

Water use regimes: Characterizing direct human interaction with hydrologic systems

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[1] The sustainability of human water use practices is a rapidly growing concern in the United States and around the world. To better characterize direct human interaction with hydrologic systems (stream basins and aquifers), we introduce the concept of the water use regime. Unlike scalar indicators of anthropogenic hydrologic stress in the literature, the water use regime is a two-dimensional, vector indicator that can be depicted on simple x-y plots of normalized human withdrawals (h_{out}) versus normalized human return flows (h_{in}). Four end-member regimes, natural-flow-dominated (undeveloped), human-flow-dominated (churned), withdrawal-dominated (depleted), and return-flow-dominated (surcharged), are defined in relation to limiting values of h_{out} and h_{in} . For illustration, the water use regimes of 19 diverse hydrologic systems are plotted and interpreted. Several of these systems, including the Yellow River Basin, China, and the California Central Valley Aquifer, are shown to approach particular end-member regimes. Spatial and temporal regime variations, both seasonal and long-term, are depicted. Practical issues of data availability and regime uncertainty are addressed in relation to the statistical properties of the ratio estimators h_{out} and h_{in} . The water use regime is shown to be a useful tool for comparative water resources assessment and for describing both historic and alternative future pathways of water resource development at a range of scales.

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1. Introduction

[2] Global concerns about the sustainability of human water use practices have grown markedly in recent years. Developments contributing to these concerns include (1) streamflow depletion and lake dessication at all scales, caused in part by human withdrawals (e.g., Yellow River, China; Colorado and Sacramento Rivers, United States; Aral Sea, central Asia; Lake Chad, central Africa); (2) regional-scale aquifer depletion due to groundwater withdrawals (e.g., High Plains, United States; North China Plain); and (3) in-stream flow needs for recreation, navigation, waste assimilation, and aquatic habitat [Poff *et al.*, 1997; Richter *et al.*, 2003; Alley and Leake, 2004]. At the global level, these concerns have prompted numerous recent assessments of human water use in relation to water availability, and the relative impacts of water use and climate change on the hydrologic cycle [Postel *et al.*, 1996; Vörösmarty *et al.*, 2000; Oki *et al.*, 2001; Alcamo *et al.*, 2003; Döll *et al.*, 2003; Gleick, 2005; Oki and Kanae, 2006]. In the United States, studies of water availability and

use historically focused on the arid West [Anderson and Woosley, 2005], although water use practices in the “water-rich” eastern United States have recently been shown to cause streamflow depletion and aquatic habitat degradation [Richter *et al.*, 2003; Armstrong *et al.*, 2004].

[3] The most widely used indicator of anthropogenic flow stress is known by a variety of names, including the withdrawal ratio [Lane *et al.*, 1999], water scarcity index [Falkenmark *et al.*, 1989; Oki *et al.*, 2001], criticality ratio [Alcamo *et al.*, 2003], level of development [Hurd *et al.*, 1999], local relative water use [Vörösmarty *et al.*, 2005], and relative water demand [Vörösmarty *et al.*, 2000], or RWD, the term used in this paper. RWD is commonly defined as the ratio of total withdrawals (H_{out}) to an estimate of natural water availability, such as average predevelopment outflow from a stream basin:

$$RWD = H_{out}/SW_{out}^* \quad (1)$$

where SW_{out}^* is predevelopment outflow, obtained through simulation models [e.g., Alcamo *et al.*, 2003], regional regression models [Vogel *et al.*, 1999], or other means. For aquifers, natural water availability is typically equated with the predevelopment groundwater recharge from all sources.

[4] RWD is well suited for measuring one important type of anthropogenic stress: depletion of system storage and outflow caused by high rates of withdrawal in relation to renewable supply. However, certain globally important

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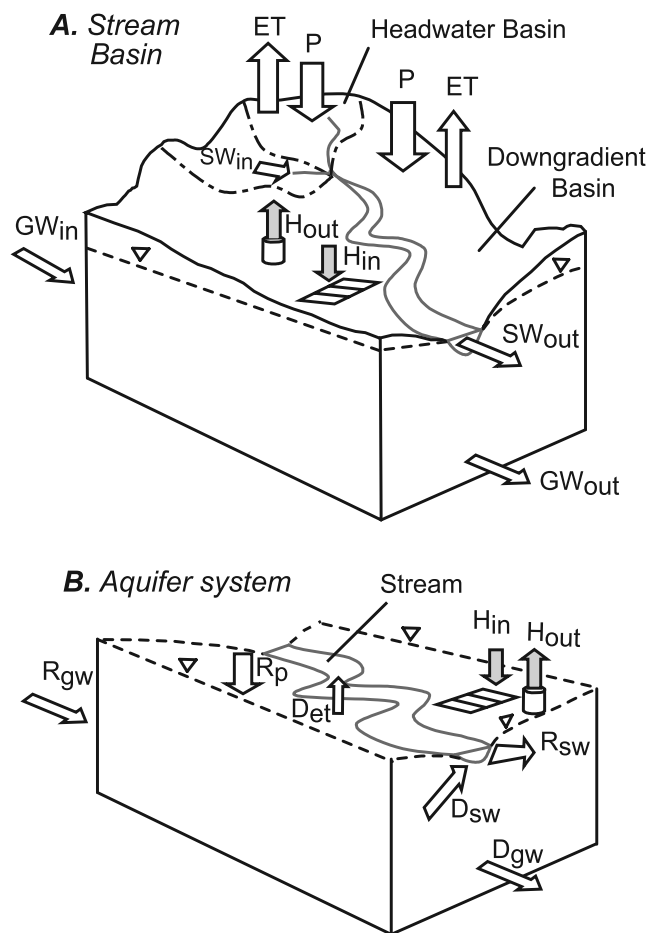


Figure 1. Water balance of (a) a stream basin and (b) an aquifer system. The “downgradient” basin receives inflow from “headwater” basins, which receive no lateral inflow. The aquifer system shown in Figure 1b is unconfined, with the dashed lines indicating the water table. See equations (4) and (5) and associated text for definition of all water balance components. Human inflows and outflows are shaded. All units are L^3/T .

anthropogenic stresses cannot be adequately characterized by RWD, because this indicator ignores return flows and water imports. For example, about 10% of the world’s 260 million hectares of irrigated agricultural land is affected by soil water logging and salinization, typically associated with high water tables caused by imports of surface water for irrigation in dry regions [Schoups *et al.*, 2005; Foley *et al.*, 2005]. Contamination of streams and shallow aquifers by high rates of domestic, irrigation, and industrial return flow [Meybeck, 2003; Foster and Chilton, 2003] is another global phenomenon not addressed by RWD.

