This is the preprint version of the contribution published as:

Carolus, J.F., Bartosova, A., Olsen, S.B., **Jomaa, S.**, Veinbergs, A., Zīlāns, A., Pedersen, S.M., Schwarz, G., **Rode, M.**, Tonderski, K. (2020):
Nutrient mitigation under the impact of climate and land-use changes: A hydro-economic approach to participatory catchment management
J. Environ. Manage. **271** , art.110976

The publisher's version is available at:

http://dx.doi.org/10.1016/j.jenvman.2020.110976

Nutrient mitigation under the impact of climate and land-use changes: A hydro-economic approach to participatory catchment management

This is a preprint of an article published in the *Journal of Environmental Management* (Vol. 271, Oct. 2020, 110976) which is accessible under the following link: https://doi.org/https://doi.org/10.1016/j.jenvman.2020.110976

Authors: Johannes Friedrich Carolus[a,b,c], Søren Bøye Olsen[b], Alena Bartosova[d], Seifeddine Jomaa[e], Artūrs Veinbergs[f], Andis Zīlāns[g], Søren Marcus Pedersen[b], Gerald Schwarz[c], Michael Rode[e], Karin Tonderski[h]

[a] Corresponding author

[b] Department of Food and Resource Economics, University of Copenhagen, Rolighedsvej 25, 1958 Frederiksberg C, Denmark.

[c] Thuenen Institute of Farm Economics, Bundesallee 63, 38116 Braunschweig, Germany

[d] Swedish Meteorological and Hydrological Institute (SMHI), 60176 Norrköping, Sweden

[e] Department of Aquatic Ecosystem Analysis and Management, Helmholtz Centre for Environmental Research, Brückstraße 3a, 39114 Magdeburg, Germany

[f] Latvia University of Life Sciences and Technologies, 19 Akademijas Street, Jelgava, LV-3001, Latvia

[g] Faculty of Geography and Earth Sciences, University of Latvia, Jelgavas iela 1, Rīga LV-1004, Latvia

[h] IFM Biology, Linköping University, SE 581 83 Linköping, Sweden

KEY WORDS: Agri-Environmental Measures; Baltic Sea Region; Catchment Water Quality Management; Cost-Effectiveness; HYPE Catchment Model; Participatory Approach.

ABSTRACT: Although nutrient loads of streams and rivers is a recognised ecological problem, implemented mitigation measures are often not cost-effective, whereas cost-effective measures are not necessarily feasible or preferred by local stakeholders. The objective of this study was to evaluate the cost-effectiveness of mitigation measures using a methodology that includes a participatory process and social learning to safeguard their successful implementation. By combining Cost-Effectiveness Analysis, case-specific hydrological modelling and a bottom-up approach, we analysed the impact of 16 nutrient mitigation measures under current and future climate and land-use conditions. The considered measures were suggested by local stakeholders in three European catchments, namely the Swedish Helge, the Latvian Berze and the German Selke rivers. The results suggest that the cost-effectiveness of measures not only depends on their design, specific location and the conditions of the surrounding area, but is also affected by the future changes the area may be exposed to. Climate and land-use changes do not only affect the cost-effectiveness of measures, but also shape the overall nutrient loads and potential target levels in a catchment. Apart from highlighting the importance of local variability and possible near-future changes, the suggested CEA approach demonstrates how nutrient mitigation can be addressed in a more participatory, cost-effective and context-specific manner.

1 INTRODUCTION

Eutrophication is mainly caused by anthropogenic nutrient inputs, namely nitrogen (N) and phosphorus (P), and represents a major problem for the water quality and ecology of streams, rivers and the Baltic or North Sea (Artioli et al., 2008; Kiedrzyńska et al., 2014; Reusch et al., 2018; Saux Picart et al., 2015). Only half of the European water bodies reached a good ecological status by 2015 (EEA, 2018; Voulvoulis et al., 2017), and the Baltic Sea is still classified as unacceptably eutrophic, with problem areas even expanding (Fleming-Lehtinen et al., 2015). Furthermore, the issue of nutrient pollution may become more severe in the future. Although Olesen & Bindi (2002) suggested that climate change can have positive impacts on agricultural production in northern Europe, they found that this would come at the cost of, amongst others, increased nutrient leaching. In addition to an intensified agricultural production (see also Wulff et al., 2014), nutrient loads may increase due to changing precipitation patterns (Bartosova et al., 2019). On top of that, the rise in temperature may also increase aquatic ecosystems' vulnerability to nutrient loads and thus further aggravate the eutrophication problem (Glibert et al., 2014).

To reach good ecological status in all European water bodies by the year 2027 at the latest, the EU Water Framework Directive (WFD) explicitly recommends Cost-Effectiveness Analysis (CEA) as one economic tool to select the most appropriate measures (European Commission, 2000). CEA is a method to assess and compare the performance of frameworks by contrasting their monetary cost per non-monetary effect unit (Pearce et al., 2006). The economic rationale is that decision-makers who wish to achieve a certain change, for instance the removal of a given amount of nutrients, should aim to do so at the lowest possible cost which is ensured by selecting the measures with the highest cost-effectiveness.

CEA is commonly applied in various settings in environmental planning and management. Apart from its wide application to assess nutrient mitigation approaches (e.g. Cardenas et al., 2011; Elofsson, 2010, 2012; Gachango et al., 2015; Hasler et al., 2014; Konrad et al., 2014), CEA has also been used to analyse fisheries policies (Kronbak & Vestergaard, 2013), water-saving measures (Berbel et al., 2011), improvements of the river morphology (Klauer et al., 2015), or in the context of both water quality management and climate change mitigation (Nainggolan et al., 2018). However, CEAs of large spatial and socio-cultural scales can typically not fully account for context-specific variabilities. Both the costs and effectiveness of measures differ across countries and regions (e.g Konrad et al., 2014; Wulff et al., 2014). These variabilities depend on the way measures are implemented and on environmental, landscape, and management characteristics (Balana et al., 2015; Martin-Ortega, 2012). Since suitable and consistently collected agri-environmental effectiveness data is scarce (Mauchline et al., 2012), CEA analyses are often developed and assessed based on standardised conditions, which may lead to wrong conclusions and thus falsely informed decision-making (Balana et al., 2011). When dealing with environmental effects, even information on standardised conditions is often not available, for instance due to unknown ecological response functions for ex-ante assessments, and must therefore be assessed based on expert or stakeholder knowledge (Oinonen et al., 2016; Perni & Martínez-Paz, 2013). When multiple environmental goals are considered, this problem becomes even more accentuated. As an example, Nainggolan et al. (2018) considered the cost-effectiveness of a joint implementation of nutrient and greenhouse gas mitigation measures in the Baltic Sea region. For practical reasons, they had to simplify their model to consider only six defined abatement measures, assuming that every measure is feasible in any catchment.

In practice, environmental policies introduced in a top-down decision-making process may be unsuitable or unaccepted in the local context where mitigation measures are realised (Carr, 2002; OECD, 2018; Smith, 2008). One reason is that the stakeholders may have different problem perceptions and favour measures that meet different goals. They may thus favour the implementation of actions "*of their own design and own initiative*" (Stupak et al., 2019, p. 301). Policy or project designs that involve local stakeholders may be superior to those set by analysts or decision-makers much further away from the issue than those most affected by the environmental problems or the implementation of measures (OECD, 2018).

Consequently, the informative value of CEA for decision-making can be improved when considering local variabilities and priorities in a more accurate way, i.e. measures which are practically feasible in and targeted to the local context, including the assessment of case-specific costs and effectiveness variables (Balana et al., 2015; Balana et al., 2011). Additionally, comprehensive analyses need to include future scenarios as changing nutrient loads due to future climate and land-use changes may affect the cost-effectiveness of mitigation measures (Jackson-Blake et al., 2013). Some previous studies have integrated hydrological modelling with cost estimates to assess the spatial variability in cost-effectiveness of measures to improve catchment water quality (cf. Lescot et al. 2013). However, few studies have used analyses of spatial cost-effectiveness integrated with a stakeholder decision making process focused on achieving good status of water. In this paper, we combine three approaches, namely CEA, hydrological modelling and participatory learning regarding river basin management and water status. We assess the nutrient reduction cost-effectiveness of stakeholder preferred management measures in specific catchments and under changing land-use and climate. Each of the applied approaches is, from a theoretical point of view, similar to other studies. For instance, Wood et al. (2015) assessed the cost-effectiveness of mitigation measures, and Jomaa et al. (2016) modelled nutrient loads under the influence of changing land-use and agricultural practices. Furthermore, Balana et al. (2015) combined hydrological modelling and CEA, and Perni & Martínez-Paz (2013) linked CEA and participatory methods, though they had to rely on how stakeholders perceive the effectiveness of potential measures.In this paper we draw on primary effectiveness estimates under the influence of both planning-related (e.g. model assumptions or design parameters, such as scope, application intensity or type of planting) and external (e.g. future climate and land-use, baseline and geophysical components) impacts and the corresponding costs of the measures. We consider measures that were suggested by local stakeholders concerned with both the water related challenges and the implementation of mitigation measures in order to improve the ecological status of water in their catchments. In other words, the starting point of the analysis was measures which are perceived as both relevant and practically feasible in the local context, and thus were more likely to be adopted by the stakeholders responsible for their successful implementation.

A case study approach was used in three catchments in Germany, Latvia and Sweden. With the underlying aims of both enhancing participatory planning and providing evidence for decision-makers, we generate assessments of the cost-effectiveness of nutrient mitigation measures selected by the involved stakeholders. The paper contributes to the literature by suggesting and applying an approach which (a) supports participatory environmental management processes and social learning, and (b) provides further evidence in terms of case-specific cost-effectiveness estimates of locally relevant nutrient mitigation measures in three different catchments while considering the implications of land-use and climate changes. The results are of direct relevance to inform stakeholders and their perceived understanding of effectiveness (Micha et al., 2018; Stupak et al., 2019), as well as to improve decision-making in the context of the WFD and its demand for finding cost-effective solutions and involving all concerned parties.

2 METHODOLOGY

The selection of the measures considered in this paper emerged from a participatory and social learning approach in three case areas within the context of the BONUS MIRACLE project. The project's overall objective was to enhance participatory environmental management by collaboratively finding management scenarios that provide multiple ecosystem service benefits including reduction of nutrient enrichment in the Baltic Sea (BONUS MIRACLE, 2018; Tonderski, 2018).

2.1 CASE CATCHMENTS

The three catchments are situated in countries of the Baltic Sea Region. While the Helge River (Sweden) and the Berze River (Latvia) discharge directly or indirectly into the Baltic Sea, the Selke River (Germany) flows indirectly into the North Sea. They were selected due to their insufficient ecological status of water according to the WFD, partly due to an excess of nutrients (cf. Abramenko et al., 2013; Jiang et al., 2014; VISS, 2017), their susceptibility to flooding, and because of the availability of abundant water quality and water flow data to facilitate hydrological modelling with a reasonable accuracy. The Swedish catchment Helge is 5 and 10 times larger than the Berze and Selke catchments, respectively, with a larger share of forest cover, particularly in the upstream part (Table 1).

Table 1: Overview of case catchments and their hydrological data availability (Richnau et al. 2013; Jiang et al. 2014; BONUS MIRACLE 2016).

Characteristics	Berze River (Latvia)	Helge River (Sweden)	Selke River (Germany)
Population (proportion living in urban areas)	26,500 (50%)	131,428 (97%)	33,000 (80%)
Size (km²)	872	4,725	463
Sub-basins	15	29	180
Elevation (m a.s.l.)	< 130	0 – 235	605 – 653
Mean precipitation (mm/year)	630	810	660
Mean temperature (°C)	5.6	7.4	9
Share of agricultural/forest land (%)	57/39	22/64	35/52
Available water quality monitoring data	1 flow and 15 water quality measurement stations	12 flow and 22 water quality measurement stations	3 flow and 3 water quality measurement stations

2.2 MEASURES AND THEIR SELECTION

A co-inquiry process with local stakeholders and researchers was initiated and focused on issues related to achieving good ecological status in each river and how to mitigate those. The approach was used i) to respond to central requests from stakeholders to gain understanding to select the measures that improve environmental quality in the most cost-effective manner, and ii) to follow the prominent role participatory processes have in the context of the WFD. Between three and five workshops per case area within the period of two years were realised between 2015 and 2017. Stakeholders discussed priority issues regarding good ecological status of water, and selected options which, in their opinion, are preferred to reduce specific environmental stressors in the local catchment and are feasible in the local context. The intention was to include stakeholders that are directly concerned with the negative state of the water resources or its resolution, and hence a broad range of different stakeholders and stakeholder agencies participated in the process. New stakeholders were invited in response to emerging issues and to suggestions from participants.

In particular stakeholders represented agriculture, water management, nature conservation and forestry[1], but in one catchment also wastewater management authorities were involved. Stakeholder priorities differed between the catchments and also included environmental problem beyond nutrient enrichment. It should be emphasized that no restrictions regarding which measures to select were set. A closer description of the co-inquiry process can be found in Powell et al. (2018).

During the workshops, the discussions to find suitable actions were stimulated by providing scientific analyses of the assessed consequences and impacts of these actions as determined from the socio-economic and hydrological modelling (BONUS MIRACLE, 2017). Those were presented to the stakeholders in an interactive visualisation tool (cf. Neset et al., 2019). Based on the new input and emerging findings, measures were added, discarded or adjusted by the stakeholders, for instance due to being considered too costly or ineffective, already implemented, or violating existing land property rights. The analysis in this paper reflects the final selection of suggested measures, restricted to those that (i) directly affect the N and P loads in the respective river catchment, and (ii) of which the impact on nutrient loads can be simulated by the utilised hydrological water quality model[2].

