# Quantification of nitrate removal by a flooded alluvial zone in the Ill floodplain (Eastern France)

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*Key words:* alluvial floodplain, nitrate removal, mass balance, groundwater modelling

## Abstract

The nitrate reducing capacity of a flooded system in the Ill floodplain (Eastern France) was investigated for a period of 2 years. The methodology used consisted of a spatio-temporal monitoring of stream flow and nitrate concentrations in the groundwater and surface water, calculation of input and output fluxes and modelling of groundwater fluxes and nitrate transfer through the alluvial area. A comparison of chloride flux (used as hydrological tracer) and nitrate flux was done to determine a floodplain effect on the retention of nitrate. We show that up to 95% of the nitrate load in the groundwater is retained by the system, whereas the retention in the stream network is very low. Ammonium fluxes increased from inputs to outputs in the stream and in the groundwater. The chloride input in the groundwater is higher than the output, whereas in the surface water the output is higher than the input, the amount evacuated in streams corresponding to the losses from groundwater. The nitrate removal rate calculated for the whole modelized surface area (40 $km^2$) represented 559 t N $yr^{-1}$ or 1397.7 kg N $ha^{-1}$ $yr^{-1}$. The ammonium fluxes exported by the system represented 102 kg N $ha^{-1}$ $yr^{-1}$. A part of nitrate is reduced and exported by the groundwater and stream network in the form of ammonium. These results can be explained by the duration of floods which controls the equilibrium between the various forms of nitrogen. Thus, long watering periods favour nitrogen removal (denitrification and plant uptake) and limit nitrate production which compensates elimination during the dry period.

## Introduction

Nowadays, groundwater quality is highly disturbed by the increased inputs of nitrogen fertilizers. The riparian forests have been demonstrated to be capable of removing nitrate from agricultural drainage and thus playing an important role in the regulation of fluxes between rivers and adjacent agrosystems (Cooper, 1990; Lowrance, 1992; Pinay et al., 1993). In the Alsace Rhine floodplain the concentrations of nitrate in the groundwater in agricultural zones are higher (4.5 – 5.6 mg $l^{-1}$ N-$NO_3^-$) than those measured under the alluvial forest - meadow system (less than 1 mg $l^{-1}$) (SEMA-DIREN, 1994). The nitrate loss through a riparian zone varies widely: the nitrate inputs were reduced by 56–100% in riparian organic soils (Cooper, 1990), by 73% in a flooded forest and by 37% in a flooded meadow in the Ill floodplain area (Takatert et al., in press) and by only 10% in the floodplain of the Adour river (southern France, Brunet et al., 1993). Moreover alluvial flooded forests greatly reduce nitrate flux of floods which infiltrate vertically through the root-soil system (Sánchez-Pérez et al., 1991a). However, this reducing capacity was modified after suppression of the natural functioning of the floodplains (Trémolières et al., 1998). The nitrate removal processes, now well-known, imply denitrification (Lowrance, 1992; Pinay et al., 1993;

Hanson et al., 1994; Lowrance et al., 1995), microbial immobilization (Groffman et al., 1992) and vegetation uptake (Sánchez-Pérez et al., 1991b; Haycock & Pinay, 1993), the importance of each of which varies according to the environmental conditions, substrate, geomorphic characteristics, the type of vegetation and hydrological regime (Pinay et al., 1995; Sánchez-Pérez & Trémolières, 1997). Catchment hydrology interacts with the biological processes responsible for nitrate depletion. Thus streams which flow through the riparian zone can have a strong influence on nitrate reduction by promoting plant uptake, as has been shown by Cooper (1990) and Fustec et al. (1991). This demonstrates that in evaluating the filtering effect of riparian forests we have to take into account the whole zone, including terrestrial and aquatic systems in the same area.

This study attempts to quantify the fluxes of water and nitrate which flow through an alluvial flooded system in the ground and surface water, and thus to evaluate the part of nitrate removal of the alluvial system. Secondly, we try to determine indicators which control the nitrate efflux from the system.

