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Forests, Land Use Change, and Water

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Introduction

A forest is a biotic community predominated by trees and woody vegetation that are significantly taller, greater, thicker, and deeper than other vegetation types and generally covers a large area (Chang, 2003). Forests cover approximately 26.2% of the world, with 45.7% of Latin America and the Caribbean being covered, 35% of East Asia and the Pacific, and 35% of the European Union. Canada and the United States (U.S.) combined account only for 6.8% of the world's forests while Africa has even less 5.7% (Forest Types of the World, 2013). In the U.S., forests cover about one-third of its land (Sedell et al., 2000; Jones et al., 2009), totaling about 300 million ha (USDA, 2001). Forested areas in the temperate zone have not changed much in

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recent decades, but continuing deforestation of tropical forests, about half of world total, is of great concern (World Resources Institute, 1996).

Land use and forests are intricately linked to how and where people live and sustain themselves (GEF, 2012). The livelihood of more than one billion people depends on tropical forests (Lynch et al., 2013). Similarly, water is critical for human life, for many human activities as well as an environmental resource (EOS, 2012). Worldwide, early human society and culture is tied to trees, forests, and water. The connections between the loss of forests, land use, streamflow, and water quality have long been recognized (de la Cretaz and Barten, 2007). An example of prehistoric Athens, was used by these authors, where the originally wooded lands were left with bare dry soil just like skin and bone resulting in flooding and dried springs due to the cutting of these forests. As a result of large-scale expansion of croplands and pasturelands at the expense of forests and grasslands, cultivated cropland and pastureland have increased globally by 460% and 560%, respectively in the past 300 years (Scanlon et al., 2007).

The major causes of land use change (LUC) since 1750 has been deforestation of temperate regions for food production and industrialization. Modern LUCs from forests to a landscape with a mosaic of agricultural, forest and urban lands have resulted in new environmental issues including landslides, flooding, soil erosion, water quantity and quality degradation, salinization, desertification, and ecosystem service losses (Amatya et al., 2009; Sun and Lockaby, 2012).

Foresters started to work proactively to better understand the relationships of forests and water in the early 20th century (Chang, 2003; Andreassian, 2004). Water and forests are recognized as two important resources that provide vital habitat for wildlife, clean air and water (Brown et al., 2008; Jones et al., 2009; FAO, 2013), recreation, timber (Prestemon and Abt, 2002), and bioenergy (King et al., 2013). Most importantly, forests are large carbon sinks and play an increasingly important role in mitigating global climate change (Bonan, 2008; Dai et al., 2013). Forest and water are interdependent natural resources; the connection between water and forests is recognized with the birth of professional forestry in the U.S. (Ice and Stednick, 2004). As a result, forest experimental watershed studies that were designed to understand forest hydrologic processes and answer forest-water relations were initiated in federal lands in the beginning of 20th century with the first one as the Wagon Wheel Gap forest in Oregon (Bates, 1921) and forest hydrologic research continues till today to refine our understanding of the water cycle in forests (Chang, 2003; Jones et al., 2009; Vose et al., 2011). Land use has changed rapidly in several parts of the world in the last few decades as a part of the global change phenomena (De Fries and Eshelman, 2004; Scanlon et al., 2007; GEF, 2012), and especially true in the U.S. (Clifton et al., 2006; Hamilton et al., 2008; U.S. Forest Service, 2011; Sun and Lockaby, 2012). Land use change may occur due to change in vegetation such as deforestation, afforestation, urbanization, and other kinds of land development including mining and construction of highways. Accordingly, there have been increased concerns about the impacts of LUC on flooding, streamflow (yields), baseflow, and quality of waters draining from the uplands into downstream water bodies. Hydrologic impacts of forest conversions are critical to issues of contaminant dilution, aquatic habitat, and public water supply and use (Wilk et al., 2001; Tang et al., 2005; Thanapakpawin et al., 2006; Clifton et al., 2006; Skaggs et al., 2011; Price et al., 2011; Vose et al., 2012). The need to increase agricultural production to feed a growing world population leads to even more concerns about environmental impacts of converting forest and pasture lands to row crop agriculture (Skaggs et al., 2011). Emphasis on growing energy crops for biofuel production will potentially increase conversion of forests and other lands to intensively cultivated fields (King et al., 2013).

In its first comprehensive forecast on southern forests, the U.S. Forest Service (2011) stated that urbanization, bioenergy use, weather patterns, land ownership changes, and invasive species will significantly alter the South's forests between the years 2010 and 2060. The area of forest land is projected to decrease by about 9.3 million ha, mainly due to population growth and urbanization.

DeFries and Eshleman (2004) suggested a need for understanding the consequences of LUC for hydrologic processes, and integrating this understanding into the emerging focus on LUC science. Scanlon et al. (2007) provided a comprehensive review and summary on global impacts of conversions from natural to agricultural ecosystems on quantity and quality of water resources for both surface and groundwater, and addressed some of those consequences in water demand, supply, and water quality. There are several studies in the literature from around the world on impacts of forest clearing on downstream hydrology and water yield (Bosch and Hewlett, 1982; Andreassian, 2004; Brown et al., 2005). Most of these studies suggest that forest management practices such as harvesting, or the conversion of forests to agricultural or other uses increase in streamflows, water table levels, and increased groundwater recharge as a result of reduced evapotranspiration (ET) (Stednick, 1996; Sun et al., 2005; Amatya et al., 2006; Abdelnour et al., 2011; Skaggs et al., 2011; Webb et al., 2012; Tian et al., 2012). However, Farley et al. (2005) and Sun et al. (2006) noted that there is only a limited knowledge on a systematic analysis of the effects of afforestation (i.e., conversion of grass, shrub, or croplands to forests) on watershed hydrology.

The impact of LUCs on water resources also depends on many factors, including the original vegetation being replaced, the vegetation replacing it, the type of change, and associated land management and application practices (Scanlon et al., 2007), upon the dominant soil type where the LUC occurs. Local climate, extreme events, and soils are important factors to consider (Jayakaran et al., 2014; Boggs et al., 2012; Caldwell et al., 2012).

Efforts to determine the hydrologic impacts of LUC have been conducted on a wide range of scales using a relatively large range of methods (Skaggs et al., 2011). Methods vary from simply monitoring precipitation, streamflow, and other basic hydrologic variables like land use/land cover (LULC), elevation, slope, etc. of the watershed during and following land use conversion, to data intensive paired-watershed approaches, to the application of models ranging from simple regression methods to process-based integrated models. However, there is only limited synthesized information on these assessment methods including the change detection. Similarly, there are knowledge gaps in understanding the effects of various specific factors including the potential evapotranspiration (PET) that varies with reference vegetation (also called reference-ET (REF-ET)) and climate change and ultimately may affect the hydrology and water quality assessments for land use conversion. Land use and climate are two main factors directly influencing watershed hydrology, and separation of their effects is of great importance for land use planning and management (Li et al., 2009).

In this paper we start by giving a brief background on the status of forest hydrologic balance and then review the current literature on available methods, scaling issues, and detection limits used for evaluating the impacts of LUC. Furthermore, specifics on effects of LUC such as water use by forests and crops or ET, change in soil hydraulic properties after forest harvesting, artificial drainage, urbanization that alters land imperviousness, and climate change, including extreme events are also considered in this synthesis.

The specific objectives are to (1) synthesize information on monitoring and modeling approaches, change detection and statistical methods in various scales including remote sensing method; (2) synthesize information on hydrologic effects of various factors including land use conversion and climate change; and (3) provide recommendations on future research directions.

Forest Water Balance

Because forests make up a relatively large portion of many of our watersheds, it is important to understand their water balance components and their flow paths and distribution for both natural forests and silvicultural operations, while considering the contribution of other land uses. Main components of the forest hydrologic balance include precipitation as input and canopy interception, throughfall, stemflow, surface runoff, quick and interflows, transpiration, understory and soil/litter evaporation, deep seepage as outputs through various pathways (i.e., forest canopy, root system, litter and soil) and change in soil-water storage (Fig. 7.1). Evapotranspiration is the sum of water loss through the process of rainfall interception from the tree canopies, transpiration

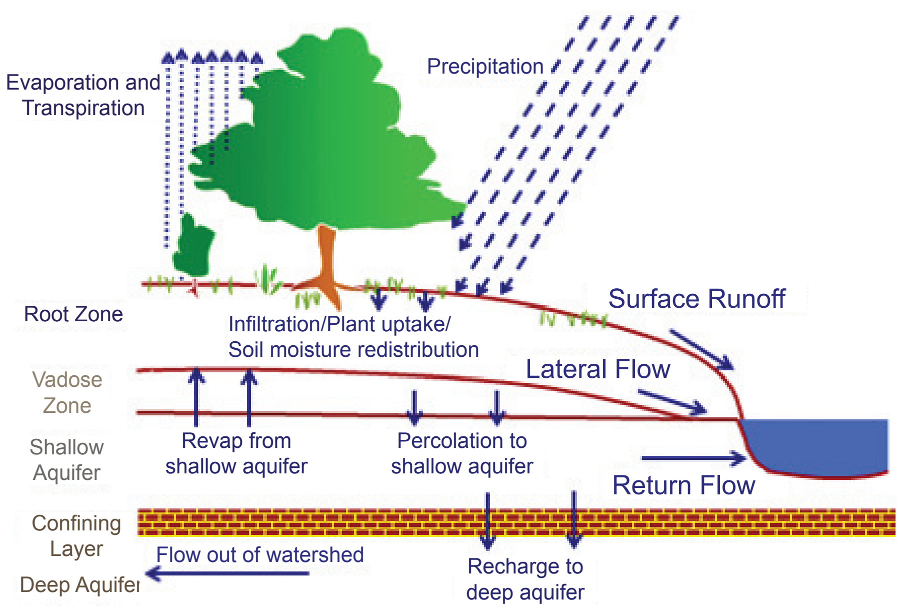

Figure 7.1. Schematic of forest hydrologic processes (Source: SWAT manual; Neitsch et al., 2005).

from foliage, and evaporation from forest floor. It is the key hydrologic flux that links water, energy, and biogeochemical cycles in forests (Sun et al., 2011a; 2011b).

