# CHAPTER 4

# Environmental implications of intensive groundwater use with special regard to streams and wetlands

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"If you don't change course, you'll end up where you're headed".
*Chinese proverb*

ABSTRACT: Groundwater is a highly vulnerable and important resource to both humans and the environment. Therefore it is essential to understand the environmental implications of groundwater intensive use. This chapter emphasizes the hydrologic fundamentals for such understanding, which involve groundwater flow system concepts, the factors controlling aquifer responses to development, and surface water-groundwater interactions, and highlights the environmental consequences of groundwater intensive use throughout the world. Groundwater intensive use can result not only in aquifer depletion and water quality degradation, but also can impact the ecological integrity of streams and wetlands, and can result in significant losses of habitat and biodiversity. It is clear that intensive use and pollution in many regions of the world are threatening groundwater resources with serious consequences for human welfare and environmental degradation. Thus it is necessary for societies to recognize the finite limits of water availability and its vulnerability, and find ways to reconcile the demands of human development with the tolerance of nature.

## 1 INTRODUCTION

For thousands of years, groundwater (GW) has been a reliable source of high-quality water for human use. Springs have always been a treasured source of potable water even in humid environments. Because GW is much less dependent on recent precipitation than surface water (SW) sources, it is a uniquely reliable source of high-quality water for human use. Today, nearly 1,500 million people rely on GW as their sole source of drinking water (UNEP 1996). But GW is being depleted and degraded in many places, and as Burke *et al*. (1999) noted, most of the major aquifer depletion and degradation has occurred in a very short space of time – over the past 50 years.

Compared with domestic and industrial water uses, agriculture has a disproportionate impact on water flow, water quality, and alteration of freshwater habitats. About 70% of all water withdrawals are for agriculture (WMO 1997), but more than half of this water never makes it to the crops because of leakage and evaporation (Postel 1995). As population grows, we will depend even more on irrigation for our food supplies (Johnson *et al*. 2001). This places extraordinary stress on freshwater systems, particularly in arid and semi-arid regions.

It is essential to recognize the indispensable role GW plays as a basis for socio-economic development especially in semi-arid and arid regions. It is equally important to recognize that its uses can harm the resource base. Most uses of GW are consumptive or involve a degradation of water quality when returned to the system of shallow GW circulation. In the longer term, prolonged abstraction at rates equal to or greater than the rates of recharge will involve fundamental changes to the dynamics of aquifer systems. As Burke *et al*. (1999) pointed out, this interdependency of uses and impacts is fundamental and should be considered in assessing the key services GW produces and the problems facing sustainable management of GW resources.

Although GW plays a fundamental role in peoples' lives, humans are not the only entities dependent on GW. Base flows in streams, wetlands, and surface vegetation are in many cases dependent on GW levels, and corresponding

GW seepages. Change in those levels or changes in GW quality can induce ripple effects through terrestrial and aquatic ecosystems. Every ecosystem remaining in the 21st century experiences some impact from humans, but the degree varies widely. Some ecosystems are obviously more heavily affected than others. More than half the world's wetlands have been converted to other uses, especially agriculture, during the 20th century (Revenga *et al*. 2000). As a result, more than 20% of the world's freshwater fish species have become extinct, endangered, or threatened in recent decades (Johnson *et al*. 2001).

As Burke *et al*. (1999) also pointed out, however, overdraft and water-level declines typically affect the sustainability of uses that depend on GW long before the resource base itself is threatened with physical exhaustion. Many uses and environmental values depend on the depth to water –not the volumetric amount of GW theoretically available. Declines in GW levels can cause wetlands and stream flows to dry up even when the total amount of GW stored in a given basin remains huge. Depth to water also affects the economics of GW extraction because it takes more fuel to pump from greater depths. In many ways, the sharpest points of competition between uses may have to do with management objectives, not with allocation of the volumes of water available (Burke *et al*. 1999).

This chapter highlights the importance and vulnerability of GW resources not only to humans but to the environment in which we live, and lays down the hydrologic basis for understanding the environmental impacts of intensive GW use. Therefore, the objectives of this chapter are to briefly explain: 1) the hydrologic fundamentals for understanding the *plethora* of environmental impacts resulting from intensive GW use; 2) the consequences of increasing exploitation of GW on the quantity and quality of that resource base; and 3) the ecological implications of intensive GW use on streams and wetlands.

## 2 THE HYDROLOGIC BASIS OF ENVIRONMENTAL IMPACTS OF INTENSIVE GROUNDWATER USE

In order to understand environmental systems, cause-and-effect relationships, and impacts of human actions on GW, some background information is necessary. The following discussion focuses on: 1) systems concepts; 2) factors controlling aquifer response to development; and 3) surface water-groundwater (SW-GW) interactions.

### 2.1 *Some basics on system concepts with special regard to groundwater flow systems*

Systems thinking is a way to understand how things work. By looking beyond events to patterns of behavior, systems theory seeks out the underlying systemic interrelationships responsible for the patterns of behavior and events (Bellinger 2000). A system is defined as an entity that maintains its existence through the *mutual interaction* of its parts (Bertalanffy 1968). Understanding the interactions of the parts is key to understanding the system. For example, one could study hydrogen and oxygen in isolation from each other forever, and never discover the characteristic of wetness (Bellinger 2000). Wetness is an emergent characteristic of the mutual interaction of hydrogen and oxygen when combined to produce a water molecule. Only by studying the system does one gets a true understanding of wetness.

Time and space scales are key to understanding system dynamics. Particular phenomena appear to be more or less important at different scales in time and space. As Klemes (1983) pointed out, it is easiest to grasp things that are within the human scale, that is, accessible directly through the unaided senses: roughly from 1/10 millimeter to a few kilometers in space, and from 1/10 second to a few decades in time (Fig. 1). For instance, such concepts as channel slope, flow in a channel cross-section in open channel flow, or hydraulic head and permeability in GW flow would be far from obvious and natural if viewed at the molecular rather than the human scale (Klemes 1983). Similarly, the confinement to the human scale during the pre-technological era made it unlikely that anybody would conceive of a hurricane as a giant vortex of air or contemplate the earth as an almost perfect sphere (Klemes 1983).

Systems in nature are generally complex, and complex systems are hierarchical, that is, they are composed of interrelated, nested subsystems, each of which in turn is made of smaller subsys-

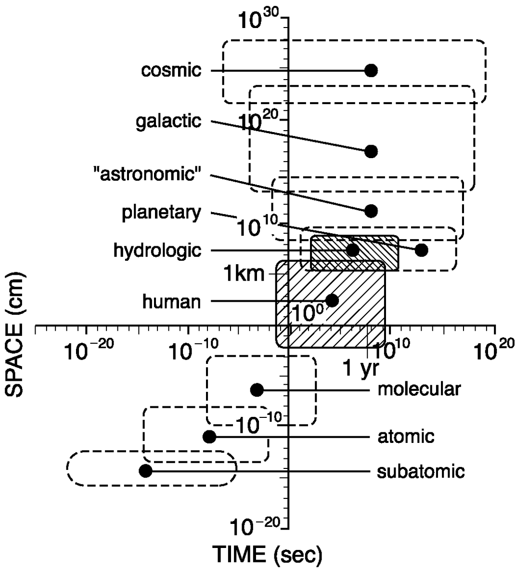


Figure 1. Space and time scales used for conceptual representations of physical processes. The term *astronomic* is used for the solar system scale. (From Klemes 1983).

tems until the lowest level is reached. A classic example of hierarchical nested structures in hydrology is Toth's (1962, 1963) GW flow system classification. GW moves along complex flow paths that are organized in space and form a *flow system*. In nature, the available subsurface flow domain of a region with irregular topography will contain a number of different flow systems of different orders of magnitude and relative, nested hierarchical order. Based on their relative position in space, Toth (1963) recognized three distinct types of flow systems: local, intermediate, and regional, which could be superimposed on one another within a GW basin. Water in a *local flow system* will flow to a nearby discharge area such as a pond or stream. Water in a *regional flow* system will travel a greater distance than the local flow system, and often will discharge to major rivers, large lakes, or to oceans. An intermediate flow system is characterized by one or more topographic highs and lows located between its recharge and discharge areas. Regional flow systems are at the top of the hierarchical organization; all other flow systems are nested within them.

Flow systems depend on both the hydrogeologic characteristics of the soil/rock material and landscape position. Zones of high permeability in the subsurface function as drains, which cause enhanced downward gradients in the material overlying the upgradient part of the high-permeability zone (Freeze & Witherspoon 1967). Areas with pronounced topographic relief tend to have dominant local flow systems, and areas that are nearly flat tend to have dominant intermediate and regional flow systems.

