Gaseous nitrogen and carbon fluxes in riparian alder stands

Ülo Mander¹, Krista Lõhmus¹, Sille Teiter¹, Veiko Uri² and Jürgen Augustin³

¹) Department of Geography, University of Tartu, 46 Vanemuise St., Tartu 51014, Estonia
²) Institute of Silviculture, Estonian University of Life Sciences, 5 Kreutzwaldi St., Tartu 51014, Estonia
³) Institute of Landscape Matter Dynamics, Leibniz Centre for Agricultural Landscape and Land Use Research (ZALF), D-15374 Müncheberg, Germany

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Riparian buffer zones are considered to be important elements of agricultural watersheds, in that they control nutrient and carbon fluxes. Although the water purification effect of riparian ecosystems has been well studied, there is little knowledge of their internal cycling, especially in the area of gaseous emissions. We measured fluxes of nitrous oxide (N₂O), dinitrogen (N₂), methane (CH₄), and carbon dioxide (CO₂) in riparian grey alder stands in southern Estonia and black alder stands in Brandenburg, Germany. Dinitrogen emission was the most important component in N retention in the studied riparian grey alder forests. In 2001–2003, the median values of N₂ emission in the grey alder sites significantly exceeded the N₂ emission rates in the black alder sites, varying from 700 to 1200 and from 360 to 400 kg N₂-N ha⁻¹ year⁻¹, respectively. In contrast, the median values of N₂O flux were higher in the black alder sites than in the grey alder sites, i.e., 0.9–2.6 and 0.2–0.7 kg N₂O-N ha⁻¹ year⁻¹, respectively. The black alder sites acted as a sink for CH₄, whereas the grey alder sites emitted a small amount of CH₄. The CO₂-C flux was higher in the black alder stands. The estimated N₂-N emission in the grey alder stands for 1994–1995 was 51.2, whereas N₂O-N emission was 0.5 kg N ha⁻¹ year⁻¹. The significant increase in N₂ emission from 1994–1995 to 2001–2003 can be related to changes in microbial activity during the succession of the pioneer grey alder stand into a more stable mixed forest community. Due to CO₂ fluxes and N₂O fluxes from sites with altered water regime, the estimated GWP of the studied riparian alder stands was relatively high. Further investigations should concentrate on the factors that regulate rates of N₂O and N₂ emission from riparian buffer zones.

Introduction

In their role as the interface between terrestrial and aquatic components of the landscape, riparian buffer zones are important ecotechnological measures to control water quality in agricultural catchments (Kuusemets and Mander 1999). Although the water purification effect of riparian ecosystems has been thoroughly studied (Haycock and Pinay 1993, Vought et al. 1994, Mander et al. 1995, 1997a), little is known about their internal cycling (Lowrance et al. 1983, Peterjohn and Correll 1984), especially concerning gaseous emissions (Groffman et al. 1991,
Teiter and Mander 2005). Moreover, some studies have shown that water purification efficiency can be less favorable in riparian zones, which function as hotspots of greenhouse gas emissions with high global warming potential (GWP; Groffman et al. 2000).

Alders are known as a host species of symbiotic dinitrogen (N$_2$) fixing bacteria (actinobacteria) from the *Frankia* group (Ryttter et al. 1989). Due to high rates of N$_2$ fixation, some authors have seen alder forests as sources of water body pollution with excess nitrogen (N) (Binkley et al. 1992). Several other studies consider riparian alder stands to be effective N removal ecosystems (Mander et al. 1995, 1997a, 1997b). This contradiction is mainly due to the positioning of alder stands in the landscape: in riparian zones the excess N is mainly denitrified, whereas in more aerated conditions of higher altitude locations (see Binkley et al. 1992) leaching takes place. Denitrification, which is generally referred to as the microbial reduction of nitrate-N (NO$_3$-N) to nitrite-N (NO$_2$-N) and further to the gaseous forms nitric oxide (NO), (nitrous oxide) N$_2$O and N$_2$ (Knowles 1982), has been found in numerous studies to be a significant process in N removal in riparian buffer zones (Groffman et al. 1991, Ambus and Christensen, 1993, Hanson et al. 1994, Weller et al. 1994, Gold et al. 1998, Hefting and Klein 1998, Groffman et al. 2000). In the majority of these studies, N$_2$O fluxes have been measured, while only a few studies pay attention to N$_2$ emission (Watts and Seitzinger 2000, Butterbach-Bahl et al. 2002).

