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Valuation of nitrogen retention as an ecosystem service on a catchment scale

Katri Rankinen, Kirsti Granlund, Randall Etheridge and Pentti Seuri

ABSTRACT

An ecosystem service approach was used to study the water purification service exemplified by impacts of land management scenarios. Nitrogen retention was calculated in two agricultural catchments by the dynamic Integrated Nutrients in Catchments (INCA)-N model. The monetary valuation was based on purification efficiency of artificial wetlands. The set of scenarios were based on existing agricultural water protection measures, and greening of the Common Agricultural Policy proposal. Scenarios were: wintertime crop cover on fields, increase in area of set aside land, decrease in nitrogen fertilization, crop diversification and nutrient recycling in organic farming. Nitrogen retention provided more value in the Yläneenjoki catchment where the main production line was animal husbandry. In the slowly flowing river Lepsämänjoki, the N retention was more effective than in the fast flowing river Yläneenjoki. When comparing measures some proved to have no value or even a negative value. Set aside had a high positive value when calculated per area, but on a catchment scale the value remained low because of the small area of implementation. Nutrient recycling and winter time vegetation cover were the scenarios that reduced N leaching from fields close to targets set in current political decisions. None of the scenarios increased greenhouse gas emissions.

Key words | agricultural scenarios, catchment scale modeling, Common Agricultural Policy (CAP), ecosystem services, nitrogen retention

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INTRODUCTION

The existence of humankind depends on nature which provides us with essential goods such as drinking water, food and energy. In the Millenium Ecosystem Assessment (MA 2005) ecosystem services are defined as the benefits people obtain from ecosystems. It deals with the full range of ecosystems from those relatively undisturbed, such as natural forests, to ecosystems intensively managed and modified by humans, such as agricultural land and urban areas. Different typologies of ecosystem services cover the broad range of services, such as providing food, fiber, shelter and available habitats, regulating carbon, water and pollination, and creating opportunities for recreation, religion and aesthetics.

On the other hand, we are changing functioning of ecosystems by our own actions. Excess use of nutrients has increased eutrophication of surface waters and combustion of fossil fuels has increased carbon dioxide content in the atmosphere leading to climate change. High nitrogen (N) losses and apparently inefficient regulation methods require immediate attention in several European countries in order for environmental water quality goals to be met (Conley *et al.* 2009; Sutton *et al.* 2011; Hong *et al.* 2012). In Finland, agriculture forms the largest source of N to waters. Despite regulatory efforts, the fluxes of N from terrestrial to aquatic systems have increased (Aakkula *et al.* 2012).

An agri-environmental subsidy program (EEC 1992) that states water protection as one of its main objectives has been in place in Finland since 1995. It is partly paid for by Finland and partly by the EU. The program attempts to influence agricultural practices such as crop choice, set

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aside, fertilizer application rates, tillage systems, and the adoption of water protection measures. At present, the EU is in the process of renewing the Common Agricultural Policy (CAP) to include some of the ecosystem protection elements (CAP Greening) (EC 2011). Further, the target of the EU's Nitrate Directive (ND) (EEC 1991) is to keep the nitrate concentration of surface and ground waters below the boundary level of 50 mg L^{-1}. Finally, the main goals of the EU Water Framework Directive are to achieve good ecological and chemical status for all inland and coastal surface waters by the year 2015 (WFD 2000).

In other regulations at an international level, the Ministers of Environment of the Baltic Sea States decided that anthropogenic loading to the Baltic Sea should be reduced by 50% from 1987 levels by the year 1995. More recently, the Helsinki Commission (HELCOM) negotiated the Baltic Sea Action Plan that aims at cutting phosphorus and nitrogen inputs to the Baltic Sea by 42 and 18%, respectively, from the average loads of 1997–2003 by the year 2016. On a national level, the Finnish Council of State issued a Decision-in-Principle on water protection targets to be met in 2005 (Ministry of the Environment 1998). These were not met and new targets to be achieved by the year 2015 were set in 2006. The key objective is that nutrient loads entering water bodies from agriculture should be reduced by a third compared to their levels over the period 2001–2005, and halved over a longer timescale. Further, Prime Minister Matti Vanhanen set a national target for nutrient re-cycling in Finland (Baltic Sea Action Group 2013) recognizing the problem that energy and nutrients are resources not to be wasted.

Environmental decision making in the EU requires understanding of policy effects on an ecosystem level. Thus a solid methodological framework for mapping and assessing ecosystem services is needed. A framework for integrating ecosystem services into decision making incorporates a variety of methods, including ecosystem service dependency, tradeoff and impact assessment, valuations, scenarios, and policies (de Groot *et al.* 2002, 2010). Ecosystem functioning or resources can be called an ecosystem service only when people recognize its value. Mapping of ecosystem services also serves the purpose of making them more visible to the public, policy makers and other stakeholders.

An ecosystem service may have ecological, socio-cultural and economic importance or value (de Groot *et al.* 2002). Further, ecosystem services can be divided into categories based on how they are economically valuated (Fu *et al.* 2011). Some of them can be directly realized in markets, such as foods or raw materials. Other ecosystem services can be indirectly realized (e.g. water purification) and valuated by methods such as avoided cost, replacement cost, and travel cost. The rest of the ecosystem services are those which are difficult to express in economic systems or are completely independent of economic systems. Methods to valuate those may be based on, for example, willingness to pay.