It is now recognized that in topography-controlled flow regimes, GW moves in systems of predictable patterns and that various identifiable natural phenomena are regularly associated with different segments of the flow systems. As Toth (1999) pointed out, such recognition was not appreciated until the 1960s (Toth 1962, 1963, Freeze & Witherspoon 1967) when the system-nature of GW flow had been understood. This recognition of the systems-nature of subsurface water flow has provided a unifying theoretical background for the study and understanding of a wide range of natural processes and phenomena (Toth 1999).

A schematic overview of GW flow distribution and some typical hydrogeologic conditions and natural phenomena associated with it in a gravity-flow environment is presented in Figure 2 (Toth 1999). On the left side of the figure, a single flow system is shown in a region with insignificant local relief; on the right side, a hierarchical set of local, intermediate, and regional flow systems is depicted in a region of composite topography. Each flow system, regardless of its hierarchical position, has an area of recharge, an area of throughflow, and one of discharge. In the recharge areas, the hydraulic heads, representing the water's potential energy, are relatively high and decrease with increasing depth, and water flow is downward and divergent. In discharge areas, the energy and flow conditions are reversed: hydraulic heads are low and increase with depth, resulting in converging and ascending water flow. In the areas of throughflow, the water's potential energy is largely invariant with depth (the isolines of hydraulic head are subvertical) and, consequently, flow is mainly lateral. The flow systems operate as conveyor belts that effectively interact with their ambient environment. The interaction produces *in situ* environmental effects, with the flow serving as the mechanism for mobilization, transport (distribution), and accumulation.

Typical environmental effects and conditions resulting from the action of GW moving in regional flow systems are illustrated in Figure 2 and enumerated by Toth (1999): 1) sub-, nor-

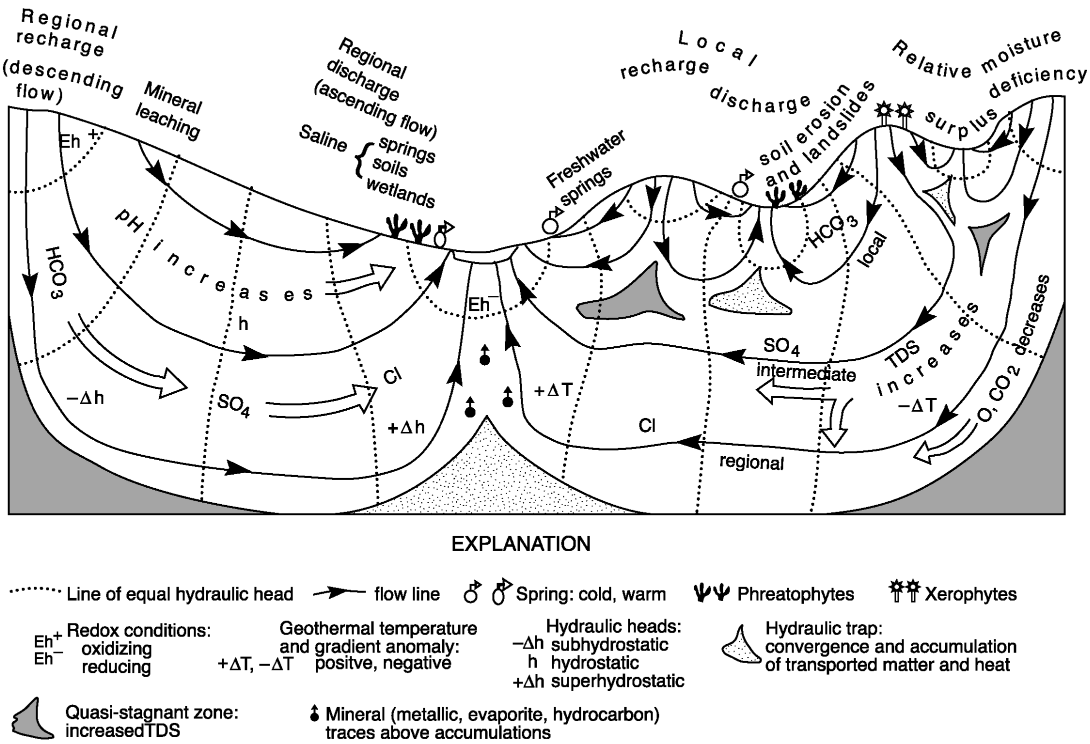


Figure 2. Effects and manifestations of gravity-driven flow in a regionally unconfined drainage basin. (Adapted from Toth 1999).

mal-, and super-hydrostatic hydraulic heads at depth in the direction of flow from recharge to discharge areas, respectively; 2) relatively dry surface-water and soil-moisture conditions (negative water balance) in recharge areas, and water surplus (positive water balance), possibly resulting in wetlands, in discharge areas; these conditions are expressed in comparison to an average water balance in the basin, which would result solely from precipitation and evapotranspiration; 3) systematic changes in the water's anion facies, from $HCO_3$ through $SO_4$ to Cl, both along flow systems and with depth; 4) chemically leached soils and near-surface rocks in areas of inflow, but increased salt contents possibly amounting to salt-affected soils or even commercial salt deposits at flow-system *termini*; 5) saline marshes in situations where wetland conditions and intensive salt supply coincide; 6) negative and positive anomalies of geothermal heat and geothermal gradients beneath areas of descending and ascending flows, respectively; 7) chemically oxidizing and reducing conditions in the near-surface environment of recharge and discharge areas, respectively; 8) identifiable response in the type and quality of vegetation cover to the contrasting nutrient and moisture conditions generated by the inflow and outflow of GW at flow-system extremities; 9) increased vulnerability of the land surface to soil- and rock-mechanical failures (such as soil erosion, slumping, quick grounds, and landslides) in areas of discharge, possibly developing into major geomorphologic features, such as gullying and stream meanders; and 10) accumulation of transported mineral matter such as metallic ions (uranium, sulfides), hydrocarbons, and anthropogenic contaminants, primarily in regions of converging flow paths (regional energy *minima*: hydraulic traps) or in regions where the fluid potential is minimum with respect to a transported immiscible fluid (oil, gas), e.g. at anticlinal structures, facies changes, grain-size boundaries, or in rocks of adsorptive minerals (local energy *minima*).

Studying flow systems in GW basins may help gain an understanding of the interrelations between the processes of infiltration and recharge at topographically high parts of the basin and of GW discharge through evapotranspiration and baseflow (Domenico 1972). For example, at least some of the water derived from precipitation that enters the ground in recharge areas will be transmitted to distant discharge points, and so cause a relative moisture deficiency in soils overlying recharge areas. Water that enters the ground in discharge areas may not overcome the upward potential gradient, and therefore becomes subject to evapotranspiration in the vicinity of its point of entry. Water input to saturated discharge areas generates overland flow, but in unsaturated discharge areas infiltrating water and upflowing GW are diverted laterally through superficial layers of high hydraulic conductivity. Further, the ramifications of human activities in discharge areas are immediately apparent. Some of these include: 1) water-logging problems associated with surface-water irrigation of lowlands; 2) water-logging problems associated with destruction of phreatophytes, or plants whose roots generally extend to the shallow GW for their water needs; and 3) pollution of shallow GW from gravity-operated sewage and waste-disposal systems located in valley bottoms in semi-arid basins where SW is inadequate for dilution (Domenico 1972).

In conclusion, GW problems must be viewed in terms of the overall system. Systems thinking is vital to the understanding of practical problems, such as GW contamination from point sources, or the hydrologic impact of a structure such as dam, waste disposal facility, or gravel pit. Many such studies suffer irreparably from the failure to place the local site in the context of the larger groundwater system of which the site is only a small part.

### 2.2 *Factors controlling the response of aquifers to development*

Our present quantitative approach to GW problems is based upon the hydrologic principles concisely stated by Theis (1940). According to Theis, the essential factors that determine the response of aquifers to development by wells are: 1) distance to and character of the recharge; 2) distance to the locality of natural discharge; and 3) character of the cone of depression in the aquifer, which depends upon the values of aquifer transmissivity ($T$) and storativity ($S$).

Under natural conditions, prior to development by wells, aquifers are in a state of approximate dynamic equilibrium: over hundreds of years, wet years in which recharge exceeds discharge offset dry years when discharge exceeds recharge. Discharge from wells upsets this equilibrium by producing a loss from aquifer storage; a new state of dynamic equilibrium is reached when there is no further loss from storage. This can only be accomplished by an increase in recharge (natural or artificial), a decrease in natural discharge, or a combination of the two.

Two possible conditions may exist in the recharge area. The potential recharge rate may seasonally (or even uniformly) exceed the rate at which water can flow laterally through the aquifer. In this case, the water table stands at or near the surface in the recharge area. The aquifer becomes overfull, and available recharge is rejected. In such locations, more water is available to replenish the flow if use of GW by means of wells can increase the rate of underground flow from the area.

