ECOLOGICAL ECONOMICS 59 (2006) 38–47

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ANALYSIS

Linking ecological footprints with ecosystem valuation in the provisioning of urban freshwater

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ARTICLE INFO

Article history:
Received 22 May 2005
Received in revised form 19 September 2005
Accepted 26 September 2005
Available online 16 November 2005

Keywords:
Ecological footprint
Urbanization
Ecosystem service
Valuation
Sustainability
Freshwater

ABSTRACT

Two prominent and alternate approaches, ecosystem service valuation and ecological footprints, link the production of ecosystem services with their consumption by societies. An overlapping goal of both approaches is to promote the sustainable use of ecosystem services such that their production rates are not compromised. Yet, little integration of these perspectives and their emphasis on distinct units, dollars and area, has been attempted. We combined these two approaches to better understand variation in the societal demand and production of freshwater, a critical ecosystem service, for 121 cities in the United States. The analysis linked previously compiled data on urban water use and the spatial distribution of run-off water throughout the conterminous United States. Incorporating the spatial distribution of water consumption and production, we computed heterogeneous urban water use footprints for all 121 cities. From the relationship between annual municipal utility expense and footprint area, the median monetary value for water footprint area was \$88,808 km^{-2} yr^{-1} from all the cities we considered. The ratio between the footprint-estimated cost and the utility-observed cost was negatively related to the local availability of water, and was independent of population size. By linking ecosystem service valuation and ecological footprint analyses into a coherent framework, we developed an integrated metric for understanding the provisioning of ecosystem services, which could help inform sustainable pricing guidelines for renewable freshwater.

1. Introduction

Human activity is connected to ecological processes through the provisioning of freshwater (Jansson et al., 1999; Falkenmark and Folke, 2003). Civilizations have long modified hydrologic cycles with dams and water diversion projects to provide freshwater for domestic, commercial, industrial, and agricultural uses (Redman, 1999; Vorosmarty and Sahagian, 2000; Meybeck, 2004). Provisioning freshwater supplies to meet societal needs includes both large-scale redistribution associated with irrigated agriculture, and more concentrated uses associated with urbanization. Water redistribution can

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doi:10.1016/j.ecolecon.2005.09.023

require extensive areas of land and is economically costly. While ecologists have had much interest in understanding water issues in the context of irrigation (Kendall and Pimentel, 1994; Matson et al., 1997; Thomson et al., 2005), much less attention has been paid to the equally challenging task of understanding municipal water supplies, particularly in the current era of rapid ecological and social change (Brookshire et al., 2002).

The growth of cities throughout the world presents new challenges for distributing water to meet societal needs, which are vulnerable to both climate and population changes (Naiman and Turner, 2000; Vorosmarty et al., 2000; Jackson et al., 2001; Fitzhugh and Richter, 2004). Half the world's population now resides in cities and the majority of projected population growth is expected to occur in cities (United Nations Centre for Human Settlements, 2001; Cohen, 2003; McGranahan and Satterthwaite, 2003). Urban water withdrawal often extends far beyond city boundaries, as municipal utilities seek to secure new water sources from their hinterland. Securing freshwater is a growing concern of many cities across the U.S., even in regions historically replete with water. For example, by 1930 Los Angeles, California had already outgrown its local water supply and was tapping the outlying San Fernando Valley (Kahrl, 1982). The area of water withdrawal for Los Angeles has grown remarkably since then, impacting the water balance of a region far beyond its border and consequently affecting urban and agricultural patterns throughout the western U.S. Currently, the Colorado River provides about 65% of the water used in Southern California (Colorado River Water Users Associated http:www.crwua.org/colorado_river.html). Scarce fresh water supplies are a global phenomenon, and local municipalities must grapple with regionally specific constraints for securing adequate water resources (Zehnder et al., 2003).

The production of renewable fresh water is an ecosystem service, i.e. an ecological process required for societal functioning. Identifying ecosystem services can help appropriately reduce the complexity of an ecosystem to those components and fluxes that are directly and indirectly related to societal sustainability (Costanza et al., 1997; Daily et al., 1997; Palmer et al., 2005). To understand the relationships between the consumption and production of ecosystem services two prominent approaches, ecosystem service valuation (Costanza et al., 1997) and ecological footprints (Wackernagel and Rees, 1996), have been developed. While these two approaches are generally used independently (although see Sutton and Costanza, 2002; Sutton, 2003), they both provide information relevant for the sustainable acquisition of ecosystem services.

Ecosystem service valuation assigns a monetary value to an ecosystem based on the supply of market and non-market ecosystem services by identifying the total cost of the service using a readily comparable metric (Costanza et al., 1997; Wilson and Carpenter, 1999; Farber et al., 2002). Valuation has been suggested as an approach that can aid decision making to promote sustainable human–ecological interactions (Costanza, 2000; Farber et al., 2002). Globally, the monetary value of the biosphere has been estimated at an annual US$33 trillion (Costanza et al., 1997). One prominent challenge with valuing ecosystem services is quantifying goods that have a disproportionately low market value compared to their contribution to societal functioning (e.g., clean air). A second challenge concerns both the spatial and temporal variability in the relationship between the supply and the demand for ecosystem services (Limburg et al., 2002; Turner et al., 2003). Locations and times of scarcity in the supply of ecosystem services should increase the economic value of these services (Sutton and Costanza, 2002), although the effects of spatial heterogeneity may be difficult to predict (Costanza et al., 1997).