2.3 Cost-Effectiveness Analysis (CEA)

In the context of this paper, CEA assesses the performance of river basin management measures by contrasting their monetary cost per non-monetary effect unit; here the effect was defined as the reduced riverine transport of N and P. By providing a ranking of the relative performance, CEA enables to find the measure with the lowest cost-effectiveness ratio (CER). Doing so will ensure that the desired target is realised at the lowest possible costs. The CER is calculated by dividing the annual average costs with the corresponding average annual effect (e.g. Berbel et al., 2011). For any measure to be assessed, C represents the costs and E_i an effectiveness indicator (with the subscript i indicating if the CER describes the removal of N or P). The CER_i is thus computed as follows:

$$CER_i = \frac{C}{E_i} \quad (1.1)$$

A higher CER indicates a lower cost-effectiveness, and vice versa. The approach implies that CEA can only provide information on which of alternative measures to select, given that one must select at least one (Kronbak & Vestergaard, 2013; Pearce et al., 2006). CEA can therefore not conclude if a measure is worth to be implemented or not, as, for instance, done by Cost-Benefit Analysis (CBA). However, given that CEA is commonly applied to find the best solution to achieve a pre-determined environmental target at the lowest cost, the starting point of CEA is typically a target which is assumed to generate benefits justifying the costs, such as the reduction of eutrophication in the Baltic Sea (cf. Hyytiäinen et al., 2013). Due to the aim of supporting both participatory processes and decision-making by providing knowledge in the form of precise numbers highlighting the variabilities of the cost-effectiveness ratios of different stakeholder-suggested measures, the analysis focuses on the mere comparison of the measures' CER. This allows to provide and compare the performance of all measures in the local context under different conditions, such as future climate and land-use change. Other than, for instance, Balana et al. (2015) or Vinten et al. (2017), we

[1] The participating stakeholder groups and agencies are listed in the supplementary material.

[2] While nutrient enrichment was considered the major issue in all case areas, the measures belong to stakeholder-suggested pathways targeting the supply of multiple aquatic ecosystem service benefits. The analysis in this paper focuses on the measures with a direct impact on the load of N and P (detailed descriptions are provided in Carolus et al., 2018; Powell et al., 2018).

therefore abstain from including an optimisation problem to find the single best and least-cost solution, but narrowly consider the suggested design of the measures, including their fixed size and specific locations.

The potential costs of the measures were assessed based on available literature and supplemented by stakeholder information, for instance in terms of land prices and gross margins which influence forgone income. The costs of a measure are typically not equally distributed throughout the years of implementation, for instance due to one-time implementation costs but recurring costs for maintenance or forgone income. Following the budgetary approach introduced in Jacobsen (2007), the annual marginal costs are generated by multiplying the investment costs related to such measure with the annuity factor and adding the results to the recurring, opportunity and/or saved costs (Formula 1.2).

$$Annual\ marginal\ costs = I \cdot \alpha_{ni} + RC + OC - SC \quad (1.2)$$

where:

I = Investment costs
α_{ni} = Annuity factor (with n = lifetime of the assessed measure, and i = interest rate)
RC = Recurring costs
OC = Opportunity costs
SC = Saved costs

The CEA in this paper draws on effectiveness indicators as simulated by the *Hydrological Predictions for the Environment* (HYPE) model.

2.4 Hydrological modelling

The effects on nutrient transport were quantified using the hydrological model HYPE, which is a dynamic, semi-distributed process-based integrated catchment model that is developed by the Swedish Meteorological and Hydrological Institute (SMHI) under a Creative Commons open source licence (Lindström et al., 2010). The model represents the spatial heterogeneity of the studied area by dividing the whole catchment into sub-basins based on the topography and the river network. Each sub-basin is, in turn, divided into different classes, where detailed spatial variability (such as soil and crop types, land-use and tile drain conditions) is considered. These classes are the smallest computational units in the HYPE model and are not assigned to specific geographic locations within a sub-basin. HYPE permits to differentiate the soil profile into up to three soil layers with different depths and characteristics, and its model parameters can be general (i.e. same values for the whole catchment), land-use- or soil-dependent, or specified for a region. The simulations are conducted on a daily time step.

HYPE simulates both water quantity and quality. It can be thus used to assess the impact of various mitigation measures on water flows and water quality concentrations and loads. A separate HYPE model application was developed for each case catchment using daily stream flow and biweekly nutrient concentrations data from the period 2005-2014 (Table 2). In this study, the total nitrogen (TN) and total phosphorus (TP) concentrations and loads as well as the streamflow were used for the CEA. Due to large variations in summer and winter, the numbers depict the annual average numbers.

Table 2: Average goodness of fit statistics for streamflow (Q), total nitrogen (TN), and total phosphorus (TP) concentration in the case study areas between 2005-2014 (validated model results). Nash-Sutcliffe efficiency (NSE) of 1 indicates a perfect match, whereas 0 is an optimal PBIAS value.

	Berze	Helge	Selke
Q: NSE	0.81	0.81	0.82
Q: PBIAS (%)	-1.7	9.2	-4.2
TN: PBIAS (%)	-0.4	3.3	-1.6
TP: PBIAS (%)	-2.4	4.5	-3.7

Two types of model outputs were used to calculate the reduction in nutrient loads as a consequence of implementing the selected measures. First, *nutrient loads at the outlet of the case catchment* represent a cumulative impact of processes in the whole catchment as well as in the streams and include retention and transformation processes that occur with nutrient transport in landscapes, soils, and the receiving waterbody. Second, *local nutrient loads* represent a load produced within each sub-basin and discharged through local tributary streams to the main river that drains each sub-basin, i.e. the effect of mitigation measures only in the sub-basin where they are implemented. HYPE simplifies the actual river network and groups all local tributary streams together to a representative “local stream” rather than simulating each tributary separately. If there is more than one tributary in a sub-basin, the local concentration produced by the model would represent a flow-averaged concentration. This might affect the evaluation of measure effectiveness on a local scale as the division does not take the locations of proposed measures within the specific sub-basin into account.

To estimate the effectiveness of the selected mitigation measures used in the CEA, the measures are mimicked by changing the different input variables and parameters of 20 processes considered in the HYPE model (see Lindström et al., 2010 for an overview of the processes, including the equations and variables behind). The processes are related to the catchment conditions, design parameters, on-site management practices, etc. For instance, three parameters are used to simulate the effectiveness of buffer strips: the fraction of a watercourse that is surrounded by agricultural land and has a buffer zone, the fraction of agricultural land that lies close to the watercourse and is thus affected by the buffer, and the filtration capacity of the buffer strip. The effectiveness can then be generated by comparing the nutrient loads of the baseline scenario with the scenario that includes the mimicked measure.

The HYPE model mimics future climate and land-use scenarios by changing the inputs that drive the model (e.g., precipitation and air temperature or land-use area), while the model parameters stay the same as in the calibrated model (cf. Lindström et al., 2010; SMHI, 2017, 2018). The potential changes in future climate and land-use which are considered in this paper and ultimately reflected in the effectiveness assessment using the HYPE model outputs are described in detail in the following sections.

A total of four different scenario combinations are considered in this study, namely (1) current climate and current land-use, (2) future climate (“RCP8.5”) and current land-use, (3) current climate and future land-use (“SCENAR2020-II”), and (4) future climate and future land-use.

2.4.1 Climate change scenario “RCP8.5”

For analysing the impact of future climate changes, we use the climate pathway RCP8.5 from the CMIP5 global climate model ensemble (e.g. Riahi et al., 2011) with the *EURO-CORDEX* projections (cf. Jacob et al., 2014) to regional climate models. The selected regional climate models, “*WRF-IPSL-CM5A-MR*” (WRF) and “*RCA4-CanESM2*” (RCA4), show the highest increase of the average precipitation and average temperature in the region from the *EURO-CORDEX* ensemble. The selection was based on changes in summer by the mid-century, and under the assumption that such changes influence the nutrient dynamics the most. 1991-2010 is selected as the current reference time period, whereas the future model period for evaluating the impact is centred around 2030 (2021-2040). The regional climate models were bias-adjusted to the local reference data using a distributed-based scaling approach (cf. Yang et al., 2010). The projected changes in precipitation and temperature in the three case catchments are quite different for the WRF model, whereas the projected changes for the RCA4 are fairly similar (Figure 1). In this study, the results are based on the average of the two regional climate models, from now on referred to as "RCP8.5".

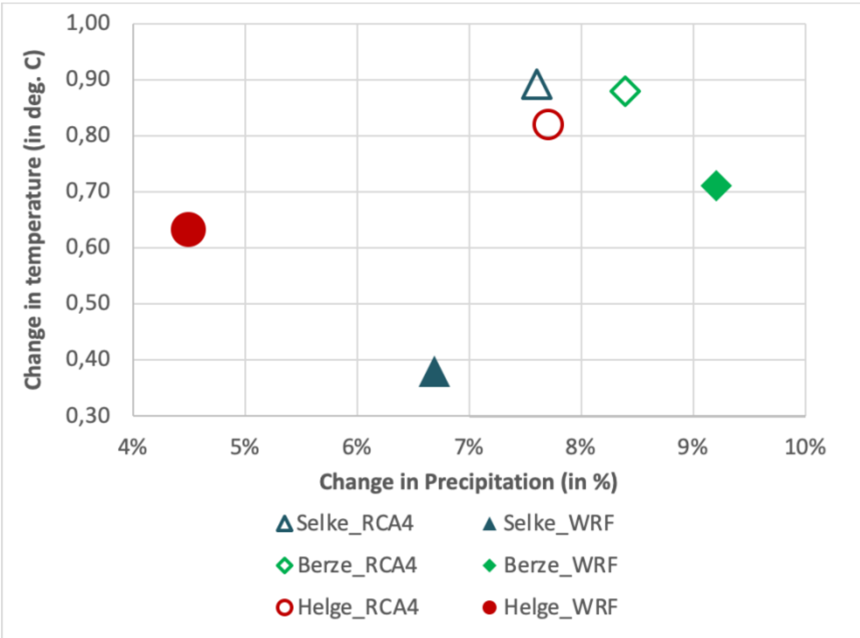

Figure 1: Change in precipitation and temperature between the current and future model period for each case study and climate model, based on the two regional climate models RCA4 and WRF.

2.4.2 Land-use scenario "SCENAR2020-II"

Land-use changes are derived from projected changes in the agricultural sector for the Baltic Sea region, as reported in Graham & Powell (2011). Those results consider the changes in agricultural land uses in the EU up to 2020, as defined in the *SCENAR2020-II* scenario (Nowicki et al., 2009). In this study, we use the projected land-use changes of the "*Reference*" pathway. This pathway projects a net loss of 1% of the agricultural areas for Helge, which was modelled as a change from agriculture area into extensive grasslands without soil cultivation and fertilisation. For the Berze catchment, the pathway implies that the area used for rape seed cultivation increases by 7%, the cereal production area decreases by 1.4%, whereas the overall land available for agricultural production decreases by 1%. In contrast, the Selke catchment would only experience a small change towards more bioenergy crops, which is considered in the modelling approach by converting 4% of the agriculture area into bioenergy crops area (mainly maize).

3 Results

To interpret the results obtained from CEA, costs and effectiveness units are considered both separately and combined. When describing the effectiveness of a measure, we thus refer to the nutrient load reduction in terms of the measure's reduction effect (e.g. kg P or N ha^{-1} $year^{-1}$). In contrast, if the costs and effectiveness units are considered simultaneously, we explicitly refer to the cost-effectiveness or the CER of a measure (where a lower CER implies a higher cost-effectiveness, cf. Formula 1.1).

3.1 THE MEASURES AND THEIR COSTS

The stakeholder-suggested measures considered in this paper are outlined in Table 3. It is interesting to note that the selection consists of rather standard measures. Some other measures were also selected to achieve multiple goals, including a better connectivity for fish, improved flood dampening and a higher biodiversity (cf. Carolus et al., 2018), but could not be included due to limitations of the current version of the HYPE model in simulating such measures. Yet, both the presented measures and the specific implementation scope are accepted and perceived as both practicable and relevant by the different local stakeholders. All selected measures but one (the small municipal wastewater treatment plants) target diffuse pollution and address nutrient loads from different sources. For instance, stormwater ponds target urban areas while contour ploughing concerns agriculture areas. While buffer strips or riparian zones were, in different forms, suggested in each of the three catchments, and the reduction of fertiliser application was suggested in the Selke and the Berze catchments, the remaining measures are diverse. Apart from the fertiliser reduction alternatives in the Berze catchment and the buffer strips with different widths in both the Selke and Berze catchments, the measures are not considered as being mutually exclusive.

The applicability of the cost structures (cf. Formula 1.2) in the respective local context was validated by local stakeholders. However, as costs of measures may vary significantly depending on the design and construction parameters, the lifetime of investments or local conditions, the upper and lower bounds of the annual costs are likewise indicated to allow for a subsequent sensitivity analysis. The bounds draw on the lowest and highest cost assumptions and predicted market prices.

Table 3: Nutrient mitigation measures for three catchments in Northern Europe, their scope and the annual average cost per unit. The underlying sources and assumptions of the cost assessment are provided in the supplementary material.

