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Coupling a spatiotemporally distributed soil water budget with stream-depletion functions to inform stakeholder-driven management of groundwater-dependent ecosystems

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[1] Groundwater pumping, even if only seasonal, may significantly impact groundwater-dependent ecosystems through increased streamflow depletion, particularly in semiarid and arid regions. The effects are exacerbated, under some conditions, by climate change. In social sciences, the management of groundwater-dependent ecosystems is generally considered a "wicked" problem due to the complexity of affected stakeholder groups, disconnected legal frameworks, and a divergence of policies and science at the cross road between groundwater and surface water, and between ecosystems and water quality. A range of often simplified scientific tools plays an important role in addressing such problems. Here we develop a spatiotemporally distributed soil water budget model that we couple with an analytical model for stream depletion from groundwater pumping to rapidly assess seasonal impacts of groundwater pumping on streamflow during critical low flow periods. We demonstrate the applicability of the tool for the Scott Valley in Northern California, where protected salmon depend on summer streamflow fed by cool groundwater. In this example, simulations suggest that increased recharge in the period immediately preceding the critical low streamflow season, and transfer of groundwater pumping away from the stream are potentially promising tools to address ecosystem concerns, albeit raising difficult infrastructure and water trading issues. In contrast, additional winter recharge at the expense of later spring recharge, whether intentional or driven by climate may reduce summer streamflows. Comparison to existing detailed numerical groundwater model results suggests that the coupled soil water mass balance—stream depletion function approach provides a viable tool for scenario development among stakeholders, to constructively inform the search for potential solutions, and to direct more detailed, complex site-specific feasibility studies. The tool also identifies important field monitoring efforts needed to improve the understanding and quantification of site-specific groundwater-stream interactions.

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1. Introduction

[2] Groundwater-dependent ecosystems (GDEs) located within streams are among several types of GDEs including peats, terrestrial systems, and springs [*Howard and Merrifield*, 2010; *Bertrand et al.*, 2012]. Significant groundwater development can lead to reduction in base flow of nearby rivers and streams. Particularly in Mediterranean and similar semiarid climates, dry, warm periods coincide with the crop growing season supported by irrigation, often with groundwater. Regions in the Western and Central U.S., Mexico, Argentina, North Africa, the Middle East, Southern Europe, Northern India, China, and Southeast Asia are widely affected by use of groundwater with major impacts

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to surface water flows [*Wada et al*., 2010, 2012; *Gleeson et al*., 2010]. Irrigated agricultural systems provide 40% of the world's crop production [*United Nations World Water Development Report*, 2009] with over 100 million ha of land equipped for irrigation with groundwater and an estimated 545 km^3 of extracted water [*Siebert et al*., 2010].

[3] Groundwater management may follow a "safe yield" approach that balances long-term, annual water extraction with groundwater recharge, yet pumping induced decrease of dry season base flow may negatively impact ecoystems [*Sophocleous*, 2000, *Jolly et al*., 2010]. Statistical analyses of long-term precipitation, pumping, and streamflow records, e.g., in the High Plains aquifer system, have been used to show significant linkages between pumping and streamflow depletion [*Burt et al*., 2002; *Wen and Chen*, 2006; *Kustu et al*., 2010]. *Zume and Tarhule* [2008] used a fully three-dimensional groundwater-surface water model to investigate the effects of basin-wide pumping reductions on streamflow depletion in Oklahoma. A similar tool was used, at a much smaller scale, to analyze the hydroecology of mountain meadows fed by groundwater [*Loheide and Gorelick*, 2007]. Significant work has been conducted on optimizing conjunctive use of groundwater and surface water [*Singh*, 2012]. But economic analysis of groundwater-surface water systems does typically not account for hydrologic regimes important to ecosystem services.

[4] Improved implementation of conjunctive use schemes of surface water and groundwater resources are an important step toward improving conditions in GDEs with opportunities for improving the economy of these systems while significantly increasing the resilience to droughts [*Lefkoff and Gorelick*, 1990; *Schoups et al*., 2006; *Bredehoeft*, 2011]. But dynamics at the interface between groundwater and streams and the combined impacts of groundwater abstraction and climate change on streamflow depletion and GDEs are legally unrecognized [*Thompson et al*., 2006] and often ignored by water managers [*Kollet et al*., 2002; *Döll et al*., 2012]. In the United States, where groundwater management is delegated to individual states, water laws largely lack a comprehensive framework for the management of GDEs and even ignore the physical connection of surface water and groundwater [*Harter and Rollins*, 2008; *Nelson*, 2012]. Human modifications of water flows at local, regional, and continental scales interject multiple conflicting objectives into water management including food production and ecosystem services [*Maxwell et al*., 2007]. Climate change promises to incur further shifts with impacts rippling throughout the water network, in unanticipated ways [*Allen et al*., 2004; *Scibek and Allen*, 2006; *Maxwell and Kollet* 2008].

[5] Such "wicked" problems are characterized by a high level of complexity, uncertainty, and conflict [*Von Korff et al*., 2012; *Ker Rault and Jeffrey*, 2008; *Kreuter et al*., 2004; *Freeman*, 2000]. Addressing wicked problems requires new participatory approaches to the decision-making process and an active role of physical/hydrologic sciences in addressing such problems. Scientific understanding of hydrologic systems is advancing rapidly, but developing tools that communicate fundamental scientific understanding to decisions makers and citizens remain a challenge at all scales (global, regional, and local) [*Reid et al*., 2010; *UNESCO*, 2010].

[6] Efforts to address wicked water problems have been or are under development in different regions of the world and at different scales [*Ostrom et al*., 1999; *Sophocleous*, 2002; *Hare et al*., 2003; *Moellenkamp et al*., 2010; *Von Korff*, 2012]. Many include an effort to integrate scientists, decision makers (at the local and regional scale), and regulators within the workflow [*Sophocleous*, 2012]. Often, collaborative solutions to such wicked problems require conceptual representations of the water management system(s) at various levels of complexity.

[7] Simple conceptual models convey fundamental insights into the dynamics of hydrologic systems to non-technical stakeholders. Such models are also useful to develop worst-case/best-case scenarios given the conceptual simplification and data limitation underlying the model. Models representing additional complexity may then be used to further constrain insights into the hydrologic system and predictions of its future state. This process enables a better understanding of water resources and leads to a more informed approach toward developing strategies and scenarios for better water resources management.

[8] In this work, we couple two low order (conceptually and geometrically simple, mass balance based) hydrologic modeling tools to investigate aquifer-stream interactions. Simplified aquifer-stream interaction models to reduce computational costs have been applied in hydro-economic modeling efforts [e.g., *Pulido-Velazquez et al*., 2008], showing that a coupled water budget-stream depletion function analysis may be useful for optimizing groundwater management under ecosystem services constraints. Here we expand the approach to investigate spatiotemporally distributed groundwater management alternatives that may improve GDE conditions in basins with significant but unmeasured groundwater extractions and recharge.

[9] The tool is applied to the Scott Valley groundwater basin, California, to (1) evaluate and demonstrate the fundamental dynamics between landuse, groundwater use, and seasonally low streamflow that is affecting stream temperature [*Caissie*, 2006] and salmonid stream habitat [*Milner et al*., 2012]; (2) evaluate the role of data in understanding the key drivers of potential stream base flow depletion during the dry season in a semiarid, irrigated agricultural region with Mediterranean climate; (3) utilize the tool to cast an overall framework for developing potential groundwater management options and for defining project-specific feasibility work; and (4) employ the tool for education and outreach to diverse stakeholders seeking common, creative solutions. Stakeholders in the Scott Valley include local landowners (farms) and groundwater pumpers, native American tribes dependent on downstream salmon fisheries, environmental groups, as well as local, state, and federal agencies representing often conflicting interests in water rights regulation, water quality control, endangered species protection, and agricultural resources management—thus representing all the ingredients to a "wicked" water management problem.

[10] In the following, we provide further details on the study area and describe the spatiotemporally distributed

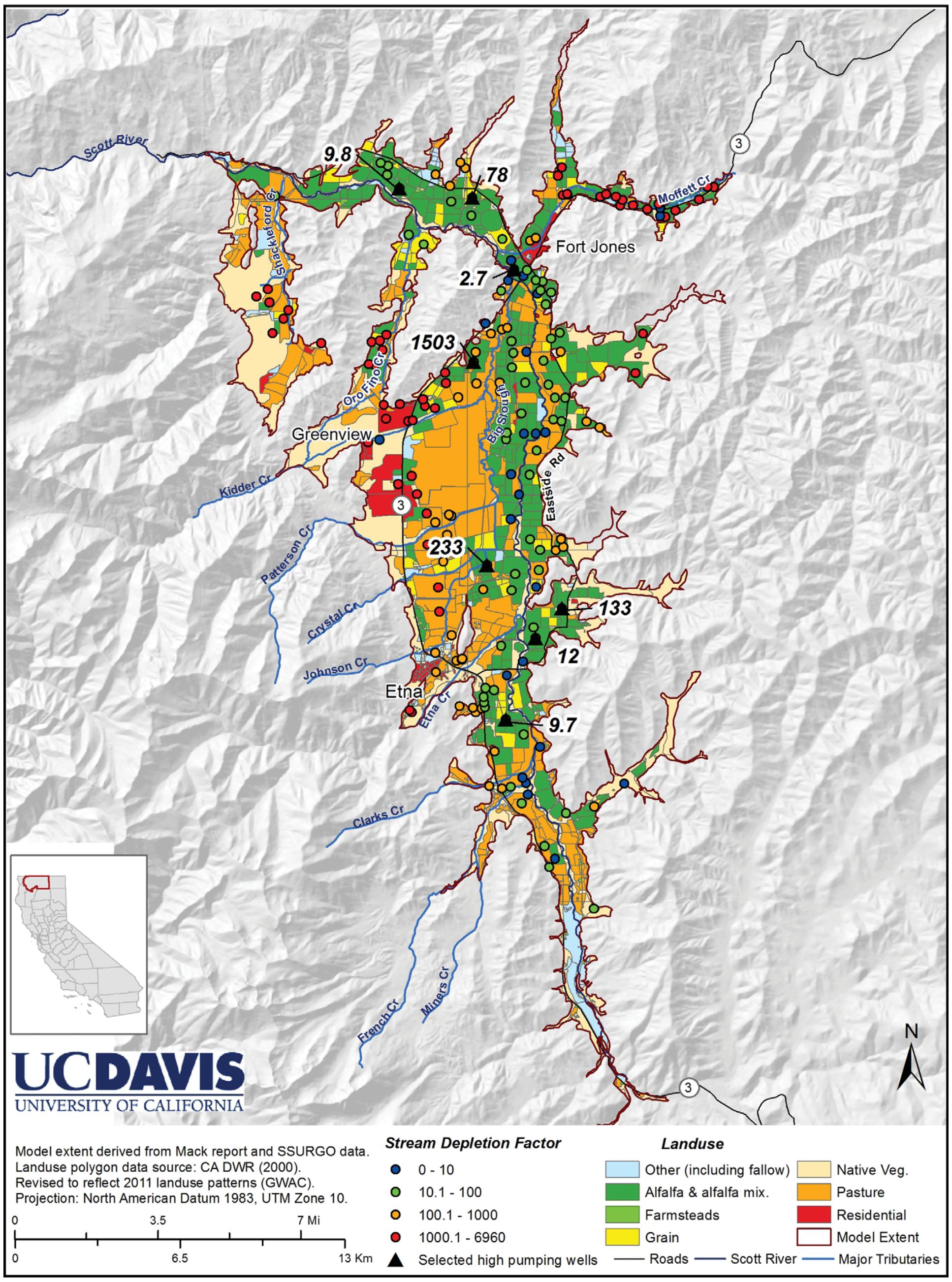


Figure 1. Map of the Scott Valley with the boundaries of the groundwater model study, landuse, and irrigation wells with their stream depletion factor (SDF in days) relative to the main-stem Scott River, calculated as described below. Highlighted are the SDF values of some wells used below for a detailed analysis (also see Table 4).

