Effect of historical changes in land use and climate on the water budget of an urbanizing watershed

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[1] We assessed the effects of historical (1931–1998) changes in both land use and climate on the water budget of a rapidly urbanizing watershed, Ipswich River basin (IRB), in northeastern Massachusetts. Water diversions and extremely low flow during summer are major issues in the IRB. Our study centers on a detailed analysis of diversions and a combined empirical/modeling treatment of evapotranspiration (ET) response to changes in climate and land use. A detailed accounting of diversions showed that net diversions increased due to increases in water withdrawals (primarily groundwater pumping) and export of sewage. Net diversions constitute a major component of runoff (20% of streamflow). Using a combination of empirical analysis and physically based modeling, we related an increase in precipitation (2.7 mm/yr) and changes in other climate variables to an increase in ET (1.7 mm/yr). Simulations with a physically based water-balance model showed that the increase in ET could be attributed entirely to a change in climate, while the effect of land use change was negligible. The land use change effect was different from ET and runoff trends commonly associated with urbanization. We generalized these and other findings to predict future streamflow using climate change scenarios. Our study could serve as a framework for studying suburban watersheds, being the first study of a suburban watershed that addresses long-term effects of changes in both land use and climate, and accounts for diversions and other unique aspects of suburban hydrology.

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1. Introduction

[2] Humans are an active and increasingly significant component of the hydrologic cycle [*NRC*, 1992]. The effects of anthropogenic changes in land use on watershed hydrology have been well studied, especially the effects of forest harvest practices, the conversion of native grasslands and forests to agriculture, and urbanization [*Jones and Grant*, 1996; *Stednick*, 1996; *Matheussen et al.*, 2000; *Dow and DeWalle*, 2000; *Beighley and Moglen*, 2002]. Forest cover strongly affects evapotranspiration (ET), snow accumulation and snowmelt processes relative to other land uses, and removal of forest is known to increase streamflow as a result of reduced ET. There has been an awareness of the effects of urbanization on watershed hydrology since the late sixties [*Leopold*, 1968]. Related studies have tended to focus on small catchments with high impervious areas or on larger catchments where urban area is typically estimated from US Census population data [*DeWalle et al.*, 2000]. In general, urbanization is considered the most dominant factor altering hydrology because of vegetation removal, creation of impervious surfaces, changes in water use and water diversions.

[3] Changes in climate can also have a major impact on the hydrology of a watershed. Studies have shown increases in mean river flow due to increases in precipitation and reductions in flow due to increases in temperature [*Frederick and Gleick*, 1999]. Elevated CO_2 concentrations may also influence the hydrologic cycle via stomatal closure, which would decrease ET [*Wigley and Jones*, 1985]. Land use change and climate change are interrelated, with land use change as an important component of anthropogenic climate change [*Kalnay and Cai*, 2003].

[4] Within the context of global climate change, there is a mounting need for predicting how watersheds will respond to these changes [*NRC*, 1992]. Unfortunately, what is lacking from most climate change assessment studies is a careful analysis of how historical changes in climate have altered basin hydrology. The same is true for aspects of land use change, especially when important activities associated with land use change (e.g., basin diversions of streamflow) have been treated minimally.

[5] Our goal is to assess the effect of historical changes in both land use and climate on the water budget of a rapidly urbanizing watershed (Ipswich River basin in Massachusetts), in particular the effect on ET. Few studies have addressed the effects of both these changes at the local scale, especially when accounting for water diversions. Urbanization in the Ipswich River basin and surrounding

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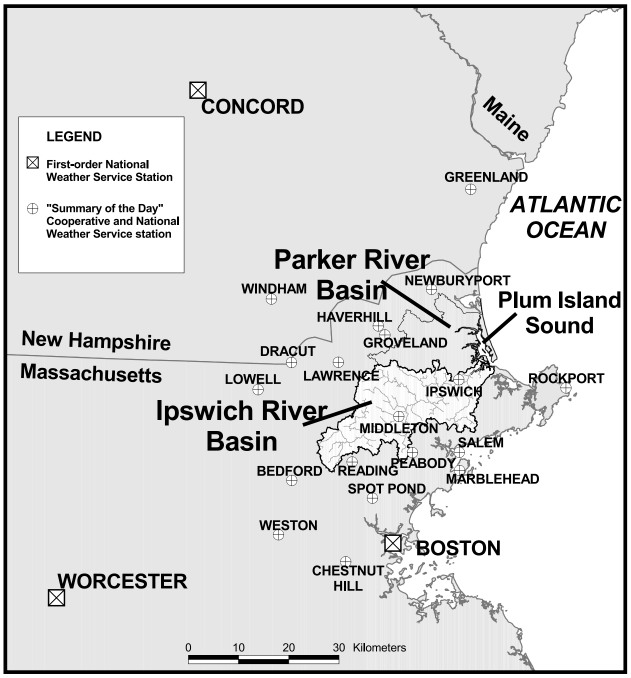

Figure 1. Map of the Ipswich River basin and meteorological stations.

areas has led to an increase in water diversions. Our study centers on a detailed analysis of these water diversions and a combined empirical/modeling treatment of ET response to changes in climate and land use. The latter allows us to separate and quantify the climate and land use change effects.

2. Data and Methods

2.1. General Approach

[6] Our approach employs a combination of empirical analysis and physically based modeling. First, we develop a long-term historical water budget (1931–1998), which includes a detailed accounting of water diversions. We corroborate our estimate of water-budget ET, by comparing it to an ET model (CRAE model). Next, we analyze the historical water budget for long-term trends. We relate trends in ET to changes in climate. Next, we separate the effect of change in land use from change in climate, through the development and application of a physically based water-balance model. This model is specifically designed for addressing differences in ET between vegetation types, and is largely built on existing models. We perform long-term model simulations (1949–1998) to address the effect of land use change. We generalize these and other findings to predict future streamflow using climate change scenarios.

2.2. Study Area

[7] The 404 km^2 Ipswich River basin (IRB) is located in the Atlantic coastal plain in Northeastern Massachusetts, 30 kilometers north of the city of Boston (Figure 1). The IRB underwent major changes in land use during the 1900s (Figure 2). While agricultural abandonment/forest re-growth characterized the first part of the century, the latter part of the century was characterized by a sharp increase in residential land use, primarily at the expense of forested areas. During this latter period, the ratio of forested to residential areas decreased from 5 in 1951 to 1 in 1999. The population in the watershed almost quadrupled over the last 80 years, and is currently estimated at 120,000 (Figure 2). The increase in population has led to an increased demand on IRB water resources and diversions have become an important issue. For example, increases in groundwater pumping have decreased low flows [*Zariello and Ries*, 2000].

[8] The climate in the IRB is humid, while the temperature is moderated by its proximity to the ocean. The average annual (1961–1990) air temperature is 10°C, ranging from −4°C in January to 22°C in July. Precipitation is distributed uniformly throughout the year, with an annual average of 1188 mm. Snowfall accounts for about 8% of total annual precipitation. On an annual basis, the amounts of precipitation leaving the basin as either ET or streamflow are similar in proportion (~45%), while net diversions leaving the basin account for the remaining portion (~10%). While precipitation does not display seasonality, there is a distinct seasonality in streamflow, driven by the strong seasonal climatological forcing on snowmelt and ET. Average monthly streamflow ranges from 107 mm in March to 8 mm in September, while monthly ET ranges from 7 mm in December to 103 mm in July.

[9] The IRB is located in a region of low relief. Elevation ranges from sea level to about 130 m, with a mean elevation of 40 m. The Ipswich River drops 34 m along its 56 km course, an average slope of 0.06%. Stratified glacial drift deposits (sand and gravel) mostly underlay the lowland areas of the basin, while glacial till mostly underlies the upland areas. Postglacial alluvial deposits are found mainly along the valley floors. The low relief of the basin is responsible for one of its main characteristics, a large expanse of wetlands. Swamps and marshes, which respectively cover 15% and 6% of the basin area, dominate the valley floors. These wetlands influence streamflow dynamics, ET and groundwater recharge [*Sammel et al.*, 1966].

