LAND USE AND HYDROLOGY

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1 Introduction

Water quality and quantity are widely recognized as important environmental resources for society (Arnell et al., 2001). Quantity refers to the presence of a sufficient supply of fresh water to support the human and natural systems dependent on it, while quality refers to the suitability of the supply for its intended use (e.g., agricultural, domestic, industrial, or natural). Water is a dynamic substance, however, and the water cycle is a series of fluxes between reservoirs of varying size, residence time, and state. This interconnected property means that there are important consequences of the life history of water on a landscape, from its first appearance in precipitation to its exit to the ocean. The water carries a signature of its history (i.e., nutrient and pollutant loads) that has impacts and consequences on the reservoirs through which it passes.

Climate change will almost certainly impact the water cycle. These effects have been studied in some detail, although largely at global/regional scales; details are presented elsewhere in this book (Bonan et al., Chapt. 17). For example, precipitation has increased 0.5-1% per decade in the 20th century in the mid-high latitudes of the northern hemisphere, with a greater frequency of heavy precipitation events, and these trends are likely to continue (IPCC Reports, 2001). This establishes the essential parameters of the water fluxes, but how that water is transported, allocated, and modified during residency on the land is an important research area for Land Cover and Land Use Change (LCLUC).

Land cover and land use are important determinates of the water supply on its transit through a landscape. Climate broadly establishes the upstream supply term of the water budget, and the effects of land cover and land use on water quality and quantity at the local level have been well established through numerous studies. For example, it is well documented that deforestation increases stream flow through decreased evapotranspiration (Bosch and Hewlett, 1982). Urban- and suburbanization also increase stream flow through increased runoff, but also decrease water quality when the amount of impervious surface in a watershed exceeds 10-15% of the total land cover (Schueuer, 1994). The demand for water in all regions, but particularly in arid and semi-arid environments, for domestic, industrial, and agricultural uses has led to water engineering projects on all scales (Rosenberg et al., 2000). The net result is that demand
often exceeds supply in these systems, resulting in some of the 20th century’s largest land transformations with consequent impacts on aquatic systems.

The case of the Aral Sea is one of the largest and well-known examples of the effects of water diversion (Micklin, 1994; Tanton and Heaven, 1999). Beginning in the 1960s, water from the two major rivers entering this inland sea (Amu Darya and Syr Darya) were diverted through engineering projects primarily for irrigation but also for storage, power generation, and flood control. This led to an enormous decrease in the area and volume of the Aral Sea over the next 35 years (Figure 1) from $6.7 \times 10^4$ km$^2$ to $3.2 \times 10^3$ km$^2$ and 1064 km$^3$ to 310 km$^3$ respectively (Saiko and Zonn, 2000). Most studies anticipate its complete disappearance in the next 25 years (e.g. Saiko and Zonn, 2000). Accompanying the shrinking sea is a long list of ecological and social impacts. For
example, local climate has been impacted as evidenced by changes in surface temperature (Small et al., 2001a), while the exposure of saline soils and desertification has been accompanied by environmental health impacts (Small et al., 2001b). The enormous expansion in irrigated agricultural production, primarily in cotton, during the 1960s and 1970s provided an economic benefit from the water engineering projects. However, agricultural production has been declining over the last two decades due to salinization and water logging resulting from poor irrigation practices, and it has been estimated that nearly half of all irrigated lands (8x10^6 km^2) in the Aral Sea basin are affected (Heaven et al., 2002).

While case studies illuminate some of the direct cause and effect relationships between LCLUC and water quality and quantity, these relationships become more difficult to assess and quantify on larger scales (regional to national to continental to global). Global assessments of water supply and consumption are useful in cataloging the major categories of water demand and in pointing to the rise in consumption against a fixed supply (L’vovich et al., 1990). Yet the distribution of water on the Earth’s surface is not uniform and neither is the distribution of population or demand. Furthermore water distribution and demand do not necessarily co-vary. Therefore, from a land cover and land use perspective there is an unequal distribution of a resource for which the demand varies greatly (Vörösmarty et al., 2000). As the regions grow larger, the effects and ability to quantify become more diffuse, limiting the usefulness of aggregated global analysis.

Our focus in this chapter is to illuminate some of the relationships between LCLUC processes and water quality and quantity on the scale relevant to local and regional issues. Those relationships that affect climate/weather are treated by Bonan et al (Chapt 17).