Case area	Measure	Scope	Unit of scope	Annual costs [c] (in €/unit) and cost range
Berze (LV)	Crop rotation	33,220	ha	434 (87 - 867)
	Grassland extensification [a]	4,131	ha	455 (304 - 612)
	Organic farming	1,305	ha	116 (232 – 23)
	Buffer strips (2+5m) [b]	540	ha	616 (297 - 1,074)
	Buffer strips (2+10m) [b]	741	ha	616 (297 - 1,074)
	N and P fertiliser reduction (5%)	31,741	ha	39 (32 – 45)
	N and P fertiliser reduction 20%)	31,741	ha	93 (76 – 106)
	Municipal Wastewater Treatment Plants (100 – 300 PE)	3	plants	14,179 (2,836 – 28,358)
Helge (SE)	Re-meandering	165	km	5,756 (169 - 11,568)
	Riparian zones in forest landscapes (30m)	442	ha	875 (44 - 1,400)
	Stormwater ponds	15	ha	17,819 (8,390 - 34,807)
	Wetlands	450	ha	1,594 (343 – 2,538)
Selke (DE)	Buffer strips (10m)	273	ha	829 (326 – 1,514)
	Buffer strips (20m)	546	ha	829 (326 – 1,514)
	Contour ploughing	23,012	ha	0
	N fertiliser reduction (20%)	23,810	ha	13 (10 – 14)

[a] The measure (and the subsequent hydrological modelling) describes extensive grassland which is converted from previously intensively managed grassland; [b] buffer strips with a width of 5m (or 10m) along the main river course, and 2m buffer strips on both sides of the melioration ditches; [c] the annual costs are calculated based on equation 1.2. The used discount rates are 2% for Selke (Albert et al., 2017), 3.5% for Helge (Swedish Transport Administration, 2016), and 5% for Berze (Ministry of Finance Latvia, 2016).

The costliest measures in terms of average annual costs per scope unit are stormwater ponds and re-meandering in the Helge catchment, as well as the municipal wastewater treatment plants in the Berze catchment. The measures with the lowest costs per hectare are contour ploughing, which was assumed to be cost-neutral, and the reduced application of N fertilisers in the Selke catchment. From the perspective of a potential budget, implementing crop rotation with the given scope results in the highest costs (€14 million), followed by the reduced fertiliser application of 20% (€3 million) and grassland extensification (€1.9 million). When comparing the annual costs for buffer strips across the catchments, the highest costs are in the Selke catchment (€829 ha^{-1}), followed by the Berze catchment (€616 ha^{-1}). The riparian zones in forest landscapes in the Helge catchment cost around €875 ha^{-1}. The different costs of some measures can be explained by the different levels of forgone income across the regions[3] (due to varying agricultural variables like cultivated crops, crop yield or market prices) and the measures' dependency upon individual characteristics (like construction parameters or labour costs).

3.2 THE EFFECTIVENESS OF MEASURES

The effectiveness of the measures was assessed based on the change of the total load of N and P at the catchment outlet (Table 4). This means that the assessment accounts for the natural retention that takes place and reduces the net effect of a measure over the entire catchment. This assessment approach is required to describe the actual consequences for a downstream receiving water body, for instance the Baltic Sea.

Table 4: Reduction effect of nutrient mitigation measures at the river outlet in three Northern European catchments (in kg year-1 scope unit-1, rounded to 3 significant figures).

		Reduction effect at the river outlet (in kg $year^{-1}$ $unit^{-1}$)							
	Climate scenario	current		RCP8.5		current		RCP8.5	
Location	**Land-use scenario**	current		current		SCENAR2020-II		SCENAR2020-II	
	Measure	N	P	N	P	N	P	N	P
Berze (LV)	Crop rotation	-0.05	0.00	-0.06	0.00	-0.06	0.00	-0.07	0.00
	Grassland extensification	0.90	0.01	0.92	0.02	0.51	0.01	0.53	0.02
	Organic farming	-1.15	0.01	-0.72	0.01	-1.07	0.01	-0.63	0.01
	Buffer strips (2+5m)	n.a.	1.31	n.a.	1.39	n.a.	1.30	n.a.	1.37
	Buffer strips (2+10m)	n.a.	1.71	n.a.	1.78	n.a.	1.69	n.a.	1.76
	N and P fertiliser reduction (5%)	1.26	0.00	1.32	0.00	1.31	0.00	1.37	0.00
	N and P fertiliser reduction (20%)	4.05	0.01	4.23	0.01	4.18	0.01	4.37	0.01
	Municipal Waste Water Treatment Plants	232	61.2	234	61.2	233	61.2	234	61.2
Helge (SE)	Re-meandering	18.2	0.00	18.2	0.00	15.1	0.00	18.2	-0.03
	Riparian zones in forest landscapes	n.a.	0.06	n.a.	0.06	n.a.	0.06	n.a.	0.06
	Stormwater ponds	65.2	6.19	65.2	6.19	65.2	6.19	65.2	6.19
	Wetlands	24.5	1.31	26.7	1.37	23.3	1.31	25.6	1.35
Selke (DE)	Buffer strips (10m)	n.a.	12.0	n.a.	14.6	n.a.	12.3	n.a.	14.9
	Buffer strips (20m)	n.a.	6.58	n.a.	8.00	n.a.	6.74	n.a.	8.19
	Contour ploughing	n.a.	0.13	n.a.	0.16	n.a.	0.13	n.a.	0.16
	N fertiliser reduction (20%)	0.77	n.a.	0.82	n.a.	0.76	n.a.	0.81	n.a.

The assessment demonstrates that the effectiveness of measures varies greatly, whereas future scenarios tend to affect the effectiveness rather marginally, with a few exemptions like a less effective grassland extensification or more effective buffer strips. The reduction effects of buffer strips differed across the catchments, reflecting the different level of implementation (e.g. scope and intensity) as well as differences in local conditions (e.g. climate, soil and slope gradient). Under current climate and land-use, buffer strips remove between 7 and 12 kg P $year^{-1}$ ha^{-1} in the Selke catchment but only between 1 and 2 kg in the Berze catchment. The riparian zones in the Helge catchment are assumed to be implemented in forest landscapes only, which results in a P removal of 0.06 kg $year^{-1}$ ha^{-1}. In the same way, a 20% fertiliser reduction (N and P fertiliser) in the Berze catchment results in a load reduction of roughly 4 kg N and 0.01 P $year^{-1}$ ha^{-1}, whereas the same reduction intensity (only N fertiliser) results in reduction of less than 1 kg N $year^{-1}$ ha^{-1} in the Selke

[3] the forgone income per hectare, in terms of the respective gross margin, amounts to €464 in the Selke, €300 in the Berze and €262 in the Helge catchment. This cost component is considered whenever the measure is implemented on land which could (and would) otherwise be used for agricultural production.

catchment. When considering the measures that target diffuse pollution, stormwater ponds in the Helge catchment show the highest N reduction effect per hectare and year, whereas the highest effect on P reduction is achieved by buffer strips in the Selke catchment. It is worth noting that the nutrient reduction achieved by the measure implementation describes the relative impact compared to the current land-use in the respective location. Therefore, measures may result in negative mitigation rates, i.e. their implementation leads to an increase in the nutrient load.

To gain insights regarding the measures' ability to improve water quality in the local setting, i.e. the change of the load produced within each sub-basin and discharged to the main river without accounting for the natural retention over the entire catchment, Table 5 provides the local load reduction of selected measures. While, on average, a larger scope of a measure results in higher load reductions, this does not apply in all cases. Due to simulating the effectiveness of measures with scopes and locations suggested by local stakeholders, measures are not evenly distributed throughout the river catchment. For instance, wetlands in the Helge catchment are implemented in 19 out of 180 sub-basins with sizes between 3 and 79 ha, whereas the effectiveness and the size of the measure does not necessarily relate in a linear way. 79 ha of wetlands in one sub-basin reduce the total local N load by approximately 11 kg year-1, whereas 18 ha wetlands in another sub-basin reduce the local load by 25 kg year-1. More precisely, the local effectiveness of wetlands ranges between 0.57 and 43.54 kg N ha-1 year-1, which corresponds to a relative cost-effectiveness range of 271% (Table 5). With more than 1000%, the highest cost-effectiveness range can be observed for the buffer strips in the Selke catchment. This can be explained due to the measure itself, such as its size and specific location, and the associated sub-basin characteristics affecting the hydrological pathways (e.g. land-use, soil, surface). Different local loads are generated in each sub-basin, i.e. each measure receives and thus abates different quantities of P and N loads. Moreover, the following examples demonstrate that further aspects determine effectiveness. In Selke, the buffer strips with a 20m width remove, on average, less P per hectare than the same measure with 10 m width, and in Berze, the effectiveness of reducing fertiliser by 5% is higher than one fourth of the effectiveness when reducing the fertiliser application by 20%. Consequently, not only the different loads within individual catchments and thus the specific location of measures, but also the construction or design characteristics (such as widths and intensities) of measures determine their effectiveness.

Table 5: The local effectiveness of selected measures. The table displays the measures of each catchment showing the highest effectiveness ranges of mitigating N and P, respectively.

Location	Measure	Local load reduction (in kg P/N ha^{-1} year^{-1})				Range (%)	Implementations in sub-catchments (total number)
			min	average	max		
Berze (LV)	Buffer strips (2+10m)	N		No effect			15 (15)
		P	0.03	1.52	4.09	268	
	N and P fertiliser reduction (20%)	N	1.12	3.51	9.21	231	15 (15)
		P	0.00	0.01	0.03	202	
Helge (SE)	Re-meandering	N	1.87	11.08	22.94	190	29 (180)
		P	0.00	0.02	0.08	530	
	Wetlands	N	0.57	15,83	43,54	271	19 (180)
		P	0.02	1.48	3.71	249	
Selke (DE)	Buffer strips (20m)	N		No effect			29 (29)
		P	0.00	26.65	276.73	1,038	
	N fertiliser reduction (20%)	N	0.02	1.90	9.08	476	26 (29)
		P		No effect			

3.3 The Cost-Effectiveness of Measures

The CER essentially reflects the cost associated with reducing the N or P load when applying a certain measure (from now on referred to as CER_N or CER_P respectively). In other words, and under current climate and land-use conditions, the removal of 1 kg N would cost €65, €274, or €317 in the Helge River catchment when implementing wetlands, stormwater ponds, or re-meandering, respectively (Table 6). CER_N and CER_P are thereby considered individually, i.e. all of one measure's costs are assumed to incur for N and P reductions separately, even if both is reduced by the same measure.

While constructed stormwater ponds belong to the most effective measures among those evaluated in terms of both N and P removal, they are also very costly and thus have a higher CER than some alternative measures with a lower effectiveness. Most notably, contour ploughing results in comparably low effects (0.13 – 0.16 kg P $year^{-1}$ ha^{-1}, Table 4) yet is assumed to be cost-neutral. Hence, its CER_P is €0. Overall, the results demonstrate that fertiliser reduction are the most cost-effective measures in the Selke and the Berze catchment to remove N, whereas wetlands is the measure with the lowest CER_N in the Helge catchment. In terms of P leaching, contour ploughing in the Selke catchment, municipal wastewater treatment plants in the Berze catchment, and wetlands in the Helge catchment reveal the highest cost-effectiveness. Crop rotation, grassland extensification and riparian zones in forest landscapes seem inappropriate for the purpose of mitigating nutrients.

Table 6: Cost-Effectiveness Ratios (in €/kg N or P removed) based on the change in total load at the river outlet under different climate and land-use scenarios and using the mean costs for the measures in each catchment (rounded to 3 significant figures). CER_N and CER_P are considered individually, i.e. all of one measure's costs are assumed to incur for N and P reductions separately. The arrows indicate the change from the current climate and current land-use.

	Cost-Effectiveness Ratio (in € kg^{-1} N or P removed)								
	Climate scenario	current		RCP8.5		current		RCP8.5	
	Land-use scenario	current		current		SCENAR2020-II		SCENAR2020-II	
Location	**Measure**	N	P	N	P	N	P	N	P
Berze (LV)	Crop rotation	-7,900	108,000	-6,870 ↑	116,000 ↑	-7,580 ↑	107,000 ↓	-6,590 ↑	114,000 ↑
	Grassland extensification	505	34,500	493 ↓	27,700 ↓	892 ↑	31,900 ↓	852 ↑	26,100 ↓
	Organic farming	-101	11,200	-162 ↓	14,200 ↑	-109 ↓	10,800 ↓	-184 ↓	13,800 ↑
	Buffer strips (2+5m)	n.a.	469	n.a.	444 ↓	n.a.	474 ↑	n.a.	449 ↓
	Buffer strips (2+10m)	n.a.	360	n.a.	346 ↓	n.a.	365 ↑	n.a.	350 ↓
	N and P fertiliser reduction (5%)	31	16,000	30 ↓	17,700 ↑	30 ↓	15,700 ↓	29 ↓	17,400 ↑
	N and P fertiliser reduction (20%)	23	10,000	22 ↓	11,300 ↑	22 ↓	10,000 →	21 ↓	11,100 ↑
	WWTPs	61	232	61 →	232 →	61 →	232 →	61 →	232 →
Helge (SE)	Re-meandering	317	n.a.	317 →	n.a.	380 ↑	n.a.	317 →	n.a.
	Riparian zones in forest landscapes	n.a.	15,500	n.a.	15,500 →	n.a.	15,500 →	n.a.	15,500 →
	Stormwater ponds	274	2,880	274 →	2,880 →	274 →	2,880 →	274 →	2,880 →
	Wetlands	65	1,210	60 ↓	1,170 ↓	68 ↑	1,210 →	62 ↓	1,180 ↓
Selke (DE)	Buffer strips (10 m)	n.a.	69	n.a.	57 ↓	n.a.	68 ↓	n.a.	56 ↓
	Buffer strips (20 m)	n.a.	126	n.a.	104 ↓	n.a.	123 ↓	n.a.	101 ↓
	Contour ploughing	n.a.	0	n.a.	0 →	n.a.	0 →	n.a.	0 →
	N fertiliser reduction (20%)	16	n.a.	15 ↓	n.a.	17 ↑	n.a.	16 →	n.a.

While measures with lower CERs are evidently preferred to those with higher CERs, the total amount of nutrients removed is additionally determined by the implementation scope of any measure. For instance, contour ploughing is assumed to be performed throughout the entire agricultural area of the Selke catchment, hence the scope and the overall effect cannot be extended. If a certain scope is reached, but the amount of removed nutrients is below a set target, other - possibly less cost-effective - measures must be implemented.