In contrast to ecosystem valuation, ecological footprints quantify human–ecosystem relationships by estimating the land area required to sustainably supply consumed ecosystem services (Wackernagel and Rees, 1996; Luck et al., 2001; Wackernagel et al., 2002; York et al., 2003). Ecological footprints explicitly recognize the interaction between production and consumption on the intensity of human impact (Erb, 2004; Jenerette et al., in press). Footprint calculations can be directly related to human consumption patterns and therefore identify benchmarks of sustainable activities. A recent estimate suggests that human global resource consumption in 2001 would require 13 billion hectares to maintain a sustainable production of ecosystem services, approximately 21% more land area than available on earth (Loh and Wackernagel, 2004). However, many questions have been raised regarding the accuracy of ecological footprint calculations (van den Bergh and Verbruggen, 1999). Criticisms can be grouped into three classes of problems: (1) production of multiple services by the same ecosystem, (2) trade of services between regions, and (3) spatial heterogeneity in both the consumption and supply of ecosystem services (van den Bergh and Verbruggen, 1999; Opschoor, 2000; Templet, 2000; Andersson and Lindroth, 2001). A newly developed method, the spatially heterogeneous ecological footprint (H-EF), address these shortcomings (Luck et al., 2001; Jenerette et al., in press). This modified footprint approach computes the area required to supply an individual ecosystem service, while incorporating spatial heterogeneity in both the demand and the supply of an ecosystem service. The sustainable supply of water seems particularly amenable to this approach as it is a single ecosystem service, generally not traded between regions, and data describing the spatial heterogeneity in rates of service production are readily available (Luck et al., 2001; Jenerette et al., in press).

Our objective was to combine ecosystem valuation and ecological footprint approaches to better understand variation in the societal consumption of freshwater within the conterminous United States. Using a data set of 121 U.S. cities, the market cost associated with supplying urban water was linked to the area required for the production of freshwater. The price individual consumers pay for water is complicated by a variety of rate structures, initial hook-up charges, and other irregularly assessed fees (Baumann et al., 1998). To simplify these charges, we used a surrogate measure of consumer cost—the revenue generated by a municipality per volume of water from supplying water to residential, commercial, and industrial users. As an alternate measure of the economic value of supplying urban fresh water, we also used the cost required by the utility to supply urban water demand, i.e., the municipal expense associated with providing water to consumers, as an estimate of the cost of supplying urban freshwater. While municipal revenue and expense are related, they are distinct estimates of the price paid for supplying urban water needs. Both measures of water costs are strictly market costs, and thus do not provide a full accounting of the value of water supplies. Nonetheless, the impact of these costs to individual consumers and municipalities is substantial, and the spatial variation of these costs can be quantified.

In conducting this research, we asked four specific questions. First, what are the determinants of urban per-capita water demand? To address this question, we tested four hypothetical sources of variation: (1) the availability of local water, (2) cost to the consumer (calculated as municipal revenue per-capita), (3) population size of the service area, and (4) conservation measures enacted by a municipality. Second, what determines the cost associated with provisioning urban water? We hypothesized the cost to supply water (i.e. municipal expense) would be coupled to the total amount of water delivered, and because water demand in part determines the area required to supply water (i.e., the footprint area), we also hypothesized municipal expense would directly relate to the water footprint size. Using the relationship between municipal expense and footprint area, we combined this estimate of observed ecosystem service valuation and ecological footprint to compute the cost/km^2 for footprint area. Third, what controls variation between the footprint-estimated cost, the price that reflects the sustainability of water use and utility expenditures, and the actual monetary costs to the utility to supply the water? We hypothesized footprint-derived costs would be greater than actual costs when local water availability of water is low, and actual costs would be less than ecological costs when population sizes are high. The final question addressed the effects of unintended water loss, an important contributor to urban water use: what factors affect the rate of urban water loss, and to what degree does this loss affect footprint area? We evaluated the effect of three alternative hypothesized factors associated with water loss: (1) local water availability, (2) total income to the utility (i.e., a surrogate measure of consumer costs), and (3) the total population served. We then calculated the footprint area associated with unintended water losses and computed the cost of this area based on our determination of cost/km^2. The systematic examination of a series of four connected questions using a hypothesis-driven framework allowed a better understanding of the variation in urban water use, the market costs associated with provisioning urban water, and the land area required to sustainably supply this ecosystem service.

2. Methods

Urban water use characteristics were obtained from a water utility operator-completed survey conducted in 2001 by Raftelis Financial Consulting Company (Raftelis Financial Consulting PA, 2002). Utility sizes spanned a broad spectrum, with service populations ranging from 13,000 (Oxford, MS) to 5,000,000 (Chicago, IL) residents. By having a large number of utilities segregated by service population and geography, potential biases in the Raftelis database due to non-random sampling were reduced. The survey responses included: total water deliveries, total utility revenues and expenditures, population size served, total water leakages in the system, and types of conservation measures enacted by the municipality (e.g., demand management, education, and plumbing retrofitting). Because the price paid by the consumer is often bundled with additional costs, and may be offset by one-time hookup charges, we used the expense paid by the utility per liter of water (total expenditures in $US/total water deliveries in liters) as our estimate of the cost of water use. From the Raftelis dataset, we extracted the cities from the U.S. for which all data fields were completed, resulting in a sample size of 121 cities and linked the urban characteristics to the location of each city.