[5] A partial solution to this limitation is to specify net demand ($H_{out} - H_{in}$) in the numerator of (1), yielding the relative net demand (RND) or “consumptive use in relation to renewable supply” [U.S. Geological Survey, 1984] (expressed here for a stream basin):

$$RND = [H_{out} - H_{in}] / SW_{out}^* \quad (2)$$

where H_{in} is total return flows plus imports of water and wastewater to the basin. Negative values of RND indicate

return flows (plus imports) in excess of withdrawals; hence RND can be used to characterize return-flow-dominated and withdrawal-dominated systems. Note, however, that RND fails to characterize the intensity of water use. Both highly developed and relatively undeveloped systems can have similar RND values, if the net human demand ($H_{out} - H_{in}$) is similar for both systems. The essential limitation of RWD and RND is that they are both one-dimensional, scalar indicators of human-induced hydrologic stress. A fully two-dimensional or vector approach, allowing for independent variation of both withdrawals and return flows relative to total system flows, is needed to adequately characterize the nature and degree of human interaction with hydrologic systems.

[6] Humans interact with hydrologic systems both directly and indirectly. For the purposes of this paper, “direct” interactions are limited to withdrawals and return flows. Indirect interactions, which nevertheless can have profound effects, include (1) anthropogenic land cover change [Foley *et al.*, 2005]; (2) dam construction for flood control and hydropower generation [Vörösmarty and Sahagian, 2000]; and (3) anthropogenic climate change [Vörösmarty *et al.*, 2000]. Conversely, some interactions between human water infrastructure and hydrologic systems are direct but unintentional. Examples include infiltration of groundwater into wastewater collection systems, conveyance losses from water distribution networks to the subsurface or the atmosphere, and evaporative losses from surface reservoirs [Weiss *et al.*, 2002]. For simplicity, only intentional withdrawals and return flows are considered in this paper.

[7] The purpose of this paper is to describe and apply a quantitative understanding of human water use, the water use regime, that accommodates the two-dimensional character of direct human interaction with terrestrial hydrologic systems. An approach is developed for characterizing the full range of anthropogenic flow stress upon hydrologic systems, in addition to certain “syndromes” of water quality degradation caused by return flows [Meybeck, 2003]. The approach is designed for hydrologists who conduct comparative water resource assessments at local, regional, or global scales [Falkenmark and Chapman, 1989; National Research Council (NRC), 2002], and who seek to define sustainable pathways of water resource development that maximize the productivity of water use while accounting for spatial and temporal variation in water availability [Loucks and Gladwell, 1999; Molden and Sakthivadivel, 1999; Falkenmark and Rockström, 2004; Rogers *et al.*, 2006].

2. Defining the Water Use Regime

2.1. Terrestrial Water Balance

[8] The water use regime is defined with respect to the water balance of an explicitly bounded hydrologic system (stream basin or aquifer; Figures 1a and 1b). It is useful to consider stream basins and aquifers separately because of their contrasting boundary conditions. A stream basin control volume is considered to include the land surface, its vegetation, streams and other surface water bodies, and both the unsaturated and saturated zones of the subsurface; it can be either a “headwater” or “downstream” basin (Figure 1a). An aquifer control volume is restricted to the

saturated portion of the subsurface (Figure 1b), and may range in scale from an individual model cell to an entire aquifer.

[9] In the case of a stream basin control volume, the total water balance can be expressed:

$$P + (GW_{in} + SW_{in}) + H_{in} - \Delta S/\Delta t = ET + (GW_{out} + SW_{out}) + H_{out} \quad (3)$$

where P is precipitation; $(GW_{in} + SW_{in})$ is groundwater and surface water inflows; ET is evapotranspiration; $(GW_{out} + SW_{out})$ is groundwater and surface water outflows; H_{in} is total return flow to the control volume from all sources, equivalent to the sum of (1) locally generated return flows from local withdrawals, (2) locally generated return flows from imported withdrawals, and (3) return flows imported from other basins through wastewater infrastructure; H_{out} is withdrawals from the control volume; and $\Delta S/\Delta t$ is the rate of change in control volume storage (surface and subsurface), all averaged over the period of interest. Constant water density is assumed. We then subtract ET from both sides of equation (3) to obtain the net water flux through the basin control volume, since only the net basin flux is directly available for human use:

$$\begin{aligned} \text{NetFlux}_{\text{basin}} &= (P - ET) + (GW_{in} + SW_{in}) + H_{in} - \Delta S/\Delta t \\ &= (GW_{out} + SW_{out}) + H_{out} \end{aligned} \quad (4)$$

For aquifer control volumes, equation (4) becomes:

$$\begin{aligned} \text{NetFlux}_{\text{aquifer}} &= (R_p - D_{et}) + (R_{gw} + R_{sw}) + H_{in} - \Delta S/\Delta t \\ &= (D_{gw} + D_{sw}) + H_{out} \end{aligned} \quad (5)$$

where R_p is aquifer recharge from precipitation; R_{gw} and R_{sw} are aquifer recharge from adjacent groundwater and surface water systems, respectively; D_{et} is groundwater ET ; D_{gw} and D_{sw} are aquifer discharge to adjacent groundwater and surface water systems; H_{in} is total return flow to the aquifer; H_{out} is aquifer withdrawals; and $\Delta S/\Delta t$ is the rate of change in aquifer storage. All units are $\text{length}^3/\text{time}$ (L^3/T) averaged over the period of interest. All flow terms are positive, except $\Delta S/\Delta t$, which can be positive, negative or zero during the period of interest. All terms in (4) and (5) except P are considered to be potentially affected by human-induced flow stress during the period of interest. In this paper, all water balance components under predevelopment conditions are denoted with an asterisk (e.g., SW_{out}^*).

[10] Normalized forms of (4) and (5) are obtained by dividing each term in the water balance by the respective net system flux, and expressing the resulting terms in lower case letters [cf. *Lent et al.*, 1997]. For example, the normalized H_{in} and H_{out} components are defined as:

$$h_{in} = H_{in}/\text{NetFlux} \quad (6)$$

$$h_{out} = H_{out}/\text{NetFlux} \quad (7)$$

where $\text{NetFlux}_{\text{basin}} = (SW_{out} + GW_{out}) + H_{out}$, and $\text{NetFlux}_{\text{aquifer}} = (D_{gw} + D_{sw}) + H_{out}$.

2.2. Water Use Regime

[11] The water use regime of a hydrologic system is defined as the set of system withdrawals, uses, and return flows during a period of interest. This paper focuses upon withdrawals and return flows, the two aspects of the water use regime that entail direct interaction with the hydrologic system, and their relative magnitude with respect to overall flow through the system. These relative magnitudes can be used to construct a water use regime plot, an x-y plot of h_{out} versus h_{in} (Figure 2). The plot domain defines the possible universe of direct flow interaction between humans and a hydrologic system. The domain is bounded by four end-member regimes (Figure 2): (1) natural-flow-dominated (or undeveloped, where $h_{out} = h_{in} = 0$), (2) withdrawal-dominated (depleted; $h_{out} = 1$; $h_{in} = 0$), (3) return-flow-dominated (surcharged; $h_{out} = 0$; $h_{in} = 1$), and (4) human-flow-dominated (churned; $h_{out} = h_{in} = 1$). Regimes characterized by net, human-induced depletion of system outflow and (or) storage plot in the lower right half of this domain ($h_{out} > h_{in}$); net accretion regimes plot in the upper left half ($h_{in} > h_{out}$).

