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Groundwater flow and transport of nutrients through a riparian meadow — Field data and modelling

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Model

Summary Groundwater flow and nutrient transport were studied in a riparian meadow during a three-year period. The meadow is situated along a first order stream in the River Gjern catchment area, Jutland, Denmark. Field data included measurements of hydraulic head, hydraulic conductivity and soil characteristics. Groundwater sampled from piezometers was analysed for nitrate, ammonium and phosphate. Nitrogen and phosphorus contents in above-ground plant biomass were also measured. For the interpretation of our data we developed a one-dimensional hydraulic-transport model for the lateral groundwater flow, transport of nitrate, and nitrate removal in the meadow. The model is based on Darcy's equation, and input data are horizontal and vertical distances, hydraulic heads, hydraulic conductivities, and nitrate concentrations. We also developed a scheme for evaluating uncertainties of the modeling results.

Annual removal of nitrate in the saturated zone of the riparian meadow was 326, 340, and 119 kg NO_3^--N ha^{-1} y^{-1} (59–68% of groundwater input) through the three-year period. The largest nitrate removal took place outside the growing seasons. Net loss of ammonium from the saturated zone was 0.4, 6.7, and 10.3 kg NH_4^+-N ha^{-1} y^{-1}. In two of the years this was counter-balanced by atmospheric nitrogen deposition. Phosphate was not retained during the first two years but lost at rates of 0.88 and 0.36 kg P ha^{-1} y^{-1}. In year 3 phosphate retention was 0.47 kg P ha^{-1} y^{-1}.

These data show how a riparian ecotone along a first order stream can reduces nitrogen pollution from agricultural areas. Also, the pronounced year to year variations in our nutrient budgets show that shorter studies, for example based on one year of observations, should be interpreted cautiously as representing a general picture of nutrient pathways.

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Introduction

Riparian wetlands or so-called buffer strips have received much attention due to their ability to remove and retain nutrients. Especially nitrate removal has been in focus (Peterjohn and Correl, 1984; Cooper, 1990; Haycock and Pinay, 1993). In riparian wetlands characterized by shallow lateral groundwater flow originating from upland areas discharging groundwater to a nearby stream, denitrification is believed to be the main process responsible for nitrate removal (Cooper, 1990; Schipper et al., 1993; Hoffmann et al., 2000; Vidon and Hill, 2004a). Denitrification has been determined using different approaches based on stable isotopes (Mariotti et al., 1988; Ostrom et al., 2002), soil sampling and immediate incubation with C_2H_2 (Cooper, 1990; Schipper et al., 1993), conservative tracer experiments with chloride and bromide (Jacobs and Gilliam, 1985; Smith et al., 1996), laboratory microcosmos experiments (Groffman et al., 1996; Hoffmann et al., 2000), and mass balance calculations (Haycock and Pinay, 1993).

Intensive field studies using multiple piezometer nests are time consuming to deploy and use, but they give detailed information about hydraulic heads, groundwater flow patterns, concentration of substances such as nitrate, biogeochemical interaction between waterborne substances, zones of enhanced denitrification, and finally, water and nutrient flows. Such quantifications are important, because they demonstrate linkages between inputs, responses and outputs of the riparian ecosystem. In other words, they throw light on the environmental benefits of natural riparian wetlands acting as buffer zones or ecotones between terrestrial and aquatic environments. This role of riparian wetlands is the focus in this paper.

Catchment area and study site Anbæk, Voldby Brook

The study was conducted in a riparian meadow at Voldby Brook (Fig. 1), a first order stream in the River Gjern catchment area, Jutland, Denmark (UTM Zone 32 ED 50 N 623689 E 552210). The catchment area of River Gjern is 114 km^2. The land use is mainly agriculture, 77.4%, while forests occupy 13.9%, towns and paved areas 4.6%, and meadows and wetlands 4.4%. Soil types are sandy loams, 61.2%, loam, 34.8% and loamy sand 4.0% (Svendsen et al., 1995 , Svendsen, personal communication).

The meadow bufferstrip is only 20—25 m wide, and is recharged by shallow lateral groundwater flow originating from an agricultural field, which was set aside in 1994. The groundwater level changes during the year between 0.2 and 1 m below the soil surface. The meadow was flooded once (approximately one month in February—March 1994) during the study period.

The meadow geology consists in the upper 30—50 cm of sandy sapric and hemic peat. Below this layer is 1—2 m of medium-grained sand with gravel and pebbles. The organic content is very sparse. Closer to Voldby Brook medium-grained sand alternates with fine-grained sand. The sandy layers are underlain by a low-permeable till, which consists of silty clayey sand (Figs. 2 and 3).

The dominant plant species in the meadow are *Dáctylis glomerata* L., *Phleum praténse* L., *Agrótis ténuis Sibth.*, *Urtica dioéca* and *Epilobium montánum* L., and with presence of *Achilléa millifólium*, *Myosótis palústris* L., *Stellaria graminea* L. and *Caltha palústris*.

Field design

A first picture of the groundwater flow through the meadow was obtained in a pilot study conducted in the period September 5, 1991 to October 24, 1991. Four piezometer nests with two piezometer levels at each station (6—9) were installed (Fig. 1). The measuring depths were 0.95—125 cm and 175—200 cm below soil surface of the meadow, respectively. The upper level was placed in a hydraulically active layer of mainly sandy deposits. The lower level was placed in a hydraulically inactive low-permeable clayey till. Hydraulic head hydrographs from the first month showed only minor gradient changes between measuring points. A horizontal equipotential map for the upper active level was then drawn. Based on this a permanent transect of four piezometer nests, which is not perpendicular to the brook, were then established along a groundwater flow path (Fig. 1). Measuring depths at these four stations (1—4) are described below. A continuous water level gauge was placed in Voldby Brook at the end of the transect (station 5). Hydraulic heads in all piezometers were then measured for half a month more and hydrographs analyzed. Head gradients were still quite stable in this period. A horizontal equipotential map for the active upper layer from the latest measuring date shows that the transect followed a groundwater flow line (Fig. 1). From the same date a vertical equipotential map (Fig. 2), was drawn along the transect from station 1 at the small hillslope to station 5 in Voldby Brook. The map shows a minor downward head gradient at the hillslope, horizontal gradients throughout the main part of the meadow, and small upward gradients close the brook.

The permanent transect established as a result of the pilot study, is 21 m long and follows the groundwater flow direction from the hillslope to the brook most of the year (i.e. there is no groundwater discharge a few days in summer). Each piezometer nest (station) was equipped with 3 or 4 polyethylene piezometers (PEH tubes) with 10 cm slotted well points (screens), placed at known depths above the low-permeable till (20—30 cm, 60—70 cm, 100—110 cm and 160—170 cm). Piezometer nest 1 was located at the top of a small slopehill (1 m high) and nest 2 only 3 m away from station 1 at the foot of the hillslope. Piezometer nest 3 was located in the middle of the riparian meadow 12 m from nest 1, while nest 4 was located next to the brook and 21 m from nest 1 (Fig. 3).

The water levels in the piezometers were measured with a Plexiglas tube with a tape measure attached. The Plexiglas tube was further connected to a piece of flexible rubber tube. By blowing air gently into the rubber tube and at the same time lowering the Plexiglas tube into the piezometer until it reaches the water table, it is possible to make accurate measurements of the water level (±2 mm), simply by placing a fingernail at the lip of the piezometer tube and press it against the tape measure, exactly when the tongue senses the air bubbles. The water level was measured in

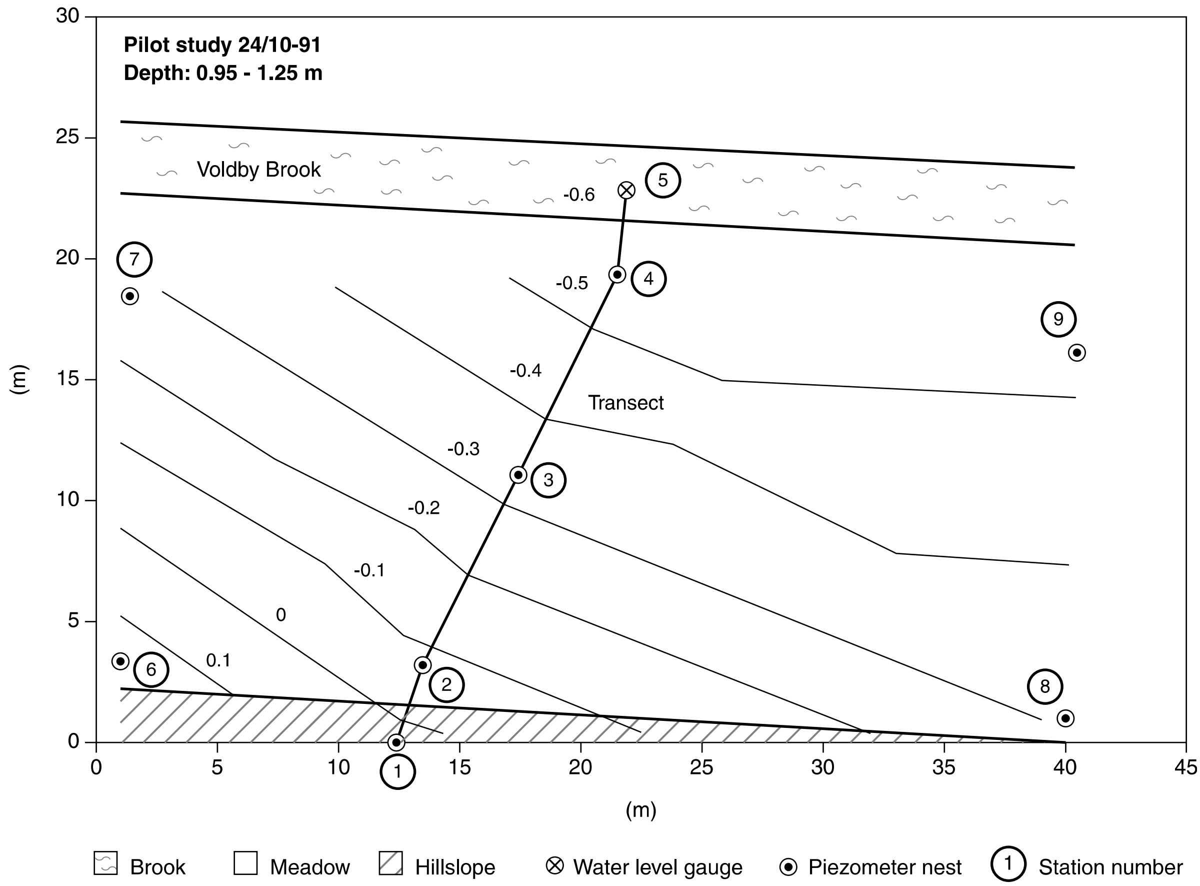

Figure 1 Horizontal equipotential map for the active upper layer. The transect with piezometer nests 1—4 was established according to the equipotential lines.

each piezometer once a week in the years 1992—1993 and once every two weeks in 1994.

Groundwater was sampled every second week in 1992—1993 and once a month in 1994. In 1992—1993 precipitation was measured weekly with a rain gauge (200 cm^2) with the orifice at ground level. Atmospheric deposition rates were determined from these precipitation measurements.

Water level in the stream was recorded weekly on a staff gauge. The topography of the riparian meadow and the stream was surveyed in detail, and the top of the piezometers were levelled in order to convert measured piezometer water levels to hydraulic heads.

Synchronous discharge measurements in Voldby Brook were performed upstream and downstream the meadow a few times in 1992 with propellers (Kleinflügel from Ott, Germany).

Methods

Groundwater was collected from piezometers using a submersible pump (Whale 921, Whale, Northern Ireland). To ensure sampling of fresh groundwater the piezometers were emptied the day before sampling. The samples were stored at 5 °C and analysed the following day as follows.

The samples were filtered through Whatman GF/C and analysed on a multi-channel flow-injection analyser for ammonium-N, nitrate-N, nitrite-N and phosphate-P (QuikChem automated ion analyzer from LACHAT INSTRUMENTS, US, methods 10-107-06-3-D for ammonium-N, 10-107-04-1-B for nitrite-N + nitrate-N and 10-115-01-1-B for phosphate-P). Nitrate $-$ N > 1 mg NO_3^--N l^{-1} was determined by ion chromatography on a Shimadzu HIC-6A chromatograph using an IC-A1S anion column (Shimadzu Corporation, Japan). pH was measured using a PHM 93 pH meter (Radiometer, Denmark). Precipitation samples were analysed for ammonium-N, nitrate-N + nitrite-N and phosphate-P using the same methods.

