Characterizing Sources of Nitrate Leaching from an Irrigated Dairy Farm in Merced County, California

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Abstract

Dairy farms comprise a complex landscape of groundwater pollution sources. The objective of our work is to develop a method to quantify nitrate leaching to shallow groundwater from different management units at dairy farms. Total nitrate loads are determined by the sequential calibration of a sub-regional scale and a farm scale three-dimensional groundwater flow and transport model using observations at different spatial scales. These observations include local measurements of groundwater heads and nitrate concentrations in an extensive monitoring well network, providing data at a scale of a few meters and measurements of discharge rates and nitrate concentrations in a tile-drain network, providing data integrated across multiple farms. The various measurement scales are different from the spatial scales of the calibration parameters, which are the recharge and nitrogen leaching rates from individual management units. The calibration procedure offers a conceptual framework for using field measurements at different spatial scales to estimate recharge N concentrations at the management unit scale. It provides a map of spatially varying dairy farming impact on groundwater nitrogen. The method is applied to a dairy farm located in a relatively vulnerable hydrogeologic region in California. Potential sources within the dairy farm are divided into three categories, representing different manure management units: animal exercise yards and feeding areas (corrals), liquid manure holding ponds, and manure irrigated forage fields. Estimated average nitrogen leaching is 872 kg/ha/yr, 807 kg/ha/yr and 486 kg/ha/yr for corrals, ponds and fields respectively. Results are applied to evaluate the accuracy of nitrogen mass balances often used by regulatory agencies to assess groundwater impacts. Calibrated leaching rates compare favorably to field and farm scale nitrogen mass balances. These data and interpretations provide a basis for developing improved management strategies.s

Key words

nitrate leaching, dairy farm, groundwater modeling

1 Introduction

Dairy farming operations produce a complex landscape of sources of groundwater nitrate (NO_3^-). Potential sources include animal waste storage ponds, animal holding areas, crop land receiving animal wastes and chemical fertilizer, surface runoff and runon containing animal wastes, septic tanks, inadvertent spills of manure, and significant atmospheric deposition of nitrogen (Canter, 1997; Karr *et al*., 2001).

The evaluation of long-term impacts from dairy farming on groundwater quality is of interest because of concerns over drinking water quality. Dairy farming has been identified as a significant source of domestic well contamination in the alluvial and fluvial fill basins of California's Central Valley (Lowry, 1987; Burow *et al*., 1998; Burow *et al.,* 2006). The issue has wider significance. Studies in the USA and indeed throughout the world indicate that livestock is a major contributor to groundwater contamination (UNESCO, 2006; Burkart and Stoner, 2002).

Nitrate pollution of groundwater, here defined as the product of recharge rate and nitrate concentration, is most often determined via general evaluation of groundwater vulnerability and regionalized assessment of nonpoint sources. This approach has been exploited in particular in the application of Geographical Information Systems to groundwater vulnerability (e.g. Evans *et al*., 1995; Holtschlag and Luukkonen, 1998; Snyder *et al*., 1998; Nolan *et al*., 2002; de Paz and Ramos, 2004; Leone *et al.,* 2007). Numerous other studies have focused on detecting or simulating nitrate losses from individual farm or land use units (e.g. Li *et al*., 2006; Bakhsh *et al.,* 2004; Nangia *et al.,* 2008;

Parker *et al.,* 1999; Cihan *et al.,* 2006). Measurements are typically taken in conjunction with field experiments, in particular those designed to evaluate specific agricultural management practices (cf. Table 1 in Harter *et al.,* 2002). There are only few studies where concentration measurements are taken across the landscape of an individual farm or farming region, especially of the confined animal holding area (Karr *et al.,* 2001; Harter *et al.,* 2002). This lack of data has hampered the development of groundwater models that account for the large amount of spatial variability in nonpoint sources across a dairy farm.

The objective of our work is to characterize the average nitrate leaching from different management units of a typical freestall dairy farm with irrigated forage crops, based on groundwater head, flux, and quality observations at various spatial scales. The present study proposes to characterize spatially distributed nitrate loading to groundwater across a dairy landscape using a process based groundwater flow and transport model. We apply a sequential procedure to calibrate the model to observations of:

- heads and nitrogen concentrations in an extensive monitoring well network; and,
- drainage fluxes and nitrogen concentrations in a tile drainage network.

The monitoring well network provides data at a scale (or measurement support as defined in geostatistics, see Isaac and Srivastava, 1990) of a few meters, with concentrations representing source areas that are a fraction of the size of an individual management unit, but may cross management unit boundaries. Head data are point-measurements at the scale of the monitoring well diameter. Tile drainage fluxes and concentrations represent an integrated measurement of recharge and nitrogen fluxes across the drainage network, which operates across multiple farms.

However, in this study we target the management unit scale. As proposed by Harter *et al.* (2002) we distinguish three management units: corrals (including animal exercise yard, freestalls, feed storage area, solid waste storage area), ponds (liquid manure storage), and manure-treated forage fields (see Figure 1). Our focus is on estimating nitrogen losses to groundwater from these management units, recognizing that nitrogen is primarily managed at that scale. For permitting, planning and assessment objectives, the nitrogen balance approach at the field and farm scale has been proposed as an alternative measurement of potential nitrogen losses to groundwater. Therefore the N-leaching obtained for the management units were scaled up and compared with the N-leaching obtained using field and whole farm mass balance approaches often considered by regulatory agencies in Europe and the United States (e.g., Oenema et al., 1998; CRWQCB, 2007; Harrison and White, 2007).

The paper is organized as follows: first we briefly describe the site and its manure management system. Then we introduce the conceptual model and the sequential calibration strategy. Next we present the calibration results and discuss the results in the context of the N-balance of individual fields and the entire dairy farm.

2 Site description

2.1 Manure management and nitrogen sources

The study area is 99 ha and encompasses two dairy farms in the San Joaquin Valley (Harter *et al.*, 2001a; Harter *et al*., 2002) . The average herd size is 1731 milk cows, 308 dry cows and heifers and 517 animals less than 1 year old (total 2069 animal units). Figure 1 shows the distribution of management units, including 88.6 ha of forage fields, 1.1 ha of animal waste holding ponds and 9.2 ha corrals. Other land uses in the area include roads, residential areas and open spaces. In early 1998, significant improvements in the manure management system were implemented on the fields F8 and F9, here referred to as targeted manure management (TMM).

Organization of the farms is typical for the region. Animals are held in freestall facilities with concrete flushlanes and surrounding exercise area with compacted bare soil. Here the entire freestall and exercise area is referred to as corrals. Excrements are collected by twice daily flushing of freestall flush lanes and monthly scraping of the surrounding exercise areas (Morse, 1997).

After mechanical separation of manure solids, the liquid manure water mixture is stored in anaerobic holding ponds. The ponds were constructed prior to 1980 with soil liners containing at least 10% clay using local soil material. The bottom of the ponds lies at or closely above the water table. Total nitrogen concentrations in these ponds typically range from 200 to 1000 mg-N/l. Ammonium-N (NH_4^+) accounts for roughly one third to one half of the total-N, with the remainder being in the organic-N form (Mathews *et al.,* 2001a). Solid manures are stored in stacks until reused for freestall bedding, field soil amendment, or sold for off-farm application (Morse, 1995). These stacks are located on concrete and since runoff is collected the losses from feed stacks to groundwater are assumed to be negligible.

Surrounding forage fields are double cropped in a rotation of summer silage corn (*Zea mays L.*) and winter cereals such as oats (*Avena sativa*), wheat (*Triticum sp.*) and sometimes in combination with Alfalfa (*Medicago sativa*) (Harter *et al., 2002*). Liquid manure from storage ponds is mixed with surface water irrigation during the summer to supplement or even replace chemical fertilizer applications. In the past, pond water was often applied undiluted, or lightly diluted as needed to reduce pond levels during the rainy winter season (Morse *et al.,* 1997). This practice has been discontinued. Harvested crops are used as part of the feed ration, which is supplemented with purchased feeds.

2.2 Climate, soils and hydrogeology

The study site has a Mediterranean climate, with 0.35 m/year precipitation (*P*), almost all occurring between October and April (UC Davis, 2000). For the predominant field crop rotation (summer corn and winter cereals), potential crop evapotranspiration (*ETp*) is 0.90 m/year (UC Davis, 2000; UCCE, 1987). Irrigation applications (*I*) were measured using a totalizing electromagnetic flow meter in three representative fields (Harter *et al.*, 2001a). Over a two year period, the average annual application was 1.23 m, resulting in a recharge rate (*P+I-ETp*) of 0.69 m/year. Overall irrigation efficiency (*ETp-P / I*) is 46 percent which is relatively low compared to other regions (e.g. Meyer and Schwankl, 2000) but typical for border flood irrigation on the highly permeable, well drained soils found at the site.

Land elevation at the study area is 25 m above mean sea level (MSL). The topography is featureless with slopes less than 0.2 %. Soils formed on flood plains and wind modified alluvial fans. The dominant surface texture is sandy loam to sand underlain by silt lenses, some of which are cemented with lime (Harter *et al*., 2002).

Groundwater occurs in the upper basin fill consisting of alluvial and fluvial deposits with some hardpan and interbedded lacustrine deposits. The Corcoran clay layer forms a continuous confining layer at a depth of approximately 33 m below MSL (Page and Balding, 1973) with an estimated conductivity between $1*10^{-6}$ m/day and $1.6*10^{-4}$ m/day (Phillips *et al*., 2007). Slug tests performed in monitoring wells at the dairy farm indicated that hydraulic conductivity in the shallow aquifer ranges from 18 to 155 m/day (H.H. Davis, California Water Quality Control Board – Sacramento, personal communication in 1998).