2 Water Quantity: Resource Allocation and Impacts

Water is a fundamental requirement of all living things, and thus water quantity has a direct impact on human and natural systems. Water vulnerability is typically cast as a human supply and demand problem, where climate and the landscape establish the supply and human systems make demands on the supply. Vulnerability is established when human demand exceeds some threshold in supply (Vörösmarty et al., 2000). However, natural systems are equally dependent on an adequate supply of water, and human systems have some dependence on natural systems for ecosystems goods and services (Costanza, 2001). In regions of the world where supply far exceeds demand, the main issues are on water engineering projects to control flooding or for power generation. The impacts from such projects primarily affect aquatic systems through dams that change the natural flow and channels of rivers (Rosenberg et al., 2000). In regions of the world where supply and demand volumes are more closely matched, the impacts are more diverse and can be more acute. This is a particularly persistent and growing problem in arid and semi-arid regions of the world, where terrestrial and aquatic biodiversity is concentrated along watercourses and in direct competition with the demands placed by the agricultural, industrial, and domestic sectors of society on the resource.
Given the scope of this topic, it is not possible to provide an exhaustive assessment of water quantity issues with respect to LCLUC. Rather, we will examine in some greater detail water allocation issues in arid and semi-arid regions to provide examples of the problems and relationships to LCLUC. Assessments of other processes have recently been covered elsewhere (e.g. Rosenberg et al., 2000). The irrigation of cropland is the primary application of water in arid and semi-arid regions, and this has resulted in an enormous benefit to food security. For example, it has been estimated that expansion of irrigated lands increased food production by 50% between 1960 and 1985. By some estimates, 40% of current global food production is generated on the 15-20% of agricultural land that is irrigated (e.g. Shiklomanov, 1997). Thus irrigated lands are 3 times as productive as non-irrigated cropland, but also provide a greater value, as the dollar value of this production is 6-7 times that of non-irrigated cropland and >30 times that of rangeland (Crosson, 1997).

Figure 2. Global irrigated land area from 1800 to the present (top) and over the last 50 years (bottom). The rate of increase over the last 50 years is $3.4 \times 10^5 \text{ km}^2/\text{yr}$. Data compiled from Brown (1985, 1994), Brown et al. (1997), Warne (1970), Eckholm (1976), Postal (1994; 1999).
2.1 IRRIGATION AND LAND USE/LAND COVER

Global land area currently classified as irrigated agriculture has been estimated to be 2.5 x 10^6 km^2. These are lands that were formerly covered by native ecosystems, used for grazing, or for dryland farming. As irrigated lands, they are now tilled, irrigated, and managed for agricultural production. The amount of irrigated land has increased dramatically over the last two centuries (Figure 2) and has more than doubled between 1950 and 2000. It is not clear from the available data whether the rate of increase has slowed over the last decade, as there is some uncertainty on the estimates of total land irrigated and there is annual variability on the specific land parcels irrigated. Nevertheless the data shown in Figure 2b exhibit an apparent linear increase in the global estimates of irrigated land of 3.4 x 10^7 km^2 y^{-1} over the last 50 years.

The specific parcels of land that are irrigated and managed for agriculture on an annual basis are not necessarily constant. Irrigated lands are abandoned due to myriad factors including loss of reliable water supply (e.g. diminished groundwater or reallocation of surface water) and degradation of the land (e.g. salinization and water logging). In the developed world the rate of abandonment is more or less balanced by the rate of development of new irrigated land resulting in a relatively stable total area of irrigated land over the last decade, while other parts of the world are experiencing explosive growth in irrigation, driven by huge government funded water engineering projects.

One area exhibiting expansion is along the Euphrates River in the Middle East (Syria, Turkey). Between 1990 and 2000, the amount of irrigated land increased 6 fold, from 175 km^2 to 1115 km^2, while the pattern of land use shifted where irrigated lands along the river bottoms and floodplains were abandoned for upland sites (Figure 3). The massive changes in water use, and attendant impacts on land cover, are being driven by a complex set of factors, including the governments’ desires to improve food security as well as to establish claims on the water resources (R. Smith, Yale University, pers. comm.).

2.2 IMPACTS OF RESOURCE ALLOCATION

The immediate environmental impacts of the reallocation of water resources (e.g. loss of wetlands, decline of aquatic species) have been well documented by a number of recent reviews (e.g. Lemly et al., 2000; Vörösmarty and Sahagian, 2000; Rosenberg et al., 2000). What is not as well understood or recognized are the impacts of shifting reallocations of water for irrigation or domestic/industrial use on native ecosystems. Particularly, what is the ability of native ecosystems to adapt and recover following abandonment of agricultural lands or through mitigation efforts to restore water to impacted systems? A series of studies by Elmore et al. (2003a; 2003b) provide important insights into this question.