3.3.1 RCP8.5 and SCENAR2020-II: The impact of climate and land-use change

Future land-use and climate conditions affects the outflow of N and P from the catchments, resulting in either increasing, decreasing or unchanged total loads of N and P at the river outlet relative to the current condition. This also leads to changes in the cost-effectiveness of the measures[4].

If no measures are implemented, the model results suggest that a change in climate alone (RCP8.5 scenario) would cause an increase in total N and P loads at the river outlet of all catchments, except for the N load in the Helge catchment which, on average, would remain unaffected by climate change[5] (Table 7). The simulated land-use changes tend to decrease the total load with the following exceptions: The N load in Berze and the P load in Selke increase, the P load in Berze remains constant. When assuming both scenarios combined (RCP8.5 and SCENAR2020-II), the total N and P loads in all three catchments increase, except for the N load in the Helge catchment which slightly decreases, compared to the current conditions.

Table 7: Total N and P loads from the study areas at the river outlets (in kg year^{-1}, rounded to 3 significant figures). The arrows indicate the change from the current climate and current land-use.

Total load from the study area at the river outlet (in kg year^{-1})										
	Climate scenario	current		RCP8.5		current		RCP8.5		
Location	**Land-use scenario**	current		current		SCENAR2020-II		SCENAR2020-II		
	Measure	N	P	N	P	N	P	N	P	
Berze (LV)	No measure	828,600	12,700	865,700 ↑	13,100 ↑	830,400 ↑	12,700 ⇨	867,500 ↑	13,000 ↑	
Helge (SE)	No measure	2,377,000	56,300	2,377,000 ⇨	59,500 ↑	2,333,000 ↓	55,900 ↓	2,334,000 ↓	59,100 ↑	
Selke (DE)	No measure	801,600	26,300	844,300 ↑	30,600 ↑	794,900 ↓	28,600 ↑	837,100 ↑	33,100 ↑	

For most of the measures, the two alternative climate and land-use scenarios improve the cost-effectiveness (i.e. lowering the CERs, Table 6). For example, in all three case areas, climate change (with current land-use) generally lowers the CERs, though with more instances of increasing CERs in the Berze River catchment. However, the impact of land-use change (with current climate) is more ambiguous; while the CERs of the measures implemented in the Helge catchment generally increase or remain constant, the impacts are somewhat more mixed for Selke and Berze catchments. When assuming both land-use and climate change, the CERs of all measures in Helge and Selke decrease or remain constant, while the impacts in the Berze catchment are more mixed. Furthermore, similar measures may have different effects across catchments. The CER_P of the riparian zones in the Helge catchment remains constant, whereas the CER_P of buffer strips in the Selke and Berze catchments decrease.

3.3.2 The impact of design and location of measures

Location matters, both in terms of where the measure is implemented and where its effect, and which one, is assessed. Given that the local effectiveness of the measures varies greatly across the different sub-basins (Table 5), the local cost-effectiveness estimates differ accordingly. Figure 2 highlights the variabilities in the local CERs which, in absolute terms, are larger for P mitigation.

[4] Due to narrowly focusing on the CERs of the measures yet considering both N and P mitigation, the analysis deals with each measure's CERN and CERP and the overall target levels separately. With the advantage of highlighting that, for instance, the impact of future scenarios is not always identical for the N and P mitigation of a single measure, considering the CER of removing just one nutrient type may give the impression of an unfavourable cost-effectiveness for measures mitigating both N and P. An alternative is the weighting of both effects, e.g. by drawing on eutrophication potential equivalents (cf. Jacobsen, 2007), such as PO4 equivalence factors (Prammer, 2009). For instance, GHK (2006) indicate that 1 kg of TN in water is equivalent to 0.42 kg PO4, and 1 kg of TP corresponds to 3.07 kg PO4 equivalents. Converting the measures' impacts on TN and TP into a common metric would therefore enable to generate CERs reflecting the mitigation potential of both nutrients of a single measure but would also increase both the uncertainty and complexity of the results.

[5] While this is the case of the considered average model (RCP8.5), the individual climate models RCA4 and WRF result in load changes.

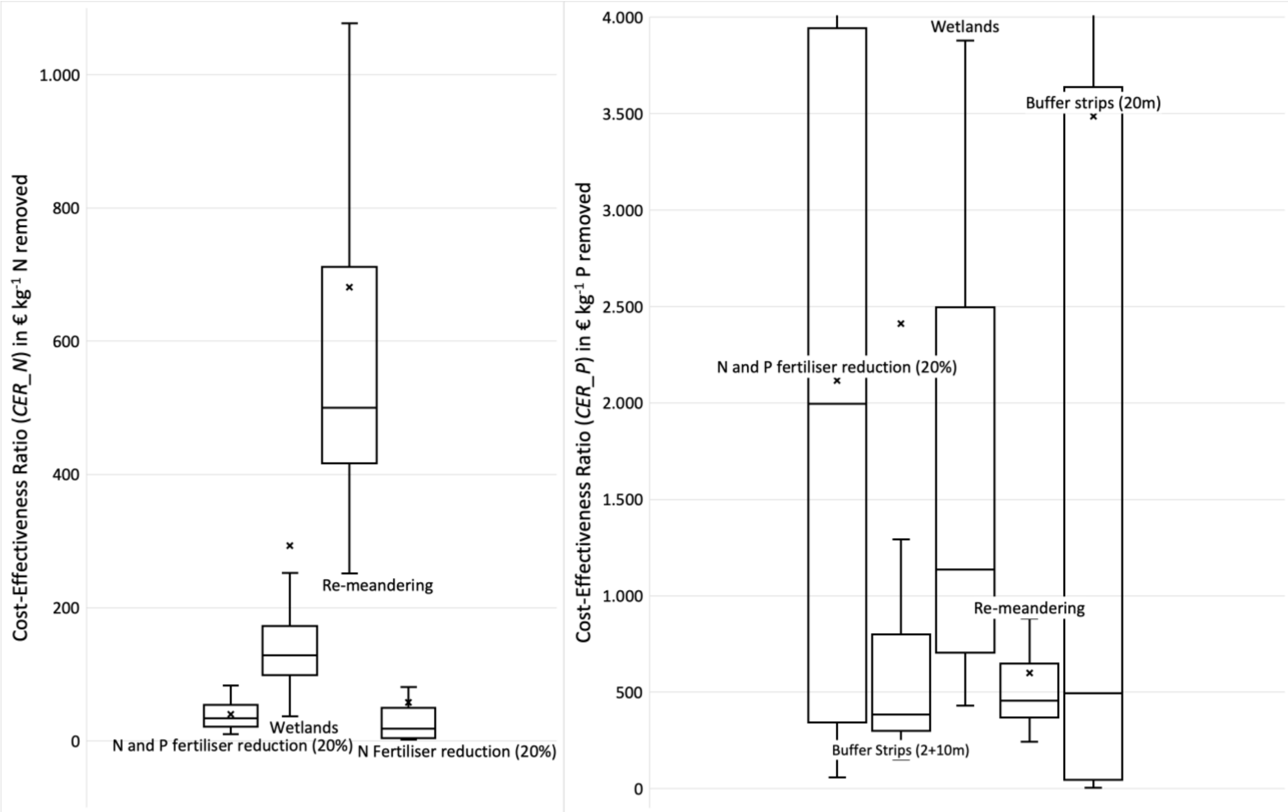

Figure 2: The variabilities in the local cost-effectiveness of mitigating N (left figure) and P (right figure) under current climate and land-use conditions. The figures display the measures of each catchment showing the highest effectiveness ranges of mitigating N and P, respectively. To avoid visual clutter, the upper part of the graph is cropped.

3.3.3 Do the measures fulfil the nutrient mitigation targets?

The focus in this paper is on measures which are accepted and perceived as relevant and feasible by stakeholders in the local context rather than achieving a pre-determined target. Still, Figure 3 and Figure 4 illustrate the total amount of N and P that would be, under current climate and land-use, removed at the respective river outlets (given that all considered measures would be implemented restricted by the scopes as defined in Table 3). Furthermore, both figures set the quantities of N and P removal in context of the associated CERs. This linking is also referred to as a cost-effectiveness ladder (e.g. Jacobsen, 2007). By ranking the measures from the lowest to the highest CER, the graphs show the increasing marginal costs per increase in the quantity of N and P removed. The ladder thus highlights the variability of the measures' performance in the different catchments according to their marginal costs per N and P removal as well as to the total amount nutrients removed at each river outlet. Still, directly comparing the measures across the case studies is hardly possible due to reflecting measures with various designs in different surroundings, and due to all rivers discharging into different water bodies and thus not equally contributing to the identical environmental target.

In terms of the total amount of N abatement at each river outlet per year, the largest quantity is removed by all measures in the Berze catchment ("*Berze_N_5*" or "*Berze_N_20*" in Figure 3), followed by the measures in the Selke and Helge catchments. With respect to the total P removal at each river outlet, the highest amount is removed by the measures in the Selke catchment ("*Selke_P_20*" or "*Selke_P_5*" in Figure 4), followed by the measures in the Berze and the Helge catchment. Interestingly, the measures in the Berze catchment rather address the removal of N than P, which is in line with the required amount of nutrient removal to achieve good ecological status in the river basin. More precisely, given the total N and P loads at the river outlet without any of the considered measures being implemented, the required N concentration of 2.5 mg/l corresponds to an additional yearly reduction of 137,000 kg N year^{-1}, whereas no further abatement of P is required to meet the target concentration set by the Latvian Environment, Geology and Meteorology Centre (2009). With a total removal of 133,000 kg N year^{-1} at the river outlet, the measures of "*Berze_N_20*" are almost reaching the target level. In contrast, implementing all considered measures in the Selke catchment would not meet the targets for good ecological status by far. In the Helge catchment, the P target is already reached without implementing further measures, whereas no N target levels are determined in Sweden. It should, however, be noted that both the target and removal levels refer to the catchment outlet, whereas nutrient loads and ecological states vary substantially across the different sub-basins[6].

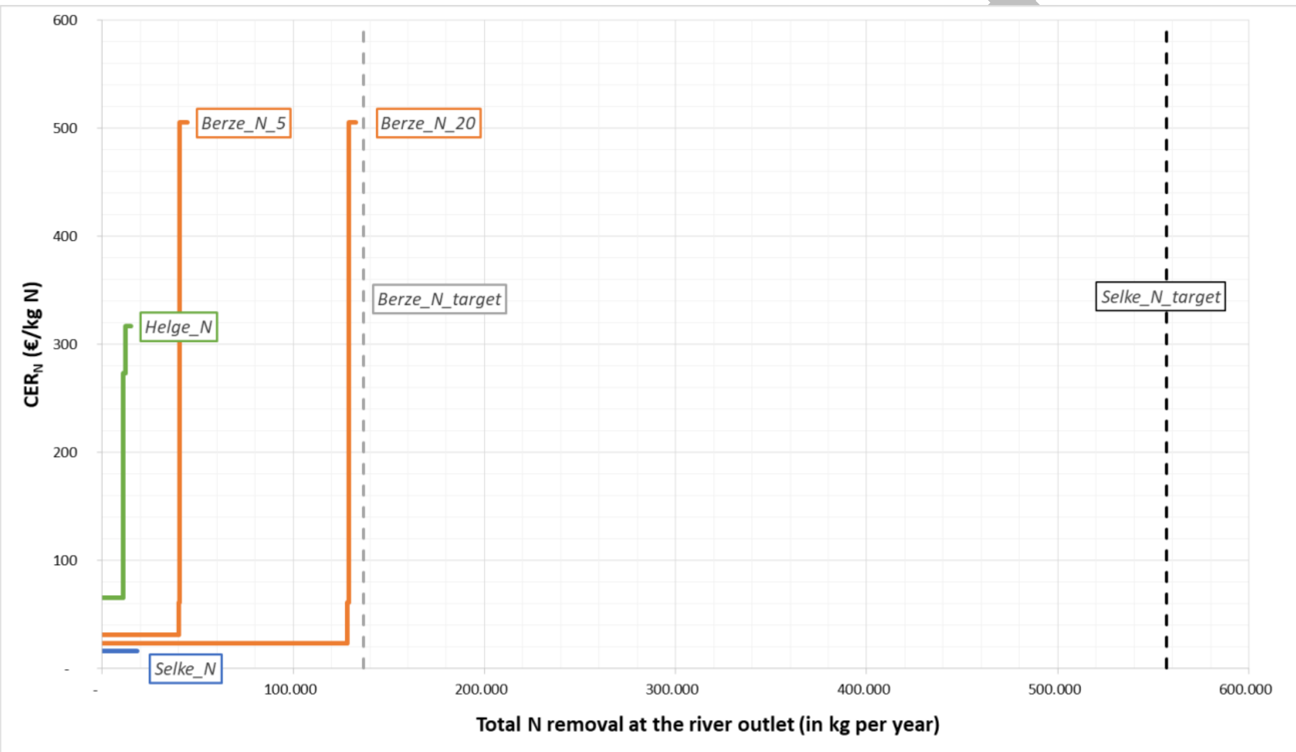

Figure 3: The total N removal (in kg N year^{-1}) at the respective river outlet in context of the measures' scopes and cost-effectiveness ratios (CERs). "Helge_N" and "Selke_N" assume all measures to be implemented in the Helge and Selke catchment, respectively. The mutual exclusive measures in the Berze case study is reflected in "Berze_N_20" (fertiliser reduction by 20% and 2+10m buffer strips, as well as remaining measures crop rotation, organic farming, WWTPs and grasslands) and "Berze_N_5" (fertiliser reduction by 5% and 2+5m buffer strips, as well as the remaining measures). "Selke_N_target" and "Berze_N_target" describe the total N removal required for reaching good ecological status at the Selke and Berze river outlet, respectively. No N target level has been set for the Helge catchment.

