WATER RESOURCES RESEARCH, VOL. 43, W03437, doi:10.1029/2006WR005486, 2007

Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality

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Received 31 August 2006; revised 9 October 2006; accepted 3 November 2006; published 27 March 2007.

[1] Past land use changes have greatly impacted global water resources, with often opposing effects on water quantity and quality. Increases in rain-fed cropland (460%) and pastureland (560%) during the past 300 years from forest and grasslands decreased evapotranspiration and increased recharge (two orders of magnitude) and streamflow (one order of magnitude). However, increased water quantity degraded water quality by mobilization of salts, salinization caused by shallow water tables, and fertilizer leaching into underlying aquifers that discharge to streams. Since the 1950s, irrigated agriculture has expanded globally by 174%, accounting for ~90% of global freshwater consumption. Irrigation based on surface water reduced streamflow and raised water tables resulting in waterlogging in many areas (China, India, and United States). Marked increases in groundwater-fed irrigation in the last few decades in these areas has lowered water tables ($\leq$1 m/yr) and reduced streamflow. Degradation of water quality in irrigated areas has resulted from processes similar to those in rain-fed agriculture: salt mobilization, salinization in waterlogged areas, and fertilizer leaching. Strategies for remediating water resource problems related to agriculture often have opposing effects on water quantity and quality. Long time lags (decades to centuries) between land use changes and system response (e.g., recharge, streamflow, and water quality), particularly in semiarid regions, mean that the full impact of land use changes has not been realized in many areas and remediation to reverse impacts will also take a long time. Future land use changes should consider potential impacts on water resources, particularly trade-offs between water, salt, and nutrient balances, to develop sustainable water resources to meet human and ecosystem needs.

Citation: Scanlon, B. R., I. Jolly, M. Sophocleous, and L. Zhang (2007), Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality, *Water Resour. Res.*, *43*, W03437, doi:10.1029/2006WR005486.

1. Introduction

[2] Water scarcity is a critical issue because 1.1 billion of the world's 6 billion people lack access to safe drinking water [*World Health Organization*, 2003]. Agricultural food production accounted for an average of ~90% of global freshwater consumption during the last century (Figure 1 and Table 1) [*Shiklomanov*, 2000]. Water requirements for food production will increase to meet demands of the projected 50% increase in global population from ~6 billion in 2000 to ~9 billion in 2050 [*U.S. Census Bureau*, 2004]. Meeting the United Nations Millennium Development Goal of reducing the proportion of people that suffer from hunger by 50% by 2015 will put additional stress on limited water resources. Changing diet preferences to increased meat consumption in developing countries will also elevate water consumption because meat production requires ~4000 to 15,000 liters/kg to produce versus ~1000 to 2000 liters/kg for grain production [*Renault and Wallender*, 2000; *United Nations*, 2003]. Agricultural production for other programs, such as carbon sequestration and energy production including biofuels, will also increase demands on water resources. Ecosystem needs are being recognized increasingly as environmental flow programs attempt to establish flow requirements to maintain ecosystem productivity and services [*Baron et al.*, 2002]. Although much emphasis has been placed on potential impacts of climate change on water resources [*Intergovernmental Panel on Climate Change*, 2001], impacts of land use changes on water resources, particularly those associated with agriculture, may rival or exceed those of climate change [*Vorosmarty et al.*, 2000]. A Comprehensive Assessment of Water Management in Agriculture will be published in 2007 (http://www.iwmi.cgiar.org) and is the product of a multi-institute process assessing approaches to managing water resources to increase food production and enhance environmental sustainability, somewhat similar to the Intergovernmental Panel on Climate Change.

[3] Large-scale changes in land use have occurred as a result of expansion of croplands and pasturelands at the expense of forests and grasslands. In the past 300 years, cultivated cropland and pastureland have increased globally by 460% and 560%, respectively (Figure 2 and Table 1)

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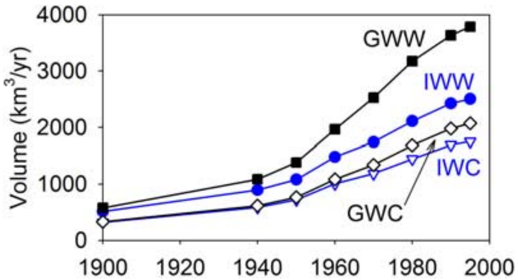

Figure 1. Irrigation water withdrawal (IWW) and consumption (IWC) relative to global water withdrawal (GWW) and consumption (GWC) [*Shiklomanov*, 2000]. Water withdrawal refers to water diverted from surface water or groundwater; however, only a fraction of this water is consumed by evapotranspiration (ET) losses or by humans or livestock.

[*Richards*, 1990; *Klein Goldewijk*, 2001]. Global cropland for food production is projected to expand by 13% (956–1076 Mha) in developing countries, primarily in sub-Saharan Africa and South America [*Bruinsma*, 2003], whereas cropland is declining in developed countries. Rain-fed agriculture accounts for ~80% of cultivated areas and produces 60% of the world's food, whereas irrigated agriculture accounts for 18% of cropland and produces 40% of the world's food [*Rockstrom and Falkenmark*, 2000] (see also Food and Agricultural Organization (FAO), FAOSTAT, http://faostat.fao.org/, accessed 2006, hereinafter referred to as FAOSTAT, 2006). Irrigated agriculture has significant impacts on water resources because it withdraws an average of ~80% of global freshwater and consumes ~90% of freshwater (1900–1995) (Figure 1 and Table 1) [*Shiklomanov*, 2000]. Irrigated agriculture has expanded by 480% from 1900 to 2000 and is projected to increase by 20% (202– 242 Mha) by 2030 in developing countries (Table 1) [*Bruinsma*, 2003; *Siebert et al.*, 2005; FAOSTAT, 2006]. Understanding impacts of past land use changes on water resources will be essential to predicting potential impacts of future changes.

[4] Because of increased stresses on limited water resources, management strategies are shifting from water supply management to water demand management. Consumptive water uses are described using the concept of blue and green water introduced by *Falkenmark* [1995]. Precipitation represents the original source of water that is partitioned into (1) water consumed in plant production and evaporation from moist surfaces such as lakes and reservoirs (green water flow) and (2) the surplus recharges aquifers and streams (blue water flow) which is available for humans and ecosystems [*Falkenmark and Lannerstad*, 2005]. Initial cultivation through expansion of rain-fed agriculture decreased green water flow through reduced ET and increased blue water flow through groundwater recharge and streamflow by up to 3800 km^3/yr around 1950 [*Falkenmark and Lannerstad*, 2005]. However, development and expansion of irrigated agriculture, mostly since the 1950s, greatly reduced blue water flow because of high consumptive use (90% of freshwater consumption) [*Shiklomanov*, 2000; *Falkenmark and Lannerstad*, 2005]. Because of large-scale impacts of land use changes, including cultivation and irrigation, on water resources, it is essential to integrate land and water resource management (ILWRM), as is recognized in the blue revolution proposed by *Calder* [2005]. The blue revolution is a conceptual framework that addresses issues related to sustainability of water resources for food production, human needs, industry, power, ecology, and the environment.

[5] This paper reviews and summarizes impacts of conversions from natural to agricultural ecosystems on quantity and quality of water resources. Unique aspects of this overview include compilation of studies conducted throughout the globe; emphasis on agriculture, which is the primary consumer of water resources; evaluation of impacts on groundwater and surface water; distinction between impacts on water quantity and water quality; and application of the current state of scientific understanding to guide sustainable water resources management within the context of future land use changes. The review is not meant to be comprehensive and limits discussion of water quality to salts and nitrate, and omits many other parameters including sediment, phosphates, pesticides, herbicides, and pathogens. Issues related to environmental flows are not examined. Although climate impacts are not discussed explicitly, many impacts of land use change vary with climatic setting and are discussed accordingly.

[6] The terms land cover and land use are used interchangeably in this paper. The distribution of land cover for 2000 is based on MODIS (1 km) Global Land Cover (Figure 3). A total of 17 International Geosphere Biosphere Program (IGBP) land cover classes were aggregated into 9 classes, including forests (5 classes, 21%), shrublands (2 classes, 19%), savannas (2 classes, 14%), grassland (9%), cropland (9%), mosaic of cropland and natural vegetation (2%), barren (14%), urban (0.5%), wetland (0.2%), and other (snow and ice, water bodies) (11%) (Figure 3) [*Giri et al.*, 2005]. The IGBP classes are defined as forests (>60% woody vegetation >2 m height), shrublands (>10% shrub canopy cover <2 m height), savannas (herbaceous and other understory cover with 10–30% forest), grassland (herbaceous cover with <10% tree and shrub cover), cropland (crops and bare soil under fallow), and mosaic of cropland and natural vegetation (no component >60%) [*Giri et al.*, 2005]. Definitions of land cover classes vary widely. For example, the *FAO* [1997] definition of forest is tree crown cover of ≥10%, area ≥0.5 ha, and canopy height >5 m at maturity. Forest plantations generally consist of forest stands established by planting and/or seeding of introduced species or intensively managed indigenous species during afforestation or reforestation [*FAO*, 1997].

2. Impact of Land Use Changes on Water Resources

[7] The impact of land use changes on water resources depends on many factors, including the original vegetation that is being replaced, the vegetation that is replacing it, whether it is a permanent or temporary change, and associated land management practices such as tillage, fallow periods, and related water and nutrient applications, such as irrigation and fertilization. A schematic of the basic natural ecosystems (forests and grasslands) and agricultural ecosystems (rain-fed and irrigated) with various inputs and outputs related to water, salt, and nutrient balances is provided in Figure 4. The primary emphasis of this section is on croplands and pasturelands because the most widespread changes in land use have occurred as a result of expansion

Table 1. Water Withdrawal and Consumption (Global, United States, Total and Irrigated Agriculture), Nitrogen Fertilizer Consumption, and Land Use (Cropland, Pastureland, and Irrigated)[a]

Year	GWW,[b] km^3/yr	GWC,[b] km^3/yr	GIWW,[b] km^3/yr	GIWC,[b] km^3/yr	IWW (United States),[c] km^3/yr	WC (United States),[c] km^3/yr	IWC (United States),[c] km^3/yr	IRF (United States),[c] km^3/yr	ICL (United States),[c] km^3/yr	Cropland (Global),[d] Mha	Pasture (Global),[d] Mha	N FC (Global),[e] Mt	Irrig. Area (Global),[e] Mha	Year	Irrig. Area (United States),[f] Mha
1700										265.6	523.7				
1750										321.3	696.9				
1800										401.6	942.0				
1850										537.1	1310.3				
1900	579	331	513	321	27.63					813.4	1954.6		47.3	1900	3.16
1910					53.89					882.8	2117.9			1910	4.73
1920					77.37					943.7	2282.0			1920	5.83
1930					82.90					1028.9	2483.0			1930	5.95
1940	1088	617	895	586	98.10					1112.2	2688.9		76.0	1940	7.24
1945					110.53									1949	10.44
1950	1382	768	1080	722	122.97					1230.0	2930.0		101.0	1954	11.98
1955					151.98									1959	13.44
1960	1968	1086	1481	1005	151.98	84.28	71.85	48.36	31.78	1361.4	3208.0	11.59[g]	138.8	1964	14.93
1965					165.80	106.39	91.19	41.45	33.16			19.10	149.8	1969	15.82
1970	2526	1341	1743	1186	179.62	120.21	100.86	48.36	30.40	1405.5	3276.3	31.76	167.7	1974	16.67
1975					193.43	132.64	110.53	51.12	31.78			44.42	188.1	1978	20.36
1980	3175	1686	2112	1445	207.25	138.17	114.68	59.41	33.16	1443.7	3356.9	60.78	209.3	1982	19.83
1985					193.43	127.53	101.97	58.86	32.61			70.35	225.2	1987	18.78
1990	3633	1982	2425	1691	189.29	129.88	105.28	46.01	38.00	1477.6	3450.3	77.18	245.2	1992	19.99
1995	3788	2074	2504	1753	185.14	138.17	112.33	37.86	34.96			78.36	263.7	1997	22.30
2000					189.29							80.95	276.3	2000	22.38

[a]GWW, global water withdrawal; GWC, global water consumption; GIWW, global irrigation water withdrawal; GIWC, global irrigation water consumption. United States: IWW, irrigation water withdrawal; WC, total water consumption; IWC, irrigation water consumption; IRF, irrigation return flow; ICL, irrigation conveyance loss. Global cropland and pastureland, nitrogen fertilizer consumption (NFC) in million metric tons (Mt), global irrigated area, and U.S. irrigated area.

