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Effects of land cover on water table, soil moisture, evapotranspiration, and groundwater recharge: A Field observation and analysis

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Abstract

The effects of land cover on water table, soil moisture, evapotranspiration, and groundwater recharge were studied with water level measurements collected from two monitoring wells over a period of 122 days. The two wells were installed under similar conditions except that one was drilled on the east side of a creek which was covered with grass, and the other on the west side of the creek which was burned into a bare ground. Substantial differences in water level fluctuations were observed at these two wells. The water level in the east grass (EG) well was generally lower and had much less response to rainfall events than the west no-grass (WNG) well. Grass cover lowered the water table, reduced soil moisture through ET losses, and thus reduced groundwater recharge. The amount of ET by the grass estimated with a water table recession model decreased exponentially from 7.6 mm/day to zero as the water table declined from near the ground surface to 1.42 m below the ground surface in 33 days. More groundwater recharge was received on the WNG side than on the EG side following large rainfall events and by significant slow internal downward drainage which may last many days after rainfall. Because of the decreased ET and increased R, significantly more baseflow and chemical loads may be generated from a bare ground watershed compared to a vegetated watershed.

1. Introduction

It is well recognized that land cover and land use change have significant effects on hydrological processes such as evapotranspiration (ET), soil moisture and groundwater recharge (Hillel, 1998; Rodriguez-Iturbe, 2000; Eagleson, 2002). Recent climate-soil-vegetation modeling (e.g. Rodriguez-Iturbe et al., 1999; Rodriguez-Iturbe, 2000; Laio et al., 2001; Guswa et al., 2002) suggest that given the same soil type under vegetated and bare soil conditions, vegetated soils with soil moisture losses from ET retain more infiltrating precipitation than bare soils with soil moisture loss from evaporation alone. Hence, vegetated soils would produce less groundwater recharge than bare soils.

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In application, differences in hydrological processing among land cover types can be used as strategy for reducing transport of pollutants to streams. For example, along stream corridors in agricultural watersheds, the use of perennial vegetation as a riparian buffer to scavenge excess water and nutrients from annual row crop fields has been long recognized (Pettyjohn and Correll, 1983; Hill, 1996; Correll, 1997; Cey et al., 1999; Lee et al., 2003). On a watershed scale, recent studies have suggested that adding perennial plants of substantial portions of agricultural landscapes offers promise for improving water quality (Nassauer et al, 2002; Coiner et al., 2001; Vaché et al., 2002). In the Walnut Creek watershed in Jasper County, Iowa, conversion of 33.7% of a 5217 ha watershed from row crop to native prairie has resulted in a 0.0028 mg/l per week decrease in stream nitrate concentrations (Schilling, 2002) and a measurable decrease in stream baseflow (Schilling, 2002). Randall et al. (1997) found that nitrate concentrations in drainage water from alfalfa and perennial grasses were 35 times lower than drainage water from corn and soybean fields.

One important benefit of using perennial vegetation in agricultural settings to reduce non-point source pollution loads relies on the fact that perennial cover increases evapotranspiration (ET) as compared to annual row crops. Perennial vegetation transpires throughout the spring, summer and fall, whereas substantial transpiration from row crops typically does not occur until mid-growing season. Vulnerable leaching periods occur in the spring and fall because crop uptake in Midwestern row crop production is not particularly well timed for utilizing available precipitation (Dinnes et al., 2002). Hence, maximizing water uptake by perennial vegetation in row crop fields can be used as an important nutrient control strategy for reducing nonpoint source pollution loads in the agricultural Midwest.

Historical evidence from Iowa illustrates how removal of perennial vegetation from an agricultural landscape profoundly affected stream flow characteristics and nitrate concentrations over the 20th century. Baseflow and the percentage of stream flow as baseflow significantly increased in Iowa over the second half of the 20th century, more than precipitation alone can explain (Schilling and Libra, 2003; Schilling and Zhang, 2003). Schilling and Libra (2003) hypothesized that one of the main reasons for increasing baseflow in Iowa over the 20th century was converting previously untilled land or other perennial cover crops to annual row crops. Increasing baseflow was found to be significantly related to increasing row crop intensity (Schilling, 2005). Because nitrate–nitrogen (nitrate) is primarily delivered to Iowa streams through baseflow discharge and tile drainage (Hallbertg, 1987; Schilling, 2002), changing watershed hydrology to more baseflow has the potential to deliver more nitrate to streams. In conjunction with the land use change in Iowa, a 2- and 3-fold increase in nitrate–nitrogen concentrations has been observed in the Cedar and Des Moines rivers during the 1940–2000 period (Iowa Geological Survey, 2001).

