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Modelled impacts of policies and climate change on land use and water quality in Austria

Martin Schönhart[a,*], Helene Trautvetter[b], Juraj Parajka[c], Alfred Paul Blaschke[c], Gerold Hepp[b], Mathias Kirchner[a,d], Hermine Mitter[a], Erwin Schmid[a], Birgit Strenn[b], Matthias Zessner[b]

[a] Institute for Sustainable Economic Development, BOKU University of Natural Resources and Life Sciences, Feistmantelstraße 4, 1180 Vienna, Austria
[b] Institute for Water Quality, Resources and Waste Management, TU Wien, Karlsplatz 13/226, 1040 Vienna, Austria
[c] Institute for Hydraulic Engineering and Water Resources Management, TU Wien, Karlsplatz 13/222, 1040 Vienna, Austria
[d] Research Group Environment, Agriculture and Energy, Austrian Institute of Economic Research (WIFO), Arsenal 20, 1030 Vienna, Austria

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ABSTRACT

Climate change is a major driver of land use with implications for the quality and quantity of water resources. We apply a novel integrated impact modelling framework (IIMF) to analyze climate change impacts until 2040 and stakeholder driven scenarios on water protection policies for sustainable management of land and water resources in Austria. The IIMF mainly consists of the sequentially linked bio-physical process model EPIC, the regional land use optimization model PASMA[grid], the quantitative precipitation/runoff TUWmodel, and the nutrient emission model MONERIS. Three climate scenarios with identical temperature trends but diverging precipitation patterns shall represent uncertainty ranges from climate change, i.e. a dry and wet situation. Water protection policies are clustered to two policy portfolios WAP_I and WAP_II, which are targeted to regions (WAP_I) or applied at the national scale (WAP_II). Policies cover agri-environmental programs and legal standards and tackle management measures such as restrictions in fertilizer, soil and crop rotation management as well as establishment of buffer strips. Results show that average national agricultural gross margin varies by ± 2%, but regional impacts are more pronounced particularly under a climate scenario with decreasing precipitation sums. WAP_I can alleviate pressures compared to the business as usual scenario but does not lead to the achievement of environmental quality standards for P in all rivers. WAP_II further reduces total nutrient emissions but at higher total private land use costs. At the national average, total private land use costs for reducing nutrient emission loads in surface waters are 60–200 €/kg total N and 120–250 €/kg total P with precipitation and the degree of regional targeting as drivers. To conclude, the IIMF is able to capture the interfaces between climate change, land use, and water quality in a policy context. Despite efforts to improve model linkages and the robustness of model output, uncertainty propagations in integrated modelling frameworks need to be tackled in subsequent studies.

1. Introduction

1.1. Land use, water quality problems and policies

In Europe, agricultural nutrient management has a considerable influence on the quality of surface and coastal water bodies. Despite some reductions in nutrient loads, agriculture is the largest contributor to nitrogen (N) pollution in more than 40% of Europe's water bodies (EEA, 2012). In Austria, concerns regarding nutrient pollution of water bodies are threefold: First, nitrate leaching from agricultural land deteriorates groundwater quality. Second, about 15% of local surface water bodies are endangered of not achieving the good water quality status due to nutrient pollution today. They are mainly located in intensively used agricultural areas with phosphate-phosphorus exceeding Water Framework Directive (WFD) environmental water quality standards (EQS) (BMLFUW, 2015). Finally, 96% of the Austrian territory is located in the Danube Basin discharging towards the western shelf of the Black Sea, which is highly vulnerable to eutrophication with phosphorus (P; Danube plume) and N (towards central Black Sea) as limiting factors of algae growth (Kroiss et al., 2006). Thereby, agriculture is the main source for N pollution in the Danube Basin. In addition to N leaching from fertilization, ammonia volatilization from animal husbandry and its deposition plays a decisive role as well (Behrendt et al., 2005). In respect to P, wastewater management is the

* Corresponding author.
E-mail address: martin.schoenhart@boku.ac.at (M. Schönhart).

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main source of water pollution in the Danube Basin, with agriculture being second. Water protection strategies for the near future are supposed to significantly reduce P pollution by enhanced wastewater treatment. However, agricultural sources are expected to be a longer lasting problem (ICPDR, 2015).

Insufficient environmental quality and high stakes for the society result in water protection policies at different governance levels. At global level, water protection is part of at least three Global Development Goals (SDGs), i.e. SDG 6 Clean Water and Sanitation, SDG 14 Life Below Water, and SDG 15 Life on Land. At the EU level, the WFD (2000/60/EC) including its influential Nitrates Directive (91/676/EEC) and Urban Waste Water Directive (UWWD, 91/271/EEC) results in national or regional policies that govern domestic, industrial, and agricultural processes to protect national and European water bodies. At the Danube Basin level, EU member states declared the basin as “sensitive area” according to the UWWD in respect to nutrient pollution. Regarding reduced N losses from agriculture at the national level, Austria implemented the Nitrates Directive with its “Aktionsprogramm Nitrat” (BMLFUW, 2012). Measures include restrictions of N-fertilizer applications in respect to timing, vulnerable locations, and amounts as well as specific requirements for manure storage and application. The main aim is to reduce nitrate pollution of groundwater. P releases from agricultural sources to surface waters are not in the focus of water protection policies in Austria so far.

1.2. Climate change and water systems

It is obvious that many countries in Europe have not achieved an area-wide socially accepted water quality status yet despite comprehensive policies and regional successes. Policies have been adapted to current socio-economic and bio-physical conditions but may become insufficient or inappropriate in the future. Growth in global population and per capita income, as well as climate change lead to direct and indirect impacts inducing land use changes (e.g. Wiebe et al., 2015). For example, climate change can increase or decrease the suitability for certain crops or land use types (e.g. Schönhart et al., 2016) or the marginal benefit of agro-chemicals. Results by Blanke et al. (2017) and Olesen et al. (2007) show large heterogeneity in N leaching changes among European regions from future wheat and maize production. Directions of change are uncertain in many regions including Austria among others due to climate model uncertainty. Rising temperatures increase biomass growth in water bodies (Zoboli et al., 2018) and changing precipitation patterns can alter nutrient emissions, soil erosion, dilution ratio, and flow regimes.

1.3. Integrated water system modelling

As pointed out in Zessner et al. (2017), 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). Future policies to protect water bodies have to be adapted to direct and indirect climate change impacts. It requires scientific evidence on the combined and mutual effects of land use choices, water protection policies, and global change. Current research is biased towards water quantity, while water quality has been insufficiently studied so far (Cai et al., 2015).

In recent years, integrated models have been developed to tackle these complexities. However, only few cover the nexus of climate change and policy impacts, land use adaptation, and its consequences on surface water quality in a consistent way. Some studies apply exogenously given land use scenarios in integrated models to assess their environmental impacts and costs but do not consider climate change impacts (e.g. Dymond et al., 2010; Bohnet et al., 2011; Polasky et al., 2011; Kling et al., 2014). Others capture land use change − though not climate change − endogenously to either search for cost-efficient spatial allocations of management options to improve water quality (Xu et al., 2018) or to simulate land use decision processes in bottom-up land use models (Lehtonen et al., 2007). Honti et al. (2017) define management scenarios to take climate change adaptation into account and model the role of climate scenario uncertainty on runoff. Molina-Navarro et al. (2018) downscaled European level storylines to a Danish catchment level and analyzed land use and climate change impacts on water quality. Both examples, as well as those of Lautenbach et al. (2009), Mehdi et al. (2015a,b), consider changes in flow conditions endogenously but assume land use adaptation to climate change exogenously. Kraucunas et al. (2015) link climate change scenarios to bio-physical and partial-equilibrium models to analyze climate change impacts and adaptation. Regional land use maps are derived via top-down spatial disaggregation but water quality impacts are not considered. To conclude, most climate change studies on water quality either keep land use invariable over time (e.g. Sinha et al., 2017) or design land use scenarios − eventually stakeholder driven − prior to modelling (e.g. El-Khoury et al., 2015; Mehdi et al., 2015a). However, ignoring climate change adaptation of agricultural land use can create inconsistencies and may lead to wrong policy conclusions.

Rare examples of combining climate change, agricultural adaptation and its corresponding impacts on water quality − for nitrate and phosphate emissions or algal production − in a consistent quantitative manner is presented in Fezzi et al. (2015) and Bateman et al. (2016). They linked spatially explicit econometric land use models with statistical surface water quality models. Barthel et al. (2012) integrated climate change and socio-economic drivers into land use modelling and related N pollution of groundwater but did not consider P or surface water quality.

Stakeholder participation − achieved in some of the cited studies above − can be crucial for the quality and social acceptance of research outcomes in management and policy processes. Volk et al. (2010) highlight the lack of stakeholder integration in decision support systems for river basin management and Martin-Ortega et al. (2015) call for transdisciplinary studies to increase the robustness of solutions to wicked environmental problems such as water pollution. Iglesias et al. (2007) emphasize the importance of knowledge transfer to stakeholders in climate change research. They highlight among others the challenge of uncertainty management und the crucial role of science communication. These are strong arguments in favor of transdisciplinary research.