When comparing the costs required in each catchment to remove nutrients at the river outlet, the cost-effectiveness ladder highlights that, up to the point at which every suggested measure is implemented in the Helge catchment, both N and P are removed with lower marginal costs in the Selke and Berze catchments, respectively. In the Selke catchment, the total amount of N is removed with marginal costs lower than the one associated with the removal of the identical quantity in the two other catchments. However, implementing all measures in the Selke catchment removes less than 20,000 kg N year^{-1} at the river outlet, and thus substantially less than the amount removed in the Berze catchment. In terms of P removal, implementing all measures in the Selke catchment leads to the highest amount of removal with the lowest

[6] The detailed calculations and results of both the total removal and the required removal to reach good ecological status at the river outlets are provided in the supplementary material of this paper.

marginal costs. This is due to removing close to 3,000 kg P year^{-1} with no additional costs when merely implementing contour ploughing, whereas the total amount of P removed when implementing all measures in the other catchments are less than 700 kg (Helge catchment) and 2,000 kg (Berze catchment) per year.

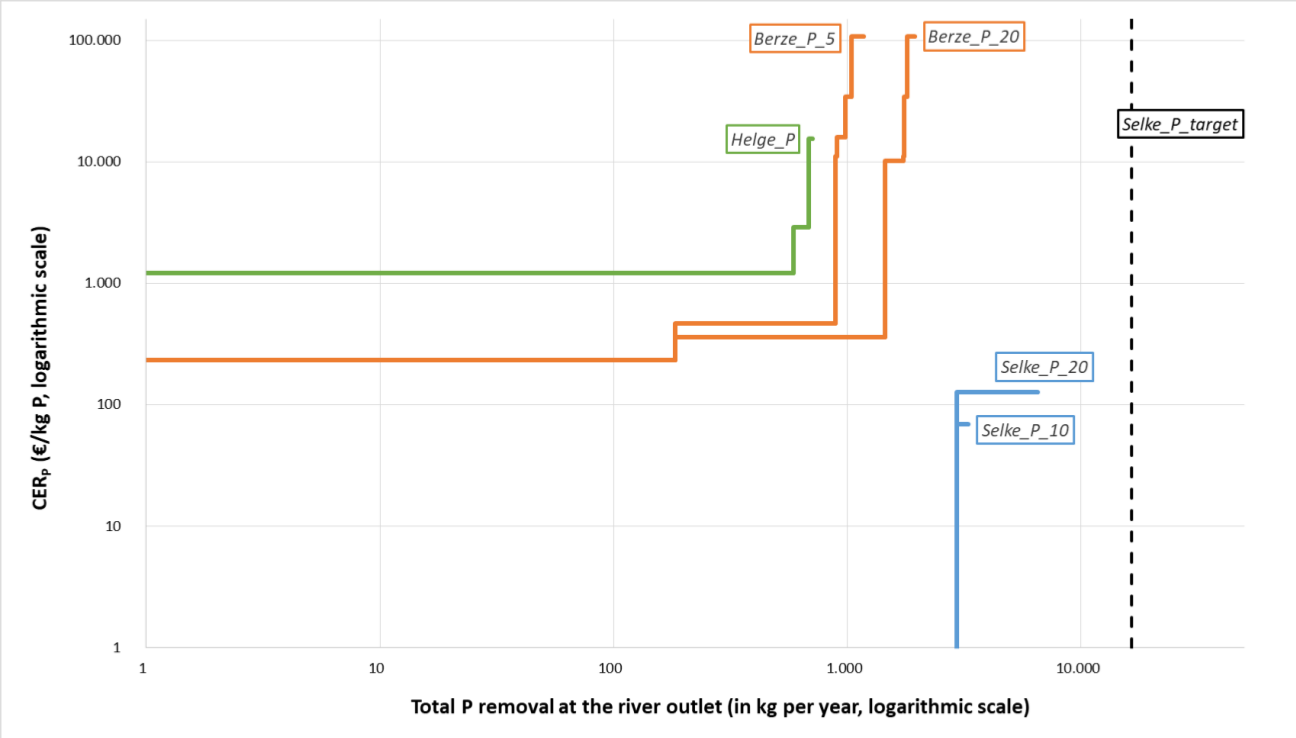

Figure 4: The total P removal (in kg P year^{-1}) at the respective river outlet in context of the measures' scopes and cost-effectiveness ratios (CERs). "Helge_P" assumes that all measures are implemented in the Helge catchment. The mutual exclusive measures in the Berze and the Selke case studies are reflected in "Berze_P_20" (fertiliser reduction by 20% and 2+10m buffer strips, as well as remaining measures crop rotation, organic farming, WWTPs and grasslands), "Berze _P_5" (fertiliser reduction by 5% and 2+5m buffer strips, as well as the remaining measures), "Selke_P_10" (10m-buffer strips and contour ploughing), and "Selke_P_20" (20m buffer strips and contour ploughing). "Selke_P_target" describes the total P removal required for reaching good ecological status at the Selke river outlet. The required removal for the Berze and Helge catchments is negative and thus excluded.

4 Discussion

It is well-known that there is a range of uncertainties that affect the the cost-effectiveness estimates of nutrient mitigation measures and, thus, the conclusions that can be obtained from a CEA. With the underlying aim of supporting participatory processes and decision-making by providing precise estimates of the consequences of stakeholder-selected solution approaches for reducing a present environmental problem, this paper explores the potential impacts of these uncertainties based on three case studies.

4.1 Using CEA in participatory approaches

The measures assessed in this study were suggested by local stakeholders using a bottom-up approach. By providing precise numbers in terms of the measures' cost-effectiveness in reducing nutrient enrichment which was perceived by all stakeholders as problematic, this paper's analysis supported the participatory and social learning process. Particularly, the results (a) demonstrate how variabilities in local conditions or assumptions lead to substantial variations in the resulting cost-effectiveness ratios, and (b) provide easy interpretable and comparable numbers and thus a way to increase stakeholders' understanding of the actual effectiveness of measures which often deviates from the one perceived (Micha et al., 2018; Stupak et al., 2019). Thus, the approach describes a different perspective than e.g. Perni & Martínez-Paz (2013) who combine CEA and participation to select cost-effective measures in context of the WFD, yet they rely on the participants to determine effectiveness of measures. In the context of the stakeholder workshops in which the development of the CEA was embedded in, the results revealed that the effectiveness of some measures is lower than often assumed, which lead to vibrant discussions in terms of both feasible measures and aiming for solutions targeting multiple benefits and not merely the mitigation of nutrients.

Interestingly, implementing all suggested measures with the associated scopes in the respective catchments would not automatically guarantee the nutrient concentrations required to achieve good ecological status (e.g. based on the WFD definition) at the river outlet. Furthermore, the total costs per year required to implement all suggested measures varies substantially across catchments (€22 million in Berze, followed by €2.3 million in Helge and €0.8 million in Selke). However, the analysis (i) only refers to the ecological state and nutrient reductions at the river outlet, (ii) does not incorporate the full set of the stakeholder-selected measures, and (iii) considers measures which also generate additional benefits beyond the mitigation of N and P. Those implications, which are in line with the ones mentioned by Martin-Ortega et al. (2015) in a similar approach in the Scottish context, thus say little about the success of the participatory approach or its relevance for finding the best environmental management solutions. The focus in this paper rather targets the challenge of increasing non-scientific stakeholders' understanding of "*the uncertainties inherent to any scientific research and its inability to provide 'ultimate' answers*" (Martin-Ortega et al., 2015, p. 43), which comes with the limitation of not providing the single best and least-cost solution due to drawing on fixed solution scenarios (e.g. in terms of the measures' selection, size and location) to narrowly reflect the stakeholder suggestions.

Despite of fostering understanding and discussions by providing straightforward numbers which correspond to the stakeholders' expressed interests of achieving environmental improvements in the most cost-effective manner, the approach may decrease the risk of selecting measures which are not relevant or accepted in the local context. For instance, the reduction of livestock, which was selected as one of six measures to reduce nutrient and greenhouse gas emissions investigated by Nainggolan et al. (2018) or Hasler et al. (2012), was not considered by any of the different stakeholders in our study. On the contrary, while cost-effectiveness studies often allocate between 0.5% (e.g. Elofsson, 2010; Gren et al., 2008) and 1% (Mewes, 2012) of the agricultural land to buffer strips, implementing buffer strips with a width of 20m in the Selke catchments, which is considered feasible by local stakeholders and consulted authorities, would represent more than 2% of the agricultural area. This indicates that drawing on knowledge of local stakeholder may not only exclude impractical measures, but also enhances the design of feasible measures. Furthermore, Stupak et al. (2019) find that farmers may follow not only economic reasoning when implementing environmental management measures, but also prefer to implement measures of their own initiative and design. When aiming to find the optimal measures, for instance to achieve the targets set by the WFD, considering stakeholder-suggested measures may thus increase their subsequent adoption. However, while the exemplified livestock plays indeed a relatively minor role in any of the case catchments, drawing on stakeholder-suggestions does not eliminate the possibility of excluding better solutions which are either not known or simply not preferred by the involved stakeholders, e.g. when going against their financial interests. Likewise, measures suggested by stakeholders may also be inappropriate to address the environmental problem of interest, as evidenced by the poor performance of crop rotation, grassland extensification or riparian zones in forest landscapes. Furthermore, measures which are adapted to the local context (and considered under local conditions) are likely to perform differently in a different setting, which reduces the transferability and generalisation of the results and conclusions (Perni & Martínez-Paz, 2013).

Finally, different stakeholders naturally have different interests and understandings (Micha et al., 2018). In this study, this is best evidenced by a strong representation of different stakeholder groups directly related to the forest sector in the Helge case study leading to the suggestion of a measure specifically targeting the forest areas. In contrast, no such measure was selected in the other cases in which only few stakeholders related to forestry participated. As forestry was perceived as key to solve the local environmental problems in the Helge catchment whereas agriculture largely dominated the other two cases, this does not imply that the selection of measures is unfeasible or biased when different sub-sets of stakeholders participate. However, the example underlines that participatory approaches like the one introduced in this paper should be interpreted as reflecting the perceptions and preferences of the included stakeholder groups, but not essentially the interests of all stakeholders or society as a whole.

4.2 The drivers of variations in cost-effectiveness of nutrient mitigation measures

The case study results suggest that the cost-effectiveness of a measure depends on (1) its design, (2) the conditions of the area in which it is implemented, (3) the future changes the surrounding area will be exposed to, (4) its location within a certain catchment, and (5) the parameters forming the basis of the effectiveness assessment.

This study allows us to consider and compare measures with different system designs, such as buffer strips with changed widths assumptions, or different reduction intensities for fertilisers application. Most notably, the effectiveness and scope of measures do not necessarily relate in a linear way and some measures show a decreasing effectiveness when the intensity or width is increased. Speaking in average terms, the CER of reducing the application of fertiliser by 20% is lower than a reduction by 5% (in the Berze catchment). A reduction of 20% on 1 ha of agricultural land would consequently remove fewer nutrients than a reduction of 5% on 4 ha of agricultural land. In the same way, the cost-effectiveness of buffer strips in the Selke catchment is decreasing when increasing their widths. In contrast, wider buffer strips in the Berze catchment lead to a larger effect in terms of nutrient removal per hectare, compared to the same measures with a lower width. While these results may have limited robustness and still depend on various assumptions, they provide useful indications for both policy-development and (economic or hydrological) modelling and confirm the conclusion by Kovacs et al. (2012) who calculate that measures targeting a "*properly selected small proportion*" of a catchment may improve the water quality remarkably (p. 74). When setting environmental targets, policymakers need to consider that achieving small changes on a larger spatial scale can be more beneficial than a change where the alternative is more radical but at a smaller spatial scale. This, however, depends on the respective measure.

Besides of the measures' design, CEA outcomes depend on the specific location in which a measure is implemented, but also on which scale the effectiveness is considered, local or total. The results reveal a substantial local variability of the effectiveness of measures, which is a result of different conditions such as land-use, soil or hydrological pathways which substantially affect a measure's performance. This implies that in environmental planning, local variabilities in terms of the cost-effectiveness need to be considered carefully, and governance instruments should allow for spatially differentiated measures (cf. Hashemi et al., 2018; Konrad et al., 2014). Moreover, the scale at which the effectiveness should be considered depends on the purpose. If the aim is to reduce the impact on a receiving water body, such as the Baltic Sea, one needs to draw on the effectiveness of reducing nutrient load at the river outlet. However, if the objective is to improve the ecological status within a local watercourse, local assessments and the focus on nutrient concentrations are required to design appropriate measures at the best possible locations.

Finally, the assessment reveals how cost assumptions affect the conclusions obtained from this and similar studies' CEAs. In our analysis, when drawing on the upper and lower bounds of the cost estimates, the cost-effectiveness of measures would differ between 48 and 164% in the Berze, 18 – 193% in the Helge, and 83 – 114% in the Selke catchment. Although the general applicability of the average costs of each measure was validated by local stakeholders, the results still depict estimates for specific catchments. And the costs really occurring when implementing measures may differ due to numerous aspects, making the transfer and comparison of results across regions and studies limited. For instance, most measures would affect agricultural production opportunities. In this study, when measures are assumed to be implemented on crop land, their opportunity costs are determined by what kind of crops are, on average, cultivated in that specific region, including their average yields and market prices. These factors differ substantially across locations. Measures may also be located only partly on arable land, as assumed for the average cost estimates of wetlands in this study. Furthermore, costs are determined by the measures' construction methods and management requirements (Balana et al., 2015). For instance, contour ploughing was indicated as cost-neutral by the involved stakeholders. This might (Posthumus et al., 2015; Stevens et al., 2009) or might not (Hein, 2007) be the case in different contexts. In light of studies providing CERs of similar measures, the performance of wetlands in the Helge catchment (€65 CER_N, €1,210 CER_P) falls into the CER-ranges described in BalticSTERN (2013), but are above other estimates in Sweden (with a CERs of €17.5 kg^{-1} N in Hasler et al., 2014; or between ca. €53 and €900 kg^{-1} P in Johannessen, 2015) and Denmark (with ranges between €3 - €31 for N, and €55 - €1150 for P in Gachango et al., 2015). To some extent, those variations can be explained by different cost assumptions. For instance, Gachango et al. (2015) calculate the CER of wetlands with costs between €34 and €194 ha^{-1}. Drawing on those cost estimates would result in comparable CER-ranges in this study.