Double counting is a term originally used in economics to refer to the erroneous practice of counting the value more than once, e.g. when an ecosystem service is valued at two different stages of the same process. Valuation of ecosystem services are especially susceptible to double counting as there is an overlap between individual ecosystem services in the MA classification system (Fu *et al.* 2011; Ojea *et al.* 2012). Further, the division between the concepts of ecosystem functioning and ecosystem services are still under debate as sometimes ecosystem functioning is used to describe the internal functioning of the ecosystem, and sometimes the benefits derived from the processes of ecosystems (de Groot *et al.* 2002). They are defined as 'the capacity of natural processes to provide goods and services that satisfy human needs, directly or indirectly'. Fu *et al.* (2011) outlined ecosystem functions to refer to the physical, chemical and biological processes that contribute to the maintenance of an ecosystem, and ecosystem services to refer to the ecosystem conditions and processes that are of direct benefit to human beings.

In MA ecosystem services are divided into four categories: provisioning (e.g. crops, livestock, fresh water, timber, bio fuels), regulating (erosion regulation, air quality regulation, water purification, pollination, carbon sequestration), supporting (nutrient cycling, primary production), and cultural (aesthetic values, recreation, ecotourism). Fisher *et al.* (2009) provide an alternative classification: abiotic inputs (sunlight and rainfall), intermediate services (soil formation, photosynthesis and nutrient cycling), final services (water regulation) and benefits (water for irrigation and drinking water). Different classifications are reviewed, for example, by Fu *et al.* (2011) and Ojea *et al.* (2012).

Other reasons for double counting occur due to, for example, inadequate recognition of spatio-temporal scale dependence of ecosystem services and overlap between ecosystem valuation methods (Fu *et al.* 2011). Ecosystem complexity should be recognized, for example, hydrological regulation, nutrient cycling and climatic regulation are interrelated with each other and affected by a common factor of vegetation cover. Examples of risks of double counting in the water sector are that water services can be evaluated as part of the process and the outcome itself (Ojea *et al.* 2012), and that water supply can be evaluated by direct market cost, replacement cost, avoided cost and travel cost (Fu *et al.* 2011). Thus, only the final ecosystem services should be included in the value of the total ecosystem service and valuation methods should be appropriate for the study context (Fu *et al.* 2011).

The ecosystem approach in policy making allows analyzing trade-offs between different services that an ecosystem provides. Gamfeldt *et al.* (2012) reported that tree species richness in production forests were positively correlated with several ecosystem services. Onaindia *et al.* (2013) concluded that though inclusion of ecosystem services in conservation planning has the potential to provide opportunities for biodiversity protection, when based only on a specific ecosystem service it may be detrimental to biodiversity and also cause other environmental problems. Burgin *et al.* (2013) presented a case study of N retention as positive and simultaneous greenhouse gas emissions in riparian zones, ponds and streams as a negative ecosystem service assessed at a landscape scale.

Most of the retention studies on a catchment scale are focused on water bodies, and only a few studies concentrate on retention of nutrients in terrestrial ecosystems. Recently, Fu *et al.* (2012) assessed the nutrient retention function of ecosystems based on the landscape source-sink theory. In a European wide scale, the effect of CAP Greening on certain ecosystem services, including water purification by N retention, was evaluated by Maes *et al.* (2012). Vigerstol & Aukema (2011) compared modeling tools and concluded that ecosystem services tools may give an overview of the processes whereas traditional hydrological models may give more information of the processes behind ecosystem services.

In this paper we evaluated the use of an ecosystem approach in an agri-environmental policy context. We simulated N retention as a water purification service in two agricultural catchments by the dynamic Integrated Nutrients in Catchments (INCA-N) model (Whitehead *et al.* 1998; Wade *et al.* 2002). The model enabled us to study both the ecosystem service as a final product and the ecosystem functioning contributing to that. The set of scenarios was based on changes in land use that arise from current water protection and agricultural policy measures, and greening measures included in the new CAP proposal. The set of scenarios included: increased area of set aside land, decrease of N fertilization, wintertime crop cover on fields, crop diversification and nutrient recycling in organic farming. As the effects of political measures can be not only positive but sometimes also negative, we aimed to provide quantitative information concerning the impact of various measures on the ecosystem service water purification. The monetary valuation of N retention was based on purification efficiency and construction costs of artificial wetlands.

MATERIALS AND METHODS

Catchment description

We selected two well monitored catchments as case study areas in Finland (Figure 1). In both catchments nutrient loading to the river originates mainly from agriculture, though there are also households that are not connected to municipal waste water treatment. The Lepsämänjoki catchment represents crop production areas, where animal density was 0.08 animal units ha^{-1} of field in 2005. In the Yläneenjoki catchment the main production line is animal husbandry. In 2005 animal density was 0.53 animal units ha^{-1} of field. Both catchments belong to the monitoring network of the follow-up project of the Finnish agri-environmental program (Aakkula *et al.* 2012).

The Lepsämänjoki catchment (214 km^2) is a sub-basin of the Vantaanjoki river basin in southern Finland. The river Vantaanjoki discharges to the Gulf of Finland outside the capital Helsinki and the area is very important for outdoor recreation. The river Lepsämänjoki is divided into two branches. The length of the main branch of the river

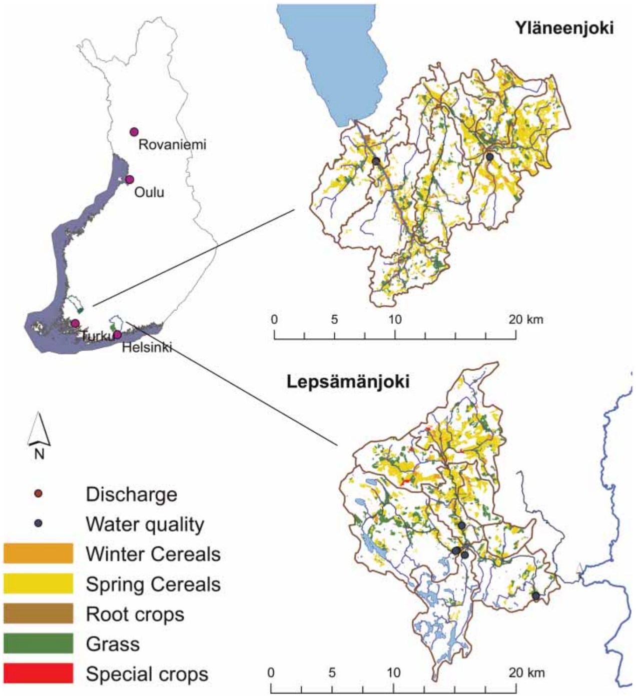

Figure 1 | Location of the study catchments.

