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A novel integrated modelling framework to assess the impacts of climate and socio-economic drivers on land use and water quality

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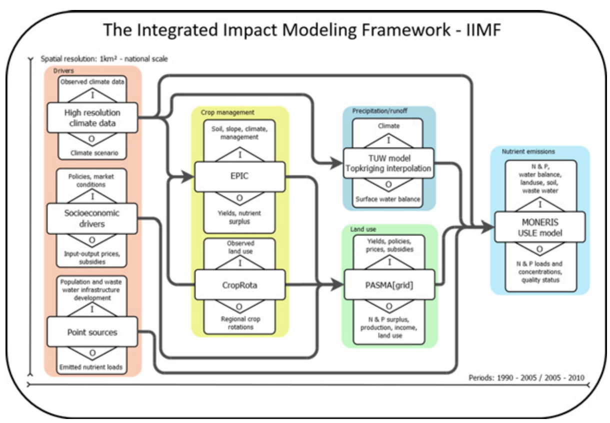
HIGHLIGHTS

- Developing an integrated impact modelling framework (IIMF) with six models.
- Application of the IIMF at various scales from 1 km pixel to the Austrian territory.
- Pollution impacts are assessed along policy-climate-agriculture-water interfaces.
- Deviations between model results and observations are assessed and discussed.
- The IIMF enables risk assessment for future water quality development.

GRAPHICAL ABSTRACT



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ABSTRACT

Changes in climatic conditions will directly affect the quality and quantity of water resources. Further on, they will affect them indirectly through adaptation in land use which ultimately influences diffuse nutrient emissions to rivers and therefore potentially the compliance with good ecological status according to the EU Water Framework Directive (WFD). We present an integrated impact modelling framework (IIMF) to track and quantify direct and indirect pollution impacts along policy-economy-climate-agriculture-water interfaces. The IIMF is applied to assess impacts of climatic and socio-economic drivers on agricultural land use (crop choices, farming practices and fertilization levels), river flows and the risk for exceedance of environmental quality standards for determination of the ecological water quality status in Austria. This article also presents model interfaces as well as validation procedures and results of single models and the IIMF with respect to observed state variables such as land use, river flow and nutrient river loads. The performance of the IIMF for calculations of river nutrient loads (120 monitoring stations) shows a Nash-Sutcliffe Efficiency of 0.73 for nitrogen and 0.51 for phosphorus. Most problematic is the modelling of phosphorus loads in the alpine catchments dominated by forests and mountainous landscape. About 63% of these catchments show a deviation between modelled and observed loads of 30% and more. In catchments dominated by agricultural production, the performance of the IIMF is much better as only 30% of cropland and 23% of permanent grassland dominated areas have a deviation of >30% between modelled

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and observed loads. As risk of exceedance of environmental quality standards is mainly recognized in catchments dominated by cropland, the IIMF is well suited for assessing the nutrient component of the WFD ecological status.

1. Introduction

Climate change is one of the major challenges of our time and adds considerable stress to the human society and environment (UNEP, 2010). A change in climate is not only restricted to a shift of seasonal weather patterns like increasing winter precipitation in Northern Europe and decreasing summer precipitation in Southern and Central Europe, but can also lead to more frequent occurrence of extreme weather events such as intense rainfall or drought (IPCC, 2007; Jentsch and Beierkuhnlein, 2008; IPCC, 2014). The most important changes in the climate system related to water resources are increases in air temperature, shifts in precipitation patterns and snow cover, and potentially an increase in the frequency of flooding and droughts (EEA, 2007). In Austria, weather station data of the last decades show a rising air temperature trend but significant changes in annual precipitation sums have not been detected in the period 1975 to 2007 (Strauss et al., 2013). For the decades to come, increasing precipitation in winter and decreasing precipitation in summer as well as increases in extreme weather events are expected (APCC, 2014). However, uncertainties and spatial heterogeneity are large, particularly in the alpine region (Gobiet et al., 2014).

Climate change has direct effects on water resources. Rising water temperatures influence biological processes and chemical conditions in surface waters, e.g. decreasing oxygen solubility, increasing growth rates of aquatic organisms and consequently increasing variability of pH-values. Since the influence of temperature and water availability is closely connected, longer dry periods leading to severe low-flow situations might affect the quality of surface waters adversely. Climate change also induces land use changes, i.e. autonomous or planned adaptation, resulting in indirect impacts on water resources. Agriculture is one of the major water consumers through either rain-fed production or irrigation, and contributes to surface and ground water pollution. Increasing yield potentials from extended vegetation periods and elevated CO_2 concentration may lead to adjustments of land cover (e.g. conversion of grassland or natural habitats to cropland, land abandonment), land use and management (e.g. choices of crops and cultivars, irrigation, fertilization, adjusted planting dates) (Olesen et al., 2011). Furthermore, climate change is accompanied with changes in socio-economic production conditions such as agricultural policy reforms and international market dynamics.

Since protecting and restoring aquatic ecosystems is a policy priority in Europe (EC, 2000), uncoordinated autonomous adaptation in agriculture can cause shortages in water supply and affects the compliance of the EU Water Framework Directive (WFD). Nutrient pollution is already considered as a global problem beyond the planetary boundaries (Steffen et al., 2015) and it is suspected that nutrient emissions will exacerbate in vulnerable European aquifers, rivers and estuaries due to climate change (Bindi and Olesen, 2010; Leclère et al., 2013).

The relationship between socio-economic conditions, climate change, agricultural production, water resources and diffuse water pollution are highly complex and require an integrated approach to assess the overall, sectoral and dissipated impacts (Dunn et al., 2012). So far, only limited information is available on the complex interactions between climate change, agriculture and water (Fallon and Betts, 2010). Using impact modelling to investigate the combination of climate change, land use and diffuse water pollution produces divergent conclusions and multiple uncertainties. Dunn et al. (2012) expressed the need for a spatially distributed approach to any large scale modelling. For this purpose, high resolution climate change data and socio-economic scenarios should be integrated in models of land use and fresh water systems for quantification of agricultural production and water resources as well as assessment of water quality. Several studies have analyzed the impacts of climate change on agricultural production (Brown et al., 2008; Fischer et al., 2005; Olesen et al., 2007) or water resources (Arnell, 2004; Bates et al., 2008; Mimikou et al., 2000; Schöner et al., 2011). A few have dealt with the linkage between agricultural production and water systems (Bindi and Olesen, 2010; Mehdi et al., 2015a; Mehdi et al., 2015b) but do not consistently combine climate change, socio-economic drivers, agricultural land use and water pollution. Though land use is considered in some modelling scenarios (e.g. Karlsson et al., 2016), agricultural land use has been rarely modelled in an integrated modelling framework combined with different climate and political scenario assumptions so far. A methodology for an integrated analysis of tradeoffs between economic and environmental indicators using bio-physical and economic models for agricultural production systems was proposed by Stoorvogel et al. (2004). A unique Australian continental model was presented by Connor et al. (2015) modelling land use change (e.g. food, carbon, water) and biodiversity ecosystem services with food price feedback. Volk et al. (2008) developed an ecological-economic modelling tool, which supports the assessment and 3-dimensional visualization of hydrological, ecological and socio-economic conditions and management effects in river basins. None of these simulation models considered climate change as integrated factor. Barthel et al. (2012) integrated climate change and socio-economic drivers into land use modelling and related nitrogen pollution of groundwater but do not consider phosphorus or surface water quality. An integration of different models combined with existing external constraints as climate change, demographic change and management practices were accomplished by Lautenbach et al. (2009) assessing impacts for the river Elbe though a direct link to climatic and socio-economic drivers was not realized within this study.

This article develops an integrated impact modelling framework (IIMF) to track and quantify direct and indirect pollution impacts along policy-economy-climate-agriculture-water interfaces in Austria. It adds important aspects to previous research by linking climatic and socio-economic boundary conditions via land use optimization and runoff-precipitation modelling to impacts on surface water quantity and quality. The IIFM models adaptation of agricultural production to climatic (e.g. temperature, precipitation) and socio-economic drivers (e.g. market prices, agri-environmental payments) and quantifies related agricultural outputs such as crop and livestock production as well as nitrogen and phosphorus emissions to surface waters, which has not been done before in integrated impact modelling. Agricultural emissions dominate pollution of surface waters in Austria (Schilling et al., 2011) and therefore significantly impact the ecological status of water bodies (BMLFUW, 2015).

The focus of this article is on the description of the IIMF and the interfaces of the single model components (Section 2) as well as the validation against observed data of single models and the IIMF (Section 3). We also quantify and discuss uncertainties relating to individual models and interface options as well as the uncertainty ranges of impacts (Sections 3 and 4). Our conclusions highlight options and procedures for the application of the IIMF in scenario studies (Section 5). A detailed scenarios assessment based on future climatic and socio-economic conditions within the IIMF will be presented in upcoming publications.

2. Material and methods

2.1. Overview of the integrated impact modelling framework (IIMF)

The IIMF has been developed in order to assess climatic and socio-economic impacts on agricultural land use, runoff and nutrient pollution of surface waters in Austria. It consists of loosely coupled models – according to the nomenclature by Antle et al. (2001) – where state or flow variables from one model are input to other models and links six independent models (see Graphical abstract and Table 1): the biophysical process model EPIC (Environmental Policy and Integrated Climate) (Williams, 1995; Izaurralde et al., 2006), the crop rotation model CropRota (Schönhart et al., 2011), the socio-economic land use optimisation model PASMA[grid] (Kirchner et al., 2016), the hydrologic rainfall-runoff model TUWmodel (Bergström, 1976; Parajka et al., 2007; Viglione and Parajka, 2014), the USLE (Universal Soil Loss Equation) (Wischmeier and Smith, 1978; Schwertmann et al., 1987) for erosion modelling, and the nutrient emission model MONERIS (Modelling Nutrient Emissions in River Systems) (Behrendt and Opitz, 1999; Venohr et al., 2009; Zessner et al., 2011).