## Study area

The study site is an area of 4000 ha which is covered by forests and meadows, flooded by the river Ill, the main tributary of the Rhine in the East France. A large network of phreatic streams and diffluents of the Ill flow in this sector over a length of 51 km and cover a surface of 52 ha (Figure 1). The study sector is surrounded by heavily fertilized crops (especially maize) which provide high concentrations of nitrate, around 5.5 mg $l^{-1}$ $N\text{-}NO_3^-$ in the groundwater under the cultivated fields. The Ill hydrological regime and consequently the groundwater regime are characterized by high water in late winter and the beginning of spring (December – March) and low water in summer. Interannual mean precipitations for the years 1994 – 1996 were 546 mm (S.E. Mean = 15 mm) and rainfall inputs during low water periods (May – September) represent almost 50% of the total annual inputs. Soils are gleysoils characterized by the presence of an hydromorphic horizon (gley or pseudogley) at a depth of 0.7m. These soils are subjected to permanent reducing conditions. Two sectors are to be distinguished, a sector located in the east with soils enriched in organic matter (about 20% in the topsoil) and in the west a sector with clayey-silty organic matter-poor soils (less than 7%). During low water period, groundwater level is at 1 m depth in the western sector where overflowing by river occurs every year and at 0.7 m depth in the eastern sector where there is only groundwater rising without overflowing.

## Material and methods

The study involved spatio-temporal monitoring of stream flow and nitrate concentrations in the ground and surface waters, calculation of input and output fluxes and modelling of nitrate transfer through the area.

### *Sampling of ground and surface water*

Groundwater samples were collected in a network of 31 piezometers installed in the gravel layer at 4.5 m depth and perforated over the forest border, 6 upstream (4 in the west sector and 2 in the east sector), and 7 downstream (5 in the west sector and 2 in the east sector) (Figure 1). Monthly surveys of concentrations of nitrate and chloride used as a hydrological tracer were made in the groundwater for the two years 1996 and 1997. The periodicity used for sampling is highly representative of the variations occurring during a water year (Sánchez-Pérez, 1992). Surface waters were sampled and the discharge measured upstream and downstream of all the streams which flow through the forest (Figure 1). Sampling was done three times in 1996 and five times in 1997. In order to evaluate the balance between inputs and outputs, ground and surface waters were sampled on the same dates. The data concerning daily discharge of the Ill River were provided by the "Service des Eaux et des Milieux Aquatiques" (SEMA), French office of waters and the aquatic environment.

Groundwater samples were collected after removal of ten times the water volume of the piezometer by a motorized pumping. Collection was made using an electrical pump. The depth of the water table was measured in each well prior to pumping. Water was pumped from each well for approximately 1 min prior to collecting the sample in order to clear the tubing of the sample from the previous well. The water samples were collected in polyethylene bottles after pumping, filtered in the laboratory through a 0.45 $\mu$m filter and stored in the dark at 4 °C until chemical analyses were performed.