Above-ground primary production was measured monthly in 1992—1993. Two plots of 0.5 m^2 were harvested each time. The harvested material was divided in dead and alive biomass and oven-dried at 105 °C for at least 16 h. The dried plant material was weighted, and analysed for total nitrogen and total phosphorus content. Nitrogen was determined as Kjeldahl nitrogen (Jackson, 1958). Phosphorus was determined as phosphate-P by the molybdate method after ashing in a muffle furnace at 550 °C, and boiling the ash with 0.1 N HCl (Andersen, 1976).

Hydraulic conductivity was measured in the field using the piezometer method developed by Luthin and Kirkham (1949) for the saturated zone, and further described by Amoozegar and Warrick (1986). Calculation of hydraulic conductivity was done using the following equation:

$$K = \frac{\pi r^2 \{\log(H - d_1)/(H - d_2)\}}{C(t_2 - t_1)}, \qquad (1)$$

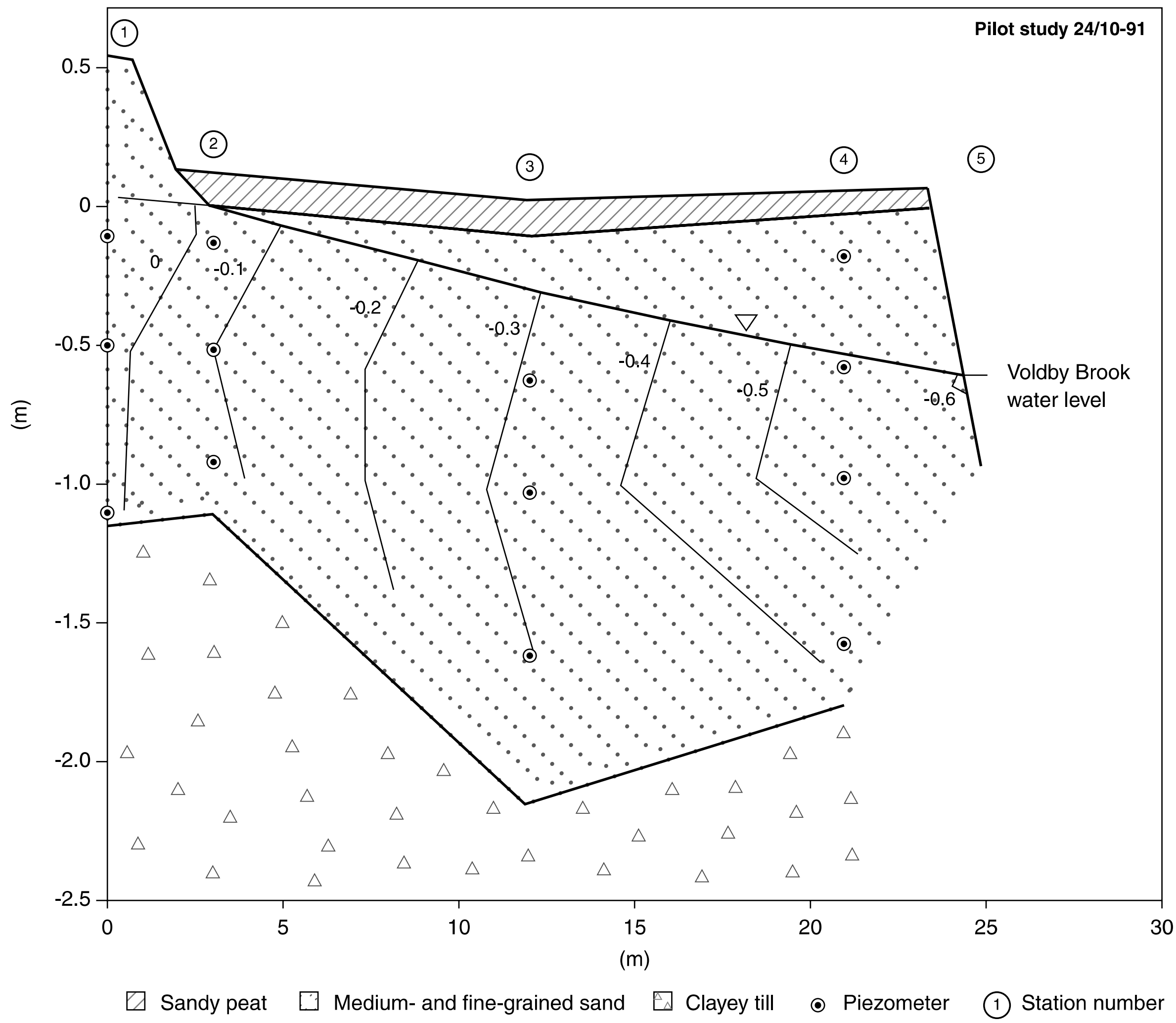

Figure 2 Vertical equipotential map, along the transect from station 1 at the small hillslope to station 5 in Voldby Brook. The map shows a small downward head gradient at the hillslope, approximately horizontal gradients through the main part of the meadow, and upward gradients close the brook.

where H is the distance from the water table to the cavity dug out under the tube and d_1 and d_2 is the distance from top of the piezometer to the water table during the measuring, at time, t_1 and time, t_2. The ''shape factor'', C, was taken from Youngs (1968).

Soil cores were sampled along the transect close to the four piezometer nests in order to get information about soil organic matter content which may be carbon source for the denitrification process. Soil sampling was performed with steel cylinders (50—60 cm long, 60 mm inner diameter). The cylinders were sharpened at the bottom end and a specially designed iron head was constructed to fit inside the top of the cylinder. The cylinder was fastened to the iron head by screws. The iron head had a hole in the middle with a rubber washer and a steel washer at the top, allowing water and air to flow out of the cylinders as soil drilling and sampling went deeper and deeper into the wetland soil. The iron head was mounted to ordinary soil drilling equipment (Eijkelkamp, Holland). The soil cores were sectioned in the field at the following depth intervals: 0—10 cm, 10—20 cm, 20—30 cm, 30—50 cm, 50—100 cm, 100—150 cm and 150—200 cm. Content of soil organic matter was determined as loss on ignition in muffle furnace maintained at 550 °C.

Dry density and porosity were determined from soil samples taken with a stainless steel cylinder as described above (500 mm long, and 60 mm inner diameter). After sampling, the cylinders were sealed at both ends with rubber stoppers, stored in an upright position and brought to the laboratory within a few hours. The soil cores were dried at 60 °C for one month, while still kept in the steel cylinders.

Hydraulic model

The model is based on Darcy's equation and calculates the horizontal groundwater flow between two piezometer nests. Fluctuations in groundwater level through the year are incorporated in this calculation. If concentrations of dissolved nutrients also are given as input, the model calculates the transport of these nutrients between the piezometer nests. Thus the model calculates the transport

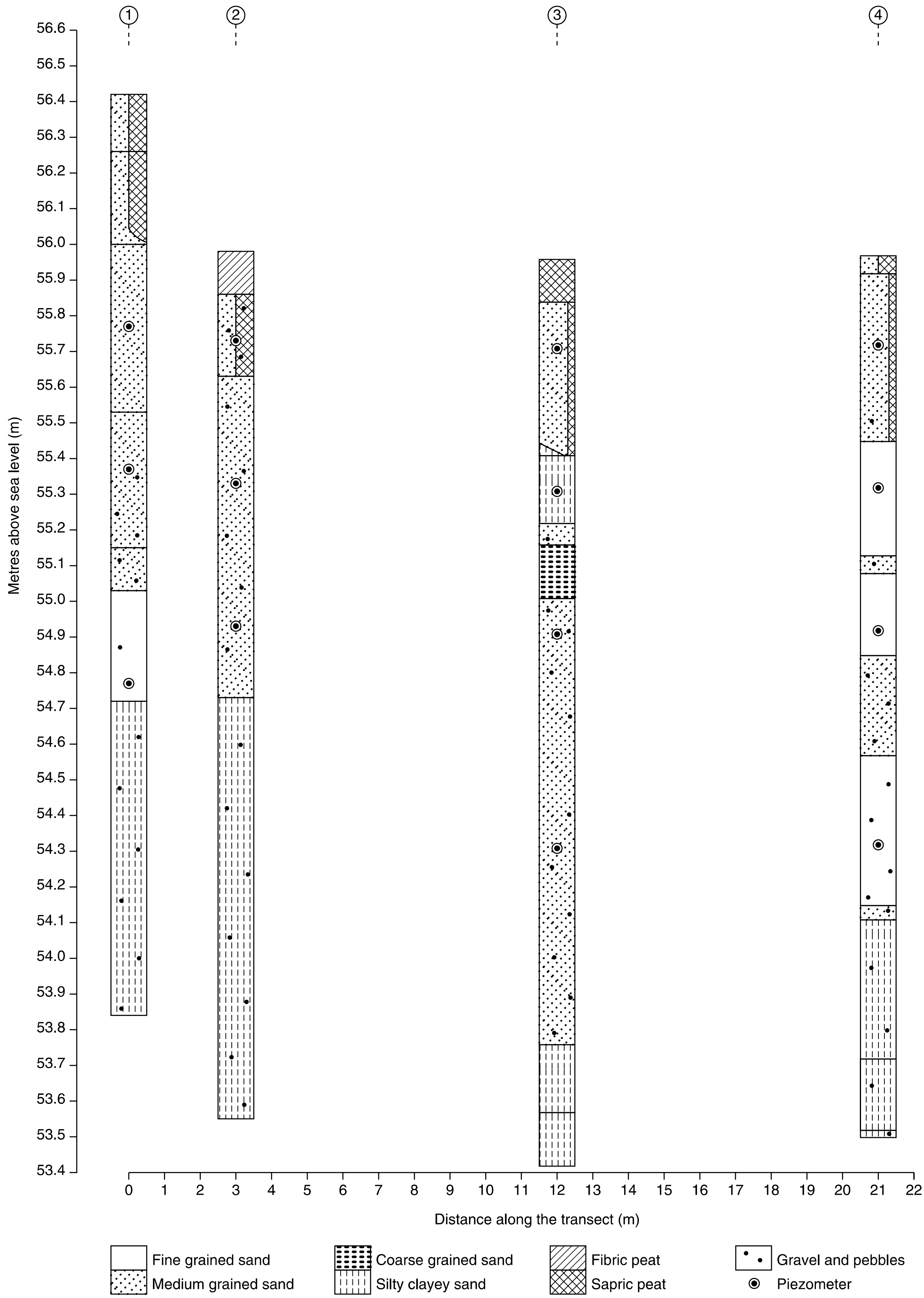

Figure 3 Geological profiles at the four piezometer nests (stations) in the riparian meadow.

of water and nutrients on measuring dates and subsequently monthly and yearly balances are calculated by linear interpolation between measuring dates. For dissolved nutrients one must consider carefully whether the element in question is removed from the system (e.g. nitrate through denitrification), retained in the system (e.g. phosphate being adsorbed to the soil matrix), accumulated in the system or leached out of the system. As a key element in our modelling effort, uncertainties of calculated water flows are estimated in the model based on assessed measurement errors of all input variables.

Necessary input data

The model input data needed to calculate the water flow and transport of a nutrient between station *a* and *b* is illustrated in Fig. 4 and listed below:

n number of layers,
L (m) horizontal distance between the two stations,
h_{ij} (m) thickness of layer for layer j at each station $i, i = a, b$ and $j = 1, 2, \ldots, n$,
K_j (m day^{-1}) hydraulic conductivity at layer $j, j = 1, 2, \ldots, n$,
φ_{ij} (m) hydraulic head at layer j and at each station $i, i = a, b$ and $j = 1, 2, \ldots, n$,
gw_i (m) groundwater level at each station $i, i = a, b$,
C_{ij} (mg l^{-1}) concentration of a dissolved substance measured at each station i for layer $j, i = a, b$ and $j = 1, 2, \ldots, n$.