On the other hand, the potential recharge rate may be less than the rate at which the aquifer can carry the water away. The rate of recharge in this case is governed by: 1) the rate at which water is made available by precipitation or by the flow of streams; or 2) the rate at which water can move vertically downward through the soil to the water table and thus escape evaporation. In recharge areas of this latter type, none of the recharge is rejected by the aquifer.

If water is rejected by the aquifer in the recharge area under natural conditions, then pumping of wells may draw more water (*induced recharge*) into the aquifer. On the other hand, no matter how great the normal recharge, if under natural conditions none of it was rejected by the aquifer, then there is no possibility of balancing the well discharge by increased recharge, except by the use of artificial recharge (such as water spreading or well injection).

Figure 3 indicates diagrammatically the difference between the two conditions. Near the mountain front where the water table is close to the surface, vegetation uses GW, and streams maintain their courses. This is the area of *rejected recharge*. If the water table in this zone is lowered, GW recharge will increase by decreasing the amount of transpiration and SW runoff. In the remainder of the area there is some recharge by rainfall, but the water table is so

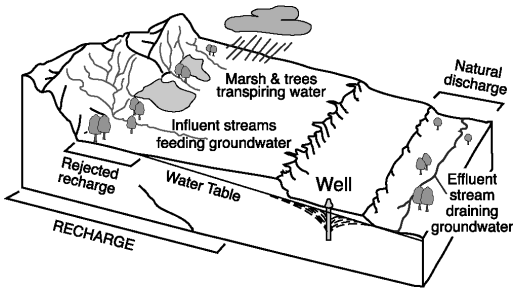


Figure 3. Factors controlling the response of an aquifer to discharge by wells. (Adapted from Theis 1940).

deep that comparatively small changes in its level will not affect the amount of recharge. Recharge is not rejected here, and when the water table is lowered by pumping, no more water will seep downward to recharge the GW body. For comprehensive outlines of GW recharge processes and estimation methodologies, see Scanlon *et al*. (2002), and Sophocleous (2002a, in press).

The above statements can be put in simple equation form (Lohman 1972, Sophocleous 1998). Under predevelopment conditions, a steady state or equilibrium condition prevails in most GW systems, and over a reasonable period of time natural recharge is equal to the natural discharge. The following equation expresses this equilibrium:

$$R = D \text{ or } R - D = 0 \quad (1)$$

where $R$ and $D$ are the natural recharge and discharge rates, respectively.

The following equation expresses the relationship between recharge and discharge after development:

$$(R + \Delta R) - (D + \Delta D) - Q + dV/dt = 0 \quad (2)$$

where $\Delta R$ = change in the mean rate of recharge; $\Delta D$ = change in the mean rate of discharge; $Q$ = rate of withdrawal from wells due to development; and $dV/dt$ = rate of change in storage in the system ($V$ is the volume of water stored in the system).

Denoting the increase in recharge $\Delta R = r$ and the decrease in discharge $\Delta D = -d$, we can rewrite Equation 2 as:

$$(R + r) - (D - d) - Q + dV/dt = 0 \quad (2a)$$

From Equation 1 and Equation 2a, we can obtain:

$$r + d - Q + dV/dt = 0 \quad (3)$$

If dynamic equilibrium can be reestablished, there will be no further withdrawals from storage; in this case, $dV/dt = 0$, and the system has reached a steady state condition, and Equation 3 can be rewritten as:

$$r + d = Q \quad (4)$$

where the sum $(r + d)$ –i.e. the decrease in discharge, $d$, plus the increase in recharge, $r$– is called *capture*.

Capture may occur in the form of pulling waters directly from streams (induced recharge), intercepting the GW discharge into streams, lakes, and the ocean, or from reducing evapotranspiration derived from the saturated zone in the riparian and other areas where the water table is near the ground surface. After a new artificial withdrawal from the aquifer has begun, the head in the aquifer will continue to decline until this new withdrawal is balanced by capture. Thus, the ultimate production of GW from wells depends on how much the rate of recharge and/or discharge can be changed, i.e. how much water can be captured.

### 2.3 *Surface water-groundwater (SW-GW) interactions*

GW and SW are closely interrelated systems (Brunke & Gonser 1997, Winter *et al*. 1998, Sophocleous 2002a). GW feeds springs and streams (have you ever wondered why streams flow during dry weather?), and SW recharges aquifers. The decline of GW levels around pumping wells located near a SW body creates gradients that capture some of the ambient GW flow that would have, without pumping, discharged as base flow to the SW. At sufficiently large pumping rates, these declines induce flow out of the body of SW into the aquifer, a process known as induced infiltration or recharge. The sum of these two effects leads to *streamflow depletion*. Stream-aquifer interactions are also important in situations of GW contamination by polluted SW, and of degradation of SW by discharge of saline or other low-quality GW.

The larger-scale hydrologic exchange of GW and SW in an area is controlled by: 1) the distribution and magnitude of hydraulic conductivities both within the channel and the associated alluvial plain sediments; 2) the relation of stream stage to the adjacent GW levels; and 3) the geometry and position of the stream channel within the alluvial plain (Woessner 2000). The direction of the exchange processes varies with hydraulic head, whereas flow depends on sediment hydraulic conductivity. Natural events, such as precipitation and seasonal evapotranspiration patterns or human-induced events, such as GW pumping, can alter the hydraulic head and thereby induce changes in flow direction. Two net directions of water flow can be distinguished: 1) the *influent* condition, where SW contributes to subsurface flow (*losing streams*); and 2) the *effluent* condition, where GW drains into the stream (*gaining streams*), thus contributing to stream *baseflow*. On the other hand, variable flow regimes could alter the hydraulic conductivity of the sediment via erosion and deposition processes, and thus affect the intensity of the SW-GW interactions. Thus, the interactions of streams, lakes, and wetlands with GW are governed by the positions of the water bodies with respect to GW flow systems, geologic characteristics of their beds, and their climatic settings (Winter *et al*. 1998, Winter 1999). Consequently, for a thorough understanding of the hydrology of SW bodies, all three factors should be taken into account.

Consider a stream-aquifer system such as an alluvial aquifer discharging into a stream, where the term *stream* is used in the broadest sense of the word to include rivers, lakes, ponds, and wetlands. A new well drilled at some distance from the stream, and pumping the alluvial aquifer forms a cone of depression. The cone grows as water is taken from storage in the aquifer. Eventually, however, the periphery of the cone arrives at the stream. At this point, water will either start to flow from the stream into the aquifer, or discharge from the aquifer to the stream will appreciably diminish or cease. The cone will continue to expand with continued pumping of the well until a new equilibrium is reached in which induced recharge from the stream balances the pumping.

The length of time, $t$, before an equilibrium is reached depends upon: 1) the aquifer diffusivity (expressed as the ratio of aquifer transmissivity to storativity, $T/S$), which is a measure of how fast a transient change in head will be transmitted throughout the aquifer system; and 2) upon the distance from the well to the stream, $x$. Thus, the time lag between the imposition of a stress in a stream-aquifer system and that system's response at a distance $x$ from the stress point is proportional to the square of the distance, $x$, and inversely proportional to the aquifer (hydraulic) diffusivity ($T/S$). For radial flow of GW, a tenfold increase in distance from the SW body causes a hundredfold delay in the response time, whereas a change in diffusivity is linearly proportional to the response time (Balleau 1988). Generally, if the wells are distant from the stream, it takes tens or even hundreds of years before their influence on streamflow is felt.

Once the well's cone of depression has reached an equilibrium size and shape, all of the pumping is balanced by flow diverted from the stream. In that case, a water right to withdraw GW from the well, as described, becomes a water right to divert from the stream at the same rate. A crucial point, however, is that before equilibrium is reached (that is, before all water is coming directly from the stream), the two rights are not the same (DuMars *et al*. 1986). Until the perimeter of the cone reaches the stream, the volume of the cone represents a volume of water that has been taken from storage in the aquifer, over and above the subsequent diversions from the river. It is this volume that may be called *aquifer-storage depletion* or *groundwater depletion*. Thus, GW sources include GW (aquifer) storage and induced recharge of SW.

The shape of the transition or growth curve for an idealized, two-dimensional, homogeneous, and isotropic system is shown in Figure 4 in nondimensional form, based on Glover's (1974) analytical solution and tabulation. In Figure 4, the percent of GW withdrawal derived from GW storage is plotted on the Y-axis against dimensionless (or normalized) time on the X-axis. For example, if GW storage is ~85% of the water source after 1 month (or 1 year) of pumping, it will end up being only ~5% of the water pumped coming from aquifer storage after 1,000 months (or 1,000 years) of pumping. The general shape of the transition curve is retained in systems with apparently different boundaries and parametric values (Balleau 1988). The rate at which dependence on GW storage (as shown at the left portion of the graph) converts to dependence on SW depletions (as shown on the right portion of the graph) is highly variable and is particular to each case.