Recently, we observed the effects of land cover on a shallow groundwater table. For a 122-day period in the late summer and fall of 2003, continuous water level measurements made in two monitoring wells, one is located on bare ground and the other under dense grass cover, revealed substantial differences in water table behavior. The purposes of this paper are to present and describe the observed data, to analyze the effects of grass cover on water table, soil moisture, ET, and groundwater recharge, and to discuss the effects of grass cover on the basin-scale water cycle and nonpoint source pollution loads. Results from this study have implications for water cycle research and for utilizing perennial vegetation in agricultural watersheds for management of nutrient losses.

2. Site description and land treatment

Our study site is situated in the central portion of the Walnut Creek watershed at the Neal Smith National Wildlife Refuge (NSNWR) in Jasper County Iowa (Fig. 1). This site was chosen because it is an area of ongoing restoration efforts by the NSNWR to restore a portion of the floodplain from vegetation dominated by reed canary grass (*Phalaris arundinacea*) to a moderately diverse sedge meadow. *P. arundinacea* is an aggressive invasive species found throughout temperate North America that has been cultivated as forage grass because it is adapted to wide extremes in soil moisture (Galatowitsch et al., 1999). *P. arundinacea* thrives in wetlands with high annual or periodic fluctuations in water levels and is

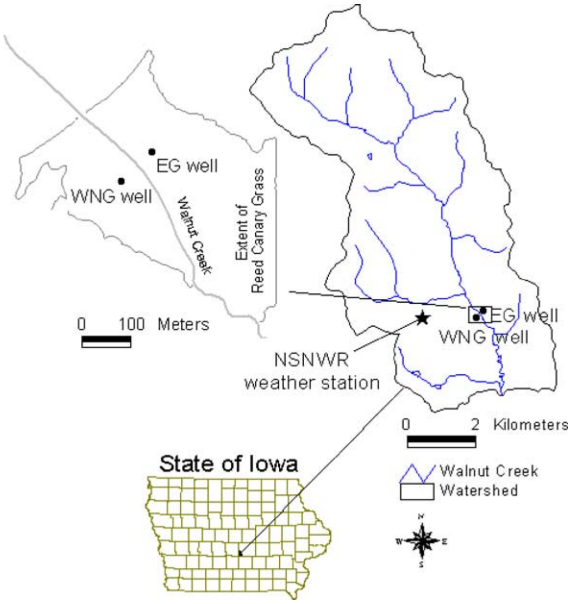

Fig. 1. Locations of the two monitoring wells and the weather station at the Neal Smith National Wildlife Refuge (NSNWR) in the Walnut Creek watershed.

considered among the most productive cool season grasses during drought (Galatowitsch et al., 1999). The rooting depth of the grass is shallow and less than 0.5 m. Two 3.2 ha sites were located on either side of Walnut Creek with the west side designated for treatment (burning, mowing, herbicide treatment and planting) and the east side designated a control (Fig. 1).

In 2001, a 550 m transect of 35 shallow monitoring wells was installed across the floodplain to evaluate floodplain hydrology (Schilling et al., 2004). Stratigraphy consisted of fine-grained (70–80% coarse and fine silt), organic rich alluvium. Based on grain size data (Schilling et al., 2004), the porosity, field capacity, and wilting point of the soil were estimated to be 0.48, 0.30, and 0.11, respectively (Saxton, 1986). Prior to treatment in spring 2002, the land cover on the two sides on the Walnut Creek was the same and consisted of densely covered reed canary grass. The groundwater levels on both sides of the creek were monitored from May to November of 2001 and they fluctuated similarly and followed nearly identical patterns (Fig. 2). The water table declined from May to early September despite significant rainfalls during this period and then increased in September 2001 in response to large rainfall events.

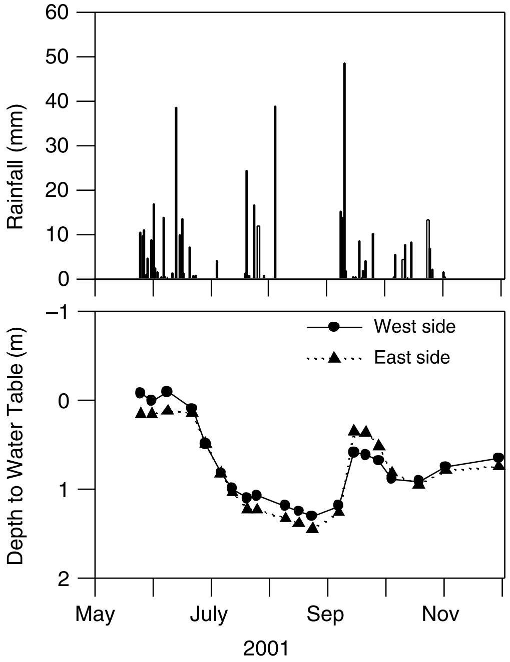

Fig. 2. Daily rainfall depth (top panel) and observed depth to the water table on west (solid dots) and east (solid triangles) side of Walnut Creek (lower panel) from May to November, 2001.

