EUROPEAN UNION
European Regional Development Fund

Interreg
Central Baltic

SEABASED
SEABASED MEASURES IN BALTIC SEA NUTRIENT MANAGEMENT

Nutrient compensation for aquatic coastal environment

— legal, ecological and economic aspects in developing an offsetting concept

Kirsi Kostamo, Sara Kymenvaara, Minna Pekkonen, Antti Belinskij

Photo: Mats Westerbom, Metsähallitus

SYKE

Ålands landskapsregering

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Authors: Kirsi Kostamo, Sara Kymenvaara, Minna Pekkonen, Antti Belinskij

Proofreader: Sonja Virta

Photos: Annica Brink, Visa Hietalahti, Essi Keskinen, Petra Pohjola, Mats Westerbom

2020

Financier/commissioner:

Abstract

Human activities have significant impacts on the coastal areas of the Baltic Sea. Land-based approaches to reduce, e.g. the nutrient loads entering the Baltic Sea and realization of the current marine protection area network have not been enough to reduce the amount of nutrients in the seawater and to stop the loss of marine biodiversity. Nevertheless, the European Union's (EU) legislation, including the Water Framework Directive (2000/60/EC) and Marine Strategy Framework Directive (2008/56/EC), requires member states to improve the state of the European waters and marine areas. These legal requirements and the deterioration of the natural environment have resulted in the development of new concepts to simultaneously allow the development of economic activities and environmental protection. Nutrient compensation, where human-induced deterioration of ecosystems due to increased anthropogenic nutrient load is offset by removing excess nutrients, has recently been presented as one possibility to overcome the harmful effects human activities may cause to water ecosystems.

At the request of the Government of Åland, for the Central Baltic project SEABASED, this report describes legal, ecological and socio-economic aspects that need to be considered when developing a scheme for nutrient offsetting in Åland. The aim of the work is to utilize the concept, described in the report, as a "practical tool" in the implementation of compensation possibilities and methods in legislation. For the Government of Åland, the report will provide input for the revision of the Water Act in Åland and for the decrees to follow. In addition, the concept could be further utilized as a tool for regional actors, e.g. environmental authorities, in assessing and choosing between different measures when planning regional water protection and necessary cost-efficient water protection measures. The basic idea of compensation, key concepts and potential risks are presented. A thorough overview is given on the legal framework covering the current situation in Åland, Finland and Sweden, and an example from the USA. From an ecological viewpoint, there are several potential measures for producing nutrient offsets both in the coastal and watershed area. Examples of these, including measures piloted within the SEABASED project, are considered for their offsetting potential and ecological impacts. Also, when relevant, risks and economic viewpoints are brought up. In general, ecological uncertainties in offsetting arise from how large an impact each measure has on the water ecosystem and where the impact is effective. Economic and societal aspects of what needs to be considered in planning the compensation scheme are briefly described. Examples and ideas are given on how compensation pools or biobanks have been organized elsewhere.

A balanced offsetting system could potentially provide a possibility for the sustainable development of economic sectors, such as marine aquaculture. However, the precautionary principle related to the ecological and socio-economic impacts of the measures must be applied when developing any compensation system. From a legal point of view, the EU law is a good starting point for developing a nutrient compensation scheme in Åland. Possible new regulatory framework should be kept as simple as possible. The whole offsetting process can be managed by the public or private sector, but the best outcome may be reached by co-operation. Governance, rules and guidelines are needed, but the implementation may be carried out by the private sector.

Keywords:

aquatic, archipelago, coast, compensation, eutrophication, fisheries, marine, nutrients, offsetting, phosphorus

Contents

1 Introduction

Human activities have significant impacts on the coastal areas of the Baltic Sea. Land-based approaches to reduce, e.g. the nutrient loads entering the Baltic Sea and the realization of the current marine protection area network have not been enough to stop the loss of marine biodiversity. However, the European Union's (EU) legislation, including the Water Framework Directive (2000/60/EC), Marine Strategy Framework Directive (2008/56/EC), Habitats Directive (92/43/EEC) and Birds Directive (2009/147/EC), require member states to improve the state of the European waters and marine areas. The Helsinki Commission's Baltic Sea Action Plan (HELCOM BSAP) is currently under revision, but it is expected that it will also require further measures from all member states to improve the state of the Baltic Sea.

These legal requirements and the deterioration of the natural environment have resulted in the development of new concepts and practices in order to simultaneously allow the development of economic activities and secure nature values and promote environmental protection. One such concept is the compensation of different characteristics of the environment. In coastal and marine areas, compensations can focus either on ecological characteristics or nutrients. Ecological compensation, where human-induced ecological loss at one location is offset by producing ecological gains elsewhere, has recently been presented as one possibility to improve the state of the environment and stop or at least slow down the deterioration of ecosystems (BBOP 2012a, b; IUCN 2016). Another approach is to improve the state of the marine environment by compensating the increase of nutrients resulting from human activities by measures aimed at removing nutrients from the ecosystem.

This report concentrates on if and how compensations could be useful in improving the environmental conditions in coastal areas of the Baltic Sea, especially in relation to the quality of water bodies. Ecological compensation, often also called biodiversity offsetting, focuses on biodiversity and ecological components of the environment, such as the loss and gain of habitats, species or ecosystem characteristics (BBOP 2012a, b; IUCN 2016). In water bodies the critical environmental component impacting the marine ecosystem is often nutrients. The excess of nutrients causes eutrophication of water ecosystems and can have adverse ecological effects. Compensation schemes can potentially be used to reduce the nutrient loads. Nutrient compensations are not, however, the same as ecological compensations. Differences and similarities between nutrient and ecological compensations are discussed further in Section 2.1.2.

If the concept of nutrient offsetting is applied to nutrient reduction in waterbodies, the actors involved in developing the compensation approach would typically be the operator of economic activity, the permitting authority and the water basin management authority as well as third parties. The main goals of this report are to collate knowledge of previously utilized offsetting models and how they could be utilized in the Northern Baltic Sea considering legislation and local environmental and socio-economic conditions. However, since the concept of nutrient offsetting is still quite new and only a few examples of realized offsets can be discovered, offsetting measure descriptions and assessing their suitability on the Northern Baltic Sea is done mostly based on literature and expert judgement. Also, the socio-economic part of developing nutrient offsetting has proven to be complicated, since the development of economically sound and socially accepted offsetting system requires the participation of decision makers, stakeholders and experts. This report contributes to forming a basis for assessing the possibility for developing a nutrient offsetting system. As an overall conclusion it can be stated that careful legislative, ecological and socio-economic considerations must be made before the realization of an operational compensation system.

2 Compensation concepts and challenges

2.1 Basic concepts related to offsetting

2.1.1 The mitigation hierarchy

Environmental damages should be addressed according to a sequential order established by the so-called mitigation hierarchy (Figure 1). Damages should firstly be avoided, thereafter minimized and remedied and lastly compensated. The mitigation hierarchy is derived from the framework of the international co-operation 'Business and Biodiversity Offsets Programme' (BBOP) as a standard for voluntary compensations (BBOP 2012a). Applying the mitigation hierarchy in compensation schemes is recommended also by the International Union for Conservation of Nature (IUCN 2016).

Both the BBOP framework and the IUCN recommendations emphasize the central role of the mitigation hierarchy when applying and working with the concept of ecological compensation and imply that all reasonable measures to avoid and minimize the impact of the exploitation should be exhausted before the need of compensation is established. The different steps of the mitigation hierarchy with practical examples are explained in more detail, for example, by Arlidge et al. (2018).

1) Avoiding damages concerns the activity's choice of location, its areal determination and the forms of its realization. Conceptually this does not imply that a project should be avoided altogether.

2) Minimization aims at reducing the damages before or when they occur. Measures to minimize damage are directly related to the location of the activity as concerns its environmental impacts and how the activity is carried out. Minimization measures are, for example, adherence to the best available technology (BAT) and best environmental practices (BEP).

3) Remediation is also carried out at the same location as the activity, usually after the activity has ended. Remediation measures can include habitat restoration or other active measures that produce ecological improvements at the activity site.

4) The compensation need should be assessed after all previous mitigation steps are planned. Ecological compensation should be used to offset the remaining environmental harm.

Adherence to the mitigation hierarchy is particularly emphasized in situations where important natural values are destroyed or ecosystem services of significant social or economic value are at risk. In practice, following the mitigation hierarchy means that all efforts are made to minimize the adverse environmental effects before compensation takes place. Depending on the case, compensating the environmental harm can be more costly than avoidance and minimization, and, more importantly, there is always a risk of failure in producing adequate compensations (Moilanen & Kotiaho 2018). Thus, it is recommended that ecological compensation should be used as the measure of last resort in minimizing biodiversity loss (Raunio et al. 2019). A similar approach should be utilized when developing nutrient compensation schemes.

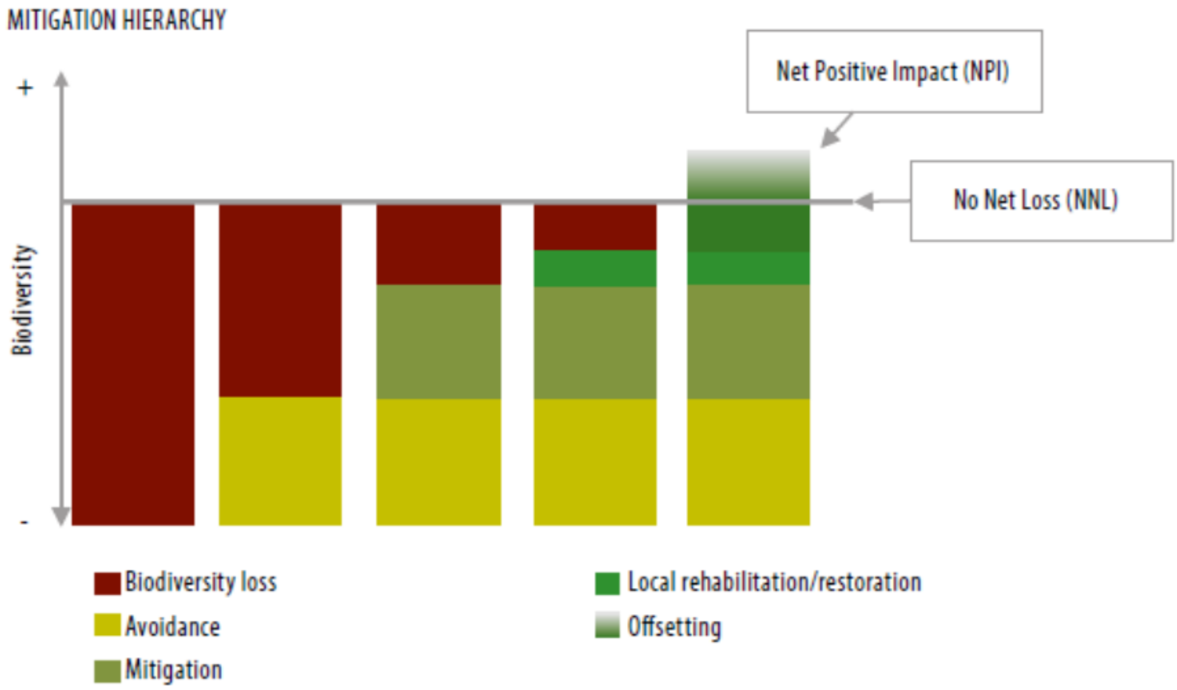

Figure 1. According to the mitigation hierarchy, the human-induced environmental degradation should always, when possible, be avoided and thereafter the inevitable impacts should be minimized. Any residual degradation (e.g. biodiversity loss or excess nutrients) is then offset outside the degraded area. The relative effect of avoidance, mitigation, rehabilitation/restoration and offsetting varies on a case-by-case basis. Due to the many uncertainties in offsetting, the achievement of no net loss (NNL) is challenging. (Figure from Raunio et al. 2019, © Kostamo et al. 2018, Finnish Environment Institute, originally adapted from BBOP 2012a.)

2.1.2 Compensation definitions and No Net Loss

The concepts ecological compensation, biodiversity offsetting and environmental compensation overlap to some extent. In compensation literature in English, the term biodiversity offsetting is widely used (e.g. BBOP 2012a, IUCN 2016, OECD 2016). The terms ecological or environmental compensation have been more common in Finnish and Swedish public discussion and recent reports (e.g. Enetjärn et al. 2015, Kostamo et al. 2018, Naturvårdsverket 2015, 2016). In Finland and Sweden ecological compensation (ekologinen kompensaatio, ekologisk kompensation) is used as a synonym to biodiversity offsetting. In some cases, ecological compensation is defined so that it covers not only biodiversity but also ecosystem services, such as recreational values (SOU 2017, see Section 4.1.1).

According to the BBOP definition, "ecological compensation" is a wider concept than the narrower 'biodiversity offsetting'. Biodiversity offsets are "*measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development after appropriate prevention and mitigation measures have been taken. The goal of biodiversity offsets is to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure, ecosystem function and people's use and cultural values associated with biodiversity*" (BBOP 2012a, p. 13).

Compensations can lead to "No Net Loss" or "Net Gain", which implies a full compensation. However, it can also lead to partial compensation in which case it does not achieve No Net Loss. Following the BBOP definition the main difference between the concepts of ecological compensation and biodiversity offsets is that biodiversity offsets always aim for no net loss of biodiversity whereas other ecological compensations may have lesser goals (partial compensation).

Environmental compensation is a wider concept than ecological compensation or biodiversity offsetting. Environmental compensation is defined in Enetjärn et al. (2015) to also include ecosystem services, that is the benefits people get from nature, in contrast to ecological compensation where the focus is restricted to biodiversity loss and gain and ecological characteristics (habitats, species) of the environment. Care must be taken when using these terms, as there are differences between these concepts (Figure 2).

As the focus of this report is the usability of compensation schemes in reducing human-induced nutrient loads from waterbodies, we also use the term nutrient offsetting. Reducing the nutrient load may also benefit the overall ecological or environmental status of eutrophicated water areas. Nutrient offsetting is not, however, the same thing as biodiversity offsetting or ecological compensation.

A nutrient offset can be defined as a unit of additional nutrient reduction. Nutrient offsets can be generated either through removing nutrients from a water body, its catchment area or another water body affecting it. The effects of nutrient offsets should be measurable and the outcomes verified. Furthermore, nutrient offsets should not cause new loads elsewhere. Nutrient offsetting may include the elements of minimization and remediation when nutrient abatement measures are taken at different sources so that the net effect of an activity to a water body would be neutral or decreasing (Belinskij et al. 2018a).

In practice, the basic concepts, challenges and possible implementation ways are similar in nutrient and biodiversity offsetting, only the focus of offsetting is different. We will utilize the principles of biodiversity offsetting where feasible and, when necessary, highlight the specific differences or special needs related to nutrient offsetting.

Additionality

One of the main principles in biodiversity offsetting / ecological compensation is that the compensation measures need to be additional. Additionality is explained in the BBOP (2012b) glossary for offsetting as follows: "*A property of a biodiversity offset, where the conservation outcomes it delivers are demonstrably new and additional and would not have resulted without the offset*".

In practice, additionality may be difficult to show. In nutrient offsetting a good example of defining what is additional is the use of the best available technique (BAT). If the use of BAT is already required for some other reason than offsetting, it is not additional. Also, according to the mitigation hierarchy, using BAT is actually a part of the basic mitigation measures and for that reason should not be considered as a way to offset residual environmental impact. Furthermore, measures included in, e.g. EU's Rural Development Programme to reduce nutrient input from agriculture can only be considered as nutrient offsets if they produce additional nutrient removal, and all targets set forth by the policy are already fulfilled. This means that developing land-based nutrient offsetting measures can be challenging.

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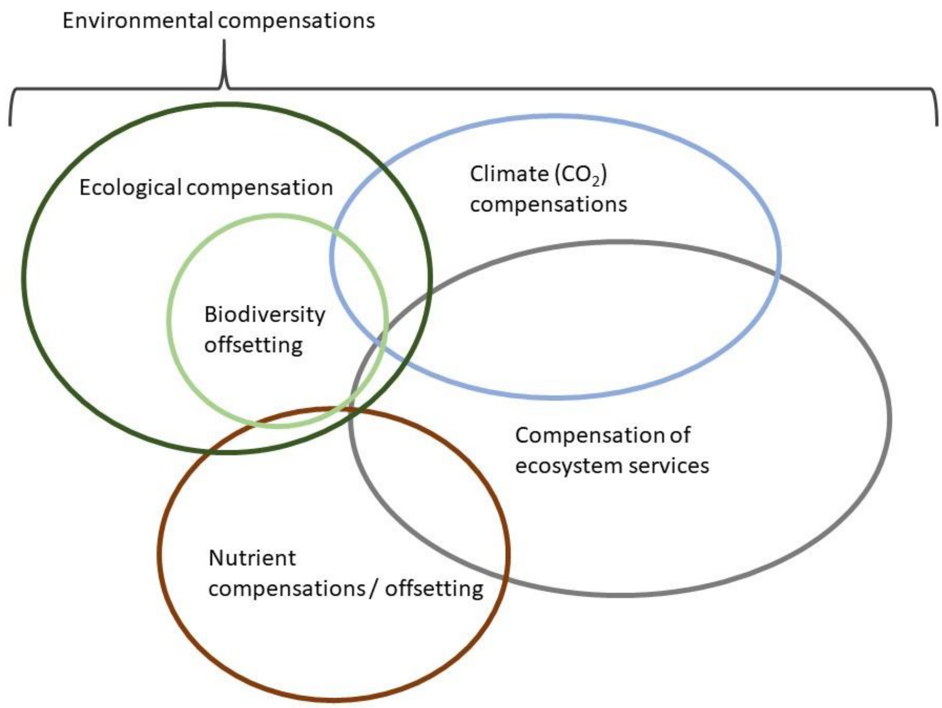

Figure 2. The focus of environmental compensation can be almost any aspect of the environment. Different compensation types overlap and interact with each other. Sometimes the interactions are mutually beneficial – for example, nutrient reduction can make the rehabilitation of an aquatic ecosystem possible. However, there are also potential conflicts of interest, like between timber production (ecosystem service) and forest biodiversity values.

2.1.3 General challenges in offsetting

Reaching the No Net Loss

The ambitious goal of biodiversity offsetting is to reach no net loss of biodiversity within the project. As biodiversity is complex and multidimensional and, in many ways, unique both locally and temporally, the no net loss is close to impossible to reach if all components of biodiversity are taken into consideration (Maron et al. 2012, Moilanen & Kotiaho 2018). Due to this intrinsic complexity of nature, only a limited number of ecological characteristics or biodiversity indicators are used in estimating loss and gain in compensation processes. Thus, in theory, no net loss can be reached for these chosen characteristics.

If the compensation measures are extensive, it is in theory also possible to reach net gain. Net gain refers to a situation where the environmental gain is larger than the human-induced environmental harm such as the loss of biodiversity. In nutrient offsetting, a net gain would mean that the amount of removed eutrophicating nutrients is larger than the human-induced environmental harm, nutrient runoff.

Reaching the no net loss is challenging. In most cases the result of offsetting is a limited loss or partial compensation. The reliability and general acceptance of compensations are at risk if the project that plans to compensate environmental harm is not clear about the objective (no net loss or limited loss) and does not openly tell about the success of the offsetting (Kostamo et al. 2018).

Measuring loss and gain and defining success

One key component in compensations is the accuracy and reliability of the loss and gain measurements. As mentioned in the previous section, biodiversity is multidimensional and, in most cases, difficult or laborious to measure (Maron et al. 2016, Moilanen & Kotiaho 2018). There are several approaches to estimating the loss and gain of biodiversity in ecological compensation. Most commonly the metrics used combine area and an estimate of environmental quality, e.g. the habitat hectares developed in Australia (Parkes et al. 2003) or different metrics developed in Europe (Wende et al. 2018).

As the focus in this report is on nutrients, the different measurement tools developed for biodiversity offsetting, and most often for terrestrial ecosystems, may not be relevant. Defining measurement units is more straightforward for nutrient offsetting in aquatic environments: the interest is in the eutrophicating nutrients, phosphorus (P) and nitrogen (N). A limited number of environmental variables makes the measurement of loss and gain easier. Still, in some cases it may be laborious to verify the impact and success of compensation measures.

Regardless of the measured units, all offsetting needs not only meticulous planning and execution of compensation measures but also adequate monitoring to verify success of the compensation. In principle, the monitoring should continue long enough to reliably show if the offsetting was successful. In case the offsetting is not adequate, there could be an obligation to take additional compensation measures. The responsibilities of monitoring and securing adequate offsetting can be arranged in various ways, and examples of these are given in Section 6.

Location, equivalence, like-for-like

Especially in biodiversity offsetting in terrestrial areas, the question of offset location draws interest. In general, there is a preference for producing the ecological gain in the proximity of the impact site, especially if there is a need to compensate not only ecological loss but also recreational values or other local ecosystem services related to the local and neighbouring impacts of an activity. From the biodiversity point of view, it would be possible to choose an offset area without considering the proximity of the impact site.

The question of where to offset is related to the choices in how flexible the compensation can be. In offset literate terms “like-for-like” or “in-kind” describe a situation where the human-induced biodiversity loss is offset by similar biodiversity (BBOP 2012b). If flexibility in trading is permitted, then the offset area should be of better or higher quality biodiversity than the lost biodiversity values. This is called “trading-up” and “like-for-better”. The like-for-like-or-better principle together with the No Net Loss goal are included in several biodiversity offsetting policies and recommendations (e.g. IUCN 2016).

The question of like-for-like may not be as fundamental in nutrient offsetting: eutrophicating nutrients are the cause of the environmental harm, and the offset should then always be “in-kind” removal of similar nutrients. The question of where to compensate is, however, relevant. If one aim of the compensations in aquatic environments is to reach a good environmental status of a water body (as meant in the WFD, see Section 4.2.1), then compensatory measures must be implemented within the same water body or monitoring area where the suggested human activity is to be located. The measures can be further away in a catchment area or another water body if it can be shown that there is an improvement effect in the area of a water body where the activity is located.

Coping with uncertainties

No matter how carefully planned and rigorously executed the compensation measures are, there is always uncertainty in the outcome. In the compensation process the loss of biodiversity or other environmental harm is known and usually certain. The success of the offsetting is uncertain for several reasons, such as the time lag in producing the target outcome, potential for technical failure in compensation measures, intrinsic complexity in ecological processes and random events that can affect the compensation outcome.

Compensation coefficients

General advice for reducing the uncertainties in producing adequate compensation gain is to use multipliers or compensation coefficients (Moilanen & Kotiaho 2018). The larger the estimated uncertainties are, the larger a multiplier should be used. In the simplest case, this would mean that if one damages a certain area of a habitat, it should be compensated by producing a much larger area of suitable offsets. There is no exact rule on how large the multiplier should be.

Mitigate and minimize the compensation need

The compensation need can be minimized by minimizing the environmental harm, in accordance with the mitigation hierarchy. The smaller the environmental harm, the smaller the offset need.

Produce compensation credits beforehand

How extensive the offset should be if all uncertainties are considered? This may vary case by case. One way to minimize the multipliers or compensation coefficients is to produce offsets successfully and credibly before the environmental harm is caused. Then the uncertainties of compensation success are smaller and thus the need for a multiplier is also smaller (Moilanen & Kotiaho 2018).

Long-term compensations

A suggested rule of thumb for biodiversity offsetting is that if the ecological harm or biodiversity loss is permanent then also the compensation should be permanent (Moilanen & Kotiaho 2019).

Back-up

Even the best efforts sometimes fail and, for an unknown or random reason, compensation may not succeed. Long-term monitoring is needed to secure the success of the compensation measures. If compensations are not voluntary but based on an obligation (e.g. related to environmental permit), there should be a back-up system to guarantee that if the original plan for compensation does not work, alternative compensations are carried out.

A recommended piece of further reading on principles related to biodiversity offsetting is ”Fifteen operationally important decisions in the planning of biodiversity offsets” by Moilanen & Kotiaho (2018, these are discussed further in Section 6). Another good review on the general obstacles and how to overcome them in biodiversity offsetting is “Taming a wicked problem: Resolving controversies in biodiversity offsetting” by Maron et al. (2016). A terminology on biodiversity offsetting with short explanations has been published by BBOP (2012b).

3 The Baltic Sea

The tideless Baltic Sea is characterized by a steep salinity gradient resulting in a variable fauna and flora, which tolerates the prevailing environmental conditions well. All of the Baltic Sea sub-basins exhibit strong gradients of wave exposure, depth and salinity. The patterns of species distribution and species richness in the Baltic Sea follow a combination of environmental gradients, with salinity appearing to be the most influential environmental factor (Zettler et al. 2013, Snoeijs-Leijonmalm et al. 2017).

The effects of eutrophication on marine ecosystems are broad. Nutrient enrichment induces enhanced pelagic primary production, leading to decreased Secchi (photic) depth and an elevated risk of low oxygen levels in the bottom water when organic matter is degraded. The depletion of oxygen in the near-bottom part of the water column can result in the release of nutrients, mostly phosphorus, into the water column, which further feeds the primary production in the photic zone. These effects have many ecosystem consequences, affecting species across photic and aphotic habitats and trophic levels (Cederwall & Elmgren 1990, Bonsdorff et al. 1997, Conley et al. 2011). In addition to the direct anthropogenic impact on the Baltic Sea ecosystem, climatic conditions have shown strong and partly unprecedented changes in recent decades (e.g. Lehmann et al. 2011) which, combined with anthropogenic pressures, have been associated with an ecosystem-wide regime shift in the higher trophic levels in the Baltic Proper (e.g. Möllmann et al. 2009 and references therein).

In the Northern Baltic Sea, there exists several macrophytes and invertebrates considered as habitat-forming species that are a precondition or promote the existence of other species that otherwise would not be present in the area (Martin et al. 2013). Benthic algae and aquatic plants serve as a spawning ground for economically important fish species like the Baltic herring (Rajasilta et al. 2006) and support a high biomass of invertebrates (Wikström & Kautsky 2007). They can also be extensively consumed by waterfowl, thus forming a substantial component of the food web (Schmieder et al. 2006). The suspension-feeding bivalves form a very important trophic link between pelagic and benthic systems (Lauringson et al. 2009, Koivisto & Westerbom 2010, 2012) and maintain self-purification and water quality in marine coastal ecosystems.

3.1 Human activities deteriorate the state of the marine environment

Eutrophication, seabed disturbance and hazardous substances along with many other human activities have resulted in the deterioration of the Baltic Sea marine environment during the last few decades. Nutrient loading has mainly been caused by land-based activities such as agriculture and forestry but also from sea-based point-source pollution sources, e.g. aquaculture. Also, the release of nutrients, especially phosphorus from the seabed sediment under anoxic conditions can contribute to the eutrophication process.