Nitrous oxide, as one of the greenhouse gases, is increasing in the atmosphere at a rate of about 0.3% year$^{-1}$ (Mosier 1998). It has an atmospheric lifetime of about 120 years, a GWP of 296 relative to carbon dioxide (CO$_2$), over a 100-year time horizon, and is responsible for about 5% of the anticipated warming (IPCC 2001). Riparian zones have the potential to be hotspots of N$_2$O production in the landscape (Groffman et al. 2000). Likewise, riparian wetlands and wet riparian forests can be sources of methane (CH$_4$) (Jones and Mulholland 1998, Rush and Rennenberg 1998), which is another greenhouse gas that is increasing in the atmosphere at a rate of about 0.8% year$^{-1}$ (Mosier 1998). Methane in the atmosphere has a lifetime of 8.4 years. On a 100-year time horizon, CH$_4$ has a global warming potential of 23 relative to CO$_2$, and is responsible for about 20% of anticipated warming (IPCC 2001).

Both denitrification and CH$_4$ formation depend on the oxygen status of the soil or sediment. As a result, the spatial and temporal variability of fluxes of both N$_2$O (Robertson and Tiedje 1984, Ambus and Christensen 1993, Augustin et al. 1998, Gold et al. 1998, Jacinthe et al. 1998) and CH$_4$ (Saarnio et al. 1997, Willison et al. 1998) are high. Denitrification rates in soils are mainly influenced by carbon availability, NO$_3$ availability, temperature and pH (Nõmmik 1956, Knowles 1982). Methane is produced in anoxic soils and sediments, while well-drained soils act as a sink for atmospheric CH$_4$ due to CH$_4$ oxidation, through either ammonia oxidizers or methanotrophs (Hanson et al. 1993).

Several studies consider CO$_2$ emissions and sequestration in riparian wetlands (Mitsch and Gosselink 1993) and buffer zones (Brumme et al. 1999, Gulledge and Schimel 2000, Tufekcioglu et al. 2001, Larmola et al. 2003, Scott et al. 2004, Teiter and Mander 2005, von Arnold et al. 2005) Depending on meteorological and hydrological conditions, riparian ecosystems, especially wetlands, can be either sources or sinks of carbon (C) (Gulledge and Schimel 2000).

The main objectives of this research were to quantify and compare N$_2$O, N$_2$, CH$_4$ and CO$_2$ emission rates in two different riparian alder forests: grey alder, *Alnus incana*, and black alder, *A. glutinosa*, and to estimate the global warming potential (GWP) of the analyzed greenhouse gases. Based on our measurements and data from an earlier study, we further estimate the role of N$_2$O and N$_2$ emission in the nitrogen budget of the grey alder stand.

**Material and methods**

**Study sites**

The Porijõgi study area represented a grey alder stand. It is situated in the moraine plain of southeastern Estonia (Tartu County, Sirvaku; 58°13′N, 26°47′E), in the riparian zone of a small river, the Porijõgi, which flows in a primeval valley where
agricultural activities ceased in 1992. The landscape study transect in this valley crosses several plant communities: an abandoned field (last cultivated in 1992) on Planosols and Podzoluvisols; an abandoned cultivated grassland (last mown in 1993) on Colluvial Podzoluvisol (dominated by Dactylis glomerata and Alopecurus pratensis); an 11 m wide wet grassland on Gleysol (two parallel communities, one dominated by Filipendula ulmaria, another by Aegopodium podagraria) and a 20-m wide grey alder stand on Mollic Gleysol. In the grey alder stand, 3 sites: Edge, Wet and Dry were chosen for gas and soil analyses. The main soil characteristics of the Porijõgi study area are presented in Table 1. For a more detailed description see Kuusemets et al. (2001).