Key forest hydrology questions identified by NRC (2008) for understanding basic processes and principles of water movement and predicting the general directions and magnitudes of hydrologic effects of anthropogenic and climate change were: (a) what are the flow paths and storage reservoirs of water in forests and forested watersheds? (b) how do modifications of forest vegetation influence water flowpaths and storage?, and (c) how do changes in forests affect water quantity and quality? de la Critaz and Barten (2007) provide a step-by-step approach on understanding hydrologic principles and processes mostly on forest landscape as a reference that govern the interactions between forest, water, and land use to experimental studies of varying scales and their management implications for the northeastern U.S. The authors also present the hydrologic and water quality principles to construct management plans for water supply watersheds on varying spatial and temporal scales.

The impact of changes in forest land use on its hydrology, in part, is reflected by the water balance components in equation 1:

$$\boldsymbol{P - E_i - E_{sl} - E_t - R_o - R_{gw} - DS = \Delta S} \quad (1)$$

Where, $\boldsymbol{P}$ = Precipitation, $\boldsymbol{E_i}$ = Evaporation from canopy interception, $\boldsymbol{E_{sl}}$ = Evaporation from soil and litter, $\boldsymbol{E_t}$ = transpiration from over and understory, R_o = surface runoff, R_{gw} = subsurface quick and return flow (baseflow), and DS = Deep seepage. ET, ET is the sum of $\boldsymbol{E_i + E_{sl} + E_t}$. The sum of the components of $\boldsymbol{R_o - R_{gw}}$ is also total streamflow. On a longer (> 1 year) term basis $\boldsymbol{\Delta S}$ can be assumed negligible.

Surface runoff seldom occurs in forests with large surface depressional storage (Amatya and Skaggs, 2011; Amatya et al., 1996), thick litter layer, and high soil infiltration rates, and thus streamflow is derived mainly from subsurface flow and groundwater. Generally, ET is a major loss of water in forest hydrologic water balance (Brauman et al., 2012; Sun et al., 2011a; 2011b; Amatya and Skaggs, 2011; Tian et al., 2012; Zhang et al., 2012). Therefore, ET is important for water resources management and development, stream ecology and fluvial geomorphology (Sun et al., 2005; Zhang et al., 2001). The basic seasonal and annual forest water balance can be dramatically shifted depending on climatic conditions, vegetation types and dynamics, and soil type.

In contrast to our knowledge about the effects of forests on peakflow or floods, impacts of forest conversion on baseflow (low flow) at the large watershed scale are not clear and have not been well documented in literature. Recently, in their study of effects of watershed land use and geomorphology on stream low flows during severe drought conditions in the Blue Ridge mountains of the U.S., Price et al. (2011) found a consistent, significant positive relationship of watershed forest cover with low flows, despite the higher ET rates associated with forests compared with other land covers and despite the relatively small range of disturbance in the study area. New activities and forest products often emerge in response to specific needs. Growth in biofuel demand could lead to increased removal of biomass from plantation forests, resulting in substantial hydrologic impacts on these lands primarily due to reduction in ET.

Methods Used for Impact Assessment

Monitoring approach

Methods of monitoring usually involve measurements of temporal variables such as precipitation, streamflow (surface runoff), and weather parameters. Spatially distributed variables from watershed characteristics include LULC, slope, elevation, altitude, and land and soil management. These variables are used for a single watershed before and after treatment or land use conversion for a single watershed to a paired watershed approach. For example, Silvera and Alonso (2008) compared flow events from the 2100 km^2 Manuel Diaz basin in Uruguay before and after afforestation (25% of the watershed area). These authors also estimated decreases in annual streamflow between 8.2% to 36.5% after afforestation. Although a long time series of streamflow data from a single watershed can be used with change detection methods, such methods can mask the effects of annual and seasonal climatic variability. On the other hand, a paired watershed approach assumes that there is a consistent, quantifiable, and predictable relationship between watershed response variables (Ssegane et al., 2013).

Paired watershed approach

A large number of small field-scale experimental studies using a paired-watershed approach have been conducted in Australia, New Zealand, South Africa, South America, Great Britain, China, Japan, and the US to better understand forest hydrologic processes, their interactions with the environment, and their ecohydrologic impacts (Swank and Douglas, 1974; Bosch and Hewlett, 1982; Amatya et al., 1996; Sahin and Hall, 1996; Fahey and Jackson, 1997; Sun et al., 2001; Worrall et al., 2003; Andreassian, 2004; Jackson et al., 2004; Brown et al., 2005; Elliott and Vose, 2005; Farley et al., 2005; Amatya et al., 2006; Edwards and Troendle, 2008; Chescheir et al., 2009; Ssegane et al., 2013; Bren and Lane, 2014; Bren and Mcguire, 2012). More than 90 years after the first paired watershed study at Wagon Wheel Gap, Colorado (Bates, 1921), forest hydrologists and natural resources managers are still working to understand the effects and the variances of forest management practices on hydrology and water quality (Zegre, 2008).

The highly variable nature of watershed responses to disturbance, by harvesting, fire, insect and disease damage, or species replacement, depends on many factors, such as watershed scale, climate, forest and vegetation types, density, geology, soils, topography, elevation, aspect, disturbance location, and type of disturbance, and vegetation. Decades of field and experimental research have been conducted to evaluate the effects of disturbance on many watershed attributes, and in response several methods have been developed and employed. The paired watershed approach offers the ability to identify roles of forest cover, internal watershed behavior, and climate variability to establish a “baseline” for reference (Zegre, 2008). This approach continues to be used on low-order watersheds as the primary method for impact assessments (Bren and Lane, 2014); its validity for predicting effects on large flooding events had been challenged (Alila et al., 2009). Andreassian (2004) presented a summary of paired watershed results to help understand contradictions of the past, as well as highlight unresolved issues in forest hydrology. Examples of the paired-watershed approach using intensively monitored, relatively small watersheds include: Swank and Crossley (1988), Amatya et al. (2006), and Boggs et al. (2012) who studied effects of harvesting (clearcutting) on North Carolina mountain, drained coastal plain, and the piedmont landscapes, respectively.

Long-term watershed studies that integrate forests, land use or land cover change, and water use in Africa include five paired watershed studies in South Africa (Van Wyk, 1987; Smith and Scott, 1992; Scott and Lesch, 1997; Scott and Smith, 1997; Scott et al., 1998; Jewitt, 2002) at experimental watersheds of Cathedral Peak (Kwazulu-Natal province), Mokobulaan (Mpumalanga province), Westfalia (Limpopo province), and Jonkershoek (Western Cape province). The watersheds were established to quantify the effects of afforestation on streamflow. The control watersheds included grasslands at Cathedral Peak and Mokobulaan, and native scrub forests at Westfalia and Jonkershoek. Treatments included afforestation with *Eucalyptus grandis* at Westfalia and Mokobulaan, *pinus patula* at Mokobulaan and Cathedral Peak, and *pinus radiata* at Jonkershoek. Reductions in streamflow due to afforestation were a function of forest type (eucalyptus or pine), location of the watershed (optimal or suboptimal growth zone) and the number of years after afforestation. Total streamflow reductions responded faster under eucalyptus (100% reduction within 8 to 9 years) than pine trees (80 to 90% reduction within 16 to 22 years) due to a faster growth rate of eucalyptus. Also, although afforestation covered only 1.2% of the land cover in South Africa, it contributed 3.2% reduction in the total annual streamflow and 7.8% reduction of the low-flows. The low-flows were defined as flows in the driest three months of an average year based on a period of 70 years.

Dagg and Blackie (1965) describe paired watershed studies in Kenya and Tanzania (East Africa). The watershed sites were located in sub-humid climatic region where less than 4% of areal land cover received more than 1250 mm annual rainfall. At the Kericho site in Kenya, the control watershed was under Montane forest while the treatment was planted with tea (54% of the watershed). At Kimakia (Kenya), the control watershed was under Bamboo forest while the treatment was under softwood plantations. At Mbeya Range (Tanzania), the control watershed was under evergreen forest while the treatment was under locally cultivated crops. According to Edwards et al. (1976), the long-term average (1958–1973) water use over the above study watersheds decreased by 8.9% at the treatment watersheds compared to the control watersheds. For example, at Kericho the water use, calculated as the difference between rainfall and streamflow, decreased by 14.4% during clearing and planting (1960–1963), increased by 2.4% during tea plantation establishment (1964–1967), and decreased by 12.1% between 1968 and 1973. Additional reports and studies on the above watersheds include works by Blackie (1972) and Edwards and Blackie (1981).

In a more recent 8-year study on the conversion of grasslands to managed pine forest on smaller paired watersheds in Uruguay, Chescheir et al. (2009) found no reduction in the third year to a 28% reduction in the fourth year since tree planting.

The year with the greatest yield reduction was characterized by a very dry period followed by a very wet one. The water yield reduction over the last three years of the study was 15%.

Flow Duration Curve (FDC) as a visual tool for change detection

A review of paired watershed studies demonstrates the relevance of the flow duration curve (FDC) as a graphical tool to detect impacts of LUC on different flow regimes (high-flows, medium-flows, and low-flows) (Best, 2003). Best (2003) examined three watersheds: Redhill (Australia; pasture to pine forest), Wights (Australia; native forest to pasture), and Glendhu (New Zealand; grassland to pine). To minimize the effects of climatic variability, daily FDCs with similar annual rainfall of about 880 mm over eight years were compared at the Redhill watershed. The two years with similar annual rainfall coincided with the first and eighth years after pine planting. Comparison of FDCs at one and eight years after pine planting showed a 50% reduction in high-flows and 100% reduction in low-flows. Also, the observed increases in streamflow magnitudes at Wights watershed due to LUC (forest to pasture) were comparable to observed reductions at Redhill for the respective flow-regimes (high and low-flows). However, the conversion of grassland to pine plantation at the Glendhu watershed on average reduced the different flow-regimes by 30%. However, Lane et al. (2003) highlight the need to improve the understanding of the impact of afforestation on the FDC. These authors found their flow reductions were in accordance with published results for paired watershed studies but with two different patterns (one with more zero flows and another with a uniform reduction across all percentiles) for 10 watersheds they studied. They also suggested the usefulness of their model in removing the effect of rainfall variability, thus making it applicable where paired watershed data are not available.