[12] A pair of descriptive regime indicators, the human water balance (HWB) and the water use intensity (WUI), may be derived from h_{out} and h_{in} as follows:

$$\text{HWB} = h_{in} - h_{out} \quad (8)$$

$$\text{WUI} = (h_{in} + h_{out})/2 \quad (9)$$

HWB ranges from -1 to $+1$, and corresponds to the distance of a regime point to the right ($-$) or the left ($+$) of the line of equality ($h_{in} = h_{out}$) on the regime plot. The magnitude and sign of HWB indicate the degree and character, respectively, of direct human alteration of the system water balance (net accretion or depletion of system outflows and storage by humans). WUI varies from 0 to $+1$, and indicates the relative magnitude of human versus natural flows through a system.

3. Applications

[13] For illustration, the water use regimes of 19 hydrologic systems representing a range of climatic zones, stress conditions, and spatial/temporal scales are plotted and briefly discussed. Hydrologic budgets for the 7 stream basins and 12 aquifers were obtained from the published literature (Tables 1–3). The stream basin budgets were estimated or simulated using methods described in the references; all of the aquifer budgets were obtained from published simulation models.

3.1. Water Use Regimes: Stream Basins

[14] The South Platte River Basin was the most intensively developed of the seven stream basins considered, as measured by water use intensity ($\text{WUI} = 0.76$; Table 2 and Figure 3a). This water use regime reflects large irrigation withdrawals, substantial water imports from the Colorado River Basin, and low water availability ($P - ET$) over most of the basin area. By contrast, the Muskegon River Basin in west central Michigan had a low-intensity regime ($\text{WUI} = 0.03$), with high water availability, low population density, and low total withdrawals and returns. Wisahickon Creek

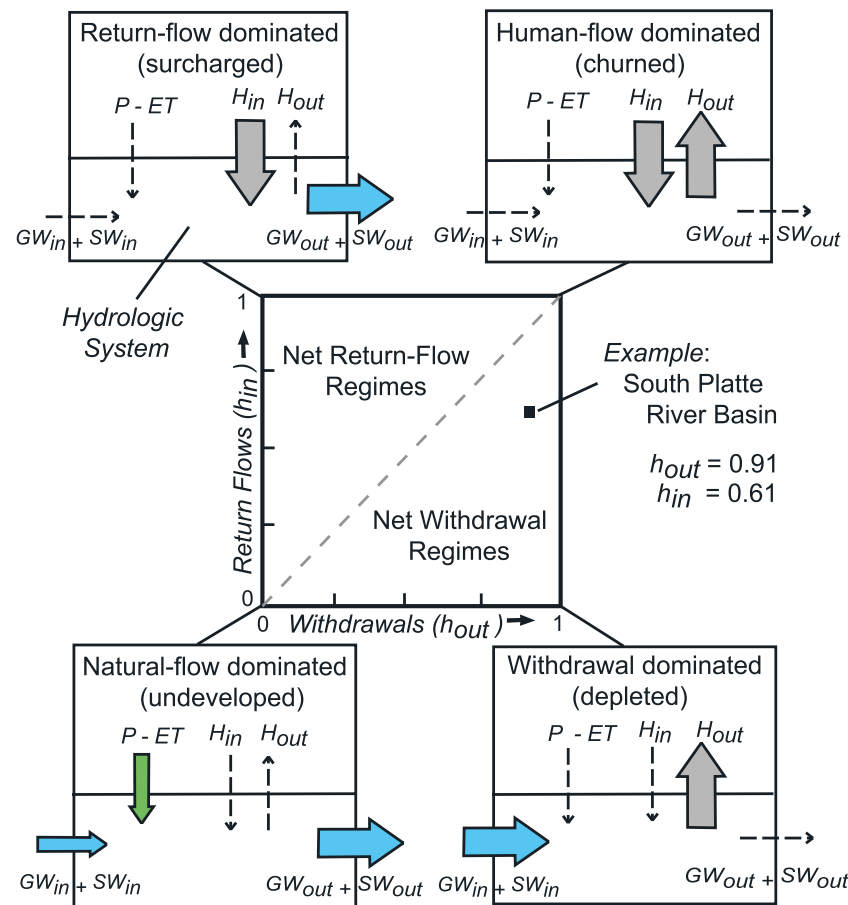


Figure 2. Human water use regimes. The relative magnitudes of normalized human withdrawals (h_{out}) versus return flows (h_{in}) are plotted on the central plot. Example regime is given for South Platte River Basin, United States, based on the work by Dennehy *et al.* [1993]. The panels show the four end-member regimes that bound the domain of possible water use regimes for a hydrologic system. Dashed arrows indicate fluxes that are either zero or very small relative to the other fluxes on each panel. For convenience, the natural-flow-dominated panel assumes humid climatic conditions ($P > ET$). See equations (3), (6), and (7) for definitions of all terms. Fluxes into and out of storage are not shown.

Basin, west of Philadelphia, Pennsylvania, had a somewhat higher water use intensity ($WUI = 0.18$), a slightly negative human water balance, and an overall water use regime typical of urbanized basins in the humid northeastern United States.

[15] In the remaining basins, human inflows and outflows were significantly out of balance under the various conditions considered. The largest of these systems is the Yellow River Basin, which drains a 865,000 km² semiarid, agricultural region in northern China. The human water balance was strongly negative ($HWB = -0.73$) during the period studied (1998–2000); the basin approached the withdrawal-dominated, or depleted, end-member regime (Figure 3a). In August 1993, the Upper Ipswich River Basin, Massachusetts, also had a very high normalized withdrawal coupled with low water availability. However, this moderately urbanized basin had higher rates of return flow ($h_{in} = 0.37$) than the Yellow River, and therefore displayed a mixed regime between the depleted and and churned end-members. Although the Upper Ipswich Basin is considered one of the most flow-stressed basins in the northeastern United States [Zarriello and Ries, 2000], only during the summer does it display a regime comparable to

the average annual regime of the Yellow River Basin, which covers an area ~ 7500 times larger.