Elmore et al. (2003a; 2003b) studied the behavior of semi-arid ecosystems in the Owens Valley of California. Owens Valley has a 130-year history of land use centered on the abundant fresh water resources in this semi-arid landscape. Agricultural activity (irrigated crops and pasture) peaked in the 1920s followed by large-scale abandonment due to a reallocation of the water resources, through interbasin transfer, for
Figure 3. Example of rapid change in the total area and location of irrigated land. The upper left image shows a Landsat TM false color scene of the region in 1990, the middle image shows the areas of irrigated agriculture, and the right image shows the areas of irrigated agriculture in 2000. Note the abandonment of agriculture along the river bottom and expansion on the surrounding upland regions. The graph shows the rapid expansion in irrigated land over the 1990s.

domestic/agricultural/industrial use by Los Angeles, CA. This water makes up a significant fraction of the fresh water budget for Los Angeles, and virtually all of the surface runoff has been exported from the valley since the 1920s. Following abandonment, much of the agricultural land was colonized by a mixture of perennial shrubs and annual grasses and plants.

The dramatic decrease in surface run-off during the drought that lasted from 1986 through the early 1990s prompted a response from resource managers to increase the amount of water extracted from groundwater reserves. At the height of the drought groundwater made up a significant fraction of the total water exported from Owens Valley. Elmore et al (2003a; 2003b) examined the response of the semi-arid ecosystems to these two forcing functions (drought, groundwater extraction) using a combination of remotely sensed and field data. They found that, not surprisingly, vegetation communities not dependent on groundwater (xeric) as well as those dependent on groundwater (phreatophytic) but where the groundwater levels remained within 3 meters of the surface were little affected by the drought. However, where groundwater dropped below 3.3 meters due to groundwater withdraw, phreatophytic meadow
communities displayed catastrophic decreases in plant cover, indicating that a threshold had been reached.

Following the return of average to above average precipitation during the mid to late 1990s, the communities most affected by the groundwater drawdown exhibited changes in plant species assemblages, with a shift towards invasive, non-native annuals and opportunistic shrubs. Furthermore, it could be clearly demonstrated that prior land use (agriculture) had a legacy that was evident 80 years after land abandonment. Previously cultivated lands showed a higher abundance of opportunistic shrubs and non-native annual plants, and lower species diversity than lands that were never cultivated. The vegetation on these lands also showed an amplified response in green activity to annual rainfall compared to the land dominated by native species. This response was detected with remotely sensed data where large increases in the amount of green cover during wet years were followed by large decreases in green cover during dry years. Thus the drought and the groundwater pumping changed the fundamental ecological functioning characteristics of these areas. It is unknown if the observed impacts represented a permanent shift as these studies only cover a twenty year period. However, ecological impacts of prior agricultural activity on lands abandoned for 80 years, as well as impacts from groundwater withdrawn in the 1970s, strongly suggest that these shifts are long-term if not permanent.

The impacts resulting from the abandonment of irrigated agricultural land have also been documented in regions where groundwater is far from the surface (Okin et al., 2001). In addition to changes in species composition similar to that documented in Owens Valley, there are significant impacts on soil properties, which might explain the persistence of the land-use legacy in Owens Valley. Soils destabilized by agriculture become susceptible to wind erosion following abandonment causing deflation in the former fields and deposition down wind. This results in changes in the natural processes of nutrient accumulation and availability through deflation and burial aggravating the impacts. Thus there are direct and indirect impacts of abandonment that persist over the long-term.

These example case studies provide detailed insight into the immediate impacts of water resource allocation and the legacy of land use. Other analyses not reported on here support these basic principles, but also extend the reach of our understanding into regions with diverse socio-economic drivers and land use histories. While the specific details in these region differ, the common threads are that in arid and semi-arid regions water vulnerability is high, agricultural abandonment is relatively common due to degradation and/or shifting priorities in allocation, and the margins for resilience (capacity to withstand climatic variability) are shrinking in the presence of high water demand.

2.3 REGIONAL TO GLOBAL IMPLICATIONS

The case studies described above provide concrete examples of the relationship between LCLUC and water resources. Recognizing that these examples are not a complete assessment of all possible drivers and impacts related to LCLUC and water quantity, it is apparent that a basic framework for assessing these problems is
understood. In humid and temperate regions where upland ecosystems as well as agriculture are sustained by rainfall, LCLUC impacts on hydrology are focused around drainage systems. Here direct impacts are on the riparian and aquatic systems through water engineering (Rosenberg et al., 2000). In regions where ecosystems are water limited and agriculture sustains a substantial benefit by irrigation, LCLUC impacts can extend many tens of kilometers from the actual water source and affect large land areas, in addition to the direct impacts on riparian and aquatic systems.