Model description

The model calculates water flux (q_j) for each layer between two stations for every measurement date as

$$q_j = -K_j \frac{\varphi_{bj} - \varphi_{aj}}{L}, \ j = 1, 2, \ldots, n. \tag{2}$$

The total water flow (Q_j) from station a to station b in j is calculated as:

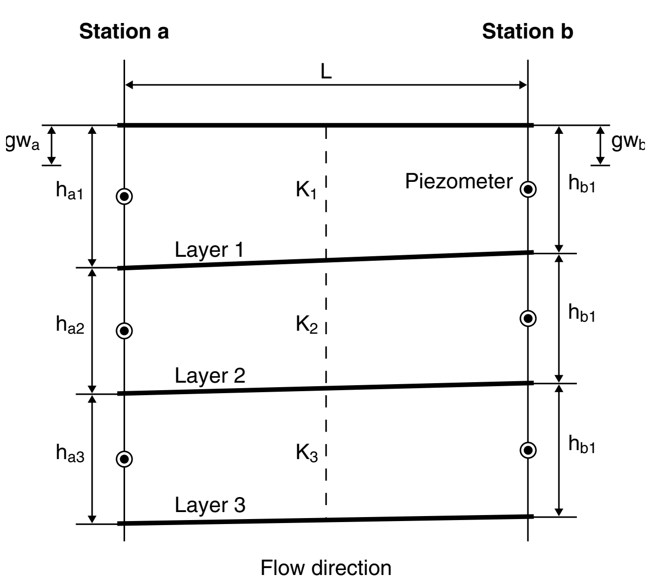

Figure 4 Schematic illustration of required model input data. Hydraulic heads are measured in piezometers located in the middle of each layer. The model calculates for each cross-sectional area groundwater flux across the vertical boundary indicated by the broken line.

$$Q_j = q_j \cdot A_j, \ j = 1, 2, \ldots, n, \tag{3}$$

where A_j is the cross-sectional area of layer j.

If the groundwater level is located in layer 1 the cross-sectional area is defined as:

$$A_1 = \frac{1}{2}(ha_1 + ha_2) - \frac{1}{2}(gw_a + gw_b),$$
$$A_j = \frac{1}{2}(ha_j + hb_j), \ j = 2, 3, \ldots, n. \tag{4}$$

If the groundwater level is located in layer 2 the cross-sectional area is defined as:

$$A_1 = 0$$
$$A_2 = \frac{1}{2}(h_{a2} + h_{b2}) - \frac{1}{2}(gw_a + gw_b) \tag{5}$$
$$A_j = \frac{1}{2}(h_{aj} + h_{bj}), \ j = 3, 4, \ldots, n,$$

and so forth. This implies that only layers that are located completely or partly beneath the groundwater level contribute to the water flow.

In situations where the groundwater level lies below a piezometer screen, the hydraulic head measured in the deeper layer is extrapolated upwards. This implies that hydraulic head for the lowest layer is always necessary for the model calculations. The model considers the measured hydraulic head as a vertical average for the respective layer. For that reason measuring heads in the middle of the layers give the most correct model calculations.

The net transport (i.e. removal, retention, accumulation or leaching) (R_j) of a dissolved substance is calculated as:

$$R_j = Q_j(C_{aj} - C_{bj}), \ j = 1, 2, \ldots, n, \tag{6}$$

where C_{aj} is the concentration at station a in layer j and C_{bj} is the concentration at station b in layer j. If a deep-lying groundwater level has prevented concentration measurements in a given layer, the concentration for the layer below is extrapolated upwards.

The total water flow for the whole riparian aquifer from layer 1 to layer n is calculated by adding the fluxes for each layer:

$$Q = \sum_{j=1}^{n} Q_j \tag{7}$$

The total net transport is calculated in the same way:

$$R = \sum_{j=1}^{n} R_j \tag{8}$$

Monthly and annual values of water flow and net transport of dissolved substances are calculated by time integrations over linear interpolations between measurement dates.

Uncertainty of model calculations

Uncertainty of the calculated water flow is estimated as follows.The water level was measured in the piezometers with an assessed uncertainty of 2 mm as mentioned above. The topographic leveling was performed with high accuracy, reading all measurements in millimeters. This is crucial because differences in water level are often very small, i.e. in the range from several millimeters to a few centimeters. The distance between the piezometers was measured using a steel measuring tape. Thus, we assumed that all distances have a measurement error of 2 mm, i.e.

$$s(L) = 0.002 \text{ m},$$
$$s(h_{ij}) = 0.002 \text{ m},$$
$$s(\varphi_{ij}) = 0.002 \text{ m},$$
$$s(gw_i) = 0.002 \text{ m},$$

where $i = a, b$ and $j = 1, 2, \ldots, n$

Furthermore, we estimated that the relative error of hydraulic conductivities is 15%. This result comes from replicate measurements of hydraulic conductivities (Table 1) using Eq. (1).

In the following we illustrate how the uncertainty for the model-calculated groundwater flow is estimated. A fundamental assumption is that all measurements needed for calculating the flow are independent. The formula for estimating the uncertainty for groundwater flux q_j according to Hoel et al. (1971) is:

$$s(q_j) = \left\{ \left(-\frac{K_j}{L} \cdot s(\varphi_{bj} - \varphi_{aj}) \right)^2 + \left((\varphi_{bj} - \varphi_{aj}) \cdot s\left(-\frac{K_j}{L} \right) \right)^2 + \left(s\left(-\frac{K_j}{L} \right) \cdot s(\varphi_{bj} - \varphi_{aj}) \right)^2 \right\}^{\frac{1}{2}},$$

where

$$s(\varphi_{bj} - \varphi_{aj}) = \{ s(\varphi_{rj})^2 + s(\varphi_{aj})^2 \}^{\frac{1}{2}} = 0.0283 \text{ m}$$

$$s\left(-\frac{K_j}{L} \right) = \frac{K_j}{L} \cdot \left\{ \left(\frac{s(K_j)}{K_j} \right)^2 + \left(\frac{s(L)}{L} \right)^2 \right\}^{\frac{1}{2}}.$$

The uncertainty for the total flow of a column, Q_j, from a to b in profile j can then be estimated as (Hoel et al., 1971):

$$s(Q_j) = \{ (q_j \cdot s(A_j))^2 + (A_j \cdot s(q_j))^2 + (s(q_j) \cdot s(A_j))^2 \}^{\frac{1}{2}},$$

where

$$s(A_1) = \left\{ \left(\frac{1}{2} s(h_{a1}) \right)^2 + \left(\frac{1}{2} s(h_{b1}) \right)^2 + \left(\frac{1}{2} s(gw_a) \right)^2 + \left(\frac{1}{2} s(gw_b) \right)^2 \right\}^{\frac{1}{2}} = 0.0283,$$

$$s(A_j) = \left\{ \left(\frac{1}{2} s(h_{aj}) \right)^2 + \left(\frac{1}{2} s(h_{bj}) \right)^2 \right\}^{\frac{1}{2}} = 0.0141, \quad \text{for } j = 2, 3, \ldots, n.$$

Finally, the total flow, Q, for the entire riparian aquifer has the uncertainty (Hoel et al. (1971) :

$$s(Q) = \left\{ \sum_{j=1}^{n} s(Q_j)^2 \right\}^{\frac{1}{2}}.$$

Results

Hydraulic conductivity, dry density and porosity

Hydraulic conductivities were measured at two locations, i.e. 2 m away from station 2 and close to station 3, respectively, and took place at four different depths (Table 1). At both locations medium-grained sand was dominating at depths 67–76 cm, 74–81 cm and 114–115 cm but with alternating content of gravel and pebbles (Table 1). Fine-grained sand dominated in the largest depth, 154–164 cm (Table 1). For all but one depth the hydraulic conductivity measurements were performed at least three times (Table 1). With the exception of the greatest depth with fine-grained sand all measurements showed some variation presumably because the cavity dug out beneath the piezometer did not maintain its shape during measuring. This problem was most pronounced at a depth of 114 cm and 115 cm. For that reason the length of the cavity at this depth had to be set to zero and the resulting calculated hydraulic conductivity from depth 114 cm had a mean of 33.5 m day^{-1} but exhibited a relative large range of 22.2–43.9 m day^{-1} (Table 1). At depth 115 cm similar problems occurred and only one measurement was performed. The total collapse of the cavity at a depth of 114 and 115 cm indicate a highly

Table 1 Hydraulic conductivity measured at different depths close to station 3 and 2 m away from station 2

Lithology	Depth (cm)	K (m day^{-1})	sd/$\bar{x}$ × 100%
Station 3: Medium-grained sand with pebbles and gravel, dark grey (2,5Y4/1,v)	67–76	37.5 (32.3–44.0)	14
Station 3: Medium-grained sand with gravel, dark grey (2,5Y4/1,v)	74–81	20.9 (18.4–22.1)	10
Station 2: Medium-grained sand with pebbles and gravel, yellow-brownish (10YR/8,v)	114	33.5 (22.2–43.9)	27
Station 2: Medium-grained sand with pebbles and gravel, yellow-brownish (10YR/8,v)	115	21.5	–
Station 2: Fine-grained silty sand with gravel and few pebbles, gleyey (5Y6/1,v–5YR4/8,v)	154–164	12.1 (11.7–12.4)	3

Code in parenthesis after lithology indicates colour according to Munsell's soil color charts (1994). Figures in parenthesis under K indicate range in measurements. The relative error (%) at different depths shown as the coefficient of variation, denoted sd/$\bar{x}$ × 100.

permeable zone. The highest conductivity was found at station 3 at a depth of 67—74 cm, with a *K*-value of 37.5 m day^{-1} (Table 1), but since this depth interval is often located above the groundwater table it is of minor significance for the overall water flow.

Wet and dry density was determined on two cores sampled at station 3 at a depth of 85—125 cm. The cores had a length of 40 cm. Wet density was 1.932 g cm^{-3} and dry density 1.496 g cm^{-3}. The porosity was calculated as 38.3%.

Loss on ignition

The organic matter content (carbon source for denitrification) expressed as loss on ignition was highest in the root zone at a depth of 0—30 cm (Fig. 5). Values between 4.4% and 13.9% were found. At all four stations organic matter content decreased with depth although values remained higher at station 3 and 4 in the depth interval of 30—100 cm as compared to stations 1 and 2. Below a depth of 100 cm loss on ignition is low varying between 1.0% and 2.4%.

Hydraulic heads and stream water level

The water level in Voldby brook and the hydraulic heads measured through the 3-year period at different depths at the four piezometer nests are shown in Fig. 6. The water in piezometer nest 1 represents groundwater recharging the meadow from the agricultural upland. Groundwater is discharged to the brook from piezometer nest 4 located 2 m from the brook. The hydraulic gradient between stations 1 and 4 shows a mean of 21‰ and a variation between 5‰ and 35‰.

All hydraulic heads show the same annual fluctuation in all years with high groundwater heads during winter, decreasing heads during spring, low heads in summer, and increasing heads during autumn. At stations 3 and 4 the water level was often below the two upper piezometric screen depths at 25 cm and 65 cm, especially in the spring, summer and autumn. For that reason there are fewer measurements from these depths. Mean groundwater level in the meadow at stations 2—4 was 0.3, 0.5 and 0.6 m below ground. It is noticeable that during summer 1992 and 1994 the brook occasionally recharges the meadow as the water level in the brook is higher than the groundwater heads at piezometer station 4, and also a few times at station 3. During these periods the main body of the meadow was recharged by both groundwater from the upland and surface water from the brook. This means that the flow path is out of function temporarily in dry periods in summer when the groundwater level is at its deepest position belowground. As the water level in the piezometers does not increase groundwater flow in the meadow takes another direction, and presumably flows along the meadow more or less parallel with the stream.

