy compared to a baseline status quo. Measured in monetary terms, costs and benefits associated with the impacts of the proposal are evaluated in terms of the public’s willingness to pay for the outputs (benefits) or willingness to pay for the inputs used up in making the outputs available (costs). Costs of project inputs are typically measured in terms of opportunity costs, namely the value in their best alternative use displaced by the proposal. The underlying principle for implementing a basin scale water policy CBA is to list all parties affected by the proposal, and place a monetary value of the effect it has on their welfare as valued by them. CBA can help inform decisions on sizing, timing, or sequencing of projects. As to sizing, CBA can show if expanding a project’s current proposed scale increases or decreases discounted net present value. Regarding timing, CBA may show that introducing a project in period Y produces greater discounted net present value than introducing the same project in period Y. For sequencing, three projects ordered in X–Y–Z may produce greater discounted net present value than in some other order, such as Y–Z–X. This paper focuses on CBA, but other economic approaches of a more limited scope exist for evaluating water policies. Examples include cost-effectiveness analysis, multi-criteria analysis, economic growth studies, input–output models, and non-monetary environmental impact assessments.

This paper presents some flexibility. It has the potential to account for impacts of a public program on people of different income levels. If desired, varying weights can be assigned to benefits and costs by income level. Lower income people could be assigned higher weights in a more comprehensive CBA. Differential weights assigned to different income classes permit assigning weighted benefits and costs accruing to people with different incomes. Higher weighted costs can be interpreted as benefits forgone for low income people than the same cost for benefits lost by the rich.
2.1. Looking forward or backwards

Many water resource applications of CBA deal with ex ante (planned) policy questions. However, policy analysis that conducts only ex ante analysis without examining the historical performance of existing programs would not learn from past mistakes. By contrast, ex post analysis looks backward and asks how well an existing project, program, or regulation performed after it was established. Ex post information can be used for three purposes: (1) to examine the stream of actual benefits and costs produced by actual projects built or policies enacted and to see if the previous ex ante CBA was accurate, and if not, identify what errors were made; (2) to revise methods, forecasts, and assumptions where mistakes were made; (3) to gain information on the existing economic impact and value relationships on which future CBAs ultimately rest.5

2.2. Comprehensive analysis

In principle, a CBA can be used to inform water managers who design, implement, and review policies. The use of CBA can sharpen the support of decisions by comprehensively accounting for economic benefits and costs of various water resource programs, plans, and policies. CBA can also identify the distribution of benefits and costs among society’s groups.

CBA’s focus on a project’s economic feasibility can contribute to the political process. CBA can be used to measure the overall relationship between benefits and costs as well as the distribution of those benefits and costs among groups. Armed with good information on benefits and costs, designing a policy that produces a positive economic efficiency gain for more interest groups can make it more politically attractive than concentrating benefits or costs on a few groups or small numbers of people.

2.3. Policy goals

An ancient challenge surrounding the design of government policy is the question of what ends are served by government activity and what programs can be established to best meet those ends. A CBA is based on the policy objective of economic efficiency. This principle states that economic efficiency is the essential standard for evaluating government regulations or programs that are proposed for adoption, maintenance, or change. So targeting the efficiency objective increases the likelihood that water policy proposals for adoption, maintenance, or change. This principle states that economic efficiency is the essential standard for evaluating government regulations or programs that are proposed for adoption, maintenance, or change. So targeting the efficiency objective increases the likelihood that water policy actions will only place burdens on businesses and consumers that are in proportion to improvements in health, safety, or the environment.6

2.4. Incremental analysis

The use of CBA can contribute to the design of more efficient policies by identifying marginal benefits and marginal costs resulting from several small changes away from the status quo rather than limiting analysis to a single large scale all-or-nothing proposal. An example of limiting comparisons to two extremes is a policy choice that compares two water policies, e.g. allocating all water to farms or all to cities. Such an all-or-nothing proposal boxes policymakers into one of two undesirable corners. They have little opportunity to compare small scale changes in plans by assessing marginal costs and marginal benefits of allocating slightly more water to cities and slightly less to cities. Information on marginal cost is valuable because it permits a comparison with marginal benefits arising from other uses of the same resources. For this reason, incremental analysis permits greater refinement in the design, review, or implementation of water resource decisions.

