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Estimation of groundwater pumping as closure to the water balance of a semi-arid, irrigated agricultural basin

Nels Ruud, Thomas Harter*, Alec Naugle

Department of Land, Air, and Water Resources, University of California, Davis, CA, USA

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Abstract

Groundwater pumping is frequently the least measured water balance component in semi-arid basins with significant agricultural production. In this article, we develop a GIS-based water balance model for estimating basin-scale monthly and annual groundwater pumping and apply it to a 2300 km^2 semi-arid, irrigated agricultural area in the southern San Joaquin Valley, California. Both, annual groundwater storage changes and pumping are estimated as closure terms. The local hydrology is dominated by distributed surface water supplies, limited precipitation, and large crop water uses; whereas basin-scale runoff generation and groundwater-to-surface water discharges are negligible. Groundwater represents a terminal long-term storage reservoir with distributed inputs and outputs. To capture the spatio-temporal variability in water management and water use, the study area is delineated into 26 water service areas and 9611 individual fields or land units. The model computes conveyance seepage losses external to districts; seepage losses within districts; and net applied surface water of each district. For each land unit, the model calculates the applied water demand; its allotment of delivered surface water; the groundwater pumping required to meet the balance of its applied water demand; and aquifer recharge resulting from deep percolation of applied water and precipitation. These spatially distributed components are aggregated to the basin scale. Estimated annual groundwater storage changes compared well to those computed by the water-table fluctuation method over the 30-year study period, providing an independent verification of the consumptive use estimation. Pumping accounted for as much as 80% of the total applied water in 'Critical' water years and as little as 30% in 'Wet' years. Pumping estimates are most sensitive to estimation uncertainty of soil available water. They show little sensitivity to estimation errors in effective root depth, irrigation efficiencies, and intra-district seepage losses, although the cumulative sensitivity is significant.

Keywords: Water balance; Agriculture; Groundwater; Recharge; Semi-arid climate; Irrigation

1. Introduction

In the semi-arid San Joaquin Valley, California, intensively developed agriculture is highly dependent on irrigation with surface water and groundwater to meet crop water demands over the often year-round growing season (California Department of Water Resources (CDWR), 1998). Surface water availability varies seasonally due to the climatic pattern and limited upstream reservoir storage capacity and spatially due to conveyance infrastructure limitations and differences in water rights. Longer periods of severe drought,

* Corresponding author. Fax: +1-530-752-5262.
E-mail addresses: thharter@ucdavis.edu (T. Harter), nruud@fugro.com (N. Ruud), awn@rb2.swrcb.ca.gov (A. Naugle).

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lasting several years, can cause groundwater basins in the San Joaquin Valley to become severely overdrafted in a matter of a few years (California Department of Water Resources (CDWR), 1980). Concerns over agricultural water use, surface water supply reliability, and groundwater storage changes in these basins have increased the demand for sustainable groundwater management. Similar concerns have been voiced in agriculturally developed semi-arid basins worldwide (Gleick, 2000; Kendy et al., 2003).

In recent years, attention has been given to evaluating the use and productivity of water at the basin scale by treating each basin as a whole unit of study (Seckler, 1996; Molden, 1997). Water accounting (e.g. estimating water depletion, beneficial and non-beneficial uses, basin inflows and outflows) is used to study water management problems by quantifying the sources, sinks, and recycling processes of water in the basin (Perry, 1996; Styles and Burt, 1999). Eagleson (1978) put forth a general conceptual and analytic framework for a physically-based dynamic water balance to estimate the two major basin fluxes, evapotranspiration and yield, from precipitation and energy (potential evaporation) data. Basin yield is defined as the sum of surface water runoff and groundwater discharge from the basin. The separation of precipitation, p, into evapotranspiration, e, and yield, y, occurs within the soil root zone, which is the key component of water balance models (e.g. Eagleson, 1978; Milly, 1994; Salvucci and Entekhabi, 1994; Jothityangkoon et al., 2001; Scanlon et al., 2002; Farmer et al., 2003)

$$\frac{\partial S}{\partial t} = p - e - y \qquad (1)$$

where S is the moisture storage in the soil root zone. Soil root zone processes, and hence the water balance, are universally controlled by vegetation, soil properties, and topography.

The water balance dynamics of many highly developed semi-arid basins ('active basins') with intensive agriculture, such as the San Joaquin Valley, do not fit the framework used in most water balance models, including those for essentially undeveloped semi-arid basins ('passive basins') (e.g. Jothityangkoon et al., 2001). While Eq. (1) does in principle apply to these active basins, the conceptual interpretation of its components is distinctly different:

1. Significant sources of water (p in Eq. (1)) include not only precipitation but also irrigation with often regulated, naturally or artificially imported surface water, and irrigation with pumped groundwater (actual groundwater pumping).
2. Consumptive use (e in Eq. (1)) is dominated by production crop consumptive use, and smaller amounts of natural vegetation and urban consumptive use.
3. Runoff generation is negligible at the basin scale due to essentially flat topography (not to confuse with the transfer of allocated surface water to downstream basins).
4. Groundwater is a terminal storage reservoir with small or negligible discharges to neighboring groundwater basins or to surface water. Since runoff generation is negligible, the effective basin yield (y in Eq. (1)) consists of groundwater recharge from precipitation and irrigation ('diffuse recharge') and groundwater recharge from seepage losses in the surface water distribution system ('localized recharge'). Annual storage changes in such closed groundwater basins reflect the annual net groundwater use, which is the difference between groundwater pumping and groundwater recharge.

As noted by Burt (1999), water balances for active basins usually include 'closure terms': components of the water balance for which data are not available and any indirect estimates would be associated with large errors. In California, climate and landuse data for defining water demands, and surface water diversions are generally available at the basin scale. On the other hand, California property rights do not require land owners in non-adjudicated basins to measure their groundwater pumping rates and publically disclose them (Harter, 2003). Because basin-wide continuous monitoring of spatially distributed groundwater extraction is expensive, a severe lack of groundwater extraction data is not unique to California. As a result, the groundwater pumping component, difficult to estimate from indirect methods, becomes the default closure term in many water balances. However, a rigorous evaluation of this approach is lacking.

Groundwater recharge and pumping are spatially distributed basin-wide processes. Data availability on the various water balance components needed for closure at sufficiently detailed spatio-temporal scales is generally limited despite the array of data available in California from federal, state, and local agencies. Most input components must be estimated, extrapolated, or aggregated from sparsely distributed pointwise measurements of several spatio-temporally varying processes. Hence, for the sake of parsimony, the choice of process complexity in the conceptual model underlying a basin water balance computation is limited, even though highly complex advanced models are readily available to estimate individual flux components in these basins. Simple 'tipping bucket' or storage capacity-based distributed hydrologic process models of the water balance have been successfully applied to compute basin water yield (e.g. Milly, 1994; Atkinson et al., 2002; Jothityangkoon et al., 2001).

In this article, we develop a GIS-based distributed hydrologic modeling approach to assemble and process available input data, to compute the key components of the basin scale water balance model and to estimate and evaluate both, groundwater storage change and groundwater pumping as closure terms in the water balance model. Available input data have varying measurement support scales (scattered points, field, soil unit, district, or basin scale). Building a hydrologic GIS system allows us to readily interpolate and dis-aggregate the data to the local land unit scale (land unit and channel segment length scales of 10^1–10^3 m), which drives the water balance process (Milly, 1994). Spatially distributed monthly water fluxes are then computed using a storage capacity-based water balance model for the soil root zone associated with each land unit and for the channel segments. Results are re-aggregated to obtain the basin scale water balance components (Flerchinger et al., 1998). The approach constitutes a loosely-coupled water balance model (Tim, 1996) and a fully non-linear upscaling method of local water fluxes to the basin scale.

We apply the model to a semi-arid, intensively-irrigated agricultural (sub-)basin in the San Joaquin Valley (Fig. 1). The base period for the water balance is 1970–1999 with monthly time steps. This period

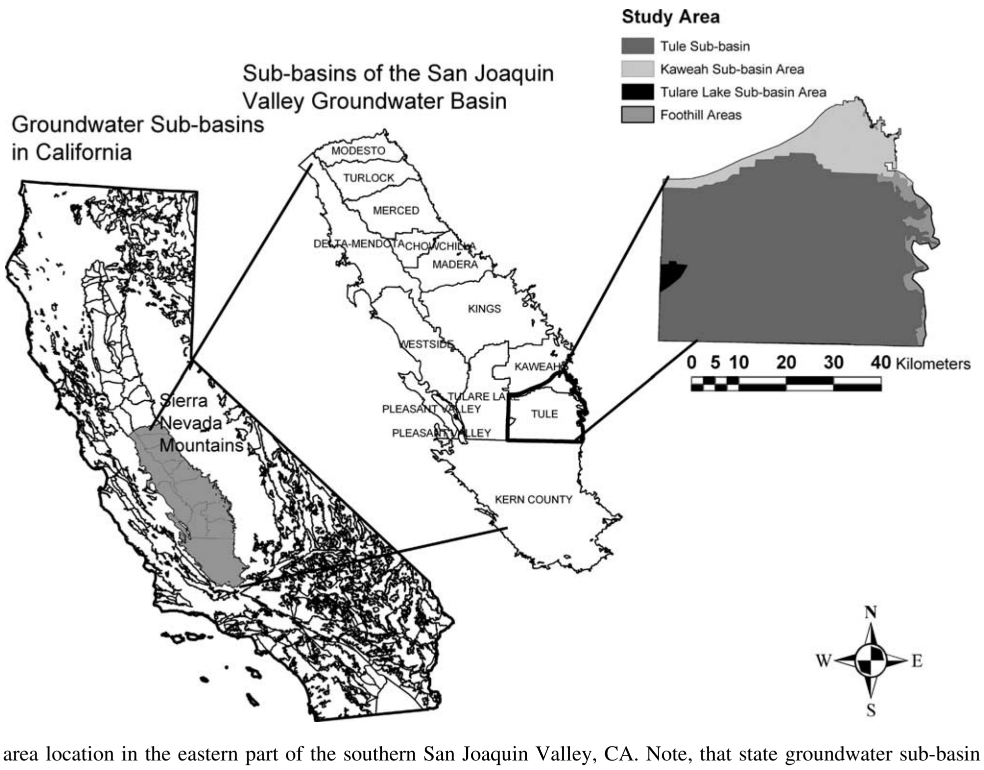

Fig. 1. Study area location in the eastern part of the southern San Joaquin Valley, CA. Note, that state groundwater sub-basin designations represent administrative boundaries. The study area itself represents a hydrologically quasi-closed groundwater basin.

includes several distinct hydrologic conditions (i.e. wet and dry periods) to adequately characterize groundwater use with respect to climate variability and surface water availability (Dutcher, 1972). Predicted basin groundwater storage changes are compared to those computed by the water-table fluctuation method (Sophocleous, 1991; Healy and Cook, 2002) as an independent check to the reasonableness of the water balance.