Uncertainty of calibration data may mask small treatment effects

Laurén et al. (2009) demonstrate how uncertainty in pre-treatment data of paired watershed studies may influence estimates of the magnitude and duration of treatment effects. The monitoring of phosphorous loads on two independent paired watersheds in Finland before and after clear-cutting demonstrated that small treatment effects may be masked by uncertainty of the pre-treatment data. Bonumá et al. (2013) state that their model simulations could not capture the runoff peaks well in the daily flow record possibly due to uncertainty in the modified CN2 method used to estimate surface runoff (Mishra and Singh, 2003). In the case where the time of concentration of the watershed is less than 1 day, the uncertainty in estimated surface runoff from daily rainfall is greater. Green et al. (2006) argue that as one value represents the range of rainfall intensities that can occur within a day, there can be a considerable uncertainty within that time period that are not captured.

The understanding of basic hydrologic processes and their interactions gained in paired watershed and other experimental studies has enabled the development of more reliable simulation models (Skaggs et al., 2011) that can capture the small treatment effects. Most recently, Andreassian et al. (2012) demonstrated how a classical hydrologic model and a paired watershed model can be associated to reach an unprecedented level of efficiency. The authors reported that such a combined method can be useful for hydrological applications including trend analysis (i.e., streamflow after LUC).

Alila et al. (2009) demonstrate how an inappropriate pairing of floods by meteorological input in analysis of covariance (ANCOVA) and analysis of variance (ANOVA) statistical tests used extensively for evaluating the effects of forest harvesting on floods smaller and larger than an average event, leads to incorrect estimates of changes in flood magnitude because neither the tests nor the pairing, account for changes in flood frequency. Similarly, Kuras et al. (2012) argued that contrary to the prevailing perception in forest hydrology, the effects of harvesting are found to increase with return period. This result is attributable to the uniqueness of peak flow runoff generation processes in snow-dominated watersheds.

Hydrologic modeling

Detecting hydrologic effects of land conversions using the 'paired watershed' approach can be time consuming, expensive and cost-prohibitive, and often limited by treatment options (i.e., watershed location and size and vegetation manipulation types) and understanding interaction of LUC and climatic variability. So, hydrological models have been widely used in such investigations and model simulation studies are frequently conducted to assess the impacts of LUC on large basins (Lorup et al., 1998; Wilk et al., 2001; Siriwardena et al., 2006; Gassman et al., 2007; Breuer et al., 2009; Simin et al., 2011).

Hydrologic models vary from lumped to physically-based, distributed watershed-scale for assessing the hydrologic impacts of LUC (Singh and Frevert, 2006). Breuer et al. (2009) examined a set of 10 lumped, semi-lumped and fully distributed hydrologic models that have been previously used in LUC studies in low mountain watersheds of Germany. The authors found a substantial difference in model performance that was attributed to model input data, calibration, and the physical basis of the models. The effect of the physical differences between models on the long-term water balance was mainly attributed to differences in how models represent ET. The authors concluded that there was no superior model if several measures of model performance were considered and that all models were suitable to participate in further multi-model ensemble set-ups and LUC scenario simulations. In a companion study with a scenario analysis, Huisman et al. (2009) reported that there was a 90% general agreement about the direction of changes in the mean annual discharge by the ensemble members.

Application of these models has been greatly enhanced by the development of GIS-based data sources for soils, stream locations and characteristics, and the type and distribution of vegetation via satellite data. A common approach is to use observed hydrologic data to calibrate a simulation model for the current land use, followed by prediction of outflow and other hydrologic variables for conditions after conversion. If a paired-watershed study is conducted, the model can be calibrated and tested for both land uses. This procedure was followed by von Stackelberg et al. (2007), who used the SWAT model to determine that afforestation of a pastured watershed in northern

Uruguay would decrease average annual outflow (yield) by 23%. A potential source of error in this approach is that model inputs, such as hydraulic properties of the soil, may be affected by the change in land use and not properly reflected in the predictions (Heuvelmans et al., 2004). This was found to be the case for hydraulic conductivity of the upper 90 cm of the soil profile as evidenced by Skaggs et al. (2006) and will be investigated herein.

Simulation results obtained by Kim et al. (2012) revealed that increased mean annual outflow was significant ($\alpha = 0.05$) for 100% conversion from forest (261 mm) to agricultural crop (326 mm), primarily attributed to a reduction in ET. While the high flow rates (> 5 mm day^{-1}) increased from 2.3% to 2.6% (downstream) and 2.6% to 4.2% (upstream) for 25% to 50% conversion, the frequency was higher for the upstream location compared to the downstream location. These results were attributed to a substantial decrease in soil hydraulic conductivity in one of the dominant soils in the upstream location that is expected after land use conversion to agriculture. As a result, predicted subsurface drainage decreased, and surface runoff increased as soil hydraulic conductivity decreased for the soil upstream. The results indicate that soil hydraulic properties resulting from land use conversion have a greater influence on hydrologic components than the location of land use conversion. Wilk et al. (2001) calibrated a rainfall-runoff model for the period prior to conversion of the 12,200 km^2 Nam Pong watershed in Thailand where the forest area was reduced from 80% to 27% of the watershed. The calibrated model was used to predict outflows after the land use conversion. Siriwardena et al. (2006) used both forward and reverse modeling strategies to analyze impacts of clearing the natural forest cover on a 16,400 km^2 watershed in Queensland, Australia, over a relatively short period of time in the mid-1960s. Application of the calibrated models led to the conclusion that the clearing resulted in increased outflow by approximately 40%.

Chappell and Tych (2012) use dynamic harmonic regression (DHR) modeling of time series (i.e., daily streamflow) to separate step changes in forested watershed hydrology due to LUC from changes due to climatic cycles and shifts. The DHR defines a low frequency component to model trend, a periodic component to model seasonal variability, and a zero mean observation error component with a constant variance. The authors note that the disadvantage of the approach is such that hydrologic shifts due to changes in LULC may be masked by errors in observed data, seasonal and inter-annual cycles in the climatic data, and a slower rate of LUC (i.e., 20% clear-cutting versus 100% clear-cutting).

Simin et al. (2012) applied a Xinanjiang model-based change detection approach on a large 1,640 km^2 watershed in China. The authors reported that the runoff has declined by nearly 25% from 1976 to 2005 attributable to a decrease in medium- to high-coverage natural forest for expansion of tea gardens and human development.

Zégre et al. (2010) use the HBV-EC hydrologic model by Hamilton et al. (2000), generalized likelihood uncertainty estimation (GLUE), and generalized least squares (GLS) regression analysis to isolate effects of forest harvesting from variations in rainfall and streamflow as an alternative approach to the paired watershed approach. The latter approach is susceptible to erroneous change detection due to variability between the paired watersheds. The HBV-EC model is used as a virtual control in place of the control watershed. The model is calibrated using pre-treatment data and, subsequently, the calibrated model is used to simulate hydrologic variables during the post-treatment period. GLUE was used to address the uncertainty of simulated streamflow due to accrued errors in model structure, model identification, and input data. The GLS was used to detect change because it accounts for auto-correlation of the daily time series.

Using comprehensive global sensitivity analysis for DRAINMOD-FOREST, an integrated model for simulating water, carbon (C), nitrogen (N) cycling, and plant growth with the 21-year of data, Tian et al. (2013) demonstrated a need for of incorporating a dynamic plant growth model for simulating hydrological and biogeochemical processes in forest ecosystems, that would have ultimate implications on applying the model in assessing impacts of conversions of forests into other land uses.

In their long-term (1973–2000) SWAT simulation on a large degrading watershed in Kenya, Odira et al. (2010) found that with the expansion of the area under agriculture, the streamflow increased during the rainy seasons and reduced during the dry seasons, whereas when the area under forest cover was increased the peak streamflow reduced. When the forest cover is reduced to almost zero there was an increased peak and mean streamflow in the basin.

Using a validated water balance model, Kuchment et al. (2011) reported 30% larger snowmelt rates and 10 days on average longer duration of snowmelt after forest cutting in the northwestern part of Russia. Although the spring flood peaks were 50% lower and started 5–7 days later in the forested basin than the clearcut one, the simulated annual runoff appeared to be about 10% higher than the one with forest cutting as a result of snowmelt effects.

Continental-scale modeling allows for the examination of the spatial and temporal variability of hydrological response to LUC due to urbanization. Sun et al. (2011b) developed a monthly scale water balance model, called Water Supply Stress Index (WaSSI) model, by incorporating remote sensing data and a set of empirical ET models derived from global eddy flux measurements. Based on the WaSSI modeling, Caldwell et al. (2012) found that impervious cover increases total water yield when compared to native vegetation, and that the increase was most significant during the growing season in general (Fig. 7.2).