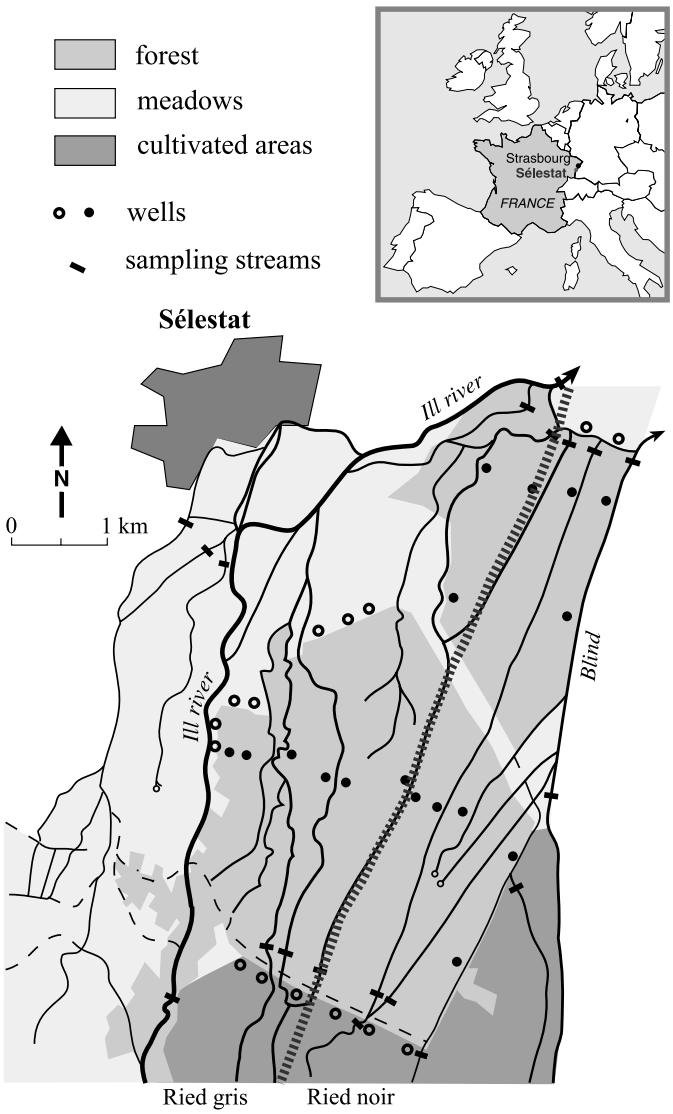


*Figure 1.* Study area, location of sampling sites in ground (piezometers) and surface water.

*Water chemistry*

We measured N-$NO_3^-$ and N-$NH_4^+$ by colorimetry with a microflux automated analyzer (Alliance instruments, Integral 4). $Cl^-$ was analyzed by ionometer with a specific Ag electrode. The procedures used are specified in APHA (1985): ammonium was determined by the indophenol blue method, nitrate by the cadmium reduction method and phosphate by the ascorbic acid method.

Standard chemical analysis procedures were followed including use of replicates and blanks. Each analysis is controlled through a series of standards, which are selected according to the measured concentrations. Data were checked and replicates with a difference of 5% were re-analyzed.

*Groundwater modelling*

A groundwater flow model based on finite elements is used to quantify the water fluxes in the groundwater and in the rivers. Groundwater modelling is made for a surface of 40 $km^2$ (Figure 2). The mesh size is 200 m for the area outside the stream network and 60 m in the stream network. The average thickness of the aquifer in this sector is about 100 m. The transmissivity varied from $1.10^4$ $m^2$ $d^{-1}$ on the west side to $6.10^4$ $m^2$ $d^{-1}$ on the east side of the site.

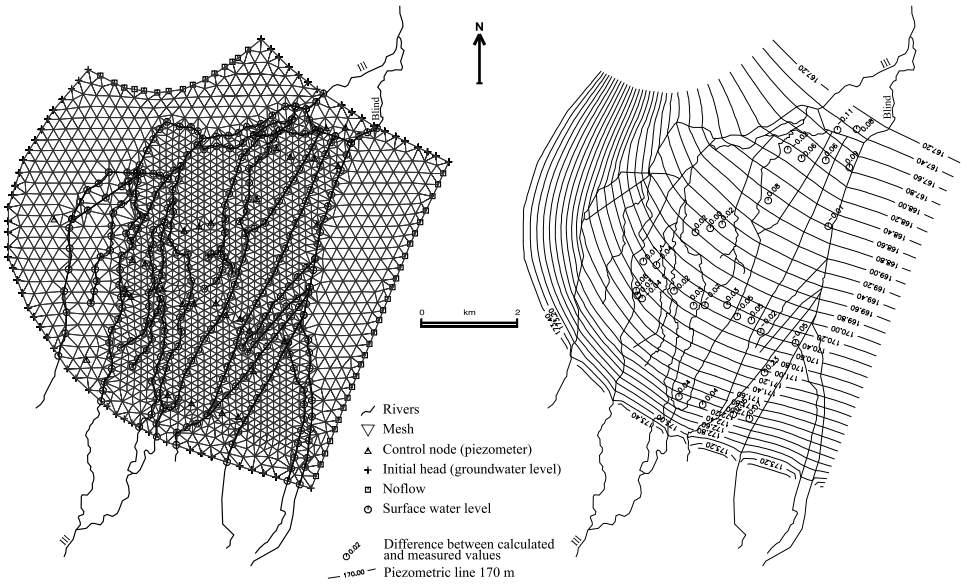


*Figure 2.* Groundwater model: control nodes, boundary conditions and contour lines of the groundwater flux.