Groundwater generally flows from the east-north-east to the west-south-west following the slope of the landscape. The water table is on average 2-3 meters below the ground surface. Fluctuations are 1 to 2 meters and are both seasonal due to rainfall, tile drainage and pumping from domestic and drainage wells and short term following irrigation. Over the past 15 years, long-term groundwater levels at the site have been constant.

2.3 Groundwater monitoring and tile drainage network

An extensive network of 47 shallow monitoring wells has been installed on and around the two dairies (Figure 1). Wells are screened from 3 to approximately 10 m below ground surface. For 13 wells, water level and sampling data are available at 43 sampling dates between 1995 and 2000. The remaining 34 wells were constructed in early 1999 and were sampled through 2000 (15 months, 10 sampling dates). The construction of these wells coincides with the construction of a tile drainage system in 1999 (Figure 1). The outflow of tile drainage is measured using a volumetric flux meter and sampled every 6 weeks at the same time as the well network. All samples are analyzed for various chemical compounds including nitrate (Harter *et al*., 2002).

2.4 *Groundwater quality*

Nitrate concentrations in monitoring wells throughout the dairy farm exceed the maximum contaminant levels of 10 mg N/l. Harter et al (2002) classified the wells according to their predominant upgradient land use. The monitoring data shows the highest nitrate concentrations in monitoring wells downgradient of corrals (87.8 mg-N/l) followed by those downgradient of ponds (70.1 mg N/l) and of fields (61.1 mg-N/l, see Figure 2). Mean concentrations downgradient of the three management units are different at a significance level below 0.03 when using 2-tailed student-t statistics. Most observed nitrogen is in the form of nitrate except in the monitoring wells downgradient of the ponds. These wells contained on average 11.5 mg NH4-N/l and no dissolved oxygen, indicating reducing conditions.

Nitrate concentrations in monitoring wells are highest in the winter period when recharge rates are low and pond manure water was often applied nearly undiluted (see pre-1998 data in Figure 3). The post-1998 data shows the response of monitoring well concentration to the introduction of improved manure management (TMM).

The average drainage flux is 2,1 10^6 m^3/year. Nitrate concentrations in the drainage water range from 46.2 mg-N/l to 51.9 mg-N/l and are on average 49.6 mg-N/l. Drainage water includes local recharge, upwelling groundwater and groundwater that recharged upgradient of the dairy farm.

3 Groundwater flow and transport model

3.1 *Modeling approach*

A sequential parameter optimization procedure was chosen to account for the varying spatial scale and temporal extent represented by the observations:

- First, a sub-regional steady state flow model of the monitored farms and surrounding area is used to calibrate the groundwater flow parameters including some recharge rates (model-1a).

- Next, a transport model at the farm scale is used to calibrate transport parameters including N-leaching from the different management units (model-2).
- Finally, a sub-regional scale transport model is used to validate the temporal and spatial dynamics of the calibrated models 1a and 2 (model 1b).

Model-1a is calibrated against average water level measurements across the dairy study site and against tile drainage outflow, representing an area larger than our study site. Tile drain outflow observations have a measurement support scale that exceeds the study dairy and requires a sub-regional scale model to prevent boundary influences at the location of the observations (see table 1). Validation data consisted of concentrations in the tile drainage outflow (model-1b).

Calibration target data for model-2 consists of nitrate concentrations measured in shallow monitoring wells that typically represent a source area of 100 to 200 m long but only several meters wide. A telescopic mesh refinement technique was used that allows for detailed spatial representation of the nitrogen sources, while the finer grid also avoids numerical dispersion.

In the process of developing a groundwater modeling concept, the principle of parsimony guides the complexity of the model. The following key simplifications were necessary to properly accommodate the limited number of data available for model calibration:

1. The complex structure of nitrogen sources across the dairy is simplified by:
    - classifying all land uses within the sub-region into 5 major source categories (‘field’, ‘corral’, ‘pond’, ‘orchard’, and ‘other’);
    - neglecting spatial variability of recharge rate (R) and concentration (c) within individual fields, corrals and ponds; and,
    - neglecting short-term or seasonal changes in nitrate recharge concentration (source strength is assumed constant except in two fields that switched from a conventional to a targeted manure management system).

2. Denitrification and other chemical reactions in groundwater are assumed to be negligible due to the oxic soil and aquifer conditions;
3. Effective horizontal and vertical hydraulic conductivity (k_x and k_z), tile drain conductance (D_{drain}), dispersivity (α), and effective porosity (n_e) are homogeneous throughout the area of interest; and,
4. Short-term and seasonal variations in water level are considered negligible (except for model-1b).

The three-dimensional finite difference code MODFLOW-88 is used to simulate flow (McDonald and Harbaugh, 1988) and MT3Dms (Zheng and Wang, 1998) for the fate of nitrate in saturated groundwater. Groundwater Vistas version 2.66 was used as pre- and post-processor (Rumbaugh, 2000).

3.2 Sub-regional scale flow model – Model-1a

For the flow model we used a 9 square km area that also includes partial areas of several adjacent dairies. Since our interest is focused on processes in the shallow-most part of the groundwater, we only consider the 58 m thick upper aquifer unit. The regional aquitard unit (Corcoran Clay) is considered the lower no-flow boundary of this upper aquifer (Davis, 1995). Horizontal discretization of the flow model ranges from 25 meters in the area of interest to 100 meters near the model boundary. There are a total of 7 layers with thicknesses ranging from 3 to 12 meters (total number of grid cells: 48,726). The size of the sub-regional model-1a is chosen sufficiently large to prevent boundary condition feedback at the location of the calibration targets.

Two long-term steady-state flow regimes are considered: the period 1995 - 1998 prior to construction of the tile drains and the period 1999 – 2000 after their construction. The instantaneous switch between 1998 and 1999 was chosen because drain fluxes are negligible during this winter period. Flux across the boundaries of the study area is considered to be driven by the regional head gradient and are simulated using constant-head boundaries that were interpolated from measured regional head data (DWR, 2000). Internal flow stresses within the study area include recharge, groundwater pumping for drainage purposes, domestic groundwater pumping, and tile drainage. Groundwater recharge at the top of the model is land use dependent using the following categories: fields, corrals, ponds, orchards, and 'other'. Recharge rates for fields, orchards, and 'other' are based on long term average irrigation and meteorological data. Land use dependent parameters were assigned to the model using a land use GIS map of the region produced from high-resolution aerial photographs (Alexander Fritz, personal communication, 1999).

Location, depth and discharge rates of drainage wells in the first aquifer have been provided by the local irrigation district (Liebersbach, Turlock Irrig. Dist., personal communication, 2000). Domestic well pumping rates at the dairy farms are estimated based on a daily use per cow of 570 liters (J. Merriam, farm advisor, University of California Cooperative Extension, personal communication, 2000).

3.3 Dairy farm scale model – Model-2

The second model comprises the 2 square km area of interest consisting of all fields, corrals and ponds for which downgradient monitoring well data are available. The depth extends to 14 m below the average water table so that the boundary does not influence simulated concentrations in the shallow monitoring wells. Vertical and horizontal discretization is finer than in the regional model. Cell sizes range from 3 meters near monitoring wells to 12 meters at the boundaries. There are 6 layers with a thickness that ranges from 1.5 m at the groundwater table to 4 meters at the bottom (total number of grid cells: 137,664). The grid is oriented on the basis of major land use characteristics to enable a good representation of pollution sources.

Groundwater heads at the boundaries were based upon simulated heads in the regional-scale model and remained unchanged throughout the simulation period. For the transport model we used no dispersive fluxes at the outflow boundaries. At the inflow boundaries we used fixed concentrations of 3 mg-N/l, which is equal to natural background concentrations. Backtracked particles from the bottom of the monitoring well screens all have their recharge source within the model boundary so the boundary concentrations have little influence on the calibration. Nitrate concentrations in recharge water are assigned based on the same land use categories used in the flow model, namely fields, corrals, ponds, orchards and 'other'. However, in the calibration we also allow for variation within farm management units so that, for example, the 7 individual fields of the dairy farm (F1, F3, F5, F7, F8, F9, F10) can have different concentrations. This reflects the different management practices in each field.

Temporal mean NO_3^--N concentrations in 47 monitoring wells were used as calibration targets. Fields are assumed to be under conventional management, which is the predominant form of management during the observation period 1995 – 2000. Samples taken after 1997 from monitoring wells located immediately downgradient of the 2 fields with TMM were excluded since these do not represent conventional management practices. Calibrated leaching rates and concentrations are assumed constant in time. Simulated, asymptotic monitoring well concentration (obtained by running the transient transport simulation for 30 years) are considered representative of and compared against the average 1995 – 2000 concentrations in each well.

3.4 Parameter estimation

The objective function to be minimized for calibrating model-1a and model-2 is defined as the sum of squared weighted residuals of all measurements:

$$OBJ : \min \sum_{i=1}^{nD} \left[w_i (O_i - S_i) \right]^2 \qquad (1)$$

With *nD* being the number of data, and O_i the average observed value in the ith monitoring location with corresponding simulated value S_i. Calibration weights w_i are equal to the inverse standard deviation of measurements at each location (Hill and Tiedeman, 2006).

During the calibration, we did not allow the flow model-1a to deviate from the drain flux measurement to prevent mass balance errors. This was achieved by adjusting the drain conductance only at the end of each parameter adjustment. Note that the effect is similar to giving the drainage flux a very large weight. This is justified because the flux data is supported by much larger area than the local head and concentration measurements.