The Gumnitz study area is located in Müncheberg (52°50´N, 14°14´E) in Brandenburg State in northeastern Germany. Vegetation cover at this site represented a black alder forest on Gleysols and Histosols, more specifically a Carici acutiformis–Alnetum community, in which Carex acutiformis and Carex canescens dominated. The Gumnitz A site represented a riparian alder stand with lowered water table due to drainage, whereas the Gumnitz B has an unchanged water regime. The main soil characteristics of the two riparian study sites Gumnitz A and Gumnitz B are presented in Table 1.

Gas sampling and analyses

For the measurement of N₂O, N₂, CH₄ and CO₂ emissions, the closed chamber method (Hutchinson and Livingston 1993) and the He-O method (Butterbach-Bahl et al. 1997, Scholefield et al. 1997, Mander et al. 2003) were used. The latter allows measurement of N₂ fluxes. Five gas samplers were installed: one at each of the 3 different sites (Edge, Wet and Dry) in the Porijõgi riparian buffer zone, and on the stream bank (A) and the alder stand edge (B) in Gumnitz. The samplers were closed chambers with a cover made of PVC, height 50 cm, Ø 50 cm, volume 65 l, sealed with a water-filled ring on the soil surface, painted white to avoid heating during application.

Gas sampling was carried out according to the following schedule: (1) in Porijõgi, once a month in May, June and September 2000, October and November 2001, March, May to December 2002, and January to March, July and October 2003; (2) in Gumnitz, 3–4 times a month from January to September 2000 and once a month in January and May 2001 and in May 2002. At the end of the 1-hr measuring period, gas samples were taken from the enclosures of samplers with previously evacuated gas bottles (100 ml; see Augustin et al. 1998). The soil temperature and redox potential, and water depth in the sampling wells was measured simultaneously, and the NH₄-N and NO₃-N concentration in soil samples was analysed using the Kjeldahl method (APHA 1989).

Intact soil cores (diameter 6.8 cm, height 6 cm) for use with the He-O method were taken from the topsoil (0–10 cm) at the gas sampler sites each time after gas sampling was completed. Soil samples were weighed, kept at low temperature (4 °C) and transported to the laboratory of the Institute of Primary Production and Microbial Ecology of the Centre for Agricultural Landscape and Land Use Research (ZALF) in Germany. At the lab, the soil cores were introduced into special gas-tight incubation vessels. In these vessels, N₂ was removed using 3 sub-