2.3. Water Budget

[10] The water budget for a river basin can be described, for any given time increment:

$$\Delta S = P - R - E - D \tag{1}$$

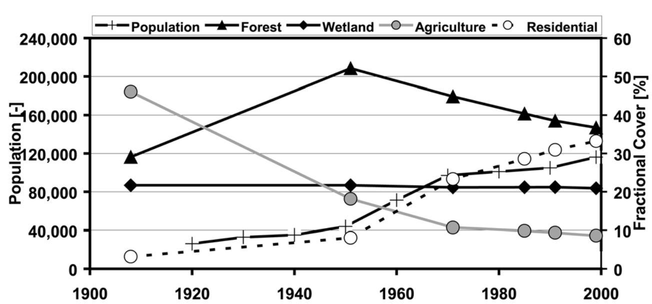

Figure 2. Population and fractional land use in the Ipswich River basin. Population data are derived from U.S. Census data. Land use estimates are based on town-level tabular data (1907 and 1951) and 1:24,000 GIS data (1971, 1985, 1991, and 1999). Symbols refer to data points; lines are for illustration only.

Table 1. Data Sets[a]

Element	Description
Precipitation and temperature	"Summary of the Day": subset of the National Climatic Data Center (NCDC) TD-3200 data set. Contains long-term records of daily observations from the National Weather Service (NWS) and cooperative station network. 31 stations are located within 25 km of the watershed (Figure 1). Data record extends as far back as 1920.
Humidity and wind speed	"First-order summary of the day": subset of the NCDC TD-3210 data set. Contains long-term records of daily observations from NWS first order weather stations. Three stations are located within the vicinity of the watershed (Boston MA, Worcester MA, and Concord NH) (Figure 1). Data record starts in 1949.
Solar radiation	Hourly estimates of global horizontal radiation, produced by the Northeast Regional Climate Center [*DeGaetano et al.*, 1993]. Estimates are based on meteorological observations from the three NWS first-order weather stations. Data record starts in 1949.
Streamflow	Daily observations from USGS gage in Ipswich. Data record starts in 1930.
Water withdrawals	Monthly data from the Massachusetts Department of Environmental Protection (MADEP), supplemented with data from municipal water departments. Data correspond to municipal withdrawals, including groundwater pumping and surface water withdrawals. MADEP data record starts in 1956; town records go back as far as 1934.
Land use	GIS data layers obtained from the Massachusetts Executive Office of Environmental Affairs (MassGIS). Land use data layers are available for 1971, 1985, 1991, and 1999. GIS data were supplemented with town-level tabular data for 1907 and 1951, obtained from records from the Board of Agriculture and Department of Agriculture, respectively.

[a]Note: Daily meteorological input data were constructed using inverse distance weighted (IDW) interpolation, followed by basin-averaging (spatial variability was not significant for estimating basin-scale ET). Because IDW interpolation of precipitation is not appropriate at a daily time step (e.g., convective storms), these surfaces were adjusted to make them consistent with IDW surfaces of monthly observations.

where ΔS is change in storage (including surface water, soil moisture and groundwater); P is precipitation; R is streamflow; E is evapotranspiration; and D are net diversions leaving the basin (including public water and sewage). The components that are relatively easy to quantify are precipitation and streamflow. The net diversions component requires a separate, detailed water budget analysis for each municipality that relies on or interacts with IRB water resources (see section 2.5). The components that are most difficult to quantify are ET and change in storage. Previous studies by *Sammel et al.* [1966] determined that underflow at the USGS IRB gaging station is negligible and that the basin did not have long-term changes in groundwater storage. Also, because wetland extent has not changed during the last three decades, we assume no long-term change in groundwater storage, as wetlands and groundwater in the IRB are hydrologically connected. Therefore, a long-term estimate of annual ET can be obtained by difference from precipitation, streamflow and net diversions ($E = P - R - D$). We constructed a historical water budget using a monthly time-step, for the period of record 1931–1998. For each year we calculated the water budget residual, which corresponds to the sum of ET and change in storage ($P - R - D = E + \Delta S$). While on a monthly and annual time step there is large variablity in the direction and magnitude of change in storage, over a longer time period (several years) it will balance to zero. Because ΔS is uncorrelated with E (results not shown), ΔS can be treated as extra noise to E. Therefore, any long-term (e.g., decadal) trend in water budget residual reflects a long-term trend in ET. Because our particular study only looks at long-term trends, we avoid the pitfalls of ET uncertainty due to interannual changes in storage. We also calculated independent estimates of ET, using physically based models, for the period of record 1949–1998. We compared both estimates to corroborate their values. The shorter period of record for the ET model simulations (1949–1998), compared to the water budget analysis (1931–1998), is because of a limited record of meteorological input data. Data sets for calculating the water budget and for model input are presented in Table 1.

2.4. Trend Detection

[11] We tested the main water budget components for long-term trends, using a rank-based nonparametric test, Kendall's tau [*Kendall*, 1975]. We choose a nonparametric test because hydrometeorological data are usually not normally distributed. Because serial correlation in hydrometeorological time series can influence trend detection [*von Storch*, 1995], we first removed the autoregressive component for time series with significant serial correlation ($p < 0.05$) by 'prewhitening' [e.g., *Yue et al.*, 2002; *von Storch*, 1995]:

$$Y_t' = Y_t - r_1 Y_{t-1} \quad (2)$$

where Y_t' is the prewhitened data, Y_t is data at time t; and r_1 is the lag-1 serial correlation coefficient.

[12] When Kendall's test indicated a significant trend ($p < 0.01$), we calculated the slope and intercept of the trend line using the method of *Theil* [1950]. This method produces a robust estimate of a nonparametric fitted line, and is closely related to Kendall's tau [*Helsel and Hirsch*, 2002]. For prewhitened time series, we estimated the slope after correcting for autocorrelation, using a method proposed by *Yue et al.* [2002].

2.5. Net Diversions: Detailed Municipal Water Budget

[13] Most of the municipalities in the IRB are served by public water. There are currently 70 public groundwater withdrawal sites and 5 surface water withdrawal sites. Public water derived from the IRB also serves communities outside the basin. To calculate net diversions we kept track of water crossing basin boundaries through three pathways: import of public water, export of public water and export of sewage (Figure 3). Long-term historical pumping records were only available at the municipal level. Analyses were

performed for the 21 municipalities that are located partially or completely in the watershed, as well as 2 municipalities (Lynn and Salem) that are located outside the watershed boundaries, but used water from the IRB. A detailed description of 1989–1993 water withdrawal operations in the IRB have been reported in *Zariello and Ries* [2000]; their study focused on short-term effects of water withdrawals on streamflow (period of record 1989–1993).

[14] Water withdrawal data were subjected to data quality checks, including temporal analyses of per-capita water use for the different communities, and a comparison against historical water use for surrounding communities (served by the Massachusetts Water Resources Authority, MWRA). Most pumping records were deemed reliable, with the exception of surface water withdrawals during the 1980s by the town of Lynn (export of surface water was severely reduced then, while there was large inter-annual variability and few reliable records). Interpolation and extrapolation of monthly water withdrawals were based on estimated per-capita water use, while decadal U.S. Census data were used to calculate annual town populations. Import and export of public water were based on town records and GIS analyses of residential land use. Outdoor usage of residential water use was calculated using a monthly irrigation ratio, based on total MWRA system withdrawals for 1995–1999 (L. T. Paszko, MWRA, written communication, 1999). The fraction of residential areas that was sewered was estimated from town records. All sewage systems export outside the IRB. To account for leaky sewer pipes, a sewage infiltration ratio of 65% was applied to all sewer systems (i.e., 65% of sewered water is from infiltrated groundwater and 35% is from household wastewater). This value is an average for the three communities sewered by the MWRA (Wilmington, Burlington, and Reading) [*Scott*, 1999] (data source is MWRA). We assumed a constant sewage infiltration ratio for the entire period of record.