In the spring of 2002, the treatment and control areas were burned and the west area was further treated by mowing and herbicide application. In the fall of 2002, the treatment area was burned again to expose trees for cutting, remove duff for spring herbicide treatment and prepare the area for planting during the growing season. Spring rains delayed glyphosate treatment on the west side until June 2003 with mowing and additional herbicide treatment with backpack sprayers completed shortly thereafter. In July 2003, two shallow wells (2 m) were installed, one on the east grass (EG) side and another on west no grass (WNG) side of Walnut Creek at a distance of 40 m from the creek edge (Fig. 1). The water levels in these two wells were automatically recorded with transducer and datalogger at 30-min intervals for a period of 122 days from July 21 to November 20, 2003. A total of 5857 water level readings were obtained from both wells and are analyzed in this paper.

3. Effect of land cover on water table fluctuations

The observed data revealed substantial differences in water table fluctuation (Fig. 3) due to the difference in land cover on the east and west sides of Walnut Creek. Over the course of monitoring, the water level in the EG well was generally lower and had much less response to rainfall events than the WNG well. We divided the hydrograph into three distinct periods to describe the water table fluctuations in more detail (Fig. 3).

3.1. Period I-July 21 to September 11

Before monitoring began, several rainfall events totaling 138.7 mm occurred on July 7 and 8 (Fig. 3), which flooded the site and resulted in ponded water conditions on both sides of the creek. The onset of monitoring (July 21, 2003) was then characterized by little precipitation and an exponential water level decline in both wells (Fig. 3). Although our monitoring began on July 21 (Fig. 3), in all likelihood the water table in the wells started falling from near ponded conditions on July 9 and kept falling until the first rainfall occurred on July 27.

The water table responses to the small rainfall events (11.2 mm) on July 27 and 28 were quite different under bare ground and grassland cover. A water table rise of approximately 120 mm was recorded in the WNG well (Fig. 3), whereas little water level change was observed in the EG well. The water table rise in the WNG well indicated that the bare ground soils were wet with high residual soil moisture content. This was evidenced by the fact that even if all 11.2 mm of rainfall infiltrated and recharged groundwater the water table would have increased by only 23.3 mm. In contrast, little water level change observed in the EG well suggested that there was no groundwater recharge and that the soil moisture content on the EG side was lower than that on the WNG side. Moreover, little water level rise was observed in the EG well even though infiltration tends to be greater on vegetated soils than bare ground. The infiltrated water may have increased the soil moisture on the east side but it did not reach the water table.

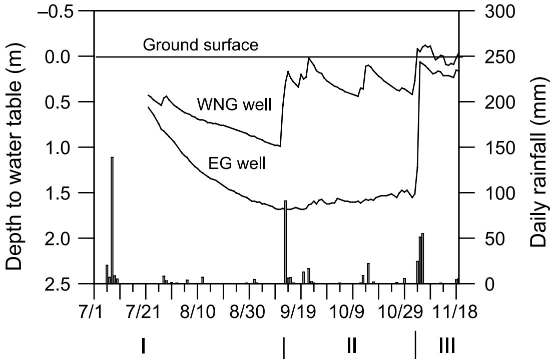

Fig. 3. Observed depth to the water table in the west no grass (WNG) well (top curve) and in the east grass (EG) well (lower curve) and daily rainfall depths from July 1 to November 20, 2003. Also marked at the bottom of the figure are the three periods discussed in Section 3.

The water levels in both wells kept falling until September 11 due to the lack of significant rainfall occurring during this period. The water table on the WNG side was lowered by lateral flow and evaporation (E) while that on the EG side by lateral flow and ET. The shallow groundwater is brought to the land surface by the capillary forces once the water table falls below the land surface. The amount of capillary rise varies with soil texture and structure (Gillham, 1984), but is estimated to be on the order of one to 2 m in the silt loam soils along Walnut Creek. Thus, considering the shallow water tables monitored during this study, the capillary fringe under the riparian sites likely extends near the ground surface and plays an important role in maintaining high soil moisture levels.