The autonomous Åland Islands are located in the Northern Baltic Proper between Finland and Sweden. The main island covers about 70 % of the total land area with 90 % of the inhabitants (Ålands landskapsregering 2012). Sixty of the largest islands are inhabited. The autonomous area is divided into 16 municipalities, most of which have direct coastline with the Baltic Sea. The export of agricultural and fish products is one of the main sources of income for the area. The mild climate and calciferous bedrock provide optimal growth conditions for vegetable and fruit production but also for a rich natural terrestrial flora. Around 60 % of the land areas are covered with forests.

The sea areas can be characterized by shallow inlets and bays but also with deep sea bottoms in the open sea areas. As a result, a great variety of underwater habitats is located within the marine areas of the Åland Islands, providing sustenance for invertebrates, fish, birds and seals. The marine environment provides important ecosystem services for inhabitants and tourists, offering possibilities for fisheries, fishing and other recreational activities.

Local sources contributing to eutrophication in the Åland Islands include aquaculture, agriculture, settlements and traffic. Furthermore, maritime traffic contributes to the nutrient content of the seawater. The annual phosphorus loading is 50 tonnes per year (tonne/y). It has been estimated that the nitrogen loading was 900 tonnes/y but was reduced to 805 tonnes/y in 2006–2012 (Ålands landskapsregering 2012). Aquaculture contributed about 65 % of the phosphorus loading in the Åland Islands, whereas agriculture contributed 10 % and settlements 9 %. Considering nitrogen-loading, aquaculture contributed 30 %, settlements 8 % and agriculture 39 % on average (Ålands landskapsregering 2012).

Aquaculture is an important livelihood on the islands, providing jobs and income for local people. The environmental impacts of aquaculture, increased nutrient and organic matter content in the seawater, have resulted in stricter permitting processes in the Åland Islands but also in other Baltic Sea countries. This has led to a situation where new concepts are needed for both developing the aquaculture industry while simultaneously protecting the already impacted marine environment. Nutrient offsetting is seen as one potential concept for addressing both of these issues.

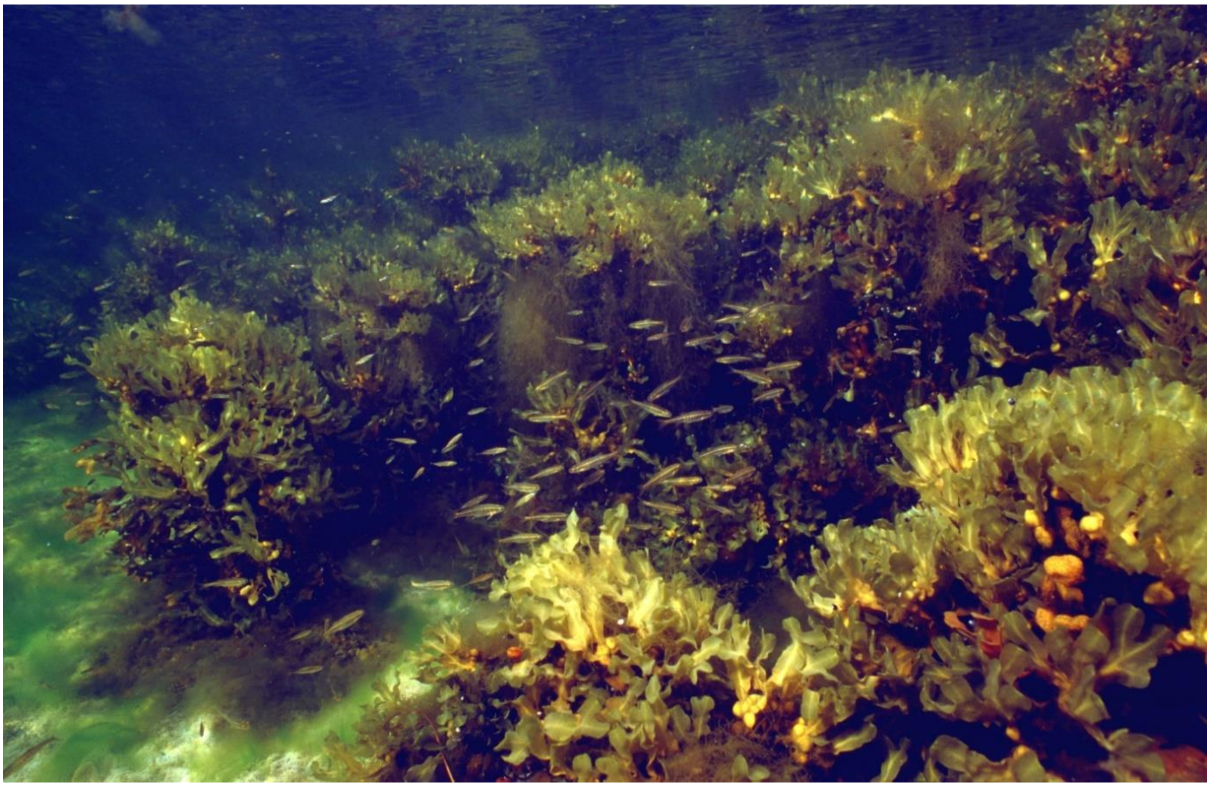

Photo: Visa Hietalahti

4 Legal aspects in developing offsetting in aquatic environments

4.1 Definitions

4.1.1 Ecological compensation and nutrient offsetting

The regulation of ecological compensation and offsetting is subject to an increasing amount of discussion in the Nordic context (Enetjärn 2015, Naturvårdsverket 2015 & 2016, Laas 2016, SOU 2017:34, Leino & Belinskij 2018, Suvantola et al. 2018). The studies have presented several definitions of ecological compensation and compensation measures that are relevant to the concept of offsetting in aquatic environments.

A 2017 Swedish investigation defines ***compensation measures*** as measures to compensate for an expected damage. ***Ecological compensation*** has been defined as 'an indemnification of the entire or partial damage to the environment or nature values such as species, nature types, ecosystem functions and recreational values' (SOU 2017:34). Yet another definition suggests that ecological compensation should concern the restoration of valuable environments, which fulfil the corresponding function of a habitat destroyed or damaged by strictly *physical* pressures (Laas 2016). In Finland, legal literature defines compensation as measures to offset a damaged natural value. Here, the starting point is that the deterioration of natural values caused by an activity is compensated by increasing or protecting natural values at another place (*ex situ*) (Leino & Belinskij 2018).

A ***nutrient offset*** has gained less attention as a concept in legal literature, but a 2018 policy brief by Finnish researchers defines it as follows:

> "*A nutrient offset typically refers to a verified, certified and registered unit that corresponds to a unit of additional nutrient reduction. There are also specific rules for verifying and measuring the generation of a nutrient offset. Most importantly the offset must generate an additional reduction in nutrient loading, i.e. reduction that would not have taken place otherwise. If nutrient offsets would be used in permitting processes, similar rules and practices should apply. In relation to environmental permitting activities, there are two potential ways to generate nutrient offsets. First, a nutrient offset could be generated through nutrient abatement in excess of an activity's legal requirements defined by environmental regulations. Second, they could be generated by removing nutrients from a water body, the amount of removed nutrients comprising the offsets. Nutrient offsets require that the effects of the actions taken can be measured and the outcomes verified. Furthermore, the reductions must be additional, and they are not supposed to induce new loads elsewhere.*" (Belinskij et al. 2018a)

Aquatic ecosystems are strongly interconnected and do not offer the same possibilities to dispersal barriers as ecosystems on land. The hydrological connectivity of aquatic ecosystems means that nutrients circulate and spread from one place to another. It also means that emissions from land-based sources have a significant impact on the environmental status of coastal and marine waters. Yet, compensation measures on land may not have the desired effect in aquatic ecosystems. The strong flow of waters, ice and the sea wind constantly modify aquatic ecosystems, adding to the complexity and uncertainty of coastal and sea-based compensation measures (UNEP-WCMC 2016, Leino & Belinskij 2018).

Because of this interconnectedness, an assessment on nutrient compensation measures should consider the environmental impacts of individual projects as a part of the cumulative effects from all emission sources in a larger area or an entire river basin district (Leino & Belinskij 2018, Suvantola et al. 2018). Also, it is important to notice that some legal principles and aspects connected to ecological compensations may not be fully applicable in the case of nutrient offsetting.

A 2016 study for the Åland Islands proposed a wider definition of compensation that is not limited to physical pressures or measures carried out *ex situ*. Instead, compensation would include measures that clearly increase the possibilities to achieve the water quality objectives of the EU Water Framework Directive when reasonable mitigation and prevention measures have been fully considered (Kymenvaara & Eklund 2016). This definition takes a recipient water body as a starting point and allows any compensation measures that contribute to meeting the established environmental objectives.

4.1.2 Mitigation hierarchy and the two step-assessment

The BBOP's standard and guidelines on ecological compensations does not enjoy any sort of legal status but can be considered a best practice and recommendations for the development of regulation. The mitigation hierarchy is well-established in legal literature and, in some cases, also enshrined by EU environmental law such as article 6 of the Habitats Directive (European Commission 2001, Soininen et al. 2019). In the so-called *Briels case* (C-521/12), the European Court of Justice (ECJ) highlighted the importance of the precautionary principle when applying derogations under the Habitats Directive and that only measures contributing to additionality may be approved as compensation. These conclusions have been re-endorsed by later case law (C-387/15 and C-388/15).

In these ecological compensation cases concerning the replacement/restoration of the loss of a specific natural value, the so-called two-step assessment of the mitigation hierarchy has been applied. It means that ecological compensations can be considered only in the second step, after the project is deemed permissible through an exemption in the first step. However, it is doubtful whether the two-step assessment can be strictly applied when dealing with nutrient offsetting and generic aquatic structures that are not threatened species or habitats (Josefsson 2019). A nutrient offset aims to ensure that an activity does not deteriorate the status of a receiving water body and can be thus permitted without an exemption.

4.2 EU Law

4.2.1 Water Framework Directive

The EU Water Framework Directive (WFD) requires that Member States reach 'good ecological status' in inland surface waters, transitional waters and coastal waters by 2015. Water bodies must be classified according to an assessment of its ecological status as regulated by WFD Annex V. Good ecological status is primarily based on three or four biological quality elements depending on the water body at issue (WFD, Annex V).

The classification is also based on a water body's physio-chemical and hydro-morphological status. Nutrient conditions constitute one of the five (5) physio-chemical quality element indicators, but the occurrence of nutrients also influence the status of the biological quality element phytoplankton for coastal water bodies. The assessment leads to assigning water bodies in the five ecological status classes (high, good, moderate, poor and bad) according to the quality element with the lowest status.

In addition to achieving good status, Member States are obliged to implement the necessary measures to prevent the deterioration of the status of all water bodies. This is the principle of non-deterioration, which constitutes the Directive's second main environmental objective. The Directive does not seek to achieve complete harmonization of the water legislations between EU Member States. Indeed, the approaches vary considerably, and the effectiveness of achieving good status is dependent on them (ECJ C-32/05).

In the 2015 Weser judgment (C-461/13), the European Court of Justice clarified that the Member States are required, unless a derogation is granted, to refuse the authorization of an activity if it may cause a deterioration of the status of a body of surface water or jeopardize the attainment of good status. Not only did the judgment confirm that the environmental objectives are legally binding, it also assigned significant legal status to the principle of non-deterioration. The Court further clarified that the deterioration of the status occurs as soon as the status of at least one of the quality elements under Annex V falls by one class, even if that decline does not result in a change in the classification of the body of surface water. The Weser judgment concerned the dredging of the German Weser river, which would influence the hydro-morphological status of the water body in question.

Soon after the Weser judgment, the application of the derogation under article 4(7) was clarified by the Court in the Schwarze Sulm case (C-346/14). The Court accepted that the promotion of renewable energy was of such an overriding public interest that a new hydropower plant could be subject to derogation under article 4(7). The Court stated that the Member States should be allowed a margin of discretion in this regard.

Initially, this derogation discretion seems wide, but for polluting activities such as nutrient emissions, the scope of applying article 4(7) of the WFD is very limited in practice (Kymenvaara et al. 2019, Soininen et al. 2019). Firstly, according to article 4(7), an activity with nutrient emissions may only lower the status of a water body from high to good because it belongs under the group of 'new sustainable human development activities'. Second, article 4(7) requires that "all practicable steps are taken" to mitigate the adverse impact of the project with nutrient emissions. Third, it requires that reasons of "overriding public interest" and/or the benefits of human health, human safety or sustainable development that comes with the new project weigh heavier than the benefits to the environment and society of achieving the environmental objectives.

4.2.2 WFD and compensation measures

The WFD does not specifically address the concept of compensation. Neither does it recognize the mitigation hierarchy or measures undertaken *ex situ* to achieve good status. Yet article 4(1)(a)(i) requires that Member States shall "implement the necessary measures" to achieve the environmental objectives. The wording suggests that there are no conceptual barriers to incorporate the mitigation hierarchy, including compensation measures, to achieve good ecological status.

Nevertheless, compliance with the two-step assessment related to the mitigation hierarchy implies practical challenges in the context of the WFD and nutrient offsetting. The two-step assessment requires that compensation is considered only in the second step – after the project is deemed permissible in the first. However, following the Weser judgment, a project with nutrient emissions may not be permitted if it risks leading to deterioration, and, thus, nutrient compensation measures cannot be taken as a second step (Kymenvaara & Eklund 2016, Josefsson 2019). In contrast to the two-step assessment, a growing body of research holds that compensation measures should belong to the full set of tools available to achieve good ecological status (Leino & Belinskij 2018, Suvantola et al. 2018, Josefsson 2019, Soininen et al. 2019).

The mitigation hierarchy serves as a useful tool in the application of compensation as a last step in the mitigation hierarchy. However, the WFD suggests that compensation should not be separated from the question of permissibility of a project with nutrient emissions. Therefore, nutrient offsetting can also be regarded as an impact mitigation measure that prevents the deterioration of the status of a water body.

In the context of derogation according to article 4(7) of the WFD, the Commission notes that there is a distinction between mitigation measures and compensation measures undertaken *ex situ* (Commission 2009). This distinction is also clearly upheld by article 6(4) in the Habitats Directive, which requires restoring or recreating habitat on a new or enlarged site that is subsequently incorporated into the Natura 2000 network. The wording "all practicable steps" under article 4(7) should be widely understood to include all mitigation measures that are technically feasible, not disproportionately costly and compatible with the new project or new sustainable human development activity. According to the Commission, mitigation measures may even be carried out in other water bodies, provided their effect occurs in the water body for which article 4(7) is applied (Commission 2009, 2017). Indeed, "all practicable steps" would seem to require that all measures, including compensation measures, should be fully explored before a derogation is granted according to article 4(7) WFD (Josefsson 2019, Soininen et al. 2019).

4.2.3 Marine Strategy Framework Directive

The EU Marine Strategy Framework Directive (MSFD) is applicable to marine waters beyond one nautical mile from the baseline. It requires Member States to achieve good environmental status in marine waters by 2020 (article 1). Good environmental status is defined by the following qualitative descriptors in the MSFD as Annex I: 1) biological diversity, 2) the level of non-indigenous species introduced by humans, 3) the limits of the populations of all commercially exploited fish and shellfish, 4) elements of the marine food webs, 5) human-induced eutrophication, 6) sea floor integrity, 7) alteration of hydrographical conditions, 8) contaminants, 9) contaminants in fish and seafood, 10) marine litter and 11) introduction of energy, including underwater noise.

For the Baltic Sea, the parties to the Helsinki Convention on the Protection of the Baltic Sea shall take all appropriate legislative or administrative or other relevant measures to prevent and eliminate pollution in order to promote the ecological restoration of the Baltic Sea Area and the preservation of its ecological balance (MSFD, art. 3). According to the Baltic Sea Action Plan, the Baltic Sea should be unaffected by eutrophication, meaning a return to 'normal' levels of oxygen and algae. The Baltic Sea Action Plan includes the HELCOM Nutrient Reduction Scheme, revised in 2013, which is a regional approach to sharing the burden of nutrient reductions and defining the country-allocated reduction targets.

In comparison with the WFD's good status, the MSFD's objective good environmental status is more vaguely framed. In addition, it has not been subject to a similar interpretation as the WFD's goal of good ecological status by the Weser judgment. It is still unclear what kind of legal character the MSFD's good environmental status has (Kymenvaara & Eklund 2019, Soininen et al. 2019).

According to article 1 of MSFD, Member States shall 'take the necessary measures' to achieve or maintain good environmental status in the marine environment by the year 2020. Taking the necessary measures is a similar obligation as the obligation on Member States to 'implement the necessary measures' under article 4 in WFD (see above). Along the lines of the discussion above, taking the necessary measures under article 1 of MSFD appears to include all the steps of the mitigation hierarchy, including compensation measures, in connection with environmental permitting of activities with impacts on marine waters.

4.3 Åland Islands

Activities with nutrient emissions on the Åland Islands are permitted according to the Water Act (Vattenlag (1996:61) för landskapet Åland) and the Environmental Protection Act

(Landskapslagen (2008:124) om miljöskydd). These two acts set out the permitting framework for nutrient emissions stemming from, for example, wastewater plants and aquaculture activities.

Under the Water Act, the government should issue so-called 'quality norms' to reduce eutrophication (5:4 §). The quality norms function as limit values for the occurrence of a pollutant such as nutrients. The norms have direct legal consequences in the water area for which they are established insofar that a new or altered activity may not be authorized if quality norms have not been issued or if they have not been met (5:9 §). The government has not issued quality norms for eutrophication, and this has brought a complete stop in the permitting of new activities with nutrient emissions. Existing activities have mainly been re-permitted at intervals of ten years since 1997.

The Åland Islands implemented the WFD and the MSFD as a separate chapter to the current Water Act (Chapter 5). Neither of the Directives are connected to the provisions on environmental permitting, and the environmental objective of good ecological status does not have legal significance in the permitting of activities with nutrient emissions.

Administrative practices of the permitting authority Ålands miljö- och hälsoskyddsmyndighet (ÅMHM) and case law by the Åland Administrative Court (ÅFD) demonstrate the complete disconnection between permitting and the WFD's environmental objectives. Even after the Weser judgment, ÅMHM and ÅFD disregarded the WFD's environmental objectives and the administrative borders of water bodies in the permitting of an aquaculture in a case decided in 2016 (See Ålands förvaltningsdomstols beslut 52/2016, diarienummer 2014/44; ÅMHM's decisions 2016-546 ÅMH-Pn 6/17 and 2016-546 ÅMH-Pn 5/17of 24 June 2017 and 2014-564 ÅMH-Pn 2/17 of 15 February 2017).

4.3.1 The 'improvement surplus'

The prohibition for permitting new or altered activities may be circumvented if it is demonstrated that an activity does not contribute to increased eutrophication, or if an improvement surplus (förbättringsöverskott) is used (Water Act 5:9 §). In other words, the Water Act allows operators to expand and initiate new activities in these situations (5:12 §). The improvement surplus is an extra improvement of water quality; the consequence of a water quality improvement measure that 'creates better water quality than required by the Act'.[1] Neither the Act nor the government bill explains what a water quality improvement measure could entail.

So far, all applications to utilize an improvement surplus to expand activities with nutrient emissions have been rejected. Applications have concerned nutrient uptake by trawling fish species subject to fishing quotas in order to reduce the excess of nutrients in waterbodies. The law requires a 'direct connection' between the improvement measure and the surplus (Water Act 5:12 §), and the Ålandic government has argued that the environmental benefits of the proposed measures (trawling) are uncertain, possible beneficial impacts could not be allocated to a specific water area, and that species subject to quota would have been taken up in any case, which gives no additional benefit in comparison with the status quo (See Decisions nr 40 (ÅLR 2011/6672 36 S40) and 41 (ÅLR 2011/6671 37 S40) of 5 June 2012 and nr 123 (ÅLR 2015/2211 249 S3). In general, these cases illustrate challenges related to nutrient offsetting.

4.3.2 The 2016 investigation on a new Water Act

In 2015, the government of the Åland Islands initiated an investigation on the revision of the existing Water Act. The revision was preceded by two decades of industry requests and political

1 The desired water quality is defined by the Water Act 1:3 §; and Water Decree (Vattenförordning (2010:93) för landskapet Åland) Chapter 7.

pressure for a renewal of the existing legal framework. Initially, the investigation concerned the provisions on the improvement surplus and correct implementation of the WFD and MSFD, but once the Weser judgment was given in July 2015 the study expanded to the permitting framework.

A 2016 study on a new Water Act proposes a legal framework connecting the WFD and MSDF environmental objectives to environmental permitting and a way to utilize compensation measures in order to make projects permissible (Kymenvaara & Eklund 2016). In line with the current Water Act, permitting under the proposed act applies to activities with physical pressures and polluting emissions alike. Both Directives' (WFD and MSFD) environmental objectives are specified by means of water quality requirements (vattenkvalitetskrav) that the government must establish for each water body and for marine waters (48 § and 49 §) to achieve good ecological status and good environmental status, respectively. While water quality requirements for good status under WFD are legally binding, the local government may choose to adopt water quality requirements for good environmental status under the MSFD as legally binding (obligation of result) or as benchmarks that should be strived for (obligation of best effort) (47 §).

Permitting under the proposed act is based on the concept of 'detrimental water impact' (negativ vattenpåverkan), which encompasses a range of negative effects on waters and aquatic environments (4 §). Along the conclusions of the Weser judgment, the concepts of 'deterioration of status' and 'jeopardize the environmental objective' can be found among the types of detrimental water impacts listed in the proposal (4 §). The established water quality requirements function as a tool with which the permitting authority evaluates if an activity leads to detrimental impacts such as deterioration of status of a specific water body.

According to the proposed act, as a starting point, detrimental water impact is forbidden, and a project may not be authorized if it leads to these negative effects as defined by 4 § (62 §). However, the study proposes a definition of compensation, which encompasses all measures that offset (uppväga) the detrimental water impact of an activity subject to permit. The proposed act and its commentary clarify that compensation measures can be undertaken once prevention and mitigation measures have been fully considered (60 §) (Kymenvaara & Eklund 2016).

If an activity, despite reasonable[2] prevention and mitigation measures, may lead to a situation where a legally binding water quality requirement cannot be met, the operator has three options. The operator may 1) apply additional mitigation measures (where such are available), 2) utilize the benefit of a compensation measure or 3) pay a 'water improvement fee' for complementary measures under the programme of measures of the river basin management plan. The complementary measures are then carried out by the government of the Åland Islands. These measures (1-3) should 'clearly improve the possibilities to meet a water quality requirement' (Kymenvaara & Eklund 2016). Thus, the proposal incorporates the mitigation hierarchy, but allows compensation to make the projects permissible.

4.3.3 Compensation measures in the new government bill (2019)

In April 2019, the government of the Åland Islands circulated among the various referee groups a draft bill on a new Water Act for the Åland Islands. The 2016 investigation provides the overarching framework for the bill, which follows the logic and structure of the 2016 conclusions as regards compensation.

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2 The wording 'reasonable' means a cost-benefit analysis that applies to general provisions on prevention and mitigation (6 §), location (7 § 1-3 mom.) and BAT or BEP for the branch of activity in question (10 §).

In the autumn of 2019, a new version of the draft bill provided amended rules on the concept of compensation in permitting (Vattenlag för landskapet Åland, version den 22 oktober 2019). In line with the 2016 study, 'harmful water impact' (försämrande vattenpåverkan, 5 §) includes the concepts of 'deterioration of status' and 'jeopardize attainment of good status' as well as 'significantly deteriorates good environmental status' (referring to the MSFD objective) (5 §)[3]. According to 47 § of the draft Water Act, an activity may not as such or jointly with other activities lead to a harmful water impact. Thus, the deterioration of status and jeopardizing good status is prohibited. However, an activity may be permitted if measures according to 48, 51 or 52 §§ counteract the harmful water impact.

1) 48 §: Additional mitigation measures

As a first step, the permitting authority may order the operator to undertake additional mitigation measures if the activity contributes to non-achievement of a water quality requirement, despite adhering to 6-9 §§, which state the following:

i. 6 §: Activities should be carried out regarding the environmental objectives, water quality requirements and sustainable development. The environmental objectives list good ecological status and good environmental status (3 §) and the water quality requirements specifying these objectives (33 §) per water body/area of marine waters.
ii. 7 §: An activity should be carried out with a knowledge of its environmental impact. The goal of the activity should be achieved with the least damage or inconvenience for the environment but without making it impossible to carry out due to costs. This implies a cost-benefit analysis of preventive measures. Harmful water impact should be avoided; thereafter any remaining damages should be restored (återställas) and lastly compensated. This codifies the mitigation hierarchy.
iii. 8 §: The location of the activity should be chosen so that the goal of the activity can be achieved with the least harmful water impact and without unreasonably high costs.
iv. 9 §: An activity should be carried out by applying the best available technology and best environmental practice.

In other words, despite reasonable prevention and mitigation measures according to 6-9 §§, an activity may be ordered to take additional mitigation measures to avoid harmful water impacts. The requirement of reasonableness is expressed in the cost-benefit analysis in both 7 § (prevention) and 8 § (location) as well as 9 § (mitigation according to BAT must be economically possible to use for the branch of activity in question) (Lagförslag om vattenlag till landskapet Åland, utkast 22 oktober 2019).

According to 12 § of the draft act, the local government may issue in a decree more detailed provisions specifying BAT/BEP for certain types of activities or the localization of certain activities in order to increase the possibilities to achieve good status or good environmental status and prevent harmful water impact according to 4 §. In 13 §, it is stated the government must issue by means of a decree the maximum allowable emissions of phosphorus and nitrogen for point and non-point sources such as animal farming, fish farming and wastewater treatment plants. In determining such provisions, the government shall adhere to the agreements with states in the Baltic Sea area that specifies limits for emissions of nutrients, i.e. the HELCOM. Both 12 and 13 §§ aim to clarify the requirements on activities with nutrient emissions as well as facilitate the work of the relevant authorities (Lagförslag om vattenlag till landskapet Åland, utkast 22 oktober 2019).

3 Lagförslag om vattenlag till landskapet Åland, utkast 22 oktober 2019.

2) 51 §: Compensation measures

When additional mitigation measures have been fully considered (51 § of the draft act[4]), the operator may utilize the benefit of a compensation measure. The proposed definition of a compensation measure follows the nature and logic of the 2016 investigation with certain modifications. A compensation measure should, as such, or jointly with other measures, influence the water status or environmental status in a way that clearly increases the possibilities to meet a water quality requirement.

Because the provision is focused on the effect of a compensation measure in the recipient in question, it allows measures to be performed outside the activity's impact area (*ex situ*) provided that it influences the activity's impact area in a way that increases the possibilities to meet the water quality requirement in the water body in question. The bill specifies that measures may be undertaken in the larger monitoring area to which the water body in question belongs, provided it has the desired effect in the water body in question.

To be approved as a compensation, a measure must meet certain requirements. It should lead to a long-term benefit (5 §, 2 mom., 1 para.), provide an additional benefit in comparison with a situation where it would not have been carried out (3 para.) and provide a benefit that may not have been accounted for in another context (4 para.). The benefit of the measure must be reasonable in relation to the cost and supervision of carrying it out (5 para.). Lastly, the measure must be authenticated in a reliable manner (6 para.).