In conjunction with data on municipal water-use characteristics, we obtained data generated by Fekete et al. (2000) on the spatial distribution of renewable water annual surface run-off, defined as the difference between precipitation and evapotranspiration. Their estimate of renewable water resources was generated through a combination of observed discharge from rivers and a climate-driven water balance model, which linked precipitation, evapotranspiration, and run-off. These data have been applied to describe global water resources available for human uses (Vorosmarty et al., 2000). From their global dataset we extracted renewable water run-off rates for the conterminous U.S. boundary. These data were then projected to an Albers equal-area projection using the same parameters as the city location database and interpolated to a 10 km grid cell size. From the overlay of water availability and city location, we identified the local run-off available for each city, and added this information to the city database.

Using the H-EF, we combined the distribution of urban water demand with the distribution of available water resources. The H-EF computes the area required to supply the water demand of a city based on the regional pattern of water run-off and its demand (Luck et al., 2001; Jenerette et al., in press). The footprint area was computed by iteratively appropriating area to each city the available water from the closest source that was not already appropriated. The footprints generated are expanding circles surrounding each city. While this technique does not identify the true locations where cities obtain their water, it does incorporate the spatial heterogeneity in ecosystem service production and demand (Luck et al., 2001). For the same level of water demand, the footprint is larger in arid regions and smaller in humid regions. All geographic analyses were conducted within a geographic information system that included commercial software (ArcInfo, ESRI) and algorithms specifically developed for this project in C++.

To analyze the patterns within these data, we used linear regression. Prior to analysis, we evaluated each variable for normality using the Kolmorgorov–Smirnov one sample test ($P<0.01$), and if necessary performed a log-transformation (Masley, 1951). To examine possible effects resulting from intercorrelation among the alternative hypothesized variables, we computed a Pearson product-moment correlation matrix. Preliminary correlation analysis suggested intercorrelation was minimal ($p>0.05$), and therefore we considered our variables to be independent.

3. Results

The urban water footprints, representing the interaction between municipal demand and water availability, exhibited a large degree of spatial variation throughout the United States (Fig. 1 and Table 1). From all cities, the mean population served was 483,868 people with a resulting mean water footprint of 611 km^2. The coefficient of variation for city population and footprint was 158% and 188%, respectively (Table 2). Correlation analysis between our selected independent variables exhibited some expected relationships (Table 3). Population, revenue, and expenses were all positively related, whereas local water availability and unintended water loss did not correlate with any of the examined variables.

Per-capita total water consumption was negatively related to local water availability (F=14.49; P<0.01; Fig. 2A). Similarly, per-capita total water consumption was negatively related to the surrogate measure of consumer cost, the revenue to the utility, calculated as utility income/water supplied (F=28.70; P<0.01; Fig. 2B). In contrast, per-capita total water use was not related to population size served (F=0.89; P=0.35; Fig. 2C). There was also no relationship detected between per-capita water use and the number of conservation measures enacted by a municipality (F=3.09; P=0.08; Fig. 2D). Annual expense to the water utility per liter of water supplied, an estimate of the total cost associated with supplying the water, was tightly coupled with the total volume of water delivered (F=519.57; P<0.01; Fig. 3A). Water footprint size was also positively correlated with the annual utility water expense (F=57.57; P<0.01; Fig. 3B).

Based on the relationship between annual utility expense and footprint area, we computed the median monetary value for water footprint area to be \$88,808 km^{-2} yr^{-1} for all cities in our data set. Arlington, TX had the highest estimated cost of footprint area at \$629,670 km^{-2} yr^{-1}, and Bismark, ND had the lowest cost at \$4121 km^{-2} yr^{-1}. The ratio of the footprint-estimated cost with the actual cost (i.e. utility expense), a unitless ratio between dollar costs, was negatively related to local water availability (F=408.17; P<0.01; Fig. 4A), but was not related to total population size (F=1.15; P=0.29; Fig. 4B).

Unintended water losses were a considerable component of urban water footprint area. Unintended water losses varied widely between municipalities and in some cases were a substantial percentage of urban water uses (mean water loss was 12±7.13%). However, this loss was not related to local water availability (F=3.382; P=0.07), total population size (F=0.02; P=0.88), or utility income (F=0.47; P=0.49) (Fig. 5A–C). The effect of water loss on footprint area corresponded with the magnitude of lost water; the mean percent change in footprint area due to water loss was 11.2±18.37% (Fig. 5D).

4. Discussion

4.1. *Variations in per-capita water use*

We examined four hypothesized determinants of per-capita water use: (1) local availability, (2) cost to the consumer, (3) population size, and (4) number of conservation measures enacted. Cities in more humid regions tend to use less water at a per-capita level than those in more arid regions (Fig. 2A). Per-capita water use is an important metric for identifying the individual contribution to urban water consumption. As our surrogate measure of consumer cost, the revenue to the utility, increases, the amount of water used per-capita declines (Fig. 2B). The size of a population, however, was not related to the mean per-capita water use, although small cities were more variable in per-capita use than larger cities (Fig. 2C).