While the impacts of LCLUC on water quantity are directly measured and understood at the local scale, assessments of water vulnerability tend to be made at the regional, national, or global level (e.g., L’vovich and White, 1990). Such assessments are typically cast as coarse supply and demand problems, the analysis of which can be conveyed conveniently in simple graph or tabular formats. However, advances in technology and analytical capabilities are opening up new avenues to assess water vulnerability. Vörösmarty et al. (2000) presented the first geographically explicit global assessment of water vulnerability. Using a water balance model validated to a

1985 database of river discharge, they compute runoff at a 30’ (latitude by longitude) resolution. A spatially explicit population distribution was derived from several sources and matched to the discharge database. The last step was to then match the population/runoff calculations with spatially explicit water demand calculations derived from county-level water withdrawal statistics. They were thus able to produce a global map of population and water demand and apply a stress threshold (demand approaching or exceeding supply) to show a global distribution of water stress as a function of population (Figure 4). The resultant map shows regions of the world where water quantity is stressed due to human demands, and where LCLUC related to water quantity issues are likely to be pervasive. This map clearly highlights vulnerable regions in western North America, areas bordering the Sahara Desert, the Arabian Peninsula, and the several densely populated areas in Asia (Pakistan, India, and northeast China). These are not surprisingly also contemporary regions of important LCLUC impacts associated with water quantity.

3 Water Quality

3.1 SOME BASIC HYDROLOGY AND BIOGEOCHEMISTRY

Stream flows typically have two main components. There is a base flow resulting from previous rain water infiltrated into soils and groundwater as well as storm flow resulting from direct rainfall inputs to the stream surface and from overland flow during rain events. Base flow is the slow, but continuous discharge generated by groundwater inputs to streams between storm events, and storm flow is a rapidly changing component of discharge which creates hydrographs responding to rain events (Figure 5).

![Storm hydrograph Feb. 1984, Choptank River](image)

Figure 5. Example of a storm hydrograph observed at USGS gauging station 01491000 at Greensboro MD from a rain event centered on 15 Feb 1984 (water year day 137). Data from Fisher et al. (1998).

Water yields are the quantity of stream flow per unit watershed area per unit time. Units for water yields are all equivalent to depth time\(^{-1}\), which is equivalent to volume discharged per unit time per unit watershed area (cm\(^3\) water cm\(^{-2}\) watershed area y\(^{-1}\) or cm y\(^{-1}\)). Water yields may also be expressed as ft\(^3\) s\(^{-1}\) mile\(^{-2}\) or inches y\(^{-1}\) (reported in USGS Water Resources Reports). Water yields reported as depth time\(^{-1}\) (cm y\(^{-1}\) or inches y\(^{-1}\)) are immediately comparable to rain inputs (also in cm y\(^{-1}\) or inches y\(^{-1}\)) and lead to a simple evaluation of the fraction of rainfall that is discharged from a watershed as surface water discharge. In mesic areas with precipitation of ~1 m y\(^{-1}\) (e.g., much of the US east of the Mississippi River), water yields are ~40 cm y\(^{-1}\) or about 40% of rainfall. This ratio varies from 0.3 to 0.6, increasing with latitude as mean annual temperature decreases (Van Breeman et al. 2002). The remainder of the rain water not exiting a basin as stream flow is returned to the atmosphere by (1) evaporation (a physical process), (2) plant transpiration (a biological process involving
water vapor loss during photosynthesis), or (3) deep seepage losses from the watershed to regional groundwater (a geological process, usually small compared to evaporation and transpiration). Evaporation + transpiration, collectively referred to as “evapotranspiration” or “ET”, are often lumped as one vapor loss term with the same units as water yield.

Elemental export from a watershed is estimated as stream flow x concentrations. Typically, continuous or near-continuous stream flow data are available, with a limited number of observed concentrations. Flow-weighting or other estimators are used to combine these data to calculate export at a time scale of months or years (e.g., export = flow \( \text{m}^3 \text{y}^{-1} \) * average concentration \( \text{g m}^{-3} \) = \( \text{g y}^{-1} \)). As for discharge, export may also be normalized to basin area. Termed “export coefficients”, the SI units are usually kg ha\(^{-1}\) y\(^{-1}\) or kg km\(^{-2}\) y\(^{-1}\); also used are lbs acre\(^{-1}\) y\(^{-1}\) (equivalent to kg ha\(^{-1}\) y\(^{-1}\)). Export coefficients are essentially areal fluxes of materials exported from a watershed, and can easily be compared with watershed inputs such as fertilizer application rates (e.g., lbs acre\(^{-1}\) y\(^{-1}\) or kg ha\(^{-1}\) y\(^{-1}\)). Thus, as for water yields, elemental or other export coefficients can also be expressed as a fraction of inputs. Export coefficients are also useful as a modeling tool because they are usually strongly influenced by land cover and land use, as described below.