External time dependent drivers in the IIMF are high resolution climate data (e.g. temperature, precipitation), socio-economic drivers (e.g. Common Agricultural Policy (CAP) reforms, market price forecasts), and wastewater infrastructure development (e.g. sewer distribution and level of wastewater treatment). Climate data are input to the hydrological precipitation/runoff calculations within the TUW model which simulates climate induced runoff and water availability at subcatchment level for the Austrian territory. After disaggregation by top-kriging interpolation, this contributes to the inputs of the MONERIS emission model for cases, where monitoring data are not available (missing gauges or future scenarios). Climate data also feed into EPIC to simulate inter alia crop growth and environmental impacts. EPIC results in turn feed into PASMA[grid], which optimizes agricultural land use choices according to socio-economic scenario assumptions. PASMA[grid] outputs include land use and management maps as well as nutrient surpluses. Land use information is used in USLE for soil loss calculations while nutrient surpluses and soil loss information are fed into the nutrient emission model MONERIS. The link PASMA[grid] to USLE and MONERIS is needed for reasons of consistency already for a reference status if future scenarios are analyzed and information from agricultural statistics is not available.

MONERIS is applied to assess the impact of land use on nutrient emissions and resulting concentrations in water bodies under specific hydrological conditions. Beside agricultural non-point pollution, MONERIS also takes into account emissions from waste water disposal such as wastewater treatment plant (wwtp) effluents and sewer overflows. Loads from wwtp effluents stem from a national emission inventory and sewer overflows are calculated with the MONERIS conceptual approach based on population connected to sewer systems and the build-up rate of combined sewer storage volume (Venohr et al., 2009). Finally, river concentrations of nutrient parameters modelled with MONERIS can be used to assess risk of failing to attain good ecological surface water status according to the Austrian implementation of EU-Water Framework Directive (BMFLUW, 2010).

A consistent spatial and temporal aggregation of model input and output data was a significant challenge for the development of the IIMF. Table 1 gives an overview on models and interfaces for the most important inputs and outputs. With respect to spatial integration, aggregation and dis-aggregation of results is necessary. For example, high resolution climate data at cluster level is disaggregated to 1 km grid size to feed into the USLE calculations. Results from PASMA[grid] at either municipality or NUTS-3 level (EU Nomenclature of Units for Territorial Statistics; NUTS-3 level represents 35 groups of districts in Austria; on average each NUTS-3 region consists of 22 municipalities) are disaggregated to a 1 km grid size and finally aggregated to MONERIS-sub-catchments to estimate nutrient emissions.

Table 1
Overview of models used in the integrated impact modelling framework (IIMF) including spatial scale, interface description and key citations.

Model	Spatial scale	Used for	Most important inputs	Most important outputs	Outputs used as input for model	Validation parameters	Key citations
EPIC	1 km grid resolution	Simulation of crop yields and environmental impacts	Data on soils, daily weather, slopes, altitudes, crops and crop management	Crop yields, nutrient uptake and nitrogen fixation for crops by alternative farm managements	PASMA[grid]	Crop yields	Williams, (1995); Izaurralde et al. (2006)
CropRota	Municipality	Simulation of typical crop rotations to support EPIC	Observed land use, expert-based values on crop combinations	Relative importance of particular crop rotations	EPIC	Crop rotations	Schönhart et al. (2011)
PASMA[grid]	Municipality up to NUTS-3; linear downscaling of results to 1 km grid	Optimization of land use and livestock production at regional levels	Crop and livestock yields, technical and bio-physical parameters on agricultural production, observed land use and livestock, parameters on agricultural policies, input and output prices	Land use and livestock production including management choices (e.g. tillage management) at 1 km grid, N and P surplus, amount of organic fertilizers	USLE, MONERIS	Land use (crop areas), livestock numbers, participating areas in agri-environmental programs, agricultural production value	Kirchner et al., (2016)
TUW-Model	Elevation zones of catchments (each 200 m), 277 catchments	Daily runoff simulation	Daily precipitation, air temperature	Daily runoff at outlets of 277 catchments	Top-kriging interpolation	Runoff	Bergström, (1976); Parajka et al. (2007); Viglione and Parajka, (2014)
Top-kriging interpolation	7774 river sections in Austria	Daily runoff estimation at ungauged locations	Runoff simulations from TUW-Model	Daily runoff at outlets of MONERIS catchments	MONERIS	Runoff	Skøien and Blöschl, (2007); Skøien et al. (2014); Parajka et al. (2015)
USLE	1 km grid	Soil loss calculations	Soil type, rain intensity, slope, slope length, crop distribution, farming practice	Soil loss aggregated to sub-catchment level	MONERIS		Wischmeier and Smith, (1978); Schwertmann et al. (1987)
MONERIS	367 sub-catchments in Austria	Calculation of loads and concentrations of N and P parameters in rivers	N and P surplus, runoff, land use, hydro-geology, soil type, connections to sewer systems and treatment efficiency of waste water treatment plants	Loads and concentrations of N and P parameters in rivers, risk assessment for exceeding environmental quality standards		Loads and concentrations of N and P parameters in rivers	Behrendt and Opitz, (1999); Venohr et al. (2009); Zessner et al. (2011)

With respect to temporal extension, long-term climate data are required for calibration and validation of the TUWmodel. In EPIC, impacts of crop management practices depend on a long-term climate scenario to account for climate variability in the performance. We applied EPIC to the period 1990 to 2005, which also serves as climatic reference period in this analysis.

On the contrary, PASMA[grid] relies on rather short-term socio-economic boundary conditions. Price and cost data as well as policy conditions are representative for the period 2005–2010. MONERIS employs land use results from PASMA[grid] and emission data from wastewater management from 2005 to 2010 as well. Consequently, runoff measured in stream concentrations for validation of river nutrient loads calculated with MONERIS were taken from the period 2005–2010. It is also called "validation period" as the performance of the IIMF is validated against nutrient river loads for this period. Models and their interfaces are described in detail next.

2.2. Precipitation/runoff modelling

Within the IIMF, the effects of changing climatic conditions on the regional water balance and river flows are estimated by the semi-distributed conceptual precipitation/runoff model TUWmodel (Viglione and Parajka, 2014). It simulates water balance and runoff generation on a daily time step by using precipitation, air temperature and potential evapotranspiration data as an input. The TUWmodel consists of snow, soil moisture and flow routing routines. The snow routine estimates snow accumulation and melt by a threshold temperature and the degree-day concept. The soil moisture routine represents changes in soil moisture storage and uses a non-linear function and a threshold limit to relate runoff generation and evaporation to the soil moisture state of the basin. Flow routing on the hillslopes is represented by an upper and a lower soil reservoir. The outflow from both reservoirs is routed by a triangular transfer function representing runoff routing in the streams.

The TUWmodel has 15 model parameters, which are typically calibrated against observed river flows. In this study, it is calibrated by using the SCE-UA automatic calibration procedure (Duan et al., 1992) in 277 basins in the period 1976–2010. The SCE-UA is a global optimization method for calibrating hydrologic models, which is based on an evolutionary algorithm combined with a simplex method. The calibration procedure and the setup of the SCE-UA algorithm is selected on the basis of prior analyses performed in different calibration studies in the study region (e.g. Parajka and Blöschl, 2008; Merz et al., 2011; Parajka et al., 2016). The performance of TUWmodel is evaluated by the Nash-Sutcliffe runoff model efficiency (M_E) and the volume error (V_E):

$$M_E = 1 - \frac{\sum_{i=1}^{n} (Q_{obs,i} - Q_{sim,i})^2}{\sum_{i=1}^{n} (Q_{obs,i} - \overline{Q_{obs}})^2} \quad V_E = \frac{\sum_{i=1}^{n} Q_{sim,i} - \sum_{i=1}^{n} Q_{obs,i}}{\sum_{i=1}^{n} Q_{obs,i}}$$

where $Q_{sim,i}$ is the simulated runoff on day i, $Q_{obs,i}$ is the observed runoff, $\overline{Q_{obs}}$ is the average of the observed runoff over the calibration period of n days.

The climate inputs for model calibration, i.e. time-series of daily precipitation and air temperature for elevation zones of each basin, have been obtained by external drift kriging interpolation (Merz et al., 2011). For interpolation, daily observations of precipitation at 1091 and air temperature at 212 climate stations were used. The potential evaporation is estimated by a modified Blaney-Criddle method (Parajka et al., 2005a). More details about the model structure and its application in Austria and Europe are given e.g. in Parajka et al. (2007, 2008), Viglione et al. (2013) and Ceola et al. (2015).

The TUWmodel is individually calibrated in selected basins, which allows to simulate river flows at gauged locations for any time period when model inputs (precipitation and air temperature) are given. In order to evaluate the effects of climate change at locations without direct flow observations, the hydrologic model simulations need to be transferred to ungauged sites. In this study the hydrologic model simulations of runoff at 277 basin outlets are transferred (regionalized) to the entire Austrian river network (7774 river sections) by using the top-kriging approach. Top-kriging is a geostatistical interpolation method that allows estimation of daily flows along the stream network. It combines two processes: local runoff generation, which is continuous in space, and runoff aggregation and routing along the stream network (Viglione et al., 2013). The river flow observations represent aggregates (linear averages) of local realizations of the process over an integral spatial support (such as runoff per unit area). The top-kriging approach assumes that the specific runoff from basins can be considered as a linear average of the runoff generated in sub-basins and that expected variance between observations is a function of separation distance. The interpolation weights are estimated by regularising the point variogram over the basin area (kriging support), which accounts for the nested structure and topology of the river network. Previous studies in Austria (e.g. Skøien and Blöschl, 2007; Skøien et al., 2008; Viglione et al., 2013; Parajka et al., 2015) show that the estimation of daily flows at ungauged locations by top-kriging is superior to methods that transfer hydrologic model parameters to ungauged sites. In the proposed framework, we apply a top-kriging approach to estimate regional patterns of observed and simulated daily river flows in the reference period. The river flows at the outlet stations of the MONERIS-sub-catchments serve then as an input for the estimation of nutrient emission loads by the MONERIS model.

