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Inorganic nitrogen in relationship with oxygen respiration and organic matter
decomposition in sandy lowland streams
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Inorganic nitrogen in relationship with oxygen respiration and organic matter
decomposition in sandy lowland streams

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1. Introduction

The development of the human society through settlement, agricultural and, industrial activities has determined and modelled on a big extent the landscape and their inherent ecological processes. Especially through agricultural activities, landscapes have been drained, stream beds been straightened and hardened and the use of excessive nutrients have altered the patterns of nutrient flux in the ecosystem and surrounding streams.

In Muensterland (Northwest Germany), for the increasing of the productivity of sandy poor nutrient soils, applications of nutrients in liquid manure form have been historically so high that considerable amount of nutrients are leached from crop field into the groundwater and further towards superficial water bodies like lakes and streams, where they have caused severe eutrophication. This problem needs to be solved, and there is considerable interest in to understand the ways and the capacity of streams to retain, process and, even remove nutrients. This thesis investigates the stream metabolism in relation to oxygen and nitrogen related processes in order to analyze through measurements, incubations and experiments in field and laboratory the nitrogen processing in a lowland stream.

1.1 Effect of climatic variables on the inorganic nitrogen concentration in lowland streams

Nitrogen concentrations in streams in the Muensterland region (North Rhine Westfalia) are among the highest in Germany, with mean nitrate concentrations between 23.3 mg L^{-1} and 30.5 mg L^{-1} . The probable causes are the washout of nitrogen soil saturation originated by the high use of reduced nitrogen in farmland, and the emission of oxidized N forms from combustion of fossilized fuels (Aschemeier 1997, Tarrason and Schaug 2000, Pott and Remy 2003, Spangenberg and Koelling 2004).

Despite the soil in this region having a low potential for agriculture, the farm ground accounts for nearly 70% of the area. In the same way, during the past 30 years the nitrogen turnout has increased from $30 \text{ kg h}^{-1}\text{y}^{-1}$ to around $130 \text{ kg h}^{-1}\text{y}^{-1}$, although lately the emissions have been slightly reduced. This phenomenon is considered the main reason for nitrogen contamination of the groundwater in the region (Henner 1984, Remy 1991, Tarrason and Schaug 2000, Behrendt et al. 2002).

Considering an enforcing effect of the global climatic change, a further increase of nutrient loading of these ecosystems can be predicted (Nilsson and Berggren 2000, Nijssen et al. 2001). Despite of the discharge is not the direct cause of nitrogen inputs in streams, the correlation between inorganic nitrogen concentration and stream discharge varies between stream and regions. Discharge in small basins is principally determined by the quantity of lateral inputs from superficial groundwater which can be strongly seasonal and dos present regional differences. An increase of the discharge can have a positive, negative or have no correlation with the stream nitrogen

concentration. The latter depends on the geology, weather, and soil use (Arheimer et al. 1996, Meyer et al. 1988, Feijoó et al. 1999). Additionally, seasonal shift of biological activity (nitrification, denitrification, and assimilation) contribute to the regulation of the stream nitrogen concentration (Johnson et al. 1997, Clenaghan et al. 1998, Melody and Dodds 2001, Lischeid et al. 2004). With this perspective, the goal of this investigation is to identify the relationship between climatic variables (precipitation, evapotranspiration and, temperature) and the inorganic nitrogen concentration of lowland streams in Northwestern Germany.

1.2 Inorganic nitrogen processing in lowland stream sediments: the nitrification process

Nitrification is the only biological mediated oxidation process which transforms ammonium ($\text{NH}_4\text{-N}$) to nitrate ($\text{NO}_3\text{-N}$). As product of a complete nitrification, the $\text{NO}_3\text{-N}$ is: a) biologically assimilated, b) subject to washout and, c) under hypoxic and anoxic conditions, substrate for the denitrification, or anaerobic oxidation. Consequence of the last two processes, losses of N from ecosystems are predetermined (Hinkle et al. 2001, Kemp and Dodds 2002).

Nitrification rate is regulated by the $\text{NH}_4\text{-N}$ availability (Triska et al. 1990, Jones et al. 1995, Starry et al. 1995), oxygen concentration (Stenston Poduska 1980, Triska et al. 1990, Kemp and Dodds 2001), substrate type (Kemp and Dodds 2002), pH and DOC concentration (Strauss et al. 2002, Strauss and Lamberti 2002). Several studies suggest that the nitrification could be responsible for the increased surface-water $\text{NO}_3\text{-N}$ concentrations observed during winter, when due to low temperature and high discharge the availability of oxygen in the hyporheic zone is enhanced (Richey et al. 1985, Strauss et. al 2006). In sandy streams, especially during high discharge events in winter, the water-flow removes and transports constantly the superficial sediment layers. Consequently, sub-superficial sediments are exposed to aerobic condition. The exposition of buried and $\text{NH}_4\text{-N}$ rich sediments to oxygen can stimulate its dormant nitrifying population and accelerate the growth rate producing high nitrification rates (Triska 1994, Kemp and Dodds 2001).

Experiments with sediment slurries showed that the enhancing of the cation-exchange capacity in sandy sediments amending it with clay minerals (Montmorillonite) increases the ammonium retention (Gardner et al. 2001). The ammonium adsorbed to clay minerals represents a storage pool for the ecosystem which can be accessible for the nitrification, when the soluble ammonium is depleted by biological uptake (Dury and Beauchamp 1991, Triska et al. 1993b, 1994). Richey et al. (1985) found in a small mountain stream that during summer and autumn a significant pool of adsorbed ammonium exists in the stream channel, which could account for the 15-25% of nitrate export during the winter months. However, Triska (1990) found in a small mountain stream that ammonium amendment to sediment slurries only slightly enhanced nitrate production indicating that sorption competed with biota for

substrate. Conversely, Triska et al. (1993b) found a nitrification rate of $3.34 \mu\text{g ml}^{-1}\text{d}^{-1}$ in clay sediments in a tropical stream. On this basis, clay minerals can stimulate or impede the nitrification.

The question to analyze in this chapter is which role plays the oxic superficial sand substrate and the contribution of the anoxic sub-superficial streams sediments on the transformation of ammonium to nitrate, specially the contribution of clay minerals as source or sink of ammonium.

1.3 Relationship between leaf-litter decomposition and nitrogen processing in a lowland stream

Leaf litter is a major source of energy for trophic webs in small streams with extensive riparian forest (Webster et al. 1995, Wallace et al. 1999, Webster et al. 1999). In temperate streams, high CPOM input occurs during the autumn-winter transition, which often coincides with high discharge events. This condition stimulates the initial phase of organic matter decomposition when well oxygenated stream water does enhance microbial activity, and the physical forces of the flowing water further accelerates the breakdown of CPOM (Braioni et al. 2001, Spaenhoff et al. 2007). However, high discharge events, on the other hand, reduce stream CPOM retention capacity, generally causing considerably more material to be floated off than being metabolized. A suchlike stream can be considered as inefficient processor of organic matter (Webster and Meyer 1997).

Following Carling (1996) and Paterson and Black (1999), fine grained sediments are continuously transported, even under base flow conditions, which, however, during high runoffs and spates are subject to massive substrate displacements. This may yield two opposite effects: CPOM can be buried and confined to anoxic conditions, or contrarily, being dug up from deeper sediment layers and abruptly being exposed to oxygenated surface water. In some systems, organic matter stored in these sub-superficial sediments can be higher than that found in the superficial layers (Herbst 1980, Metzler and Smock 1990).

Leaf litter plays an important role as regulator of the nitrogen balance in streams. It takes part on the short term nitrogen immobilization e.g. the microbiological assimilation during the first stages of organic matter decomposition, and at the end, the nitrogen release through the mineralization process. Nitrogen immobilization by litter makes use of nutrient sources and may temporarily reduce the nitrogen content in interstitial water. As the major CPOM inputs into streams occur during the autumn, nitrogen retention in the litter fraction is strongest during the winter period. This process coincides with soil washout and high nutrient inputs from agricultural areas (see chapter 1, Burt and Arkell 1997). However, high discharges and the consequent substrate instability may limit the microbial colonization on CPOM.

The efficiency of oxic vs. anoxic decomposition and their role in the carbon preservation and nitrogen balance in streams have been in debate for more than two decades (Hendrichs and Reeburg 1987, Canfield 1989, Pedersen and Calvert 1990). One theory is that anoxic decomposition is slower than oxic one. Therefore, organic matter accumulates in anoxic sediments and water (Metzler and Smock 1990). However, also under anoxic conditions most substrates can be degraded (Schink 1988). As a product of the organic matter mineralization ammonium is released, which under oxic conditions is rapidly converted to nitrate by nitrifying bacteria, while under anoxic conditions ammonium tends to accumulate.

In this chapter we evaluated the importance of the organic matter on the ammonium content in stream sediments under oxic and anoxic conditions. Burying and digging up of CPOM was experimentally simulated and the ammonium assimilatory activity under anoxic conditions was compared with that of CPOM exposed to an oxic environment. We investigated the relationship between the amount of accumulated organic matter in the sediment and the resulting (1) ammonium concentration, (2) changes of the C:N ratio, (3) nitrogen metabolism (nitrification, assimilation or ammonification) in leaf packs, incubated in the oxic and anoxic environment of a sandy lowland stream.

1.4 Oxygen production and respiration and the relation to light, inorganic nutrient, and discharge

Nutrients do significantly affect the ecological processes in streams, and primary production and respiration are useful indicators of carbon and nutrient cycling in these streams and other ecosystems. Light, temperature, organic matter, nutrients, and flow have been demonstrated to be the principal variables, which determine the production and respiration in streams. However, the effect of each one varies between streams, catchments, and regions as a consequence of, among others, differences in riparian vegetation, channel form, geology, discharge regime, and land use (Bott et al. 1985, Mulholland et al. 2001). In addition, annual variation is substantial due to seasonal changes of vegetation phenology, discharge, nutrients concentrations, light availability, and temperature (Naiman 1983, Young and Huring 1996, Sabater et al. 2005, Acuña et al. 2008).

Despite of the accumulated knowledge about this topic, sandy lowland streams of Central Europe have been poorly studied. Only few studies have been published (e.g. Christensen et al. 1990, Kaenel et al. 2000, Fischer and Push 2001, Sand-Jensen and Pedersen 2005, Sand-Jensen et al. 2007). The goal of this study was to examine the principal variables, which control the rate of stream metabolism at a reach scale. We report the observed production and respiration rates in a sandy lowland stream, following the annual seasonal cycle and its relationship with inorganic nitrogen, organic matter, discharge and temperature.

2. Study site

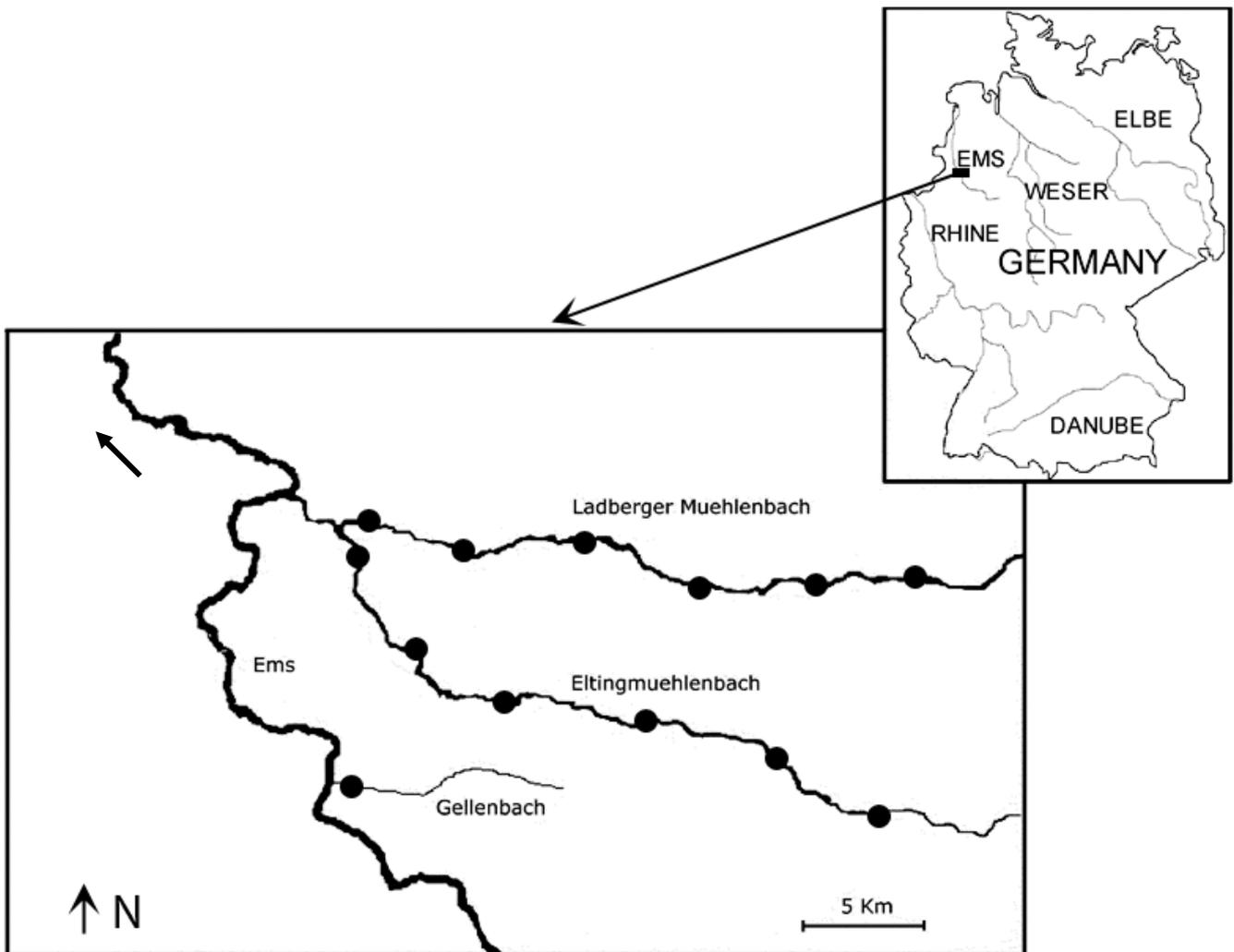


Figure 1. Location of the study site area in Muensterland (NRW, Germany Lat. $51^{\circ}58'$ N, Long. $7^{\circ}37'$ E) in the river Ems basin. Black dots denote sites where the samples have been taken.

This work was realized in Northwest Germany, Lat. $51^{\circ}58'$ N, Long. $7^{\circ}37'$ E. It is a lowland region at an altitude of around 48 m.a.s.l. and a slope of 1 ‰ (Fig. 1).

The investigation was carried out at three lowland streams Eltingmuehlenbach (EMB), Ladberger Muehlenbach (LMB), and Gellenbach, all tributaries of the river Ems (Northwest Germany). The first two flow together near of the confluence with the Ems, forming the Glane. The EMB is 51.4 km long, the LMB 20,4 km and the Gellenbach 12 Km, and the catchments area covers approx. 231.51 km² (Fig. 2) (Landesamt für Wasser und Abfall Nordrhein-Westfalen 1986).

The Muensteraner basin represents the most closed ground water landscapes of North West Germany. It is 150 km long, 80 km wide and has an asymmetrical hollow

structure. Its axis passes of Burgsteinfurt over Muenster after Guetersloh. The southern flank runs out very flatly, that tips over north flank very steeply until. It is to be compared to a large tub, in which three layers determine the flow of the ground water. The lowest layer consists of a crevice ground water leader, who emerged during the lower chalk to the lower coniac (145 million years until 88 million years). The Emschermergel (middle coniac until santon, 86 to 83 million years) forms an almost impermeable separation layer. The layer of the Emschermergel becomes more and thicker in the direction of the center of the basin to a thickness of 800 ms. It separates the lower crevice ground water leader who is filled usually with Sole, from the superficial pore ground water leaders. At some places, this barrier is interrupted reaching the surface. A sodium-hydrogencarbonate-water develops from the sodium containing Sole and the surface near hydrogencarbonate water (fresh water) (Geological state office NRW 1995). The Emschermergel reaches the surface in the Ems region and is a component of the sediments of the Gellenbach, Eltingsmuehlenbach and Landberger Muehlenbach neat to the river Ems.

The soil types are podsol and gleysol, characterized by a low nutrient content and a high hydraulic permeability. The region has been used by man since prehistoric times. The first agricultural activity started in the early middle Ages, and from about 1200 years ago they were used on communal crop field situated on the best drained parts of the landscape, mostly the higher flanks of glacial moraines and on sand dunes. To make the poor sandy soils more fertile, sods with sheep excrements were put on the communal crop fields. On these fields organic matter and nutrients accumulated over centuries, creating fertile soils with layers rich in organic mater eventually reaching a thickness of about 1 meter. During the last 30 years intensive manure practices has yielded locally to a soil eutrophication.

Nowadays, these soils are annually exposed to two phases of nitrogen enrichment which are fertilized mostly with liquid manure: the first, in spring (April and May) during the growth phase, and the second in autumn (September and October) when crop roots and dead leaves break down (Remy 1991).

The annual mean precipitation (1990-2007) is 780 mm, 51 % of which fall during the summer half year (April to September) and 49 % during the winter half year (October to March) (German Weather Office (DWD), Muenster-Osnabrueck Airport Weather station). However, flow discharges show their maximum in winter and their minimum in summer. The groundwater table fluctuates between 50 and 100 cm beneath the terrain surface. It drops below 100 cm after dry periods and frequently forms wetland patches after heavy rainfalls during winter (Cleven and Meyer 2003).

The riparian vegetation has been reduced to a minimum, the 81% of the region is used for agricultural activities, and the stock farming represents an important production with bovine and swine as the major products. The 72% of the riparian vegetation corresponds to the strong impacted class, where well developed riparian vege-

tation is almost inexistent. The streams have been straightened and deepened so that the water flows faster, to avoid flooding during winter, being so the 85% of the streams course heavily impacted (STUA Münster 2001)

The streams sediments are composed of a layer of sand of 5 to 30 cm profundity with a mean grain size of 125 μm , deposited on a clay layer (Emschermergel) which isolates the stream water from the aquifer (Struckmeier 1990).

The experiments and measurements made for the methodology subchapters 3.2 to 3.8 where done in the Gellenbach, where it flows through the Bockholterberge natural reserve, there, the stream reach presents almost natural conditions, the stream structure is meandered and abundant vegetation composed of european beech (*Fagus sylvatica*), oak (*Quercus robur*), alder (*Alnus glutinosa*), elder (*Sambucus nigra*) and some willow (*Salix sp.*) dominate the stream reach, which are the major CPOM source for these stream.