### Table 1. Main soil characteristics of the study sites.

<table>
<thead>
<tr>
<th>Study site</th>
<th>Average water table (m)</th>
<th>pH</th>
<th>Dry matter (%)</th>
<th>N (%)</th>
<th>NH₄⁺-N (mg 100 g⁻¹)</th>
<th>NO₃⁻-N (mg 100 g⁻¹)</th>
<th>C (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gumnitz A</td>
<td>0.60</td>
<td>5.2</td>
<td>39.8</td>
<td>1.6</td>
<td>0.43</td>
<td>20.90</td>
<td>20.1</td>
</tr>
<tr>
<td>Gumnitz B</td>
<td>0.02–0.04</td>
<td>5.7</td>
<td>25.7</td>
<td>1.9</td>
<td>0.46</td>
<td>14.30</td>
<td>39.5</td>
</tr>
<tr>
<td>Porijõgi Wet</td>
<td>0–0.05</td>
<td>6.5</td>
<td>55.0</td>
<td>0.4</td>
<td>1.88</td>
<td>0.09</td>
<td>4.0</td>
</tr>
<tr>
<td>Porijõgi Dry</td>
<td>0.45–0.95</td>
<td>6.3</td>
<td>44.4</td>
<td>0.8</td>
<td>0.33</td>
<td>0.14</td>
<td>4.5</td>
</tr>
<tr>
<td>Porijõgi Edge</td>
<td>0.45–0.95</td>
<td>6.3</td>
<td>65.5</td>
<td>0.3</td>
<td>0.61</td>
<td>0.11</td>
<td>5.3</td>
</tr>
</tbody>
</table>
sequent slight evacuation/flushing cycles with an artificial gas mixture (21.3% O₂, 78.6% He, 337 ppm CO₂, 374 ppb N₂O, 1882 ppb CH₄ and approximately 5 ppm N₂). This was followed by the establishment of a new flow equilibrium by continuously flushing the vessel headspace with the gas mixture at 10 ml per minute for 12 hours. For the start value, N₂ and the greenhouse gas concentration in the gas mixture was measured. The gas concentrations in the incubation headspace were measured (final value) after closing the incubation headspace for one hour to accumulate the emission of N₂ and the greenhouse gases. The final accumulation value minus the start continuous flow value served as the basis for the calculation of the emission rates. During the flushing the redox potential of the soil cores was regularly measured and regulated so that it was comparable with the field conditions. The gas concentration in the collected air was determined by using gas chromatography (electron capture detector and flame ionization detector; Loftfield et al. 1997) in the lab of the Institute of Primary Production and Microbial Ecology, Centre for Agricultural Landscape and Land Use Research (ZALF), Germany. The procedures used for the determination of the emission rates of gases are described by Mander et al. (2003).

Water sampling and analyses

In Porijõgi, shallow groundwater samples from the upper aquifer were collected once to twice a month from 6 piezometers, 3 of them installed on the upper border and 3 within the grey alder stand. The depth of ground water varied from 10–80 cm. Filtered water samples were analysed for NH₄-N, NO₂-N, NO₃-N, total Kjeldahl-N, phosphate (PO₄³⁻-P), total phosphorus (total-P), sulphate (SO₄²⁻), iron (Fe) and calcium (Ca²⁺) in the Laboratory of Plant Biochemistry of the Estonian Agricultural University following standard methods for the examination of water and wastewater quality (APHA 1989). Groundwater discharge was estimated on the basis of both Darcy’s law and through gauging with weirs installed in groundwater seeping sites. Average annual input discharge was estimated to be 8.3 ± 3.5 m³ h⁻¹ d⁻¹. Meteorological analyses are based on precipitation, air temperature, wind velocity and humidity data, measured six times a day, as well as daily, monthly and annual averages from the Ülenurme Meteorology Station of the Estonian Meteorology and Hydrology Institute (EMHI). The station is located near the weir of the Porijõgi transect. Mean annual precipitation for the study period varied from 555 to 711 mm.

Soil sampling and analyses in the Estonian study area

In Porijõgi, three soil samples of 50 cm³ were taken at two depths (0–10 cm and 10–20 cm) from all plant communities through the riparian buffer zone. Sampling was carried out twice a year: in spring (May) and autumn (October). Soil pH value, organic matter (loss of ignition), Kjeldahl-N, and lactate-soluble P concentrations were analyzed in all soil samples using the standard methods (APHA 1989).

Calculations and statistical analyses

In order to estimate the final N₂-N flux values, we used the N₂:N₂O ratio calculated using the He-O method. The GWP of the studied systems was calculated by converting the fluxes of N₂O and CH₄ into CO₂ equivalents (IPCC 2001).