2.6. Climatological Evapotranspiration (CRAE Model)

[15] To corroborate our estimates of water-budget ET, we compared them to modeled ET estimates using the Complementary Relationship Areal Evapotranspiration (CRAE) model [*Morton*, 1983; *Morton et al.*, 1985]. The CRAE model is based on the complementary relationship in regional ET [*Bouchet*, 1963], which assumes complementarity between actual and potential ET:

$$E_A + E_P = 2E_W \tag{3}$$

where E_A is actual ET, E_P is potential ET and E_W is wet environment ET. Potential ET is calculated similar to the *Penman* [1948] equation, and wet environment ET through an adaptation of the *Priestley and Taylor* [1972] equation. See *Morton* [1983] for details.

[16] The CRAE model considers meteorological observations to reflect the interaction between evapotranspiring surfaces and the near-surface atmosphere. The complexities of the soil-vegetation-atmosphere system are avoided because the model only requires input of standard meteorological observations (solar radiation, humidity and average temperature) and standard location descriptors (elevation, latitude, time of year). The model does not require calibration. *Claessens* [1996] found overall good agreement between CRAE model ET estimates and independent water-budget ET estimates for numerous river basins across the conterminous United States.

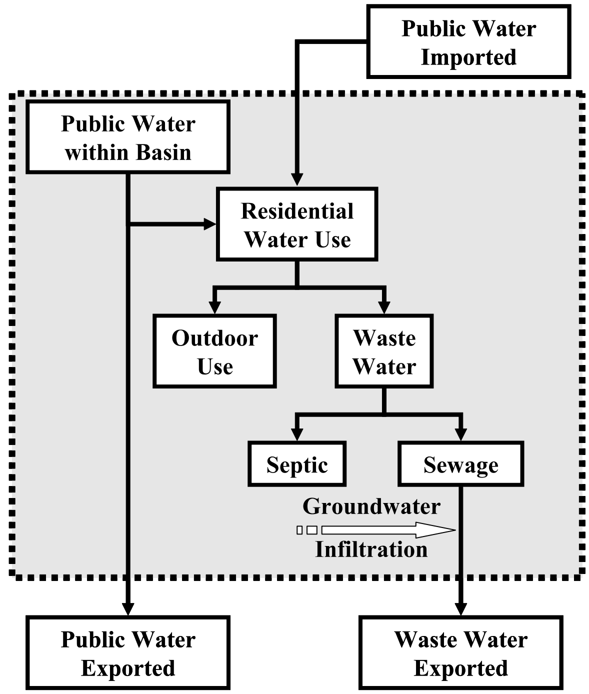

Figure 3. Schematic of municipal net diversions calculations. The shaded box represents the basin control volume for each municipality. Net diversions are the difference between public water imported, public water exported, and waste water exported. Import and export refer to the basin boundary instead of the municipal boundary.

2.7. Land Use Specific Evapotranspiration (Water-Balance Model)

[17] To evaluate the effect of land use change on ET, we developed a physically based water-balance model. The model was specifically designed for addressing differences in ET between vegetation types, and is largely built on existing models [e.g., *Federer et al.*, 1996; *Choudhury and DiGirolamo*, 1998]. Our choice of model parameterization was guided by parsimony; it has few parameters, while most parameter values are based on literature or field observations (i.e., no adjustments or calibration). The model performs detailed, daily ET calculations for vegetated and nonvegetated surfaces, based on resistance-type model formulations. We did not model snow processes, because we are mainly interested in estimating ET during the growing season (i.e., all precipitation is considered rainfall).

[18] The water-balance model can be summarized as follows. Part of the rainfall is intercepted and subjected to interception evaporation. The remaining net rainfall (i.e., throughfall) either discharges as surface runoff or recharges the soil moisture reservoir. The latter is depleted by both transpiration and soil evaporation. Excess soil moisture percolates to the groundwater reservoir, which subsequently discharges as streamflow. Net water diversions are taken from the groundwater reservoir. Where applicable (see below), irrigation of residential lawns is simulated, based on observed lawn management practices. Computational details of the water-balance model are presented in Appendix A. The

model was run at a daily time-step for the period 1949–1998. Storages were initialized using a 10-year spin-up cycle. Only one parameter was calibrated, the groundwater drainage rate constant (K_g). This parameter has no effect on modeled ET estimates. It was calibrated from deciduous forest simulations (see next), by optimizing the Nash-Sutcliffe coefficient for monthly streamflow ($R^2 = 0.71$).

[19] The water-balance model was run separately for each of 4 main vegetation types: (1) deciduous forest; (2) pasture; (3) crops; and (4) lawn. Wetlands were not simulated using the water-balance model for two reasons: (1) wetland area in the IRB did not change during our period of study, thus reducing the need for including them in land use change scenarios; and (2) our model is one-dimensional, and is therefore not suitable for modeling wetlands, which receive most of their water through lateral flow. To calculate total basin ET, we roughly estimated wetland ET using the CRAE model. With a long-term increase in precipitation (discussed later), increased water delivery to the wetlands will increase ET. Our CRAE model estimates do not display this relative increase in wetland ET, because the meteorological observations used as input into the model are from upland locations, where the atmosphere does not reflect these relatively higher ET rates. Therefore, long-term increases in wetland ET are underestimated.

[20] Land use fractional cover was obtained from 1:24,000 digital land use maps (Office of Geographic and Environmental Information (MassGIS), Commonwealth of Massachusetts Executive Office of Environmental Affairs, http://www.state.ma.us/mgis). Based on field observations and analyses of digital orthophotos for four residential areas of varying age and density, we partitioned residential land use into lawn (65%), forest (10%) and impervious (25%) land-cover. Evaporation from impervious land-cover was calculated in a separate water-balance model simulation. Model simulations were performed for both nonirrigated and irrigated lawns. The fraction of lawns that are irrigated was estimated by combining model predictions of lawn water requirements, with data on outdoor usage of residential water use (see section 2.5) and total area of lawns; we adjusted the value upward to account for the redistribution of precipitation from impervious surfaces to lawns. We estimated a current fractional lawn irrigation of 25%. Because the area of watered lawns gradually increased during the 1960s and 1970s, we assumed a linear ramp in fractional lawn irrigation during this period. Based on county-level agriculture census data, we partitioned agricultural land use into pastureland (41%) and cropland (59%). Irrigation of agricultural land is insignificant, and therefore not modeled. For land use change, we assumed that residential development first occupies abandoned agricultural land before converting forested land. Soil types in the watershed are equally distributed between loamy sand and fine sandy loam, based on an analysis of STATSGO soils data [*U.S. Department of Agriculture*, 1991]. In our study we used a single soil type for the entire watershed, sandy loam. Sensitivity analyses were conducted on several key parameters (Appendix B).

3. Results and Discussion

3.1. Net Diversions

[21] Diversions from the IRB have been large (Figure 4). Earlier in the record, diversions were mainly through surface-water withdrawals to supply communities located primarily outside the IRB (Lynn, Salem and Beverly). These withdrawals remained fairly constant until the 1960s, and increased through the 1970s. During the 1980s, Lynn discontinued withdrawal operations, causing a significant variation and drop in total water export from the IRB. Consequently, the current total export of public water is below its 1930s level, but is increasing. While surface-water withdrawals have dropped, groundwater withdrawals have increased (in 1998, 67% of public water derived from the IRB was from groundwater, compared to 17% in 1931; results not shown). Per capita water-use increased about 50%, mainly during the 1960s and 1970s (results not shown). This period is associated with a rapid increase in residential development and lawn watering practices. Starting in the late 1950s several communities in the IRB switched from septic to sewered systems that discharge outside the IRB. Over the last 40 years this export of sewered wastewater has become an important component, currently accounting for about 45% of the total diversions from the IRB. Net diversions have increased substantially since the 1930s. By the late 1970s, net diversions had increased by 75% above the 1930s levels, after which it dropped and slightly rebounded again in the 1990s, to an increase of 40% above 1930s levels. Overall, net diversions constitute about 10% of total precipitation or about 20% of total streamflow.