Considering that lateral flow was equivalent on both sides of the creek prior to the land cover change (Fig. 2), differences in water table behavior are primarily due to the differences between E and ET and their effects on soil moisture. ET losses on the EG side should be larger than E losses on the WNG side because evaporation would likely diminish much faster than ET as the water table was lowered. As a result, the soil moisture on the EG side would be lower than that on the WNG side. While unfortunate that soil moisture was not measured directly in this study, we can infer much about soil moisture conditions from the amount and timing of leakage through the soil zone captured by high-resolution water table measurements.

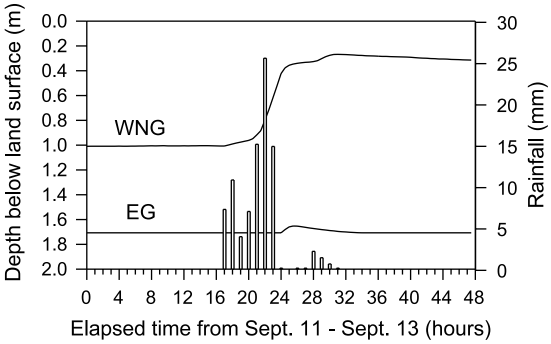

Fig. 4. Observed depth to the water table in the WNG well (top curve) and in the EG well (lower curve) and hourly rainfall depths in 48 h from 0:00 of September 11.

3.2. Period II-September 11–November 2

From 17:00 to 23:00 on September 11, 2003, 85.6 mm of precipitation raised the water table in the WNG well approximately 674 mm in the span of 8 h (Fig. 4). In contrast, the water table in the EG well increased little (less than 30 mm) during the same time period (Fig. 4). Differences in groundwater recharge between WNG and EG wells on September 11 were most likely attributable to differences in antecedent soil moisture conditions under grass and bare ground. ET demands by the grass resulted in a soil moisture deficit compared to the WNG side of Walnut Creek. Combined with a lower water table (thicker unsaturated zone), the capacity of the soil at the EG side to retain infiltrating water was considerably greater than the WNG side. The soil moisture on the WNG side before the rainfall event on September 11 was much higher than that on the EG side, since only a small portion of the infiltrated water reached the water table on the EG side. Evidently, most infiltrated water was trapped in the vadose zone to replenish soil moisture.

After the rainfall event on September 11, several other rainfalls occurred during this period (a total of 176.5 mm of precipitation occurred during this period). Soil moisture was replenished to saturation on the WNG side and the water table increased to near the ground surface. On the EG side, it was evident that soil moisture also increased during this period. Although the water table in the EG well increased slightly (Fig. 3), soils remained largely unsaturated.

3.3. Period III-November 2–20

A total of 130.6 mm of precipitation fell in three days from November. 1–3 and resulted in a rapid water table increase on both sides of Walnut Creek (Fig. 3). On the WNG side, the excessive precipitation resulted in ponded water above the land surface, whereas on the EG side, the water table increased 1414 mm in the span of several hours, indicating a near saturated soil on both sides of the creek. Additional rainfall events after November 4 produced similar water table responses in both EG and WNG wells.

Based on the above observation and discussion, we may conclude: (1) land cover has significant effect on water table fluctuations and soil moisture. In general, grass cover reduced soil moisture through ET, resulting in less recharge and a lowering of the water table; (2) soil moisture remained high on the WNG side because of less E on this side than ET on the EG side. Small rainfalls caused a large rise in the water table in the near saturated soils on the WNG side; (3) the soil moisture content on the EG side was significantly lowered through ET, as evidenced by the lack of water level response to several large rainfall events from September 11 to November 3.

4. Effect of land cover on evapotranspiration (ET)

The water table was lowered by evaporation (E) on the WNG side and by ET on the EG in addition to lateral flow which drained water from both sides. The amount of lateral flow is different on the two sides and will be discussed later. The amount of ET from the EG side should be larger than that of E from the WNG side, as indicated by a much lower water table and little response to the rainfall events at the EG well during most of our monitoring (from July 21 to November 2) (Fig. 3). The difference in ET between the two sides was estimated with a water level recession model based on measured groundwater level data.