3.2 EFFECTS OF LAND USE ON WATER YIELDS

Water yields are strongly influenced by land use and land cover. Due to higher plant biomass, forests usually have much higher rates of evapotranspiration than agricultural or urban land uses, leaving less water available for groundwater or overland flows to streams. As a result, conversion of forest land cover to anthropogenic land uses in mesic areas of the temperate zone usually results in increased water yields (Figure 6). These observations are based on forest logging in mesic to wet areas (Bosch and Hewlett 1982), and there was no significant difference between removal of conifer and hardwood forests. Water yields increased by an average of 20 cm y\(^{-1}\) in watersheds with complete forest removal. Considering that watersheds with mixed land cover currently have water yields of ~40 cm y\(^{-1}\) (e.g., Van Breemen et al. 2002), removal of forest land cover clearly has significant impacts on water yields.

Urban land cover also influences water yields. Impervious surfaces such as roads, roofs, and parking lots cause storm responses to be faster, with higher discharges and power for bank erosion. Impervious surfaces directly shunt rainfall to overland flow and prevent infiltration to groundwater. The decreased groundwater inflows cause a rapid drop in flows after the end of the rain event, and the reduced abundance of plants results in much lower evapotranspiration. The net effect is that storm hydrographs have a larger volume of water over a shorter period of time in urbanized areas, with reduced base flows between storm events.

Forested land cover is the most retentive of water as well as particulates and dissolved materials. It is clear from the above examples that forested landscapes produce the most steady and smallest stream discharges of any land cover. Conversion of forest to anthropogenic land uses increases both the total flow (Figure 6) as well as the volume and erosive power of storm flows while decreasing base flows. Similarly, forested
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Landscapes also export the smallest amount of dissolved and particulate material, whereas anthropogenic land uses such as agriculture and urban areas export much higher amounts, as described below.

3.3 EFFECTS OF AGRICULTURE ON NUTRIENT YIELDS

The agricultural production of food for human populations is an essential need of modern societies. Within the last 100 years, two major changes in agriculture have occurred in North America and Europe: (1) since the beginning of the 20th century a largely agrarian population has gradually shifted to an urban base, with a large fraction of the population in urban areas dependent on a small number of agricultural workers; and (2) after World War II there were large increases in crop yields from smaller areas of land using commercial fertilizers containing primarily nitrogen (N), phosphorus (P), and potassium (K). These changes have concentrated the production of food into smaller areas than in the 19th century, reduced the number of people involved in food production, and greatly increased the intensity of agricultural land use.

One of the unintended consequences of the increased intensity of agricultural land use has been the contamination of shallow groundwater with nitrate (Figure 7). Inexpensive commercial fertilizers became available after World War II as munitions factories were converted to the production of agricultural N fertilizers, and application rates on agricultural lands increased exponentially for 30 years (Figure 7, lower panel). Sampling and dating of groundwater on the agriculturally dominated Delmarva Peninsula by Bohlke and Denver (1995) showed historical increases in groundwater
Nitrate that largely paralleled the fertilizer N application rates (Figure 7, upper panel). Nitrate concentrations of 10-20 mg NO$_3$-N L$^{-1}$ are frequently observed in most shallow aquifers in North America in agricultural areas (http://waterdata.usgs.gov/nwis/gw), and these waters are undrinkable due to the tendency of nitrate concentrations >10 mg NO$_3$-N L$^{-1}$ to cause methemoglobinemia in infants and to form carcinogenic nitrosamines in the human intestine (US EPA 1976, Heathwaite et al 1993).