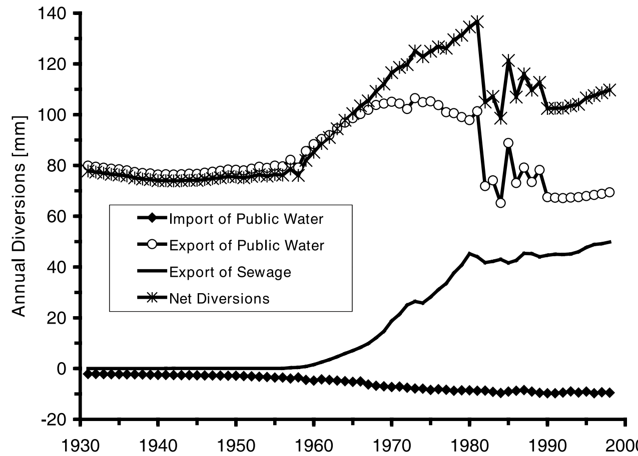

Figure 4. Annual totals of the main diversion components for the Ipswich River basin. Export (positive values) and import (negative values) refer to basin boundaries. Net diversions refer to water leaving the basin boundaries, and are calculated as sewage exported plus public water exported minus public water imported. The irregular pattern during the 1980s is due to Lynn discontinued withdrawal operations.

3.2. Water Budget

[22] Over the long-term, water-budget residual reflects ET (see section 2.3); hereafter we refer to water-budget ET instead. On a long-term annual scale, precipitation and streamflow are highly variable (Figure 5). Precipitation increased at an average rate of 2.7 mm per year ($p < 0.01$) (Table 2). Neither streamflow nor runoff ($R + D$) displays a significant trend. Water-budget residual (i.e., ET) ($P - R - D$) increased at an average rate of 1.7 mm per year

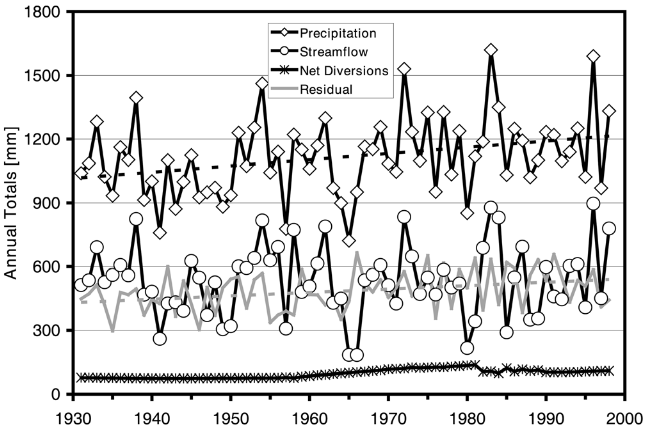

Figure 5. Annual totals of the main water budget components for the Ipswich River basin; precipitation (*P*), streamflow (*R*), net diversions (*D*), and water-budget residual (*P-R-D*). Over the long-term, water-budget residual reflects ET (see section 2.3). Both precipitation and water-budget ET display significant, positive trends (p < 0.01); corresponding Theil trend lines are shown.

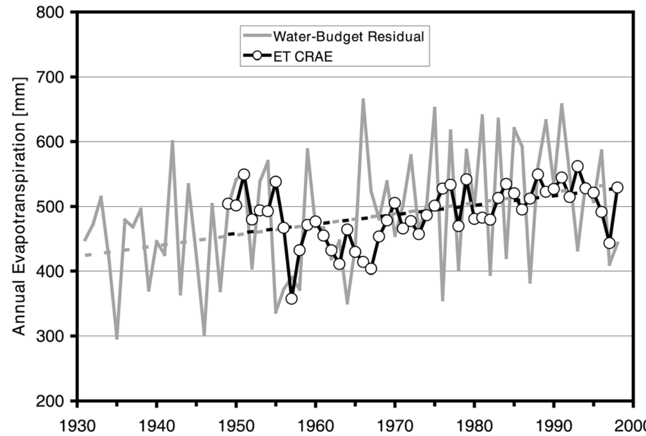

Figure 6. Long-term annual ET, comparing the empirical water-budget residual (*P-R-D*) against the CRAE-model ET estimate. Over the long-term, water-budget residual reflects ET (see section 2.3). Both ET estimates display a similar, significant, positive trends (p < 0.01); corresponding Theil trend lines are shown.

(p < 0.01). Comparing the average annual values of 1998 to 1931, this amounts to a precipitation increase of 187 mm (18%) and an ET increase of 113 mm (27%). The trend in precipitation and ET may be a short-term increase in a cyclic climate pattern and therefore not necessarily an indication of long-term ''climate change''; See *Potter* [1976] and *NERA* [2001] for longer-term climate assessments. For precipitation, we also performed trend analyses on a monthly basis (results not shown here). October and December displayed positive trends at medium significance (p < 0.05).

[23] We corroborated the long-term water-budget ET estimates, by comparing them to ET estimates predicted with the CRAE model. The two independent estimates of ET compare well over the long-term, having almost identical magnitude and trend (Figure 6); in both estimates, ET for 1949-1998 increased significantly at a rate of 1.5 mm per year (p < 0.001) (Table 2).

[24] The ratio of runoff to precipitation (i.e., the runoff ratio) and ET to precipitation did not change (results not shown; runoff ratio is ~56%). Therefore, the average partitioning of precipitation between runoff and ET did not change, despite significant precipitation increases. This finding is important because it allows for simple projection of average runoff response to change in precipitation (see 3.6).

Table 2. Trend Analysis Results for Main Water Budget Components (Theil Slope, Kendall's Probability, and Lag-1 Serial Correlation Coefficient)[a]

	Theil Slope, mm/yr	Kendall P	AR r_1	Note
Water Balance (1931–1998)				
Precipitation (*P*)	**2.7**	0.006	0.06	
Streamflow (*R*)	0.3	0.379	0.12	n.s.
Net diversions (*D*)	**0.7**	<0.001		
Runoff (*R* + *D*)	1.1	0.114	0.13	n.s.
Residual ($E + \Delta S = P\text{-}R\text{-}D$)	**1.7**	0.005	−0.10	
CRAE Model (1949–1998)				
Evapotranspiration (*E*)	**1.5**	<0.001	0.55	prewhitened

[a]Note: Bold slopes indicate significant trend (p < 0.01). Serial correlation coefficient is for original time series (i.e., before prewhitening). Net diversions were not prewhitened.

3.3. Masking Effect

[25] Generally, the conversion of forest to residential land cover leads to a decrease in ET and a corresponding increase in runoff [e.g., *Dow and DeWalle*, 2000]. This decrease in ET is primarily due to a decrease in interception evaporation and an increase in impervious area. Why then has IRB ET increased instead of decreased? We suspect the 50-year trend in climate is masking the effect of land use change.