3. Methodology

3.1 Dissolved inorganic ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$) nitrogen and discharge

Water chemistry parameters were obtained from the database of the Department of Limnology, Institute for Evolution and Biodiversity, University of Muenster. The chemical parameters $\text{NH}_4\text{-N}$ (Ammonium nitrogen) and $\text{NO}_3\text{-N}$ (Nitrate nitrogen) were measured monthly (excepting for the lack of sampling during the months February, June, and September) during 1992, results where analyzed on a seasonal basis taking according to the discharge regime as following for spring (March, April and May), summer (July, august), autumn (October and November), and Winter (December and January).

From six sampling sites along the streams the $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ water concentrations were analyzed according to German standard methods ($\text{NH}_4\text{-N}$: DIN 38 406 E 5-1, $\text{NO}_3\text{-N}$: DIN 38 405 D 9-2) (Aschemeier 1997). All the sampling sites were characterized to have open reaches, where the riparian vegetation has been cleared and agricultural activities specially corn crops dominates the landscape.

The potential evapotranspiration was determined following Thornthwaite (1948). Precipitation and temperature data were obtained from the German Weather Office (DWD). The daily discharge data were provided by STUA (Staatliches Umweltamt, Muenster).

Water samples were collected monthly during 2002 for each climatic season. Concentration of dissolved inorganic nitrogen (DIN), composed of ammonium ($\text{NH}_4\text{-N}$) plus nitrate ($\text{NO}_3\text{-N}$), and orthophosphate ($\text{PO}_4\text{-P}$) were determined the same day of sam-

pling. Ammonium was analyzed by the Indofenol-Blue method, nitrate by the Griess-Ilosvay colorimetric method. PO₄-P was analyzed with the ascorbic acid method. Concentrations were determined with a UV-visible recording spectrophotometer (UV-160, APHA 1992).

In the river Gellenbach during summer and winter of 2005, the same measurements were realized in order to compare the seasonal pattern to the observed in the EMB and LMB. Moreover, hourly discharge and stream temperature data were obtained from a permanent gauging station located in the study site.

3.2 Determination of organic matter content in sediments and sediment grain size

For the chapter 4.2 (Potential nitrification), contents of CPOM (Coarse Particulate Organic Matter, > 0.5mm) and FPOM (Fine Particulate Organic Matter < 0.5mm) (following Cummins 1974) were determined for superficial sediments (1-5cm depth) and sub-superficial sediments (5-10 cm depth) for three different stream sectors, including riffles, pools, and transition zones, for which three replicate sediment samples by means of a Perspex core sampler (50 cm long and 10 cm diameter) were sampled monthly from May 2005 to September 2005. Sediment samples were conserved at 5 °C conditions in hermetic plastic containers. At the laboratory, the sediment was sieved through a 0.5 mm mesh, dried at 70°C during 24 hours, weighed, muffled at 560 °C during 4 hours, and finally reweighed again, after cooling to constant weight, to calculate the content of total ash free dry matter given in g AFDM kg⁻¹ sediment. For the same sediments, the grain size spectrum was determined by dry sieving the sediment sample through several standard mesh sieves (500, 250, 125, 63 µm) following Matthes and Ubell (1983) and Tucker (1996). In total 15 superficial and 15 sub-superficial sediment samples were analyzed.

For the chapter 4.3 (Relationship between organic matter and nitrification) 9 samples were sampled, 3 from each stream section (riffle, pool and transition), following the method described in the section 3.2 Superficial and sub-superficial sediments where sieved through a sieve of 0.5mm in order to determine the CPOM content (< 0.5mm) and FPOM content (> 0.5mm), samples were taken during March and May 2007.

For the chapter 4.4 (Oxygen metabolism in a lowland stream), each two months from January to November 5 sediments cores of 5 cm profundity in sediment were extracted randomly from an area of 10 m², CPOM and FPOM was determined following Cummins 1994 described as above, organic matter contents in sediment are expressed in terms of AFDM kg⁻¹.

3.3 Extraction and determination of inorganic nitrogen and DOC in sediment

For the above sediment samples (for the chapter 4.2 Potential nitrification and 4.3 relationship between organic matter and nitrification), from 10 g wet mass sediment from each layer, the inorganic nitrogen content ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$) was extracted in a 100 ml CaCl_2 solution (0.01 mol L^{-1}) placed into a 300 ml Erlenmeyer flask, extracted at low speed (120 rev./min.) on a horizontal shaker during 1 hour. ($\text{NH}_4\text{-N}$) was determined by the Indofenol Blue method and $\text{NO}_3\text{-N}$ by the Griess-Ilosvay colorimetric method (Kenney and Nelson 1982), the concentrations was determined with UV-Visible Recording Spectrophotometer UV-160. Finally, DOC was analyzed with a Shimadzu TOC analyzer (TOC-V CSN). The dissolved $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, and DOC of stream water were determined the same way.

3.4 Oxygen determination in sediments

In 5 transversal transects, each 60 cm of distance, the oxygen concentration was measured using a fibre optic adapted in a pipe of 5 mm diameter and 30 cm long with a net of $125 \mu\text{m}$ mesh size on the extreme. The device was inserted into the sediment at 3, 6, and 9 cm. Subsequently, the oxygen was measured through the contact of the sensor with the surrounding interstitial water. The signal was interpreted by atmospheric press and temperature compensated PreSens Microx oxygen meter (Precision Sensing GmbH).

3.5 Determination of the nitrification rate in sediments

Each month from May 2005 to September 2005 from 3 sediment samples from three different stream sectors, including riffles, pools, and transition zones the potential nitrification was determined for superficial sediments (1 - 5 cm depth) and sub-superficial sediments (5-10 cm depth). In total 15 superficial and 15 sub-superficial samples were analyzed. The nitrification mesocosms were prepared as follow: To 1 L unfiltered stream water 0.5 ml of Nitrapyrin (Sigma-Aldrich Chemical) (which blocks the NH_4N nitrification) was dissolved in DMSO (dimethyl sulfoxide Sigma-Aldrich) at a final concentration of 10 mg L^{-1} . From each sediment sample, two sub-sample of 20 ml were placed in two separate 300 ml Erlenmeyer flasks, to one of them 100 ml unfiltered stream water at 10 mg L^{-1} Nitrapyrin was added and to the other 100 ml unfiltered stream water with DMSO following Strauss and Lamberti (2000). In total 60 sediment samples were analyzed, 30 with nitrapyrin+DMSO and 30 with only DMSO, 15 from superficial and 15 from lower sediment layers. To eliminate photosynthesis and the subsequent $\text{NH}_4\text{-N}$ assimilation, the flasks were covered with aluminium foil.

The flasks were incubated at $17 \text{ }^\circ\text{C}$ (summer mean stream temperature obtained from our permanent gauging station) on a horizontal shaker at low speed (aprox. 120

rev/min.) under darkness for three days. Dissolved oxygen and pH were measured with conventional WTW instruments for each flask at the beginning, during, and at the end of the incubation to identify shifts in the environmental conditions. After one hour of mixing on the horizontal shaker, and at the end of the three days incubation, 30 ml water of each flask was removed for determination of initial and subsequent final $\text{NH}_4\text{-N}$ concentration.

Nitrification rate was calculated subtracting the observed change of total extractable $\text{NH}_4\text{-N}$ in the flasks containing only DMSO from the $\text{NH}_4\text{-N}$ in the flasks that contained nitrapyrin. Nitrification rate is expressed as nitrogen change (N) per volume of sediment and time of incubation ($\mu\text{g N ml}^{-1}\text{d}^{-1}$). Dissolved oxygen and pH were measured with conventional Clark-type and glass probe instruments (WTW) at the beginning, during, and at the end of the incubation to control for the environmental conditions (Fig. 2).

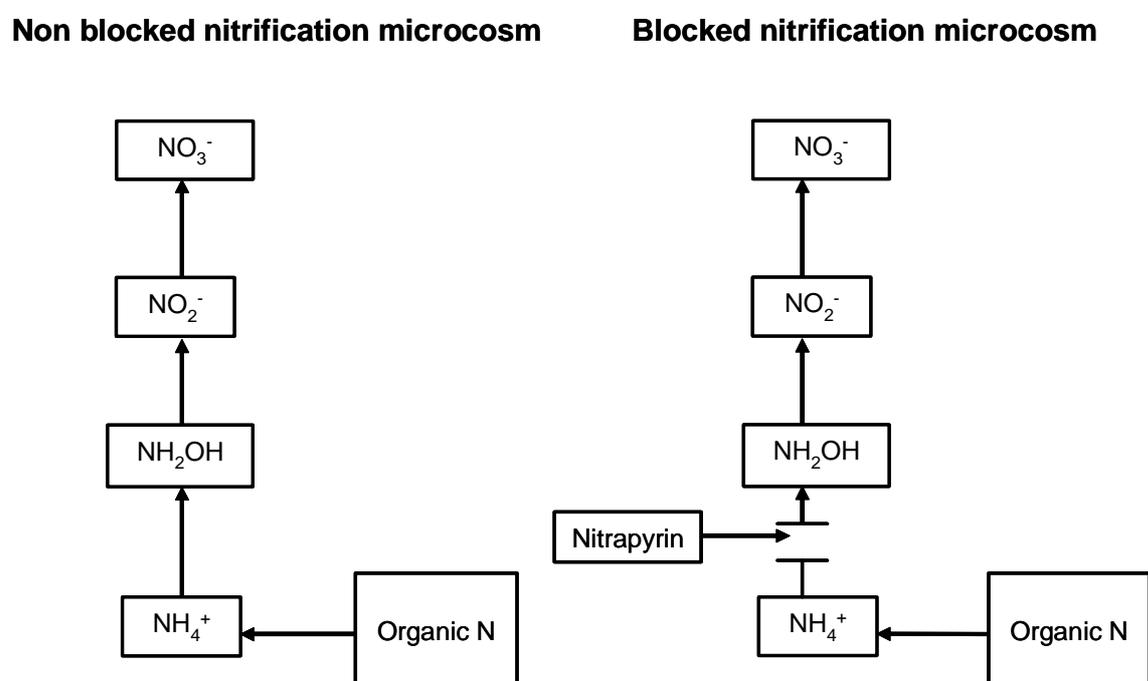


Figure 2. Schematic representation of the microcosms for the determination of nitrification rate in sediments and by CPOM.

3.6 Effect of silt-clayed sediments on the nitrification

From 12 replicates of 20 ml the effect of the $\text{NH}_4\text{-N}$ concentration was analyzed from sand (diameter: 0.063 mm to 0.5 mm) and Silt (diameter < 0.063 mm) sediments at different $\text{NH}_4\text{-N}$ concentrations, adding $\text{NH}_4\text{-N}$ to unfiltered stream water to a final concentration of 0 additions (stream concentration), 0.25, 0.5, and 1 mg L^{-1} .

Sediments were sampled the same as in the chapter 3.5, but in this case, exclusively silt sediments were isolated to be compared to sand sediments.

Organic matter and Nitrification rates were determined using the method proposed in the chapter 3.5.

3.7 Incubation of CPOM under oxic and anoxic environment

Autumn dried leaves of the European beech (*Fagus sylvatica*) were packed in 60 bags (20 x 20 cm size) with a mesh size of 0.3 mm and exposed during 90 days in the river Gellenbach. 30 bags were incubated in an oxic environment (fixed on the sediment surface), the other 30 in an anoxic over a period of 90 days through May, April, and May. Each 15 days CPOM mass loss, C:N ratio (Isotope Ratio Mass Spectrometry, Eurovector S.p.A.), potential nitrification, NH₄-N assimilation, and nitrification rates were determined with the same procedure as the used to determine the nitrification in sediments (Fig. 2).

3.8 Determination of the stream metabolism

Oxygen metabolism was determined using an open reach method where daily changes of oxygen concentrations and temperature were measured continuously at 15 minute intervals in the middle reach of the stream at approximate half depth of the water column, using a Clark-type oxymeter (4002-HP-FS-SP, Syland Scientific), calibrated for altitude and air saturation. Measurements were logged on a digital datalogger (LI-1000, Licor Inc). Data were averaged for one hour. At the same time, the photosynthetically active radiation (PAR) was measured every 15 min. with a quantum sensor (LI-190SA, Licor Inc.) placed over the middle of the stream. Measurements were performed at intervals of two months from January to November during 2004.

Mean daily ecosystem respiration (ER) and the reaeration coefficient (k) were determined using the night time regression method, which uses only data collected in the dark ($\text{PAR} < 2 \mu\text{mol m}^{-2} \text{s}^{-1}$) (Owens 1974). The hourly rate of change of oxygen concentration is regressed against the oxygen deficit (difference between the oxygen concentration at saturation at a given temperature and the observed concentration in the water) to generate the relation: $dO/dt = R + kD$, where dO/dt is the rate of change of oxygen concentration, R is the ecosystem respiration (ER) ($\text{g O}_2 \text{ m}^{-2} \text{d}^{-1}$), k is the reaeration coefficient (s^{-1}), and D is the oxygen deficit ($\text{g O}_2 \text{ m}^{-3}$). The factor k is the slope of the regression line, whereas y is its intercept (Kosinski 1984).

The calculated respiration rate and reaeration coefficient were used to determine the gross photosynthetic rate over 1 hour intervals using $Pt = dO/dt + R - kD$, where Pt is the gross photosynthetic rate over time interval t . Daily gross primary production (GPP) ($\text{g O}_2 \text{ m}^{-2} \text{d}^{-1}$) was estimated as the mean of all temperature corrected photosyn-

thetic rates during daylight, multiplied by the duration of the daylight period. Respiration rates were measured per unit volume (Wiley et al. 1990). An areal estimate was obtained by multiplying the volume-based estimates by average reach depth (m), which did allow direct comparison of measurement samples. Finally, the production-respiration ratios (P/R), were calculated to determine whether the stream reach is autotrophic (P/R > 1) or heterotrophic (P/R < 1), and the net daily metabolism (NDM) as the arithmetic difference between GPP and ER (Odum 1956).

3.9 Statistical analysis

The fit of the data for normal distribution was determined with the standardized Kurtosis and skewness tests with a range of tolerance between -2 and +2. However, in all cases, data did not fit for a normal distribution. Due to that, for all statistical analysis non parametric statistics was used. Mann-Whitney test was used to determine differences between medians of two independent samples, and the Kruskal-Wallis test for multiple independent samples. Spearman Rank Correlation was employed to determine the relationship between two independent variables (Zar 1999). Analysis were done using SPSS ver. 11.0.1 statistic program. Moreover, PCA (Principal Component Analysis) procedure was performed for the temporal distribution of the in-stream concentration of NO₃-N and NH₄-N from the Eltingmuehlenbach and Ladberger Muehlenbach using the statistical program Statgraphics ver. 2.1. (Data were standardized subtracting the mean to each value and dividing it by the standard deviation of the sample data, in order to set them into a comparable scale).

4. Results

4.1 Influence of season and discharge on the concentration of inorganic nitrogen in two low-land streams

4.1.1 Discharge, precipitation, and potential evapo-transpiration

Despite the relative low fluctuation of the monthly precipitation along the course of the year, the mean monthly discharge exhibited a pronounced temporal dynamic, being highest values observed during winter (January and February) and the lowest during summer (July and August) (Fig. 3). On this basis, a direct dependence of mean monthly discharge and monthly precipitation was not observed. However, short strong rain events have evident effect on the discharge, but this effect is more evident during winter than during summer. Contrary to, the cumulative monthly potential evapotranspiration presented an opposed pattern being the highest observed during summer and the lowest during winter, being the pattern opposed to the presented by the discharge. Consequently, there was no correlation between the cumulative monthly precipitation and the mean monthly discharge (Fig. 4a). How-

ever, the cumulative monthly potential evapotranspiration showed a negative and significant correlation with mean monthly discharge ($r = -0.9$, $p < 0.001$) (Fig. 4b).

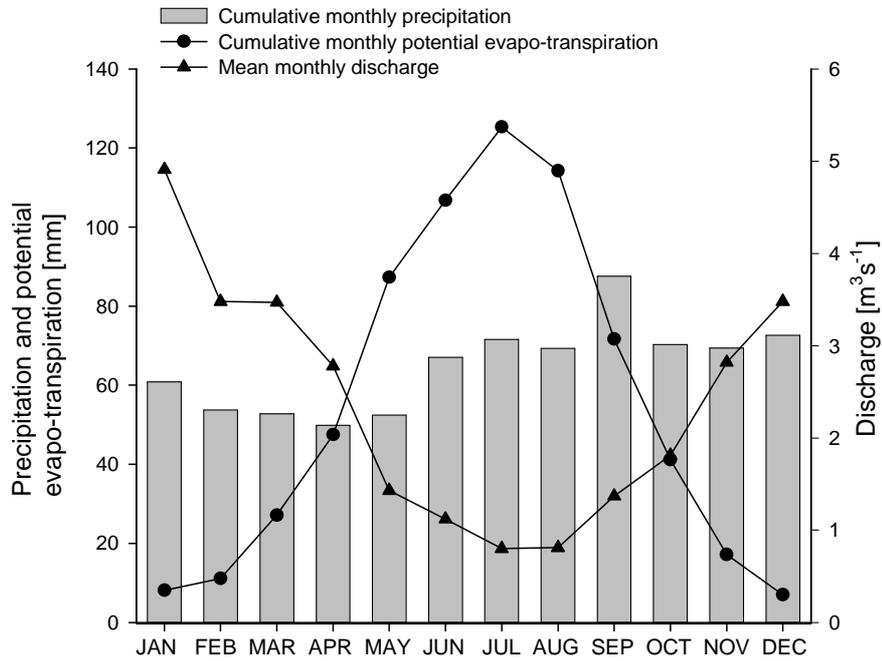


Figure 3. Monthly precipitation, Monthly potential evapotranspiration from the Muenster region (Muenster-Osnabruek Airport weather Station), and mean monthly discharge from the river Glane gauging station located 200 m. downstream the confluence with the Larberger Muenlenbach. Data from 1992 (Data from DWD, STUA Münster).