[b]*Shiklomanov* [2000].

[c]*Council of Environmental Quality* [1995] and *Solley et al.* [1998].

[d]*Klein Goldewijk* [2001].

[e]FAOSTAT (2006).

[f]USDA Census of Agriculture (http://www.nass.usda.gov).

[g]Data for 1961 rather than 1960.

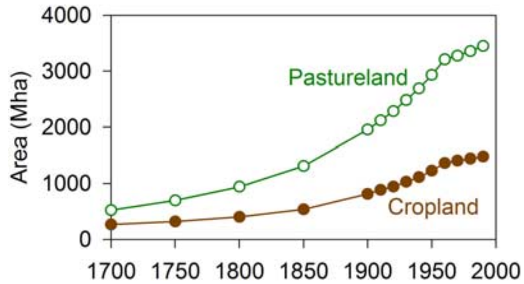

Figure 2. Temporal variation in global cropland and pastureland [*Klein Goldewijk*, 2001].

of these classes. Time series variations in croplands and pasturelands are based on a historical database of the global environment (HYDE) [*Klein Goldewijk*, 2001] that used land use estimates from FAO (arable land and permanent crops; permanent pasture) and hindcasted to 1700 (Figure 2) on the basis of historical population densities. Cropland increased by 460% between 1700 and 1990, whereas pastureland increased by 560% (Figure 2 and Table 1). Time series in cropland and pastureland follow similar trends. Increases in cultivation occurred primarily in developed countries in the 1800s and more in developing countries in the 1900s [*Klein Goldewijk*, 2001]. Impacts of nonirrigated (rain-fed) and irrigated agriculture are discussed separately because of large differences in water withdrawal between these types of agriculture.

[8] Changes in land use impact water quantity in part by controlling the partitioning of water near the land surface, which can be considered using simplified water balance equations:

$$\mathrm{P - ET - Q = 0} \tag{1a}$$

$$\mathrm{P - ET - Q = \Delta S} \tag{1b}$$

$$\mathrm{P - ET - R_0 - R_{gw} = \Delta S} \tag{1c}$$

where P is precipitation, ET is evapotranspiration, Q is streamflow, ΔS is change in water storage, R_0 is runoff, and R_{gw} is groundwater recharge. The simplest form of the water balance equation (1a) assumes that changes in soil water storage are zero, which is appropriate for long times (5–10 years) [*Brown et al.*, 2005]. Equations (1a) and (1b), unlike (1c), do not partition excess water (P − ET) between runoff and recharge and assume that groundwater recharge ultimately discharges to streams (base flow), where the two systems are hydraulically connected.

2.1. Impact of Conversion to Rain-Fed (Nonirrigated) Agriculture on Water Resources

2.1.1. Water Quantity

[9] Nonirrigated or rain-fed agriculture almost doubled during the last century (Figure 1 and Table 1) [*Klein Goldewijk*, 2001] (also FAOSTAT, accessed in 2000). Conversion of natural ecosystems to rain-fed agriculture has had large-scale impacts on water quantity by modifying partitioning of water at the land surface. In many areas natural vegetation was originally forests, which generally have higher ET than other types of vegetation (1.6 times higher than grasslands) [*Zhang et al.*, 2001]. Deforestation reduced ET by 4% globally [*Gordon et al.*, 2005]. Reduced ET in cultivated areas converted from forests provides more water for groundwater recharge and streamflow.

[10] The classic example of impacts of cultivation on water resources is provided by conversion of eucalyptus mallee forests to cropland and pastureland in Australia in the late 1800s and early 1900s. Mallee vegetation originally covered an area of ~30 Mha in S Australia [*Noble*, 1984]. Conversion to rain-fed agriculture decreased ET by about 10% and increased downward water fluxes below the root zone by 1 to 2 orders of magnitude from ≤0.1 mm/yr to 1 to 50 mm/yr in SE Australia [*Allison et al.*, 1990; *Petheram et al.*, 2000; *Gordon et al.*, 2003]. In mallee areas, detailed chemical and isotopic analyses indicate that soil water and groundwater are naturally saline as a result of evapoconcen-

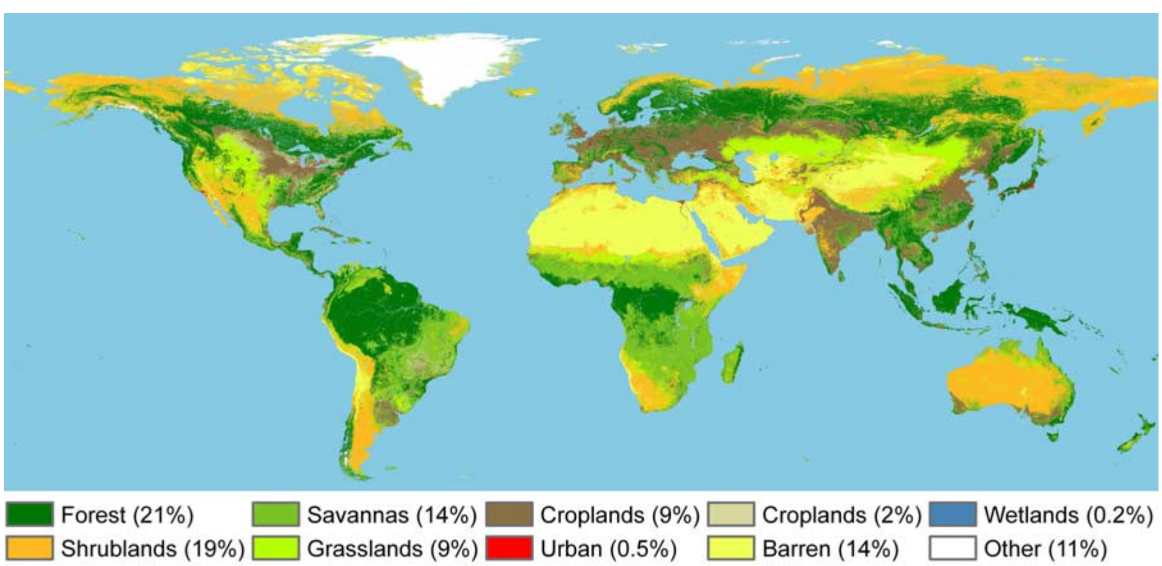

Figure 3. Global distribution of land cover based on MODIS (1 km) satellite data using International Geosphere Biosphere Program land cover classes prepared by Boston University (Earth Observing System (EOS) Data Gateway, http://edcimswww.cr.usgs.gov/pub/imswelcome/). Cropland (2%) refers to cropland/natural vegetation mosaic.

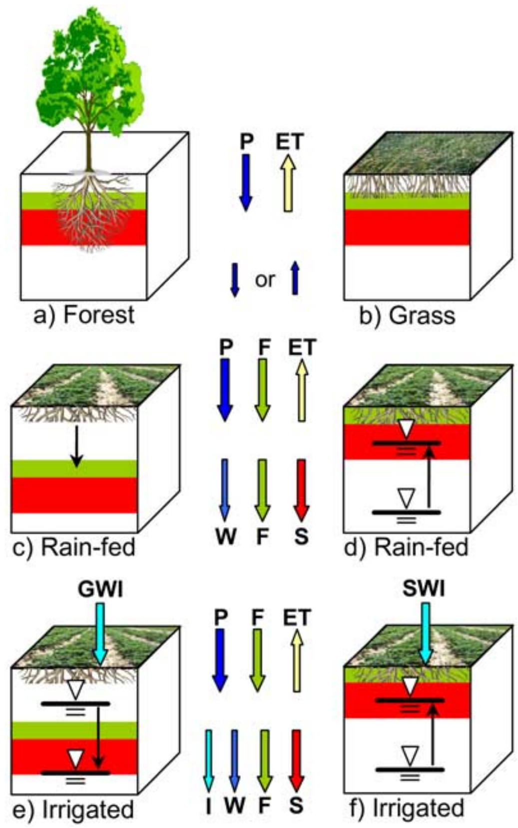

Figure 4. Schematic of different land use settings: natural ecosystems including (a) forest and (b) grassland and agricultural ecosystems including (c, d) rain-fed and (e, f) irrigated agriculture. The top set of arrows refers to inputs and outputs at the land surface and the bottom set of arrows refers to outputs at the base of the profile. There is little or no recharge beneath natural ecosystems in semiarid and arid regions and salt (S), and often nutrient (nitrate; N) reservoirs are found in the unsaturated zone. Conversion of natural ecosystems to rain-fed agriculture results in decreased ET and increased runoff (not shown) or recharge (W, water), raising water tables and mobilizing salts (S) in semiarid regions and nutrients from fertilizers (F) in semiarid or humid regions. Salts are derived from precipitation and fertilizers, and nitrate is derived from precipitation, N-fixing vegetation, and fertilizers. Irrigated agriculture has an additional input of irrigation water that can be derived from groundwater (GWI) or surface water (SWI). Irrigated agriculture generally results in increased recharge from irrigation return flow (I). Salts are mobilized by increased drainage in semiarid regions in groundwater-fed systems. Surface water-fed irrigation systems result in raised water tables causing waterlogging and soil salinization. Salts are derived from precipitation, fertilizers, and irrigation, whereas nitrate is derived from precipitation, N-fixing vegetation, fertilizers, and irrigation water.

tration of salts from precipitation by deep-rooted (≥5–30 m) mallee vegetation during the past 25,000 years or longer, when the climate was semiarid [*Cook et al.*, 2001; *Herczeg et al.*, 2001]. Increased recharge is attributed to replacement of deep-rooted perennial eucalyptus by shallow-rooted annual crops and grasses and associated fallow periods in crop rotations when there is no vegetation. In areas of shallow water tables (≤30 m), increased water fluxes related to land clearance have reached the water table, as shown by increasing groundwater levels up to ~1 m/yr; however, monitoring data are limited [*Allison et al.*, 1990; *Simpson and Herczeg*, 1991; *Bari and Schofield*, 1992; *Ghassemi et al.*, 1995]. In areas of deeper water tables (≥30 m), increased water fluxes have not reached the water table and are restricted to the unsaturated zone. Evidence of increased water fluxes is provided by downward mobilization of salts that previously accumulated beneath native vegetation (Figure 5a) [*Allison and Hughes*, 1983; *Jolly et al.*, 1989]. Long time lags (decades to centuries) between increased water fluxes below the root zone in cleared areas and groundwater recharge result from low absolute water fluxes in cleared areas (≤50 mm/yr) and deep water tables (≥30 m) [*Allison et al.*, 1990]. Therefore the effect of increased recharge beneath cleared areas on the groundwater system has not been fully realized and may take tens to hundreds of years [*Allison et al.*, 1990].