1.4. Added value and article concept

In this article, we tackle the identified methodological concerns and knowledge gaps, i.e. coarse spatial resolution and inconsistent representation of land use in water quality modelling under climate change, lacking knowledge on effective policies to govern climate change impacts and autonomous adaptation, and missing stakeholder engagement. Our major applied research objective is to assess the land use, farm economic, and environmental effects of stakeholder driven water protection policies to maintain water quality in Austrian rivers under climate change. From a methodological perspective, we test the applicability of a novel quantitative spatially explicit integrated impact modelling framework (IIMF) in a scenario context. The IIMF has been presented for the first time and applied on a reference scenario in Zessner et al. (2017). It combines among others a bio-physical process model to simulate crop yields, an economic land use optimization model to derive efficient land use choices, a precipitation/runoff model to compute flows, and a nutrient emission model to quantify environmental impacts. The IIMF provides consistent nutrient emission outcomes from socio-economic, climate change, and water protection policy scenarios. Climate change could improve or deteriorate the current status of Austrian water resources. The policies designed for water protection will be scrutinized for their effects on water quality, agricultural producer surplus, and private land use costs under

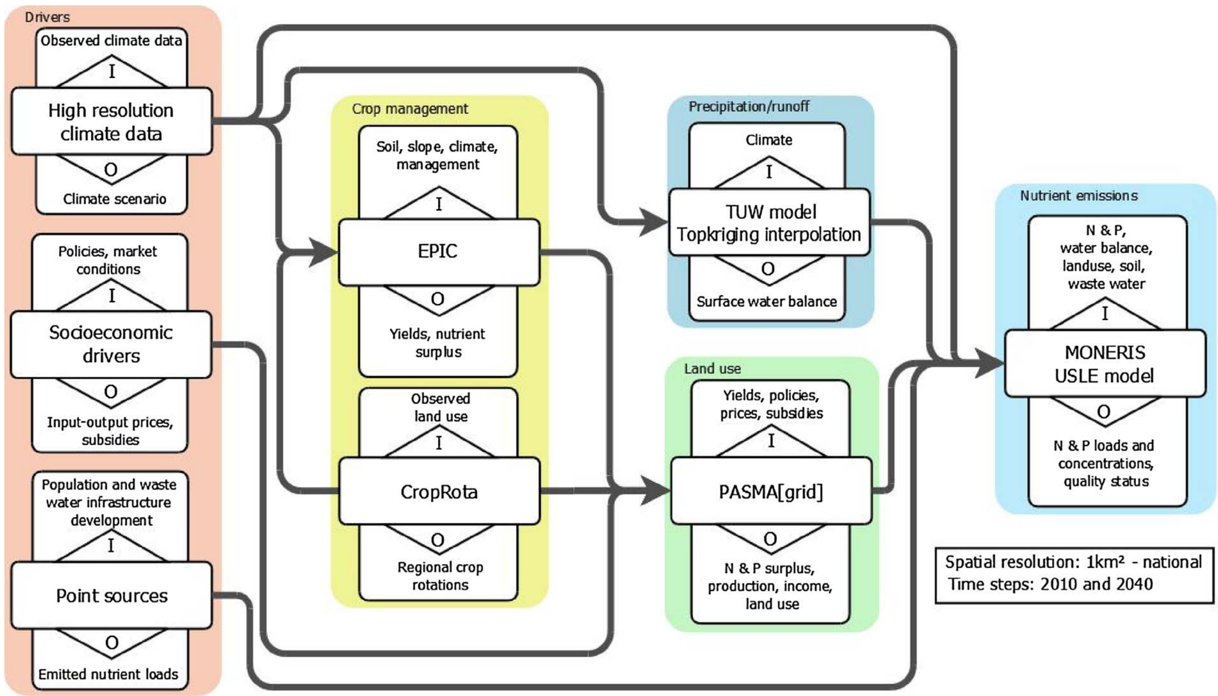

Fig. 1. The Integrated Impact Modelling Framework on land use and water quality in Austria.
Source: adapted from Zessner et al. (2017)

contrasting climate change scenarios and results shall contribute to risk management in the WFD. Stakeholders from public administration, agricultural organizations and research institutes were engaged to accompany the research process.

Section 2 provides an overview on the methodology, i.e. IIMF, data sources and scenario development. Section 3 presents model results. We discuss results with respect to their policy implications and critically review major model assumptions and uncertainties. Finally, we draw conclusions on promising policy actions and further research demand. Supplementary material provides details on methods and results.

2. Material and methods

2.1. Integrated impact modelling framework (IIMF)

The core IIMF applied in our research has been evaluated in Zessner et al. (2017). The following section summarizes the basic model structure and data from the cited publication. The IIMF includes six sequentially coupled models (Fig. 1), i.e. the crop rotation model CropRota (Schönhart et al., 2011), the bio-physical process model EPIC (Environmental Policy and Integrated Climate; Izaurralde et al., 2006; Williams, 1995), the socio-economic land use optimization model PASMA[grid] (Kirchner et al., 2015, 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), and the nutrient emission model MONERIS (Modelling Nutrient Emissions in River Systems; Behrendt and Opitz, 1999; Venohr et al., 2009; Zessner et al., 2011). Each model is a stand-alone tool with complementary characteristics in the IIMF and has been chosen due to its unique properties and requirements for model set-up including data processing, calibration, and validation. Compared to other crop rotation generators (e.g. ROTOR by Bachinger and Zander, 2007), a unique feature of CropRota is its utilization of observed cropping patterns and its reduction of large numbers of potential crop rotations to a hierarchy of typical rotations based on agronomic criteria. EPIC may be replaced by other bio-physical process models able to model the observed set of crops. A major constraint is the required experience with a particular crop model in situations with lacking calibration data, which is typical for large scale high resolution applications. PASMA[grid] represents the Austrian agricultural and forestry sector at high spatial resolution. There is no alternative model available due to high costs in data collection and processing. The role of TUWmodel within the IIMP is to transform the climate signal into runoff calculations. Regionalization of simulated runoff to the sub-catchment level for the Austrian territory by top-kriging interpolation (Skøien et al., 2014) provides the inputs for the scenario calculations with the MONERIS emission model. The major advantage of the TUW-model is its elaborated representation of Austrian hydrological regime conditions. Land use information from PASMA[grid] is used in USLE for soil loss calculations. Nutrient surpluses and soil loss information together with river runoff at sub-catchment scale are input to the nutrient emission model MONERIS. It is needed to calculate river loads and concentrations indicating risks for not achieving the good status of water bodies due to nutrient pollution. The availability of MONERIS adapted to emission modelling on a sub-catchment scale all over Austria was the main reason for its choice.

Exogenous drivers in the IIMF are high-resolution climate data, socio-economic drivers (e.g. agricultural policies, input and output prices), and wastewater infrastructure (e.g. sewer distribution and level of wastewater treatment). A major source for land use data is the Integrated Administration and Control System (IACS) provided by BMLFUW (s.a.).

Mean daily air temperature and precipitation are input to the rainfall-runoff modeling by TUWmodel. Climate data, i.e. minimum and maximum temperature, precipitation, solar radiation, and wind speed at daily and 1 km grid resolution, also feed into EPIC to simulate crop growth and environmental impacts of management options, such as crop rotations, management intensity, irrigation, soil management, and cover crops. EPIC considers the CO_2 fertilization effect. The management options are based on expert knowledge and meet the CAP cross compliance standards (e.g. WFD). EPIC is applied at a spatial grid resolution of 1 km representing homogeneous response units (HRUs). HRUs are homogeneous with respect to soil type, slope and altitude (Stürmer et al., 2013) and are merged with climate clusters (Strauss

Table 1
Overview on land use management measures and policy instruments in the IIMF.

Management measure	Policy instruments	Representation in PASMA[grid]
Establishment of buffer strips	AEP − preventive surface water protection	Financial incentive for set-aside land use in particular pixels
Implementation of diverse crop rotations (e.g. restrictions on row crops)	Cross compliance standard − row crop limits Greening standard − crop rotation restriction	Constraints on particular crops (see Table A1 in Appendix A)
Adaptation of medium management intensity	AEP − Environmentally sound and biodiversity-promoting management	Financial incentive to choose a particular management intensity
Adaptation of low management intensity	AEP − limitation of yield-increasing inputs AEP − organic farming	Financial incentive to choose a particular management intensity
Establishment of set-aside land	Greening standard − set-aside	Constraint on minimum share of set-aside
Planting of cover crops	AEP − greening of arable land	Financial incentive to choose cover crops
Reduced tillage	AEP − direct and mulch seeding	Financial incentive to choose reduced tillage

Note: AEP agri-environmental program. Table A1 in Appendix A presents more details on the policy instruments.

et al., 2013), crop rotations and crop management options. With respect to management intensity, we define three levels, i.e. high, medium, and low. They are based on expert assumptions and include portfolios of management choices on N, P, and potassium (K) fertilization levels, pesticide applications and cropping schedules. The major difference are fertilization levels, which correspond to legal thresholds from WFD (high) and assumptions on the compliance with measures of the Austrian agri-environmental program (medium, low). Such correspondence enables a consistent representation of policies in PASMA[grid].

EPIC results at HRU level are input to PASMA[grid], which seeks to find optimal agricultural and forestry land use choices following socio-economic and climate change scenario drivers. PASMA[grid] maximizes total gross margin from agriculture and forestry at NUTS-3 level (i.e. several districts in the EU Nomenclature of territorial units for statistics) subject to grid land endowments as well as regional livestock housing capacities and feed and fertilizer balances. Gross margins are calculated by subtracting variable production costs from market revenues and agricultural policy payments. PASMA[grid] models land use choices at HRU level and linearly disaggregates results such as land use and management as well as nutrient surpluses to 1 km grid resolution subsequent to the optimization process.

The USLE model considers the spatial extend of the main arable crops, tillage choices and permanent grassland from PASMA[grid] to calculate soil loss. The amount of nutrient surpluses as well as changes of agricultural land use and soil loss feed into the nutrient emission model MONERIS.