soil water budget approach and the theory of stream depletion analysis. We use the coupled water budget and stream depletion analysis to explore the role of groundwater pumping in the Scott River Valley with respect to late summer base flow in the Scott River. We then identify broad options for potential alternative water management scenarios to improve summer streamflow as a basis for discussion with stakeholders and for directing the selection and assessment of specific projects including necessary field work and higher level, more complex hydrological modeling efforts.

2. Methods

2.1. Study Basin

[11] The coupled water budget and stream depletion analysis is applied to the 202 km^2 Scott River Valley, Siskiyou County, Northern California. The major landuse is pasture, alfalfa hay, and grain farming (approximately 140 km^2) supported by summer irrigation with stream water and with groundwater. The valley is part of the Klamath Basin watershed straddling the California-Oregon border (Figure 1). The Scott River is one of four undammed

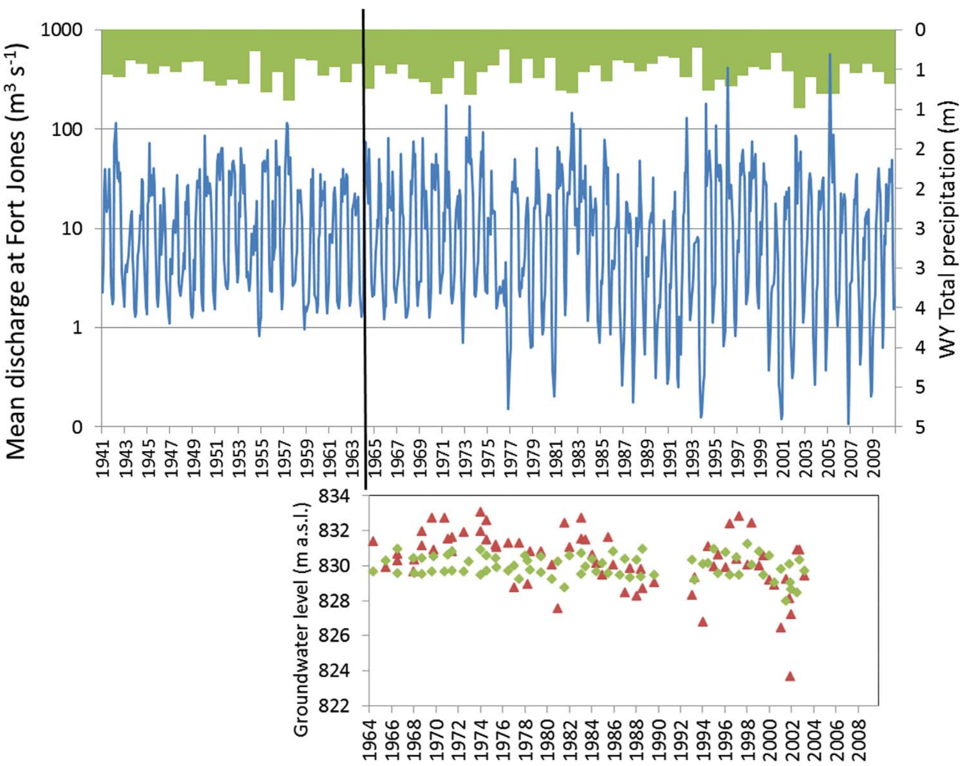


Figure 2. (a) Daily mean discharge ($m^3\ d^{-1}$) of the Scott River recorded at the USGS gauge near Fort Jones. Since the mid-1970s, dry year low flows (1977, early 1990s, 2001, 2007–2008) have been about half an order of magnitude lower than during the 1941–1976 measurement period (1945, 1955). (b) Scott River Valley well levels and precipitation, 1965–2012 [*California Department of Water Resources* (*CDWR*), 2012]. Beginning with the drought-year 1977, summer water levels in some dry years were lower than during the 1964–1976 period.

tributaries to the Klamath River. It provides key spawning habitat for salmonid fish in the Klamath Basin, including *Oncorhynchus tschawytscha* (Chinook salmon) and federally protected threatened *Oncorhynchus kisutch* (Coho salmon). The Scott River has been mapped as medium to high ranking for the presence of base flow-dependent ecosystems [*Howard and Merrifield*, 2010].

[12] Scott Valley overlies an intermontane alluvial basin within the Klamath Mountains Province, created by faulting along its northwestern outlet, and subsequent alluvial deposition during the late Tertiary and Quaternary. The alluvial fill, consisting of gravel, sands, and also silts and clays, may exceed 100 m thickness at the center of the basin and decreases in thickness to the valley margins [*Mack*, 1958]. Groundwater pumping is limited to the upper 60 m of the alluvial fill. Spring groundwater levels, while slightly variable from year to year, have not experienced a long-term decline that would indicate systemic overdraft [*Harter and Hines*, 2008; *S.S. Papadopoulos & Associates (SSPA)*, 2012].

[13] The climate is Mediterranean. Precipitation predominantly occurs during winter and early spring months but is negligible between June and September. Average July temperature is 21°C and average January temperature is 0°C. Total annual rainfall on the valley floor is 500 mm. Mountain ranges surrounding Scott Valley reach elevations of 2500 m with much higher precipitation rates than the valley. Annual runoff from the 1700 km^2 watershed is 560 Mm^3 [*U.S. Geological Survey* (*USGS*), 2012]. Winter flows in the main stem of the Scott River, immediately downstream of the groundwater basin, may exceed 1,000,000 $m^3\ d^{-1}$ (400 cfs) during winter months, but are as low as 25,000–125,000 $m^3\ d^{-1}$ (10–50 cfs) during the later summer months (July–September) (Figure 2a).

[14] During the dry summer, streamflow in the Scott River system significantly relies on groundwater return flow (base flow) from the alluvial aquifer system underlying Scott Valley. Historic records show that summer base flows in dry years prior to 1977 (1945, 1955) have been higher than during later dry years (1977, early 1990s, 2001, 2009) (Figure 2a). The decrease is generally attributed to climate change [*Drake et al.*, 2000], but also to increased groundwater pumping for irrigation [*Van Kirk and Naman*, 2008]. As a result of lower summer/fall base flow, but also due to the lack of widespread riparian vegetation, temperatures in the Scott River may exceed critically high levels during the summer months [*NCRWB*, 2011]. Yet, ecologically necessary minimum flow requirements remain uncertain.

[15] Under regulatory efforts driven by federal *Clean Water Act* [1972] provisions (33 U.S.C. §1251 et seq., 1972, and 40 C.F.R. 130.2), stakeholders have agreed that better knowledge of the hydrology and the alluvial aquifer system is needed to develop a possible array of solutions to water issues and associated problems [*Harter and Hines*, 2008]. Siskiyou County has management jurisdiction over

groundwater and is taking a community-based approach to implementing groundwater management.

[16] Water and groundwater management is also affected by recent enforcement actions under the California Endangered Species Act (CESA; California Fish and Game Code, Sections 2050 et seq.), which allows the State of California to curtail diversions of irrigation water if instream flows are considered critically low with respect to threatened or endangered salmon species in the river system (California Fish and Game Code, Section 5937). Finally, a lawsuit has been brought against the County (as the groundwater management agency) and the State (as the licensor of water rights) to protect groundwater-dependent ecosystems under the so-called Public Trust doctrine [*Hart*, 1996]. If successful, this may give the State an unprecedented legal tool to enforce limits on current groundwater pumping not already controlled under existing adjudications. An existing groundwater adjudication in the Scott Valley, dating to the 1970s, prescribes the amount of groundwater that is reasonably required to irrigate within a groundwater—surface water "interconnected zone" (California Water Code 2500.2) extending approximately 500–1000 m from the main-stem Scott River [*California State Water Resources Control Board* (*CSWRCB*), 1980]. Elsewhere in Scott Valley, as is customary in California, groundwater pumping for overlying uses does not require state permitting [*California Department of Water Resources* (*CDWR*), 2003].

2.2. Soil Water Budget Model

[17] Land use specific water budgets have been used to allow for a better understanding of landuse linkages to groundwater and provide the basis for distributed groundwater-stream models [e.g., *Ruud et al*., 2004; *Faunt*, 2009; *Chung et al*., 2010]. In the study area, measurement data on groundwater extraction and recharge do not exist. Hence, a soil water budget model is used to estimate spatially and temporally varying recharge and pumping across the groundwater basin.

[18] The spatial resolution for the analysis is determined by the size of individual fields and other landuse parcels defined in a recent landuse survey [*CDWR*, 2000] that was further refined using aerial photo analysis and on-the-ground verification. A total of 2119 landuse parcels overly the Scott Valley groundwater basin (Figure 1). Of those, 710 parcels (70 km^2) are alfalfa/grain, typically on an 8 year rotation with 1 year of grain crops followed by 7 years of alfalfa, 541 parcels (67 km^2) are pasture, 451 parcels (58 km^2) belong to landuse categories with significant evapotranspiration but no irrigation (e.g., cemeteries, lawns, natural vegetation), and 417 parcels (6.8 km^2) represent landuses with no evapotranspiration or irrigation (e.g., residential, parking lots, roads, and—most significantly—historic mine tailings). For each landuse parcel, the soil water budget is computed with daily time steps [e.g., *Gassman et al*., 2007] for the period from 1 October 1990 to 30 September 2011, a period that includes several dry years as well as average year and wet year periods.