[11] Raised water tables increased hydraulic gradients to streams. Early work in SW W Australia showed that deforestation raised groundwater levels ~1 m/yr and increased streamflow by ~30% of annual precipitation (770 to 1147 mm/yr) (Wights catchment, 100% cleared [*Ruprecht and Schofield*, 1989; *Ghassemi et al.*, 1995]). Cultivation increased surface runoff and base flow to streams that were hydraulically connected to groundwater where annual precipitation is high (1350 mm/yr). Time lags between land use changes to rain-fed agriculture (water

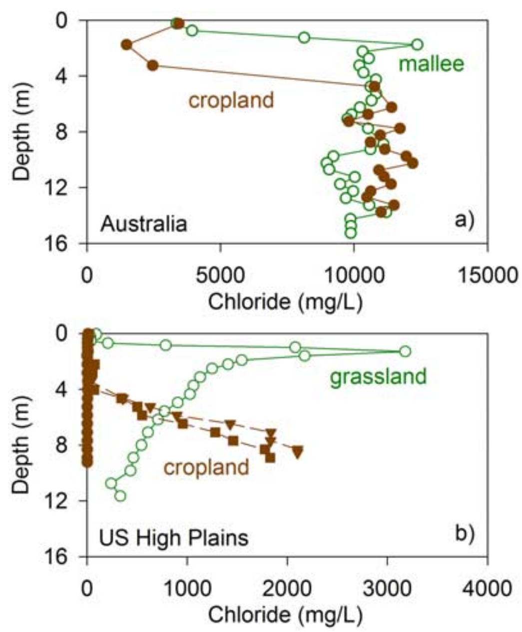

Figure 5. Chloride reservoirs in unsaturated zones in (a) Australia and (b) U.S. High Plains. Conversion of natural ecosystems (mallee and grasslands) to cropland results in mobilization of salt reservoirs. High chloride concentrations at depth in most profiles beneath cropland indicate that increased drainage below the root zone has not reached the water table. An exception is a profile in the U.S. High Plains where the water table is shallow.

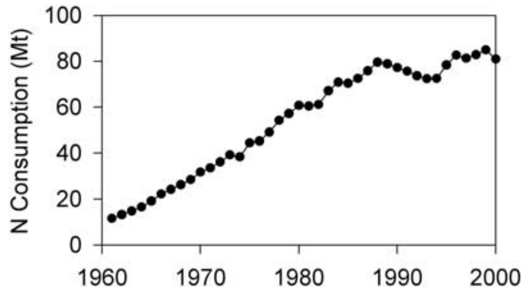

Figure 6. Temporal variability in global nitrogen fertilizer consumption (FAOSTAT, 2006).

fluxes ≤50 mm/yr) or irrigated agriculture (water fluxes ≤175 mm/yr) and base flow to streams are controlled by distance to streams and aquifer transmissivity and specific yield. In large basins time lags can be up to hundreds of years, such as the Murray Basin [*Jolly and Cook*, 2002; *Knight et al.*, 2005].

[12] Similar increases in recharge related to cultivation were found in the High Plains (45 Mha area), which is one of the largest agricultural areas in the United States. Although much of the High Plains is dominated by effects of irrigation, recent studies show that conversion of natural grasslands to rain-fed agriculture in the Southern High Plains changed the direction of water flow from upward (discharging through ET) beneath natural ecosystems to downward, or recharge (median recharge rate 21 mm/yr; 5% of precipitation), beneath cultivated areas [*Scanlon et al.*, 2005]. Evidence of upward flow beneath natural ecosystems is provided by large chloride concentrations in unsaturated zone pore water that have accumulated since the Pleistocene glaciation about 10,000 to 15,000 years ago (Figure 5b) [*Scanlon and Goldsmith*, 1997]. Increased recharge beneath rain-fed agriculture is shown by low chloride concentrations in unsaturated zone profiles (Figure 5b). Increased recharge resulted in an average groundwater level rise of 7 m over a 0.34 Mha area of rain-fed agriculture, over an average of 43 years. In areas of coarse-grained soils underlain by shallow water tables (≤10 to 15 m deep), chloride is flushed throughout the unsaturated zone, and groundwater levels have risen (Figure 5b). However, in areas of finer textured soils and deeper water tables, increased water fluxes related to cultivation have not reached the water table and are shown by downward displacement of salts that previously accumulated under natural ecosystems (Figure 5b).

[13] In Niger (Africa) conversion of natural savannas to rain-fed millet crops increased recharge by about an order of magnitude, from 1 to 5 mm/yr to 10 to 47 mm/yr, as shown by large groundwater level rises from 1963 to 1999 (0.01 to 0.45 m/yr), despite severe droughts during the 1970s and 1980s [*Leduc et al.*, 2001; *Favreau et al.*, 2002]. Increased recharge is attributed to enhanced overland flow, resulting from soil crusting and fallow periods, and focused recharge beneath endorheic ponds and primary gullies [*Leduc et al.*, 2001; *Massuel et al.*, 2006].

[14] In more humid settings, cultivation has enhanced recharge, resulting in increased base flow to many streams in the midwestern United States. The Mississippi River Basin (MRB) drains 41% of the United States and consists of 58% cropland. Conversion of perennial pastures and crops (oats, alfalfa, hay) to annual row crops (mostly soybeans) in headwater areas of the basin and associated soil conservation practices (terracing, contour cropping, and conservation tillage) decreased ET and increased groundwater recharge and base flow discharge to streams [*Zhang and Schilling*, 2006]. For example, in four selected gauging stations in the Upper MRB, where the largest increases (343%; 2.8 to 12.4 million hectares, Mha) in annual row crops (soybeans) occurred between 1940 and 2003 (Iowa, Illinois, Indiana, and Ohio states), annual streamflow increased by 9 to 102% from 1940 to 2003, mostly as a result of increases in base flow (28 to 134%) [*Schilling*, 2005; *Zhang and Schilling*, 2006]. The water demand for annual crops is much less than that for perennial crops because of their shorter growth period (3 to 4 months). Increased recharge associated with fallow periods is supported by studies that show 48 mm higher recharge beneath bare ground than beneath grass for a 122 day monitoring period [*Zhang and Schilling*, 2006]. The period of highest water demand for annual crops (midsummer) is not synchronized with peak precipitation (spring and early summer), which results in annual crops being highly vulnerable to leaching in early spring, when these areas are plowed or fallow. For example, average increases in streamflow and base flow between 1940 and 2003 in Keokuk station were highest in April and May [*Zhang and Schilling*, 2006].

2.1.2. Water Quality

[15] Conversion of natural to agricultural ecosystems impacts water quality in semiarid and arid regions primarily by mobilizing natural salts that accumulated in the subsurface in many semiarid regions; by dissolving minerals such as gypsum; by raising water tables which results in salinization through ET; or by adding nutrients such as nitrate to the system. Impacts in more humid settings are generally related to transport of applied fertilizers. Global nitrogen fertilizer consumption increased by ∼600% from 1961 to 2000 (Figure 6 and Table 1) and has large-scale impacts on nutrient cycles. The 50% decrease in global wetlands [*Revenga et al.*, 2000] strongly decreases nitrate sinks because denitrification is widespread in these systems [*Woltenmade*, 2000].

2.1.2.1. Salinization

[16] Although cultivation has greatly increased water quantities in Australia, it has negatively impacted water quality. Increased recharge flushes salts that previously accumulated over millennia in the vadose zone as a result of evapotranspirative concentration of atmospherically derived salts (Figure 4c). Profiles of chloride in the vadose zone show various stages of flushing through the unsaturated zone (Figure 5a). Increased recharge also raised water tables to within a few meters of the land surface in many areas, resulting in waterlogging and soil salinization, known as "dryland salinity" (Figure 4d) [*Allison et al.*, 1990]. An assessment of dryland areas indicates that 5.7 Mha of cropland and pastureland are affected by or are at risk of developing dryland salinity from shallow water tables [*National Land and Water Resources Audit* (*NLWRA*), 2001]. This area could increase to 17 Mha by 2050 unless effective strategies are implemented [*NLWRA*, 2001].

[17] Historical catchment salt output/input (O/I) ratios and stream salinity trends can be used to assess the degree to which land use has impacted water quality in catchments with high natural reservoirs of salt in soils (Figure 5a) and in

groundwater, as found in much of S Australia [*Jolly et al.*, 2001]. Output is calculated from streamflow and chloride concentrations in streamflow at a gauging station, and input from precipitation and chloride concentrations in precipitation. Under natural conditions in flow through basins mean catchment salt O/I ratios are approximately unity and there are no long-term rising trends in stream salinity. Land use change mobilizes salt stored in the soil profile into groundwater through increased recharge. Elevated recharge increases hydraulic gradients, resulting in increased discharge of saline groundwater and hence salt loads to streams. Studies of incipient salinization in a 10^4 km^2 area in the Murray Basin indicate that groundwater salinities could increase by factors of 2 to 6 (total dissolved solids, TDS, 850 mg/L to 1550–5000 mg/L) within the top 10 m of the water table in the next 100 years in some areas, given unsaturated zone salt inventories [*Leaney et al.*, 2003]. Salt O/I ratios were as high as 10 in catchments of the Murray-Darling Basin in SE Australia [*Jolly et al.*, 2001]. Salt O/I ratios were highest in medium-rainfall areas (500–800 mm/yr) and minimal in areas of high rainfall (>800 mm/yr). Chloride concentrations increase from ~2 mg/L in the headwaters (1978–1986; kilometer point (KP) 2322) to 170 mg/L toward the downstream end (KP 87) of the 2500 km long channel of the Murray River, which is much greater than would be predicted from assuming concentration of chloride from precipitation by evapotranspiration (factor of 25) and input from the Darling tributary [*Simpson and Herczeg*, 1991]. Monthly chloride concentrations increased by 80% from 1938 to 1981 at KP 322. Linear increases in salt discharge with river flow discharge indicate that salts are readily available for mobilization under varying flow regimes. The most statistically significant rising stream salinity trends were also in the medium-rainfall areas. High groundwater salinities (Cl, 5000–10,000 mg/L) require influx of only 1–2% of river discharge volume to account for the measured increase in chloride in the river; these low groundwater discharge levels cannot be quantified because they are within uncertainties in river gauging measurements (±5 to 10%) [*Simpson and Herczeg*, 1991].