Calibration was hand-operated by trial-and-error adjustments of parameters. Composite scaled sensitivity (*CSS*) was regularly evaluated to make sure only those parameters for which the measurements provide enough information are calibrated (Hill and Tiedeman, 2006). Parameters with low sensitivity were estimated directly based on prior information. The model performance was evaluated using graphical analysis such as scatter plots and statistical analysis.

4 Results and discussion

4.1 Parameter optimization

Parameter sensitivity guided the calibration process. The sensitivities of the final models are shown in Figure 4. Low sensitivity indicates that the data do not contain enough information to calibrate the parameter. For these parameters the values are estimated from literature sources as shown in Table 2.

For the steady-state regional flow model only the regional aquifer hydraulic conductivity (k_x) and the tile-drain conductance (D_{drain},) are calibrated. An optimal fit is found at a k_x of 21 m/day. The calibrated value is low compared to other field and modeling studies in the region such as Phillips *et al.* (2007) who found average conductivities of 40 m/day, but this may be due to lower fractions of coarse grained materials at our site.

Groundwater recharge from fields is a moderately sensitive parameter, but is not estimated by parameter optimization because we have confidence in the prior estimation with meteorological and irrigation data. Recharge rates from ponds and corrals are very insensitive parameters in the flow model-1a. Water level response is minimal over the tested range due to the small area of ponds and corrals in combination with relatively high conductivity of the aquifer.

Alternatively, to estimate pond recharge, Harter *et al* (2002) pointed out that reduced nitrogen species are only present throughout 3 meter screen length of the monitoring wells immediately adjacent to the ponds. Since reduced nitrogen was not observed elsewhere, they argued that the entire source area of the monitoring well is likely to be within the pond. Pathline analysis with MODPATH on the calibrated flux field of model-1a indicates that this constraint is fulfilled if the average pond recharge rate is at least 0.8 meters per year (=2.2 mm/d), confirming preliminary estimates of Harter *el al.* (2002). The obtained pond leaching rate is similar to seepage rates observed on other dairy manure ponds after initial liner development (no very coarse soil, no frost, see for instance Barrington and Jutras, 1983; Davis, 1973; DeTar, 1979; Ham and DeSutter, 1999; Korom and Jeppson, 1994; Meyer et al., 1972).

Corral recharge is calibrated with model-2 because the parameter is sensitive to nitrogen concentration data. In the process of calibrating the transport model, concentrations and recharge rates from corral C1 are found to be consistently higher than from corral C2. A possible explanation is that corral C1 areas are not constructed under a slope to enhance runoff and reduce infiltration. Other factors such as soil type, management practices and soil are not apparent. The optimum model fit for recharge rate is 2.4 m/year in corral-1 and 0 m/year in corral C2. However, both the high recharge rate for corral-1 and the zero recharge rate for corral C2 is outside the reasonable range of recharge values for this management unit: daily liquid excretion by an adult Holstein cow is 0.085 l/kg animal weight; hence, the total water available for groundwater recharge is 0.49 m/year from cow excrement and 0.35 m/year precipitation. During the summer months, evaporation is likely to consume much of the applied liquid in the corral area. Hence, actual recharge is significantly lower than 0.84 m. Zero recharge is also unlikely based on the analysis of other water quality parameters such as EC that show elevated values downgradient of C2 compared to other, upgradient monitoring wells (Harter *et al.,* 2002). We therefore decided to use the initial estimate of 0.29 m/year in corral C2 and twice the original value in corral C1.

At these recharge rates, the average calibrated nitrate leaching is 486 kg/ha (70 mg-N/l) for fields, 872 kg/ha (298 mg-N/l) for corrals and 807 kg/ha (98 mg-N/l) for ponds.

We note that - in most cases – simulated management unit-specific recharge (input) concentrations are higher than the simulated concentrations in the downgradient monitoring wells used as calibration targets. This is due to the fact that monitoring wells (either actual or simulated) do not measure recharge directly from one management unit but instead may capture a mixture of water that originates from different management units.

The dilution and mixing process is illustrated in Figure 5 where the source area of individual monitoring wells are determined using pathline analysis in MODPATH. For example, water sampled in the monitoring wells downgradient of corral C2 is also influenced by recharge from field F8. As a result, parameters of different management units are often either positively or negatively correlated (see Table 3) and cannot be estimated independently. Correlation is highest between recharge rate and concentrations of corrals (-0.71) but remain within reasonable bounds for acceptable model calibration.

4.2 Model fit and uncertainty

Both, flow and transport models show a good fit for the observed heads and nitrate concentrations in monitoring wells. Residuals are generally small (Figure 6) with the largest differences occurring in monitoring wells downgradient of corrals. The largest residual occurs in a monitoring well with simulated nitrate concentrations that are nearly twice the measured concentration: 231 mg-N/l versus 121 mg-N/l. A possible mechanism not accounted for in the model that makes this observed concentrations lower is that the upgradient corrals are relatively new and roofed, thus reducing recharge.

The higher residuals for monitoring wells downgradient of corrals is also reflected in the larger linear confidence intervals for recharge concentration from corrals as shown in Figure 7. The figure also illustrates that the head data do not contain enough information to allow for a reliable estimate of recharge rates. However, for these we have other, more reliable prior information available. Note that the linear confidence intervals in Figure 7 are based on unconstrained corral recharge calibration results, not on the final calibration values for corral recharge rate, which we limited given other information available.

The standard error of regression should ideally be close to 1, meaning that the fit achieved by regression is consistent with the data accuracy reflected in the weighting (Hill and Tiedeman, 2006). The high standard error of regression of the transport model, in combination with the high number of parameters possibly indicates over-parameterization (see Table 4). We therefore recalibrated the transport model with a single average recharge concentration and rate for each management unit. However, this leads to a higher standard error of regression of 2.25 instead of 1.30 and lower model efficiency of 0.56 instead of 0.89. Also, for this latter exercise, total calibrated recharge losses from the dairy (518 kg/ha) are similar to the 525 kg/ha found with the optimally calibrated value.

4.3 Validation

The simulated concentration in the tile drainage system of 54 mg-N/l is only 10 percent higher than the average measured concentration of 49.6 mg-N/l. Differences may be caused by lower leaching rates in the area surrounding the study dairy than accounted for in the model-1b. Simulated response of nitrate concentrations in monitoring wells downgradient of fields with Targeted Manure Management is also in good agreement with the measurements (see figure 3). Overall we consider the testing results satisfactory given the reasonably simple model and the complexity of the system.

4.4 Spatial and temporal dynamics of N-leaching

Fields: The range of calibrated recharge concentrations for the various fields (F1 through F11) varies from 30.6 mg-N/l to 110.0 mg-N/l with associated N leaching rates ranging from 211 kg/ha/year to 758 kg/ha/year. The calibrated recharge concentrations are on average much lower for the fields that do not receive liquid manure (F4 and F10), due to lower overall application rates under commercially fertilized management.

Corrals: Existing research indicates that timing and the amount of N-application via livestock urine has a profound impact on concentrations leached to the aquifer (Pakrou & Dillon 2004; Shorten & Pleasant, 2007). Spatial variability within management units and the transient nature of the nitrate applications are not accounted for in our model. The highest residuals are found for concentrations in monitoring wells downgradient of corrals indicating that spatial variability of the leaching rate and or concentration is highest within this management unit.

Existing research on feedlots and corrals suggests that leaching of N is limited because reducing conditions keep nitrogen in the top layer in the ammonium form. Nitrified nitrogen is almost immediately denitrified. (e.g. Elliott *et al*, 1972). N movement can be very high in abandoned feedlots after oxidation of the organic mat (Mielke & Ellis, 1976). This conflicts with the findings in our study where we calibrated high N-concentration in corral recharge. Our findings are in line with previous research by Harter *et al.* (2002) who identified corrals as source based on the higher N-concentration in downgradient monitoring wells compared to upgradient monitoring wells.

Ponds: The calibrated recharge concentration from ponds (98 mg-N/l) is much lower than the measured total-N concentration in ponds, which ranged from 200 to 1000 mg-N/l. This means that part of the pond's N-recharge is lost due to sorption, volatilization, or denitrification in the shallow aquifer. The presence of both NH_4^+ and NO_3^- indicates that the monitoring well water is a mixture of pond leachate that travels under entirely anaerobic conditions, nitrified pond leachate and underlying groundwater with nitrate.

Denitrification may occur in the transition zone between the pond leakage and underlying groundwater. Alternatively, it has been observed that cracks develop in the lining after the twice-annually emptying and cleaning of the ponds allowing recharge of aerobic water until the pond is filled and anaerobic conditions are restored. This may cause an alternating aerobic and anaerobic environment that would allow for nitrification and subsequent denitrification of pond leachate. This is consistent with independent findings of Singleton *et al.* (2007) who found localized denitrification at ponds using isotope analysis of groundwater samples.

4.5 Comparison with Nitrogen mass balances at the field and farm scale

<u>Case specific field N-balance</u>: We compare the simulated nitrate leaching with an annualized nitrogen budget of fields F8 and F9 (see Table 5). Both fields were under conventional manure management until spring 1998 when a targeted manure management trial began (Harter *et al*., 2001a). Of the mass balance components, the manure nitrogen application prior to 1998 is considered to be the least reliable estimate and ranges from 900 kg/ha/year to 1,200 kg/ha/year (*ibid.*). Manure application on these two fields is not representative for all fields since it received relatively frequent irrigation with pond water.

The total N-loss to groundwater under conventional management was estimated by the N-balance to be at least 660 kg/ha/year compared to 760 kg/ha/year in both fields in the calibrated model. N-loss under targeted manure management in 1998-2000 was estimated 280 kg/ha/year compared to 300 kg/ha/year with the calibrated model. The N-losses to groundwater agree reasonably well given the significant uncertainty about actual manure applications prior to 1998 and given the confidence intervals of calibrated N-losses.