For the water reallocation decision, maximum economic efficiency is achieved when marginal benefit per added unit of the resource used is equal for all uses.7 This ideal has been called the equimarginal principle for water reallocation. One good example of a policy that reallocated water among competing users is the US Central Valley Project Improvement Act of 1992, that reallocated 800,000 acre feet of water from irrigated agriculture to improve fish and wildlife habitat (US Fish and Wildlife Service, 2004). As a consequence of changing the water allocation, the benefits gained in added fish and wildlife habitat supplied exceed the benefits lost in reduced agriculture, so that the total economic benefit of water use is raised and marginal benefits in each of the two uses are nearly equal. Furthermore, in the real world of water policy formulation, non-marginal changes in water supply, for example through increases in the size of water treatment plants, are more common than marginal ones.

Benefits and costs are measured with the program compared to benefits and costs that would have occurred without it. Use of this principle ensures that measured benefits (or costs) are due solely to the program or project, rather than changes that would have occurred even without it. For the analysis of water policies at the river basin scale, “with versus without” is an important guiding principle. Carrying it out requires defining a clear baseline policy against which to measure the incremental benefits and costs of a proposed policy. Defining that baseline that would occur without the project guards against falling into the double counting trap. For example, some water resource policy analyses have mistakenly counted as benefits those changes that would have occurred even without the policy. A classic example is the mistake of crediting increased crop yields to an irrigation project that would have occurred even without the project.

The total economic efficiency achieved by a series of water projects can vary according to the order in which they are carried out. Implementing one program right away may influence the benefits or costs produced by a related program in a later period. For example, consider the case of a plan to enact drinking water quality regulations that improves human health before introducing a program whose output is aesthetic such as instream flow protection for recreation benefits. That sequence of programs may produce greater economic efficiency benefits than introducing the programs in reverse order. Introducing the health program first likely produces more total net benefits because there will be more healthy people to enjoy the improved aesthetics from the second program. If regulatory resources are scarce, then where one program’s outputs influence other programs’ benefits or costs, it may be most efficient to examine several possible time sequences for introducing the programs.

5 For example, if ex post analysis showed that reducing each ton of emissions into a river connected to a drinking water source reduced hospital admissions by source X, that fact tells us something about the future value of controlling emissions by a different source Y.

6 This assumes that all costs and benefits arising from a policy program can be measured in the CBA framework. Even if those impacts can be measured, critics of the use of CBA state that it ignores the distribution of benefits and costs, and that a simple summing of costs and benefits across all affected individuals ignores important considerations of equity. CBA might be more politically acceptable and become more widely embraced if water resource programs are designed to account for equity. One way to accomplish equity is to design programs for which benefits exceed costs to most groups, such as people in most tax brackets, time periods, geographic locations, ethnic backgrounds, and economic sectors.

7 For a single time period, achieving maximum economic efficiency is equivalent to securing the maximum value of benefits minus costs.
Fig. 1. Structure of environmental policy analysis.
2.5. Linking physical and economic effects

Fig. 1 shows the steps required to conduct a basin scale environmental analysis: policy scenario, flows, conditions, physical effects, and economic outcomes. Each of the five are conducted both for a baseline policy and for a proposed policy. Beginning in the upper left, a baseline policy is defined as the one that would occur into a specified future under status quo conditions (no new policy). For a river basin, that policy produces a series of future physical flows, such as a base streamflow and a base level of pollution emissions into various water bodies. These flows give rise to conditions at various locations in the basin, including reservoir levels and pollution concentrations in water bodies. These conditions produce short term and long term effects, such as a base level of human health, drinking water quality, commercial fish harvests, and the like. On the far right are economic costs and benefits associated with these effects, including base costs of ill health, base costs of poor drinking water, and the like. The bottom half of the diagram shows the same five steps going from a proposed policy at the left to its economic consequences at the far right. Basin scale water policy analysis requires a baseline series of five steps as well as the same five steps for each policy being considered.

The top half of the figure presents a major challenge. It requires a forecast of impacts of the status quo policy, if continued, on flows, conditions, effects, and on economic outcomes for all relevant future years. The bottom half is an even larger challenge, since it deals with a future counter to the historical facts. It requires forecasting impacts of a proposed policy, not currently practiced, on the same four outcomes, also for the same future years.

3. Methods

3.1. Hydroeconomic model structure

A hydroeconomic model is an organizing framework managers can use for implementing comprehensive river basin scale analysis. For example, the European WFD provides a guideline for such a comprehensive economic analysis. Water researchers and managers can achieve that integration by building, verifying, testing, and applying results of a hydroeconomic model. Despite the widely-recognized integration capability hydroeconomic models present, water managers often find that if they wish to learn more about its details, much is packed away on somebody’s hard disk, with detailed description rarely available for open critique.