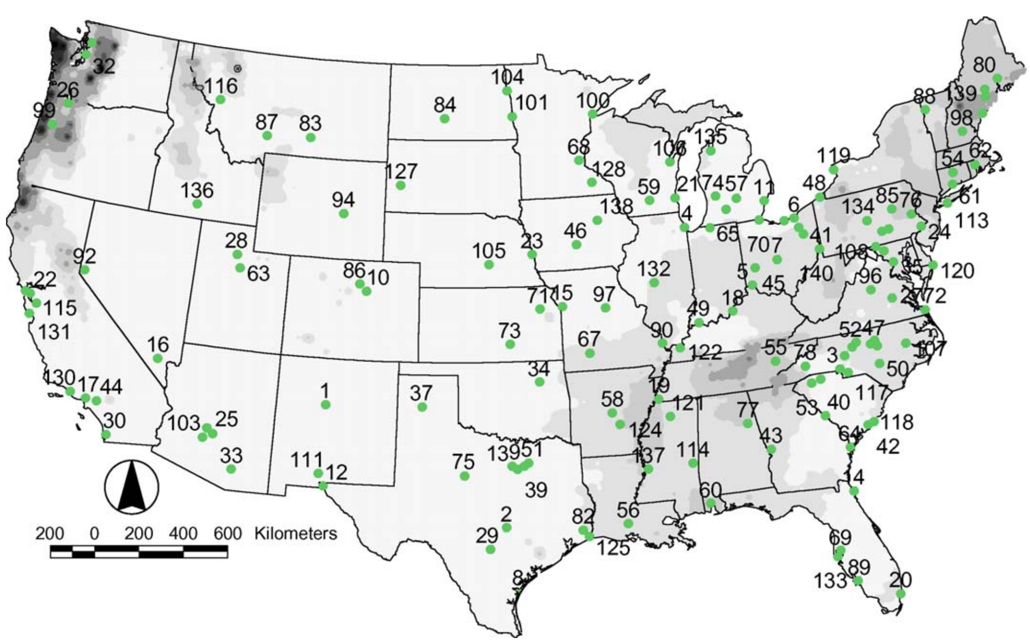

Fig. 1 – Map of the conterminous U.S. background shading indicates increasing local runoff (darker colors). Locations of the cities used in the analyses are indicated with dots. See Table 1 for a list of the cities and their corresponding ID codes.

Table 1 – List of cities used in the analysis by population size and their H-EFs

City ID	City	Pop. (1000)	H-EF (km^2)	City ID	City	Pop. (1000s)	H-EF (km^2)
121	Oxford, MS	13	25	43	Columbus, GA	200	125
91	Carlisle, PA	20	25	48	Erie, PA	200	100
139	Waterville, ME	22	25	51	Garland, TX	206	300
79	Augusta, ME	25	25	67	Springfield, MO	216	100
135	Traverse City, MI	27	25	59	Madison, WI	218	175
136	Twin Falls, ID	35	100	39	Arlington, VA	238	375
129	Salisbury, NC	35	25	57	Lansing, MI	250	200
98	Concord, NH	40	25	36	Akron, OH	276	125
113	Massapequa, NY	43	25	44	Covina, CA	297	350
96	Charlottesville, VA	45	25	60	Mobile, AL	300	125
140	Wheeling, WV	48	25	28	Salt Lake City, UT	315	750
116	Missoula, MT	50	125	38	Arlington, TX	317	100
99	Corvallis, OR	52	25	47	Durham, NC	318	125
94	Casper, WY	53	800	58	Little Rock, AR	368	200
92	Carson City, NV	53	125	46	Des Moines, IA	370	275
80	Bangor, ME	53	25	55	Knoxville, TN	382	75
119	Niagara Falls, NY	54	75	61	New Haven, CT	384	125
104	Grand Forks, ND	55	150	73	Wichita, KS	390	725
124	Pine Bluff, AR	55	25	54	Hartford, CT	400	100
125	Port Arthur, TX	59	650	42	Charleston, SC	401	250
127	Rapid City, SD	59	75	72	Virginia Beach, VA	401	125
84	Bismarck, ND	60	650	68	St. Paul, MN	415	325
118	Mount Pleasant, SC	60	50	45	Dayton, OH	422	225
102	Frederick, MD	60	25	70	Toledo, OH	437	425
134	State College, PA	62	25	69	Tampa, FL	450	375
115	Milpitas, CA	63	150	1	Albuquerque, NM	480	1350
122	Paducah, KY	64	50	8	Corpus Christi, TX	500	5250
107	Greenville, NC	69	50	35	Washington, DC	519	300
138	Waterloo, IA	70	75	27	Richmond, VA	529	325
95	Chapel Hill, NC	70	50	23	Omaha, NE	560	1075
112	Lorain, OH	72	50	15	Kansas City, MO	600	525
97	Columbia, MO	81	75	62	Providence, RI	632	125
111	Las Cruces, NM	85	725	34	Tulsa, OK	650	650
128	Rochester, MN	88	75	33	Tucson, AZ	675	1375
131	Santa Cruz, CA	90	100	2	Austin, TX	691	475
101	Fargo, ND	91	225	3	Charlotte, NC	700	475
83	Billings, MT	94	750	12	El Paso, TX	725	1450
100	Duluth, MN	97	100	18	Louisville, KY	780	375
110	High Point, NC	98	50	26	Portland, OR	780	175
106	Green Bay, WI	102	100	21	Milwaukee, WI	833	600
130	San Buenaventura, CA	104	100	14	Jacksonville, FL	840	1175
123	Peoria, AZ	105	100	13	Fort Worth, TX	870	2175
93	Cary, NC	105	50	5	Cincinnati, OH	900	400
63	Provo, UT	113	250	19	Memphis, TN	900	325
78	Asheville, NC	116	75	16	Las Vegas, NV	948	5875
86	Boulder, CO	120	200	10	Denver, CO	1000	1300
65	South Bend, IN	122	125	7	Columbus, OH	1014	375
76	Allentown, PA	123	50	29	San Antonio, TX	1100	1225
66	Spartanburg, SC	125	250	22	Oakland, CA	1200	1900
50	Fayetteville, NC	130	100	32	Seattle, WA	1316	250
75	Abilene, TX	132	775	30	San Diego, CA	1357	3625
56	Lafayette, LA	135	75	6	Cleveland, OH	1500	675
41	Canton, OH	150	100	25	Phoenix, AZ	1631	6000
49	Evansville, IN	155	200	24	Philadelphia, PA	1650	450
132	Springfield, IL	159	75	9	Dallas, TX	1979	3250
71	Topeka, KS	160	125	20	Miami, FL	2047	1150
85	Bloomsburg, PA	171	50	31	San Francisco, CA	2412	3175
40	Augusta, GA	180	275	17	Los Angeles, CA	3829	2200
64	Savannah, GA	181	175	11	Detroit, MI	3886	4300
126	Portland, ME	186	50	4	Chicago, IL	5000	4275
74	Wyoming, MI	200	175				