Lepsämänjoki is 33.5 km. It meanders slowly in the middle of the fields while the tributary collects waters from forested upland area with lake percentage up to 13%. The main river is also prone to floods, especially during snow melting in the spring. The mean annual precipitation in the area is 650 mm, and mean annual temperature is $+4\,^{\circ}$C (data from the Finnish Meteorological Institute). The mean discharge in the river Lepsämänjoki was 2.2 $m^3\,s^{-1}$ in 2000 (Korhonen & Haavanlammi 2012).

The main soil types in the Lepsämänjoki catchment are clay (*Vertic Cambisol*) and rocky soils (*Dystric Leptosol*) (Lilja *et al*. 2006). Fields cover 23% of the area, the rest being mainly forest. The main crops are spring cereals, but at the upper reaches of the catchment there is also some cabbage cultivation (about 3% of the area).

The Yläneenjoki catchment (233 km^2) is located in the upper reaches of the Eurajoki catchment in southwestern Finland. The river Yläneenjoki is 33 km long and it discharges to the lake Pyhäjärvi, which is an important lake for water supply, commercial fishing and recreational use (Ventelä *et al*. 2007). The mean discharge in the river Yläneenjoki was 1.94 $m^3\,s^{-1}$ in 1991–2010 (Korhonen & Haavanlammi 2012). There are several small rapids in the river Yläneenjoki. Thus nutrient retention is increased by building bottom dams in the river and several small wetlands in the surrounding areas (Ventelä *et al*. 2010). The mean annual precipitation is 700 mm and the mean annual temperature is 4 $^{\circ}$C (data from the Finnish Meteorological Institute).

The main soil types in the catchment are clay (*Eutric Cambisol 2, Vertic Cambisol*) and till and rock (*Lithic Leptosol 1, Haplic Podzol 1,2*), but there are also some organic soil types (*Fibric/Terric Histosol 1*) (Lilja *et al*. 2006). Fields cover 25% of the catchment area, and the main crops are spring cereals, though there is also some sugar beet cultivation in the lower reaches. In the Yläneenjoki catchment, manure is extensively used as fertilizer. A common practice is to spread manure on fields in the autumn when it is available for leaching with autumn rains.

Observation data about river discharge (daily values) and inorganic N concentrations (grab samples: on average 21 samples per year in Lepsämänjoki and 36 in Yläneenjoki) were available from national monitoring programs (Niemi 2006). Discharge in both rivers showed the same seasonal pattern so that the growing season was the low flow period (Figures 2(a) and 2(b)). The snow melt period in the spring was not emphasized, but there were also high flows in the autumn in both rivers. Nitrate concentrations were somewhat higher, especially in the autumn in the river Yläneenjoki (Figures 2(b) and 2(c)) probably reflecting manure use as fertilizer in the fields. In the river basin plan of the Water Framework Directive (WFD 2000), the river Lepsämänjoki was estimated to achieve good ecological condition by 2021 and the river Yläneenjoki by 2027 (Salmi & Kipinä-Salokannel 2010).

Scenarios

The Finnish Agri-Environmental Programme (FAEP) together with the ND defined the current situation in agricultural water protection (Business As Usual). ND contains an upper limit to spread manure on fields (170 kg N ha^{-1}) and restrictions on spreading times. ND is valid in the whole of Finland.

Although the FAEP is voluntary, over 90% of farmers have joined it. Obligatory basic measures contain maximum allowed fertilization levels for different crops. Further, a