The control nodes (piezometers used to calibrate the model) and the flow boundary conditions are presented in Figure 2. The first step is the calibration of a steady state flow model. The hydraulic conductivity and the thickness of the aquifer are taken from numerous modelling studies of groundwater flow done in the same area. The calibrated parameters are the river bed conductance and the prescribed head at the upstream boundary. Calibration is performed on average piezometric head values measured within the domain (29 piezometers used as control nodes) and average water fluxes in the rivers. The double calibration (groundwater heads and surface water fluxes) is expected to ensure the reliability of the estimate of water fluxes.

*Determination of ground and surface water fluxes*

The effects of the vegetation and the soil on nitrate transport in the groundwater is estimated by the difference between the input fluxes at the upstream boundary and the output fluxes at the downstream boundary. These fluxes depend on the water fluxes and nitrate concentrations in the river and streams, and in the groundwater. The different data required are obtained by *in situ* measurements ($N\text{-}NO_3^-$ concentrations in the surface and ground water, flow rate of surface water) or by modelling (which provides the water input and output flow rates in the aquifer). The mass balance of chloride, used as a conservative tracer, is performed to estimate the balance error due to insufficient sampling, measurement and/or model errors. We assume that, if the $N\text{-}NO_3^-$ mass balance error is significantly different from the chloride mass balance error, $N\text{-}NO_3^-$ cannot be considered as conservative and that the decrease in the fluxes are due to additional processes (absorption, degradation,....).

Groundwater fluxes per surface unit (kg N or $Cl^-$ $ha^{-1}$ $d^{-1}$) are determined by coupling the concentrations of $N\text{-}NO_3^-$, $N\text{-}NH_4^+$ and $Cl^-$ measured in the monitoring wells with groundwater flux estimated from the groundwater modelling. The concentrations are average concentrations over the cross-sectional area defined by the dimensions of the model. The volume of the groundwater leaving the control volume per day ($Q$ = daily flow rate $m^3$ $d^{-1}$) was determined by the hydrogeological model for the section area.

The flux of ($N\text{-}NO_3^-$, $N\text{-}NH_4^+$ and $Cl^-$) ($M$) leaving the system was calculated by:

$$M = M_{\text{input}} - M_{\text{output}} =$$
$$(([C_{\text{input}}]\text{x } Q_{\text{input}}) - ([C_{\text{output}}]\text{x } Q_{\text{output}})),$$

where $M$ = mass flux of element leaving (kg $d^{-1}$), $[C_{\text{input}}]$ = average concentration of the element in the input monitoring well (g $cm^{-3}$), $[C_{\text{output}}]$ = average concentration of the element in the output monitoring well (g $cm^{-3}$), $Q_{\text{input}}$ = daily flow rate in the input section ($m^3$ $d^{-1}$), $Q_{\text{output}}$ = daily flow rate in the output section ($m^3$ $d^{-1}$).

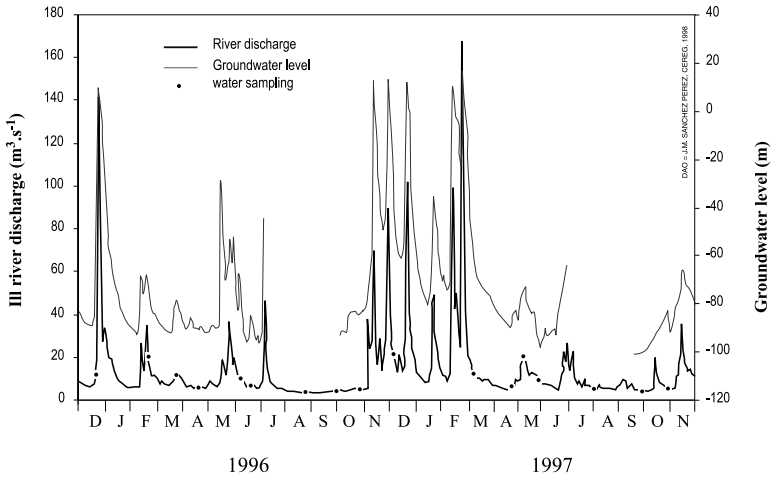


*Figure 3.* Discharge of the river Ill over the study period (January 1996–December 1997) and corresponding dates of ground and surface water sampling.

The groundwater removal rate per surface unit was then obtained by dividing the mass removed per day ($M$) by the surface area (40 km$^2$).