2.3. Crop management and land use modelling

The bio-physical process model EPIC simulates processes in the soil-water-crop-atmosphere system at daily time steps for individual plots (Williams, 1995). Major model components include weather, hydrology, nutrient cycling, crop growth and crop management. In the IIMF, the EPIC simulations at 1 km grid size represent spatial stratifications of soil quality, altitude, slope (data based on e.g. BFW, 2016), and climate data. Climate data with a spatial and temporal resolution of 1 km and 1 day are provided by Strauss et al. (2013) and include minimum and maximum temperature, precipitation, solar radiation, relative humidity, and wind speed for the period 1990–2005. Strauss et al. (2013) clustered weather station data to 60 climate clusters – homogenous with respect to mean annual precipitation sums and mean annual temperatures – and applied linear regression modelling and repeated bootstrapping.

EPIC output consists of annual dry matter arable crop and grassland forage yields as well as environmental outcomes (e.g. nutrient losses). We ran EPIC for a combination of management alternatives. Management components for arable crop include typical crop rotations, three tillage options, three fertilization intensity levels (high, moderate, low), and irrigation. Due to lacking empirical data on actually applied crop rotations, we apply CropRota to model typical crop rotations for each municipality. CropRota combines crops to rotations and computes its relative area coverage in a municipality by maximizing the total agronomic value of the modelled crop rotations and by representing the observed relative shares of crops in the municipality (e.g. 50% winter wheat and 50% maize). In the IIMF, we consider 22 crops, which represent about 90% of the Austrian cropland (see Schönhart et al. (2011) for further details on CropRota). Tillage options in EPIC comprise conventional tillage, reduced tillage, and reduced tillage in combination with winter cover cropping. The tillage options mainly differ in the applied farm machinery and the crop residue on soil surface before planting (see Mitter et al. (2014) for specifications of the tillage options). The fertilization intensity levels differ by crop and are based on legal standards and policy guidelines, i.e. the Guidelines for Appropriate Fertilization (BMLFUW, 2006) and the Action Program Nitrate 2012 (BMLFUW, 2012), to meet low, medium and high crop yield potentials, respectively. The average nitrogen and phosphorus fertilizer rates are decreased by about 20% for the moderate and by about 40% for the low fertilization intensity level compared to the high fertilization intensity. The timing of fertilization is determined by fractions of total heat units a particular crop requires to reach maturity. With respect to irrigation, we assume that it is only available in combination with high fertilization intensity. Availability of irrigation water is unlimited up to the maximum annual irrigation volume of 500 mm. For permanent grassland, management decisions encompass fertilization intensities, mowing frequencies, pasture use, and irrigation. Due to its plot level structure, we do not model management options that are effective to control nutrient flows between plots. However, buffer strips are considered in PASMA[grid] with respect to their area coverage in a grid cell and their effectiveness in emission reduction is implemented in MONERIS for scenario calculations.

With the EPIC results, we calculate multi-year average crop yields and environmental outcomes for spatially explicit homogenous response units (HRU) consisting of grid cells within the same soil, slope and altitude class in a municipality as well as each of the management options. The bottom-up economic land use model PASMA[grid] is applied to choose the economically optimal crops and management variants from all available management options within an HRU in the municipality. This choice also determines the environmental outcomes such as nutrient surpluses. Furthermore, PASMA[grid] depicts optimal livestock and forestry production choices for each grid cell taking into account variable production costs for agricultural land use and livestock as well as gross margin annuities for short rotation coppice plantations and afforestation measures. Livestock production is represented at NUTS-3 level and considers 20 different livestock activities. Both, crop and livestock production is distinct for conventional and organic production. However, low intensity management and organic farming have the same fertilizer rates.

PASMA[grid] is set up as a linear optimization (LP) model, maximizing regional producer surplus (i.e. the sum of gross margins) for each NUTS-3 region separately. Optimal land use, livestock and policy choices (e.g. participation in an agri-environmental program) in the model are affected by crop and livestock yields, commodity prices, production costs, and policies (e.g. agri-environmental payments). Choices are constrained by regional endowments (land and livestock housing capacities) and feed and fertilizer balances. Duality constraints utilize mixes of crops, land uses, and livestock, which are based on observations. They shall avoid extreme corner solutions typical to LP models while still providing a reasonably large solution space. Prices are exogenously given as we can apply the small country assumption for Austria. PASMA[grid] is a static model at annual temporal resolution. In the IIMF, input data and constraints represent the period from 2005 to 2010 such that the model output is representative for this period as well.

PASMA[grid] has been applied to regional and national case studies (Schmidt et al., 2012; Kirchner et al., 2015). The validation routine of PASMA[grid] compares land use and economic model outputs to statistical land use and agricultural data at NUTS-3 up to national level combining EU integrated administration and control (IACS) data and land use surveys. Crops are aggregated to groups with similar bio-physical characteristics. Surrogate indicators are applied, when statistical data on fertilization management are lacking. Observed participation of farms in agri-environmental measures, i.e. organic farming, renunciation of agro-chemical inputs, environmentally friendly management, as well as no participation is compared to the participation modelled in PASMA[grid]. This is justified because fertilization is a major determinant of agri-environmental program participation in PASMA[grid]. Kirchner et al. (2016) have shown that the model can adequately represent land use and production choices in the agricultural sector for the reference year 2008 and in addition provide detailed sensitivity analyses and a complete mathematical formulation of the model.

2.4. Interface for land use and emission modelling

PASMA[grid] provides land use details in high spatial resolution, which impact water quality in subsequent models. Data flows to MONERIS and to the USLE model in the IIMF include i) the amount of nutrient surplus (nitrogen and phosphorus) and nitrogen from applied organic fertilizers to calculate nitrogen losses and stock changes for phosphorus as well as changes in nitrogen deposition, and ii) arable crop and tillage choices as well as the extent of permanent grassland to calculate soil sediment losses from agricultural land. Results from PASMA[grid] of an average year in the validation period 2005–2010 are disaggregated to 1 km grid cells by distributing land use and nutrient information from each HRU in a municipality equally to its grid cells.

The nutrient surplus for each grid cell is the sum of applied fertilizers, organic nitrogen fixation by legumes, and nitrogen deposition minus the nutrient uptake by crops. Nutrient surpluses are specific to crops and management intensities. Fertilizer quantities are modelled at NUTS-3 level resulting from the crop and crop management choices in PASMA[grid], where crop nutrient demand at HRU and municipality level is aggregated to the NUTS-3 level and balanced to the supply by organic and mineral fertilizers. Organic fertilizers result from livestock production choices and feed residues in each NUTS-3 region, while mineral fertilizers are purchased on markets in PASMA[grid]. Both, organic and mineral fertilizers are adjusted for losses from nutrient emissions during storage, transport, and application. Extensive permanent grassland, such as alpine meadows, does not receive fertilizers except excretion from grazing livestock during the grazing period. Parameters for organic nitrogen fixation on grassland result from the literature, while fixation of legumes on cropland is based on EPIC results. Data for nitrogen deposition is derived from EMEP (2015). Nutrient uptake on permanent grasslands is based on expert and literature values (LFL, 2008) due to large uncertainties in grassland modelling. Uptakes on cropland result from EPIC. Finally, we calibrate the location and management specific EPIC results on nitrogen surplus to results from the national nitrogen balance by Thaler et al. (2014). This procedure takes account of the heterogeneity in bio-physical production conditions and crop management practices and assures consistency with the highly aggregated nitrogen surplus values applied in MONERIS.

We take the significance of the nitrogen surplus on emission estimations and the uncertainties of surplus calculations into account by considering two variants for emission modelling. Besides PASMA[grid] nitrogen surplus results (here “PASMA”), “OECD” estimates are based on the OECD method (Parris, 1998; OECD, 2013). It requires statistical data on agricultural production and land use as well as coefficients from agricultural handbooks (e.g. nutrient requirements and uptake of crops, N-fixation of crops; LFL, 2008; Fachbeirat für Bodenfruchtbarkeit und Bodenschutz, 2006). The long-term nitrogen surplus is the PASMA[grid] nitrogen surplus multiplied with the ratio of long-term to current nitrogen surpluses already existing in the adapted MONERIS version by Gabriel et al. (2011).

2.5. Emission modelling

The role of MONERIS (in combination with USLE) in the IIMF is to transfer land use signals as changes of nutrient surpluses, land cover, and crop categories into nutrient emission loads and in stream concentrations. MONERIS is well established in the field of emission modelling and several publications have shown its ability to reproduce regional differences in emissions and river loads (Venohr et al., 2011; Zessner et al., 2011) as well as temporal developments of nutrient pollution in catchments at a large (Behrendt et al., 2005; van Gils et al., 2005) and medium scale (Zessner et al., 2016). Furthermore, a model comparison conducted in the EU-project EUROHARP showed that the MONERIS model delivered more balanced results as compared to other models (Kronvang et al., 2009) and showed a large spectrum of potential applications with respect to landscape and climate (Schoumans et al., 2009).

Gabriel et al. (2011) estimated nitrogen and phosphorus emission loads via diffuse and point sources, phosphorus retention, nitrogen denitrification and hence nutrient loads in Austrian rivers using the empirical model MONERIS for the period 2002–2006. As MONERIS was originally developed for German river systems, Gabriel et al. (2011) advanced the MONERIS version 2.14 by implementing several adaptions for its application in Austria, which are summarized in Zessner et al. (2011) and Gabriel et al. (2011). Adaptions include modification of coefficients for denitrification in the sub-surface of mountainous areas, suspended solids exports form glaciers and P concentrations in suspended solids from mountainous areas as well as the implementation of a module to calculated river concentrations of nutrient parameters NO_3-N and PO_4-P as 90%-percentiles.