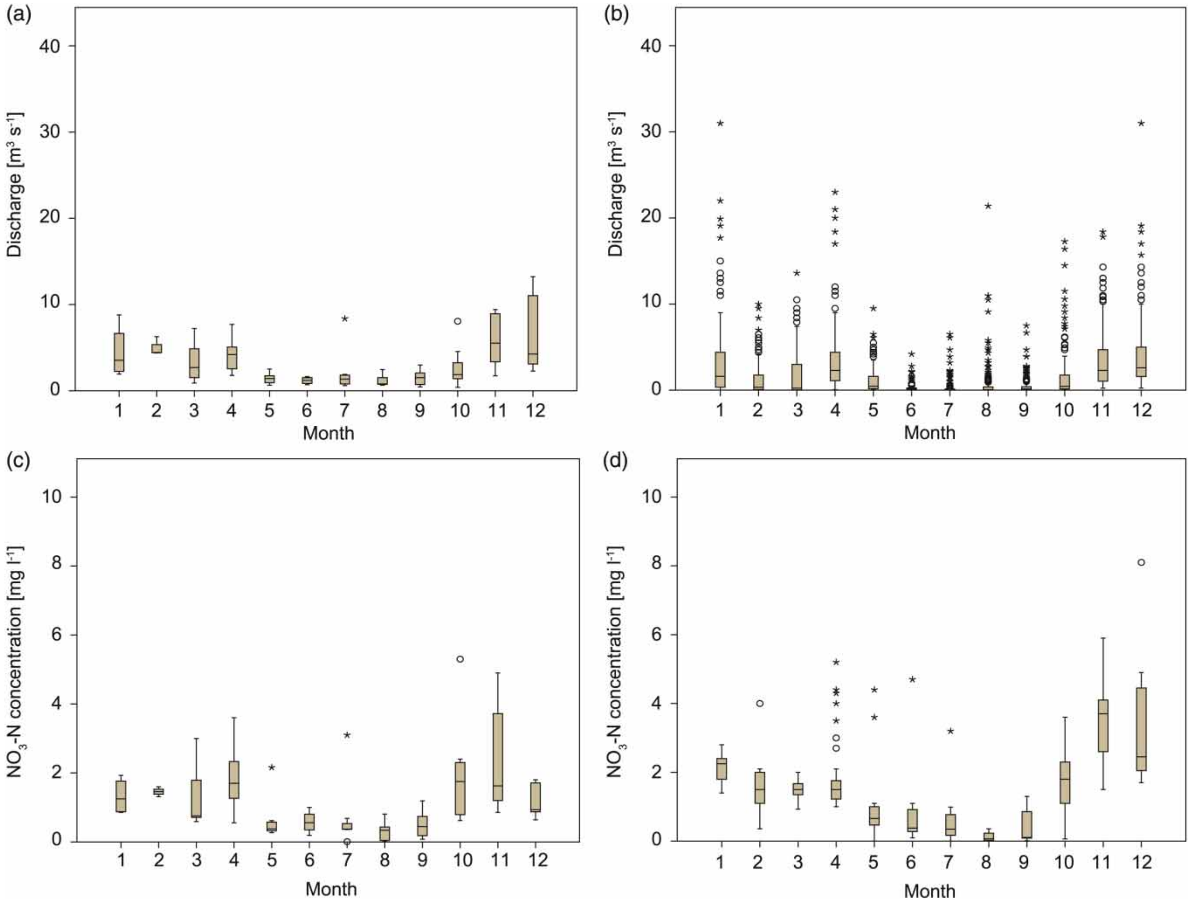

Figure 2 | Seasonality of discharge in the river (a) Lepsämänjoki and (b) Yläneenjoki, and NO_3-N concentrations (c) Lepsämänjoki and (d) Yläneenjoki in 2003–2008/2009.

minimum 5% of field area should be left as set aside. Farmers should also choose some of the additional measures, which contain, for example, reduced fertilization and winter-time vegetation cover. The FAEP also involves clearly targeted measures such as wetlands or 15 m wide buffer zones.

According to basic measures, maximum N fertilization on clay soils in southern Finland is 100 kg N ha^{-1} for barley, 120 kg N ha^{-1} for spring wheat and 200 kg N ha^{-1} for grass (www.mavi.fi/attachments/mavi/ymparistotuki/5xQHBic3n/Ymparistotuen_sitoumusehdot_2011.pdf). When using manure as fertilizer, only soluble N is included when calculating its N content. If manure is spread in autumn only 50% (until 2006) or 75% (after 2006) of soluble N is taken into account.

In the WFD river basin management plans, there are more specific catchment level targets for water protection measures than in the FAEP. In the Yläneenjoki catchment the suggested measures are 50% of total field area under wintertime vegetation cover and 100% of area under reduced fertilization (Salmi & Kipinä-Salokannel 2010). The same measures are mentioned in the water protection plan for the Lepsämänjoki catchment, but without any clear areal targets.

The scenarios for Finnish case studies were based on water protection plans of WFD and CAP Greening. The scenarios were as follows:

1. 'Wintertime vegetation cover', in which stubble remains over winter in all spring cereal fields.
2. 'Reduced fertilization', in which the N balance decreases to 20 kg N ha^{-1} for all crops, and if manure is used as fertilizer (Yläneenjoki) its spreading is allowed only during the growing season.
3. 'Ecological set aside', in which set aside area increases up to 10–15% of the total field area.

4. 'Crop diversification', in which at least three crops are cultivated, spring cereals cover <40% of the field area and grasses cover >10% of the field area.
5. 'Ecological Recycling Agriculture (ERA)' is based on local nutrient resources, integrating animal and crop production on farms or in their proximity. A theoretical crop rotation was developed to represent ERA cultivation. Nitrogen fixation and soil N mineralization provided N for all crops except fodder cereal, which received manure.

Modeling of nitrogen retention

Changes in N retention were simulated by the INCA-N (Whitehead *et al.* 1998; Wade *et al.* 2002) model combined with a Monte Carlo generalized sensitivity analysis of Spear (1970) and Spear & Hornberger (1980). The INCA-N model integrates hydrology and N processes in the river and soil compartments. The model is semi-distributed in that the land surface is not described in detail, but rather by land-use classes in sub-basins. These classes are based on current land use from CORINE 2006 data and filed parcel database (2010) of the Information Centre of the Ministry of Agriculture and Forestry. Hydrological input is calculated by the conceptual WSFS model (Bergström 1976; Vehviläinen 1994), which is in operational use for flood forecasting in Finland.

In the INCA-N model, sources of N include atmospheric deposition, leaching from the terrestrial environment and direct discharges. In the river the key N processes are nitrification and denitrification. River flow velocity was calibrated against a measured hydrograph.