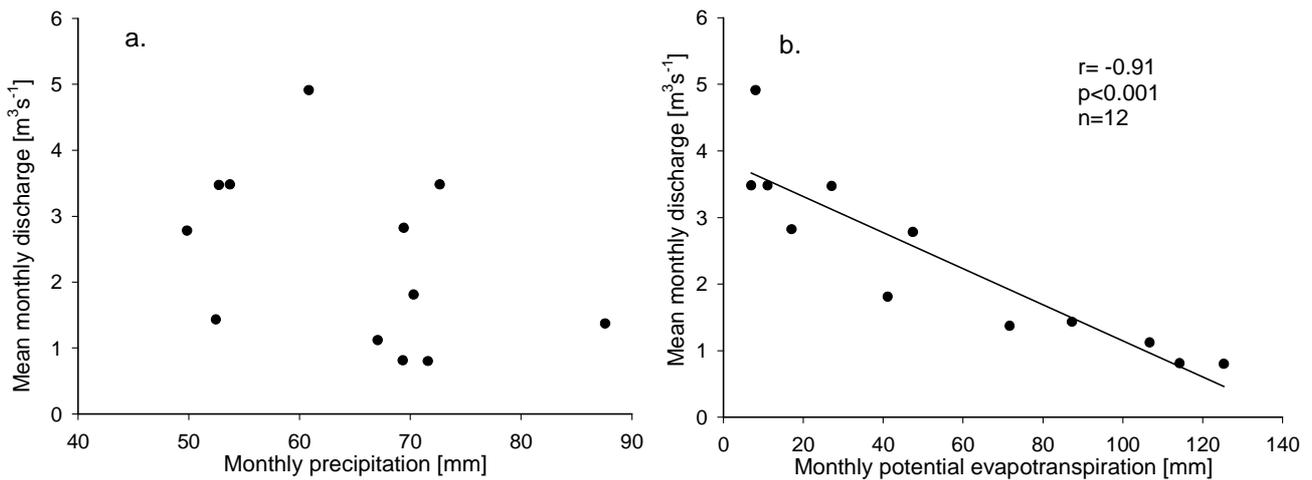


Figure 4. Spearman Rank Correlation between the precipitation (a) and potential evapotranspiration (b) with discharge in the Eltingmuehlenbach and Ladberger Muehlenbach. Data from from the Muenster region (Airport Muenster-Osnabruek weather Station) and mean monthly discharge from Eltingmuehlenbach Gauging station located 200 m. downstream the confluence with the Larberger Muenlenbach during 2002 (Data from STUA, Münster).

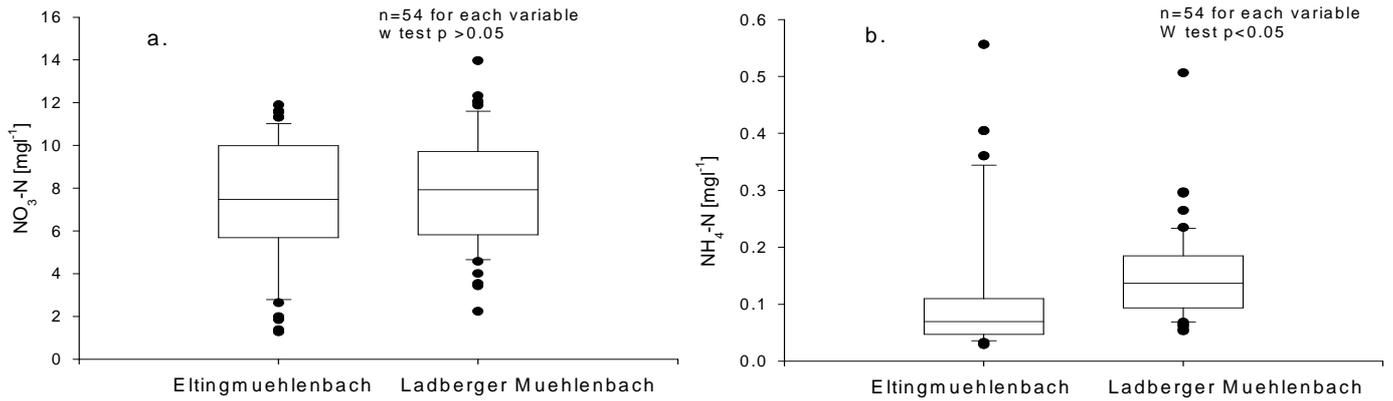


Figure 5. Box and Whisker plot of the total variation of $\text{NO}_3\text{-N}$ (a) and $\text{NH}_4\text{-N}$ (b) in the Eltingmuehlenbach and Ladberger Muehlenbach during 1992. Monthly data from six sampling stations along the two rivers (Data from database Department of Limnology, University of Muenster).

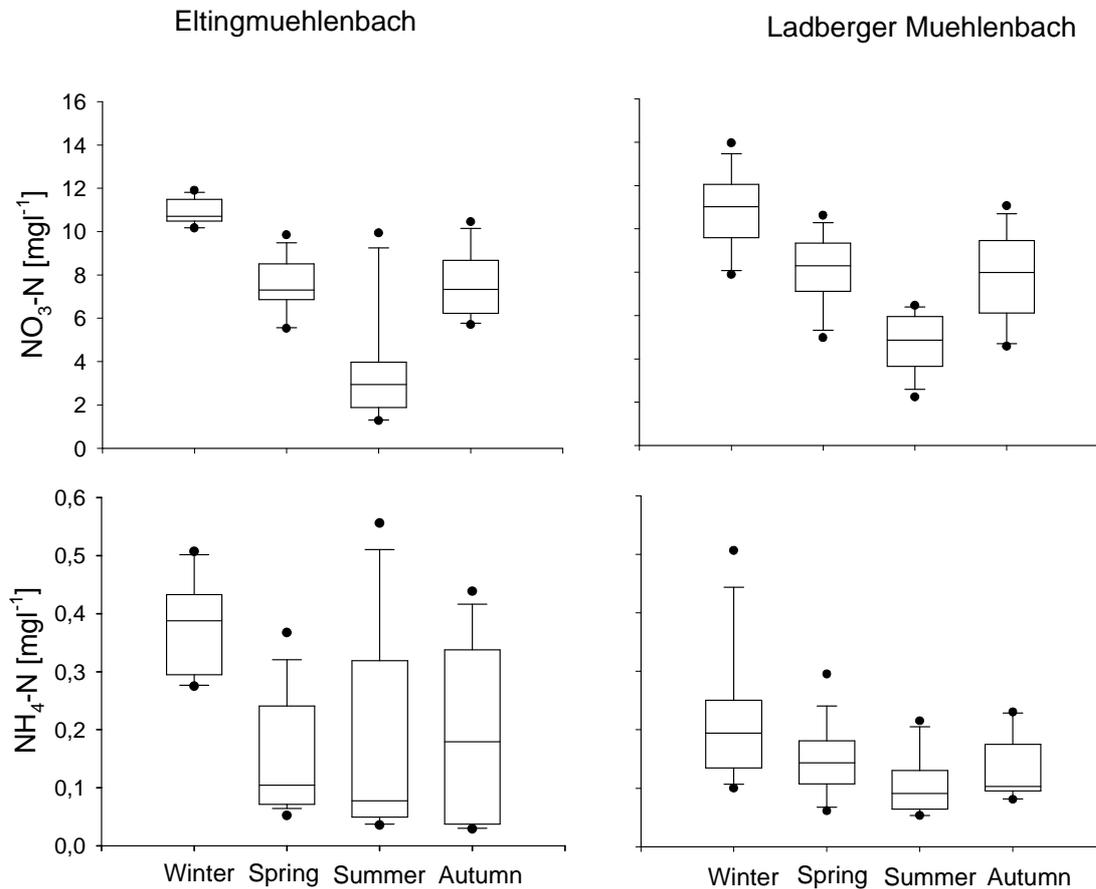


Figure 6. Box and Whisker plot of seasonal distribution of $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ concentrations from six sampling stations in the Eltingmuehlenbach and Ladberger Muehlenbach, during 1992, $n = 12$ for each season, excepting Spring with $n = 18$ (Data from database Department of Limnology, University of Muenster).

The concentrations of nitrogen fractions were in the same order of magnitude in both streams. They showed a high monthly variability, being more variable in the EMB ($\text{NO}_3\text{-N}$: 40 %, $\text{NH}_4\text{-N}$: 108 %) than in the LMB ($\text{NO}_3\text{-N}$: 33 %, $\text{NH}_4\text{-N}$: 53%). There was not any significant difference on the $\text{NO}_3\text{-N}$ concentration between the two rivers, but the $\text{NH}_4\text{-N}$ concentration was significantly higher in the Eltingmuehlenbach (Fig. 5).

4.1.2 Seasonal variation

Both $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ concentrations showed a similar temporal pattern in the two streams, characterized by a significant difference between winter and summer ($p < 0.05$, U-test, two-tailed). The highest concentrations were found during winter (median for the two streams $\text{NO}_3\text{-N}$: 10.9 mg l^{-1} , $\text{NH}_4\text{-N}$: 0.29 mg l^{-1}), whereas the lowest were observed during summer (median for the two streams $\text{NO}_3\text{-N}$: 4.2 mg l^{-1} , $\text{NH}_4\text{-N}$: 0.13 mg l^{-1}), differing almost by the factor two (Fig. 6).

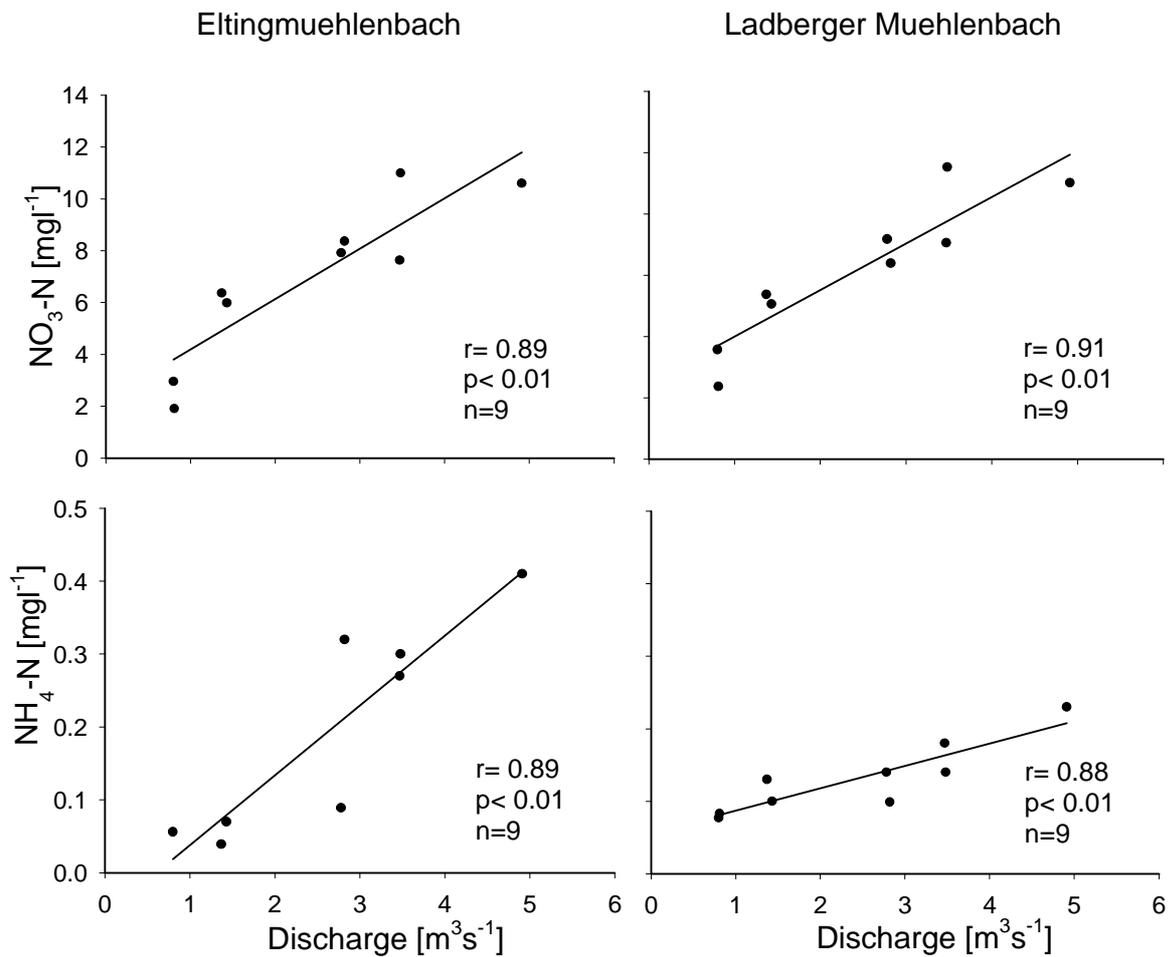


Figure 7. Spearman Rank Correlation between $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ concentration with the discharge in the Eltingmuehlenbach and Ladberger Muehlenbach during 1992. Median monthly (calculated from the monthly median of the six sampling stations for each stream) $\text{NH}_4\text{-N}$ and $\text{NO}_3\text{-N}$ concentration was used for the analysis (Data from database Department of Limnology, University of Muenster and Stua Münster).

During spring and autumn, inorganic nitrogen concentrations are the transitions between summer and winter (decreasing during spring and increasing during autumn). Both $\text{NO}_3\text{-N}$ and the $\text{NH}_4\text{-N}$ concentrations were positively correlated with the discharge ($r = 0.912$, $p < 0.01$ and $r = 0.9$, $p < 0.01$, respectively) (Fig. 7).

4.2 Potential nitrification in sediments

4.2.1 Extractable and dissolved $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, DOC, POM, sediment grain size, and interstitial oxygen concentration

Extractable inorganic $\text{NH}_4\text{-N}$ concentration in sediment varied locally (U-test, $p < 0.05$) with a minimum content in superficial sediments (0.01 mg kg^{-1}) and maximum $\text{NH}_4\text{-N}$ content in lower sediment layers (0.56 mg kg^{-1}). The opposed pattern was observed in the $\text{NO}_3\text{-N}$ content. It varied significantly among the sediment layers (Mann-Whitney test $p < 0.05$) with a minimum of $< 0.001 \text{ mg kg}^{-1}$ in lower sediment layer and a maximum 1.76 mg kg^{-1} in the superficial sediment layers.

Table 1. Total $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, and DOC concentrations and % POM content in superficial and lower sediments from the riffle, pool and transition sectors, sampled each month from May to September during 1999 in the river Gellenbach ($n = 54$).

	$\text{NH}_4\text{-N}$ [mg kg^{-1}]	
	Superficial	Sub-superficial
Average	0.07	0.19
Minimum	0.01	0.02
Maximum	0.16	0.57
	$\text{NO}_3\text{-N}$ [mg kg^{-1}]	
	Superficial	Sub-superficial
Average	0.72	0.49
Minimum	n.d.	n.d.
Maximum	1.76	1.00
	DOC [mg kg^{-1}]	
	Superficial	Sub-superficial
Average	2.65	2.83
Minimum	1.77	1.50
Maximum	4.05	4.55
	POM [%]	
	Superficial	Sub-superficial
Average	0.50	0.70
Minimum	0.14	0.30
Maximum	1.12	1.25

Hence, the amount of extractable $\text{NO}_3\text{-N}$ content in superficial and lower sediment layers was significantly higher than the $\text{NH}_4\text{-N}$ content (Mann-Whitney test, $p < 0.05$) (Table 1).

The DOC content did not differ significantly between sediment layers, being the minimum 1.492 mg kg^{-1} in lower and maximum in lower sediment layers 4.546 mg kg^{-1} (Tab. 1). The POM in the lower sediment layer did not differ significantly among the sediment layers. The maximum POM content was 0.70 % of total sediment mass in lower sediments, whereas the minimum content was 0.514 % in superficial sediment layers (Table 1).

About 50 % of the sediment grain size observed varied from 63 to 125 μm diameter size class. The grain size class < 63 was significantly higher in the lower sediment layers with 39% compared to the 5.8% in the superficial sediment layers (U-test, $p < 0.05$) (Fig. 8a).

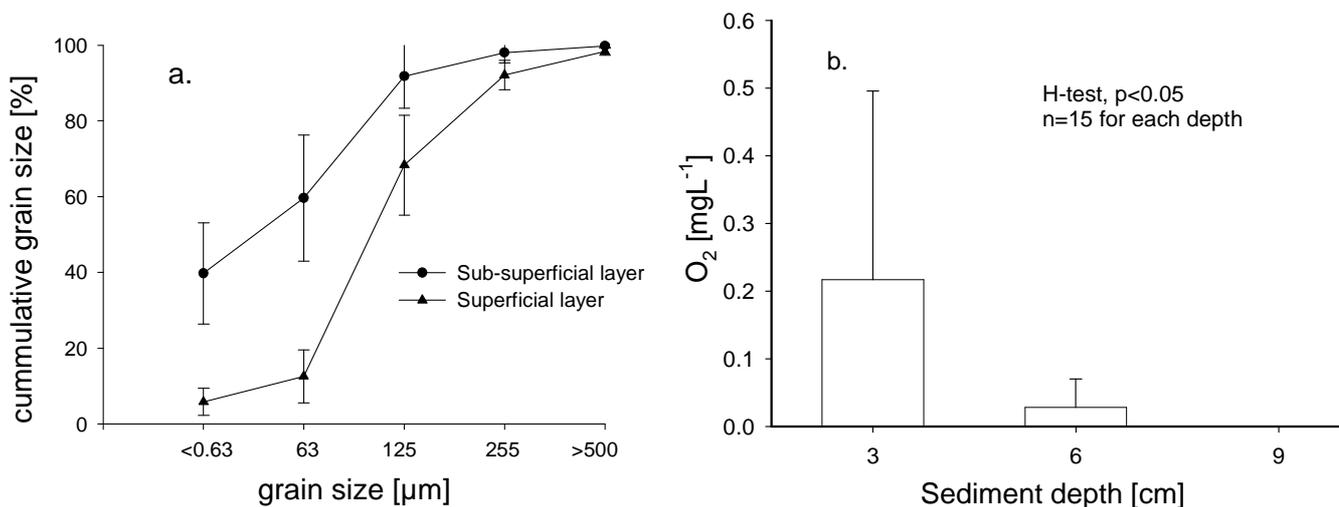


Figure 8. Grain size composition (a) of the superficial (0-5 cm) and sub-superficial sediment layer (5-10 cm sediment depth). Error bars ± 1 are shown. Vertical profiles of oxygen concentration (b) in the sediments at different depths measured direct in sediments in a riffle section in the river Gellenbach. Error bars ± 1 are shown. At 9 cm depth the oxygen concentration was 0 or undetectable. Mann-Whitney test (determined for three different stream sectors, including riffles, pools, and transition zones, for which three replicate sediment samples).

The stream water, the dissolved $\text{NH}_4\text{-N}$ varied from 0.041 to 0.136 mg L^{-1} . DOC in stream water varied from 8.76 mg L^{-1} to 11.02 mg L^{-1} (Table 2). The oxygen concentration was always near of saturation with a mean of 8.32 mg L^{-1} . In sediment, oxygen concentrations displayed significant differences between the lower and superficial sediment layers were evidenced (U-test, $p < 0.05$) with a minimum of 0.0 mg L^{-1} at 9 cm depth and a maximum of 0.71 mg L^{-1} at 3 cm depth (Fig 8b).

Table 2. NH₄-N, NO₃-N, and DOC concentrations from monthly measurements from May to September in the river Gellenbach (n = 5).