[19] The soil water budget approach includes the managed components of the surface water system (diversions) and of the groundwater system (extraction), as well as groundwater recharge from managed and unmanaged land-uses. The budget does not account for stream recharge or for groundwater discharge downstream resulting from stream recharge upstream. It also does not account for evapotranspiration due to root water uptake from the water table by nonirrigated crops or in natural landscapes with shallow water table. A complete surface watershed or groundwater basin budget requires a more complex, integrated groundwater-surface model.

[20] To compute the soil water budget, each landuse polygon is characterized by a set of properties (attributes) assembled from existing databases, through field work, survey, and by applying spatial analysis within a geographic information system (GIS). The concepts applied represent some simplification over detailed root zone water models, but are commensurate given available data and the overall framework of the approach:

[21] 1. Daily precipitation for 1990–2011 is obtained as the average of records at two rainfall gauges located in the northeast and southern-most portions of the valley floor [*National Oceanographic and Atmospheric Administration (NOAA)*, 2012].

[22] 2. Streamflow for 1990–2011: Daily discharge data for the Scott River downstream of Scott Valley are available from the U.S. Geological Survey [*USGS*, 2012]. Streamflow data on ten tributaries, including the two main stem forks of the Scott River, at locations immediately upstream of the valley floor (i.e., upstream of the groundwater basin) have been collected at various times by local and state agencies. But no long-term records exist. Missing data on tributary inflows into the valley at the upgradient boundaries of the groundwater basin are estimated by performing a regression analysis of measured tributary flow against downstream flow, snowpack, and precipitation as independent variables (see supporting information).

[23] 3. Landuse: Digital land use survey maps for the year 2000 [*CDWR*, 2000] identify individual landuse parcels (polygons) and their landuse. The information was updated and corrected via interviews with landowners (Figure 1). Landuse is then aggregated into four major categories for purposes of computing the soil water budget: (1) Alfalfa/grain rotation in an 8 year cycle (each field is randomly assigned one of the 8 years in the cycle during which it goes into "grain" rotation), (2) pasture, (3) landuse with evapotranspiration but no irrigation (includes natural vegetation, natural high water meadow, misc. deciduous trees, trees), and (4) landuse with no evapotranspiration and no irrigation, but with potential recharge from precipitation via soil moisture storage (barren, commercial, dairy, extractive industry, municipal, industrial, paved, gravel mine tailings, etc).

[24] 4. Soil type: Digitally mapped soil type information is available from the U.S. Soil Survey Geographic (SSURGO) database [*Natural Resources Conservation Service (NRCS)*, 2012a, 2012b]. Soil type information includes water holding capacities at 0.9 m and 1.5 m depth. For the soil water budget, water holding capacity is computed as the average of these values assuming that average effective root-zone depth for alfalfa is approximately 1.22 m (4 ft) [*Luo et al*., 1995]. Here we use the same depth for grain and pasture. Each landuse polygon is associated with the soil type present at its centroid location.

[25] 5. Crop coefficients (k_c) and reference ET (ET_0): estimation methods of actual crop ET are primarily designed for

Table 1. Total Areas of Subwatersheds, Total Area for Various Irrigation Types, Total Area for Various Irrigation Water Sources, and Total Area of Landuse, in Square Kilometers[a]

Subwatersheds Name	Area (km^2)	Irrigation Type	Area (km^2)	Water Source	Area (km^2)	Landuse	Area (km^2)
Etna Creek	17	Non-irrigated	75	DRY	14	Water	1
French Creek	2	Flood	44	GW	67	Alfalfa/Grain	71
Kidder Creek	38	Sprinkler	51	MIX	16	Pasture	67
Mill Creek	9	Center Pivot	28	SUB	9	ET/No irrigation	57
Moffett Creek	10	Unknown	4	SW	31	No ET	7
Patterson Creek	16			None/unknown	67		
Scott River	84						
Scott River tailings	14						
Shackleford Creek	12						
Study area total	202	Total	202	Total	202	Total	202

[a]All values represent 2011 conditions. Note that not all areas in the alfalfa/grain and pasture category are irrigated.

irrigation scheduling purposes but are here applied to estimate daily varying actual crop ET (equations (4)–(6)). Daily reference ET is estimated from study area climate data [*Hargreaves and Samani*, 1982; *Snyder et al*., 2002]. Crop coefficients vary by crop, by stage of crop growth, and by cultural practices. For alfalfa, a crop coefficient of 0.95 was fitted to field data from the study area [*Hanson et al*., 2011b], since we did not simulate alfalfa cutting dates individually at each field. For grain (variable k_c) and pasture ($k_c = 0.9$), state agricultural extension recommendations were applied [*University of California Cooperative Extention (UCCE)*, 2012].

[26] 6. Irrigation type: The year 2000 landuse survey by CDWR [*CDWR*, 2000] identified the irrigation type associated with each landuse polygon. In the Scott Valley, flood, center pivot sprinkler, and wheel-line sprinkler irrigation are used almost exclusively. Over the past 25 years, significant conversion from wheel-line sprinkler (but also from flood irrigation) to center pivot sprinkler has occurred. The location (extent) and year of such irrigation type conversions are mapped to landuse polygons by reviewing 1990–2011 aerial photos. Total areas for 2011 are shown in Table 1.

[27] 7. Irrigation efficiency is assumed to be a function of irrigation type. It accounts for irrigation nonuniformity and deep percolation losses to below the root zone. Delivery and interception losses are not accounted for. Efficiencies are based on informal surveys of local growers and expertise of local agricultural consultants, although they do not account for unintended underirrigation or deficit irrigation: 90% for center pivot sprinkler, 75% for wheel-line sprinkler, and 70% for flood irrigation (University of California Cooperative Extension (UCCE), personal communication, 2011).

[28] 8. Water source for irrigation: Water source is identified for each landuse polygon by the year 2000 landuse survey [*CDWR*, 2000] and is updated through landowner survey. Water sources include groundwater, surface water, subirrigated (shallow groundwater table), mixed groundwater-surface water, and nonirrigated (dry land farming) (Table 1).

[29] 9. Surface water diversion allocation: Each landuse parcel is associated with one of nine subwatersheds corresponding to the various tributaries to the main stem Scott River (Table 1). Discharge on these tributaries defines available maximum diversion rates (see below).

[30] The soil water budget for each landuse polygon is performed using a storage routing approach with soil water inputs from precipitation and irrigation [e.g., *Neitsch et al*., 2011]. Adjusted daily precipitation (P_{adj}) is the portion of daily precipitation (P) that infiltrates into the soil and is available for daily evapotranspiration (ET) or recharge [*Allen et al*., 1998]:

$$P_{adj}(i) = P \quad \text{if } P(i) > 0.2^{*}\text{ET}_0(i) \tag{1a}$$

$$P_{adj}(i) = 0 \quad \text{if } P(i) \le 0.2^{*}\text{ET}_0(i) \tag{1b}$$

where ET_0 (i) is the daily reference evapotranspiration on day i, assumed uniform across the valley floor due to the size of the study area and its level topography. The storage routing mass balance for the 1.22 m thick root zone is then computed as:

$$\theta(i) = \max(0, \theta(i-1) + P_{adj}(i) + \text{Irrig}(i) - \text{actual ET}(i) - \text{Recharge}(i)) \tag{2}$$

$$\text{actual ET}(i) = \min\left(\text{ET}(i),\ \theta\ (i-1) + P_{adj}(i) + \text{Irrig}(i)\right) \tag{3}$$

$$\text{Recharge}\ (i) = \max(0,\ \theta\ (i-1) + P_{adj}(i) + \text{Irrig}(i) - \text{actual ET}(i) – \text{WC4}(i)) \tag{4}$$

where $\theta(i)$ is water content at the end of day i, $P_{adj}(i)$ is precipitation on day i, Irrig(i) is irrigation on day i, ET(i) is evapotranspiration on day i, computed from potential ET as: $\text{ET}(i) = \text{ET}_0(i)^{*}k_c(i)$, $k_c(i)$ is crop coefficient, Recharge(i) is deep percolation to groundwater, to below the 1.22 m thick root zone, and WC4 is water holding capacity of the 1.22 m root zone.

[31] Runoff, particularly during the irrigation season, is considered negligible due to the low land surface gradient. The algorithm intrinsically exerts complete mass balance control on each landuse polygon:

$$P_{adj}(i) + \text{Irrig}(i) - \text{actual ET}(i) - \text{Recharge}(i) = \theta(i) - \theta(i-1). \tag{5}$$

[32] Furthermore, we can compute the amount of water deficit relative to optimal growing conditions as follows:

$$\text{Deficiency}(i) = \text{ET}(i) - \text{actual ET}(i). \tag{6}$$

[33] The source of irrigation water, Irrig(i), depends on the water source and landuse specified for an individual landuse polygon. For pasture, irrigation water is most often

exclusively supplied from surface water. Alfalfa/grain landuse polygons are most often irrigated from groundwater. Based on information from stakeholders, alfalfa/grain fields with a surface water source are treated as if equipped for a mixed source.

[34] For mixed sources of irrigation water, the decision process that leads to a landuse polygon switching from surface water irrigation to groundwater irrigation is simulated based on the available surface water supply: if the total surface water irrigation demand within a subwatershed, in a given month, exceeds stream discharge, groundwater is used to make up the landuse polygon-specific difference between surface water available and the irrigation demand. The available surface water is distributed to all polygons designated for use of surface water at equal water depth (water volume proportional to polygon size).

3. Irrigation Scheduling Simulation

[35] Surface water delivery and groundwater pumping rates are driven by daily precipitation and evapotranspiration. Urban and domestic pumping are small in comparison and are here neglected. Irrigation water demand is calculated following FAO guidelines [*Allen et al*., 1998]. The approach computes irrigation timing and demand as a function of climate, soil, crop type, irrigation type, and water source.

3.1. Alfalfa/Grain and Pasture

[36] Alfalfa irrigation in polygon k starts on the first day i after 24 March 24, on which the soil water content has dried to less than 45% of field capacity (*ibid*., Table 22):

$$\theta(i) < (1-0.55)^{*}\mathrm{WC4}(k). \tag{7}$$

[37] 25 March is the earliest reported irrigation date. The last alfalfa irrigation application in Scott Valley typically occurs before 5 September. For the water budget computations, irrigations are assumed to occur daily through 5 September based on perfect farmer foresight of crop water demand.

[38] For grain, the first irrigation on a field k is determined exactly as for alfalfa but the reported earliest starting date is 15 March. The last day of continuous irrigation on grain is assumed to be 10 July, after which the grain crop is harvested.