The initial and final phases of the transition curve (Fig. 4), representing aquifer storage depletion on the left and induced recharge on the right, are separated in time by a factor of nearly 10,000. As the example above showed, full reliance on induced recharge takes an extremely long time. The distinct category of GW mining depends entirely upon the time frame. Initially, all GW developments mine water but ultimately they do not (Balleau 1988, Sophocleous 1997).

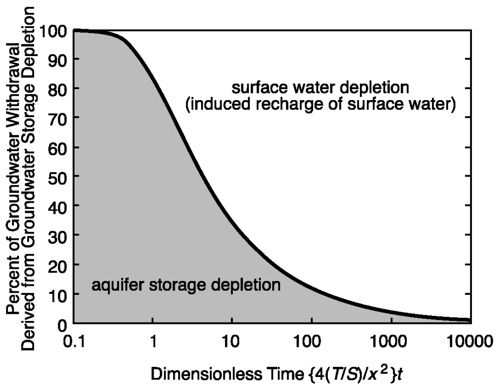


Figure 4. Transition or growth curve for an idealized aquifer, representing the transition from reliance upon groundwater storage to induced recharge of surface water. $T$ is transmissivity, $S$ is storativity, $x$ is the distance from pumping well to stream, and $t$ is time.

## 2.4 *An example of the dynamic response of a groundwater system to development*

To illustrate the dynamics of a GW system under development, Bredehoeft *et al*. (1982) chose a simple, yet realistic, system for analysis –a closed intermontane basin of the sort common in the Western USA (Fig. 5). Under predevelopment conditions, the system is in equilibrium: phreatophyte evapotranspiration in the lower part of the basin (the natural discharge from the system) is equal to recharge from the two streams at the upper end. Pumping in the basin is assumed to equal the recharge. This system was simulated by a finite-difference approximation to the equations of GW flow (Bredehoeft *et al*. 1982) for 1,000 years. Stream recharge, phreatophyte-water use, pumping rate, and change in storage for the entire basin were graphed as functions of time. Two development schemes were examined: Case 1, in which the pumping was more or less centered within the valley; and Case 2, in which the pumping was adjacent to the phreatophyte area (Fig. 5).

The system does not reach a new equilibrium until the phreatophyte-water use (i.e. the natural discharge) is entirely salvaged or captured by pumping (Fig. 6). In other words, phreatophyte water use eventually approaches zero as the water table drops and plants die. In Case 1, phreatophyte-water use is still approximately 10% of its initial value at year 1,000 (Fig. 6). In Case 2, it takes approximately 500 years for the phreatophyte-water use to be completely captured. These curves are similar to the transition or growth curves referred to earlier (Fig. 4), where initially most of the water pumped was coming out of aquifer storage, whereas at later times it was coming from capturing GW discharge.

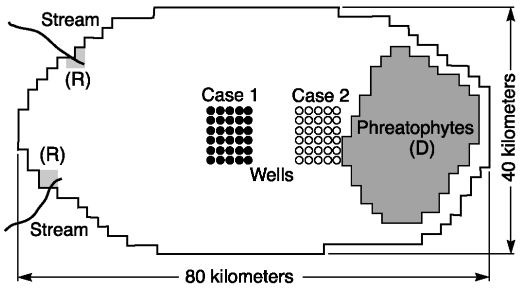


Figure 5. Schematic map of an intermontane basin showing areas of recharge (R) discharge (D), and two hypothetical water-development schemes, Case 1 and Case 2, described in the text. (Adapted from Bredehoeft *et al*. 1982).

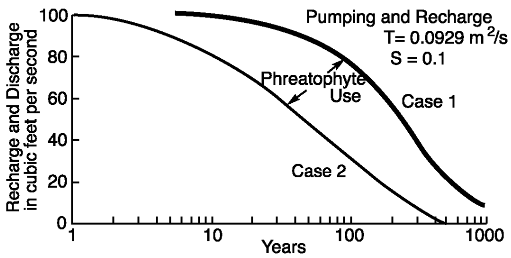


Figure 6. Plot of the rate of recharge, pumping, and phreatophyte water use for the system depicted in Figure 5. To convert cubic feet per second to liters per second multiply by 28.3. (Adapted from Bredehoeft *et al*. 1982).

This example illustrates three important points (Bredehoeft *et al*. 1982). First, the rate at which the hydrologic system can be brought into equilibrium depends on the rate at which the discharge can be captured. Second, the placement of pumping wells changes the dynamic response and the rate at which natural discharge can be captured; and third, some GW must be mined before the system can approach a new equilibrium. Steady state is reached only when pumping is balanced by capturing discharge and, in some cases, by a resulting increase in recharge. In many circumstances, the dynamics of the GW system are such that long periods of time are necessary before any kind of an equilibrium condition can develop. In some circumstances the system response is so slow that mining will continue well beyond any reasonable planning period.

## 3 ENVIRONMENTAL CONSEQUENCES OF INTENSIVE GROUNDWATER USE

### 3.1 *Groundwater mining and water-quality degradation*

With the hydrologic basis outlined in the previous section at hand, we are now ready to better assess the environmental consequences of human-imposed stresses on GW systems. There are two major consequences of increasing exploitation of GW supplies (Revenga *et al*. 2000). One is *GW mining*, in which GW abstraction consistently exceeds the natural rate of replenishment. This can result in *land subsidence*, *saltwater intrusion*, and *GW supplies becoming economically and technically unfeasible for use as a stable water supply*. The second major consequence is the *degradation of water quality* resulting from a variety of point and nonpoint source pollutants, including agricultural runoff, sewage from urban centers, and industrial effluents.

GW overpumping and the consequent aquifer depletion are now occurring in many of the world's most important crop-producing regions (Table 1). As Postel (1996) pointed out, this not only signals limits to expanding GW use, it means that a portion of the world's current food supply is produced by using water unsustainably –and can therefore not be counted as reliable over the long term.

In Europe (Revenga *et al*. 2000), the European Environment Agency (EEA) shows that nearly 60% of the cities with more than 100,000 people are located in areas where there

Table 1. Groundwater depletion in major regions of the world, circa 1990. (Adapted from Postel 1996).

| Region/Aquifer | Estimates of Depletion |
|---|---|
| High Plains Aquifer System, USA | Net depletion to date of this aquifer that underlies nearly 20% of all USA irrigated land totals some 325,000 $Mm^3$, roughly 15 times the average annual flow of the Colorado River. More than two thirds of this occurred in the Texas High Plains, where irrigated area dropped by 26% between 1979 and 1989. Current depletion rate is estimated at 12,000 $Mm^3$/yr. |
| California, USA | GW overdraft averages 1,600 $Mm^3$/yr, amounting to 15% of the state's annual net GW use. Two-thirds of the depletion occurs in the Central Valley, the country's vegetable basket. |
| Southwestern USA | Water tables have dropped more than 150 m east of Phoenix, Arizona, and resulted in land subsidence and fissuring causing damage to buildings and sewer systems. Projections for Albuquerque, New Mexico, show that if GW withdrawals continue at current levels, water tables will drop an additional 20 m by 2020. |
| Mexico City and Valley of Mexico | Pumping exceeds natural recharge by 50-80%, which has led to falling water tables, aquifer compaction, land subsidence, and damage to surface structures. |
| Arabian Peninsula | GW use is nearly three times greater than recharge. Saudi Arabia depends on nonrenewable GW for roughly 75% of its water, which includes irrigation of 2-4 million tons wheat/yr. At the depletion rates projected for the 1990s, exploitable GW reserves would be exhausted within about 50 years. |
| African Sahara | Vast non-recharging aquifers underlie North Africa. Current depletion is estimated at 10,000 $Mm^3$/yr. |
| India | Water tables are falling throughout much of Punjab and Haryana states, India's breadbasket. In Gujarat, GW levels declined in 90% of observation wells monitored during the 1980s. Large drops have also occurred in Tamil Nadu. |
| North China | The water table beneath portions of Beijing has dropped 37 m over the last four decades. North China now has eight regions of overdraft, covering 1.5 million ha, much of it productive irrigated farmland. |
| Southeast Asia | Significant overdraft has occurred in and around Bangkok, Manila, and Jakarta. Overpumping has caused land to subside beneath Bangkok at a rate of 5-10 cm/yr for the past three decades. |

is GW overabstraction (EEA 1995). GW overexploitation is also evident in many Asian cities. The cities of Bangkok, Manila, Tianjin, Beijing, Madras, Shanghai, and Xian, for example, have all registered a decline in water table levels of 10–50 m (Foster *et al*. 1998). In the USA, Albuquerque, Phoenix, and Tucson are among the larger cities that are overdrafting their aquifers. In Latin America, Mexico City, San Jose, Lima, and Santiago are among the cities known to be highly dependent on GW (UNEP 1996). This overexploitation in many cases is accompanied by water-quality degradation and land subsidence. For instance in Mexico, where many cities have experienced declines in GW levels, water levels in the aquifer that supplies much of Mexico City fell by 10 m as of 1992, with a consequent land subsidence of up to 9 m (Foster *et al*. 1998). As Revenga *et al*. (2000) also pointed out, one of the worst cases of GW overexploitation is Yemen where, in some areas, the rate of abstraction is 400% greater than the rate of recharge.