	NH ₄ -N [mg L ⁻¹]	NO ₃ -N [mg L ⁻¹]	DOC [mg L ⁻¹]
Average	0.08	2.56	9.90
Minimum	0.04	1.34	8.76
Maximum	0.14	5.53	11.02
Standard deviation	0.04	2.00	1.59

4.2.2 Potential nitrification in superficial and sub-superficial sediments

The potential nitrification in sediment varied from 0 to 4.04 $\mu\text{g N ml}^{-1}\text{d}^{-1}$, it differed significantly among the sediments layers being higher in the superficial (mean = 1.86 $\mu\text{g N ml}^{-1}\text{d}^{-1}$) compared to the lower layers (median = 1.01 $\mu\text{g N ml}^{-1}\text{d}^{-1}$) (U-test, $p < 0.05$). In lower sediments layers, the potential nitrification was significantly lower with 0.49 $\mu\text{g N ml}^{-1}\text{d}^{-1}$ compared to the superficial ones (U-test, $p > 0.05$) (Fig. 9a, 9b). In superficial sediments, the potential nitrification showed a positive and significant correlation with the initial extractable NH₄-N content (Spearman rank correlation $r = 0.688$, $p < 0.05$, $n = 13$) (Fig. 10a). However, in the sub-superficial sediments this relationship was not observed (Fig 10b).

4.2.3 Importance of sand and silt-clayed substrates on nitrification rates

Silt-clayed sediments presented a higher, around three times, FPOM content compared to sand sediments (Fig. 11b). The response of the nitrification to NH₄-N additions was higher on sand substrate than on clay substrate (U-test, $p < 0.05$). On sand substrate, the nitrification increased from 0.33 $\mu\text{g N ml}^{-1}\text{d}^{-1}$ at 0.1 mg L⁻¹ NH₄-N to a maximum rate of 0.912 $\mu\text{g N ml}^{-1}\text{d}^{-1}$ at 0.5 mg L⁻¹ NH₄-N. Despite at 1 mg L⁻¹ NH₄-N, the nitrification was reduced to some extent. Contrarily, on silt+clay substrate, only at 1 mg L⁻¹ NH₄-N nitrification increased significantly to a maximum of 0.330 $\mu\text{g N ml}^{-1}\text{d}^{-1}$, which is significantly lower than the observed on sand substrate under the same treatment (U-test, $p < 0.05$) (Fig. 11a). It is important to note that silt+clay sediments content at least 4 times more FPOM than in sandy sediments (Fig. 11b).

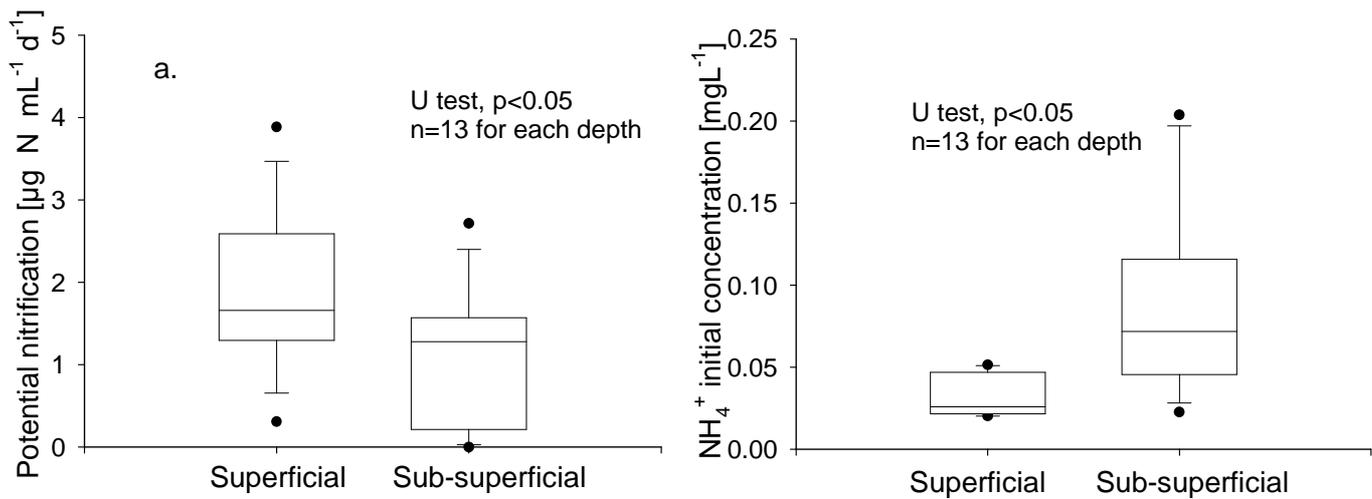


Figure 9. Box and Whisker plot from the Potential nitrification (a) and initial $\text{NH}_4\text{-N}$ concentration (b) in microcosms from superficial and sub-superficial sediments.

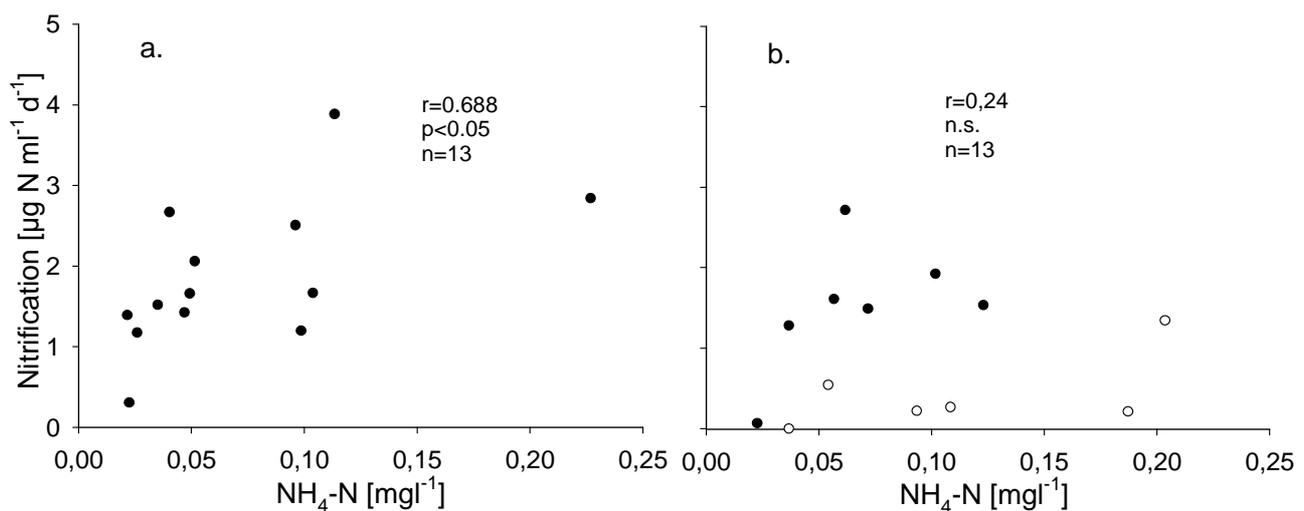


Figure 10. Spearman Rank correlation of the response of the nitrification to the $\text{NH}_4\text{-N}$ concentration in superficial (a) and sub-superficial (b) sediment layers. White dots denote sediment with the $< 63 \mu\text{m}$ (silt + clay) fraction higher than 30%. Each point represents a different sediment sample. The number r is the Spearman Rank correlation.

4.3 Relationship between organic matter and nitrification

$\text{NH}_4\text{-N}$ content in the stream sediment showed a more than 90-fold variation over the 36 sediment samples analyzed, ranging from $0.0062 \text{ mg kg}^{-1}$ to $0.4465 \text{ mg kg}^{-1}$. Measurements from May (mean: 0.138 mg kg^{-1}) were significantly different and higher than those from March (mean: $0.0163 \text{ mg kg}^{-1}$) (Kruskall Wallis test $n = 9$, $p < 0.01$).

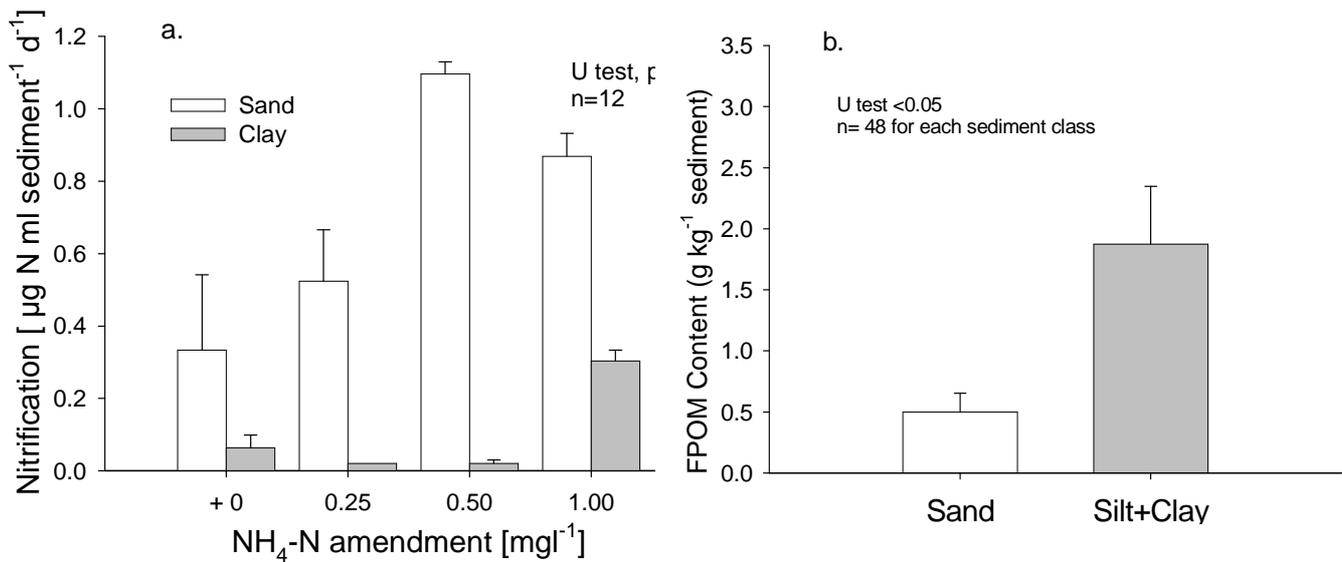


Figure 11. Response of the nitrification process to the increase of dissolved NH₄-N concentration (a) for mesocosms composed stream water plus sand and silt-clayed substrates. FPOM content in stream sandy and silt and Clay substrates (b). Vertical bar shows the median. Errors estimated as the upper 75 percentile. Significant differences for each treatment was determined with the Man-Whitney U-test (** p < 0.05)

Ammonium content did not differ between superficial and buried sediments in March (U-test, n = 9, p > 0.05).

However, during May, ammonium content in sub-superficial sediment was significantly higher (U-test, n = 9, p < 0.05) (Fig. 12). Despite of the significant temporal variation of the NH₄-N content in sediments, a similar pattern was not detected in the stream water, where differences between March and May (mean = 0.125 mg L⁻¹ versus 0.123 mg L⁻¹) were negligible.

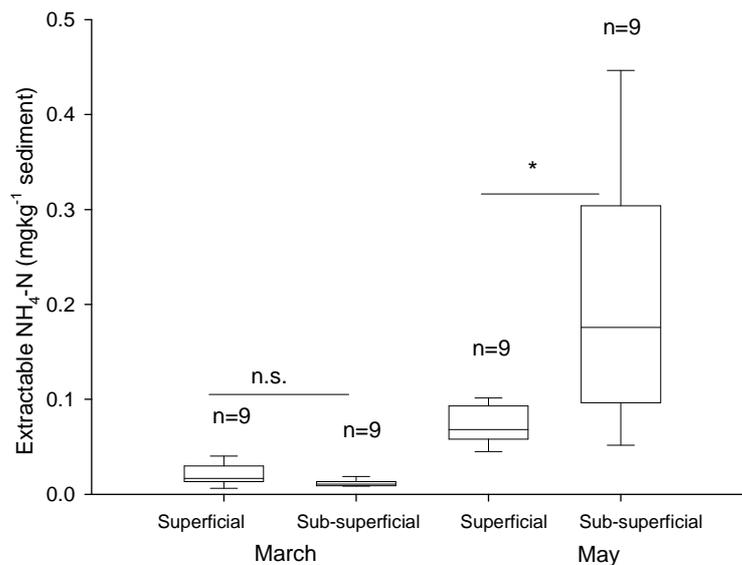


Figure 12. NH₄-N content of superficial and sub-superficial sediments of the study stream (horizontal lines indicate significance levels for differences between sample fractions with ** p < 0.05).

4.3.1 Accumulated organic matter in superficial and sub-superficial sediments

CPOM content of the sediment presented a mean of 2.3 g AFDM kg⁻¹ sediment, with a minimum of 0.057 g AFDM kg⁻¹ and a maximum of 9.1 g AFDM kg⁻¹. Mean measurement readings from May were significantly higher than those from March (3.52 g AFDM kg⁻¹ versus 1.54 g AFDM kg⁻¹) ($W = 61$, $n = 18$ test $p < 0.05$). On the other hand, CPOM content in superficial and sub-superficial sediments did not show significant differences within the respective sampling periods (Fig 13a).

The mean FPOM content in sediment was 1.12 g AFDM kg⁻¹, ranging from a minimum of 0.25 g AFDM kg⁻¹ to a maximum of 4.1 g AFDM kg⁻¹. There was no statistical difference in measurements from March and May. Although, superficial sediment presented a significantly higher FPOM content of 2.15 g AFDM kg⁻¹ than buried sediment with 0.76 g AFDM kg⁻¹ ($n = 18$, $p < 0.01$), likewise in both sampling periods (Fig. 13b).

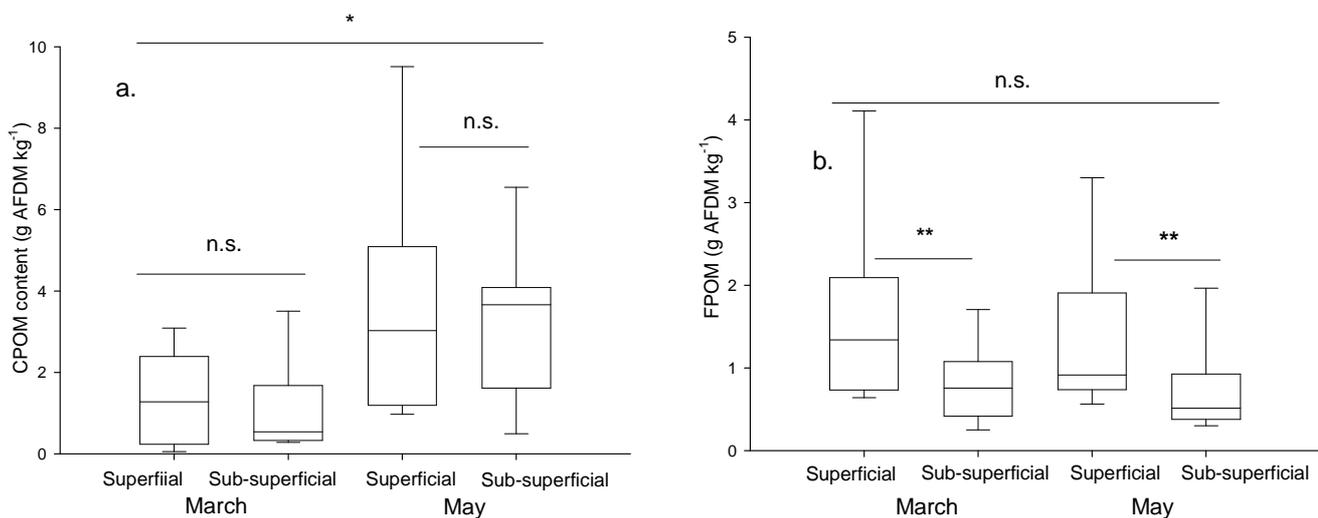


Figure 13. Organic matter content in superficial and sub-superficial stream sediments. Measurements were taken during March and May 2007. CPOM (a), FPOM (b) (horizontal lines indicate significance levels for differences between sample fractions with U-test ** $p < 0.01$, * $p < 0.05$, $n = 9$ for each samples).

Total, superficial and buried CPOM content in sediment correlated significantly with the ammonium content (Fig 14a, b, c), whereas in FPOM no significant correlation with ammonium was found (Fig. 14d, e, f).

4.3.2 CPOM mass loss and change of the C:N ratio in oxic and anoxic environments

The mass loss of CPOM, observed during the first 30 days of in-stream exposure of the leaf packs, was slightly greater under oxic than anoxic conditions, with a decrease of 13.6 % and 10.4 %, respectively. After 90 days after incubation, mass loss of the leaves exposed to oxic conditions did reach a faster rate resulting in a final amount of 39 %, whereas leaves exposed to anoxia remained relatively stable with a

final loss of 15 %. The variation of the mass loss was statistically significant during the sampling period (H test < 0.05, n = 5 for each sampling period). After 45 days of exposition, differences between the mass loss between oxic and anoxic expositions began to be significant, being faster the mass loss on oxic expositions (U-test < 0.05, n = 5 for each sampling period (Fig. 15a).