The organization of the article is as follows. A general description of the study area setting is followed by a description of the surface water supply system. Monthly distributed channel seepage losses, intra-district conveyance seepage losses, and district surface water deliveries are then computed. We then describe the land unit water balance model, which computes monthly recharge and groundwater pumping for individual crop and urban land units. The section provides a detailed description of the estimation methods used for the various components of the mass balance. The results and discussion highlight key findings with respect to the basin water balance components, particularly groundwater storage changes and groundwater pumping rates. We also analyze the sensitivity of the results to potential errors introduced by model assumptions and parameter uncertainties.

2. Setting

The study area is located in the southwest corner of Tulare County, California and is 2300 km^2 (229,384 ha) in size. The San Joaquin Valley groundwater basin is delineated into several basins of which the study area includes the entire Tule groundwater basin and smaller portions of the Kaweah and Tulare Lake basins (California Department of Water Resources (CDWR), 1980) (Fig. 1). Most of the study area is topographically flat with westward down-slopes of approximately 0.1%. Near the eastern boundary, alluvial terraces provide limited relief.

2.1. Climate

The area climate is semi-arid with most precipitation falling between November and March. From 1970 to 1999, the annual precipitation varied between 13 and 56 cm with a mean of approximately 23 cm. The annual pan evaporation rate, measured by CDWR at a southern San Joaquin Valley field station, ranged from 140 to 178 cm with a mean of 163 cm. Average monthly daytime temperatures vary from 13 °C in December to 36 °C in July. Average monthly night-time temperatures vary from 2 °C in December to 17 °C in July. The region experiences alternating periods of drought (1975–1977, 1987–1992) and wet conditions (1973, 1978, 1982–1983, 1995, 1998).

2.2. Water service areas

The study area is delineated into 26 surface water management districts: 21 irrigation, water, or public utility districts; two major cities; two private contractors; and one water company (Fig. 2). The remaining area is predominantly unincorporated agricultural land and native vegetation. Not all districts completely reside within the study area (Fig. 2).

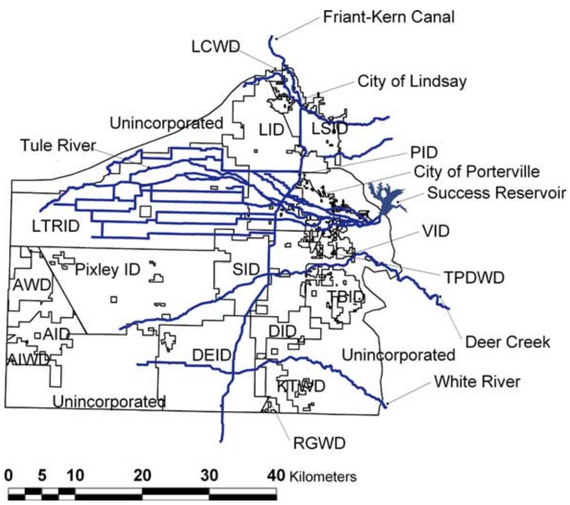

Fig. 2. Water service areas, and major natural and constructed surface water channels. Note that Lindmore Irrigation District (LID), Lindsay-Strathmore ID (LSID), Lewis Creek Water District (LCWD), and the City of Lindsay are located within the Kaweah basin. Small fractions of Angiola WD (AWD) and Alphaugh ID (AID) are located within the Tulare Lake basin. All other districts are partially or entirely located in the Tule groundwater basin.

2.3. Land use

Agriculture is the largest land use, comprising approximately 71% of the study area. Native vegetation and urban land use comprise 21 and 4% of the study area, respectively. The remaining 4% consists of surface water bodies, fallow lands, dairy feedlots, farm operations and farm buildings, and other miscellaneous land uses. Twelve crops account for 95% of the area under agricultural production. Cotton, grain and grass hay, citrus, and vineyards represent 21, 18, 16, and 12% of the total agricultural acreage, respectively. For the water balance, the study area is delineated into 9611 landuse units using a GIS coverage of a 1993 land use survey obtained from the CDWR (Fig. 3). Each land unit belongs to one of 61 differentiated land use types (California Department of Water Resources (CDWR), 1999).

2.4. Soils

Soil textures in the study area range from low-permeable clays to highly-permeable sands (Table 3). A digitized 1993 soils survey developed by the USDA Natural Resources Conservation Service was used to assign minimum, intermediate, and maximum available water contents (i.e. field capacity minus permanent wilting point) to each land unit in the digitized land use survey.

2.5. Groundwater

The aquifer system is comprised of late Tertiary and Quaternary age unconsolidated continental and alluvial deposits with interbedded zones of lacustrine and marsh deposits (Hilton et al., 1963; Croft, 1969; Lofgren and Klausing, 1969). In the western part of the study area, the aquifer system is divided into three

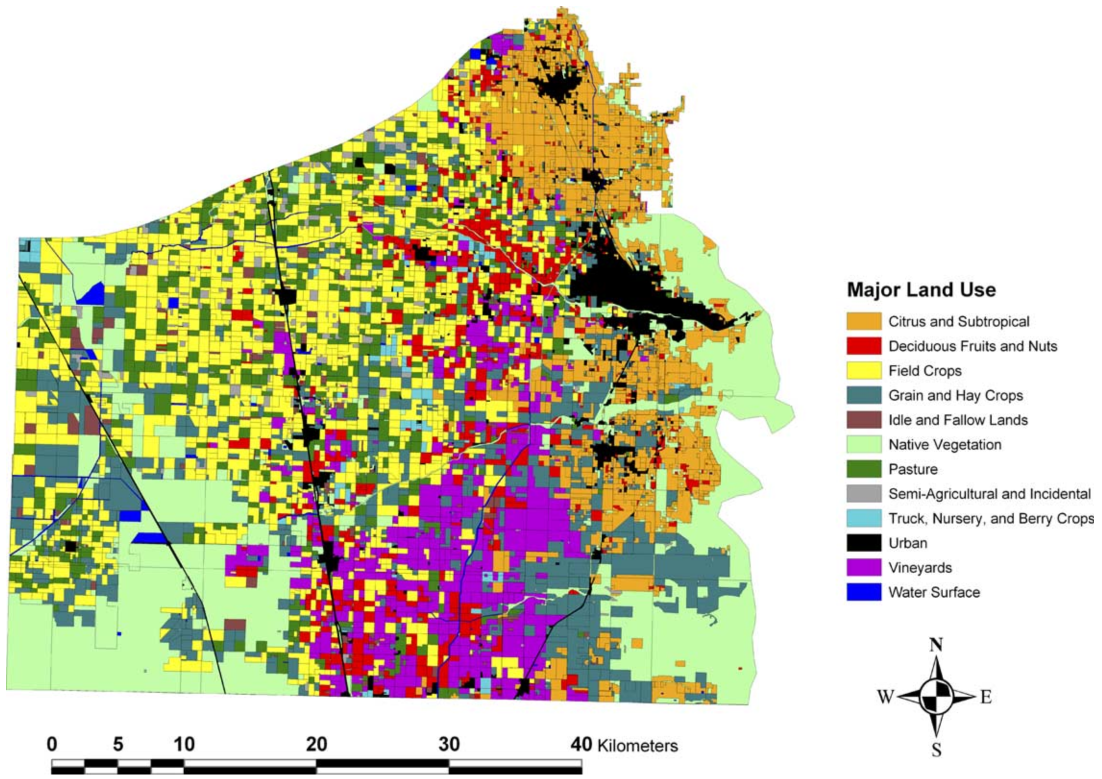

Fig. 3. Major land use from a 1993 land-use survey.

hydrogeologic units: an unconfined aquifer, an underlying aquitard (i.e. the Corcoran Clay Member of the Tulare Formation), and a confined aquifer below the aquitard. In the eastern part of the study area, alluvial and continental deposits form a single thick unconfined to semi-confined aquifer. The bottom boundary of the aquifer system is formed by consolidated marine rocks of late Pliocene age. Near the eastern border, the unconsolidated deposits of the aquifer unconformably overly the Sierra Nevada mountain range basement complex of metamorphic and igneous rocks of very low permeability. Water level depths range from less than 1 m in the immediate vicinity of some river channels to over 50 m in the most overdrafted portions of the basin. Since the depth to groundwater generally exceeds 5–10 m, interactions of the water table with the root zone are considered negligible.

While the study area is hydrogeologically unbounded at its northern, western, and southern boundaries, its extent was defined such that long-term hydraulic gradients at those boundaries are negligible and any groundwater fluxes computed using reasonable permeability values are small in comparison to the total changes in storage due to vertical stresses applied to the entire study area (e.g. groundwater pumping, evapotranspiration, applied surface water, channel seepage). Hence, for purposes of computing a water balance, it is appropriate to assume that the groundwater basin underneath the study area behaves as a closed system.