City ID codes are used in Fig. 1.

Table 2 – Mean and standard deviation for data used in this analysis

Variable	Mean (SD)
Population served (pop)	483,868 (763,981)
Utility revenue ($ yr^{-1})	51,073,279 (7,464,785)
Utility expense ($ yr^{-1})	36,352,702 (55,065,343)
Local water production (Million L km^{-2} yr^{-1})	19.02 (16.12)
Loss (%)	11.96 (7.13)
Water footprint (km^2)	682.77 (1239.89)
Total water used (million L yr^{-1})	113,528 (189,562)

The lack of a relationship between population size and per-capita water use is evidence against increased urban efficiency or economies of scale associated with resource use; small cities had similar per-capita water usage as large cities. The observed water-use increases in more arid regions may result from increased outdoor uses; however, our data did not separate water into outdoor and indoor uses and thus we cannot test this hypothesis directly. Finally, whereas entire municipality per-capita water use captures the total urban water use, it may obscure some individual practices, as a proportion of water consumed is not for residential uses.

In contrast to the hypothesis that the enactment of water conservation programs would result in reduced per-capita water use, we observed no relationship between the number of water conservation methods enacted and per-capita water use (Fig. 2D). This analysis does not evaluate the efficacy of individual conservation programs; however, it suggests alternative conservation programs are non-additive, and the cost of water appears to be the most important factor regulating water use (Fig. 2B). Previous studies examining the effect of non-price based conservation strategies on reducing water demand have also shown mixed results (Nieswiadomy, 1992; Michelsen et al., 1999). Our analysis suggests the implementation of many different conservation strategies may not be more effective than concentrated focus on particularly effective strategies. However, per-capita spending on conservation methods maybe a better estimate of municipal effort to promote conservation, and may relate significantly to per-capita water use.

4.2. *Variations in the cost of water supply and water footprints*

We hypothesized that the total utility's expenses supplying water would be tightly related to the total water delivered. Not surprisingly, the results supported this hypothesis (Fig. 3A). Furthermore, the hypothesized relationship between expense and the land area required to provision the supplied water demand was also supported (Fig. 3B). As expected, the utility's expenditures, i.e., cost to supply water, and the amount of water delivered were positively related (Fig. 3A). Because footprint area is in part determined by the water demand, we also observed a consistent relationship between footprint area and utility expenditures (Fig. 3B). From this comparison, we generated an estimate of the cost for each km^2 included in the footprint area. Corroborating the hypothesis that this new metric would decrease with local water availability and increase with the size of population served, we observed a negative relationship between local availability of water to both the ratio of footprint-estimated cost and the utility-observed cost (Fig. 4A), and was independent of population size (Fig. 4B). The pattern of this ratio suggests that where the availability of water was higher, the utility expenditure/area decreased substantially.

The cost per footprint area metric estimates the value of an ecosystem service such that the spatial variation in the supply and demand of the service is explicitly considered. This metric is a potentially useful source of ecological information that can be incorporated into water management and pricing structures. Such use would acknowledge the increased cost associated with providing water to cities in more arid regions. However, as these footprint-derived costs are related only to market values, the costs for sustainable water consumption should also include the non-market and instream services provided by freshwater (Duffield et al., 1992; Costanza et al., 1997). Future research linking the regional patterns of urban and agricultural water demand with its supply would be appropriate next steps in developing water management strategies that account for the diversity of water uses and its often limited supply.