[39] For pasture, the Scott Valley irrigation season is typically from 15 April to 15 October (184 days). Simulated irrigation is applied daily based on ET demand and irrigation efficiency. However, on pasture that is surface water irrigated (which represents most pasture), no irrigation occurs once surface water supplies become unavailable. For each polygon k and for each day i, the daily irrigation amount is calculated as:

$$\mathrm{Irrig}_k(i) = (\mathrm{Ieff}_k)^{-1*}\big(\mathrm{Max}\big(0,\ \big(\mathrm{ET}_k(i) - P_{\mathrm{adj}}(i)\big) \tag{8}$$

where Ieff_k is the irrigation efficiency in polygon k. We assume that there is no contribution to plant evapotranspiration from groundwater. To the degree that groundwater irrigated areas are subject to direct groundwater uptake by crops, the uptake is implicitly accounted for in the net stress estimated with this approach. It is the difference between estimated groundwater pumping and recharge from polygon k.

3.2. Evapotranspiration (ET) Losses Without Irrigation

[40] The main assumption is that, at all times:

$$\mathrm{Irrig}(i) = 0. \tag{9}$$

[41] In this category, ET computed from the soil water budget model does not include direct ET from groundwater (e.g., wetlands, riparian vegetation).

[42] In the first step, we use the soil water budget model to compute daily ET (on day i):

$$\mathrm{ET}(i) = k_c{}^{*}\mathrm{ET}_0(i) = 0.6^{*}\mathrm{ET}_0(i) \quad \text{subject to}: \ \mathrm{ET}(i) \le \theta(i-1) + P_{\mathrm{adj}}(i). \tag{10}$$

[43] This latter constraint distinguishes this category from an irrigated crop.

3.3. No Irrigation/No ET Category

[44] Landuse categories of this type do not receive irrigation, and they also are not subject to evaporation or evapotranspiration from plants:

$$\mathrm{Irrig}(i) = 0 \text{ at all times} \tag{11a}$$

$$\mathrm{ET}(i) = 0 \text{ at all times.} \tag{11b}$$

[45] Given the flat topography of the valley floor, runoff is here considered negligible and recharge is equal to the adjusted precipitation:

$$\mathrm{Recharge}(i) = P_{\mathrm{adj}}(i). \tag{12}$$

4. The Analytical Solution for Stream Depletion

[46] Following *Jenkins* [1968], *Wallace et al*. [1990], and *Bredehoeft* [2011], we simplify the groundwater system and assume a semi-infinite, homogeneous and isotropic aquifer, with transmissivity constant in time and space; recharge to the aquifer is not considered prior to the time of interest, hence the water table is horizontal; the stream is considered to fully penetrate the aquifer; wells also fully penetrate the aquifer; and constant rate pumping starts at time $t = 0$.

[47] Under those assumptions, stream depletion due to pumping is given by *Jenkins* [1968]:

$$\boldsymbol{\frac{q}{Q} = erfc\left(\frac{t_a}{4t}\right)^{1/2} \quad t < t_p} \tag{13}$$

where: $t_a = \frac{a^2 S}{T}$ is the Stream Depletion Factor (SDF) defined by *Jenkins* [1968] and used by *Bredehoeft* [2011]; q is the change in rate of streamflow caused by the well pumping; Q is the rate of pumping; a is the distance of the well from the stream; S is the aquifer storativity and a value of 0.12 is used for the (unconfined) Scott Valley system; T is the aquifer transmissivity; t is time since pumping began; and t_p is the duration of pumping.

[48] The stream depletion after pumping stops at $t = t_p$ is calculated following *Wallace et al*. [1990]:

$$q = Q\left(erfc\left(\frac{t_a}{4t}\right)^{\frac{1}{2}} - erfc\left(\frac{t_a}{4(t-t_p)}\right)^{\frac{1}{2}}\right) \quad t_p \le t < \infty. \quad (14)$$

[49] The rate of stream depletion due to nonsteady, annual cyclical pumping is calculated using (equation (15)) and the principle of superposition. As shown by *Wallace et al*. [1990], for constant t_p and t_d, the stream depletion corresponds to:

$$q = \sum_{i=0}^{N-1}\delta(t - t_d i)\left(Q\ erfc\left(\frac{t_a}{4(t-t_d i)}\right)^{\frac{1}{2}}\right) - \sum_{i=0}^{N-1}\delta(t - t_p - t_d i)\left(Q\ erfc\left(\frac{t_a}{4(t-t_p-t_d i)}\right)^{\frac{1}{2}}\right) \quad 0 \le t < \infty \quad (15)$$

where δ is the unit step function which has a value of 1 when its argument is greater than zero and a value of zero when its argument is equal or less than zero; N is the number of time the pump is turned on; t_d is the interval at which the pattern repeats itself.

5. Coupling Soil Water Budget and Stream Depletion Model

[50] Analytical solutions for simplified stream-aquifer depletion evaluation were originally developed and used for investigations that were lacking today's computer resources. These analytical tools remain attractive, partly because of the computational efficiency and relative ease of implementation, typically with spreadsheets or simple computer programs. More importantly, they are powerful tools that provide fundamental, rigorous theoretical insight into the physical behavior of the groundwater-stream system, even if under highly simplified, hypothetical conditions [*Jenkins*, 1968; *Glover*, 1974; *Wallace et al*., 1990; *Hunt*, 2003; *Bredehoeft and Kendy*, 2008]. In a complex and often misunderstood management system such as the aquifer-stream system, these simplified approaches allow for quickly establishing major operational constraints imposed by basic system variables. Here coupling the water budget model with the analytical solution for stream depletion provides a framework (a) to estimate the magnitude of streamflow depletion and its sensitivity to key system parameters, (b) to implement a benchmark test against an existing numerical model, and (c) to develop management scenarios for discussion and analysis with stakeholders.

[51] The soil water budget model and the streamflow depletion model are coupled, first, by assigning the estimated pumping in each field to its nearest existing well. Active wells in Scott Valley are identified through a review of well drilling permits, GIS analysis, and partial, randomized on-the-ground verification. If multiple wells are located within one landuse polygon, the total pumping is evenly split between wells, while the pumping from a well that is serving multiple polygons is the sum of all daily water needs in the associated fields. Secondly, recharge from each polygon is similarly assigned to the nearest well, but as an injection rate (negative pumping rate). Then, 1990–2011 net daily groundwater pumping rates at each well are computed as the difference between daily groundwater pumping and groundwater recharge assigned to the well.

[52] The distance of each well to the stream is computed as the orthogonal distance from the well to the Scott River, not to the nearest tributary. Here streamflow in the main stem Scott River is the key concern (Figure 1). Transmissivity is obtained from *Mack* [1958] and *SSPA* [2012].

[53] Finally, the superposition principle (equation (15)) is applied to show the effect of transient, combined recharge and pumping on the total streamflow depletion rate along the integrated length of the Scott River within the Scott Valley. We apply each well's average, yearlong net pumping time series cyclically until a dynamic (cyclical) steady state is achieved in annual stream depletion rates. Convergence is considered to be achieved once all wells exhibit less than 1% change in relative depletion on all calendar days. Using the results of the final cyclical year, the 163 wells' computed daily stream depletion (or stream replenishment) rates are summed to obtain a time series of the net total daily stream depletion of the Scott River ("base scenario").

[54] We apply the tool to several additional scenarios to demonstrate the sensitivity of the solution to the SDF parameters, to compare the estimated streamflow impacts from changes in pumping and recharge stress with those obtained with a fully three-dimensional groundwater model, and to outline potential impacts of alternative groundwater management practices that affect timing and amount of additional recharge when additional surface flows are available and the distribution of groundwater pumping.

6. Results and Discussion

6.1. Soil Water Budget

[55] The water budget simulation provides daily soil water fluxes in water years 1991 through 2011, which are aggregated to monthly, yearly and long-term averages. Table 2 summarizes average annual fluxes, by landuse category. The total amount of annual recharge (groundwater system input) from the irrigated landscape is on the order of 46 $Mm^3\ y^{-1}$ (37 thousand acre-feet per year [TAF y^{-1}]). Groundwater pumping (groundwater system output) is about 25% larger, nearly 55 $Mm^3\ y^{-1}$ (44 TAF y^{-1}). Surrounding non-irrigated landuses, including dry land farming and riparian vegetation, contribute 26 $Mm^3\ y^{-1}$ (21 TAF y^{-1}) to basin recharge, mostly from winter precipitation, with 23 $Mm^3\ y^{-1}$ (18 TAF y^{-1}) of water uptake by natural vegetation and dry land crops (not including direct groundwater uptake). The ET demand from natural vegetation and dry land farming (274 mm) is provided through

Table 2. Average Annual Soil Water Fluxes, Water Years 1991–2011, for Irrigated Crops, for Dry Land Farming and Natural Vegetation Areas ("ET noIRR"), and for Areas With No Consumptive Water Use ("noET noIRR")[a]

	Crop ET	Actual ET	Irrigation	SW Irrigation	GW Pumping	Recharge	Deficiency	Area (ha)
mm y^{-1}								
Alfalfa	1068	1018	840	104	736	370	49	5,622
Grain	411	409	358	55	303	467	2	803
Pasture	1017	861	755	528	228	437	155	4820
ET noIRR	284	274				273	10	8240
noET noIRR						547		686
Mm y^{-1}								
Alfalfa	60.0	57.3	47.2	5.8	41.4	20.8	2.8	5622
Grain	3.3	3.3	2.9	0.4	2.4	3.8	0.0	803
Pasture	49.0	41.5	36.4	25.4	11.0	21.1	7.5	4820
ET noIRR	23.4	22.6				22.5	0.8	8240
noET noIRR						3.8		686

[a]"SW": surface water, "GW": groundwater. Deficiency refers to the difference in ET between optimal water supply ("Crop ET") and actual, limited water supply ("Actual ET").

spring precipitation with the dominant source coming from root zone water storage filled during the cold winter rainy season. The nominal deficit in natural vegetation is small, but for this category, recharge and deficit are highly sensitive to the selected k_c (0.6): if k_c values are chosen higher, the deficit is correspondingly higher (due to water availability being limited) with no simulated impact on groundwater; if k_c values are chosen lower, the simulated deficit decreases or disappears and additional groundwater recharge would occur, depending on the annual dynamics of the crop coefficient.

[56] Early spring groundwater levels in the basin do not experience a long-term declining or increasing trend indicating a balanced groundwater budget (Figure 2b). The net surplus of 17.1 Mm^3 y^{-1} (14 TAF y^{-1}) between recharge and pumping across the basin indicates a net inflow from the groundwater basin to the Scott River. However, the model does not account for annual direct recharge from the stream system to groundwater that is subsequently discharged back to the stream. Both, actual recharge from and groundwater discharge to the stream are likely larger, due to the complex interaction of the groundwater system with streams and tributaries that are not accounted for here. This includes hyporheic zone exchanges due to streambed topography and groundwater-surface water exchanges due to the larger scale streambed and water table variability [e.g., *Wondzell et al*., 2009; *Boano et al*., 2010].