The normality of variable distributions was checked using the Kolmogorov-Smirnov, Lilliefors, and Shapiro-Wilk’s tests. In most cases for gas analyses, the distribution differed from the normal distribution, and hence non-parametric tests were performed. Medians, 25% and 75% percentiles and non-outlier range values of variables are presented. We used the Kruskal-Wallis ANOVA and multiple comparison of mean ranks to check the significance of differences between the gas fluxes at different sites. For CH₄-C, the Duncan test was used. The Mann-Whitney U-test was used to check the difference between the gas fluxes in the study areas in Estonia and Germany. The statistical analysis was carried out using Statistica ver. 7.1 (StatSoft Inc.). The level of significance of α = 0.05 was accepted in all cases.
Results

Gaseous emissions

We found remarkable differences in the median values of gaseous emissions between different study sites (Fig. 1). Riparian grey alder sites emitted significantly less N\textsubscript{2}O-N than black alder sites: the median values for N\textsubscript{2}O flux varied from 0.2 to 0.7 and from 0.9 to 2.6 kg N ha\textsuperscript{-1} year\textsuperscript{-1} respectively, and for N\textsubscript{2} flux from 700 to 1200 and from 360 to 400 kg N ha\textsuperscript{-1} year\textsuperscript{-1} correspondingly (Fig. 1). No significant difference was found between different sites in the grey alder forest, while the N\textsubscript{2}O-N flux in black alder stand with altered water regime (Gumnitz A: 2.6 kg N ha\textsuperscript{-1} year\textsuperscript{-1}) was significantly higher than in the black alder stand with unchanged water regime (Gumnitz B: 0.9 kg N ha\textsuperscript{-1} year\textsuperscript{-1}; Fig. 1). No significant differences between N\textsubscript{2}:N\textsubscript{2}O-N ratios in Estonian grey alder stands were found.

The CH\textsubscript{4} emission showed elevated values in the grey alder sites, and was highest in wetter conditions (Porijõgi Wet: 2.1 kg C ha\textsuperscript{-1} year\textsuperscript{-1}). At the same time, the black alder forests acted as a methane sink (–0.2 and –2.4 kg C ha\textsuperscript{-1} year\textsuperscript{-1} in Gumnitz B and A, respectively; Fig. 1).

The emission of CO\textsubscript{2}-C was highest in the black alder forest with lowered water table (Gumnitz A: 21 t C ha\textsuperscript{-1} year\textsuperscript{-1}), whereas no significant differences were found between the study sites in the grey alder forest (2.2–2.9 t C ha\textsuperscript{-1} year\textsuperscript{-1}; Fig. 1).

Global warming potential of studied systems

The studied riparian buffer zones did show relatively high GWP values: from 1356–1547 kg CO\textsubscript{2}-C equivalents (eq) ha\textsuperscript{-1} year\textsuperscript{-1} in Porijõgi to 4870–11 130 CO\textsubscript{2}-C eq ha\textsuperscript{-1} year\textsuperscript{-1} in Gumnitz.
Carbon dioxide constituted the largest part of the total GWP of the riparian alder forest, showing average values of 1473, 1105 and 1226 kg CO$_2$-C eq ha$^{-1}$ year$^{-1}$ in Porijõgi Edge, Wet and Dry sites, respectively, and 10413 and 4604 kg CO$_2$-C eq ha$^{-1}$ year$^{-1}$ in Gumnitz A and B sites, correspondingly (Fig. 2). The share of N$_2$O in GWP was significantly higher in the black alder stands in Gumnitz (290–791 kg CO$_2$-C eq ha$^{-1}$ year$^{-1}$) than in the grey alder stand in Porijõgi (68–202 kg CO$_2$-C eq ha$^{-1}$ year$^{-1}$; Fig. 2). The share of methane fluxes in GWP was very low, while it was negative in the black alder stands at both Gumnitz sites (–4 to –55 kg CO$_2$-C eq ha$^{-1}$ year$^{-1}$).