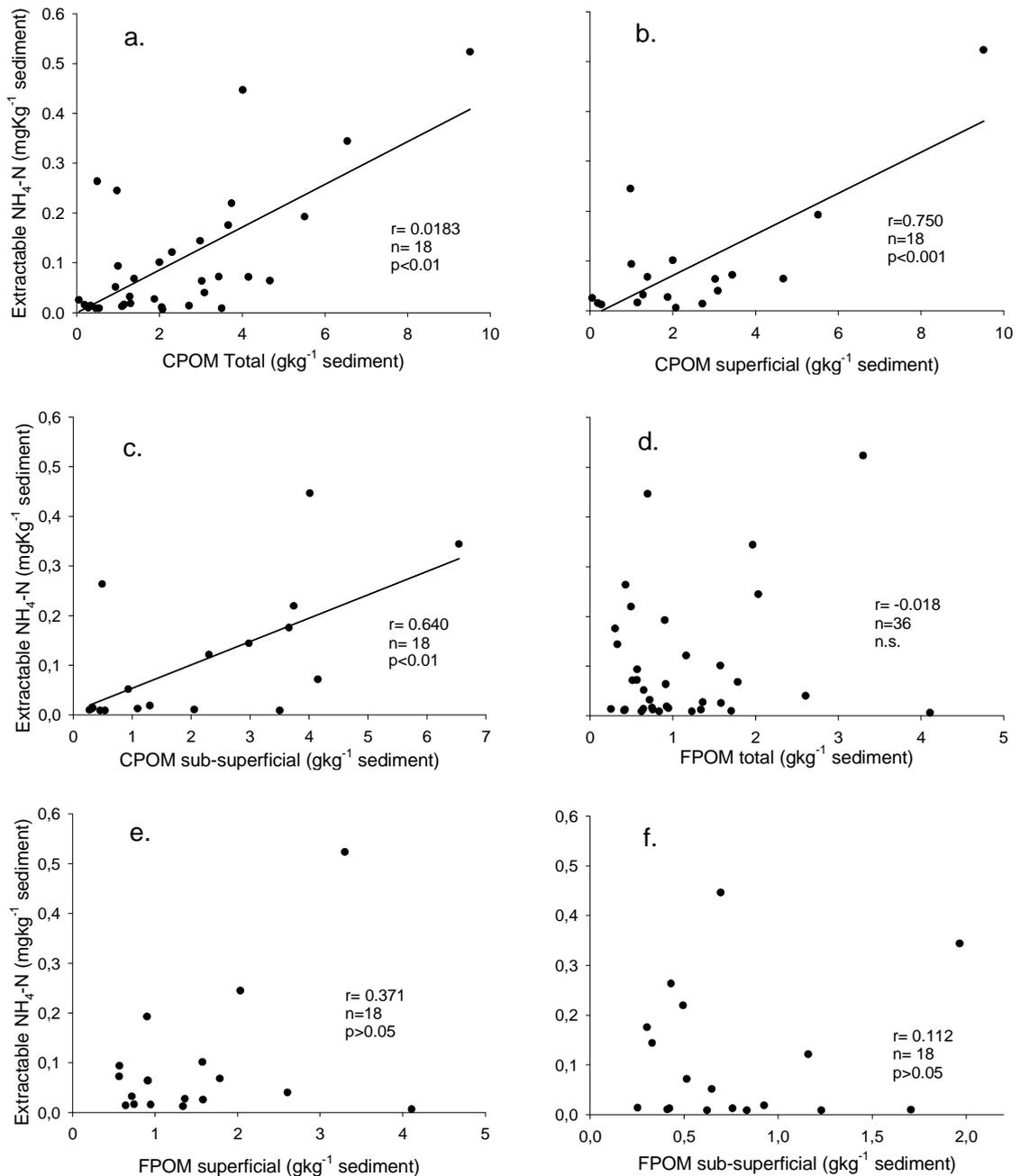


Figure 14. Spearman Rank Correlation between extractable $\text{NH}_4\text{-N}$ and POM contents in sediment. Total CPOM (a), superficial CPOM (b), sub-superficial CPOM (c), total FPOM (d), superficial FPOM (e), sub-superficial FPOM (f). Monthly measurements from March to May 2007. Data from the river Gellenbach

C:N ratio of the leaves exposed to oxic conditions decreased at a faster rate than those exposed to anoxia with differences statistically significant (H test < 0.05, n = 5 for each sampling period) (Fig. 15b). In oxic environment the major C:N changes occurred during the first 30 days, starting from an initial ratio of 37.7 and ending up at 24.4. After the whole period of 90 days of exposure, the C:N ratio had decreased to final value of 19.4. Thus, breakdown rate was 2.7 times faster during the first 30 days of exposition than during the rest of 60 days under oxic conditions (Fig. 15b). Leaves exposed to anoxic conditions reached a final C:N ratio of 31.9 after 90 days of exposure. Similar to the process in oxic environment, the greatest decrease was observed during the first 30 days. Differences between oxic and anoxic exposition began to be significant after the first 15 days of exposition (U-test < 0.05, n = 5 for each variable and sampling period (Fig. 15b).

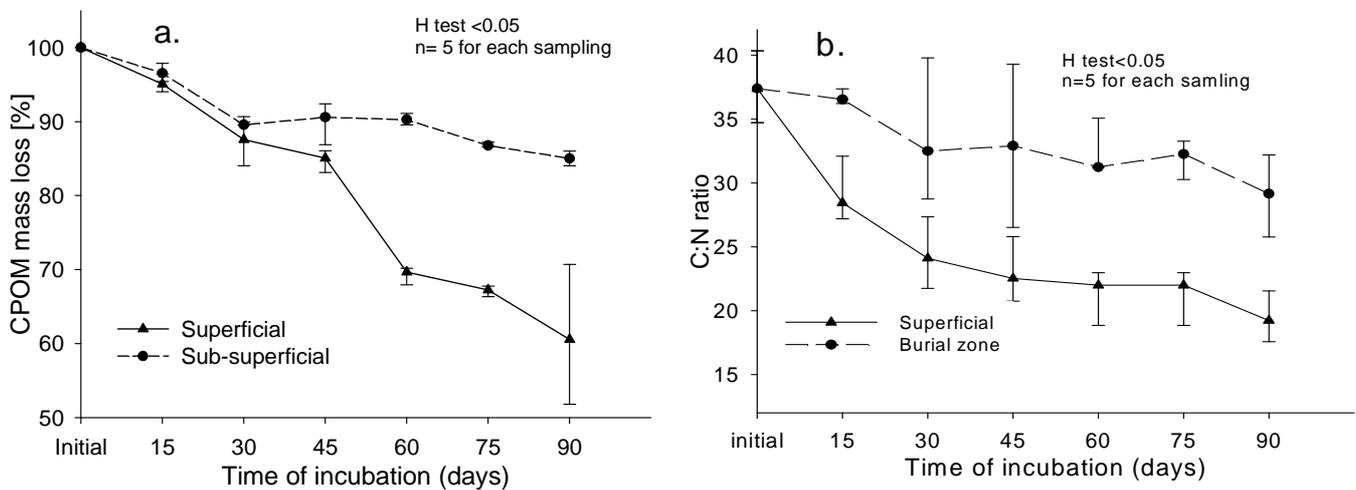


Figure 15. CPOM mass loss (a), and change of C:N ratio (b) of *Fagus sylvatica* leaves during 90 days (from March to May 2007) incubation in superficial (oxic) and sub-superficial (anoxic) sediment layers; n = 5 for each sampling period and each treatment. Data from the river Gellenbach.

4.3.3 Nitrogen processing b CPOM incubated under oxic and anoxic environments

Leaf packs, exposed to surface water and incubated in anoxic interstitial water, in neither case showed nitrification even so mineralization. Instead, assimilation was the only $\text{NH}_4\text{-N}$ process observed in the experiments. During the first 30 days of incubation, assimilation rates did not differ much between leaves exposed to either oxic or anoxic conditions. In both cases, the rate of $\text{NH}_4\text{-N}$ assimilation remained low at 5.20 and 7.49 $\mu\text{g N g}^{-1} \text{d}^{-1}$, respectively (Fig. 16).

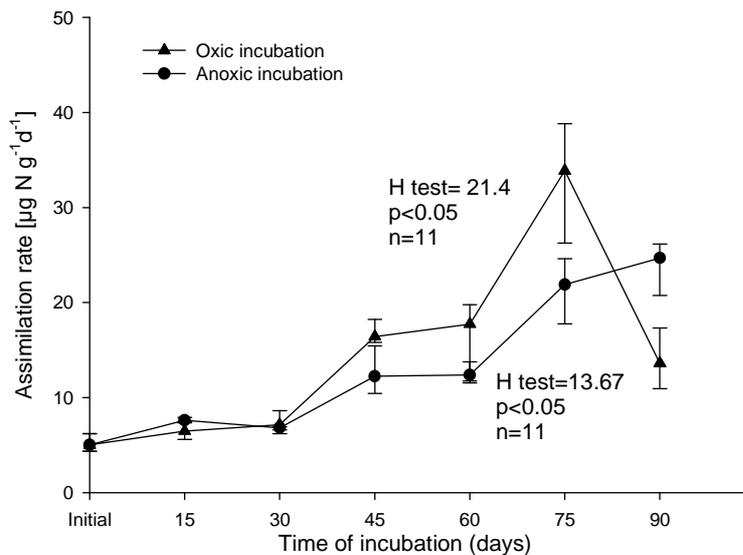


Figure 16. NH₄-N assimilation rate of *Fagus sylvatica* leaves exposed to superficial water (oxic incubation) and sub-superficial in sediment (anoxic incubation) during March, April, and May 2007. During 90 days (from March to May 2007); n = 11 for each sampling period and each treatment. Data from the river Gellenbach.

From day 30 to 75, assimilation rates of the treatments increased stronger in both treatments, however, being clearly higher in leaves exposed to oxygen. There, assimilation rose at double the rate from 16.4 to 32.2 μg N g⁻¹ d⁻¹ but decreased again to 14.25 μg N g⁻¹ d⁻¹ at the day 75. In leaves packs, which were exposed to anoxia, assimilation rates readily increased at a smaller rate reaching a final rate of 23.1 N g⁻¹ d⁻¹, which was the double of final rate of 12.7 N g⁻¹ d⁻¹ in the leaf packs maintained under oxic conditions. Only after the 75 days of exposition significant differences between the treatments could be observed (U-test = 110, p < 0.05, n = 11) (Fig. 16).

NH₄-N assimilation correlated negatively with the C:N ratio and positively with mean temperature during the period of incubation in both superficial and buried sediments (Fig. 17a, b, c, d).

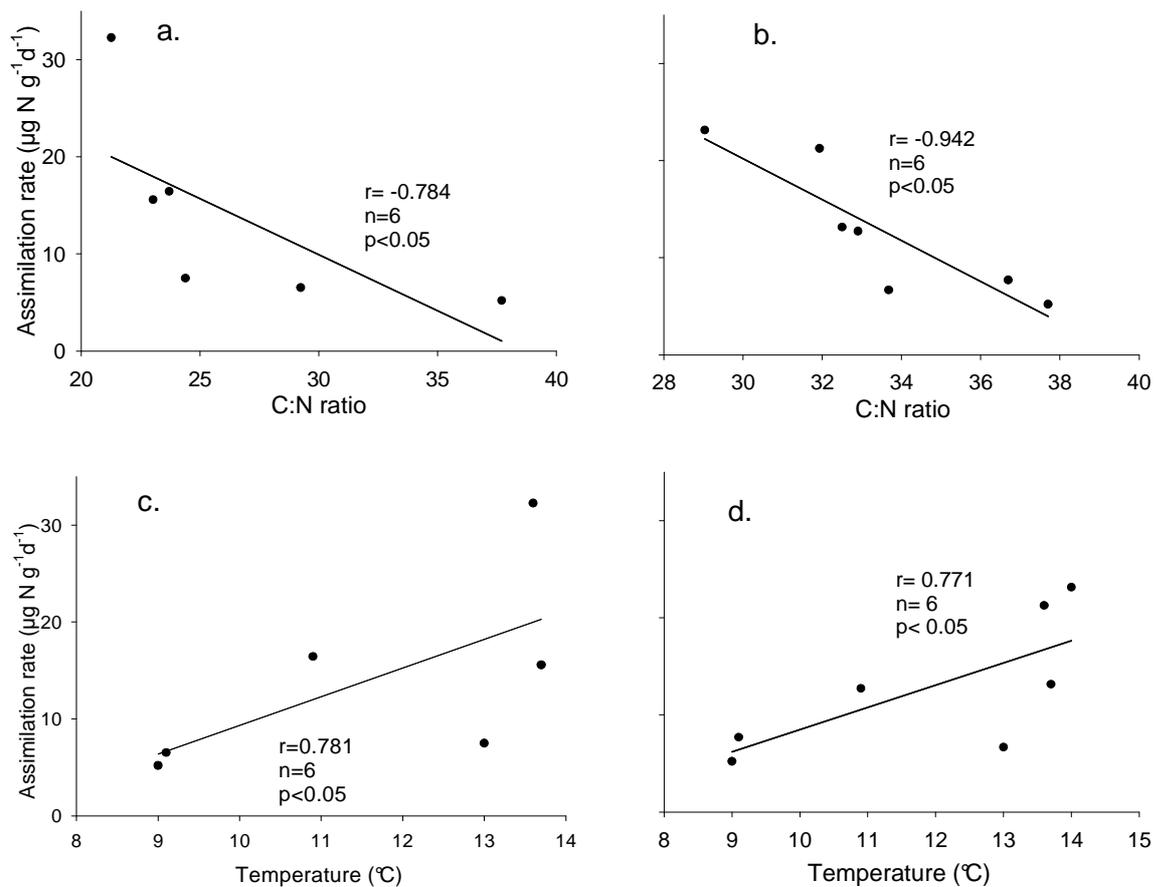


Figure 17. Spearman Rank Correlation between extractable $\text{NH}_4\text{-N}$ and POM contents in sediment. Total CPOM (a), superficial CPOM (b), sub-superficial CPOM (c), total FPOM (d), superficial FPOM (e), sub-superficial FPOM (f). Data from the river Gellenbach.

4.4 Oxygen metabolism in a lowland stream

4.4.1 Physical and chemical variables

Mean annual discharge of the river reach was 158.17 L s^{-1} . The highest daily mean was observed during January (512.94 L s^{-1}), whereas the lowest occurred during July (28.24 L s^{-1}) (Fig. 18a). Conversely, the lowest daily mean water temperature was measured during January ($3.0 \text{ }^{\circ}\text{C}$) and the highest during August (16.2°C). The yearly observed mean temperature in the stream was $9.1 \text{ }^{\circ}\text{C}$ (Fig. 18b).

Photosynthetically active radiation (PAR) showed a clear seasonal variation. The highest PAR was measured during March (mean = 104.61 , maximum = $251.57 \mu\text{mol m}^{-2} \text{ s}^{-1}$) and May (mean = 145.71 , max = $350.975 \mu\text{mol m}^{-2} \text{ s}^{-1}$), whereas in July irradiation was clearly reduced (mean = 9.12 , max = $11.77 \mu\text{mol m}^{-2} \text{ s}^{-1}$). In November a small increase was observed (mean = 17.03 , max = $30.85 \mu\text{mol m}^{-2} \text{ s}^{-1}$) (Fig. 18c).

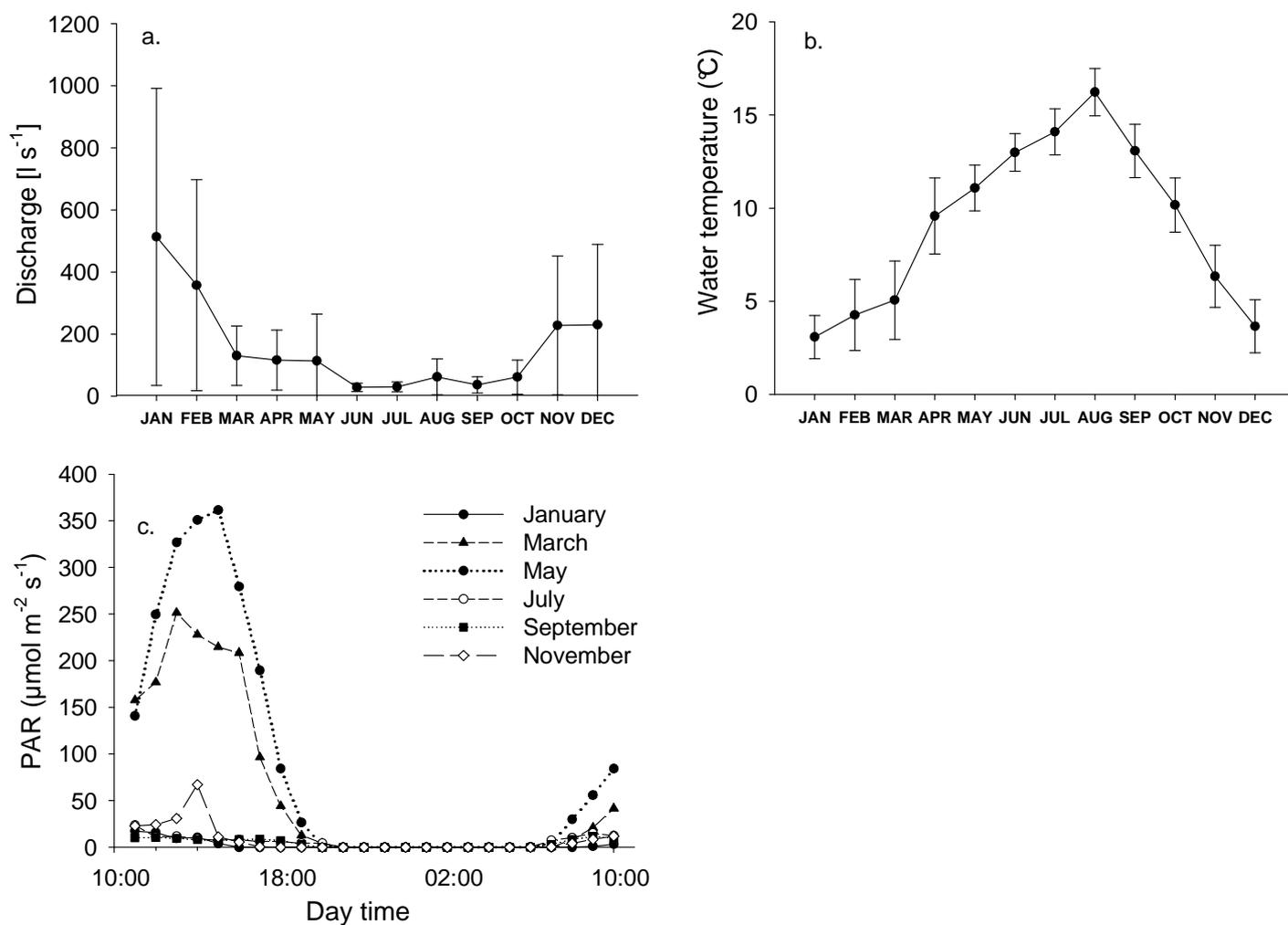


Figure 18. Annual characteristics of discharge (a) and temperature (b). Bimonthly records (c) for one day of the dial characteristics of hourly photosynthetically active radiation (PAR) of the Gellenbach all measured during 2004. Points represent the means and bars the standard deviation. Data from the figure 6 are for one day each one.

The measured mean yearly FPOM content in the sediment was 1.97 g kg⁻¹. The highest amount was observed during September (4.17 g AFDM kg⁻¹), the lowest, however, during January (0.51 g AFDM kg⁻¹) (Fig. 19a). The benthic CPOM constant had a yearly mean of 9.3 g kg⁻¹. The higher content was observed during November (20.75 g AFDM kg⁻¹), whereas the lowest during July (4.18 g AFDM kg⁻¹) (Fig. 19b). Concentration of nitrogen in the NO₃-N and NH₄-N fractions displayed a distinct seasonal variation, reaching the highest concentrations during November and January, whereas the lowest concentrations were observed during July and September. These two variables correlated significantly with the discharge ($r = 0.943$, $p < 0.01$ and $r = 0.886$, $p < 0.05$, respectively). Contrarily, orthophosphate did not show any seasonal variation, nor correlated with any other variable (Tab. 3).