4.3 THE IMPACT OF FUTURE CLIMATE AND LAND-USE CHANGES

This study's results confirm literature suggesting that changes in climatic conditions and land-use will affect the performance of environmental management measures (Perni & Martínez-Paz, 2013; Zanou et al., 2003). However, the results reveal that the impact of future land-use and climate on the cost-effectiveness of the considered measures is evident, but not necessarily negative. While climate and land-use changes do not affect most measures' CER strongly, especially when comparing the changes to other accrued variabilities, such as the local cost-effectiveness, the impact in subsequent assessments may nevertheless be significant. In other words, while the future scenarios appear to not change the conclusions obtained from the relative performance of the considered measures in terms of their CERs, the total cost to reach a specific nutrient reduction level may change substantially. For instance, future climate leads to grasslands mitigating 0.5 kg N ha-1 instead of 0.9 kg under current climate conditions, which results in an increase of the CER from €500 to almost €900. As this is of direct relevance for both scientific impact assessments and decision-making in regard to finding the most cost-effective solution, both climate and land-use changes should be considered in such analyses.

Furthermore, addressing the challenges which occur due to future land-use and climate change could benefit from applying CEA. The increase of future nutrient loads as suggested by, for instance, Olesen & Bindi (2002) or Reusch et al. (2018) is also reflected by this study's results which demonstrate that the nutrient loads are increasing in all considered case areas due to the impact of climate change, whereas the consequences of land-use changes are more ambiguous. The findings highlight that future changes are not merely affecting the (cost-)effectiveness of measures, but also have a direct impact on politically or ecologically determined nutrient mitigation targets which should get adjusted accordingly. For instance, given the overall load increase, more nutrients than originally assumed would need to be mitigated to reach desired target levels, and targeted reduction quantities may need to increase to meet desired ecological states. This is best supported by the Helge case area in which the reduction in the total P load as a consequence of implementing the suggested measures is below the load increase which occurs due to the change in climate. In other words, even if all suggested measures would be implemented, the P load at the river outlet would still increase.

5 Conclusions

This paper provides a comprehensive cost-effectiveness (CEA) assessment of a range of measures selected by local stakeholders to reduce nutrient emissions in three catchments in the Baltic Sea Region, and of the impact that future changes in climate and land-use may have on the CEA estimates.

The results obtained from the three case studies highlight how local variations substantially affect the cost-effectiveness ratios of nutrient mitigation measures. Their cost-effectiveness depends on each measure's design and assumed cost, its specific location and the conditions of the surrounding area, as well as the future changes this area will be exposed to. Among the measures chosen, fertiliser reduction abates N emissions at the river outlet most cost-effectively in the Selke and the Berze catchment, whereas creating wetlands reduce the N transport in the Helge catchment with the lowest cost. In terms of P transport, contour ploughing in the Selke catchment and municipal wastewater treatment plants in the Berze catchment should be the prioritised measures. The removal of P in the Helge catchment with the means of any of the suggested measures appears to be relatively costly, with wetlands being the measure of choice given that one must be selected. Crop rotation, grassland extensification and riparian zones in forest landscapes are not practical to mitigate nutrients, keeping in mind that the effect estimates are very uncertain for the last one.

Changes such as future climate and land-use, do not only affect the cost-effectiveness of measures, but also determine the overall catchment nutrient loads. Thorough and case-specific assessments are therefore required to target nutrient mitigation in the most cost-effective manner. Hydrological modelling provides a way of assessing both the local effects of measures in their actual settings and the effect on nutrient transport at the catchment outflow. In this study, future climate and land-use changes did not affect the above ranking of the considered measures, but there was a clear yet ambiguous impact on the cost-effectiveness ratio and not necessarily negative. However, the assessments of changes expected due to different climate and land-use conditions are based on predicted scenarios of the near future (2021-2040). Looking at the far future may change the conclusions substantially. Future research is thus necessary to both refine the cost-effectiveness models and assessment frameworks and respond to emerging observations, for instance erratic or extreme changes in temperature, precipitation or land-use.

Implementing all suggested measures would not fulfil the targets to achieve good ecological status at the river outlets of the Berze and Selke catchments, but the applied approach enhances participatory planning and social learning by providing fairly precise numbers on measures considered as relevant by the most concerned stakeholders. This may bridge the gap of the perceived and actual effectiveness of certain measures. Apart from potentially decreasing the risk of selecting solutions which are not feasible in the local context, the gained insights further highlight the implications for both decision-making bodies and (economic) modelling. In addition to accounting for the variabilities affecting measures' performance, finding the most cost-effective solutions to achieve politically or ecologically determined nutrient reduction targets does therefore necessitate to adjust those targets based on the expected nutrient loads. In the face of these variabilities and future climate change, assuming that CERs and rankings of measures can merely be transferred from one region to another will likely result in wrong conclusions and less-optimal solutions, and "top-down"-selected measures may not be practicable in a different context.

The results in the current study may not be generalised and directly transferrable to other catchment areas, but the chosen methodology as a basis for decision-making can be transferred. In other words, using HYPE or another similar dynamic model to quantify impacts of stakeholder-suggested measures and subsequently assessing the suitability of these measures using a CEA could be implemented anywhere. Despite the limitation of not ending up with a single-best and least-cost set of measures to fulfil specific environmental targets, drawing on CERs of measures proves to be a useful approach to support participatory processes by stimulating discussions and enhancing social learning. Furthermore, the variability in cost-effectiveness highlights the need to carefully consider which measures perform most cost-effectively under specific conditions and locations.

6 REFERENCES

Abramenko, K., Lagzdiņš, A., & Veinbergs, A. (2013). Water quality modeling in bērze river catchment. Journal of Environmental Engineering and Landscape Management, 21(4), 316-324.

agrarheute. (2018). Marktpreise Stickstoffdünger (Market price in March 2018). https://markt.agrarheute.com/duengemittel-4/stickstoffduenger-20 (Accessed 03/2018).

Albert, C., Schröter-Schlaack, C., Hansjürgens, B., Dehnhardt, A., Döring, R., Job, H., Köppel, J., Krätzig, S., Matzdorf, B., & Reutter, M. (2017). An economic perspective on land use decisions in agricultural landscapes: Insights from the TEEB Germany Study. Ecosystem Services, 25, 69-78.

Artioli, Y., Friedrich, J., Gilbert, A. J., McQuatters-Gollop, A., Mee, L. D., Vermaat, J. E., Wulff, F., Humborg, C., Palmeri, L., & Pollehne, F. (2008). Nutrient budgets for European seas: A measure of the effectiveness of nutrient reduction policies. Marine Pollution Bulletin, 56(9), 1609-1617. doi:https://doi.org/10.1016/j.marpolbul.2008.05.027

Balana, B. B., Jackson-Blake, L., Martin-Ortega, J., & Dunn, S. (2015). Integrated cost-effectiveness analysis of agri-environmental measures for water quality. Journal of environmental management, 161, 163-172. doi:https://doi.org/10.1016/j.jenvman.2015.06.035

Balana, B. B., Vinten, A., & Slee, B. (2011). A review on cost-effectiveness analysis of agri-environmental measures related to the EU WFD: Key issues, methods, and applications. Ecological Economics, 70(6), 1021-1031.

BalticSTERN. (2013). Costs of mitigating eutrophication. Background Paper. Havs- och vattenmyndighetens rapport 2013:4.

Bartosova, A., Capell, R., Olesen, J. E., Jabloun, M., Refsgaard, J. C., Donnelly, C., Hyytiäinen, K., Pihlainen, S., Zandersen, M., Arheimer, B. (2019). Future socioeconomic conditions may have a larger impact than climate change on nutrient loads to the Baltic Sea. Ambio, 48(11), 1325-1336.

Berbel, J., Martin-Ortega, J., & Mesa, P. (2011). A cost-effectiveness analysis of water-saving measures for the water framework directive: the case of the Guadalquivir River Basin in Southern Spain. Water Resources Management, 25(2), 623-640.

BONUS MIRACLE (2016): BONUS MIRACLE project. Deliverable 2.2. Modelled baseline flooding and eutrophication based on data verified by stakeholders.

BONUS MIRACLE. (2017). BONUS MIRACLE project. Deliverable 2.3. Report and data set on scenario modelling of measures suggested by stakeholders to reduce flooding, eutrophication, enhance biodiversity and contribute to other goals, as well as climate change scenarios.

BONUS MIRACLE. (2018). Project Final Report (2015.04.01 to 2018.07.31). BONUS MIRACLE project. Mediating integrated actions for sustainable ecosystems services in a changing climate.

Cardenas, L. M., Cuttle, S. P., Crabtree, B., Hopkins, A., Shepherd, A., Scholefield, D., & del Prado, A. (2011). Cost effectiveness of nitrate leaching mitigation measures for grassland livestock systems at locations in England and Wales. Science of the total environment, 409(6), 1104-1115. doi:https://doi.org/10.1016/j.scitotenv.2010.12.006

Carolus, J. F., Hanley, N., Olsen, S. B., & Pedersen, S. M. (2018). A Bottom-up Approach to Environmental Cost-Benefit Analysis. Ecological Economics, 152, 282-295. doi: https://doi.org/10.1016/j.ecolecon.2018.06.009

Carr, A. (2002). Grass roots and green tape: principles and practices of environmental stewardship: Federation Press.

Collentine, D., Johnsson, H., Larsson, P., Markensten, H., & Persson, K. (2015). Designing cost efficient buffer zone programs: An application of the FyrisSKZ tool in a Swedish catchment. AMBIO, 44(2), 311-318.

EEA. (2018). European Waters. Assessment of status and pressures 2018. EEA Report No 7/2018. Retrieved from

Elofsson, K. (2010). Cost-effectiveness of the Baltic Sea Action Plan. Marine Policy, 34(5), 1043-1050. doi:https://doi.org/10.1016/j.marpol.2010.03.003

Elofsson, K. (2012). Swedish nutrient reduction policies: an evaluation of cost-effectiveness. Regional Environmental Change, 12(1), 225-235. doi:10.1007/s10113-011-0251-8

Environmental Agency. (2015). Delivering benefits through evidence. Cost estimation for land use and run-off - summary of evidence. Report –SC080039/R12. Flood and Coastal Erosion Risk Management Research and Development Programme.

European Commission. (2000). Directive 2000/60/EC of 23 October 2000. Establishing a framework for community action in the field of water policy. Official Journal of the European Communities, L 327.

Fleming-Lehtinen, V., Andersen, J. H., Carstensen, J., Łysiak-Pastuszak, E., Murray, C., Pyhälä, M., & Laamanen, M. (2015). Recent developments in assessment methodology reveal that the Baltic Sea eutrophication problem is expanding. Ecological Indicators, 48, 380-388.

Gachango, F. G., Pedersen, S. M., & Kjaergaard, C. (2015). Cost-Effectiveness Analysis of Surface Flow Constructed Wetlands (SFCW) for Nutrient Reduction in Drainage Discharge from Agricultural Fields in Denmark. Environmental Management, 56(6), 1478-1486. doi:10.1007/s00267-015-0585-y

GHK. (2006). A study to examine the benefits of the End of Life Vehicles Directive and the costs and benefits of a revision of the 2015 targets for recycling, re-use and recovery under the ELV Directive. Final Report to DG Environment. ANNEX 5: Environmental impacts analysed and characterisation factors.

Glibert, P. M., Icarus, A. J., Artioli, Y., Beusen, A., Bouwman, L., Harle, J., Holmes, R., & Holt, J. (2014). Vulnerability of coastal ecosystems to changes in harmful algal bloom distribution in response to climate change: projections based on model analysis. Global Change Biology, 20(12), 3845-3858. doi:doi:10.1111/gcb.12662

Graham, M., & Powell, N. (2011). Agriculture in a Changing Context - Projected Land Use Changes in the Baltic Sea Region 2020/2050. Baltic Compass report.

Gren, I.-M., Jonzon, Y., & Lindqvist, M. (2008). Costs of nutrient reductions to the Baltic Sea - technical report. Swedish University of Agricultural Sciences (SLU) Working Paper Series 2008:1.

Grunert, M. (2015). Herausforderung Stickstoff-Effizienz - Welche Optimierungspotenziale bietet die Düngung? Paper presented at the Fachtagung „Nährstoffeffizienz in der Landwirtschaft“, Landesarbeitskreis Düngung Brandenburg/Sachsen/Thüringen, Nossen (Germany), 09.07.2015.

Gulbis, K., & Ruža, A. (2012). The Utilization of Nitrogen Fertilizer for Winter Rape at LLU RSF „Vecauce” (Slāpekļa mēslojuma izmantošana ziemas rapša sējumos LLU MPS „Vecauce”).

Hashemi, F., Olesen, J. E., Hansen, A. L., Børgesen, C. D., & Dalgaard, T. (2018). Spatially differentiated strategies for reducing nitrate loads from agriculture in two Danish catchments. Journal of environmental management, 208, 77-91. doi:https://doi.org/10.1016/j.jenvman.2017.12.001

Hasler, B., Smart, J. C. R., & Fonnesbech-Wulff, A. (2012). Structure of BALTCOST Drainage Basin scale abatement cost minimisation model for nutrient reductions in Baltic Sea regions Deliverable 8.1. RECOCA. Retrieved May 8, 2018, from http://nest.su.se/recoca/deliverable_8.1.pdf.