As described above, the water table was at the ground surface and soil was saturated after several large rainfalls ended on July 8 (Fig. 3). The water table then fell during the subsequent dry period. The water table recession following a recharge event may

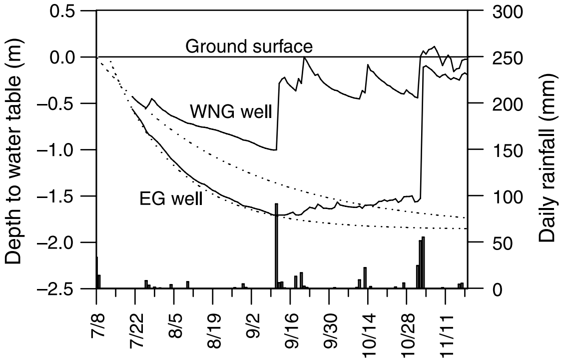

Fig. 5. Observed depths to the water table in the WNG well (top curve) and in the EG well (lower curve) and daily rainfall depths from July 8 to November 20, 2003. The top dashed curve is fitted by Eq. (2) with $d_0 = 1.86$ m and $\alpha = 0.0203$ day^{-1} and the lower dashed curve by Eq. (2) with $d_0 = 1.86$ m and $\alpha = 0.0425$ day^{-1}.

be described by a simple model

$$\mathrm{d}(t) = d_0(1 - e^{-\alpha t}) \tag{1}$$

where d(t) is the depth to the water table, d_0 is the asymptotic depth at which the water table becomes stable, α is the decline coefficient, and t is time. Although not measured, the water level in the WNG well should have started falling right after the large rainfalls (138.7 mm) ended on July 8. A small rainfall on July 27 caused a small increase in the water level at WNG well but had little effect on EG well. Eq. (1) was fitted to the observed water level decline at WNG well during the first 6 days of our monitoring from July 21 to 27 (Fig. 5). The best fitted (top dashed) curve to the water level decline from July 21 to 27 in the WNG well is given by $d_0 = 1.86$ m and $\alpha = 0.0203$/day with the starting time ($t = 0$) of July 8 on which the water level in the WNG well was at the ground surface. For the same value of $d_0 = 1.86$ m, the best fit to the observed water levels at the EG well is obtained by the lower dashed curve (Fig. 5) with $\alpha = 0.0425$/day and the starting time ($t = 0$) of July 12. The later starting time for water level decline on the east side suggests the grass covered east side started to become dry a few days later than the bare west side. This was likely due to the insolating capacity of the dense grass cover to reduce immediate evaporative losses. Field observations were consistent with this phenomenon. Once the east vegetated side became dry, the water level in the EG well fell at a faster rate than that in WNG well due to ET by the grass. Both curves eventually approach an equilibrium depth of 1.86 m if there is no additional recharge.

We initially believed the fitting presented in Fig. 5 seemed rather arbitrary, especially the dash curve fitted to the WNG well since, only 6 days of the monitoring data (July 21–27) are available. Our results, however, show otherwise: the two fitted curves are rather unique because the time at which the water level started falling is clearly known (at least for the WNG well, i.e. July 8). Once the starting time is fixed, there is a small range of α and d_0 values in Eq. (1) that one may change to obtain a curve that fits well to the 6-day measured water levels at the WNG well. With the value of d_0 obtained from the best fit to the WNG well (the value of d_0 should be more or less the same for both sides of the creek since it is mainly controlled by the lateral flow), one does not have much leeway in adjusting the values of α and t to gain the best fit to the observed water level changes from July 21 to September 11 at the EG well. Therefore, we are confident that the fitted curves presented in Fig. 5 are rather unique.

Based on the two fitted water table recession curves (two dashed lines in Fig. 5), the amount of ET on the EG side can be estimated with the difference between these two curves, i.e.

$$\mathrm{ET}(t) = (d_\mathrm{W} - d_\mathrm{E})\theta_\mathrm{a} = d_0(e^{-\alpha_\mathrm{E} t} - e^{-\alpha_\mathrm{W} t})\theta_\mathrm{a} \tag{2}$$

where d_W and d_E are the water table recession on the WNG and EG side, respectively, given by Eq. (1), θ_a is the available water supply for the soil or the difference between the field capacity and wilting point, $d_0 = 1.86$ m and $\alpha_\mathrm{E} = 0.0425$/day and $\alpha_\mathrm{W} = 0.0203$/day. The daily and cumulative ET values were estimated with a value of $\theta_\mathrm{a} = 0.19$ which is calculated based on the estimated filed capacity (0.30) and wilting point (0.11) from the grain size data (Schilling et al., 2004). A maximum ET of 7.6 mm occurred in the first day (July 12) when the water table was near the ground surface (Fig. 6(a)). The amount of ET then decreased exponentially to zero around day 33 (August 11). The cumulative ET for the 33-day period was 93.9 mm or 2.84 mm/day (Fig. 6(b)). Daily ET decreased almost linearly as the depth to the water table increased (Fig. 6(c)) and became negligible at $d = 1.42$ m. The finding that ET varies