In the current MONERIS version, Austria is subdivided into 367 catchment areas with an average size of about 200 km^2. Nutrient emission loads are calculated for each catchment area via different diffuse (groundwater, erosion, surface runoff, tile drainage, deposition, combined sewer overflow, rainwater sewers disposal systems of population not connected to sewer systems) and point pathways (wastewater treatment plant effluents). River loads are modelled by summing up the different pathway emission loads to total emission loads and reducing them by an empirical retention factor. Subsequently, river concentrations are derived from the river loads using flow data. The calculated results are compared to measured river concentrations and loads to validate the accuracy of the MONERIS results and to identify uncertainties.

Input data, which do not change over time like the size of the (sub-)catchments and flow routing for each catchment as well as hydrogeological data, soil data, slope or extent of tile drainages originate from the MONERIS version from Gabriel et al. (2011). For the assessment of nutrient emissions corresponding to actual conditions all time dependent input data into MONERIS including river flows as well as observed nutrient concentrations and loads were derived from following sources for the validation period 2005–2010:

Population originates from the census provided by Statistic Austria (2006) at a 1 km grid resolution for the year 2006. The census data were aggregated to the MONERIS catchment areas. Emission loads via urban areas are based on inhabitants connected to sewer systems estimated by Fenzl and Gruber (2011), inhabitants connected to wastewater treatments plants via sewer systems (Überreiter and Schwaiger, 2014) and the area proportion of combined sewer systems (Clara et al., 2014). Information about nutrient emission loads and wastewater discharge into the river system is taken from the Austrian data base EMREG (2015) for 638 municipal wastewater treatment plants with ≥ 2000 pe (pe: population equivalents). The discharges were allocated to the MONERIS catchments by their geographical position and adjusted using orthophotos (basemap.at, 2015). The European Monitoring and Evaluation Program (EMEP, 2015) provides modelled air concentrations and depositions for NH_3 and NO_x within a grid scale of 50 km. Area-weighted means of NH_3 and NO_x deposition rates over the years 2005 to 2010 were disaggregated to each catchment area.

Further time dependent input parameters for MONERIS are provided by the PASMA[grid] model. They include the extent of cropland and permanent grassland, crop distributions as well as nitrogen and phosphorus surplus on agricultural land. PASMA[grid] delivers crop distribution and crop management practices (tillage type) to calculating soil loss. PASMA[grid] outputs are provided for each grid cell and aggregated to MONERIS sub-catchments. As MONERIS divides the agricultural land in five categories depending on the slope, the PASMA[grid] data are categorized accordingly.

Soil loss from agricultural land is an important input in MONERIS emission modelling. It has been calculated according to the USLE approach (Wischmeier and Smith, 1978; Schwertmann et al., 1987). The calculation of the rainfall-runoff erosivity factor (R) is based on the linear relationship between long-term annual R and annual precipitation (Strauß et al., 1995). For this purpose, a 1 km grid of mean annual precipitation has been employed (Parajka et al., 2005b). The USLE factors soil erodibility (K), slope length (L), slope steepness (S) and cover-management (C) have been derived from a database related to the Integrated Administration and Control System (IACS) of the European Commission (Hofer et al., 2014). This database contains detailed information (size, slope, cultivated crops etc.) on the vast majority of agricultural fields in Austria on an annual basis to estimate USLE factors (wpa and BAW, 2009). It was possible to calculate S separately for cropland and permanent grassland from the area-weighted mean slopes of the corresponding field plots of each cell according to McCool et al. (1987). The mean grid cell values for K and L could only be obtained by disaggregating values of the database at municipality level which does not differentiate between cropland and permanent grassland. Finally, area-weighted mean soil loss per grid cell has been determined utilizing the reference crop C values of the before mentioned database in combination with the crop distribution output of PASMA[grid]. P has been globally assumed as one (no contouring present) in this process. A summary of the USLE factors applied in all grid cells can be found in Table A1 of the Appendix.

River discharges at the outlets of the 367 sub-catchments considered in MONERIS were derived by top-kriging as described in Parajka et al. (2015) (see Section 2.2). Precipitation and evapotranspiration for each 1 km grid have been provided by Parajka et al. (2015) as well. The data were aggregated to MONERIS catchment areas. For current mean yearly precipitation, summer half-yearly precipitation (April to September) and evapotranspiration were averaged over the years 2005 to 2010. The averages over the years 1990 to 2005 were calculated for the long-term yearly and summer half-yearly precipitation.

For the validation of calculated river loads we used observed instream river loads and concentrations for dissolved inorganic nitrogen (DIN = NO_3-N + NO_2-N + NH_4-N), orthophosphate phosphorus (PO_4-P) and total phosphorus (TP) for the period 2005–2010. Measured concentrations originate from two Austrian monitoring programs for water quality in rivers (GZÜV, 2015; AIM, 2015) and monitored discharges are available at the Hydrographical Service of Austria (HZB, 2014). The discharge and water quality monitoring stations were allocated to the outflow of MONERIS catchment areas. Nutrient loads were calculated in accordance to the methodology agreed upon by the members of the International Commission for Protection of the Danube River (ICPDR, 2001), where monthly loads are calculated by mean concentrations and flows of each month and summed up to yearly loads. Additionally, in stream measured loads were only calculated if data existed for at least nine months per year and for two years within the period 2005–2010. Hence, for 122 MONERIS catchment areas the measured DIN load and for 121 catchment areas the measured TP load could be calculated as annual means. Further, mean values and the 90% percentiles of measured in stream concentrations for PO_4-P, TP and DIN were calculated. Data of the water quality monitoring program had to meet the following criteria for this calculation: For the substances at least 24 measured concentrations within the period should exist. Furthermore, at least half of these measured concentrations should be above the limit of detection and quantification.

The model performance of MONERIS is tested by comparison of modelled nutrient loads against nutrient loads from observations. The tests include the correlation coefficient r^2, the steepness of the linear regression and the Nash-Sutcliffe model efficiency (ME, see Section 2.2). With respect to steepness of linear regression, the best model performance would be 1 as this indicates no systematic deviation between modelled and observed loads. Additionally, the share of catchments is indicated for which the deviation between modelled and observed river loads is >30%.

3. Results

The presentation of results is structured according to the main components of the IIMF, i.e. precipitation/runoff modelling (Section 3.1), crop management and land use modelling (Section 3.2) and emission modelling (Section 3.3). The main focus of this chapter is the presentation of both individual model and the IIMF performance. The latter is ultimately reflected in the model performance of MONERIS as its output is affected by the other IIMF-components.

3.1. Precipitation/runoff modelling

The performance of TUWmodel in the calibration period 1976–2010 is presented in Fig. 1. This figure shows cumulative distribution functions of the Nash-Sutcliffe runoff model efficiency (ME, left panel) and volume error (VE, right panel) in 277 Austrian river basins. ME varies between 0.4 and 0.9, the median ME is 0.71, which indicates a good regional agreement between observed and simulated daily runoff in Austria. The median VE is −1.5% and the absolute VE of 85% of the river basins is <5%, which also indicates an accurate simulation of runoff volumes and an unbiased model calibration in most of the selected river basins.

Results indicate that there are regional differences in the runoff model performance (compare Fig. A1 in the Appendix). Runoff is more accurately simulated in wetter alpine basins with dominant snow accumulation and melt regime. In drier lowland basins, the runoff generation is highly nonlinear and hence less accurately simulated by a conceptual hydrologic model. This pattern is consistent with previous regional assessments (Parajka et al., 2005a).

Daily model simulations at 277 gauged locations are used to estimate runoff over the entire river network by using top-kriging interpolation. As an example, Fig. 2 shows the spatial variability of long-term mean annual runoff (MAR) in Austria. Such regional patterns of river flows are used as an input for the estimation of nutrient emission loads and instream concentrations by MONERIS.

3.2. Crop management and land use modelling

A comparison of the two major land use categories cropland and permanent grassland at NUTS-3 level shows a very good fit between model results and observations. It results from a low degree of freedom with respect to land cover choices in PASMA[grid], which are justified by legal constraints and rules on subsidies. Fig. 3 compares model results on crop choices to observations. There is considerable degree of freedom in PASMA[grid] to choose the most profitable crops in each NUTS-3 region. Consequently, larger deviations emerge between model results and observations but major crop categories such as grains and maize are reproduced well in most regions. In general, PASMA[grid] overestimates the extent of protein crops and set-aside land (not presented in the figure) and slightly underestimates grains, maize and temporary grasslands.

A major model output with respect to the IIMF interface is the production intensity – mainly represented by fertilization levels. Fig. 4 compares participation of modelled and observed agricultural areas under different agri-environmental measures. While the area for organic farming is well reproduced, other categories deviate from observations. In general, PASMA[grid] overestimates participation in the agri-environmental program. Category "no participation" represents area that does not participate in any of the other three measures.

The fertilization intensity is further validated by comparing PASMA[grid] results with calculations based on the OECD method aggregated to the whole Austrian territory (see Section 2.4). Both methods show good agreement to each other with respect to organic nitrogen fixation by legumes and nutrient uptake. However, slightly higher mineral fertilizer levels and organic nitrogen fixation on cropland in PASMA[grid] lead to higher nitrogen input levels and a corresponding higher nitrogen surplus (see Appendix, Fig. A2).

3.3. Emission modelling

A comparison of the modelled DIN loads between the MONERIS variants "OECD" and "PASMA" to observed river loads for the validation period 2005–2010 is shown in Fig. 5. The catchments are marked differently depending on their dominant land use. The division depends on the share of intensive agricultural land use (agricultural area without alpine pastures and meadows) on the total catchment area and the share of permanent grasslands on total intensive agricultural land in the catchment. Additionally, low impact from wastewater treatment plant (WWTP) effluents is required for identification of the domination of a specific land use. If no clear allocation to a land use type is possible and WWTP influence is not negligible, catchments are marked as "no clear domination". Table 2 presents a summary of the criteria for subdivision of the catchments.