The key N processes in soil are nitrification, denitrification, mineralization, immobilization, N fixation and plant uptake. These N processes are calibrated against measured values in Finland (Martikainen *et al.* 1994; Syväsalo *et al.* 2006; Regina & Alakukku 2010, annual yield statistics) so that simulated leaching from fields is at the same level with leaching from field experiments (Salo & Turtola 2006; Puustinen *et al.* 2010). Results from the research program HYÖTYLANTA (sustainable utilization of manure, Luostarinen *et al.* 2011), aiming at discovering reuse options for manure, were also utilized (e.g. information about nutrient content in manure). Simulated average N field balances (Table 1) were higher in the Yläneenjoki catchment where manure was used as fertilizer than in the Lepsämänjoki catchment, indicating higher potential for N losses to waters. Simulated N balances represented well the field N balances calculated for different crops in the agri-environmental follow up study (Mattila *et al.* 2007). Average inorganic N leaching from the fields was 21 kg ha^{-1} in the Lepsämänjoki and 35 kg ha^{-1} in the Yläneenjoki. Nutrient loading from forested areas was in general very low (Lepistö 1996; Mattson *et al.* 2002).

The starting point of the Monte Carlo simulations was the model application calibrated against observed discharge and NO_3-N concentrations in the river. The goodness of fit for simulations was evaluated (Table 2) by calculating the coefficient of determination (R^2), root mean squared error (RMSE), relative error (RE) and Nash–Sutcliffe efficiency criteria (N–S) (Nash & Sutcliffe 1970). The calibrated parameter sets were allowed to vary in between the maximum and minimum values of field measurements, or if those did not exist, by ±20% from the calibrated value. As the main interest was in agricultural fields, forests were excluded from the analysis. During Monte Carlo simulations, 250 iterations of a Latin hypercube with 20 divisions were produced for a total of 5,000 simulations.

Most of the measured NO_3-N concentrations fell between the 95% uncertainty bounds, if the discharge was correctly simulated (Figure 3). Especially, in the river Yläneenjoki the measured NO_3-N concentrations were higher than simulated ones during low flow periods, possibly indicating higher N inputs from point sources than assumed in the modeling or insufficient retention processes in the river. The most influential parameters

Table 1 | Simulated average N balance (N input minus crop N uptake) of main crops in the catchments

	Lepsämänjoki N balance (kg/ha)	Yläneenjoki N balance (kg/ha)
Spring cereals	40	81
Winter cereals	45	96
Grass	50	100
Cabbage	145	–
Sugarbeet	–	68

Table 2 | Model performance in INCA-N applications in the three study catchments

	Discharge			NO_3-N concentration			NO_3-N daily load		
Catchment	R^2	N–S	RMSE	R^2	RE	RMSE	R^2	RE	RMSE
Lepsämänjoki	0.753	0.741	48.318	0.368	–123.81	76.299	0.579	–5.354	17.382
Yläneenjoki	0.627	0.556	113.8	0.520	–613.64	66.60	0.27	–2120.6	335.78

R^2 = coefficient of determination; N–S = Nash–Sutcliffe coefficient; RE = relative error; RMSE = root mean squared error.
Values are given for modeled discharge, stream NO_3-N concentration and daily load.