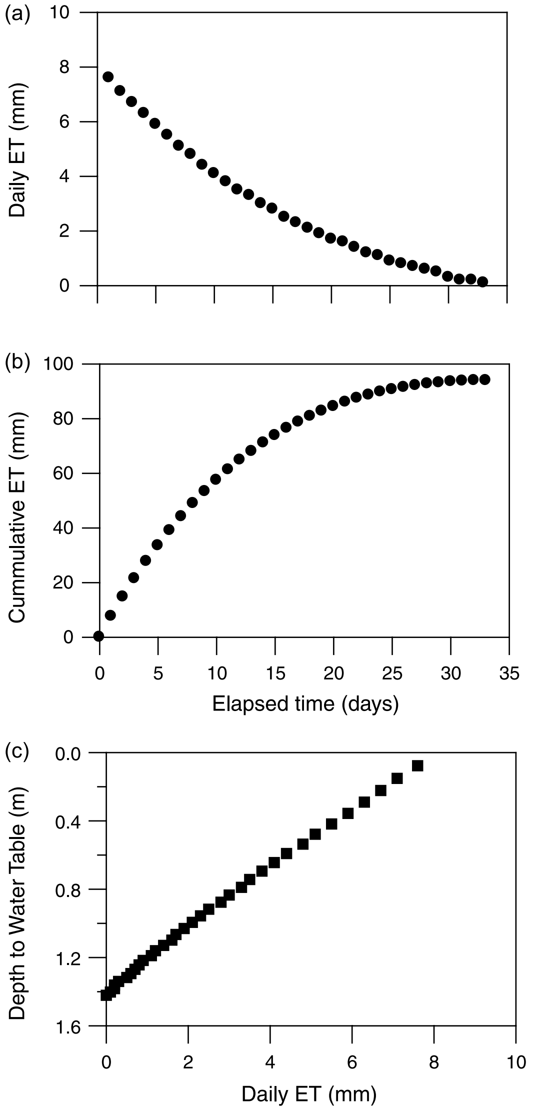

Fig. 6. Estimated (a) daily ET, and (b) cumulative ET by the grass cover on the EG side in 33 days from July 8 when the water table was near the ground surface to August 11 when the water table declined to 1.42 m below the ground surface, and (c) change of the estimated ET rate with the depth to the water table. Note: the ET estimated here based on Eq. (2) is actually the difference between the amount of ET on the EG side and that of E on the WNG side, mainly the transpiration by the grass.

linearly with the depth to water table supports the simple ET model implemented in the popular groundwater flow modeling code MODFLOW.

It should be pointed out that the values of ET estimated with Eq. (2) and presented in Fig. 6 are actually the differences between the amount of ET on the EG side and that of E on the WNG side, mainly the transpiration rates by the grass on the EG side (although they are not exactly transpiration rates because E on the EG side is different from E on the WNG side). The estimation was made under the assumption that the amount of the lateral flow on both sides is the same. In reality, the lateral flow on the WNG side should be larger than that on the EG side due to the higher water level or hydraulic gradient on the west side. As a result, we likely underestimated the actual difference between ET on the EG side and E on the WNG side. In other words, the difference in the land cover on the two sides results in at least a 93.9 mm or 2.84 mm/day difference in the amount of ET compared to E.

5. Effect of land cover on groundwater recharge

Groundwater recharge can be estimated with observed water level fluctuations, since water levels usually rise in monitoring wells due to infiltration in through the vadose zone. One approach to estimating groundwater recharge is to simply compute the difference between the two consecutive water level measurements multiplied by the soil moisture deficit before a recharge event, i.e.

$$R_i = (d_i - d_{i-1})(n - \theta_{i-1}) \tag{3}$$

where R_i is the recharge rate on day i, d_{i-1} and d_i are the observed depth to water table on day $i-1$ and day i, respectively, n is porosity, and θ_{i-1} is the soil moisture content on day $i-1$. This approach may provide a good estimate in the ideal situation of a flat water table with no ET or other source and sink terms. In reality, there usually is a hydraulic gradient and water levels in wells would fall due to lateral flow, which in our case is the discharge to Walnut Creek. In such a case, the value of R_i obtained with Eq. (3) would be underestimated. More accurate estimation for R_i may be obtained by the following equation

$$R_i = (d_i - d_i^*)(n - \theta_{i-1}) \tag{4}$$

where d_i^* is the depth to water table that would had been observed on day i if there was no recharge between day $i-1$ and day i. Typically, Eq. (4) would be difficult to use because the value of d_i^* cannot be measured. However, the value of d_i^* in our case can be obtained with the simple water table recession model (Eq. (1)). In doing so, first the time t_{i-1} at which the depth d_{i-1} was observed, is calculated with

$$t_{i-1} = \frac{-1}{\alpha} \ln\left(1 - \frac{d_{i-1}}{d_0}\right) \tag{5}$$

which was obtained by rearranging Eq. (1). Then the value of d_i^* is calculated with

$$d_i^* = d_0(1 - e^{-\alpha(t_{i-1}+\Delta t)}) \tag{6}$$

where Δt is the time interval between the two measurements d_{i-1} and d_i (Δt is one day in our case).