The benefit of a compensation measure may be transferred, implying that actors other than the operator of an activity may undertake compensation measures. The bill reads that the evaluation of a benefit should consider the measure's significance for the quality elements and harmful substances according to the water quality requirement for a certain water body. Regarding the effect of a measure, the assessment is more about an estimation and less about exact science. The effect must, nevertheless, be sufficiently palpable in order to "clearly increase the possibilities to meet a water quality requirement" (Lagförslag om vattenlag till landskapet Åland, utkast 22 oktober 2019).

3) 52 §: Improvement surplus

The new draft act allows the use of an improvement surplus, which follows the nature and logic of the current Water Act. An improvement surplus is defined as an extra improvement of water quality beyond what is required by the water quality requirements under the proposed Act, and which creates an additional, lasting improvement, more than what is achieved through compensation. Thus, despite restrictions and prohibitions in the Act, a new activity or the expansion of existing operations can be permitted if it is directly connected to the creation of an improvement surplus.

The requirement of a direct connection is unclear. Up to two thirds of the improvement may be utilized and it can be transferred to be used by another person. As a main rule, it may only be utilized in the water body where the activity is carried out. Yet, the permitting authority may allow an operator to use the improvement surplus in another water body if it is demonstrated that the measure has created an improvement in the latter water body. The government aims to delegate the powers to legislate in a decree on the documentation required to determine whether an improvement surplus has been created.

4 The draft act version from October 2019.

4.4 Sweden

4.4.1 Environmental goals and quality standards

Sweden works for achieving the country's environmental objectives through its national system of environmental quality goals (miljökvalitetsmål). The system is governed by the Swedish Environmental Protection Agency (EPA) and engages a range of state and regional actors such as the permitting authorities, which are the regional county boards (länsstyrelserna) and the municipalities.

Ecological compensation relates to many of the 16 environmental quality goals such as those connected to wetlands, lakes and water areas. The goals have no direct legal status and cannot serve as a legal basis for obligations on private persons or authorities. Rather, they set the political direction for the national work on improving the environment and instruct the authorities' work at a general level. Yet, the goals may be relevant to the application of the law in environmental permitting, particularly when the rules allow for different interpretations. The goals also indicate the legislator's view on what constitutes important public interests, and this may be relevant to the extent to which compensation can be used and required (Michanek & Zetterberg 2017). One of the environmental quality goals is 'no eutrophication', and the relevant quality requirements for this goal are specified by the environmental quality standards for good ecological status under the WFD and good environmental status under the MSFD.

Following the entry into force of the WFD and the MSFD, Sweden incorporated the Directives' environmental objectives into its national system of environmental quality standards (miljökvalitetsnormer) under the Environmental Code (1998:808) and its interconnected decrees and regulations. The environmental quality standards for water derive their normative content from Annex V of the WFD and apply to each water body, while the descriptors of the MSDF apply to marine waters.

Previously, the environmental quality standards for water were not considered legally binding in permitting. Instead, obligations on operators with impact on waters were subject to a cost-benefit analysis under the Environmental Code 2:7 §. Following an amendment of the Code as of 1 January 2019, they are legally binding insofar that the Code exempts the requirement of reasonable permit conditions (5:4 §, 5:5 §, 2:7 §, see more below).

4.4.2 Permitting and environmental quality standards for water

The main substantive provisions for environmental permit regulations applicable to all activities including nutrient emissions are found in the 2nd chapter of the Environmental Code. These include the general rules of consideration (allmänna hänsynsregler) that concern the choice of location (6 §), the best available technique (3 §), the obligation to be aware of the activity's environmental impacts (2 §), the efficient use of resources (5 §) and a cost-benefit analysis (rimlighetsavvägning) (7 §). Permit regulations (tillståndsvillkor) derive their substantive contents from these general provisions, which constitute the basis of the assessment on permissibility.

The cost-benefit analysis in 7 § stipulates that application of the general rules of consideration should assess the risk of damage or inconvenience in relation to the impact on human health and the environment. The assessment should be based on a measure's ability to prevent or limit damage or inconvenience and the cost of such a measure. National environmental objectives defined by the parliament should steer the evaluation of a preventive or mitigative measure's benefit for human health and the environment (Government bill 1997/98:45). As of 1 January 2019, the cost-benefit assessment may not lead to permits being authorized if they cause prohibited degradation or put at risk the possibility to reach good water status or potential. The wording of the provision is derived from the Weser judgment (Government bill 2017/18:234). In relation to nonwater environmental quality standards, compensation measures may be ordered to meet the requirements (2:7 §).

The consequences of the recent amendments to the Environmental Code mean that only prevention and mitigation measures may be ordered to meet the requirements of environmental quality standards for the WFD, not compensation measures.[5] Nevertheless, the law states that all the necessary conditions to meet the requirements of a water quality standard should be prescribed by authorities in permits given (5:5 §). This could include compensation measures to offset nutrient emissions or other nuisance, but this is mentioned only in the bill (Government bill 2017/18:234), not in the Environmental Code. It has been criticized that the Environmental Code does not specifically mention compensation measures in relation to water environment (Josefsson 2019), that being questionable from the perspective of the WFD wording "necessary measures" and "all practicable steps" (art. 4(1) and 4(7))

A possibility to compensate

The concepts of ecological compensation and offsetting are not defined by Swedish legislation. However, the Environmental Code contains certain provisions on compensation measures. While compensation measures are not related to reaching the water environment quality standards, compensation or offsetting is applicable to all sorts of environmental damage, including nutrient emissions in aquatic environments.

According to 16:9 § of the Environmental Code, a permit or dispensation may be combined with an obligation to carry out or pay for special measures to compensate for an activity's infringement of a public interest. This is a general provision with a wide area of application that grants the authority a *possibility* to oblige a permit holder to take certain measures to counteract or compensate for environmental damage or intrusion caused by its activity. The provision concerns violations of environmental public interests as well as other types of public interests. A *possibility* means that the permitting authority must not require such measures in connection with each permit or dispensation. The authority should rather perform an assessment of the severity of the damage caused by the activity in comparison with the benefit that the permit holder can be expected to gain from the activity (preparatory works of the Environmental Code).

The provision allows the authority to order the payment of economic compensation in lieu of taking concrete action. The wording of 16:9 § could also be interpreted so that it does not provide a substantive basis for ordering compensation measures but merely informs the authority on a possibility to do so. This has caused discussions on the suitable legal basis for compensation measures (Moksnes et al. 2016).

Compensation in nature protection areas

In relation to nature protection areas (naturreservat) including Natura 2000 areas, 7:7 § of the Environmental Code allows the authority to grant an exemption or dispensation from regulations in these areas in case of special reasons. A decision on dispensation may only be granted if the violation of the value of the nature affected (naturvärde) is reasonably compensated in the protected area or in another area. In other words, a dispensation from regulations requires special reasons, and in this case, compensation *must* be carried out to a reasonable extent. Thus, compensations are obligatory and are applied also in the case a decision on a protected area is repealed (7:7 §). The wording 'reasonable extent' implies an assessment on reasonable compensation and that the loss in values of nature need not to be entirely compensated. The provision does not allow economic compensation, only concrete measures.

5 The previous wording of 2:7 § enabled compensation measures for all types of environmental quality standards.

If the dispensation concerns an activity subject to permit, 16:9 § of the Environmental Code is applied in parallel. The provision on compensation requirements in 7:7 § is limited to infringements of natural values in the nature protection area. Compensating infringements of other interests, such as recreational values in protected areas, are, instead, governed by 16:9 §.

Legal basis for compensatory measures

In environmental permitting, compensation measures could, in theory, be ordered based on the general rule in 2:3 §, which requires an actor to "perform protective measures, apply limitations and take *other precautionary actions* that are required to prevent and *counteract damage* or inconvenience to human health or the environment".

One could argue that "other precautionary actions" to "counteract damage" would include compensation measures. In this way 2:3 § could be applied in line with the mitigation hierarchy through ordering compensation measures *after* assessing permissibility according to the most suitable location and fully exhausting protective measures and other limitations. However, in this case compensation measures would not be separated from the issue of permissibility of the activity. That is not aligned with the mitigation hierarchy, which states that an activity should be assessed in two steps to settle the question on compensation after permissibility.

Yet the preparatory works for 2:3 § state that compensation measures according to 16:9 § or 7:7 § may be ordered if protective measures are not sufficient for permissibility. It would seem to suggest that the substantive legal basis for compensation measures is not 2:3 §, but 16:9 § or 7:7 § as the wording *protective* and *precautionary action* (skyddsåtgärder och försiktighetsmått) in 2:3 § may be considered too narrow to also cover compensation measures (SOU 2017:34).

In view of the above ambiguities, the Swedish Environmental Protection Agency has suggested that the substantive legal basis for compensation measures should be 16:9 § (SOU 2017:34, Naturvårdsverket 2016). In this way, the question on permissibility would be separated from that on compensation, and permit consideration would be conducted in two steps (SOU 2017:34).

In summary, the effect of compensation measures ordered as a part of the general permit consideration under chapter 2 of the Environmental Code would influence the permissibility of the activity. This would challenge the correct application of the mitigation hierarchy and an assessment in two steps. Nevertheless, this way of reasoning is not applicable for offsetting nutrients in aquatic environments. In the case of nutrients, compensations should belong to the full set of tools to make projects permissible.

4.4.3 Examples of compensations in Sweden

The possibility to require ecological compensations in connection with permitting has been sparsely used thus far in Sweden. Most compensations carried out have been related to nature conservation areas. The largest ecological compensations have concerned violations of protected nature due to railway construction.

Compensation measures as enshrined in the Environmental Code are equally applicable to land and water areas. In the marine environment the largest case so far was carried out to compensate environmental damage caused by harbour activities. In older water permits, conditions may refer to 'compensation measures' or 'measures to rectify damages'. Today, these measures would belong to the category of minimization or prevention measures (as a part of BAT or BEP). The dividing line between what measures are considered customary measures to prevent and minimize damage (skyddsåtgärder eller försiktighetsmått) and what are considered compensation measures has not been clearly upheld in case law.

In an aquaculture case decided upon in the Land and Environment Court of Appeal, compensation measures were ordered in a permit authorizing fish farming. Despite the reference to 16:9

§, it is not clear what legal basis the Court used to order the compensation and if/how the mitigation hierarchy was applied. The compensation was carried out through building wetlands at a place other than the location of the activity (case law from the Land and Environment Court of Appeal MÖD 2005:5).

4.4.4 Study proposes changes to the Environmental Code

According to a 2017 state investigation, a consistent and systematic use of *ecological* compensation requires certain clarifications in Swedish law (SOU 2017:34). To this end, the study proposes the following amendments to the Environmental Code:

i. An obligatory requirement to assess the need of compensation measures. This requires: a) an amendment of existing provisions on what material should be presented to the authority as a part of the permit application. By adding 'compensate' to 6:7 § 2 p., an environmental impact assessment should clearly contain a description of the planned measures that may compensate for significant environmental impact; and b) an addition to 16:9 § that obliges the authority to assess if special measures for the activity's (permit or dispensation) violation of public (environmental) interests should be ordered.

ii. A codification of the mitigation hierarchy and a permit assessment in two steps. As a first step, the evaluation should consider the location and reasonable prevention and mitigation measures. If the environmental impact cannot be sufficiently limited in the first step, compensation measures should not be ordered to make the activity permissible. Instead, the permit should be rejected in that situation. To this end, a new 8a § is proposed to the Code's 2nd chapter. According to the provision, reasonable compensation measures are assessed once the activity is considered permissible, and the compensation should correspond to the violation of the environmental damage / lost natural value. The new 8a § prescribes the following priority order to address the activity's impact: 1) avoidance, 2) limitation, 3) restoration and 4) compensation.

iii. A possibility to order compensation measures in connection with permit supervision. An addition to 22:25 § clarifies that a permit decision or judgement should contain the necessary conditions that are required to avoid, limit, restore and, lastly, compensate for damage or loss. This strengthens the role of the mitigation hierarchy in the permit assessment and allows the supervisory authority to undertake supervision to ensure that compensation measures fulfil their intended purpose. Such conditions could also contain information on the consequences of not achieving the intended function of effect of such measures.

Referral round highlights the lack of relevance for aquatic environments

In the referral round, the following statements were provided on the 2017 state investigation:

i. The study's main focus is on land-based ecosystems with a lack of relevance for compensation in aquatic environments and compensation to achieve good ecological status. In the light of the conclusions of the Weser judgment, the special prerequisites for compensation in relation to the environmental quality standards for water and aquatic ecosystems should be further investigated.[6]

6 Remissyttrande av Havs- och vattenmyndigheten, 2018-10-02: https://www.regeringen.se/4a8129/contentassets/2528ad4c3b1243ec84df1cedc0e19e48/havs-och-vattenmyndigheten.pdf.

ii. The relationship between compensation measures in permitting and the measures of the programme of measures in a river basin management plan require further analysis.[7]

iii. It is generally viewed as important to codify the mitigation hierarchy. However, doubts have been raised on whether an assessment in two steps is necessary and functional in contexts other than in Natura 2000 matters. An assessment in two steps would also risk to further complicate and lengthen permitting procedures.[8]

iv. There is a risk of hollowing the provisions on preventive and mitigation measures (BAT/BEP, location). This could be addressed through the new 2:8a §, which should clarify the difference between prevention and mitigation to earn a permit on one hand and compensation on the other. The new provision should explicitly state that compensation measures may not affect the question on an activity's permissibility.[9]

4.5 Finland

4.5.1 The legal status of the environmental objectives under the WFD and MSFD

Finland implemented the WFD by establishing a new framework for river basin management planning governed by the Act on River Basin Management and Marine Strategy[10] and the Water Management Decree.[11] The legislation focuses primarily on the procedural aspects of river basin and marine strategy planning and less on their respective environmental objectives, their legal effect and their enforcement. The quality elements of ecological status follow the descriptive definitions of WFD Annex V and are described in a ministerial guidance document, thus not established by legislation.

4.5.2 Two types of permits: environmental and water permits

Under Finnish law, an activity with nutrient emissions is typically subject to permits under two legal acts. According to the Environmental Protection Act, activities causing risk of environmental pollution require an environmental permit (27 §). Under the Water Act a water management permit is required for activities that come with structural changes to waters (3:2 §), which would include any type of constructions placed in or close to water. The legal frameworks are procedurally combined and apply to activity with impacts in all coastal and marine waters.

According to the Environmental Protection Act, an operator must organize its activities so that pollution of the environment can be prevented. If pollution cannot be completely prevented, it should be minimized as far as possible (7 §). The general obligations under the Water Act

7 Remissyttrande av Uppsala universitet, 2018-10-04: https://www.regeringen.se/4a8126/contentassets/2528ad4c3b1243ec84df1cedc0e19e48/uppsala-universitet.pdf; The regional water management authority abstained from commenting on the investigation with reference to the study's lack of relevance for implementation of the WFD in Remissyttrande av vattenmyndigheten för Södra Östersjöns vattendistrikt: https://www.regeringen.se/4a8105/contentassets/2528ad4c3b1243ec84df1cedc0e19e48/vattenmyndigheten-for-sodra-ostersjons-vattendistrikt.pdf.
8 Remissyttrande av Mark- och miljödomstolen Nacka tingsrätt, 2018-08-31: https://www.regeringen.se/4a8107/contentassets/2528ad4c3b1243ec84df1cedc0e19e48/mark--och-miljodomstolen-nacka-tingsratt.pdf; Remissyttrande av Vattenfall AB, 2018-10-03: https://www.regeringen.se/4a8126/contentassets/2528ad4c3b1243ec84df1cedc0e19e48/vattenfall.pdf.
9 Remissyttrande av Havs- och vattenmyndigheten.
10 Laki vesienhoidon ja merenhoidon järjestämisestä 1299/2004.
11 Valtioneuvoston asetus vesienhoidon järjestämisestä 1040/2006.

require that projects use water resources so that public and private interests are not violated in a way that may be avoided if the purpose of the project can be achieved without unreasonable cost increase in relation to the total costs and the damage caused (2:7 §).

Both acts appear to incorporate the first two steps of the mitigation hierarchy. However, neither of them defines the last step of the mitigation hierarchy. They do not contain provisions that explicitly allow an operator to compensate environmental pollution or the harm caused on public and private interests.

4.5.3 WFD and MSFD environmental objectives in permitting

The binding character of the WFD's environmental objectives is not reflected in Finnish legislation. The conclusions of the Weser judgment significantly deviate from the presumptions about the nature of the WFD framework and its environmental objectives when the Directive was implemented in Finland. At the time, it was considered a planning instrument for good water status without a direct legal effect on other decision-making (Government bill 120/2004, Constitutional Law Committee statement 45/2004). The Act on River Basin Management and Marine Strategy as well as the Environmental Protection Act and Water Act stipulate that the river basin management plans (which include the environmental objectives) must be 'taken into account' within decision-making, including permitting.

As a main rule, an environmental permit is granted if the project does not cause health hazard or significant pollution of the environment (Environmental Protection Act 49 §). A water management permit is granted if the benefits to public and private interests outweigh the harm to these interests (Water Act 3:4 §). An authority or court must rely on these provisions of sectoral legislation to refuse a permit to an activity that may cause deterioration or jeopardize the WFD environmental objectives.

Both Acts emphasize the assessment of a project's local impacts as well as the possibilities of preventing and mitigating such impacts by means of permit conditions. This includes an assessment on how the project impacts a recipient water body. By taking the river basin management plans into account in the permitting of a project, the assessment could also consider the impacts of other activities on the receiving water body (Suvantola et al. 2018). In theory, this could allow the use of compensation measures carried out at another location (*ex situ*) that have an impact on the water body at issue. However, in practice, the time lag/shift and uncertainties of the effect of compensation measures as well as widening the assessment in general would seem to be challenges for the permit consideration.

4.5.4 Compensations in aquatic environments

Thus far, the Weser judgment has not led to any amendments of the Finnish legislation although a study funded by the Government points to the need of strengthening the role of environmental objectives in permitting (Belinskij et al. 2018b). Currently, the substantive legal basis for the decisions derives from sectoral legislation, but the environmental objectives have an important role in environmental and water permit decision-making when interpreting the provisions of the Environmental Protection Act and Water Act.

The Finnish Supreme Administrative Court has given the WFD environmental objectives a significant role in permitting in line with the conclusions of the Weser judgment (SAC 2014:176; 14.2.2018 t. 608; 20.8.2010 t. 186; 2017:87; 31.3.2017 t. 1484). Lately, this was confirmed in December 2019 through the so-called *Finnpulp case* (SAC 2019:166). In Finnpulp, an exceptionally large biomass plant did not receive a permit because it would have been at risk of deteriorating the quality element "phytoplankton" in the water body in question. The decision of the Supreme Administrative Court was based on the Environmental Protection Act, but its interpretation followed the Weser judgment. Thus, it can be stated that in practice the environmental objectives of the WFD are legally binding in Finland although this is not reflected in national legislation.

Against this background and very limited possibilities to derogate from the environmental objectives, compensation measures have been presented as a potential way to reconcile new projects with the WFD environmental objectives (Leino & Belinskij 2018, Suvantola et al. 2018, Soininen et al. 2019). Compensation measures could offset nutrient emissions and thus allow the authorization of a new project without a derogation from the environmental objectives (Soininen et al. 2019). However, the main legal constraint for ordering compensation measures is that legislation does not contain any explicit provision on them (Suvantola et al. 2018).

4.5.5 Discussions on development of Finnish law

Incorporating the possibility to use compensation measures in Finnish permitting legislation has been increasingly discussed in legal literature and research reports, particularly after the Weser judgment (Leino & Belinskij 2018, Suvantola et al. 2018, Soininen et al. 2019). A study from 2018 suggests that the programme of measures of the river basin management plan would offer an existing planning system to consider compensation measures. The programme of measures is developed for a river basin district or its part, and the need and effect of compensation measures could be assessed for the whole area concerned. This type of approach could create the prerequisites for approving projects with the help of compensation measures undertaken outside the project area (*ex situ*) but affecting the same water body (Leino & Belinskij 2018). River management planning would also allow the 6-year river basin management cycle for the compensation measures of the environmental impacts of new projects (Suvantola et al. 2018).

Wide and systematic use of compensation in connection with environmental permitting would also require amendments to the relevant acts (Leino & Belinskij 2018, Suvantola et al. 2018). Compensation measures and their use in relation to the WFD environmental objectives should be defined in the Environmental Protection Act and Water Act (Leino & Belinskij 2018).

4.6 U.S. example

4.6.1 Permit system

The federal Clean Water Act[12] establishes a basic structure for water quality standards and water pollution control in the USA. Its objective is to restore and maintain the chemical, physical and biological integrity of the Nation's waters (Sec. 101(a)). The Act requires a permit to discharge any pollutant from a point source into surface waters. Industrial, municipal and other facilities must obtain permits if their discharges go directly to surface waters.

The Act delegates permitting responsibility to the states. The states must also adopt water quality standards for lakes, streams and estuaries. These standards are expressed as maximum allowable concentrations of pollutants to assure that they do not impair the designated uses of the waters (Sec. 303(c)). Water quality standards must also include a so-called anti-degradation policy (Sec. 303(d)), which sets a level of water quality protection to prevent degradation. Yet, the Clean Water Act has been criticized for insufficient focus on anti-degradation. The lack of a

12 The Federal Water Pollution Control Act 33 U.S.C. §1251 et seq. (1972) as amended through P.L. 107–303, November 27, 2002.

common definition of degradation and inconsistent design and enforcement across the states lead to ambiguities and irregularities in the case of anti-degradation (Glicksman & Zellmer 2013).[13]

When a water body does not meet its water quality standard, a state shall list the water as impaired or in danger of becoming impaired (Sec. 303(d)(1)(A)-(C)). For such waterbodies, states calculate and allocate Total Maximum Daily Loads (TMDLs), which establish the maximum amount, i.e. quantity of a pollutant allowed in a water body. The TMDL serves as a planning tool for meeting the approved water quality standards. In the TMDL, the state allocates the daily load of nutrients among the various point sources and non-point sources within that area. Point sources receive a waste load allocation (WLA) and unregulated non-point sources a load allocation (LA).[14]

Permits for point sources are issued through the federal Environmental Protection Agency's (EPA) national pollutant permit programme.[15] These are, for example, wastewater treatment plants, certain storm water discharges and animal feeding operations. WLAs are implemented through this permit system, and the point sources are controlled by means of effluent limits in permits for such point sources (Sec. 402). The effluent limits must be "consistent with the assumptions and requirements" of WLAs in the TMDLs.[16] The revision of a permit's effluent limitation based on a TMLD must also be consistent with an anti-degradation policy established under the Clean Water Act by the state in question (Sec. 303(d)).

The EPA, which also administers implementation of the Clean Water Act, is obliged to develop programmes for preventing, reducing or eliminating the pollution of surface waters (Sec. 102(a)). The EPA supports, for example, the use of water quality trading and offsets of nutrients to meet the requirements of the Clean Water Act, such as the TMDLs for phosphorus and nitrogen.[17] Market-based mechanisms such as nutrient trading and offsets allow a permit holder to comply with an effluent limitation in a pollutant permit.

The Chesapeake Bay nutrient trading scheme

In December 2010, the states of Delaware, Maryland, New York, Pennsylvania, Virginia, West Virginia and the District of Columbia entered an agreement with the federal EPA to establish a TMDL for the Chesapeake Bay under Section 303(d) of the Clean Water Act. The TMDL identifies the necessary pollution reductions across the jurisdictions. Essentially, it is a comprehensive "pollution diet" with limits for nitrogen, phosphorus and sediment to meet the water quality standards in the Bay. The seven jurisdictions are allocated a share of the total yearly limit.[18] The TMDL is designed

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13 The EPA regulations require states to adopt anti-degradation policies protecting water quality to 1) maintain existing uses, 2) support recreation and propagation of fish and wildlife unless a lower water quality is necessary to accommodate important economic and social development and 3) maintain water resources of exceptional recreational and ecological significance.
14 Agricultural runoffs are exempted from permitting according to Sec. 502(4) of the Clean Water Act.
15 The Environmental Protection Agency, Overview of Identifying and Restoring Impaired Waters under Section 303(d) of the Clean Water Act: https://www.epa.gov/tmdl/overview-identifying-and-restoring-impaired-waters-under-section-303d-cwa (accessed 8 November 2019).
16 The Environmental Protection Agency, Overview of Total Maximum Daily Loads (TMDLs): https://www.epa.gov/tmdl/overview-total-maximum-daily-loads-tmdls (accessed 8 November 2019).
17 The Environmental Protection Agency, Collaborative Approaches to Reducing Excess Nutrients: https://www.epa.gov/nutrient-policy-data/collaborative-approaches-reducing-excess-nutrients#creating (accessed 8 November 2019)
18 The total yearly limit is 185.9 million pounds of nitrogen, 12.5 million pounds of phosphorus and 6.45 billion pounds of sediment per year, which means a 25 percent reduction in nitrogen, 24 percent reduction in phosphorus and 20 percent reduction in sediment.

to ensure that all pollution control measures required to fully restore the Chesapeake Bay are in place by 2025, with at least 60 percent of the necessary actions completed by 2017.[19]

As a policy instrument, the TMDL is primarily an "informational tool", which requires implementation by federally regulated point sources (pollution permits), state or local plans for point and non-point source pollutant reduction, to meet the water quality standards.[20] The TMDL's implementation plans detail how and when the seven jurisdictions will meet the pollution allocations.

When the TMDL does not account for new or increased loadings of nutrients, a jurisdiction may accommodate such loadings only through offsets necessary to meet the TMDL and applicable water quality standards in the Chesapeake Bay. The offsets must be additional to reductions already needed to meet the allocations in the TMDL.

The Nutrient Credit Exchange Program in Virginia

Ahead of the 2010 Chesapeake Bay TMDL, the Chesapeake Bay Watershed Nutrient Credit Exchange Program authorized nutrient trading in Virginia's portion of the Bay in 2005.[21] As a main rule, the legislation allows a regulated point source permit (e.g. a wastewater treatment plant) to either purchase credits[22] or upgrade technology processes on-site to comply with permit regulations.

However, existing and new or expanding point sources are treated differently. If an existing point source's discharge exceeds its WLA (allocated under the TMDL), it must seek credits from another point source within the same river basin. Only if no such credits are available, a point source may pay a per pound fee to an offset fund administered by the state. Existing point sources must exhaust all available nutrient point source credits before turning to the fund, even if the fund is cheaper. New or expanding point sources must offset all its new nutrient loads through the following sequential offsetting hierarchy. First, such permit holders must purchase WLAs or credits from an existing point source or, second, fund measures reducing nutrients[23] from non-point sources. Third, a permit holder must fund nutrient reductions by other means approved by Virginia's competent authority. The fourth option is to purchase credits from the offset fund.

Legislators anticipated that this type of regulatory context, with uncontrolled non-point sources (typically farmers) and regulated point sources (permit holders), would spur the demand for farmers' non-point source credits. Permit holders were expected to pay for such credits if prices were lower than reducing nutrients through on-site technology.