[18] Stream salinization also increased markedly in SW W Australia, where catchments changed from net salt accumulating prior to clearing to salt exporting after clearing, with salt O/I ratios up to 20 [*Ruprecht and Schofield*, 1991]. The degree of stream salinization associated with forest clearing is related to the proportion of the catchment that is cleared and to salt storage in the catchments, which varies inversely with precipitation [*Schofield and Ruprecht*, 1989]. Clearing of native vegetation had minimal impacts on stream salinity in catchments with precipitation ≥1,100 mm/yr (increase from 144 mg/L TDS (0% cleared) to 233 mg/L TDS (100% cleared)). In contrast, clearing in low-precipitation catchments (500–700 mm/yr) increased TDS from 70 mg/L (freshwater) (0% cleared) to 3,488 mg/L (brackish water) (100% cleared) [*Schofield and Ruprecht*, 1989].

[19] Increased recharge related to cultivation in the Southern High Plains, United States, degraded groundwater quality [*Scanlon et al.*, 2005]. Increases in groundwater salinity (43%; 150 to 214 mg/L) have been attributed to flushing of salts that had accumulated in soils during the Holocene, ~10,000 to 15,000 years ago [*Phillips*, 1994; *Scanlon et al.*, 2005].

2.1.2.2. Nutrient Loading

[20] Increased recharge related to cultivation in semiarid regions can degrade water quality by flushing nutrients, in addition to salts, into underlying aquifers. In many semiarid regions, unsaturated zone nitrate reservoirs, similar to chloride reservoirs, have built up as a result of evapotranspirative concentration of nitrate from precipitation and dry fallout over millennia. High nitrate levels (≤680 mg/L) in soils beneath native mallee vegetation have been mobilized by cultivation, as shown by low nitrate levels (≤54 mg/L) beneath cropland and pastureland (Borrika site, Murray Darling Basin [*Cook*, 1992]). In contrast, nitrate levels were generally low beneath native vegetation in the Southern High Plains [*Scanlon et al.*, 2005; *McMahon et al.*, 2006]. Increased recharge related to cultivation in Niger has degraded groundwater quality by leaching nitrate from the unsaturated zone, as shown by order of magnitude higher nitrate levels (≤45 mg/L nitrate-N) in groundwater near recharge zones relative to areas distant from recharge zones [*Favreau et al.*, 2004]. Monitoring groundwater nitrate levels shows sharp increases in nitrate (e.g., 10 to 45 mg/L nitrate-N in 2000) related to recharge pulses. The primary source is natural nitrate reservoirs in the unsaturated zone beneath native bush (0.1 to 0.3% by mass) because fertilizer application is negligible and septic tanks are limited. Nitrogen and oxygen isotopes in groundwater and soil water support the natural source of nitrate (δ^{15}N 6.6–8.8‰ and δ^{18}O of oxygen in nitrate 7–14‰) [*Favreau et al.*, 2004].

[21] Nutrient loading in more humid settings has generally been related to fertilizer applications. Land use changes in the Mississippi River Basin (MRB) impacted nutrient loading to groundwater, streams, and coastal waters. An ~500% increase in agricultural nitrogen fertilizer application (<1 to >6 metric tons) from 1950–1970 to 1980–1996 is the primary contributor to a 200% increase in nitrate export from the MRB to the Gulf of Mexico, resulting in seasonal bottom water hypoxia [*Goolsby et al.*, 1999; *Goolsby and Battaglin*, 2001]. States within the U.S. corn belt are major contributors to nutrient loading [*Schilling and Zhang*, 2004]. Nitrate export from the Raccoon River catchment (Iowa) was highest out of 42 Mississippi River subcatchments evaluated in a Gulf of Mexico hypoxia study [*Schilling and Zhang*, 2004]. Base flow contributed ~66% (17.3 kg/ha) of the long-term nitrate export from the Raccoon River and was greatest (80%) in spring and late fall, when crop uptake was minimal, and was lowest in summer during the crop growing season.

[22] Nutrient loading of streams in the NE United States is also highly correlated with land use change and contributes to eutrophication in Chesapeake Bay [*Bratton et al.*, 2004]. Variance in flow-weighted mean nitrate concentrations is explained by percentage cropland (51% of variance explained), base flow index (69%), and crop and base flow combined (85%), on the basis of monitoring water and nutrients in 27 catchments for 1 year [*Jordan et al.*, 1997]. Increased stream nitrate concentration with base flow percent indicates that nitrate contamination results from nitrate leaching and recharge to groundwater, followed by base flow discharge to streams [*Jordan et al.*, 1997]. Catchments with lower base flow indices retain nitrate near the land surface where it can be used by plants [*Jordan et al.*, 1997].

[23] Agriculture and fertilizer application generally control nutrient contamination of groundwater and surface

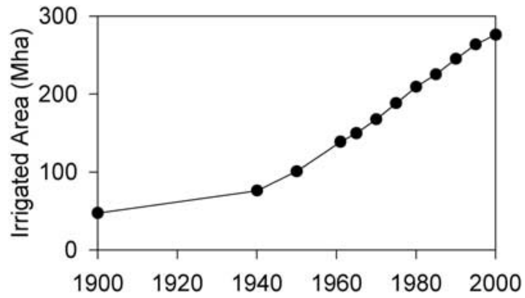

Figure 7. Temporal variation in global irrigated land area (FAOSTAT, 2006).

water in Europe. The percentage of rivers exceeding a mean annual total nitrogen concentration of 2.5 mg/L nitrate-N in Denmark is among the highest of 28 EU countries [*Kronvang et al.*, 1995]. Agriculture was found to be responsible for 65 to 83% of yearly riverine total nitrogen transport in Denmark on the basis of results from monitoring 130 stations at river mouths during a 4 year period (1989–1992). From 1989 to 1992, annual median nitrogen loss to 115 rivers draining agricultural catchments (23.4 kg N/ha) was 14 times greater than that (1.7 kg N/ha) to 7 rivers draining undisturbed catchments in Denmark [*Kronvang et al.*, 1995]. Median nitrate concentrations were much higher (18 mg/L nitrate-N) beneath cropland than beneath forests (1.5 mg/L nitrate-N), according to a national Danish survey of nitrate concentrations in soil water at 1,000 sites [*Callesen et al.*, 1999]. In the Netherlands, agricultural land (2 Mha) represents 60% of the land cover and accounts for 70% of the nitrogen losses to the environment (soil and groundwater) [*van Eerdt and Fong*, 1998]. Increased groundwater nitrate contamination in sandy soils in the east and south (37% of agricultural area) is attributed to intensification of agriculture and deeper groundwater levels because these regions are above mean sea level (MSL) and are freely draining [*Oenema et al.*, 1998]. Low groundwater nitrate concentrations in other regions result from denitrification associated with peaty soils and shallower water tables in finer textured soils because these regions are below MSL and are artificially drained. Water tables in agricultural areas are generally shallow in autumn and winter (50% <0.4 mm, wet soils; 35% 0.4–1.2 m, medium wet soils; 15% >1.2 m, dry soils) and denitrification and drainage to surface waters is widespread in wet soils.

[24] Nitrate contamination in groundwater and surface water in the UK is also related to land use. Nitrate-N concentrations in the Tweed catchment (0.4 Mha area) in Scotland increase from 0 to 5 mg/L in the upland moorland and pasture areas to 5 to 33 mg/L in the lowland cropland areas [*Jarvie et al.*, 2002]. Increased nitrate-N levels from 3.5 to 6.4 mg/L in the Kennet catchment (0.12 Mha area, tributary of the Thames River) is attributed to an increase in cropland (25–60%) and corresponding decrease in grassland (64–37%) and forest (11–3%) from 1931 to 1991 on the basis of modeling analysis [*Whitehead et al.*, 2002].

2.2. Impacts of Conversion to Irrigated Agriculture on Water Resources

[25] Irrigated agriculture has large-scale impacts on water resources because it consumes 90% (1900–1995) of global freshwater [*Shiklomanov*, 2000]. Irrigated areas have increased by 480%, from 47.3 Mha in 1900 to 276.3 Mha in 2000, and currently represent 18% of global croplands (Figure 7 and Table 1). The percentage of global irrigated area is greatest in Asia (69%), followed by America (17%), Europe (9%), Africa (4%), and Oceania (1%) [*Siebert et al.*, 2005]. Countries with the largest fraction of irrigated agriculture include India (57.3 Mha), China (53.8 Mha), and the United States (27.9 Mha), representing 50% of global irrigated areas [*Siebert et al.*, 2005].

[26] Irrigation causes redistribution of water between surface water and groundwater. Irrigation based on surface water transfers water from streams to groundwater through leakage from conveyance structures and enhanced recharge of irrigation return flow. Groundwater-fed irrigation may ultimately result in transfer of surface water to groundwater also. The following water balance equation describes the various stages of the process:

$$\mathrm{R} = \mathrm{D} \tag{2a}$$

$$(\mathrm{R} + \Delta\mathrm{R}) - (\mathrm{D} + \Delta\mathrm{D}) - \mathrm{Q} = \mathrm{dV/dt} \tag{2b}$$

$$(\mathrm{R} + \Delta\mathrm{R}) = (\mathrm{D} + \Delta\mathrm{D}) + \mathrm{Q} \tag{2c}$$

Prior to groundwater development for irrigation, the system is at steady state, and groundwater recharge (R) equals groundwater discharge (D) [*Sophocleous*, 2000]. Groundwater pumping (Q) due to irrigation initially removes water from storage in the aquifer (V), lowering water tables. Reduced water tables may result in increased recharge (ΔR) or decreased discharge (ΔD) through groundwater ET and/or base flow to streams. Ultimately, a new steady state is established, and groundwater recharge (diffuse, areally distributed recharge and focused recharge from streams) balances groundwater discharge (pumpage, base flow, ET). At this time water for irrigation pumpage can be derived from increased recharge (irrigation return flow or focused recharge from streams) or decreased discharge (reduced ET and/or reduced base flow to streams).

2.2.1. Water Quantity

[27] There are numerous examples on the impacts of irrigation and associated dams and diversions on water quantity, resulting in little or no flow of many major rivers reaching the sea during parts of the year, such as the Nile (NE Africa), the Amu Darya and Syr Darya (central Asia), the Ganges and Indus (southern Asia), and the Colorado (United States) [*Postel*, 1999]. In the case of the Amu Darya and Syr Darya, large diversions of water from these rivers to irrigate an ~8 Mha area reduced the volume of the Aral Sea by ~70% [*Ghassemi et al.*, 1995; *Postel*, 1999]. The following section focuses on examples of impacts of irrigation on water quantity in the most heavily irrigated countries (China, India, and United States).