For this reason the next section presents a simple expandable prototype hydro-economic model. Its structure is so simple that it requires only minimal detailed documentation, summarized in the appendix. Yet, despite its simplicity, considerable important detail on hydrology, economics, and institutions potentially can be added to the model. That added detail could make the model sufficiently subtle, complex, and realistic to support informed water management decisions that account for unique hydrologic, economic, and institutional characteristics of a watershed under study.

3.2. River basin structure

Fig. 2 illustrates the simple structure of an expandable prototype basin. It contains the following basic elements summarizing the major supply and demand nodes:

- watershed inflows measured in water volume per unit time, rainfall plus snowmelt
- agricultural water use, river diversions
- urban water use, river diversions
- streamflows at river gauges
- minimum delivery requirement to downstream location (e.g. an international treaty)

While this basin diagram is simple, it contains many of the important elements to support a full basin scale analysis. The skeletal structure can be expanded to include things like minimum stream flow requirement for species protection, water pollution emissions and their downstream tracking of concentrations, and various water chemical indicators for water quality. Several farming areas, urban nodes, storage reservoirs, tributaries, national borders, environmentally sensitive areas, and other kinds of water use nodes could be added to this diagram.

3.3. Economic efficiency as an objective for water management

The model described here is based on the principle of maximizing total basin-wide economic efficiency, subject to hydrologic and institutional constraints. The WFD emphasizes the importance of the economic efficiency objective in meeting its aims, including measures to improve water quality. Still, allocation of water quantity among competing uses subject to environmental constraints is a large economic and political issue for countries in southern Europe and in many other dry places (Gomez-Limon et al., 2002).

This simple model described in this paper allocates water among uses and time periods (Eq. (5)) so that discounted total net economic benefits are maximized (Eq. (12)) consistent with available water supplies (Eq. (1)). The water allocation is also consistent with various institutional constraints that limit the freedom to allocate water (Eq. (6)). Streamflow is measured at the various stream gauges (Eqs. (2)—(4)).

Agricultural benefits of irrigation diversions are commonly measured by use of farm family crop enterprise budgets, in which farm income is compared with and without additional stream diversions. Data required are crop prices, costs of production, crop yields, and crop water use. Urban benefits are often measured as the water bill paid plus any unpriced consumer surplus from urban water use. Consumer surplus can be quite high for households for whom prices are set considerably below the costs of supply for equity reasons.

3.4. Institutional constraints

Major institutional constraints limit or define the allocation of water in many river basins. Globally, there are hundreds of transboundary rivers shared by more than one country. As described in the comprehensive database published at Oregon State University (Wolfe, 2007), countries often have formal or informal agreements laying out rules for sharing a river’s water quantity and/or on sharing responsibility to avoid polluting activities. So for historical, moral, military, or legal reasons, a single country or region rarely has the right to use all the water flowing through its boundaries or produced by its watersheds. The mathematical appendix shows the parameters, variables, and equations used to specify and solve the model. The simple model presented here has only a single institutional constraint, namely the requirement that a set quantity of water be delivered downstream of all this community’s water users. However, considerable detail can be added to this constraint, in which various water use or water delivery patterns are either required or prohibited for political, legal, or ecological reasons.

Numerous institutional constraints face water managers, politicians, and treaty negotiators. Examples include water sharing arrangements, sharing of water

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8 The famous “picnic table” talks are a good example of an informal method of allocating water. At these talks, water experts from Jordan and Israel have periodically met for years to discuss how to manage the Jordan River.
### Table 1