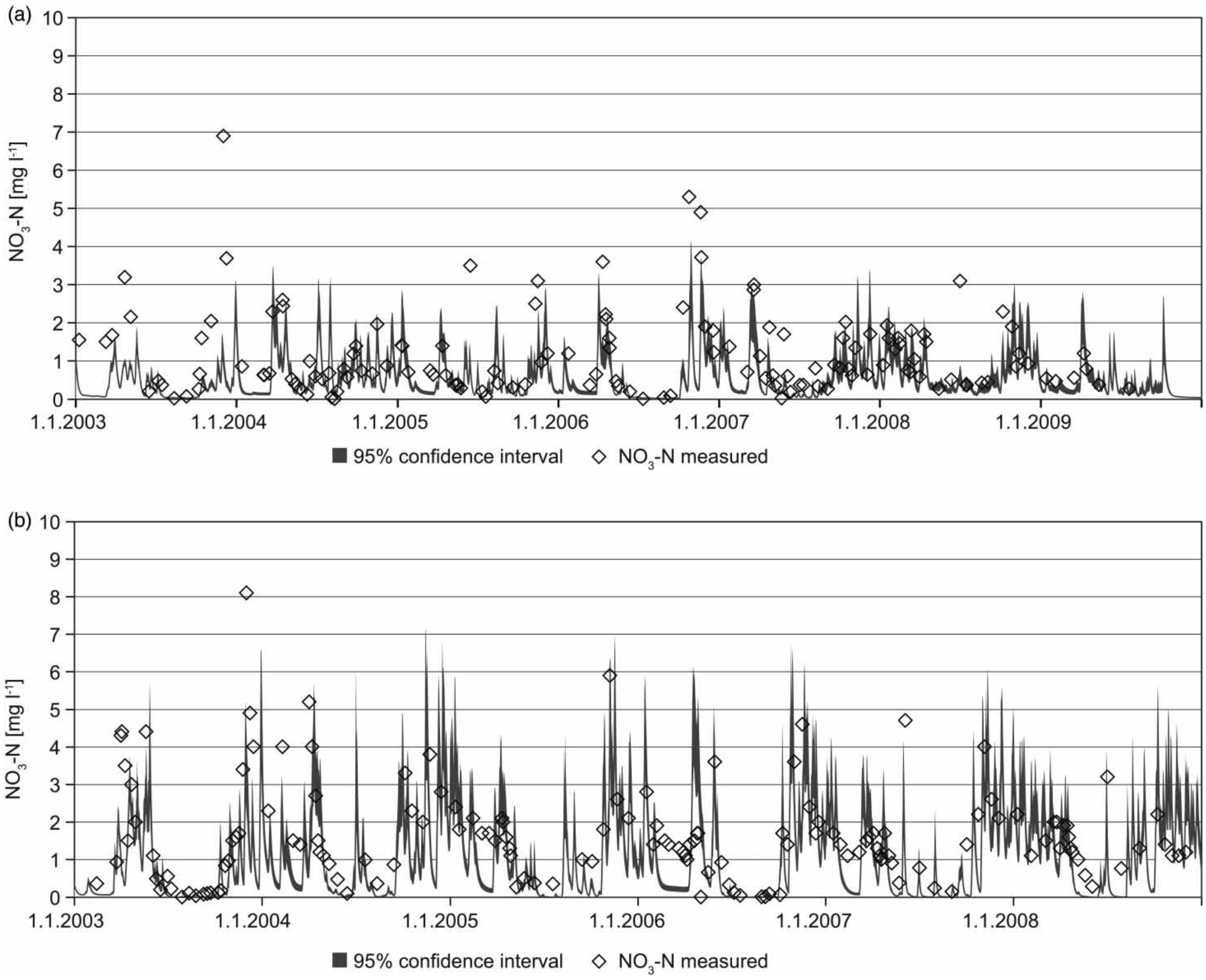

Figure 3 | Modeled 95% uncertainty bounds and measured NO_3-N concentrations (a) in the river Lepsämänjoki and (b) in the river Yläneenjoki.

were related to temperature dependency of terrestrial N processes and parameters defining river velocity (Figure 4). The results were in line with earlier results by Rankinen *et al*. (2013) where terrestrial N processes were less sensitive to parameters defining process rates than to parameters regulating process rate dependency of environmental conditions. Also, according to Jin *et al*. (2012), simulated in-stream N transformation rates

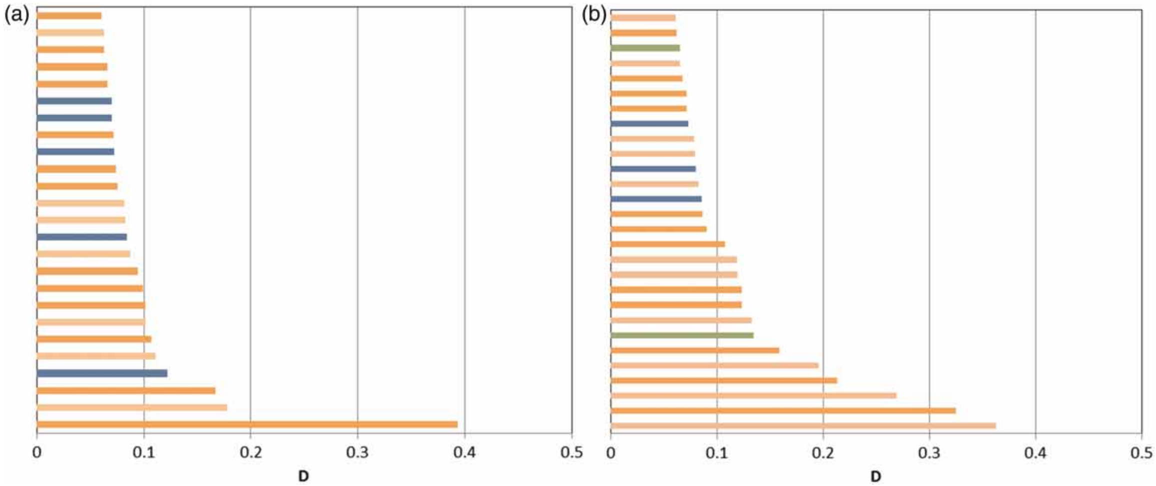

Figure 4 | Influential parameters in INCA-N applications to (a) Lepsämänjoki catchment and (b) Yläneenjoki catchment. Orange: temperature dependency, blue: river flow velocity, and green: soil N process.

were sensitive to flow velocity. The parameter uncertainty bounds were included in scenario runs to represent model uncertainty.

Monetary valuation

We used replacement cost as a common criterion for monetary valuation, assuming that N retention can be replaced by constructing an artificial water protection wetland. To avoid double counting we valuated only N retained in the whole catchment. Out of curiosity, for the 'business as usual' scenario we calculated the value of N retained in river itself, but in the scenario evaluation that value was included in the scenarios.

The price per kg of N retained by artificial wetlands (25 € kg^{-1} N) was based on studies on wetlands of different size (Majoinen 2005; Väisänen 2008) carried out in Finland. These studies took into account observed retention and construction and maintenance costs for a 10–15 years operational cycle of the wetland, but not the value of land. In Sweden, Byström (1998) and Byström *et al.* (2000) estimated that the retention cost would be 113–495 SEK kg^{-1} N if the retention percentage of the wetland was about 50–75%. With the current rate (1 € = 8.8 SEK) this value is close to what was estimated in Finland. We used these values when calculating uncertainty bounds for monetary valuation.

RESULTS AND DISCUSSION

Nitrogen loading in the 'business as usual' scenario was 119 t a^{-1} from the Lepsämänjoki catchment and 237 t a^{-1} from the Yläneenjoki catchment. Nitrogen loading decreased in other scenarios except 'crop diversification' (Table 3). The average retention percentage and the total value of N retention in the river Lepsämänjoki was higher than that in the river Yläneenjoki (Figure 5). Retention was calculated to the main branch only, but it might implicitly contain retention in the small lakes as well. The retention percentages (including modeling uncertainty) of the rivers did not change considerably between the scenarios.