3. Water balance model

The study area hydrologic system is conceptualized as consisting of three compartments: (1) a surface water supply system; (2) a soil root zone with associated land uses; and (3) an underlying aquifer system. The surface water supply system is the network of natural and constructed channels whose function is to import surface water into the study area and to deliver it to contracting districts and their individual land units. Incidental and intentional seepage losses from surface water channels and conveyance structures are a significant source of groundwater recharge. The soil root zone of each land unit together with its associated land use controls water demands (surface water deliveries and groundwater pumping), consumptive use, and groundwater recharge. The deep vadose zone between the root zone and the groundwater table is considered to be at steady-state (negligible interannual storage changes).

3.1. Surface water supplies

Surface water features include natural and constructed channels (Fig. 2) and a number of smaller recharge basins, which are typically incorporated into the channel network. The region has two external sources of surface water: (1) The Tule River and a small number of other western flowing streams and creeks (natural channels) carry runoff from adjacent Sierra Nevada mountain catchments to the east into the study area. The Tule River is regulated through Success Reservoir just outside the basin boundaries. (2) Significant surface water imports are conveyed via the Friant-Kern Canal, which traverses the study area north to south. The Friant-Kern Canal is part of the federally-operated Central Valley Project (CVP), a massive inter-regional water development project.

From these sources, water distribution is achieved through a system of hydraulically inter-connected canals and ditches (constructed channels). Some natural channels are used as an intrinsic part of the highly controlled distribution system.

We distinguish three types of natural and constructed surface water channels: (1) source channels; (2) diversion channels; and 3) distribution channels. Source channels import developed surface water or natural runoff into the study area and deliver it directly to the contracting districts, release it to diversion channels for later delivery, or allow it to infiltrate through its channel bed into the subsurface as recharge. Diversion channels convey the surface water releases from the source channels to the borders of contracting districts or redirect it to other diversion channels for later delivery. Source and diversion channels constitute the inter-district surface water conveyance network and are considered explicitly as spatially distributed sources of recharge (Fig. 2). In contrast, distribution channels deliver surface water from district boundaries to individual land units within each district and constitute the intra-district surface water distribution system. They are not explicitly modeled as individual canals. Distribution is driven by the water demands of individual land units within a district and surface water availability; while intra-district seepage losses are distributed uniformly across the agriculturally developed portions of a district.

Accordingly, we obtain three designations of surface water: (1) surface water diversions (2) surface water deliveries; and (3) applied surface water. Diversions are an amount of surface water released from the source channels for delivery to the receiving districts. Deliveries are the actual amount of the diversions received at the district borders after accounting for conveyance seepage losses in the inter-district network. The applied surface water is the amount of surface water applied to the land units within each district after accounting for conveyance seepage losses in the intra-district distribution systems.

A monthly mass balance is computed for inter-district channel segments based on reported gauging and diversion information. Monthly surface water diversions from the Friant-Kern Canal and the Tule River were provided by the US Bureau of Reclamation (USBR) and the Tule River Association, respectively. Other reported smaller sources of surface water for districts on the western side of the study area include the California State Water Project and the Kings River. Natural runoff data for the Tule River, Deer Creek, and White River were obtained from the US Geological Survey (USGS), which operates gauging stations on these streams.

3.2. Seepage losses

3.2.1. Inter-district channel seepage losses

The Tule River and Deer Creek convey flows which exceed the measured amounts of surface water diverted from them or into them for delivery to receiving districts. In these two channels, gauged flows in excess of the total diversions are assumed to be lost as natural channel seepage. The White River is not known to be a diversion channel; consequently, all of its gauged flows are assumed to recharge the aquifer system as natural channel seepage. Since the major natural channels are predominantly ephemeral (Sophocleous, 2002), inter-district channel seepage is assumed to directly recharge the unconfined aquifer. The Friant-Kern Canal is concrete-lined and is considered to have negligible seepage losses.

For districts which do not intersect the source channels, seepage losses in their unlined diversion channels are estimated as 1.6% of the monthly surface water diversion per kilometer of that channel. This seepage rate was adapted from an engineering study of unlined canal seepage in Tulare Irrigation District (ID) (CH2MHILL Inc., 1998), which is in a hydrogeologically similar area immediately to the north of the study area. Districts with source channels inside their boundaries have no inter-district seepage losses.

3.2.2. Intra-district seepage loss

Intra-district seepage losses for districts with predominantly unlined distribution systems are estimated as the difference between the delivered surface water and the applied surface water (i.e. farm head-gate deliveries) to the individual farms. We lack both farm head-gate delivery data and an explicit characterization of the channels constituting the district distribution systems. Instead, we rely on estimates of intra-district seepage losses obtained from studies performed in hydrogeologically similar districts located along the eastside of the southern San Joaquin Valley.

Two independent groundwater studies performed in the neighboring Kaweah basin north of the study area estimated intra-district seepage losses that ranged from 10 to 40% of surface water deliveries (Bookman and Edmonston Inc., 1972; Fugro West Inc., 2003). Another groundwater study for the Alta ID in the Kings basin north of the Kaweah basin estimated seepage losses to be approximately 25% of district deliveries (Kings River Conservation District, 1992). The seepage losses within each district vary due to differences in district area and shape, distribution system type, spatial distribution of subsurface sediments, and management of canal operations.

For districts known to have predominantly piped distribution systems, we assumed a nominal seepage loss of 2% of diversions. Districts known to have predominantly unlined distribution systems were assigned seepage loss rates relative to their total acreage. For the two largest districts (Lower Tule River ID, Pixley ID), we assume an intra-district seepage loss of 25% of diversions. For the four smaller districts (Alpaugh ID, Atwell Island Water

District, Angiola ID, Porterville ID), we assume a seepage loss of 15% of diversions. The remaining districts are piped systems. The uncertainty about unlined distribution system seepage rates and its effects on the basin water balance are evaluated later through the sensitivity analysis.

3.3. Crop and urban water balances

For vegetative land uses, the change in storage of available water in the effective soil root zone for the jth land unit (Fig. 3) during the ith month is computed by an explicit form of Eq. (1)

$$\Delta D_{\mathrm{aw}(i,j)} = P_{(i,j)} + S_{w(i,j)} + G_{w(i,j)} - \mathrm{ET}_{(i,j)} - D_{p(i,j)} \quad (2)$$

where D_{aw} is the depth of available water in the effective soil root zone (L), P is precipitation (L), S_w is applied surface water (L), G_w is the surface applied pumped groundwater (L), ET is evapotranspiration (L), and D_p is the percolation from the soil root zone into the water table (L). The effective soil root zone is defined to be that portion of the soil root zone from which the crop extracts the majority of its water (Evans et al., 1996). We assume that no lateral flow occurs between the soil root zones of adjacent land units due to the large size of individual land units relative to the thickness of the root zone and due to the lack of significant topographic gradients.

For urban land uses, the monthly soil root zone storage change of each land unit is

$$\Delta D_{\mathrm{aw}(i,j)} = P_{(i,j)} + S_{w(i,j)} + G_{w(i,j)} - M_{(i,j)} - D_{p(i,j)} \quad (3)$$

where M is the urban water demand (L). The inputs and outputs for Eq. (3) are the same as in Eq. (2) except that the water demands of urban land units are calculated differently than ET. Changes in available soil root zone water are computed by estimating the components on the right-hand sides of Eqs. (2) and (3).

A similar threshold water holding capacity (tipping bucket) model was successfully used for estimating basin yield across the eastern US (Milly, 1994) and its use confirmed in an evaluation of various conceptual modeling approaches by Jothityangkoon et al. (2001), Atkinson et al. (2002) and Farmer et al. (2003).

3.3.1. Crop water demands

In this article, unless specified otherwise, we refer to any vegetative land use as a 'crop'. Two concepts are used for calculating the water needs of a cropped land unit: (1) crop evapotranspiration and (2) the theoretical applied water demand. Crop evapotranspiration is defined as the cumulative amount of water transpired by the crop, retained in its plant tissue, or evaporated from adjacent soil surfaces during its growing season. The evapotranspiration is given by

$$\mathrm{ET}_{(i,j)} = 0.95 k^*_{c(i,j)} \mathrm{ET}_{0(i)} \quad (4)$$

Table 1
For the 12 major crops: reported value or range of annual ET; estimated range and average annual ET from 1970 to 1999; and maximum root depths for some crops

Major crop	Reported annual ET (cm)	Estimated range of annual ET 1970–1999 (cm)	Estimated average annual ET 1970–1999 (cm)	Maximum root depth (m)
Cotton	69.6–90.2	66.5–86.9	78.7	1.19
Grain and grass hay	38.1–43.2	28.2–44.2	38.4	1.19
Citrus	73.4–96.8	71.9–97.5	88.4	1.4
Vineyards	60.5–79.5	59.2–77.2	70.1	1.49
Alfalfa	103.9–135.9	100.6–135.6	124.0	1.49
Grain and corn	91.4	72.6–97.3	88.4	–
Olives	99.6	82.0–111.0	100.8	1.49
Almonds	98.3	78.7–104.9	95.0	1.8
Corn	69.6	60.5–77.5	69.9	1.31
Plums	85.9–110.2	78.7–104.9	95.0	–
Walnuts	106.2	85.6–112.8	102.1	2.01
Pistachios	103.4	84.1–109.2	98.6	–

where ET_0 is the evapotranspiration of the grass reference crop (L). It is obtained from climate data using the Penman–Monteith method (Allen et al., 2001; California Department of Water Resources (CDWR), 2000). The modified crop coefficient k_c^* is defined as

$$k^*_{c(i,j)} = k_{c(i,j)} d_1 \quad (5)$$

where k_c is the crop coefficient (Allen et al., 2001) and d_1 is a landuse change adjustment factor explained below. The factor of 0.95 in Eq. (4) imposes a 5% reduction in the estimated evapotranspiration for each land unit. The reduction accounts for areas within the digitized land units occupied by access roads, uncropped field margins, and other areas that do not have an applied water demand but have not been explicitly delineated.