Yet, the expectation did not materialize. A weak demand for non-point source credits may be explained by a complex permitting structure and multiple regulatory requirements (Stephenson & Shabman 2017). Another issue is the so-called "severance costs", i.e. costs associated with defining, enforcing and transacting a commodity, the nutrient credit. The complexity involved in calculating reductions to create a credit, in combination with the regulatory complexity of pollution control laws, creates high severance costs for nutrient credits. This may negatively impact institutional credibility and trust (Pappas & Flatt 2018). Permit holders have tended to prefer on-site technology compliance instead of nutrient trading.

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19 The Chesapeake Bay Total Maximum Daily Load for Nitrogen, Phosphorus and Sediment, Section 1, December 29, 2010.

20 Ibid., p. 48.

21 Article 4.02 of the Code of Virginia established the Chesapeake Bay Watershed Nutrient Credit Exchange Program in September 2006: https://www.deq.virginia.gov/Portals/0/DEQ/Water/PollutionDischargeElimination/GM07-2008.CB_Watershed_Facilities_Permitting-Amd-2.pdf (accessed 8 January 2020).

22 A nitrogen and phosphorus credit is defined as an annual one pound (0.454 kg) reduction.

23 These are often voluntary adoption of agricultural conservation practices called best management practices or BMPs, Virginia Department of Environmental Quality.

Photo: Visa Hietalahti

5 Ecological aspects in developing nutrient offsetting in the Northern Baltic Sea

The successful realization of nutrient offsetting to reduce human impacts requires solid and specific understanding of the ecosystem and its components and functions. The Baltic Sea is a globally unique brackish water ecosystem with its environmental gradients and unique species composition. This means that the usability of offsetting measures used outside or even within the Baltic Sea region must be carefully assessed case by case before they are taken into local operative use. Furthermore, each case is unique, so assessing environmental impacts and potential for offsetting should be project-specific and include a thorough use of the mitigation hierarchy. Relevant monitoring measures need to be developed to assess the success of planned and executed compensation measures.

5.1 Nutrient offsetting in the marine ecosystem

In order to achieve the targets set by the MSFD, WFD and the HELCOM Baltic Sea Action Plan, there may currently exist limitations in developing human activities which directly contribute to eutrophication of the Baltic Sea. Deteriorating the ecological status of water bodies is not allowed, and therefore activities resulting in an increase in the nutrient content of the seawater are restricted. Removing nutrients from the ecosystem through locally tailored nutrient offsetting measures at sea or with a combination of land- and sea-based measures can provide a possibility to develop human activities at coastal regions and offset the harmful impacts human activities cause for the coastal and marine ecosystem in the Northern Baltic Sea.

The removal of nutrients from an ecosystem must be well-planned and based on adequate measures, which, in turn, must be based on reliable data and long-term monitoring of the effectiveness of the chosen measures. Planning and executing efficient nutrient offsets requires comprehensive knowledge on the locally prevailing physical and chemical parameters as well as marine ecosystem components. This is necessary because otherwise local environmental conditions cannot be taken into account sufficiently. The data on the physical and chemical parameters of the area is needed, for example, on bathymetry, currents, water quality parameters and ecosystem components such as food web structure and its functioning groups. Furthermore, for some measures, information on potential hazards, like the presence of heavy metals in the sediment or cyanobacterial toxins in the water column, are also required. Existing knowledge can be inquired from the environmental authorities, but in most cases supplementary field inventories are needed to achieve all necessary data needed for planning a successful compensation procedure.

To reach the target level of nutrient offsetting, a high enough quantity of nutrients must be removed from the ecosystem. In practice, the impacts of the offsetting measures must be adequate both spatially and temporally. Spatial in this context means that nutrient removal must be targeted to an area that is ecologically but also through WFD legislation directly linked to the area receiving the nutrient input from the planned human activities. The temporal aspect of measures means that the nutrient offsetting must occur before or at the same time as the planned nutrient increase, otherwise there will be an interim or, if the offsetting fails, a permanent increase in nutrient load at the target area.

5.1.1 Ecological compensation in the marine ecosystem

The main goal for ecological compensation is to offset biodiversity losses caused by human activities. The possibility to develop ecological compensation in the Northern Baltic Sea as well as the potential for biodiversity offsets of natural habitats in Finland have been studied already to some extent (Kostamo et al. 2018, Raunio et al. 2018, respectively). These reports conclude that by carefully developing and implementing compensation measures it is possible to offset the ecologically harmful impacts of human activities in the coastal and marine areas of the Northern Baltic Sea. However, applying ecological compensation sustainably both in terrestrial and marine environments requires still more research and pilot studies, e.g. in developing metrics for calculating ecological losses and gains, in practical, effective and implementable ecological restoration or remediation measures, and in monitoring. Measures that reduce nutrients and are potential nutrient offsets could be included in an ecological compensation if they aid in achieving the set biodiversity targets. For example, the reduction of excess nutrients can improve water quality and water transparency, which in turn can result in successful establishment or restoration of valuable seabed habitats.

5.1.2 Trade-offs and synergies between nutrient and ecological compensations

Reducing the amount of nutrients from the marine environment may provide a possibility to also produce ecological improvements that could be considered ecological (biodiversity) offsets. The execution of these simultaneously requires an ecological offset specific approach along with nutrient offsetting. In practice this means that the measures must be assessed both through the nutrient and ecological offsetting frameworks, i.e. measures need to be assessed on their nutrient offsetting capacity but also on their ecological impacts. Furthermore, since the targeted direct effect for the marine ecosystem differs between nutrient and ecological compensation procedures, nutrient decrease versus biodiversity increase, it is very likely that integrated approaches where both compensation types can be used may prove difficult to plan, execute and monitor. Also, if the same measure is used both as nutrient and biodiversity offset, the question of additionality becomes more complicated (*additionality*, see Chapter 2.1.2 and Moilanen & Kotiaho 2018).

In some cases, measures aiming at nutrient reduction in a marine environment may have unintended, surprising and potentially negative ecological effects to the aquatic ecosystem. The general expectation is that if an aquatic habitat is deteriorated due to eutrophication, the removal of excess nutrients will have a positive ecological effect. But the eventual outcome is dependent on the chosen measure. If, for example, nutrient removal is done by fishing large quantities of selected fish species, there can be cascading effects through changes in the food web that may cause alterations in the ecosystem functioning. The unintended negative ecological effects and potential trade-offs need to be kept in mind when planning the practical measures for nutrient offsetting.

5.2 Potential offsetting measures in the coastal areas

5.2.1 Seaweed biomass removal

Theory

Seaweeds and macroalgae bind nutrients in their biomass during the growing season. Some seaweeds are perennial with slower growing rates while annual species are often opportunistic with a rapid growth rate and a short life cycle. Macroalgae inhabit hard littoral substrates and have great ecological importance because they are one of the primary producer groups in shallow coastal areas and supply oxygen to the sea. Seaweed communities also provide important habitats and food for invertebrates, fish and even some birds.

Eutrophication has resulted in the increase of opportunistic annual species with a high capacity for annual growth (Bonsdorff 1992, Gubelit et al. 2015). Furthermore, annual filamentous algal species usually have a very short life cycle resulting in detached algal biomass gathering in shores and seabed depressions. Decaying biomass in seabed depressions can result in an increased nutrient load in the water column when nutrients are released from the algal biomass or from the bottom sediment due to anoxic conditions. Decaying biomass can also prevent the recreational use of beaches.

The harvesting of wild perennial macroalgal stocks is not likely to be allowed in the Baltic Sea Region, but the cultivation and removal of beach-cast and free-floating algal mats remains a possible option (Ikonen & Hagelberg 2007). Furthermore, seaweed cultivation may provide an opportunity to both reduce nutrient input locally and simultaneously produce biomass for economic utilization.

When planning seaweed cultivation as a nutrient removal measure, the great differences in nutrient removal capacity among macroalgal species must be considered. The nutrient removal capacity varies depending on the species but also on environmental conditions (Table 1).

Table 1. The nutrient removal capacity of common filamentous macroalgal species grown in artificial substrate in the Northern Baltic Sea. The amount of removed nutrients is presented as grams (g) per kilogram of macroalgae (kg) in dry weight (DW). POP = particulate organic phosphorus, POC = particulate organic carbon, PON = particulate organic nitrogen. Source: Suutari et al. 2017.

Species	**POP** [g kg^{-1} DW]	**PON** [g kg^{-1} DW]	**POC** [g kg^{-1} DW]
Ulva spp.	2.31 ± 1.4	28.89 ± 15.7	343.32 ± 32.2
Cladophora glomerata	2.76 ± 0.8	23.83 ± 8.1	264.46 ± 56.5
Polysiphonia fibrillosa	2.22 ± 0.5	42.17 ± 11.3	322.27 ± 54.6
Ceramium tenuicorne	2.74 ± 0.7	30.35 ± 4.9	286.61 ± 48.8
Pylaiella/Ectocarpus	3.00 ± 0.7	25.38 ± 6.1	237.56 ± 38.6

Ecological impacts of biomass removal

In the Baltic Sea region, attached perennial algal biomass including species like *Fucus vesiculosus* or *Furcellaria lumbricalis* form important underwater habitats on the seafloor. Therefore, the removal of the biomass of these species from the marine ecosystem would result in strong negative ecosystem impacts. Furthermore, perennial species generally have quite a poor dispersal capacity and growth rate, so the restocking of perennial species would take several years and thus the ecosystem impacts would also last a long time. Thus, for ecological reasons, the removal of attached perennial algal biomass is not a recommended nutrient removal measure. However, there is currently some interest at least in Finland by the Origin by Ocean company in cultivating bladder wrack for commercial purposes and an ongoing effort to develop an economically sustainable business ecosystem around compounds extracted from algal biomass.

However, the cultivation and removal of annual filamentous algal biomass, including annual species like *Cladophora glomerata*, *Ulva* spp. *Pylaiella littoralis* and *Ectocarpus siliculosus*, would most likely only reduce the amount of nutrients from the seawater. Annual species have excellent dispersal capacity and rapid growth rate, so they provide an opportunity for cultivation applications. Some tests on their cultivation were performed by Suutari et al. (2017) and further investigations on the topic are currently ongoing within the Baltic Region (see, for example, Interreg GRASS project[24])

Loosegrowing *Fucus vesiculosus*. Photo: Visa Hietalahti

Potential for offsetting

Nutrient removal capacity

Seaweeds provide a possibility for nutrient offsetting due to their capability to bind nutrients from the water column.

Risks and uncertainties

Developing practical solutions for removing enough algal biomass to produce nutrient offsets might prove to be difficult. This is because in pilot studies algal cultures originating from spores dispersing in the water column have consisted of several macroalgal species but also of invertebrates. Moreover, the biomass of invertebrates growing on artificial substrata in pilot studies has exceeded algal biomass several-fold (Suutari et al. 2017), so that the actual biomass obtained from cultures has mainly consisted of invertebrates. Thus, it would be difficult to calculate beforehand the exact amount of nutrient removal that algal cultivations might result in and to estimate the overall food web effect of the mixed cultures. Furthermore, after removal the algal biomass needs to be transported and stored without nutrient leaks back into the marine ecosystem. This, along with efficient cultivation and harvesting technologies, requires further studies.

[24] More information on GRASS project, see: https://projects.interreg-baltic.eu/projects/grass-176.html

Timescale of measures

Annual removal of cultivated macroalgal biomass is required to obtain nutrient offsets.

Biomass utilization

There are several potential applications for the use of macroalgal biomass harvested from the sea. The use as a fertilizer is one potential option (Gren et al. 2009, Lill et al. 2012, Alobwede et al. 2019), but macroalgae can also be used as biosorbents in metal removal (Carrilho & Gilbert 2000, Lill et al. 2012). Macroalgae are potential material for the production of feed and food (Prou & Goulletquer 2002, Jönsson & Holm 2010, Rebours et al. 2014, Bikker et al. 2016) and as a source for health-promoting products (Grienke et al. 2014, Parjikolaei et al. 2016). Seaweeds can also be used as raw material for biofuels, and many species are currently utilized for chemicals and cosmetics (Fitton et al. 2015, Bikker et al. 2016). However, instead of actual tests or applications, evaluations on the use of biomass and algae collected are mostly based on their chemical composition (Petersen et al. 2014, McCauley 2016, Biancarosa et al. 2018) although some pilot tests have been run especially on biofuel research (Barbot et al. 2016).

There are several factors that can restrict the economic use of macroalgae in the Baltic Sea region. Some macroalgal species, especially the green alga *Cladophora glomerata*, accumulate heavy metals, which may restrict the use of algal biomass in some applications (Chmielewská & Medved 2001, Akcali & Kucuksezgin 2011). Furthermore, the quantity and quality of macroalgal biomass varies during the growing season due to environmental factors: each annual species has its own specific life cycle which is influenced by annually varying environmental conditions, resulting in several generations of algae at the same site (Kiirikki & Lehvo 1997, Kraufvelin et al. 2007). Thus, the variability of biomass among years can be strong due to environmental variability. Obtaining biomass from only one target species is not likely in natural conditions, where the surface of all available growing substrata is covered by a variable community consisting of algae and invertebrates (Suutari et al. 2017) and where the structure of the community is governed by both environmental factors (e.g. salinity, waves, depth) and biological interactions (e.g. competition, grazing). Thus, producing economically valuable macroalgal biomass requires the development of cultivation and harvesting technologies as well as biomass applications.

5.2.2 Mussel biomass cultivation and biomass removal from the sea

Theory

Blue mussel (*Mytilus trossulus*) forms extensive mussel reefs on rocky seabed (> 4 salinity in PSU, practical salinity units), filtrating the surrounding water for phytoplankton and other organic particles. Mussels harvest the nutrients through their food intake, which results in increased water transparency and improved coastal water quality (Edebo et al. 2000, Makarewicz et al. 2000, Idrisi et al. 2001, Newell 2004, Lindahl et al. 2005, Lindahl 2011). Cultivating blue mussels and removing the biomass could provide a measure to remove nutrients from the Baltic Sea because farmed mussels, in contrast to most farmed fish, do not require addition of feed, and therefore the nutrients that are incorporated in the mussel meat and shell can be considered as a net nutrient removal from the ecosystem when harvested. The first full-scale trial of blue mussel farming as a nutrient abatement method in Sweden was an attempt to extract nutrients from the sea as a cost-effective alternative to improve the local sewage treatment plant in Lysekil, Sweden (Lindahl 2008). Even though this particular attempt did not succeed, further trials to farm blue mussel with the primary purpose to extract nutrients from the sea have been conducted since then, mainly in

Sweden and Denmark (Gren et al. 2009, Lindahl & Kollberg 2009, Petersen et al. 2014; reviewed by Minnhagen 2017).

Nutrient removal capacity

The blue mussels in the Baltic Proper differ in morphology and physiology from the blue mussels in the Atlantic (e.g. in Skagerrak and Kattegat). The low salinity affects the growth rate, maximum size, byssus production, shell formation and meat/shell ratio of mussels (Kautsky et al. 1990, Riisgård et al. 2014, Maar et al. 2015). Blue mussels living in low salinity generally have lower uptake rate and a higher excretion rate of nitrogen than blue mussels in high salinity (Livingstone et al. 1979, Tedengren & Kautsky 1987). When the salinity is low, the mussels need to allocate a lot of energy for osmoregulation, which means that they have less energy available for growth compared to mussels in high salinity. As a result, reported discrepancies exist between the expected mussel biomass yield and the total harvest in field conditions (Table 2, Hedberg et al. 2018). The estimation of nutrient removal potential of mussel farms should therefore be based on data on the actual nutrient removal capacity of mussel occurring in the region in question, not on reported estimates from other parts of the Baltic Sea.

Table 2. Mussel farms in the Baltic Sea, Kattegat and Skagerrak. Data on reported total harvest, growth time, area occupied (or measured from Google Earth Map) was compiled from both commercial and mitigation farms, and the harvest recalculated to tonnes of wet weight per hectare and year (tonne ww / ha y). Farm size in hectares (ha), farming cycle (months), total harvest in tonnes (tonne) and projected and actual annual yields per hectare (tonne / ha y), when not reported = no data (n.d.). Source: Hedberg et al. 2018.

	Salinity [PSU]	Location	Farm size [ha]	Farming cycle [months]	Total harvest [tonne ww]	Projected yield [tonne ww/ha y]	Actual yield [tonne ww/ha y]
Baltic Proper	6.5	Kumlinge	0.45	30	14.4	20	12.6
Baltic Proper	6.5	Hållsviken	n.d.	n.d.	6	50	n.d.
Baltic Proper	7	St Anna	4	19	15	4.5	12
Baltic Proper	7	Byxelkrok	1	n.d.	n.d.	25	n.d.
Baltic Proper	7.3	Hagby	3.5	24	n.d.	50–90	7–40
Öresund	12	Malmö	6	21	9	15–80	0.85
Baltic Proper	14	Kiel	0.6	21	35	30-50	30
Belt Sea	16	Musholm	1	7	14.6	200	25
Kattegatt	25	Skive fjord	18.8	12	1100		60–90
Skagerrak	30	Tjörn	2	12	200		50
Skagerrak	30	Tjärnö	0.46	22	160		190
Skagerrak	30	Mollösund	13	12–18	1500–2000		75-150

Developing a mussel culture consisting mainly of blue mussels seems quite difficult. In field studies, the invertebrate community forming on artificial growing substrata has consisted of a mixed community of blue mussels, bay barnacles (*Amphibalanus improvisus*) and the hydroids *Cordylophora caspia* and *Gonothyraea loveni* (Suutari et al. 2017). The dry weight and the nutrient content of the community varied between the two sampled years in the Archipelago Sea and in the Gulf of Finland (Table 3), being on average 172.12 g/kg particulate organic carbon (POC), 18.0 g/kg particulate organic nitrogen (PON) and 2.29 g/kg particulate organic phosphorus (POP) in the entire invertebrate fraction. Furthermore, even though the macroalgal colonization of culturing substrata was relatively slow, barnacles and blue mussels colonized all available growing substrata rapidly, producing a total biomass of more than 1000 g/m^2 within five months of the experiment and close to 2400 g/m^2 after 14.5 months of incubation in 2012 (Suutari et al. 2017). Since the invertebrate biomass does not annually detach from the growing substrata, it could be harvested at an optimum level of nutrient uptake and practically at any time of the year.

Table 3. The nutrient content of the invertebrate community in the Archipelago Sea (Rymättylä) and in the Gulf of Finland (Tvärminne) in 2011 and 2012. Particulate organic carbon (POC), particulate organic nitrogen (PON) and particulate organic phosphorus (POP) grams per kilogram of invertebrates (g/kg). Source: Suutari et al. 2017.

Location	**2011**	**2012**	
Rymättylä	153.74 ±24.9	187.00 ± 24.9	POC g/kg
	13.88 ± 7.8	23.03 ± 6.7	PON g/kg
	2.00 ±0 .7	2.47 ± 1.5	POP g/kg
Tvärminne	176.6 ± 15.3	170.83 ± 10.6	POC g/kg
	18.97 ± 5.3	16.13 ± 2.5	PON g/kg
	2.98 ± 0.9	1.72 ± 0.6	POP g/kg

Ecological impacts

Mussel farms not only remove nutrients through harvesting the mussels that filtrate phytoplankton and organic particles from the seawater but also have positive effects on the environment by reducing phytoplankton and particle concentrations, and thereby they improve water transparency (EUCC 2019). This can in turn improve the conditions for macrophytes, resulting in improvements in the condition and distribution of seabed habitats. However, in the western Baltic Sea (salinity 14.3 in PSU), the effect of a blue mussel farm on water clarity in the Kiel Fjord was investigated by Schröder et al. (2014). This relatively small farm (harvesting 30 tonnes in wet weight of mussels per year) improved the Secchi depth by only 30 cm within the farm area and by 5 cm in a 10 km^2 area around the farm. Intensification of mussel farming in order to improve water clarity may also lead to food limitation for the mussels in the farm (cf. Rosland et al. 2011) and since food-limited mussels grow less, this may ultimately lead to smaller harvests and less nutrients removed from the ecosystem than expected. It should be remembered that as blue mussels are efficient grazers, their capacity to filter food to limiting levels may also mean food limitation for, e.g. zooplankton and other pelagic organisms like the meroplanktonic larvae of benthic organisms and, consequently, zooplankton predators like pelagic fish and their juveniles.

Mussel cultivation can also have harmful impacts on the marine environment. One of the major objections against using mussel cultures as a nutrient offsetting measure is their impact on entire biochemical cycles of nutrients (Stadmark & Conley 2011, Hedberg et al. 2018). Over the entire farming cycle, only about 25 % (5–45 %) of the nutrients that are contained in the plankton and organic matter consumed by the mussels are removed at harvest (Folke & Kautsky 1989, Cranford et al. 2007, Brigolin et al. 2009, Jansen et al. 2012). The remaining nutrients are deposited as feces and pseudofeces to the seabed below the farm or excreted as dissolved nutrients to the water. A considerable part of the nutrients is released as eggs and sperm during spawning in spring (Hedberg et al. 2018). According to Hedberg et al. (2018) farmed mussels excrete both dissolved nutrients at the farm site and nutrients bound in feces and pseudofeces that can generate anoxic bottoms, which in turn can lead to the leakage of ammonium and phosphate from the sediment. They also point out, that increased sedimentation may impact denitrification and nutrient burial rates, resulting in enforced or weakened nutrient mitigation effects depending on local conditions. This impact is known to depend on mussel density and prevailing local environmental conditions such as hydrography, water exchange rate, sediment type and the eutrophication level of the area in question.

Furthermore, 3D ecosystem modelling in recent studies has demonstrated that mussel growth and filtration of microplankton within a cultivation experiment may deplete Chlorophyll a from the water column (EUCC 2019). The depletion varies over time due to changes in the water current direction and speed, and the highest depletion is observed in the downstream current direction away from the farm. Although the depletion of microplankton will lead to increased seawater transparency, the ecosystem effects of the depletion may be profound, since the removal of one trophic level, especially in primary production, may alter the whole trophic system. Furthermore, increased plankton turnover and a change in species composition may possibly lead to unwanted plankton blooms (Cranford et al. 2007, 2008, Guyondet et al. 2015).

Potential for offsetting

The nutrient removal capacity of mussel farms depends on the size of the harvest, the time period between harvests and the nutrient content in the mussels at the harvesting time. As in all measures based on biomass removal, the principle of using mussel cultures in nutrient offsetting should be based on a mass balance perspective. This means that one tonne of excess nitrogen in the marine environment will be compensated when one tonne of nitrogen stored in the harvested mussels is brought back to land (EUCC 2019). The mass balance principle was adopted in 2017 into Danish legislation as a mechanism to offset for new sources of nutrients. In particular, the new law demands that any further expansion of fish farming in coastal and offshore waters and the nutrient emissions it produces must be compensated for by marine measures, i.e. mussel cultivation. The mussel culture is not considered a physical filter, removing precisely the nutrient molecules released from the fish farm, but as a means to 'balance the nutrient budget'. During the implementation of the legislation, several aspects of the potential of using mussels as an offsetting measure for expansion of fish production were debated in the Danish Parliament. The four main points of discussion were:

1) if impact mitigation by mussel cultivation in offshore areas is effective enough to compensate for the excess nutrients without covering large areas,
2) if there are enough natural mussel beds in the offshore areas to provide sufficient recruitment of mussel larvae for the mitigation mussel cultures,
3) risk of loss of mussels due to predation by, e.g. eider ducks and
4) the effects caused by the increased sedimentation (mussel feces) on the seabed underneath the mitigation mussel culture.

Gren et al. (2009) assessed the economic potential for using mussel farming as a nutrient reduction measure in the brackish water Baltic Sea and concluded that it could be a cost-efficient measure. However, the assessment did not fully consider the slow growth and lower nutrient content of blue mussels at low salinities, nor did it internalize any potential environmental costs (Hedberg et al. 2018). So far, none of the blue mussel farm trials in salinities between 6 and 12 PSU in the Baltic Sea have met the expectations set up in Gren et al. (2009). The yields have generally been much lower than expected, which has been explained by low growth, severe ice winters, storms, unexpected technical problems, fouling by epiphytes or eider predation (Minnhagen 2017). Several of the experienced problems relate to the special environmental conditions in the Baltic Sea, and recent publications conclude that the low salinity in the Baltic Proper is a major limitation for blue mussel farming (Stadmark & Conley 2011, 2012, Rose et al. 2012, Maar et al. 2015). Also, in many northern areas of the Baltic Sea, phytoplankton productivity levels or species composition may be suboptimal to mussel growth.

Harvesting farmed blue mussels will remove nutrients from a water body, just as fishing or any other harvesting of biomass (Hedberg et al. 2018). However, in the brackish Northern Baltic Sea, the low salinity restricts the production of blue mussel biomass, and it is technically more challenging to farm mussels. This lowers the efficiency and increases the costs of using mussel farming in nutrient offsetting, which the current economic estimates have not fully considered. Finally, the potential negative environmental effects of large-scale-farming must be taken into account in the economic calculations before any large-scale operations are realized to avoid major environmental impacts on the marine ecosystem.

Biomass utilization

Blue mussels are potential material for production of feed and food (Prou & Goulletquer 2002, Jönsson & Holm 2010, Rebours et al. 2014, Bikker et al. 2016). However, the slow growth rate of mussels in the Baltic Sea results in only small-sized mussels, which most likely will not become a major food product within the region. The other potential uses, including, for example, animal feed and biogas production, have not been studied extensively in the region. The lack of profitable circular economy applications for mussel biomass is probably one of the main reasons why mussel farming for nutrient recycling has not yet proceeded beyond the pilot stage.

Blue mussel (*Mytilus trossulus*) forms extensive mussel reefs on rocky seabed. Photo: Visa Hietalahti.

5.2.3 Common reed biomass removal from the sea

Theory

Common reed (*Phragmites australis*) is a perennial grass that forms dense monocultures along the shallow freshwater and brackish coastlines of the Baltic Sea. It is one of the key species for wetlands affecting coastal habitats through its efficient dispersal capability. Although common reed provides important habitats for fish and birds, native plant species can be outcompeted by rapidly expanding reed beds.

There exists a strong link between aquatic macrophytes and eutrophication (Maristo 1941, Toivonen & Huttunen 1995), which is obvious in water bodies that are naturally eutrophic, resulting in a strong positive correlation between nutrient levels and the biomass of aquatic macrophytes at the land-water interface. The nitrogen content of reed stems is also higher in the shore areas of eutrophic lakes probably because reed retains nutrients coming from the catchment area (Kvét 1973, Sandström 2007). Therefore, the removal of reed biomass might provide a possibility to remove nutrients from the coastal ecosystem.

Table 4. The aboveground biomass of common reed (*Phragmites australis*) in different parts of the Baltic Sea region. Average yield in dry matter in tonnes per hectare (t/ha), source: Ikonen & Hagelberg 2007.