**Basin scale cost benefit parameters, results, and sensitivities.**

<table>
<thead>
<tr>
<th>Parameter value</th>
<th>Water diverted</th>
<th>Gross benefit</th>
<th>Gross cost</th>
<th>Net benefit</th>
<th>Shadow price</th>
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</thead>
<tbody>
<tr>
<td></td>
<td>Ag</td>
<td>Urban</td>
<td>Ag</td>
<td>Urban</td>
<td>Ag</td>
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<td></td>
<td>Per acre (a-f/year) Total (ka-f/year)</td>
<td>Per HH (a-f/year) Total (ka-f/year)</td>
<td>Per acre (US$/1000 HH) Total (US$)</td>
<td>Per acre (US$/1000 HH) Total (US$)</td>
<td>Per acre (US$/1000 HH) Total (US$)</td>
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<tr>
<td>Hydrology</td>
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<tr>
<td>Watershed run off</td>
<td>600</td>
<td>3.2</td>
<td>142</td>
<td>0.5</td>
<td>58</td>
</tr>
<tr>
<td>Stream flows</td>
<td>600</td>
<td>4.5</td>
<td>202</td>
<td>0.5</td>
<td>58</td>
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<tr>
<td>Stream diversions</td>
<td></td>
<td></td>
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<tr>
<td>Evaporation</td>
<td>540</td>
<td>1.8</td>
<td>82</td>
<td>0.5</td>
<td>58</td>
</tr>
<tr>
<td>Water applied (environmental, urban, agriculture)</td>
<td>400</td>
<td>3.2</td>
<td>142</td>
<td>0.5</td>
<td>58</td>
</tr>
<tr>
<td>Water used (depleted)</td>
<td>400</td>
<td>2.3</td>
<td>102</td>
<td>0.5</td>
<td>58</td>
</tr>
<tr>
<td>Seepage to aquifer</td>
<td>360</td>
<td>4.0</td>
<td>182</td>
<td>0.5</td>
<td>58</td>
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<tr>
<td>Surface return flow</td>
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<td>Groundwater recharge to aquifer</td>
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<tr>
<td>Aquifer flow</td>
<td>600</td>
<td>3.2</td>
<td>142</td>
<td>0.5</td>
<td>58</td>
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<tr>
<td>Aquifer storage</td>
<td>600</td>
<td>4.5</td>
<td>202</td>
<td>0.5</td>
<td>58</td>
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<tr>
<td>Reservoir area</td>
<td>540</td>
<td>1.8</td>
<td>82</td>
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<td>Aquifer depth</td>
<td>400</td>
<td>3.2</td>
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<td>Institutions</td>
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<td>International treaties</td>
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<td>National legislation</td>
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<td>Intra-regional agreements</td>
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<td>Water transfer limitations</td>
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<td>Economics</td>
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<tr>
<td>Water price</td>
<td>660</td>
<td>3.6</td>
<td>202</td>
<td>0.5</td>
<td>58</td>
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<tr>
<td>Cities’ population and growth</td>
<td>440</td>
<td>2.3</td>
<td>102</td>
<td>0.5</td>
<td>58</td>
</tr>
<tr>
<td>Crop prices and water production relations</td>
<td>360</td>
<td>4.0</td>
<td>182</td>
<td>0.5</td>
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<td>Irrigated and irrigable acreage</td>
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<td>Price and income elasticity of demand</td>
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<td>Pumpsing and surface treatment capacity/cost</td>
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<td>Recreation site facilities</td>
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<td>Water’s value: urban, agriculture, recreation</td>
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**Fig. 3. Comprehensive structure of a hydroeconomic model.**
development, and responsibilities for dealing with climate change. Other institutional arrangements that constrain or require water use patterns could also be analyzed by extending the model described here. Examples include water rights adjudication, historical water use patterns, international delivery obligations, instream flow requirements for endangered species habitat, transboundary water sharing plans, water pollution regulations, and minimum guarantees to supply piped water to secure human rights. For the simple model presented, we specify a single downstream requirement, a volume of water, which must be delivered in every period.

3.5. Hydrologic constraints

This model uses a simple hydrologic constraint defined by a known volume of water runoff available from the various watershed inflow points. For ex post analysis, past streamflow patterns would normally be used. For ex ante analysis, such as climate change analysis, streamflows forecasts would be used. However, ex ante analysis could simply ask how historical water decisions could have been improved while constrained by the historical streamflow record.

These inflow points are typically defined as being natural or near-natural runoff points above all major water uses. One classic way to implement the hydrologic constraints is to use historical gauged inflows for the available period of record. While the model shown here has no reservoirs, reservoir evaporation, return flows, groundwater pumping, aquifer levels, or fate and transport of pollutants, these could be added with sufficient time and money. Fig. 3 presents an example of a basin scale model with the kinds of extensive hydrologic, economic and institutional detail relevant to the kinds of important questions often asked in policy debates.

3.6. Water use and its economic benefits

For this prototype hydroeconomic model, the only two uses requiring water diverted from the stream are irrigated agriculture and urban water use. It is assumed that previous analyses have been conducted from which total economic benefit functions can be algebraically specified for each use. Economic values for urban uses can be found by analyzing urban demands as a function of water's price and price structure, size and structure of households, climate, nature of landscape, income and urban population levels. One example for finding these economic values for urban, agricultural, and environmental uses is presented in Ward et al. (2006).