A number of resources was available from which to choose monthly crop coefficients. For the 12 major crops which constitute 95% of the agricultural area, experimentally derived values or ranges of annual ET in the region were obtained from publications and through personal communications with agricultural industry professionals (Table 1). Crop coefficients were chosen and in some cases adjusted such that the computed average annual ET for each crop from 1970 to 1999 was similar to its derived value or within the range of the values given in Table 1. Monthly crop coefficients for citrus, cotton, field corn, alfalfa, and vineyards were adapted from Letey and Vaux (1984). Crop coefficients for olives, plums, almonds, walnuts, pistachios, and grain and grass hay were adapted from Goldhammer and Snyder (1989). Grain and corn refers to the double cropping of silage corn and winter grain. All other crop coefficients were obtained from Naugle (2001). The estimated ranges and averages of annual ET for the 12 major crops from 1970 to 1999 are presented in Table 1.

The crop type of individual land units generally changes over time. However, the overall cropping pattern in the study area has been relatively stable over the 1970–1999 period. Since landuse maps are not available to track those changes, we chose a lumped, uniformly applied adjustment of crop ET in Eq. (4) to reflect annual changes in reported county areas of the 12 major crops. For each year, the total acreage of each crop is multiplied by a typical value of

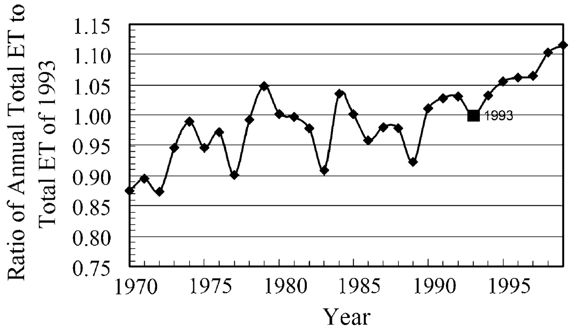

Fig. 4. Factors used to adjust the monthly consumptive use of each land unit for annual changes in crop acreage.

its annual ET to produce an estimate of its total ET demand in the county. The estimated annual ET of the 12 crops are summed to produce a total annual ET for the entire county. The total county ET of each year is then divided by the total ET calculated for 1993, the year of the land use survey (Fig. 3). The resultant ratios are the values of d_1 in Eq. (5) and are plotted in Fig. 4. The annual adjustment assumes that relative land use changes in the study area are directly proportional to those of the entire county. This is a reasonable assumption given that the study area encompasses approximately half of the agricultural production area of Tulare County with a cropping pattern that is very similar to that in the other half.

The theoretical applied water demand, w^*, (L) accounts for ET and inefficiencies in irrigation

$$w^*_{(i,j)} = \left[\frac{ET_{(i,j)} - D_{aw(i,j)}}{IE} \right] \quad (6)$$

where IE is the crop irrigation efficiency (Tables 2 and 4). Furthermore, not all crops receive surface applications sufficient to satisfy their theoretical applied water demand. The amount of Eq. (6) satisfied by surface water or pumped groundwater is a function of land use

$$w_{(i,j)} = w^*_{(i,j)} \lambda_j, \qquad \lambda_j = \begin{cases} 1, & u_j = 1\text{–}45, \\ 0.25, & u_j = 46\text{–}50 \\ 0, & u_j = 56 \text{ or } 60 \end{cases} \quad (7)$$

where $w_{(i,j)}$ is the adjusted applied water demand (L) and u_j is the land use identification number of the jth land unit (Table 2). In Eq. (7), crops used primarily for food and fiber ($u_j = 1\text{–}45$) receive 100% of their

Table 2
Land use, identification number (ID), acreage, percentage of study area, percentage of cropped area, and potential irrigation system used by the crop

Land use	ID, *u*	Area (ha)	Study area (%)	Crop area (%)	Irrigation system
Grapefruit	1	97.6	<0.1	<0.1	Trickle-point source emitters
Lemons	2	639.9	0.3	0.4	Trickle-point source emitters
Oranges	3	20,217.8	8.8	12.4	Trickle-point source emitters
Avocados	4	130.8	0.1	0.1	Trickle-point source emitters
Olives	5	4561.0	2	2.8	Trickle-point source emitters
Misc. subtropical fruits	6	21.4	<0.1	<0.1	Trickle-point source emitters
Kiwis	7	510.0	0.2	0.3	Trickle-point source emitters
Eucalyptus	8	27.6	<0.1	<0.1	Trickle-point source emitters
Apples	9	227.3	0.1	0.1	Trickle-point source emitters
Apricots	10	71.5	<0.1	<0.1	Trickle-point source emitters
Cherries	11	1.7	<0.1	<0.1	Trickle-point source emitters
Peaches	12	457.4	0.2	0.3	Trickle-point source emitters
Pears	13	4.0	<0.1	<0.1	Trickle-point source emitters
Plums	14	2858.4	1.2	1.7	Trickle-point source emitters
Prunes	15	323.6	0.1	0.1	Trickle-point source emitters
Misc. deciduous fruits	17	851.1	0.4	0.6	Trickle-point source emitters
Almonds	18	4054.8	1.8	2.5	Trickle-point source emitters
Walnuts	19	2690.7	1.2	1.7	Surface-furrow
Pistachios	20	1604.8	0.7	1.0	Surface-furrow
Cotton	21	33,859.0	14.8	20.8	Surface-furrow
Safflower	22	1393.8	0.6	0.8	Surface-furrow
Flax	23	11.3	<0.1	<0.1	Surface-furrow
Sugar beets	24	417.4	0.2	0.3	Surface-furrow
Corn	25	12,701.5	5.5	7.7	Surface-furrow
Sudan	26	356.3	0.2	0.3	Surface-furrow
Dry beans	27	1073.2	0.5	0.7	Surface-furrow
Misc. field crops	28	4.1	<0.1	<0.1	Surface-furrow
Sunflowers	29	10.2	<0.1	<0.1	Surface-furrow
Grain and grass hay	30	29,667.3	12.9	18.1	Surface-border
Grain and corn	31	3930.8	1.7	2.4	Surface-border/furrow
Alfalfa hay	32	17,553.3	7.7	10.8	Surface-border
Green beans	33	338.1	0.1	0.1	Sprinkler-solid set
Cole crops	34	413.1	0.2	0.3	Sprinkler-solid set
Lettuce	35	60.5	<0.1	<0.1	Sprinkler-solid set
Melons	36	451.6	0.2	0.3	Sprinkler-solid set
Onions	37	155.6	0.1	0.1	Sprinkler-solid set
Tomatoes	38	96.3	<0.1	<0.1	Sprinkler-solid set
Flowers and nursery	39	63.4	<0.1	<0.1	Sprinkler-solid set
Misc.s truck crops	40	107.1	<0.1	<0.1	Sprinkler-solid set
Peppers	41	133.2	0.1	0.1	Sprinkler-solid set
Lawn areas	42	90.4	<0.1	<0.1	Sprinkler-traveling gun
Golf courses	43	3.2	<0.1	<0.1	Sprinkler-traveling gun
Cemeteries	44	32.7	<0.1	<0.1	Sprinkler-traveling gun
Vineyards	45	20,295.2	8.8	12.4	Trickle-point source emitters
Mixed pasture	46	764.3	0.3	0.4	Surface-basin
Native pasture	47	244.5	0.1		Surface-basin
Farmsteads	48	786.7	0.3		
Dairies	49	2221.6	1.0		
Poultry Farms	50	135.8	0.1		
Fruit and vegetable canneries	51	453.0	0.2		

(*continued on next page*)

Table 2 (*continued*)

Land use	ID, u	Area (ha)	Study area (%)	Crop area (%)	Irrigation system
Misc. high water use	52	10.7	<0.1		
Sewage treatment plants	53	13.2	<0.1		
Fallow land	54	2502.4	1.1		
Idle land	55	981.8	0.4		
Native vegetation	56	47,687.0	20.8		
Surface water	57	1916.0	0.8		
Feed lots-livestock	58	319.6	0.1		
Urban (unspecified)	59	7478.4	3.3		
Cemeteries—not irrigated	60	5.1	<0.1		
Urban vacant (unspecified)	61	1286.8	0.6		

theoretical applied water demand. The water use in semi-agricultural and incidental to agriculture land uses ($u_j = 46–50$), such as native pasture, dairies, feedlots, and farmsteads, has negligible effect on the basin water balance computation due to their limited acreage (less than 2%) within the study area. They are assumed to use a nominal 25% of the equivalent ET_0 to account for animal water use, pond evaporation, and irrigated lawns, which are typical of those landuses. Native vegetation ($u_j = 56$) and non-irrigated cemeteries ($u_j = 60$) do not receive any irrigation water; their evapotranspiration rates are equal to those of pasture, but cannot exceed available soil moisture contents. Their sole source of moisture is precipitation.

3.3.2. Urban water demands

The applied water demands of urban municipal and industrial land units ($u_j = 51$, 52, 53, and 59) are estimated using water influent and effluent data for the city of Porterville, the largest urban area in the basin. From 1995 to 1999, the average net water use, m_P, (L^3) for the ith calendar month in Porterville is

$$m_{P(i)} = m_{I(i)} - m_{E(i)}, \qquad i = 1, \ldots, 12 \tag{8}$$

where m_I is the monthly average total water influent (L^3) and m_E is the monthly average total water effluent used for recharge (L^3). The monthly average net water use per hectare, M, (L) is computed by

$$M_{P(i)} = \frac{m_{P(i)}}{a_P}, \qquad i = 1, \ldots, 12 \tag{9}$$

where a_P is the area of Porterville in 1995.

The applied water demand, w, is equated to the average net water use per acre

$$w_{(i,j)} = M_{P(i)} \tag{10}$$

for $u_j = 51$, 52, 53, and 59.

For all urban land units, most of which encompass entire communities, we assume that none of the applied water demand in Eq. (10) is satisfied by soil moisture. Hence, all precipitation on urban land is assumed to become recharge either by direct infiltration or by recharge through unspecified recharge basins distributed throughout urban areas. Except for land units residing within the city of Lindsay, the water demand in Eq. (10) is satisfied exclusively with pumped groundwater. We only consider the net consumptive urban water use (Eq. (8)), which includes consumptive use of lawns and parks within urban areas. Wastewater effluent is typically recharged back to groundwater through recharge basins within the urban area and is therefore not explicitly considered. The city of Lindsay has an existing CVP water contract and provides intentional recharge of excess surface water supplies as computed by the surface water supply model.