The proportion of streamflow that occurred as baseflow decreased somewhat, even though total water yield increased as a result of impervious cover. Water yield was most sensitive to changes in impervious cover in areas where annual ET is high relative to precipitation (i.e., the southwestern states: Texas and Florida). Water yield was less sensitive in areas with low ET relative to precipitation (i.e., Pacific Northwest and Northeastern States). Additionally, water yield was most impacted when high ET land cover types (i.e., forests) were converted to impervious cover than when lower ET land cover types (i.e., grassland) were converted. Using projections of future impervious cover provided by the U.S. EPA Integrated Climate and Land Use Scenarios project, this study predicts that water yield in urban areas of the Southwest, Texas, and Florida will be the most impacted by 2050, in part because these areas are projected to have significant increases in impervious cover and their unique climate. This study suggests that maintaining vegetation ET in urbanizing watershed is important for reducing hydrologic impacts. At a regional scale, watershed management should consider the

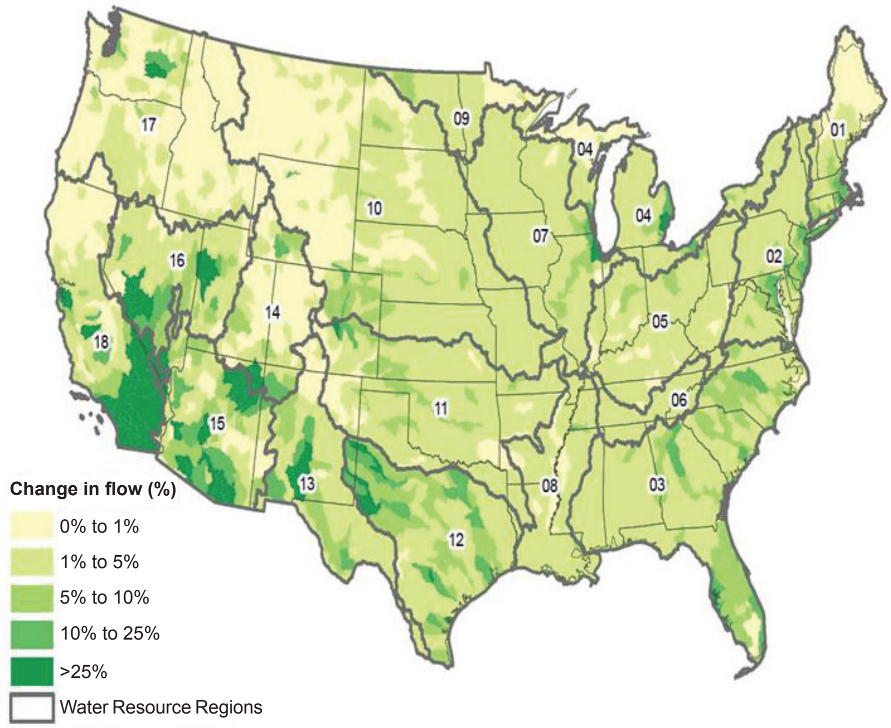

Figure 7.2. Simulated effects of impervious cover on annual water yield (%) in 2010 in the U.S. by a monthly scale water balance model, WaSSI (Caldwell et al., 2012).

climate-driven sensitivity of water yield to increases in impervious cover and the type of land cover being converted in addition to the magnitude of projected increases in impervious cover (Caldwell et al., 2012).

A SWAT simulation study with limited data in Arroio Lino in the state of Rio Grande do Sul in Brazil, that is covered by annual crops (tobacco) and forest at rates of approximately 30% and 50%, respectively, demonstrated that changed land uses impacted the hydrologic distribution (Bonumá et al., 2013). The SWAT model helped to demonstrate that baseflow is an important component of the area's hydrology and continued cropping on steep slopes would lead to greater erosion but also that a smaller scale model would possibly have caught the undulations in topography that the SWAT model missed. This led to an overestimation of hillslope runoff and; therefore, the water balance was incorrect.

Beyond multiple linear regressions for developing empirical relationships

Poor and Ullman (2010) indicate that statistical models developed using regression trees better predicted stream concentrations of nitrate and chloride than use of multiple linear regression in the Willamette River Basin (Oregon, U.S.). The predictive power as quantified by the coefficient of determination (R^2) increased from 0.35 to 0.75 for nitrate simulations and from 0.6 to 0.9 for chloride simulations. Their findings were consistent with earlier work by Creed et al. (2002) whose regression tree model explained 67% of the variability of observed soil nitrogen compared to 23% by the multiple linear regression model at Turkey Lakes Watershed, in central Ontario, Canada. However, Agren et al. (2010) show that if the independent variables are not highly correlated, multiple linear regression is still a powerful predictive tool for developing empirical relationships that simulate transport of nutrients in forested watersheds. They used readily accessible baseflow chemistry data (dissolved organic carbon [DOC]) and watershed landscape characteristics as explanatory variables to predict concentrations of DOC during peak flood conditions. Their parsimonious model comprised of log-transformed baseflow DOC, runoff, and percent wetlands; it explained 87% of total variability of the log-transformed peak flood DOC. They used principal component regression analysis to verify that covariance between explanatory variables did not influence the developed multiple linear regression model. In another study Amatya et al. (2007) compared four different empirical methods to estimate ET based on annual precipitation, PET, and watershed characteristics such as landcover, plant-water available coefficient, elevation, slope, and latitude that was used to estimate/validate annual streamflow for a forested watershed in coastal South Carolina.

Hydrologic recovery after disturbance

Sun et al. (2006) hypothesized that the impacts of forestation in China varied across five regions. The variation in rates of hydrologic recovery was attributed to effects of climate, soils, and vegetation. Troendle and King (1985) observed higher annual flows over a post-harvesting period of 30 years; however, the storm event peak flows over the same period were lower than pre-harvesting peak flows in some of the years. Their findings demonstrate how the effects of harvesting on different hydrologic responses (i.e., annual water yield, peak flow, and water table elevation) may be mediated by other factors such as rainfall intensity and duration. A review of hydrologic recovery on three watersheds in the Northeastern U.S. (Fernow, Leading Ridge, and Hubbard Brook) by Andreassian (2004) demonstrated that it took seven to 25 years for the annual water yields to return to pre-harvesting volumes. The difference in hydrologic recovery rate was attributed to difference in the percent composition of tree species that regrew after 100% clear-cutting (i.e., conversion of pre-treatment hardwood to post-treatment coniferous). Further review of the original work by Hornbeck et al. (1993) indicates that prolonged hydrologic recovery on the above three watersheds was due to application of herbicides to control natural regrowth. For watersheds where natural regrowth was not controlled (Hubbard Brook Catchments 4 and 5), pre-treatment annual water yields were attained within three to four years after harvesting (Hornbeck et al., 1986; Hornbeck et al., 1993; Hornbeck et al., 1997). Analysis by Swank et al. (2001) of 20 year annual water yields following clear-cutting of mixed hardwood forest in the southern Appalachian Mountains (Coweeta Hydrologic

Laboratory, USA), showed hydrologic recovery after five years. This observation was attributed to the rapid regrowth of the herbaceous species in the first three years after clear-cutting. Sun et al. (2000) showed spatial and temporal variation in the hydrologic response of forest clear-cutting on the southeastern coastal flatwoods of Florida. For example, clear-cutting of wetlands and uplands significantly increased the water table elevations. However, clearing cutting of only wetlands did not significantly affect water table elevations in the uplands. For both cases, the treatment effects were more pronounced during dry than wet periods. The same temporal (seasonal) effects were observed by Miwa (2004) and Blanton et al. (1998). A study by Grace et al. (2003) on hydrologic effects of harvesting a mainly hardwood forest stand in North Carolina (U.S.) indicated increases in event outflow, event peak flow and number of flow days. However, their results indicated no significant difference in daily outflow and daily water table depths about two years after harvest. The effect of forest harvesting on daily water table depth and outflow is contrary to results by Sun et al. (2001) for low gradient coastal watersheds. Grace et al. (2003) attributed the high variability of the two hydrologic responses to the record dry spell in 2001.

Scaling issues from plot to watershed scale and beyond

Spilsbury (2002) noted that the hydrological processes related with forests and water may be better understood at a plot level, or for a particular LULC type at a watershed scale but processes become more uncertain at greater spatial scales or smaller scales where the same processes operate across multiple land uses and management regimes. Liang et al. (2012) couple a landscape model (LANDIS) and an ecosystem process model (LINKAGES) to demonstrate that predictions of forest response to climate change at a watershed/landscape scale based on plot scale data are controlled by the sensitivity of forest species to heterogeneity of environmental factors such as temperature, precipitation, and soils. The study demonstrates that for forest species in a natural temperate forest (in China) that are highly sensitive to heterogeneity of environmental controls (i.e., Spruce and Birch), more experimental plots are required to accurately scale species distribution from plot to the watershed/landscape scale. While for species that are less sensitive to environmental controls (i.e., Larch and Fir), the choice of plot location is more instrumental than the number of experimental plots.

Bloschl (2001) suggested maybe, instead of trying to capture everything when upscaling, methods should be developed to identify dominant processes that control hydrological response in different environments and scales; then develop models to focus on these dominant processes, a notion that is called as the 'Dominant Processes Concept (DPC).' This may help with the generalization problems that have haunted hydrologists since the science began. Because most of the LUC effects are generally of interest and are assessed in the scales exceeding plot (< 0.1 ha) and field (< 100 ha) scales, some of the fine processes with less significant effects compared to others may be ignored for large scale assessments. That is how the models like SWAT (Arnold et al., 1998) has been developed and applied in large landscape scale assessments (Gassman et al., 2007; Arnold et al., 1998).

Remote sensing-based approach

Analysis of large complex landscapes with multiple land uses including forests and their effective management for sustainable development involves dealing with large scale spatial and tabular (attribute) data management. These data are becoming increasingly available worldwide with the advancement in satellite-based remote sensing technology. The need for assessing forest health by color changes due to chlorophyll loss has been stated by the United Nations REDD+ framework (Reducing Emissions from Deforestation and Forest Degradation). Carbon budget components such as biomass, dead organic matter, and soils need to be accounted for as well (Lynch et al., 2013). However, with the simultaneous advancements in high speed computing technology and geospatial technology, including geographical information systems (GIS), assessments of effects of land use conversion on various environmental and ecosystem functions are becoming possible for management decisions at various spatial levels from regional, continental, and global scales. Remote sensing uses various types of sensors such as multispectral, hyperspectral, ultraviolet, thermal sensors, light detection and ranging (LiDAR), radio detection and ranging (RADAR), and synthetic aperture radar (SAR), as well as others for collecting intricate information or attributes of forestry management (Franklin, 2001) including harvesting, plantation, and effects of disturbances like invasive species, hurricanes, and droughts. Similarly, time series or temporal analyses of satellite imageries and remote sensing data have been widely used for mapping, monitoring, and post-disturbance including LUC assessments (Pereira et al., 1997; Chuvieco et al., 2005; Chuvieco et al., 2007; Mitri and Gitas, 2010).