4.3. *Water lost from deliveries*

Water lost during distribution to consumers may have strong effects on water production and consumption patterns. Here we examined three factors that may affect unintended water loss: local water availability, utility total revenue, and size of population served. We observed a weak decline in water loss associated with an increase in water availability. Where there was more water locally available, there was less loss, possibly due to reduced levels of evaporation and an overall positive net water balance. In contrast, the percent water loss was not related to the annual revenue of a utility. Although percent water loss was not related to the size of the population served, the variability of water loss generally decreased in cities with smaller revenues serving smaller populations. The effect of water loss on footprint area resulted in a non-linear increase in footprint area, thus with increasing water losses the human impact to regional ecosystems increases disproportionately.

Table 3 – Correlation matrix between potential explanatory variables

	Population	Revenue (cost to consumer)	Expense (cost to utility)	Local water availability
Revenue (cost to consumer)	0.96*			
Expense (cost to utility)	0.91*	0.96*		
Local water availability	−0.16	−0.18	−0.18	
% Water loss	−0.08	−0.11	−0.06	0.17

* Significant correlations, Bonferonni correction $P<0.05$.

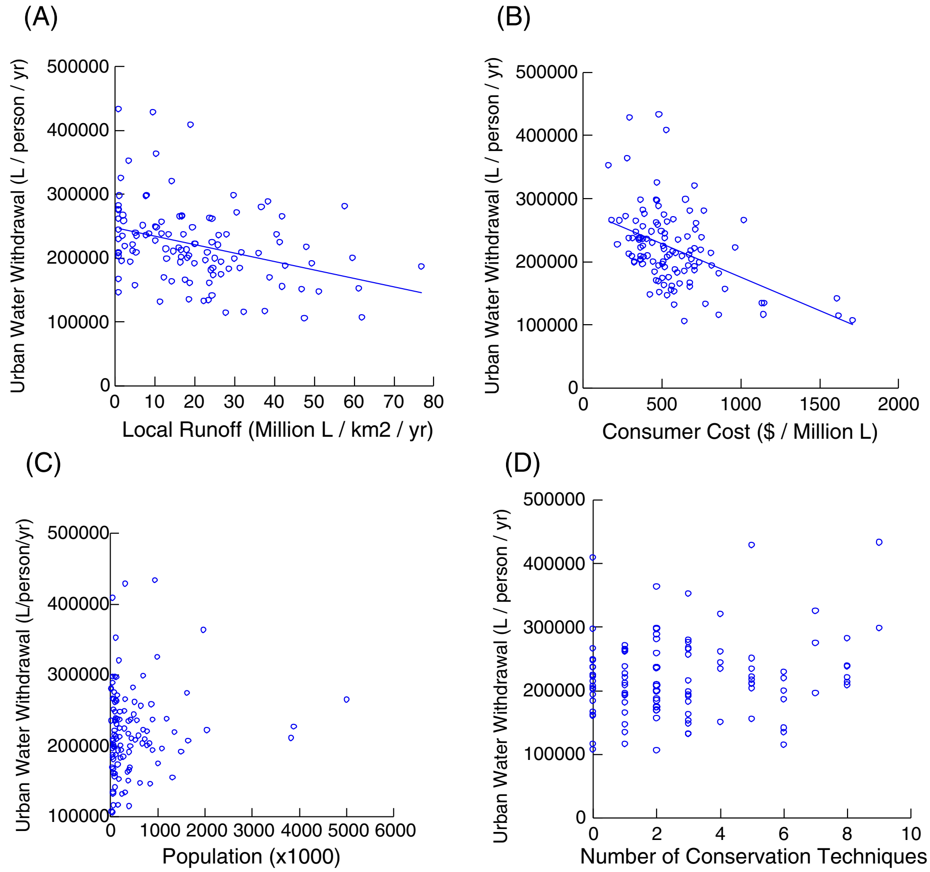

Fig. 2 – (A) Urban per-capita water use (liters/person/year) vs. local water availability (millions of liters/km^2/year) by city. (B) Urban per-capita water use (liters/person/year) vs. cost of water to the consumer (US\$/millions of liters) by city. (C) Urban per-capita water use (liters/person/year) vs. population size (in thousands) by city. (D) Urban per-capita water use (liters/person/year) vs. number of water conservation techniques and programs enacted by city.