3.3.3. Others water demands

Other land uses such as ‘fallow land’ ($u_j = 54$), ‘idle land’ ($u_j = 55$), ‘livestock feedlots’ ($u_j = 58$), and ‘unspecified urban’ ($u_j = 61$) are assigned an adjusted applied water demand of zero. As a result, they do not receive any surface water or groundwater. These land units may, however, experience bare-soil evaporation. Therfore, their ET is equated to the available soil water content, where the sole source of moisture is precipitation.

3.3.4. Precipitation

The basis for estimating the spatio-temporal distribution of precipitation is a sparse network of three gauging stations (temporal distribution) and a regional isohyet map developed by CDWR (spatial distribution). We assume that overland runoff and non-beneficial losses of precipitation by evaporation are negligible and that 100% of the precipitation infiltrates into the soil root zone. Overland runoff is neglected due to the relatively flat topography of much of the study area. While runoff may occur locally after heavy winter rains, it is typically collected and recharged to groundwater within the field boundaries or not far thereof. Transfer of runoff at the district and basin scale is therefore negligible. While local (land unit scale) runoff from precipitation is unavailable to meet crop water demand its effect is also negligible because local runoff is limited to winter months when ET is very limited and soils are near field capacity.

Evaporation of precipitation is implicitly accounted for by crop ET. It is negligible during the winter months. The precipitation for the ith month for the jth land unit is then estimated as

$$P_{(i,j)} = P_{a(j)}\left[\frac{p_{0(i)}}{P_0}\right] \tag{11}$$

where $P_{a(j)}$ is the average annual precipitation for the jth land unit (L), $p_{0(i)}$ is the ith monthly reference precipitation (L), and P_0 is the average annual precipitation at the reference location (L). Monthly records of precipitation from 1970 to 1999 are not always complete at all gauging stations. Consequently, the time series of $p_{0(i)}$ was developed by concatenating the monthly time series for the following periods: 1970–1973 (Vestal station: Southern California Edison Co.), 1974–1995 (Tulare ID station: CDWR), and 1996–1999 (Visalia station: National Weather Service). The spatial distribution of $P_{a(j)}$ is defined by the isohyet map.

3.3.5. Surface water allocation and groundwater pumping

Solving Eqs. (7) and (10) for each land unit, the total adjusted applied water demand of the kth district, $W_{(i,k)}$, (L^3) is then computed using

$$W_{(i,k)} = \sum_{j=1}^{n(k)} w_{(i,j)} a_{(j)} \gamma_{(j)},$$
$$\gamma_{(j)} = \begin{cases} 0, & u_j = 51\text{–}53,\ 59 \text{ and } k \neq 4, \\ 1 & \text{otherwise} \end{cases} \tag{12}$$

where a is the land unit area and $n(k)$ is the number of land units in the kth district. Eq. (12) sums the applied water demands of those land units which are eligible to receive surface water allocations. Not included in Eq. (12) are those urban land units which rely solely on groundwater pumping.

If the total available surface water to the kth district during the ith month, $S_{d(i,k)}$, is greater than or equal to the total applied water demand (i.e. $S_{d(i,k)} - W_{(i,k)} \geq 0$) then each land unit receives an allotment of surface water equal to its applied water demand, $w_{(i,j)}$. The remaining surplus surface water, $S_{d(i,k)} - W_{(i,k)}$, is distributed uniformly over the land units in the kth district for which $1 \leq u_j \leq 45$ or for urban land units in the city of Lindsay using

$$s'_{(i,j)} = \frac{(S_{d(i,k)} - W_{(i,k)})}{A^*_{(k)}} \tag{13}$$

where $s'_{(i,j)}$ is the applied surplus surface water (L) and

$$A^*_{(k)} = \sum_{j=1}^{n(k)} a_{(j)} \gamma_{(j)},$$
$$\gamma_{(j)} = \begin{cases} 1, & u_j = \{1-45\} \text{ or } \{51, 52, 53, \\ & \text{or } 59 \text{ and } k = 4\} \\ 0, & \text{otherwise} \end{cases} \tag{14}$$

is the total area of land units in the kth district (L^2) eligible to receive 100% of the available surface water to them. The total allotted surface water for the jth land unit is

$$S_{w(i,j)} = \begin{cases} s'_{(i,j)} + w_{(i,j)}, & 1 \leq u_j \leq 45 \\ w_{(i,j)}, & 46 \leq u_j \leq 50 \end{cases} \tag{15}$$

If the total available surface water is less than the total applied water demand (i.e. $S_{d(i,k)} - W_{(i,k)} < 0$) then the fractional amount of the total applied water demand which will have to be satisfied by groundwater pumping is

$$c_{(i,k)} = (W_{(i,k)} - S_{d(i,k)})/W_{(i,k)} \tag{16}$$

The groundwater pumping demand for the jth land unit becomes

$$G_{w(i,j)} = c_{(i,k)} w_{(i,j)} \tag{17}$$

and its allotment of surface water is

$$S_{w(i,j)} = (1 - c_{(i,k)}) w_{(i,j)} \tag{18}$$

3.3.6. Soil root zone percolation

Percolation from the soil root zone to the water table, D_p, (L) is calculated using a simple tipping bucket model given by

$$D_{p(i,j)} = D_{\mathrm{aw}(i-1,j)} + P_{(i,j)} + S_{w(i,j)} - \mathrm{ET}_{(i,j)} - d^{\max}_{\mathrm{aw}(j)} b_{\mathrm{s}} \tag{19}$$

where b_{s} is the effective soil root zone thickness and $d^{\max}_{\mathrm{aw}}$ is the maximum available water per unit volume, defined as the difference between the field capacity and the permanent wilting point (Martin et al., 1991). The initial soil moisture is estimated as 50% of the soil available water, a commonly used value of allowable depletion (Martin et al., 1991).

Substitution of Eqs. (4), (11), (17)–(19) into Eq. (2) yields the soil root zone storage change during the ith month for the jth cropped land unit. Substitution of Eq. (9) into Eq. (3) yields the corresponding storage change for urban land units.

3.4. Basin scale aquifer storage change

The net aquifer recharge, Q^T, is computed by subtracting the groundwater pumping from the aquifer recharge of each land unit, aggregating these differences to the basin scale, and then adding the contribution to aquifer recharge from channel seepage

$$Q^T_{(i)} = q^T_{s(i)} + \sum_{j=1}^{n} a_{(j)} [D_{p(i,j)} - G_{w(i,j)}] \tag{20}$$

where q^T_s is the combined total seepage from the inter-district surface water network and the intra-district distribution systems (m^3), and n is the total number of land units. The monthly net aquifer recharge can be summed to produce a cumulative annual storage change from the 1970 baseline year to each fiscal water year from 1971 to 1999.

3.5. Water-table fluctuation method

As an independent check, we compare the computed groundwater storage change in Eq. (20) with that obtained by the water-table fluctuation (WTF) method (Healy and Cook, 2002). The WTF method computes cumulative annual groundwater storage changes in the unconfined aquifer from 1970 to 1999 using annually measured hydraulic heads from production wells and point estimates of specific yield (available from CDWR). Hydraulic heads represent spring water levels since they are measured annually between early January to late March. The specific yield values represent a depth of 100 m in the unconfined aquifer. For the WTF method, a grid of uniformly sized squared grid cells was superimposed on a GIS coverage of the study area. Cell-length is $\Delta x = \Delta y = 1000$ m. Scattered point estimates of specific yield were then interpolated to the grid cell centroids. A set of spatially distributed hydraulic head measurements for each year were also interpolated to the grid cell centroids. The cumulative groundwater storage change in ijth cell from 1970 to the year l was estimated using

$$\Delta s^l_{ij} = (h^l_{ij} - h^{1970}_{ij}) S_{y_{ij}} \Delta x\, \Delta y, \quad l = 1971, \ldots, 1999 \tag{21}$$

where h^{1970}_{ij} is the spring-measured hydraulic head of 1970, h^l_{ij} is the hydraulic head of the year l, and $S_{y_{ij}}$ is the unconfined aquifer specific yield. The cumulative storage change in the unconfined aquifer from 1970 to the year l is

$$\Delta S^l = \sum_{i=1}^{n_x} \sum_{j=1}^{n_y} \Delta s^l_{ij} \tag{22}$$

where n_x is the number of grid cells in the $x$direction and n_y is the number of cells in the $y$direction.

The WTF method neglects storage changes in the confined aquifer system. This simplification is justified since the storage coefficient of the confined aquifer system is nearly three orders of magnitude smaller than the specific yield of the unconfined aquifer. Hence, even if potentiometric water level changes in the confined aquifer significantly exceeded those in the unconfined aquifer, the effective confined

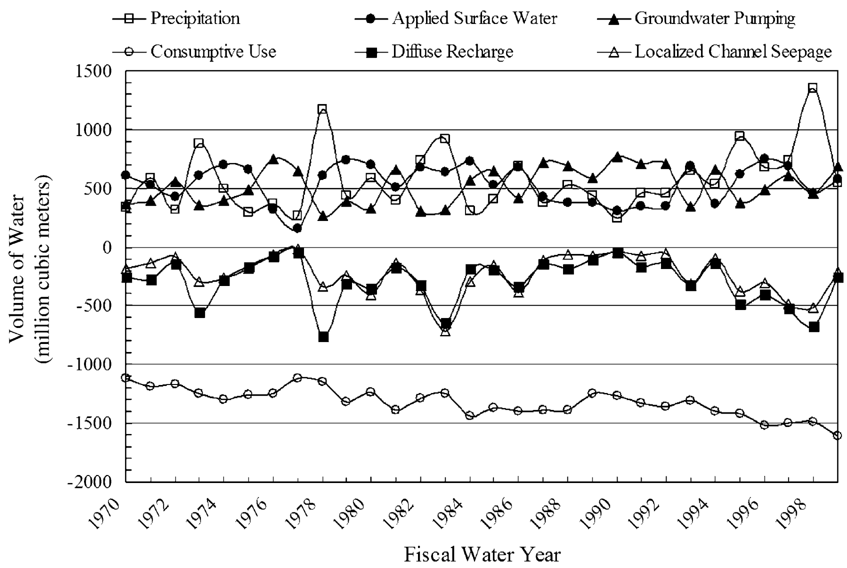

Fig. 5. Annual water balance components of the study area from 1970 to 1999. Fluxes to the land surface are positive, fluxes away from the land surface are negative.

water storage change would be but a small fraction of the changes in unconfined groundwater storage.