In general the hydraulic heads at the different depths at each of the piezometer nests are almost identical. However, at few dates there are vertical gradients at some stations. For example, at station 1 in summer and autumn of 1992 there seems to be a vertical gradient especially between depth 105 cm and depth 165 cm during the decrease and increase in the groundwater table, but this pattern does not continue in the following years. This might be caused by malfunctioning of one of the screens (e.g. clogging). At station 3, at a depth of 25 cm the hydraulic head is also different at certain dates, but the difference makes no clear pattern as the hydraulic head sometimes is above and sometimes below the hydraulic heads from the other depths. The most likely explanation is that the screen at depth 25 cm was malfunctioning at certain times during the study. Comparing the hydraulic heads at station 4 at depths 165 and 105 cm show that the head at depth 165 cm is higher 122 times out of 139 (Fig. 6). Mean difference in hydraulic head between the two depths is 6.3 ± 0.5 cm (±SE, n = 139). The upward

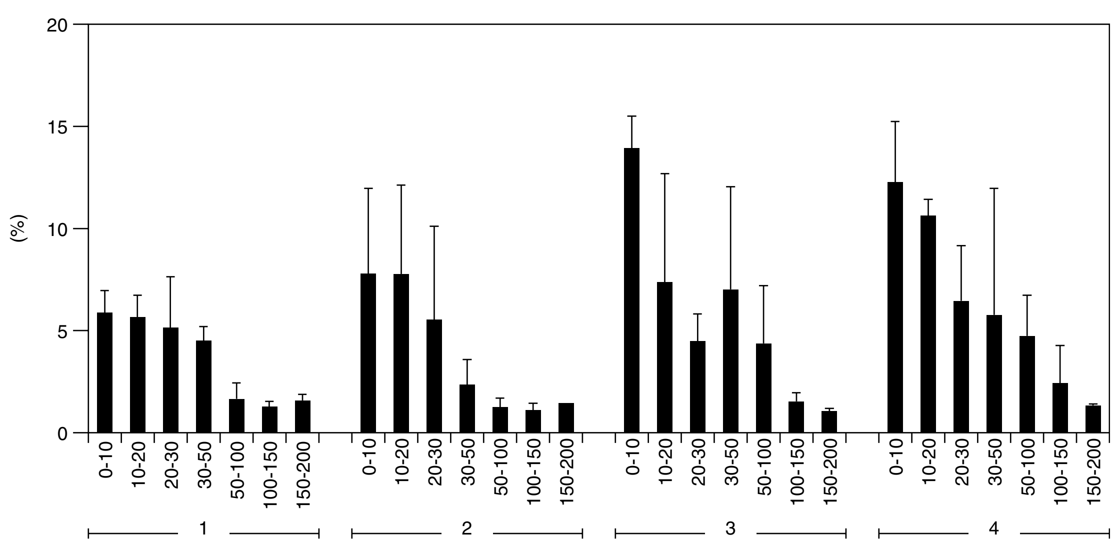

Figure 5 Loss on ignition at each station measured in the following depths: 0—10, 10—20, 30—50, 50—100, 100—150 and 150—200 cm. Number of samples n = 3, except for station 2, depth 150—200 cm. Shown with 95% confidence limits.

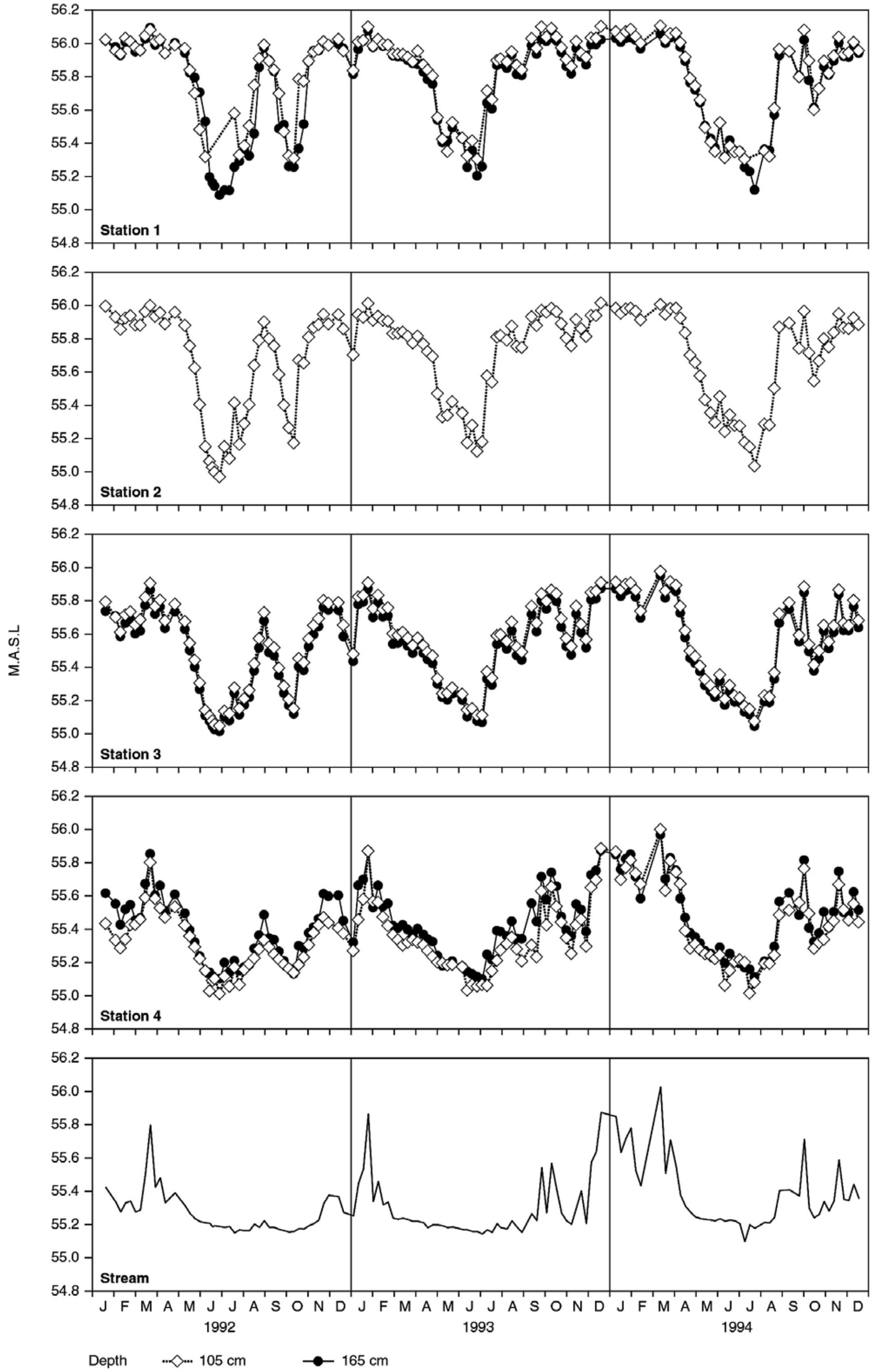

Figure 6 Hydraulic heads at different depths at piezometer nest 1—4 (stations 1—4) and water level in Voldby Brook. At station 1 the ground surface lies at 56.41 m.a.s.l., and at stations 2—4 the soil surface lies at 55.97, 55.92 and 55.93 m.a.s.l., respectively.

move of groundwater at this location of the meadow, only 2 m away from the brook indicates that groundwater is discharged to the brook. This is in accordance with the results from the pilot study (Fig. 2). An exception from this are the dry periods in summer 1992 and 1994 mentioned above. Further, it seems reasonable to model the groundwater flow as horizontal one-dimensional flow along the transect from the hillslope to the stream (i.e. from stations 1 to 4) above the low-permeable bottom till layer, although the two dry periods have to be treated separately as the fate of groundwater recharging the meadow from the upland field and stream water recharging the meadow from the brook remains unknown.

Precipitation

The annual precipitation in 1992 and 1993 was 711 mm and 852 mm, respectively. These numbers agree well with data from a nearby meteorological station (Hammel) where the annual precipitation was measured to 755 mm and 892 mm, (Danish Meteorological Institute). The 30-year annual normal for this station is 809 mm. Precipitation data for 1994 used in this study was acquired from Hammel meteorological station, and amounted to 1076 mm per year.

Nutrients in groundwater

Concentrations of nitrate-N at the four stations and in Voldby Brook are shown in Fig. 7. All stations showed an annual pattern with low concentrations in summer and beginning of autumn while concentrations were higher in winter and spring. From autumn 1993 through 1994 there were generally lower nitrate concentrations at all four stations. This overall pattern coincide with a change in farming practice as the upland field recharging the meadow was set aside in 1994; i.e. the field was neither fertilized nor tillaged after autumn 1993. Comparing nitrate concentrations between stations and depths reveals that in year 1992 station 1 and 2 had very similar concentrations at depths 65 and 105 cm. In 1993 and 1994 nitrate concentration at station 2 were a little lower at all depths compared to station 1. Moving from station 2 to 3 and from station 3 to 4 nitrate concentrations gradually decrease at depth 105 cm. The same pattern was also seen at depth 165 cm.

Relative to nitrate, ammonium was only found in small concentrations below the groundwater table (Fig. 8). At station 3, depth 65 cm, ammonium concentration fluctuated around 0.5–0.6 mg NH_4^+-N l^{-1}. This is significantly higher than values from all other stations and depths. At station 4, depth 105 cm mean ammonium concentration increase from 0.19 in 1992 to 0.25 and 0.41 mg NH_4^+-N l^{-1} in 1993 and 1994, respectively. At the other depths at stations 2–4 ammonium concentration rarely exceeded 0.05 mg NH_4^+-N l^{-1}.

Phosphate concentrations were generally very low. Most depths show mean concentrations of phosphate-P below 0.010 mg PO_4^{3-}-P l^{-1}. At station 3, depth 65 cm, phosphate concentration was higher in autumn 1993 (0.008–0.103 mg PO_4^{3-}-P l^{-1}). At station 4, depth 165 cm, the mean annual concentration exceeded 0.02 mg PO_4^{3-}-P l^{-1}, with 0.021, 0.027 and 0.021 mg PO_4^{3-}-P l^{-1} for the years 1992–1993–1994, respectively. For all three years mean concentration (±SE) at stations 1–4 was 0.007 ± 0.001 (n = 110), 0.009 ± 0.001 (n = 126), 0.012 ± 0.001 (n = 151) and 0.018 ± 0.002 ($n = 82$) mg PO_4^{3-}-P l^{-1}.

Nitrate concentration in precipitation was stable during the whole three year study period, with a mean of 0.674 ± 0.069 mg NO_3^--N l^{-1} (n = 85 ± SE). Ammonium concentration was 1.035 ± 0.115 mg NH_4^+-N l^{-1} (n = 33 ± SE) in 1992 and increased to 1.825 ± 0.504 mg NH_4^+-N l^{-1} (n = 21 ± SE) in 1993 and decreased to 0.506 ± 0.113 mg NH_4^+-N l^{-1} (n = 27 ± SE) in 1994. Phosphate concentration in precipitation was 0.063 ± 0.014 mg PO_4^{3-}-P l^{-1} (n = 33 ± SE) and 0.054 ± 0.014 mg PO_4^{3-}-P l^{-1} (n = 21 ± SE) in 1992 and 1993, respectively. In 1994 mean annual phosphate concentration decreased to 0.018 ± 0.007 mg PO_4^{3-}-P l^{-1} (n = 27± SE).

Nutrients in above-ground biomass

Alive and dead aboveground biomass was measured monthly from April 1992 to October 1993. Both fractions varied between the years (Table 2), probably because the meadow was grazed until the end of summer 1991, but was neither grazed nor cut during the study period from 1992–94. The amount of dead biomass was much smaller in 1992 as compared to 1993 with maximum dead aboveground biomass of 680 g m^{-2} in March 1993 (Table 2).

The nitrogen content in alive above-ground biomass increased from 2.28 g N m^{-2} on April 8, 1992 to 12.98 g N m^{-2} on October 19, 1992, which equals a daily uptake rate of 55 mg N m^{-2} day^{-1} (Table 2). In 1993 the lowest content was found on March 23 with 4.97 g N m^{-2} and the highest content on August 9 with 9.07 g N m^{-2}, equivalent with a daily uptake rate of 29 mg N m^{-2} day^{-1} (Table 2). The higher amount of dead aboveground biomass on the meadow in 1993 was also reflected in the amount nitrogen, which was much higher, especially in summer and autumn (Table 2).

In 1992 the lowest phosphorus content, 0.15 g P m^{-2}, in above-ground alive biomass was found on April 8 and the highest content, 1.46 g P m^{-2}, on July 27. This corresponds to a daily uptake rate of 11.9 mg P m^{-2} day^{-1} (Table 2). In 1993 the phosphorus content in above-ground alive biomass increased from 0.64 g P m^{-2} on March 23 to 1.38 g P m^{-2} on August 9, equivalent with a daily uptake rate of 5.3 mg P m^{-2} day^{-1} (Table 2).

Model calculations

Groundwater flow

The model was used to calculate the groundwater flow across the boundary between two stations, forming one sectional area. The flow calculations between station 1 and 2 (*flow1 of sectional area 1*), between station 2 and 3 (*flow2 of sectional area 2*), and between station 3 and 4 (*flow3 of sectional area 3*) were carried out independently of one another.

Input data to the three model sectional areas were: horizontal distance between stations (Fig. 3), vertical extent of each layer included in the model (Table 3). Because the width of the transect was set to 1 m groundwater flow is given in m^3 per meter meadow.