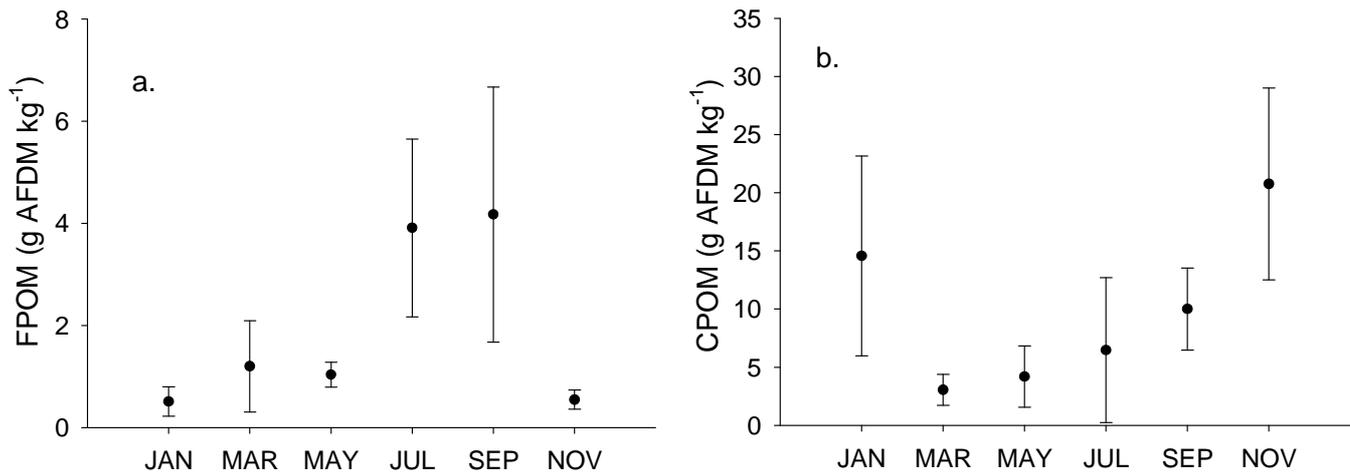


Figure. 19. Benthic particulate organic mater (POM) content in sediments in the Gellenbach, FPOM (a), CPOM (b). Points represent the means and bars the standard deviation; $n = 5$ for each sampling period during 2004.

4.4.2 Stream metabolism

The daily rate of metabolism varied considerably among the seasons. The GPP ranged from 0 to $3.9 \text{ g O}_2 \text{ m}^{-2} \text{ day}^{-1}$, with January having the lowest and May the highest GPP rate (Fig. 20a).

There was substantially lower variation in ER with values ranging from 0.03 during January to $2.1 \text{ g O}_2 \text{ m}^{-2} \text{ day}^{-1}$ during September (Fig 20b). NDM showed negative ratios for all months except during May (Fig. 20c). The same pattern was observed in P/R, which was > 1 (autotrophic characteristics) only during May (Fig. 20d).

Table 3. Main nutrient concentrations in the river Gellenbach during the period of the oxygen dial curves measurements for the determination of the stream metabolism. Punctual bi-monthly measurements from January to November during the year 2004 in the river Gellenbach.

	$\text{NH}_4\text{-N} [\text{mg l}^{-1}]$	$\text{NO}_3\text{-N} [\text{mg l}^{-1}]$	$\text{PO}_4\text{-P} [\text{mg l}^{-1}]$
January	0.42	10.8	0.03
March	0.28	8.0	0.02
May	0.08	6.4	0.02
July	0.05	3.9	0.03
September	0.04	6.6	0.03
November	0.34	11.1	0.02
Average \pm std. dev.	0.20 ± 0.17	7.8 ± 2.8	0.025 ± 0.005

The daily rate of GPP was significantly correlated with daily PAR (Fig 21a). The correlation between GPP and other physical and chemical variables (water temperature, discharge, $\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, and orthophosphate concentrations) was not significant. Multiple regression analysis indicated that 90% of the variation of GPP could be explained by a model that included PAR (Tab. 4).

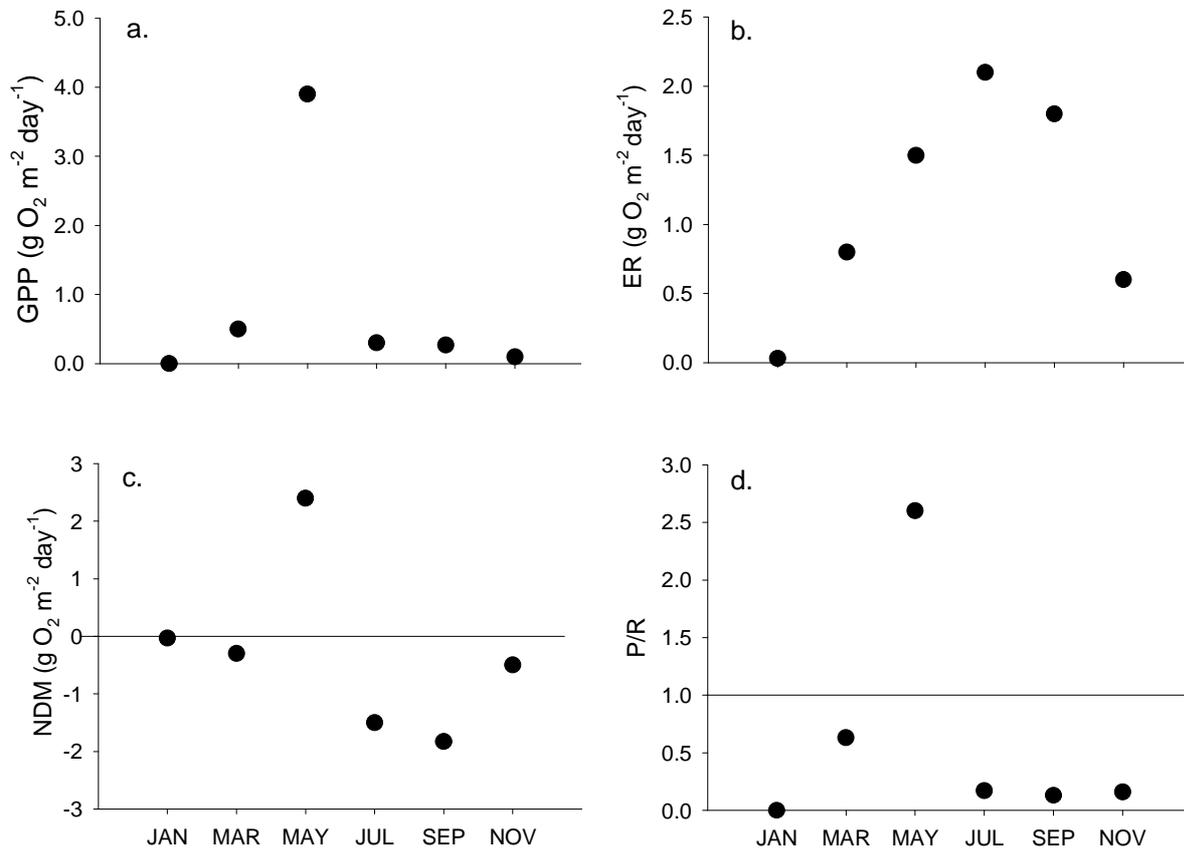


Figure 20. Temporal variation of daily rates of the gross primary production (GPP) (a), ecosystem respiration (ER) (b), net daily metabolism (NDM) (c), and production-respiration ratio (P/R) (d) during 2004 for a 30 m² stream reach in the river Gellenbach.

The daily rate of ER was significantly correlated with water temperature, FPOM content of the sediment, and discharge (Figs. 21b, c, d). Correlations between ER and other physical and chemical variables were not significant.

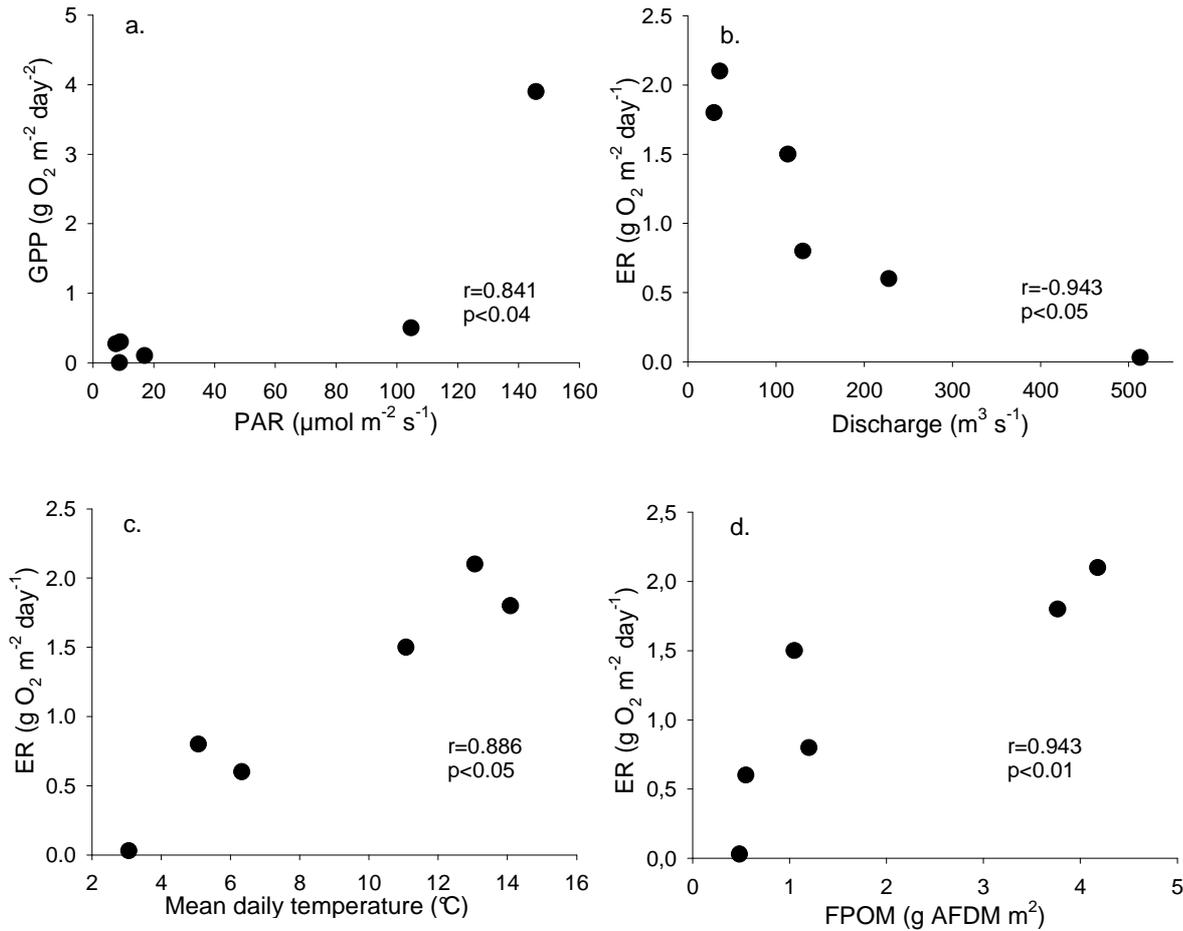


Figure 21. Spearman rank correlations between gross primary productivity (GPP) and photosynthetically active radiation (PAR) (a), ecosystem respiration (ER) and discharge (b), ER and temperature (c), ER and fine particulate organic matter (FPOM) (d) during 2004 for a stream reach of 30 m² in the river Gellenbach.

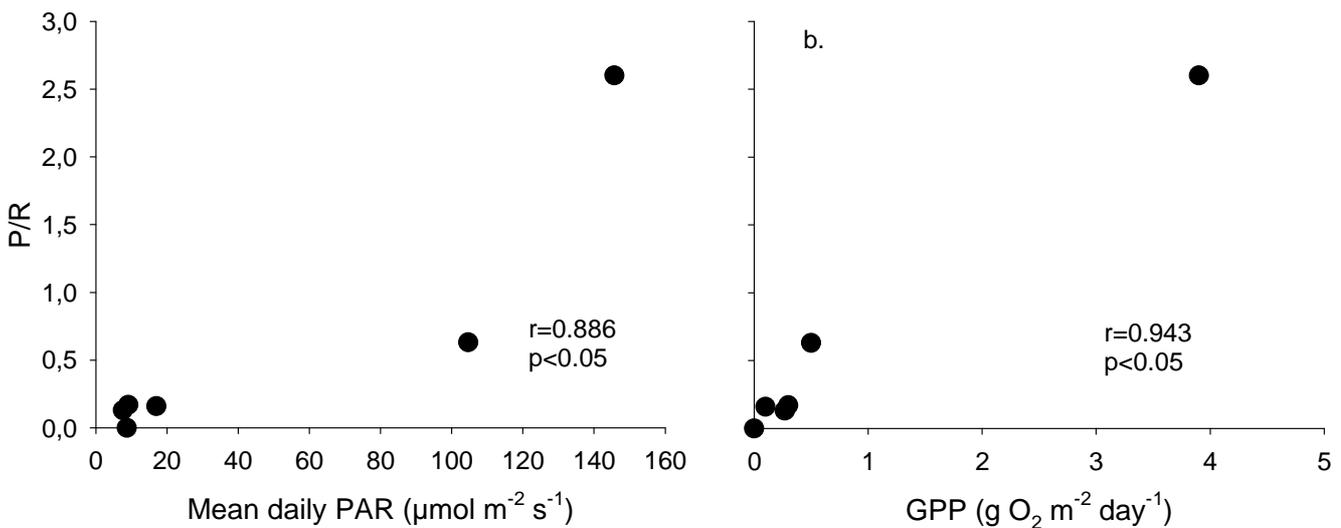


Figure 22. Spearman rank correlations between production/respiration ratio (P/R) and photosynthetically active radiation (PAR) (a), and P/R and gross primary productivity (GPP) (b). Measurements done during 2004 in a stream reach of 30 m² in the river Gellenbach, Germany.

5. Discussion

Temporal variation of NO₃-N and NH₄-N concentrations in stream water

According to the findings of this study, the mean monthly precipitation did not show a correlation with the mean monthly discharge, yet with the monthly potential evapotranspiration (Fig. 4a, b). This characteristic can be explained satisfactorily by the effect of the evapo-transpiration, i.e., the estimated loss of water to the atmosphere. As described by Arheimer et al. (1996) and Burt (1996), when the evaporation rate is lower than the precipitation during winter (in this region great part of the precipitation during winter occurs as rain form and not in snow form), the water table rises near to the soil surface, and forms local swamp patches and drains to the streams. As a consequence, the discharge increases. During summer the opposite effect can be observed, the evapo-transpiration exceeds the precipitation, hence the soils have a water deficit and, consequently soil water does not drain to streams, or only in small extent.

The discharge and inorganic nitrogen concentration in these streams presented the same temporal variation and hence a positive correlation between them (Fig. 7). This relationship is probably a direct consequence of soil washout. During winter, the high level of soil water drains to streams transporting the labile nitrate with. The fact that in this farm region NO₃-N concentration in soil reaches its maximum through soils fertilization in ammonium-rich manure form (despite the crop production and nutrients assimilation) which in contact with oxygen, in the sandy soils, is rapidly nitrified this helps to explain the high nitrate concentrations observed in soils at the end of summer. The minimum concentrations at the end of winter is the result of that soils become water saturated and anoxic conditions, ideal environment for the nitrate-ammonification and denitrification, adding the soils washout these processes contribute to explain the lowest nitrate concentrations observed in soils at the end of winter (Remy 1991, Dorge 1994, Arheimer et al 1996, Collins and Jenkins 1996, Paul 2002, Hefting et al. 2004).

During spring and summer, the nutrient demand from the vegetation, specially from the agricultural crops is the highest of the year, and as a consequence of the water deficit in soils, the transfer or washout of nitrate from soils to streams is minimized. In case of heavy thunderstorms in summer, nitrate NO₃-N can be washed to streams but it is rapidly assimilated by aquatic plants, which proliferate during this season especially in the open canopy sectors (Strand and Weisner 2001, Jarvie et al. 2002, Pott and Remy 2003, Cedergreen and Madsen 2004). Moreover, the loss of nitrate by the denitrification and dissimilatory nitrate reduction under anaerobic conditions can play an important role on the nitrate availability (Lischeid et al. 2004, Storey et al. 2004).

In other regions the results are divergent: In an agricultural region of Argentina no correlation was found between discharge and the nitrate concentration (Feijó et al.

1999), whereas in rivers of Kansas (US) a negative correlation existed between these parameters (Melody and Dodds 2001). The latter finding was attributed to the dilution by discharge of pristine upland waters. Unequally, the EMB and the LMB run from the source to the mouth through a homogeneous agricultural area in which the dynamics of dilution does not change. Generally, $\text{NH}_4\text{-N}$ is characterized by lower mobility in the ground than $\text{NO}_3\text{-N}$, due to its tendency to form associations with clay and organic matter (Wang and Alva 1996 2000). Despite this characteristic, $\text{NH}_4\text{-N}$ followed the same dynamics as $\text{NO}_3\text{-N}$. In this study, Collins and Jenkins (1996) reported that when the ammonium reaches saturation levels in soils, it cannot be retained, and therefore it can be washed out at the same way as the nitrate, this scenario is possible as in winter the soils become water saturated, therefore anoxic condition, which favors the denitrification, but also the nitrate ammonification. In addition, the higher ammonium concentration during winter could be the effect of the exposition of lower and anoxic sediments, where the ammonium normally accumulates and can therefore be a source of nitrogen during the high discharge period

However, to clarify the origin and the relative contributions of fertilizers, waste waters, and autochthonous processing as stated by Sørensen (1978), Ostrom et al. (1988), and Chang et al. (2002) it is necessary to use isotope techniques and to quantify in-stream nitrification and denitrification processes to have a better knowledge of the factors that control the nitrogen concentration in this streams.

Potential nitrification in sediments

Early studies have indicated that *Nitrosomonas europaea* and *Nitrosococcus oceanus* are able to grow at oxygen concentrations as low as 1%. They can survive as viable inactive cells during periods of low substrate concentrations or poor growing conditions. Especially, *N. europaea* is capable of maintaining an anaerobic metabolism using pyruvate as an electron donor and nitrite as an electron acceptor (Carlucci and McNally 1960, Goreau et al. 1980, Abeliovich and Vonshak 1992, Verhagen et al. 1992, Schramm et al. 1996). Dormant nitrifiers (nitrifying bacteria under inactivity as cause of the anoxia) can exponentially increase nitrification rates as response to the exposure to oxygen and $\text{NH}_4\text{-N}$ as suggested by Kemp and Dodds (2001). It has been demonstrated in this study that in sub-superficial sediments with less than 30% silt-clayed content, nitrification rates were as high as observed in superficial sediment (Fig. 7b). Although clay sediments in this stream content a high Ca^+ saturation, the probability for ammonium absorption is rather low. It can probably be a consequence of the FPOM content which is 4 times higher in silt-clayed sediments than in sandy ones (Fig 11b), this FPOM may offer a substrate for the microbial $\text{NH}_4\text{-N}$ assimilation, or consequence of the a-priori reduction condition could favour the $\text{NH}_4\text{-N}$ adsorption to this organic matter. Experiments of Macura and Stotzky (1980) showed that additions of montmorillonite and kaolinite to clay free soils increase the mineralization and nitrification, and at the same way, this phenomenon was observed by

Triska et al. (1993b). However, in this study the clay-silt substrate with high FPOM content acts as a sink for the $\text{NH}_4\text{-N}$.