Discussion

Several investigations into N$_2$O and CH$_4$ emissions from riparian zones show significant variability in emission rates, as well as in the percentage share of N$_2$O-N in the N load (Table 2). The N$_2$O and CH$_4$ fluxes have ranged from −0.22 to 20 kg N$_2$O-N ha$^{-1}$ year$^{-1}$ and from −4.4 to 36 kg CH$_4$-C ha$^{-1}$ year$^{-1}$, while the share of N$_2$O-N emission in the total N input has varied from 0.02% to 4.2%. Likewise, our earlier studies showed a remarkable variability of greenhouse gas emissions from the Porijõgi alder forest (Mander et al. 2005, Teiter and Mander 2005).

Dinitrogen emission was the most important component of N retention within the Porijõgi riparian grey alder forest. The estimated N$_2$-N emission for 1994–1995 was 51.2 kg N ha$^{-1}$ year$^{-1}$ (Lõhmus et al. 2002), while the latest measurements in 2001–2003 gave median values as high as 700–1200 kg N ha$^{-1}$ year$^{-1}$. The intensive N$_2$ emission from the Porijõgi test area can be related to the high microbial activity in alder forests (Hart et al. 1997, Dilly et al. 2000, Lõhmus et al. 2002), which could be assumed to lead to higher denitrification activity. One of the N$_2$ sources could be denitrification (degassing) in the groundwater, although this has been measured in riparian wetlands (Blicher-Mathiesen et al. 1998, Mookherji et al. 2003). Mogge et al. (1998) found that denitrification in a black alder forest was approximately 15 times greater than in beech forest. The comparison of nitrogen-fixing red alder and Douglas fir forests has shown that denitrification enzyme activity was greater in the alder forest. Also, in alder forest denitrification was limited only by organic matter and not by NO$_3^-$, whereas in Douglas-fir soils it was frequently limited by both NO$_3^-$ and energy (Griffiths et al. 1998). On the other hand, the CH$_4$ oxidation capacity in alder forests has
been found to be significantly lower than in other forests (Reay et al. 2001, 2005). This explains the higher CH$_4$ emission values in alder stands (see the Wet site in Porijõgi; Fig. 1).

To determine the possible impact of stand age and site conditions on the forms of N emission, we measured gaseous emissions from additional grey alder stands in southern Estonia in July and October 2003. In a Viiratsi riparian grey alder stand (Kuusemets et al. 2001, Lõhmus et al. 2002, Mander et al. 1997) two sites in the 50-year-old part and one in the <10-year-old buffer strip were involved in the study. In Holvandi, emissions were measured in a 9-year-old grey alder plantation on an abandoned field (Uri et al. 2003a) where the soil N pool was 3–4 times lower than in riparian grey alder stands. In all of the grey alder stands studied (Porijõgi, Holvandi, and Viiratsi) the N$_2$-N:N$_2$O-N ratio was very high.

In riparian grey alder stands N$_2$ emissions were significantly lower in 50-year-old sites than in younger ones, which indicates that N$_2$ emission in alder stands could be higher in earlier successional phases. This is most likely related to changes in microbial activity during the succession of a pioneer grey alder stand into a more stable mixed forest community.

Based on the total inputs and outputs as well as accumulation and transformation rates of nitrogen estimated in our earlier publications (Lõhmus et al. 2002, Mander et al. 2005) the N budget of the grey alder stand in Porijõgi was established (Fig. 3). Total inputs (kg N ha$^{-1}$ year$^{-1}$), which consist of symbiotic N$_2$ fixation (184.6), subsurface and overland flow (25.6), deposition of mineral nitrogen (6.4), and non-symbiotic N$_2$ fixation (0.5), were 217.1 kg N ha$^{-1}$ year$^{-1}$. This relatively high value is due to high symbiotic fixation, whereas the non-symbiotic N$_2$ fixation was estimated to be negligible.