Subsidence can occur where the land surface compacts and permanently lowers the storage capacity of the aquifer. Adams & MacDonald (1998) succinctly summarized the land subsidence mechanism. In general, the weight of the overburden compresses the underlying strata. This weight is balanced by the effective intergranular stress in the skeleton of the underlying aquifer in combination with the pore-water pressure. GW abstraction has the effect of decreasing the pore-water pressure, thus increasing the effective stress from the overlying strata on the aquifer matrix. When the increase in effective stress is greater than a critical value (the preconsolidation stress), the resulting compaction of the sediments is mainly inelastic and therefore not recoverable (Adams & MacDonald 1998). Coarse-grained sandy aquifer strata form a rigid matrix skeleton that generally resists compaction, whereas finer-grained clayey strata are more compressible and hence more prone to compaction. Where relatively coarse-grained aquifers are bounded by fine-grained aquitards or confining layers, GW abstraction from the coarse layers can induce leakage from the aquitards; the resulting delayed dewatering of the aquitards can result in greater compaction than occurs in the aquifer. Thus, in a multilayered system consisting of coarse-grained aquifers separated by clayey aquitards, cumulative compaction of the aquitards layers can result in significant subsidence at the ground surface, as occurred in the Mexico City basin we mentioned earlier, in the San Joaquin Valley of California –the largest known area of subsidence (10,875 km$^2$ has subsided more than 0.3 m)– where land levels have fallen as much as 9 m, and in many other places around the world (Table 2).

The second major consequence of the overuse of GW is the degradation of water quality. This degradation results from a variety of point and nonpoint source pollutants from agriculture, industry, untreated sewage, saltwater intrusion, and natural GW contamination. Once pollutants enter an aquifer, the environmental damage can be severe and long lasting, partly because of the extremely long time needed to flush pollutants out of the aquifer.

In coastal zones, GW overabstraction can reverse the natural flow of GW into the ocean, causing saltwater to intrude into inland aquifers. This can be briefly explained as follows (Goudie 2000). Freshwater has a lower density than salt water, such that a column of sea water can support a column of freshwater approximately 2.5% higher than itself (or a ratio of about 40:41). So where a body of freshwater has accumulated in a coastal aquifer that is also open to penetration from the sea, it does not simply lie flat on top of the salt water but forms a lens, whose thickness is approximately 41 times the elevation of the water table above sea-level. This is called the *Ghyben-Herzberg principle*. The corollary of this rule is that if the hydrostatic pressure of the freshwater falls as a result of overpumping in a well, then the underlying salt water will rise by 40 units for every unit by which the freshwater table is lowered.

Scheidleder *et al*. (1999) pointed out that because of the high salt content, a concentration of only 2% seawater in an aquifer is enough to make GW supplies unusable for human consumption. Thus saltwater intrusion is a major problem in regions that depend on coastal aquifers for their water supply or irrigation. Scheidleder's *et al*. (1999) study of GW resources in Europe shows that saltwater intrusion as a consequence of overabstraction is most prevalent in the Mediterranean countries, particularly along the coastlines of Spain, Italy, and Turkey.

Saltwater intrusion can also occur in inland

Table 2. Areas of major land subsidence due to groundwater overdraft. (Adapted from Poland 1972).

| Location | Depositional Environment and Age | Depth range of compacting beds m | Maximum subsidence m | Area of subsidence km$^2$ | Time of principal occurence |
|---|---|---|---|---|---|
| Japan | | | | | |
| Osaka | Alluvial and shallow marine; Quaternary | 10-400 | 3 | 190 | 1928-1968 |
| Tokyo | As above | 10-400 | 4 | 190 | 1920-1970 |
| Mexico | | | | | |
| Mexico City | Alluvial and lacustrine; late Cenozoic | 10-50 | 9 | 130 | 1938-1970 + |
| Taiwan | | | | | |
| Taipei basin | Alluvial and lacustrine; Quaternary | 10-240 | 1.3 | 130 | 1961-1969 + |
| USA | | | | | |
| Arizona, central | Alluvial and lacustrine; late Cenozoic | 10-550 | 2.3 | 650 | 1948-1967 |
| California | | | | | |
| Santa Clara Valley | Alluvial and shallow marine; late Cenozoic | 55-300 | 4 | 650 | 1920-1970 |
| San Joaquin Valley (three subareas) | Alluvial and lacustrine; late Cenozoic | 60-1,000 | 2.9-9 | 11,000 (> 0.3 m) | 1935-1970 + |
| Lancaster area | Alluvial and lacustrine; late Cenozoic | 60-300 (?) | 1 | 400 | 1955-1967 + |
| Nevada | | | | | |
| Las Vegas | Alluvial; late Cenoizoic | 60-300 | 1 | 500 | 1935-1963 |
| Texas | | | | | |
| Houston-Galveston area | Fluvial and shallow marine; late Cenozoic | 60-600 (?) | 1-1.5 | 6,860 (> 0.15 m) | 1943-1964 + |
| Louisiana | | | | | |
| Baton Rouge | Fluvial and shallow marine; Miocene to Holocene | 50-600 (?) | 0.3 | 650 | 1934-1965 + |

areas where GW overexploitation leads to the rise of highly mineralized water from deeper aquifers. This problem has been reported in the Central USA (Oklahoma, Texas, New Mexico, Colorado, and Kansas), where active dissolution of Permian-age salt beds at relatively shallow depths (< 300 m) is especially pronounced; in Europe (Latvia, Poland, and the Republic of Moldova); and other areas of the world.

### 3.2 *The impact of agriculture and increasing world population and needs*

Crop production is a highly water-intensive activity, using about 65% of all the water removed from rivers, lakes, and aquifers for human activities worldwide, compared with 25% for industries and 10% for households and municipalities (Postel 1996). It takes about 1,000 tons of water to produce a ton of harvested grain. (This figure includes the moisture transpired by crops and evaporated from the surrounding soil, but not the water that is lost because of inefficiencies in irrigation methods. As such, it represents an approximate minimum water requirement for the production of grain, the source of roughly half of human calories). Irrigated lands account for only 16% of the world's cropland, but they yield some 40% of the world's food (Postel 1996).

Each year, some 2,700 km$^3$ of water –about five times the annual flow of the Mississippi, are removed from the earth's rivers, streams, and underground aquifers to water crops (Postel 1993). Practiced on such a large scale, irrigation has had a profound impact on global water bodies and on the cropland that is watered. Waterlogged and salted lands, declining and contaminated aquifers, shrinking lakes and inland seas, and the destruction of aquatic habitats are some of the environmental costs of irrigation (Postel 1993). A well-known case of river water overabstraction for irrigation is the remarkable shrinkage and salinization of the Aral Sea and the consequent loss of fish species and fishing livelihoods. Mounting concern about this damage is making large new water projects increasingly unacceptable.

Nitrogen fertilizers and pesticides are a significant source of widespread contamination of both SW and GW resources in both industrialized and industrializing countries. A brief look at *nitrate pollution* ($NO_3$) will illustrate contamination problems affecting GW resources around the world. In regions where well-drained soils are dominated by irrigated cropland, there is a strong propensity towards development of large areas with GW that exceeds 45 mg/L $NO_3$ (or 10 mg/L $NO_3$-N), which is the U.S. Environmental Protection Agency's maximum contaminant level for drinking water (Spalding & Exner 1993).

In China, nitrogen fertilizer consumption increased sharply in the 1980s and now equals application rates in Western Europe (Revenga *et al*. 2000). Fourteen cities and counties in Northern China, covering an area of 140,000 km$^2$, were sampled to assess the extent of nitrate contamination of GW supplies (Zhang *et al*. 1996). This area had over 20,000 Mm$^3$ of GW withdrawals in 1980, mostly for irrigation but with large amounts used for drinking water supplies as well (UN 1997). Over one-half of the sampled areas had nitrate concentrations that were above the allowable limit for nitrate in drinking water.

Pollution of GW supplies from synthetic fertilizer application is also a problem in parts of India (Revenga *et al*. 2000). GW samples in the states of Uttar Pradesh, Haryana, and Punjab were found to have between 5 and 16 times the prescribed safe amount of nitrate, with one site in Haryana almost 30 times the prescribed limit. GW in these areas is also being progressively depleted because of overabstraction for irrigation.