Country/Region	Average yield in dry matter (t/ha)
Estonia	7.4–9.1
Curonian Lagoon (Lithuania)	5–40
Niedermoor (Germany)	12–20
Hirvensalo (Finland)	5–12

Nutrient removal capacity

The aboveground biomass of common reed varies according to the environmental conditions prevailing locally (Table 4). Biomass studies in Finland provide estimates that the aboveground dry biomass of reed varies in the Hirvensalo study area from 4 to 12, on average 6–7, tonnes per hectare (t/ha) and in reed beds along Halikonlahti Bay from 3 to 12, on average 5–6, t/ha (Ikonen & Hagelberg 2007). The biomass of reed could be calculated based on existing high-resolution Earth Observation data on the distribution of reed in coastal areas and locally performed biomass measurements (Kauppi et al. 1983).

Nutrient accumulation in the aboveground part of the reed plants peaks in July or August (Björndahl & Egnéus 1980, Hansson & Fredriksson 2004, Komulainen et al. 2008) and is later reversed when nutrients are returned to the roots for storage until the next growing season. Only 10–20 % of the nutrients present in August remain in the dead aboveground shoots over winter (Granéli 1990, Hedelin 2001). Therefore, the extraction efficiency of nutrients by the harvesting of reed biomass is highly dependent on the cutting time (Table 5). The dead aboveground biomass is cut mostly in the winter when the ground is frozen. The winter harvesting of a reed bed in south Sweden with a standing crop of 7.4 t/ha and annual harvestable crop of 5 t/ha/y extracted 20 kg/ha/y of nitrogen, 1 kg/ha/y of phosphorus and 8 kg/ha/y of potassium. Summer harvesting of the same reed bed yielded 10 t/ha/y of biomass and removed 92 kg/ha/y of nitrogen, 9 kg/ha/y of phosphorus and 66 kg/ha/y of potassium (Granéli 1990).

Table 5. An overview of nutrient extraction in aboveground biomass from the summer and winter harvesting of natural wetlands. n.d. = no data. Table published in Köbbing et al. 2013.

Table 3. Overview of nutrient extraction in above-ground biomass by summer and winter harvesting of natural wetlands. n.d. = no data.

Country	Sweden[1]		China[2]		Finland[3]		Estonia[4]		Seasonal ranges
Habitat	Coastal and littoral stands		Eutrophic lake		Coast of the Baltic sea		Lake reed		
Harvest time	Feb	Aug	Nov–Jan	Sep	Mar	Jul	Winter	Summer	*Winter* Summer
Yield of dry mass (t ha^{-1} y^{-1})	5	10	10	20	4.5	5	8.1	7.4	*4.5–10* 5–20
N (%)	0.2	0.24	n.d.	n.d.	0.33	1.0	0.3	1.0	*0.2–0.33* 0.24–1.0
P (%)	0.02	0.08	0.028	0.09	n.d.	0.095	n.d.	n.d.	*0.02–0.028* 0.08–0.095
K (%)	0.16	0.5	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	*0.16* 0.5

[1]Granéli (1990), [2]Hedelin (2001), [3]Komulainen et al. (2008), [4]Kask (2011).

Ecological impacts of biomass removal

Aquatic macrophytes stabilize the seabed sediments and reduce the impacts of water movements on the sediment (Vermaat et al. 1990). They also reduce the growth potential of algae (Phillips 2006). Aquatic macrophytes can also have an indirect effect on the nutrient levels of a water column because they improve the habitat of predatory fish, which, in turn, can have a significant impact on the biomass of cyprinid fish that enhance the eutrophication (e.g. bream and roach) (Jeppesen & Sammalkorpi 2002). Aquatic vegetation also suppresses the impacts of planktivorous fish on zooplankton by providing the zooplankton increased shelter. Micro-crustaceans living in aquatic vegetation can reduce the biomass of phytoplankton remarkably (Vakkilainen 2005). Aquatic macrophytes can also shade the phytoplankton and thereby reduce its growth and biomass (Jeppesen & Sammalkorpi 2002, p. 298). Aquatic macrophytes that have both roots and aerial shoots, such as reed, can keep the phosphorus locked in the sediment. The release of oxygen from the roots can lead to the oxygenation of iron, and this in turn leads to the retention of phosphates. Aquatic macrophytes can also increase the phosphorus levels in the water because of the decomposing plant mass followed by sedimentation and an increase in organic matter (Moss et al. 1986).

Early summer cutting of the reed increases the leakage of nutrients into the environment. This leakage stops later during the growing season (Güsewell 2003, Kojo 2006). Continued intensive cutting in June over several years caused a decline of reed stands because the nutrient storages in the root system were exhausted. Cutting at the end of August will not impact the growth of reed stands because enough nutrients have already been stored in the root system for the next growing season (Weisner & Granéli 1989). The effects of the cutting on the water quality are also dependent on the size of the cut area, the total surface area of the reed bed and its location. The removal of reed beds from the shore areas of the islands closest to the mainland must be performed with caution because reed has a positive impact on the retention of nutrients in coastal areas. However, reed areas further from mainland are most likely not as important for controlling the nutrient levels in water (Lindholm et al. 1989) although the role of reed in the nutrient dynamics of the brackish water ecosystem is not yet totally understood. In the littoral zone of oligotrophic water bodies, the role of narrow reed belts, especially in rapidly deepening shores, in retaining nutrients in the sediment is not significant (Nurminen 2003).

Removing the reed during the wintertime increases the above-surface biomass and the density of the shoots of reed during the next growing season. However, winter cutting increases the density of shoots, not their speed of growth the next summer (Haslam 1971). In narrow reed beds ice often cuts the reeds, but this has no impact on the following season's growth. The positive impact of winter cutting on the fitness of reed beds was obviously the result of a decrease in the number of hibernating insects and an increase in the amount of light available to the emerging shoots. The reed stems which were cut from the ice often have a dense population of wintering insects. If the reed, is cut these insects will not graze the reed and the other aquatic macrophytes in the next growing season (Granéli 1989). On the other hand, this weakens the diversity of invertebrate populations and will also have a negative impact on bird populations in a reed bed (Ditlhogo et al. 1992). It seems that winter cutting has a positive impact on the vitality of the reed bed, which in turn may improve the reed bed's ability to retain the nutrient load from the catchment area.

Potential for offsetting

Nutrient removal capacity

The harvesting of reed for use in agriculture is reasonable mainly in situations when the reed growth is a problem or when there is a need to decrease the concentrations of nutrients in the water (Hansson & Fredriksson 2004). Without these positive effects included in the calculations, alternatives of biomass utilization, such as biogas production, utilizing fresh reed as a fertilizer on field or composting, become quite expensive. It is therefore important to consider both the nutrient offsetting capacity of the biomass removal and the utilization of biomass when calculating the costs and benefits of the measure. Reed differs from macroalgae in its nutrient uptake system, which is not based on direct uptake from the surrounding water. Thus, the removed nutrients mainly originate from the accumulated sediment storages. To compare this to the direct nutrient load to the water mass is a difficult exercise, and further information on the efficiency as a compensation tool is required.

Risks and uncertainties

The plant mass that is removed should be removed from the shoreline immediately after it has been cut to prevent nutrient leakage back to the marine ecosystem.

Timescale for measures

Common reed biomass needs to be removed annually to achieve offsetting goals.

Biomass utilization

Common reed has been utilized in the Baltic region for centuries for construction, fodder for the cattle, biofuel, raw material in cellulose production and feed for the people (Ikonen & Hagelberg 2007). More recently, the possibilities to utilize reed as a resource for energy production including biogas, bioethanol and in heating plants has been studied (Allirand & Gosse 1995). There is also potential for the commercial use of common reed as a substrate for growing plants where

Common reed (*Phragmites australis*) forms dense stands along the brackish coastline and bays of the Northern Baltic Sea. The utilization of reed biomass has been studied in Finland and other Baltic Sea countries, for example, for biogas production. Photo: Essi Keskinen, Metsähallitus.

reed could substitute peat in soil products (for an example on the commercial use of common reed, see Kiteen mato ja multa[25]).

The reed on the edges of the bed is usually thicker and more crooked and can be used for energy purposes. The edges are usually richer in other vegetation because they are brighter, warmer and drier. As a result, reed material harvested from the edges requires more labour for separating it from other plants shoots. Inside of a reed bed, wind intensity declines, wetness increases and reed tends to be more dense, thin and straight and, hence, might be better suited for thatching (Ikonen & Hagelberg 2007).

Several strategies are possible for the handling of reed if the aim is to use it as a nutrient source in agriculture (Hansson & Fredriksson 2004). The less complicated strategy is to chop the harvested material and spread it directly on farmland without further treatment. The alternative of chopping and spreading the reed directly as green manure does not require large investments or complicated processing plants, but produces no useful energy, and the risk of nitrogen leakage is higher than for the biogas alternative. This strategy is, however, not optimal with respect to the availability of the nutrients to the crop (Steineck et al. 2000).

A more complicated alternative is to compost the biomass and spread the processed material on the farmland. The compost alternative has the least favourable characteristics among the three

25 Short explanation on how the Kiteen mato ja multa company uses common reed in their products (in Finnish only): https://www.biotalous.fi/kiteen-mato-ja-multa-hyodyntaa-ruovikkojen-ravinteet-ja-suopeltojen-ruokohelvet/ and the company website: https://www.matojamulta.com/

strategies of biomass use. The operations at the compost plant are costly, and no useful energy is produced. A third possible strategy is to use the harvested biomass as raw material for biogas production and use the by-product (sludge) as an organic fertilizer. Harvesting reed for biogas production produces both large amounts of energy but also nutrients in a form that is easily available for agricultural plants. The energy balance of the biogas alternative is very favourable, whereas the economics of the system are sensitive to changes in income from the gas produced and in the costs of the chopping operation. The three strategies have different characteristics and in order to determine if any of them offers advantages for farmers and for the environment, the system needs to be comprehensively evaluated in terms of the environment and the economy (Hansson & Fredriksson 2004).

5.2.4 Fish biomass removal from the sea

Theory

Fish populations in the Baltic Sea have undergone major fluctuations over the past five centuries (MacKenzie et al. 2002). During the 20th century, while the Baltic underwent eutrophication, the biomass and landings of three fish species, cod (*Gadus morhua*), herring (*Clupea harengus*) and sprat (*Sprattus sprattus*), all increased. By the end of the 20th century, the drastic decrease of major predatory fish cod caused profound impacts on the pelagic food webs, cascading down to all ecosystem levels (Casini et al. 2008). Furthermore, there have also been changes in coastal fish communities, where the biomasses of cyprinids and three-spined stickleback (*Gasterosteus aculeatus*) have increased due to eutrophication (Olsson et al. 2019).

It has been estimated that fishery concentrating on herring and sprat annually removes 1.4 % and 7 % of the total nitrogen and phosphorus load to the Baltic Sea (Hjerne & Hansson 2002). Moreover, compared with the anthropogenic load of nutrients that reaches the open sea, the fishery removes 2.4 % and 18 % of the nitrogen and phosphorus. Therefore, the summer increase of fish biomass can explain up to one third of the summer decrease in ''total phosphorus'' in the upper 40 metres of the water column. This suggests that fish may store a substantial amount of phosphorus, which is not then available for primary producers (particularly cyanobacteria) (Bartell & Kitchell 1978, Kraft 1992, Hjerne & Hansson 2002). Fish and fishery can thus substantially influence nutrient dynamics in marine systems. Therefore, removing fish is also a potential nutrient offsetting measure.

Three-spined stickleback is one of the most abundant fish species in the Baltic Sea after herring and sprat. The stickleback has an influential position in the ecosystem as a food source for piscivorous fishes and as an important predator on grazers, such as crustacean *Idotea baltica*, inhabiting the coastal algal belts. Stickleback utilizes both shallow coastal habitats (for spawning and as nursery habitats) and offshore areas (for feeding) in the Baltic Sea and may serve as a vector linking the two habitats (Williams & Delbeek 1989, Flinkman et al. 1992, Leinikki 1995, Jurvelius et al. 1996, Eriksson et al. 2011, Bergström et al. 2015, Olsson et al. 2019). It is well documented that stickleback can have negative effects on coastal predatory fish via predation on and competition with early life stages and, in turn, cause trophic cascades in the food webs in the coastal Baltic Sea by inducing an increase in the growth of ephemeral algae (Eriksson et al. 2011, Sieben et al. 2011, Bergström et al. 2015, Byström et al. 2015, Donadi et al. 2017). Harvesting the vast open sea populations might provide a possibility to remove nutrients from the ecosystem with three-spine stickleback biomass.

Increased nutrient availability, decreased water transparency and increased biomass of plankton, filamentous algae and zoobenthos as well as an increased proportion of roach in the fish community has been observed along the coastal areas of the Northern Baltic Sea during 1975–

1994 (Bonsdorff et al. 1997). Similar development, especially the increase of cyprinid fish species, has been reported from temperate lakes undergoing eutrophication (Svärdson & Molin 1981, Persson et al. 1991) as well as from other eutrophicated coastal areas of the Baltic Sea (e.g. Hansson 1987, Lappalainen 2002). Harvesting the coastal roach (*Rutilus rutilus*) and common bream (*Abramis brama*) populations could provide not only a possibility to remove nutrients from the Baltic Sea, but also impact the structure of fish communities and potentially reduce the impacts of eutrophication by reducing fish species that actively contribute to eutrophication.

Nutrient removal capacity

In freshwaters, fish can be important in nutrient dynamics, and sometimes most of the pelagic phosphorus is stored in fish biomass (Bartell and Kitchell 1978, Schindler et al. 1993). Fish stock fluctuations can influence the availability of nutrients to phytoplankton and especially young-of-the-year (YOY) fish can be an important phosphorus sink relative to sedimentation losses (Bartell and Kitchell 1978, Kraft 1992). The removal of nutrients by fisheries' landings can be substantial (Sarvala et al. 1984). Similar effects have also been discovered for the Baltic Sea (Hjerne & Hansson 2002). Furthermore, nutrient transport between coastal and open sea areas with migrating fish, such as herring and three-spined stickleback, can be annually substantial (Deegan 1993).

Hjerne & Hansson (2002) have studied the role of fish and fisheries in Baltic Sea nutrient dynamics. They estimate that the content of N and P in herring is 2.4 % (standard deviation, SD 0.2 %) and in sprat 0.43 % (SD 0.07 %) of the wet biomass, which corresponds well with values from other teleosts (Hjerne & Hansson 2002, cybrinids, see Sterner & George 2000). No obvious differences in nutrient concentrations among species, fish size, seasons or areas were recorded by Hjerne and Hansson (2002), and, therefore, an assumption of constant N and P content has been made in calculations. In sticklebacks the N and P content have been estimated to be 2.24 % and 0.71 %, respectively, in the Archipelago Sea (LUKE).

Calculating the potential for nutrient removal is easier for the quota-fished species, like herring, sprat and cod, than for example on roach and three-spined stickleback. This is due to the fact that more precise data on fish biomass and catch exist for the commercially fished species over decades. Some examples from the Northern Baltic Sea on fish catch per unit effort based on HELCOM (2006) are presented in Table 6, although the data is lacking on three-spined stickleback. However, according to a current estimation of the number and biomass of three-spined stickleback in the Baltic Sea, there exists great annual variation in fish abundance, spatial distribution and biomass within the Baltic Sea (Olsson et al. 2019, Figure 3 & 4).

Table 6. An example of mean fish catch per unit effort, in number of individuals, in coastal fish monitoring areas in 2004 in the Northern Baltic Proper. One unit of effort is defined as one fishing night using one gillnet. No correction of differences due to length, depth or mesh-size of the gillnet has been made. Source: HELCOM 2006.

	Lagnö (SWE)	**Forsmark (SWE)**	**Finbo (AX)**	**Kumlinge (AX)**	**Brunskär (FIN)**
Roach (*Rutilus rutilus*)	9.2	8.5	28.4	0.3	8.8
Three-spined stickleback (*Gasterosteus aculeatus*)		< 0.1			
Baltic herring (*Clupea harengus*)	4.6	4.7	7.9	2.2	1.9
European sprat (*Sprattus sprattus*)	1.4	1.1	0.4	1.5	0.1

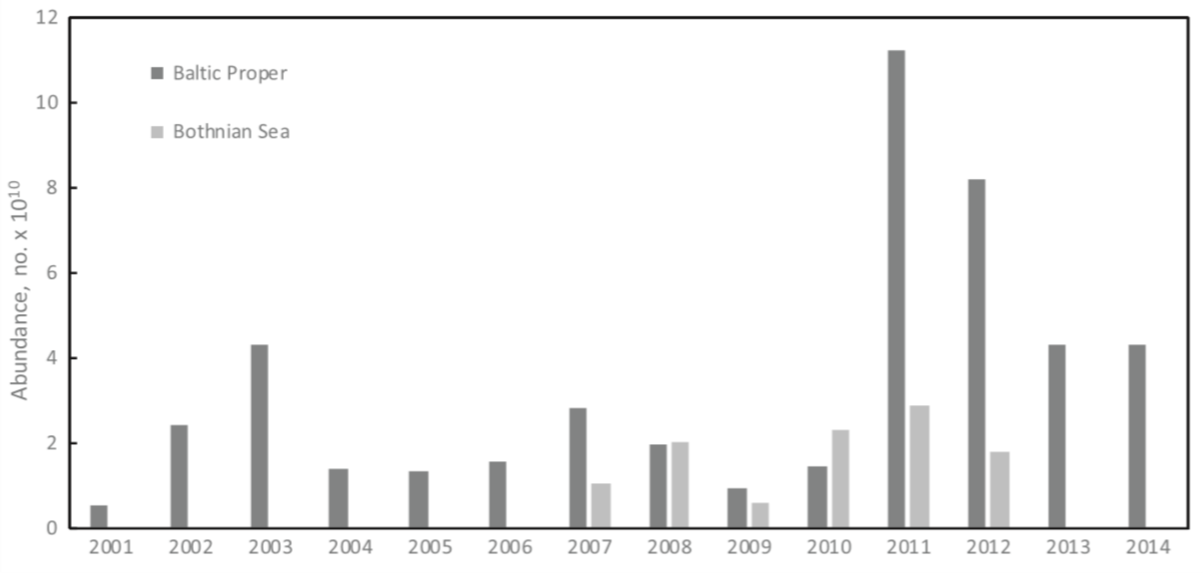

Figure 3. Total abundance of stickleback in the rectangles covered by the surveyed area in the Baltic Proper (SD 25–29) and Bothnian Sea (SD 30) during 2001–2014. The data used for the figure are derived from the "standard BIAS method." Source: Olsson et al. 2019.

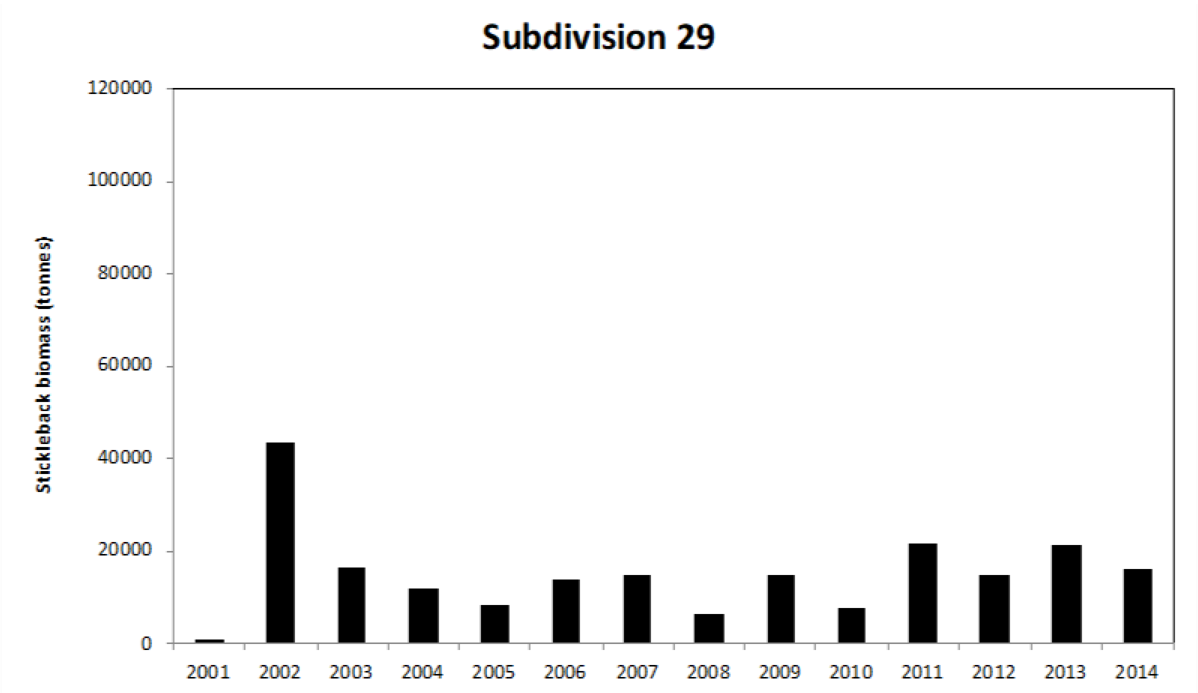

Figure 4. Total biomass (tonnes) of three-spined sticklebacks in the rectangles covered in the survey area for the ICES subdivision 29 (Åland Islands and the Archipelago Sea, Baltic Sea) in 2001–2014. Annual values represent the sum over the included ICES rectangles covered within a subdivision. Source: Olsson et al. 2019.

Ecological impacts

Before advocating intensive fishing as a measure to counteract eutrophication, it must be understood that sustainable and rich catches can only derive from large and well-managed fish populations. In the Baltic Sea today, the cod stock is seriously depleted, herring is at record low biomass, and the fishing for sprat is unusually intensive (ICES 2000). As discussed earlier in this chapter, fishing those species that are not commercially used in a large scale and thus currently under lower fishing pressure could potentially be used in nutrient offsetting. However, the precautionary principle must be used in any further plans to exploit any fish stocks. There are still uncertainties regarding the relative role of fisheries, effects of marine mammal predation, environmental variability due to eutrophication, major inflows of saline water and climate change on the long-term dynamics of key fish species.

Another important assumption is that the fishery does not, directly or indirectly, increase the negative effects of eutrophication in other ways. The dynamics of herring, sprat and cod (Jarre-Teichmann et al. 2000, Köster et al. 2001, Peltonen et al. 2004) and those of their major prey, crustacean copepods (Dippner et al. 2000, Möllmann et al. 2000), are governed by external physical forcing, biological interactions (Sparholt 1994, Köster & Möllmann 2000a, 2000b) and anthropogenic influences, particularly fishing activities and habitat alteration (Alheit & Hagen 2001). Herring, sprat and cod make up to 80 % of the total fish biomass (Elmgren 1984). The highest abundance of three-spined stickleback has been discovered in the central parts of the Baltic Proper and the Bothnian Sea (Olsson et al. 2019). The proportion of stickleback biomass in the total planktivore biomass increased from 4 % to 10 % in the Baltic Proper and averaged 6 % of the total planktivore biomass in the Bothnian Sea. In some years, however, stickleback biomass has ranged from a half to almost twice that of sprat in both basins. Other fish species, such as salmon (*Salmo salar*), flounder (*Platichthys flesus*), gobies (Gobiidae), are present, but their biomass is much lower. Understanding the complex dynamics between the fish community and the marine ecosystem is vital when planning any new measures to remove fish biomass from the sea because otherwise measures aimed at removing nutrients from the marine ecosystem can cause profound impacts on the marine ecosystem or processes within it.

Eutrophication affects the diets of both demersal and pelagic fish species in the Baltic. For example, diets of demersal fish species in the eastern Baltic (ICES Subdivisions 25–32) and the Kattegat change because hypoxia alters the infaunal species that are prey for demersal fish (Bagge et al. 1994, Pihl 1994) or causes fish to feed on pelagic prey in shallower water (Bagge et al. 1994). As a result, the intensity of the cascading top-down effect exerted by planktivorous fish on the phytoplankton abundance and community structure is of importance, which has been demonstrated in mesocosm experiments on, e.g. three-spined sticklebacks (Jakobsen et al. 2004).

Fish can influence nutrient dynamics through excretion (Schindler 1992, Schindler et al. 1993), especially benthic-feeding fish, which mobilize nutrients from sediments (Lamarra 1975, Schaus et al. 1997). Furthermore, fish can influence nutrient dynamics indirectly through their effects on zooplankton. Intensive zooplanktivory can decrease nutrient regeneration, but size-selective predation can also alter the zooplankton community structure, and increases in small species with high metabolic rate can increase nutrient regeneration (Bartell & Kitchell 1978, Carpenter et al. 1992).

During the summer, phosphorus-limited cyanobacteria are important primary producers in the Baltic Sea (Larsson et al. 2001). The occurrence and intensity of cyanobacterial blooms (including toxic species) vary considerably between years and areas, and our understanding of the controlling and factors is limited (Paerl & Millie 1996). Hjerne and Hansson (2002) suggest that variation in fish stock sizes may influence the nutrient conditions for cyanobacteria (e.g. the termination of blooms). Consequently, the fishery may, by changing stock sizes, influence the production and blooms of cyanobacteria. The overall picture is, however, complicated by the possibility that fish can also influence phytoplankton and nutrient dynamics through nutrient transport with migrating fish and by changing the behavior or structure of the gazer community, which results in changed nutrient transport and recycling.

Potential for offsetting

Nutrient removal capacity

There exists a possibility to develop nutrient offsetting measures based on removing fish biomass from the Baltic Sea. However, herring, sprat and cod are already fished according to a quota system, which is based on the carrying capacity of each species. Any additions to the fish catch would therefore have a strong impact on the already heavily fished populations. Developing measures based on other fish species, such as three-spined stickleback or cyprinids, would therefore provide a more sustainable option for removing nutrients from the Baltic Sea. However, developing large-scale fisheries on these species would require extensive environmental impact assessment on the effects of fisheries on the marine ecosystem. It should also be pointed out that fisheries are seldom a local method. Most of the fish biomass is removed from open pelagial waters. The efficiency of nutrient removal is also highly dependent on the nature of fish prey. Nutrients originating from benthic feeding utilize the vast stores, having little effect on eutrophication-enhancing nutrient load effects in water mass.

Risks and uncertainties

Further exploitation of herring, sprat or cod stocks would require careful planning so that the carrying capacity of stocks will not be exceeded. Ecosystem impacts of large-scale nutrient removal measures based on fish biomass should be assessed before the execution of measures. Furthermore, the removal of fish biomass should target the same water body where the predicted nutrient increase is assumed to occur. This might be difficult because fish move along the coastline and between coastal and open water area. In case of cyprinids the eutrophication-enhancing nutrient process is remineralization by fish. If this is increased by the changing population structure in, e.g. food-limited systems, the outcome may be more remineralization and the promotion of algal production during the remineralization-based summer period.