We calculated both unit value (€ km^{-2} of treatment, Figure 6) and total value (€, Figures 7(a) and 7(b)) of N retention. As a measure 'set aside' had a high unit value, but because implementation areas were small its total value remained low. In the Lepsämänjoki catchment the 'wintertime vegetation cover' scenario gave the highest total value (€) but in the Yläneenjoki catchment the highest total value was gained in the 'reduced fertilization' scenario, probably because

Table 3 | Percentual changes in process loads in different scenarios compared to 'business as usual'

Catchment	Scenario	N input (%)	Leaching from fields (%)	Denitrification (%)	Terrestrial N retention (%)	N export (%)
Lepsämänjoki	Wintertime vegetation cover	0	–16	3	3	–14
	Reduced fertilization	–14	–8	–9	–15	–7
	Crop diversification	3	1	3	3	2
	Ecological set aside	–6	–5	–2	–6	–4
	ERA	–64	–28	–40	–72	–23
Yläneenjoki	Wintertime vegetation cover	0	–18	–6	6	–18
	Reduced fertilization	–11	–20	–8	–7	–21
	Crop diversification	5	5	2	4	5
	Ecological set aside	–6	–7	–3	–6	–7
	ERA	–75	–29	–11	–90	–30

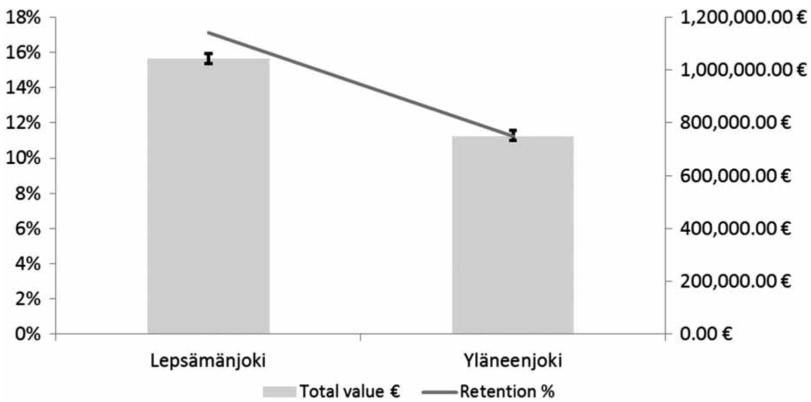

Figure 5 | Nitrogen retention and its value in the rivers.

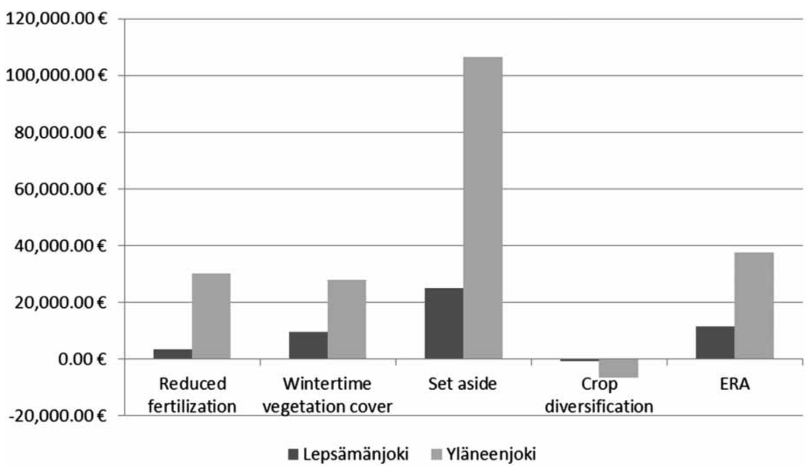

Figure 6 | Nitrogen retention in the catchment, unit value (€ km^{-2} treatment area).