With correlation coefficient r^2 of 0.74, a slope of the regression line of 0.89 and a Nash-Sutcliffe efficiency of 0.73 for the modelled area specific DIN river loads compared to observed ones, the model performance of variant "OECD" is better than for the variant "PASMA" ($r^2 = 0.65$, slope = 0.75, and Nash-Sutcliffe efficiency = 0.6). Further, in the variant "OECD" only 30% of the catchments show deviations of >30% of modelled area specific river loads as compared to the observed area specific river loads. In the variant "PASMA" this is the case for 40% of the catchments. Variant "PASMA" shows a systematic underestimation for catchments dominated by agricultural land (either dominated by permanent grassland or cropland) for catchments with observed DIN-loads >10 kg ha^{-1} y^{-1}. This is less pronounced in the variant "OECD". On the contrary, at observed loads around 5 kg ha^{-1} y^{-1} the modelled DIN-loads based on PASMA[grid] results tend to be higher than the ones based on "OECD". Apart from that, there is no apparent impact of the categories on model performance.

With respect to nitrogen, the surplus in soils due to agricultural management and deposition from the air is an important controlling factor of nitrogen inputs into rivers. Nevertheless, denitrification in the soil subsurface system can predominate this impact in conditions favorable for subsurface denitrification (e.g. low groundwater recharge rates leading to high nitrate concentrations, heavy soils and porous

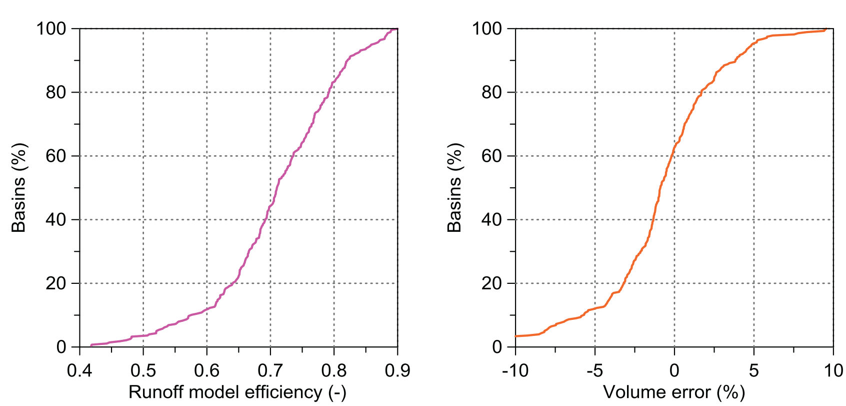

Fig. 1. Cumulative distribution function of ME (left panel) and VE (right panel) obtained for selected 277 Austrian river basins in the calibration period 1976–2010.

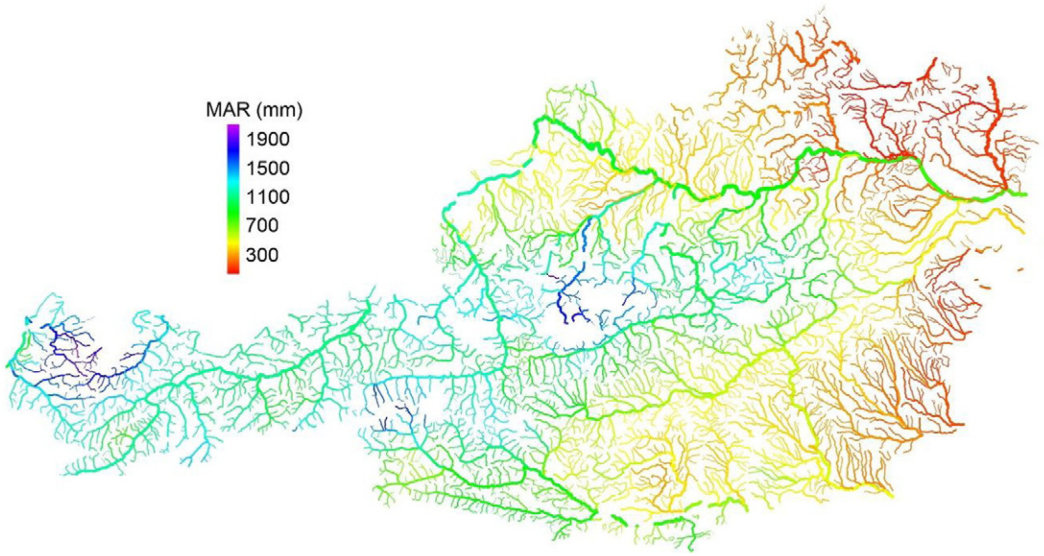

Fig. 2. Long-term mean annual runoff (MAR, 1976–2010) in Austria, interpolated by using top-kriging.

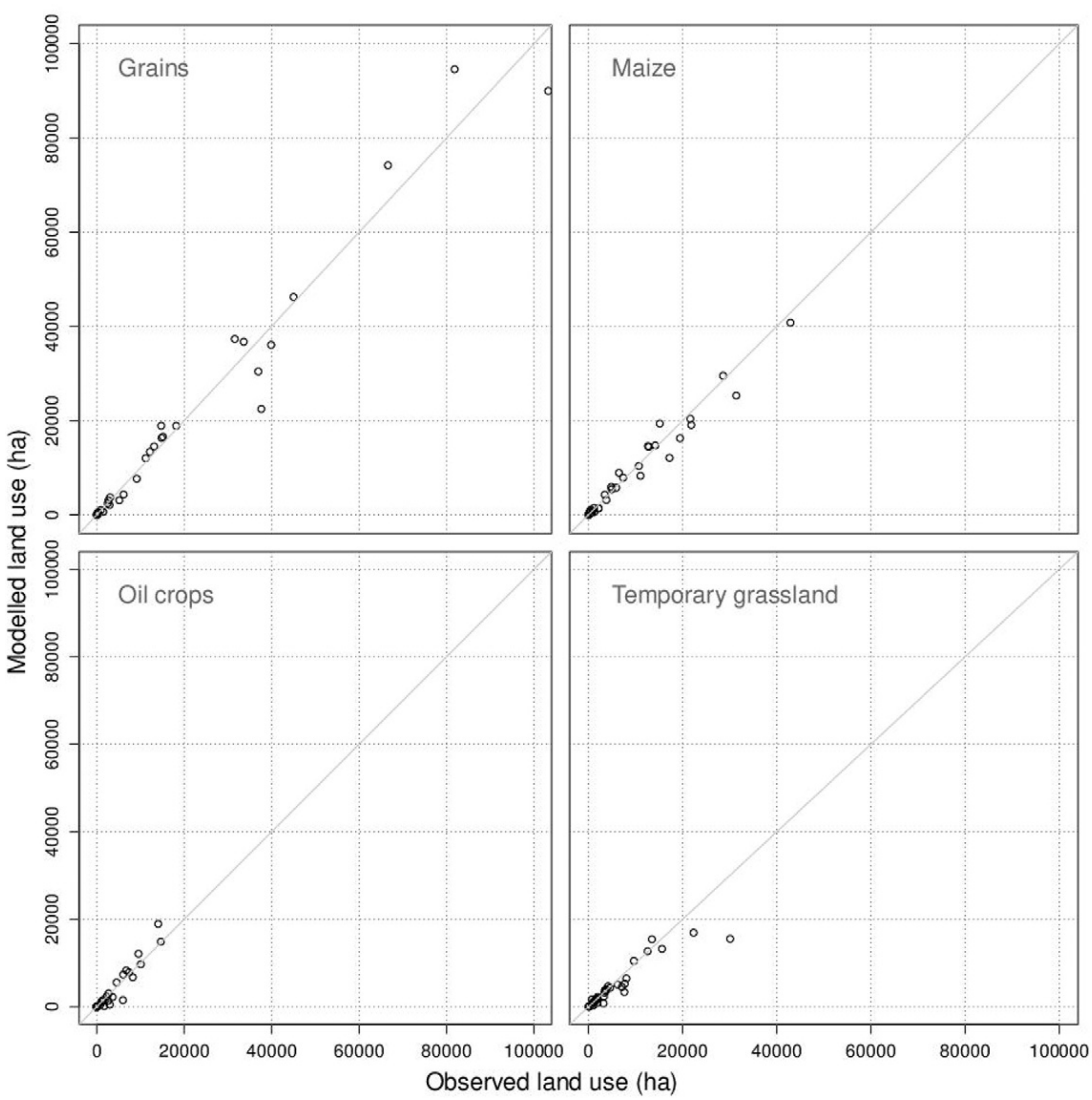

Fig. 3. Comparison between selected modelled and observed crop categories at NUTS-3 level (in ha; $N = 35$).