[16] The Sacramento River Basin in California, like the Yellow River Basin, is a globally important agricultural region with high withdrawal rates per unit basin area (240 mm/yr), mostly for irrigation and urban uses. However, because average water availability (418 mm/yr) was over 6 times greater in the Sacramento Basin than in the Yellow Basin, (Table 2), h_{out} was smaller (Figure 3a), and the water use regime was more balanced ($HWB = -0.37$). The moderately urbanized Upper Assabet Main stem River Basin in east central Massachusetts, simulated for average September conditions during 1997–2001, was the only stream basin considered with a positive human water balance during the period of interest ($HWB = +0.31$). This regime reflects imports of treated municipal wastewater to the main stem river in excess of local withdrawals, combined with low summer baseflows.

3.2. Water Use Regimes: Aquifers

[17] The selected aquifers showed an equally wide diversity of water use regimes (Figure 3b). The California Central Valley Aquifer most closely approximates a churned

Table 1. Hydrologic Systems Selected for Water Use Regime Analysis

Hydrologic System	Source
Yellow River Basin, China	<i>Cai and Rosegrant</i> [2004]
Sacramento River Basin, CA	<i>Yates et al.</i> [2007]
South Platte River Basin, CO, NE, WY	<i>Dennehy et al.</i> [1993]
Muskegon River Basin, MI	R. Vogel (manuscript in preparation, 2006)
Wissahickon Creek Basin, PA	<i>Sloto and Buxton</i> [2005]
Upper Assabet River Basin, MA	<i>DeSimone</i> [2004]
Upper Ipswich River Basin, MA	<i>Zarriello and Ries</i> [2000]
Central Valley Aquifer, CA	<i>Johnston</i> [1999]
Southern High Plains Aquifer, TX, NM	<i>Johnston</i> [1999]
Mississippi River Alluvial Aquifer, AR	<i>Reed</i> [2003]
Floridan Aquifer, FL, AL, GA, SC	<i>Johnston</i> [1999]
Eastern Snake River Plain Aquifer, ID	<i>Garabedian</i> [1992]
Long Island Aquifer, NY	<i>Buxton and Smolensky</i> [1999]
La Crosse County Aquifer, WI	<i>Hunt et al.</i> [2003]
Paradise Valley Aquifer, NV	<i>Prudic and Herman</i> [1996]
Cape Cod Aquifer, MA	<i>Walter et al.</i> [2004]
Upper Charles River Aquifer, MA	<i>Eggleson</i> [2003]
NE Antelope Valley Aquifer, CA	<i>Nishikawa et al.</i> [2001]
Irwin Basin Aquifer, CA	<i>Densmore</i> [2003]

regime, in which withdrawals and return flows dominated the overall water balance ($WUI = 0.87$). By contrast, a group of aquifers from the humid northeastern and north central United States (Cape Cod, Upper Charles, and La Crosse County) could be considered natural-flow-dominated ($WUI = 0.05$ to 0.08). The Floridan and Long Island Aquifers displayed more developed regimes ($WUI = 0.15$ and 0.25 , respectively), while the Northeast Antelope Valley Aquifer in the Mojave Desert, California, approached a purely withdrawal-dominated or depleted regime, where essentially all outflows from the system were captured for human use ($HWB = -0.83$). By contrast, the Eastern Snake River Plain Aquifer, Idaho, had a positive human water balance ($HWB = +0.45$). In this case, infiltration of surface irrigation water imported to the aquifer from adjacent mountain areas substantially exceeded local withdrawals.

[18] The remaining aquifers displayed mixed regimes involving two developed end-members. For example, the 75,000 km² Southern High Plains Aquifer was pumped at very high rates during the period of interest relative to natural recharge from precipitation ($H_{out} = 115$ mm/yr;

$R_p \approx 3$ mm/yr). However, unlike some other heavily pumped aquifers, (e.g., the Northeast Antelope Valley), the Southern High Plains Aquifer derived significant inflow from irrigation return flow as well as from storage depletion, placing it midway between the depleted and churned end-members. The Irwin Aquifer, California, had a contrasting type of mixed regime—midway between the surcharged and churned end-members. In this case, large wastewater imports were balanced by both withdrawals and accretion of storage.

3.3. Spatial Variation in Water Use Regime

[19] Water use regimes and their derived indicators (HWB and WUI) may be mapped at any spatial scale for which required data or model output are available. Regimes for stream basins may be spatially discretized by subbasin (Figure 4), or by model cell if a gridded model is used. Subbasins in the the Assabet River Basin, for example, showed significant variation in human water balance and water use intensity (Figures 4a and 4b). A series of main stem subbasins, extending from the southwestern headwaters to the confluence with the Sudbury River in the northeast (Figure 4), all had moderately positive HWB values (+0.15 to +0.31). This reflects net import of wastewater from adjacent tributary subbasins, which, in turn, were relatively depleted due to net wastewater export (HWB values of -0.02 to -0.26). Water use intensity is greatest in the main stem subbasins, where WUI ranges from 0.15 to 0.34.

3.4. Long-Term Temporal Change: Water Resources Development Pathway

[20] The position vector connecting the origin of a regime plot ($h_{out} = h_{in} = 0$) to a regime point depicts the average water resources development pathway of a hydrologic system over its history. The actual pathway to a particular regime can be expected to be circuitous, due to historical changes in withdrawals, return flows, and climatic conditions. The Mississippi River Alluvial Aquifer of northeast Arkansas, as simulated by *Reed* [2003], serves to illustrate the pathway concept (Figure 5). Significant withdrawals from the aquifer for agricultural irrigation began in the early 1900s, and averaged 27 m³/s from 1918 to 1957. By 1998, withdrawals had increased to 207 m³/s, due mainly to the rapid expansion of irrigated rice agriculture. Until 1972,

Table 2. Hydrologic Budgets of Selected Stream Basins, Averaged Over the Periods Specified^a

Stream Basin	DA, km ²	P	Inflows, ^b m ³ /s			Outflows, m ³ /s				Net				
			GW + SW _{in}	H _{in}	Total in	ET	GW + SW _{out}	H _{out}	Total out	ΔS/Δt	h _{out}	h _{in}	HWB	WUI
Yellow, 1998–2000	865,000	11,395	0	270	11,664	9,708	216	1,579	11,502	162	0.88	0.15	−0.73	0.52
Sacramento, ^c 1962–1998	72,000	2,087	0	140	2,227	1,113	565	549	2,227	0	0.49	0.13	−0.37	0.31
South Platte, ^{c,d} 1990	62,900	784	0	113	897	709	16	170	896	0	0.91	0.61	−0.30	0.76
Muskegon, 1995	5,390	124	0	1.72	126	76	65	2.9	144	−17	0.04	0.03	−0.02	0.03
Wissahickon, 1987–1998	166	6.3	0	0.53	6.8	3.1	3.0	0.8	6.9	−0.02	0.21	0.14	−0.07	0.18
U. Assabet., Sep 1997–2001	27	1.0	0.13	0.27	1.4	1.3	0.45	0.10	1.7	−0.31	0.18	0.49	0.31	0.34
U. Ipswich, Aug 1993	115	0.15	0	0.011	0.16	0.31	0.002	0.03	0.33	−0.17	0.93	0.37	−0.56	0.65

^aSee Table 1 for sources and text for definition of budget terms. Flows are in m³/s; h_{out} and h_{in} are dimensionless. Basins are ranked by H_{out} .

