GES REG

Indicators of the Good Environmental Status of food webs in the Baltic Sea

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GES-REG project final report on food web indicators, September 2013

1. Introduction

Laura Uusitalo, SYKE

The European Union's Marine Strategy Framework Directive (2008/56/EC) (MSFD), and the accompanying Commission Decision (2010/477/EU) set requirements for the assessment of Good Environmental Status (GEnS)[1] in the European marine areas. The directive lists 11 descriptors; descriptor 4 addresses marine food webs:

> *(4) All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.*

The Commission Decision elaborates this by defining three criteria (4.1, 4.2 and 4.3), each with a related conceptual indicator (4.1.1, 4.2.1 and 4.3.1). The third criterion (4.3) includes in addition to indicator 4.3.1. six suggested indicator topics not specified in detail:

> *Descriptor 4: All elements of the marine food webs, to the extent that they are known, occur at normal abundance and diversity and levels capable of ensuring the long-term abundance of the species and the retention of their full reproductive capacity.*
>
> *This descriptor concerns important functional aspects such as energy flows and the structure of food webs (size and abundance). Additional scientific and technical support is required, at this stage, for the further development of criteria and potentially useful indicators to address the relationships within the food web* (15)*.*
>
> *4.1. Productivity (production per unit biomass) of key species or trophic groups*

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[1] Following recommendations by Borja et al. (in press), instead of the widely used abbreviation GES we use the abbreviation GEnS for Good Environmental Status. This is to avoid confusion between the Good Environmental Status of the MSFD and the Good Ecological Status (also commonly abbreviated to GES) of the EU Water Framework Directive."

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To address energy flows in food webs, adequate indicators need to be developed further to assess the performance of the main predator-prey processes, reflecting the long-term viability of components in the part of the food web that they inhabit, based on the experience in some sub-regions in selecting appropriate species (e.g. mammals, seabirds).

— Performance of key predator species using their production per unit biomass (productivity) (4.1.1).

4.2. Proportion of selected species at the top of food webs

To address the structure of food webs, size and abundance of components, there is a need to assess the proportion of selected species at the top of food webs. Indicators need to be further developed, based on the experience in some sub- regions. For large fish, data are available from fish monitoring surveys.

— Large fish (by weight) (4.2.1).

4.3. Abundance/distribution of key trophic groups/species

— Abundance trends of functionally important selected groups/species (4.3.1).

It is necessary to identify changes in population status potentially affecting food web structure. Detailed indicators need to be further specified, taking account of their importance to the food webs, on the basis of suitable groups/species in a region, sub-region or subdivision, including where appropriate:

— groups with fast turnover rates (e.g. phytoplankton, zooplankton, jellyfish, bivalve molluscs, short-living pelagic fish) that will respond quickly to ecosystem change and are useful as early warning indicators,

— groups/species that are targeted by human activities or that are indirectly affected by them (in particular, by-catch and discards),

— habitat-defining groups/species,

— groups/species at the top of the food web,

— long-distance anadromous and catadromous migrating species,

— groups/species that are tightly linked to specific groups/species at another trophic level.

These criteria and indicators are based on the report by Rogers *et al.* (2010), which also includes background and justification for the selected approaches.

1.1 Structure of the report

In this report we first identify the key elements of the northern Baltic Sea and analyse the pressures and risks on the selected central food web components: based on scientific research, which pressures are likely to be the most crucial on affecting the central food web components?

Secondly, we review the food web indicators that exist or have been proposed in the Baltic Sea: the proposed core indicators of the HELCOM CORESET project (HELCOM 2012, 2013) that can be suitable as food web indicators, and food web indicators identified or proposed in the Marine Strategies of the Member States (downloadable from http://cdr.eionet.europa.eu/recent_etc?RA_ID=608). We catalogue these indicators and their properties (Annex 1), analyse the gaps in the indicator coverage, and make note of similarity and dissimilarity of the indicators across the Member States.

We focus on questions deemed especially challenging, or in need of development in the Baltic Sea, and therefore chosen as foci in the GES-REG project. These include a way to assess energy flows in the food web based on stable isotope analysis; the questions about the large fish indicator, by-catch and discards, and indicators related to zooplankton and phytoplankton, which have fast turnover rates and can therefore potentially give early warning signs of the ecosystem change.

2. Analysis of pressures and risks related to descriptor 4 – marine food webs

Laura Uusitalo and Heidi Hällfors, SYKE; Pekka Jounela and Eero Aro, RKTL

2.1 Introduction

The aim of this chapter is to identify the key elements of Baltic food webs, and the pressures and risks that can be considered relevant to these species as the basis for further indicator development.

The approach of this assessment is to

1. identify the key food web components of the northern Baltic Sea

2. map how pressures and impacts, as defined in Table 2 in Annex III of the Marine Strategy Framework Directive, affect these food web components, by identifying observed changes in the food webs

This work is based primarily on scientific publications. It has to be noted that no final, established truth about the Baltic Sea food webs exists; we have focused on the food web elements that are uncontroversial and commonly considered important.

2.2 Key food web components of the Baltic Sea

Mammals

The Baltic Sea hosts four marine mammal species: the grey seal, ringed seal, harbour seal, and the porpoise (Furman *et al.* 1998). The harbour seal and porpoise populations are small, and cannot be considered to have major impacts on the food webs, whereas the population size of grey seals and ringed seals in the beginning of the 20th century (88 000–100 000 and 190 000–220 000, respectively; Harding and Härkönen 1999), had a strong effect on the food web structure (Österblom *et al.* 2007). In 2011 approximately 24 000 grey seals were counted in the Baltic Sea (RKTL 2012). According to HELCOM, there are at present about 4 000 ringed seals in the Gulf of Bothnia, 200–300 in the Gulf of Finland and about 1 400 in the Gulf of Riga (http://www.helcom.fi/environment2/biodiv/seals/en_GB/ringed/). The species is also found in the eastern Baltic Proper, and in small numbers in the Archipelago Sea. The seals feed on available fish species, such as Baltic herring (*Clupea harengus membras*), smelt (*Osmerus eperlanus*), three-spined stickleback (*Gasterosteus aculeatus aculeatus*) (Routti *et al.* 2005, Suuronen and Lehtonen 2012).

Birds

The coasts, islands and islets of the Baltic Sea host a wide variety of birds. The outer archipelago hosts for example several species of terns, gulls, warders, and ducks, while the inner archipelago and coast host even wider variety of birds feeding from the sea (Furman *et al.* 1998). Common eider (*Somateria mollissima*), a common species breeding on offshore islands, can be considered as a keystone species (HELCOM 2010). It feeds mainly on blue mussel.

Fish

The dominating fish species in the Baltic Sea pelagial/benthic ecosystem are cod (*Gadus morhua*), sprat (*Sprattus sprattus*) and Baltic herring (*Clupea harengus membras*), forming approximately 80% of the total fish biomass (Elmgren 1984 ref. Österblom *et al.* 2007; Thurow 1984 ref. Österblom *et al.* 2007). Cod is a benthic predatory species that feeds on benthic meio- and macrofauna and fish, such as juvenile herring and cod, and juvenile and adult sprat (e.g. Uzars 1994, Harvey *et al.* 2003).

Small herring and all size classes of sprat are strictly zooplanktivorous, and the copepods *Pseudocalanus elongatus, Temora longicornis,* and *Acartia* spp. are considered to be their main zooplankton prey items (e.g. Szypula *et al.* 1997a, Flinkman *et al.* 1998, Möllmann and Köster 1999, 2002, Viitasalo *et al.* 2001, Casini *et al.* 2004, Rönkkönen *et al.* 2004, Casini *et al.* 2006). Larger herring feed also on nektobenthos, i.e. *Mysis mixta*, amphipods and polychaetes (Casini *et al.* 2004, Möllmann *et al.* 2004).

Herring is also an important zooplanktivore in the coastal ecosystems (Hansson *et al.* 1990). There is a smaller body of research on the roles of the Baltic Sea coastal fish species in the food webs, and as coastal systems vary spatially, common features of coastal food webs are harder to identify.

Invertebrates

In the coastal community, blue mussel (*Mytilus trossulus × M. edulis*; Väinölä and Strelkov 2011) is one of the key species of the food web (HELCOM 2010). They form dense beds on the shallow shores of the Baltic Sea, and provide a number of resources for over 40 associated macrofaunal species. Changes in the structure of this community have the potential to alter the assemblages and diversity of the related fauna (Koivisto 2011).

In the open-water ecosystem, copepods such as *Pseudocalanus elongatus, Temora longicornis,* and *Acartia* spp. are key species due to their importance as planktivorous fish food. They themselves are omnivorous, feeding on algae, and small heterotrophic organisms (e.g. Gasparini and Castel 1997, Setälä *et al.* 2009).

Primary producers and microbes

The majority of energy entering the Baltic Sea food webs comes from planktonic algae, i.e. phytoplankton (Furman *et al.* 1998). The bacteria that decompose soluble organic matter, ensuring efficient recycling of energy, are also crucial especially in the open sea ecosystem (Furman *et al.* 1998). Ratios of various phytoplankton groups, such as the dinoflagellate to diatom ratio or the proportion of cyanophytes to the whole phytoplankton community, have been suggested to reflect ecosystem state and the quality of the phytoplankton community as food for zooplankton. No single phytoplankton species can, however, be named as a keystone species.