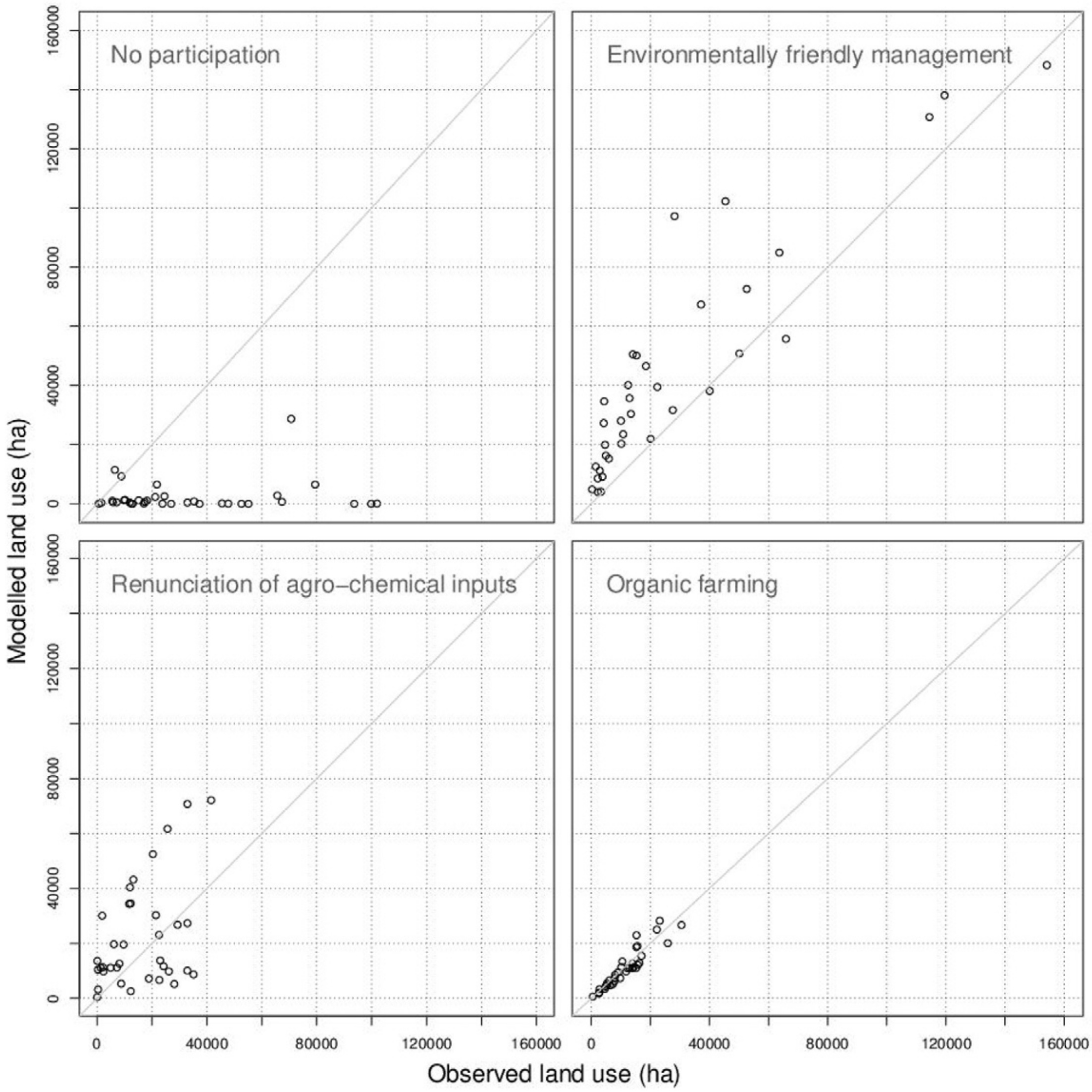

Fig. 4. Comparison between modelled and observed participation in selected agri-environmental measures at NUTS-3 level (in ha; $N = 35$). Category "no participation" represents area that does not participate in any of the other three measures.

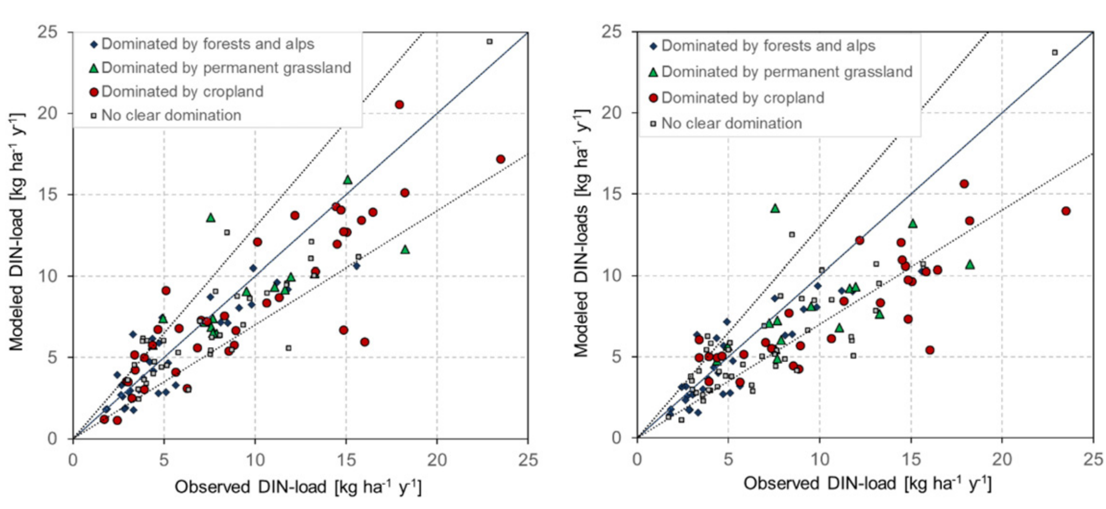

Fig. 5. Observed versus modelled area specific DIN loads for MONERIS variant "OECD" (left) and "PASMA" (right) for catchments in different categories; the full grey line indicates a slope of 1 of the regression line between observed and modelled loads; the dotted lines show a deviation of modelled loads from observed ones of 30%.

Table 2
Criteria for subdivision of catchments according to the dominating land use category.

Category	Share of WWTP effluents on total river flow*	Share of intensive agricultural land on total catchment area	Share of permanent grassland on intensive agricultural land
Dominated by forests and alps	<5%	<10%	
Dominated by permanent grassland	<5%	>20%	and >60%
Dominated by cropland	<5%	>20%	and <40%
No clear domination	>5%	and 10–20%	or 40–60%

* wastewater treatment plant.

aquifers with low permeability) (Zessner et al., 2005). The nitrogen surpluses in catchments dominated by cropland are clearly higher than in other catchments (Table 3). This is not reflected in the observed and modelled river loads (Fig. 5). Area specific river loads in catchments dominated by cropland are not generally higher than in other catchments. On the contrary, NO_3-N concentrations are highest in catchments dominated by croplands (Table 3). In Austria, NO_3-N environmental quality standards (EQS) are defined as 90% percentiles (c90) in assessing the ecological status of rivers. Determination of EQS is site specific and varies across Austrian rivers between 3 and 7 mg L^{-1} (c90). Exceedance is observed predominantly in catchments dominated by cropland. This is well represented in the modelling results (Table 3).

The comparison of modelled to observed river loads for TP can be found in Fig. A3 of the Appendix. Only variant "PASMA" has been calculated as differences in phosphorus surplus between "OECD" and "PASMA" are not decisive for the model outcome. The model performance shows a correlation coefficient r^2 of 0.82, a slope of the regression line of 0.60 and a Nash-Sutcliffe efficiency of 0.75 for the modelled area specific TP river loads as compared to observed ones. These results mainly depend on an alpine catchment with a very high area specific load resulting from glaciers. Without this catchment the model performance is lower ($r^2 = 0.57$, slope $= 0.92$, and Nash-Sutcliffe efficiency $= 0.51$). A deviation of modelled area specific loads from observed ones of >30% can be found in 48% of the catchments. This deviation predominantly appears in catchments dominated by forests and alps (in 63% of catchments). In case of catchments dominated by croplands, 30% of the modelled specific river loads deviate >30% from the observed ones. Hence, model performance for catchments dominated by croplands is much better than those for alpine dominated catchments. Nevertheless, in 3 out of 27 cases of this category, high deviations between modelled and observed loads with a factor of >2 are recognized. The model performance is best in catchments dominated by permanent grassland with 23% of catchments having a deviation between modelled and observed specific loads of >30%.

Apart from few catchments where wastewater emissions are dominating phosphorus emissions, inputs from erosion are the main pathways for phosphorus emissions into Austrian rivers (Schilling et al., 2011). Area specific soil losses generally are highest in catchments dominated by cropland. In some catchments dominated by forests and alps, an even higher area specific erosion is identified, which leads to very high phosphorus loads (up to 12 kg ha^{-1} y^{-1} in one specific case; Fig. A3 and Table 4). As in these extreme cases, phosphorus is almost entirely in particulate and hardly in soluble form (apatite), low PO_4-P concentrations are measured in rivers with catchments dominated by forests and alps. Site specific EQS for PO_4-P to support the determination of the ecological status of rivers vary between 0.015 (central alps) and 0.2 (eastern lowlands) mg L^{-1} (c90) in Austria. Exceedance is measured predominantly in catchments dominated by cropland or in catchments with no clear domination, in cases where emissions from wastewater treatment plants and erosion from croplands play an important role. In case of emissions from erosion, particulate phosphorus emissions contribute to PO_4-P concentrations after resolution of particulate P-forms into orthophosphate. Hence, PO_4-P concentrations are well represented in the modelling results across different types of catchments (Table 4). However, it is not adequately represented by the model in single cases with extreme exceedance of EQS. This can be seen from the relation between c90 river concentrations of PO_4-P and EQS which in catchments dominated by cropland has its maximum at 4.3 for observed c90 values while for model results the maximum is 2.5 (Table 4).

The model results are also used for risk assessment of exceedance of EQS. Fig. 6 illustrates the regional distribution of the risk for exceedance of type specific NO_3-N- and PO_4-P-EQS. A share of modelled c90 values to EQS of <0.7 for NO_3-N and PO_4-P is indicated as no risk of exceedance of EQS. Shares of 0.7–1.3 for one or both of the parameters and no share of >1.3 are indicated as potential risk of exceedance. Shares of >1.3 for one of the parameters are indicated as risk of exceedance of EQS. Risk of exceedance is almost always provoked by high PO_4-P concentrations (c90), indicating that Austrian catchments are especially vulnerable to phosphorus pollution with respect to their ecological status. Modelling results for NO_3-N and PO_4-P are compared to the risk assessment of the Austrian government for the implementation of the EU Water Framework Directive (EU-WFD) (BMLFUW, 2009). This assessment is based on monitoring data of river concentrations and where such data are absent, on expert judgement. Both assessments show a high degree of agreement.

4. Discussion

The main purpose of this article is to present an IIMF, which is able to quantify direct and indirect impacts on water quality along the policy-

Table 3
Nitrogen surplus of total catchment area; observed area specific river loads; 90% percentiles of measured NO_3-N concentrations (c90); relation between c90 in rivers and site specific NO_3-N environmental quality standards (EQS) for measured (mea.) and modelled (mod.) river concentrations; for different categories of catchments.