MONERIS assesses the impact of land use on nutrient emissions and 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 waste water treatment plant effluents and sewer overflows. Data are available from the emission calculations for the reference period (Zessner et al., 2017). As a major output indicator, river concentrations of nutrient parameters from MONERIS can be used to assess risks of missing a good ecological surface water status according to the Austrian implementation of the EU WFD.

2.2. Scenarios

IIMF outputs result from contrasting scenarios. They include three major components, i.e. socio-economic framework conditions, climate change, and portfolios of water protection policies. The time steps of the scenarios are the present reference period (2010, scenario *REF*) with markets and policies as observed and contrasting situations in 2040. This date is determined by the climate scenarios and appropriate as a reasonable period for economic decision making and policy planning. *REF* has been developed to calibrate and validate the IIMF as described in Zessner et al. (2017).

Socio-economic framework conditions are different for 2010 and 2040. They include changes in agricultural policies, market prices for inputs and outputs as well as agricultural productivity. With respect to the Common Agricultural Policy (CAP), we assume that present policies continue until 2040. We take current requirements for direct payments (greening measures) into account by forcing the model to devote at least 5% of the arable land to ecological focus areas, e.g. fallow and buffer strips. Other CAP elements with immediate impacts on nutrient emissions such as agri-environmental payments are part of the water protection policy scenarios. Commodity price assumptions until 2040 are based on forecasts of OECD-FAO (2013) but adapted to national circumstances assuming linear trends between 2023 and 2040. Assumptions on production costs and productivity developments follow similar procedures. Socio-economic framework conditions do not vary between the different scenarios in 2040 to reveal the impacts of climate change and water protection policies.

We selected three climate change scenarios from Strauss et al. (2013) based on discussions with stakeholders on sensible weather conditions for the quality and quantity of groundwater and surface water. The scenarios have one common temperature trend of about +1.5 °C up to 2040 compared to the past climate and three contrasting precipitation patterns. Scenario *similar* represents a similar distribution of precipitation sums compared to the past. *Wet* represents increases of daily precipitation sums by 20% and *dry* decreases of daily precipitation sums by 20%. These scenarios thus provide a range of possible future climates typical to realizations of regional climate simulation models (Ahrens et al., 2014) and allow to explore possible but eventually critical water system conditions in the future (see Supplementary material B).

Stakeholders supported the choice and definition of potential land use management measures and policy instruments in the IIMF − either economic incentives (e.g. agri-environmental payments) or legal constraints (e.g. environmental regulation such as nutrient limits). Criteria for this choice were the expected ecological effectiveness, cost-effectiveness, and stability of effects. The objective was to include measures that improve Austria's compliance to the EU WFD under climate change, i.e. alleviate eventual mal-adaptation to climate change with negative effects on surface water. Table 1 gives an overview on management measures and their corresponding policy instruments in the IIMF. PASMA[grid] selects land use management measures − under ceteris paribus market and climate conditions − due to the incentives and restrictions from policy instruments. Further measures such as a transition of arable land to permanent grassland or forests, or irrigation of arable land are feasible in the IIMF but do not have particular policy triggers. We clustered the selected instruments to two contrasting water protection policy portfolios (Table A1 in Appendix A).

The combination of socio-economic framework conditions, climate change scenarios, and water protection policy portfolios leads to five contrasting scenarios (including *REF*) that feed into the IIMF. Narratives (Supplementary material D) facilitated discussions on these scenarios in the stakeholder workshop.

Business as usual (*BAU*) implements the socio-economic framework conditions for 2040 but no further water protection policies. It is combined with climate change scenario *similar*. We compare most IIMF scenario results with *BAU*. To reveal climate change impacts, scenario *IMP* has policy conditions like *BAU* but is combined with climate scenarios *wet* and *dry*. The water protection scenarios *WAP_I* and *WAP_II* represent two consecutive levels of water protection policy portfolios. In *WAP_I* premium levels for certain agri-environmental measures are increased regionally by 25% and legal standards are adapted. *WAP_II* builds on *WAP_I* but further tightens thresholds, e.g. the legislation on crop rotations, and increases premium levels. For example, while some policies in *WAP_I* are applied to target regions with insufficient water quality according to the *REF* model results, i.e. areas with exceedance of EQS for phosphate, *WAP_II* offers these policies for the entire Austrian territory (see Table A1 in Appendix A). Fig. 2 summarizes the final scenarios applied in the IIMF. Details on the technical scenario implementation in the IIMF are presented in Supplementary material E.

3. Results

3.1. Precipitation/runoff

The climate scenarios are input to the TUW model to estimate surface water balances for 277 basins in Austria (Fig. 3). Scenario *similar* leads to an increase of winter and spring flows, particularly for alpine catchments with winter low flows. Rivers with predominant low flow in summer face a decrease of flows throughout the year. While vulnerability on low flow is decreased in the first case, it is increased in the second. Vulnerability of rivers dominated by summer low flow is further increased in scenario *dry*. Modelled flow reductions are in the range of 40–60% throughout the year. In the context of this study this is of specific interest as in these flatland basins agricultural land use is prevailing. Surface water balances are a major input to MONERIS because flow is an important component for the modelling of nutrient emissions, river loads and river concentrations.

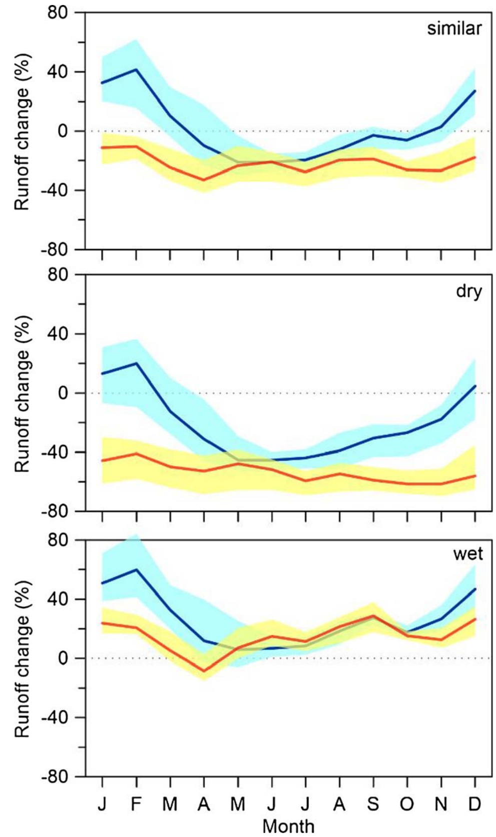

Fig. 3. Seasonal differences in runoff (in %) for three climate change scenarios compared to the past climate. Note: blue = winter (alpine basins) low flow regimes, red = summer (flatland basins) low flow regimes; line = median, shading = 25%- and 75%- percentile.

3.2. Land use results

3.2.1. Crop yields

Simulated crop yields from EPIC are available at 1 km grid resolution stratified by three soil management measures (conventional and reduced tillage as well as cultivating winter cover crops), three intensity levels (high, medium, low) under rain-fed conditions, one intensity level (high) under irrigated conditions and four climate scenarios (*past*, *similar*, *wet*, *dry*). The major difference between intensities is the amount and timing of fertilizer applications. Fig. 4 presents the distribution of relative crop yield changes from *past* with medium intensity. Climate change and intensities mutually impact crop yields. For example, winter wheat yields are more frequently increasing under declining precipitation (*dry*) than maize or rapeseed. For most crops the

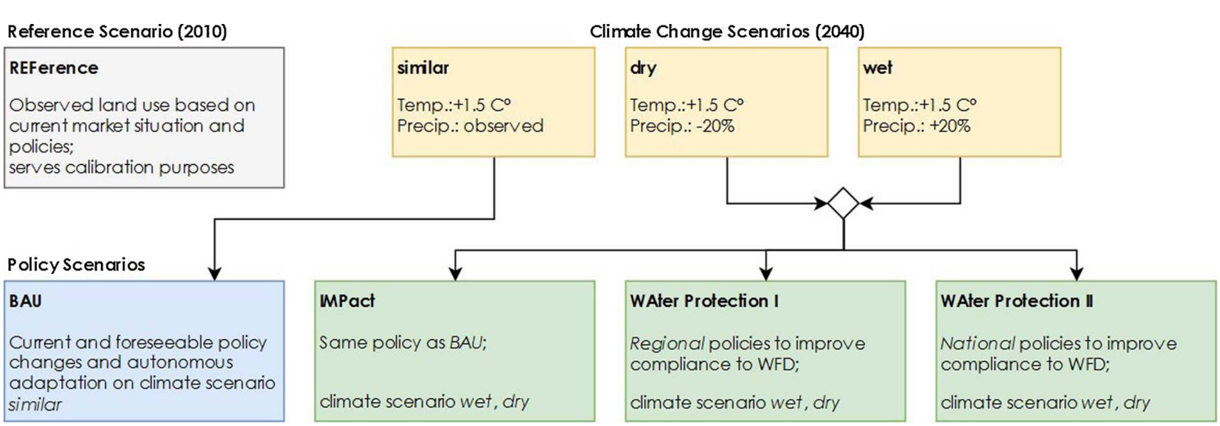

Fig. 2. Scenario overview (see Table A1 in Appendix A for details).