The agreement between the mass balance results and the calibrated model results suggest that the volatization and denitrification losses (set to zero in the mass balance) from below the root zone of the fields and from the top part of the aquifer are not significant. This is consistent with a field study by Singleton *et al.* (2007) in the same area who only found denitrification under ponds and under fields in a laterally extensive anoxic zone 5 m below the water table: below the bottom of the monitoring wells. Hence, under the absence of significant atmospheric losses (volatilization and denitrification), a field mass balance approach combined with known recharge rates provides a reasonable estimate of shallow groundwater concentration.

Whole farm N-balance by regulatory guidelines: Recently, The University of California Committee of Experts developed new guidelines to determine nitrogen losses to groundwater (UCCE, 2005). The guidelines are based on a whole farm N-mass balance in combination with field-by-field N-mass balances. Mass balance components are preferably measured on-farm but can also be estimated based on literature values.

In Table 6 we compared N-leaching estimates using the UCCE figures with the previous guidelines recommended by the regional water authority (CRWQCB). The new guidelines result in a much higher estimate. The difference is caused by the inclusion in the UCCE method of on-farm measurements of crop N-yield and use of more recent (higher) estimates of N-excretion rates per cow. Also, the new guidelines assume 30% volatilization losses from ponds, instead of 75% in the CRWQCB numbers.

The relatively close match between the calibrated N-loss of the groundwater model and UCCE guidelines indicates that farm N-balances, coupled with recharge estimates, provide a reasonable approximation of shallow groundwater nitrate or farm-scale nitrate losses in groundwater recharge.

Case-specific whole farm N-balance: We compiled a detailed mass balance of the dairy by combining the calibrated leaching rates and concentrations for groundwater losses with farm records and available literature data, in particular the UCCE (2005) report (see Table 7 and Figure 10). With groundwater losses known, volatilization losses from ponds were estimated as closure term in the mass balance. The 35% volatilization loss in the animal production area (including the ponds) is consistent with the estimated range of 20% to 40% given by UCCE (2005) and other literature reviews such as Liu *et al.* (1996) and vanHoorn *et al.* (1994).

Within the overall whole farm nitrogen balance, nitrogen leaching from fields comprises 45% of the whole farm field applied N [=P/ (B+C+S)]. This figure is much higher than the optimum 10% found with a modeling study of Feng et al. (2005). The inefficiency of the dairy farm is mainly caused by the low irrigation efficiency. The border-check flood irrigation system results in a 55% leaching loss of irrigation water. The 10% difference between nitrogen and irrigation losses is possibly caused by limited macropore flow. Other studies have found that this process enhances leaching of applied N to greater depth (e.g.Pakrou and Dillon, 1995). In these flood irrigation systems on relatively sandy soils, leaching losses must also be controlled by managing the nitrogen using split applications of manure and fertilizer.

5 Conclusion

In this paper, we propose a sequential calibration procedure for nitrate loading estimation in a dairy farm setting. The calibration is designed to accommodate a variety of groundwater monitoring data available at various spatial and temporal measurement scales. Data include long-term average head, and average as well as transient nitrate concentrations in an extensive monitoring well network; and fluxes and nitrate concentrations in a tile-drain network. Moreover, the various monitoring or measurement scales are different from the spatial scales of the calibration parameters, which are the recharge and nitrogen leaching rates from individual sources. The sequential calibration procedures with steady-state sub-regional flow and farm scale transport models provides a spatially varying map of dairy farming impacts on groundwater nitrogen. Thus, the physical groundwater flow and transport model offers a conceptual framework to cross-scale the multitude of field measurements for estimation of recharge N concentrations at the management unit scale (field, corral, pond).

Average nitrate-N losses are calibrated to 486 kg/ha/yr for fields, 872 kg/ha/year for corrals and 807 kg/ha/year for ponds. While the calibration provides recharge concentrations with relatively high confidence, the overall N loading rate (kg/ha/year) is strongly influenced by uncertainty of recharge rates, particularly for the corral area. Hence, loading rates are most accurately estimated for manure-treated fields, where recharge rates are known with relatively high accuracy. Independent measurements of corral and pond leaching are needed to better assess the loading rates from these manure management units.

We applied the results to evaluate the accuracy with which groundwater losses can be predicted from nitrogen mass balances, often used by regulatory agencies to assess potential groundwater impacts from nitrogen management. For individual fields the results of the N mass balance approach are in good agreement with those from the calibrated groundwater model. We also compare the up-scaled calibration results to a whole-farm mass balance approach developed for the Central Valley region by UCCE (2005) and find the results (434 kg/ha/yr) similar to those from the calibrated groundwater model. The amount of manure recovery in corrals averaged 70% and the 35% volatilization losses from ponds is well within the 20% to 40% regulatory recommendations, thus independently confirming these literature-based approaches.

The good agreement of the calibrated N leaching losses with those estimated from field and whole farm mass balances is partly due to the fact that apparent atmospheric losses in the field area (during and after manure application) are relatively small (5% or less). In areas with significant atmospheric losses (due to ammonia-volatilization and denitrification), mass balance approaches may significantly over-estimate groundwater losses unless atmospheric losses are measured independently.

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8 Figure captions

Figure 1. Monitoring wells and land use at and around the dairy farm in 1999.

Figure 2. Mean, standard error of the mean, and standard deviation of total-N in monitoring wells for each management unit.

Figure 3. Observed and simulated (model-1b) nitrate concentration in monitoring well downgradient of field F8.

Figure 4. Sensitivity of model parameters

Figure 5. Source area of monitoring wells, calculated with pathline analysis in MODPATH.

Figure 6. Simulated residual versus observed values from all monitoring well data.

Figure 7. Estimated average recharge rates and concentrations and their 95% linear confidence intervals for the parameters of the final calibrated model-2.

Figure 8. Nitrogen mass balance for dairy under investigation (kg N/yr/ha-forage field)

9 Table captions

Table 1. Overview of the applied models (TMM = targeted manure management)

Table 2. Model parameters estimated with prior information

Table 3. Parameter correlation matrix

Table 4. Summary statistics for the calibrated models (Hill and Tiedeman, 2006; Spitz & Moreno, 1996)

Table 5. Overview of individual field mass balance [kg N/ha/yr]. Adapted from Harter *et al*. (2001b)

Table 6. Generic nitrogen mass-balance of the dairy fields (kg N/ha/yr)

Table 7. Estimation of nitrogen mass balance components

Figure 1
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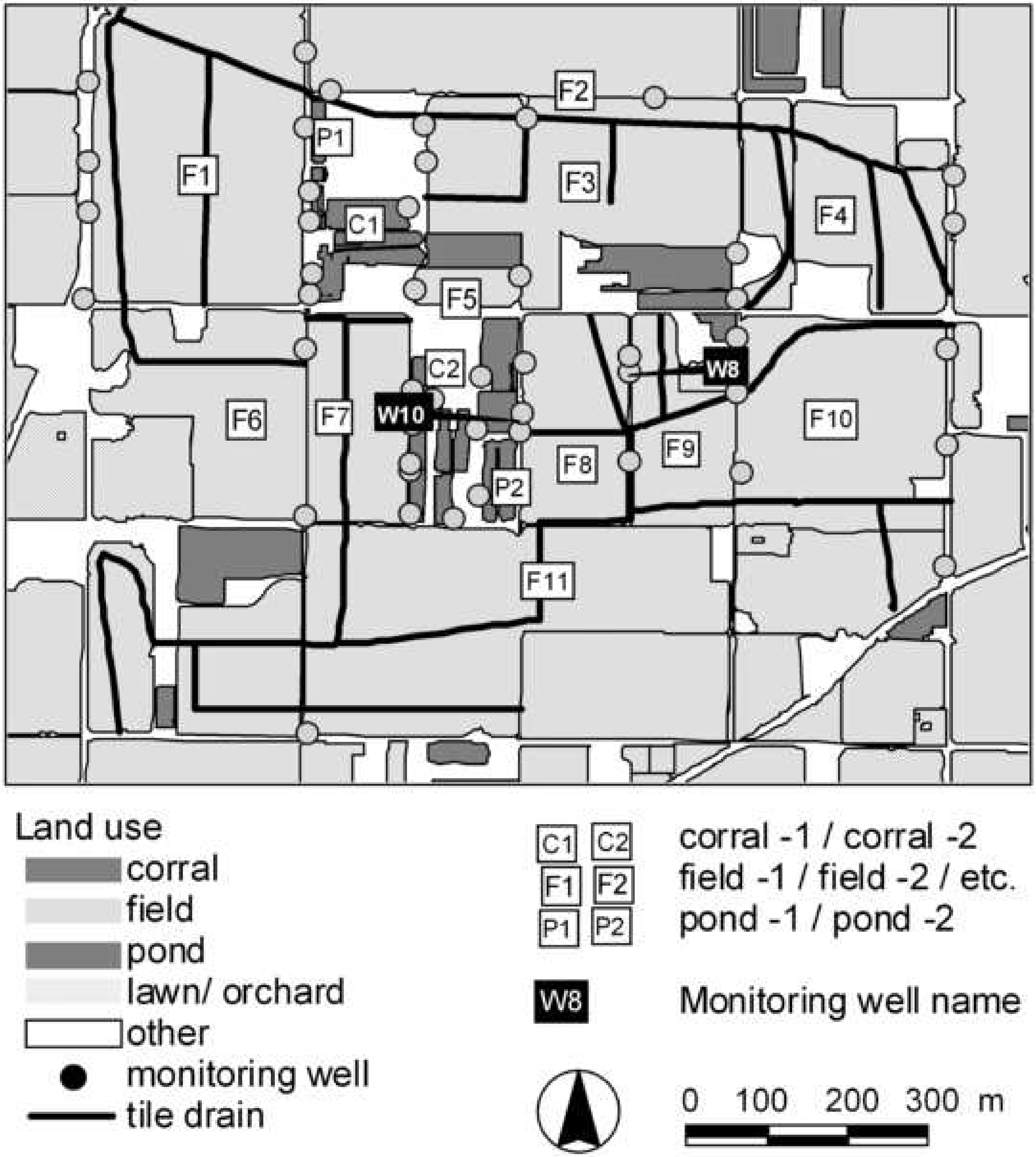