Researchers have found the utility of multitemporal medium/coarse satellite imagery from sensors such as Landsat, MODIS, SPOT-VGT and NOAA-AVHRR to assess fire severity (Veraverbeke et al., 2011) and monitor vegetation phenology and regrowth in burned areas (Goetz et al., 2006; Casady et al., 2010). Vegetation indices (VIs) such as NDVI and Soil Adjusted Vegetation Index (SAVI) developed by Huete (1988) were used in these post-fire monitoring studies and they provided accurate analysis on forest cover changes and classification over time. Besides these indices, remote sensing data are being used to derive the climatic (i.e., surface temperature, albedo) and vegetation (i.e., moisture, LAI, height) parameters for water balance, hydrologic, ecosystem, and climate change impact assessments. Panda et al. (2004) studied forest degradation using remote sensing and GIS in Indian forest ecosystems. They suggested that deforestation can be interpreted in terms of the conversion of forestland to other uses such as shifting agriculture (cropping or grazing followed by extended periods of fallow), permanent agriculture (cropping or grazing with little or no fallow), or urban uses. A comprehensive hydrological assessment study using data from a pair of gravity-measuring NASA satellites found that large parts of the arid Middle East region lost freshwater reserves during the past decade (www.jpl.nasa.gov/news/news.php?release=2013-054).

According to de Beurs and Henebry (2004), when change detection techniques using satellite images are based on short time series information, there is a greater risk that seasonal variation can be interpreted as change. For example, if the two different time periods in two different years are used in the analysis, the yearly vegetation cover changes (i.e., crop harvesting) may be considered as vegetation cover degradation. Therefore, caution should be taken while using remote sensing method for forest management and land use assessment studies because of the specific thresholds or change trajectories used in change detection for different spectral and phenological characteristics of land cover types (Lu et al., 2004; Verbesselt et al., 2009). Lynch et al. (2013) state that optical measurements should be taken every one to two weeks to achieve sufficient annual coverage to identify potential forest damage and possible warning signs for future prevention in detection of LULC. As reported by Verbesselt et al. (2009), a newly introduced method Breaks For Additive Seasonal and Trend (BFAST) approach enables the iterative decomposition of time series into trend, seasonal, and noise components resulting in the detection of gradual and abrupt changes in ecosystems and providing accurate data. Together, radar and optical systems can be used to create an early warning system that allows for daily scanning of forests thereby forming a 5–20 meter resolution that monitors logging in real time (Lynch et al., 2013). These authors state an alternative approach to this high cost method. A much less expensive choice is for cheaply made low-resolution optical satellites to monitor forests at a more sparse time scale (i.e., MODIS, DMC, SPOT, Landsat).

Effects of Various Factors in Land Use Change Impact Assessments

Thus far we have described the effects of deforestation and/or clear-cutting forests for land use conversion to agricultural crops or vice versa (i.e., afforestation or reforestation by planting forest on water yield, streamflows, peak flow rates, low flows, and water table dynamics). Almost all of these studies conclude and/or implicate this effect to reduce in ET as a result of canopy removal for deforestation/clear-cutting (Bosch and Hewlett, 1982; Brown et al., 2005). Similarly, the hydrologic effect of replacing pasture or other short crops with trees is reasonably well understood on a mean annual basis (Lane et al., 2003). Higher water yield from croplands/grasslands has been attributed to the lower ET from short crop/grass as compared to taller vegetation, which means that afforestation of grasslands would likely result in reductions in water yield. While it is true that ET is the largest component of the forest water balance, it is also a major component of the hydrologic cycle with direct impacts on water quantity, water quality, and net ecosystem and agri-ecosystem primary productivity. ET is influenced by parameters that vary across multiple scales—from site-specific variables such as soil, vegetation type, and localized weather conditions (PET), across the spatial heterogeneity of land use management at the landscape scales, to regional scales controlled by broad climatic conditions. Furthermore, the effects of LUC also depend on many other characteristics like water use and/or uptake (i.e., transpirational rates) by various vegetation types, percent imperviousness, soil types and hydraulic properties, and land management which may be even more complicated by climate change. There are several studies in the recent literature that estimate mean annual or annual ET as a difference of only annual precipitation and streamflow assuming no change in soil water storage (Lu et al., 2003; Sun et al., 2005; Amatya and Trettin, 2007; Amatya and Skaggs, 2011). Such estimates and other empirical models that include annual potential ET (PET) and vegetation factors (Sun et al., 2005; 2006; 2011b; Lu et al., 2003; Zhang et al., 2001; Turner et al., 1991; Calder and Newson, 1979) to assess streamflow for watersheds and its associated impacts. We synthesize below information on studies conducted to evaluate the effects of ET, methods of estimating ET, soil types and properties, land management, and climate change.

Effects of methods of estimating evapotranspiration

Most of the studies related to assessing effects of forest conversion to agricultural cropland have been conducted using hydrologic models. Breuer et al. (2009) reported that the magnitude of simulated effects depends substantially on the structure/method used in simulating ET. For example, in 10 models the authors evaluated, six of them used the Penman-Monteith method, one used Jensen-Haise method, one used Penman-Monteith, temperature driven monthly factors, one used solar radiation based, and the last one used an empirical temperature and precipitation driven method (Huisman et al., 2009). The authors concluded that although there was a general agreement among the models about the direction of changes in the mean annual discharge and 90% discharge, there was a considerable range in magnitude of predictions. Differences in the magnitude of flow increase were attributed to the different mean annual actual ET simulated by these models for each land use type. Similar findings were reported by Kim et al. (2012) who found the simulated drainage outflow sensitive to the method of estimating PET used in the DRAINWAT model. Similarly, Rao et al. (2011) examined three PET models (FAO P-M grass reference, Hamon, and Priestly Taylor) for possible applications in two mature forests in western North Carolina. The authors concluded that the first two models might underestimate the actual forest ET and thus might underestimate hydrologic effects of forest conversions. The Priestly-Taylor equation gave reasonable annual PET values, but applying the model to estimating actual ET requires calibration; it is unknown how the model performs at finer temporal scales since actual ET data are rarely available for forests. However, using three available PET methods in the SWAT model, Wang et al. (2006) reported that the AET values estimated by the three methods shared a concurrent spatial pattern and temporal trend and were insignificantly different from each other ($\alpha = 0.05$). The results indicated that after calibration, using the three ET methods within SWAT produced very similar hydrologic (AET and discharge) predictions for the studied watershed.

Gordon et al. (2005) as cited in Scanlon et al. (2007) reported a 4% reduction in global ET due to deforestation. Converting forested lands to the production of agricultural crops nearly always reduces ET and increases runoff (Skaggs et al., 1991; Skaggs et al., 2011; Sun et al., 2005; Amatya et al., 2008; Amatya and Trettin, 2007; Scanlon et al., 2007). Besides decreased ET, effects vary widely in quantity and in the timing based on the type of land, crop, and water management including the types of site preparation and the timing of such management during the year (Skaggs et al., 2011; Rab, 2004; Grace et al., 2006). In their study of assessing the impacts of reduction in forest cover on mean annual runoff using two empirical methods involving annual precipitation, land cover, elevation, and precipitation that fit the best with annual streamflow, Amatya and Trettin (2007) found an increase of as much as 62% runoff as a result of removal of 90% forest cover on the study watershed. Data in Table 7.1 present results from various types of studies that assess the effects of forest clearing and/or land use conversion from forests to agricultural croplands around the world.

Effects due to changes in type of crop, vegetation, and their water use

Forests generally have higher ET than other types of vegetation (1.6 times higher than grasslands (Zhang et al., 2001), as cited in Scanlon et al. (2007). In recent years, a need to better understand the relationship between watershed vegetation type and the variability of annual runoff as affected by vegetation manipulation for ET has found important implications for water resources management and development, stream ecology and fluvial geomorphology (Williams et al., 2012; Sun et al., 2005; De Wit, 2001). Holmes and Sinclair (1986) and Zhang et al. (2001) developed a relationship between annual ET and annual rainfall for various types of vegetation including grass and trees. Accordingly, in their study using worldwide fluxnet data, Williams et al. (2012) reported that grasslands on average have a higher evaporative index (ET/P) than forested landscapes, with 9% more annual precipitation consumed by annual ET compared to forests. The authors stated that while the Budyko framework's assumption of using mean annual precipitation and net radiation as two variables controlling mean annual ET and streamflow, vegetation type may well be another control. Brauman et al. (2012) also found that modeled PET from pasture was higher than that for the forest. This finding, according to the authors, was due to a balance between aerodynamically and stomatally controlled ET that differs significantly between two vegetation types, changing weighted sum of the two components yields, and lower PET at the forest sites.

Based on the SWAT model simulations, Schilling et al. (2008) concluded that historical LULC change in the U.S. Corn Belt region impacted the annual water balance in many Midwestern basins by decreasing annual ET and increasing streamflow and baseflow. Consistent with historical observations, their modeling results indicated increased corn production would decrease annual ET and increase water yield and losses of nitrate, phosphorus, and sediment, whereas increasing perennialization with grasses for ethanol biofuel would increase ET and decrease water yield and loss of nonpoint source pollutants. Global eddy flux ET data for different ecosystems have gradually become available for a general understanding of the environmental control of ET processes and validating hydrological models. Brauman et al. (2012) noted that concerns about reductions in water yield due to afforestation are likely to be relevant only in systems in which wind speeds are high and water stress limits ET.