^bInflows may not sum to outflows plus change in storage, due to independent rounding.

^cSteady state flow conditions assumed by source reference

^dSource reference used long-term-average values of all budget components except for human flows, which are for 1990.

Table 3. Hydrologic Budgets of Selected Aquifers, Averaged Over the Periods Specified^a

Aquifer System	Inflows, ^b m ³ /s				Outflows, m ³ /s				Net $\Delta S/\Delta t$	h_{out}	h_{in}	HWB	WUI
	R_p	R_{gw+sw}	H_{in}	Total in	D_{et}	D_{gw+sw}	H_{out}	Total out					
CA Central Valley, 1961–1977	58.6	19.5	367.0	445.0	0	11.6	465.0	477.0	−31.4	0.98	0.77	−0.21	0.87
So. High Plains, 1960–1980	7.6	44.1	115.0	166.0	0	3.4	273.0	276.0	−110.0	0.99	0.41	−0.57	0.70
Mississippi R. Alluvial, 1994–1998 ^c	45.9	79.2	86.9	212.0	0	36.7	207.0	243.0	−31.6	0.85	0.41	−0.44	0.63
Floridan, 1980 ^{d,e}	598.0	0	81.2	679.0	0	563.0	116.0	679.0	0	0.17	0.12	−0.05	0.15
Eastern Snake R. Plain, ID, 1980	27.5	98.5	189.0	315.0	0	277.0	44.6	321.0	−6.2	0.14	0.59	0.45	0.36
Long Island, NY, 1968–1983 ^d	45.8	0	10.9	56.7	0	38.9	17.9	56.7	0	0.31	0.19	−0.12	0.25
La Crosse County, WI, 2003 ^d	16.2	8.3	0	24.5	0	21.7	2.8	24.5	0	0.12	0.00	−0.12	0.06
Paradise Valley, NV, 1981–1982	0	2.7	0.4	3.0	1.9	0.1	1.8	3.8	−0.7	0.95	0.19	−0.76	0.57
Cape Cod, 2003 ^{d,e}	18.7	0	0.97	19.6	0	18.5	1.1	19.6	0	0.06	0.05	−0.01	0.05
Upper Charles R., MA, 1989–1998	1.19	3.61	0.42	5.21	0.27	4.57	0.37	5.21	0.01	0.07	0.08	0.01	0.08
NE Antelope Valley, CA, 1996	0	0.04	0.01	0.05	0	0.03	0.18	0.21	−0.16	0.88	0.04	−0.83	0.46
Irwin Basin, CA, 1999	0.002	0	0.055	0.057	0	0.004	0.031	0.035	0.022	0.53	0.97	0.43	0.75

^aAll budgets were obtained from simulation models; see Table 1 for sources and text for definition of budget terms. Aquifers are ranked by H_{out} .

^bInflows may not sum to outflows plus change in storage, due to independent rounding.

^c H_{in} not provided by source reference; it is estimated as total inflow minus ($R_p + R_{gw+sw}$), and may represent an overestimate.

^dSteady state flow conditions assumed by source reference.

^eSource reference used long-term-average values of all budget components except for human flows, which are for year specified.

return flows were simulated to be relatively small; most of the withdrawal demand was met by increased recharge from, and decreased discharge to, adjacent streams and adjacent aquifer units, accompanied by modest depletion of aquifer storage. After 1972, return flows were estimated to be a significant fraction of the total budget. The development pathway shifted upward from the h_{out} axis, and proceeded toward a relatively high-intensity regime by 1998 (WUI = 0.63).

3.5. Short-Term Temporal Change: Effects of Seasonality

[21] The Upper Charles River Aquifer had a highly seasonal pattern of simulated natural recharge, natural discharge, and human withdrawal [Eggleston, 2003], similar to the pattern previously documented in a New England glacial valley aquifer by Barlow and Dickerman [2001]. Although precipitation was evenly distributed throughout the year, natural recharge from precipitation (R_p) occurred mainly from October to May, when ET from the unsaturated zone is low. Net withdrawals ($H_{out} - H_{in}$), by contrast, were greatest from June to September, when R_p is very low due to high unsaturated zone ET. Consequently, summer withdrawal demands were met largely by depletion of aquifer storage. The net result was an essentially balanced annual regime (Figure 6), with peak water use intensity in September (WUI = 0.16), and a slightly negative human water balance in the summer months (HWB = −0.01 to −0.04).

4. Data Availability, Model Simulation, and Regime Uncertainty

4.1. Data Availability and the Role of Simulation Models

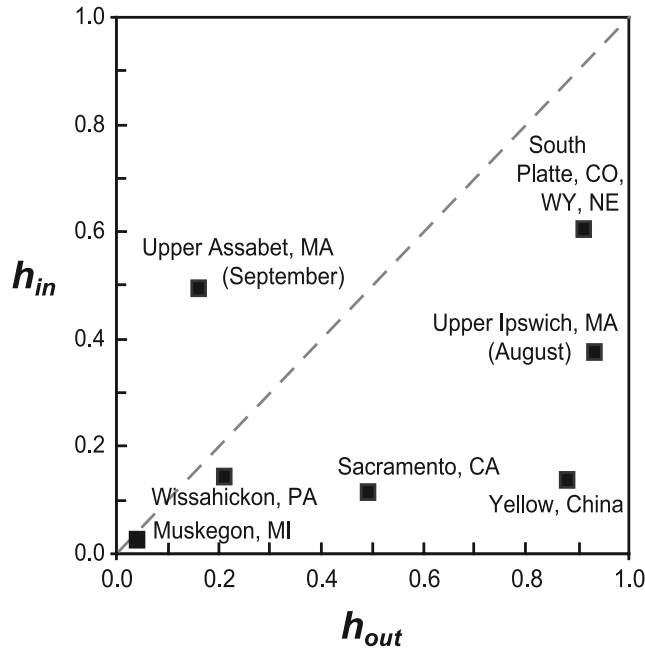
[22] Only three types of data are required to specify the water use regime of a hydrologic system: (1) net system outflow under stressed conditions ($SW_{out} + GW_{out}$ for stream basins or $D_{sw} + D_{gw}$ for aquifers); (2) withdrawals (H_{out}); and (3) return flows from local sources plus imports to the system (H_{in}); see (3) through (9). The most widely available data type, by far, is net basin outflow. In the United States, the U.S. Geological Survey presently oper-

ates about 7300 continuous record stream gages in a wide variety of basins where SW_{out} may be quantified at hourly to decadal timescales, depending upon the period of record (see <http://water.usgs.gov/nsip/>). In many basins, GW_{out} is either very small relative to SW_{out} or close in magnitude to GW_{in} . In such cases, SW_{out} approximates net basin outflow. In many aquifer systems, D_{gw} is either small relative to D_{sw} , or close in magnitude to R_{gw} . In such cases, stream baseflow (D_{sw}) approximates net aquifer outflow. Baseflow may be estimated from streamgage records using a variety of manual and automated hydrograph separation methods [Rutledge, 1998].