Nitrate (NO$_3$) is almost never applied as N fertilizer. Urea or ammonia (NH$_3$) are more common, but bacterial processes in soils oxidize urea and NH$_3$ to produce highly soluble NO$_3$. This process, termed “nitrification”, is a natural heterotrophic response to excess N, which yields energy for the nitrifying organisms (e.g., Aber 1992). The result is that readily sorbed forms of N (urea, NH$_3$) are converted to highly soluble NO$_3$ by biological activities in soils. During late fall and early winter when hydrologic connections are reestablished after the seasonal lowering of groundwater in summer, rain water infiltrating to groundwater may carry high concentrations of nitrate (1.5-15 mM NO$_3$ or 20-200 mg NO$_3$-N L$^{-1}$) from the root zone of heavily fertilized agricultural areas into shallow aquifers (Staver and Brinsfield 1998). This transfer of nitrate from
the root zone to groundwater is a flushing of excess N, which can be greatly reduced by winter cover crops that maintain some plant N uptake during the cold season (Staver and Brinsfield 1998).

Over the last twenty years, fertilizer applications in North America appear to have largely stabilized (e.g., Figure 7). Furthermore, in the late 1990's concerns about nitrate in groundwater have prompted the US Dept. of Agriculture (USDA) to require nutrient management plans to reduce groundwater nitrate in agricultural areas, and the Conservation Reserve Enhancement Program (CREP) now provides funds to create riparian zones along streams to intercept nitrate in contaminated groundwater. However, the multi-year residence time of most groundwater (Focazio et al. 1998) means that a decade will probably be required for the effects of nutrient management to be readily observable.

Other land uses also influence groundwater nitrate (Figure 8). Although agricultural land use has the highest nitrate concentrations in groundwater, unsewered urban areas have similarly high NO₃, whereas forested areas have the lowest amounts of NO₃ due to lower inputs and uptake by plants. The production of human food (agriculture) and the disposal of human waste in septic systems are clearly the principal causes of elevated NO₃ in groundwater.

Due to the large differences in groundwater NO₃ under varying land uses (Figures 7-8), the NO₃ concentrations of streams reflects the spatially varying distributions of land uses in watersheds (Figure 9). These stream concentrations from the Choptank River basin on the Delmarva Peninsula are annual means of TN; NO₃ is the single largest component of the TN, represents ~70% in streams on the Delmarva Peninsula (Norton and Fisher 2000, Fisher et al. 1998). Note that the effect of decreasing forest cover and increasing agriculture is not linear, particularly at high % agriculture and low forest. As agriculture expands to >70% of land cover, normal landscape traps for NO₃ (e.g., wetlands, forested areas along streams) are converted to agriculture, and NO₃ concentrations in streams rise exponentially. The effect of agriculture on stream chemistry is not limited to the agriculturally dominated Delmarva Peninsula; in a watershed survey of the mid-Atlantic region, Jordan et al. (1997) showed relationships in coastal plain streams similar to those in Figure 9, with similar concentrations, although the data of Jordan et al. (1997) are <70 % agriculture and appear linear. Better drained piedmont streams have even higher concentrations of NO₃ for the same amount of agriculture (Jordan et al. 1997).
Despite the strong relationships described above, not all NO$_3$ in groundwater appears in baseflow. Using the average land use specific NO$_3$ concentrations in Figure 9, Lee et al. (2000) have estimated the land use adjusted NO$_3$ in surface groundwater of sub-basins of the Choptank River watershed and compared these to base flow nitrate of streams draining these sub-basins. In 34 sub-basins, baseflow nitrate was less than the estimated groundwater nitrate, and the fractional decrease in observed baseflow nitrate was a linear function of the amount of hydric soils in each sub-basin. These water-saturated soils have little air space, are often anoxic, and are probably sites of denitrification (conversion of NO$_3$ to N$_2$ gas) by soil bacteria using NO$_3$ as an alternate electron acceptor in the absence of O$_2$. Thus, there appears to be some natural attenuation of anthropogenic NO$_3$ by hydric soils in watersheds, and NO$_3$ export may be reduced by as much as 80% below the expected concentrations based on land use (Lee et al. 2000).

![Effect of Land Use in the Choptank River Basin](image)

Figure 9. The effect of varying land use (agriculture and forest) on stream nitrate concentrations in subbasins of the Choptank River watershed. Data of Norton and Fisher (2000).