Sun et al. (2011a) conducted a synthesis of ET studies for 13 worldwide intensively measured sites and found that monthly leaf area index (LAI) was the single most useful variable to explain ET variability across ecosystems over time, and PET and precipitation were additional key climatic variables for predicting monthly ET. There is a large variability in ET in space and time, and vegetation's influences on ET can be masked by climatic factors that are rather complex. Using similar eddy flux data from Ameriflux, a recent analysis suggests that forests use more water than grasslands

Table 7.1. Summary of studies on effects of land use change on streamflow.

Study	Site Name	Site Area, km^2/Vegetation type	Method Used	Data/Simulation period	Mean annual rainfall/Runoff, mm	Increase or Decrease n Streamflow, mm (%)
Bonumà et al. (2013)	Rio Grande do Sul, Brazil	4.8 km^2/ Tobacco	SWAT model	2001–2005		Overpredicted sediments
Kim et al. (2012)	S4 coastal forest watershed, NC, U.S.	29.5/100% forest	DRAINWAT model	1951–2000	1290	65 (25%) increase for 100% conversion to cropland
Caldwell et al. (2012)	Continental scale for U.S.	Impervious area conditions in 2010 compared to past 30 years	WaSSI model	2010	Variable	mean 9.9% increase of flow in 2099 watersheds, median 2.2%
Kuchment et al. (2011)	North-western part of Russia		Water balance model			30% larger snowmelt rates; however, 10% higher runoff from forested than clearcut due to snowmelt
Odira et al. (2010)	Degrading watershed, Kenya		SWAT model	1973–2000		Streamflow (both peak and mean) increase in rainy season and decrease in dry season with increase in ag-land; increase in forest reduces peak flow
Qi et al. (2009)	Trent River watershed, Coastal NC	377/66% forest	USGS PRMS model	20 yrs (1981–01)	1300/426	59 (14% increase in flow)
Dai et al. (2009)	Control watershed, WS80, Coastal SC	1.6/100% forest	DHI- MIKESHE model	3 yrs (2003–06)	1270/269	113 (30% increase in flow)
Chescheir et al. (2009)	El Cerro basins, Uruguay	0.76 to 1.08	Paired watersheds	2000–2007		15% reduction for three years after planting
Dai et al. (2008)	Control watershed, WS80, Coastal SC	1.6/100% forest	DRAINMOD model	3 yrs (2003–06)	1270/269	122 (35% increase in flow)

Silverra and Alonso (2008)	Manual Diaz basin, Uruguay; Before and after afforestation (25% area)	2100	Measured data			8.2–36.5% flow decrease
Amatya and Trettin (2007)	Turkey Creek watershed, Coastal SC	72/96	EMPIRICAL: Rain, Canopy, PET	13 yrs (1964–76)	1320/350	208 (60% increase in flow)
Fernandez et al. (2007)	S4 watershed, Parker Tract, Coastal NC	30/50	DRAINMOD-based		1354/437	208 (16% increase in flow)
von Stackelberg et al. (2007)	Pastured watershed planted with pine, Uruguay	0.7–1.08	SWAT model	2000–2004	1487	23% reduction in mean annual flow
Siriwardena et al. (2006)	Natural forest, Queensland, Australia	16,400	Modeling startegy	Mid-1960s		Increase in outflow by 40%
Best et al. (2003)	Australia (pasture to pine and native forest to pasture;	0.94–3.10	Flow Duration Curve method	1974–1994	876	50% reduction in highflows and 100% reduction in low flows and for grass to pine and similar increase for vice versa; 30% reduction in New Zealnd
	New Zealand (grassland to pine)			1980–1999	1290–1310	
Wilk et al. (2001)	Nam Pong forested watershed, Thailand	12,200/80	Rainfall-runoff model			

Table 7.1. contd....

Table 7.1. contd.

Study	Site Name	Site Area, km^2/Vegetation type	Method Used	Data/Simulation period	Mean annual rainfall/Runoff, mm	Increase or Decrease n Streamflow, mm (%)
Scott et al., 1998	South Africa watersheds; Grasslands planted to pine, eucalyptus, and wattle	2612–378,243 (1.2% forest)	Robust empirical model	Various lengths	448/37	3.2% reduction in mean annual flow
Jewitt (2002)						7.8% reduction in low flows
Scott and Smith (1997)	5 South African watersheds planted to pine and	0.26 to 2.0	Empirical relationships		1135–1611	Variable reductions with higher on Eucalyptus than the pine
	Eucalyptus				217–742	Higher percent reductions in low flows than in high flows
Dagg and Blackie (1965), Blackie (1972), Edwards et al. (1976)	Kenya & Tanzania:	0.544–0.702	Paired watershed approach and water balance	1958–1973	1500–2880	Longterm streamflow reduction of 8.9%
	Kericho (Montane forest to 54% tea estate) Kimakia (Vegetables to mature pine: 35 feet)					Water use at Kericho (Kenya) decreased by 14.4% after clearing and planting and increase by 2.4% with tea plantation
	Mbeya range (Evergreen forest to cultivated crops					
Dung et al. (2012)	Japan (43.2% thinning of a cypress forest)	0.002–0.004	Paired watershed approach	2004–2009	1732	240.7 mm increase in mean annual flow (36.7%) of 2 years

in general. The ET/P is much higher in deciduous forests than in grasslands. Irrigated croplands can have similar ET to forests (Fig. 7.3).

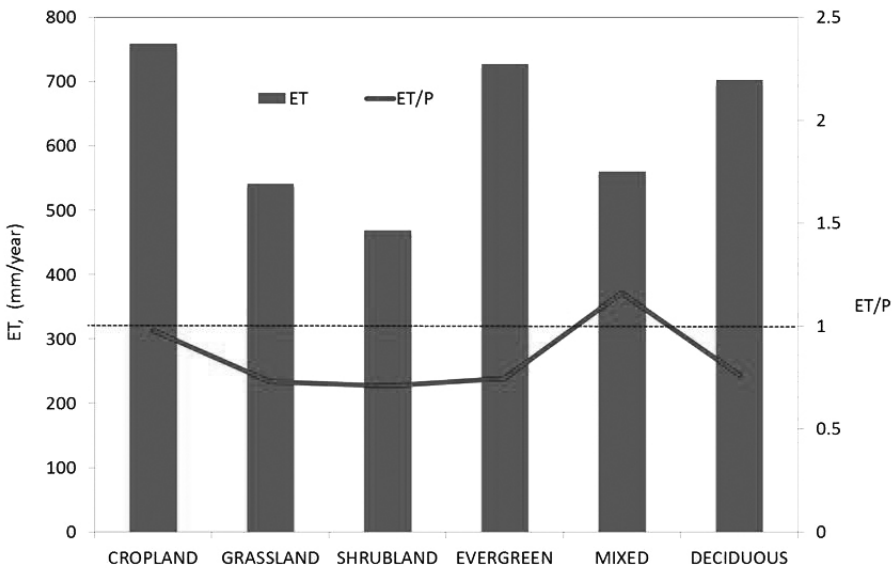

Figure 7.3. A comparison of annual ET and ET/P measured by the AmeriFlux network (Ge Sun, unpublished data).

Effects of land, soil and water management practices

In their simulation study on effects of land use on soil properties and hydrology of drained coastal plain watersheds using a validated DRAINMOD model, Skaggs et al. (2011) reported that the higher ET on an artificially drained pine forest resulted in reduced drainage outflow and deeper water table depth compared to an agricultural cropland site in North Carolina lower coastal plain in the U.S. The authors also argue that the assumption of approximation of soil-water properties based on soil type, independent of crop or land use, may not always be valid. For example, field effective hydraulic conductivity in the top 70 cm of the drained forest site was more than two orders of magnitude greater than that of the corresponding layers of soil on the agricultural site. Drainable porosity was much higher for the forested sites. As a result of these and large surface depressional storage, predicted surface runoff from the forested site was nil. Harvesting using heavy machines have the potential to disturb the forest soil surface including their structure and soil hydraulic properties (Rab, 2004; Skaggs et al., 2006; Grace et al., 2006). Skaggs et al. (2006) observed 20 to 30 times higher values of saturated conductivity (K_s) at 90 cm depth than those given in the NRCS Soil Survey Report for a Deloss fine sandy loam after harvest of a poorly drained mature pine plantation in coastal North Carolina. The authors noted that harvest did not appear to have affected those values, but site preparation for regeneration, including bedding, reduced the effective K values. A study by Grace et al. (2006) stated that soil compaction during harvesting of hardwoods on poorly drained soils of a Tidewater region of North Carolina increased the soil bulk density from 0.22 to 0.27 g cm^{-3} and decreased saturated hydraulic conductivity from 397 to 82 cm hr^{-1}. Rab (2004) reported that 10 years after timber harvesting in Australia, there was a 22–68% difference in soil physical properties of harvested compared to undisturbed soils.

Brauman et al. (2012) noted that in addition to afforestation, other changes in land use such as increased grazing intensity, can have an unexpected and pronounced role on soil properties. The authors reported while grazing reduces PET in temperate climates, an opposite effect was found in their humid site where short grazed grass increased PET rates. Grazing that reduces understory ET in forest may reduce water use. In a study on riparian deforestation effects, Greenberg et al. (2012) used LiDAR data to estimate changes in insolation that affect stream water temperatures and ecology.