Category	N-surplus [kg ha^{-1} y^{-1}] mean (min-max)	DIN river loads [kg ha^{-1} y^{-1}] mean (min-max)	NO_3-N, C90 [mg L^{-1}] mean (min-max)	NO_3-N, mea. C90 to EQS (−) mean (min-max)	DIN, mod. C90 to EQS (−) mean (min-max)
Dominated by forests and alps	10 (5–20)	5 (2–16)	0.9 (0.2–2.5)	0.2 (0.1–0.4)	0.2 (0.0–0.3)
Dominated by permanent grassland	20 (10 − 30)	10 (4–18)	1.8 (0.7–3.8)	0.4 (0.2–0.7)	0.4 (0.1–0.8)
Dominated by cropland	30 (20–50)	11 (3 − 22)	3.9 (1.6–8.2)	0.8 (0.4–1.5)	0.7 (0.3–1.1)
No clear domination	20 (10 − 30)	7 (2 − 23)	2.2 (0.7–7.1)	0.4 (0.1–1.3)	0.4 (0.1–1.0)

Table 4
Soil loss of total catchment area; observed area specific river loads; 90% percentiles of measured PO_4-P concentrations (C90); relation between C90 in rivers and site specific PO_4-P environmental quality standards (EQS) for measured (mea.) and modelled (mod.) river concentrations; for different categories of catchments.

Category	Soil loss [t ha^{-1} y^{-1}] mean (min-max)	TP river loads [kg ha^{-1} y^{-1}] mean (min-max)	PO_4-P, C90 [mg L^{-1}] mean (min-max)	PO_4-P, mea. C90 to EQS [−] mean (min-max)	PO_4-P, mod. C90 to EQS [−] mean (min-max)
Dominated by forests and alps	0.6 (0.0–8.8)	0.7 (0.0–11)	0.02 (0.01–0.04)	0.5 (0.0–0-9)	0.5 (0.1–1.2)
Dominated by permanent grassland	0.5 (0.0–1.4)	0.2 (0.1–0.4)	0.03 (0.01–0.09)	0.4 (0.1–1.1)	0.6 (0.2–1.7)
Dominated by cropland	2 (0.3–5.6)	0.4 (0.1–1.0)	0.08 (0.01–0.22)	1.2 (0.1–4.3)	1.2 (0.3–2.5)
No clear domination	0.6 (0.1–1.6)	0.4 (0.1–1.6)	0.06 (0.00–0.24)	0.7 (0.1–2.7)	0.7 (0.2–3.2)

economy-climate-agriculture-water nexus. Here, we discuss methodological achievements and challenges, restrictions of the IIMF and its to support policy and climate change scenario assessments.

The TUWmodel transforms the climate (precipitation, temperature) signal into river flow and its runoff components. These are used in MONERIS for emission modelling as well as for calculations of river loads and concentrations. TUWmodel results in MONERIS are especially required for catchments where flow measurements do not exist as well as for scenario calculations. Several studies evaluate the runoff model performance and uncertainty of TUWmodel simulations in Austria (Parajka et al., 2005a, 2007; Merz et al., 2011; Parajka et al., 2016). The runoff model performance in the analyzed period 2005–2010 is very similar to the previous assessments in Austria and almost identical to that obtained for the entire calibration period (i.e. 1976–2010) and forms a well-established basis for IIMF calculations.

PASMA[grid] fulfills two major roles in the IIMF: (i) it provides outputs where observations are insufficient in the reference situation and (ii) it provides ex-ante scenario results on land use. With respect to (i) spatial resolution of land use data is insufficient for some management alternatives such as fertilization intensity. Bottom-up economic land use models can fill such gap. Here, PASMA[grid] provides efficient land use and livestock choices from an economic perspective. Land management results are driven by bio-physical yield potentials modelled in EPIC, variable production costs (e.g. fertilizer prices), resource endowments (e.g. land, livestock housing capacities), or agricultural policies (e.g. agri-environmental payments). Uncertainties are inherent to several input parameters and model assumptions though. Even a high spatial resolution of 1 km grid size leads to aggregation biases with respect to soil quality or slope. Calibration of all EPIC outputs to all bio-physical production conditions and management variants is impossible due to the large number of alternatives and lacking observations. Experiences in bio-physical modelling, calibration for individual alternatives and sites, and validation with statistical data and stakeholder experiences are options to limit uncertainties with respect to bio-physical model output. Data on management variants in PASMA[grid] also include production cost estimates. Expert-based calculated cost estimates for each management alternative shall represent an average situation, but likely deviate from individual farming situations. This can be a reason for the deviation between observed and modelled participation in agri-environmental measures. Calibration techniques on production costs, such as positive mathematical programming (e.g. Cortignani and Severini, 2012), are available for economic models in agriculture. Although helpful in many situations, they require observational data, need assumptions on cost functions, and can be demanding with respect to computational requirements. Above all, the assumption on constrained profit maximizing behavior of land users is well established in the literature (e.g. Janssen and van Ittersum, 2007) but usually does not represent all land use and livestock choices and restrictions. It is one of several simplifications of real-world decision making required to establish a quantitative land use model.

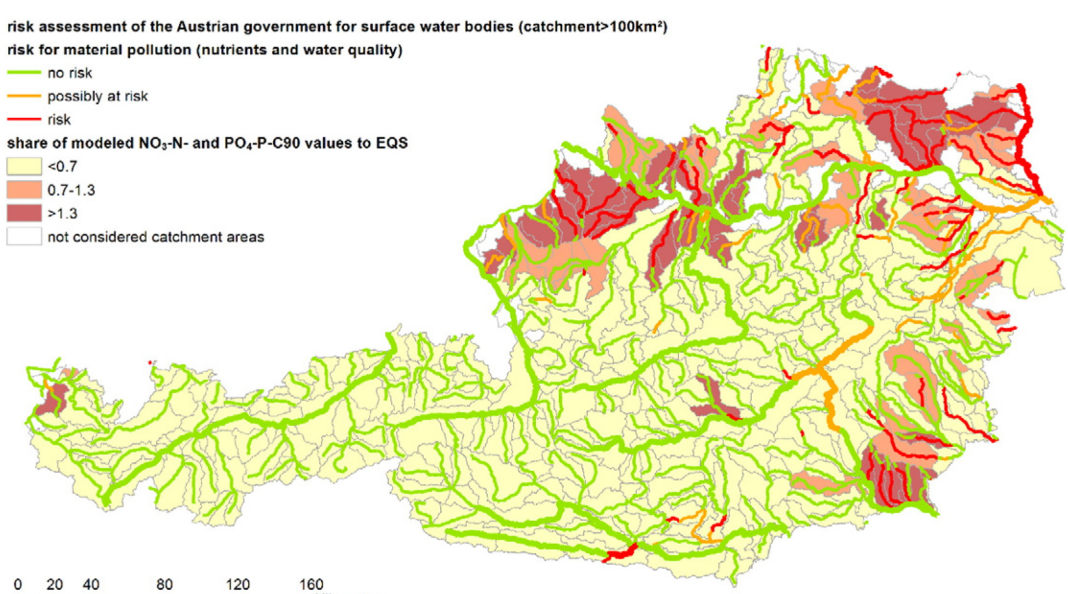

Fig. 6. Risk for exceedance of type-specific target values for NO_3-N and PO_4-P (national implementation of EU-WFD), model results as compared to official national risk assessment.

Our investigations confirm the capability of MONERIS in modelling river loads and concentrations for medium to large scale catchments with varying regional boundary conditions with respect to crop cover, land use intensity, air deposition, wastewater management, hydrogeology and hydrology. The model performance decreases slightly if PASMA[grid] outputs for nitrogen surpluses on agricultural lands are used as input data instead of surpluses derived from statistical data based on the OECD method. Main problems with respect to model performance are related to phosphorus inputs from mountainous areas, despite the fact that MONERIS has been updated by Zessner et al. (2011) specifically in this respect. Especially, phosphorus inputs from glaciers are extremely high. These inputs dominate in regions with a significant share of glaciers and lead to extremely heterogeneous sediment and phosphorus exports. Phosphorus input from mountainous regions stems from weathering of rocks containing apatite, which is hardly soluble under surface water conditions. This is demonstrated by very low PO_4-P concentrations in these rivers. Therefore, the shortcoming of the model in catchments dominated by glaciers does not impact risk assessment with respect to achieving EQS for PO_4-P in surface waters. With respect to future scenario investigations with the IIMF, catchments dominated by agriculture are of specific interest for which model performance is good in most cases at the scale of catchments with an average size of 200 km^2. Therefore, a regional assessment of the risk of failing EQS is possible. Nevertheless, single outliers show that some local peculiarities are not covered by the model.

The stand-alone models in the IIMF are specialized on and designed for particular system components, such as bio-physical crop growth processes, economic land use choices, or land use impacts on water quality. Their explanatory power results from detailed systems understanding and long-term model development. The IIMF integrates such models to utilize the strengths while reducing costs of model development. This integration is a source of uncertainty. Major parameters on nutrient surpluses are modelled in EPIC and required in MONERIS. Two interfaces are required to transfer these parameters via PASMA[grid]. For example, we processed EPIC results to MONERIS parameters to both i) maintain the data and parameter consistency in MONERIS, which is adapted to Austrian conditions and ii) to keep the spatial and management heterogeneity provided by EPIC. However, any processing of single EPIC results also impacts the overall consistency of the EPIC/PASMA[grid] model interface. With respect to aggregation and dis-aggregation biases, livestock is modelled at NUTS-3 level in PASMA[grid], but organic fertilizer inputs are required at grid level for calculation of mineral fertilizer requirements. Disaggregation of organic fertilizers according to nutrient uptake in each 1 km grid cell can lead to spatial biases of nutrient surpluses in heterogeneous regions with uneven livestock distributions. The choice of NUTS-3 levels for optimization in PASMA[grid], i.e. rather small spatial aggregates, is a compromise between data availability particularly in spatial resolution and modelling assumptions.