A recent European assessment of GW resources (Scheidleder *et al*. 1999) showed that in some regions of France, the Netherlands, and Slovenia, nitrate concentrations exceeded 50 mg/L in 67% of sampling sites. In Poland and Moldova, GW wells with nitrate concentrations in excess of 45 mg/L could be found across all parts of both countries (Scheidleder *et al*. 1999).

In the USA, an analysis of nutrients and pesticides in GW shows that high nitrate concentrations in shallow GW are widespread and closely correlated with agricultural areas in the USA Great Plains, the Central Valley of California, and parts of the north-west and mid-Atlantic regions (USGS 1999).

By far the most pervasive damage to GW stems from waterlogging and soil salinization brought about by poor water management (Postel 1993). Without adequate drainage, seepage from unlined canals and overwatering of fields causes the water table to rise. Eventually, the root zone becomes waterlogged, starving plants of oxygen and inhibiting their growth. In such cases, intensive GW use could have a positive environmental impact, as occurred in Punjab, Pakistan (van Steenbergen & Oliemans, in press), where intensive GW use for irrigation induced drainage movement and alleviated waterlogging problems. Such drainage, in turn, resulted in richer diversity of grasses for livestock, and also reduced water-born vectors (van Steenbergen, pers. comm.)

In dry climates, evaporation of water near the soil surface leaves behind a layer of salt that also reduces crop yields and eventually, if the buildup becomes excessive, kills the crops. Salts also get added to the soil from the irrigation water itself. Even the best water supplies typically have concentrations of 200-500 mg/L. (For comparison, ocean water has a salinity of about 35,000 mg/L; water with less than 1,000 mg/L is considered fresh; and the recommended limit for drinking water in the USA is 500 mg/L). Applying 10,000 m$^3$/ha/yr of such freshwater, a fairly typical irrigation rate (Postel 1993), thus adds between 2 and 5 tons of salt to the soil annually. If it is not flushed out, enormous quantities can build up in just a few decades, greatly damaging the land. Aerial views of abandoned irrigated areas in the world's dry regions reveal vast expanses of glistening white salt, land so destroyed it is essentially useless (Kovda 1983). Approximately 25 million ha, more than 10% of world irrigated area, suffer from yield-suppressing salt buildup, and this figure is expanding each year (Postel 1993).

In the Western USA, excessive irrigation and poor drainage have spawned another set of problems (NAS 1989). There, scientists have linked alarming discoveries of water-bird embryo deformities and death, and reproductive failure in fish, birds, and other wildlife to agricultural drainage water laced with toxic elements. Selenium had leached from agricultural soils, moved through drainage systems, and became concentrated in the Kesterson National Wildlife Refuge ponds in the San Joaquin River Valley in California. Irrigation has washed more selenium and other toxic chemicals out of the soil in several decades than natural rainfall would have done in centuries (Postel 1993). Since 1985, intensive investigations throughout the region have found lethal or potentially hazardous selenium concentrations at 22 different wildlife sites, including Kesterson National Wildlife Refuge.

In the USA, more than 4 million ha –roughly a fifth of USA's irrigated area– are watered by pumping in excess of recharge (Postel 1993). By the early 1980s, the depletion was already particularly severe in parts of Texas, California, Kansas, and Nebraska, four important food-producing states. As Postel (1993) pointed out, current year-to-year fluctuations in the USA irrigated area reflect crop prices and government farm policies more than water availability and cost. But overpumping cannot continue indefinitely. The 4 million ha currently watered unsustainably will eventually come out of irrigated production.

As of 1995, the world consumed directly or indirectly (through animal products) an average of just over 300 kg/yr of grain per person (Postel 1996). At this level of consumption, growing enough grain for the 90 million people now added to the planet each year (Postel 1996) requires an additional 27,000 $Mm^3$ of water annually, roughly 1.3 times the average annual flow of the Colorado River, or about half that of China's Huang He (Yellow River). Assuming the global average grain consumption remains the same as today, Postel (1996) estimated that it will take an additional 780,000 $Mm^3$ of water to meet the grain requirement of the projected world population in 2025 –more than nine times the annual flow of the Nile River.

Much of the crop production needed to meet future food needs would thus seem to depend on an expansion of irrigation. But it is becoming increasingly difficult to supply additional water for agriculture. To achieve a secure future, societies need to recognize the finite limits of water availability and bring human numbers and demands into line with them.

## 4 ECOLOGICAL IMPLICATIONS OF INTENSIVE GROUNDWATER USE ON STREAMS AND WETLANDS

### 4.1 *Streams, with some examples from Kansas, USA*

Freshwater ecosystems, including rivers, floodplains, lakes, swamps, wetlands, and deltas, perform a host of vital functions (Postel 1997). Rivers, for example, deliver nutrients to the seas and so nourish marine food webs. They sustain fisheries, dilute our waste products, provide convenient shipping channels, create habitat for a rich diversity of aquatic life, maintain soil fertility, and offer us some of the most inspirational natural beauty on the planet. These functions are easy to take for granted because they are rarely priced by the market, and they require virtually no investment on our part. Their value to us, however, is enormous (Postel & Carpenter 1997). The ecological integrity of GW and fluvial systems is often threatened by human activities, which can reduce connectivity, alter exchange processes, and lead to toxic or organic contamination.

Probably the best known example of intensive GW use is the Ogallala or High Plains aquifer, where declines of more than 30 m over a 30 years period (1950 to 1980) were common in parts of Texas, New Mexico, and Kansas (Fig. 7). Maps comparing the perennial streams in Kansas in the 1960s to those in the 1990s (Fig. 8) show a marked decrease in the length (in the order of many kilometers) of flowing streams in the western third of the state (Sophocleous 1998, 2000a, 2002b). Figure 9 shows a graph of median annual discharge in the Arkansas River in Western Kansas, based on daily streamflow records for the stream gauging stations at Garden City, Dodge City, Kinsley, and Great Bend for the period 1950-1995. The pattern in reduction of stream discharge since the mid-1970s is clearly visible. This pattern of streamflow decline is typical of most Western and Central Kansas streams (Sophocleous 1998, 2000a, b, 2002b).

These modified streamflow regimes have greatly altered the composition of the riverine community. Species most highly adapted, morphologically and behaviorally, to plains rivers have been decimated (Cross & Moss 1987). Table 3 lists fishes reported by Hay (1887, cited in Cross & Moss 1987) and their subsequent fates at two Western Kansas streams –the Smoky Hill River, near Fort Wallace, Wallace County; and the North Fork of Solomon River at Lenora, Norton County. Cross & Moss (1987) attributed the loss of diversity evident in Table 3 to reduced seepage flow into the Smoky Hill and Solomon Rivers (see location map in Fig. 9), that in turn were caused by the falling GW lev-

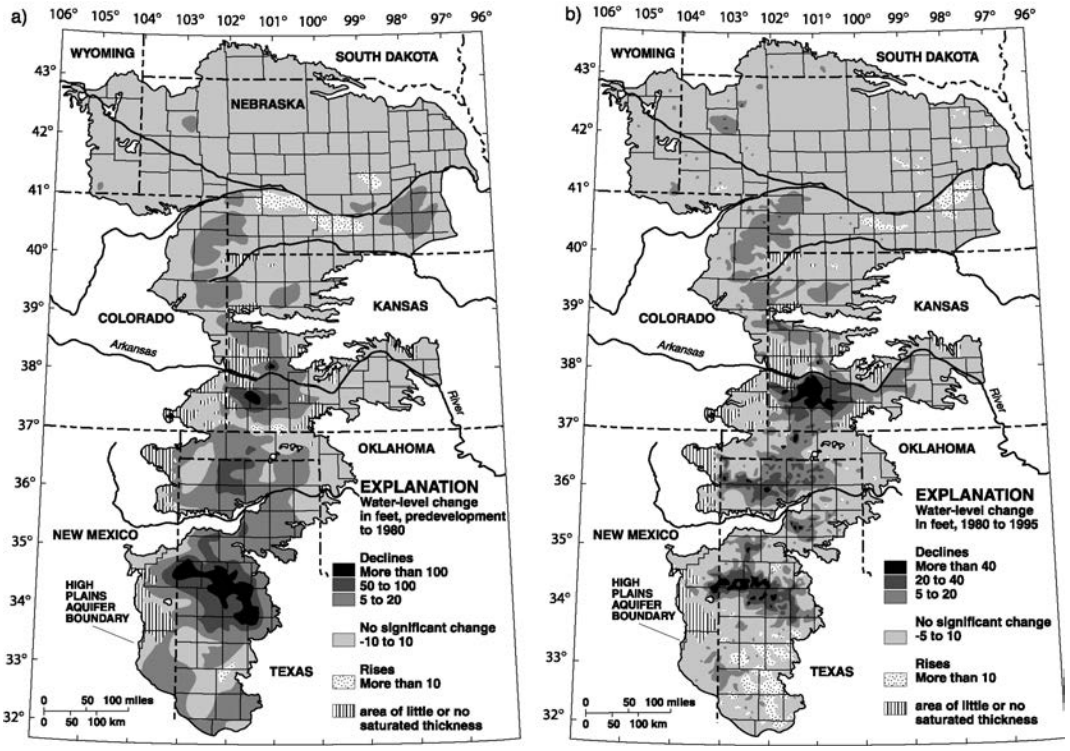


Figure 7. Water-level changes (in ft) in the High Plains aquifer: a) predevelopment to 1980; b) 1980-1985. To convert to meters multiply by 0.3048. Adapted from U.S. Geological Survey. (http://www-nc.cr.usgs.gov/highplains/hpactivities.html)

els due to intensive GW use associated with agricultural land use. Such use explains the gradual desiccation of streams in Western Kansas, whereas precipitation patterns do not account for it (Cross & Moss 1987). As a result of these GW-level declines, streamflows in Western and Central Kansas streams have been decreasing, especially since the 1970s, as mentioned previously.