Effects of climate change

The global climate change has direct (precipitation input, ET, and extreme events) and indirect (fires, insect disease, plant growth, invasive species) impacts on watershed hydrology, and has consequences of future LUC. In the southern U.S., climate warming is likely to increase ET, and thus decrease water yield if precipitation does not change (Sun et al., 2012). An increase in storm intensity and frequency is likely to increase in stormflow and peak flow rates and soil erosion potential (Sun et al., 2012; Dai et al., 2011). In such a case, the effects of deforestation or forest clearing for developments resulting in higher streamflows may be reduced due to higher soil and vegetation evaporation as a result of warming temperatures for prolonged growing season. At the same time seasonal flows may be further exacerbated due to projected increased intensity of storms. However, watershed water yield is most sensitive to precipitation change in a wet environment as shown in Fig. 7.4. Recently, Patterson et al. (2012) argued that whether the decrease in temperature with increase in observed precipitation and streamflow in the South Atlantic from 1964 to 1969 but with opposite trends from 1970 to 2005 have been driven by climatic or anthropogenic changes poses a great challenge to water resources managers.

Land use change and climate change do not occur independently. Lettenmaier et al. (1994) proposed that where streamflow does not follow climatic indicators, the cause is likely anthropogenic, although the LUC due to urbanization may confound climate-streamflow relationships (Shrestha et al., 2012). In an analysis of climate and streamflow data from six gauging stations from 1961 to 2006 in northeast China, Zhang et al. (2011) found that climate variability was estimated to account for 43% and human activities accounted for about 57%, respectively, of the reduction in the annual streamflow. Climate change often interacts with forest cover change to affect streamflow (Ford et al., 2011). Forest management may aggravate or mitigate climate change impacts on water yield. Urbanization generally increases stormflow and peakflow, impacts of which may likely be exacerbated due to extreme storm intensities projected as a result of climate change. Alternatively, climate change induced droughts

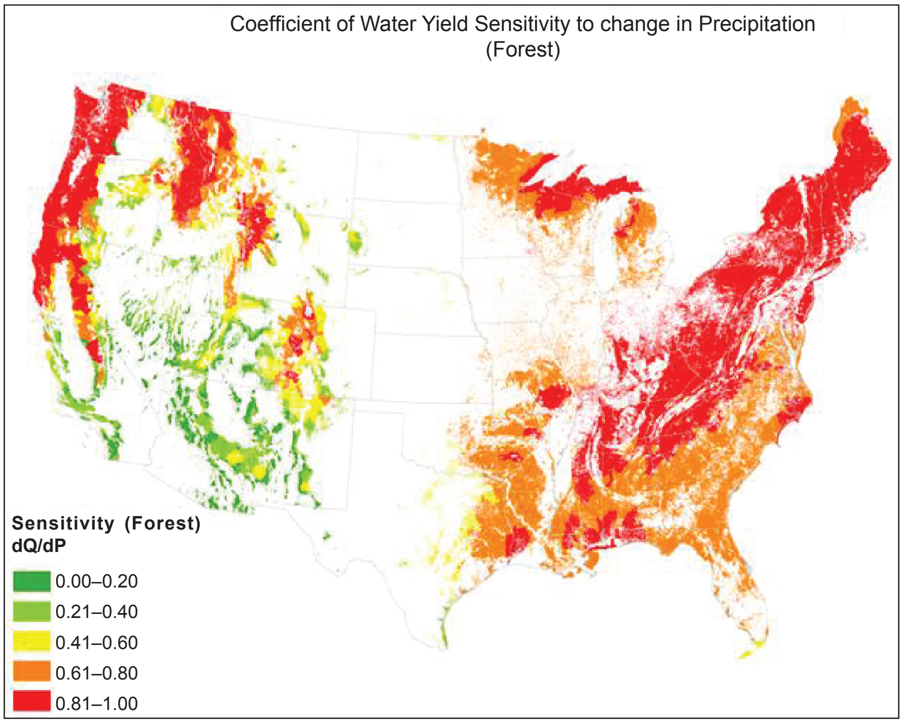

Figure 7.4. Forest water yield sensitivity (dQ/dP) to precipitation change as modeled using a method in Ma et al. (2008).

may aggravate the hydrologic influences of reforestation (i.e., water yield reduction). A parameter output sensitivity analysis suggests that with a reduction of leaf area index of 50% may double flow in the coastal plain in the humid southern U.S., but with minimal effects in the dry region (Fig. 7.5). In another study in an agricultural watershed on the Loess Plateau of China, Li et al. (2009) reported that the integrated effects of LUC and climate variability decreased runoff, soil water contents and ET. LUC increased ET by 8% while climate variability decreased by 103%. Similarly, a recent study by Shrestha et al. (2012) reported that the effects of changes in climatic variables on nutrient transport need to be considered with possible future changes in land use, crop type, fertilizer application, and transformation processes in the receiving water bodies.

Forests for bioenergy in a changing climate

The environmental value of forests will increase as climate change accelerates. While sustainably managed forests are encouraged for climate protection by carbon sequestration (UN General Assembly, 1994), the climate benefits of forest biogeophysical processes may equal those of carbon in the tropics (Bonan, 2008; Betts,

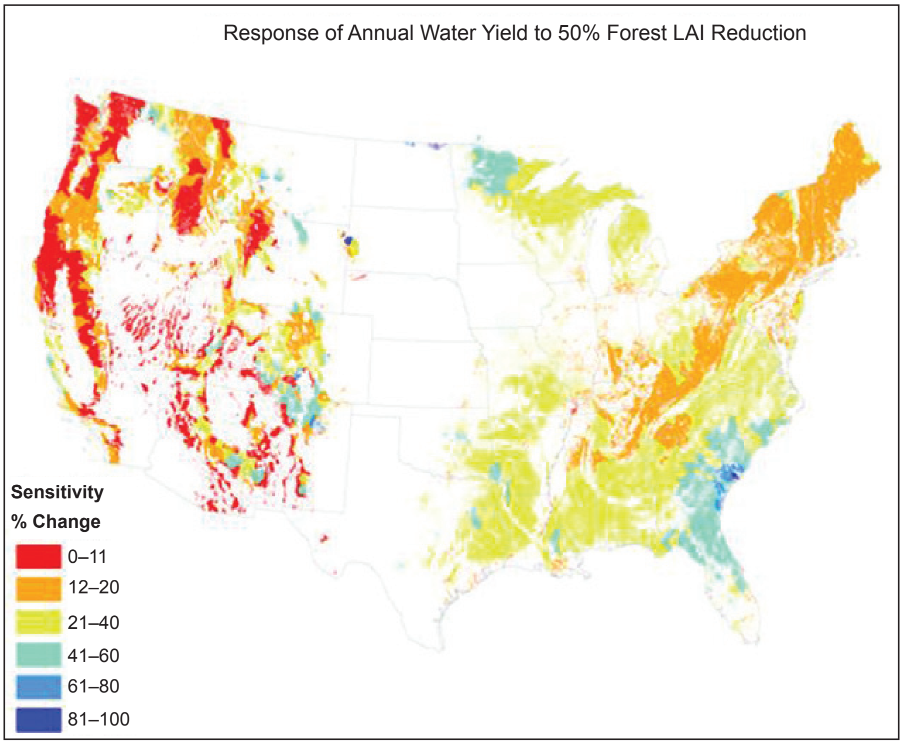

Figure 7.5. WaSSI modeled sensitivity of water yield to 50% reduction of forest leaf area index (LAI).

2011), although this diminishes in temperate and boreal forests. Cellulosic biofuel from forest-based bioenergy crops—whether short rotation woody crops, energy grass intercropping, or higher utilization of harvests—can increase again the value of forests and significantly reduce greenhouse gas emissions by replacing fossil fuels.

Human and natural disturbances are expected to become increasingly common; these alterations will impact forests (Dale et al., 2001), and be mitigated by them. By restricting pollutant movement due to vegetative ground cover, providing protection as a wind barrier, or acting as a filter by which chemicals are retained by the soil and/or organic debris or are remediated into alternative byproducts over time, forests are an *in situ* alternative to chemical treatment. Forests have been shown to intercept harmful radioactive elements (Kato et al., 2012), phytostabilize various metals and salts (shallow and deep rooted trees), and mineralize organic pollutants to carbon dioxide and salts. Trees reduce mass wasting and water held in fog moisture can be captured a increasing water resources in montane areas. Riparian forests protect watercourses from upland chemical uses (pesticides, fertilizers) and as a filter and depository for sediment (USFS, 2013). Forests cool nearby air and water bodies through shading and evaporation, reducing their temperature during hot season, providing a favorable habitat for aquatic life.

Sustainably managed forests operate under prescriptions that maximize economic return while protecting environmental resources. Many planning and operational guidelines are for water, with the majority being for water quality. BMPs have been shown to protect water resources, including aquatic life, by practices that vary in detail and implementation from state to state (Ice et al., 2010) but work as a system (Ice and Stednick, 2004). Minimizing net greenhouse gas emissions from forests and forest products may reprioritize forest benefits in a way that forces changes in BMPs. For example, current prescriptions call for limiting stream crossings, minimizing operational tract size, and disconnecting harvests. These inefficiencies in the harvesting process increase fuel consumption, and may need to be re-evaluated through the carbon life cycle analysis in order to provide the maximum benefit for biofuel solutions.

Several forest biogeophysical processes interact with climate, and are especially affected by albedo and vegetative ET. Strong forest ET cycles cool the atmosphere, while relatively low albedo (compared to agriculture) can have a warming effect. Arora and Montenegro (2011) introduce the concept of the “temperature effectiveness of afforestation” to aggregate the multiple effects of forests on climate, and demonstrate that forests can have either positive or negative effects on temperature, dependent on the characteristics of local climatic regime. Climate models predict that wetter areas will get wetter, although PET may outpace precipitation. In addition, storminess (Muschinski and Katz, 2013) and rainfall variability show signs of increase. More planted forests in wet areas could stabilize water yield, in addition to providing cooling, but forests in drier areas will be subject to die off due to great drought stress under climate change (Williams et al., 2013). These site-specific considerations will only become more important for intensive forest-based biofuel.