Hasler, B., Smart, J. C. R., Fonnesbech-Wulff, A., Andersen, H. E., Thodsen, H., Blicher Mathiesen, G., Smedberg, E., Göke, C., Czajkowski, M., Was, A., Elofsson, K., Humborg, C., Wolfsberg, A., & Wulff, F. (2014). Hydro-economic modelling of cost-effective transboundary water quality management in the Baltic Sea. Water Resources and Economics, 5, 1-23. doi:https://doi.org/10.1016/j.wre.2014.05.001

Hein, L. (2007). Assessing the costs of land degradation: a case study for the Puentes catchment, southeast Spain. Land Degradation & Development, 18(6), 631-642. doi:10.1002/ldr.802

Hyytiäinen, K., Hasler, B., Ericsdotter, S., Nekoro, M., Blyg, K., Artell, J., Ahlvik, L., & Ahtiainen, H. (2013). Worth it: Benefits outweigh costs in reducing eutrophication in the Baltic, BalticSTERN Summary Report: BalticSTERN Summary Report for HELCOM 2013 Ministerial Meeting: HELCOM.

Jacob, D., Petersen, J., Eggert, B., Alias, A., Christensen, O. B., Bouwer, L. M., Braun, A., Colette, A., Déqué, M., Georgievski, G., Georgopoulou, E., Gobiet, A., Menut, L., Nikulin, G., Haensler, A., Hempelmann, N., Jones, C., Keuler, K., Kovats, S., Kröner, N., Kotlarski, S., Kriegsmann, A., Martin, E., van Meijgaard, E., Moseley, C., Pfeifer, S., Preuschmann, S., Radermacher, C., Radtke, K., Rechid, D., Rounsevell, M., Samuelsson, P., Somot, S., Soussana, J.-F., Teichmann, C., Valentini, R., Vautard, R., Weber, B., & Yiou, P. (2014). EURO-CORDEX: new high-resolution climate change projections for European impact research. Regional Environmental Change, 14(2), 563-578. doi:10.1007/s10113-013-0499-2

Jacobsen, B. H. (2007). In search of cost-effective measures: Danish report on the use of cost-effectiveness analysis when implementing the EU Water Framework Directiv. . Institute of Food and Resource Economics, Report No. 191.

Jackson-Blake, L.A., Dunn, S.M., Hershkovitz, Y., Sample, J., Helliwell, R., Balana, B.B., 2013. Title: biophysical catchment-scale modelling in the River Dee catchment, Scotland. Deliverable 5.3 of the EU FP7 REFRESH project (www.refresh.ucl.ac.uk).

Jiang, S., Jomaa, S., & Rode, M. (2014). Modelling inorganic nitrogen leaching in nested mesoscale catchments in central Germany. Ecohydrology, 7(5), 1345-1362.

Johannessen, Å. (2015). Integrating flood risk reduction, river basin and resilience management in planning: a case study of Kristianstad, Sweden: Citeseer.

Jomaa, S., Jiang, S., Thraen, D., & Rode, M. (2016). Modelling the effect of different agricultural practices on stream nitrogen load in central Germany. Energy, Sustainability and Society, 6(1), 11. doi:10.1186/s13705-016-0077-9

Kārkliņš, A., & Ruža, A. (2013). Fertiliser Recommendations for Agricultural Crops (Lauku kultūraugu mēslošanas normatīvi). Jelgava: Latvia University of Life Sciences and Technologies.

Kiedrzyńska, E., Kiedrzyński, M., Urbaniak, M., Magnuszewski, A., Skłodowski, M., Wyrwicka, A., & Zalewski, M. (2014). Point sources of nutrient pollution in the lowland river catchment in the context of the Baltic Sea eutrophication. Ecological Engineering, 70, 337-348. doi:https://doi.org/10.1016/j.ecoleng.2014.06.010

Klauer, B., Schiller, J., & Bathe, F. (2015). Concept for cost effective improvement of river morphology in the context of the European Water Framework Directive. Journal of Environmental Planning and Management, 58(11), 1944-1960. doi:10.1080/09640568.2014.969833

Konrad, M. T., Andersen, H. E., Thodsen, H., Termansen, M., & Hasler, B. (2014). Cost-efficient reductions in nutrient loads; identifying optimal spatially specific policy measures. Water Resources and Economics, 7, 39-54. doi:https://doi.org/10.1016/j.wre.2014.09.001

Kovacs, A., Honti, M., Zessner, M., Eder, A., Clement, A., & Blöschl, G. (2012). Identification of phosphorus emission hotspots in agricultural catchments. Science of the total environment, 433, 74-88. doi:https://doi.org/10.1016/j.scitotenv.2012.06.024

Kronbak, L. G., & Vestergaard, N. (2013). Environmental cost-effectiveness analysis in intertemporal natural resource policy: Evaluation of selective fishing gear. Journal of environmental management, 131, 270-279. doi:https://doi.org/10.1016/j.jenvman.2013.09.035

Latvian Environment Geology and Meteorology Centre. (2009). LIELUPES UPJU BASEINA APGABALA APSAIMNIEKOŠANAS PLANS. 2010. – 2015. GADAM. Lielupes upju baseinu apgabala apsaimniekošanas plans 2010.- 2015. gadam.

Lescot, Jean-Marie; Bordenave, Paul; Petit, Kevin; Leccia, Odile (2013): A spatially-distributed cost-effectiveness analysis framework for controlling water pollution. In Environmental Modelling & Software 41, pp. 107–122. DOI: 10.1016/j.envsoft.2012.10.008.

Lindström, G., Pers, C., Rosberg, J., Strömqvist, J., & Arheimer, B. (2010). Development and testing of the HYPE (Hydrological Predictions for the Environment) water quality model for different spatial scales. Hydrology Research, 41(3-4), 295-319. doi:10.2166/nh.2010.007

Martin-Ortega, J. (2012). Economic prescriptions and policy applications in the implementation of the European Water Framework Directive. Environmental Science & Policy, 24, 83-91. doi:https://doi.org/10.1016/j.envsci.2012.06.002

Martin-Ortega, J., Perni, A., Jackson-Blake, L., Balana, B. B., McKee, A., Dunn, S., Helliwell, R., Psaltopoulos, D., Skuras, D., Cooksley, S., & Slee, B. (2015). A transdisciplinary approach to the economic analysis of the European Water Framework Directive. Ecological Economics, 116, 34-45. doi:https://doi.org/10.1016/j.ecolecon.2015.03.026

Mauchline, A. L., Mortimer, S. R., Park, J. R., Finn, J. A., Haysom, K., Westbury, D. B., Purvis, G., Louwagie, G., Northey, G., Primdahl, J., Vejre, H., Kristensen, L. S., Teilmann, K. V., Vesterager, J. P., Knickel, K., Kasperczyk, N., Balázs, K., Podmaniczky, L., Vlahos, G., Christopoulos, S., Kröger, L., Aakkula, J., & Yli-Viikari, A. (2012). Environmental evaluation of agri-environment schemes using participatory approaches: Experiences of testing the Agri-Environmental Footprint Index. Land Use Policy, 29(2), 317-328. doi:https://doi.org/10.1016/j.landusepol.2011.07.002

Mewes, M. (2012). Diffuse nutrient reduction in the German Baltic Sea catchment: Cost-effectiveness analysis of water protection measures. Ecological Indicators, 22, 16-26. doi:https://doi.org/10.1016/j.ecolind.2012.01.006

Micha, E., Roberts, W., Ryan, M., O'Donoghue, C., & Daly, K. (2018). A participatory approach for comparing stakeholders' evaluation of P loss mitigation options in a high ecological status river catchment. Environmental Science & Policy, 84, 41-51. doi:https://doi.org/10.1016/j.envsci.2018.02.014

Ministry of Finance Latvia. (2016). Makroekonomisko pieņēmumu un prognožu skaitliskās vērtības. Retrieved from http://www.fm.gov.lv/files/publiskaprivatapartneriba/2016-09-12_12_06_37_160711_info_ES%20FEA.pdf

Nainggolan, D., Hasler, B., Andersen, H. E., Gyldenkærne, S., & Termansen, M. (2018). Water Quality Management and Climate Change Mitigation: Cost-effectiveness of Joint Implementation in the Baltic Sea Region. Ecological Economics, 144, 12-26. doi:https://doi.org/10.1016/j.ecolecon.2017.07.026

Neset , T.-S., Wilk, J., Navarra, C., Capell, R., & Bartosova, A. (2019). Visualization-supported dialogues in the Baltic Sea Region. *Ambio, 48*, 1314-1324. doi:https://doi.org/10.1007/s13280-019-01250-6

Nicolle, A. (2013). Bakgrundsdokument klassning näring Helge å punkt nationell flodmynning. Länsstyrelsen i Skåne län. Retrieved from https://viss.lansstyrelsen.se/ReferenceLibrary/51986/SE620276-140087.pdf (Accessed 01/02/2019)

Nowicki, P., Goba, V., Knierim, A., van Meijl, H., Banse, M., Delbaere, B., Helming, J., Hunke, P., Jansson, K., & Jansson, T. (2009). Update of Analysis of Prospects in the Scenar 2020 Study – Contract No. 30–CE-0200286/00-21. . European Commission, Directorate-General Agriculture and Rural Development, Brussels.

NWRM. (2014a). Natural Water Retention Measures (NWRM). DGENV Pilot Project - Atmospheric Precipitation - Protection and efficient use of Fresh Water: Integration of Natural Water Retention Measures (NWRM) in River basin management. Individual NWRM - Crop rotation.

NWRM. (2014b). Natural Water Retention Measures (NWRM). DGENV Pilot Project - Atmospheric Precipitation - Protection and efficient use of Fresh Water: Integration of Natural Water Retention Measures (NWRM) in River basin management. Individual NWRM - Meadows and pastures. Retrieved from http://nwrm.eu/sites/default/files/nwrm_ressources/a1_-_meadows_and_pastures_0.pdf

OECD. (2018). Cost-Benefit Analysis and the Environment: Further Developments and Policy Use, OECD Publishing, Paris. doi:doi.org/10.1787/9789264085169

Oinonen, S., Hyytiäinen, K., Ahlvik, L., Laamanen, M., Lehtoranta, V., Salojärvi, J., & Virtanen, J. (2016). Cost-Effective Marine Protection - A Pragmatic Approach. PLOS ONE, 11(1), e0147085. doi:10.1371/journal.pone.0147085

Olesen, J. E., & Bindi, M. (2002). Consequences of climate change for European agricultural productivity, land use and policy. European Journal of Agronomy, 16(4), 239-262. doi:https://doi.org/10.1016/S1161-0301(02)00004-7

OpenStreetMap contributors. (2018). OpenStreetMap (https://www.openstreetmap.org/copyright/en)

Pearce, D., Atkinson, G., & Mourato, S. (2006). Cost-benefit analysis and the environment: recent developments: Organisation for Economic Co-operation and development.

Perni, A., & Martínez-Paz, J. M. (2013). A participatory approach for selecting cost-effective measures in the WFD context: The Mar Menor (SE Spain). Science of the total environment, 458-460, 303-311. doi:https://doi.org/10.1016/j.scitotenv.2013.04.029

Posthumus, H., Deeks, L., Rickson, R., & Quinton, J. (2015). Costs and benefits of erosion control measures in the UK. Soil use and management, 31, 16-33.

Powell, N., Do, T., Osbeck, M., Tonderski, K., Olsson, O., Schwarz, G., Tonderski, A., Zilans, A., & Veidemane, K. (2018). BONUS MIRACLE project. Deliverable 5.3. Report on Reconciling Multiple Demands.

Prammer, H. K. (2009). Integriertes Umweltkostenmanagement: Bezugsrahmen und Konzeption für eine ökologisch-nachhaltige Unternehmensführung (Vol. 369): Springer-Verlag.

Reusch, T. B. H., Dierking, J., Andersson, H. C., Bonsdorff, E., Carstensen, J., Casini, M., Czajkowski, M., Hasler, B., Hinsby, K., Hyytiäinen, K., Johannesson, K., Jomaa, S., Jormalainen, V., Kuosa, H., Kurland, S., Laikre, L., MacKenzie, B. R., Margonski, P., Melzner, F., Oesterwind, D., Ojaveer, H., Refsgaard, J. C., Sandström, A., Schwarz, G., Tonderski, K., Winder, M., & Zandersen, M. (2018). The Baltic Sea as a time machine for the future coastal ocean. Science Advances, 4(5). doi:10.1126/sciadv.aar8195

Riahi, K., Rao, S., Krey, V., Cho, C., Chirkov, V., Fischer, G., Kindermann, G., Nakicenovic, N., & Rafaj, P. (2011). RCP 8.5—A scenario of comparatively high greenhouse gas emissions. Climatic Change, 109(1), 33. doi:10.1007/s10584-011-0149-y

Richnau, G., Angelstam, P., Valasiuk, S., Zahvoyska, L., Axelsson, R., Elbakidze, M., Farley, J., Jönsson, I., & Soloviy, I. (2013). Multifaceted value profiles of forest owner categories in South Sweden: The River Helge å catchment as a case study. AMBIO, 42, 188-200.

Richter, R. (2016). Landesanstalt für Landwirtschaft und Gartenbau. Prozesskosten im Ackerbau in Sachsen-Anhalt. Ausgabe 2016 (Durchschnittswerte ausgewählter Produktionsverfahren).

Ruža, A. (2014). “Minerālmēslu maksimālo normu noteikšana kultūraugiem”. ZM subsīdiju projekts Nr. 140814/S157.