4.4. *Synthesis*

This study begins an integration of two prominent approaches for linking human and ecological systems, ecosystem service valuation and ecological footprints. Through this integration, we developed a method for estimating the cost of the area sustainably provisioning an ecosystem service required for a city. By combining estimates of the volume of water consumed and produced with its economic cost, this approach can facilitate the use of hydrologic models to assess the sustainability of water resources management decisions. Future climate scenarios suggest large spatial variation in precipitation responses to climate change (Douville et al., 2002; Giorgi and Bi, 2005); developing tools to manage water resources for many competing uses will become more important with these projected future changes. Furthermore,

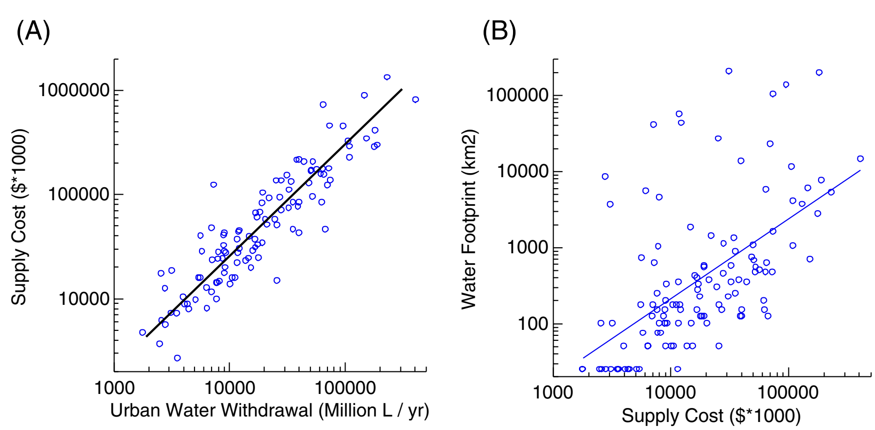

Fig. 3 – Cost to supply water, i.e., expense to utility (US\$), vs. (A) total urban water use (millions of liters/year), and (B) footprint size (km^2).

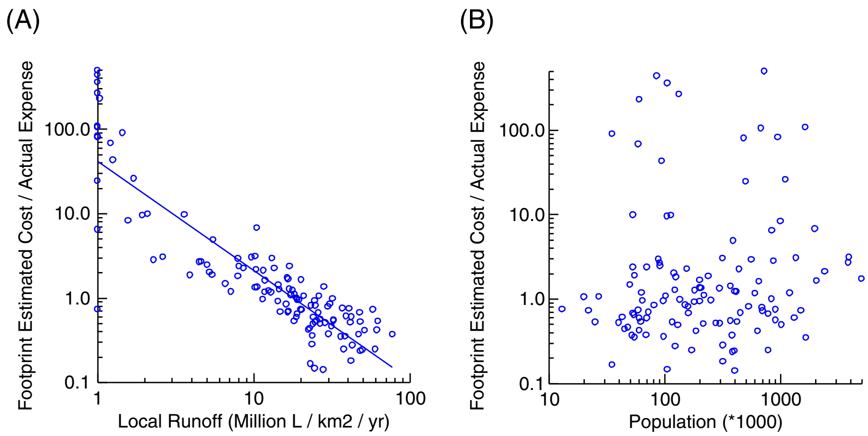

Fig. 4 – (A) Ratio (unitless) of footprint-estimated cost and actual cost to the utility for supplying water to a city vs. local water availability (millions of liters/km^2/year) by city. (B) Ratio (unitless) of footprint-estimated cost and actual cost to the utility for supplying water to a city vs. population size (in thousands) by city.

this study addresses and highlights two of the challenges for both valuation and footprint methods (van den Bergh and Verbruggen, 1999; Limburg et al., 2002; Turner et al., 2003). (1) By relying only on market costs, the valuation scheme described here uses market expenditures as an empirical observation of actual ecosystem service valuation. This is an underestimate of the cost; for example in Montana's Big Hole and Bitteroot rivers, substantial economic value of the water was associated with instream uses for recreation and hydroelectric power production (Duffield et al., 1992). (2) Whereas H-EF footprints improve upon traditional footprint calculations by explicitly incorporating the heterogeneity in ecosystem service supply, there is still imprecision in the delineated water footprints. Future studies that compare H-EF delineated