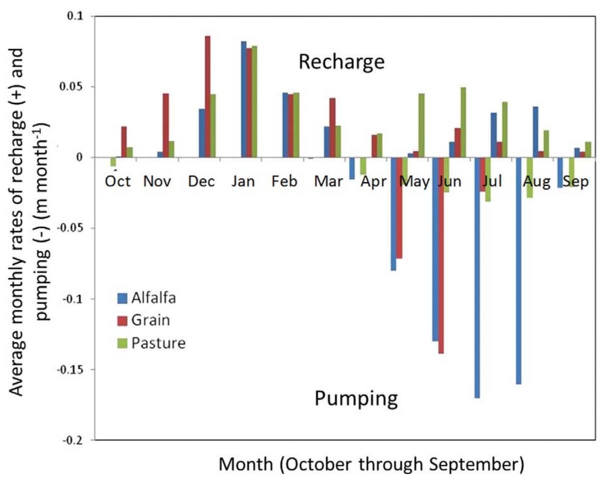


Figure 3. Simulated monthly rates of recharge and pumping (m month^{-1}) for each of the three main landuses as calculated with the water budget model.

[57] Irrigation amounts are highest in alfalfa, 840 mm y^{-1}, due to continuous availability of groundwater (736 mm y^{-1} of simulated groundwater pumping) (Table 2). Grains have an early and much shorter cropping season than alfalfa, with lower ET rates and, hence, lower irrigation (358 mm y^{-1}). Pasture, while irrigated much more generously when surface water supplies are available and with crop ET rates comparable to alfalfa (Figure 3), has a lower average annual irrigation rate (755 mm y^{-1}) than alfalfa. This is due to the surface water limitations on this predominantly surface water irrigated crop. Some pasture areas near the western margin of the valley are subject to direct groundwater uptake (not accounted for here).

[58] Average monthly recharge and pumping rates indicate strong seasonal variations. Most pumping occurs during the summer months. Most recharge occurs in the late winter and early spring (Figure 3). On pasture, significant recharge also occurs during the irrigation season due to widespread surface water flooding at rates that are significantly higher than crop water use (relatively lower irrigation efficiency). In August–September, streamflow available for flood irrigation decreases significantly, thus lowering recharge in pasture. Few pasture fields, often wheel-line sprinkler irrigated, switch to groundwater as a water source. Recharge in alfalfa is highest in July and August, when all fields are fully irrigated. Fields in grains (12.5% of the alfalfa/grain cropping area) are fallow after their harvest in July, which causes recharge and pumping in those areas to become nearly negligible after harvest. During the winter months, differences in the amount of recharge between the three landuses reflect varying levels of soil moisture depletion and slight differences in average soil characteristics across each landuse type, in particularly water holding capacity. Although very different in seasonal dynamics (Figure 3), annual average recharge in alfalfa/grain fields and pasture is not dissimilar (Figure 4b). Alfalfa has a simulated average recharge of 370 mm y^{-1},

Table 3. Total Amount of Simulated Irrigation Water Applied to Alfalfa, Grain and Pasture in a Typical Dry (2001) and Typical Wet Year (2003) in mm y^{-1}

	Dry Year		Wet Year	
	Ground Water Applied (mm y^{-1})	Surface Water Applied (mm y^{-1})	Ground Water Applied (mm y^{-1})	Surface Water Applied (mm y^{-1})
Alfalfa	862	50	723	83
Grain	419	29	397	55
Pasture	178	361	167	636
Total	701	326	596	573

about 20% lower than the average grain and pasture recharge of 467 and 437 mm y^{-1}, respectively (Table 3), but significant between-field variability exists due to varying soil water holding capacity.

[59] Few field data exist to confirm the soil water budget results. While simulated ET in alfalfa is consistent with *Hanson et al*. [2011a], the simulated average annual irrigation amounts for alfalfa (840 mm) and grain (358 mm) are found to be significantly higher than reported by growers in the study area: Preliminary field monitoring data for the 2012 irrigation season and interviews with growers on irrigation practices indicate that actual irrigation rates may be on the order of 500–600 mm in alfalfa and 150–200 mm in grain. Lower irrigation rates, when using groundwater for irrigation, may be due to overestimation of ET due to deficit irrigation, direct groundwater uptake by the crop, not accounted for in the model, or due to underestimating root zone depth and, hence, soil moisture storage capacity. Deficit irrigation has been found to lower ET by as much as 55 mm in Scott Valley and up to 200 mm elsewhere [*Hanson et al*., 2011b]. Lower ET would lower the net stress on groundwater. Direct groundwater uptake, where it occurs in groundwater irrigated areas, does not change the simulated net stress to the aquifer obtained from the soil water budget model unless it also affects crop ET. Doubling the water holding capacity (effectively assuming a thicker root zone) reduces simulated irrigation requirements by 3% in alfalfa and only 1% in grain, thus not explaining the discrepancy with observed irrigation rates. New field work was initiated among the study area stakeholders to obtain representative measurements of soil water dynamics, irrigation rates, evapotranspiration and the occurrence of deficit irrigation that can be used in the future to improve soil water budget simulations.

[60] Analysis of the spatial distribution of annual average values over the 21 year period for surface water irrigation, recharge, pumping, and pumping minus recharge (Figure 4) provides useful insight to evaluate the differences in irrigation amount and pumping based on landuse and water source. Some key observations include:

[61] 1. Highest recharge rates (Figure 4b) occur in polygons with pasture as landuse and with groundwater as water source due to relatively low irrigation efficiency and long irrigation season; also in the non-vegetated mine tailings at the southern end of the valley and in areas with very small water holding capacity;

[62] 2. Highest pumping rates occur in the few polygons with pasture as landuse and groundwater as water source

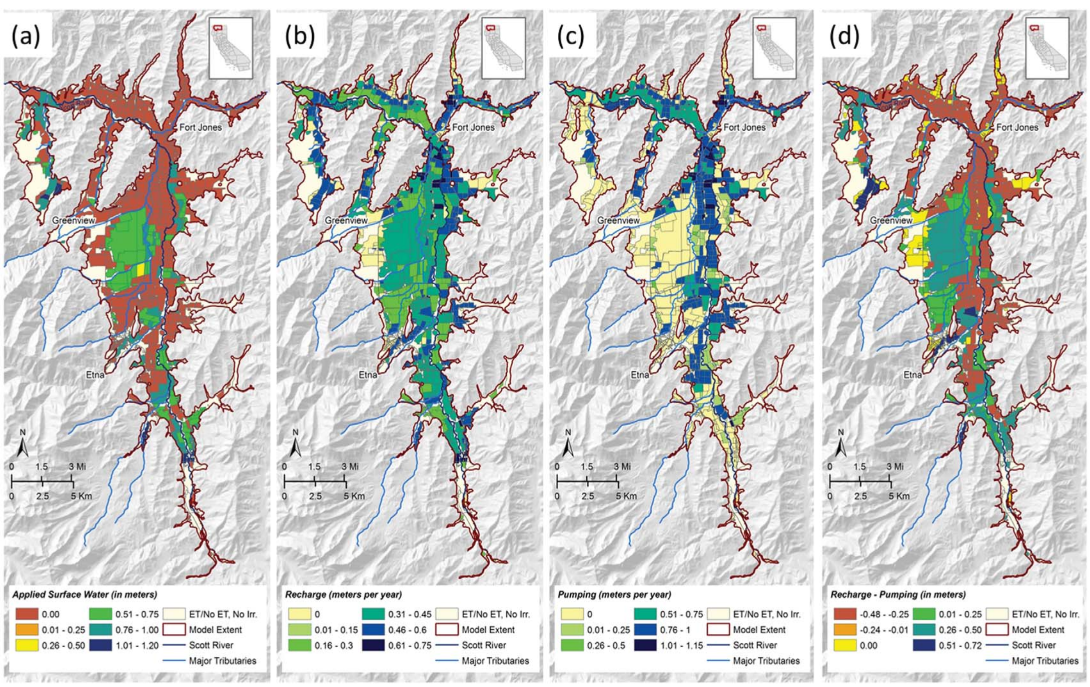


Figure 4. Water budget simulation results: (a) Average annual applied surface water rates (m y^{-1}) in irrigated crops between October 1990 and September 2011; (b) Average annual recharge (m y^{-1}) in irrigated areas between October 1990 and September 2011; (c) Average annual irrigation pumping rates (m y^{-1}) between October 1990 and September 2011; (d) Average annual difference between recharge (positive) and pumping (negative) (m y^{-1}) in irrigated areas between October 1990 and September 2011.

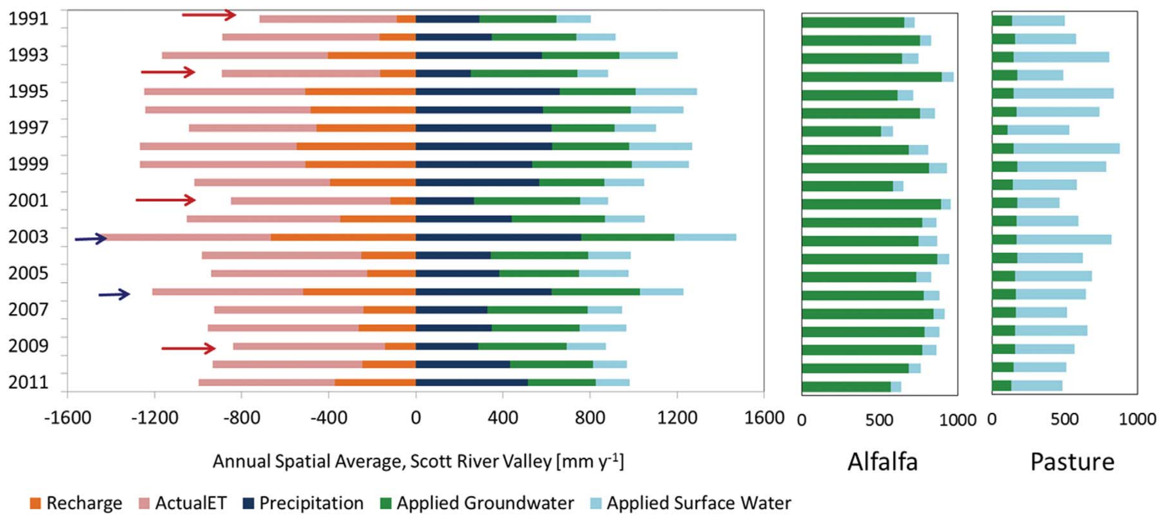


Figure 5. (a) Annual soil root zone water budget (mm y^{-1}), area-weighted average for the alfalfa/grain and pasture area in the Scott Valley. Input to the root zone shown as positive values (precipitation, applied groundwater and applied surface water). Outputs from the root zone are shown as negative values (actual ET and recharge). Annual applied surface water and annual applied groundwater (mm y^{-1}) for (b) alfalfa/grain and (c) pasture, area-weighted average over all alfalfa/grain landuse polygons in the project area. Critically dry years are highlighted in red and wet years are highlighted in blue.