A common method used to estimate economic values for agricultural irrigation is to develop and use representative farm budgets to identify costs and returns of various agricultural enterprises. For known prices of crop and livestock activities and for known costs of production, surface and/or groundwater supplies can be varied from wet to dry conditions to see how farm income varies in the face of water supply changes. Data used to characterize these relations are shown in Appendix A. The agricultural and urban benefits equations are shown in appendix Eqs. (7) and (8). Parameters used for those two benefit functions came from empirical analysis of urban water use patterns under varying water prices along with data from farms that considers budgets adjusted to various water supply conditions.

Numerically, the parameters, variables, and equations presented in Appendix A, water-related total and marginal benefit functions are derived and specified for the basin model. From those benefit functions, the model’s optimal solution identifies what combination of stream withdrawals at each use node maximizes the basin’s total economic benefits, subject to relevant constraints that limits those withdrawals. In Appendix Eq. (7), per user benefits by use type are quadratic functions of water diverted. Quadratic benefit functions that top out at a finite level of water diversion mean that marginal benefit functions per user (not shown) are linear and downward sloping. A linear marginal benefits function simplifies computations needed to find the level of water diversion at which marginal benefits are zero. Unless specifically constrained, an optimization framework will inform managers to avoid diverting more water than the level at which marginal benefits are zero. The current model is solved with simple calculus. Larger models are normally solved with optimization software, such as the General Algebraic Modelling System.

Economically efficient water planning at the basin scale can be found by identifying maximum discounted net present value of water uses summed over time periods and users. Without groundwater or reservoir storage, each period’s effective supply equals its headwater flows minus its downstream delivery obligations, shown in Eq. (13). Deciding how to best use water supplies that are both hydrologically available and legally allowed is the challenging facing water policy makers that can be informed by integrated basin analysis.

4. Model results

Table 1 shows results used by solving the simple optimization model documented in Appendix A, whose parameters used for the model are also summarized in the table. The parameters described in this table are based on a regional water policy analysis of the Rio Grande Basin in North America (Brinegar, 2007; Ward et al., 2006). Results come from constrained optimization using the standard calculus-based Lagrangian multiplier approach. No special software is required for this model other than a spreadsheet. The tables show the constrained efficient solution for several variables. These include water diverted for each type of use, gross economic benefits by use, gross economic costs by use, positive shadow price for valuing added headwater flows, and negative shadow price (marginal cost) for meeting downstream delivery requirements.

The shadow price measures the net economic value to the basin associated with one additional unit of water if it could be secured. It is an important economic indicator that informs the conduct of any water policy for which a water supply augmentation could be secured through human action. For example suppose the basin is endowed with baseline headwater flows of 600,000 acre feet per year. Imagine that one additional acre foot could be secured by some measure such as negotiating a treaty from an upstream country, or by increasing that part of snowmelt that makes its way to the river. In that case, Table 1’s first row shows that $US 20 is the economic value of the measure used to make the added supply available. The next two rows show how that shadow price changes as the basin’s headwater flow is increased or decreased by 10%. That shadow price, when compared to the marginal cost of securing added supplies, can be used to inform decisions on whether or not it is economically worthwhile to secure those additional headwater flows. The shadow price measures the marginal benefit of the added water supply, which means that it can be compared with the marginal cost of securing the added water. The comparison of the marginal benefit to the marginal cost informs the decision on whether the action should be taken or avoided.

Results also show that as urban population grows, water is transferred from agriculture to urban use. Price elasticity of demand for urban use is not very sensitive to price changes, with a base-case elasticity of ~0.04, typical of households in the Rio Grande Basin (Ward et al., 2006). That is, increased urban demands will bring about water transfers from agriculture, in which per capita urban use falls only very slightly. Moreover, since urban treatment costs are a high percentage of the total price of water ($400 per acre foot typical for the Rio Grande Basin), only a very small part of the price of urban water comes from its absolute scarcity (shadow price). By contrast, agricultural demands are much more price-sensitive than urban demands, with a price
elasticity in the range of $-0.79$ for our base case, typical of irrigated agriculture in the Rio Grande Basin. Almost none of the price of agricultural water comes from treatment costs (only $10$ per acre foot) since irrigation water often needs only little treatment. The upshot of these economic facts is that growing urban water demands to meet the needs of rising urban populations can be met with only a small increase in price to agriculture. That small increase in water price facing agriculture is enough to bring forth sufficient amounts of water transferred from agriculture to provide adequate supply for those growing urban demands.