[23] In areas of the world with sparse streamflow data, or in areas with substantial regional groundwater recharge or discharge, the GW_{out} and GW_{in} (or D_{gw} and R_{gw}) terms cannot be neglected and simulation models may be required to estimate SW_{out} . At global and continental scales, however, gridded, steady state, meteorologically driven water balance models of the global land surface have recently been developed to estimate SW_{out}^* , both with and without calibration to streamflow data [Vörösmarty et al., 2000, 2005; Oki et al., 2001; Alcamo et al., 2003; Döll et al., 2003].

[24] The remaining two data types required, withdrawals and return flows, are less widely available than streamflow data in most regions. In the United States, the U.S. Geological Survey compiles withdrawal (H_{out}) estimates at 5-year intervals for thermoelectric, irrigation, public supply, self-supplied industrial, self-supplied domestic, and other water use sectors, aggregated most recently at State, County, and principal aquifer levels [Hutson et al., 2004; Maupin and Barber, 2005] (see <http://water.usgs.gov/watuse/>). The U.S. Department of Agriculture (USDA) also assesses U.S. irrigation withdrawals at 5-year intervals [USDA, 2004], and the States collect a wide range of aggregated and site-specific water use data [NRC, 2002]. Recently, global water resources assessments have used georeferenced population and irrigated area data to estimate withdrawal rates, by major sector, for use in gridded models [e.g., Alcamo et al., 2003]. Periodic, worldwide estimates of withdrawals are also available by country [Gleick, 2005].

A. Stream basins



B. Aquifer systems

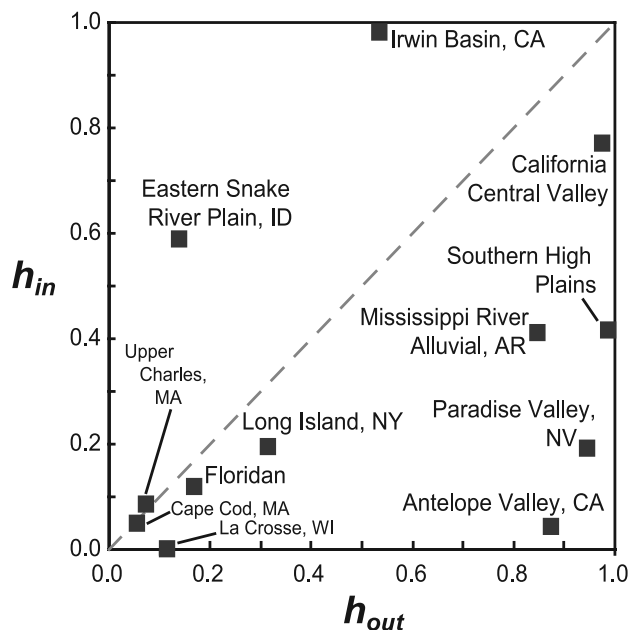


Figure 3. Water use regimes of selected (a) stream basins and (b) aquifer systems. See Tables 2 and 3 for source data and text for definition of normalized human withdrawal (h_{out}) and return flow (h_{in}) terms.

[25] Throughout the world, return flows (H_{in}) are generally less well characterized than withdrawals. In most developed countries, programs such as the U.S. National Pollutant Discharge Elimination System (NPDES) program <http://cfpub.epa.gov/npdes/> track large return flows from municipal and industrial water use sectors. However, non-point and unregulated point returns from these and other

sectors are usually poorly known, and are typically estimated using empirical consumptive use coefficients. Coefficient errors [Solley *et al.*, 1998] are generally unknown but potentially large. Recently, improved estimates of irrigation return flow have been obtained using georeferenced withdrawal data in concert with models that simulate irrigation requirements as a function of climate and crop type [Döll and Siebert, 2002; Schoups *et al.*, 2005].

4.2. Water Use Regime Uncertainty

[26] All water resources assessment approaches are subject to uncertainty, due to measurement error, sampling error, and model error in cases where models are used. Although a comprehensive uncertainty analysis of water use regimes is a topic for future research, we briefly describe one approach for estimating likelihood intervals for estimated values of the ratio estimators h_{in} and h_{out} , where $h_{in} = H_{in}/(SW_{out} + H_{out})$ and $h_{out} = H_{out}/(SW_{out} + H_{out})$. Vogel and Wilson [1996] and others have found that a normal distribution provides a good approximation to the probability density function (pdf) of annual streamflows (SW_{out}) for most temperate regions, whereas a Gamma or Pearson type III distribution is needed in regions of greater hydrologic variability. In this initial study, we begin by assuming a normal pdf for estimates of H_{out} and H_{in} , as well as SW_{out} . Since Geary [1930], numerous investigators have studied the statistical properties of the ratio of two normal random variables. The pdf of $R = X/Y$ is given by Öksoy and Aroian [1994]. In our case, $X = H_{out}$ and $Y = SW_{out} + H_{out}$, and they are considered to be bivariate normal variables (see Appendix A).

[27] Figure 7 shows a set of hypothetical 90% confidence intervals around the previously plotted (h_{out} , h_{in}) positions of Figure 3a, based on this analysis. These intervals were calculated using hypothetical coefficients of variation of 0.05, 0.1, and 0.15 for SW_{out} , H_{out} , and H_{in} respectively. The relative magnitude of these C_v values reflects one possible set of assumptions concerning these variables, namely, the suspected low, moderate, and high degree of uncertainty concerning SW_{out} , H_{out} , and H_{in} . Note that h_{in} and h_{out} are least sensitive to error when near 0 or 1, and most sensitive to error toward the middle of the regime plot. The exact location of the zone of maximum error sensitivity will depend upon the relative magnitude of the respective C_v values. Improvements in water use regime uncertainty analysis should result from (1) further exploration of the statistical properties of H_{in} , H_{out} , and SW_{out} (or D_{sw} in the case of aquifer systems), (2) better characterization of H_{out} and H_{in} variability and error (because error for SW_{out} is already well characterized), and (3) extensions which treat R as the ratio of two Gamma or Pearson type III variables [Loaiciga and Leipnik, 2005].

5. Conclusions

[28] The study leads to the following conclusions.