We emphasize that all relevant aspects along the policy-economy-climate-agriculture-water interfaces are considered in the IIMF in a way that makes it possible to integrate assumptions on climate and policy scenarios into the calculation of nutrient pollution impacts. EPIC and TUWmodel are driven by climate signals (e.g. geo-referenced time series on precipitation and air temperature) and PASMA[grid] by economic and policy incentives (e.g. prices, costs, policy payments and legislative rules) which are transformed via MONERIS into river loads and concentrations. All models and interfaces are built upon well-established knowledge from scientific literature and these models are able to reproduce decisive environmental state variables from monitoring to a satisfactory extent. This is demonstrated with river runoff for the TUWmodel and with distribution of crop cultivation for EPIC/PASMA[grid]. River loads, concentration and risk assessment of not achieving EQS for N- and P-parameters are used in the validation of MONERIS and the IIMF.

Nevertheless, uncertainties of the assessment are obvious throughout the model chain. A full sensitivity analysis of all model components in the IIMF is currently not feasible due to high number of input variables and model coefficients as well as lacking of automation of model interfaces. However, two procedures will be considered for the forthcoming scenario calculation and assessment to take uncertainty into account: (a) comparison of model results using significantly different variants for the interfaces between models, and (b) qualitative assessments of the statements derived from model outcomes based on experiences from different variants and expert judgement.

5. Conclusions

We presented an integrated impact modelling framework (IIMF) for assessing regionalized impacts of climatic and socio-economic drivers on choices of crops, fertilization intensity and soil management as well as on river runoff and river vulnerability for exceedance of EQS for assessment of the nutrient component of the good ecological water quality status. Since little scientific information has been available on the context of climatic and socio-economic change and its consequences for land use, water resources and the ecological status of surface waters (Dunn et al., 2012), this IIMF provides a useful interdisciplinary methodology for evaluating these interactions at regional scale with Austria as example.

In order to show the capability of the IIMF to depict pollution impacts along the policy-economy-climate-agriculture-water interfaces, all single models and the IIMF have been validated based on observed state variables of land use, river flow as well as nutrient concentrations and loads in rivers. The overall performance of the IIMF for calculations of river nutrient load (120 monitoring stations) shows a Nash-Sutcliffe Efficiency of 0.73 for nitrogen and 0.51 for phosphorus. The largest deviations between model results and observations are for phosphorus loads in alpine catchments. Among them, 63% of catchments show a deviation between modelled and observed loads of >30%. In catchments dominated by agricultural production, the performance of the IIMF is much better as only 30% (domination of cropland) and 23% (domination by permanent grassland) have a deviation of >30% between modelled and observed loads. As risks of exceedance of environmental quality standards are largest for catchments dominated by cropland, the IIMF has shown to be well suited for risk assessment of not achieving nutrient criteria for the good ecological status in most Austrian regions.

Modelling proves to consistently generate scenario results. Major advantages are the reproducibility compared to expert judgements. Complex system behavior can be quantified, which may be beyond the experiences or imagination of experts. However, the validity of results is subject to a sufficient understanding of the modelled system. The outcome of this article supports using the IIMF for a set of selected climate and socio-economic scenarios to be analyzed for the period 2025 to 2040 in an upcoming article. The analysis will focus on a risk assessment for exceedance of nutrient requirements beyond the good ecological water quality status under changing climatic and socio-economic boundary conditions as well as on assessing of the cost-effectiveness of water protection. Therefore, the upcoming article has the potential to support relevant policy decisions, e.g. on the effect of forthcoming agricultural policies on water quality under changing climatic conditions.

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Appendix A

Table A1
Summary of the USLE factors of all grid cells as used for calculating the soil loss.

	R-factor	K-factor	L-factor	S-factor	C-factor crop land	C-factor grassland
Min	15.23	0.05	0.63	0.03	0.02	0.02
Max	297.89	0.78	3.50	11.32	0.40	0.02
1st Quartile	71.84	0.18	1.41	0.40	0.06	0.02
3rd Quartile	118.75	0.46	1.86	1.70	0.18	0.02
Median	94.05	0.31	1.59	0.89	0.10	0.02
Mean	97.22	0.32	1.63	1.25	0.14	0.02

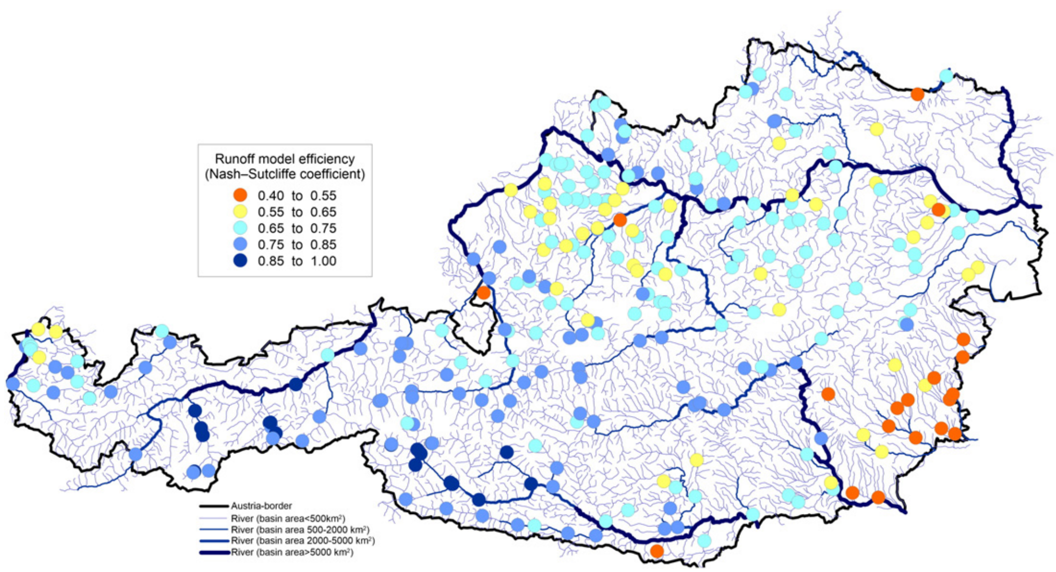

Fig. A1. Spatial patterns of ME in Austria in the calibration period 1976–2010.

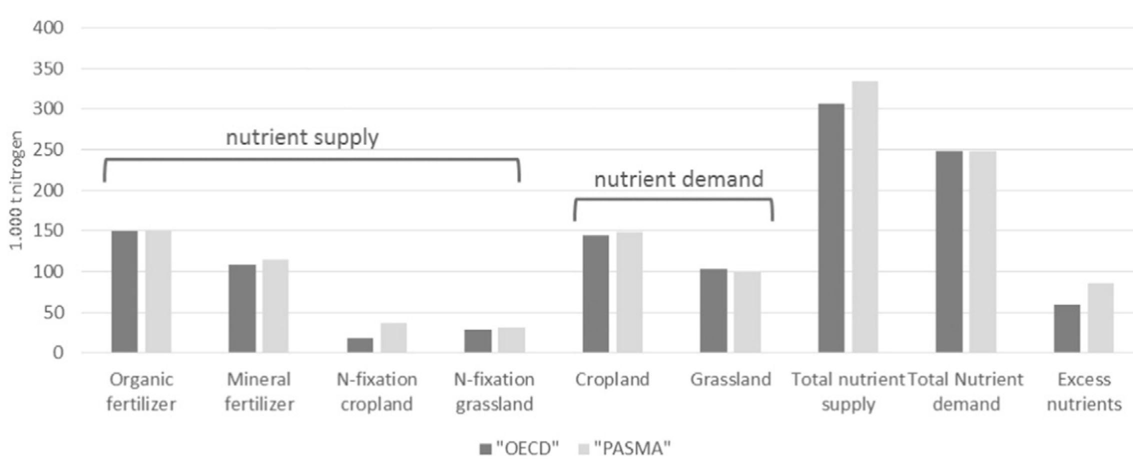

Fig. A2. Comparison of nitrogen balance components at national level with results from PASMA[grid] ("PASMA") and calculations based on the OECD method ("OECD"). Note: Atmospheric deposition is included at a later stage in the modelling chain.

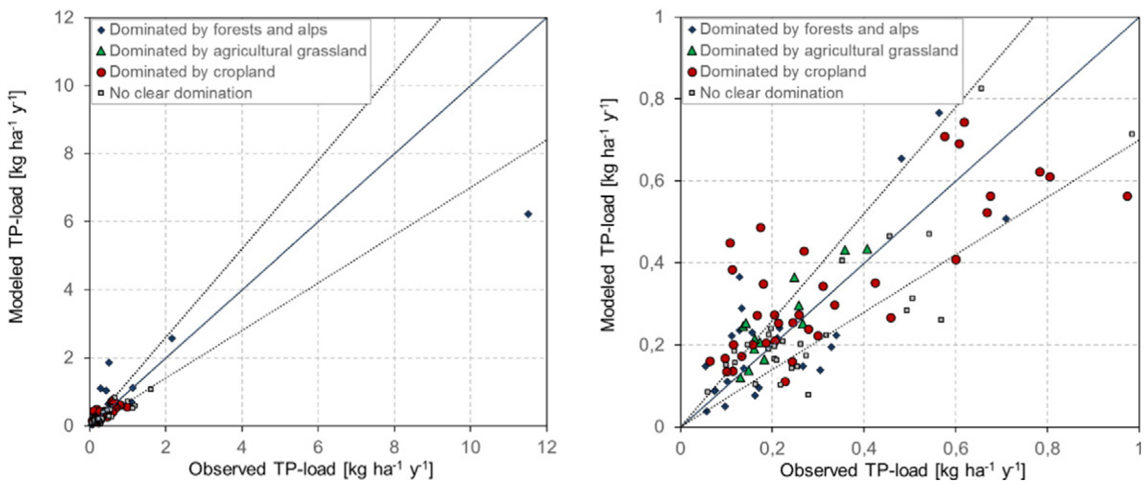

Fig. A3. Observed versus modelled TP loads, full range (left) and zoom to TP load <1 kg ha^{-1} y^{-1} (right) for catchments in different categories; the full grey line indicates a slope of 1 of the regression line between observed and modelled loads; the dotted lines show a deviation of modelled loads from observed ones of 30%.

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