4. Results and discussion

The following sections summarize the annual and intra-annual water balance components obtained and discuss the validity and sensitivity of the estimated annual groundwater storage changes and groundwater pumping rates.

4.1. Water balance components

The annual water balance components for the study area are displayed in Fig. 5. The total annual consumptive use ranged from $1138 \times 10^6\ m^3$ in 1970 to $1635 \times 10^6\ m^3$ in 1999. Urban consumptive use constitutes only $23 \times 10^6\ m^3$ of this total. The upward trend in consumptive use is due to a 20% increase in the amount of land put into agricultural production from 1970 to 1999 (Fig. 4), while inter-annual fluctuations are due to climate variations as represented by ET_0.

Localized recharge from the combined inter-district and intra-district channel seepage (‘localized channel seepage’ in Fig. 5) averaged $234 \times 10^6\ m^3$/year but varied over almost two orders of magnitude from a practically negligible $18.5 \times 10^6\ m^3$ in 1977 to $718 \times 10^6\ m^3$ in 1983. Intra-district and inter-district channel seepage accounted for an average of 39 and 61% of the total localized recharge, respectively, over the 30-year period. In wet years, inter-district seepage generally accounts for more than 80% of localized recharge. Most of the additional wet-year seepage represents mountain front recharge derived from upstream runoff in the Tule River, Deer Creek, and White River, in part supported by intentional recharge facilities along these rivers.

Diffuse recharge from deep percolation of applied water and precipitation is generally of the same magnitude as localized recharge. On average, it accounts for 57% of the annual total recharge. Annual variations in diffuse and localized recharge are highly correlated due to the high correlation between valley floor precipitation and runoff from the eastern foothills and the Sierra Nevada mountains. Wet years are associated with high natural channel surface water inflows and correspondingly increased channel seepage in them. In addition, high precipitation results in increased diffuse recharge compared to normal years (e.g. 1973, 1978, 1983, 1998).

The three water sources, average annual precipitation, applied surface water, and groundwater pumping—are of similar magnitude with an annual average of 571×10^6, 538×10^6, and $522 \times 10^6\ m^3$, respectively. However, precipitation varied widely between a minimum of $245 \times 10^6\ m^3$ in 1991 and a maximum of $1345 \times 10^6\ m^3$ in 1998. Applied surface water varied between $160 \times 10^6\ m^3$ in 1977 and $747 \times 10^6\ m^3$ in 1996, while groundwater pumping ranged from $266 \times 10^6\ m^3$ in 1978 to $768 \times 10^6\ m^3$ in 1990. As expected, annual applied surface water and groundwater pumping are negatively correlated, with increased pumping occuring in years of reduced surface water availability.

For each water year, CDWR assigns a hydrologic classification index which is a measure of the relative amount of annual unimpaired runoff from the major rivers which drain into the San Joaquin Valley river basin. The indices representing lowest to highest runoff are 'Critical', 'Dry', 'Below Normal', 'Above Normal', and 'Wet'. In Critical and Dry years, groundwater pumping accounted for 55–80% of the total applied water and in Above Normal and Wet years it accounted for 30–54%. Only 1971 was classified as a Below Normal year during which pumping accounted for 43% of the total applied water.

The intra-annual water balance components vary significantly between different water year indices (Fig. 6).

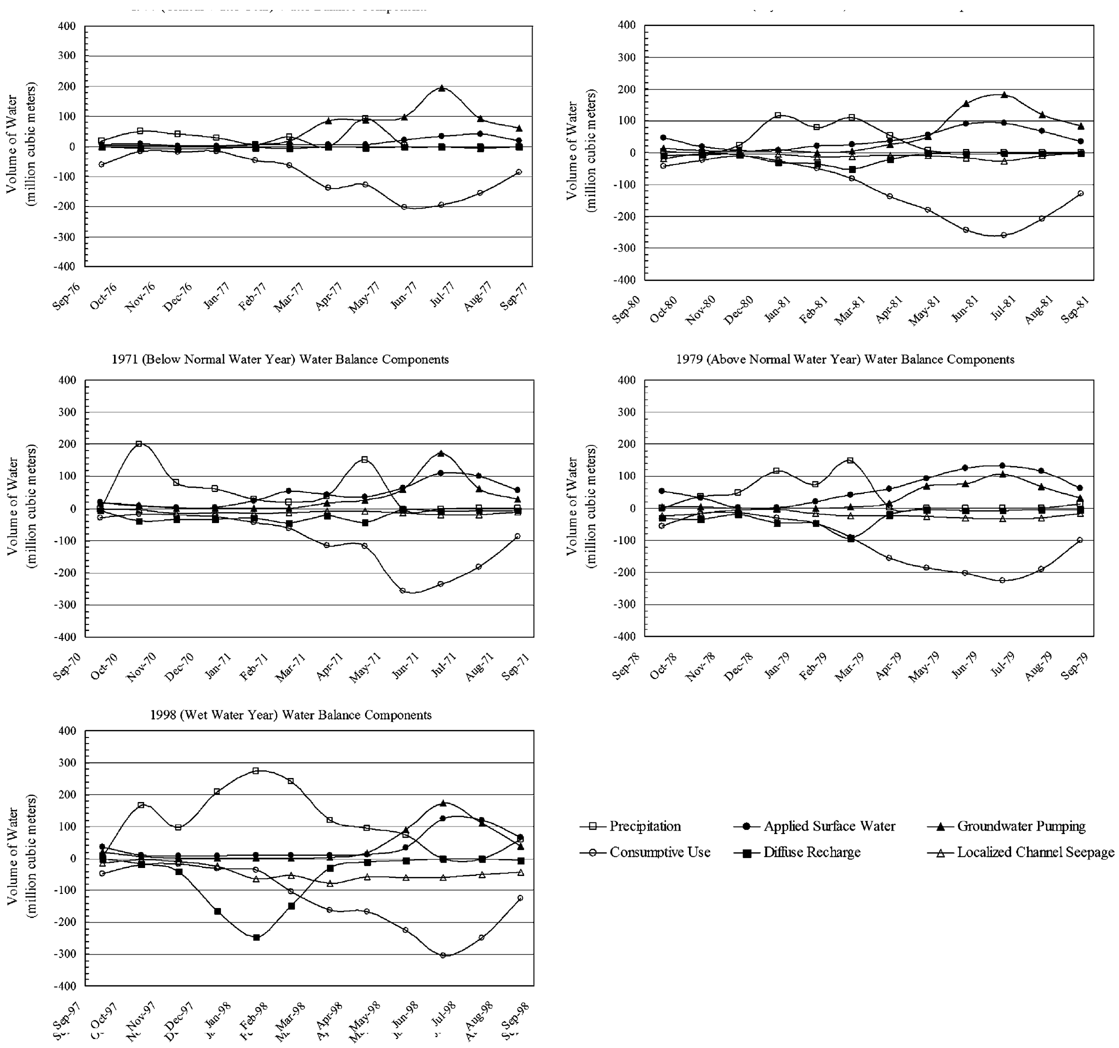

Fig. 6. Monthly water balance components for representative 'Critical', 'Dry', 'Below Normal', 'Above Normal', and 'Wet' water year types. Fluxes to the land surface are positive, fluxes away from the land surface are negative.

Precipitation is nearly absent in Critical years but extends over several months, from late fall into early summer, in Wet years. Diffuse recharge is generally highest during the early spring months when soils are saturated with precipitation and surface water in excess of storage capacity is available for intentional recharge and pre-irrigations. The additional diffuse recharge during Above Normal and Wet years occurs almost entirely during the late winter and early spring; while channel seepage is relatively constant during the spring and summer months but nearly absent in the late fall and early winter. Groundwater pumping occurs from April through September with a peak in July. However, in Above Normal and Wet years the onset of pumping can be delayed by 1–2 months. Seasonal variations in consumptive use (i.e. high during the summer, low during the winter) are similar regardless of water year classification.

4.2. Validation of groundwater storage changes

Annual groundwater storage changes (positive or negative) account for a significant portion of the annual water balance: The absolute annual storage change varies from 21×10^6 m^3 to 1042×10^6 m^3 and averages 402×10^6m^3 or nearly one-third of the annual consumptive use. The average is of nearly the same magnitude as annual precipitation and surface water inflows.