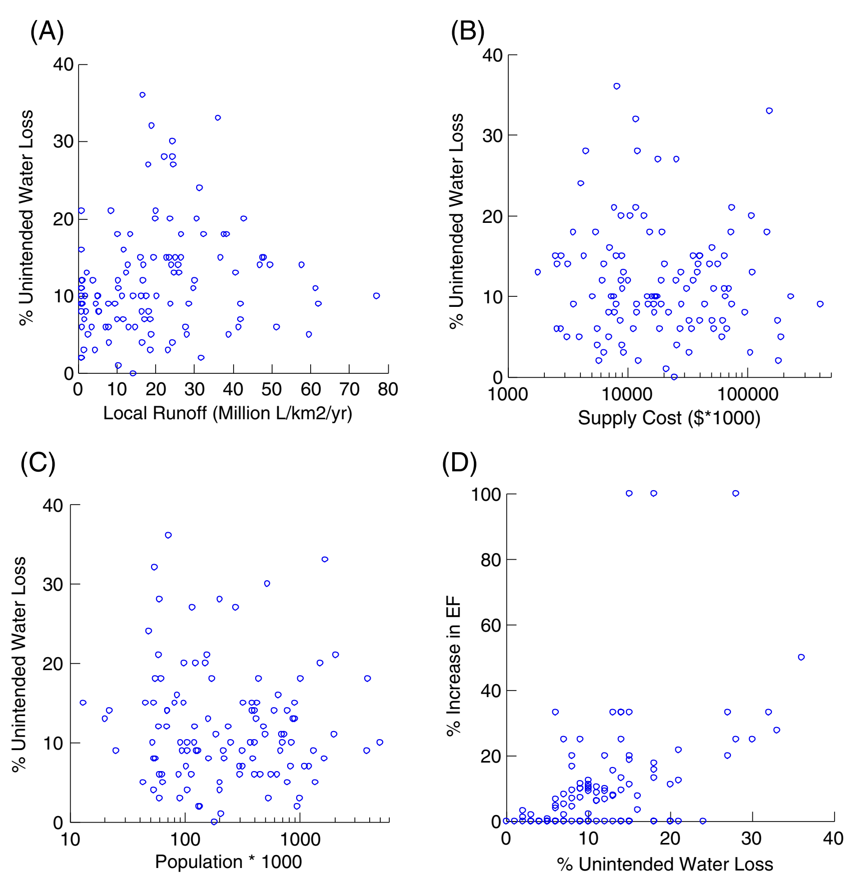

Fig. 5 – Unintended urban water loss. Relationships between unintended water loss and (A) local run-off, (B) supply cost, (C) population served, and (D) footprint area.

water source areas and actual water source areas would improve the H-EF algorithm, and may provide insight into resolving controversies in water allocation between multiple cities, agricultural, and in-stream uses. By computing the valuation of footprint areas an important research need is met — the identification of the spatial variation in ecosystem value such that differences in consumption and production are included (Costanza et al., 1997; van den Bergh and Verbruggen, 1999).

The relationship between urbanization and water appropriation is complex and varies substantially within the United States. We might have expected some cities to be consistent outliers, however, different cities were outliers in almost all of our analyses. Furthermore, some cities we had expected to be outliers, e.g., large, desert cities such as Phoenix, or wet, coastal cities such as Seattle, were surprisingly consistent with national patterns. The overall relationship between footprint area and utility expenditures for all cities in our analysis allowed us to calculate a general cost of footprint area for the supply of water for 121 cities in the U.S. Here we estimate that for cities in the U.S., the cost per km^2 of land included in the water footprint was \$88,808 km^{-2} yr^{-1}. After adjusting prices for inflation to 2004 dollars, our estimate of cost per area for services associated with freshwater production is slightly lower than the value reported by Costanza et al. (1997) for terrestrial ecosystems, and substantially lower than the value of rivers and lakes based solely on water supply. However, our methods differed from Costanza et al. (1997) in that we considered the available water supply to be produced as run-off from terrestrial ecosystems and deposited in lakes and rivers, whereas they considered water to be produced solely within the confines of river and lake ecosystems.

In conducting our study, we were confronted with the complexity of water pricing. The cost of water for the consumer varies substantially in response to many idiosyncratic factors (Baumann et al., 1998). The cost of water often varies by season, with higher costs associated with summer usage compared to winter usage. Water rates are often graded based on amount of water used; in some cases the pricing structure could serve as incentives to increase water use, because higher uses may have discounted prices on a per liter basis. The actual price paid by users may be subsidized by costs charged for initial hook-up to the utility. Adding further complexity, in some cases water prices are bundled with sewage services. In many rental properties, water price is incorporated in the rental fees, and thus, in these cases the residents do not directly experience the costs of water usage. Finally, water supplied for different uses, e.g., agricultural, commercial, industrial, or residential, can have different rates. Our solution to this problem, computing a per-capita cost per liter from the total utility revenue is a reasonable first approximation to the complexity of conducting a national scale analysis. However, much future research remains to understand the linkages between human economic decisions and the consumption of ecosystem services at a national scale. The methodology and insights from this analysis of freshwater consumption not only provides information for a critical ecosystem service, but also serves as an example for understanding the appropriation of other ecosystem services.

5. Conclusions

We developed a multiple city national analysis of linked societal demand and the production of ecosystem services. While there are difficulties in such an analysis, its use lies in identifying the variation in human–ecological linkages and the controls on such linkages (Karl et al., 1988; Folke et al., 1997; Decker et al., 2000; Luck et al., 2001; Jenerette et al., in press). Complexity in evaluating the effects of spatial heterogeneity on ecological economic interactions was accounted for in the spatial variation of the value, consumption, and production of an ecosystem service. In particular, linking footprint area, an estimate of the scale of consumption–production relationships, with economic valuation, may provide a useful metric for estimating the spatial variation in the value of the ecosystem service being provided. This valuation can then be properly accounted for in water management and pricing strategies. Among the 121 U.S. cities presently analyzed, the local supply of water and the per-capita demand for water varied greatly. This variation was partially described by a few hypothesized mechanisms; however, there is much idiosyncratic variation between the cities. This idiosyncrasy is likely an intrinsic component of coupled human–ecological systems. In understanding the link between the consumption and production of ecosystem services, two predominantly separated approaches have been developed, ecosystem service valuation and ecological footprints. Here we linked these two approaches by calculating the price per footprint area using an empirical analysis of the revenue and expenditures from water utilities of cities and the spatial pattern of renewable water production. For the 121 cities in our study, the mean cost associated with footprints was \$88,808 km^{-2} yr^{-1} and the variations were in part correlated with hypothesized controlling mechanisms. Identifying spatial heterogeneity in the interactions between the demand, economic costs, and area required for the appropriation of ecosystem services can enhance our understanding of how societies interact with the biosphere.

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