In addition to providing a major part of the energy available for higher trophic levels, there is another aspect to phytoplankton community that can be highly relevant to food web functioning: toxins. Several toxin-producing species occur commonly in the northern Baltic Sea (Hällfors 2007). Because of their key position as primary producers, the effects of phytoplankton and their toxins on the Baltic Sea food web has been investigated intensively for the past 15 years or so (e.g. Koski 1999, Sipiä 2001, Engström-Öst 2002, Kozlowsky-Suzuki 2004, Karjalainen 2005, Uronen 2007, Sopanen 2009). Although current knowledge suggests that the transfer rate of phytoplankton toxins through the food web is low (Karjalainen *et al.* 2005, 2007, Setälä *et al.* 2011), toxic phytoplankton are considered a potential risk for co-occurring organisms, as well as for high-trophic-level consumers through toxin bioaccumulation in the food web (cf. Kuuppo *et al.* 2006, Sipiä *et al.* 2006, Setälä *et al.* 2009, Hakanen *et al.* 2012). Dinoflagellate and/or cyanophyte toxins have been found in e.g. copepods (Lehtiniemi *et al.* 2002, Setälä *et al.* 2009, Sopanen *et al.* 2011), bivalves (Pimiä *et al.* 1997, Sipiä *et al.* 2001), flounder and roach, as well as eider (Sipiä *et al.* 2006). The immediate effects of toxins vary from reduced feeding and growth rates in fish larvae exposed to cyanophyte toxins (Karjalainen *et al.* 2007), to mortality in copepods (Sopanen *et al.* 2008) and fish (Lindholm and Virtanen 1992) exposed to prymnesiophyte toxins.

Habitat-forming species

In the coastal zone, bladder wrack algae (*Fucus spp.*) and eelgrass (*Zostera marina*) are among the keystone species of the coastal zone (HELCOM 2010), forming habitats for various species of fish and invertebrates and hence enabling the functioning of the coastal food webs. The bladder wrack algae require hard substrata and salinity above 4 (PSU, Practical Salinity Units), although isolated, sparse populations have been reported in salinities down to 2 (Martin 2012). *Fucus* belts play an important structuring role in the coastal ecosystems, forming habitats for species-rich epiphytic and epibenthic communities (e.g. Haage 1975, 1976; Kautsky and Kautsky 1989; Wallentinus 1991 ref. Martin 2012), as well as shelters and feeding habitats for fish.

Eelgrass meadows typically occupy shallow soft bottoms in estuaries and coasts, usually at the depth of 2–4 m. They require salinity of approximately 5. *Zostera* meadows stabilise the substrate and create habitats for various benthic and fish species, including nursery habitats for commercially exploited species (Möller 2012).

Based on these considerations, we discuss the species/groups presented in Table 2.1. We do not intend to imply that the selected organisms are a complete representation of Baltic Sea food webs; rather, they constitute examples of key elements. If some of these species/groups undergo major changes due to human-induced pressures, the effects are likely to be seen in other parts of the food web as well.

Table 2.1. *The species/groups discussed in this report.*

Species	**Group**	**Habitat**	**Prey to**	**Preys on**
Grey seal	mammal	archipelago		fish
Common eider	bird	archipelago		blue mussel
Cod	fish	benthic, pelagial	seals, cod, sprat (as eggs/larvae)	benthic fauna, fish
Herring	fish	coastal, pelagial	seals, birds, fish	zooplankton, nektobenthos
Sprat	fish	pelagial	seals, birds, fish	zooplankton
Blue mussel	invertebrate	coastal zone	birds, fish	plankton, detritus
Copepods (*Pseudocalanus*, *Temora*, *Acartia*, *Eurytemora*)	zooplankton	open water	fish	phytoplankton, small-sized zooplankton
Phytoplankton (diatoms, dinoflagellates, cyanophytes)	phytoplankton	open water	zooplankton, filtering macroinvertebrates	*)
Bladder wrack	macrophyte	coastal zone	*habitat-forming species*	
Eelgrass	macrophyte	coastal zone	*habitat-forming species*	

*) In this report we focus on the role of phytoplankton as primary producers; as food for others. Heterotrophic and mixotrophic species belonging to e.g. dinoflagellates are however traditionally counted along with photosynthetic species in phytoplankton analyses. These feed on bacteria and other phytoplankton.

2.3 Effects of pressures and impacts on the identified elements

In this section, we focus on the pressures and impacts on marine food webs mentioned in the Marine Strategy Framework Directive (Table 2.2). The adopted approach was to review literature for results showing or suggesting effects of these pressures on the selected food web components (Table 2.1). No unpublished expert assessment was used at this time. In addition to the anthropogenic MSFD pressures, also the effects of climatic and hydrological factors are noted if they are identified in the literature as main factors affecting the species.

Table 2.2. Pressures and impacts on the marine environment according to the Marine Strategy Framework Directive (Table 2, Annex III).

Table 2 of MSFD. Pressures and impacts.	
Physical loss	— Smothering (e.g. by man-made structures, disposal of dredge spoil),
	— sealing (e.g. by permanent constructions).
Physical damage	— Changes in siltation (e.g. by outfalls, increased run-off, dredging/disposal of dredge spoil),
	— abrasion (e.g. impact on the seabed of commercial fishing, boating, anchoring),
	— selective extraction (e.g. exploration and exploitation of living and non-living resources on seabed and subsoil).
Other physical disturbance	— Underwater noise (e.g. from shipping, underwater acoustic equipment),
	— marine litter.
Interference with hydrological processes	— Significant changes in thermal regime (e.g. by outfalls from power stations),
	— significant changes in salinity regime (e.g. by constructions impeding water movements, water abstraction).
Contamination by hazardous substances	— Introduction of synthetic compounds (e.g. priority substances under Directive 2000/60/EC which are relevant for the marine environment such as pesticides, antifoulants, pharmaceuticals, resulting, for example, from losses from diffuse sources, pollution by ships, atmospheric deposition and biologically active substances),
	— introduction of non-synthetic substances and compounds (e.g. heavy metals, hydrocarbons, resulting, for example, from pollution by ships and
	— introduction of radio-nuclides.
Systematic and/or intentional release of substances	— Introduction of other substances, whether solid, liquid or gas, in marine waters, resulting from their systematic and/or intentional release into the marine environment, as permitted in accordance with other Community legislation and/or international conventions.
Nutrient and organic matter enrichment	— Inputs of fertilisers and other nitrogen — and phosphorus-rich substances (e.g. from point and diffuse sources, including agriculture, aquaculture, atmospheric deposition),
	— inputs of organic matter (e.g. sewers, mariculture, riverine inputs).
Biological disturbance	— Introduction of microbial pathogens,
	— introduction of non-indigenous species and translocations,
	— selective extraction of species, including incidental non-target catches (e.g. by commercial and recreational fishing).

Grey seal

In the beginning of the 20th century the estimated grey seal population in the Baltic Sea was 88 000–100 000 individuals (Harding and Härkönen 1999). In the 1960s the population collapsed mainly due to reproductive failure caused by poisonous substances and due to state-subsidised hunting. By 1976 the grey seal population size was approximately 2 000–3 000 individuals and hence, the financial compensation for killing a seal was abolished and hunting was finally banned in 1982.

During the 1990s the improved fertility contributed to an increase in grey seal population size. The improved reproductive health was caused by a decreased prevalence of uterine obstructions due to a decline of the chlorinated substance concentrations (Jensen *et al.* 1969, Almkvist 1978, 1982, Bergman and Olsson 1986, Kokko *et al.* 1997, Bergman 1999, Harding and Härkönen 1999, Bäcklin *et al.* 2003, Nyman *et al.* 2003). Thus, since 1999 in Finland and since 2001 in Sweden, grey seals were again legally hunted. The annual hunting quotas were about 1 000 in Finland (including Åland) and 180–230 in Sweden and about 50% of quota were used (Anon. 2007).

Another important human-related cause of death may be drowning in fishing gear. The extent of grey seal by-catch, however, is not well known. It has been suggested that together with hunting, it may have a marked effect on the population size (Lunneryd and Westerberg 1997, Harding *et al.* 2007, Bäcklin 2011). The current population size of approximately 24 000 individuals is considered sustainable although variable opinions exists on the reference population levels based on target, limit and precautionary approach (HELCOM 2011).

The liver parasite (*Pseudamphistomum truncatum*) has been found in an increased proportion of grey seals in a Swedish study in the years 2007-2008 compared to previous years (Bäcklin *et al.* 2010). Cyprinid fish act as intermediary hosts of this parasite, and it is possible that changes in the fish community may have contributed to the increase of the parasite in seals.

The noise caused by wind farms has been suggested to cause stress to the seals and possibly have negative effects on their sound-based hunting, orienting, and communication (Vehanen *et al.* 2010).

No negative effects have been attributed to the increased concentrations of heavy metal contaminants in the Baltic grey seal population (Bergman *et al.* 2001).

Common eider

The Baltic/Wadden Sea flyway common eider population increased steadily from the 1970s to 1990s, but between 1991 and 2000 the wintering population decreased by about 36% (Desholm *et al.* 2002). The reasons for the decline are unclear, but possible causes include the increase in numbers of American mink, a predator of common eider (Desholm *et al.* 2002), lead poisoning (Franson *et al.* 2000, 2002, Hollmén 2002 ref. Desholm *et al.* 2002), by-catch in gill-nets (Desholm *et al.* 2002), as well as collisions with man-made structures and high-speed vessels (Desholm *et al.* 2002). Decreasing salinity has also been suggested as a reason for the drop in the common eider population in the Gulf of Finland (Westerbom 2006), the mediating factor being common eiders' main food item, blue mussel, which lives at the edge of its salinity tolerance in the area.