[28] Irrigation in China (960 Mha area) has been practiced for the past 4000 years [*Framji*, 1981]. China has 154 Mha of cultivated land (2000) (FAO AQUASTAT, http://www.fao.org/AG/AGL/aglw/aquastat/main/index.stm, accessed in 2006, hereinafter referred to as AQUASTAT, 2006). The irrigated area increased from 16 Mha in 1949 (~16% of cultivated land) to 53.8 Mha (2000; 35% of

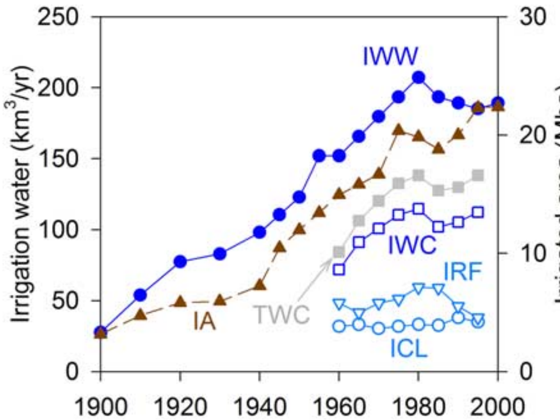

Figure 8. Temporal variability in irrigation water withdrawal (IWW), irrigated area (IA), total freshwater consumption (TC), irrigation freshwater consumption (IC), irrigation return flow (IRF), and conveyance loss (CL) in the United States. Sources: IWW, *Council of Environmental Quality* [1995] and USDA Census of Agriculture (http://www.nass.usda.gov); TC, IC, IRF, and CL, *Solley et al.* [1998].

cultivated land), and accounts for 68% of water withdrawal [*Ghassemi et al.*, 1995; AQUASTAT, 2006]. Much of the early irrigation was derived from surface water, including rivers and reservoirs. Diversion of surface water for irrigation has resulted in river desiccation. For example, doubling of irrigated agriculture in the Yellow River Basin accounts for ~91% of surface water abstractions and resulted in drying up of the lower reach of the Yellow River since 1972, termed "duanliu" in Chinese [*Chen et al.*, 2003]. The duration and length of dry channels in the vicinity of the lower reaches of the Yellow River (Lijin station) have increased from an average of 12 d/yr and 163 km/yr in the 1970s to 100 d/yr and 352 km/yr in the 1990s, up to a maximum of 226 d and 700 km in 1997 [*Liu and Xia*, 2004]. Irrigation based on surface water has resulted in groundwater level rises related to seepage from rivers, water losses along conveyance structures, and deep percolation on irrigation fields (Figure 4f). Examples of groundwater level rises include the Jinghuiqu canal irrigation district (Shaanxi Province) where diversion of water from the Jinhe River raised groundwater levels from 15 to 30 m in the 1930s to <1 to 2 m in the 1950s [*Sheng and Xiuling*, 2001]. Increases in groundwater levels from >7 m in the 1950s to 1 m in 1989 were recorded in parts of Xinjiang Province after diversion of river water for irrigation [*Sheng and Xiuling*, 2001]. These large groundwater level rises have resulted in extensive waterlogging in these irrigation districts (Figure 4e).

[29] Irrigation has rapidly expanded in the North China Plain (NCP, 32 Mha area, population 213 million), which includes the Yellow (Huang), Huai, and Hai river basins [*Liu et al.*, 2001]. The NCP, one of the largest agricultural areas in China, includes 18 Mha of cropland (56% of the NCP area), which produces 50% of the wheat and 33% of the maize in China [*Liu et al.*, 2001; *Kendy et al.*, 2003]. The irrigated area has increased from 1.4 Mha (1950s) to 4.9 Mha (1990s), and consumes up to 90% of the water supply [*Liu and Xia*, 2004]. Precipitation ranges from 800 mm/yr in the south to 500 mm/yr in the north [*Liu et al.*, 2001]. The monsoon climate results in about 70% of precipitation occurring in the summer (June–September), which generally exceeds the water requirements for summer maize (360 mm/yr) [*Liu et al.*, 2001; *Zhang et al.*, 2004]. However, water requirements for winter wheat (450 mm; October–May growth period) greatly exceed winter precipitation (100–150 mm), requiring intensive irrigation [*Yang et al.*, 2000; *Zhang et al.*, 2004]. Large increases in yield of winter wheat from 1000 kg/ha in the 1950s to 4000 kg/ha in the 1990s are attributed to irrigation, fertilization, and improved crop strains [*Foster et al.*, 2004].

[30] Irrigation was primarily supplied by surface water originally with large-scale diversions from rivers and reservoirs in the late 1950s and early 1960s. Irrigation in Liuyuankou irrigation system (Henan Province) raised groundwater levels within a meter of land surface [*Khan et al.*, 2006]. Groundwater-fed irrigation has been expanding since the mid-1960s and currently supplies an average of 64% of irrigation water in the NCP and up to 80% in some regions (Hebei Province) [*Zhang et al.*, 2004]. Most irrigation is provided by a shallow, unconfined aquifer [*Foster et al.*, 2004]. Water table depths in this aquifer were predominantly <4 m in the Hebei Plain in 1964 (730 ha); however, by 1993, 18% was <4 m, 45% between 4 and 10 m, 37% > 10 [*Liu et al.*, 2001]. Groundwater level declines were up to 0.7 m/yr (Luancheng Station, water table 3 m in 1950s to 30 m in 1990s) [*Zhang et al.*, 2004]. The large increases in water use and declines in groundwater levels have resulted in stream channels drying up.

[31] Irrigation is very important for agricultural food production in India. India's large population (~1 billion) and temporal variability in precipitation related to monsoons (e.g., ~90% of precipitation from June to September in western and central India is related to the southwest monsoon) result in heavy demands on water resources for food production [*Kumar et al.*, 2005]. Irrigated cropland has more than doubled, from 25 Mha (15% of cropland) in 1960 to 57 Mha (34% of cropland) in 2000 (AQUASTAT, 2006). Irrigation is supplied by surface water through canals in government-funded programs. Overapplication of surface water relative to crop water demands has resulted in depletion of surface water resources and rising water tables in several states. For example, large rises in groundwater levels in Punjab during 1941 to 1963 resulted in extensive waterlogging. Use of groundwater for irrigation has expanded greatly, from 30% of the irrigated area in 1960 (~7 Mha) to ≥50% in 1995 (~27 Mha (AQUASTAT, 2006)). The number of mechanized wells and tube wells increased from less than 1 million in 1960 to about 19 million in 2000 [*Deb Roy and Shah*, 2003]. However, the booming groundwater-based agrarian economy in many parts of India is under serious threat of resource depletion and degradation. Throughout India, declining groundwater levels have resulted in increased pumping costs and have put many wells out of commission. In W India, where depletion is greatest, over half of the wells are out of commission, and the proportion steadily rises as water tables continue to decline [*Deb Roy and Shah*, 2003].

[32] Irrigation is the largest consumer of freshwater in the United States, accounting for an average of 83% of total freshwater consumption between 1960 and 1995 (Figure 8 and Table 1) [*Solley et al.*, 1998]. Irrigation water withdrawal

increased from 28 km^3/yr in 1900 to a maximum of 207 km^3/yr in 1980 (640% increase) and decreased slightly to 189 km^3/yr in 2000. Irrigation water consumption follows trends similar to those of withdrawals. Irrigation application (withdrawal–conveyance loss) decreased in the last few decades, although irrigated area increased. Irrigation return flows (withdrawal–consumption–conveyance losses) decreased in the last couple of decades as the efficiency of irrigation systems improved. Irrigation by groundwater increased from 23% in 1950 to 42% in 2000, with a corresponding decrease in the fraction based on surface water [*Hutson et al.*, 2004]. Irrigation has expanded greatly in more humid regions in the eastern United States in the last few decades as irrigation technology has advanced and as benefits of supplemental irrigation in more humid regions are recognized [*Gollehon and Quinby*, 2006]. Surface water based irrigation has reduced streamflow and changed the seasonal distribution of streamflow [*Kendy and Bredehoeft*, 2006]. There are many examples of groundwater-fed irrigation impacting base flow, changing streams from perennial to ephemeral [*Wahl and Tortorelli*, 1997; *Sophocleous*, 2000]. Irrigation pumpage in western Oklahoma decreased mean annual streamflow, from 37 to 91% in different stations relative to predevelopment streamflow [*Wahl and Tortorelli*, 1997]. Additional examples of the impact of groundwater pumpage on streamflow were provided by *Glennon* [2002]. One of the largest irrigated areas in the United States is the High Plains (45 Mha area), which represents 27% of the irrigated land, 30% of groundwater use for irrigation in the United States, and 94% of groundwater use in the High Plains [*Qi et al.*, 2002]. Although irrigated agriculture accounts for only 11% of land area in the High Plains, irrigation has resulted in large groundwater level declines ($\geq$30 m; average 43 m) over 10,500 ha of the Southern High Plains, 1945–2003 [*Scanlon et al.*, 2005].

[33] Irrigation in California is derived from both surface water (63%) and groundwater (37%) [1995] [*Solley et al.*, 1998]. Groundwater-fed irrigation in California accounts for 24% of groundwater consumptive use in the United States [*Solley et al.*, 1998]. The most heavily irrigated area is the California Central Valley. Interbasin transfer projects, including the California Aqueduct (1953, 715 km long, concrete lined 12 m wide, 9 m deep) and Delta-Mendota Canal (1967, 193 km long), transport water from humid northern California to the Central Valley [*Schoups et al.*, 2005]. Overapplication of irrigation resulted in water tables rising to near the land surface in parts of the San Joaquin Valley [*Schoups et al.*, 2005]. Groundwater-fed irrigation has resulted in large water table declines that exceed 120 m in parts of the Central Valley and have caused maximum levels of land subsidence of 3–9 m that correlate with groundwater pumpage [*Williamson et al.*, 1989]. Comparison of numerical simulations representing predevelopment and postdevelopment (1961–1977) in the Central Valley indicates that 80% of the water for pumpage is derived from increased recharge of irrigation return flow, 14% from reduced discharge, and 6% from groundwater storage [*Johnston*, 1997]. Increased recharge occurs in many irrigated areas as excess irrigation water drains below the crop root zone.

2.2.2. Water Quality

[34] Irrigation area losses resulting from salinization are estimated to be 1.5 Mha/yr, totaling ~45 Mha globally, which represents ~16% of the global irrigated area [*Ghassemi et al.*, 1995; *Wood et al.*, 2000].