Biomass utilization

The use of cyprinids, especially roach and common bream, in food industry has been studied actively over the last two decades in Finland. Finding new ways for fish biomass utilization has been a part of, for example, the NutriTrade[26] and Local Fish[27] projects. New products made of fish from the Baltic Sea or Finnish lakes are now commercially produced and readily available in stores[28]. New applications for commercially unexploited fish have also been studied, e.g. fish gelatin and its processed products like biodegradable membranes and gels, which have applications in food and packaging industry as well as in encapsulation. Fish gelatin produced from roach and common bream can also be used in cosmetics and food supplements. Three-spined stickleback was used earlier as a source for fish oil to replace linseed in production of oil, which was used to protect

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26 More on NutriTrade (in Finnish): https://nutritradebaltic.eu/pilots/pilot-fish/
27 John Nurminen Foundation’s Local Fish Project: https://johnnurmisensaatio.fi/en/projects/local-fishing-project/
28 Two relatively new trademarks for local fish products are JärkiSärki (http://www.jarkisarki.fi) and Luonnonkalasäilyke (http://www.pielisenkala.fi). Another example, Pirkka Saaristolaiskala, has been developed as a part of the Local Fish project

wood products. Exhausted biomass was used for chicken and pig feed. Fish biomass was regularly used in agriculture as a fertilizer in coastal areas.

Marine biorefinery concepts utilizing fish and fish waste can be used to produce biogas through mono-digestion (Eiroa et al. 2012, Regueiro et al. 2012, Kafle et al. 2013, Wu et al. 2014, Hagman et al. 2018, Vivekanand et al. 2018, Bücker et al. 2020). Additionally, new regulations in the European Union regarding unwanted catch favour the development of biorefineries and biogas solutions. However, biogas solutions are not a common waste treatment in the fish industry since fish waste is more commonly upcycled to fodder or oils (Antelo et al. 2015). There have been several studies combining fish and especially fish waste with other substrates like whey, pig manure and biodiesel wastes. The fraction of biomass that is not utilized in food or feed production could thus be used in biogas applications to produce renewable energy.

Three-spined stickleback. Photo: Visa Hietalahti

5.2.5 Irrigation water from eutrophicated waterbodies

Theory

Coastal bays annually receive a high amount of organic loading and nutrients from the watershed area. Although a part of the loading will disperse to the outer archipelago and the open sea by water flow and currents, a part of the organic fraction and nutrients remains in the innermost bays. The degradation of annual organic loading and the release of nutrients from the anoxic seabed may result in high concentrations of nutrients in the near-bottom water. Removing nutrient-rich bottom waters from eutrophicated closed or semi-enclosed coastal bays for agricultural use may provide an opportunity to recycle nutrients especially in areas where freshwater for irrigation is in demand.

Nutrient removal capacity

The nutrient removal capacity of the measure depends on the prevailing nutrient concentration of the bottom water. This can vary during the growing season depending on, for example, the amount of freshwater inflow containing organic substances and nutrients from the watershed, the outflow and exchange of water from the bay with the outer archipelago and the open sea nearby and the release of nutrients from the seabed during anoxic conditions. Therefore, the nutrient removal capacity of the measure should be firstly studied *in situ* by measuring the changes of nutrient content and other substances in the near-bottom layer of the water column and secondly by monitoring the nutrient content in seawater used in irrigation during the growing season. These two approaches can give a precise enough estimate on the nutrient removal capacity but also on the real efficiency of nutrient removal. A pilot study on the nutrient removal capacity was performed in 2019–2020 in the Åland Islands, Northern Baltic Sea, and the preliminary results indicate that using brackish water from coastal bays in irrigation removes nutrients from the ecosystem (Table 7)

Table 7. Nutrient removal capacity from irrigation by brackish, eutrophicated water in two pilot study sites in the Åland Islands, Northern Baltic Sea. The salinity of the irrigation water and the total amount of salt acquired were also measured during the experiments. Source: SEABASED project[29].

	Kaldersfjärden					**Ämnäsviken**		
	2019				**Total**	**2019**		
	Jul 7	*Jul 9*	*Aug 5*	*Aug 25*	**Total**	*Jul 11*	*Jul 29*	**Total**
Ntot (µg/L)	1940	2100	1640	1880		1200	1360	
Ptot (µg/L)	179	157	94	148		109	139	
Salt (g/L)	2.8	3.4	3.7	4.1		4.7	4.9	
Irrigated volume (m^3)	1080	1080	1080	1080		2960	2960	
Uptake Ntot (kg)	2.1	2.27	1.77	2.03	**8.16**	3.55	4.03	**7.58**
Ntot kg/ha	0.78	0.84	0.66	0.75	**3.02**	0.48	0.54	**1.02**
Uptake Ptot (kg)	0.19	0.17	0.1	0.16	**0.62**	0.32	0.41	**0.73**
Ptot kg/ha	0.07	0.06	0.04	0.06	**0.23**	0.04	0.06	**0.10**
Uptake salt (kg)	3 024	3672	3996	4428	**15120**	13912	14504	**28416**
Salt kg/ha	1120	1360	1480	1640	**5600**	1180	1960	**3840**

[29] SEABASED project website: https://seabasedmeasures.eu/

Eutrophicated near-bottom seawater could potentially be used in the irrigation of crops. Photo: Annica Brink

Ecological impacts

The Baltic Sea

The removal of nutrient-rich near-bottom water reduces the amount of nutrients in the local ecosystem and can improve the water quality locally. This, in turn, can reduce local anoxia occurring in the seabed and improve water transparency. Both factors can result in the restoration of bottom fauna and flora of the bay, thus providing a possibility to achieve a better state of the marine environment within the area impacted by the measure. Due to the closed or semi-enclosed geological structure of the inner bays, it is unlikely that the effects of the measure will be directly noticeable on a wider scale. On the other hand, if the bay is more open and thus there is more water exchange, the effects of the efforts done to reduce nutrients might not become visible at all. Indirectly, both positive and negative changes in the nutrient load within semi-enclosed areas will eventually have an impact also on the water further out from the coast.

In the pilot study (Table 7), the impact of nutrient-rich irrigation water on crops was also monitored. The irrigation using the bottom water was done on fields at Ämnäsviken twice, and it increased the production of ley with 60 %. At Kaldersfjärden the irrigation was done four times, and the increase in ley production was 170 %. Though these results are still preliminary, they indicate that irrigation with the nutrient-rich bottom water can have a positive effect on the yield.

Potential for offsetting

Nutrient removal capacity

Using eutrophicated near-bottom seawater in irrigation of crops may be considered a nutrient offsetting measure if it can be verified that a significant amount of nutrients is removed from the sea and this in turn reduces the use of artificial fertilizers in agricultural practices. The method is efficient as long as the anoxic situation continues. The relative effect of removing near-bottom seawater decreases considerably when the sea area recovers from eutrophication. In stratified areas the eutrophication influence of deep water varies according to mixing conditions: in seasonally stratified basins the bottom water nutrients do not necessarily participate in algal production processes during the summer. Thus, the surface production may continue without change, and the symptoms of eutrophication continue as previously.

Risks and uncertainties

The ecological impacts of irrigation with brackish water can vary from increased soil salinity to accumulation of heavy metals and other contaminants in the crop. Furthermore, in some cases the leaching of salinity can also cause problems with groundwater.

Increase in soil salinity. An increase in the sodium absorption ratio (SAR) in the soil is reported to decrease the soil hydraulic conductivity (McNeal et al. 1968, Adhikari et al. 2012b). This means that the capacity of water to flow through the soil is reduced, and the renewal rate of water within the soil as well as the amount and flow of dissolved gases and nutrients can be impacted. Irrigation water salinity can subsequently cause seed dormancy, which relies on physical defense to exclude predators and pathogens (Dalling et al. 2011). Varying results can be found from literature on the impacts of irrigation with brackish water on crop plants (see e.g. Bernstein 1975, Allen et al. 1998, El-Dardiry 2007, Diaz et al. 2013) although in most cases the water used in growing experiments has been more saline than the 3–5.5 occurring, for example, in the inner bays of the Åland Islands. Some earlier experiments on brackish water irrigation from Sweden suggest that although short-term accumulation of salt could be seen during some years of the 10-year experiment, no long-term increase in soil salinity could be detected (Persson & Wesström 1991). Globally soil salination has resulted in the development of more salinity-tolerant crop plant varieties and halophyte farming (Browning et al. 2006, Picchioni et al. 2014). However, most agriculture occurring in salinized soils occurs in arid warm or tropical climates, where the growing season is longer than in Scandinavia. Therefore, most of the cultured halophyte species and salinity-tolerant varieties of crop plants would most likely not be successful in Scandinavia.

Accumulation of heavy metals in soil and crop. Irrigation with surface water or waste water containing heavy metals may lead to heavy metal accumulation in soil and crop plants (see e.g. Yang et al. 2011, Leblebici & Kar 2018, Chaoua et al. 2019, Hussain et al. 2019). However, the risks associated with heavy metal contamination in the inner archipelago bays of the Åland Islands and Northern Baltic Sea are low because no polluting heavy industry has existed in these areas. Some toxins are still expected to be found in the sediment, originating from anti-fouling agents, agriculture, atmospheric deposition and roads/traffic. Most of these toxins are bound to particles in the sediment and not easily dissolved into the pore water. An ecotoxicological assessment is however needed before water is taken into irrigational use to estimate the risks of contamination.

Accumulation of cyanotoxins in crop. Eutrophication leads to increased cyanobacterial growth, which may result in the occurrence of harmful cyanotoxins in the water column. The most common cyanotoxins encountered in the Baltic Sea are microcystin, anatoxin and nodularin. Besides being toxic to invertebrates and vertebrates, many cyanotoxins can bind covalently and/or accumulate through, e.g. protein incorporation in numerous members of the Poaceae family such as *Triticum aestivum* (wheat), *Zea mays* (corn) and *Oryza sativa* (rice) (Machado et al. 2017, Contardo-Jara et al. 2018). This raises a concern that terrestrial crop plants irrigated with water contaminated with cyanotoxins might induce toxic accumulation effects consisting of various cyanotoxins in food. Häger (2018) has collated a short report concerning the bioaccumulation of cyanotoxins in crop and concludes that some cyanotoxins, such as cyanobacterial microcystin-LR (MC-LR), can efficiently accumulate in a wide variety of agricultural plants. Furthermore, numerous studies have shown that the physiology and metabolism are affected if plants are exposed to sufficient levels of MC-LR, risking loss of crops as a result of inhibited germination, alteration of chlorophyll as well as decreased growth and total yield (Manning & Nobles 2017). Exposing terrestrial plants for nodularin have in studies shown to increase plant oxidative stress, to lower the general fitness of individuals and to reduce growth (Lehtimäki et al. 2011). A better understanding regarding the accumulation and impacts of cyanotoxins in terrestrial crops is the key for enabling the safe utilization of irrigation water from potentially cyanotoxin-polluted water sources.

Increased salinity in the groundwater. Understanding the impacts of irrigated agriculture on hydrological systems is fundamental for implementing management programmes that are effective in maintaining water resources (Pulido et al. 2018). It is estimated that about 30 % of global irrigation water withdraws back to local hydrological systems by return flows and conveyance losses to groundwater and rivers (Scanlon et al. 2007). However, most scientific literature is concentrating on areas with arid and tropical to subtropical climate conditions. The transfer of salt from soil surface is impacted by the soil composition, which is typically moraine, clay or bedrock in the Nordic countries. In the light of current knowledge, it is difficult to predict at which level (depth) in the soil the water and salinity transfer will occur, i.e. if the water will travel downwards until reaching the groundwater or if it will be transferred horizontally within the catchment area until reaching the bay of origin. However, it can be concluded that water generally tends to move underground vertically in moraine whereas it travels horizontally after encountering clay or bedrock. Due to the lack of knowledge, the impacts of irrigation with brackish water in large quantities should be studied before any large irrigation systems are realized.

An irrigation water cannon. Photo: Annica Brink

5.2.6 Removal of nutrient-rich seabed sediment layer

Theory

Along the Northern Baltic coastline, there are thousands of relatively small local basins, in depths usually less than 25 m, with thick soft sediment layers. As a result of oxygen-consuming decomposition of organic matter, the seabed depressions are often annually affected by anoxic conditions occurring in late summer and autumn. Seabed anoxia results in the release of phosphorus into the water column and as a result the water quality of coastal archipelago areas has not improved even though external nutrient loading from the watershed area has decreased. In lakes it has been estimated that highly P-saturated sediments have a slow response to reduced external loads because sedimentary storage of P can act as a buffer to changes in water column P concentrations (Søndergaard et al. 2005). A decrease can be expected some years after external reductions when the P content of the top sediment layer has moved towards equilibrium with the concentrations in the overlying water column (Kagalou et al. 2008). The time needed for this recovery can vary considerably between lakes, and it is assumed that in some cases it can even take decades. Shallow anoxic areas might also be sources of methane to the atmosphere. The removal of the sediment surface layers from shallow coastal anoxic basins could thus decrease nutrient and methane release into the water column and atmosphere and improve water quality.

Nutrient removal capacity

The extent of anoxic seabed areas above 25 m in the Archipelago Sea was estimated based on a spatial model developed by Virtanen et al. (2019) (Figure 5). The model extends over the entire Finnish marine area, excluding the exclusive economic zone (EEZ). Based on the morel the anoxic conditions vary on a regular basis: severe hypoxia ($O_2 < 2$ mg/l) covers occasionally 1.4 % and frequently 2.3 % of the seabed, whereas moderate hypoxia ($O_2 < 4.6$ mg/l) covers occasionally 8.3 % and frequently 1.5 % of the seabed. However, the model does not consider local factors contributing to the formation of anoxia, such as point sources of nutrients or the impacts of currents impacting the nutrient content of both the water column and the accumulating organic particles. Suitable site selection should therefore be based on models and field sampling data so that optimum sites for sediment removal can be discovered.

A sediment removal pilot is planned within the SEABASED project. Some sediment samples for monitoring have been collected in May 2019. Based on these sediment samples, there is a preliminary indication that the removal of 10 cm sediment layer from 2 hectares could correspond to about 300 kg P and 2500 kg N. More data and large-scale piloting are however needed to make more accurate estimates and conclusions.

Ecological impacts

The main objective of the measure is to eliminate the organic oxygen consumption processes in the sediment and, consequently, decrease the leakage of phosphorus from anoxic sediments into the water column. This may lead to improved water quality locally and, depending on the prevailing currents, also on a wider area. However, soft sediment removal in shallow coastal areas may cause extensive sediment blooms that disperse from the dredging site via coastal currents. The release of organic matter and nutrients into the water column can result in algal blooms and the siltation of underwater habitats. Impacts on, e.g. coastal reproduction sites of fish also have to be assessed because both the fish spawn and larvae can be adversely impacted by sediment dispersal.

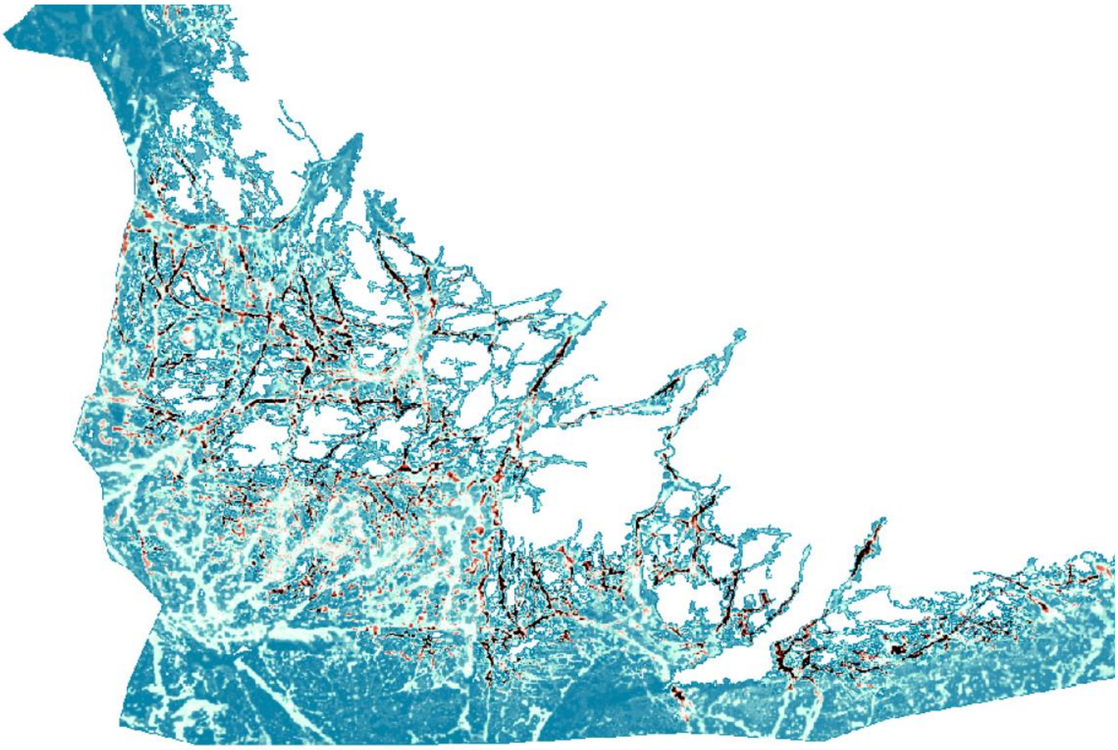

Figure 5. Predicted extent of anoxic seabed areas above 25 m in the Archipelago Sea. Source: Virtanen et al. 2019.

Potential for offsetting

Nutrient removal capacity

There exists a possibility to develop a nutrient compensation measure from removing a nutrient-rich sediment layer from the seabed. However, in order to do this, a technical solution to prevent extensive sediment and nutrient dispersal in the water column must be developed. Furthermore, a methodology needs to be developed to assess the actual nutrient removal capacity case-by-case based on field inventory data and modelling. The efficiency of the method to control eutrophication is not directly proportional to the amount of nutrient removed but to the amount of decreased nutrient release from sediments and of the entrainment of released nutrients to a productive surface water layer. Thus, the efficiency needs to be carefully estimated before its benefits in nutrient compensation can be evaluated.

Risks and uncertainties

Seabed sediment removal may form a sediment bloom that spreads with water currents locally or even further. The potential dispersal of sediment should thus be studied, so that no adverse impacts on the marine ecosystem are caused. Furthermore, the removal activity can cause underwater noise locally, and the resulting physical seabed disturbance can have an impact on fish. Technical choices in dredging can greatly impact the environmental effects of sediment removal, i.e. suction dredging is likely the least harmful method due to its very localized impacts. Furthermore, local seasonal environmental and ecologic conditions, e.g. bird and fish reproduction periods, must be taken into account when planning sediment removal activities to avoid harmful impacts on biota.

The sediment dry matter consists also of chemical compounds other than phosphorus. Parameters such as dry-to-weight ratio, organic content and the content of iron, aluminium, manganese, calcium, clay and other elements with the capacity to bind and release phosphorus may all influence sediment-water interactions (Søndergaard et al. 2003). While the concentrations of phosphorus (P) and nitrogen (N) vary widely in the sediment pore water, it also contains, e.g. phosphate. Along with seabed sediment matter, the suction dredging also removes substantial amounts of water with varying concentrations of P and N. The dynamics of both nutrients between the sediment and near-bottom water are constant processes, and the contents both in the sediments and in the water vary markedly in time and space – even over short times and small areas. As a result, if nutrients of both the sediment and the seawater are removed before the water is discharged back into the sea, a great variability in acquired amounts of both nutrients can most likely be observed. This might make it difficult to calculate the actual gain of nutrients for developing economically profitable applications.

Sediment utilization

Only a small proportion of phosphorus is currently recycled, apart from manure in animal husbandry, which is almost fully recycled in Finland. Many waste flows in different sectors contain high phosphorus concentrations, which could be potential reserves for the future. Wastewater sludge and biodegradable solid wastes in the food production chain provide the highest recycling potential (van Dijk et al. 2016). Phosphorus concentration, chemical quality (e.g. heavy metals), spatial location and technological costs are important aspects affecting P recyclability (Oenema et al. 2013). Utilization possibilities of phosphorus stored in aquatic sediments has only recently been studied (e.g. Laakso et al. 2016, Laakso et al. 2017a, 2017b). In principle, it is possible to utilize the sediment nutrient reserves although developing an economically cost-effective process requires more research and experiments. Furthermore, there is a need for developing applications based on removed seabed sediment. Sediment could be used in forestry improvement but possibly also in biogas production. However, in experiments conducted with freshwater sediments, it has been discovered that the plant availability of P in sediments is very low due to the high concentrations of clay, and Al and Fe (hydr)oxides in sediments (Laakso et al. 2017b). Bringing sediments to fields in large quantities is therefore likely to decrease the amount of P readily available for crop uptake. However, the application of dredged sediments can be expected to immobilize soil P and decrease non-point source P loads when applied to critical source areas with environmentally problematic P saturation. A practical rate of sediment addition to the surface soil layer could be approximately 5 % (by fresh volume).

5.2.7 Permanent binding of nutrients in the sea bottom

Theory

Chemical restoration methods mainly aim at reducing the phosphorus release from sediment by improving the sediment P binding capacity, reducing the amount of soluble P in the sediment pore water and thus creating P limitation for the phytoplankton (Zamparas & Zacharias 2014). During the last two decades, several solid adsorbents have been considered in freshwater environments to bind phosphorus into the sediment (Zamparas et al. 2013). Some of the materials studied in the freshwater environments include aluminium, iron oxides, red mud, fly ash and carbonates (De-Bashan & Bashan 2004, Huang et al. 2008, Wang et al. 2008, Rydin et al. 2017). It was estimated in 2017 that 46 % of the soft sea bottoms of the actual Baltic Sea, including the Gulf of Finland and Gulf of Riga, suffered from the impacts of hypoxic or anoxic conditions (Hansson et al. 2017). Utilizing maerl or limestone residue or its derivatives to bind phosphorus to the sediment in highly phosphorous sediment areas in the Baltic Sea could therefore provide a possibility to decrease nutrient release into the water column under anoxic conditions and reduce the development of extensive algal blooms.

Maerl is formed of the dead deposits of calcareous red algae (Corallinaceae) found growing in shallow waters around the coasts of north-west Europe and the western Mediterranean. It has been demonstrated earlier that it effectively removes phosphorus from sewage (Gray et al. 2000), and it also performed well in an excess conventional waste water treatment system as a trickling filter and activated sludge plants (Horan 1990).

Activated limestone is produced from a side product of industrial lime production. During the production of lime, stone material is crushed, and particles with size less than 25 mm are not used in the process but piled as a by-product of the process. Sifting and heat treatment could be used to improve the P binding capacity of the material by over 600 % (Blomqvist et al. 2019). There is currently an ongoing field experiment in two Swedish coastal bays concerning the impacts of activated limestone on the P release from sediments and overall environmental impacts of the treatment (Blomqvist et al. 2019).

Nutrient removal capacity

There have been numerous studies on potential chemical compounds that could be used to reduce the release of phosphorus from sediments in freshwater environments (Table 8). However, there are much fewer studies on potential compounds and their environmental impacts in marine or brackish environments.

Table 8. Phosphorus removal capacity of some materials used in treating bottom sediments. The nutrient removal efficiency is dependent on the prevailing experimental and environmental conditions, such as the amount of sorbent.

Material	Removal capacity (Total P %)	Experimental system	References
Maerl	98	Sewage nutrient removal in flow-through tanks	Gray et al. 2000
	90	Trickling filters containing maerl in waste water treatment systems	Horan 1990
	25	Activated sludge plant	Horan 1990
Activated limestone	30*	Inner bay experiments in Baltic Sea	Blomqvist et al. 2019
Gravel bed	94	Removal of nutrients in wetlands	Breen 1990
	95	Removal of nutrients in wetlands	Rogers et al. 1990
Red mud	100	Removal of nutrients in wetlands	Lopez et al. 1998
LECA	98	Removal of nutrients in wetlands	Maehlum et al. 1995

*Estimation based on laboratory experiments

Maerl has been studied in water purifications, and it is successful as a P adsorbing substrate due to its high surface area, providing increased contact time with the effluent and many sites for adsorption, while the high calcium carbonate content enables the chemical precipitation and subsequent sedimentation of phosphate (Gray et al. 2000). However, the prevailing temperature influences the adsorptions process, so in very low temperatures the process occurs relatively slowly.

The utilization possibilities of activated limestone in sediment P binding are currently investigated in the Northern Baltic Sea in field experiments. The experiments are based on laboratory studies, and it is expected that first results will be gained in 2020 on the P binding capacity and environmental impacts of the treatment.

The use of aluminium (Al) in eutrophicated lake or sea bottoms is based on the idea that if the supply of a binding agent that is naturally involved in the permanent burial of phosphorus in the sediment, such as Al, is increased, it strengthens the binding of P in anoxic sediments (Rydin et al. 2017). The method has been used in lakes but is a potential measure also in the Baltic Sea. It has been tested in a large-scale experiment in Björnöfjärden in Sweden. There the dissolved aluminium was injected into the anoxic sediment during 2012–2013 and the effects were followed until 2016 (Rydin et al. 2017). As a result, during the 40 months' follow-up period, 1.3 tonnes of dissolved phosphorus has been trapped by the added Al in the 0.7 km^2 treated anoxic sediment area. Rydin et al. (2017) conclude that phosphorus "no longer supports eutrophication in the bay, turning the treated bay into a sink for P in the Baltic Sea ecosystem". There was a time lag before the phosphorus binding effects of aluminium were detected. Rydin & Kumblad (2019) conclude based on data analysis and modelling that "the lag in recovery of water quality after eutrophication in enclosed water bodies is largely dependent on the limited sediment-P burial capacity". Though the results from Björnöfjärden are promising, the authors highlight the importance of also mitigating the external nutrient loads.

Ecological impacts

Reductions in the release of phosphorus from the seabed sediment would result in a decrease in algal blooms and improve water quality. The improvement of water quality can have positive impacts on, e.g. coastal aquatic macrophyte communities, invertebrates and fish.

<u>Maerl</u>

In an experimental study conducted on the adsorption capacity of P and environmental effects of maerl utilization, reed growth was sustained throughout the experiment with evidence on horizontal spread of rhizomes (Gray et al. 2000). Furthermore, all experimental tanks were colonized by various species of algae, which were grazed by gastropods. Thus, it could be hypothesized, that utilizing maerl would not result in any extensive environmental impacts.

There are, however, conservation issues associated with the extraction of maerl (Birkett et al. 1998, Gray et al. 2000). The Habitats Directive requires all EU member states to protect a sufficient amount of their maerl beds as a subtype of the marine habitat Reefs (1170). Maerl beds have a high conservation value because of their very high diversity of organisms (Birkett et al. 1998). Due to the fact that maerl beds form extremely slowly, it has been estimated that in the UK the annual sustainable extraction rate of maerl is 4 000 t. As a result, sustainability issues have to be considered when developing applications based on maerl in water purification.

<u>Activated limestone</u>

Aerial distribution of activated limestone or other similar substrates can cause a short-term decrease in water transparency before the particles have settled on the seabed. Laboratory experiments have demonstrated that a slight increase in pH may occur in water close to the seabed sediment (Blomqvist et al. 2019). The effect varies depending on the substrate that the limestone is being spread on.