## Results

### *River discharge*

During the sampling low water period, the total river flow range was 8.9 – 22.3 m$^3$ s$^{-1}$ for the inputs and 12.8 – 27.6 m$^3$ s$^{-1}$ for the outputs, which corresponds to a river discharge of around 20 m$^3$ s$^{-1}$. We do not measure the discharge during the overflowing period. According to the daily discharge monitoring of the river Ill, discharge ranges from 160 m$^3$ s$^{-1}$ during the flood events to 3.8 m$^3$ s$^{-1}$ during low water (Figure 3). When we compare the occasional measurements in the river and diffluents to the daily discharge of the river Ill in the study site, the river discharge measured in the network during low water period represented 50% of the total surface water fluxes including the flood water fluxes for the study period (December 1995 – November 1997).

### *Groundwater modelling*

The water table computation presented in Figure 2 by the contour lines at intervals of 0.2, indicated a groundwater flux SW – NE; differences in control nodes were less to 0.1 m. Groundwater flow calculated by the modelling represented 3.98.10$^5$ m$^3$ d$^{-1}$ for the inputs and 1.06.10$^5$ m$^3$ d$^{-1}$ for the outputs.

*Table 1.* Concentrations of $N\text{-}NO_3^-$, $N\text{-}NH_4^+$ and $Cl^-$ in the groundwater, expressed in mg l$^{-1}$

| Date | $N\text{-}NO_3^-$ (mg l$^{-1}$) | | $N\text{-}NH_4^+$ (mg l$^{-1}$) | | $Cl^-$ (mg l$^{-1}$) | |
|---|---|---|---|---|---|---|
| | Input | Output | Input | Output | Input | Output |
| 23/12/95 | 2.71 | 0.31 | 0.03 | 0.57 | 72.1 | 64.5 |
| 24/02/96 | 5.87 | 0.85 | 0.03 | 0.41 | 77.7 | 61.3 |
| 22/03/96 | 3.63 | 0.52 | 0.18 | 0.60 | 79.7 | 71.5 |
| 22/04/96 | 4.02 | 0.91 | 0.07 | 0.08 | 79.3 | 80.1 |
| 30/05/96 | 6.03 | 0.80 | 0.02 | 0.06 | 83.2 | 78.1 |
| 24/06/96 | 2.66 | 0.59 | 0.08 | 0.07 | 77.8 | 68.7 |
| 20/07/96 | 5.87 | 0.63 | 0.06 | 0.44 | 86.0 | 67.0 |
| 27/08/96 | 4.46 | 0.46 | 0.02 | 0.22 | 82.2 | 68.7 |
| 6/10/96 | 5.81 | 1.15 | 0.01 | 0.68 | 82.1 | 77.4 |
| 1/11/96 | 5.36 | 0.91 | 0.04 | 0.10 | 81.3 | 84.3 |
| 8/12/96 | 1.37 | 1.38 | 0.05 | 0.07 | 76.8 | 89.7 |
| 23/03/97 | 2.68 | 0.69 | 0.03 | 0.03 | 68.7 | 67.4 |
| 26/04/97 | 2.48 | 0.43 | 0.01 | 0.08 | 69.0 | 74.7 |
| 10/06/97 | 3.30 | 0.53 | 0.06 | 0.57 | 80.5 | 69.7 |
| 19/07/97 | 4.80 | 0.65 | 0.03 | 0.66 | 68.1 | 68.3 |
| 14/09/97 | 4.63 | 1.34 | 0.03 | 0.35 | 79.5 | 74.8 |
| 19/10/97 | 2.29 | 0.34 | 0.08 | 0.50 | 81.5 | 80.0 |
| 23/11/97 | 4.79 | 0.58 | 0.01 | 0.33 | 80.3 | 81.6 |
| Mean | 4.04 | 0.73 | 0.05 | 0.32 | 78.1 | 73.8 |

Drainage represented 2.92.10$^5$ m$^3$ d$^{-1}$ (or 3.4 m$^3$ s$^{-1}$) and corresponded to the differences of surface flow measurements between upstream and downstream in the river network (3.2.10$^5$ m$^3$ d$^{-1}$ or 3.7 m$^3$ s$^{-1}$, Table 2).