Agricultural land use also influences the other two major components of commercial fertilizer. Potassium (K$^+$) is also highly soluble, and Driscoll and Whitall (unpub.) have reported a strong positive correlation between potassium concentrations in streams and % agricultural land use in New England basins (D. Whitall, pers. com.). Phosphorus, however, unlike the highly soluble NO$_3^-$ and K$^+$, accumulates in surface soils due to strong sorption reactions of H$_3$PO$_4^-$ with positively charged iron oxyhydroxide and aluminum oxide in soils, particularly when excess animal manures are applied (Sims and Wolf 1994). Although the sorbed P is not transported vertically downward to groundwater, erosion of the highly enriched surface layer of agricultural soils results in considerable horizontal P transport during storm events. NO$_3$ concentrations are often reduced during the storm flow because overland flows contain less NO$_3$ than the groundwater which contributes baseflow; however, the overland flows transport eroded soils with high P concentrations, increasing stream P dramatically during storms (e.g., Fisher et al. 1998). Thus N, P, and K are transported from watersheds in very different modes; N and K are continuously transported primarily in baseflows, whereas P transport is more episodic and related to storm events.
A review of studies of relatively small areas of nearly uniform land use (Beaulac and Reckhow 1982) has provided a synthesis of land use specific N and P yields. As for water yields, forests have the lowest export coefficients of N and P by any land cover, and anthropogenic land uses along an approximate gradient of increasing disturbance show increasing export coefficients for N and P over several orders of magnitude. Small, but severely disturbed areas such as animal feedlots have export coefficients ~100 X higher than other types of agricultural land use.

3.4 EFFECTS OF HUMAN POPULATIONS ON NUTRIENT EXPORT

Human populations also directly influence nutrient concentrations and export from basins. As shown above in Figure 8, septic systems in less densely populated areas oxidize organic N from human waste and produce NO$_3^-$, enriching local groundwater. Valiela et al. (1992) have shown that increasing housing density in unsewered areas on the sandy soils of Cape Cod results in increasing NO$_3^-$ in groundwater. In more densely populated areas served by sewage systems, N may be discharged in many forms, usually to surface waters (e.g., Ryding and Rast 1989, Fisher et al. 1998), and these discharges may represent a large fraction of the N and P inputs to aquatic systems (e.g., Lee et al. 2001). Efforts to reduce eutrophication of aquatic systems usually begin with diversion or upgrading of treatments in sewage systems to reduce N and P inputs (e.g., Ryding and Rast 1989). In large river basins, the integrated impacts of human populations (both from agricultural production of food and disposal of human wastes) result in strong positive correlations between human population density and river NO$_3^-$ concentrations and export (Peierls et al. 1991).

3.5 MODELING OF WATERSHED HYDROLOGY AND CHEMISTRY

The empirical studies described above provide a strong conceptual basis for hydrochemical modeling of watershed export. While it can be argued that models tell us only what we already know, it is also true that models can be used to conduct “experiments” (model scenarios) that are empirically impractical, but which incorporate empirically derived principles such as those summarized above. For example, a hydrochemical model which has been calibrated under current conditions may be used to simulate the effects of manipulations which could never be done in practice within a watershed; e.g., removal of human populations, changes in human waste disposal, conversion of all land uses to forest, urbanization, etc. While the results of such model scenarios are predictions based on inputs often well outside of the calibration conditions, the projected conditions may be useful to select the best of several possible options, as in EPA’s current TMDL process.

We have used the hydrochemical model Generalized Watershed Loading Functions (GWLF) to examine the effects of land use/cover change in the Choptank River basin. The model was carefully calibrated at a USGS gauging station over an 11 water year period with detailed stream chemistry data using many of the empirical principles described above (Lee et al. 2000), and at the annual time scale the model has estimated validation errors of 5-10% for export of water and N and ~35% for export of P (Lee et al. 2001). We have used this model to estimate the biogeochemical effects of land use changes in the Choptank basin over the last 150 years observed in historical maps,
aerial photographs, satellite imagery, and human population data from the US Bureau of the Census (Benitez 2002, Benitez and Fisher in revision). Using the estimated land cover and population changes, we have performed model manipulations to isolate the effects of fertilizer applications, human waste, and urbanization on watershed nutrient export. Below we describe the model results from a sub-basin within the Choptank, which has experienced fertilizer applications as well as urbanization and growth of human populations.

Tilghman Island is an island of 5.9 km\(^2\) connected by a bridge to a peninsula of land which lies between the main axis of Chesapeake Bay and the Choptank estuary, a tributary of Chesapeake Bay. In 1850 Tilghman Island was primarily forest and agricultural lands, with small amounts of wetlands and urban areas, and a human population density \(~0.05\) ha\(^{-1}\) (Figure 10). Between 1850 and 1980 there was a net loss of forest and increases in agriculture and urban areas as the population density increased to \(~0.5\) ha\(^{-1}\). During this period, application of fertilizers to agricultural lands increased by several orders of magnitude at a rate similar to other regions of the US (Goolsby et al. 1999). However, beginning in \(~1980\), the prime location of Tilghman Island on the Chesapeake encouraged the development of tourism, and urbanization and human population densities increased to \(~30\%\) and \(>1\) ha\(^{-1}\), respectively, primarily at the expense of agricultural land. Thus, this small region of the Choptank Basin has experienced both the conversion of the original forest cover to agriculture as well as significant urbanization.