3.4. Climate Effect on Evapotranspiration

[26] Apart from precipitation, we assessed how other climate variables have changed in the IRB. During the 1950s the estimated amount of solar radiation reaching the land surface decreased, after which it remained relatively constant (Figure 7a). While maximum and average temperature remained fairly constant throughout the period of record, minimum temperature slightly decreased starting in the 1950s (Figure 7b). During this same period, dewpoint temperature slightly increased, and minimum temperature and dewpoint temperature gradually converged. This pattern is an indication that the near-surface atmosphere gradually became more humid, perhaps because of a change in regional circulation patterns or in precipitation and a resulting increase in ET. An increase in residential lawn watering might have played a role as well.

[27] To assess the climate effect on ET, we used the CRAE model. An important aspect of the CRAE model is the underlying assumption that meteorological observations are a reflection of the interaction between regional evapotranspiring surfaces and the overlying atmosphere. For example, dry air is indicative of low ET rates, and vice versa for humid air. Therefore, the three components of the complementary relationship (equation (3)) should display a distinct response to changes in climate drivers. Wet envi-

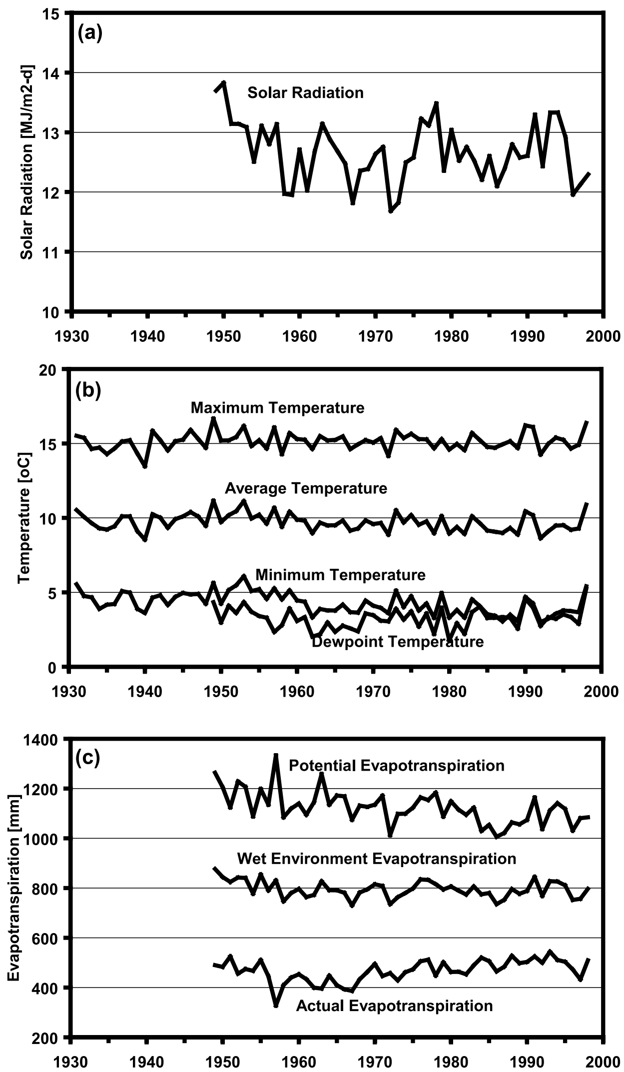

Figure 7. (a) Annual average daily solar radiation; (b) annual average daily temperatures, including maximum, minimum, average, and dewpoint temperature; (c) CRAE-model estimates of annual ET, including potential, wet environment, and actual ET.

ronment ET has remained fairly uniform throughout the record (Figure 7c), a direct reflection of a similar lack of trend in its main driver, solar radiation. Potential ET on the other hand, declined significantly. This decline is directly related to the gradual increase in humidity of the air. With a significant decrease in potential ET and no change in wet environmental ET, the CRAE estimate of actual ET has significantly increased over time. This is consistent with the complementary relationship in regional ET, with increased humidity being indicative of higher ET rates.

3.5. Land Use Change Effect on Evapotranspiration

[28] We assessed the effect of land use change using our physically based water-balance model (Figure 8a). Forest ET displays a long-term positive trend (0.9 mm per year; $p < 0.05$), while the positive trend for agriculture is not significant (0.8 mm per year; $p = 0.15$). Residential land use initially displays a lower ET rate compared to forest and agriculture. However, with increase in lawn watering, residential ET rates go up, and surpass rates from forest and agriculture during years of low soil moisture availability. Overall, forests have higher ET rates compared to the nonirrigated shorter-stature vegetation types. This pattern can be attributed to the importance of the interception component in forests, as well as to the deeper rooting depth. Interception evaporation accounts for 22% of total ET for forest, 13% for agriculture and 3% for lawn (results not shown). Irrigated lawn has the highest value for transpiration, due to high moisture supply.

[29] Next, we combined the per-unit-area ET estimates (Figure 8a) with their associated land use fractional cover (Figure 2). During the period of record, basin ET from both forest and agriculture decreased, while basin ET from residential increased (Figure 8b). The figure also shows an estimate of total basin ET, produced by summing basin ET from forest, agriculture, residential and wetlands. Recall that for wetlands the long-term increase in ET is underestimated (Section 2.7). This explains why the trend in total basin ET from the water-balance model (1.0 mm per year; $p < 0.05$) is lower than the trend in ET from the empirical water-budget (1.5 mm per year; 1949–1998).

[30] To assess the effect of a change in land use, separate from the change in climate, we simply compared water-balance model simulations for two land use change scenarios. The first scenario corresponds to the observed historical (1949–1998) land use trajectory (Figure 8b), i.e., conversion of forest and agriculture land to residential. The second

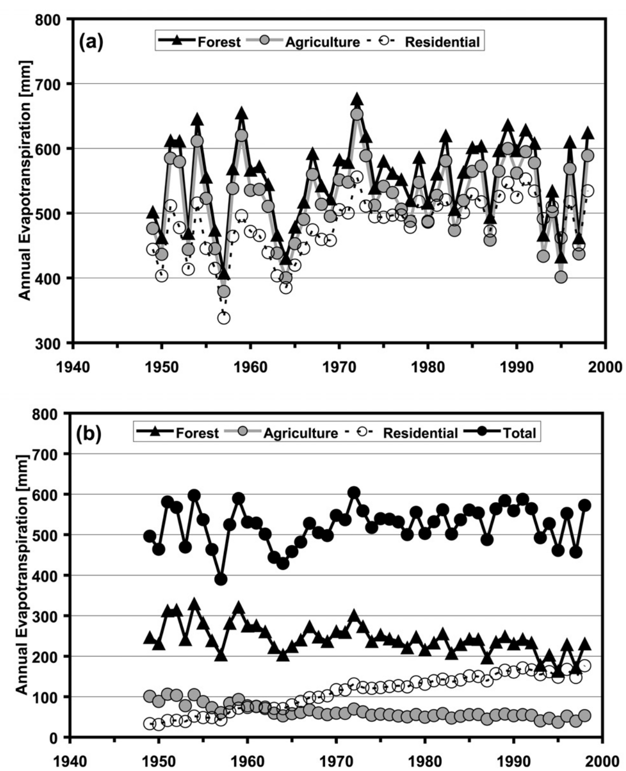

Figure 8. (a) Water-balance model estimates of annual ET for the upland land use types (values per unit area); (b) basin-totals of above estimates. Also shown is the total basin estimate of ET (which includes wetlands; see text for detail).

scenario is a nonconversion scenario, which maintains the 1949 land use during the entire simulation. When comparing results from the two simulations, land use conversion has a negligible effect on total basin ET (Figure 9). During the first part of the period of record, land use conversion produces a slight reduction in ET, because of relatively low ET from lawns and impervious areas. Later in the record this reduction disappears because of the effect of lawn watering, which results in relatively high ET during periods of low soil moisture availability (compared to forest and agriculture). Our estimated fractional lawn irrigation of 25% is most likely an upper bound value; actual irrigation fraction might be lower during extended periods of droughts, as some municipalities introduce watering restrictions. Though not common [e.g., *Dow and DeWalle*, 2000], increases in ET associated with residential development have been found in a few other studies [e.g., *Grimmond and Oke*, 1986; *Stephenson*, 1994].