[29] 1. Human water use may be characterized as a two-dimensional process, entailing both withdrawals from and return flows to hydrologic systems. The water use regime framework provides a more complete representation of this process than commonly used one-dimensional indicators. The framework specifies four end-member regimes: natural-flow-dominated (undeveloped), human-flow-dominated

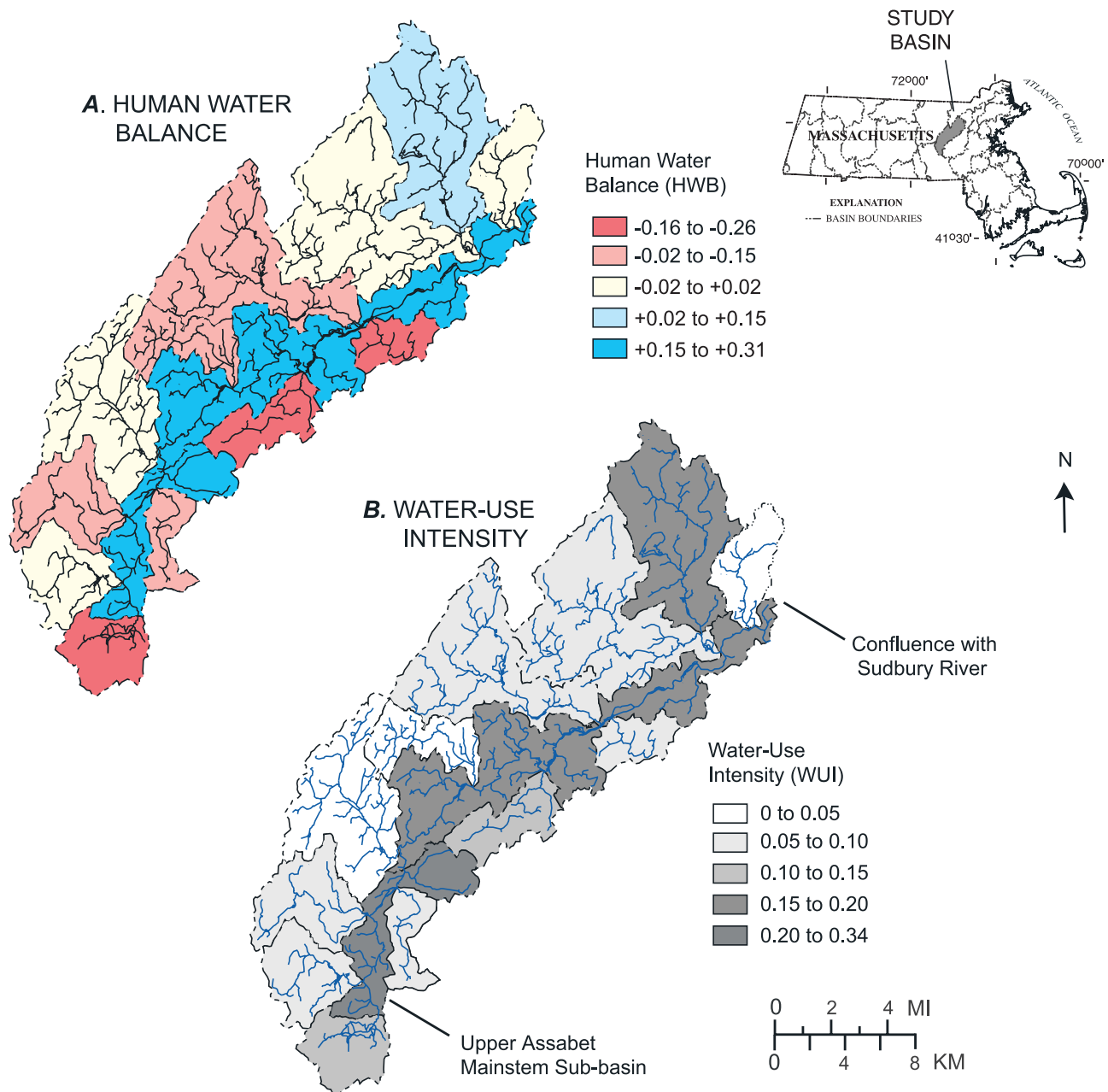


Figure 4. Average September water use regimes, Assabet River Basin, Massachusetts, 1997–2001, as indicated by the (a) human water balance and (b) water use intensity indicators, defined by (8) and (9), based on model simulation results of *DeSimone* [2004].

(churned), withdrawal-dominated (depleted), and return-flow-dominated (surcharged). Regime plots can be used for comparative analysis of developed hydrologic systems, and for interpreting their seasonal dynamics and long-term historical development.

[30] 2. Regional-scale hydrologic systems can be highly impacted by human water use, even when the effects are spatially and temporally averaged. The 52,000 km² California Central Valley Aquifer and the 63,000 km² South Platte River Basin, for example, both displayed average water use regimes approaching the churned end-member. The 865,000 km² Yellow River Basin, China, approached the depleted end-member on an annual basis. Typically,

highly impacted regional systems have low water availability ($P - ET$) combined with large consumptive losses ($H_{out} - H_{in}$) from irrigation, although consumptive losses and return flows were found to vary widely.

[31] 3. Characterization of water use regimes is limited by data availability and uncertainty. In particular, human return flows (H_{in}) are often poorly estimated or not adequately differentiated from natural inflows to a system. Improved procedures for site-specific estimation of withdrawals, return flows, and their variability are a high-priority research need. Although subject to additional forms of uncertainty, gridded water balance models at the basin, continental, and global scales [Alcamo *et al.*, 2003; Vörösmarty *et al.*, 2005], as

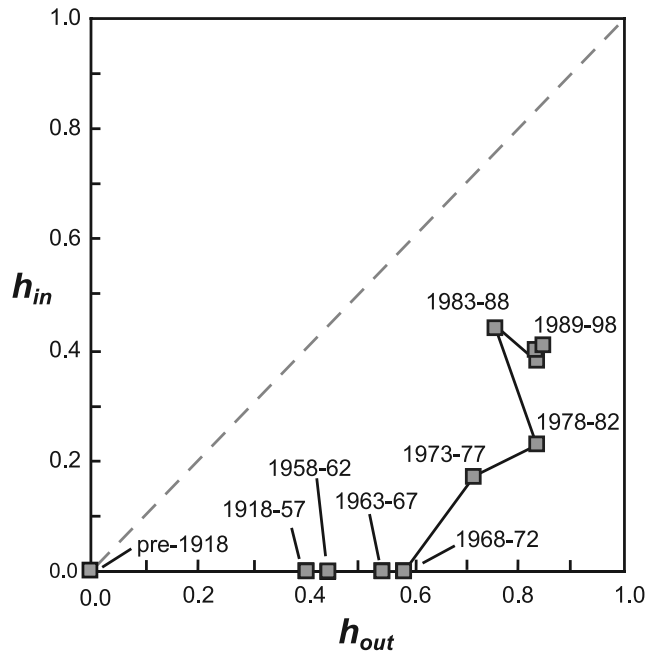


Figure 5. Water resources development pathway for the Mississippi River Alluvial Aquifer, Arkansas, predevelopment conditions (1918) to 1998. Each point represents the average water use regime during the stress period indicated, based on transient simulation results of Reed [2003].

well as groundwater flow models (Table 1 sources) are useful tools for future mapping of water use regimes.