Cod, herring, and sprat

The Baltic Sea open-water fish community composition has undergone changes in the last century: the previously cod-dominated community is now dominated by the clupeids sprat and herring (Österblom *et al.* 2007). The decline of the cod populations has been attributed to too high fishing mortality levels, combined with poor reproduction success of cod due to the decreased salinity and oxygen concentrations in the spawning areas, resulting in poor recruitment of cod (Aro 2000). To survive, the Baltic cod eggs need to develop in water with salinity > 11, temperature > 1.5 °C, and dissolved oxygen > 2 ml/l (Plikshs *et al.* 1993, 1999, MacKenzie *et al.* 2000). Due to hydrographic conditions and increased oxygen consumption as a consequence of eutrophication, the water volume in which these criteria are met (the reproductive volume) has decreased.

The Bornholm Basin has traditionally been the most important area for cod reproduction, while the Slupsk Furrow, Gdansk Deep and the southern Gotland Basin each regularly contributed about half the size of the reproductive volume observed in the Bornholm Basin. The central Gotland Basin is often oxygen depleted, but it has contributed a volume equivalent to that of the Bornholm Basin in some years, and may on those occasions be an important habitat (Plikshs *et al.* 1999).

The reproductive volume has fluctuated strongly. However, it was on average much higher before 1980 than thereafter. In 1985–1992, the Bornholm Basin represented almost the only suitable area for cod reproduction. In the most recent decade the reproductive volume has fluctuated widely.

In recent years, however, the combination of an increasing cod stock and a lack of prey in the main cod distribution area have resulted in locally high predation mortality of forage fish and in cannibalism of cod (Eero *et al.* 2012). While the cod abundance in the southern Baltic Sea (Bornholm Basin) is currently the highest recorded since the 1970s, the biomasses of sprat and herring, major prey for adult cod, are at historic lows in the area. Consequently, the body weight and nutritional condition of cod has drastically declined because the majority of clupeids are currently found in the northern areas of the Baltic Sea.

According to the present knowledge of cod and seal interactions, seal predation has been found to have a much lower impact on cod recovery, compared to the effects of fishing and salinity (MacKenzie *et al.* 2011). These results suggest that the recovery of both seal and cod populations is realistic but success in achieving these goals will also depend on how climate change affects cod recruitment.

Changes in clupeid (herring and sprat) nutritional conditions and weight-at-age have also been observed in the Baltic Sea. Food availability is commonly agreed to be the main determinant of clupeid condition in the Baltic Sea, and changes in the abundance and structure of zooplankton has been attributed the reason of changes in clupeid weight-at-age (e.g. Flinkman *et al.* 1998, Cardinale *et al.* 2002, Rönkkönen *et al.* 2004, Casini *et al.* 2006). Casini *et al.* (2006), on the other hand, point out that clupeid total biomass seems to be the best explaining factor for clupeid condition, i.e. weight compared to length.

Cod has been found to react to the increase of anoxic habitats by shifting from a benthic lifestyle and feeding to inhabiting intermediate water layers and feeding on pelagic prey, such as crustaceans and sprat (Uzars, 1994).

Changes in fish stocks have cascading effects: for example, reduction in the fledging body mass of common guillemot (*Uria aalge*) chicks has been attributed to a parallel drop in the condition factor of its main prey item, sprat (Österblom *et al.* 2001). It has also been observed that the M74 syndrome of Baltic Sea salmon, which causes high mortality of yolk sac larvae, is more common when the diet of adult salmon contains a high amount of sprat (Mikkonen *et al.* 2011).

Blue mussel

The blue mussel is a marine species, and changes in salinity are among the main threats to its populations near the borders of its distribution range in the Baltic Sea, and the size, density and biomass of blue mussels increase as the salinity increases (Westerbom 2006, Westerbom and Jattu 2006, Koivisto 2011). In addition to salinity, wave exposure is critical to blue mussel distribution (Westerbom and Jattu 2006). Other factors affecting the blue mussel distribution and population dynamics negatively are eutrophication, siltation due to construction work, oil accidents, and invasive species (Kostamo and Ekebom 2012).

Copepods (*Pseudocalanus, Temora, Acartia, Eurytemora*)

Changes in the biomasses of the copepod genera *Pseudocalanus*, *Temora*, *Acartia,* and *Eurytemora* can be explained by climatic/hydrographic factors: *Pseudocalanus* biomasses decrease due to decreasing salinity, while the biomasses of the other copepod species increase with increases in temperature (Möllman *et al.* 2000, Rönkkönen *et al.* 2004, Suikkanen *et al.* 2013). This shift in the species composition is suggested as a reason for decreasing growth rates of clupeids; but also, that increased clupeid biomasses were at least a partial reason to low mesozooplankton biomass in the 1990s.

Recently, Bickel *et al.* (2011) found that boat-generated turbulence caused copepod mortality. They suggest that this anthropogenic disturbance may be an important source for zooplankton mortality in areas with heavy boat traffic (such as the Gulf of Finland), to the degree that it potentially alters trophic interactions in the pelagial (Bickel et al. 2011).

Phytoplankton

Climatic/hydrographic processes and anthropogenic eutrophication form the major drivers of long-term changes in the phytoplankton community in the northern Baltic Sea (Suikkanen *et al.* 2007, 2013, Fleming-Lehtinen *et al.* 2008, Hällfors *et al.* 2013a). A significant increase in phytoplankton biomass has been observed since the 1970s (Suikkanen *et al.* 2007, 2013, Fleming-Lehtinen *et al.* 2008), and changes in the species composition have occurred both on decadal (Suikkanen *et al.* 2007, 2013) and centurial (Hällfors *et al.* 2013a) scales. The changes in community composition vary somewhat with investigated area and period, but in recent decades, of the three (by biomass) most important phytoplankton groups, cyanophytes have increased, while the abundances of dinoflagellates and diatoms displayed alternating oscillations (cf. Suikkanen *et al.* 2007, 2013, Jaanus *et al.* 2011, Wasmund *et al.* 2011).

Several invasive alien phytoplankton species occur in the Baltic Sea; one of these, the dinoflagellate *Prorocentrum minimum*, has been identified as an alien causing a recognizable effect to its environment (Olenina *et al.* 2010). During blooms it may form up to 98% of phytoplankton biomass, a level of dominance which constitutes a major change to the structure of the phytoplankton community (Olenina *et al.* 2010).

Ilus and Keskitalo (2008) noted that the thermal effluents of power plant cooling waters can cause phytoplankton biomasses to increase and the species dominance to change. However, these changes are likely to remain local.

Bladder wrack algae

Light is the main factor affecting the distribution of macroalgae, and also the depth limits of bladder wrack algae (*Fucus*) have been shown to correlate well with light attenuation in the Baltic Sea (Kautsky 1999, Bäck and Ruuskanen 2000). The depth limits of *Fucus* have moved up in the 20th century, and the development is attributed to eutrophication, causing turbidity and decreasing the depth to which light penetrates in the water (Torn *et al.* 2006).

Eelgrass

The requirement of light penetration to the bottom is concerns eelgrass as well. The growth depth and distribution of eelgrass has decreased in many parts of the Baltic Sea (Möller 2012), and the change is attributed to eutrophication (Boström *et al.* 2002). On the other hand, a positive food web change is also attributed to eutrophication in the eelgrass meadows: faunal changes indicate increased food availability for animals feeding in these habitats (Boström *et al.* 2002).

2.4 Discussion

While changes in the salinity and thermal regime were identified as the factors behind many changes in the Baltic Sea food web, they fall outside the impacts and pressures as listed in the MSFD (Table 2.2): they are due to large-scale climatic/hydrological changes rather than changes that are due to clearly defined construction work, power plants, etc.

The results reflect the common understanding that eutrophication is the major human-induced threat for the Baltic Sea ecosystem. Partly these results may also reflect a bias: eutrophication may also be over-represented in these results since it has been intensively studied in the past decades.

The results indicate that physical damage, physical loss, or man-made changes in hydrological regime, are not considered as major threats for the Baltic Sea food webs. Introduction of non-natural substances do not show up in the identified threats, either; however, this may be partly due to the lack of research and information, e.g. regarding impacts of marine litter on food webs.

3. Overview of food web indicators in the Baltic Sea

Laura Uusitalo, SYKE

In this chapter, we review the food web indicators that exist or have been proposed for the Baltic Sea. The reviewed indicators constitute those core indicators of the HELCOM CORESET project (HELCOM 2013a) that are identified as suitable as food web indicators, and food web indicators identified or proposed in the Marine Strategies of the Member States. Of the Baltic Sea countries, Finland, Estonia, Sweden, Latvia, Lithuania, and Denmark have to date reported indicators (article 10 of the MSFD) in their Marine Strategies; these indicators were considered in this overview. Germany and Poland have not defined indicators at the time of writing this report. This overview is based on the Roof Report Questionnaire made in GES-REG project's Work Package 2, and on documentation of the Marine Strategies of the Member States.

For each food web health related indicator, we sought to answers the following questions (Annex 1):

- What is the status of the indicator: is it operative, or proposed?
- Does the indicator have GEnS boundaries, or protocols to define the GEnS boundaries?
- Can information regarding the indicator be acquired from existing monitoring programmes?
- Does the indicator correspond to the indicators as required in the Commission Decision 2010/477/EU (EU 2010)?
- Which pressures does the indicator respond to?