Despite of the higher $\text{NH}_4\text{-N}$ content in the sub-superficial sediment microcosms, nitrification rates were inferior to the observed in the superficial sediments. Early studies have suggested that DOC at concentrations of 30 mg L^{-1} reduces significantly nitrification rates (Strauss and Lamberty 2000, Strauss et al. 2002). Nevertheless, in our incubations, the maximum DOC concentration was 11.02 mg L^{-1} which could not inhibit to nitrification in the lower sediment layers (Table 1, 2). The pH during the incubation varied always between 7.5 and 8.5 (data not shown) being the optimal pH for the nitrification (Strauss et al. 2002). The temperature was constant through the incubations. The anoxic condition ($\text{O}_2 < 1 \text{ mg L}^{-1}$) of lower sediment layers can limit the nitrifying bacteria populations. However, the oxygen absence does not implicate the non-existence of nitrifiers.

Relationship between CPOM and nitrification

The structural characteristic of the studied stream, such as low slope, low water velocity, sandy substrate, and the absence of oxygen in the sediment deeper than 5 cm could typically favor the observed major differences in $\text{NH}_4\text{-N}$ content between superficial and sub-superficial sediment layers. It suggests that disturbance of superficial and sub-superficial sediments, which occurred during March measurements when discharges reached high levels and extreme variability, lead to a homogenization of the layers (Carling 1996, Paterson and Black 1999). As a consequence, this process could have made dissolved oxygen of the surface water available to sub-superficial sediments and subsequently favors the oxidation of the interstitial $\text{NH}_4\text{-N}$ to $\text{NO}_3\text{-N}$. Contrarily to these processes, in the May indicate the lack of sediment mixing, and consequently oxygen depletion stimulated the accumulation of ammonia in anoxic sub-superficial sediments. Nitrification of this $\text{NH}_4\text{-N}$ took place only in the oxygenated upper sediment layer (Patrick and Redy 1976). This presumption was confirmed by the May measurements when discharges were low, stable, and sub-superficial sediments tended to contain more $\text{NH}_4\text{-N}$ than superficial ones (Fig. 12).

As consequence of leaf litter input from the upstream riparian zone and the stability of runoff, we observed a significant greater CPOM content in sediments during May than in March. The organic matter present in the stream at that period, for the most part, were residuals of the autumn leaf fall that had passed leaching (N release) and immobilization (N uptake) phases during winter and were conditioned for a mineralization phase as described by Berg and Staaf (1981). The results support this assumption by the significantly positive correlation between CPOM and $\text{NH}_4\text{-N}$ content in both superficial and sub-superficial sediments (Fig. 13). Generally, the nutrient content of FPOM is low and thus does not offer an optimal substrate for the mineralization process. Although, the surface/volume ratio favors bacterial colonization and can be an optimal nutrient basis for macroinvertebrates, especially burrowers

and detritus feeders, whose excretes can increase the nitrogen content in sediments (Grimm 1988).

However, as the density of macroinvertebrates in the sandy substrate of our study stream is rather low, the actual situation deviates from the optimal condition for the metabolization of FPOM mentioned before (Sabarth 1999).

The results of this study support the assumption, according to which organic matter decomposition in anoxic sediments is supposed to be slower compared to the oxic ones (Metzler and Smock 1990). Basically, organic particles are deposited and degraded initially in an oxic environment on the sediment surface where aerobic decomposers colonize on more labile substrates, before these are buried into the deeper anoxic sediment layers (Henrichs and Reeburgh 1987, Kristensen 1993). It should be taken into account that the mechanical energy of flowing water and the abrasive effect of sand mobilized by the flow accelerates the fragmentation and mass loss of organic matter (Braioni et al. 2001, Spaenhoff et al. 2007). Despite of that, the faster decrease in the C:N ratio of the CPOM exposed to surface water suggests that nitrogen immobilization was faster in superficial sediments, although this process also occurred in sub-superficial ones and should not be underestimated. For ecosystems driven by allochthonous inputs, the terrestrial vegetation provides large amounts of carbon-rich, N-poor materials for heterotrophic community (Dodds et al. 2004). The stoichiometry of this important energy component, which mostly shows relatively high C:N ratios of > 20 , suggests that N must be acquired by the heterotrophic community of the stream from a different source, in order to be able to grow and exploit carbon compounds. This relation was supported by the negative correlation between the C:N ratio and assimilation rates observed in leaf packs, which were incubated in both superficial and sub-superficial sediment layers (Fig. 17a. b). In addition, these findings correspond to the experimental results by Webster et al. (2000). Consequently, neither mineralization nor nitrification could be observed in the buried leaves packs. The phase of initial nitrogen retention that we observed during the incubations corresponded to the metabolic activity of heterotrophic organisms.

Facultative aerobic and anaerobic bacteria are widespread in stream sediments. They are characterized by their ability to quickly change their metabolism under highly variable oxic or anoxic conditions (Rosswall 1981, Jesse et al. 1983, Stanford et al. 2002). This immediate response of the micro-organisms to shifts from anoxic to oxic environmental conditions was observed in this study during the exposition of the leaf packs (Fig. 16). Sand substrates are constantly perturbed and transported, especially during high discharge events, so that formerly superficial sediments can quickly be buried and enclosed in anoxia, whereas otherwise sub-superficial sediments can abruptly be exposed to an oxygenized environment. Under these conditions, facultative biota presents their ecological advantages. The incubation showed, however, that assimilation rates are potentially higher on litter exposed to aerobic in comparison to anoxic conditions. On the other hand, substrate stability in buried and anoxic sediments can be advantageous, especially when high discharge events occur,

by which biofilms are destroyed through abrasion by suspended particles in water flowing close above the sediment surface. This assumption is evidenced by the smaller assimilation rate, which was observed on litter exposed to surface water after 75 days of exposition, when after a period of stable flow a high discharge event occurred, which affected only the leaf packages exposed to surface water (Fig. 16).

Organic matter accumulated in stream sediments serves as a storage and buffer of nitrogen for the stream, initially through the retention and assimilation of ammonium to its later release and nitrification to nitrate. The lower rate of mass loss of organic matter observed in sub-superficial sediments, compared with superficial ones, clearly indicates that anoxic sediments act as a temporal retention system, which any time can be activated for nitrogen release when exposed again to oxic conditions.

The results of this study suggest that PAR is the only variable that influences the GPP rates. Though, several authors have reported that GPP increases when stream vegetation cover is reduced and PAR intensity increases (Naiman 1983, Bott et al. 1985, Young and Huryn 1999, Roberts et al. 2007). At the same time, a sharp decrease in GPP was observed during July, as a consequence of a 93 % reduction in PAR. The low GPP rates observed despite of the opened canopy conditions during January probably could be considered as a consequence of low PAR intensity during high discharges (Fig. 18a). However, discharge did not explain the variability of GPP in this study. On the other hand, Atkinson et al. (2008) found that primary producers were notably lacking in a sandy lowland stream in all seasons. Then this observation explained by the continuous motion of the sand grains in the uppermost sediment layer, which inhibits growth of photosynthetic algal communities on the streambed surface.

The stream bed studied here was dominated by sediments of 250 μm mean grain size, where sand movements were evident in riffles and in transition zones, although in banks and pools, high algal densities did establish well during May, i.e. during much lower discharges than in November and January (Fig. 17a). Moreover, as a consequence of the small slope (approximately 0.1 %) and slow water velocity, the algal communities can reach high GPP rates (Bott et al. 1985).

Experiments with nutrient additions have demonstrated an increase of biofilm production, when light is not the limiting factor (Carrick and Lowe 1989, Lowe et al. 1986, Guash et al. 2006). Nevertheless, we did not find any significant correlation between nutrient concentration and stream metabolism, despite of the observed strong seasonality in $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ concentrations in the streams. Also, nutrient concentration obviously was not a determinant factor for the production and respiration in the stream studied here because of the lack of correlation of metabolic parameters with inorganic nitrogen in water, and the relatively high inorganic nitrogen concentrations during all seasons. Based on this finding, it can be assumed that higher concentrations of nutrients did not increase stream metabolism (Bushong and Bachmann

1989). Therefore, owing to the environmental properties of the Gellenbach, the seasonal changes of PAR and phenology of vegetation could be understood as the principal factors controlling GPP rates.

Three general patterns of metabolism were evident: (i) relatively high GPP rates, a positive rate of NDM, and $P/R > 1$ when little canopy cover and high PAR occurred, (ii) a relatively high respiration rate, a negative rate of NDM, and $P/R < 1$, when the stream was very shaded and received low PAR, (iii) and very low stream metabolism when high discharge occurred.

Based on our results, the Gellenbach could be classified as a heterotrophic stream, as cause of the predominance of respiration over production during the most of the year (Fig. 20c), though ER rates were relatively low compared with other forested streams (Tab. 4).

The variability of respiration was best explained by the FPOM content of the sediments, as the great amounts were observed during July and September, when simultaneously highest ER rates were measured. Peters et al. (1989) and Yoshimura et al. (2008) reported that, compared to CPOM, the fraction FPOM is of low nutritional quality because of its high lignin and cellulose contents. Metabolism determined based by FPOM the mineralization of glucose using C^{14} (Peters et al. 1987). Results suggest that FPOM potentially supports lower metabolism rates than CPOM. At the same way, Yoshimura et al. (2008) reported smaller respiration rates of FPOM compared to CPOM. Nevertheless, the surface/volume ratio in FPOM allows for a higher bacterial biomass to develop than in CPOM, and, as bacteria can utilize dissolved organic carbon and nutrients directly from water, the nutrient quality of FPOM should not be of vital importance for the bacterial colonization (Stuart et al. 1993, Jones 1995). On the other hand, FPOM plays a crucial role as a food basis and substrate for microbes and invertebrates (Cummins and Klug 1979, Hall 1995). Studies by Naimann (1983) on a forest boreal stream, and Bott et al. (1995) on four temperate streams, highlighted that community respiration is broadly supported by FPOM as a consequence of its ubiquity, large standing stock, and associated microbiota. Deposition and accumulation of FPOM in streams occurred at low runoff levels, a condition that was observed in the Gellenbach during July and September. Against this background, both factors, discharge and, depending thereupon, the FPOM content were considered the main components which control the respiration rates in the Gellenbach (Uehlinger and Naegeli 1998, Atkinson et al. 2008).

It has been reported in several studies that leaching of DOC from CPOM induced an augmentation of bacterial biomass and by this stimulating respiration (Bott et al. 1985, Crocker and Meyer 1987, Pusch and Schwoerbel 1994, Meyer et al. 1998, Crenshaw, et al. 2002, Roberts et al. 2007). Despite of the higher content of CPOM observed in the sediment during November and January, we did not find any increase in the respiration rate. A possible explanation is that this finding coincided with a period of extremely high and variable discharges (flash pulses and spates),

which cause a decline of leaf litter retention and substrate stability (Speaker et al. 1988, Jones and Smock 1991, Uehlinger 2000, Mathooko et al. 2001, Acuña et al. 2004). However, the importance of CPOM, to ecosystem respiration, in relation to FPOM, should not be discarded, as it may have contributed certain portion to the high respiration rates measured in the Gellenbach during July and September.

In various studies, little response of respiration to temperature changes has been reported. Mulholland et al. (2001) did not find any relationship between temperature and ER, probably as cause of the small temperature range analyzed, and differences of organic matter supply and nutrient concentration between the analyzed streams overlapping the effect of temperature. Bott et al. (1985) found temperature as a good predictor of the respiration, although only 22 % of ER variability is explained. According to Sinsabaugh (1997) temperature accounted only 30 % of the variation of ER, whereas he mentioned the latitudinal and altitudinal shift in vegetation type as a possible cause. Likewise, Uehlinger et al. (2000) found a positive effect of temperature on ER, though explaining only 22% of variation of ER. Conversely, Acuña et al. (2004) found in a Mediterranean stream higher respiration rates during winter than summer, despite of the significantly lower temperature observed (5-15°C) compared to summer (20-25 °C). Here, enough discharge to remove organic matter from sediments was discussed as the main cause of the lower respiration rates observed during summer. In this major context, our results conform to the finding of Acuña et al (2004). We observed a negative correlation between discharge and ER, although in the Gellenbach, both lowest discharges and highest water temperatures occurred during summer. Thus, the combination of high temperature and low and stable discharge conditions were the principal factors influencing ER during summer, despite of the considerable lower OM content compared to autumn and the beginning of winter.

Ecosystem respiration dominated gross primary production most of the year. Only during May, when high PAR was observed, the stream showed autotrophic characteristics ($P/R > 1$). Based on these findings, a high potential for a strong onset of GPP can be expected when light is available to the stream bed. The importance of POM can be disregarded for both ER and GPP, because the bulk of CPOM occurred during a period of high discharges and low temperature. Releases of DOC by primary production are assumed to be an additional carbon source and to fuel heterotrophic respiration during March and May, which in combination with respiration of primary producers make up to the whole stream respiration.

Table 4. Comparison of metabolism rates in different low to medium sized streams taken from literature.

Author	Method	Stream	GPP [g O ₂ m ⁻² d ⁻¹]	ER [g O ₂ m ⁻² d ⁻¹]	P/R
This study	Single-station	Gellenbach	0.85 ± 1.5	1.13 ± 0.79	0.61 ± 0.99
Chessman 1985	Single-station	La Trobe river	0.15 - 1.90	3.0 - 4.6	
Edwards and Meyer 1987	Single-station	Oegeche river	0.5 - 15.0	3.7 - 11.5	0.25
Mollá et al. 1996	Single-station	Montesina Stream	2.4 ± 2.1	2.2 ± 1.6	0.59 ± 0.69
Uehlinger and Naegeli 1998	Single-station	river Necker	2.5	3.5	0.73
Young and Huryn 1999	Two-station	Three O'Clock	3.07 ± 0.9	2.7 ± 0.9	1.5 ± 0.2
Kaenel et al. 2000	Single-station	Mühlbach	12.5 ± 4.5	8.9 ± 6.84	1.07 ± 0.08
Fellows et al. 2001	Two-station	Gallina Creek	1.7	0.1	
Uehlinger et al. 2002	Single-station	Hassayampa river	0.3 ± 0.1	1.65 ± 0.13	0.17 ± 0.05
Uehlinger 2006	Single-station	river Thur	5.0 ± 0.6	6.2 ± 1.4	0.53 - 1.0

The persistent negative NEP values and relatively low P/R registered during the study emphasize the importance of allochthonous sources of carbon to the stream metabolism. This fact is in concordance with the river continuum concept (Vannote et al. 1980) as the expected heterotrophy for low order streams has been met. However, production and respiration rates in this study were relatively low compared with other studies (Table 5).

The generality of our results is relatively limited, because we studied only one stream, and measurements were made each two months during one year. Consequently, the amount of data did not allow for robust statistical tests. However, to date there have been only very few studies that have focused on stream metabolism in sandy lowland streams through all seasons.

6. Synthesis

This study contributes to the understanding of the oxygen and inorganic nitrogen metabolism in Central European lowland streams. The main topic is the temporal variation of the inorganic nitrogen concentration in stream water, its processing involving sediments, organic matter (nitrification and assimilation) and the influence on the oxygen production and consumption. Nutrient and oxygen metabolism studies are scarce in Central European lowland streams although in other regions of the world this topic has been studied extensively. Here the importance of this contribution is that it provides information from the basin, reach to sediment micro-scale, not only increasing the background knowledge for the limnological science but also opening a source of discussion about this topic.

The following sections resume the principal questions and conclusions of this research discussed and synthesized in the context of the current stage of knowledge.

What is the mechanism to explain the dissolved inorganic nitrogen in flow water in this case study?

The sole reason that the high evaporation does concentrate the nutrients in the stream, and that precipitation dilutes them, are for this stream by far not a sufficient explanation for the temporal variation of the inorganic nitrogen concentration in these streams. Moreover, it must be taken account that the information obtained and analyzed in this study does not offer a direct response for this question, but gives some sights for the explanation of the causes. Here, the geology, land use and weather play the key roles on the observed trend.

From the measured physical and chemical variables, discharge and potential evapotranspiration covariate with high probability with ammonium and nitrate concentration in these streams, therefore they are the initial trace to begin to elucidate the explanation (Fig. 23).

The literature presents different points of view concerning the relationship discharge-dissolved inorganic nitrogen, where discharge does not present any correlation with nitrogen concentration, or can have a positive or negative relationship. Negative relationship is as due to dilution by upland pristine stream sectors diluting the downstream reaches affected by agricultural activities (Feijó et al. 1999, Melody and Dodds 2001, Mourad and Perk 2009).

The results of this study are in concordance with the conclusions done by Mourad and Perk (2009), meaning that exist connectivity between the stream water, the surrounding groundwater and the land use explaining this kind of relationship.