For validation, the cumulative annual groundwater storage changes are compared against those of the WTF method (Fig. 7). The differences between them range from −4.4 cm of water in 1983 to 58.1 cm in 1991. The correlation coefficient between them is 0.8. On average, model estimates are only 62% of WTF method estimates, due possibly to errors in the specific yield values used by the WTF method. As a result, the model estimates of cumulative storage changes from 1970 are higher than those of the WTF method in most years (Fig. 7). Nevertheless, the shape of the cumulative storage change curves are qualitatively similar with differences often less than 20 cm. Cumulative storage changes encompass the same range as those of the WTF method, and are subject to similar annual variations. The agreement of the cumulative probability of annual storage changes (Fig. 7) is comparable to that for annual variations in basin yield found with conceptually similar multiple

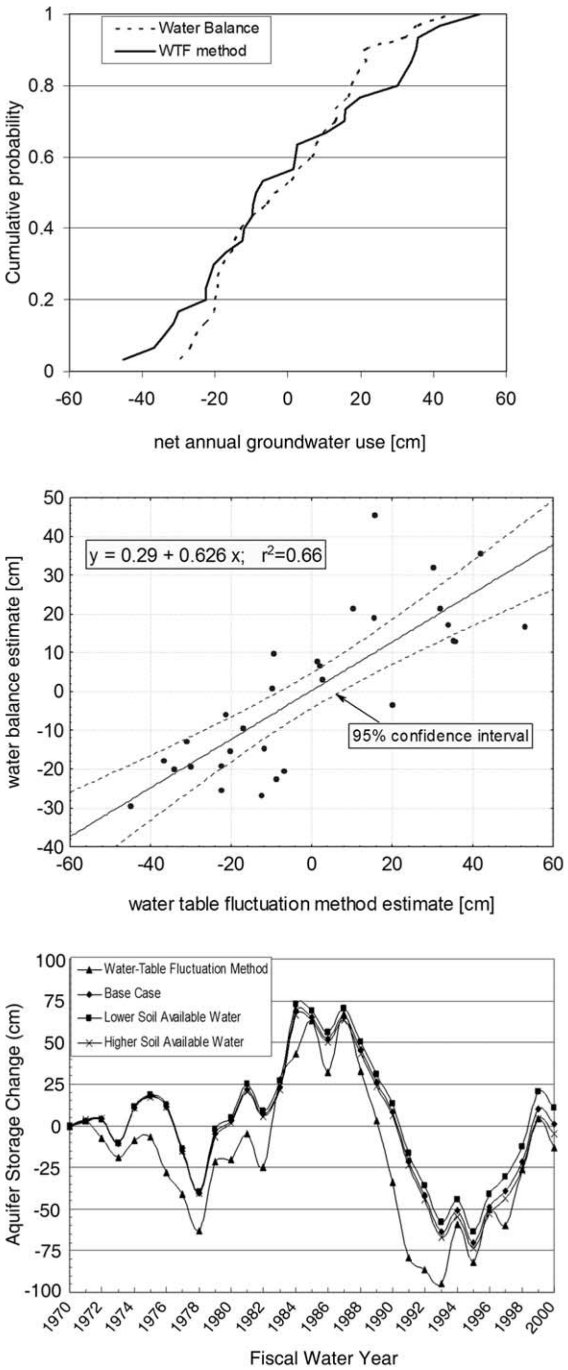

Fig. 7. Annual groundwater storage changes for the study area as computed by the water balance model and the water-table fluctuation method. Top: cumulative probability distribution functions; center: Scatterplot and linear regression; cumulative changes from 1970 to 2000.

'parallel soil storage' models in other semi-arid basins (Farmer et al., 2003). Although the WTF method is itself only an estimator of groundwater storage change, it does provide a reasonable check for the water balance model results. In particular the 30-year total groundwater storage change computed by the WTF method—unlike that of the water balance method—is not subject to compounding errors due to systematic methodological bias, because water levels in 1970 and 1999 are similar.

Annual groundwater storage change is estimated as the difference between annual precipitation and surface water inflows on one hand and annual consumptive use on the other. Of the three components, surface water inflows are the most reliably quantified. The estimated spatio-temporal distributions of precipitation and crop consumptive use may possess systematic errors which cannot be easily quantified. However, Jothityangkoon et al. (2001) and Farmer et al. (2003) have shown that monthly precipitation and consumptive use values estimated from sparsely measured data using methods comparable to ours are sufficient for computing annual and seasonal water balances for semi-arid basins. Their work also emphasized the importance of accounting for soil, vegetative, and climate variability within the basin at scales that are comparable to that of our land units.

Over the 30-year period, the water balance totals of precipitation, surface water, and consumptive use amount to 17.13×10^9, 23.15×10^9, and 40.35×10^9 m^3, respectively, leaving a difference between these inputs and outputs of -0.07×10^9 m^3 (-3.2 cm). The latter is within 25 cm of the total groundwater storage change observed by the WTF method for 1970–1999. Therefore, on a long-term basis, systematic errors in groundwater storage change estimates amount to less than 1 cm/year or less than 20×10^6 m^3/year. This corresponds to less than 5% of the average annual surface water supply, less than 5% of the average annual precipitation, or less than 2% of the average annual consumptive use. The accuracy of the long-term groundwater storage change estimation indicates that the precipitation and consumptive use components of the water balance model provide sufficiently accurate estimates of inter-annual variations in water availability and water use. It is conceivable but unlikely that methodological bias in the precipitation and consumptive use estimates coincidentally cancel each other. Further data collection is needed to provide an independent check of this possibility.

The agreement with the WTF method provides significant confidence in the use of the 5% reduction in Eq. (4) for non-consumptive areas of land units and the adjustment factor in Eq. (5) for annual land use changes over the base period. They also support the assumption that nearly all precipitation contributes either to consumptive use or groundwater recharge, confirming recent field findings by Green et al. (2003) who show that evaporative losses of (winter-dominated) precipitation at the land surface (other than those accounted for by the consumptive use method) are negligible despite the semi-arid climate.

4.3. Sensitivity of estimated groundwater pumping

The estimation of annual groundwater storage changes is independent of the estimated groundwater pumping (i.e. excess applied groundwater deep percolates back into the water table resulting in no change in long-term aquifer storage). However, reasonable estimates of basin scale pumping require accurate knowledge of the efficiency with which crops are irrigated, the amount of soil moisture available for crop uptake, and the amount of surface water actually available for irrigation. The model parameters that define these variables are: (1) crop irrigation efficiencies; (2) the effective soil root zone depth; (3) the available soil water; and (4) the intra-district seepage in districts with predominantly unlined distribution systems.

To understand their effects, we evaluated the model by specifying lower, intermediate, and higher values or spatially distributed sets of values for these four parameters (Tables 2–4). The set of intermediate values for these parameters was taken as a base case for comparison purposes. The study area pumping was estimated for both the lower and higher values of each parameter, while maintaining the other three parameters at their intermediate values. In addition, the widest possible range of model pumping estimates was also calculated by selecting combinations of higher and lower values for the four parameters which would produce

Table 3
Soil series name, low value of soil available water, intermediate average value of soil available water, high value of soil available water, and associated depth of soil profile

Soil series	Low value	Intermediate average	High value	Depth (cm)
Akers–Akers	0.13	0.17	0.21	152
Armona sandy loam	0.02	0.08	0.14	152
Atesh-Jerryslu association	0.06	0.11	0.16	109
Auberry-rock outcrop complex	0.1	0.14	0.18	109
Biggriz–Biggriz	0.12	0.145	0.17	165
Blasingame sandy loam	0.1	0.14	0.18	91
Blasingame-rock outcrop complex	0.1	0.14	0.18	91
Calgro–Calgro	0.11	0.145	0.18	64
Centerville clay	0.12	0.135	0.15	94
Cibo clay	0.1	0.125	0.15	89
Cibo-rock outcrop complex	0.1	0.125	0.15	89
Cieneba-rock outcrop complex	0.09	0.11	0.13	41
Clear lake clay	0.12	0.14	0.16	168
Coarsegold loam	0.12	0.15	0.18	79
Coarsegold-rock outcrop complex	0.12	0.15	0.18	79
Colpien loam	0.14	0.175	0.21	152
Crosscreek-Kai association	0.06	0.12	0.18	140
Delvar clay loam	0.1	0.14	0.18	152
Excelsior fine sandy loam	0.05	0.1	0.15	152
Exeter loam (western soils map)	0.14	0.16	0.18	71
Exeter loam (eastern soils map)	0.14	0.155	0.17	76
Flamen loam	0.1	0.145	0.19	109
Friant-rock outcrop complex	0.08	0.105	0.13	46
Gambogy loam	0.11	0.14	0.17	119
Gambogy-Biggriz	0.11	0.14	0.17	119
Gareck-Garces association	0.1	0.15	0.2	119
Gepford silty clay	0.08	0.105	0.13	127
Grangeville fine sandy loam	0.08	0.11	0.14	203
Grangeville silt loam	0.13	0.15	0.17	163
Greenfield sandy loam	0.1	0.13	0.16	178
Hanford sandy loam	0.07	0.11	0.15	152
Havala loam	0.09	0.135	0.18	163
Honcut sandy loam	0.1	0.115	0.13	178
Houser fine sandy loam	0.01	0.08	0.15	152
Houser silty clay	0.01	0.075	0.14	152
Kimberlina fine sandy loam	0.02	0.05	0.08	147
Las Posas loam	0.12	0.145	0.17	81
Las Posas-rock outcrop complex	0.12	0.145	0.17	81
Lethent silt loam	0.06	0.1	0.14	165
Lewis clay loam	0.06	0.12	0.18	97
Nahrub silt loam	0.08	0.11	0.14	168
Nord fine sandy loam	0.05	0.1	0.15	127
Porterville clay	0.1	0.125	0.15	183
Porterville Cobbly clay	0.08	0.11	0.14	175
Posochanet silt loam	0.12	0.14	0.16	152
Quonal-lewis association	0.09	0.13	0.17	104
San Emigdio loam	0.1	0.13	0.16	168
San Joaquin loam	0.04	0.105	0.17	64
Sesame sandy loam	0.1	0.135	0.17	79
Seville clay	0.14	0.155	0.17	74

(*continued on next page*)

Table 3 (*continued*)

Soil series	Low value	Intermediate average	High value	Depth (cm)
Tagus loam	0.12	0.135	0.15	160
Tujunga loamy sand	0.02	0.05	0.08	178
Tujunga sand	0.05	0.065	0.08	152
Vista coarse sandy loam	0.07	0.095	0.12	69
Vista-rock outcrop complex	0.07	0.095	0.12	69
Wasco sandy loam	0.06	0.085	0.11	152
Westcamp silt loam	0.06	0.115	0.17	152
Wyman Gravelly loam	0.1	0.125	0.15	102
Wyman loam	0.14	0.165	0.19	175
Yettem sandy loam	0.09	0.11	0.13	178
Youd loam	0.05	0.105	0.16	152

the minimum and maximum amounts of pumping. The minimum pumping is expected by choosing values which reduce intra-district seepage losses, increase the amount of available moisture in the soil root zone, and increase the efficiency of irrigation applications. Conversely, maximum pumping is expected for large intra-district seepage losses, low available moisture contents, and lower irrigation efficiencies.