Not all of these questions could be answered based on the source documents. There was missing information or ambiguity in some of the source documents regarding the following points (Annex 1):

- The status of the GEnS targets: do they exist already, have they been proposed, or has a methodology for setting the GEnS targets been proposed? In some cases, indicators were operational, but used with either qualitative or trend-based intermediate targets.
- The existence of sufficient monitoring: is current monitoring sufficient? If not, which additions are required?
- To which Commission Decision indicator does this indicator correspond to? In some documents these were clearly identified, in others not. Some proposed indicators did not fit the strict definition of any of the Commission Decision indicators; some of them did not fill them strictly, but bore resemblance to the idea behind the Commission Decision indicator.
- Which pressures are relevant for the indicator? In some of the source documents these were clearly identified, in others not. In these cases, we have identified the relevant pressures based on ecological understanding.
- Which ecological areas or habitats does the indicator cover? It was not always clear whether the indicator was applicable in the coastal zone, open sea, or both; in some cases, as in with marine mammals, this distinction is not relevant.

In addition, it was sometimes difficult to determine whether some indicators in two Member States are essentially the same indicator, just phrased differently, or whether there are some relevant differences.

3.1 Background: HELCOM CORESET and national indicators

The HELCOM CORESET project approaches biodiversity broadly, so that the biodiversity indicators encompass the descriptors 1, 2, 4, and 6 (biodiversity, non-indigenous species, food webs, and sea-floor integrity) of the MSFD The HELCOM CORESET project has the ambitious aim of providing a common set of indicators that are scientifically justified, linked to an anthropogenic pressure, have existing monitoring data or a proposal for monitoring, and have policy relevance (HELCOM 2013a). The aim of the HELCOM core set of indicators is to enable the follow-up of the Baltic Sea Action Plan, and to be utilizable also for the other international requirements, such as the MSFD (HELCOM 2013a). A large number of experts from the Baltic Sea countries have participated in the indicator development work.

The HELCOM CORESET project introduces 20 core indicators for biodiversity, 15 of which are also identified as potentially addressing the MSFD descriptor 4, food webs (Table 17 in HELCOM 2013a). In addition, the interim report (HELCOM 2012) identifies several candidate indicators which are to be developed into core indicators. These candidate indicators are, however, at this stage missing some properties required from a proposed core indicator, such as validation of the scientific basis, links to pressures, or suggested GEnS boundaries (HELCOM 2012). The details of the HELCOM core indicators will be available in the HELCOM web site in the future; at the moment, they can be downloaded as pdf files. The following document includes links to all of the core indicator description sheets: http://meeting.helcom.fi/c/document_library/get_file?p_l_id=18975&folderId=2258447&name=DLFE-54162.pdf

At this stage of the MSFD implementation process, the approaches to define the national indicators, and the national indicators themselves, vary strongly, as can be seen from the national Marine Strategies. The proposed indicators seem to reflect not only the marine environment of the Member States, but also the special fields of expertise of the experts working on the indicators, and presumably also the interpretation of the Commission Decision requirements. For example, Lithuanian, Latvian, and Estonian food web indicators deal exclusively with plankton and fish, while Finnish and Danish indicators range from plankton to marine mammals. Swedish indicators focus on fish, birds, and mammals. It is also possible that the selection of approaches and organisms groups reflect the competence of the institute or institutes that have been, in each country, tasked with the responsibility of drafting the indicators and following them.

There are also obvious differences in the interpretation of the task of setting the indicators. Sweden and Estonia have taken the list of required indicators in the Commission Decision as the starting point and identified or developed one or a few suitable indicators for each; whereas Germany and Finland have drafted their own vision of the healthy sea, which, while related to the MSFD descriptors and Commission Decision criteria, do not follow their structure. Therefore, the correspondence between the Finnish national indicators and those required by the Commission Decision is sometimes hard to see. It is also noteworthy that all of Sweden's food web indicators act simultaneously as indicators for biodiversity. Denmark has two indicators that are defined as indicators on pressure on food webs; namely, harbour porpoise by-catch, and plankton biomass.

The discussion below is a summary of findings of the indicator review; for more detailed information about the indicators, the reader is referred to Annex I.

3.2 Food web indicator status and coverage in the Baltic Sea

Do the indicators cover the requirements of the Commission Decision?

Not all of the indicators required in the Commission Decision were covered by the indicators reported by the Member States in their Marine Strategies (Table 3.1). Especially indicator 4.2.1, *Proportion of selected species at the top of food webs: large fish (by weight)*, was represented by well-fitting indicators in the Marine Strategies of Sweden and Denmark only, and with somewhat-fitting indicators in the Marine Strategy of Lithuania. We discuss the challenges relating to this indicator in the Baltic Sea region in Chapter 5.

Latvia and Lithuania proposed only three and four food web indicators, respectively, and no indicators that would strictly respond to Commission Decision indicators 4.1.1 and 4.2.1.

Table 3.1. *Number of indicators in each Member State that correspond well (no parenthesis) or somewhat (in parenthesis) to the indicators of the Commission Decision. Some indicators may appear in more than one category.*

	Commission Decision indicator			Other food web indicator	Total number of indicators	Number of operative indicators
	4.1.1*	4.2.1**	4.3.1***			
Sweden	2	2+(1)	1+(1)	0	7	1
Estonia	1	2	4	0	7	6
Finland	5+(1)	0	5+(3)	8	20	4
Latvia	0	0	3	0	3	2
Lithuania	0+(2)	0+(2)	2+(1)	0	4	4
Denmark	1	1	2	2	6	?

* 4.1.1*: Productivity (production per unit biomass) of key species or trophic groups: Performance of key predator species*

**4.2.1: *Proportion of selected species at the top of food webs: large fish (by weight).*

*** 4.3.1: *Abundance/distribution of key trophic groups/species: Abundance trends of functionally important selected groups/species*

In addition to the indicators required in the Commission Decision, Finland and Denmark proposed eight and two additional food web related indicators, respectively. The Danish indicators were indicators of pressure towards food webs, while part of the Finnish indicators could be described as pressure indicators (such as number of hunted seals), and others that may summarize changes that have taken place in the food web, such as health and mortality of seals and salmon.

Finland proposed the highest number of food web indicators, a total of 20; however only four of these are operational at the moment. The rest are to be developed either by 2014 or 2018. Denmark had the smallest set on indicators that however covers all of the requirements listed in the Commission Decision; the information whether these are operational at the moment could not be found in the source document.

It has to be noted that one indicator might not cover all parts of the ecosystem; for example, the open sea and the coastal ecosystems and their energy transfer paths can be quite independent and need to be considered separately. These considerations appear to be one of the main reasons for the high number of Finnish indicators; for example, there are separate indicators for the breeding success of water birds feeding on coastal fish, off-shore fish, and mussel.

3.3 Monitoring requirements

We attempted to determine whether the existing monitoring programmes are sufficient for the purposes of the HELCOM CORESET and national food web indicators; however this was often difficult or impossible to determine from the source documents. Even in cases where the indicators were operational, it might be that additional spatial or temporal coverage would be required. In general, however, most of the indicators can be assessed based on the existing monitoring programmes. However, it is apparent that some additional monitoring will be required at least in Finland and Sweden.

3.4 Recommendations on the harmonized use of food web indicators

Some indicators were found in the indicator set of more than one Member State and/or HELCOM's core set (column G in Annex 1). Sometimes it was difficult to judge by the short descriptions whether two indicators were essentially the same or not; therefore this information in the table must be considered preliminary at best. However, the fact that the same indicators come up in several Baltic Sea Member States is indicative of successful co-operation between the scientists and managers of the Baltic Sea states. It is desirable that in the later cycles of the MSFD implementation, steps will be taken toward the further harmonization of indicators between the Member States.

The HELCOM core set of indicators will be monitored by all of HELCOM's Contracting Parties, i.e. all Baltic Sea countries (HELCOM 2013a). Since the HELCOM CORESET indicators aim to be useable also in the monitoring of the Good Environmental Status according to the MSFD, it is likely that the Member States will increasingly adopt HELCOM core indicators into their MSFD indicator suite, which will naturally lead to harmonization of indicators in the Baltic Sea countries. The GEnS targets of these indicators must be defined for each sea area separately, according to its characteristics;

however a common procedure for the determination of the targets is suggested by HELCOM CORESET project, giving rise to harmonized interpretation of these common indicators.

However, the HELCOM core set of indicators still has gaps in relation to the ecosystem components and relevant pressures (HELCOM 2013a). Therefore the GES-REG project encourages the Baltic Sea states to evaluate their national set of food web indicators also from the point of view of whether they cover the central food web components, and whether they are responsive to the key pressures (Chapter 2 of this report). If gaps in the ecological coverage are revealed, the indicator catalogue compiled in this project (Annex I) may be used to see if other Member States are using or developing an indicator that could be applied in other countries as well.

4. Energy flows: food web structure assessment by stable isotope analysis

Heikki Peltonen and Mikko Kiljunen, SYKE

4.1 Introduction

Food webs belong among the 11 descriptors of the MSFD, but it is evident that the current knowledge about food web structures in different sea areas is scattered and few cost-efficient tools can be applied to enhance our knowledge in this issue. Besides, in ecology a wide variety of methods can be applied to evaluate the structure, functioning and dynamics of food webs. In this chapter, we aim to evaluate stable isotope analysis (SIA) as a potential tool to support the needs of the MSFD in analysing food web structure, energy flow and dynamic changes in key trophic interactions in food webs.

We describe, based on a literature review, how the stable isotopes of carbon and nitrogen in particular can be applied in analyses of trophic interactions and energy flows in the food web. Besides, as a part of the stable isotope work in the GES-REG we compiled a large amount of existing SIA material collected in various research projects to address specific management tasks and ecological questions. For the compiled material, we applied up-to-date methods in data analyses, to explore the utilities of this approach in exploring the food webs of the Baltic Sea. Analyses of this data have enabled construction of diagrams (Figures 4.1 and 4.2) of the food web structure and energy pathways as well as isotopic niches of species, making it possible to evaluate the potential merits of the approach in indicating the health of marine food webs.