[35] Information on the impact of irrigation on water quality in China is limited. Salinity of the Yellow River has been increasing in the last 4 decades as a result of saline irrigation return flows [*Chen et al.*, 2003]. For example, sodium in the Yellow River increased by 100% (40–79 mg/L) and TDS by 29% (431–557 mg/L) after flowing through two large irrigation districts in Ningxia and Inner Mongolia Provinces [*Chen et al.*, 2003]. A total of 70% (42 out of 63) of stream stations in the Yellow River Basin show an upward trend in TDS between 1960 and 2000, including all mainstream stations downstream of Qingtongxia irrigation district (Nignxia Province) [*Chen et al.*, 2003]. Waterlogging and salinization are widespread in China and are associated with canal irrigation and seepage from rivers. Examples include the Jinghuiqu canal irrigation district, Shaanxi Province (30% salinized in 1954), parts of Xinjiang Province (14,000 ha, 1963), and the Liuyuankou irrigation system (Henan Province) where water tables have been raised within a meter of land surface and salinization occurs as a result of capillary flow [*Sheng and Xiuling*, 2001; *Khan et al.*, 2006]. Salinization is a critical issue in the North China Plain, with 29% seriously affected and 14% moderately affected [*Ghassemi et al.*, 1995]. Parts of the Hebei Province have water tables within 3 to 5 m of the ground surface and TDS ranges from 3000 to 50,000 mg/L [*Ghassemi et al.*, 1995]. The large expansion of groundwater wells from the mid 1960s has lowered water tables in many areas, reducing salinization. Parts of the NCP have shallow saline groundwater overlying fresh groundwater; however, overdevelopment of the freshwater zone has lowered the saline/freshwater interface by an average of 10 m to a maximum of 30 m [*Liu et al.*, 2001]. Seawater intrusion has resulted in salinization of groundwater. For example, in Laizhou the rate of seawater intrusion has increased from 50 m/yr (1976–1979) to 405 m/yr (1988) [*Liu et al.*, 2001]. Nitrate contamination ($\geq$10 mg/L nitrate-N) was found in 50% of 69 wells sampled throughout the NCP [*Zhang et al.*, 1996]. A more recent study evaluated the spatial distribution of nitrate and found high nitrate to 50 m depth in the recharge zone in the west (26% of samples $\geq$10 mg/L), very little in the intermediate flow zone, and low levels of contamination (10% $\geq$10 mg/L) in the discharge zone in the east [*Chen et al.*, 2005]. Increased nitrate contamination is attributed to increased application of nitrogen fertilizers from 130 kg/ha in the 1980s to 211 kg/ha in 2005.

[36] Irrigation in India has exacerbated many water quality problems by causing secondary salinization in canal-irrigated areas, depleting uncontaminated groundwater sources, and causing mixing of uncontaminated and contaminated groundwater. Secondary salinization of groundwater and soils arises from waterlogging in canal-irrigated areas, particularly in Punjab (25% of irrigated area) and Sindh (60%), resulting in abandonment of many irrigated areas [*Shah et al.*, 2001]. In many coastal aquifers subject to intensive groundwater development, seawater intrusion is a devastating problem, as in the Saurashtra region of Gujarat state and in the Minjur aquifer in Tamil Nadu [*Shah et al.*, 2001]. Fluoride contamination in groundwater is a major problem in two thirds of India, resulting in dental and skeletal fluorosis [*Gupta et al.*, 2005]. For example, in the north Gujarat region,

increasing fluoride concentrations were noted during the past 3 to 4 decades, which may be caused by expansion of irrigated areas by a factor of three and by water table declines of up to 100 to 150 m, resulting in pumpage of higher fluoride concentrations from deeper groundwater [*Gupta et al.*, 2005]. Widespread groundwater arsenic contamination in West Bengal and Bangladesh, with concentrations from <5 to 4,100 μg/L, has resulted in about 50 million people being at potential risk from arsenic exposure [*Kinniburgh and Smedley*, 2001; *Smedley and Kinniburgh*, 2002]. Introduction of high-yielding, dry season rice (Boro) accelerated the demand for irrigation [*Harvey et al.*, 2005]. Contaminated aquifers are generally shallow (<10 to 150 m deep) [*Smedley and Kinniburgh*, 2002]. Various studies have suggested deeper groundwater as an alternate, safe drinking water source [*Kinniburgh and Smedley*, 2001; *Zheng et al.*, 2005]. However, pumping of deeper aquifers for irrigation and other uses may have caused downward migration of arsenic contamination from shallower aquifers, as suggested by relationships between land use and vertical distribution of arsenic [*Zheng et al.*, 2005] and groundwater particle tracking modeling [*Mukherjee*, 2006].

[37] In the central High Plains (United States), water quality degradation in 20 out of 30 sampled wells is attributed to irrigation, based on higher TDS (median 675 mg/L relative to 374 mg/L), higher nitrate-N levels (median 7.1 mg/L relative to 2.5 mg/L), and pesticide detects [*Bruce et al.*, 2003]. The dominant source of nitrate is fertilizer, according to nitrogen isotope data, and nitrate-N concentrations exceeded the MCL of 10 mg/L in 30% of the wells [*Bruce et al.*, 2003]. Evidence of water quality degradation related to irrigation in the southern High Plains includes a 34% increase in TDS and 221% increase in nitrate-N% over an average of 34 years beneath irrigated areas [*Scanlon et al.*, 2005]. Unsaturated zone profiles of chloride, nitrate, and tritium indicate that irrigation return flow has not reached the water table in some areas of the Southern High Plains because the rate of water table decline from irrigation pumpage is greater than that of soil water drainage from irrigation return flow [*McMahon et al.*, 2006].

[38] Cultivation and irrigation have resulted in nitrate-N contamination ($\geq$10 mg/L nitrate-N) in about 25% of domestic wells in the eastern San Joaquin Valley in California, which is attributed in part to a 500% increase in nitrogen fertilizer application from 1950 to 1980 [*Dubrovsky et al.*, 1998]. Historical development of water quality problems on the west side of the San Joaquin Valley (south part of Central Valley) is related to variations in irrigation water source from predominantly groundwater in the 1920s to mostly surface water supplied by interbasin transfers from the north [*Schoups et al.*, 2005]. Low-salinity water supplied by the California Aqueduct initially resulted in leaching of previously accumulated salts. However, reduction in groundwater pumpage and application of water from the north raised water tables to near the land surface, causing salinization (Figure 4e). Additional salt inputs are provided by dissolution of gypsum. Subsurface drainage of irrigated areas into San Luis Drain and ultimately into evaporation ponds in Kesterson National Wildlife Refuge in the San Joaquin Valley resulted in contamination with selenium $\leq$1400 ppb and other trace elements including boron, chromium, molybdenum, and vanadium. Selenium contamination is attributed to mobilization of salts from seleniferous soils and transport by irrigation drainage to ponds and wetlands on the refuge resulting in selenium toxicosis [*Seiler et al.*, 1999]. Decadal timescales are required to develop equilibrium salt balances and remediation of these salt problems will also require decades. Salinization issues are critical to sustainability of irrigated agriculture in this region [*Schoups et al.*, 2005]. Evaluation of 26 sites throughout the western United States identified key factors that result in susceptibility to selenium contamination: saline marine sedimentary rocks of Cretaceous or Tertiary age containing selenium, oxidized soils resulting in selenate (more mobile), evaporation much greater than precipitation resulting in salinization of soils, irrigation for leaching salts, and drainage into terminal lakes or ponds [*Seiler et al.*, 1999]. The study identified 1.1 Mha of land susceptible to selenium contamination.

3. Sustainable Water Resources Within the Context of Land Use Changes

[39] Developing sustainable water resources requires a thorough understanding of impacts of land use changes on water quantity and quality. Various approaches are being considered to alleviate negative environmental impacts of past land use changes and to minimize such problems in the future. As discussed in the Comprehensive Assessment of Water Management in Agriculture, we need to think differently about water (http://www.iwmi.cgiar.org). Sectoral divisions between rain-fed and irrigated agriculture need to be abandoned in favor of supplemental irrigation in rain-fed systems (such as rainwater harvesting) and deficit irrigation instead of overirrigation in traditional irrigation areas. Divisions between surface water-fed and groundwater-fed irrigation also need to be reevaluated because conjunctive use of both can often reduce negative environmental impacts of each in many areas. Transporting water from more humid to more arid regions is also being considered in large interbasin transfers in China and India to alleviate water scarcity issues in more arid settings. Water resource problems could also be reduced by growing more food in regions with high water productivity and exporting virtual water in food to lower water productivity regions leading to a saving in global water resources through virtual water trade [*Yang et al.*, 2006]. The latter topic is outside the scope of this review. The following sections describe some approaches that can be applied to remediate negative environmental impacts of agriculture and to avoid such problems in the future.

3.1. Remediation of Water Resource Problems Associated With Rain-Fed Agriculture

3.1.1. Afforestation and Reforestation

[40] To remediate existing problems associated with rain-fed agriculture, mostly related to water quality, various changes are being proposed for rain-fed agricultural areas. In Australia, afforestation or reforestation is one of the options being considered to reverse dryland salinity problems by lowering water tables. Evaluation of 80 sites in W Australia indicates that reforestation of about 70 to 80% of a catchment would be required to achieve significant reductions in groundwater levels and salinity control [*George et al.*, 1999]. Studies by *Salama et al.* [1999] indicate that

reforestation of 45% of a catchment in southeastern Australia would be required to reduce salt loads by 30%. Similar results are documented in field studies where groundwater levels declined by 0.6 to 3 m at six sites in proportion to the area reforested (8–70%) [*Bell et al.*, 1990] and by 5.5 m at a site where 70% of the area was reforested [*Bari and Schofield*, 1992]. However, modeling analysis at a site in E Australia indicates that reforestation would substantially reduce river flows: a 10% increase in forest cover at this site would result in a 17% decrease in river flow into a downstream dam [*Herron et al.*, 2002]. Conversion of grassland to pine (*Pinus radiata*) plantation (100% conversion) in 1986 to 1987 in a small upland catchment in Australia (Pine Creek, 320 ha area, 785 mm precipitation) decreased streamflow substantially and salt load by a factor of six and increased the number of zero flow days [*Lane et al.*, 2005]. Salt O/I ratios decreased from 6.6 (1989–1991) to 1.0 (1992–2003). Modeling analysis shows that surface runoff is halved within 5 years; however, groundwater discharge takes about 15 years to halve, which accounts for the initial increases in stream salinity in the first 5 years, followed by gradual decreases [*Dawes et al.*, 2004]. Salt concentrations flowing from streams can be diluted by precipitation. For example, catchments with similar salt exports would have lower stream salinities in catchments with higher precipitation and vice versa. Therefore, from a water quality perspective, afforestation should not be conducted in high-precipitation catchments because it would reduce the dilution effect of high precipitation on stream salinity concentrations. This is because afforestation would reduce surface runoff much sooner than groundwater discharge, resulting in a reduction in dilution flow, hence increased stream salinity. Large-scale reforestation is infeasible because previously cleared regions account for most of the agricultural production in the country; e.g., the Murray-Darling River Basin (100 Mha) represents 45% of Australia's agricultural production [*Pierce et al.*, 1993].

[41] Programs have been proposed to double the area of forests in Ireland and the UK by 2035 to 2045 because forests are thought to reduce nitrate pollution associated with agriculture [*Allen and Chapman*, 2001; *Calder et al.*, 2003; *Calder*, 2005]. Groundwater in the Triassic Sandstone in the British Midlands is close to or exceeds the MCL of 50 mg/L nitrate (equivalent to 11.3 mg/L nitrate-N) in many areas [*Calder*, 2005]. However, afforestation or reforestation may negatively impact water quantity as shown by studies in the UK and South Africa [*Dye*, 1996; *Calder*, 2005]. Discrepancies in results among different studies in the UK make it difficult to determine potential impacts of forests on water resources. Various studies show reductions in recharge related to afforestation ranging from close to 0 to 75% [*Calder*, 2005]. Some field studies showed low nitrate levels under oak but high levels under pine, attributed to increased nitrate deposition in pine forests [*Calder*, 2005].