Figure 2
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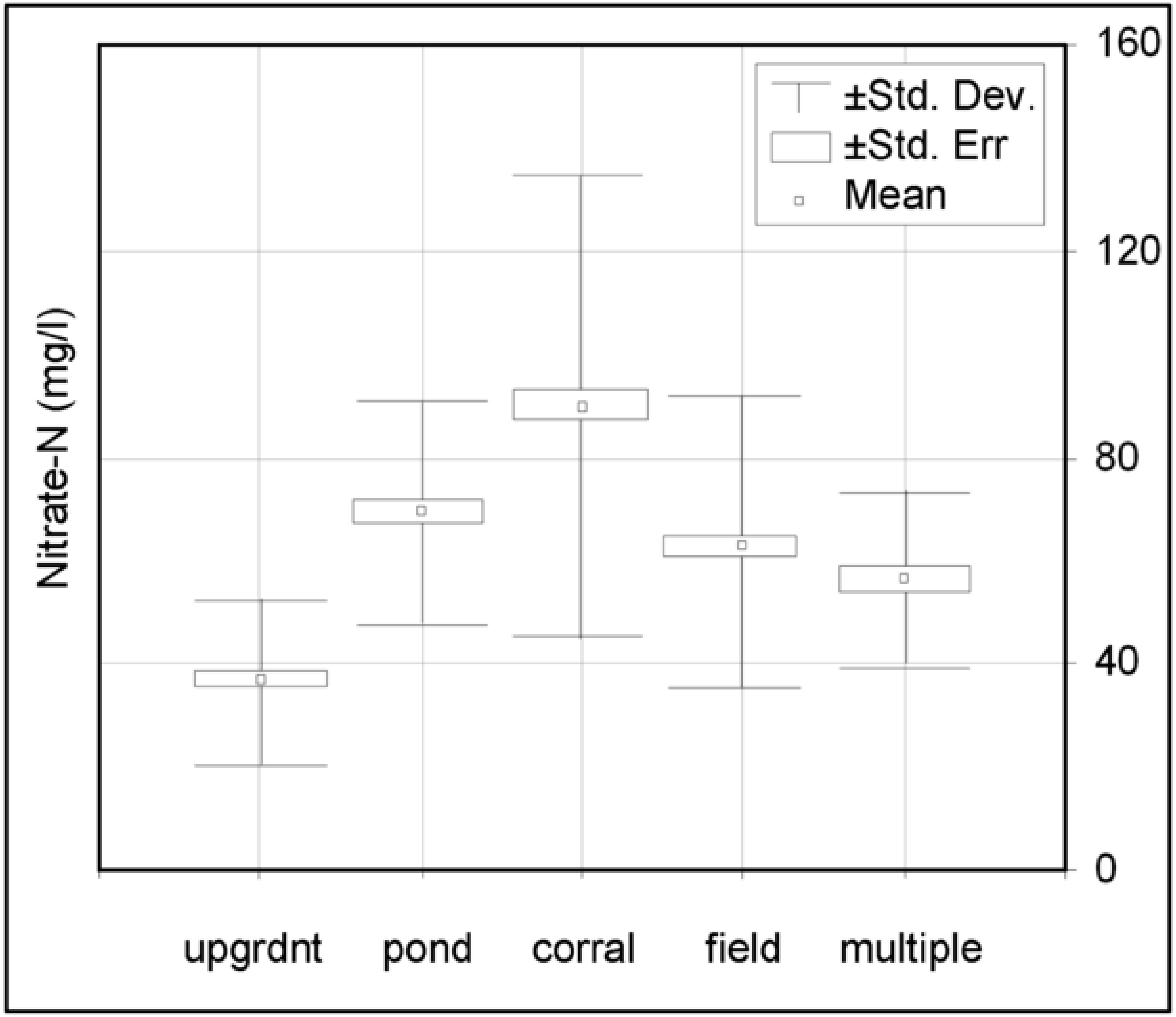

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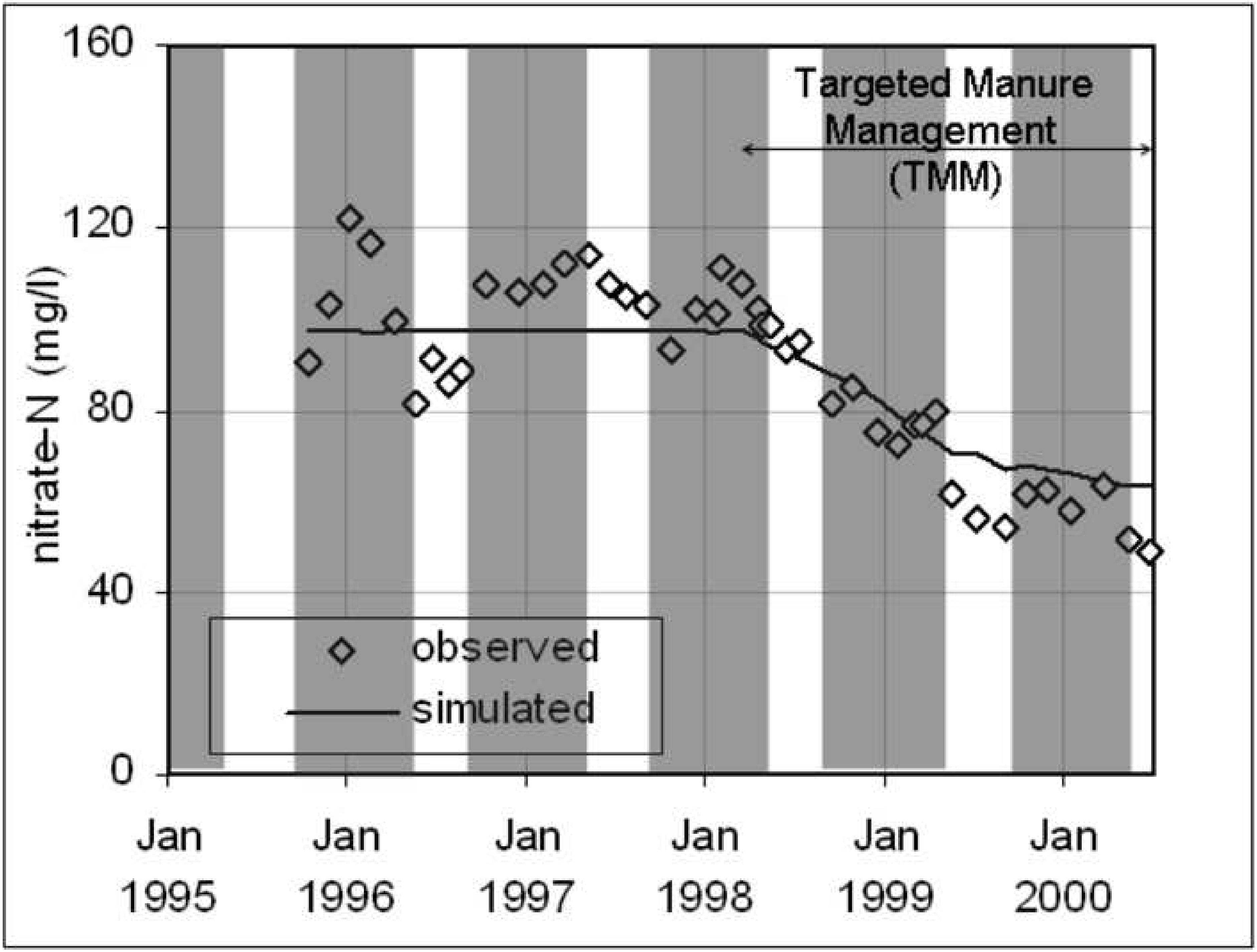