*Table 2.* Mean daily fluxes in streams and groundwater over the study period

| | Inputs (I) | Outputs (O) | O - I | Δ(O-I/I) |
|---|---|---|---|---|
| **Streams** | | | | |
| Flow ($m^3.d^{-1}$) | 1 347 840 | 1 667 520 | 319 680 | 24% |
| N-$NO_3^-$ ($kg.d^{-1}$) | 3986.9 | 4209.3 | 222.4 | 6% |
| N-$NH_4^+$ ($kg.d^{-1}$) | 102.9 | 198.5 | 95.6 | 93% |
| Cl- ($kg.d^{-1}$) | 80 730 | 104 663 | 23 933 | 30% |
| **Groundwater** | | | | |
| Flow ($m^3.d^{-1}$) | 398 000 | 106 000 | –292 000 | –73% |
| N-$NO_3^-$ ($kg.d^{-1}$) | 1608.7 | 77.0 | –1531.8 | –95% |
| N-$NH_4^+$ ($kg.d^{-1}$) | 18.2 | 34.4 | 16.2 | 89% |
| Cl- ($kg.d^{-1}$) | 31 085 | 7819 | –23 267 | –75% |
| **Groundwater + Streams** | | | | |
| Flow ($m^3.d^{-1}$) | 1 745 840 | 1 773 520 | 27 680 | 2% |
| N-$NO_3^-$ ($kg.d^{-1}$) | 5595.6 | 4286.3 | –1309.4 | –23% |
| N-$NH_4^+$ ($kg.d^{-1}$) | 121.1 | 232.9 | 111.8 | 92% |
| Cl- ($kg.d^{-1}$) | 111 815 | 112 482 | 666 | 1% |

*Inputs and outputs of nitrate and ammonium nitrogen and chloride in ground and surface water*

Concentrations of nitrate nitrogen in the groundwater input collected through the study period can be relatively high and vary widely from 1.37 to 6.03 mg $l^{-1}$ whereas the output concentrations are very low, always less than 1.38 mg $l^{-1}$ (Table 1). The groundwater in flux into the system was 546.9 – 2399.9 kg $d^{-1}$ and the efflux was 32.9 – 146.3 kg $d^{-1}$. The nitrate nitrogen fluxes in the stream network are evaluated at 3986.9 kg $d^{-1}$ input and 4209.3 kg $d^{-1}$ output (Table 2). Ammonium nitrogen inputs fluctuate over the study period between 8.2 and 440.6 kg $d^{-1}$ (mean 102.9 kg $d^{-1}$) in the surface water and between 2.9 and 70.6 kg $d^{-1}$ in groundwater. Outputs are always higher than inputs, between 19.5 and 922.9 kg $d^{-1}$ (mean 198.5 kg $d^{-1}$) in the surface water and 2.8 – 71.8 kg $d^{-1}$ in groundwater flow. Outputs by streams are close to the inputs except in April and June 1996. The chloride fluxes represented an input of 80 730 kg $d^{-1}$ and 104 663 kg $d^{-1}$ output in the streams. In the groundwater, the concentrations, about 75 mg $l^{-1}$, do not vary between inputs and outputs. Expressed in terms of the fluxes, 27 110 – 34 228 kg $d^{-1}$ enter the system and 6498 – 9503 kg $d^{-1}$ flow out the system (Table 2).

*Variations of inputs and outputs*

The discharge of the river Ill is well correlated with the groundwater level and shows several overflow peaks in winter (Figure 3). The lowest drainage effect is observed for the highwater period (December survey) when the difference of discharge between input and output is the lowest for the whole period. In the surface water, the change of nitrate fluxes over the whole period shows a trend to the elimination of nitrate from the system by streams. In winter, nitrate nitrogen output over input rises to 36%, whereas in summer we measured retention in the system of the nitrate brought by streams, (around 10–20%) (Figure 4). In contrast to nitrate, variations of ammonium fluxes show periods of elimination in summer and periods of retention in the system in winter. The mean nitrate evacuation in the stream system is negligible (Table 2). About 30% (15 – 43%) of chlorides are evacuated from the stream system which is close to the increase of output discharge (Figure 4). Chloride flux evacuated by stream system represented 23 933 kg $d^{-1}$.