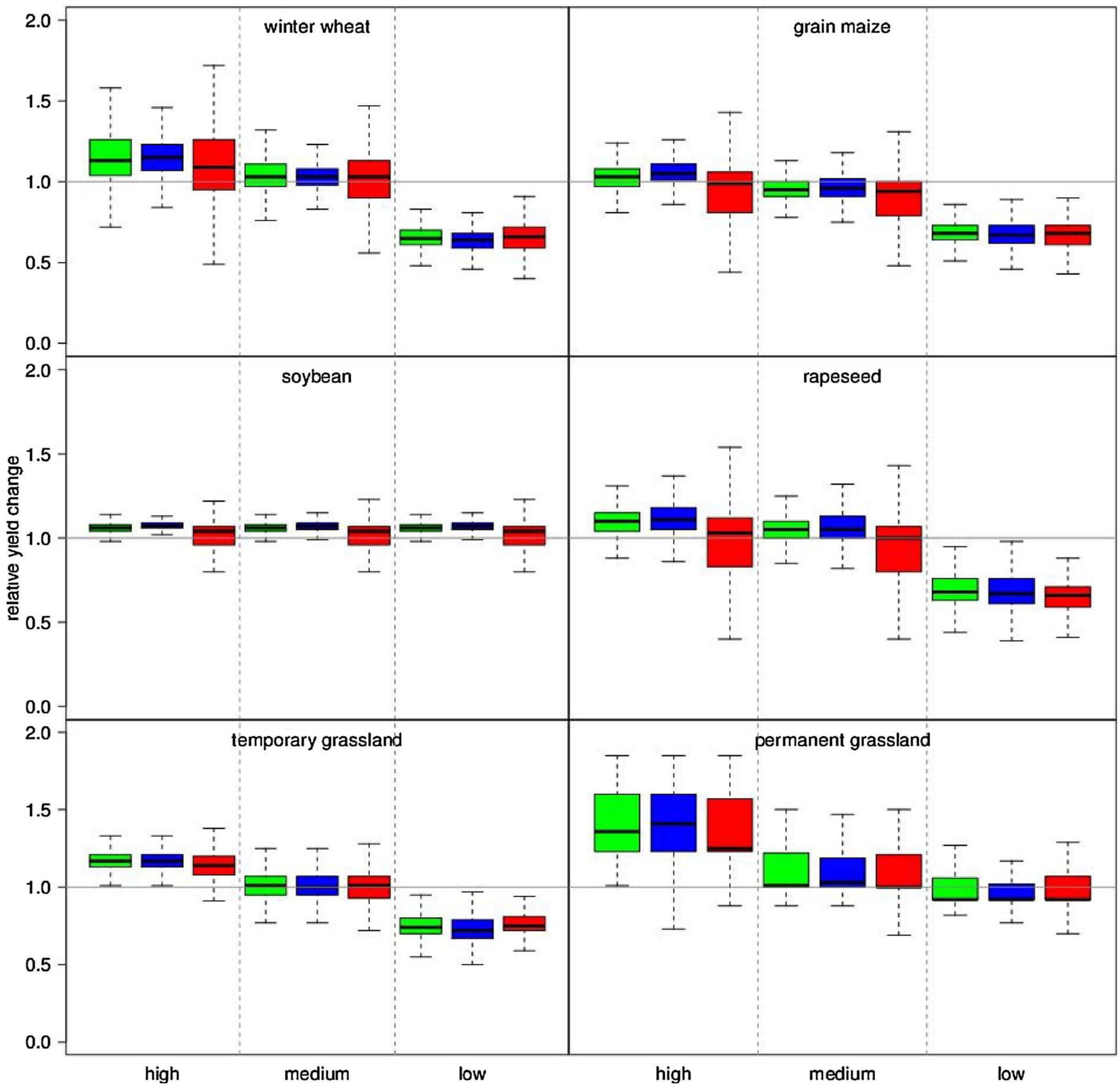

Fig. 4. Modelled multi-year average relative yield changes for six crops and three rainfed intensities. The reference is past climate with medium intensity. Colors indicate climate scenarios similar (green), wet (blue) and dry (red). Box plot statistics indicate spatial variability with respect to 1 km grid resolution. This figure and others in this article result from the R software (R Development Core Team, 2014). Note: The whiskers' length is up to 1.5 times the range of the 25%–75% quartiles. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

impacts of climate change (including CO_2 fertilization) are stronger under high and medium intensities. EPIC shows large yield changes on permanent grassland. They are driven by higher temperatures rather than precipitation changes, i.e. median yield changes for high and medium intensities decrease with decreasing precipitation (*dry*) but remain positive. The boxplots in Fig. 4 also indicate the considerable spatial heterogeneity in crop yield changes among HRUs.

3.2.2. Land use

Average annual crop yields from EPIC as well as policy specifications from all policy scenarios are input to the economic land use model PASMA[grid]. The choice of crops, the intensity levels and soil management are three major land use components impacting water quality. Fig. 5 compares areas of two decisive crop categories for water quality, i.e. maize and set-aside, of the two climate and three policy scenarios to *BAU* in each NUTS-3 region. Set-aside includes buffer strips, which are part of agri-environmental measures in *WAP_I* and *WAP_II*. Climate change impacts crop choices in some regions. Maize areas decline while set-aside slightly increases with decreasing precipitation but *WAP* policies are a more important driver of crop choices in PASMA[grid] in most regions. The policies focus on decreasing maize and increasing set-aside areas with increasing ecological effectiveness of *WAP_II* compared to *WAP_I*. In some regions, however, a drying climate (*IMP_dry*) results in stronger reductions of maize and increases of set-aside than impacts of *WAP_I*.

PASMA[grid] seeks optimal land use from three pre-defined intensity levels, i.e. high, medium and low (see section 2.1). Each level has crop specific fertilization rates and corresponds to particular agri-environmental policy measures. Consequently, a crop area is eligible for a measure only if it remains below the particular fertilization thresholds. For example, high intensity is incompatible with any agri-environmental measure. Fig. 6 compares the spatial extent of intensity levels from alternative policy and climate scenario combinations with *BAU*. More favorable growth conditions under *IMP_wet* slightly increase production intensities (intensity "high") in many regions, while scenario *IMP_dry* with generally less favorable growth conditions decreases

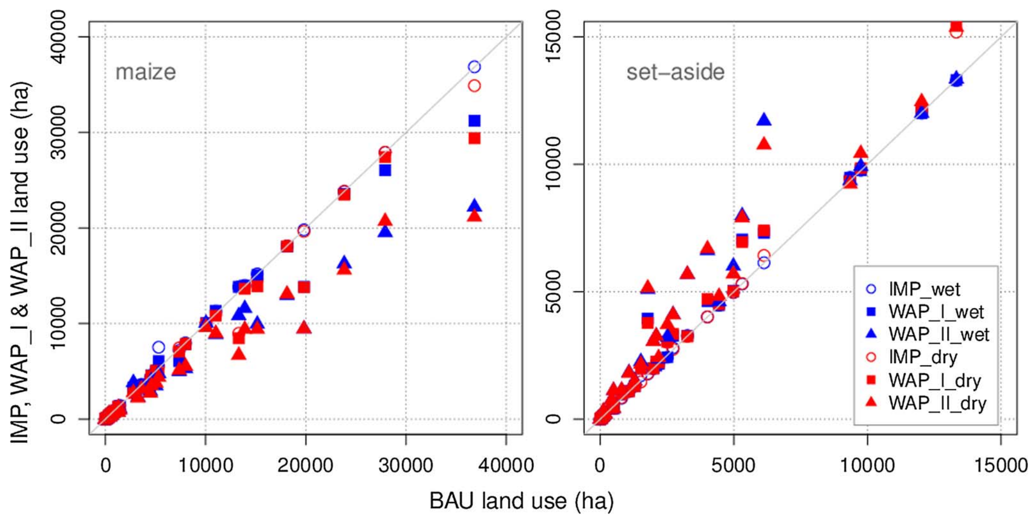

Fig. 5. Comparison of maize and set-aside area of two climate and three policy scenarios with the BAU scenario for 35 Austrian NUTS-3 regions (Data: Supplementary material F).

intensity (Fig. 6). Changes in production intensity result from climate induced changes of the marginal values of farm inputs – mainly but not only fertilizers in case of the IIMF. Better growth conditions increase yields from a given amount of farm inputs and vice versa. At the national level, modelled mineral N fertilizer inputs are 125 10^3t N in *BAU_similar*, 126 10^3t N in *IMP_wet*, and 117 10^3t in *IMP_dry* with more pronounced changes at regional level (see Fig. 7 and Supplementary material H). The water protection policies (*WAP_I* and *WAP_II*) reduce land use intensity in many regions compared to *IMP*. There is interaction between climate and policies in some regions, where areas with low intensity – induced by policies – increase more under *dry* than *wet* conditions. It indicates lower opportunity costs of intensive agricultural production under unfavorable climate conditions.

The N balance is the major interface between PASMA[grid] and MONERIS. Fig. 7 shows the modelled annual national N cycle for agricultural soils. N inputs result from organic and mineral fertilizer production and biological N fixation in PASMA[grid]. Atmospheric N deposition complements the N inputs in MONERIS. N removal results from uptakes by arable crops, permanent grasslands, and permanent crops. The resulting N surplus in Fig. 7 slightly decreases from *IMP* to *WAP_I* and *WAP_II* and is lower for *wet* than *dry*. The latter results from higher nutrient uptakes despite higher fertilization and N fixation in *wet*.