In addition to stream-aquifer interactions related to water quantity, organic and toxic contamination in SW or GW can also be transferred to the GW in influent stream reaches or to SW, respectively.

The quality of the downwelling SW is normally altered during its passage through the first few meters of the infiltrated sediments. However, this may not be the case for persistent organic compounds, such as chloroform and inorganic pollutants, which may contaminate extensive areas of GW (Schwarzenbach *et al.* 1983, Santschi *et al.* 1987).

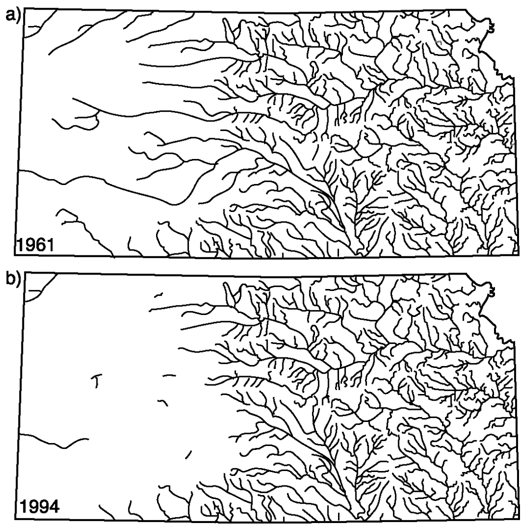


Figure 8. Major perennial streams in Kansas, 1961 (a) *versus* 1994 (b). (Adapted from Angelo 1994).

Table 3. Fishes Reported in 1885 from the Smoky Hill River at Wallace, Kansas, and/or North Fork of the Solomon River at Lenora, Kansas, by Hay (1887). Species are grouped according to the sequence of their extirpation, on the basis of subsequent collections from the sites. (From Cross & Moss 1987).

| | |
|---|---|
| *Phoxinus erythrogaster*<br>*Nocomis biguttatus*<br>*Notropis heterolepis*<br>*N. topeka*<br>*N. umbratilis*<br>*Etheostoma nigrum* | Not subsequently captured: disappeared before 1935. |
| *Notropis cornutus*<br>*Hybognathus hankinsoni*<br>*Pimephales notatus*<br>*Catostomus commersoni* | Last captured in 1950-58: disappeared before 1961. |
| *Phenacobius mirabilis*<br>*Hybognathus placitus*<br>*Ictalurus melas*<br>*Noturus flavus*<br>*Lepomis cyanellus*<br>*L. humilis* | Last captured in 1961-74: disappeared before 1978. |
| *Semotilus atromaculatus*<br>*Notropis lutrensis*[a]<br>*N. stramineus*[a]<br>*Pimephales promelas*<br>*Campostoma anomalum*<br>*Fundulus zebrinus*<br>*Etheostoma spectabile* | Recorded at Lenora in 1978-85. |

[a]Not found in 1985.

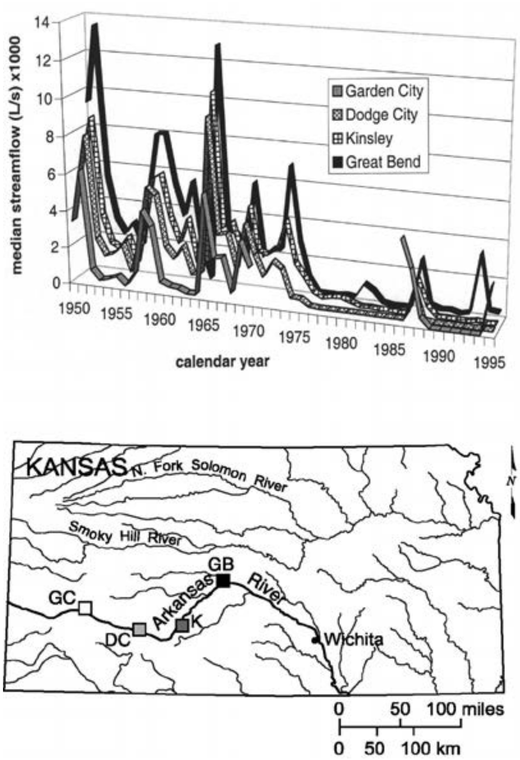


Figure 9. Median annual discharge of the Arkansas River based on daily streamflow records from 1950 to 1995 at Garden City (GC), Dodge City (DC), Kinsley (K), and Great Bend (GB) stream gauging stations, and location map. The Garden City station daily record from 1971 to 1985 is nonexistent (from Sophocleous 2000a, b).

For example, one of the most saline rivers in the USA is the Arkansas River in Southeast Colorado and Southwest Kansas. The dissolved constituents (mainly sulfate, sodium, bicarbonate, and calcium) are considered to originate from soils and bedrock in Colorado. The dissolved salt concentration in the river water greatly increases across Eastern Colorado as evapotranspiration from ditch diversion for irrigation and storage systems consumes water, while the dissolved salts remain in the residual water. The dissolved solids content of the river water in the year 2000 averaged over 3,000 mg/L at the Kansas-Colorado state line (Whittemore *et al*. 2000). The discharge into Kansas is saline during both high and low flow periods, although the salinity decreases with increasing flow. The major constituent, sulfate, reaches a maximum concentration of about 2,600 mg/L in low flows, suggesting limitation by gypsum precipitation (Whittemore *et al*. 2000). The river salinity has generally increased at the state line during the last few decades. Shallow, saline groundwater in the alluvium of the Arkansas River and under fields irrigated with river water has penetrated to various depths in the underlying High Plains aquifer in Southwest Kansas as shown in Figure 10 (Whittemore *et al*. 2000). Much of the saline water migration deep into the freshwater aquifer has occurred since the 1970s. Due to intensive GW use in the past three decades, the water level in the main aquifer declined below the alluvium and shallow parts of the High Plains aquifer, changing the average river flow condition from one of GW feeding the river to one in which river water recharges the aquifer. During this time, when the rate of GW use peaked, essentially all of the salt mass has remained in Southwest Kansas because there has often been little or no river flow exiting the area. Based on recent conditions and existing contamination, Whittemore *et al*. (2000) estimated that in about 50 years, river water seepage has the potential to contaminate all of the High Plains aquifer underlying

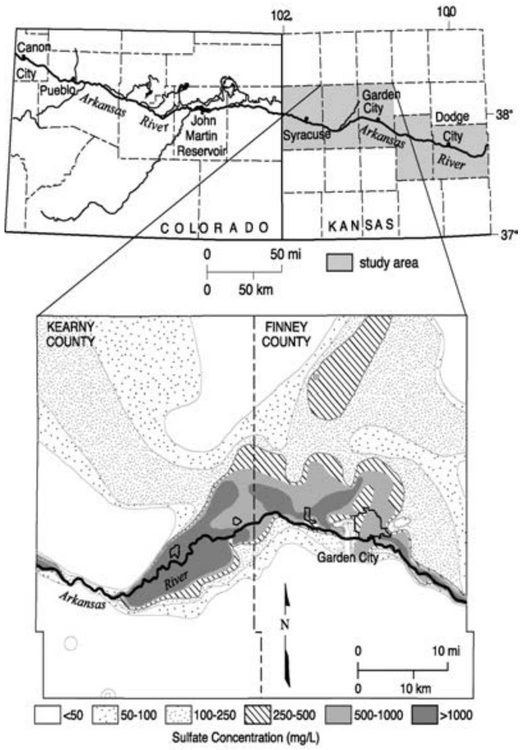


Figure 10. Natural drainage and major canals in the Arkansas River system in Southeast Colorado and Southwest Kansas. Sulfate concentration in groundwater of the Arkansas River corridor in Kearny and Finney Counties, Kansas. The data used are for wells >18 m deep for 1986-1995 (adapted from Whittemore *et al*. 2000).