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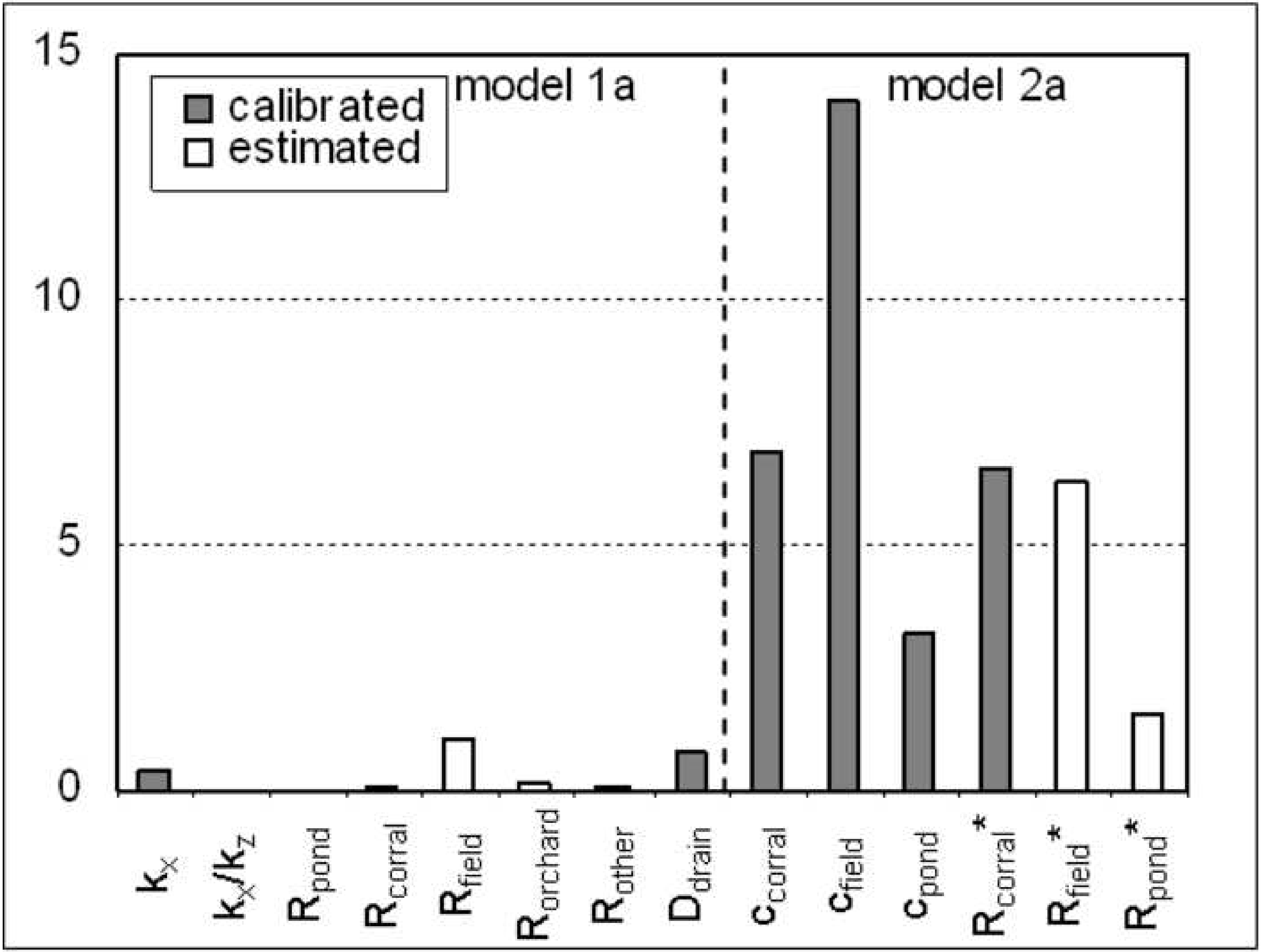

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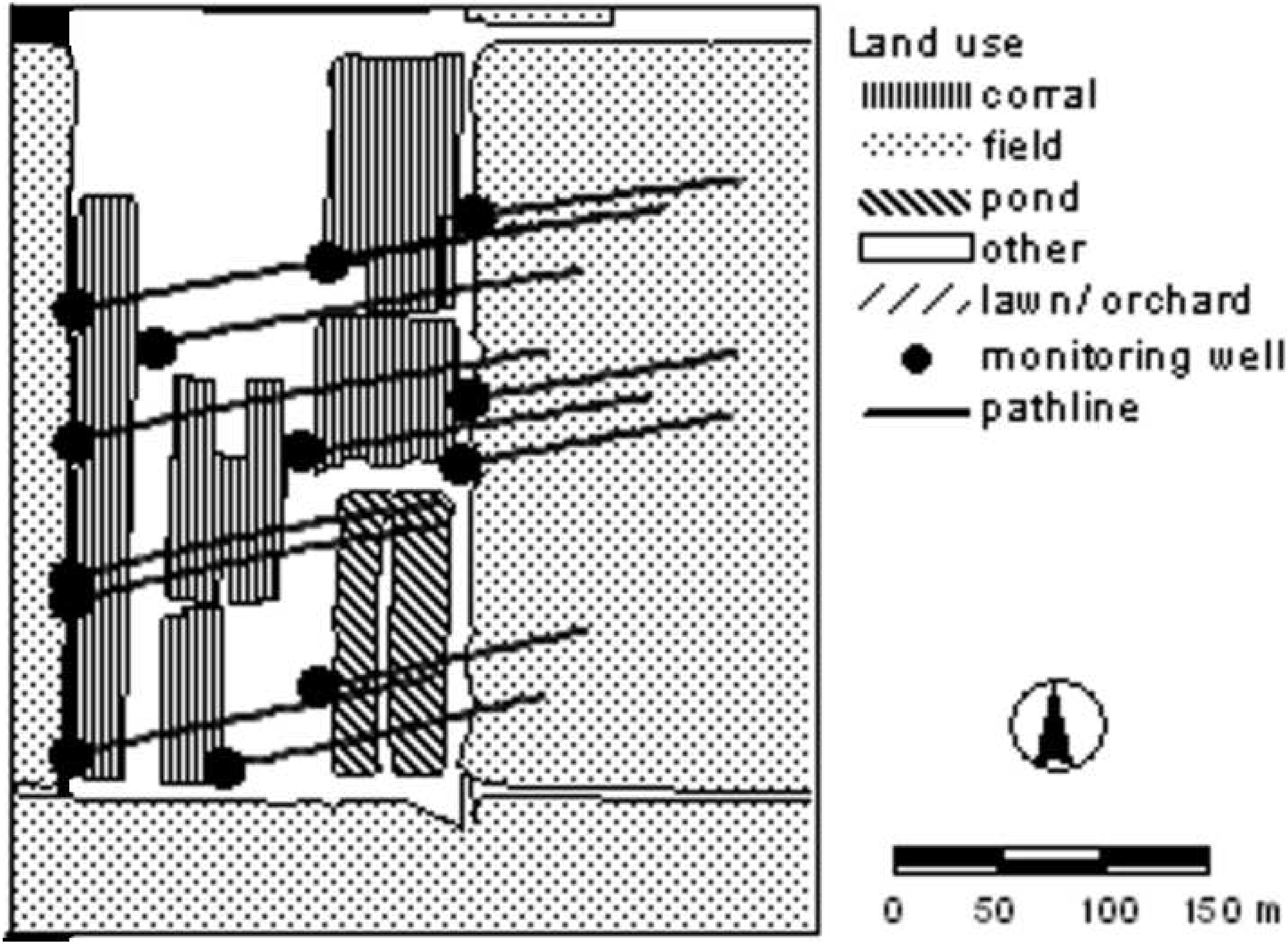

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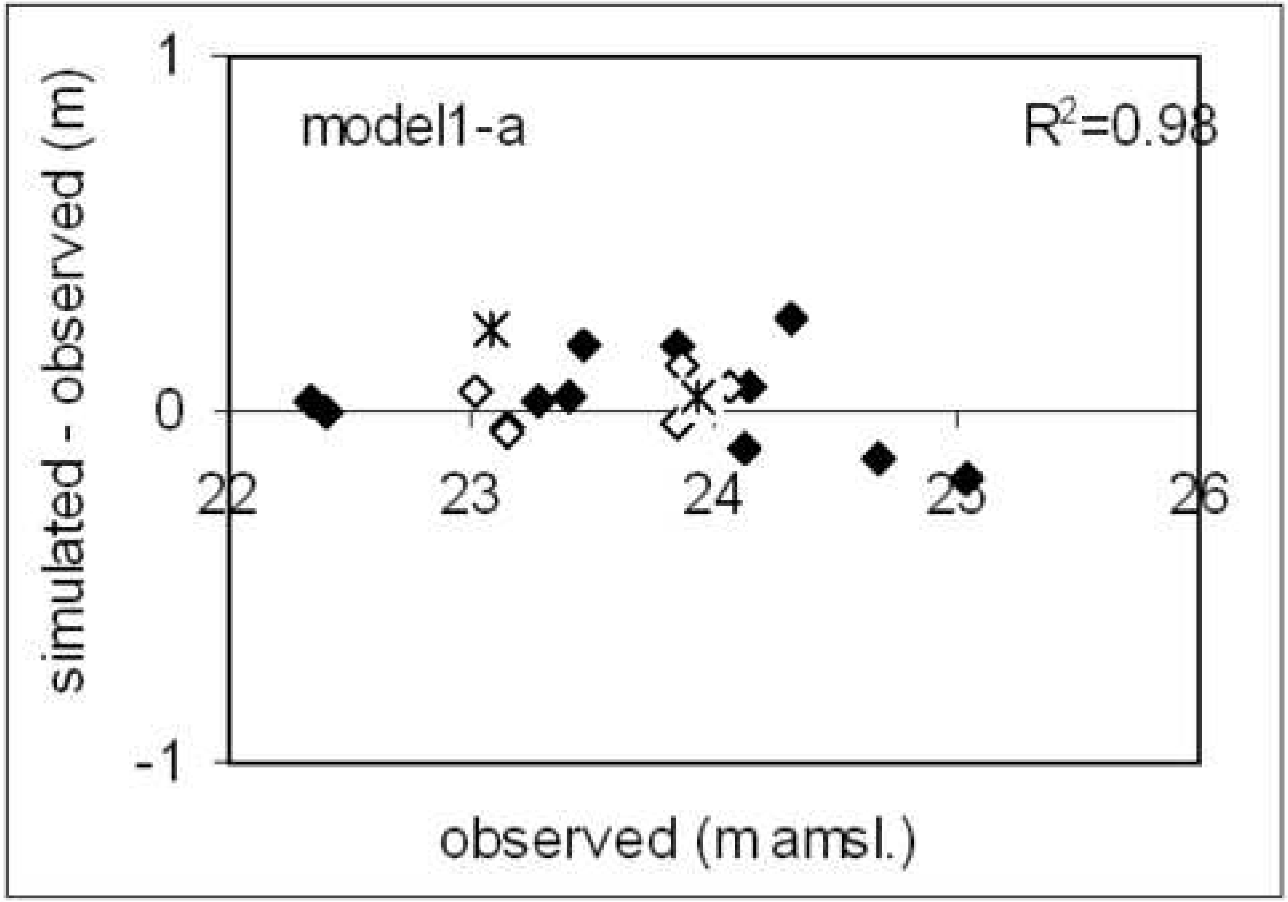