As activated limestone is a by-product of the mining industry, there are ecological effects in the production chain. However, the mining industry does usually not affect marine or coastal areas directly and is thus not discussed further in this report.

Aluminium

Ecological effects were clearly detectable in the so far largest experiment on adding dissolved aluminium to the Baltic Sea sediment (Björnöfjärden). Rydin et al. (2017) report the main findings. In the Al-treated area the water quality changed, which caused changes both in the benthic vegetation, benthic macrofauna and fish species abundance and diversity (for water quality parameters, see Table 2 in Rydin et al. 2017). Increased water transparency affected not only the benthic ecosystem but also the bottom water, where photosynthesis increased and the development of sulfate-reducing bacteria decreased.

Though the results from Björnöfjärden are promising, the use of geoengineering techniques to enhance the state of the Baltic Sea are also criticized. For example, in the critical paper by Conley (2012), the main concerns regarding using aluminium include 1) the cost of the chemicals, 2) the uncertainties of how chemical binding used in lakes will work in salty water areas, 3) how long the Al-bound P will stay in the sediment and 4) are there effects on water acidity. In general, Conley's critique is centred on the potential risks of rapid oxygenation of deep waters using geoengineering techniques. He concludes that a better solution would be to work harder to reduce nutrient flows to the Baltic Sea. Rydin et al. (2017) agree with the need to reduce the nutrient supply but see that addressing the problem of already existing sediment P release is needed, and aluminium could be a cost-effective tool to help in this, especially in enclosed coastal areas.

Potential for offsetting

Nutrient removal capacity

There exists potential for phosphorus reductions in using suitable laboratory and field-tested chemical compounds to reduce phosphorus release from the seabed sediments. However, in order to be able to calculate the nutrient offsetting capacity of the measure, more long-term *in situ* knowledge is needed to assess the long-term impacts on nutrient release and on the marine environment.

Risks and uncertainties

Since most of the studies have so far been conducted in freshwater environments, more knowledge is needed to develop measures based on materials such as aluminium, maerl or activated limestone in the marine environment. In freshwater it has been discovered that if the chemical properties of the water phase change, e.g. redox or pH, the adsorbed phosphate can be released (Zamparas & Zacharias 2014). Some results exist from laboratory experiments conducted on seabed sediments treated with activated limestone in the Baltic region (Blomqvist et al. 2019), providing a starting point for field experiments, which were started in 2019 in two bays in Sweden.

Circular economy applications

Activated limestone is produced from a by-product of lime production, at least in Finland and in Sweden. By using an industrial by-product in the production of a solid absorbent for binding seabed sediment phosphorus, the extensive use of maerl and related environmental risks to biodiversity could be avoided.

5.3 Potential offsetting measures in the watershed area

Agriculture provides livelihoods for farmers, food for society, and rural landscapes for both humans and wildlife. However, current agricultural activities result in the release of nutrients and soil off fields into the aquatic environment. These cause eutrophication and increased turbidity in both freshwater and marine ecosystems. The leaching of phosphorus is strongest from fields that are susceptible to erosion or that have abundant phosphorus reserves. Such fields can be found in the river basins of many coastal areas of the Baltic Sea.

When considering land-based measures as a part of a nutrient offsetting plan, several things have to be taken into account. Local environmental conditions have to be considered and also used when assessing the projected impacts of measures on the targeted marine area. One of the most important questions is linked to the additionality of the measures, which means that measures have to be additional to all EU policies required by, e.g. EU's Rural Development Programme. Only after the measures required by the EU's Rural Development Programme have been fully implemented can measures for nutrient compensations be realized.

Another issue is locality. All measures in the watershed area have the common feature of having positive effects due to decreased river loading. However, the riverine effect is restricted to the coastal areas with large rivers flowing, and even in them the effect is restricted to near the coast. Watershed measures rarely affect outer archipelago areas or areas without large river loading.

The use of gypsum as a soil amendment, manure processing and crop rotation are potential tools to reduce the phosphorus load and potentially also erosion on agricultural areas. These three measures are addressed in more detail. In addition, structural liming is one way to reduce the nutrient runoff and erosion in agricultural areas. It is suitable for clay soils and has been tested, for example, in Björnöfjärden (the same area where aluminium was tested in binding sediment phosphorus, see 5.2.7). Compared, e.g. to using gypsum, soil liming is more expensive, but then again, the positive effect on the soil and reducing nutrient leakage should be long-term[30].

5.3.1 Gypsum

Theory

Gypsum is calcium sulfate dihydrate ($CaSO_4 \cdot 2H_2O$). As a soil amendment, it significantly reduces the agricultural phosphorus load and erosion in the watershed area (Ekholm et al. 2011, 2012, Iho & Laukkanen 2012, Uusitalo et al. 2012). Gypsum amendment as an agricultural water protection measure has been investigated at various scales in Finland in the recent years (Ollikainen et al. 2018). It has been concluded that the soil type, cultivation practices as well as weather conditions all have an influence on the impacts of gypsum on soil. Gypsum is particularly suitable for reducing phosphorus runoff on clay soils. Several large-scale experimental studies were carried out in Southern Finland to study the impacts of gypsum as a water protection measure[31]. It has been estimated that the method is suitable for over 500 000 hectares of arable land in Finland located in the river basins of water bodies flowing into the Baltic Sea (Ollikainen et al. 2018). Based on the promising results, the method of spreading gypsum on fields to reduce phosphorus runoff is currently in use as a water protection measure at some suitable locations in Southern Finland. The long-term monitoring of the impacts also continues.

30 Short introduction to structure liming and the pilot project in Björnöfjärden: http://www.balticsea2020.org/english/bibliotek/32-eutrophication/320-structure-liming-reduces-the-leakage-of-nutrients-to-bjoernoefjaerden

[31] 31 The gypsum pilot is ongoing, treated area locates at the Vantaa river catchment area, more information on the John Nurminen foundation webpage: https://johnnurmisensaatio.fi/en/projects/the-river-vantaa-gypsum-treatment-project/

Nutrient removal capacity

The effects of gypsum treatment on soil solid matter, particulate and dissolved phosphorus, organic carbon, calcium, magnesium and potassium have been described by Ollikainen et al. (2018).

Solid matter. According to *in situ* measurements, gypsum treatment reduced the amount of solid matter running off the treated fields by around a half in the first two years. Gypsum amendment therefore markedly reduces the amount of solid matter in the aquatic environment and reduces turbidity.

Particulate phosphorus. The leaching of particulate phosphorus was reduced in proportion to the leaching of solid matter from the gypsum-treated fields. A reduction of around 50 % in particulate phosphorus content was achieved during the SAVE project and 60 % by the TraP project. Varying field characteristics could explain at least a part of this difference in test results, so local conditions have to be taken into account when considering the possible impacts of measures.

Dissolved phosphorus. Gypsum reduces the leaching of dissolved phosphorus. A reduction of around 25 % has been achieved by the TraP project.

Organic carbon. The increased ionic strength in soil due to gypsum treatment reduces the runoff of dissolved organic carbon. As a result, gypsum has been tested as a means of reducing organic carbon leaching in Australia. In the Savijoki river pilot project in Finland, gypsum significantly reduced the leaching of organic carbon. It was estimated that gypsum reduced the leaching of carbon bound to soil by around 50 %.

Calcium, magnesium and potassium. Gypsum contains calcium, which is absorbed by the surface of soil particles. It can also displace other cations in the soil, such as magnesium and potassium, which can be released into soil water. In the Savijoki river, it was discovered that calcium and soluble sulfate leached in mostly equivalent amounts. Only a relatively small portion of the calcium therefore remained on the surface of the soil particles. There was a slight increase in the runoff of other cations as well.

Based on recent studies it can be concluded that gypsum provides efficient phosphorus reductions with lower costs than any other water protection measure currently used in agriculture (Ollikainen et al. 2018). According to the tests conducted in Finland, a suitable dose of gypsum is 4 tonnes per hectare. In a large-scale pilot in 2016, gypsum was transported from eastern Finland to southwestern Finland (about 500 km). The total cost of gypsum amendment was EUR 220 per hectare. In reducing the phosphorus load from agriculture, the cost was EUR 60–70 per kilo of P — significantly less than the costs of other currently available measures to reduce nutrient leaching from fields to the aquatic environment.

Ecological impacts

Gypsum amendment significantly reduces erosion and the leaching of both dissolved reactive phosphorus and particulate phosphorus which is bound to the soil (Ekholm et al. 2011, 2012, Iho & Laukkanen 2012, Uusitalo et al. 2012). Dissolved phosphorus can be directly used by algae. Particulate phosphorus only affects eutrophication when it dissolves in water (Ollikainen et al. 2018). The increased ionic strength in soil due to gypsum also reduces the runoff of dissolved organic carbon. In addition, when the erosion rate is reduced, less carbon bound to the soil leaches into the aquatic environment. Similarly to phosphorus and nitrogen, carbon is an important factor in eutrophication. To improve the soil structure and mitigate climate change, carbon should be bound to fields rather than escaping into water bodies. Gypsum treatment also reduces carbon leaching from the soil into the aquatic environment. If the method is used near lakes, there is a risk of increased eutrophication due to the input of sulphate ions into the lake. Thus, the whole landscape needs to be considered when deciding where gypsum could be spread. The current information indicates that the impact of gypsum amendment on water quality lasts around five years.

Potential for nutrient offsetting

Gypsum treatment can prove to be an effective nutrient offsetting measure on suitable areas. The cost of gypsum amendment in proportion to its ability to reduce the phosphorus load in agriculture is around EUR 60 to 70 per kilogram of phosphorus reduced. Existing means of reducing the phosphorus load, such as the addition of protection strips and wetlands, are considerably more expensive.

Gypsum treatment is not currently in the measures in the EU's Rural Development Programme, so it could be used directly as a nutrient offsetting measure. However, there is an ongoing discussion on whether the method should be included in the list of measures of the Rural Development Programme. It can therefore be concluded that if additionality in nutrient reduction is achieved on top of the policy required measures, it could be considered as a nutrient offsetting measure.

Circular economy applications

From the viewpoint of general applicability of the gypsum amendment, sources of suitable gypsum should be found locally. Gypsum occurs in nature as a mineral which can be mined, but also an industrial by-product. It can also be easily recycled. When the gypsum's origin is known and it is confirmed that its contents are pure, it is safe for use in agriculture (Ollikainen et al. 2018).

Phosphogypsum. In Finland, gypsum is produced as a by-product of the phosphoric acid industry. Gypsum is formed through a process in which locally mined apatite is dissolved in sulfuric acid. Because apatite from Siilinjärvi contains no heavy metals or radioactivity, the gypsum generated through the process is safe to use. Globally, the possibilities to utilize phosphogypsum are restricted due to possible content of heavy metals or radioactivity. Still, it might be possible to develop methods to purify phosphogypsum from harmful substances and use it in agriculture.

FGD gypsum. Another industrial and commonly known by-product gypsum is FGD gypsum (flue-gas desulfurization). It is a by-product of the energy industry, originating from a process of purifying sulfur oxide from fossil-fuel power plant emissions. FGD gypsum is common in the USA, where gypsum is a well-known soil treatment additive.

Recycled gypsum. Recycled gypsum from finely grounded wallboards is also a suitable material for agricultural use, used, for example, in the USA.

Mined gypsum mineral. Gypsum is also a natural mineral, which is mined globally. Mined gypsum is suitable for organic farming. However, the use of industrial by-products and recycled gypsum can support circular economy approaches and reduce the need to use pristine natural resources.

5.3.2 Manure processing

Theory

Manure has a relatively low nutrient content and its P:N ratio is not optimal for agricultural use. Manure processing, such as mechanical and chemical separation and anaerobic digestion in biogas plants, may provide profitable solutions for improving manure utilization in fertilization and reduce nutrient leaching from fields into the aquatic environment. The efficient use of manure energy and nutrient content is a prerequisite for sustainable food production and decreasing the agricultural nutrient load to the environment. There have been several recent projects investigating the processing possibilities of manure to improve its usability in farming. Questions related to the mechanical and chemical separation technologies, use for biogas applications, composting and commodifying into commercial fertilizers have been investigated. The key question is how to achieve the optimal fertilization ratio of P:N for different crop plants. Also, the nutrient balances need to be considered. If too much manure is produced compared to the field area, cost-efficient solutions are needed to the problem of how the excess manure could be transported to other areas needing nutrients.

The fertilizing value of cattle slurry, digestate from a farm scale biogas plant and the separated solid and liquid fraction of the digestate, has been studied in barley and grass production (Virkajärvi et al. 2016). The use of organic fertilizers was also compared to that of mineral fertilizers in field experiments. In the barley experiment, digestate gave similar yields as a comparable dose of mineral soluble nitrogen (N), except in the dry year of 2010 when the yield was 10 % lower. Raw slurry yielded only 85 % of the yield from digestate. Separation and use of the fractions did not show particular benefits for barley. In the grass experiment, drought caused a larger difference in N uptake than the use of digestate or separated digestate: in the dry year, raw manure and digestate gave significantly lower yields than mineral fertilizer. However, when using the liquid fraction of digestate, no such effect was noticed. Raw manure and digestate did not differ in the fertilizing effect. With barley, N balance was mostly positive, i.e. the N removal in harvested crop was lower than input N as slurry and fertilizers. The N balance of digestate was lower than that of raw slurry in two of the years studied. With grass, the N balances of the first cut were usually negative, which lead to negative annual balances. On the grass plots, digestate produced a lower N balance than raw slurry only in 2012 when its proportion of soluble nitrogen of the total nitrogen content was higher than that of raw slurry. The plots fertilized with a liquid fraction of digestate received a higher dose of total N than the other plots, which was also seen as positive balances each year. Also, the P balances differed significantly between the plant species. With barley, the P balance was almost always positive, as with mineral fertilizers, while with grass, it was nearly without exception negative. The fertilizing effect, nutrient balances and soil nitrogen cycle of organic fertilizers are clearly different in grass production than in cereal cultivation. This is mostly explained by differences in cultivation methods (e.g. the number of fertilizing and harvesting times, harvesting the grains or the entire biomass) and by the different nutrient uptake ability of the plants. According to this study, the benefits of using digestate are clearer for barley cultivation than for grass production.

Nutrient removal capacity

Developing manure processing would reduce the need for chemical fertilizers and allow the development of more efficient, crop species-specific fertilization practices in agriculture. For example, it has been discovered that plant-availability of phosphorus is higher in biogas digestate than in raw manure with both oat and grass cultivation (Virkajärvi et al. 2016). Furthermore, it was shown that the improved nitrogen and phosphorus uptake when using digestate was also linked to the smaller excess of nitrogen and phosphorus in the soil than when using raw manure. Biogas digestate was more homogeneous and fluid compared to raw manure and its nutrient content more stable. Therefore, regardless of the limiting factor in manure fertilization (manure/digestate total nitrogen or soil phosphorus), digestate allowed for a slightly higher dose of soluble nitrogen (7 %) per hectare.

Ecological impacts

Developing novel solutions for manure use in agriculture would allow its more efficient use in agriculture and reduce nutrient leaching from fields into the aquatic environment. Reductions in the amount of nutrients in the aquatic environment results in improved water quality. Water turbidity decreases when harmful algal blooms are not formed. Cascading impacts of nutrient reductions reach all levels of the Baltic Sea ecosystem, impacting the heavily eutrophicated coastal areas most strongly. Furthermore, manure processing could provide novel energy sources on a local level and reduce the need for unrenewable energy. Regional nutrient balances need to be considered when novel solutions are developed.

Potential for nutrient offsetting

Manure processing is one of the measures included in the EU's Rural Development Programme to reduce the inflow of nutrients from the watershed, so it could most likely not be used directly as a nutrient offsetting measure. However, if the measure would result in additional nutrient reductions on top of the policy-required measures, it could be considered as a nutrient offsetting measure.

Circular ecology applications

Studies exist on how to develop biorefinery concepts for animal husbandry, combining, e.g. crop rotation including perennial grasses, grass biorefining and use as pig feed, biogas production and recycling of the residues (Molina-Moreno et al. 2017, Yazan et al. 2018, Tampio et al. 2019). Biorefinery concepts can provide an energy source for the farm and be economically viable for vehicle fuel production with a slight increase in the price of the fuel, a moderate increase in the price of grass liquid fraction or with a better optimized production, starting with grass cultivation and processing. Moreover, the profitability of an activity increases with operational size, for example, by scaling up production by co-operation between two or more farms. When evaluating the concept for an individual farm, other measures such as the requirements of the agri-environmental support scheme need to be taken into consideration. More knowledge and potentially new regulations are needed on spreading residues from biogas production to fields. The risks of increasing the nutrient content of the fields too much and causing additional nutrient leakage need to be addressed properly.

5.3.3 Change in land use through crop rotation

Theory

The majority of nutrient runoff into the aquatic environment originates from agriculture (see, for example, Karonen et al. 2015, Westerberg et al. 2015, Laine et al. 2016). Of the total Finnish nutrient load to the sea, 53 % of P and 38 % of N is estimated to come from agriculture (HELCOM 2018). There are two major sources of nutrient runoff: (1) nutrient leaching from the fields and (2) the accumulation of P in the fields, which becomes a part of nutrient runoff through erosion. Along with culturing practices, annual hydrographical conditions strongly impact the rate of nutrient release from soil. Changing land use in agriculture through, for example, five-year crop rotation, could prove to be an efficient nutrient compensation measure in cultured areas with high nutrient release capacity, while simultaneously providing a carbon sink and improving soil quality and water holding capacity (Granlund et al. 2015). Moreover, grass has higher P requirements than grain. On fields with high P levels, it is possible to lower the P content of the soil by cultivating grass instead of grain (Valkama et al. 2015). Crop rotation would allow reductions in soil P content while simultaneously improving soil quality for crop plant cultivation.

Nutrient removal capacity

Grass cultivation should not include any chemical or organic fertilization.

Ecological impacts

Growing a wider diversity of crops and perennial forage crops results in a more diverse agroecosystem with an increased soil biodiversity and improved soil structure as a consequence (Tiemann et al. 2015). This also increases the soil's capacity to store carbon and improves soil structure. Introducing less intensive crops, such as cereals, grass and clover species, in the crop rotation increases the carbon content in the soil through the extensive rooting system.

As an example, a 12-year study was conducted in the USA to study the impacts of monocultures and crop rotation on soil characteristics (Tiemann et al. 2015). It was discovered that the carbon and nitrogen concentrations of the soil increased with rotational crop diversity across both sizes of aggregate. For instance, soil carbon increased by 33 % in mega-aggregates in soils planted with diverse crops compared with carbon in soils used to grow monoculture corn. These changes under high-diversity rotations were associated with the increased stability of mega-aggregates, which is an indication of soil organic material (SOM) formation and accumulation. This is because higher crop diversity increases the quality and quantity of crop residues that can be incorporated into the soil, which then become available to the microbial communities. As a result, microbial activity is increased, which enhances soil clumping into mega-aggregates where SOM becomes protected and can accrue. Microbial activity in micro-aggregates is also enhanced, further promoting micro-aggregate formation and increases in the amount of organic carbon and nitrogen that can be stored.

As the number of crops grown on a plot increased, the structure of the microbial community shifted, with the most diverse rotations having a greater abundance of fungi relative to bacteria. This result highlights the importance of soil fungi in the development of stable soil structures, as fungal hyphae play an important role in the binding of soil particles together. Soil aggregates help to store and protect organic carbon and nitrogen in the soil. Furthermore, a high level of SOM enhances soil fertility. The results demonstrate that rotational diversity enhances soil microbial diversity, with related increases in organic carbon, nitrogen and SOM. Particularly in agricultural systems practicing conservation techniques with minimal or no chemical inputs, growing a diverse range of plants in rotation can improve soil structure and boost soil fertility.

Currently there is an ongoing project, SoilDriver AGRO, funded by the EU Horizon 2020 programme, where questions related to soil diversity and agriculture are addressed across Europe. The project aims at describing soil microbial communities' characteristics but also provides crop management practices and cropping systems to enhance soil biodiversity, studies environmental, economic and social costs and benefits of proposed management practices and sets operational biodiversity targets for cultured soil across Europe.

Potential for nutrient offsetting

Moving from crop plant cultivation to, e.g. crop plant-grassland cultivation with five-year crop rotation would both reduce nutrient release from soil and increase soil carbon storage. In practice this would mean that agreed changes in agricultural practices and the loss of income would be compensated financially for the farmer by companies requiring nutrient offsetting measures. The measures should be temporally long to achieve the goals set for nutrient reductions.

Circular economy applications

Besides only receiving financial compensation from agricultural land turned from crop cultivation into growing grass as a part of crop rotation, there also exists novel circular economy applications for grass biomass. During the last few years there have been studies on the usability of grass in various farm-scale applications. Tampio et al. (2019) describe a new farm-scale grass biorefinery concept to sustainably intensify local feed and food production and how to potentially combine it with energy production on a pig farm. In this approach the grass for a green biorefinery can be used either fresh or after ensiling. The typical first processing step is liquid-solid separation. Furthermore, the liquid and solid fractions can have multiple uses depending on the processes involved (Corona et al. 2018). In the study of Tampio et al. (2019), grass silage was used as the raw material, the liquid fraction for feeding the pigs, and the solid fraction, together with pig slurry, for co-digestion in a biogas plant on the farm. Grass juice obtained from grass silage is rich in amino acids and suitable as a part of liquid feed for pigs (Hulkkonen 2019). There has been a similar approach based on producing pig feed from grass silage in Norway (Orkel 2017), whereas in Denmark fresh grass has been used and the protein from grass juice has been further precipitated and used as a dry feed product (Corona et al. 2018). The concept could also be applied to other northwestern European countries where grasses grow well in the cold and humid climate conditions with abundant solar radiation during the growth season (Manevski et al. 2017). Finnish pig production is often based on feeding the pigs barley, which is typically grown on the farm's own fields (Tampio et al. 2019). Pig slurry is used as a fertilizer for barley. Other common crops in the crop rotation include, e.g. wheat and rape, part of which may be used as feed, with the rest being sold. Additional feeds, particularly those with high crude protein content, are bought in. The introduction of grass cultivation and the biorefining of grass silage could contribute to increasing the protein, energy and nutrient self-sufficiency of pig farms and to enhancing farm-scale nutrient reuse, including manure.

Irrigation water uptake in Kaldersfjärden. Photo: Annica Brink

6 Economic and other aspects in establishing an offsetting system

The offsetting of human-induced environmental degradation can be obligatory or voluntary depending on the situation. The whole offsetting process can be managed by the public or private sector, or both of these in complementing roles. The way the responsibilities are shared among the private or public sector can affect compensation credibility, acceptability and compensation costs, among other things. How different responsibilities are allocated, benefits shared and the whole compensation process governed depends on, at least, the objectives, motivation, the existing legal framework and practical alternatives, but also on convenience and opinion.

In a simplified scenario there are two motivations for environmental compensations: 1) they are voluntary and done, for example, to gain social license to operate, fulfil corporate responsibility goals or reputational benefits or 2) they are obligatory and required by law (more on the legal aspects, see Section 4). Many of the questions on how the offsetting should and could be done in practice, how to secure long-term success of the compensation measures, or what metrics or units to use in compensation estimation are the same irrespective of compensation motivation. The solutions may differ, though. For example, a large enough demand for offsets is necessary for an economically sound compensation market (Kangas & Ollikainen 2019), and the demand for offsets is more predictable if compensations are obligatory. If the compensation system relies on the activity of private sector consultants or mediator companies, a large enough demand is needed. If compensation cases are expected to be occasional and rare, the system could be managed by the public sector only. Even if compensations were voluntary and the system market-based, the comparison of existing biodiversity offsetting systems indicates that the role of the public sector is important in securing the rights and responsibilities of different actors (Koh et al. 2019). Also, based on workshops and stakeholder interviews on biodiversity offsetting in Finland, common rules and the role of the public sector in defining those rules are considered very important (Suvantola et al. 2018, Primmer et al. 2019). The compensation procedure can involve several actors. First and foremost, there are those needing compensation, like a company that causes environmental degradation, and those providing offsets, for example landowners. They can use an intermediary to ease the compensation process. An authority is needed to monitor the process and define the rules. Citizens are important in defining the public acceptance of compensations, but they are also the ones who eventually may end up paying potential additional costs of compensations if the costs go into retail prices. These actor groups and their roles are schematically described in Figure 6.

6.1 Decisions that need to be made when planning offsetting

The focus in the compensation discussion is often on the ecological aspects, on how an offset can technically be produced or how the environmental loss and gain should be estimated and measured. There is also debate and research on how the whole compensation process should be governed. A recent and quite a thorough list on operational decisions that are needed in biodiversity offset planning with possible solutions has been published by Moilanen and Kotiaho (2018). Another interesting solution-oriented review on controversies in biodiversity offsetting is written by Maron et al. (2016). They list 13 common difficulties or challenges in biodiversity offsetting and propose potential solutions to these. Challenges are divided into categories (philosophical/ethical,

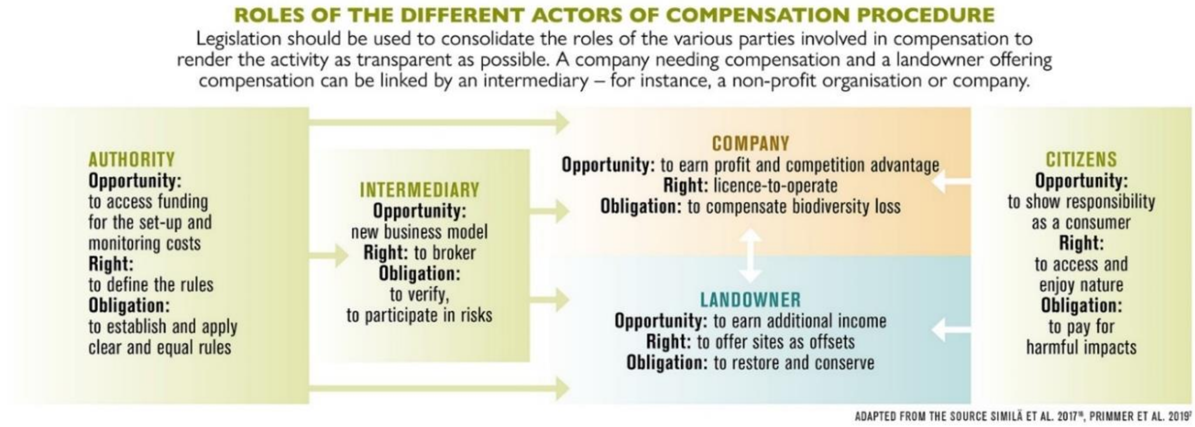

Figure 6. Several different actors can be involved in biodiversity offsetting. Each actor has both rights and obligations but also opportunities if they participate in the compensation procedure (copyright SYKE Policy Brief 20.11.2019, Pekkonen et al. 2019, original source: Primmer et al. 2019).

social, technical, governance), estimated for tractability, and for each the needed response is suggested and potential barriers identified. For example, “applying the mitigation hierarchy” is considered a technical challenge with high tractability. Suggested response to overcome this specific challenge is “*Ensure offsets reflect full replacement cost; develop clear guidelines on mitigation hierarchy application*”. So, the needed responses are not implementation guides but give an idea of what could be done. In a similar fashion Moilanen and Kotiaho (2018) identify 15 factors or decisions that need to be considered in biodiversity offsetting, and many of these are relevant also for nutrient offsetting (Figure 7).