[42] Evaluation of afforestation of cropland in the Netherlands based on measured nitrogen concentrations and modeled water fluxes indicates that recharge would decrease from 485 mm/yr beneath cropland to 172 mm/yr beneath 18 year old oak stands and 100 mm/yr beneath 14 year old spruce stands [*Van der Salm et al.*, 2006]. Leaching of nitrogen was negligible under spruce stands and decreased from 16 kg/ha/yr under young oak stands to 8 kg/ha/yr under the 18 year old stand.

[43] Time lags between land use changes and system response (streamflow and water quality) are greater for afforestation (annual to decadal) than deforestation (days to months) because of the time required for trees to reach maximum water use. Annually the response time in streamflow can be regarded as the time delay for a catchment to reach a new equilibrium state following a disturbance. Data from field experiments in South Africa suggest that the time for catchments to reach a new equilibrium following afforestation varies from 10 to 20 years [*Scott and Smith*, 1997]. Independent estimates of response time using data from Australia are similar [*Lane et al.*, 2005].

3.1.2. Agricultural Land Use Management

[44] Changes in agricultural land use management are also being considered to remediate dryland salinity in Australia. Continuous or perennial cropping that eliminates fallow periods in crop rotations is being evaluated to reduce deep drainage and groundwater recharge. Replacing an 18 month fallow period with mustard in a crop rotation reduced drainage by 2 mm/yr and reduced cumulative soil water storage change by $\sim$2 mm/yr during an 8 year study over a wide range in annual precipitation (134 to 438 mm/yr) [*O'Connell et al.*, 2003]. However, elimination of the fallow period caused marked reductions in grain yield during drought conditions because fallow periods build up soil moisture for future rain-fed crops [*O'Connell et al.*, 2002]. Replacing winter fallow with a cover crop is also being evaluated in the High Plains, United States, to sequester nitrate and reduce nitrate leaching to the underlying aquifer. Eliminating fallow periods reduced water tables in the northern Great Plains and diminished saline seeps in this region [*Black et al.*, 1981; *Halvorsen*, 1984]. In general, most of the land use changes proposed to improve water quality by reducing salinization generally decrease water quantity.

[45] The most widespread negative impact of rain-fed agriculture in humid regions is reduction in water quality related to fertilizer applications. A European-wide study was conducted to assess different strategies for reducing nitrogen loading through land use changes and fertilization using a semidistributed integrated nitrogen model for multiple source assessment in catchments (INCA) [*Neal et al.*, 2002]. The INCA model showed that converting all cropland to unfertilized and ungrazed vegetation decreased nitrate-N levels in the Lee River (0.14 Mha area; tributary of the Thames, UK) by up to 7 mg/L [*Flynn et al.*, 2002]. Various national-scale models (e.g., NIRAMS, MAGPIE) have been developed to predict nitrate concentrations and fluxes from cropland to groundwater and surface water [*Lord and Anthony*, 2000; *Dunn et al.*, 2004]. Results from these models are needed to meet the requirements of the European Union Nitrates and Water Framework Directives [*Dunn et al.*, 2004]. These screening codes are useful for understanding sources and transport processes related to nitrate and predicting nitrate contamination related to LU/LC change.

[46] Various agricultural approaches have been proposed for reducing nitrogen loading to the Mississippi River Basin and, ultimately, the Gulf of Mexico [*Mitsch et al.*, 2001]. Converting from annual row crops (corn and soybeans) to

perennial crops (alfalfa and grass) should reduce nitrogen loading because of extended periods of water and nutrient uptake of perennial crops [*Mitsch et al.*, 2001]. Previous studies have shown 20 to 50 times greater nitrate losses from annual versus perennial crops [*Randall et al.*, 1997]. Other approaches include reducing nitrogen applications, changing timing of application from fall to spring, accounting for other sources of nitrogen from fertilizers and applied animal manure, and use of nitrification inhibitors [*Mitsch et al.*, 2001].

[47] Additional approaches for reducing nitrate contamination of streams and coastal zones include restoration of riparian buffers and wetlands, because of documented denitrification in these zones [*Jordan et al.*, 1993; *Woltenmade*, 2000]. In the Upper Mississippi River Basin, 14.1 Mha of wetlands were lost between the 1780s and 1980s, mostly as a result of agricultural drainage [*Dahl*, 1990]. Restoration of wetlands and creation of new wetlands and installation of riparian buffers between agricultural fields and adjacent streams should greatly reduce nitrate loading to the streams [*Mitsch et al.*, 2001]. Various factors can influence the ability of these systems to reduce nitrate loading. Distribution of flow relative to riparian zones or wetlands may be more important than width of riparian buffer strips [*Devito et al.*, 2000]. Shallow aquicludes that force groundwater to discharge through organic-rich riparian zones increase plant uptake and denitrification, such as in coastal plain soils along the eastern United States, whereas absence of aquicludes may allow water to bypass riparian zones completely by moving deeper ($\leq$30 m) in the profile, as in adjacent piedmont soils [*Jordan et al.*, 1993, 1997]. Nutrient removal increases with retention time in riparian zones or in wetlands and with the ratio of riparian zones or wetlands to the contributing zone [*Woltenmade*, 2000].

3.2. Optimization of Rain-Fed Agriculture

[48] Increased emphasis on rain-fed agriculture is projected for the future as water supplies for irrigated agriculture continue to decline, particularly in India and China. Approximately 95% of current population growth occurs in developing countries, which rely heavily on rain-fed agriculture [*Rockstrom and Falkenmark*, 2000; *Rockstrom et al.*, 2003]. Because 90% (1600 m^3/yr per capita) of human freshwater needs are water for food, increased productivity of rain-fed agriculture will be required to meet food demands of the higher population. Productivity can be increased either through extensification or expansion of agricultural land or through intensification related to limited irrigation, fertilization, and crop breeding [*Tilman et al.*, 2001].

3.2.1. Expansion of Rain-Fed Agriculture

[49] Expansion of agricultural areas has accounted for much of the increase in food production in developing countries in the last 2 decades [*Rockstrom and Falkenmark*, 2000]. Rain-fed agriculture is projected to increase in developing countries from 754 Mha (1997–1999) to 834 Mha (2030) (11% increase) [*Bruinsma*, 2003]. Agricultural expansions should occur primarily in sub-Saharan Africa and in Latin America (particularly Brazil) because of availability of suitable land in these regions [*Bruinsma*, 2003]. Because cultivation often increases groundwater recharge and many semiarid regions have large reservoirs of chloride and nitrate in unsaturated zones beneath natural ecosystems (Figure 4), conversion of new areas to agriculture may result in degradation of water quality, as is seen in parts of Australia, the U.S. High Plains, and Africa [*Allison et al.*, 1990; *Edmunds and Gaye*, 1997; *Scanlon et al.*, 2005]. It will be important to characterize the reservoirs of salts, particularly nitrate, in unsaturated zones in areas that are going to be converted to agriculture in order to assess the potential for mobilizing salts into underlying aquifers.

3.2.2. Intensification of Rain-Fed Agriculture

[50] Intensification of existing and newly developed rain-fed agriculture will be required to increase food productivity. The phrase "more crop per drop of rain" is used to reflect this enhanced productivity [*Falkenmark and Lannerstad*, 2005]. Various techniques can be used to enhance water use efficiency (kilogram of crop per unit of water) in rain-fed agriculture [*Rockstrom et al.*, 2003]. Critical issues include nonuniform spatial and temporal distribution of rainfall and low soil fertility. Rainwater harvesting is being promoted to provide supplemental irrigation of rain-fed agriculture that can address 2 to 4 week intraseasonal dry periods, which have large-scale effects on crop productivity, but it is generally insufficient to address long-term meteorological droughts [*Fox and Rockstrom*, 2000]. Rainwater harvesting generally consists of intercepting and concentrating runoff and storing it in the soil profile or in wells, tanks, or small reservoirs for crop production [*Ngigi*, 2003]. Systems for rainwater harvesting generally include a catchment area for collecting runoff, a storage device (ponds, subsurface tanks, 100–1000 m^3 etc.), and a distribution system (gravity-fed irrigation or low-technology drip systems) [*Boers and Ben-Asher*, 1982; *Rockstrom et al.*, 2003]. Water use efficiency increased by 38% with rainwater harvesting in study regions in Burkina Faso and Kenya and is further improved with the addition of fertilizers [*Rockstrom et al.*, 2003]. Rainwater harvesting is widely practiced in the Gansu Province in China using low-cost mobile or semifixed microdrip irrigation systems for water distribution [*Li et al.*, 2000]. Studies show that rainwater harvesting (35–105 mm/yr of water) increased yields of spring wheat by 28–50% in this region [*Li et al.*, 2000]. Rainwater harvesting systems are generally designed for small-scale applications, and upscaling may be difficult, particularly considering potential conflicts between upstream and downstream water users [*Ngigi*, 2003]. For example, the Rajasthan communal rainwater-harvesting system was destroyed because of potential reductions in water availability for the downstream irrigation department [*Ngigi*, 2003].

[51] In addition to traditional rainwater harvesting through runoff collection, various land management techniques, such as terracing, earth bunds on contours $\sim$5 to 10 m apart, retention ditches, and conservation tillage, can be considered in situ water-harvesting methods [*Rockstrom and Falkenmark*, 2000]. Conservation tillage includes reduced tillage or zero tillage, which decreases soil moisture loss because soil is not overturned, as in conventional tillage approaches. Crop yields have increased by up to 50% as a result of conservation tillage in Kenya [*Ngigi*, 2003]. Crop residue is generally retained at the soil surface, resulting in increased soil water storage because crop residue provides a barrier to soil water evaporation and can also reduce surface

runoff [*Hatfield et al.*, 2001]. Conservation tillage may not provide sufficient soil moisture in soils with low water-holding capacity to span the typical 1 to 2 week intraseasonal dry periods. Water-holding capacity may be enhanced by addition of organic matter in manure, which also has the advantage of slowly releasing fertilizers to crops [*Tilman et al.*, 2001].

3.3. Remediation of Water Resource Problems Associated With Irrigated Agriculture

[52] Irrigation has had large-scale impacts on global water resources (quantity and quality) and land resources (waterlogging and salinization). Strategies to remediate problems associated with existing irrigated areas can also be applied to optimize new areas that are converted to irrigated agriculture. Irrigated agriculture is projected to increase by 20% from 202 Mha (1997–1999) to 242 Mha [2030] in developing countries [*Bruinsma*, 2003]. The following remediation approaches, such as interbasin transfers, conjunctive use of surface water and groundwater etc. are not intended to provide a comprehensive list but are representative of different strategies to reduce negative environmental impacts of irrigated agriculture.