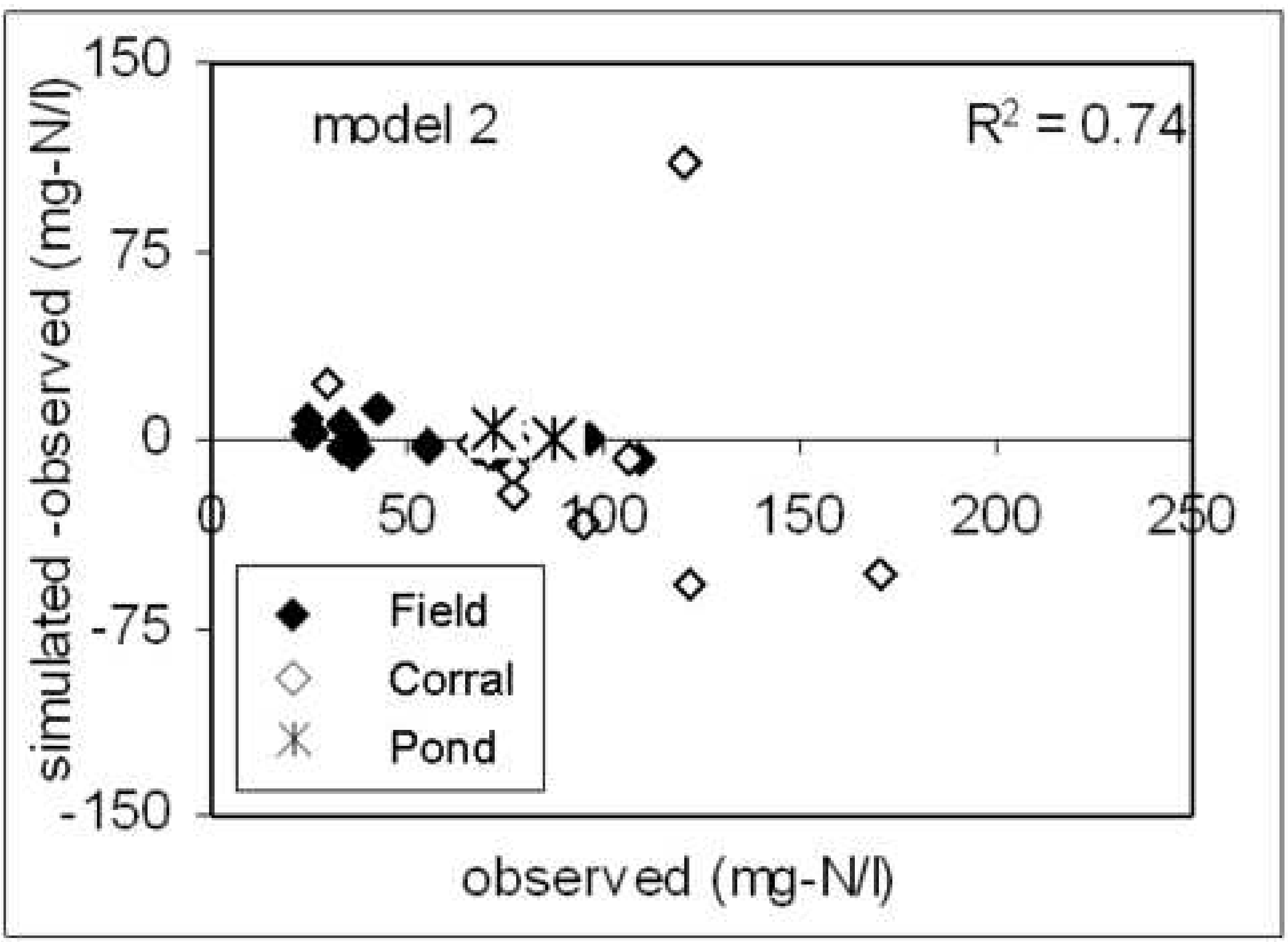

Figure 7

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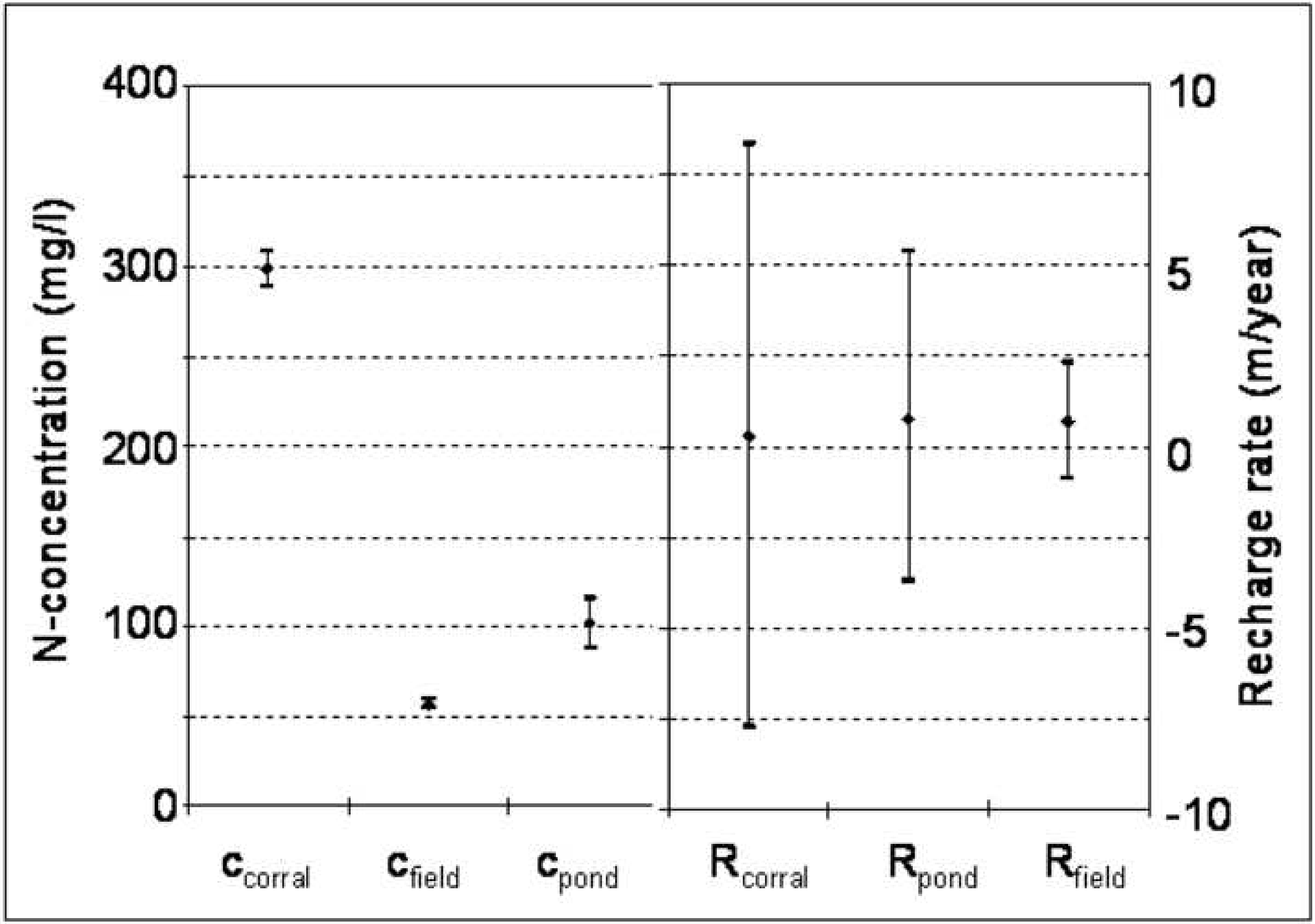