(Figure 4c): this can be explained by the fact that pasture has the longest irrigation season. In polygons with groundwater as water source, the estimated irrigation rate is equal to the estimated pumping rate and it is not limited by (surface) water availability;

[63] 3. The lowest recharge rates occur in polygons that correspond to dry land farming or natural vegetation. They rely on precipitation as water source for plants, which are effective at extracting available moisture;

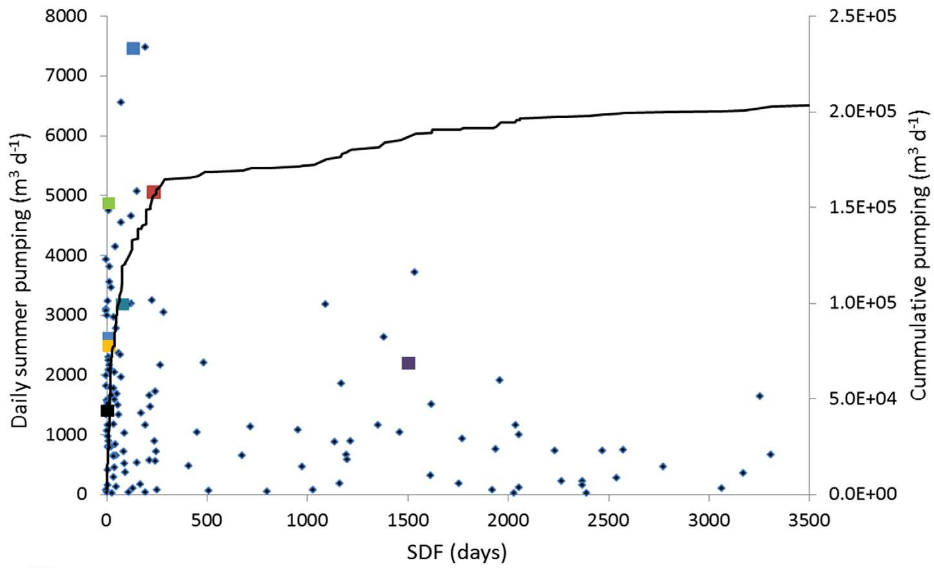
[64] 4. Since irrigation is driven by ET and irrigation efficiency, there is no water deficiency during the irrigation season. The water deficiency shown in Table 2 occurs mostly in the months immediately following the end of the irrigation season (September, October, and November) and prior to winter dormancy. In practice, much higher deficiencies may occur in wheel-line and center pivot sprinkler irrigated crops, as possibly indicated by preliminary data on field irrigation rates.

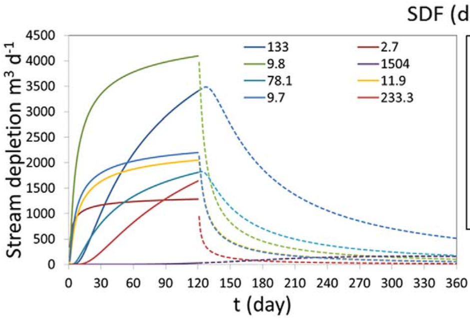
[65] Significant differences in water flows are found between dry years and wet years (Figure 5 and Table 3). Valley wide recharge to groundwater is significantly lower in dry years (as little as 100 mm y^{-1}) than in wet years (over 600 mm y^{-1}). Low recharge in dry years is mostly due to lack of streamflow from the surrounding watershed and, hence, lower amounts of applied surface water (Table 3). Dry year surface water irrigation is only 60% of wet year surface water irrigation. Changes in groundwater pumping due to dry year conditions are relatively small when compared to the large reductions in surface water irrigation, as is common in semiarid regions [*Ruud et al*., 2004]. Dry years, therefore, significantly affect the agricultural productivity of the Scott Valley with most impact focused on pasture areas (Figure 5c).

[66] Simulated groundwater use in alfalfa, on average, is about 16% higher in dry years than in wet years. Higher groundwater use in dry years is driven mostly by higher evapotranspiration from alfalfa/grain landuses early in the growing season, demanding a higher irrigation amount. Less importantly here, higher groundwater use in dry years is also due to limited surface water availability on those fields equipped to switch from surface water to groundwater (Figure 5b). Groundwater irrigated pasture land is the exception (Figure 5c). The amount of applied groundwater, driven by spring precipitation, ET, and soil moisture availability, varies within a limited range throughout the 21 year period because there are no significant differences in the

Table 4. Summary of the Data on the Eight Wells Selected for the Analysis (for Location, See Figure 3)

SDF (d)	Polygon	HK (m/d)	Storage Coefficient	Aquifer Thickness (m)	Transmissivity (m^2/d)	Distance From the River (m)	Daily Pumping (m^3/d)
2.7	595	45	0.12	45.4	2042	215	1400
9.7	88	45	0.12	40.5	1821	385	2620
9.8	46	45	0.12	44.7	2013	405	4870
12	414	45	0.12	44.7	2013	446	2490
78	226	45	0.12	42.3	1905	1114	3180
133	103	45	0.12	39.6	1782	1407	7460
233	617	12	0.12	66.8	801	1248	5060
1503	1728	12	0.12	32.5	390	2211	2200





	# wells	%	total pumping	%
SDF < 10 d	34	24	67,566	33
10d < SDF < 1000 d	83	59	126,815	61
SDF > 1000 d	24	17	13,164	6

Figure 6. Daily cumulative summer pumping as a function of the stream depletion function (SDF, blue dots) for all 163 wells (Figure 1). For eight wells shown in larger colored squares, the graph on the lower left shows simulated stream depletion over 1 year, assuming 120 days of constant pumping and 240 days without pumping (in corresponding colors). Solid lines represent the pumping period and dashed line the subsequent period without pumping. The eight wells are labeled by their SDF (d) (also see Figure 1 for location).

length of the irrigation season between different years. Where the water source is groundwater, irrigation continues for the entire irrigation season, unaffected by surface water availability. This does not account for grower responses to climate, such as increasing/decreasing deficit irrigation.

6.2. Scott River Stream Depletion Dynamics

[67] The stream depletion factor, t_a [SDF; *Jenkins*, 1968] associated with each of the 163 wells identified (Figure 1) varies from less than 1 day to over 3600 days. High SDF values lead to slow stream depletion and vice versa. The SDF increases (stream depletion slows down) with increasing aquifer storage coefficient and distance. But the SDF decreases (stream depletion occurs more rapidly) with higher transmissivity between the well and the stream (equation (13)). Distance, varying over orders of magnitude from few meters to several kilometers is the key controlling variable for the variability of the SDF across Scott Valley. In contrast, the storage coefficient, here assumed constant, has been found to vary within a relatively narrow range throughout most of the valley (7–15%) [*Mack*, 1958]. Regional hydraulic conductivity varies by about half an order of magnitude between subareas, significantly influencing SDF. Hydraulic conductivity has been estimated from short-term pump tests to evaluate the specific capacity of wells, typically performed during well construction [*Mack*, 1958; *SSPA*, 2012]. Accuracy of these estimates may be limited, as they reflect local conditions in the immediate vicinity of the well, rather than effective conditions. However, total (integrated) stream depletion in the Scott River is less sensitive to random errors of local transmissivity estimates than to systematic under or overestimation of transmissivity across multiple wells, especially those with small SDF. This suggests that further field evaluation of hydraulic conductivity is needed, particularly near high capacity wells in close proximity to the river.

[68] Spatial distributions of crop type and the SDF values show some similarities: alfalfa/grain fields are concentrated in the vicinity of the Scott River, where well capacity is likely higher due to coarser and thicker sediments with higher aquifer transmissivity and with low SDF (Figure 6, equation (13)). Pasture fields are often located away from the Scott River in areas with higher SDF (Figure 1), and are irrigated with surface water from tributaries emanating off the surrounding canyons.

[69] Considering stream depletion due to average seasonal pumping at eight selected wells with a wide range of

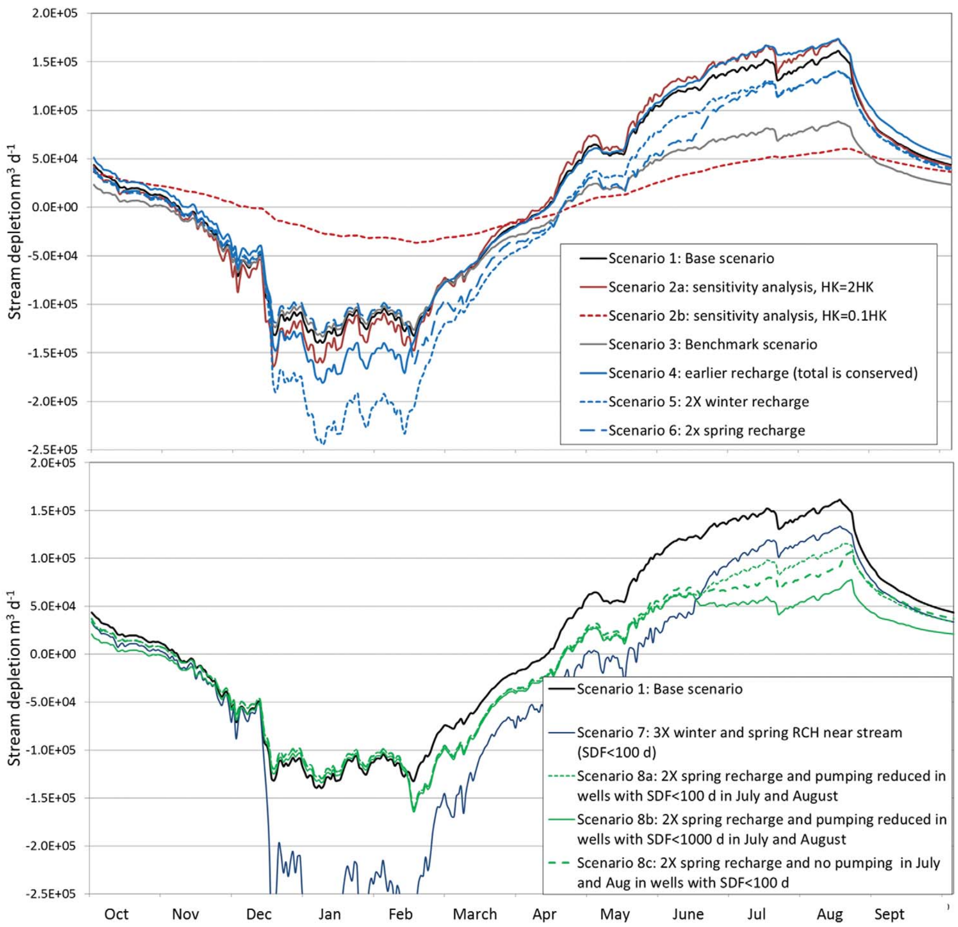


Figure 7. Simulated total daily stream depletion of the Scott River in response to 1991–2001 average daily varying net stress (pumping minus recharge), spatially distributed across the Scott Valley. Results represent a cyclical, dynamic equilibrium. Absolute stream depletion values are subject to significant uncertainty due to parameter uncertainty (compare Scenarios 1, 2a, 2b) and the simplicity of the conceptual approach, but relative changes in stream depletion over time and between management scenarios (Scenarios 3–8) provide guidance on the magnitude of stream depletion changes affected by managed changes in recharge and pumping. Note: 100,000 m^3 d^{-1} corresponds to approximately 40 cfs.