With this approach we estimated the daily changes in the depth to the water table ($d_i - d_i^*$) due to recharge at both WNG and EG wells (Fig. 7). Several observations can be made. First, the large daily water level increases in the WNG well (Fig. 7(a)) correspond well to large rainfall events (Fig. 7(c)). For example, the water level increase in the WNG well as results of recharge on September 12 and 21, October 14, and November 2 are 734, 319, 334, 475 mm, respectively, all of which were recorded 1 day after large rainfall events. On the other hand, only one large rise (1414 mm) was observed in the EG well on November 3 (Fig. 7(b)) in response to the large rainfall event on November 1. There was no significant response to large rainfalls before November 1 in the EG well because of the soil moisture deficit caused by ET. Secondly, there were many small daily increases in the water level observed at both wells. This was probably due to internal downward drainage which could last many days after a rainfall event (Hillel, 1998). The small daily rise or recharge to groundwater occurred more often at the WNG well (Fig. 7(a)) because of a relatively shallower water table and higher soil moisture content. On the EG side, the small daily rises occurred mainly after September 11. In both cases, these small daily rises in water table represent significant amounts of recharge to groundwater. For example, the summation of the small changes in the EG well was 1553 mm which was more than half of the total increase (3070 mm) in the water level. Thirdly, there were a few negative values of ($d_i - d_i^*$). The most noticeable negative ($d_i - d_i^*$) occurred in the WNG well probably due to additional ET on November 14 when the water table was at the ground surface.

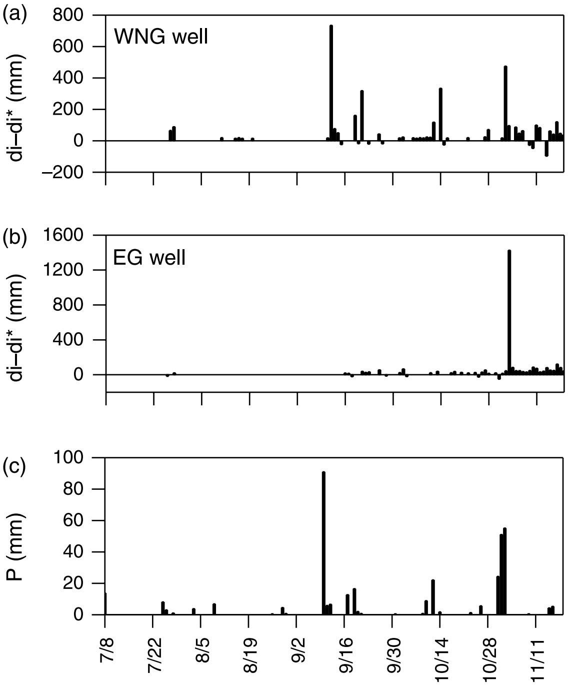

Fig. 7. Estimated daily recharge to groundwater in the WNG well (top panel) and in the EG well (middle panel), and the daily rainfall depths (bottom panel) from July 8 to November 20, 2003.

The total rise in the water level, i.e. summation of the daily values, was 3668 mm for the WNG well and 3070 mm for the EG well. The difference in the total rises at the two wells was 598 mm. One has to know the value of the soil moisture before each recharge event in order to use Eq. (4) to calculate the amount of recharge to groundwater. Since soil moisture was not measured in this study, we assume three possible soil moisture values: 0.2, 0.3, and 0.4. Using these values, the differences in groundwater recharge between the two sides are calculated to be 47.8, 107.6 and 167.4 mm, respectively, with Eq. (4) and the estimated porosity of 0.48. In other words, the WNG side of Walnut Creek received at least 47.8 mm more recharge than the grass covered east side during the monitored period of 122 days.