For every measuring date (i.e. once a week in 1992–1993 and every two weeks in 1994) the groundwater table is input to the model, and thus the upper boundary of the model changes (or may change) from measuring date to measuring date. The dynamic water table is obviously an approximation to increasing or decreasing groundwater flow (recharge from the upland) as well as precipitation or evapotranspira-

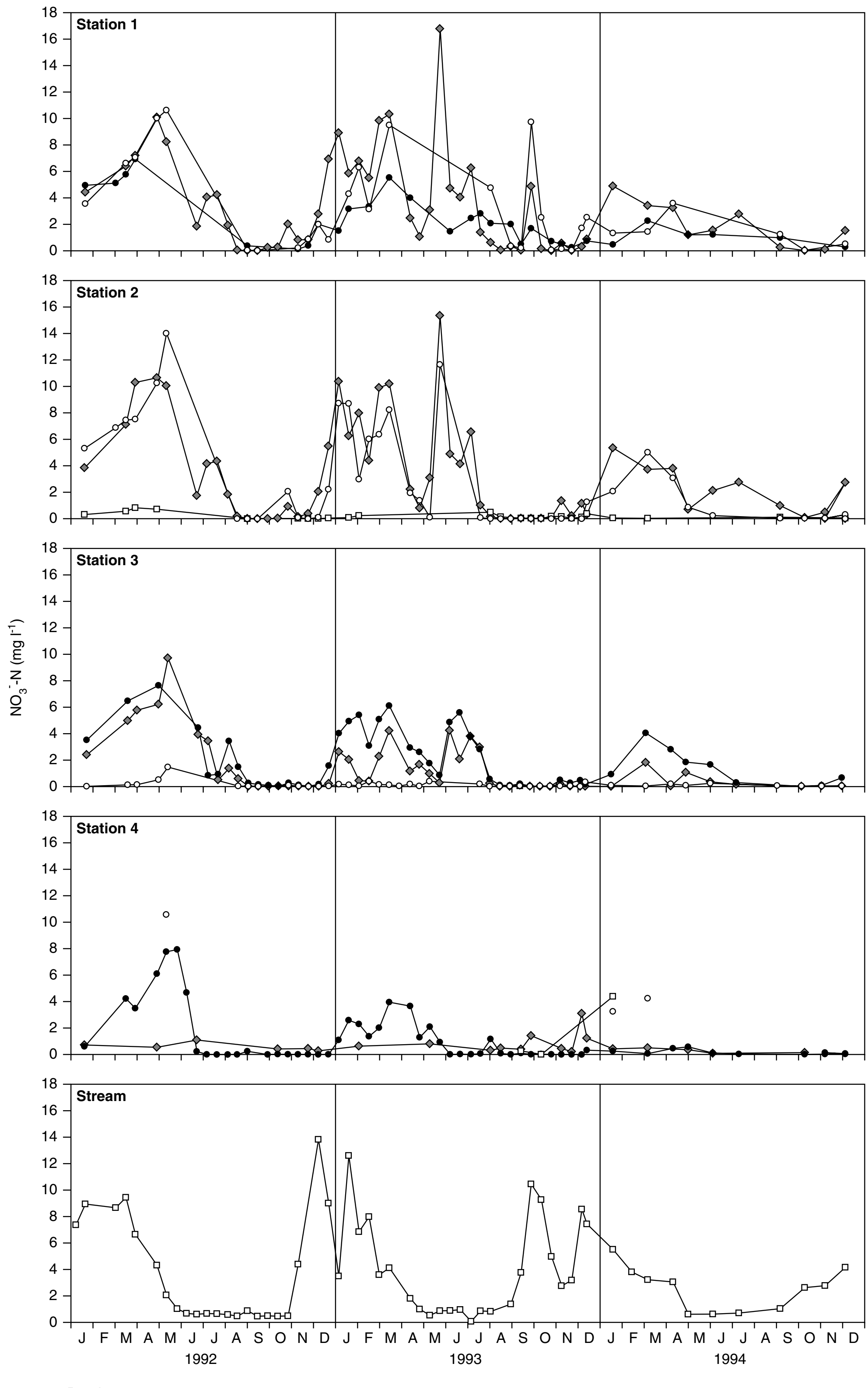

Figure 7 Nitrate-N concentration in different depths at piezometer nest 1—4 and in Voldby Brook.

tion in the meadow. Further, it should be noticed that the summer situation in 1992 and 1994 where the meadow in a short period is recharged by both groundwater from the upland and by stream water from Voldby Brook, is handled separately as the usual flow path at this time is temporary interrupted.

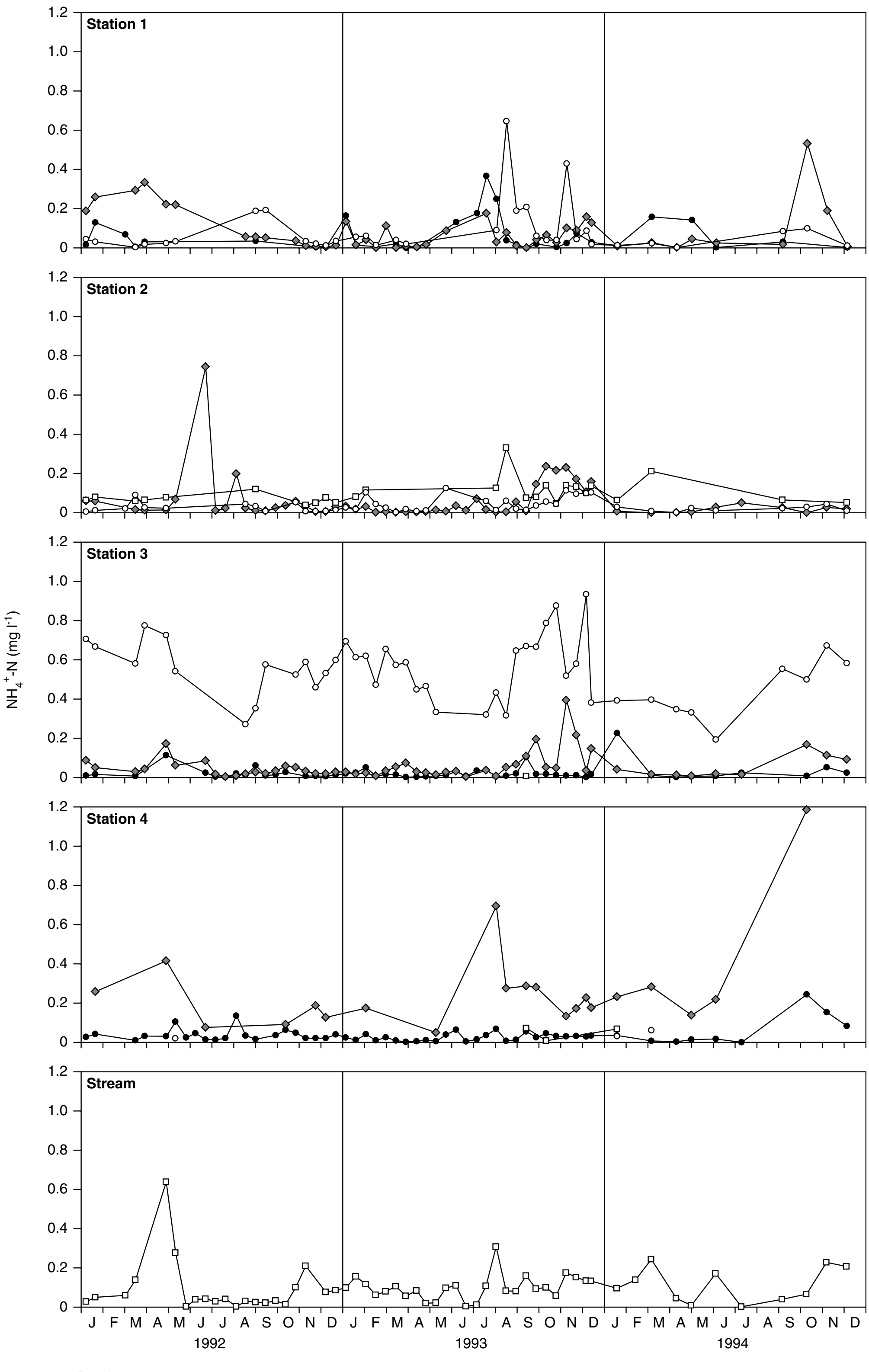

Figure 8 Ammonium-N concentration in different depths at piezometer nest 1—4 and in Voldby Brook.

Because the hydraulic conductivities were measured with the highest mean relative uncertainty (15%) of all variables, the K-values were used to calibrate the models in order to consistently get the same groundwater flow moving from the hillslope to the stream for all three model sectional areas.

The model was tested in a four layer version and a three layer version. The results obtained with the two versions

Table 2 Dry weight (DW) of alive and dead above ground biomass, nitrogen content (N) in alive and dead above ground biomass and phosphorus content (P) in alive and dead above ground biomass

Year	Biomass DW – N – P	Month 1	2	3	4	5	6	7	8	9	10	11	12
1992	DW Alive	–	–	–	84	172	425	574	341	730	771	625	550
1992	DW Dead	–	–	–	143	14	0	0	17	21	33	36	58
1993	DW Alive	324	–	247	–	–	282	510	360	318	362	–	–
1993	DW Dead	281	–	680	–	–	98	278	142	348	396	–	–
1992	N Alive	–	–	–	2.28	4.31	5.84	8.35	7.07	12.42	12.98	11.98	10.66
1992	N Dead				3.35	0.35	0	0	0.21	0.24	0.45	0.65	1.08
1993	N Alive	7.12	–	4.97	–	–	6.26	8.34	9.07	6.98	7.83	–	–
1993	N Dead	5.93		13.56			1.73	4.83	2.59	6.43	7.24	–	–
1992	P Alive	–	–	–	0.15	0.33	0.58	1.46	0.94	0.83	0.91	0.70	1.20
1992	P Dead				0.23	0.02	0	0	0.04	0.02	0.03	0.04	0.13
1993	P Alive	0.79	–	0.64	–	–	0.83	1.33	1.38	1.07	1.19	–	–
1993	P Dead	0.73	–	1.98	–	–	0.21	0.72	0.38	0.96	0.97	–	–

Units: g DW m^{-2}, g N m^{-2} and g P m^{-2}.

Table 3 Input data to three-layer model set-up

H	*Flow1* St. 1 m	K_{1-2} m d^{-1}	St. 2 m	*Flow2* St. 2 m	K_{2-3} m d^{-1}	St. 3 m	*Flow3* St. 3 m	K_{3-4} m d^{-1}	St. 4 m
1	0.89	25.5	0.35	0.35	37.2	0.74	0.74	21.1	0.84
2	0.50	31.1	0.59	0.59	33.3	1.15	1.15	25.1	0.56
3	0.31	16.1	0.31	0.31	18.4	0.31	0.31	28.5	0.50
$\sum$	1.70	–	1.25	1.25	–	2.20	2.20	–	1.90

Layer number, *H*, vertical extent of a layer at each station, *St.*, and sum of all layers, $\sum$, all in meter. Hydraulic conductivity, *K*, in meter per day, for each layer used for calculating groundwater flow between stations in the meadow.

showed that the calculated groundwater flows were almost identical. However, the four layer model failed when nutrient transport was included, because the number of water samples at stations 3 and 4, at a depth of 25 cm, were too scarce to support this higher depth-resolution. Thus, the results presented below are based on the three layer version of the model.

Results from three-layer model set-up

In the three-layer model set-up the vertical extent of the different layers at stations 1–4 and the applied hydraulic conductivities are given in Table 3. The flows, referred to as *flow1*, *flow2*, and *flow3*, calculated in this calibration are shown in Fig. 9 for the three years. The groundwater flow was calculated for 138 dates. Although some discrepancy is seen in day to day comparisons of the three sectional areas, the overall pattern through the seasons of the years agrees well. More specific, most calculated values lie within the 95% confidence limits also shown in Fig. 9. The individual discrepancies were found in February 92, June 92, January 93, December 93, and October 94. In June 92 stream water recharged the meadow, which means that *flow2* and *flow3* are negative due to opposite direction of water compared to *flow1* recharging the meadow from the hillslope.

The mean groundwater flow through the riparian meadow can be calculated based on the following two assumptions: (i) the flow of groundwater from hillslope to stream is strictly horizontal, (ii) *flow1*, *flow2* and *flow3* are equal but with random variation around the true value due to geological heterogeneity (Fig. 9). For that reason, the mean value of *flow1*, *flow2* and *flow3* was used to calculate the mean monthly groundwater flow through the meadow. An exception from this are the two summer periods in 1992 and 1994, where stream water also recharges the meadow. In these periods only *flow1*, or *flow1* and *flow2* were used to calculate the groundwater flow recharging the meadow from the upland, while flow3 was used to estimate the flow of stream water recharging the meadow. Thus, in those two periods, 27 days in 1992 and 20 days in 1994, the model only calculates the input of water to the meadow, but what further happens to the recharged water was not addressed.