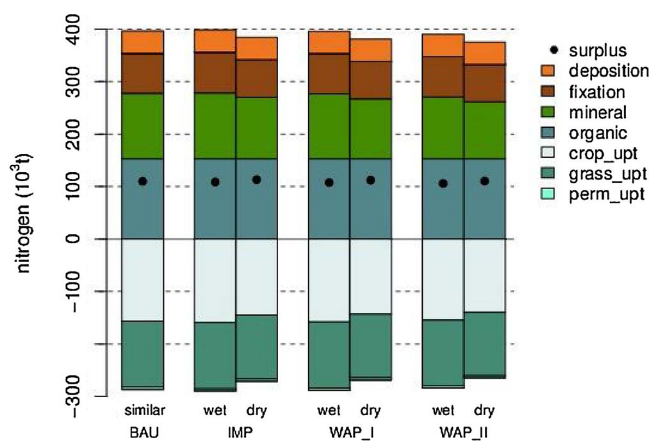

Fig. 7. Annual agricultural N cycle at national level for BAU with climate change scenario similar and IMP, WAP_I and WAP_II with wet and dry. Components are: N surplus, atmospheric N deposition, biological N fixation, mineral and organic fertilizer inputs, N uptake by arable crops, permanent grasslands, and permanent crops (Data: Supplementary material H).

3.2.3. Economic effects of water protection policies

Climate change and policies impact agricultural producer surplus but the effects are heterogeneous among Austrian regions. In *IMP*, either a *wet* or *dry* climate lead to ± 0% or −2% changes in producer surplus at the national level with larger regional disparities. In the eastern regions, for instance, changes from *dry* are up to −10% (for further details see Supplementary material I). In most regions, the water protection policies increase producer surplus while some show hardly

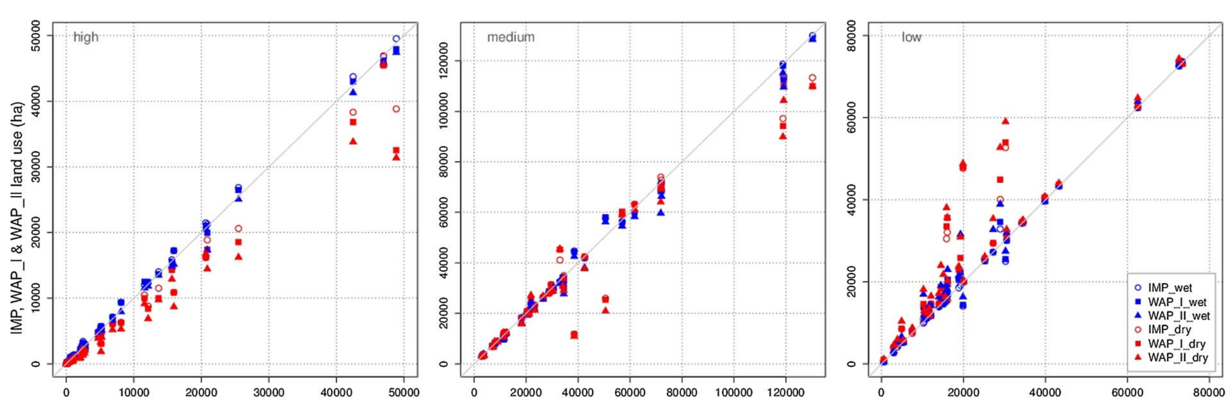

Fig. 6. Comparison of modelled agricultural land use of two climate and three policy scenarios with modelled agricultural land use in the BAU scenario. Each plot is specific to an intensity level (high – left; medium – middle; low – right) for 35 Austrian NUTS-3 regions (Data: Supplementary material G).

any changes. Most policies in *WAP_I* and *WAP_II* are part of the agri-environmental program, and if payments at least cover opportunity costs they incentivize management change in PASMA[grid], improve water quality (see 3.3) and may increase agricultural producer surplus. In *WAP_I*, the national agricultural producer surplus amounts to +1% compared to *IMP*. Increasing premium levels and spatial coverage lead to +2% in *WAP_II* compared to *IMP*. It indicates modelled windfall profits from agri-environmental payments.

The *WAP* policy portfolios − representing legal standards and voluntary agri-environmental measures − affect agricultural producer surplus by raising production costs, decreasing market revenues and increasing revenues from agri-environmental payments. Consequently, private land use costs of the *WAP* policy portfolios in the IIMF are the change in market revenues and variable production costs compared to *IMP* under the same climate conditions. We focus on land use as this is the strength of the IIMF and do neither consider costs nor emission effects of waste water treatment plants in this section. As *WAP_I* focuses on catchments with EQS exceedance and *WAP_II* on the total agricultural area in Austria, total private land use costs of *WAP_I* are significantly lower than those of *WAP_II*. With respect to climate change, total private land use costs are higher under the *dry* than *wet* climate (Table 2).

3.3. Emissions

PASMA[grid] land use and management results and TUW-model runoff results are input to MONERIS to model changes in water quality at (sub-)catchment level. TP (total phosphorus) annual export loads at the outlet of MONERIS catchments show a high variety of < 0.05 to 10 kg/ha (Fig. 8). The highest export loads stem from catchments with high shares of glaciers and corresponding high suspended solids concentrations from rock weathering but they have low relevance in the context of this study. With climate change, scenario *IMP_wet* leads to a significant increase of export loads. Policy measures can significantly impact TP loads in rivers as well. *WAP_I* is targeted towards hot-spot regions with respect to the exceedance of EQS for phosphate. *WAP_II* further tightens crop rotation limitations for maize production and offers additional agri-environmental measures all over Austria. Mainly effective for TP is a reduced nutrient loss from erosion by restricting maize and soybeans on steep fields close to water courses and by establishing buffer strips. A nation-wide adoption rate of water protection measures (*WAP_II_wet*) can partly outweigh increasing runoff and leads to an overall reduction of river loads close to *BAU* results. Scenario *dry* tends to decrease river loads. Combined with a nation-wide water protection policy (*WAP_II_dry*), this leads to a reduction of export loads of some catchments by more than 50% as compared to *BAU*.

Phosphate concentrations in rivers − modelled as 90%-percentiles (c90) − are presented in relation to nutrient type-specific EQS to indicate their risk of EQS exceedance (Fig. 9). Due to model uncertainties of about ± 30% (Zessner et al., 2017) ratios between 0.7 and 1.3 are considered as potential exceedance of EQS. Ratios between 1.3 and 2.0 are considered as exceedance, ratios > 2.0 as severe excceedance, and ratios < 0.7 as no exceedance. The vulnerability of rivers is clearly increasing under *dry* (e.g. *IMP_dry*). Risks of failing EQS can be reduced below the level of *BAU* in case of a high level of water protection (*WAP_II_dry*). A *wet* scenario would be favourable in respect to achiving EQS in Austrian rivers. Nevertheless, even in scenario *WAP_II_wet* the risk of exceeding EQS remains in some rivers (for spatial details, see Supplementary material J).

Table 2 shows the overall scenario results for P (TP loads and phosphate concentrations) and N (TN (total nitrogen) loads and NO_3-N concentrations). While the policy portfolios of *WAP_I* and *WAP_II* are effective in several regions to reduce P emissions, the effects on N emissions are generally low. On average for Austria, TN emissions decline by 1% to 2% in *WAP_I* and *WAP_II* under the *wet* and 0% and 1% under the *dry* climate scenario. However, reductions of aggregated N surplus in agriculture are 1% (*WAP_I*) and 4% (*WAP_II*) under both climate situations. In the *BAU* scenario, an exceedance of EQS for NO_3-N occurs only in few Austrian rivers (1% of the catchments). Exceedances disappear in *wet* but they increase in *dry* to 2% of the catchments without additional measures (*IMP*) and are reduced to 1% under both *WAP* scenarios.

Fig. 10 relates private land use costs of the *WAP* policy portfolios (see Section 3.2.3) to the annual changes in TN and TP emission loads into surface waters at NUTS-3 level, while Table 2 presents national averages (last two columns). As expected, *WAP_II* leads to lower average nutrient emissions. The national coverage of *WAP_II* policies increases private land use costs per unit reduction of TP emissions in both climate scenarios and of TN emissions under *wet* conditions. However, it has the opposite effect for TN emission reductions under *dry* conditions where private land use costs per unit TN emission reduction slightly decrease when policies are applied at the national level (*WAP_II*) instead of regional targeting (*WAP_I*). This is indicated by the dashed red trend line in Fig. 10 (left), which is below the solid red trend line. At the national level total private land use costs decrease from 205 €/kg TN to 168 €/kg TN with a shift from *WAP_I* to *WAP_II* (Table 2).

Table 2
Results at national level for annual total N and P export loads and share of catchments with EQS exceedances (modeled concentrations to EQS > 1.3) of scenarios BAU, IMP_wet and IMP_dry and load reductions by measures implemented in WAP_I_wet, WAP_II_wet, WAP_I_dry and WAP_II_dry; exceedances of EQS; total annual private land use costs as compared to IMP and per unit reduction of exported load under these scenarios.