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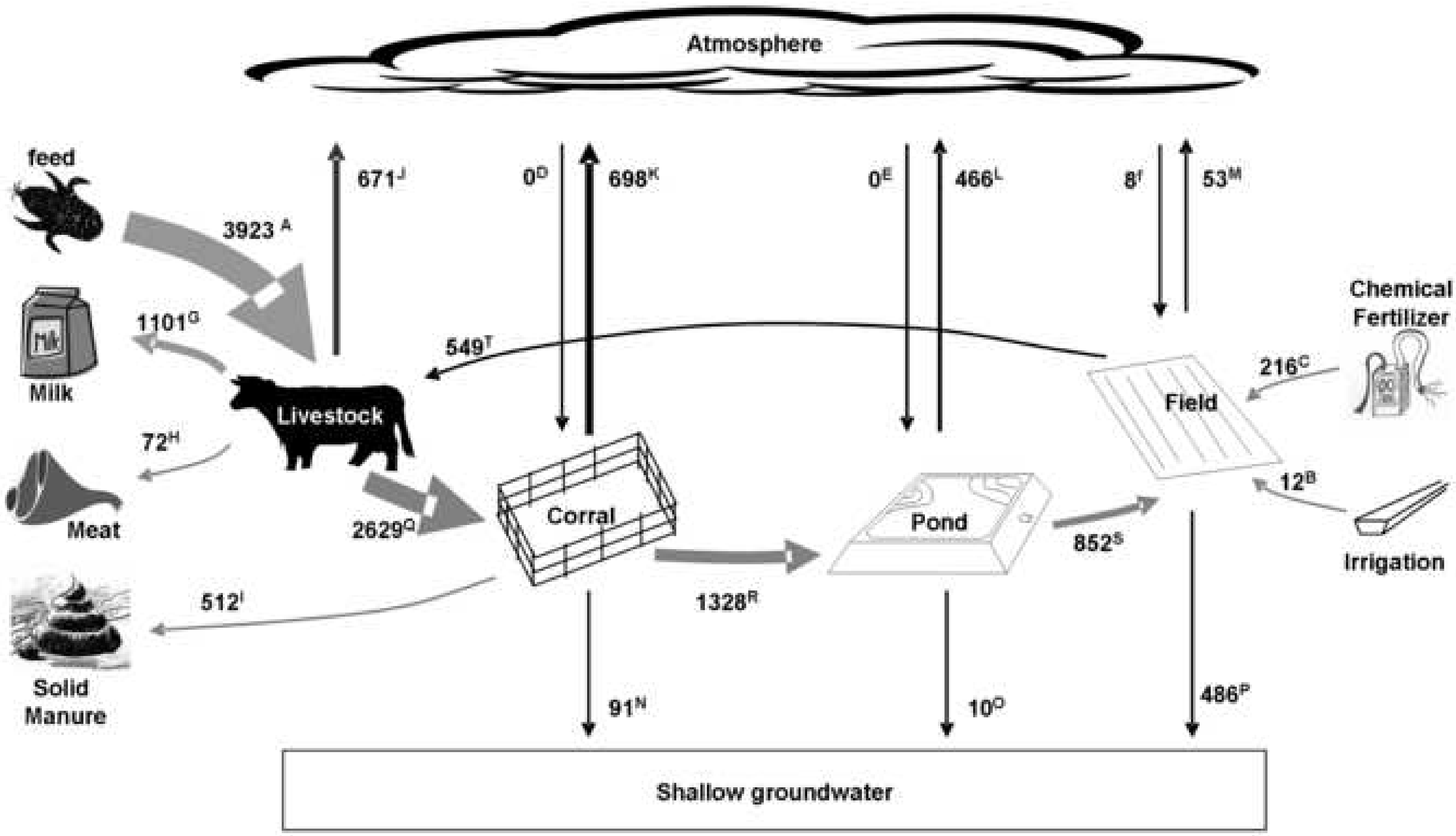

Table 1

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Table 1. Overview of the applied models (TMM = targeted manure management)

Model	*Type*	*Stress periods*	*Purpose*	*Calibration data*
Sub-regional scale model				
1a	Steady state flow	2 scenarios: pre- and post-drainage	Calibrate flow parameters (k_x, D_{drain}, R_{corral}, R_{pond})	Pre- and post-drain mean heads in 10 monitoring wells.
1b	Steady state flow Transient transport	3 consecutive scenarios: pre drainage without TMM, pre-drainage with TMM, post-drainage with TMM	Model validation	Average NO_3^- concentrations in tile drainage system; 122 transient NO_3^- observations in 6 monitoring wells.
Dairy scale model				
2	Steady state flow and long-term transient transport	1	Calibrate transport parameters (C_{field}, C_{corral}, C_{pond} , α_L)	Average NO_3^- concentrations in 47 monitoring wells.

Table 2

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Table 2. Model parameters estimated with prior information

Symbol	Description	Value	Comment and data source
R_f	Field recharge rate	0.69 m/yr	UCCGI, 2001; UCCE, 1987
R_o	Orchard recharge rate	0.46 m/yr	Estimated on basis of author's observations and meteorological data (UCCGI, 2001)
R_{ot}	Other recharge rate	0.29 m/yr	UCCGI, 2001
k_z	Vertical hydraulic conductivity	0.1 k_x	Spitz and Moreno, 1996
c_o	Orchard recharge concentration	3 mg-N/l	Natural background concentrations for dry and wet atmospheric deposition. ARL, 2001; NAPD, 2001
c_{ot}	"Other" recharge concentration	3 mg-N/l	*Ibid.*
n_e	Effective porosity	0.25	Spitz and Moreno, 1996; Validated with model-1b
α_L	Longitudinal dispersivity	1.5 m	Spitz and Moreno, 1996; Validated with model-1b
α_T	Transverse dispersivity	α_L /10	Spitz and Moreno, 1996; Validated with model-1b

Table 3

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Table 3. Parameter correlation matrix

	C_{corral}	C_{field}	C_{pond}	R_{corral}	R_{field}	R_{pond}
C_{corral}	1.00					
C_{field}	-0.31	1.00				
C_{pond}	-0.31	-0.15	1.00			
R_{corral}	-0.71	0.33	0.31	1.00		
R_{field}	-0.21	-0.48	0.12	0.12	1.00	
R_{pond}	0.25	0.14	-0.69	-0.25	-0.19	1.00

Table 4

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Table 4. Summary statistics for the calibrated models (Hill and Tiedeman 2006, Spits & Moreno 1996)

Statistics	Equation	Model-1a	Model-2
Residual mean	$\left(\sum_{i=1}^{nD} O_i - \sum S_i\right) \Big/ nD$	-0.1 m	2.1 mg-N/l
Model Efficiency	$EF = \dfrac{\sum_{i=1}^{n}(w_i(O_i - \bar{o}))^2 - \sum_{i=1}^{n}(w_i(O_i - S_i))^2}{\sum_{i=1}^{n}(w_i(O_i - \bar{o}))^2}$	0.95	0.89
Standard error of regression	$s^2 = \sqrt{\dfrac{\sum[w_i(O_i - S_i)]^2}{nD + nPR - nP}}$	0.56	1.30
Correlation coefficient	$R = \dfrac{\sum(O_i - \bar{O})(S_i - \bar{S})}{\sqrt{(O_i - \bar{O})^2(S_i - \bar{S})^2}}$	0.98	0.89

Table 5
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Table 5. Overview of individual field mass balance [kg/ha/yr]. Adapted from Harter et al., (2001b)

Mass balance component	Conventional	Targeted manure management
Manure application	900	620
Chemical fertilizer	280	130
Irrigation	12	12
Atmospheric deposition	8	8
Corn Yield	- 320	- 320
Winter grain Yield	- 220	- 170
Total (mass balance)	660	280
Total (model calibration)	760	300

Table 6
Click here to download Table: Table 6.doc

Table 6. Generic nitrogen mass-balance of the dairy fields (kg/ha/yr)

Mass balance component	UCCE 2005	CRWQCB 2001	Ground water model
Manure application to fields	929	414	
Chemical fertilizer	216	216	
Irrigation water	12	12	
Gaseous losses from fields	-174	0	
Corn Yield	-325	-280	
Wheat Yield	-225	-196	
Leaching losses	434	166	486

Table 7
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Table 7. Estimation of nitrogen mass balance components

ID	Component	kg N/year	kg N/ha/year	Remarks	Data source
External sources					
A	Purchased feed	347,659	3,923	Cow intake - harvest of forage crop (Y). Cow intake is estimated 0.563 kg/day (lactating cows), 0.270 (dry calves)	UCCE 2005
B	Irrigation water	1,099	12	Based on measured N-content of irrigation water (1 mg-N/l) and application of 1.24 m/year	Harter, 2001b; Lowry, 1987
C	Chemical fertilizer	19,142	216	Farm records	This study
D	Atmospheric deposition to corrals	0	0	Included in estimation of volatilization losses	
E	Atmospheric deposition to ponds	0	0	Included in estimation of volatilization losses	
F	Atmospheric deposition to fields	709	8	Literature values	ARL, 2000; NADP, 2001
Exports					
G	Milk production	97,527	1,101	Baed on estimated milk cow production of 0.154 kgN/day	UCCE 2005
H	Meat production	6,398	72	Product of annual cow sales (517) and N-content of cow (12.37 kg)	Belyea et al., 1978; Wright and Russel, 1984; Gibb et al, 1992.
I	solid manure sales	45,385	512	Estimated 50% of solid excrements are separated and sold	UCCE 2005
Losses					
J	Cow volatilization	59,447	671	15% of feed intake	Liu et al., 1996
K	Volatilization from corrals	61,844	698	Q + D - I - N – R	
L	Volatilization from ponds	41,267	466	R + E – O -S	
M	Gaseous losses from fields	4,730	53	Denitrification, soil NH3 loss, plant NH3 loss (5% of applied N)	
N	corral leaching	8,038	91	Groundwater model	This study
O	pond leaching	892	10	Groundwater model	This study
P	field leaching	43,081	486	Groundwater model	This study
Internal fluxes					
Q	Cow excrements	232,939	2,629	Product of cow excretion rates and population size. Cow excretion rates are 0.376 kgN/day (lactating cows) 0.184 kgN/day (Dry cows and bred heifers) 0.053 kgN/day (calves and cows < 1 year old)	UCCE 2005
R	Stored manure	117,673	1,328	70% of cow excrement - solid manure sales (I)	UCCE 2005
S	Manure application to fields	75,514	852	M + P + T – B – C – F	
T	harvest of forage crop	48,652	549	Onsite field measurements: 325 kg/ha/yr silage corn + 225 kg/ha/yr wheat	Harter 2001b
U	Mineralization	0	0	The soil N-content is considered at long term equilibrium	UCCE 2005