Apart from summer, the groundwater flow through the meadow varies between 17 and 33 m^3 m meadow^{-1} month^{-1}. Groundwater flow in summer varies between 7 and 13 m^3 per m meadow^{-1} month^{-1}. In June 1992 the meadow was recharged by both groundwater and stream water. Stream water recharge lasted 18 days and amounted to 1.7 m^3 m meadow^{-1} month^{-1}. Stream water recharge continued for 9 days in July 1992 and amounted to 0.5 m^3 m meadow^{-1} month^{-1}. In June and July 1993 there

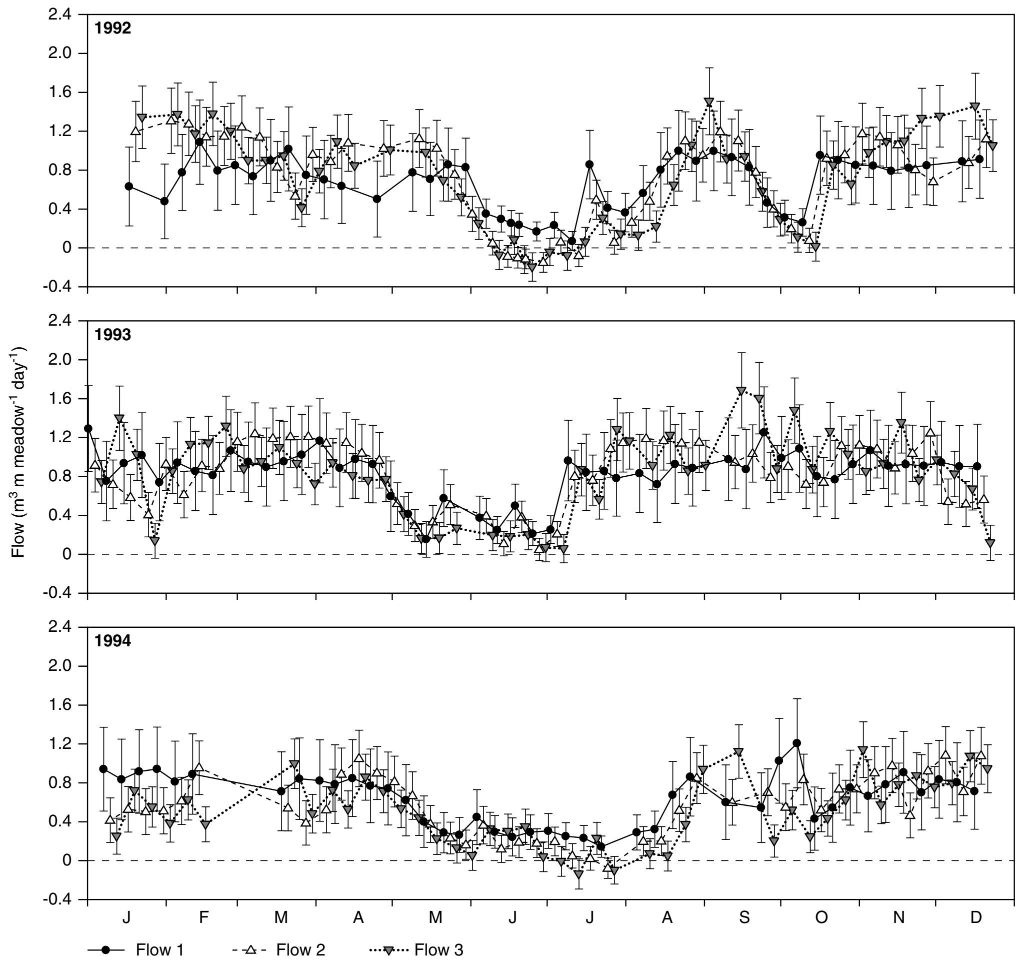

Figure 9 Calculated groundwater flow with three-layer model. *Flow1* is from stations 1 to 2, *Flow2* is from stations 2 to 3, and *Flow3* is from stations 3 to 4. Calculated 95% confidence limits are included for each of the three model sectional areas.

was a net discharge of groundwater to the stream, but on single measuring dates the discharge was close to zero indicating that recharge by stream water may have taken place in this period. In July 1994 there was recharge by stream water to the meadow for 20 days, which amounted to 1.2 m^3 m meadow^{-1} month^{-1}. At the end of June and beginning of August discharge of groundwater to the stream was close to zero.

The mean annual groundwater recharge to the meadow during the years 1992—1993—1994 was 288, 304 and 215 m^3 m meadow^{-1} y^{-1}, respectively.

There are a few synchronous discharge measurements from this reach of Voldby Brook which can be compared to the model predictions (Table 4). On March 10, 1992 the flow was measured to 1.35 m^3 m^{-1} day^{-1} assuming equal discharge of groundwater from both sides of the stream. On

Table 4 Synchronous discharge measurements, in Voldby Brook, mean of three replicas with uncertainty in parenthesis, performed along the meadow site in 1992

Date	Length (m)	Q_{us} l s^{-1} (%)	Q_{ds} l s^{-1} (%)	δQ l s^{-1}	δQ (%)	Q (m^3 day^{-1})	Model date	Q Model (m^3 day^{-1})
10-03-92	240	79.2 (2.3)	86.7 (3.5)	7.5	9.5	1.351	09-03-92	0.829
17-06-92	240	12.9 (0.4)	14.2 (1.8)	1.3	10.2	0.233	15-06-92	0.075
01-07-92	160	10.0 (1.2)	9.7 (0.5)	−0.3	−2.6	−0.082	29-06-92	−0.041
10-09-92	160	11.2 (1.8)	11.4 (3.6)	0.2	1.9	0.054	07-09-92	0.841

The table shows the dates when the discharge measurements were performed, the length of the stretch of the brook, Q_{us} = stream discharge upstream, Q_{ds} = stream discharge downstream, δQ = increase or decrease in discharge shown as litre second and relatively (%), Q = discharge to stream or recharge of meadow (—) m^3 m stream bank^{-1} day^{-1}. The two right-hand columns show dates for the model calculated discharge of water from the meadow to the stream or recharge of the meadow (—). Model calculations are corrected with sine to 67° for comparison with stream discharge measurements.

March 9, 1992 the model predicted $0.901 \pm 0.244\ m^3$ m meadow^{-1} day^{-1} but as the transect is not at right angles to the stream but has an angle of 67° it has to be corrected with sine to 67 degrees for comparison, which gives $0.829 \pm 0.225\ m^3\ m^{-1}\ day^{-1}$. On June 17, 1992 stream discharge was $0.233\ m^3\ m^{-1}\ day^{-1}$ (assuming equal discharge of groundwater from both sides of the stream) while the model calculation on June 15, 1992 gave a groundwater discharge of $0.082 \pm 0.152\ m^3$ m meadow^{-1} day^{-1}, which corrected with sine to 67° gives $0.075 \pm 0.140\ m^3\ m^{-1}\ day^{-1}$. On July 1, 1992 synchronous discharge measurements showed a loss of $0.08\ m^3\ m^{-1}\ day^{-1}$ (Table 4), while the model calculation estimated stream recharge of the meadow on June 29, 1992 to be $0.044 \pm 0.142\ m^3$ m meadow^{-1} day^{-1} or $0.041 \pm 0.131\ m^3\ m^{-1}\ day^{-1}$ (sine corrected). On July 6, 1992 the meadow was still recharged by stream water amounting to $0.073 \pm 0.143\ m^3\ m^{-1}\ day^{-1}$ (sine corrected). These data correspond satisfactorily, considering that the relative increase or decrease (δQ) in stream discharge measurements is small.

Residence time

The total volume of the meadow considered in the model is calculated as follows: At stations 1–4 the vertical extent is 1.70 m, 1.25 m, 2.20 m and 1.90 m (Table 3). The corresponding lengths along the transect are: 3 m, 9 m and 9 m (Fig. 3). Thus the total volume of the meadow transect is $38.4\ m^3$. With a mean groundwater level of 0.5 m below ground and a porosity of 38.3% the total volume of water considered in the model is $10.7\ m^3$. Based on this estimate, the mean groundwater residence time was 14, 13 and 18 days, for the years 1992, 1993 and 1994, respectively. Based on model predicted flows, the shortest groundwater residence time was approximately 10 days, and the longest 42 days. The two periods in summer 1992 (27 days) and 1994 (20 days), when the meadow was both recharged by groundwater from the upland and by stream water are excluded from the above rough estimate of residence time as the fate of groundwater flow and residence time in these two periods remain unknown.

Mass balances of nitrate, ammonium and phosphate

The nitrate load, i.e. input of nitrate with groundwater recharging the meadow, was calculated as the amount of nitrate entering the meadow at station 1, which was located at the boundary to the upland field (Fig. 3). Technically the calculation was done by setting the nitrate concentration at station 2 to zero.

The nitrate load fluctuated during the year and exhibited the same overall seasonal pattern for all three years (Table 5). In summer and autumn the nitrate load was low. In December or January the nitrate load increased and remained high through the winter until beginning of summer. In January 1994 the increase in nitrate load was smaller than in January 1992 and 1993, and further, in the summer of 1994 the nitrate load was lower than the two preceding years (Table 5). This could most likely be explained by the fact that the upland field recharging the meadow was set aside from autumn 1993.

The model estimated nitrate-N removal (Table 6) reflects the variation in both concentration pattern during the year (Fig. 7) and the annual variation in groundwater flow passing through the meadow (Fig. 9). Despite the short distance of 3 m, nitrate removal was detectable already between station 1 and 2, especially in 1993 where 298 g N m meadow^{-1} y^{-1} was removed (Table 6; *flow1*). This is probably due to a significant shift in redox conditions from oxidizing to reducing conditions at this location. For all three years the highest removal of nitrate took place between station 2 and 3 (*flow2*), where 61–74% of the total nitrate removal took place (Table 6). In the winter 1993/1994 the nitrate removal did not increase as in the two preceding winters and the nitrate removal continued to be smaller in 1994, only amounting to 35% of the annual removal in 1992–1993. This is probably due to the fact that the agricultural field uphill to the meadow was set aside in 1994 and it was already grass-covered in late summer 1993. This can also be seen from the annual load of nitrate in 1994, amounting to 364 g NO_3^--N m meadow^{-1} y^{-1} (Table 5). This figure is only one third of the loads in 1992–1993, which were 1156 and 1104 g N m meadow^{-1} y^{-1}, respectively (Table 5). The annual removal of nitrate in 1992 and 1993 was 685 and 715 g NO_3^--N m meadow^{-1} y^{-1}, respectively (Table 6), followed by a drop to 249 g NO_3^--N m meadow^{-1} y^{-1} in 1994. The percentage removal of nitrate seems not to be related to the nitrate load, but seems to vary randomly from 59% to 68% during the years.

A closer inspection of the results for each layer revealed that the percentage nitrate removal decreased gradually with depth. *Flow2* between stations 2 and 3, covering the area with the highest nitrate removal in the meadow, showed for 1992 that the nitrate removal was 83 g NO_3^--N y^{-1} (59% of the load) in layer 1, 289 g NO_3^--N y^{-1} (40% of the load) in layer 2 and 48 g NO_3^--N y^{-1} (26% of the load) in layer 3, which adds up to 420 g NO_3^--N y^{-1} (Table 6). In 1993 the removal was 119 g NO_3^--N y^{-1} (94% of the load) in layer 1, 275 g NO_3^--N y^{-1} (55% of the load) in layer 2 and 65 g NO_3^--N y^{-1} (38% of the load) in layer 3, which adds up to 459 g NO_3^--N y^{-1} (Table 6). In 1994 which had the smallest

Table 5 Monthly and annual nitrate load in groundwater recharging the meadow

Year	Month												Total Year
	1	2	3	4	5	6	7	8	9	10	11	12	
1992	143	147	165	222	192	41	45	12	2	20	28	140	1156
1993	166	168	274	86	111	54	61	19	76	50	8	30	1104
1994	57	61	67	64	16	14	18	15	11	2	10	29	364

Units: g NO_3^--N m meadow^{-1} month^{-1} (month columns) and g NO_3^--N m meadow^{-1} year^{-1} (total year column).