Scenarios	Emissions		Exceedance of EQS in % of catchments (367 in total)		Total private land use costs compared to IMP	Total private land use costs per unit of emission reduction	
	TN (t/y)	TP (t/y)	NO_3-N (%)	PO_4-P (%)	1000 €/y	€/kg TN	€/kg TP
BAU	51,500	3150	1	13			
IMP_wet	57,500	3510	0	11			
IMP_dry	45,000	2990	2	16			
WAP_I_wet	57,000	3240	0	5	31,400	64	118
WAP_II_wet	56,300	3020	0	5	86,400	70	176
WAP_I_dry	44,800	2720	1	11	37,700	205	137
WAP_II_dry	44,400	2590	1	11	101,600	168	254

4. Discussion

4.1. Policy implications

4.1.1. Adaptation to climate change

The model results confirm previous studies on climate change impacts for Austria, i.e. strongly heterogeneous impacts between the semi-arid eastern and humid western regions (Mitter et al., 2015; Schönhart et al., 2014). Particularly the eastern parts of Austria are sensitive to changing precipitation patterns. An increase in temperature likely leads to increasing drought stress, which has been modelled for central Europe for the previous decades (Trnka et al., 2016). However, *wetter* conditions and higher temperatures extend the vegetation period and elevated CO_2 concentration can stimulate crop growth. These results are limited to the modelled climate change scenarios and time frames until mid-century, while further temperature increases may lead to more pronounced negative effects (Ciscar et al., 2011).

Changing bio-physical production conditions likely trigger land use changes. Increasing yield potentials raise the marginal value product of farm inputs, which stimulates for example more fertilizer use or higher livestock densities. The IIMF shows average climate induced changes in N inputs from fertilization and biological fixation of −5% to +1% with larger variation among NUTS-3 regions. Similar results can be found in

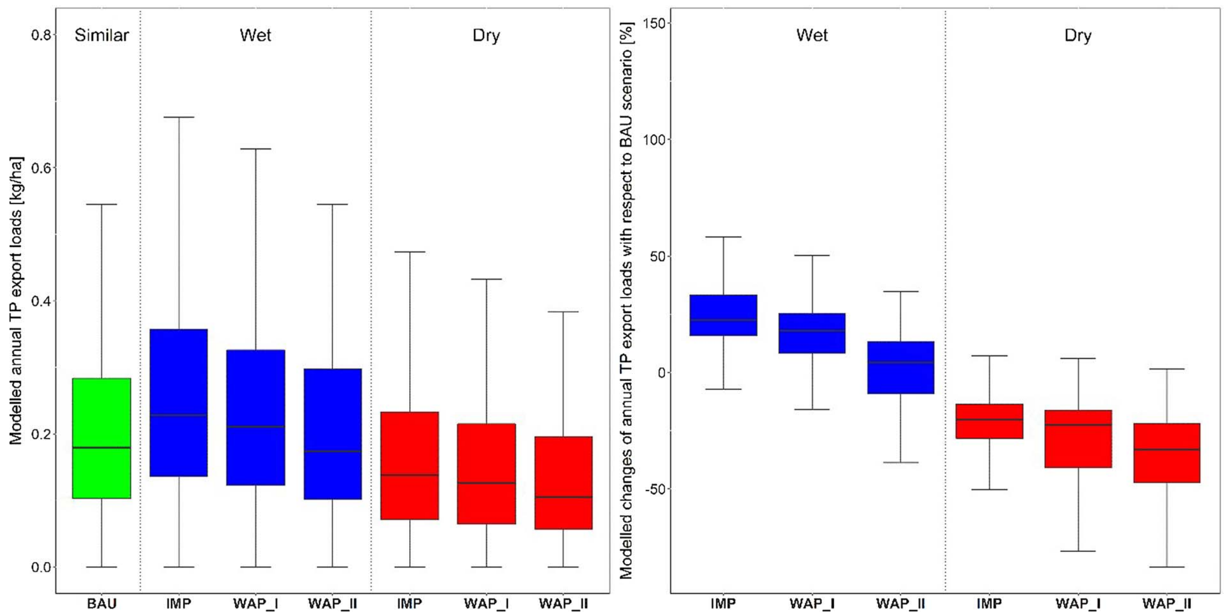

Fig. 8. Comparison of modelled annual TP export loads between scenario BAU (with climate change scenario similar) and alternative policy and climate scenario combinations. The left figure shows boxplots of total TP export loads for each catchment. The right figure shows the relative changes of a pairwise comparison between BAU and the other scenarios for all the catchments. Note: The whiskers' length of the boxplots is up to 1.5 times the range of the 25%–75% quartiles; outliers are excluded.

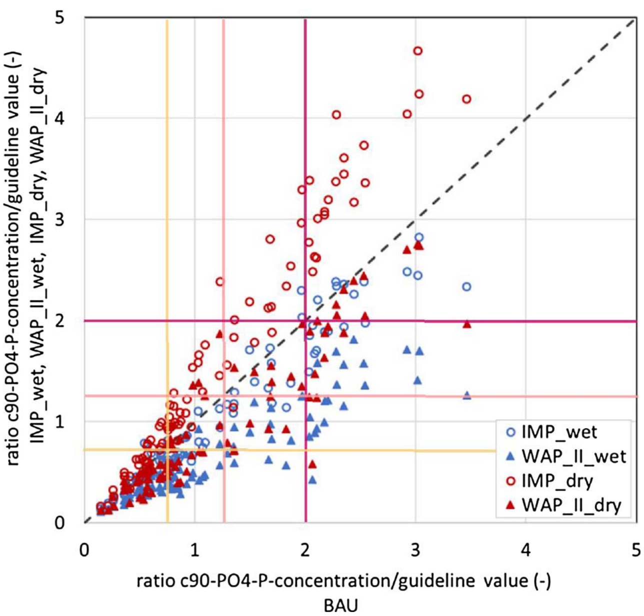

Fig. 9. Comparison of the ratio between modelled phosphate concentrations and environmental quality standards (EQS) in catchments dominated by agricultural land use between BAU and alternative policy and climate scenario combinations. Horizontal and vertical lines indicate borders between no exceedance of EQS and potential exceedance (model concentrations to EQS = 0.7, orange), potential exceedance and exceedance of EQS (modeled concentration to EQS = 1.3, pink) and exceedance to severe exceedance of EQS (modeled concentration to EQS = 2.0, red). (Spatial details: Supplementary material J). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

the scientific literature. For example, Finger et al. (2010) modelled −28% and +39% in autonomous adaptation of N fertilization for two contrasting climate scenarios in a Swiss case study until 2100. Climate change adaptation may increase pressures on water quality from increasing livestock density and fertilization as well as conversion from grassland to arable land in the UK (Fezzi et al., 2015). For the EU level, Leclère et al. (2013) model increasing fertilization rates of up to 22% as adaptation response to climate change. Variations are between 0% and 32% for the first and third quartile among the EU regions and two climate scenarios.

Adaptation to more favorable production conditions does not necessarily increase nutrient emissions if adaptation measures improve plant nutrient uptake (Huttunen et al., 2015). PASMA[grid] is driven by outputs from the bio-physical process model EPIC and shows declining intensity for several regions under the *dry* scenario. However, the opposite appears under *wetter* conditions in most regions. This has − ceteris paribus − two major implications for future policy design. On the one hand, there may be efficiency gains from adapting agri-environmental premium levels towards changing productivity in order to reduce windfall profits and to maintain participation rates − with likely implications on the social justice of payment distributions. On the other hand, changing yield potentials and corresponding nutrient uptake may require adaptation of legal nutrient thresholds and eventually fertilization schedules within the WFD legislation.

4.1.2. Climate change and policy interference

Results show the need to consider climate change impacts when designing water protection policies. We applied two policy portfolios that represent current trends in policy design, i.e. a mix of environmental subsidies and technical thresholds, and hence seem to be both realistic and accepted by stakeholders. The major difference between *WAP_I* and *WAP_II* is the spatial coverage with a focus on water quality hot-spots in *WAP_I* and the national territory in *WAP_II*. *WAP_II* leads to higher total emission reductions. However, despite the large heterogeneity in private land use costs and environmental effects among NUTS-3 regions (Fig. 10) private land use costs per unit reduction in nutrient emissions show diverging patterns for both nutrients under either *wet* or *dry* conditions. Costs increase by 50% (TP) and 10% (TN) from *WAP_I* to *WAP_II* under *wet* conditions. Under *dry* conditions, however, private land use costs per unit emission reduction increase from *WAP_I* to *WAP_II* by about 80% for TP but decrease by 20% for TN.

Different patterns for TN and TP result from distinct emission pathways and the regional targeting in *WAP_I*. High phosphate concentrations and exceedances of EQS for phosphate are mainly found in catchments with high shares of arable land vulnerable to soil erosion, which is a function of slope, field length, soil type and share of erosion-prone crops (Zessner et al., 2017). *WAP_I* supports measures on erosion

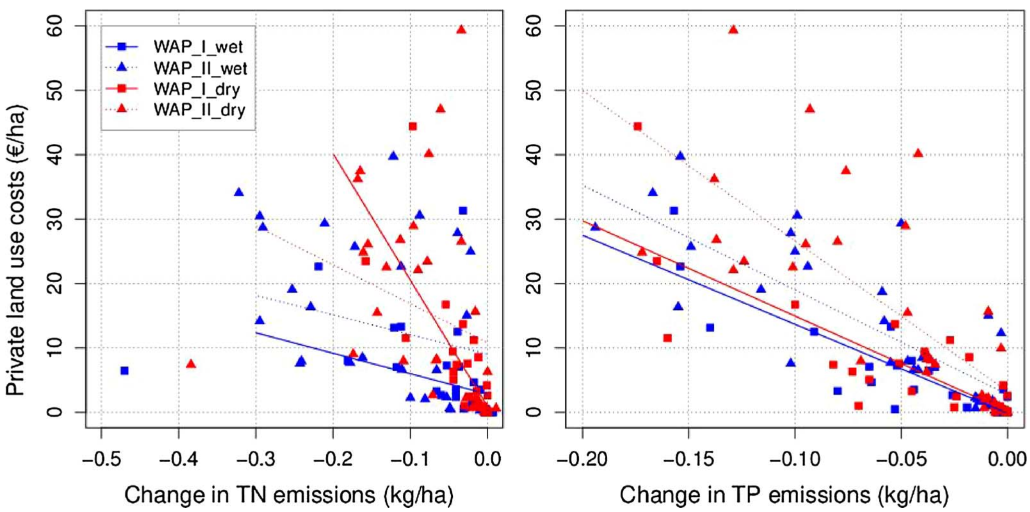

Fig. 10. Annual private land use costs for reductions in total nitrogen (TN) emissions (left) and total phosphorus (TP) emissions (right) of WAP_I and WAP_II policies for a wet and dry situation compared to the respective IMP scenario at NUTS-3 level. Note: Lines indicate linear trends of the respective scenario. R^2 for TN/TP are 0.17/0.80 (WAP_I_wet), 0.18/0.69 (WAP_II_wet), 0.62/0.69 (WAP_I_dry), 0.10/0.57 (WAP_II_dry) (Data: Supplementary material K).