Nitrogen accumulation in plants after litter fall was 139.4 kg N ha$^{-1}$ year$^{-1}$ (Fig. 3). The annual N accumulation in soil was remarkably high (98.3 kg N ha$^{-1}$ year$^{-1}$), about half of the fixed N$_2$-N. The total N demand in the Porijõgi grey alder stand in 1994 was 285.3 kg N ha$^{-1}$ year$^{-1}$; 81.9 kg N ha$^{-1}$ year$^{-1}$ was transformed and assimilated by plants, whereas the litter amounted to 132 kg N ha$^{-1}$ year$^{-1}$ (see Lõhmus et al. 2002).

### Table 2. Emission of nitrous oxide and methane in riparian buffer zones.

<table>
<thead>
<tr>
<th>Reference</th>
<th>Country</th>
<th>Ecosystem type</th>
<th>N$_2$O emission (kg N ha$^{-1}$ year$^{-1}$)</th>
<th>CH$_4$ emission (kg C ha$^{-1}$ year$^{-1}$)</th>
<th>N$_2$O% of N input</th>
<th>Comment</th>
<th>Initial load N$_2$O (kg N ha$^{-1}$ year$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Weller et al. 1994</td>
<td>USA</td>
<td>Riparian hardwood forest (hyporheic)</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>range</td>
<td>24.1–102</td>
</tr>
<tr>
<td>Jones et al. 1995</td>
<td>USA</td>
<td>Riparian (hyporheic) zone in an arid area</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>range</td>
<td>65.7–298</td>
</tr>
<tr>
<td>Jaschke et al. 1998</td>
<td>USA</td>
<td>Riparian forested wetlands</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>range</td>
<td>475</td>
</tr>
<tr>
<td>Groffman et al. 2000</td>
<td>USA</td>
<td>Riparian black alder forest</td>
<td>20.0</td>
<td>-</td>
<td>-</td>
<td>range</td>
<td>11.0–292</td>
</tr>
<tr>
<td>Hefting et al. 2005</td>
<td>Germany</td>
<td>Riparian black alder forest</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>range</td>
<td>475</td>
</tr>
<tr>
<td>Tietz &amp; Mander 2005</td>
<td>Estonia</td>
<td>Riparian grey alder forest</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>range</td>
<td>-4.4–3.1</td>
</tr>
</tbody>
</table>

* original data given in μg m$^{-2}$ h$^{-1}$ or in mg m$^{-2}$ d$^{-1}$.
Nitrogen output into groundwater and streams was 13.2 kg N ha\(^{-1}\) year\(^{-1}\) (Fig. 3). The estimated N\(_2\)-N emission for 1994–1995 was 51.2, whereas emission of N\(_2\)O-N, which contributes to global warming and damages the ozone layer, was 0.5 kg N ha\(^{-1}\) year\(^{-1}\) (Fig. 3). Thus the nitrous oxide flux was 3 orders lower than the dinitrogen emission.

Considering all inputs and outputs, the N removal efficiency in grey alder stands decelerates with increasing age (see also Mander et al. 2005). Thus grey alder buffer communities should be managed by regeneration cutting and tending so as to keep their N removal rate high.

Due to intensive N and C retention, the P retention in alder forests is also remarkable (Giardina et al. 1995, Zou et al. 1995, Uri et al. 2003b, Mander et al. 2005). Thus alder forests with intensive internal nutrient cycling work as effective soil producers without significant leaching of nutrients. In riparian zones denitrification is the key factor in balancing N fluxes.

The relatively high CO\(_2\) emission from the Gumnitz study area is related to the significantly higher total C concentration of soils in Gumnitz as compared with that in the Poriõgõi study area (20%–40% and 4.0%–5.3%, respectively). In our study, the CO\(_2\) emission is not connected with fluxes related to plant photosynthesis. Therefore only data for cold periods can be considered as losses to the atmosphere. In the calculation of net ecosystem CO\(_2\) exchange, a more advanced measurement technique is required.