This chapter aims in particular to evaluate whether SIA could support applicable indicators to observe shifts in the structure and function of marine food webs in the context of the EU Marine Strategy Framework Directive, particularly considering the key groups as defined in the directive. The focus is in the applications of isotope ratios found in nature, while application of isotopes in experimental work is not considered.

4.2 Overview of the Stable Isotope Analysis

Knowing how marine ecosystems are structured and how they function is essential to detect changes in their status and health. Stable isotope analysis (SIA) has been used in numerous applications to increase ecological knowledge. In particular, it is exceedingly gaining popularity in the evaluation of food web structure and of the flow of organic matter and energy in ecosystems (e.g. Michener and Kaufman 2007). The ratios of stable isotopes can be cost-efficiently analysed from biological material using mass spectrometry (Michener and Kaufman 2007). SIA can provide approaches for example to observe shifts in trophic position of species and in species interactions in food webs (e.g. Gorokhova *et al.* 2005) which would be very difficult to observe with other methods. Recent developments in analysing the data have made the SIA increasingly attractive in the exploration of food webs.

Which isotopes?

Isotopes are variants of given elemental atoms with an equal number of protons and electrons but with different numbers of neutrons. Thus, different isotopes have different atomic weights. Stable isotopes in contrast to radioactive ones, do not decompose but persist in element-specific frequencies. Trophic transfer causes predictable fractionation of isotope ratios in consumers relative to their diets (DeNiro and Epstein 1978, 1981), which allows inference of a consumer's diet or trophic position by comparing its isotopic ratios with those of other species in its food web. The method is most powerful when several isotope ratios are studied simultaneously (Harvey et al. 2002). Many animals are highly opportunistic foragers and their diet may vary substantially over time (Pinnegar and Polunin 1999). Stable isotopes link consumers via their diet also e.g. to sources of contamination.

The reasons for the differences in the reaction rates of the isotopes and their compounds are discussed by Sulzman (2007). The SIA approach focusing on trophic interactions is based on the differences in isotope ratios of certain atoms among food sources available for consumers and the predictable manner in which isotope ratios are transformed from dietary sources to consumers.

SIA reflects the diet assimilated during a long time, whereas e.g. identification of prey from gut content analysis only reveals diet prior to capture. Furthermore, SIA can support analyses of the diet of animals that crush their food beyond recognition and additionally, analyses of gut content can be impractical or impossible in small animals.

In general, the most applicable isotopes have low atomic mass and a large mass difference between the rare and abundant isotope (Sulzman 2007). Ratios of naturally occurring stable isotopes of especially carbon and nitrogen, and also of sulphur, hydrogen and oxygen, can be used to trace energy flow in ecosystems (Jardine *et al.* 2006). The isotope ratios are defined for carbon $^{13}C/^{12}C$ or $\delta^{13}C$, for nitrogen $^{15}N/^{14}N$ or $\delta^{15}N$, for sulphur $^{34}S/^{32}S$ or $\delta^{34}S$, for hydrogen $^{2}H/^{1}H$ or D/H or δD and for oxygen $^{18}O/^{16}O$ or $\delta^{18}O$. In marine ecology, nitrogen and carbon are the elements that most often form the basis for the SIA.

Definitions of isotopic composition

In SIA, the isotopic composition in the analysed material (δ X) is usually reported relative to an internationally accepted standard. As the isotopic differences are in general small between various materials, the isotopic composition is usually expressed in the delta notation in parts per thousand following the formula:

$\delta X (‰) = 1000 (R_{sample}/R_{standard} - 1)$,

where R_{sample} is the ratio of heavy-to-light (typically, but not always, rare-to-abundant) isotopes in the sample, and $R_{standard}$ is the heavy-to-light ratio in the standard (e.g. Peterson and Fry 1987, Sulzman 2007). Thus, increases in these δ values denote increases in the proportion of the heavy isotope components (Peterson and Fry 1987).

For example for nitrogen the standard represents the ratio of the isotopes in air, and the fractionation (altering the ratio of heavy to light isotopes) is calculated as

$$\delta^{15}N \ = 1000\left(\frac{^{15}N_{sample}/^{14}N_{sample}}{^{15}N_{air}/^{14}N_{air}} - 1\right)$$

Standards for common systems include Vienna-Pee Dee Belemnite limestone (V-PDB) for carbon, atmospheric N_2 for nitrogen, and Vienna Standard Mean Ocean Water (VSMOW) for hydrogen and oxygen, the numerical values for the standards being for example for nitrogen $^{15}N/^{14}N = 0.0036765$ and for carbon $^{13}C/^{12}C = 0.0112372$ (e.g. Sulzman 2007).

Isotopic composition turnover rates

Isotopic composition of animals is almost constantly changing at least to some degree. Changes in the isotopic composition will take place due to growth of new tissue and metabolic loss of tissue materials (Fry and Arnold 1982). Increases in the mass of tissues through growth induce dilution which results in faster equilibration to the new diet than would occur by metabolic turnover alone (Phillips and Eldridge 2006). Following a step change in the isotopic composition of their diets, animal tissues eventually come to isotopic equilibrium with their new diet (Fry and Arnold 1982). Due to species-specific differences in metabolic activity and due to variations in metabolic activity during the life history of animals, different isotopic turnover rates can be found in different species and development stages of animals.

Different animal tissues incorporate the isotopic signatures of resources at different rates (Tieszen *et al.* 1983, Phillips and Eldridge 2006). In general, the rates are high in metabolically active tissues such as blood plasma and liver, somewhat lower in muscle and low rates in long-lived tissue such as bone (Tieszen *et al.* 1983). The composition of diet influences the isotopic turnover in animals. In particular feeding on a protein deficient diet is likely to induce long retention of ^{13}C and ^{15}N (Martínez del Rio 2009 and references therein).

The varying isotopic turnover rates in multiple tissues have been applied as a chemical clock to estimate the time elapsed since a diet shift, and the magnitude of the isotopic shift in the tissues (Phillips and Eldridge 2006). Another approach to detect temporal variation in diet is to analyse tissues which have progressive growth and retain isotopic values in a chronological order (e.g. Dalerum and Angerbjörn 2005). Such tissues with chronological records also keep record of the movement of animals among habitats or food webs with different isotopic composition. The fact that isotopic composition informs of the diet during a long time can be a major advantage of SIA in comparison to traditional dietary proxies, such as foraging observation or analysis of gut contents (Dalerum and Angerbjörn 2005) but it is also a factor which in certain cases can confound isotopic analyses of food web structure.

As regards the impacts of diet shift on isotopic composition in crustacean species common in the Baltic Sea, *Mysis mixta* and *Neomysis integer,* the $\delta^{13}C$ composition in muscle tissue had not reached an isotopic equilibrium with an altered diet in 12 weeks, while the exoskeleton came into equilibrium with the food in 2–3 weeks and the $\delta^{15}N$ in the muscle tissue integrated the isotopic signal over 6–8 weeks (Gorokhova and Hansson 1999).

Estimation of the trophic level with stable nitrogen isotopes

The availability of nitrogen plays a central role in regulating biological productivity in marine environments. The distribution of the stable isotopes of nitrogen within marine ecosystems can provide critical insights into the sources of N supporting production and the pathways and mechanisms of movement of nitrogen through those ecosystems (Montoya 2007).

DeNiro and Epstein (1981) observed the enrichment of the ^{15}N relative to ^{14}N towards higher trophic levels and stated that *"the dependence of the $\delta^{15}N$ values of whole animals and their tissues and biochemical components on the $\delta^{15}N$ value of diet indicates that the isotopic composition of animal nitrogen can be used to obtain information about an animal's diet if its potential food sources had different $\delta^{15}N$ values"*. Thereafter, the $\delta^{15}N$ of animal tissues has been often used as an indicator of trophic position in marine food webs (Peterson and Fry 1987, Post 2002). A model for estimating the trophic position of a secondary consumer is: trophic position = $\lambda + (\delta^{15}N_{sc} - \delta^{15}N_{base}) / \Delta_n$, where λ is the trophic position of the organism used to estimate $\delta^{15}N_{base}$ (e.g., $\lambda = 1$ for primary producers), $\delta^{15}N_{sc}$ is measured directly, and Δ_n is the enrichment in $\delta^{15}N$ per trophic level (Post 2002). The trophic enrichment of nitrogen per trophic level is generally assumed to be between 3‰ and 4‰ (Peterson and Fry 1987). Post (2002) found a general value of 3.4‰ which has been subsequently applied widely although also wide variations may occur. Gorokhova and Hansson (1999) found that in the muscle tissue of mysids from the Baltic Sea become enriched in $\delta^{15}N$ relatively to the food by +3.6‰ when fed with *Artemia* and +2.7‰ when fed with *Enteromorpha*.

For $\delta^{15}N$ to be a useful metric of trophic position and variation, it is critical to understand factors aside from trophic enrichment that can influence isotopic values (Ingram *et al.* 2007). Such factors include nitrogen metabolism, food quality and temperature (Ingram *et al.* 2007). While the fractionation associated with the assimilation of food nitrogen is small, the significant isotopic fractionation associated with the excretory loss of NH_4+ appears to be the primary factor contributing to the isotopic enrichment of an animal's tissues relative to its food (Post 2002 and references therein).