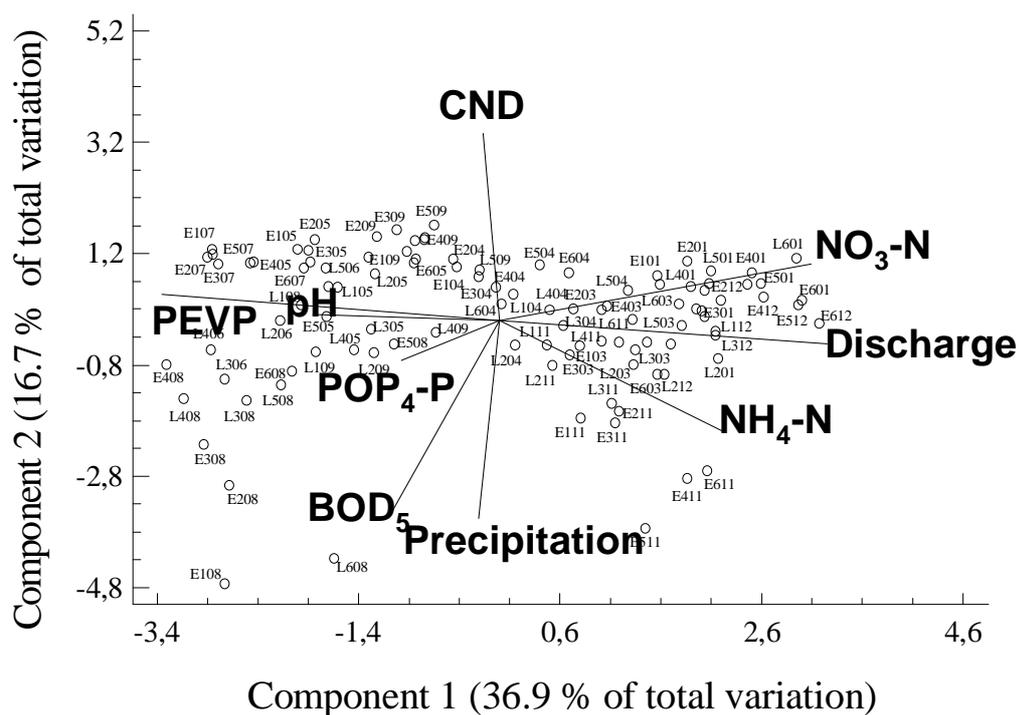
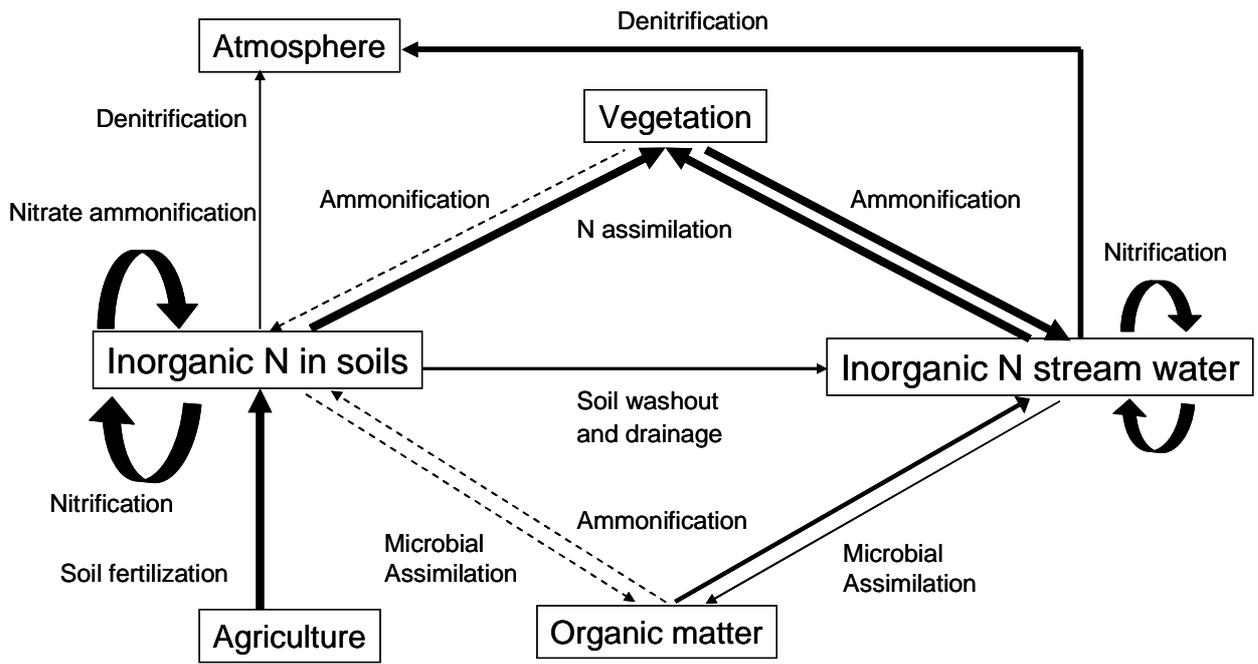


Figure 23. PCA of the physical and chemical variables related to climatic variables from monthly samples along the rivers Eltingmuehlenbach (E) and Ladberger Muehlenbach (L), 01 to 12 denotes the months of the year (from 01 to 12 representing the month of the year). CND: conductivity, PEVP: Potential evapo-transpiration.

Basically, soils should have an excess of nutrients to be exported, which is the case in this basin where the nutrient poor soils have been conditioned for crop practices. Secondly a relatively superficial clay layer (“Emschermergel”) isolates the superficial groundwater from the profound ones, resulting in a fast superficial groundwater flow from soils to streams, finally the temporal variation of water deficit and excess in soils determines the groundwater path to the streams and lastly discharge display a hydrological connectivity, which determines the temporal hot spot for the inorganic nitrogen transfer from soils to streams (Creed et al. 1996, Sickman et al. 2003, Burns 2005, Rozemeijer and Broers 2007, Mourand and Perk 2009).

The temporal variation of the inorganic nitrogen concentration in these streams is described in figure 6, where the relationship between agricultural practices, climatic regime, phenology of the vegetation, and in-stream and soils nitrogen processes determine the inorganic nitrogen concentration in the streams and can be resumed in figure 24. In summer the inorganic nitrogen assimilation by the vegetation and crops are high, soils are not water saturated so that oxygen can reach the soils pore and promotes the nitrification, which can be again reduced to nitrate. In stream water, low inorganic concentrations are expected by the high demand for the biota, and low lateral inputs from the agricultural zone.

a. Summer



b. Winter

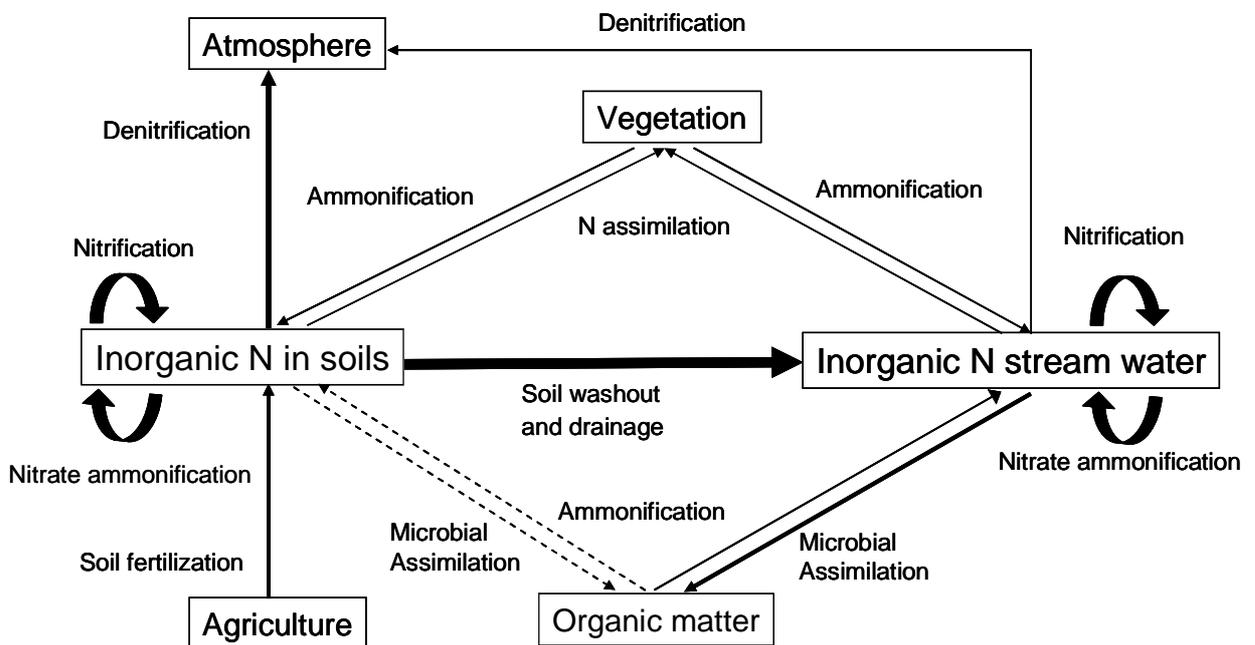


Figure 24. Schematic representation of the hydrological connectivity for summer (a), and winter (b), in relationship and the inorganic nitrogen concentration in the study streams.

During winter, biological processes are reduced in streams and soils, the lack of soil vegetation and reduced demand for the inorganic nitrogen allows the washout of inorganic nitrogen from soils to stream, which occurs when the higher organic matter in form of CPOM occurs, stimulating assimilatory processes (Fig. 24).

Which is the importance of the concentration of inorganic nitrogen for the oxygen production and respiration for this stream?

Although dissolved nutrients stimulate the respiration and production in stream ecosystems (Carrick and Lowe 1989, Lowe et al. 1986, Guash et al. 2006), and based on our results, due to the lack of correlation between respiration and production rates with dissolved ammonium and nitrate, it appears that nitrogen does not have any influence on these two processes. However, the lowest observed $\text{NH}_4\text{-N}$ and nitrate concentrations observed during summer are sufficient to be non limiting factors for the stream productivity. Paradoxically, in the Gellenbach occurs the same process as in many other temperate streams: during winter abundant CPOM inputs from autumn accumulating in stream sediments, while at the same time, inorganic nitrogen loads are the highest of the year, theoretically leading to high respiration rates. But low temperatures, which reduce the potential for the production, and the high discharge, that reduces the substrate stability for the microbial colonisation, are the real factors that limit the respiration during winter. For the light dependent production, the light intensity plays the principal role. As the maximal light inputs occur during spring, with the increase of the temperature and reduction of discharge, optimal conditions for the production are present. Consequently, other variables such as the light intensity, temperature, discharge, and to some extent the FPOM in sediment are playing the determinant factor for the production and respiration for this stream and not the nutrient availability.

Which role plays the sediment for inorganic nitrogen transformation to the stream?

Stream sediments belong to the hyporheic zone, where important nitrogen transformation in streams take place. It is the transition from superficial water to groundwater, and important redox gradients are present, which determine the reduced or oxidized stages of inorganic nitrogen and the processes which take place in this compartment (Triska et al. 1993a, Triska et al. 1989, Jones et al. 1995, Hedin et al. 1998, Dahm et al. 1998, Baker and Vervier 2004, Clivert et al. 2009). Sand sediments for its compactness does not permit the oxygen penetration into deep sediment layers, as we found that in the first 3 cm the oxygen average is 0.25 mg L^{-1} , at 6 cm 0.001 mg L^{-1} , and at 9 cm of deep oxygen is not detectable anymore.

Under these conditions, in anoxic sediment layers nitrification is inhibited and ammonium tends to accumulate. For that reason, anoxic sediments can be considered as the ammonium reservoir for the stream (Triska et al. 1993b).

The continuous sand removal by the flow, especially during high discharge events, exposes sub-superficial and anoxic sediments to oxic conditions, whereas at the same time superficial sediments layers are buried into anoxic ones. In this way, ammonium exposed to surface waters through contact with oxygen is quickly transformed to nitrate. However, in this stream substrate type plays an important role in the ammonium availability for the nitrification. Silt sediments with high FPOM content, and coarse organic matter (CPOM) at the initial decomposition phase out-compete the nitrificant bacteria for the dissolved ammonium and therefore the nitrification is inhibited competitively

Despite of the inhibition of the nitrification, it is evident that the exposition of large amounts of sub-superficial ammonium rich sediments and consequent nitrification can be an important variable which helps to explain the high nitrate and ammonium concentrations observed during the high discharge events in winter.

Is the organic matter a nutrient source for the stream metabolism?

CPOM is the principal energy source for the heterotrophic production, nutrient retention and bank stability (Decamps 1997). In the streams studied here the bulk of organic matter is described in the figure 25, where input occurs during autumn and

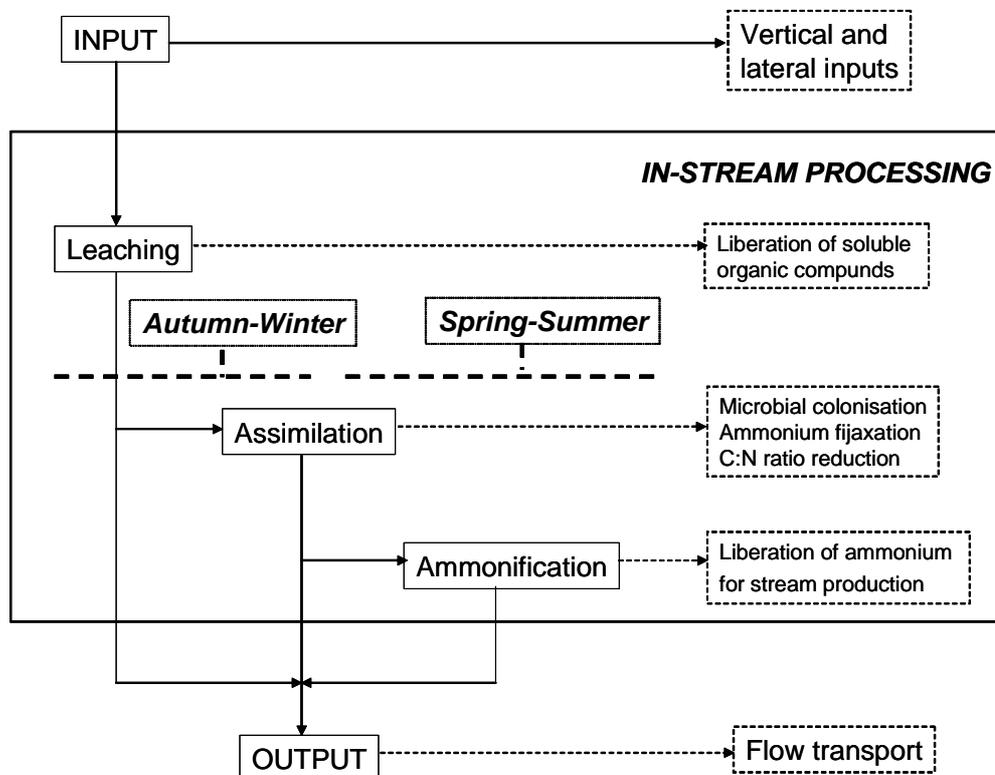


Figure 25. Schematic representation of the organic matter processing in temperate lowland streams.

the beginning of winter, when low temperatures and high discharges are typical, as well as the high inorganic nitrogen concentrations (Fig. 3, 5). The organic matter accumulated in stable stream zones such as pools or was trapped in debris dams is used as substrate by the microbial community as described by Allan (1995) for the organic matter decomposition and subsequent nitrogen assimilation and posterior release to stream and sediments through the ammonification process (Fig 24).

Fagus sylvatica leaves are the major CPOM inputs into this stream. Shortly after falling from the trees, C:N ratio are relatively high (around 34.8%), which configures as not a suitable substrate for a fast decay (Taylor et al. 1989, Hendriksen 1990). This disadvantage for the bacterial community is, at the same time, an advantage for the stream nutrient budget as the slow decomposition rates does mean a late release of nitrogen through the ammonification process, being so the ammonium available for the stream production during spring and summer, when the lowest available nitrogen concentration as inputs from agricultural fields occurs.

7. Summary

In a large part of the lowland region streams from Central Europe especially Germany, Belgium and Holland have lost their natural conditions, as cause of the landscape transformation for the settlement and development of human communities. In this region, rivers flow over a sand dominated soil, characterized by low nutrient contents which in some places has been eutrophicated through excessive soil fertilization during the last 30 years, a high permeability to the surrounding aquifer. In this scenario, loads mainly of nitrogen and phosphorus compounds originating from agriculture activity cause a considerable effect on the in-stream nutrient balance and metabolic processes of the biological community.

In order to understand the manifold physical and physiological variables that control the nitrogen content in water and stream sediments, field and laboratory experiments on the temporal variation of in-stream concentration of inorganic nitrogen, oxygen production and respiration, nitrification, and organic matter decomposition were carried out. The study mainly focused on one stream, the Gellenbach. Two more streams, the Eltingmuehlenbach and the Ladberger Muehlenbach, were included for the investigation of the temporal variation of inorganic nitrogen. All streams belong to the Ems river basin and are similar in terms of climate, geology, and human intervention.

A common and specific feature of the studied streams was the pronounced temporal variation of the nitrate and ammonium load, showing higher concentrations during winter and low concentrations during summer. A significantly positive correlation with discharge was observed, which, on the one hand, suggested the influence of the surrounding groundwater, contaminated with considerable amounts of inorganic nitrogen. On the other hand, the greater biological demand for nutrients during

spring and summer, compared to autumn and winter, to some extent, could also have influenced this pattern.

Sandy sediments of the streams are relatively unstable and constantly eroded by the flow, which, as observed here, resulted in massive displacements during spates. This could drive to the exposition to surface waters of sub-superficial sediments layers, while superficial ones are buried into sub-superficial zones. Ammonium tended to accumulate under anoxic conditions, but as soon as brought to the oxic environment, nitrification increased to high rates. However, in the presence of CPOM with a high C:N ratio, nitrifying bacteria were competitively inhibited by the heterotrophic assimilatory activity. Likewise, nitrification was negatively affected by the presence of silt-clay sediments with high FPOM content. However, as sand is still the most abundant substrate in these streams, highest nitrification rates occurred when ammonium-rich sediments is exposed to oxygen-rich surface waters. Against this background, the elevated nitrate concentrations during high discharge events and the role of the massive substrate removal could be plausibly explained.

High quantities of CPOM were observed in autumn and beginning of winter. During this period, the nutrient quality of CPOM was enriched through microbial colonization and the assimilation of inorganic nitrogen. Although, more than 90 % of these CPOM inputs were retained in deep sediment layers after winter, mineralization of the remaining organic matter fueled the production of the streams. The major part of the organic matter, which had been buried in deeper anoxic sediments, decomposed at slower rates, compared to that exposed to oxic conditions. Specifically the retained CPOM was identified as a reservoir, which, under certain circumstances, was disclosed and utilized for spring and summer production, when CPOM inputs from the riparian vegetation had become a limiting factor.

Paradoxically, the inorganic nitrogen concentration seemed not to affect the stream metabolic activity, as no correlation was found with production and respiration. Apparently, base concentrations were already enough to sustain relatively high production and respiration rates. Primary production, to a great extent, was limited by the intensity of photosynthetically active radiation, while respiration was negatively controlled by discharge, but enhanced by temperature and FPOM concentration. During winter, the in-stream metabolism was extremely low as a consequence of the high discharges, which reduced the stability of deposited organic matter. Under this level of disturbance and partially due to the low maximum water temperature of 3.7 °C, microbial colonization and activity was limited. Taking into account the production-respiration balance, the stream showed autotrophic characteristics during spring and a dominance of heterotrophic processes during the rest of the year.

In conclusion, discharge was the principal variable, which determined the metabolic processes in the studied sandy lowland streams. Being induced by the rising groundwater table during high runoff, the stream water received the inputs of inor-

organic nitrogen from the surrounding land. It further controlled the stability of organic substrates for the activity of microorganism and the relocation of the sandy sediments, which permitted to conserve CPOM packs buried them in deep sediment layers for the posterior oxic decomposition, and the exposition of ammonium for the following and rapid nitrification.

The present study could only cover a small aspect of the wide and complex topic of stream metabolism, but the results shall be a useful contribution to the still vague knowledge of the sandy lowland streams of Central Europe.

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