Outcomes associated with this large difference in price elasticity and large difference in treatment cost are shown in Table 1 for the case when urban population grows from its base level (107,000 households) to a 10% higher level (117,700 households). This growth is accommodated by a small increase in the shadow price of headwater flows, increasing from $US520$ to $US521$ per acre foot of headwater supply. This means that the economic value of an additional unit of headwater supply increases as population grows. That higher value is a signal of the increased economic value of transferring water from irrigated agriculture to the growing city. Total urban withdrawals increase by 6000 acre feet per year from 58,000 acre feet to 64,000 acre feet per year to support the growing population. Agricultural water demands fall by the same 6000 acre feet per year from 142,000 to 136,000, which maintains the required hydrologic balance. Agricultural water use falls from 3.2 to 3.0 acre feet per acre in the face of the increase in water’s shadow price.

Table 1 is designed to show the effects of a large number of changes in critical parameters shown at the row heads in which each parameter could be increased or decreased by 10%. This table summarizes the effect of changes in each of the parameters on several hydrologic and economic variables shown at the column heads. Nevertheless, while the table is compact and comprehensive, it lacks power to express impacts of a single parameter changing over a very wide range by more than 10% up or down. To rectify this limitation, Figs. 4–6 show detailed impacts of large changes on important parameters affecting the allocation of water in most river basins, namely the absolute scale of the urban population. Other parameters could have also been varied, but because of limited journal space, this analysis focuses on variations in urban population alone. To illustrate the importance of urban population on the allocation of water between agriculture and cities, the basin’s urban population is varied over a wide range. Population is varied from the considerable decline to just 50% of original base population (down from 107,000 to 53,500) to an increase to 400% of that same population (up from 107,000 to 428,000).

Fig. 4 shows impacts of that population change on total water use by sector. Total urban water use increases nearly linearly with a growing population, since per household use is only cut back slightly because of slight price increases (not shown). Agricultural use falls from about 165,000 acre feet per year when urban population is 53,500 to zero at an urban population of 428,000. With no water use in agriculture, further increases in urban population will call forth considerably greater price increases if there are no alternative water supplies, as water is allocated from discretionary urban uses to higher-valued required uses such as drinking, cooking, washing, and flushing.

Fig. 5 shows the results of the identical urban population growth assumptions as presented in Fig. 4, but Fig. 5 also shows the results from the view of per capita and per acre use in the two sectors. For the reasons described above, urban population growth has little effect on water use per household because of the very low price elasticity of demand for urban uses. However even small increases in water’s shadow price facing irrigated agriculture has considerable impact on reduced water use per acre in agriculture. As cities grow, farmers with established water rights find it financially attractive to sell or rent water or otherwise transfer water rights to cities. In addition, the normal market water transfer process can expect to see farmers investing in water conservation measures such as alternatives to conventional flood irrigation, including water saving irrigation technology, reduced acreage, a changed crop mix, or fallowed fields. Growing water demands by growing cities will result in greater on-farm financial incentive to invest in water conserving measures, showing in agriculture a general substitution of land, capital, and labor for water. The higher shadow price resulting from urban population growth results in economic benefits gained by urban areas that are larger than benefits lost from reduced use of water in irrigated agriculture.

Similarly a reduction in urban population would reduce the value of water for urban use and would signal a call for transferring some urban water back into agricultural use.
Fig. 6 shows impacts of the same urban population scenarios on total net economic benefits by sector. As expected, total urban benefits show very large increases with population growth. But, because of a high water use per dollar of farm income, comparatively small amounts of water can be transferred from agriculture at small associated losses in net farm income. In fact if farm income includes the income from water transferred to cities, agricultural benefits would be shown increasing greatly rather than falling slightly as in Fig. 6.

Results shown in Table 1 and Figs. 4–6 are consistent with expectations. With regard to possible variations of the parameters shown in Table 1, use by agriculture increases with more acres served while the quantity of water supplied to urban areas increases with increased population. Water supplied to either agriculture or cities falls with an increase in marginal cost of either use.\(^{10}\) Generally because of purification requirements, the marginal cost of bringing urban water supply up to acceptable health and safety standards is considerably higher than marginal costs for supplying irrigated agriculture. Results are based entirely on observable variables unique to a given river basin, and would be adaptable to hydrologic conditions, economic relations, or institutional constraints facing a given basin. For example, increases in food prices facing agriculture increase the total economic benefits of water in agriculture, which justifies on economic grounds increased water allocated to irrigation uses. Population growth in the urban sector increases economic benefits of urban water uses, reduces allocation of water to agriculture in favor of city use, provided the economic losses in agricultural water use are less than the additional economic benefits in the urban water use.