SDF values (Table 4) indicates that wells with very small SDF (<10), lead to measurable stream depletion within hours to few days after the onset of pumping. About half of the full depletion effect occurs within approximately one week. Within 2 months, the stream is affected at 90% of the full depletion rate (Figure 6). For SDFs on the order of 100, significant effects on stream depletion are observable within less than 1 month and increasing impacts occur throughout the 4 month pumping season. Only wells with SDF $>$ 1000, have limited effect on stream depletion during the 4 month pumping season. Climate variability would therefore exacerbate stream depletion: dry years lead to more stream depletion during the later summer months due to reduced basin-wide spring and summer recharge (total runoff in the Scott River, e.g., in 2009, was less than 45% of average), while groundwater pumping to support crop irrigation remains unchanged or maybe even somewhat higher than in average or wet years due to increased crop ET.

[70] Wells with SDF of less than 10 days represent 24% of all wells, but 33% of the total pumping. This is consistent with the alluvial hydrogeology of the valley, which dictates that larger capacity wells are located closer to the river, where aquifer thickness is large and sediments are coarsest. For the same reason, wells with an SDF of over 1000 days represent less than one-fifth of all wells (17%) delivering merely one-twentieth (6%) of the total pumpage (Figure 6).

[71] Cyclical simulations based on average daily pumping rates converge to a dynamic steady state only after 20 years, due to the long-term effects of wells with high SDF on stream depletion. The CPU time for computing 20 years of stream depletion due to daily varying net pumping stresses across 163 wells and for performing the convolution is 470 s (0.13 h) on a PC with Intel(R) Core^{TM} i7–3520M CPU @ 2.90GHz and 64-bit operating system. In comparison, a fully integrated, three-dimensional numerical hydrological model with sufficient resolution to resolve individual landuse parcels requires about 8 h using monthly stresses for 21 years on the same platform.

[72] For the base scenario, the maximum total stream replenishment (negative depletion) in the study area occurs

from mid-December through mid-February, at approximately 125,000 $m^3\ d^{-1}$ (50 cfs), while the largest stream depletion occurs in August, at approximately 150,000 $m^3\ d^{-1}$ (60 cfs) (Figure 7). The latter represents slightly more than one-third of the simulated peak groundwater pumping rate, nearly 400,000 $m^3\ d^{-1}$ (160 cfs) in July.

[73] Summed over the entire year, the stream depletion model, which assumes an infinite aquifer, yields a small net annual stream depletion despite the water budget of the study area showing more recharge than pumping (Table 2). Due to the high streamflows during November through June (in excess of 250,000 $m^3\ d^{-1}$ [100 cfs]), stream depletion is here only of concern during the summer period. During that period, existing winter and spring recharge is not sufficient to offset summer groundwater pumping effects on stream depletion due to the large number of wells with SDF $\ll$ 1000 days and especially those with SDF $<$ 10 days.

[74] If the selected transmissivity values for the base scenario consistently underestimated actual aquifer transmissivity by a factor 2, actual stream depletion during the critical period in July and August would be about 9200 $m^3\ d^{-1}$ (3.8 cfs) more than estimated with the base scenario (Figure 7, Scenario 2a). Similarly, if actual transmissivity in the Scott Valley consistently were only half of the values assumed for the base scenario, actual stream depletion due to the same stresses would be 9200 $m^3\ d^{-1}$ (3.8 cfs) lower than in the base scenario (not shown). The transmissivity term in (13) is an effective transmissivity for the flow between a well and the stream. If the aquifer is heterogeneous or flow paths are constricted, especially near the stream, the lowest transmissivity values along the flow path between a well and a stream would dominate the effective value. If such factors reduced the effective field transmissivity between Scott River and wells to 10% of that assumed in the base scenario, actual stream depletion in July and August would be about 80,000 $m^3\ d^{-1}$ (33 cfs) less than in the simulated base scenario. This shows that estimated stream depletion is highly sensitive to actual hydraulic conductivity and flow configuration, especially near the stream.

[75] To understand the accuracy of predictions based on equation (15), *Sophocleous et al*. [1995] analyzed the predictive accuracy of the *Glover* [1954] stream-aquifer analytical solution with a numerical groundwater flow model. Across a range of aquifer conditions, assumptions in the analytical solution were tested, e.g., by removing the hydraulic equilibrium conditions. Generally, the analytical solution overestimated stream depletion suggesting that the analytical solution approach leads to a relatively conservative assessment in guiding decisions about water rights administration. A rank of the importance of the various assumptions involved in the derivation of the analytical solution was presented and the three most significant factors were: (1) streambed clogging, as quantified by streambed-aquifer hydraulic conductivity contrast, (2) degree of stream partial penetration, and (3) aquifer heterogeneity. Aquifer width, not considered by the SDF, has also been demonstrated to be important [*Miller et al*., 2007].

[76] Streambed clogging or low streambed hydraulic conductivities (relative to the aquifer) may be addressed by applying the method of additional seepage resistance [*Sophocleous et al*., 1995] to raise the SDF value. In our study area, it is unlikely to play an overriding role due to the absence of fine materials in the streambed and frequent scouring and redeposition of streambed materials during the high flow season. The effect of partial well penetration on stream depletion has also been shown to be small [ibid].

[77] The range of maximum stream depletion obtained from this sensitivity analysis (54,000–143,000 $m^3\ d^{-1}$ (22–55 cfs)) provides a coarse approximation of possible actual stream depletion in July and August given the pumping and recharge distribution simulated for the Scott Valley. This range would be proportionally lower, if actual ET, especially in alfalfa, will be shown to be lower in the Scott Valley than simulated here, due, e.g., to deficit irrigation.

[78] A benchmark test (Scenario 3) is used to perform an independent assessment of the order of accuracy provided by this simplified stream depletion analysis, when used to provide predictions of changes in stream depletion due to certain changes in pumping and recharge. For the benchmark test, results of the coupled water budget-stream depletion model are compared against a third-party fully 3-D, numerical, cyclical steady-state groundwater model that represents year 2000 conditions in groundwater pumping and recharge. The spatial distribution of pumping and recharge is qualitatively similar to that of our soil water budget model, but not identical [*SSPA*, 2012]. The numerical model simulates a partially penetrating streambed and its streambed hydraulic conductivity has been calibrated against measurements of well water levels. Aquifer hydraulic conductivities vary across the valley, but are of similar magnitude in both models (7–45 m d^{-1}). For the benchmark test, basin-wide net groundwater extraction (pumping minus recharge) is reduced by approximately equivalent amounts, 12.0 $Mm^3\ y^{-1}$ (13.5 cfs) in the numerical model, and 14 $Mm^3\ y^{-1}$ (15.7 cfs) in the analytical model. The resulting late summer reduction in streamflow (July–September) depletion reported for the numerical groundwater model is 39,000 $m^3\ d^{-1}$ (16 cfs). The corresponding reduction estimated with our simple analytical model is 50,000 $m^3\ d^{-1}$ (21 cfs). The analytical model results, while exceeding the numerical estimates by 25%, are sufficiently consistent with the numerical results to consider this tool useful for evaluating broad options for pumping and recharge that can guide preliminary planning for alternative groundwater management practices to evaluate.

6.3. Groundwater Management Scenarios

[79] With surface water storage not available at the scale required for agricultural water use in the basin, the groundwater basin is the *de facto* storage basin to hold water from winter and spring recharge for irrigation water use during the summer. As in other semiarid and arid basins, groundwater is a key local water management instrument to extend the cropping season beyond that possible without power pumps, especially in dry years.

[80] The water budget model indicates that there are broad opportunities to redistribute surface water available during the wetter periods of the year for irrigation water use during the dry season. Alternative management practices may include those affecting groundwater recharge, practices affecting groundwater pumping, or both. In the

past, changes in recharge have occurred due to changes in landuse, and due to changes in irrigation efficiency and methods in the Scott Valley. Given the soil water budget results, switching from mostly flood irrigation to wheel-line sprinkler irrigation between the 1950s and the 1970s had a significant impact on the timing and amount of recharge. It also incentivized the much increased use of groundwater since pumps were needed to pressurize wheel-line sprinklers and, later, center pivot sprinklers (introduced during the late 1990s and 2000s). *Van Kirk and Naman* [2008] suggested considering the difference in irrigation efficiency between flood irrigation and sprinkler irrigation.

[81] Management scenarios 4 to 7 highlight potential benefits to stream depletion during the critical summer months by managing groundwater recharge during seasons with high streamflow. Scenario 4 illustrates the effect of recharge timing, while keeping the total annual recharge amount the same as in the base scenario: recharge timing is moved from spring and early summer months to January–February, a difference that may occur naturally between individual years due to interannual climate variability. Having recharge occur earlier in the year, albeit at the same total amount, increases stream depletion in July and August by nearly 10% (by 15,000 m^3 d^{-1}, 6 cfs) over the base scenario (Figure 7). In contrast, hypothetically doubling the amount of (already high) recharge in January–February while keeping recharge during other months identical to that in the base scenario (Scenario 5) reduces July and August stream depletion by 16,000 m^3 d^{-1} (7 cfs) (Figure 7). Additional recharge in January and February would not significantly interfere with agronomic practices as crops are dormant, if aquifer storage capacity is available.

[82] Stronger reduction in streamflow depletion may be expected when increasing the amount of recharge closer to the period of high stress in July and August. Indeed, doubling recharge in March through June rather than in January and February (Scenario 6) substantially decreases stream depletion (relative to Scenario 5) during the months with additional recharge (by as much as 30,000 m^3 d^{-1}, 12 cfs), but 3–4 weeks after the additional recharge seizes, there are no observable differences between Scenarios 5 and 6 (Figure 7).