Terrestrial versus marine food sources

Compared to stable nitrogen, there is much less enrichment of stable carbon per trophic level, but the ratio of ^{13}C to ^{12}C can be used to find the carbon flow to consumers and diet composition especially in cases when the diet sources have relatively large differences in $\delta^{13}C$ values (DeNiro and Epstein 1978, Post 2002). In particular, as species in the coastal waters often have differences in isotopic $\delta^{13}C$ values depending on the contribution of energy from terrestrial versus marine food sources, stable C isotope ratio can also highlight utilization of terrestrial versus marine food sources or movements of animals between coastal water and open sea (Hobson *et al.* 1996).

Lipid normalization

Lipids are known to be ^{13}C-depleted (have lower $\delta^{13}C$) relative to other major tissue constituents (e.g. Kiljunen *et al.* 2006 and references therein).For example a mixing model (see below) will produce biased results of contribution of different food sources in a predators diet if carbon isotope ratios of the predator and prey have not been corrected for lipid content. This is due to fact that one food source may be rich in carbon due to lipids, which leads to a proportionate increase in the contribution of that food source to the predator for that element relative to the nitrogen. Lipid-normalization of samples can be done either through chemical extraction which physically removes lipids from samples, or by applying a mathematical model which uses the carbon-to-nitrogen (C/N) ratio of a sample to normalize $\delta\ ^{13}C$ after analysis (Kiljunen *et al.* 2006, Ingram *et al.* 2007). While some studies have found that chemical extraction of lipids may also influence $\delta^{15}N$, Ingram *et al.* (2007) stated that as long as appropriate techniques are used, effects of chemical lipid extraction on $\delta^{15}N$ of aquatic consumers need not be a major consideration in the SIA of food webs.

Baseline

Estimation of trophic position or carbon source requires the isotopic signature of a consumer together with an appropriate isotopic baseline (e.g. Post 2002). Herbivorous zooplankton or filter-feeding mussels can serve as the baseline organisms for the pelagic food web and snails have been applied as a baseline for the littoral food webs (Post 2002). However, establishing the baseline can be a demanding task. For example, some archived material could be used to define an appropriate isotopic baseline, but it can be difficult to obtain such material over the same period which has preserved the original isotopic composition. Anyhow, it is essential to find out if there have been changes in the isotopic composition of the baseline organism. Voss *et al.* (2000) observed increasing $\delta\ ^{15}N$ and $\delta\ ^{13}C$ values in Baltic Sea sediments and suggested that the increase was due to elevated primary production — in the case of nitrogen, also by an increase in $\delta^{15}N$ values due to nitrogen inputs into the ecosystem via urban wastewater or soil runoff from fertilized fields.

Mixing models

Ratios of carbon and nitrogen isotopes are often used in mixing models to calculate proportions of different food sources in the diet of a consumer and to construct food web models (e.g. Phillips and Gregg 2003, Kiljunen *et al.* 2006). The methods to establish contributions of different food sources work best when the food sources differ substantially in isotopic composition, but show low variance (Phillips and Gregg 2003).

The trophic position of an animal utilising two food sources can be calculated as: trophic position = $\lambda + (\delta^{15}N_{sc} – [\delta^{15}N_{base1}\,\alpha + \delta^{15}N_{base2}\,(1-\alpha)]) / \Delta_n$, where α is the proportion of nitrogen in the consumer ultimately derived from the base level of the first food web (Post 2002). It is often assumed that trophic fractionation of carbon is insignificant. In such a case, α can be estimated using carbon isotopes such that: $\alpha = (\delta^{13}C_{sc} - \delta^{13}C_{base2}) / (\delta^{13}C_{base1} - \delta^{13}C_{base2})$, where sc denotes a secondary consumer (Post 2002). Kiljunen *et al.* (2006) recommended using concentration-weighted models if C/N ratios differ substantially between the modelled species, and cautioned about application of mixing models that do not allow for concentration dependency. Besides, since differential digestibility of carbon and nitrogen among species may alter these ratios after ingestion, this should also be taken into account (Phillips and Koch 2002).

Distinct isotopic signatures generally persist for only one or two elements per system, so relative source contribution determinations are often limited to estimates for two or three sources. Although resolution of the contribution of relatively few sources can be useful for specific applications, the inherent complexity of natural systems often requires the inclusion of a larger number of sources.

Recent development of new Bayesian based stable isotope models such as MixSIR and SIAR (Moore and Semmens 2008, Inger *et al.* 2010) have revolutionized the stable isotope ecology. These models solve for the most likely set of dietary proportions given the isotopic ratios in a set of possible food sources and a set of consumers. These models allows all sources of uncertainty such as in the sources or trophic fractionation values to be propagated through the model to return a true probability distribution of estimated dietary proportions. Previously it was necessary to ignore variation and uncertainty and work only with mean estimates of isotope ratios (Inger *et al.* 2010).

Isotopes to track animal movement

In analysing marine foodwebs there can be uncertainty about the feeding areas of migratory species. SIA can support useful information to evaluate the mobility of species. Several factors such as altitude, latitude, distance from the coast, and temperature, influence the isotopic composition of rainwater and create the broadly predictable geographical patterns in oxygen ($\delta^{18}O$) and deuterium (δD) in precipitation (Bowen and Revenaugh 2003). The “isoscapes” arising from spatial patterns in isotopic composition have been used widely to track animal movements (Rubenstein and Hobson 2004).

Isotopic vs. ecological niche

Analysing isotopic niches is a potentially useful approach to investigate ecological niches (Newsome *et al.* 2007). Both the ecological niche and isotopic niche are comparable as they are influenced by a consumer’s diet as well as the habitat in which it lives (Newsome *et al.* 2007). Similarly to the definition of an ecological niche, isotopic niche can be defined as a multidimensional space (e.g. $\delta^{13}C$ versus $\delta^{15}N$ plot) with coordinates that are directly influenced by diet and habitat. The identification of niche shifts by SIA can have important implications for conservation (Newsome *et al.* 2007) and it could be a useful concept to adopt to support the implementation of the EU Marine Strategy Framework Directive (see below).

4.3 Previous examples of Baltic Sea applications

Some studies have investigated the features of the Baltic Sea food webs with the SIA. Application of SIA on size-specific level could be useful in certain cases. For example, Rolff and Elmgren (2000) found with stable isotopes of C and N the presence of two separate trophic structures, one in the smaller size classes (0.7-5 pm), and another in the larger (>50 pm) and they suggested that that these structures corresponded to the microheterotrophic food web and the classic phytoplankton-based grazing food web. Enrichment of $\delta^{15}N$ in size classes of plankton was found to be a linear function of logarithmic organism size from 20 to 500 µm, reflecting size-related consumption patterns of marine plankton food webs (Rolff 2000).

Seasonal patterns can occur in stable isotope values although they are so far poorly known. In a Baltic Sea coastal plankton food web, the $\delta^{13}C$ showed a bimodal annual cycle with two local maxima, the first coinciding with the spring bloom and the second with the autumn bloom (Rolff 2000), but in $\delta^{15}N$, the annual cycle was trimodal.

Voss *et al.* (2005) concluded that stable isotope data indicated that the Baltic Proper is separated into two subsystems (coastal rim and the central Baltic Proper) with limited dissolved nitrogen exchange. Thus coastal eutrophication was stated to be driven by river nutrients whereas eutrophication in the open Baltic Sea apparently is dominated by excess phosphate (of riverine and sedimentary origin) driving nitrogen fixation by cyanophytes.

Gorokhova and Hansson (2006) applied two-source isotope-mixing models for stable N, with micro- and mesozooplankton as prey for the invasive predatory cladoceran *Cergopagis pengoi*, and mesozooplankton and *C. pengoi* as prey for young-of-the-year herring. Mesozooplankton was the major food source of both species while microzooplankton was important prey for young stages of *C. pengoi*. Herring trophic position increased from 2.6 to 3.4 after the invasion of *C. pengoi*, indicating substantial alterations in the food web structure (Gorokhova and Hansson 2006).

4.4 Compilation of the existing material and data analyses in GES-REG

The previous examples have dealt with some specific group of animals or small functional part of the Baltic Sea food web. In order to investigate or monitor the energy transfer through the whole system, an ecosystem-based approach has to be applied. To test this approach, we used stable isotope ratios of carbon and nitrogen to elucidate the ecological frameworks of the trophic structure and energy flow of the Bothnian Sea food web. Large amounts of existing SIA material was obtained from various research projects collected in the past ten years and mixing models and trophic position models presented in earlier sections were applied to the data.

Analyses of these data enabled reconstruction of the Bothnian Sea pelagic food web and nicely illustrates some key food web parameters, namely trophic position of the species and the pathways of the energy (Figure 4.1). These metrics are good measures of food web functioning and therefore may act as potentially useful ecosystem indicators. Based on the exercise, mysids and amphipods appear to be major integrators of the benthic and pelagic systems. Roughly 35–50% of their energy originated from the benthic zone which was further transferred to higher trophic levels by herring. Such interactions may provide a basis for using some species as indicators of ecosystems change, since fundamental changes in the ecosystem should be reflected in these parameters.

Closer examination of the benthic invertebrate species using the isotope niche approach reveals that even species previously considered as ecologically relatively similar can differ in their feeding behaviour (Figure 4.2). Isotopic niche is tightly linked to diet and therefore any change in the ecosystem which will affect species dietary composition should be seen as a change in the isotopic niche.