Table 6 Mass balance for nitrate-N showing monthly and yearly rates of nitrate-N removal or gain in the saturated zone

Nitrate-N removal	Month 1	2	3	4	5	6	7	8	9	10	11	12	Total Year
1992													
Flow1	12	−27	9	53	−73	−10	−1	1	2	3	19	60	48
Flow2	58	58	47	62	103	4	13	−7	−1	18	9	56	420
Flow3	47	63	24	58	23	−4	4	8	−3	−1	−5	4	218
Total	117	95	79	173	52	−10	16	3	−1	19	23	120	685
1993													
Flow1	6	29	45	14	31	−3	16	18	75	49	5	15	298
Flow2	113	109	137	23	62	17	−14	−2	−1	1	1	12	459
Flow3	5	−4	0	−29	1	14	45	−5	−16	−7	−9	−35	−41
Total	124	134	181	8	94	28	47	10	58	42	−4	−8	715
1994													
Flow1	25	8	−5	4	5	3	0	3	6	1	5	15	69
Flow2	29	35	43	41	−2	5	3	9	3	1	5	13	185
Flow3	−22	−2	0	7	7	2	−0	1	1	−1	1	3	−4
Total	31	42	39	52	10	9	3	13	10	0	10	30	249

The results are made up from all three model sectional areas. *Flow1* is from station 1 to station 2, Flow2 is from stations 2 to 3 and Flow3 is from stations 3 to 4. Units: g NO_3^--N m meadow^{-1} month^{-1} (month columns) and g NO_3^--N m meadow^{-1} year^{-1} (total year column).

groundwater recharge and the lowest nitrate load, nitrate removal in layer 1, 2 and 3 was 21 g NO_3^--N y^{-1} (86% of the load), 125 g NO_3^--N y^{-1} (73% of the load) and 39 g NO_3^--N y^{-1} (45% of the load), respectively, and this adds up to 185 g NO_3^--N y^{-1}.

The atmospheric bulk deposition (wet and dry) of nitrate was of minor importance (Table 7). Although 33–69% of the total N-deposition was nitrate, it only constituted 1% of the total nitrate load in 1992–1993 (1166 and 1117 g N m meadow^{-1} y^{-1}, respectively) and roughly 5% of the total load in 1994 which amounted to 385 g N m meadow^{-1} y^{-1}.

Input of nitrate to the meadow during stream water recharge of the meadow amounted to 7.4 g NO_3^--N in 1992 (June–July, 27 days) and 0.4 g NO_3^--N in 1994 (20 days in July). Groundwater recharge in the same two periods was 20.1 and 12.2 g NO_3^--N, respectively. Thus, the total amount of nitrate which cannot be accounted for in 1992 is 27.5 g NO_3^--N and it constitutes 2.3% of the total load in

Table 7 Mass balances for nitrate-N, ammonium-N and phosphate-P in g m meadow^{-1} year^{-1}

	1992	1993	1994
Nitrate load saturated zone	1156	1104	364
Atmospheric nitrate deposition	10	13	21
Total nitrate load	1166	1117	385
Groundwater discharge of nitrate	471	389	115
Nitrate removal in saturated zone	685	715	249
Net removal of nitrate	695	728	270
Ammonium load saturated zone	27.6	23.5	16.4
Atmospheric ammonium deposition	15.8	26.9	10.7
Total ammonium load	43.4	50.4	27.1
Groundwater discharge of ammonium	28.5	37.5	38.1
Ammonium retention in saturated zone	−0.9	−14.0	−21.7
Net retention of ammonium	14.9	12.9	−11.0
Phosphate load saturated zone	2.65	2.27	2.05
Atmospheric phosphate deposition	0.92	1.03	0.46
Total phosphate load	3.57	3.30	2.51
Groundwater discharge of phosphate	5.42	4.06	1.52
Phosphate retention in saturated zone	−2.77	−1.79	0.53
Net retention of phosphate	−1.85	−0.76	0.99

For all three elements are shown load and removal/retention in the saturated zone and discharge from the saturated zone, atmospheric deposition, total load and total balance (i.e. net retention). The meadow equals 21 m^2 along the transect.

groundwater recharging the meadow in 1992. The 20 days in July 1994 make up 12.6 g NO_3^--N (3.5% of total load in groundwater), which cannot be accounted for, but also the flooding period in February—March 1994 adds to the uncertainty of the nitrate balance this year.

Ammonium was leached from the meadow on an annual basis (Table 7). Although the load of ammonium input with recharging groundwater in the years 1992 to 1994 decreased from 27.6 to 16.4 g NH_4^+-N m meadow^{-1} y^{-1}, leaching of ammonium-N to the stream increased from 0.9 to 21.7 g NH_4^+-N m meadow^{-1} y^{-1} (Table 7). In 1992 ammonium was retained in seven out of 12 months and leaching was close to zero (Table 8). In 1993 ammonium was only retained in January, May and June, while leaching in August and September was high amounting to 5.1 and 5.5 g NH_4^+-N m^{-1} meadow month^{-1}, respectively. In 1994 there was no retention of ammonium at all, and September — October showed the highest leaching 3.1 and 6.3 g NH_4^+-N m meadow^{-1} month^{-1}, respectively (Table 8).

The atmospheric deposition of ammonium varied from 10.7 to 26.9 g NH_4^+-N m meadow^{-1} y^{-1} (Table 7), and made up 36%, 53% and 39% of the total ammonium load for the years 1992, 1993 and 1994, respectively. Including the atmospheric deposition in the ammonium mass balance for the meadow results in a net retention of ammonium amounting to 14.9 g NH_4^+-N m meadow^{-1} y^{-1} in 1992, and a net retention of 12.9 g NH_4^+-N m meadow^{-1} y^{-1} in 1993, while 1994 had a loss of 21.7 *g* NH_4^+-N m meadow^{-1} y^{-1} via groundwater discharge to the stream and an atmospheric deposition (wet and dry) of 10.7 g NH_4^+-N m meadow^{-1} y^{-1} resulting in an overall net loss of 11.0 g NH_4^+-N m meadow^{-1} y^{-1} (Table 7).

Input of ammonium to the meadow during stream water recharge of the meadow amounted to 0.07 g NH_4^+-N in 1992 (27days) and 0.03 g NH_4^+-N in 1994 (20 days). Groundwater recharge in the same two periods was 0.28 and 0.09 g NH_4^+-N, respectively. The amount of ammonium which cannot be accounted during these two summer periods only constitutes 1% of the total ammonium load in groundwater (Table 7).

Phosphate exhibited the opposite pattern relative to ammonium. In 1992 phosphate was leached during the whole year, except for September which showed a small retention (Table 9). Phosphate was leached both between station 1 and 2 (*flow1*) and between station 3 and 4 (*flow3*). In 1993 this pattern continued but less pronounced, and there was a net retention of phosphate in January, August, September and December (Table 9). This resulted in a decrease in annual loss of phosphate (Table 7). The decrease in phosphate leaching continued in 1994, and although there still was a net loss of phosphate during several months, the annual balance showed net retention of phosphate (Tables 7 and 9). It was especially the decrease in phosphate leaching between stations 3 and 4 (*flow3*), which was significant. In 1992 and 1993 leaching of phosphate was 1.75 and 2.34 g PO_4^{3-}-P m meadow^{-1} y^{-1}, respectively. In 1994 only 0.66 g PO_4^{3-}-P m meadow^{-1} y^{-1} was leached. Comparing the mass balances for all three years revealed, that there was always retention of phosphate between stations 2 and 3 (*flow2*) and always net loss of phosphate between stations 3 and 4 (*flow3*; Table 9).

Atmospheric deposition of phosphate constituted 18—31% of the total P-load to the meadow (Table 7). In 1992 leaching of phosphate via groundwater discharge was 2.77 g P m meadow^{-1} y^{-1}, but atmospheric deposition compensated for 0.92 g P m meadow^{-1} y^{-1}, resulting in a net loss of 1.85 g P m meadow^{-1} y^{-1} (Table 7). The same calculation for 1993 resulted in a net loss of 0.76 g P m meadow^{-1} y^{-1}. In 1994 there was retention of phosphate in the saturated zone, 0.53 g P m meadow^{-1} y^{-1} and

Table 8 Mass balance for ammonium-N showing monthly and yearly rates of ammonium-N retention or loss in the saturated zone

NH_4^+-N retention	Month 1	2	3	4	5	6	7	8	9	10	11	12	Total Year
1992													
Flow1	3.0	2.4	2.6	3.4	2.1	0.3	0.0	0.4	1.7	0.1	−0.2	0.3	16.0
Flow2	−6.6	−2.0	−2.7	−6.7	−3.0	0.0	0.2	0.2	−1.4	−0.6	−3.9	−4.3	−30.8
Flow3	3.5	−0.9	3.0	1.7	0.9	−0.0	−0.1	0.6	1.4	0.4	1.2	2.0	13.9
Total	0.0	−0.5	2.9	−1.7	0.0	0.3	0.1	1.2	1.8	−0.0	−2.9	−2.1	−0.9
1993													
Flow1	0.7	−0.2	0.9	−0.0	0.2	1.1	2.5	1.7	0.5	−1.0	0.3	−0.3	6.4
Flow2	−3.7	−3.6	−5.1	−2.6	0.4	0.1	−0.3	−1.5	−6.8	−6.3	−5.4	−2.6	−37.4
Flow3	3.8	2.6	4.1	2.5	−0.0	−0.1	−2.8	−5.3	0.6	5.4	4.9	1.4	17.1
Total	0.8	−1.2	−0.1	−0.1	0.6	1.2	−0.6	−5.1	−5.7	−1.9	−0.3	−1.6	−14.0
1994													
Flow1	−0.5	−0.5	−0.1	0.5	0.4	−0.1	−0.2	−0.1	1.9	6.3	2.1	−0.3	9.3
Flow2	−2.6	−2.0	−2.1	−1.8	−0.0	0.1	0.0	−0.4	−2.8	−3.2	−4.1	−5.2	−24.1
Flow3	0.6	0.3	0.5	0.9	−1.2	−1.1	−0.0	0.1	−2.1	−9.4	1.4	3.2	−6.8
Total	−2.6	−2.2	−1.7	−0.4	−0.8	−1.1	−0.2	−0.5	−3.1	−6.3	−0.6	−2.2	−21.7

The results are made up from all three model sectional areas. *Flow1 is* from station 1 to station 2, *Flow2* is from station 2 to station 3 and *Flow3* is from station 3 to station 4. Units: g NH_4^+-N m meadow^{-1} month^{-1} (month columns) and g NH_4^+-N m meadow^{-1} year^{-1} (total year column).

Table 9 Mass balance for phosphate-P showing monthly and yearly rates of phosphate-P retention or loss in the saturated zone

PO_4^{3-}-P retention	Month												Total Year
	1	2	3	4	5	6	7	8	9	10	11	12	
1992													
Flow1	−0.09	−0.05	−0.02	−0.14	−0.61	−0.11	0.00	−0.14	−0.05	−0.10	−0.10	−0.06	−1.47
Flow2	0.21	0.10	−0.25	−0.49	0.13	0.01	0.00	0.12	0.14	0.12	0.19	0.19	0.45
Flow3	−0.18	−0.11	−0.17	0.10	−0.14	0.02	−0.05	−0.04	−0.09	−0.35	−0.50	−0.23	−1.75
Total	−0.06	−0.06	−0.45	−0.53	−0.62	−0.09	−0.05	−0.06	0.01	−0.33	−0.42	−0.11	−2.77
1993													
Flow1	0.49	−0.08	−0.10	−0.01	−0.00	0.00	−0.43	0.13	−0.00	−0.03	0.03	−0.08	−0.09
Flow2	0.11	0.14	0.18	0.11	0.03	−0.04	0.28	0.32	0.10	−0.15	−0.30	−0.16	0.63
Flow3	−0.48	−0.28	−0.73	−0.38	−0.09	−0.10	−0.61	−0.15	0.10	−0.21	0.22	0.37	−2.34
Total	0.12	−0.22	−0.65	−0.28	−0.06	−0.13	−0.76	0.30	0.20	−0.39	−0.05	0.13	−1.79
1994													
Flow1	−0.32	−0.19	−0.02	−0.02	−0.03	0.00	0.00	0.01	−0.03	0.19	0.96	−0.01	0.54
Flow2	0.23	0.15	−0.07	−0.10	0.06	−0.00	−0.00	0.01	0.04	0.07	0.11	0.14	0.65
Flow3	−0.00	−0.07	−0.08	−0.19	−0.02	0.01	−0.01	−0.08	−0.11	−0.09	−0.14	0.11	−0.66
Total	−0.09	−0.11	−0.17	−0.30	0.01	0.01	−0.01	−0.06	−0.10	0.17	0.93	0.24	0.53

The results are made up from all three model sectional areas. *Flow1* is from station 1 to station 2, *Flow2* is from station 2 to station 3 and *Flow3* is from station 3 to station 4. Units: g PO_4^{3-}-P m meadow^{-1} month^{-1} (month columns) and g PO_4^{3-}-P m meadow^{-1} year^{-1} (total year column).

together with atmospheric deposition of 0.46 g P m meadow^{-1} y^{-1} net total retention added up to 0.99 g P m meadow^{-1} y^{-1} (Table 7).