In groundwater, nitrate fluxes are reduced by 95%. The ammonium output fluxes are 89% higher than the input flux. Chloride concentrations show little differences between input and output from the area over the study period whereas fluxes of the inputs were reduced by 75%, i.e. by 23 267 kg $d^{-1}$ (Table 2).

It is assumed that the groundwater fluxes remain constant over time. In this domain, groundwater level fluctuations are small due to the numerous rivers which stabilize the water level by infiltration/drainage. Therefore, this assumption may be considered as valid. For chloride, it is assumed that the efflux of chloride from surface waters were 30% higher than the influx and was compensated by the reduction of chloride in groundwater fluxes. The loss of nitrate in the groundwater is significantly higher, about 95% on average. If we consider that the loss due to the rivers is negligible, then nitrogen influx is largely absorbed or degraded by the vegetation and/or the soil.

*Estimate of nitrate removal*

To evaluate the nitrate removal by the system we calculate balance of inputs and outputs of nitrate and chloride by including fluxes in the streams and in groundwater. The concentrations used were those measured at 5 m depth, but we assume that these measurements are representative of the concentrations measured at a depth of less than 50 m (our measurements correspond to the measurements of the monit-

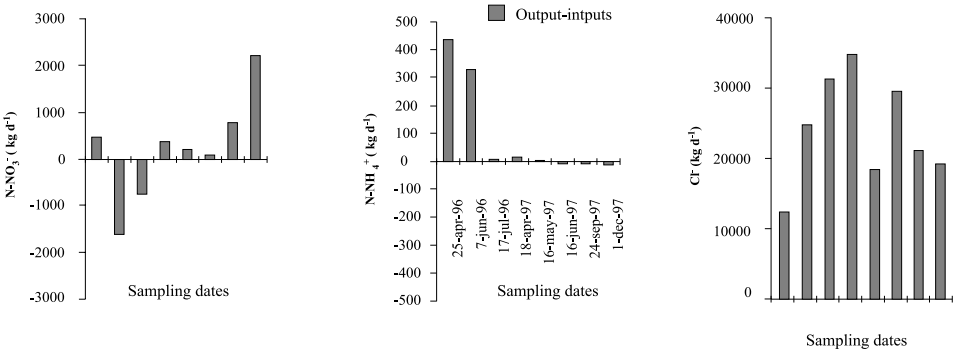


*Figure 4.* Nitrate, ammonium and chloride fluxes in the stream network of the study area.

oring network done by French office of waters and the aquatic environment – APRONA at around 50 m depth). Under these conditions, we can calculate the nitrate removal at a depth of 100 m (depth used in the model) and the surface area of the model (40 $km^2$); the estimation of nitrate removal by the alluvial zone studied was 559 t N $yr^{-1}$, which represented a global $NO_3^-$-N removal capacity of about 1397.7 kg N $ha^{-1}$ $yr^{-1}$.

## Discussion

### *Nitrate depletion in groundwater*

Nitrate removal seems to be particularly efficient with up to 95% reduction when the groundwater flux flows through the system, even in winter when there is no vegetation effect. The chloride used as hydrological tracer varies between 60 and 90 mg $l^{-1}$ in the zone and is preserved globally during the transfer in groundwater through the system. Comparisons of chloride and nitrate fluxes showed that the alluvial floodplain effectively retained nitrate entering the system, findings which were in good agreement with studies on the fate of nitrate in groundwater using $Br^-$ as a tracer (Hubbard & Lowrance 1996; Starr et al., 1996).

By contrast, ammonium fluxes increase from upstream to downstream. This result demonstrates that a part of nitrate is reduced (Schipper et al., 1994), but this cannot explain the overall removal. Denitrification (measurements in progress) might be involved in nitrate removal by a permanent reducing horizon of soil in our study area as has commonly been shown by numerous studies (e.g. Lowrance, 1992; Pinay et al., 1995). This process was demonstrated as the main process in the elimination of nitrate in flooded zones (Leonardson et al., 1994).

However, a large network of streams in this sector drains the groundwater and thus can have an effect on nitrate removal from groundwater. In this study we try to combine the relative roles of the soil-forest system and the drainage by streams.