abatement and sediment trapping (e.g. restriction of maize, buffer strips) in these catchments. Area specific effects of P emission reduction measures therefore are higher compared to regions less vulnerable to soil erosion and to exceedance of EQS. This is well reflected in the scenario results for both climate scenarios *wet* and *dry*. Case study results confirm that P emission reduction measures (erosion abatement, buffer strips) on a small but selected portion of the land may lead to a disproportional reduction in nutrient loads (Kovacs et al., 2012).

For N the situation is more complex. Areas with high N-surpluses and low precipitation and runoff are most sensitive to high nitrate concentrations in rivers and thus to EQS exceedance (Zessner et al., 2017). However, denitrification during soil-subsoil passage may significantly reduce N emissions into surface waters (Zessner et al., 2005). It tends to increase under dry conditions due to an increasing groundwater retention time (Behrendt et al., 1999) resulting in lower amounts of soil N surplus reaching the surface water. Therefore, private land use costs such as generated by the *WAP* policy portfolios to reduce soil N surplus and corresponding N concentrations in surface waters tend to decrease under *dryer* conditions. Targeting policies to regions vulnerable to high N-concentrations, i.e. regions with high soil N surpluses and low runoff, is reasonable from the standpoint of ambient water protection but may become less effective in case of transported loads because of higher soil/subsoil denitrification. Significantly reduced runoff in scenario *dry* further increases this tendency. It finally leads to lower private land use costs per unit reduced emissions even if regions not vulnerable against EQS exceedance are included into a management concept (*WAP_II*). Nevertheless, the overall effect of the chosen *WAP* scenarios to reduce N emissions is very limited in the IIMF.

To conclude, future precipitation patterns affect total nutrient emission levels as well as the effectiveness of emission abatement policies. Both TN and TP emissions are substantially lower under *dry* than *wet* conditions for all policy scenarios. However, risks for exceeding EQS and private land use costs for TN and TP emission reductions per unit nutrient are higher under *dry* than *wet* (Table 2). Long-term budget planning for water protection policies, including payments for environmental services, needs to acknowledge climate change impacts.

4.1.3. Regional targeting for different environmental objectives under climate change

Another policy implication is the need for well-defined environmental objectives. At least two need to be taken into account: i) protection of ambient water quality to achieve EQS in local groundwater and rivers, and ii) reduced long distance transport of nutrients − here to avoid Black Sea eutrophication. With respect to the first objective, a regional targeting strategy such as implemented in *WAP_I* obviously is advantageous. Enlarging the policy measure to regions with already acceptable EQS by definition cannot have any effect on this specific objective. With respect to the second objective − i.e. to manage total nutrient loads in a large watershed such as the Danube basin − regions with low costs for nutrient load reductions need to be detected. As shown in this study, these regions coincide with EQS hot spots with respect to P emissions and the same regional targeting strategy can contribute to both objectives. However, it is not the case for N emissions under certain climatic conditions (cf. Zessner et al., 2005). *WAP_II* results in lower private land use costs per unit N emission reduction under *dry* than *WAP_I*. Consequently, local water quality protection may require a different regional targeting than the protection of receiving seas in the case of N emissions. With respect to loads exported towards the Black Sea, soil N surpluses in regions with high runoff, i.e. regions with low likelihood of local EQS exceedance, have to be specifically targeted as these regions deliver a significant share of the loads while private land use costs per unit emission reduction are relatively lower than in dryer regions (cf. Kroiss et al., 2006).

Previous studies confirm the importance of regional targeting of policies to reduce nutrient emissions from agriculture but do rarely highlight the challenge of diverging water quality objectives (e.g. Kuhr et al., 2013). In comparison to our results, Gren et al. (1997) modelled similar patterns of cost reductions for P but different results for N when enlarging targeting areas. They showed increases of costs per unit emission reduction of up to 300% for both P and N emissions with a shift of coordinated nutrient reduction policies from the international to the national level in the Baltic Sea region. It indicates considerable differences in nutrient reduction costs among nations and large cost reduction potentials from international cooperation. Fröschl et al. (2008) modelled cost-effectiveness of technology-oriented N management measures for the Danube watershed. Costs of N surplus reductions − which is different from emission reductions − are between 5 and 24 €/kg compared to private land use costs of 30–40 €/kg in our analysis. The management measures triggered by *WAP* policy portfolios impact crop choice and intensity (e.g. reduced cropping intensity, altered crop rotations) and combine measures for N emission reductions with measures for P emission reductions in bundles, which is typical for agri-environmental programs. This obviously leads to higher private land use costs per unit emission reduction than improved nutrient management technologies such as presented by Fröschl et al. (2008).

We present private land use costs and environmental effects for both TN and TP separately although nutrient emission reductions are complementary. Some synergies are likely for N and P reduction policies from agricultural sources due to the interdependencies of nutrients in plant physiology. It would meet claims to control both nutrients to prevent eutrophication of fresh water resources (Paerl et al., 2014). Typical to agri-environmental programs in the CAP, the policy portfolios of *WAP_I* and *WAP_II* do not only tackle N and P emissions but other environmental objectives such as habitat quality or greenhouse-gas emissions. This may explain the rather low nutrient emission reductions from *WAP* policies in some regions. Even dedicated measures to control nutrient emissions can serve other environmental objectives (Balana et al., 2011). For example, buffer strips impact the appearance of cultural landscapes, determine habitat quality, protect flood prone areas and can supply agro-fuels (Christen and Dalgaard, 2013). Reduced maize shares in crop rotations are a mean to manage pest infestations such as Diabrotica virgifera virgifera – a challenge to some Austrian regions with risks of EQS exceedance (cf. Feusthuber et al., 2017). Some Austrian provinces reacted with legal limits on maize shares in crop rotations. This example proves that maize share restrictions assumed in WAP_I and WAP_II are plausible policies. To conclude, our results compare *WAP* policy effects across regions under climate change. An example of an alternative and, with respect to multiple environmental benefits, more meaningful cost-benefit analysis is provided by Martin-Ortega et al. (2015).

Blanke et al. (2017) analyze the single and combined impacts of climate change and fertilization on N leaching from maize and wheat fields for different European agro-climatic zones in the year 2040. Two zones that represent Austria, i.e. Alpine and Continental North, show slightly more importance of future N fertilization as driver for changes in leaching than climate change. Results from the IIMF confirm that land use is a decisive driver of water quality. P emissions are mainly determined by crop choices and resulting soil cover. N emissions mainly result from soil N surplus. However, changes in runoff – impacted by future precipitation patterns – can overrule effects from land use change. Although both researchers and stakeholders in our study regarded the *WAP* policy portfolios – particularly *WAP_II* – as rather tight, impacts on nutrient emission loads and finally river loads and concentrations appear limited especially in respect to N. Again, changes in runoff patterns induced by climate change can even have a higher impact. One reason is that the mainly voluntary measures do not induce sufficient land use changes in all vulnerable zones. Another reason are the combined impacts of nutrient surpluses, emissions, and dilution. *Dry* conditions tend to reduce the diffuse emission loads of nutrients but concentrations in rivers at low flow may increase due to reduced dilution of point emissions from waste water treatment plants. For example, Sinha et al. (2017) show substantially increasing N loads from anticipated increases in precipitation sums and extreme events under climate change in the US. We show that higher soil erosion and nutrient leaching under *wet* conditions could be partially offset by dilution effects from higher runoff. It may improve local water quality but may not contribute to total nutrient load reductions to the Black Sea.

4.2. Methodological limitations

A thorough discussion of the IIMF is presented in Zessner et al. (2017). Here, we will focus on uncertainties resulting from the model interfaces and the implementation of climate and policy scenarios.