[31] Overall, our results suggest that the long-term increase in basin ET can almost entirely be attributed to the trend in climate, while change in land use has had little effect. Our results also indicate that the land use change effect is not being masked by the climate trend per-se, because land use change did not lead to a significant reduction in ET. As part of a sensitivity analysis (Appendix B), we evaluated the effect of independently changing parameter values (within range ± 50%), including depth of soil layer, maximum leaf area index and fractional lawn irrigation. Our conclusions are robust with respect to these ranges of parameter values.

3.6. Addressing Both Climate and Land Use Change Effects

[32] The results from our historical analysis can be generalized to predict the effect of future changes in climate and land use on long-term streamflow. We compared two climate change projections for the 21st century, reported by the VEMAP Data Group [*Kittel et al.*, 2000]: the Canadian Centre for Climate Modelling and Analysis model and the Hadley Centre model. We adjusted the 0.5° gridded VEMAP data to the IRB, based on anomalies from the historical VEMAP record and IRB climatology. For the 21st century, the Hadley model projects a significant increase in precipitation (2.2 mm per year; $p < 0.001$), while the Canadian model indicates no trend (results not shown). Compared to 1998, this corresponds to a precipitation increase of 19% for the Hadley model. Because the 1931–1998 historical water budget showed a significant increase in precipitation (18%) but no change in runoff ratio, we simply assumed the runoff ratio to remain unchanged at 56%. This is a conservative estimate for ET, which most likely will increase with increases in temperature. An alternative approach would be to use VEMAP data as input into an ET model. However, an analysis of the historical VEMAP record revealed problems with the humidity data, which in the case of the IRB results in an erroneous long-term ET trend (L. Claessens, unpublished data). Therefore, we based evapotranspiration on the above empirical runoff-ratio approach instead. Net diversions can be estimated from residential land use; the detailed net diversion budget for the 1990s shows a significant positive trend between net diversions and residential land use (3.2 mm per % residential land use) (results not shown). Land use

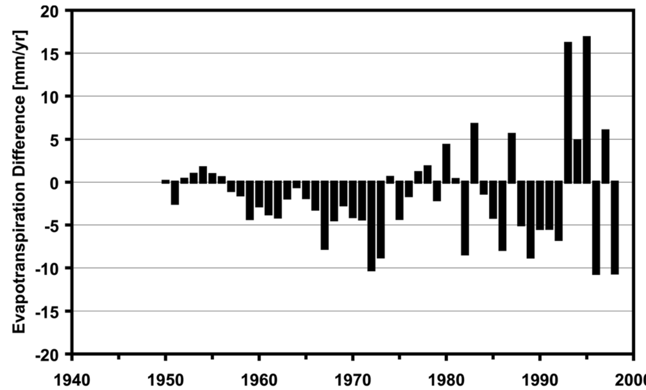

Figure 9. Difference in total annual ET between change (1949–1998) and no-change (1949) land use scenarios. Positive values indicate increased ET due to land use change, negative values vice versa.

change projections for the IRB for the 21st century are reported by *Pontius et al.* [2000], based on a GIS-based land use change model. The model projections show an increase in residential land use from 33.2% in 1998 to 50.5% in 2101. By combining the above changes in climate and land use, and assuming no long-term change in storage, we predict streamflow from equation (1). Results show that, compared to 1998, by the end of the 21st century the average change in streamflow ranges between −10% (Canadian model) to +12% (Hadley model). However, because of our conservative estimate for ET, the change in average streamflow most likely will be more negative. Interestingly, net diversions play a large role in these estimates. If net diversions would have remained at their 1998 levels, by the end of the 21st century the average change in streamflow would range between 0% (Canadian model) to +22% (Hadley model). This illustrates why it is important to account for net diversions when predicting the effect of future changes in climate and land use on streamflow.

4. Conclusions

[33] We presented an assessment of the effects of historical changes in both land use and climate on the water budget of a rapidly urbanizing watershed. Our study illustrates the importance of addressing an important activity associated with land use change: basin diversions. Diversions are a general problem in suburban watersheds, where there can be significant cross-basin transfer of public water and/or sewered water. In the IRB, net diversions increased ~40% (mostly due to groundwater pumping and export of sewage) and currently constitute a major component of runoff (20% of streamflow). Without a detailed accounting of this water budget component, inferences regarding streamflow or ET response to changes in land use or climate could be invalid. For our historical analysis, not accounting for diversions would have overestimated water budget ET (both magnitude and trend). Conversely, overestimating increases in diversions would not have shown the ET trend. Accuracy of data is important; because we were able to corroborate long-term ET estimates (both magnitude and trend) from empirical (water budget) and model (CRAE) approaches, we have confidence in our results. For future

predictions, diversions are important as well. Our results show that not accounting for future changes in diversions could overpredict streamflow by ∼10%.

[34] Our analysis mostly addressed annual patterns. Urbanization impacts streamflow at shorter timescales as well. For example, *Zariello and Ries* [2000] found that groundwater withdrawals in the IRB decreased summertime lowflows. Their short-term study did not address climate-trend effects. Apart from changes in ET, urbanization also leads to an increased demand on public water resources. Our results indicate that on an annual average basis, the increase in net diversions has been negated by increased precipitation. However, seasonal and shorter-term effects are important as well. For example, did the precipitation increase partially offset summertime low-flows? As such, evaluating impacts from changes in both land use and climate is not only important for understanding long-term average response, but is particularly important for understanding responses at shorter timescales.

[35] We showed that land use change had a negligible effect on ET, while the increase in ET could be attributed entirely to the change in climate. This lack of a land use change effect is different from ET trends commonly associated with urbanization [e.g., *Dow and DeWalle*, 2000]. Forest-residential conversion generally leads to a significant decrease in ET, whereas in the IRB the change in ET due to land use change alone was found to be insignificant. This finding is specific for our study area. Land use change effects may be more important in locations where urbanization leads to significant changes in consumptive use. For example, medium- to low-density residential developments with lawn watering in semi-arid areas should increase ET; conversely, the conversion of forests to high density residential generally decreases ET (see also *Grimmond and Oke* [1986]). This refers to land use change effects only; changes in climate could amplify or reverse these effects, as we showed in this study.

[36] Our study illustrates that to understand effects of changes in either land use or climate on watershed hydrology, both effects have to be addressed in tandem. For example, in the IRB the lack of streamflow response to historical increases in precipitation could easily be attributed to an increase in net diversions associated with urbanization. However, our results showed that a change in climate played an important role in this lack of streamflow response, through inducing an increase in ET. Thus, to understand future effects of urbanization on streamflow, one has to consider not only changes in diversions, but also changes in ET as a result of changes in both land use and climate. The latter is a complex task, and generally requires a coupled land-atmosphere model, accounting for feedbacks between the evapotranspiring surface and the atmosphere, as well as for local and regional forcing effects. For our suburban watershed, the combined empirical/modeling approach we employed was well suited for assessing past effects and for predicting possible future consequences. We addressed several potentially important components of the suburban water cycle (e.g., export of public water and sewage, change in land cover and ET). To apply this approach to other areas, other aspects might have to be addressed that could be important in those areas, but were not important in the IRB (e.g., changes in groundwater levels and surface water storage). Being the first study of a suburban watershed that addresses long-term effects of changes in both land use and climate, and accounts for diversions and other unique aspects of suburban hydrology, our study could serve as a framework for similar assessments.

Appendix A: Water Balance Model—Computational Details

[37] The water-balance model is largely built on existing models, mainly *Federer et al.* [1996] and *Choudhury and DiGirolamo* [1998]. The reader is referred to these and other sources for additional computational details and corresponding units.