### *Effect of drainage*

Overall, the discharge of the streams was high and thus streams transport high quantities of chloride and nitrate, about 10 and 50 times more, respectively, than in the groundwater. Moreover these streams drain the groundwater with a regular (or constant) increase of about 24% of flow entering the system. Globally over the whole study period, nitrate input corresponds to nitrate output. The influence of the high waters is low. In fact, during the flood period the inputs of chloride and nitrate are diluted by the large water volumes brought by the flood (see the episode of December 1996) The retention of nitrate by streams seems to be zero, as has been shown by Jansson et al. (1994a) who measured total retention of less than 3% of the total nitrogen transported in the stream. By contrast, chloride output values from streams are closely correlated with those of discharge increase and, consequently, the overall balance is always positive and indicates an elimination of the pollution by drainage. So the differences observed between the behaviour of these two elements show that the nitrate balance could express a retention of nitrate in streams, albeit low. In fact, in summer the trend is to retention in the streams and could reflect a stream plant uptake (Cooper, 1990) and/or of terrestrial plant uptake in groundwater. Svendsen & Kronvang (1993) show that in-stream retention can effectively reduce nutrient export to downstream aquatic systems in low water periods and that areas with macrophytes have the highest potential for storage nutrients. It is also suspected that the decrease of ni-

trate concentration in surface water is the result of upwelling of nitrogen-poor groundwater rather than nitrate removal by aquatic plant and/or denitrification (Trémolières et al., 1994; Jansson et al., 1994b). The efficiency of nitrate depletion in streams is influenced by nitrate inputs, discharge and consequently by water residence time (Hill, 1988). In spite of absence of measurement during the highwater period, we observe that the highest nitrate evacuation by streams of our study site corresponds to the highest discharge (December, 1997).

*Nitrate removal rates from the groundwater*

The differences in nitrate input and output fluxes may be due to drainage by the surface water, mass balance error, nitrate reduction or plant uptake. The first assumption can be rejected, the effect of drainage is significant for choride fluxes and negligible for nitrate fluxes. However, the differences found in the nitrate balance in the groundwater are significant and a forest – meadow effect (plant uptake and denitrification) could be proposed for nitrate removal. Thus we can conclude *a priori* that the drainage of groundwater has a reduced effect on nitrate removal from groundwater. Moreover the groundwater dilutes nitrate concentrations in streams.

The estimated nitrate nitrogen removal rate is 559 t N $yr^{-1}$ in the study area. This calculation represents a global elimination from the study area (40 $km^2$), and thus corresponds to 1397.7 kg N $ha^{-1}$ $yr^{-1}$ within the area. This rate seemed high when we compared it with a riparian poorly drained zone which retains 120 kg $ha^{-1}$ $yr^{-1}$ (Nelson et al., 1995). Leonardson et al. (1994) report retention varying from 390 to 490 kg N $ha^{-1}$ $yr^{-1}$ in wet meadows whereas in artificially flooded meadows retention was 0 – 100 kg $ha^{-1}$ $yr^{-1}$ in spite of a plant uptake amounting to 50–70 kg N $ha^{-1}$ $yr^{-1}$ and high denitrification losses between 190 and 220 kg N $ha^{-1}$ $yr^{-1}$.

In flooded zones, the nitrate removal depends more particularly on the flood conditions (duration, alternating periods of wet and dry) which control the equilibrium between the various forms of nitrogen. Thus long watering periods favour nitrogen retention or elimination and limit nitrate production which compensates elimination during the dry period. In fact, it is highly probable that the nitrate loss is compensated during the flow by nitrification which increases the amount of nitrate in the groundwater, in spite of favourable conditions for nitrate elimination (permanent hydromorphic horizon and, in some parts of the system, organic soils which increase denitrification potential).

The data collected in this study need to be supplemented by data for the highwater periods in order for the influence of stream drainage on nitrate removal efficiency to be clearly determined. The modelling of nitrate transfer could solve some uncertainties concerning the part of drainage and that of nitrate removal by alluvial system. Moreover, denitrification measurement and monitoring of groundwater in wells installed at various distances from the entrance to the alluvial system will allow us to confirm and improve our initial data. Moreover we have to take into account the role of an unsaturated zone which could provide nitrate in the alluvial soils.

## Acknowledgements

This research was supported by IFARE (Institut Franco-Allemand de Recherche sur l'Environnement), PNRH (French National Programme of Research in Hydrology) and PIR Environment SEAH (Systèmes Ecologiques et Actions de l'Homme).

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