The range of available soil water for the soil series present in the study area were obtained from the Natural Resources Conservation Service (Table 3). The effective root depth is defined as 70% of the maximum root depth (Evans et al., 1996). The maximum root depths of some of the major crops range from 1.2 m for cotton to 2.0 m for walnuts (Table 1) (Walker, 1989). Consequently, the low, intermediate, and high values for the effective soil root zone were chosen as 0.76, 0.91, and 1.07 m, respectively (Table 4). Typical irrigation systems for crops grown in the study area are listed in Table 2 and their range of attainable irrigation efficiencies are given in Table 4 (Solomon, 1988). As mentioned earlier, the assignment of intra-district seepage losses was based on estimates obtained from previous groundwater studies in adjacent sub-basins (Bookman and Edmonston Inc., 1972; Kings River Conservation District, 1992; Fugro West Inc., 2003). For districts with predominantly unlined distribution systems, we assumed an intermediate seepage rate of 15% of monthly surface water deliveries for four small-sized

Table 4
Low, intermediate average, and high values of model parameters used to evaluate the range of groundwater pumping estimates

Model parameter	Low value	Intermediate average	High value
Effective soil root zone thickness, b_s (m)	0.76	0.91	1.07
Intra-district seepage rate (% of district surface water delivery):			
Small districts	5	15	25
Large districts	15	25	35
Irrigation systems/irrigation efficiency (%):			
Surface-basin	80	85	90
Surface-border	70	77.5	85
Surface-furrow	60	67.5	75
Sprinkler-solid set	70	75	80
Sprinkler-traveling gun	60	65	70
Trickle-point source emitters	75	82.5	90

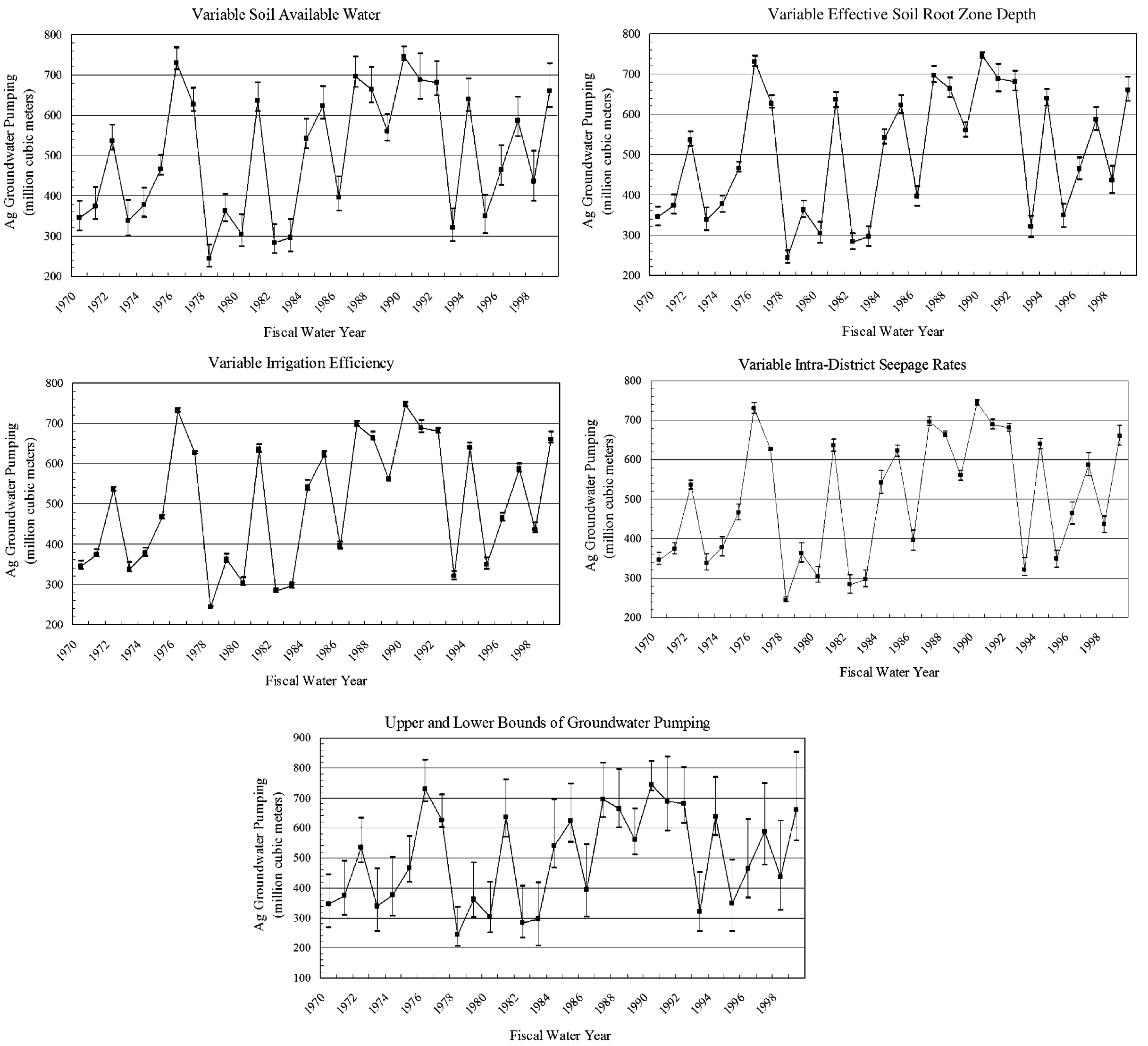

Fig. 8. Estimated annual groundwater pumping for the study area for variable soil available water, effective soil root zone depth, irrigation efficiency, and intra-district seepage rates.

districts and 25% for two larger districts. The lower, intermediate, and higher intra-district seepage rates are given in Table 4.

The estimates of annual agricultural groundwater pumping for the various scenarios are presented in Fig. 8. Of the four parameters varied, annual pumping was most responsive to changes in soil available water. The spatial distributions for the low and high ends of the available water range produced estimates which varied about the base case estimate by an annual average of 10.6 and −6.9%, respectively. As expected, cumulative groundwater storage changes are also minimally affected by changes in these parameters, as demonstrated for available water (Fig. 7). For the low and high values of the other three parameters, estimates varied from the base case by a 30-year annual average of 5.4 and −4.7% for effective root depth; 2.4 and −1.4% for irrigation efficiency; and −3.7 and 4.3% for intra-district seepage loss rates. The parameter set chosen to minimize pumping produced estimates which on average were 15.3% less than the intermediate base case; while the set chosen to maximize pumping produced estimates which were 28% higher than the base case. The difference between the minimum and maximum pumping ranged from 98×10^6 m^3 in 1990 to 297×10^6 m^3 in 1998, with an annual average of 195×10^6 m^3. Although the minimum and maximum scenarios represent extreme cases, they merely illustrate the wide range of pumping estimates which may result from unrealistic choices of parameter values in the model.

The results confirm studies by Milly (1994), Salvucci and Entekhabi (1994), Muttiah and Wurbs (2002) and Scanlon et al. (2002) who found that basin water yield is very sensitive to the storage capacity and thickness of the root zone, particularly in semi-arid and arid climates. In our water balance, groundwater recharge constitutes the basin yield. Given that the water demand (consumptive use) is fixed through active management, groundwater pumping is directly proportional to recharge, and therefore an indirect measure of basin yield.

5. Conclusions

Groundwater storage changes and agricultural groundwater pumping in active semi-arid basins are significant yet little understood components of the basin water balance. Here, we developed and evaluated a distributed dynamic water balance approach that estimates intra-annual and interannual components of the water balance including groundwater pumping and groundwater storage changes via closure of the water balance.

The water balance model does not distinguish the contributions to ET from precipitation, applied surface water, and applied groundwater. For example, it does not compute 'effective precipitation', that is the amount of precipitation consumed by production crops. The model assumes that 100% of the monthly precipitation infiltrates into the soil root zone and is available for crop uptake or recharge (i.e. no regional surface runoff generation and no direct evaporation of precipitation). Changes in the soil root zone moisture content are calculated using a simple tipping-bucket model. As a result, infiltrated precipitation will be stored as soil moisture until it is either consumed by the crop or percolates below the root zone once the field capacity is exceeded. Since flows in the soil root zone are not explicitly modeled, field capacity can be maintained in the root zone for months, due to high monthly precipitation inputs and low crop water demands.

We applied the model to an agriculturally highly developed groundwater basin within the San Joaquin Valley, California. Groundwater pumping accounted for as much as 80% of the total applied water in Critical dry water years and as little as 30% in Wet water years. Annual groundwater storage changes for the basin varied from less than 5 to over 40 cm/year and are therefore a significant portion of the basin water balance. Comparison to annual groundwater storage change data obtained independently from distributed groundwater level measurements indicates that the sub-models we used to estimate annual basin precipitation, allocated surface water inflows and crop consumptive use from available data are associated with estimation errors less than 2–5%.

Much of the uncertainty in model input data is associated with the quantification of soil root zone available moisture and with the actual amounts of imported surface water which is applied as irrigation. The data uncertainty leads to potential errors in estimating intra-annual and inter-annual variations in groundwater pumping. Sensitivity analysis indicates that the estimation of agricultural groundwater pumping is associated with estimation errors on the order of ±10%.

While the uncertainty in some model data (e.g. applied surface water, crop consumptive use) will likely decrease in the future with the wider implementation of improved measurement technologies (e.g. remote sensing, head-gate flowmeters), this study underscores the need for better spatial characterization of soil hydraulic properties and irrigation management practices to improve basin scale groundwater pumping estimates.

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