A1. Soil Drainage and Groundwater Discharge

[38] The soil moisture and groundwater reservoirs are updated at the end of each time step:

$$\begin{aligned} \frac{dS}{dt} &= P - I - Q_s - E - Q_d \\ \frac{dG}{dt} &= Q_d - Q_g - D \end{aligned} \tag{A1}$$

where S and G are the soil moisture and groundwater reservoir, respectively; P is precipitation (including irrigation); I is interception; Q_s is surface runoff; E is evapotranspiration; Q_d is soil drainage (i.e., percolation to groundwater); Q_g is groundwater discharge; and D is net diversions. Storage units are in mm; fluxes in mm/day.

[39] The soil moisture reservoir has a fixed maximum water holding capacity, prescribed by the soil layer depth (D_s [mm]) and the maximum available soil moisture content between field capacity (θ_f [mm^3/mm^3]) and wilting point (θ_w [mm^3/mm^3]). Where applicable, irrigation of residential lawns is simulated when available soil moisture drops below 10%, by raising soil moisture to field capacity. The 10% low-moisture limit was chosen to reduce moisture stress and produced irrigation frequencies in accordance with observed lawn watering practices. Other land-cover types are not irrigated.

[40] Soil drainage (Q_d) from the soil reservoir to the groundwater reservoir is estimated at hourly time step from soil hydraulic conductivity, calculated using Campbell's equation [*Rawls et al.*, 1993] (used in *Choudhury and DiGirolamo* [1998]):

$$Q_d = K(\theta) = K_{sat}\left(\frac{\theta}{\theta_p}\right)^{3+\frac{2}{\lambda}} \tag{A2}$$

where $K(\theta)$ is hydraulic conductivity [mm/h], K_{sat} is saturated hydraulic conductivity [mm/h], θ is soil moisture content [mm^3/mm^3], θ_p is porosity [mm^3/mm^3] and λ is pore size distribution index.

[41] Surface runoff (Q_s) is calculated using an analogy of the SCS curve number equation [*Schaake et al.*, 1996] (used in *Choudhury and DiGirolamo* [1998]):

$$Q_s = \frac{(P-I)^2}{(P-I)+S_d} \tag{A3}$$

Table B1. Vegetation-Specific Parameters for the Water-Balance Model[a]

	Deciduous Forest	Grassland	Crops	Lawn
L_{mx} [-]	4.4[b]	3.0[c]	3.0[c]	2.9[d]
H [m]	17[e]	0.5[e]	0.3[e]	0.12[d]
G_{lmx} [m/s]	0.0053[c]	0.0080[c]	0.0110[c]	0.0143[d]
G_{lmn} [m/s]	0.0003[c]	0.0003[c]	0.0003[c]	0.0003[c]
Z_{0G} [m]	0.020[c]	0.010[c]	0.005[c]	0.010[c]
D_s [m]	1.0[f]	0.8[g]	0.8[g]	0.25[g]
A [mm]	1.02[h]	0.42H[h]	0.42H[h]	0.42H[h]
B [-]	0.18[h]	0.26H[h]	0.26H[h]	0.26H[h]
N [-]	1[h]	1[h]	1[h]	1[h]

[a] L_{mx} is maximum leaf area index; H is height of vegetation; G_{lmx} is maximum leaf conductance; G_{lmn} is minimum leaf conductance; Z_{0G} is roughness parameter for ground; D_s is depth of soil layer; and a, b, and n are interception model parameters.
[b] Average of maximum LAI for established oak stands in Harvard Forest (source: B Felzer using data from C Barford, Harvard Forest NIGEC).
[c] From *Federer et al.* [1996].
[d] From *Jensen et al.* [1990, Table 6.3].
[e] Estimated (based on Harvard Forest data).
[f] From *Aber and Federer* [1992].
[g] Estimated (from various sources).
[h] From *Bras* [1990, Table 5.11].

where S_d is the infiltration capacity or soil moisture deficit ($S_d = D_s(\theta_p - \theta)$).

[42] Groundwater discharge (Q_g) to the stream is calculated as a linear reservoir:

$$Q_g = K_g D_g \tag{A4}$$

where K_g is groundwater drainage rate constant [day^{-1}] and D_g is the equivalent depth of water stored in the groundwater reservoir [mm].

A2. Canopy Phenology and Interception

[43] Forest regrowth is based on field observations from nearby New England forests, indicating about five years till maximum canopy cover in early stand development [*Bormann and Likens*, 1979]. Consistent with these observations, leaf area index is calculated using a linear ramp during years 1–4, reaching its maximum value at year 5. Leaf onset in the spring is based on a degree-day method [*Goulden et al.*, 1996]; after calendar day 100, degree-days are calculated by summation of average daily temperatures >0°C; leaf onset starts at degree-day 300 and ramps linearly to degree-day 650, after which leaf area index (L) is set at its maximum value (L_{mx}). In the fall, leaf drop is controlled by minimum temperature (M. Williams, personal communication, 2000); after calendar day 250, once minimum temperature < 10°C, leaf drop occurs linearly over the next 44 days.

[44] Fractional vegetation cover (F_{veg}) is estimated as [*Choudhury and DiGirolamo*, 1998]:

$$F_{veg} = 1 - \exp(-0.67L) \tag{A5}$$

Interception (I) is calculated using the equation [*Bras*, 1990]:

$$I = MIN[P, a + bP^n] \tag{A6}$$

where a, b and n are model parameters. These parameters are a function of vegetation type and precipitation characteristics. Interception is expressed per unit fractional vegetation cover.

A3. Evapotranspiration

[45] Evapotranspiration is calculated as a combination of interception evaporation, transpiration from vegetation and soil evaporation. Evaporation from impervious cover is calculated similar to potential interception evaporation. The following evapotranspiration equations are expressed per unit fractional cover (either vegetation or soil), except for total evapotranspiration (equation (A12)).

[46] Both potential interception evaporation and potential transpiration are calculated using the Penman-Monteith combination equation [*Monteith*, 1981]:

$$\lambda E = \frac{\Delta(R_n - G) + \rho c_p (e_s - e_a) r_a^{-1}}{\Delta + \gamma(r_a + r_c) r_a^{-1}} \tag{A7}$$

where λ is latent heat of vaporization; Δ is the gradient of the saturation vapor pressure curve; γ is the psychrometric constant; R_n is net radiation; G is ground heat flux; ρ is density of air; c_p is specific heat of moist air; e_s is saturation vapor pressure; e_a is actual vapor pressure; r_a is the aerodynamic resistance; and r_c is the canopy resistance. Net radiation calculations are based on the CRAE model [*Morton*, 1983] for shortwave radiation and *Brutsaert* [1982, chapter 6] for longwave radiation. Ground heat flux is assumed to be negligible, and has been ignored in further calculations.