Suppose that a re-negotiated treaty results in more water that must be delivered at the downstream international border, raising required deliveries from 400,000 to 440,000 acre feet per year. That act reduces supplies available to domestic uses. It also reduces both agricultural and urban uses (demand side adjustments). Table 1 shows that the re-negotiated treaty increases the scarcity of water available for domestic beneficial use, indicated by the higher shadow price of water. The shadow price of water increases from the baseline $US20 to an increased $US30 per acre foot. The higher shadow price associated with the re-negotiated treaty signals a $US10 per acre foot economic value of measures designed to find alternative sources of supply for domestic uses (supply side adjustments).

For example, under baseline water supply conditions, each acre foot of headwater supply is worth $US20 per acre foot (one acre foot is 36,000 cubic meters). That economic value informs both economic and policy choices. It says that policy measures taken to secure more water at the top of the basin, such as new water developments, expanded reservoir capacity, reduced evaporation, greater deliveries negotiated from upstream countries, reduced snowmelt losses that previously failed to reach the river, and the like are worth $US20 per acre foot salvaged. That value per acre foot salvaged can be compared with the cost of salvaging that water. That comparison informs the decision on whether or not it is good economics to salvage the water. Similarly, the negative shadow price associated with downstream delivery requirements means that each extra acre foot exported from the basin costs $US20 in economic benefits displaced under baseline conditions. The $US20 can be compared with alternative measures for delivering that downstream water delivery obligation, such as additional groundwater pumping to meet the increased delivery obligations.

5. Conclusions

Growing water scarcity and increasing competition among water uses, locations, and time periods point to the need for more comprehensive analyses that can identify economic efficiency gains and other impacts of policy proposals. This paper responds to previous gaps in the literature in which little work has been done that illustrates simply and clearly how costs and benefits can be incorporated into an integrated physical, institutional and economic model to support water resources planning and policy. For this reason, its contribution is to describe and illustrate a method by which costs and benefits can be systematically integrated into a physical, institutional and economic analysis for a river basin. In meeting this aim, a prototype integrated economic-hydrologic river basin model is developed and described. Its size is small enough to build, solve, understand, and interpret with no sophisticated computer analysis required. However, its structure is flexible enough to permit expansion for comprehensive policy design that rests on a foundation of a basin's hydrology, institutions, and economic relations.

Integrated basin scale models offer a measure to efficiently and consistently analyze a basin’s hydrology, institutions, and economics. For this kind of analysis to be used by policymakers to inform emerging complex policy questions, integrated analysis requires accounting for additional water uses, reservoirs, return flows, groundwater management, minimum stream flows for water quality and natural habitats. It also requires application of computer techniques and software, by which increasingly complex interdependencies at the watershed level can be accurately represented. This integration has considerable potential to support basin scale environmental economic policy assessments that inform the design of objective and informed environmental policy.

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Appendix: Prototype hydroeconomic model

Parameters

\[ X_{D_{t}} \quad \text{Headwater runoff, year} \ t, \text{ set to 600,000 ac-ft/year (Table 1).} \]

This requires measurement of historical headwater runoff flows in the basin of interest. For analysis of future policy choices, it also requires forecasting those flows into future years.

\[ X_{D_{a}} \quad \text{Delivery requirement at gauge 3, period} \ g, \text{ set to 400,000 ac-ft/year (Table 1).} \]

This term can be derived from international treaties or transboundary agreements in which one region or country has agreed to delivery water to a downstream community. It could also come from a transboundary water agreement under debate. Results from basin a scale model could be used to inform transboundary water allocation debates as treaties are being negotiated.

\[ B_{a}, B_{u}, B_{u} \quad \text{Intercept, linear, and quadratic term for a-th use's benefit function, agricultural and urban.} \]

These 3 terms reflect the typical character of such a benefits function, showing benefits that increase at a decreasing rate, reaching a maximum, then possibly being reduced at very high use levels. Requires data on the incomes earned in irrigated agriculture, typically comes from information on prices, costs of production, crop yields, and water use rates per unit land. Price elasticity set at -0.79 for

\[ 10 \quad \text{The marginal cost of supplying urban water to acceptable health and safety standards is typically much higher than marginal costs for supplying water to irrigated agriculture. See the marginal cost parameter in Table 1. The fact that bottled water sells for what amounts to several million US dollars per acre foot indicates the high willingness to pay for tasty, healthy, safe, treated drinkable water.} \]
agriculture and –0.04 for urban uses (Table 1). Details on estimation are in Section 3.6.