Coupling of stand-alone modelling tools is difficult due to deviating model structures and objectives, systems boundaries, and spatial and temporal dimensions. The major interfaces in the IIMF link crop yields from EPIC with PASMA[grid], runoff from the TUW model with MONERIS, and nutrient surpluses and land use from PASMA[grid] with MONERIS. The latter integrates all information to estimate future nutrient emissions and loads at catchment level. Error propagation from one model to the other decreases the robustness of results along the modelling chain, which challenges integrated impact modelling in general (Wilby and Dessai, 2010). The calculation of nutrient surpluses has been particularly demanding. For example, PASMA[grid] in its initial version prevents nutrient surpluses in most cases by exactly matching crop nutrient demand and fertilization while optimizing regional land use and livestock production. It is unclear whether the diffusion of precision farming, technological change and market developments allow current farming systems to approach towards such efficient nutrient management in the future. We adapted the nutrient balances in PASMA[grid] to achieve a more realistic representation of surpluses. It combines the high spatial resolution of PASMA[grid] data with MONERIS nutrient demand parameters that have been developed from expert knowledge and empirical data. Nevertheless, challenges with respect to nutrients modelling remain. Blanke et al. (2017) highlight the importance of robust N intensity trajectories in studies on N leaching potentials. EPIC and PASMA[grid] consider three intensity classes that are distinct mainly by fertilizer rates and timing of application. The optimal choice is determined by socio-economic and biophysical conditions including market price assumptions, climate change impacts, and water protection policies. It is consistent with hydrological modelling – a major advantage of the IIMF. Nevertheless, real fertilization levels are specific to each farm. Model validation shows a good fit to observed fertilizer application rates (Zessner et al., 2017) but aggregation biases are still possible. PASMA[grid] covers a broad set of adaptation measures. However, lacking coverage of all plausible measures that improve nutrient uptake (e.g. soil improvements by liming, pest management, cultivar improvements, inter-annual adaptation of fertilizer application) can overestimate nutrient emissions from agricultural fields (cf. Huttunen et al., 2015).

The IIMF is based on sequentially coupled stand-alone models. It allows to utilize the benefits of single models while keeping the integrated model structure simple. Disadvantages are the lacking representation of feedbacks among systems. We compare the two policy portfolios under climate change with respect to their private land use costs in PASMA[grid], i.e. the changes in market revenues and variable production costs in complying with the policies, and their nutrient emission effects in MONERIS. Comparing the cost-effectiveness of particular management practices to achieve particular nutrient emission targets (e.g. Balana et al., 2015), or computing marginal abatement cost curves would either require a demanding iterative modelling procedure with the existing IIMF or a fully integrated model instead of the rather loosely coupled models in the IIMF (cf. Antle et al., 2001).

PASMA[grid] represents land use and livestock production in detail but does only take a limited number of stylized farming system characteristics into account (e.g. organic and conventional farming, farm size, crop rotations). In PASMA[grid], all farm resources are portrayed at a region level. It represents regional heterogeneity but not heterogeneity among farm types or even individual farms (Balana et al., 2011). Regional case studies with bio-economic farm models can represent individual farms (e.g. Schönhart et al., 2016) and are complementary tools and enable testing of aggregation biases. Surveys are a mean to empirically analyze adaptation behavior of farmers (e.g. Mitter et al., 2018). They complement modelling studies and may support the choice of assumptions such as required in the IIMF.

Impacts of water protection policies and climate change can be systematically assessed by keeping certain policies and market prices unchanged among the policy scenarios and by comparing results to the *BAU* scenario. This is a major advantage to comparisons based on a historic reference scenario. However, it only holds if the specific policies and climate beyond *BAU* do not impact the market price and policy assumptions. With respect to market impacts, alternative policies in Austria can be seen as unimportant to international agricultural markets under the small country assumption. With respect to climate change impacts, this assumption is less robust. Climate change will be observed globally and will have impacts on global markets. Assuming different climate change scenarios but equal market conditions – as is typical to climate change impact and adaptation studies – biases results. Consistent alternative price scenarios have been absent so far but may be available in the future within the framework of shared socio-economic pathways (Riahi et al., 2017).

A methodological shortcoming that limits the applicability of the IIMF to certain management measures and that increases the uncertainty of scenario outcomes is the restriction of spatial details in

Table A1
Policy measures of four scenarios.

	Policy	BAU IMP	WAP_I	WAP_II
Market **regulation and direct payments (CAP 1. pillar)**	Production quotas (e.g. dairy quota)	Not available		
	Coupled direct payments	Not available		
	Single farm payment	Regional payments		
	Cross compliance: e.g. Nitrate directive[1] N.... Nitrogen at field level (ha)	Max. 100 kg N/ha per application	Max. 80 kg N/ha per application	Max. 80 kg N/ha per application
		Max. N according to Annex 3	Max. N according to Annex 3	Max. N according to Annex 3
		Max. 170 kg N/ha with organic fertilizers	Max. 150 kg N/ha	Max. 150 kg N/ha
			No maize, soybeans, sugar beets, potatoes, and pumpkin on areas > 8% slope close to surface waters[2,4]	No maize, soybean, sugar beets, potatoes, and pumpkin on areas > 8% slope close to surface waters[3,4]
	Greening	Maintenance of permanent grassland	Maintenance of perm. grassland	Maintenance of perm. grassland
		5% ecological focus areas	5% set-aside	5% set-aside
		Crop rotation restrictions	Crop rotation restrictions like BAU and max. 50% maize	Crop rotation restrictions like BAU and max. 33% maize
Rural development (CAP 2. pillar)	Less favored area payments	Available	Available	Available
	Agri-environmental program (ÖPUL)	*Premium levels and standards according to ÖPUL for the following measures[5]:*	Like BAU, *additionally (regional):* +25% premium levels[2] for greening of arable land, direct and mulch seeding, preventive surface water protection, limitation of yield-increasing inputs, and organic farming	Like BAU, *additionally (national):* +25% premium levels[3] for greening of arable land, direct and mulch seeding, preventive surface water protection, limitation of yield-increasing inputs, and organic farming
		Environmentally sound and biodiversity-promoting management		
		Limitation of yield-increasing inputs		
		Greening of arable land − intermediate crops		
		Direct seeding and seeding on mulch		
		Preventative surface water protection on arable land		
		Organic farming		
Waste water treatment		Total phosphorus < 1 mg/l N removal > 70% (current standards)	Total phosphorus < 0,5 mg/l[2] N removal > 85%[3]	Total phosphorus < 0,5 mg/l[3] N removal > 85%[4]

[1] The IIMF does not consider the following standards for manuring: fertilization schedule, slope limitations, water logging, minimum distance to open water, manure storage.
[2] Target regions: insufficient water quality according to *REF* model results.
[3] Available on all Austrian territory.
[4] Target regions: regions with these crops available in *IMP*.
[5] See Table 1 for the resulting list of management measures in the IIMF.

MONERIS to sub-catchment levels. We define HRUs with bio-physical data at 1 km resolution and model land use at sub-municipality level with a post-optimization downscaling procedure to 1 km in PASMA[grid]. Despite such high resolution model results, assumptions on the spatial land use distributions within an HRU may bias model results. Biases from localization in MONERIS may be overcome with raster based sediment and nutrient transport models (Verstraeten et al., 2006; Kovacs et al., 2012) but they are not available on a country scale so far. Future investigations shall address sub-catchments identified as vulnerable in respect to water pollution from erosion and shall improve routines to localize erosion abatement measures. It shall include short-term extreme rainfall events, which are not covered by the IIMF's daily to annual temporal resolution.

The IIMF and most of its components are static. Model results for the reference year and 2040 shall represent two equilibrium states. This simplification may bias results if substantial cause-effect delays are longer lasting than climate change or market dynamics. In real world policy making, cause-effect delays challenge the implementation of water protection policies (Volk et al., 2009).

5. Conclusions

A major advantage of the IIMF applied in this article is its ability to quantify combined impacts of climate change, policies and economic framework conditions in a consistent way. Particularly the endogenously modelled agricultural adaptation increases consistency compared to previous work. This shall facilitate the communication of results relevant to decision makers. The scenario assessment allows conclusions on runoff, land use, and water quality under climate change and for particular policies.

The IIMF results in large spatial heterogeneity of flow regimes and crop yield potentials − two major inputs to model land use and water quality in the IIMF. Consequently, the currently rather uncertain changes in precipitation patterns will be decisive for future water quality. The bio-physical production conditions of a region determine the magnitude of impacts from policies and climate changes on land use. Choices on crop rotations appear to be less impacted by climate change than choices of intensities. Water protection policies based on voluntary agri-environmental programs impact land use by reducing cropping intensity (i.e. mainly fertilization) and increasing direct seeding, cover crops, and buffer strips, but regions with high opportunity costs, eventually under *wet* climate conditions, require substantial payments to spur changes. On the contrary, policies should take declining opportunity costs of extensive production under a *drier* future climate situation into account as well.

Future development of precipitation is decisive for nutrient concentrations and loads in Austrian rivers. Increasing precipitation tends to increase river nutrient loads with increasing impacts on receiving standing water (i.e. Black Sea). Decreasing precipitation leads to increasing river concentrations and tends to increase vulnerability of local water bodies in respect to the exceedance of EQS for nutrients. Climate change induced autonomous adaptation measures in agriculture seem less important than potential impacts of changes in precipitation and runoff according to the IIMF results. Nevertheless, implementation of specific water protection measures can overcome negative climate impacts and significantly reduce river loads and concentrations in case of P. Examples are bans of maize and soybean on steep fields close to rivers to control emission sources or buffer strips to control nutrient transport.

IIMF results show the interrelation of water protection policies and climate change. While *drier* climatic conditions increase the vulnerability of local water bodies, targeting of policies to vulnerable regions gains in importance. In case of *wetter* conditions, exported loads tend to increase while local concentrations tend to decrease. In such a situation, the protection of local water bodies is less demanding, but water protection policies to reduce exported loads and therefore incentives on a national level gain in importance.

The IIMF proved to be suitable to quantify climate change and policy impacts on nutrient emissions and water quality. Future research will focus on the improvement of the technical implementation of model interfaces. Uncertainty assessments will include enhanced sensitivity analysis and ensemble modelling.

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Appendix A

Appendix B. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.landusepol.2018.02.031.

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