1,300 km$^2$ of the corridor to a sulfate concentration more than 1,000 mg/L (i.e. four times the level recommended for drinking water). Therefore, management and protection of fresh GW in the region will be critical for maintaining water quality for municipal, agricultural, and industrial uses.

As previously mentioned, desiccation of floodplains due to GW-level declines endangers aquatic and riparian vegetation, reduces the connectivity and spatio-temporal heterogeneity of former channels, and ultimately alters biodiversity patterns (Dister *et al*. 1990, Allan & Flecker 1993, Bornette & Heiler 1994). The vegetation contributes to the resisting forces by stabilizing the bank material with roots and decreasing the velocity of floodwaters. Thus, riparian vegetation that has been impacted by a lowered water table enhances the danger of stream-bank erosion during flooding (Keller & Kondolf 1990). Changes from perennial to intermittent flow may alter bank vegetation and moisture content, and hence fluvial geomorphology (Keller & Kondolf 1990).

### 4.2 *Wetlands*

Wetlands are a key component of freshwater ecosystems worldwide (Revenga *et al*. 2000). They include highly productive habitat types, ranging from flooded forests and floodplains to shallow lakes and marshes. Wetlands provide a wide array of benefits, including flood control, nutrient cycling and retention, carbon storage, water filtering, water storage and aquifer recharge, shoreline protection and erosion control, as well as food and material products, such as fish, shellfish, timber, and fiber. Wetlands also provide habitat for a large number of species, from waterfowl and fish to invertebrates and plants. In North America, for instance, 39% of plant species depend on wetlands (Myers 1997). Wetlands also have aesthetic and recreational values, which are harder to quantify, such as birdwatching, hiking, fishing, and hunting.

Not only are wetlands highly productive and biologically rich, but much of the world's population lives in or near floodplain areas, where the soils are rich in nutrients and, therefore, very fertile. As a result of their potential as agricultural land (and also because they are feared as places that harbor disease), wetlands have undergone massive conversion around the world (Revenga *et al*. 2000). Sometimes, this has come with considerable ecological and socioeconomic costs.

Historically, wetlands have been viewed as a resource to be converted to more *productive* uses. In the USA, for example, the Federal Agricultural Stabilization and Conservation Services promoted drainage of wetlands through cost-sharing programs with farmers as recently as the 1970s (Gleick *et al*. 1995). Failure to quantify the real value of these natural resources resulted in significant losses. Myers (1997) estimated that half of the wetlands of the world were lost in the 20$^{th}$ century. More than half of USA wetlands have been lost, with an average annual loss of about 185,350 ha from the mid-1950s to the mid-1970s, 117,360 ha from 1974 to 1983, and 48,560 ha/yr from 1982 to 1991 (GAO 1993, cited in Gleick *et al*. 1995).

In California, where approximately 95% of its wetlands have been lost, conditions are even worse. That state has also lost more than 90% of its riparian forests in the Central Valley, 80% of its salmon and steelhead population since the 1950s, and 95% of the anandromous fish-spawning habitat in the Central Valley (Gleick *et al*. 1995). No rivers are untouched by dams, reservoirs, or major water withdrawals for human use, including those that now have protection under federal and state law (California State Lands Commission 1993, cited by Gleick *et al*. 1995). According to the California State Lands Commission (1993), over two-thirds of the 116 native California fish populations have declined sufficiently to raise concerns. California has lost at least 21 naturally spawning Pacific salmonid stocks, and an additional 39 are threatened. This decline is indicative of serious habitat degradation, as summarized in Table 4.

Table 4. Changes in aquatic and other ecosystems in California (adapted from Gleick *et al*. 1995).

| | Pre-settlement estimates | Current estimates | Percentage lost |
|---|---|---|---|
| Wetlands area in the Central Valley (ha)[a] | > 1.6 million | < 121,500 | 95% |
| Salmon and steel head population[b] | N/A | N/A | 80% |
| Sacramento/San Joaquin salmon population[b] | 600,000 | 272,000 | 55% |
| Anadromous fish spawning habitat along rivers and streams in the Central Valley ($km^2$)[b] | 9,660 | 480 | 95% |
| Riparian forest area in the Central Valley (ha)[b] | 373,000 | 41,000 | 89% |

[a] Of the remaining wetlands, 30% are within the boundaries of National Wildlife Refuges and State Wildlife Areas, and 70% are privately owned and managed. Nationally, 75% of the remaining wetlands are privately owned.
[b] Of the approximately 41,280 ha of riparian forest that remain, about half are in a highly degraded condition. The problem may be even worse, as reflected by the results when one uses the higher original riparian forest area estimate of 650,000 ha (which means that we have lost approximately 94%).
N/A = not available

According to Gleick *et al*. (1995), until recently, only a small portion of the water used by fish, wetlands, migrating birds, and other components of the environment was explicitly included in state water management plans. Instead, water for human uses was identified and allocated, and whatever was *left* was implicitly assumed to be available for the environment. The result of this approach was that the environment over time received a smaller and smaller share of the state's limited water. As a result, Gleick *et al*. (1995) concluded that the severe impacts of water shortages on California's natural ecosystems in the last several years are the direct result of these policies.

In addition to those in California (as well as other states in the USA), wetland and riparian ecosystems throughout the world have been altered or destroyed because of GW depletion and stream dewatering. In Europe, wetland loss is severe (Revenga *et al*. 2000). Estimates show, for example, that Spain has lost more than 60% of all inland freshwater wetlands since 1970 (EEA 1995); Lithuania has lost 70% of its wetlands in the last 30 years (EEA 1999); and the open plains of Southwestern Sweden have lost 67% of their wetlands and ponds to drainage in the last 50 years (EEA 1995). Overall, drainage and conversion to agriculture alone has reduced wetlands area in Europe by some 60% (EEA 1998).

Despite the previously cited statistics on wetlands, however, wetland loss data for many regions of the world, as well as data on more gradual modifications of the hydrological regime of wetlands, are hard to obtain (Revenga *et al*. 2000). For example, because wetland ecosystems depend on a shallow water table and are generally very sensitive to changes in water levels, GW overabstraction can cause substantial damage to wetlands by causing the underlying water table to drop. If water withdrawals are large enough, wetlands can be permanently destroyed (EEA 1995). In 1995, the European Environment Agency estimated that 25% of the most important wetlands in Europe were threatened by GW overexploitation (EEA 1995).

Dewatering continues to threaten riparian ecosystems throughout the world, in part because of the paucity of riparian protection measures (Stromberg & Tiller 1996). For example, although some states in the USA regulate GW extraction, the regulations typically do not provide for riparian protection (Lamb & Lord

1992). Instream flow rights are granted for fish and wildlife habitat in most USA western states, but these are often junior rights (Stromberg & Tiller 1996).

## 5 EPILOGUE

Growing human population, expanding urbanization and industrialization, and increasing demands for food production are placing more pressure on the world's GW supplies. At the same time that water demand is increasing, pollution from industry, urban centers, agriculture, mining, and other sources is limiting the amount of water available for domestic use and food production. In addition to water pollution, habitat degradation and loss, physical alteration, overexploitation, and the introduction of nonnative species have taken a toll on freshwater biodiversity.

Because of the interdependence of SW, GW, and water-reliant ecosystems, changes in any part of the system have consequences for the other parts, and therefore the GW system cannot be managed by itself in isolation of the rest of the environment. For example, what may be established as an acceptable rate of GW withdrawal with respect to changes in GW levels, may reduce the availability of SW to an unacceptable level. In addition, many of the effects of GW development manifest themselves slowly over time, so that pumping decisions today may affect SW availability many years in the future. Consequently, a comprehensive, long-term, and integrated approach to the management of GW resources is required if the water quality and supply are to be sustained in the longer term, and other ecosystems dependent on water are to be protected.

Groundwater management responses in areas of over-extraction must include bringing use back to sustainable or at least community-acceptable levels while exploring more sustainable options. In other areas, groundwater management needs to adapt to working within the finite limits set by the goal of sustainability and rooted on hydrologic principles of mass balance. Wise management of water resources needs to be approached not only from the viewpoint of focusing on the volume of water available for sustainable use, but also from the impact of groundwater exploitation on the natural environment.

Economic development and human well-being will depend in large part on our ability to manage ecosystems more sustainably. To achieve a secure future, we need to recognize the finite limits of water availability, and bring human numbers and demands in line with them. As the President of the World Resources Institute aptly noted in his Foreword to the Pilot Analysis of Global Ecosystems: Freshwater Systems (Revenga *et al*. 2000):

"…Human dominance of the earth's productive systems gives us enormous responsibilities, but great opportunities as well. The challenge for the 21st century is to understand the vulnerabilities and resilience of ecosystems, so that we can find ways to reconcile the demands of human development with the tolerance of nature..."

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