Input of phosphate to the meadow during stream water recharge of the meadow amounted to 0.018 g PO_4^{3-}-P in 1992 (27days) and 0.012 g PO_4^{3-}-P in 1994 (20 days). Groundwater recharge in the same two periods was 0.043 and 0.033 g PO_4^{3-}-P, respectively. Compared to the total load of phosphate in groundwater recharging the meadow (Table 7) the amount of phosphate which cannot be accounted for in these two summer periods only constitutes 2%. The lack of information about the flooding period in February—March 1994 especially data on sedimentation makes the total phosphate balance for 1994 uncertain.

Discussion

The saturated hydraulic conductivities were estimated to have a relative error of 15% (coefficient of variation). This degree of accuracy is small compared to other studies, which have shown variations from tens of per cent to more than 150% (Päivänen, 1973; Andersen, 2004). According to the soil profile description the main body of the meadow consists of medium grained sand, and this may be the reason for the relative small error in measured hydraulic conductivity. Furthermore, in this study the measured hydraulic conductivities only vary between 12.1 m day^{-1} (i.e. 1.4×10^{-4} m s^{-1}) and 37.5 m day^{-1} (i.e. 4.3×10^{-3} m s^{-1}). Other studies have shown hydraulic conductivities that varied 3—4—5 orders of magnitude (Päivänen, 1973; Maître et al., 2003; Andersen, 2004; Angier et al., 2005) indicating much more heterogeneous soil profiles as compared to this study.

In some riparian areas precipitation may contribute significantly to the total water input and may dilute chemical compounds such as nitrate (Maître et al., 2003). Precipitation was not included in our estimates, but this error committed was small: In the years 1992—1993—1994 total precipitation was 14.9, 17.9 and 22.6 m^3 in the meadow transect (21 m^2). Compared to the groundwater flow this constitutes 5.2%, 5.9% and 10.5% for the three years. If evapotranspiration is subtracted, the relative contribution from net precipitation for the three years is in the range of 3—5%, as net precipitation from 1990 to 2000 in this area amounts to approximately 400 mm y^{-1} and potential evapotranspiration to 560 mm y^{-1} (Scharling and Kern-Hansen, 2002).

In 1994 the groundwater discharge to the brook was low, only 215 m^3 per meter meadow but precipitation of 1076 mm was the highest in the three-year period. Precipitation in January, February and March 1994 was very high, 350 mm, compared to 168 and 170 mm in 1992 and 1993, respectively. Due to the very high precipitation and snowfall in the whole catchment from December 1993 to March 1994 the meadow was completely flooded in the period February to March 1994 (approximately one month). For that reason no measurements were taken in this period which prohibited trustworthy model predictions. It is not possible to determine if this significant flood event was caused by high groundwater discharge or impounded stream water caused by surface runoff as a result of heavy rainfall and snowmelt.

During summer months the meadow was also recharged by stream water (Fig. 7). This observation is supported by synchronous discharge measurements along the same reach of Voldby Brook in the River Gjern catchments area showing that along this reach Voldby Brook recharges the riparian areas in low flow periods (Clausen, 1995; Kronvang et al., 1997; personal communication Clausen, Kronvang and Svendsen).

The nitrate removal from the groundwater moving through a riparian meadow as we report here, agree generally with other studies (Peterjohn and Correl, 1984; Cooper, 1990; Haycock and Pinay, 1993; Cey et al., 1999; Clément et al., 2003; Maître et al., 2003;). However, unlike some of these studies, nitrate was not completely removed at our site. A possible explanation is that the width of the meadow is only 21 m and this is too short of a distance to give a complete nitrate removal, or formulated in another way, the groundwater residence time is too short to facilitate a complete removal. Mean groundwater residence time varied between 42 days during summer and 10 days during winter. Despite the longer summer residence time nitrate removal is smaller because both nitrate concentration and nitrate load is lower during the summer. At wintertime when plant uptake of nitrate is insignificant, nitrate is most likely denitrified. Although the annual nitrate load decreased from 1156 to 1104 g NO_3^--N m meadow^{-1} y^{-1} (i.e. 550–525 kg NO_3^--N ha^{-1} y^{-1}) in 1992–1993 to 364 g NO_3^--N m meadow^{-1} y^{-1} (i.e. 173 kg NO_3^--N ha^{-1} y^{-1}) in 1994 the removal efficiency only increased by a few percent from 59 and 65% in 1992–1993 to 68% in 1994. A possible explanation is that the denitrification process was limited by carbon availability. The meadow aquifer sediments are sandy and the content of organic matter (Fig. 5) is low in the saturated part. In the meadow at a depth of 50–100 cm the mean content of organic matter was 3.2% and deeper at a depth of 100–150 cm the mean content decreased to 1.6%. This is in line with the finding of Brettar and Höfle (2002) who found a close correlation between organic matter content and denitrification rates in a forest soil. Likewise, Hoffmann et al. (2000) showed for a riparian fen, that denitrification was non-detectable when the content of organic matter was below 3%.

As pointed out by Correll (1996), there is considerable uncertainty on the exact role of riparian vegetation in nitrate removal. Some studies have shown that plant uptake of nitrogen and phosphorus can make a significant contribution to the retention of these nutrients in riparian wetlands (Peterjohn and Correl, 1984; Lowrance et al., 1984; Sánchez-Pérez et al., 1991). Others claim that topography (Cirmo and McDonnell, 1997; Clément et al., 2002; Vidon and Hill, 2004b) and carbon availability (Cooper, 1990; Schipper et al., 1993; Brettar and Höfle, 2002) are more important factors for retention of nutrients in wetlands. In this study the annual variation in nitrate concentration in groundwater recharging the meadow as well as the annual variation in groundwater flow indicate that plant uptake of nitrate is of limited importance for nitrate removal in the meadow. Further, the position of the groundwater table, which is close to the ground surface in winter and may decrease to 1 m below ground in summer, suggests that denitrification is more important than plant uptake.

The atmospheric deposition of nitrogen and phosphorus is of course available for plant uptake, but P-deposition only constitutes 3–4% of total-P in above-ground standing stock and total-N deposition constitutes 9–21% of N in standing stock. Examining the position of the groundwater table reveals that in the beginning of the growing season — April and May — and occasionally also later in the growing season, the groundwater table is situated in the root zone (Fig. 7). The nitrate load in groundwater recharging the meadow in April and May is also high enough to support plant uptake of N (Table 2) even though only approximately 22% of the groundwater flow occurs in the upper part of the geological profile (layer 1) carrying 17–24% of the nitrate load. For a 60 day period (e.g. April and May) nitrogen uptake amounts to 69 and 37 g (21 m^2 meadow) in 1992 and 1993, respectively. Approximately 20% of the nitrate load recharging the meadow, 80 and 40 g NO_3^--N in April–May 1992–1993, respectively, is transported in layer 1, and these amounts can fully sustain N-uptake in plants. On the contrary phosphate uptake in above-ground biomass exceeds the phosphate load, and at the same time there is a net loss of phosphate via groundwater discharge to the stream in 1992–1993. The meadow was used for cattle grazing in summer until start of the study, and if the cattle received additional feed, for example commonly used concentrates, this may have influenced P leaching from the meadow, because P-content in cattle manure is high i.e. 2.0–2.6 kg P by the animal grazing 165 days a year (Poulsen et al., 2001).

The consistently high concentration of ammonium at station 3 at a depth of 65 cm may be a result of denitrification, as the bacteria needs easy degradable carbon as an energy source, and mineralization of organic matter liberates ammonium. Comparing nitrate loads (Table 5) during the year with ammonium concentration (Fig. 8) reveals that ammonium concentration is higher during months with high nitrate load (e.g. January 92–May 92 and January 93–March 93) and lower during summer when nitrate load is small. This coincides with the result of the nitrate calculations, which showed that *Flow2* (stations 2–3) always exhibited the highest nitrate removal (Table 6). Though the increase in ammonium concentration in the autumn months is not accompanied by the same increase in nitrate load and therefore other processes may be involved, such as mineralization of dead plant material when the growing season ends.

The high nitrate load in 1992 and 1993 may have contributed to the loss of phosphate from the saturated zone. Verhoeven and Arts (1987) have found phosphorus mineralisation rates up to 2.3 kg P ha^{-1} y^{-1} in a fen with high nitrate load and groundwater through flow. In this meadow study phosphate is retained in 1994 and this coincides with a decrease in nitrate load to 364 g NO_3^--N m meadow^{-1} y^{-1} (i.e. 173 kg NO_3^--N ha^{-1} y^{-1}). But also the groundwater table may influence mobilisation of phosphorus. Meissner et al. (2004) found that fluctuations in groundwater table to be the driving force for phosphorus mobilisation at four different study sites. Although the annual fluctuations in groundwater table are significant in this study the annual pattern from year to year remains the same, and therefore fluctuation in groundwater level cannot alone explain why phosphate is retained in 1994, but leached from the meadow in 1992–1993. There is a striking difference in nitrate load and nitrate removal between the first six months and the last six month every year in the study period. Nitrate removal in the first six months of 1992–1993–1994 is 506, 569 and 183 g NO_3^--N m meadow^{-1}, respectively, and during the last six months of 1992–1993–1994 it is 180, 145 and 66 g NO_3^--N m meadow^{-1}. Comparing these results with phosphate reveal that phosphate retention (or leaching) is opposite. During the first six months of 1992–1993–1994 phosphate retention in the saturated zone amounts to −1.82, −1.22 and −0.56 g PO_4^{3-}-P m meadow^{-1} (i.e. leaching), while retention in the last six months of 1992—1993—1994 is −0.96, −0,57 and (+) 1.17 g PO_4^{3-}-P m meadow^{-1}. Thus, the pattern is that when nitrate removal is high leaching of phosphate is high and when nitrate removal is lower phosphate leaching decreases with approximately 50% in 1992—1993 and results in retention of phosphate in 1994.

The flooding event in February—March 1994 is an element of uncertainty to the results of the mass balance calculations in 1994. We observed that the water level in some of the piezometers which were not drowned (i.e. still functional) showed the same water level as the floodwater, and therefore neither downwards nor upwards movement of water took place. But exchange of dissolved nutrients may have taken place by diffusion due to differences in concentration of dissolved substances in surface water and soil water. However, it is unclear exactly how this exchange mechanism may have acted both with respect to direction of concentration gradients and amount of substance exchanged. Deposition of sediment and sediment associated phosphorus may have taken place at the flooding event in February—March 1994. Kronvang et al. (2002) estimated for a ten year period the sedimentation rate in the floodplain at the mouth of River Gjern, to 2.1 kg dry weight $m^{-2}\,y^{-1}$ while phosphorus deposition was estimated to 8.2 g P $m^{-2}\,y^{-1}$. But whether these figures reflect the deposition rates in the study site for this isolated event is difficult to evaluate. The fate of the deposited sediments and sediment associated nutrients in the study site at Voldby Brook is unknown but if the sediment associated nutrients were mineralised and liberated in the saturated zone after the flooding event it is included in the mass balance for the saturated zone.

Conclusion

Our study documents the importance of riparian ecotones along first order streams to reduce diffuse pollution from agricultural areas. The hydrogeologic setting with an aquitard already at 2—3 m depth below the meadow surface allow nitrate from shallow nitrate-rich groundwater to be removed as water flows through the riparian aquifer. Although conditions for denitrification were not optimal due to scarcity of organic matter, nitrate removal was significant. Construction of a model to calculate water and mass balances was a valuable tool to study variation in transport both in time and space. Plant uptake of nitrogen was only significant in the beginning of the growing season while phosphorus uptake by plants was more difficult to validate, because the first two years of our study revealed a net loss of phosphorus from the meadow. Still plant production is important as a carbon source for denitrification as the organic matter content in the meadow sediments is scarce. Periods with high nitrate removal was accompanied by high phosphate leaching from the saturated zone, while periods with low nitrate removal was accompanied by significant lower phosphate leaching or even retention of phosphate in the saturated zone. The pronounced year to year variations in our nutrient budgets indicate that shorter studies, for example based on only one year of observations, should be interpreted cautiously as representing a general picture of nutrient cyclings.

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