Figure 10. Historical changes in land use, human populations, and fertilizer application rates at Tilghman Island estimated from historical maps, aerial photographs, satellite imagery, and census data (Benitez 2002, Benitez and Fisher in review).
We used the GWLF model to estimate the effects of these changes on export of water, N, and P from this basin. We used the observed land use/cover changes (Figure 10) as inputs to GWLF to estimate the overall effect of the land use changes, and then we used model scenarios in which we manipulated the history of Tilghman Island for comparison with the full model scenario to separate the individual effects of conversion to agriculture, application of fertilizers, and disposal of human wastes (Figure 11).

The first model scenario was the estimation of N and P export under forest prior to European colonization. Using the advantage of modeling, we removed the people, agriculture, and urban areas, and reforested the entire basin. The estimated export coefficients under these conditions were quite low, 0.7 kg N and 0.09 kg P ha\(^{-1}\) y\(^{-1}\), approximately 20 and 4 times lower than those currently estimated (18 kg N and 0.4 kg P ha\(^{-1}\) y\(^{-1}\)). Although this model scenario is farthest from the calibration conditions, these estimated increases in nutrient export indicate the magnitude of the anthropogenic effects on Tilghman Island and are consistent with currently observed exports of forests (Clarke et al. 2000). Other, less manipulative model scenarios were used to estimate the individual effects of conversion of forest to agriculture, application of fertilizers, and disposal of human wastes.

Figure 11. Changes in N and P export from Tilghman Island using the hydrochemical model GWLF.
Two effects of agriculture were examined. Prior to 1850, conversion of forest to agriculture was not accompanied by application of fertilizers, although manures, guano, and lime were increasingly employed later in the 19th century (Benitez and Fisher in review). Thus, the model output for 1850 conditions under low population density and ~60% agriculture is an estimate of the effect of the conversion of forest to a primarily agricultural landscape. Both N and P export appear to have increased ~2 fold from the all-forest scenario (green arrows in Figure 11) to ~1.2 kg N and 0.23 kg P ha\(^{-1}\) y\(^{-1}\). The subsequent application of fertilizers to agricultural lands in the second half of the 20th century had a much greater impact on N export yields than the initial conversion to agriculture, although this was not true for P export. Comparing the full model scenario with the one in which fertilizers were withheld, the model results suggest that application of agricultural N fertilizer was responsible for 30-70% of the historical increases in N yields. P export was little affected by withholding of fertilizers because storm flows do not erode large amounts of surface soils under the relatively low relief of Tilghman Island (maximum elevation 3 m asl). In areas with greater topographic relief, a larger effect of storm flows would be expected. Thus conversion of forest to agriculture in the 18th and 19th centuries appears to have increased N and P export coefficients by a factor of 2, but application of fertilizers in the 20th century resulted in a factor of ~5 increase in N export and an almost undetectable effect on P export.

The effect of waste disposal from human populations was estimated in another model scenario. Human populations were permitted to develop as observed in census data, but all waste discharges were set to zero. For simplicity in Figure 11, we show the combined effects of withheld fertilizer applications and human waste (lowest lines in each panel). For both N and P, increases in export yields were small compared to the 1850 conditions (<2 kg N and ~0.3 kg P ha\(^{-1}\) y\(^{-1}\)), indicating that most of the estimated increases in nutrient yields from Tilghman Island for the 1990's was due to application of fertilizers to agricultural lands and disposal of human wastes. The effect of the latter was estimated to be about a doubling of N export yields and an increase of ~30 % for P export yields.

4 Summary

The specific drivers and effects of land use and land cover change on water quantity and quality tend to be specific to each region. For example, water diverted from a river for domestic use may be returned to the same river through municipal effluent, used for irrigation and thus largely evaporated, or removed entirely from the system by interbasin transfer. Nevertheless, as we have outlined this chapter, there are commonalities relevant to the impacts of LCLUC on hydrology: land cover directly affects stream and river flow, and land use directly impacts water quality. In particular, we have identified the production of food in agricultural areas and the disposal of human waste from populated areas as the primary causes of degraded water quality. As a result, many regions of the world are reaching a state of vulnerability with respect to water quantity and quality, and this situation is likely to worsen over the next 25 years, particularly if human populations continue to increase.

5. References