Appendix A: Probability Density Function of the Ratio of Two Normal Variables

[32] In this initial study we focus on the statistical properties of h_{out} , however, the exact same approach may

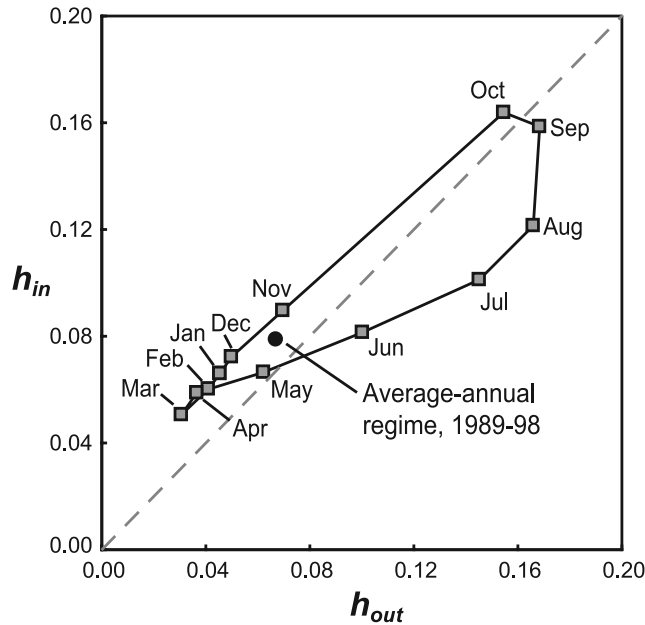


Figure 6. Average monthly water use regimes, Upper Charles River Aquifer, Massachusetts, 1989–1998, based on the transient simulation results of Eggleston [2003]. Average annual regime for this period is also shown.

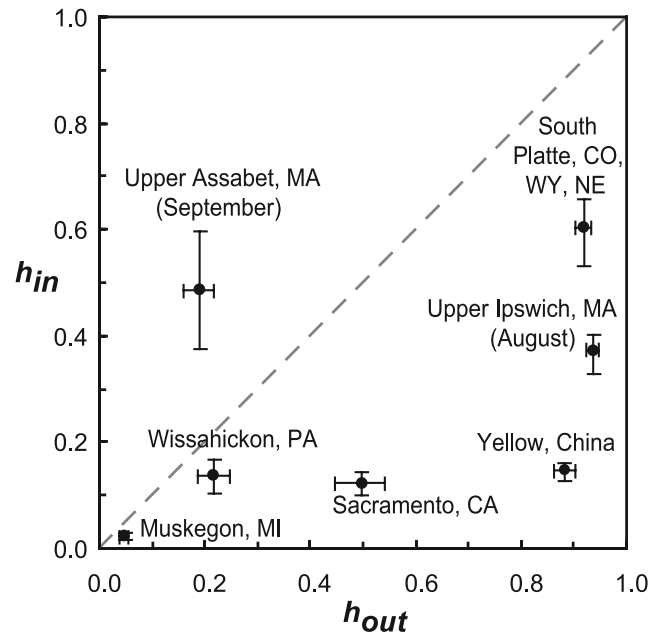


Figure 7. Sensitivity of water use regimes to errors in system outflow (SW_{out}), withdrawals (H_{out}), and return flows (H_{in}) for coefficients of variation of 0.05, 0.10, and 0.15, respectively. Error bars show 90% confidence intervals for resulting estimates of h_{out} and h_{in} for watersheds of Figure 3a.

be applied to h_{in} . In the case of $h_{out} = H_{out}/(SW_{out} + H_{out}) = X/Y$ the mean of X and Y , μ_x and μ_y , are given by $\mu_x = \mu_{H_{out}}$ and $\mu_y = \mu_{SW_{out}} + \mu_{H_{out}}$ and their variances σ_x^2 and σ_y^2 , are given by $\sigma_x^2 = \sigma_{H_{out}}^2$ and $\sigma_y^2 = \sigma_{H_{out}}^2 + \sigma_{SW_{out}}^2$. Here we assume, initially, that H_{out} and SW_{out} are independent, in which case it can also be shown that the correlation of X and Y , is equal to $\rho = 1/\sqrt{1 + (\sigma_{SW_{out}}^2/\sigma_{H_{out}}^2)}$. Interestingly, even though

H_{out} and SW_{out} are assumed to be independent and thus uncorrelated, the numerator X and denominator Y in $R = X/Y$ are correlated. One can easily show that the correlation between H_{out} and $H_{out} + SW_{out}$ increases as their ratio, h_{out} , increases and as the coefficient of variation (C_v) of H_{out} increases, relative to the C_v of SW_{out} .

[33] Öksoy and Aroian [1994] compare and contrast a number of different, yet equivalent approaches for expressing the exact pdf of $R = X/Y$ where X and Y follow a bivariate normal pdf. The simplest exact result from Öksoy and Aroian [1994, equation [8]]:

$$f_R(r) = \frac{\sigma_y}{\pi\sigma_x(1+t^2)\sqrt{1-\rho^2}} \exp\left[-\frac{(a^2+b^2)}{2}\right] (1+c\Phi(q)) \quad (A1)$$

where

$$a = \frac{\mu_x - \rho\mu_y}{\sigma_x\sqrt{1-\rho^2}}; \quad b = \frac{\mu_y}{\sigma_y}; \quad c = \frac{q}{\phi(q)};$$

$$t = \frac{\sigma_y r - \rho}{\sigma_x\sqrt{1-\rho^2}}; \quad \text{and } q = \frac{b+at}{\sqrt{1+t^2}}$$

$\phi(z)$ and $\Phi(z)$ are the pdf and cdf of a standard normal random variable z and $r = x/y$ is a realization of the random variable $R = X/Y$. A number of investigators have introduced approximations to the pdf of R , however, Öksoy and Aroian [1994] show that such approximations can lead to gross errors. Interestingly, all moments of R are undefined yet its median is equal to μ_x/μ_y . The distribution of R is rarely symmetric and can even exhibit bimodal behavior. One may compute the likely interval of values for the ratio R using

$$\int_{R_{\text{lower}}}^{\mu_x/\mu_y} f_R(r) dr = \frac{1-\alpha}{2} \quad \text{and} \quad \int_{\mu_x/\mu_y}^{R_{\text{upper}}} f_R(r) dr = \frac{1-\alpha}{2} \quad (\text{A2})$$

with $f_R(r)$ given in (A1) and $\alpha = 0.10$ to obtain a 90% likelihood interval $[R_{\text{lower}}, R_{\text{upper}}]$.

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