6. Implication for the basin water balance and pollutant loads

The effect of vegetation on evapotranspiration and groundwater recharge is an important consideration in the basin-scale water cycle. Our monitoring data provide baseline information linking land use differences to groundwater recharge and ultimately baseflow to streams. A water balance equation for an aquifer system (assuming no well pumping) can be written as

$$R - \mathrm{ET} - \mathrm{BF} = \Delta S \tag{7}$$

where R is recharge to groundwater, ET is evapotranspiration, BF is baseflow, and ΔS is the change in groundwater storage. To understand the important role of vegetation in the basin-scale water cycle and provide end member constraints on maximum differences, one may write two water balance equations, one for a watershed that is covered entirely by vegetation and another for a watershed that is totally left as bare ground. Groundwater storage stays more or less the same over a long period of time in both watersheds, e.g. one or more years, and thus ΔS may be neglected in Eq. (7) for a long-term water balance. Then the difference in baseflow of these two watersheds can be estimated by subtracting one equation for the other, i.e.

$$\mathrm{BF_B} - \mathrm{BF_V} = (R_\mathrm{B} - R_\mathrm{V}) + (\mathrm{ET_V} - E_\mathrm{B}) \tag{8}$$

where the subscript ‘B’ stands for a bare watershed and ‘V’ for a vegetated watershed. Based on the results obtained at our site, the difference in recharge between the two land covers was at least 47.8 mm and that in ET was 93.9 mm for the monitored period of 122 days. This means that there would be a total of 141.7 mm more baseflow in a bare ground watershed than in a vegetated watershed. For comparison, this magnitude of increase in the Walnut Creek watershed would nearly double the baseflow since the estimated baseflow for the basin is 145 mm/year (Schilling, 2002). For the Walnut Creek watershed with a drainage area of 52.17 km^2, the increase of 140.7 mm in the baseflow is equivalent to a total baseflow volume of 7.28* 10^6 m^3. More significantly, the total nitrate load carried to the creek by this amount of baseflow would be 72,800 kg or 13.8 kg/ha with an average nitrate concentration of 10 mg/L. This calculation clearly demonstrates the effect of land cover on the basin-scale water cycle and its important implications for reducing pollutant loads to streams.

While it is clear that bare soil conditions do not represent the land cover of annual crop fields, it is illustrative to consider their similarity during non-growing seasons. If one considers the bare soil condition (WNG) to represent the land cover of agricultural fields during vulnerable leaching periods in the spring and fall, and the grass condition (EG) to model the land cover of perennial vegetation on the agricultural landscape (buffers, filter strips, pasture), the benefits of perennializing portions of agricultural landscapes become apparent. Our study suggests that locating perennial grass cover in agricultural watersheds, or encouraging the use of annual cover crops, could substantially reduce groundwater recharge compared to the bare ground alone, and thus reduce subsurface pollutant loads delivered to streams via baseflow. Results from this study suggest that utilizing perennial vegetation in agricultural watersheds can significantly reduce baseflow that would in turn reduce the nitrate–nitrogen loads delivered to rivers and streams.

7. Summary and conclusions

Two wells were installed under similar conditions except that one well was installed on the east side of a creek which was covered with grass, and the other on the west side of the creek which was burned into a bare ground and had no vegetation cover. Water level measurements were collected from two monitoring wells over a period of 122 days. While limited in duration, we observed substantial differences in water table behavior under two land cover scenarios and assessed the effects of land cover on evapotranspiration (ET), soil moisture, and groundwater table and recharge (R). The following main conclusions may be drawn from this study:

1. Land cover had a significant effect on soil moisture and the water table. In general, the grass cover reduced soil moisture through ET, resulting in less water to recharge groundwater and therefore lowering the water table. This conclusion is consistent with the climate-soil-vegetation model developed by Rodriguez-Iturbe et al. (1999).
2. The water recession model presented in this paper suggested that ET decreased exponentially from

7.6 mm/day to zero as the water table declined from near the ground surface to 1.42 m below the ground surface in 33 days. The cumulative ET for the 33-day period was 93.9 mm or 2.84 mm/day.

3. The estimated ET with the water recession model varied linearly with the water table depth, which supports the simple ET model implemented in the popular groundwater flow modeling code MODFLOW.
4. While large daily water level increases in the WNG well corresponded well to large rainfall events, small daily recharge to groundwater was observed at both wells during non-raining periods. This was probably due to internal downward drainage that could last many days after a rainfall event (Hillel, 1998). The small daily rise or recharge to groundwater occurred more often at the WNG well (Fig. 7(a)) because of the relatively shallower water table and higher soil moisture content. The small daily rises in the water table represented a significant amount of recharge to groundwater.
5. The differences in ET and in groundwater recharge due to the difference in land cover on the two sides were at least 93.9 and 47.8 mm, respectively, for the monitored period of 122 days. Given a watershed entirely in bare soil, there would be 141.7 mm more baseflow compared to a watershed entirely in grass. In a watershed the size of Walnut Creek, this magnitude of increase would nearly double the baseflow and thus dramatically increase the total nitrate carried to the creek by baseflow. This calculation clearly demonstrates the effect of land cover on the basin-scale water cycle and its important implication for managing vegetation in landscapes to reduce pollutant loads to streams.

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