From the ecological factors in Figure 7, the biodiversity axis is less relevant to nutrient offsetting. For nutrient offsetting the measurement units are less complex than for biodiversity. Also, the trading up (Factor 10.), which in biodiversity offsetting means a situation where typically the loss of a common habitat is compensated with restoring a rare habitat is not that relevant in nutrient offsetting. Due to simpler units in nutrient offsetting, it is easier to define the no net loss target (Objective 2).

In both nutrient and biodiversity offsetting, decisions need to be made on the following factors, actions or objectives:

<u>The mitigation hierarchy – how strictly is it followed (Objective 1)?</u>

- For nutrient offsets the question of mitigation hierarchy has already been discussed in Section 4 from the legal point of view. How strictly the mitigation hierarchy is followed may be relevant to the social acceptance of the environment-degrading activity. In addition, it may be difficult to draw the line between mitigation and compensation measures.

<u>Is the goal No Net Loss or partial compensation (Objective 3)?</u>

- For nutrient offsetting the compensation goal is set in practice by WFD (see Section 4.2).

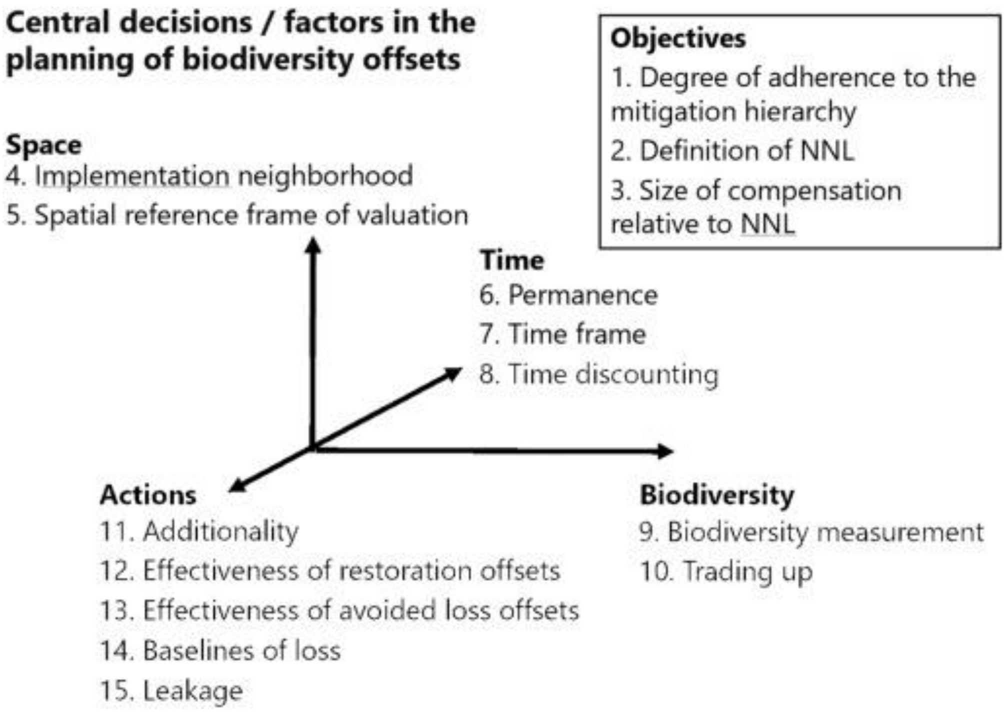

Figure 7. According to Moilanen & Kotiaho (2018) there are 15 decisions or factors that need to be considered when planning biodiversity offsets. They group these factors or decisions around objectives, offset actions and the three ecological relevant axes (space, time, biodiversity). Some of these, but not all, are relevant also for nutrient offsetting. For more on mitigation hierarchy, see Section 2.1.1, Figure 1. NNL = No Net Loss. Source and copyright: Moilanen & Kotiaho 2018.

<u>Where can the offset be produced (Decisions on Space 4 & 5)?</u>

- This is a highly relevant question for terrestrial habitats but also more straightforward. In aquatic environments the spatial question is more complex.

<u>Should the offset be permanent (Decision on Time 6)?</u>

- A good rule of thumb is that if the environmental loss or damage is permanent, then also the offset should be permanent. How to secure an offset in perpetuity raises questions on the responsibilities and costs in offsetting.

<u>When is and should the offset be done in relation to the degradation (Decision on Time 7)?</u>

- Ideally, the offset should be ready before the degradation takes place. In practice this is possible to reach only if there is a portfolio of offsets that are produced in advance. The BioBanking scheme in New South Wales (described in more detail later) is an example of a system where biodiversity offsets can be produced upfront.

<u>Is the time discounting used if the offsetting happens after the degradation (Decision on Time 8)?</u>

- If compensation measures are done after the degradation happens, one way to minimize the loss is time discounting; an offset now is more valuable than an offset that will be realized in the future. Following this logic, compensating becomes more expensive if the time discounting is used (Kangas & Ollikainen 2019). This is one of the reasons why habitat banking or a compensation pool type of arrangement may reduce compensation costs.

How is additionality secured (Action 11)?

- Additionality (Section 2.1.2) is a tricky thing in offsetting. Some measures that could be used for nutrient offsetting are already done for other reasons, such as the fishing of species where fishing quotas are already regulated. The risk in failing to find additional measures to produce offsets is that compensations replace existing environmental responsibilities and nature conservation practices. It is important to meticulously assess the potential compensation measures from the additionality point of view.

Compensation measures, avoided loss and restoration (Actions 12, 13).

- The discussion on what is a more suitable compensation measure (avoided loss vs. restoration) is less relevant in nutrient offsetting than in biodiversity offsetting. In biodiversity offsetting avoided loss is, for example, a situation where a forest area is not used for timber but protected in perpetuity as an offset area. The concept of avoided loss is not generally used in nutrient offsetting where the aim is to improve or retain the current situation. Active measures to minimize the nutrient load or remove nutrients from the target area (Chapter 5) are appropriate in nutrient offsetting. Regardless of the chosen compensation measures, the effectiveness of offsets needs to be controlled and verified.

What is the baseline against which loss (and gain) are compared (Action 14)?

- For nutrient offsetting this is a slightly simpler question than for biodiversity offsetting. The water quality standards (WFD) are the basis for the baseline. Nevertheless, decisions need to be made on the time-point and location where the baseline is defined.

How to prevent or consider leakage (Action 15)?

- In ecological compensations, leakage refers to a situation where an offset is produced based on avoided loss, but the avoided loss-causing activity is transferred to some other location. The typical scenario for leakage, e.g. in forestry is one forest area being protected as an offset site and not used for timber production, but the need for timber staying the same and the "avoided" logging eventually taking place at some other location. The risk of leakage needs to be considered also in nutrient offsetting if the avoided loss is considered as an option to produce offsets. If potential leakage is not included in the estimation of avoided loss offsets, there is a risk of overestimating the benefits of offsetting.

There are also several other practical matters that must be decided in operationalizing offsetting. The examples in the following list are based on a Suvantola et al. (2018) delfoi-questionnaire and Primmer et al. (2019) workshops where stakeholders were asked questions on how compensations should be put to practice.

- What kind of contracts are needed between the offset producer and the offset buyer?
- Who verifies that compensations or offsets are adequate?
- Who keeps a record of produced offsets?
- Who estimates environmental harm (biodiversity loss or nutrient load) and the adequacy of compensation measures?
- Is an intermediary needed to bring together the offset buyer and seller in the compensation market?
- If so, who should be the intermediary?

- Who should manage compensations? Should it be the same administration that does environmental permitting or a separate, even a new entity as a part of the environmental administration?
- How are benefits, rights and liabilities shared in a just manner?

The list is not comprehensive but gives an idea of matters that may come up in offsetting. How these practical questions are solved varies depending on the compensation objectives, legal framework and the governance of the compensation system.

6.2 Compensation pools and biobanking

One option in organizing the supply of suitable offsets are the so-called compensation pools (SOU 2017), habitat banks or biobanks. The basic idea is that offsets (nutrient reductions, biodiversity increases) are sought for and produced beforehand and managed and marketed in a joint effort. Then the offsets are sold to those who need to compensate their actions. Different names are used for this type of system in biodiversity offsetting or ecological compensation literature, most commonly species, habitat or environmental banking. In the biodiversity offsets paper by OECD (2013), biobanks resemble the compensation pools of SOU (2017).

A framework on how these “kompensationspooler” could work for ecological compensation within Swedish governance system is described in a Swedish compensation report (SOU 2017, pp. 147–162). Three options on organizing ecological compensations in Finland and stakeholder views on these are described in the Suvantola et al. (2018) report. Both reports include summaries of international biodiversity offsetting programmes. The description and discussion on compensation pools in this report mainly utilizes reports by SOU (2017), Suvantola et al. (2018) and OECD (2013). Also, a thought provoker for this report has been the informal discussions and pondering of the Habitat Bank consortium[32] on how a broker-like habitat bank could function in Finland if biodiversity offsetting was voluntary. Part of the governance-related work of the Habitat Bank consortium is published in Primmer et al. (2019). A market analysis of ecological offsetting in Finland is published in Kangas & Ollikainen (2019).

Compensations can be done without an organized compensation pool as a direct exchange between a compensation buyer and a seller. This straightforward option can be a good choice, especially if there are very few compensation cases. Also, according to some views, the so called over-the-counter contracts could be a tempting choice for offsetting in the private sector, where both the buyer and seller are companies (Laininen & Matthies 2017). Companies make contracts with each other on various matters, and environmental offsets would not be overwhelmingly complicated for the business sector and from the contract point of view. If, however, the demand for compensations increases, the compensation pool has several advantages over one-on-one deals.

6.2.1 Advantages of a compensation pool

A compensation pool can work both in a voluntary or an obligation-based compensation market. The key point is that there is enough demand for compensations. Handling offsets in a combined fashion is expected to produce cost-efficiency to a compensation market if the compensation market is large enough. If there is very little demand for compensations, the costs of building and upholding habitat banks or compensation pools may exceed the potential profits, and the system becomes economically unsustainable. A legal obligation to compensate environmental damages creates demand for offsets. In most international examples on compensation pools, habitat banks or similar systems, there is some compensation obligation that guarantees the demand.

[32] More information on the Habitat Bank consortium: https://blogs.helsinki.fi/habitaattipankki/?lang=en

In previous sections of this report, both technical questions but also governance or social acceptance related challenges in biodiversity or nutrient offsetting have been brought up. Some of these may need to be solved on a case-by-case basis, but compensation pools can help in overcoming others. According to SOU (2017), the following benefits could be achieved with compensation pools:

1) Compensation pools could seek suitable offsets and implement restoration or other compensation measures proactively. The possibility to estimate compensation success increases if offsets are produced before the environmental degradation happens. Uncertainties related to technical, often ecological questions in producing the compensation can be solved in advance. When uncertainties are diminished, the compensation need also diminishes. The alternative to upfront-compensation is to counterbalance uncertainties in compensation success by overcompensation, as explained earlier in this report (see Section 2.1.3, compensation coefficients).

2) The quality of offsets is evaluated in advance. This increases the security of the system especially for the buyer but also for the seller. It may also decrease transaction costs.

3) The matching of compensation demand and offset supply is easier. At least in theory, a compensation pool can have a portfolio of different types of offsets. Usually, in the case of biodiversity offsetting, a portfolio could include offsets produced in different habitats and/or in various geographical locations. For nutrient offsetting, the portfolio could include offset areas located in different water areas and/or offsets produced using different methods (examples in Section 5).

4) Entering the compensation market becomes easier. With compensation pools it is possible for even small-scale offset producers to enter and participate in the compensation market. Of course, this depends on the organization and rules of the compensation pool.

5) A compensation pool can support regional planning. When offset areas, measures and locations are handled as a larger combination, it is easier to incorporate them in regional planning and, for example, consider competing or mutually beneficial interests in different areas.

6.2.2 Options for organizing a compensation pool

A compensation pool can be organized in several ways. The compensation pool can be a middleman or a broker, as envisioned in the Habitat Bank project, that brings together the offset need and the supply. The compensation pool can also have a more comprehensive role than being just a mediator.

The following list on potential, alternative or partly overlapping roles or functions of a compensation pool is based on SOU (2017):

1) A compensation pool produces nature values on their own or someone else's land (or water area), and can mediate, sell or buy offset areas or compensation measures.
2) A compensation pool acts as a caretaker who does not own the offset areas but is responsible for producing and, if necessary, maintaining the offsets. The pool can include one or several landowners (or owners of water areas). The profits are divided

based on contracts between the owners of the area where offset is produced and the compensation pool.

3) A compensation pool is a shared effort of several actors who own areas where offsets can be produced (usually landowners in biodiversity offsetting) and who provide areas and expertise for compensation.
4) A compensation pool mediates offsets or other compensation measures and/or facilitates contacts between offset producers and buyers.

It is worth to note that the list has been compiled for ecological compensation which mostly only considers terrestrial areas. In marine and coastal areas and in nutrient compensations, the “landowner” could include farmers who participate in producing nutrient offsets on their land areas but also “water owners”, meaning all potential actors who have ownership or other legitimate right to make decisions regarding the use of water areas that are relevant to the compensation process.

The potential roles or activities of a compensation pool described by SOU (2017) overlap to some extent. In the first option, the merchandise of the compensation pool could be both the already produced offsets and the know-how on how to produce offsets. In the second, the pool takes a larger responsibility on the long-term maintenance of the offsets. Some compensation measures need to be taken only once, but some others need to be repeated. An offset that needs long-term and recurrent management is likely to be more expensive than an offset that can be achieved with one-off measures. One of the technical challenges is solved if the compensation pool takes care of the offsets in perpetuity. The main difference between the second and the third option is the ownership of the land or, more generally, the area where the offset is produced. One co-operation benefit in the third option for the offset producers comes from sharing general expenses. The fourth option is closest to an offset-broker, where a compensation pool does not own the offset areas or take responsibility of carrying out compensation measures or maintaining offsets. These different types of compensation pools could also work together, like the broker helping the landowners in finding buyers for their offsets.

The compensation pool could be organized as a private sector consultant-type enterprise, as a foundation or as a loosely co-operative cluster of private offset producers working together. Regardless of how a compensation pool is organized or what activities it carries out, common rules and principles for offsetting are needed. Some control procedure or a certificate system is needed to secure that offsets deliver their promises, that is offsets produce the biodiversity values or reduce nutrient levels as promised.

Examples of joint efforts in offsetting

A private fund — Vattenkraftens Miljöfond (Sweden)

Hydropower companies in Sweden have together founded an environmental fund, called Vattenkraftens Miljöfond Sverige AB[33]. The idea of the fund is that the Swedish hydropower industry together through the fund covers expenses of the required environmental measures of individual hydropower installations in Sweden. The potentially fundable environmental investments should help to achieve national and international environmental targets in hydroindustry but also benefit fishing, tourism and local development. Thus, the fund is not limited to compensations or offsetting but considers environment, ecosystem services and social aspects more widely. Vattenkraftens Miljöfond will be one of the largest investors in the environmental sector in Sweden. The funding comes from the eight hydropower companies that operate in Sweden. The first round

[33] The homepage of the Swedish hydropower environmental fund: https://vattenkraftensmiljofond.se

of funding applications will open this year, in 2020. The guidelines and instructions for applying are already available on the fund website.

There are no examples of funded projects yet. The overall aim of the fund seems to be cost-sharing and possibly also knowledge-sharing of solutions that enable hydropower companies to reach their environmental and social responsibility targets.

Established biodiversity offsetting scheme — Biodiversity Banking and Offsets Scheme, BioBanking (New South Wales, Australia)

New South Wales in Australia has been one of the forerunners in biodiversity offsetting. The BioBanking offset scheme was launched in 2008, and the aim was to help to address the biodiversity loss due to habitat degradation and habitat loss. The concerning legislation was approved in 2006. The BioBanking Scheme has been replaced by the Biodiversity Offsets Scheme under the Biodiversity Conservation Act 2016 in 2017. The new scheme is also market-based and operates mostly in a similar way to the BioBanking. Main elements have been kept: the standardized metrics for measuring biodiversity loss and gain, permanent protection of the offset sites and possibility for the landowners to get monetary compensation for providing areas for offsets (Suvantola et al. 2018). The Biodiversity Offset Scheme brings together the offset seller and the offset buyer. There are several actors in the BioBanking: the public sector that sets the rules and guidelines and verification criteria, an offset buyer who needs to meet the criteria of the Biodiversity Offset Scheme, an offset seller who also needs to meet relevant eligibility criteria, a conservation fund and also private consultants. The New South Wales BioBanking system seems well established, but has been underused, which is one of the reasons for its recent reform.

The role of the landholders (offset producers, sellers) is described as follows on the BioBanking information website: "*Landholders can establish Biodiversity Stewardship Agreements to create offset sites on their land to generate biodiversity credits. These credits are then available to the market for purchase by developers, landholders or the Biodiversity Conservation Trust to offset the impacts of development or clearing. Sufficient funds are held to support the long-term management of the biodiversity stewardship sites*."

A Biodiversity Stewardship Agreement is done between the conservation fund and the offset producer. The agreement includes a management plan, with annual management actions and permanent expenses for 20 years. These sum up to a Total Fund Deposit. The agreement is registered by the local environmental officer and real-estate register. The landowner can either find the buyer themselves, use a broker or use public register kept by the local environmental officer. For the offset seller there are several options for selling the offset: they can 1) sell offset credits either to the Biodiversity Conservation Trust or a private purchaser, 2) transfer the Total Fund Deposit to the Biodiversity Conservation Trust's Stewardship Payments Fund using the Biodiversity Offsets and Agreements Method System (BOAMS) or 3) transfer the ownership of the credits to the buyer using BOAMS. As the BioBanking information page states: "*The landholder is only likely to sell the credits at a price that allows them to recoup the full Total Fund Deposit amount*"[34]. The offset can also be bought by someone willing to do voluntary compensations.

A private sector enterprise — the Environment Bank (UK)

The Environment Bank Ltd[35] is an example of a consultant type company that works as an intermediator between offset supply and need. The Environment Bank was established in 2006, and it

34 Detailed information on how the Biodiversity Offset Scheme works: https://www.environment.nsw.gov.au/topics/animals-and-plants/biodiversity/biodiversity-offsets-scheme/how-it-works
35 Homepage of the Environment Bank: https://www.environmentbank.com/

operates in the United Kingdom. According to the listing on their website, the services they provide include 1) biodiversity accounting, 2) bespoke offset research, 3) offset delivery, 4) habitat banks, 5) planning advice and 6) biodiversity metrics. The metrics are based on the Defra metrics, where the condition and distinctiveness are used in estimating the overall quality of a given habitat (DEFRA 2012). The offset delivery service also includes management, regular monitoring and reporting through the agreed scheme term. The services are marketed for developers, planning authorities and landowners.

Information sharing platform — Speciesbanking.com (USA, potentially global)

The mitigation and compensation of wetland degradation or loss of endangered species or their habitats has been possible for some time. The approaches and legislation vary between states. To increase openness and help offset buyers and sellers to find each other, the Ecosystem Marketplace[36] launched in 2011 the SpeciesBanking.com website, which was described as "*an online information hub for bankers, buyers, researchers, and regulators*". Currently the website is not active, but it was operational at least for some time and produced at least material for research (see, e.g. Pawliczek & Sullivan 2011).

The relevance of the previous examples on different ways of organizing biodiversity offsetting and ideas presented in SOU (2017) for nutrient offsetting in Åland:

- If there are only very few compensation cases and they occur rarely and/or are small-scale projects, it is not likely that it is profitable to start new business enterprises that focus solely on nutrient offsetting.

- Nevertheless, there is a business opportunity in nutrient offsetting for already existing environmental consultant companies even if the market were to remain small. For example, knowhow in techniques on how offsets can be produced, matching offset supply and demand, environmental measurements, mapping and monitoring that are needed in the compensation process can potentially be handled by private sector actors.

- A private fund could steer funding for offsetting costs, as in the case of hydropower companies covering environmental responsibility actions together in Sweden. The underlying idea in (biodiversity) offsetting is like the polluter pays principle: the one who causes the harm also pays for the offsetting. Still, there is no reason why the extra costs that offsetting is likely to cause could not be covered jointly by those actors who benefit (economically) from the activity that causes the environmental degradation.

- There are several options for the offset producers to work together and enter the compensation market. Co-operation can decrease expenses in offset production.

- The control and verification of offsetting success is crucial for the success of the whole offsetting scheme. Long enough monitoring is necessary to verify compensation success.

36 More on Ecosystem Marketplace, see: https://www.ecosystemmarketplace.com/

- Building a comprehensive system to govern and enable the offsetting process may take up to ten years as in the New South Wales case. Starting with small steps and enabling flexibility and self-correction in the process is recommended by Suvantola et al. (2018) for biodiversity offsetting.

- An easily accessible, open platform on existing offsets and on-going projects helps both in sharing information on potential compensation measures and in learning where they could be used. Openness to the general public can also increase social acceptance of the compensations.

A field near Ämnäsviken. Photo: Annica Brink

7 Conclusions

Developing an operational nutrient offsetting system to reduce the impacts of human activities on the Baltic Sea marine ecosystem requires the consideration of several aspects related to the legislation, the carrying capacity of the marine environment and the possibility to develop an economically viable offsetting system. Furthermore, to obtain social acceptance the offsetting system needs to gain the trust of relevant stakeholders and general public.

Developing a functional, large scale offsetting scheme can take several years. However, it is not necessary to build the whole system at once, but in small steps, so that the system can be revised and modified along the way.

7.1 Recommendations from the legal point of view

Based on the legal analysis above, we present the following recommendations to be considered when developing legislation on nutrient offsetting in the Åland Islands:

1) Start with EU law. First and foremost, the need for nutrient offsetting stems from the environmental objectives of the Water Framework Directive. However, the Directive also provides wide leeway for national legal solutions including compensation measures if the status of a water body does not deteriorate and the good status can be achieved. The Directive does not prevent using nutrient offsetting as a measure to allow the permitting of projects. Take the objectives of the Marine Strategy Directive also into account.

2) Keep it simple. Simplicity in regulatory frameworks allows the applicability of nutrient offsetting. The legal boundaries and possibilities for nutrient offsetting remain unclear in many countries since it has not been specifically allowed (Finland, Sweden) or because the legal system is too complex in that regard (Sweden, USA, Åland Islands). The legislator's intentions and exemplifying the concrete application of the rules for certain situations can be further explained in a government bill commentary.

3) If different compensation mechanisms (compensation, improvement, joint improvement) are included in the Act, they need to be simple and straightforward. One option is to use only one type of measure in addition to the avoidance and mitigation of environmental impacts – it is a challenge to have both compensation and improvement measures in the same package. This may eventually lead to unclarity and difficulties in applying the legal rules.

4) Set clear definitions. It is useful to define what nutrient compensation or offsetting means exactly and how it can be implemented in a permit process. Also, the share of the benefit that may be utilized as a compensation, e.g. such as in the current act 2/3, is a policy and law matter – it should not depend on *in casu* assessments and calculations.

5) Use a planning or market instrument to support nutrient offsetting. A permit process concentrates on a single project. It is difficult and time-consuming to try to analyze the nutrient abatement capacity of other activities or measures taken outside the area of operation in the process. Thus, nutrient offsetting measures in a permit decision need

to be supported by planning instruments such as the programme of measures or there needs to be market-based solutions where nutrient credits (or offsetting measures) can be bought.

6) Lessons learned from the USA; i) trading within a "cap" set for a large area (such as the different states' portion of the entire Chesapeake Bay TMDL) may not give rise to environmental benefits, ii) too complex a regulatory setting may inhibit actors from entering into a credit system, iii) setting a quantitative cap on the basis of the quality standard "good ecological status" aids the permitting authority in determining whether deterioration or jeopardization occurs.

7.2 Measures for ecologically sustainable nutrient offsetting

There is a variety of actions that potentially could produce nutrient offsets. Based on the review on scientific literature, data on pilots and case studies, we suggest that removing nutrients by harvesting common reed should be further studied and has potential as an offset measure.

Removing nutrients by fishing or binding excess nutrients to the sea sediment are both actively piloted. More data is needed on their overall effects. For example, though there is even economic potential in removing nutrients by fishing non-commercial fish stocks, there is also difficulty in verifying how large and where the actual nutrient removal takes place. In addition, the manipulation of fish stocks can cause unexpected changes in ecosystem functioning.

General principles

To be effective, the offsetting should target the premises of eutrophication by decreasing the amount of limiting nutrient(s) during the growth period (April-September). It should be long-term in its effect, and the effect should be calculable. Possible ecosystem effects should be known and evaluated before the offsetting.

Sea-based measures

The variability of sea-based measures ranges from those directly targeting nutrient (mostly phosphorus) dynamics to a range of biomass-based measures. Their efficiency requires throughout study, and food web effects especially must be known before the measures are feasible. In some measures the question of local vs. the Baltic Sea in general must be considered as, e.g. fishery takes place largely in the wide-open sea areas, and the offsetting is optimally considered to affect the local environment.

Land-based measures

Land-based measures target the actual loading of nutrients. If effective, this is a very positive approach as nutrient loading is the basis of the eutrophication problem. However, land-based measures mostly affect the coastal areas near large rivers and the inner archipelago. They are not efficient in offsetting the local nutrient loading outside these restricted coastal areas.

7.3 Co-operation and guidelines are needed in offsetting

The whole offsetting process can be managed by the public or the private sector, but the best outcome may be reached by co-operation. Governance, rules and guidelines are needed, but the implementation may be carried out by the private sector. A large enough demand is needed if the compensation system relies on the activity of private sector consultants or mediator companies. The decision on who pays the costs of nutrient offsetting is a political one. If the costs are shared, a private fund is one potential option to collect and share assets to cover the costs of compensation measures. If the nutrient offsetting scheme or programme is built on the polluter pays principle, there are several options how the offset demand and the supply can be brought together. Co-operation between offset suppliers increases knowledge sharing and decreases the expenses of offset production.

For the success of the compensation scheme, the control and verification of offsetting are very important. In many cases, long-term monitoring is needed to verify if the compensation measure delivers the planned benefits. Patience and planning are needed in building a comprehensive system to govern and enable the offsetting. When planning and establishing nutrient compensation, it is good to learn from already existing systems, proceed with small steps and enable flexibility and self-correction in the process. Openness to the general public and a close co-operation with key stakeholders are also important as they can increase the social acceptance of compensations.

Photo: Petra Pohjola, Metsähallitus

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