In addition to direct climate effects, the substitution of sustainably produced biofuel for fossil fuel can reduce greenhouse gas emission. Kior, the first commercial-scale producer of cellulosic biofuel, reports a reduction of 80% in emissions from their product made from wood chips over fossil-fuel emissions (Kior, 2013). While first-generation biofuels reduce emissions, the rain fed water use is a huge advantage over irrigated crops now often used for biofuel. The economic benefits of a shorter rotation crop may make conversion of marginal agricultural land feasible, moving land use from annual to perennial plantings and reducing erosion rates. Energy grasses have high water use efficiency and after establishment will protect soil from erosion. Higher harvest utilization takes advantage of an existing source of biomass. All of these possibilities are dependent on technology and economic conditions; however, research needs to continue into all possible scenarios to allow society to make the best choices that protect local environmental conditions and prioritize concerns.

A Conceptual Model

Based on our synthesis of information presented above on hydrologic impacts of forest removal as well as afforestation and effects of various factors including climate variability and change, a conceptual model was built with various types of stressors like LUC, climate change, and others interacting with various eco-hydrologic processes and functions with overall impacts on water quantity, streamflow timing and distribution, and water quality (Fig. 7.6).

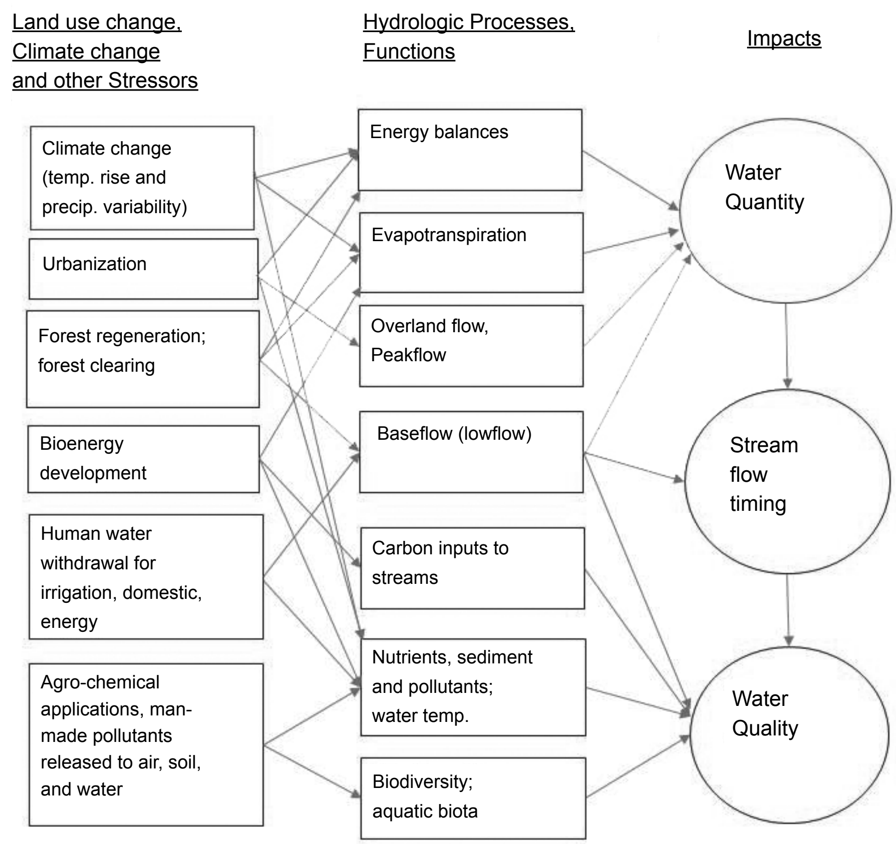

Figure 7.6. Factors affecting quantity, quality, and timing of streamflow.

Future Directions in Advancing Understanding the Interactions of Forests, Water, and Land Use Change

Water from forested landscapes is critical for supporting ecological systems and surrounding communities. Many factors that affect water quantity and quality, including LUC and climate change are summarized in a conceptual model (Fig. 7.6) that can help resource managers understand the interactions and pressures of various influences. Forest hydrology must advance to address current complex issues, including climate change, wildfires, changing patterns of development and ownership, and changing societal values (NRC, 2008; Jones et al., 2009). Sound science is needed to support wise decisions and appropriate responses to contentious water policy issues (USFS, 2011). The complex water and forest issues must be addressed using a 'one land' or a landscape approach. There is an urgent need for better understanding of the interface between forests/trees, and water, for awareness raising and capacity building in forest hydrology, and for embedding this knowledge and research findings in policies (Hamilton et al., 2008).

Future forest hydrologic science and related studies should move beyond 'forests'. Modern forest hydrology research should address issues on multiple spatial scales covering multiple landuses, including larger-scale contemporary water resource issues (i.e., water supply, instream flow, floods, droughts, beneficial water uses), climate change, hydrologic processes under disturbances (i.e., invasive species, fire, etc.) and urbanization (Sun et al., 2008; Amatya et al., 2011; Dai et al., 2011). Future forest land conversions and changing climate have a series of cascading effects (i.e., invasive species, fire, etc.) besides eco-hydrologic processes (Fig. 7.6; Vose et al., 2011; Brauman et al., 2012). To better understand their natural variability and for accurate impact assessments in field and landscape scales, longterm measurements at forest landscapes of varying ecological regions/characteristics accompanied by innovative monitoring technologies including remote sensing and advanced modeling are critical.

Defries and Eshelman (2004) call for a multidisciplinary approach with a comprehensive view towards the hydrologic processes that maintain ecological health and human requirements for food, water, and shelter. While the authors elaborated on available methods for assessing the effects of LUC from controlled experiments like paired watershed approach and mathematical modeling and their strengths and limitations, they expected that experimental approaches combining measurements from paired watersheds with process modeling will serve to unravel rapidly the response to LUC of watersheds of varying size, topography, and spatial configuration. Generally modelers calibrate and validate a model for current conditions of a watershed/landscape and predict the conditions of future treatment or calibrate the model for the first few years for pre-treatment and validate for the remaining time period for the treatment on the same watershed (Skaggs et al., 2011; Kuchment et al., 2011). A more novel approach for an accurate assessment of hydrologic and water quality impacts is to simulate, not only the reference conditions, but also the processes and interactions involved after the treatments. In that context, the applicability of the paired watershed approach has also been questioned in accurately predicting the peak flow rates, particularly their frequency and return periods (Kuras et al., 2012; Alila et al., 2009). Alila et al. (2009) argued that the science of forests and floods is in an urgent need of reevaluation of past studies in light of changing climates, insect epidemics, logging, and deforestation worldwide.

Spilsbury (2002) noted that most watershed management projects give little priority to research and monitoring, even though these are essential to effective watershed management and essential for establishing the efficacy of water management interventions. The author, therefore, concluded that future research efforts should focus on ways to maximize provision of environmental services in mixed land use mosaics, and strive to influence and inform public policy debates.

A scientific investigation of the causes and consequences of LULC requires an interdisciplinary approach integrating both natural and social scientific methods, which has emerged as the new discipline of land-change science (Ellis and Pontius, 2010). For more than a century agricultural and biological engineers have provided major advances in science, engineering, and technology to increase food and fiber production to meet the demands of a rapidly growing global population. Much of our agricultural land base originates from historically forested lands (Amatya et al., 2009), which have experienced dramatic declines and resurgence over the past century, including the Southern U.S. (USFS, 2011). The resulting landscape is a mosaic of agricultural, forest and urban lands that may not be sustainable with respect to the expected goods and ecosystem services including water quantity and quality (Amatya et al., 2011).

Water and nutrient balances quantified using long-term hydrology and water quality data from forested watersheds with minimal anthropogenic disturbances can serve as a reference for assessing the effects of LUCs into croplands and/or urban areas (Amatya and Skaggs, 2011). In this context, additional research is needed to advance our current understanding of forest hydrologic processes, especially the detail of stream networks, topographic depressions, floodplain and wetland functions, preferential flow characteristics into and within the forest soil profile, shallow and deep water table influences, flow generation in low-gradient watersheds, and the ET process for various forest types and species including the understory, which has received limited attention in the literature. An accurate understanding of these processes in a reference forest system is critical to the evaluation of impacts of all disturbances to the system. Hamilton et al. (2008) provide recommendations for a number of special forest situations important for water resources and their management. Those situations include management of cloud forests, swamp forests, riparian forest buffers, headwater forests for clean drinking water, and other forests to minimize salinization, erosion, etc.

Recent advancements in electronic sensor/digital monitoring, mapping, and remote sensing technology together with computing speed should also be used as opportunities to address these complex ecologic, hydrologic, and biogeochemical processes during land use conversion in a changing climate. Shuttleworth (2012) stated that future research is also required to fully validate recent interactive vegetation models, perhaps using remote sensing data.

Lockaby et al. (2011) reported that water stress will likely increase significantly by 2050 under four climate change scenarios largely because higher temperatures will result in more water loss by ET and because of decreased precipitation in some areas. Williams et al. (2012) concluded that climate type and vegetation should be considered in assessing ET, when streamflow is being regarded. However, the degree to which we can estimate changes in vegetation type with new ET requirements are speculative at best. The same authors also concluded that water stress due to the combined effects of population and LUC will increase by an average of 10% in the southern U.S. by 2050. Additional research is needed to identify innovative solutions and methodologies for mitigating potential impacts of climate and LUC for sustainable management of water resources on large prior converted agricultural landscapes that include forested watersheds. Furthermore, research needs more accurate information about the quantification of relationships between ecosystem attributes and forest management, including biomass production and harvest in a multi-dimensional context (Loehle et al., 2009).

According to the 4th Assessment Report of the Intergovernmental Panel on Climate Change (IPCC), the links between water and climate change are undeniable, with water predicted to be the primary medium through which early climate change impacts will be felt by people, ecosystems, and economies (WWC, 2009). Moreover, these climate change impacts will compound other existing pressures on water resources such as population growth, LUC, and changes in consumption patterns. As a result, further research is warranted to determine whether the impacts on streamflow and/or water availability trends have been driven by climatic or anthropogenic effects (i.e., LUC) posing a greater challenge to water resource managers and planners.

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