\( \gamma_u \) Cost per unit water supplied to the u-th use, agriculture and urban, set to $US1 0 per ac-ft for agriculture and $US4 00 per ac-ft for urban (Table 1). Requires data on the cost of securing deliveries to irrigated agriculture. Where infrastructure like dams, canals, and aqueducts are built these capital costs can be quite high. Cost should include the opportunity cost of water displaced from other uses. For urban uses, data are required on the cost of developing and delivering safe and healthy water to urban users. Costs of building infrastructure to secure, convey, and purify water are typically high. Marginal cost can be low once infrastructure is in place.

\( \delta_u \) A scale parameter for the u-th use. For agriculture, the scale is total amount of land served, set to 45,000 acres for this analysis (Table 1). This requires measuring total irrigated land available to which water can be put to beneficial use. For urban use, the scale is the total number of urban households served, set to 107,000 (Table 1). Requires measuring the number of households served.

Unknown variables (determined by the model)

**Hydrologic**

- \( X_{h, t} \): Headwater runoff, pd t
- \( X_{da, t} \): Total u-th use, ag and urban, pd t
- \( Y_{da, t} \): Per unit u-th use, per ac or per household, pd t
- \( X_{g, t} \): Gauge no. 1 streamflow, pd t
- \( X_{g2, t} \): Gauge no. 2 streamflow, pd t
- \( X_{g3, t} \): Gauge no. 3 streamflow, pd t

**Economic**

- \( \text{Benefit}_{u, t} \): Per acre or per household benefits, pd t
- \( \text{T_Benefit}_{u, t} \): Total benefits, agriculture or urban, pd t
- \( \text{Cost}_{u, t} \): Per unit costs of supply, per ac or per household, pd t
- \( \text{T_Cost}_{u, t} \): Total costs of u-th use, agriculture or urban, pd t
- \( \text{PVNB} \): Discounted net benefits over uses and periods

**Equations**

**Hydrologic**

(1) \( X_{h, t} = X_{h0, t} \): Headwater flows
(2) \( X_{g1, t} = X_{g1, t} - X_{da, t} \): No. 1 gauge flows, pd t
(3) \( X_{g2, t} = X_{g2, t} - X_{da, t} \): No. 2 gauge flows, pd t
(4) \( X_{g3, t} = X_{g3, t} - X_{da, t} \): No. 3 gauge flows, pd t
(5) \( Y_{da, t} = X_{da, t} / \delta_u \): Ag use per acre, pd t
(6) \( Y_{da, t} = X_{da, t} / \delta_u \): Urban use per household, pd t

**Institutional**

(7) \( X_{g3, t} > X_{g3, t}^0 \): Delivery requirement, pd t

**Economic benefit**

(8) \( \text{Benefit}_{u, t} = b_0 + b_1 Y_{da, t} + b_2 Y_{da, t}^2 \): Ag benefits per acre, pd t
(9) \( \text{Benefit}_{u, t} = b_0 + b_1 Y_{da, t} + b_2 Y_{da, t}^2 \): Urban benefits per household, pd t
(10) \( \text{T_Benefit}_{u, t} = \delta_u \text{Benefit}_{u, t} \): Total ag benefits, pd t
(11) \( \text{T_Benefit}_{u, t} = \delta_u \text{Benefit}_{u, t} \): Total urban benefits, pd t

**Economic cost**

(12) \( \text{Cost}_{u, t} = \gamma_u Y_{da, t} \): Ag water cost per acre, pd t
(13) \( \text{Cost}_{u, t} = \gamma_u Y_{da, t} \): Urban water cost per household, pd t
(14) \( \text{T_Cost}_{u, t} = \delta_u \text{Cost}_{u, t} \): Total Ag water cost, pd t
(15) \( \text{T_Cost}_{u, t} = \delta_u \text{Cost}_{u, t} \): Total urban water cost, pd t

**Economic discounted net benefits**

(16) \( \text{PVNB} = \sum_{t} \text{T_Benefit}_{u, t} - \text{T_Cost}_{u, t} (1+r)^t \)

**Langragian expression for constrained maximization**

(17) \( \max \phi = \text{PVNB} + \sum_{t} \left[ \text{Benefit}_{u, t} - \text{Cost}_{u, t} (1+r)^t \right] \)

**References**