However, some studies on C sequestration in wetlands and forest ecosystems (Butnor et al. 2003) allow us to estimate that about 50% of the CO\(_2\) released during soil respiration, will in the vegetation period be assimilated by trees through the photosynthesis. Therefore this 50% reduction has been taken into account in our results on CO\(_2\) emissions from forest ecosystems (Fig. 3). In comparing the greenhouse potential of CH\(_4\) and N\(_2\)O over a long time scale (100–500 years), one can speculate that due to the short adjustment time for CH\(_4\) in the atmosphere (8.4 years; IPCC 2001), the radiative forcing of CH\(_4\) will fall relative to CO\(_2\) (Whiting and Chan- ton 2001). Nitrous oxide, with its atmospheric lifespan of about 120 years and GWP value of 296, however, has a more significant impact. Our data show a minor CH\(_4\) emission from riparian alder forests. Likewise, the N\(_2\)O emission was relatively low as compared with emissions from constructed wetlands for wastewater treatment (Teiter and Mander 2005). However, the high radiative forcing value of N\(_2\)O makes its share in total GWP remarkable, being highest in the riparian black alder forest with altered water regime (771 kg CO\(_2\)-C eq ha\(^{-1}\) year\(^{-1}\) in Gumnitz A; Fig. 3). The lowering of the water table in wetlands is a well-known reason for N\(_2\)O emission (Martikainen et al. 1993). The planned increase in fertilization intensity and reconstruction of abandoned drainage systems in several Eastern European countries after the political and socio-economic changes of the 1990s (see Mander et al. 2000, Stålnacke et al. 2003) may change the N balance of riparian ecosystems. However, its dynamics and especially the change in gaseous N fluxes are relatively unpredictable. Therefore
further investigations should concentrate on the factors that regulate N\textsubscript{2}O and N\textsubscript{2} emission rates from riparian buffer zones.

Conclusions

Gaseous emissions from the studied riparian alder stands showed significant spatial variation. The median values of N\textsubscript{2}O-N, N\textsubscript{2}-N and CH\textsubscript{4}-C ranged from 0.2 to 2.6, from 360 to 1200 and from –2.4 to 2.1 kg ha\textsuperscript{-1} year\textsuperscript{-1}, respectively. The CO\textsubscript{2}-C fluxes varied from 2.2 to 21 t ha\textsuperscript{-1} year\textsuperscript{-1}. In the grey riparian alder forest sites the CH\textsubscript{4} and N\textsubscript{2} fluxes were higher and the CO\textsubscript{2} and N\textsubscript{2}O fluxes were lower than in the black alder sites.

Dinitrogen emission was found to be the most important component in N retention from the studied riparian grey alder forests (up to 1200 kg N\textsubscript{2}-N ha\textsuperscript{-1} year\textsuperscript{-1}). On the other hand, we found a significantly higher N\textsubscript{2}-N emission and lower N\textsubscript{2}O-N flux from the grey alder stand on mineral soils than from the black alder stands on organic soils. Nitrous oxide emission was significantly higher in a study site with a lowered groundwater table, while the methane emission showed higher values in wetter conditions at Porjõgi Wet Site.

Carbon dioxide constituted the largest part of the total GWP of riparian alder forests, being significantly higher in black alder sites (from 4604 to 10413 kg CO\textsubscript{2}-C eq ha\textsuperscript{-1} year\textsuperscript{-1}) as compared with that in the grey alder sites (from 1105 to 1473 kg CO\textsubscript{2}-C eq ha\textsuperscript{-1} year\textsuperscript{-1}). The share of N\textsubscript{2}O in the GWP was also significantly higher in the black alder stands: 290–771 and 68–202 kg CO\textsubscript{2}-C eq ha\textsuperscript{-1} year\textsuperscript{-1} for black alder and grey alder stands, respectively.

Riparian grey alder forests as pioneer communities are optimal buffer zones, and their N removal capacity is high despite the N accumulation in soil due to symbiotic N\textsubscript{2} fixation. Instead of leaching, in these ecosystems denitrification works down the soil N storage accumulated during succession.

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