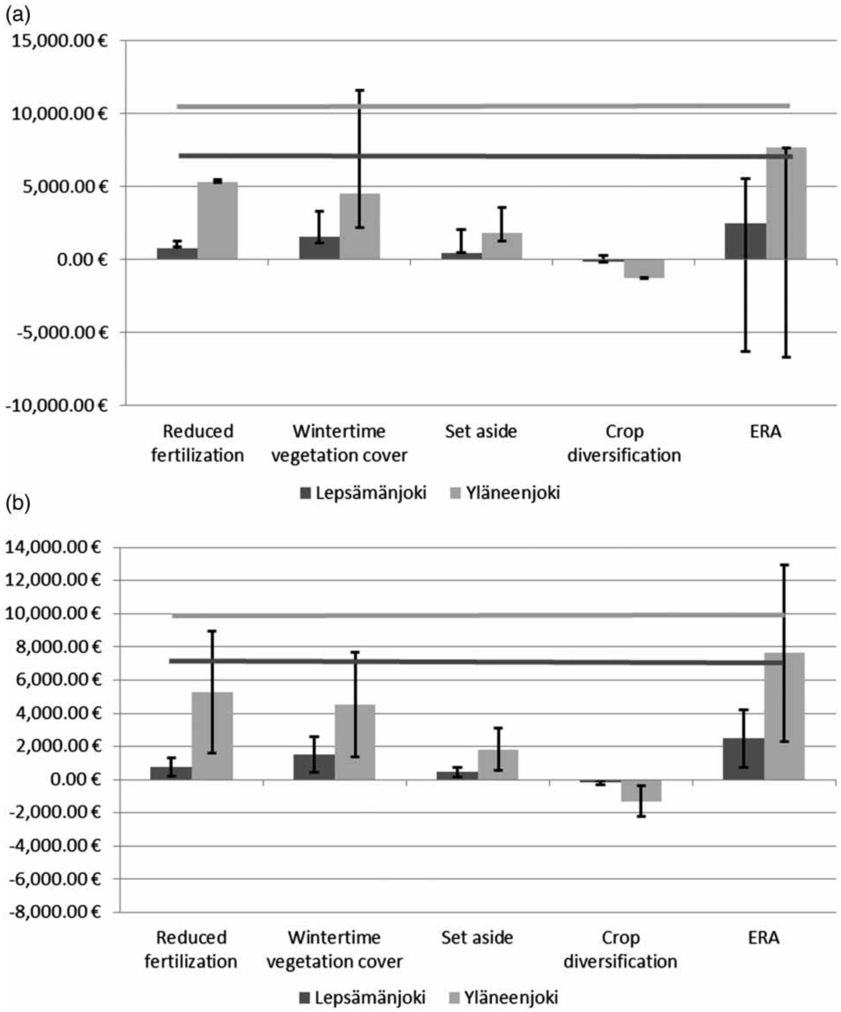

Figure 7 | Nitrogen retention in the catchments as total value (€ km^{-2} catchment area) (a) with modeling uncertainty and (b) with valuation uncertainty.

of high amounts of nutrients spread on fields in manure. 'Crop diversification' seemed to have either no value or even a negative value. In the 'business as usual' scenario spring cereals were the main crop, and according to FAEP they have relatively strict fertilization levels. When the area of other crops with higher fertilization levels increased, areal N balance and thus leaching from fields also increased.

The process based model allowed examination in more detail of the mechanisms that reduced N export from fields (Table 3). Land use/cover changed in all other scenarios except in 'reduced fertilization'. In this scenario the only reason for decreased N export was decreased N input to the fields. Manure spreading in autumn was assumed to be prohibited, which yielded a high reduction in N leaching. Manure carries organic material to the soil, which is known to have an influence on soil functioning (e.g. Mattila 2006). Further, changes in mineral fertilizer levels alone are observed to influence soil activity (Söderberg & Bååth 2004; Enwall *et al.* 2007; Piotrowska & Wilczewski 2012).

The N input was also reduced in the 'ERA farming' scenario, and in 'ecological set aside' there was assumed no fertilization at all. Also, crop composition was changed leading to a different N cycle in these land uses. In 'ERA farming' the inputs of N were manure, which was spread once in 5 year crop rotation, and N fixing crops cultivated in 4 years out of 5.

The 'crop diversification' scenario had the potential to lead to a slight increase in N retention in the terrestrial system, but at the same time it increased the area of crops with a high N balance (fertilization minus crop uptake) so the net result may be zero or even negative (Figures 6 and 7). In the 'winter time vegetation cover' scenario, N transport from the catchment was decreased only due to increased retention in the terrestrial ecosystem, or more correctly, due to decreased mineralization (Puustinen *et al.* 2010) when fields were not ploughed in autumn.

'ERA farming' and 'winter time vegetation cover' were the scenarios that reduced N leaching from fields close to the targets set in current political decisions. On the other hand, the highest model uncertainty was connected to those scenarios (Figure 7(a)). In the 'ERA farming' scenario there was even risk of increased N export from the catchment. Field studies by Nykänen (2008) on the N dynamics of red clover-grass leys and subsequent cereals showed a substantial spatial variation of all measured parameters, between and within fields and also temporally. This variation may lead to problems to control the system in models but also in real life.

None of the scenarios appeared to increase greenhouse gas emissions by substantially increasing denitrification in river or in soil. Currently, modeling of denitrification is implicitly based on literature studies. Denitrification is also largely dependent on the carbon content of the substrate (e.g. Burgin *et al.* 2013) so, to accurately model this, the already existing INCA nitrogen and carbon (Futter *et al.* 2007) models should be combined.

The cost of N retention in artificial wetlands formed the basis of the monetary valuation as they form one method to remove N from runoff waters. Artificial wetlands are supported in FAEP so they can be considered to represent N purification costs to society even though the accurate value of removed N is difficult to estimate. The valuation uncertainty was in most scenarios higher than the model uncertainty. The average cost of wintertime vegetation cover and targeted fertilization is relatively low, 50 € ha^{-1} per year (Hjerppe *et al.* 2012), but they may also cause savings for farmers as reduced purchasing of fertilizers and fuels, and saving of working time when fields are not ploughed in autumn, leading to double counting problems.

Further, in this study the monetary valuation of N retention is not supposed to be taken literally, but more as a method to equalize and to make visible the effect of retention processes in different ecosystems and in different areas. This approach is more general than the use of economical optimization methods in environmental decision making (e.g. O'Shea & Wade 2009), where the aim is to find a cost-efficient combination of water protection methods to achieve targets in water quality.

The monetary valuation of N retention allowed a comparison between the river and soil compartment, but also between different catchments. For example, in the scenario 'ecological set aside', N loading decreased by an almost equal percentage but the monetary value was clearly higher in the Yläneenjoki catchment. In general the monetary values of different scenarios were higher in the Yläneenjoki catchment than in the Lepsämänjoki catchment. That reflected both high N loading in the area due to animal husbandry, but also the lower retention percentage of the river. In the river Lepsämänjoki, the value of N retention was clearly higher than the value of retention in soil in any of the studied scenarios.

CONCLUSIONS

This study showed that the ecosystem approach has potential in evaluating the effects of different policies. 'ERA farming' and 'winter time vegetation cover' were the scenarios that reduced N leaching from fields close to targets set in current political decisions. As the target of most policies to reduce nutrient loading is set as reduction percentage or as kilograms per land area, current monitoring methods are restricted to provide only total nutrient loading at the outlet of the river basin. The ecosystem approach in combination with dynamic modeling reveals more information of ecosystem functioning than the current methods. It clearly shows some differences between ecosystem types, thus allowing the most beneficial water protection measures to catchments to be targeted. Later on, the ecosystem approach can be used to study the trade-offs of water protection measures between different ecosystem services.

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