[83] Tripling the amount of recharge during the entire first half of the year, but only in areas near the Scott River (SDF $<$ 100 d, Figure 1), yields large stream replenishment (negative depletion) for most of the winter months and into May (Scenario 7), much longer than in the base scenario. Also, through much of July and August, stream depletion is much lower than in the base scenario and never reaches base scenario levels. Although additional recharge in this scenario occurs only near the Scott River and ends on July 1, stream depletion is consistently smaller (by 8000 m^3 d^{-1}, 4 cfs) in July and August when compared to Scenario 6. A significant delay in the onset of strong stream depletion could benefit other streamflow management scenarios that rely on the enhancement of instream flows: later onset of stream depletion would result in shorter periods where additional instream flow requirements are needed. Later spring recharge (April–June) could therefore provide a particularly important management tool to limit stream depletion during the critical period of July and August. Additional surface water could be obtained through acquisition of surface water rights from the valley margin (where a discontinuation of recharge during the summer months has no detrimental effect on Scott River flow), or by creating an external surface or subsurface storage capacity [*Schneider*, 2010].

[84] Groundwater management options may not only include additional recharge, but also altered groundwater pumping patterns. These scenarios are designed following the classification of SDF values by *Bredehoeft and Kendy* [2008]:

[85] 1. Wells with SDF $>$ 1000 d (17% of the wells, Figures 1 and 6, representing 6% of the total pumping) present the most interesting pool of wells for the design of mitigation strategies. Significant recharge occurring in the areas between the wells and the stream during the spring months is sufficient to offset potential long term, delayed stream depletion from pumping during the summer months.

[86] 2. Wells with 10 d $<$ SDF $<$ 1000 d (59% of the wells, Figures 1 and 6, representing 61% of the total pumping) represent the most uncertain situation. The pumping causes significant seasonal fluctuations. Different patterns of streamflow depletion can be produced depending on the SDF value, which is subject to uncertainty due to varying aquifer properties and boundary conditions not considered in the analytical model. For example, a combination of significant additional late spring and early summer recharge, switching from groundwater pumping to surface water irrigation or increasing already ongoing surface water irrigation, while streamflows are high, may significantly dampen effects of summer pumping from these wells. In the Scott Valley case, more detailed analysis using a numerical groundwater-surface water model and additional data collection will further guide specific future decision making.

[87] 3. Wells with SDF $<$ 10 d (24% of the wells, Figures 1 and 6, representing 33% of the total pumping) have quick impact on streamflows and produce large annual fluctuations in stream depletion. Pumping may be offset by additional streamflow, which would require additional surface water rights. Pumping may also be offset by groundwater transfers that replace groundwater pumping from wells with SDF $<$ 10 d with groundwater pumping from wells with SDF $\gg$ 100 d, at least during the most impacted season (July–August).

[88] Scenarios 8a–8c investigate potential benefits obtained by jointly managing groundwater recharge and groundwater pumping. Increased recharge during spring and early summer delays the onset of significant stream depletion, while the translocation of pumping away from the river during the sensitive summer period mutes the groundwater stresses that impact streamflow most immediately. A 50% reduction of July and August pumping in the wells closest to the river (SDF $<$ 100 d, Figure 1), and replenishment of that water by additional pumping (1.6 fold) outside that zone (Scenario 8a) would potentially yield reductions in July and August streamflow depletion of 42,000 m^3 d^{-1} (17 cfs). Expanding to a hypothetical 75% reduction of pumping in the zone with SDF $<$ 1000 d (Figure 1), yields additional July and August streamflow reductions of another 37,000 m^3 d^{-1} (16 cfs) when compared to Scenario 8a (Figure 7, Scenario 8b). Alternatively, an additional streamflow depletion of 12,000 m^3 d^{-1} (5 cfs), when compared to Scenario 8a, are obtained when

completely replacing groundwater pumping in the zone with SDF < 100 d and providing that irrigation water by transporting additional groundwater pumping from outside that zone to those fields (Scenario 8c). The latter two scenarios are hypothetical designs to estimate the magnitude of possible reductions in streamflow depletion. But Scenarios 8b and 8c would impose unachievable pumping requirements on outlying areas (3.5 fold and 2.3 fold pumping increases, respectively). Reductions in streamflow depletion achieved by these scenarios therefore reflect unrealistic goals.

[89] The scenario analysis indicates that both, recharge alone and the combination of recharge and selective changes in groundwater pumping patterns yield some reductions in streamflow depletion, which is here hypothesized to yield equivalently larger instream flows. The magnitude of the simulated reductions in streamflow depletion is significant. Potential streamflow increases are on the same order as current summer flow rates in the Scott River, which sometimes fall below 24,000 m^3 d^{-1} (10 cfs) suggesting that measurable gains in streamflow can be made. Stream temperature modeling indicates that a 50% increase of these low summer streamflows may substantially reduce the extent of Scott River reaches that are above 25°C, considered lethal for salmon habitat [*North Coast Regional Water Quality Control Board (NCRWQCB)*, 2005]. Flow increases also create opportunities for creating additional local habitat.

[90] Regulatory agencies have not defined numeric objectives regarding streamflow, largely because streamflow management to protect salmonid habitat via groundwater management remains an emerging research arena [*Malcolm et al*., 2012; *Milner et al*., 2012]. Salmonid ecosystem responses to streamflow are highly variable and confounded by other factors. Local investigations of flow impacts and solutions were identified as most promising [*Milner et al*., 2012]. In the case of managing the salmonid GDE in Scott Valley, regulators envision a broad range of measures and assessments across hydrologic and ecological disciplines [*NCRWQCB*, 2007].

[91] All scenarios are based on average monthly 1991–2011 recharge and pumping conditions. Other scenarios that could be considered with this tool may account for climate variability, the transient effects of consecutive dry or wet years, as have occurred in the recent past, and artificial aquifer recharge (AR) and aquifer storage and recovery (ASR) projects [*Nelson*, 2011; *Sophocleous*, 2012]. Scenarios may include sensitivity analysis to parameters in the soil water budget model. And the analytical stream depletion model can also be implemented as a fully transient, long-term impact analysis model.

[92] The scenarios presented here are purposefully designed to mimic relatively simple, extreme management cases. While not considered accurate and subject to significant uncertainty, such scenarios enable scientists and stakeholders to better understand the relationship between management outcome (the amount of reduction in stream depletion) and the associated magnitude of specific management changes needed to affect the outcome (change in pumping and recharge operations). Such scenarios may also enhance the interaction between stakeholders and scientists [*Margerum*, 2008]. For example, the scenario analysis has prompted stakeholders to identify large tracts of alfalfa that have suitable infrastructure to use a combination of in lieu recharge (switching from groundwater pumping to surface water irrigation) and increased recharge via lowering irrigation efficiencies, during spring months while streamflows are high. Stakeholders are further considering to reintroduce beaver dams as a way to increase recharge to groundwater in the immediate vicinity of the stream, while also creating potential salmonid habitat improvements.

[93] Other issues and limitations will need to be considered in the process: Implementation of programs to translocate summer pumping toward the valley margins would require further feasibility analysis with a hydraulic groundwater model to assess the limitations imposed, for example, by the aquifer geometry and heterogeneity, with often lower transmissivity near the valley margins. The scenarios also sketch out potential routes for an assessment of legal and political issues related to transferring groundwater across property boundaries, and applying surface water to increase groundwater recharge. The economic feasibility of such management strategies would further require an assessment of infrastructure needs and costs to install the required groundwater pumping capacity and distribution system.

[94] The approach presented here identifies important groundwater management options that warrant additional analyses including the design of useful scenarios to be simulated with a fully developed numerical groundwater-surface water model [*Sophocleous*, 1995; *Neupauer and Cronin*, 2010]. The approach must therefore be considered as only one of a broader range of tools that support monitoring and assessment programs and adaptive management of groundwater-dependent streamflows under complex conditions and at multiple scales. One potential option that warrants further research is the application of this computationally efficient methodology in automated multiobjective groundwater management optimization that considers various management constraints and uncertainties. Such an application would be particularly relevant because future groundwater management in systems like the study area typically consists of a portfolio of multiple management options that optimize for economic cost, political acceptability, and desired ecologic outcome within the hydrologic constraints of the basin.

7. Conclusion

[95] The modeling approach presented here, a combination of a spatiotemporally distributed soil water budget model and an analytical streamflow depletion model, represents a powerful, computationally efficient, while conceptually simple means to effectively integrate science into a social network watershed process driven by legal and policy decisions. The tool has been applied to the Scott Valley watershed in Northern California, a groundwater-dependent ecosystem that relies on sufficient groundwater discharge into the stream during July–September. The estimation of spatiotemporally distributed recharge and pumping stresses with the soil water budget model allowed us to develop and implement a range of groundwater management scenarios to broadly bracket options that can serve as catalyst to direct stakeholder discussions, and to

demonstrate the potential range of beneficial impacts from groundwater management on stream depletion. The scenarios provide significant insights into spatial and temporal scales of measures and potential venues needed to mitigate existing conflicts between stakeholders representing local farms and those representing downstream fisheries:

[96] 1. Increased groundwater storage of winter and spring streamflow, especially near the Scott River, may significantly decrease the impact of the pumping season on streamflow depletion during the critical summer period.

[97] 2. Groundwater pumping effects in August and July could be further mitigated by transferring groundwater pumping in the most sensitive areas to wells that are some distance away from the Scott River. This would require water trading and transport infrastructure. But the analysis also identified significant limitations on the amount of stream depletion reduction that can realistically be expected.

[98] 3. Addressing uncertainty about the effective hydraulic conductivity between the stream and the aquifer due to geologic heterogeneity, due to geomorphologic complexity, and the unknown complexity of the flow field between groundwater and the stream is critical to better quantify actual stream depletion impacts. We also found that the soil water budget significantly overestimates currently reported farm irrigation rates in center pivot and wheel-line sprinkler systems, possibly due to significant, but unreported deficit irrigation. Sensitivity analysis yields a measure of uncertainty. More importantly it provides direction for critical field measurement programs and the design of more complex hydrologic models for site-specific assessment and feasibility studies of specific recharge and pumping management projects.

[99] The approach has broad merit in the initial phases of a stakeholder driven process to address groundwater-stream interactions through groundwater management, to identify broad areas of potentially feasible projects, and to convey information on the scope of potential projects and expected outcomes. The approach may possibly also be applicable, e.g., for computationally demanding complex management systems optimization applications. Further research on such applications is warranted. The approach is not intended as a tool to provide accurate, quantitative answers for site-specific assessments. Some of its components, especially in the soil water budget, can be significantly improved (e.g., by addressing ditch and canal losses, potential winter runoff, deficit irrigation and reduced ET).

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