[53] To alleviate water quantity problems associated with irrigated agriculture, large-scale interbasin transfer projects have been developed for China and India. The South-North Water Transfer Scheme (SNWTS) will divert $\sim$45 $\times$ 10^6 m^3 of water from the Yangtze River north to the Yellow River [*Stone and Jia*, 2006]. This flow represents $\sim$5% of the flow in the Yangtze River and $\sim$100% of the flow in the Yellow River. The system includes three separate routes, the eastern route (1150 km, to Tianjin, 14.8 $\times$ 10^6 m^3 of water), the central route (1277 km, to Beijing and Tianjin, 13 $\times$ 10^6 m^3 of water) and the western route (headwaters of the Yangtze and Yellow Rivers (17 $\times$ 10^6 m^3). The National River Linking Project has been proposed in India to transfer $\sim$178 $\times$ 10^6 m^3 of water from monsoon runoff in 12 rivers in eastern India to semiarid regions in the west using 12,500 km of canals [*Bagla*, 2006; *Ghassemi and White*, 2006].

[54] Conjunctive use of surface water and groundwater instead of sole use of either resource can improve water quantity and quality aspects of irrigation in many regions. An excellent example of the benefits of conjunctive use of surface water and groundwater is provided by the Indus Basin Irrigation System (IBIS) in Pakistan [*Qureshi et al.*, 2004]. IBIS irrigates 16 Mha and consists of $\sim$ 63,000 km of canals [*Ghassemi et al.*, 1995]. Preirrigation groundwater levels were $\sim$ 30 m deep in the Indus Plain; however, poor irrigation management resulted in groundwater levels rising (0.15 to 0.7 m/yr) to land surface, resulting in waterlogging and salinization [*Ghassemi et al.*, 1995]. Large-scale groundwater pumpage was initiated in the 1960s as part of the Salinity Control and Reclamation Projects (SCARPS) [*Qureshi et al.*, 2004]. A total of 20,000 large capacity, public tube wells and $\sim$600,000 small capacity, private tube wells have been installed in the Pakistani Punjab to lower water tables and reduce waterlogging and salinization. Large groundwater level declines associated with groundwater-fed irrigation systems can also be reduced by supplementing with surface water-based irrigation. Therefore conjunctive use of surface water and groundwater for irrigation should help alleviate negative impacts associated with either irrigation source alone.

[55] Improving water use efficiency of irrigation systems may also reduce negative environmental impacts. Lining canal systems reduces conveyance losses in surface water-based irrigation systems. For example, lining $\sim$80% of an extensive canal system in the Shiyan River basin (Gansu Province) increased the water efficiency of the canal system from $\sim$0.30 in the 1950s to 0.54–0.72 in 2000; however, groundwater recharge was reduced 53% resulting in $\sim$15 m groundwater level declines in the past 40 years [*Kang et al.*, 2004]. These groundwater level declines have resulted in die-off of sand fixation and wind break vegetation, resulting in desertification. Therefore total system performance needs to be considered. Replacing flood irrigation, which is still widely used in China and elsewhere, with more efficient center pivot and/or drip (surface and subsurface) systems would greatly reduce water losses that do not result in biomass production. There is increased emphasis on subsurface drip irrigation in the High Plains (United States) to reduce evaporation that does not result in biomass production (nonbeneficial evaporation) or deep water drainage through more frequent irrigation applications that could result in fewer negative impacts on groundwater quality. Although the trend is toward more reduced irrigation applications to conserve limited water resources, the impact of such deficit irrigation on soil and water salinization by nonnutrient salts should be monitored [*Scanlon et al.*, 2005]. Flood irrigation systems were generally considered to be 60% efficient; i.e., 40% of the water either runs off or drains below the root zone. If we assume that excess water drains below the root zone, this level of irrigation efficiency would result in an increase in salinity by a factor of 2.5 relative to the applied water. However, because subsurface drip irrigation systems are considered to be $\geq$95% efficient, salinity of the resultant drainage would be at least 20 times greater than that of the applied water. Quality of the applied irrigation water is also an important factor to consider in evaluating salt balances in irrigated regions. Supplemental irrigation in more humid settings may reduce negative water quality problems associated with nonnutrient salts because large irrigation applications are not required, reducing the salt load; precipitation is higher, which can dilute salts in soil water; and evaporation and transpiration are much lower than in more arid regions, reducing salt buildup. Irrigated agriculture is increasing in humid regions in the United States [*Gollehon and Quinby*, 2006].

[56] Many other approaches may be used to reduce irrigation water use. Irrigation scheduling based on actual ET and soil moisture information should optimize irrigation applications. Reducing the size of bare rows between crops or uniform sowing of seeds rather than row cropping, as suggested for irrigated winter wheat in China, should reduce nonbeneficial evaporation [*Zhang et al.*, 2004]. Optimizing root distributions to maximize soil water use is also important. Deep plowing to break up hard pans was evaluated in China to increase deep root penetration that would use soil water in this zone [*Zhang et al.*, 2004]. The greatest water conservation can be achieved by eliminating irrigation and focusing on synchronizing crop production with seasonal and interannual precipitation variability. One of the options suggested for the North China Plain is to focus on summer maize and spring-summer cotton, which are grown during periods of moderate to high precipitation, and eliminate

winter wheat, which is grown during periods of low precipitation [*Zhang et al.*, 2004]. Nonsustainable groundwater-fed irrigation is generally self limiting because, as water tables decline and aquifer resources are depleted, irrigation is discontinued. Therefore many areas that are currently under irrigation have been or will be converted to rain-fed agriculture in the future.

[57] Remediating water quality problems associated with agrichemicals may be achieved by reducing fertilizer application rates and improving timing of fertilizer applications relative to crop nutrient requirements. Multiple fertilizer applications in irrigation water rather than a single large application should reduce nitrate leaching into underlying aquifers. Precision agriculture adjusts inputs of water and nutrients differentially across a field to meet crop needs, which is possible with advances in technology such as Geographic Information Systems, Global Positioning Systems, and automatically controlled farm machinery. Using centimeter-scale resolution with GPS, nutrients and water can be applied directly to crop roots to maximize uptake and minimize leaching.

4. Policy Implications of Integrated Land and Water Resources Management

[58] Linkages and feedbacks between land use and water resources should be strongly considered when developing policies related to either or both. *Calder* [2005] dispelled the myth that forests are always good for the environment. This myth has led to the notion that increases in forests result in conservation and rehabilitation and absence of forests results in degradation of water resources. The various improvements attributed to forests include increased rainfall and runoff, flow regulation, erosion reduction, and sterilization of water supplies [*Calder*, 2005]. The myth has resulted in widespread funding of afforestation and reforestation programs by governments, World Bank, and United Nations organizations to improve water resources. One of the largest reforestation programs is the Sloping Lands Conversion Program in China that involves conversion of cultivated land with slopes $\geq 25°$ to perennial vegetation. About 8 Mha of farmland has been converted to forests in this program, and trees have been planted on an additional 11 Mha of bare hills between 1999 and 2004 [*Calder*, 2005; *Xu et al.*, 2004]. However, forests generally have higher ET (4% on a global scale) relative to grasses and crops as a result of increased interception and greater rooting depths, resulting in less water being available for runoff or recharge [*Zhang et al.*, 2001; *Gordon et al.*, 2003, 2005]. Therefore reduced water availability should be considered in programs that promote afforestation or reforestation for erosion control, carbon sequestration, or for water quality improvement.

[59] Policies that promote land abandonment, particularly in developed countries such as the Conservation Reserve Program (CRP) in the United States, should also consider impacts of these land use changes on water resources. Since its inception in 1986, the CRP has retired ~14 Mha of environmentally sensitive cropland under 10 to 15 year contracts [*Hellerstein*, 2006]. Conversion of rain-fed agriculture to natural ecosystems could decrease groundwater recharge according to results of previous studies in Australia, United States, and Africa. Land abandonment outside defined policy programs might also have the unintended consequence of reducing groundwater recharge and should be monitored [*Gallart and Llorens*, 2003]. Alternatively, policies could be developed in suitable areas to encourage conversion of natural ecosystems to rain-fed agriculture through incentives because of potential positive impacts on water quantity through increased recharge. Comprehensive evaluation of environmental impacts of such conversions should be considered, such as the potential for erosion and salinization.

[60] An example of an advanced water policy program that integrates land use is provided by the South African National Water Act (NWA, Act 36, 1998). One of the advantages of the South African system is that water resource and forestry programs are integrated into the Department of Water Affairs and Forestry. The National Water Act includes a levy known as a resource conservation charge on activities that are considered to reduce streamflow (Streamflow Reduction Activities), such as forest plantations [*Calder*, 2005]. The NWA mandates reserving water for environmental flows.

5. Summary

[61] Major impacts of conversions of natural to agricultural ecosystems on water resources can be summarized as follows.

[62] 1. Conversions to rain-fed agriculture generally increase water quantity but degrade water quality. For example, conversion of eucalyptus mallee vegetation to pastures and crops in Australia increased recharge by one to two orders of magnitude, mobilizing salt reservoirs in soils into rivers and raising water tables, causing waterlogging and soil salinization (dryland salinity). Similar increases in recharge have been recorded in semiarid regions in the SW United States and Niger, Africa, that have degraded water quality by mobilizing salt reservoirs in unsaturated zones. Cultivation in more humid settings has resulted in increased water quantity (streamflow) and degradation of water quality, mostly related to large increases in fertilizer application (600% increase globally from 1961 to 2000), as in the Mississippi Valley and northeastern United States and Europe.

[63] 2. Conversions to irrigated agriculture decrease water quantity by consuming ~90% of global freshwater (1900–1995). Surface water based irrigation reduces streamflow and often raises water tables, resulting in waterlogging. Groundwater-fed irrigation lowers water tables up to 1 m/yr in many areas, also reducing streamflow(North China Plain; India; Central Valley and High Plains, United States).

[64] 3. Impacts of conversions to irrigated agriculture on water quality are similar to those of conversions to rain-fed agriculture: waterlogging and salinization in surface water-based systems and salt mobilization and increased fertilizer application in surface-water-and groundwater-based systems.

[65] 4. Remediation of water quality problems caused by rain-fed agriculture through reforestation/afforestation and agricultural land management (e.g., perennial versus annual crops) will decrease water quantity.

[66] 5. Remediation of water quantity and quality problems associated with irrigated agriculture can be achieved through interbasin transfers, conjunctive use of surface water and groundwater, and improved irrigation water use efficiency.

[67] 6. Optimization of agricultural land use management with respect to water resources will require removal of sectoral divisions between rain-fed and irrigated agriculture through supplemental irrigation of rain-fed agriculture (e.g., rainwater harvesting) and deficit irrigation in traditional irrigated areas.

[68] 7. Long time lags between land use change and water resource impacts need to be considered in determining when full-scale impacts of a land use change will be realized and in setting up remediation programs.

[69] **Acknowledgments.** We would like to acknowledge assistance with data compilation from Noel Gollehon (USDA), Susan Hutson (USGS), Navin Ramankutty (McGill University), and Chunmiao Zheng (University of Alabama). Daniel Kurtzman and Robert Reedy provided helpful reviews of earlier drafts of this manuscript. Comments from Eloise Kendy and an anonymous reviewer greatly improved the manuscript. The senior author would like to thank the Bureau of Reclamation and the Jackson School of Geosciences for providing financial support for this study.

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