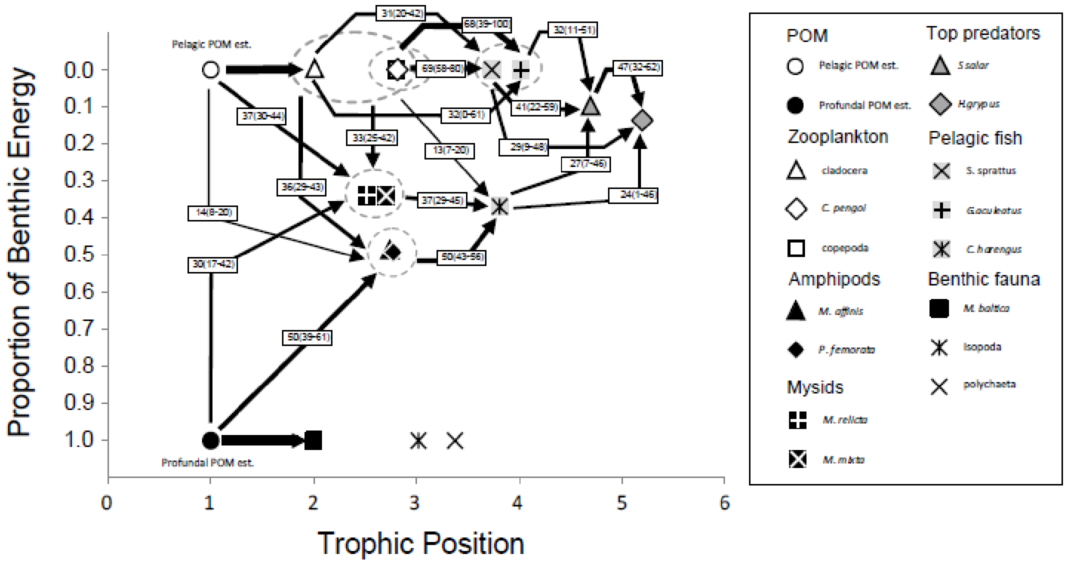

Figure 4.1. *Proportion of benthic energy in different trophic levels in the Bothnian Sea. The figure is based on stable isotope (carbon and nitrogen) composition of presented food web components.*

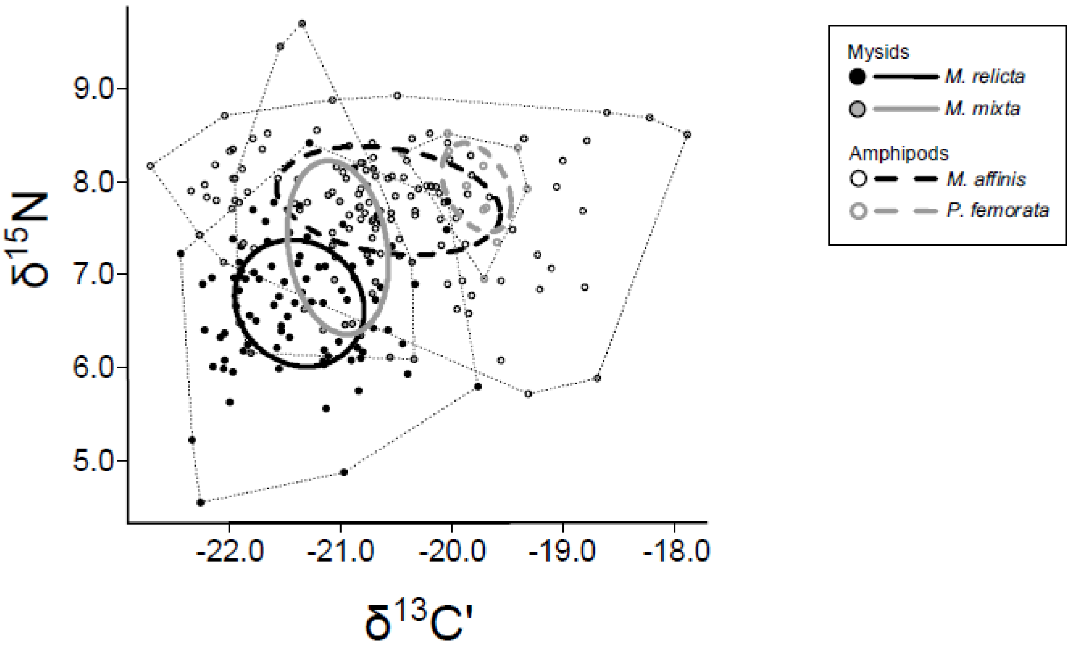

Figure 4.2. *Dietary niches (standard ellipse areas) of amphipods and mysids in the Bothnian Sea. Markers represent the location of an individual* $\delta^{15}N$ *and* $\delta^{13}C$ *space and dotted lines the minimum area in which the individuals were located.*

4.4 Can SIA support indicators for MSFD?

Descriptor 4 in the MSFD addresses marine food webs, and more exact definitions are given in the Commission Decision (2010/477/EU) (see Chapter 1 above). Here we will discuss the applicability of SIA to support monitoring of the food webs in the Baltic Sea considering the directive and the commission decision.

The first criterion in the Commission Decision is the *Productivity (production per unit biomass) of key species or trophic groups*, with the indicator "*Performance of key predator species using their production per unit biomass (productivity) (4.1.1)*".

Performance of predators can be expressed e.g. as productivity, reproduction success or mortality rates, but a key point limiting the applicability of this indicator is that unless predators are food delimited, this indicator infers little about the food web processes on lower levels (Rombouts *et al.* 2013). Rombouts *et al.* (2013) emphasize the importance of understanding the predator-prey relationships to ensure that predator performance provides a measure of food web functioning, and they also underscore the need to know how variations in prey abundance influence consumption of certain prey. Further on, they suggest stomach analyses and SIA as potential tools to estimate the trophic structure of the food web, and application of the statistical analysis of Rossberg *et al.* (2012) to average over many species located high in the food web rather than attempting to identify the ideal indicator species. Anyhow, in a species-poor ecosystem such as the Baltic Sea averaging over many species high in the food web may not be possible. However, SIA can produce information about the pathways of energy trough the food web to key predators. For example, the analyses performed within GES-REG project enabled estimating the contribution of benthic and pelagic energy sources to the nutrition of the key pelagic fish species herring, sprat and the three-spined stickleback, and further on to higher trophic levels.

Focusing on trophic position and isotopic niche (Newsome *et al.* 2007, see above) could enable determining the structure of the food web and observing dynamic shifts in the function of the ecosystem. Apparently, SIA would be a particularly useful tool for analyzing the Baltic Sea food webs and isotopic niches because there are few key species in the ecosystem and few or even a single species can uphold essential functions in the food web (see Johannesson *et al.* 2011). Thus, major shifts in the niches of a small number of species can indicate fundamental changes in the ecosystem function. Anyhow, in analysing the compiled SIA material it became apparent that all the essential trophic groups should be adequately sampled, because omitting essential elements can give distorted views about the structure of the food web. This holds e.g. for analyses on grey seal, where either the migratory characteristics of the species and/or missing prey categories did not enable reliable estimates of the trophic position of this seal species in the food web. Thus it is evident, that more analysis with SIA should be performed to better estimate the trophic position of seals, and also of other key species.

The second criterion addresses the structure of the food webs, with an indicator "*Large fish*". Rombouts *et al.* (2013) highlighted the applicability of the "*Large fish*" –indicator in their review on food web indicators. In particular, they concluded that this indicator is currently better applicable for the demersal fish community but to be suitable as a food web indicator, it should also include some pelagic species. Since the demersal fish are currently virtually absent in large areas of the open Baltic Sea, this indicator cannot be applied like e.g. in the North Sea. Besides, shifts in the proportion of large fish may not distinguish between demographic changes within a population or changes in proportions of different-sized species (Rombouts *et al*. 2013). Moreover, since fish growth rates are very plastic, nutrition is an essential element to consider in the context of fish growth. This is an important issue in the Baltic Sea where the growth rates of clupeid fish and cod apparently have changed more than in any other marine ecosystem. SIA could provide information about how predators utilize their prey resources, and thereby support finding out the reasons for changes in fish growth rates and in shifts in the size composition of the fish community.

The third criterion deals with functionally important species or groups, fundamental for the structure and function of the food web. The Commission Decision (2010/477/EU) also announces the indicator for this criterion *"Abundance trends of functionally important selected groups/species".*

(4.3.1) Several selection criteria for key trophic groups/species were mentioned:

- *groups with fast turnover rates (e.g. phytoplankton, zooplankton, jellyfish, bivalve molluscs, short-living pelagic fish) that will respond quickly to ecosystem change and are useful as early warning indicators,*
- *groups/species that are targeted by human activities or that are indirectly affected by them (in particular, by-catch and discards),*
- *habitat-defining groups/species,*
- *groups/species at the top of the food web,*
- *long-distance anadromous and catadromous migrating species,*
- *groups/species that are tightly linked to specific groups/species at another trophic level*

SIA should well fit into the toolbox when the trophic interactions are evaluated. SIA could support evaluation of the food web structure and detection of shifts in the structure and function of the food web ranging over many trophic levels even from the bottom to the top.