[47] Potential interception evaporation (E_{Pincp}) is calculated using equation (A7), without canopy resistance (i.e., r_c = 0). Aerodynamic resistance calculations are based on *Federer et al.* [1996]. Actual interception evaporation (E_{incp}) is limited by the amount of intercepted rainfall:

$$E_{incp} = MIN\left[E_{Pincp}, I\right] \tag{A8}$$

[48] Potential transpiration is calculated using equation (A7). Both aerodynamic resistance and canopy resistance calculations are based on *Federer et al.* [1996]. Their

Table B2. Soil-Specific Parameters for the Water-Balance Model[a]

	Sandy Loam
θ_p [mm^3/mm^3]	0.45[b]
θ_f [mm^3/mm^3]	0.21[b]
θ_w [mm^3/mm^3]	0.10[b]
λ [-]	0.38[b]
K_{sat} [cm/h]	2.18[b]
ψ_f [MPa]	0.01[c]
ψ_w [MPa]	2.5[c]
K_g [day^{-1}]	0.03[d]

[a]θ_p is porosity; θ_f and θ_w are moisture content at field capacity and wilting point, respectively; λ is pore size distribution index; K_{sat} is saturated hydraulic conductivity; ψ_f and ψ_w are water potential at field capacity and wilting point, respectively; and K_g is groundwater drainage rate constant.
[b]From *Rawls et al* [1993].
[c]Assumed.
[d]Calibrated (see text).

formulation was designed for nonlimiting soil moisture conditions, i.e., potential transpiration. To calculate actual transpiration (E_{trns}) we multiply potential transpiration (E_{Ptrns}) by a soil moisture reduction factor (F_θ):

$$E_{trns} = F_\theta E_{Ptrns} \tag{A9}$$

$$F_\theta = \begin{cases} 0 & \theta \leq \theta_w \\ 1 - \left(\dfrac{\Psi_f}{\Psi_w}\right)^{\frac{\theta-\theta_w}{\theta_f-\theta_w}} & \theta_w < \theta \leq \theta_f \\ 1 & \theta > \theta_f \end{cases} \tag{A10}$$

where Ψ_f and Ψ_w are the water potential at field capacity and wilting point, respectively.

[49] We estimate a transpiration factor (F_{trns}) to adjust for reduced transpiration rates when leaves are wet from intercepted rainfall [*Choudhury and DiGirolamo*, 1998]:

$$F_{trns} = 1 - F_{incp}\left\{\frac{E_{incp}}{E_{Pincp}}\right\} \tag{A11}$$

where F_{incp} is the daytime fractional interception [-]. Values for daytime fractional interception were derived from modeled interception estimates for a deciduous forest, using 1980–1997 summertime hourly precipitation data for a station located in the IRB.

[50] Soil evaporation (E_{soil}) is calculated using an approach reported by *Choudhury and DiGirolamo* [1998], which distinguishes between energy limited and exfiltration limited rates of soil evaporation. The energy-limited rate uses an adaptation of the *Priestley and Taylor* [1972] equation, while the exfiltration limited rate is based on *Philip* [1957].

[51] Total evapotranspiration (E) for each vegetation type is calculated as:

$$E = F_{veg}E_{incp} + F_{veg}F_{trns}E_{trns} + \left(1 - F_{veg}\right)E_{soil} \tag{A12}$$

Appendix B: Water Balance Model—Sensitivity Analysis

[52] We evaluated the sensitivity of ET and the sensitivity of climate effect on ET, to changing parameter values. Three key parameters were analyzed, including depth of soil layer, maximum leaf area index, and fractional lawn irrigation (ET was found to be most sensitive to these parameters). The

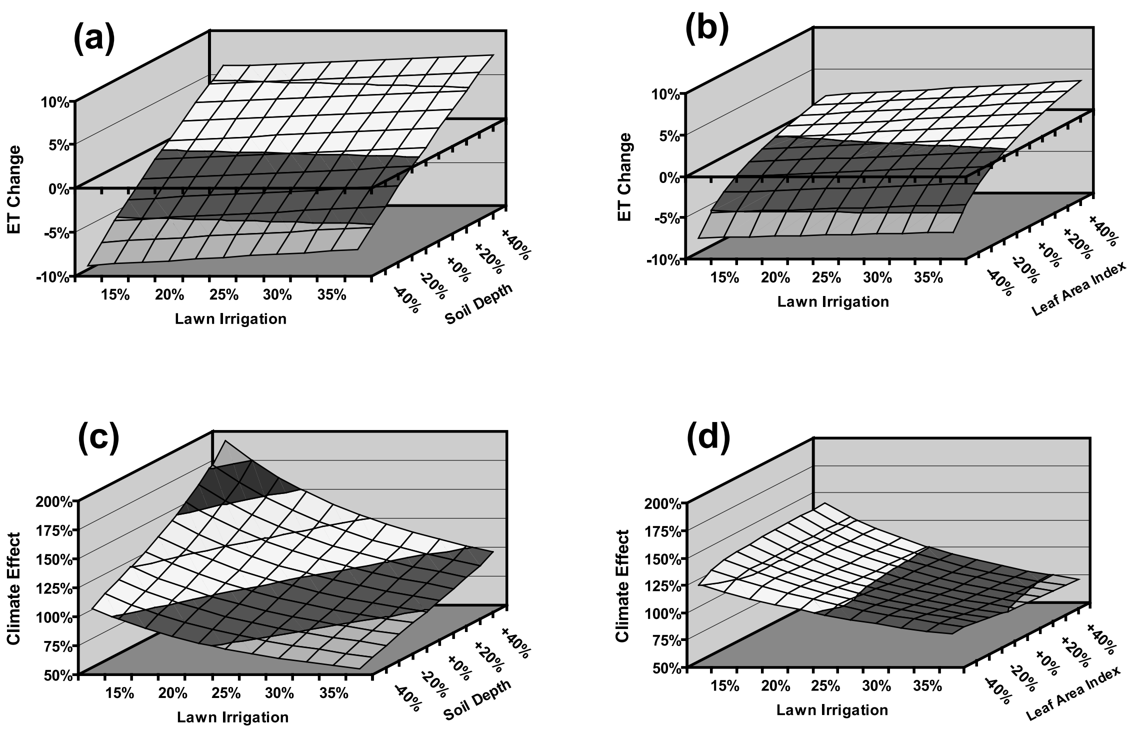

Figure B1. Sensitivity analysis of: (a) soil depth and fractional lawn irrigation on basin ET; (b) leaf area index and fractional lawn irrigation on basin ET; (c) soil depth and fractional lawn irrigation on climate effect; and (d) leaf area index and fractional lawn irrigation on climate effect.

reported set of parameter values was used as a baseline (Tables B1 and B2). We performed sensitivity analysis within ±50% of these values (using 10% intervals). However, when interpreting these results, one should keep in mind that the expected range of values probably falls within ±25%. We independently changed the parameter values, using a 2-way combination. We did not change parameter values independently for the different vegetation types; instead, we adjusted each vegetation type similarly (e.g., 50% decrease in leaf area index for all vegetation types). We analyzed the sensitivity on basin ET and climate effect on ET. For basin ET, we report percentage change in ET. Climate effect on ET is estimated as:

$$1 - \frac{S[E_{1949-1998}] - S[E_{1949}]}{S[E_{1949-1998}]} \tag{B1}$$

where $S[]$ is the slope of the trend line; $E_{1949-1998}$ is ET with land use change (i.e., with 1949–1998 land use); and E_{1949} is ET without land use change (i.e., with 1949 land use). A low value for climate effect indicates that land use change has a strong effect on ET. A value over 100% suggests that the land use change effect is being masked by a climate trend.

[53] Basin ET shows a low-level sensitivity to fractional lawn irrigation (Figures B1a and B1b). There is a high-level sensitivity to soil depth, but it is mostly within ±5% for the range of soil depths (Figure B1a). There is a medium-level sensitivity to leaf area index (LAI), also mostly within ±5% for the range of LAIs (Figure B1b). Overall, basin ET falls within ±5% for the given parameter space.

[54] Climate effect on ET shows a medium- to high-level sensitivity to fractional lawn irrigation, depending on soil depth (Figure B1c). For large soil depths, the climate effect increases rapidly with increasing fractional lawn irrigation (the land use effect decreases because lawns are kept longer at nonoptimal soil moisture conditions). Similarly, there is a medium- to high-level sensitivity to soil depth, depending on fractional lawn irrigation (Figure B1c). The climate effect is small (i.e., the land use effect is large) for high fractional lawn irrigation with small soil depths (frequent lawn watering maintains optimal soil moisture conditions). There is a low-level sensitivity to LAI, regardless of fractional lawn irrigation (Figure B1d). Overall, for the expected range of parameter values, the climate effect falls within 75%–125%, indicating that the effect of land use change on ET is small and/or being masked by the climate effect.

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