Saux Picart, S., Allen, J. I., Butenschön, M., Artioli, Y., de Mora, L., Wakelin, S., & Holt, J. (2015). What can ecosystem models tell us about the risk of eutrophication in the North Sea? Climatic Change, 132(1), 111-125. doi:10.1007/s10584-014-1071-x

Schmidt, T. G., Röder, N., Dauber, J., Klimek, S., Laggner, A., Witte, T. d., Offermann, F., & Osterburg, B. (2014). Biodiversitätsrelevante Regelungen zur nationalen Umsetzung des Greenings der Gemeinsamen Agrarpolitik der EU nach 2013. Thünen Working Paper 20. Braunschweig/Germany, March 2014. Retrieved from

SMHI. (2017). HYPE model description. Swedish Meteorological and Hydrological Institute. Retrieved from http://www.smhi.net/hype/wiki/doku.php?id=start:hype_model_description (Accessed 12/02/2019)

SMHI. (2018). HYPE variables. Swedish Meteorological and Hydrological Institute. Retrieved from http://www.smhi.net/hype/wiki/doku.php?id=start:hype_file_reference:info.txt:variables (Accessed 12/02/2019)

Smith, J. L. (2008). A critical appreciation of the “bottom-up” approach to sustainable water management: embracing complexity rather than desirability. Local environment, 13(4), 353-366.

Stevens, C. J., Quinton, J. N., Bailey, A. P., Deasy, C., Silgram, M., & Jackson, D. R. (2009). The effects of minimal tillage, contour cultivation and in-field vegetative barriers on soil erosion and phosphorus loss. Soil and Tillage Research, 106(1), 145-151. doi:https://doi.org/10.1016/j.still.2009.04.009

Stupak, N., Sanders, J., & Heinrich, B. (2019). The Role of Farmers' Understanding of Nature in Shaping their Uptake of Nature Protection Measures. Ecological Economics, 157, 301-311. doi:https://doi.org/10.1016/j.ecolecon.2018.11.022

Swedish Transport Administration. (2016). Analysmetod och samhällsekonomiska kalkylvärden för transportsektorn: ASEK 6.0. Chapter 20, English summary of ASEK Guidelines. Version 2016-04-01. Trafikverket.

Tonderski, K. (2018). The Final Publishable Summary Report (2015.04.01 to 2018.07.31). BONUS MIRACLE project.

Tucker, G. (2011). Final report. Green Infrastructure Implementation and Efficiency. Annex III. Costs of Green infrastructure. Annexes to the Final report for the European Commission, DG Environment on Contract ENV.B.2/SER/2010/0059. Institute for European Environmental Policy, Brussels and London. . Retrieved from

UBA. (2017). Gewässer in Deutschland: Zustand und Bewertung. German Environment Agency (Umweltbundesamt – UBA): Dessau-Roßlau.

Vinten, A., Sample, J., Ibiyemi, A., Abdul-Salam, Y., & Stutter, M. (2017). A tool for cost-effectiveness analysis of field scale sediment-bound phosphorus mitigation measures and application to analysis of spatial and temporal targeting in the Lunan Water catchment, Scotland. Science of the total environment, 586, 631-641. doi:https://doi.org/10.1016/j.scitotenv.2017.02.034

VISS. (2015). Vatteninformationsystem Sverige (VISS). Anpassade skyddszoner på åkermark. ID: VISSMEASURETYPE000794. Retrieved from https://viss.lansstyrelsen.se/Measures/EditMeasureType.aspx?measureTypeEUID=VISSMEASURETYPE000794 (Accessed 10.01.2019)

VISS. (2016a). Vatteninformationsystem Sverige (VISS). Återställning av rätat vattendrag. ID: VISSMEASURETYPE000823. Retrieved from https://viss.lansstyrelsen.se/Measures/EditMeasureType.aspx?measureTypeEUID=VISSMEASURETYPE000823 (Accessed 10.01.2019)

VISS. (2016b). Vatteninformationsystem Sverige (VISS). Dagvattendamm. ID: VISSMEASURETYPE000785. Retrieved from https://viss.lansstyrelsen.se/Measures/EditMeasureType.aspx?measureTypeEUID=VISSMEASURETYPE000785 (Accessed 10.01.2019)

VISS. (2016c). Vatteninformationsystem Sverige (VISS). Ekologiskt funktionella kantzoner. ID: VISSMEASURETYPE000890. Retrieved from https://viss.lansstyrelsen.se/Measures/EditMeasureType.aspx?measureTypeEUID=VISSMEASURETYPE000890 (Accessed 10.01.2019)

VISS. (2017). Vatteninformationsystem Sverige (VISS). Helge å - Huvudavrinningsområde. Nyhet i VISS - information utifrån förvaltningscykel. Retrieved from https://viss.lansstyrelsen.se/areastatisticsform.aspx?area=2,210&managementCycleName=Cykel_2 (Accessed 06.02.2019)

VISS. (2018). Vatteninformationsystem Sverige (VISS). Våtmark för näringsretention. ID: VISSMEASURETYPE000725. Retrieved from https://viss.lansstyrelsen.se/Measures/EditMeasureType.aspx?measureTypeEUID=VISSMEASURETYPE000725 (Accessed 10.01.2019)

Voulvoulis, N., Arpon, K. D., & Giakoumis, T. (2017). The EU Water Framework Directive: From great expectations to problems with implementation. Science of the total environment, 575, 358-366. doi:https://doi.org/10.1016/j.scitotenv.2016.09.228

Wood, A., Blackhurst, M., Hawkins, T., Xue, X., Ashbolt, N., & Garland, J. (2015). Cost-effectiveness of nitrogen mitigation by alternative household wastewater management technologies. Journal of environmental management, 150, 344-354. doi:https://doi.org/10.1016/j.jenvman.2014.10.002

Wulff, F., Humborg, C., Andersen, H. E., Blicher-Mathiesen, G., Czajkowski, M., Elofsson, K., Fonnesbech-Wulff, A., Hasler, B., Hong, B., Jansons, V., Mörth, C.-M., Smart, J. C. R., Smedberg, E., Stålnacke, P., Swaney, D. P., Thodsen, H., Was, A., & Żylicz, T. (2014). Reduction of Baltic Sea Nutrient Inputs and Allocation of Abatement Costs Within the Baltic Sea Catchment. AMBIO, 43(1), 11-25. doi:10.1007/s13280-013-0484-5

Yang, W., Andréasson, J., Phil Graham, L., Olsson, J., Rosberg, J., & Wetterhall, F. (2010). Distribution-based scaling to improve usability of regional climate model projections for hydrological climate change impacts studies. Hydrology Research, 41(3-4), 211-229. doi:10.2166/nh.2010.004

Zanou, B., Kontogianni, A., & Skourtos, M. (2003). A classification approach of cost effective management measures for the improvement of watershed quality. Ocean & Coastal Management, 46(11), 957-983. doi:https://doi.org/10.1016/j.ocecoaman.2004.01.003

SUPPLEMENTARY MATERIAL

A: Participating stakeholder groups and agencies

Stakeholder Group	Agencies (Berze)	Agencies (Helge)	Agencies (Selke
Public Sector	▪ Ministry of Environmental Protection and Regional Development ▪ State Environmental Services ▪ Ministry of Agriculture ▪ Rural Support Service ▪ Latvian Environment, Geology and Meteorology Centre ▪ Health Inspectorate ▪ Municipalities ▪ County Councils, Jelgava and Dobele	▪ Water Association, Helge River ▪ District Council, Kristianstad ▪ Swedish Forest Agency ▪ Initiative Model Forest in Helge å ▪ Environment Committee, District of Hässleholm ▪ Nature Management School in Osby ▪ Municipal Office of Osby	▪ State Agency for flood protection and water management, Saxony-Anhalt (LHW) ▪ Ministry of Agriculture and Environment, Saxony-Anhalt ▪ Nature Protection Authority, Landkreis (County) Harz ▪ Nature Protection Authority, Salzlandkreis ▪ Lower Water Authority, Landkreis (County) Harz ▪ Lower Water Authority, Salzlandkreis ▪ Office of Agriculture, Land Consolidation and Forestry (ALFF Mitte) ▪ State Institute for Agriculture, Forestry and Horticulture, Saxony-Anhalt
Civil Society	▪ Latvian Fund for Nature ▪ Baltic Environmental Forum	▪ Swedish Society for Nature Conservation (SSNC), Kristianstad ▪ Swedish Society for Nature Conservation (SSNC), Kronoberg	
Private Sector	▪ Farmers Parliament and Farmers ▪ Hydropower plant operators ▪ Wastewater treatment plant operators ▪ Property Management Group	▪ Farmers ▪ Forest-Owner association ▪ Regito Research Center on Water and Health ▪ Hydropower plant operators	▪ Farmers Union Nordharz e.V. ▪ Farmers Union, Saxony-Anhalt e.V.

B: Cost assumptions and sources of the considered nutrient mitigation measures:

Case area	Measure	Lifetime (years)	Sources and explanation
Berze (LV)	Crop rotation	1	Investment cost: Tucker (2011) via NWRM (2014a) Recurring cost: based on a letter from Indulis Abolins, Assistant Director, Rural Support Service of Latvia (dated 18/03/2016), and validated by key stakeholders in context of the MIRACLE project. Cost range: assumed standard cost range of 20-200 %.
	Grassland extensification [(a)]	10	NWRM (2014b) Cost range: assumed standard cost range of 20-200 %.
	Organic farming	1	Recurring costs: EU support payment proxy (Support per hectare of agricultural area to convert to or maintain organic farming practices and methods, including relevant minimum requirements for fertiliser and plant protection products use as well as other relevant mandatory requirements, established by Latvian national law; based on a letter from *Indulis Abolins*, Assistant Director, Rural Support Service of Latvia (dated 18/03/2016), and validated by key stakeholders in context of the MIRACLE project); Cost range: assumed standard cost range of 20-200 %.
	Buffer strips (2+5m) [(b)]	5	Investment cost: Environmental Agency (2015) Recurring/opportunity cost: Survey amongst stakeholders (farming sector) in context of the MIRACLE project (11/2016); Cost range: VISS (2015)
	Buffer strips (2+10m) [(b)]	5	
	N and P fertiliser reduction (5%)	1	Own calculation based on crop cultivations, fertiliser application and yield declines in the Berze river catchment in 2017. The decline in yield is based on Kārkliņš & Ruža (2013), Ruža (2014) and Gulbis & Ruža (2012), the 2017 crop prices of 143 €/t (wheat and barley) and 350 €/t (rape) and fertiliser prices of 236 €/t (NH4NO3) and 245 €/t (P2O2) are based on stakeholder information. The cost range is the one considered for the fertiliser reduction in the Selke catchment (see below).
	N and P fertiliser reduction (20%)	1	
	Municipal Wastewater Treatment Plants (100 – 300 PE)	30	Based on the average costs of three comparable WWTPs (in Strauti, Apgulde and Krimuna settlements, Latvia), provided by stakeholder from the water and wastewater company SIA "DOBELES ŪDENS" (03/02/2017) in context of the MIRACLE project (with the cost range being based on the example with the highest and lowest cost, respectively).
Helge (SE)	Re-meandering	30	VISS (2016a)
	Riparian zones in forest landscapes (30m)	30	Collentine et al. (2015); VISS (2016c)
	Stormwater ponds	25	VISS (2016b)
	Wetlands	30	VISS (2018)
Selke (DE)	Buffer strips (10m)	5	Investment cost: Environmental Agency (2015) Recurring cost: Own calculation of the average gross margins based on the cost indications in Richter (2016) which were applied to the land-use in the Selke catchment, based on the data of the 2010 Agricultural Census of the statistical office of Lower-Saxony, Germany ("Reihe CIV 3j/10"); Cost range: VISS (2015)
	Buffer strips (20m)	5	
	Contour ploughing	1	No additional costs are expected, provided by stakeholders (agricultural sector) in context of the MIRACLE stakeholder workshop in Magdeburg, Germany (11/2016).
	N fertiliser reduction (20%)	1	Recurring cost: own calculation of the average gross margins based on the cost indications in Richter (2016) which were applied to the land-use in the Selke catchment, based on the data of the 2010 Agricultural Census of the statistical office of Lower-Saxony, Germany ("Reihe CIV 3j/10"); the reduced cost are based on N fertiliser market price (agrarheute, 2018) and forgone income based on yield decline in Grunert (2015) Cost range: The cost range includes the fertiliser price range (agrarheute, 2018) yet assumes a constant yield decline (due to missing information)

C: The required yearly removal of N and P to reach good ecological status (GES) at the river outlets

Acronym	**Description**	**River outlet**			**Source/Calculation**
		Selke	**Helge**	**Berze**	
Q	Water flow at river outlet (m^3/s)	3.11	42.2	8.78	HYPE model results
tc_N	Target N concentration (mg/l) to reach GES	2.50	not set	2.50	Selke: UBA (2017); Berze: Latvian Environment Geology and Meteorology Centre (2009); Helge: Nicolle (2013) via VISS
tc_P	Target P concentration (mg/l) to reach GES	0.10	0.045	0.075	
max_N	Maximum amount of total N to reach GES at the river outlet (kg N/year)	245,000	not set	692,000	*Q*tc_N* (converted to kg/year)
max_P	Maximum amount of total P to reach GES at the river outlet (kg P/year)	9,790	60,000	20,800	*Q*tc_P* (converted to kg/year)
base_N	Baseline load at river outlet (kg N/year)	801,600	2,377,000	828,600	HYPE model results
base_P	Baseline load at river outlet (kg P/year)	26,300	56,300	12,700	HYPE model results
target_N	**Required reduction of total N at the river outlet to reach GES (kg N/year)**	**557,000**	**0**	**137,000**	*base_N – max_N (0 if negative)*
target_P	**Required reduction of total P at the river outlet to reach GES (kg P/year)**	**16,500**	**0**	**0**	*base_P – max_P (0 if negative)*