As food webs are complex entities it may be difficult to find a simplistic approach to evaluate their structure and dynamics in order to evaluate the environmental status of marine waters. While SIA can support useful information which would be difficult and costly to acquire with other methods, an optimal choice might be a combination of different approaches (monitoring of species abundance and distribution, food web modelling, SIA and perhaps biochemical analyses). Also Rombouts *et al.* (2013 and references therein) suggested incorporation of modelling and empirical studies: ”*Before responses of food webs to change can be anticipated, they need to be both unravelled, i.e. to identify appropriate levels of detail, and rewoven, i.e. to capture the essential elements for simplification and generalisation. Theoretical and empirical models may then help to identify potential impacts and elucidate key properties that should be monitored, thereby promoting the development of more effective and comprehensive operational food web indicators. The combination of theoretical modelling with empirical analysis offers the potential for testing theoretical considerations whose findings may then be used for practical ecological and management applications, and policy strategies. The complementary use of empirical and modelling approaches to derive population, community and ecosystem indicators is key to the development operational food web indicators for ecosystem-based management in the marine environment. To accommodate the difficulty of integrating across so many levels of organisation, however, requires new information as well as new methods.*”

4.5 Suitability of current monitoring and assessment methods

Currently, few efforts are being made to apply SIA as a monitoring method for the Baltic Sea food webs. However, in the future SIA methods could enable more explicit perception of ecosystem structure and function than many of the current food web indicators, in particular as SIA can be applied to find out the key species, to resolve how trophic levels and food webs (e.g. coastal and pelagic or benthic and pelagic) are coupled, to observe shifts in trophic niches of species and to determine dynamic changes in food webs.

If SIA would be applied for food web monitoring, in the initializing phase of the work, temporal data from many species should be collected and analysed, and also archived material should be fully utilised to better understand the utilities and limitations of SIA to support evaluation of the food webs in the context of the MSFD.

Sampling

Sampling of the material for SIA would be quite straightforward, as it could be incorporated with the annual monitoring cruises of research vessels. The Cooperative Monitoring in the Baltic Marine Environment (HELCOM COMBINE), established in 1992 through integration of different national programs into a common structure would be of specific interest in this context. These cruises annually collect mesozooplankton and macrozoobenthos samples, covering the Baltic Sea extensively. After sampling with a plankton net, SIA can be applied directly to the dried bulk sample. Since the required sample size is very small (ca 0.3 mg, dry weight), SIA can be applied to a selected taxonomic category hand-picked from the sample or to a size-fraction of the sample (Rolff 2000).

Fishery research cruises coordinated by the Baltic International Fish Survey Working Group (ICES WGBIFS) annually cover almost the whole Baltic Sea. These cruises could be utilised to collect material especially of marine fish species from the offshore areas. Other types of monitoring surveys or sampling of commercial catches could be utilised to collect species e.g. from coastal areas.

While the material from the monitoring cruises would enable the determining of the trophic space of species, it is also necessary to monitor the isotopic baseline for the analysed food webs by collecting and analysing the isotope ratios in desired species of zooplankton or macrozoobenthos.

Ecological knowledge should be fully utilized in sampling design so that e.g. the same populations and development stages are sampled each time. Whether a species is a feeding specialists or generalists (Reynolds 2008) can influence on the variations in SIA, and some species have temporal variations in diet composition. Although it is well known that these and several other potential sources of uncertainty exist, they can be overcome by careful planning of the sampling, and by applying robust methods in laboratory analyses and in the scrutiny of the data.

Analyses

Commercial SIA laboratories are increasing available around the world and analytical costs, especially for the most common isotopes (nitrogen and carbon), have come down drastically in past two years. Also many universities and research institutes have facilities for stable isotope analyses and may provide analytical services. In the laboratories standardisation is based on common IAEA standards and laboratories are generally well harmonized and standardized. At the time of writing, the price for a sample analysed for carbon and nitrogen varies between 10–100€, depending on how much sample pre-preparation is done. C/N ratio, used for e.g. lipid normalization, can be usually obtained from the same run and separate lipid analysis is therefore not needed. SIA can be a very cost-effective method to obtain monitoring data. E.g. trophic position estimates and dietary information can be obtained by a fraction of the cost of more conventional methods.

Statistical methods for analyzing stable isotope data have improved drastically in past few years. Especially introduction of Bayesian mixing and niche models have been great improvement to the earlier models (Moore and Semmens 2008, Inger *et al.* 2010, Jackson *et al.* 2011). These models have been used by the scientific community for some years now and have been shown to provide unbiased results even with small sample size, and their estimation via Bayesian inference allows robust comparisons to be made among data sets comprising different sample sizes. Selecting the statistical methods for food web monitoring purposes would be relatively easy, since standard statistical methods within the discipline are quite well harmonized.

5. Large fish (by weight): Problematic indicator in the northern Baltic Sea

Pekka Jounela and Eero Aro, RKTL

5.1 Introduction: indicators of fish in food webs

The primary fish-related indicator standards that are related to the MSFD criteria on GEnS give emphasis to healthy fish stocks, which should be characterised by a high proportion of old, large individuals. The primary indicators based on the relative abundance of large fish include: 1) *Proportion of fish larger than the mean size of first sexual maturation*, 2) *Mean maximum length across all species found in research vessel surveys*, and 3) *95% percentile of the fish length distribution observed in research vessel surveys*. The secondary indicator standard that relates to the MSFD criteria on GEnS includes: 1) *Size at first sexual maturation*, which may reflect the extent of undesirable genetic effects of exploitation. In the northern Baltic Sea the aforementioned indicators of GEnS can be assessed for the monitored stocks, namely cod, salmon, herring, and sprat. These species are also covered by the Common Fisheries Policy (CFP) of the EU, and harmonization of the requirements of the MSFD and CFP is required. Furthermore, the commercially exploited species are covered by the MSFD descriptor 3; the relationship between D3 and D4 indicators needs to be taken into account as well, if the commercially exploited species are to be used in the D4 assessment.

5.2 Large clupeoids

The HELCOM monitoring strategy is designed to prove the impact of exposure to eutrophication, habitat alteration, climate change, toxic substances and overexploitation. So far, there have been more or less well-developed models predicting the reactions of the Baltic Sea coastal freshwater fish community to these factors (Hartmann 1977, Neuman and Sandström 1996), but regarding marine fish such as clupeoids in the northern Baltic Sea, however, there is still a considerable lack of knowledge.

In the northern Baltic Sea, fishing mortality (F) estimates for clupeoids are very low (e.g. for Baltic herring in the Bothnian Sea: approximate F =0.14 and assumed natural mortality (M) =0.2). It is very difficult to distinguish between the very low fishing mortality rates from natural mortality rates, resulting in the use of agreed approximations of fishing mortality at maximum sustainable yield (FMSY), rather than FMSY. The low F estimate suggests that approximately 80% of the Bothnian Sea herring stock is presently outside commercial fishery. Assessing food web interactions from this information would only give implications for the remaining 20% of the stock. Thus, estimating undesirable effects of exploitation in terms of food web impacts would be somewhat subjective and, assessment of food web pressures on commercially most important large fish stocks (by weight) in the northern Baltic Sea would be doubtful. So far reliable reference values with scientific agreement for assessment of undesirable effects of exploitation have not been made. Most pelagic commercial fisheries in the northern Baltic Sea take a mixture of herring and sprat with varying proportions according to area and season. The actual species composition is somewhat uncertain because estimates of pelagic catch compositions are mainly based on logbooks and landing declarations, with limited supplementary sampling of catches.

The reduced mean weight at age of Baltic herring in the northern Baltic Sea may have been indirectly affected by the increased abundance of sprat, which in turn may have been induced by overexploitation of the cod stock in the past. An additional reason for the reduced mean weight at age of Baltic herring may have been anoxid water that does not sustain production of benthic food (e.g. mysids) for consumption of herring. In the northern Baltic Sea, the major reduction of cod abundance around 1988 was also followed by a major increase in recruitment of Baltic herring, which reduced mean weight at age. That is, mean weight at age of Baltic herring seems to follow both bottom-up and top-down forces and thus, it could be an appropriate indicator of GEnS in the northern Baltic Sea. In recent years the growth of Baltic herring has increased in the northern Baltic Sea.

Pauly *et al.* (1998, 2005) found that the mean trophic level of the species groups reported in Food and Agricultural Organization (FAO) global fisheries statistics reflected a gradual transition in landings from long-lived, high trophic level, piscivorous bottom fish toward short-lived, low trophic level invertebrates and planktivorous pelagic fish. The ‘fishing down the food webs’ was one of the main reasons behind the use of mean trophic levels as an index of biodiversity (the Marine Trophic Index, MTI) by the Convention on Biological Diversity. Pauly has also shown that the Baltic Sea has been one of the areas experiencing the largest reduction in the Marine Trophic Index. Note, however, that the reduction in MTI illuminated mostly overfishing of cod in the Baltic Sea Main Basin. Hence, it does not suggest that MTI would be an appropriate indicator of GEnS in the northern Baltic Sea.

5.3 Large coastal fish

The aim of the coastal monitoring is to describe the long-term trends in coastal fish populations and to link them to natural and anthropogenic pressures on large fish. Three main methods have been used to monitor coastal fish communities around the Baltic Sea. In the Baltic Proper, the longest time series are from monitoring using Net series, and further north in the Gulf of Bothnia, using Coastal survey nets (pre-fixed mesh sizes). The present northern monitoring network covers areas of Finland, Estonia, Latvia, Lithuania and Sweden. In Sweden there are also additional monitoring sites and nationally financed environmental monitoring. Coastal fish communities further south in the Baltic Sea areas of Denmark, Germany and Russia are also monitored, but are not included in the present HELCOM assessment. The coastal monitoring is carried out in August and is mainly directed towards demersal and bentho-pelagic fish living in coastal areas during the warm season. The guidelines for coastal fish monitoring are presented in Neuman *et al.* (1999).

In the northern Baltic Sea (Finland and Sweden) coastal monitoring of fish stocks is presently carried out each year using Nordic multi-mesh gill-nets and gill-net series. The coastal fish monitoring using

Nordic coastal multi-mesh gill-nets was introduced in 2001 and is only used in Finland and Sweden and in some areas in parallel with monitoring using Net series or Coastal survey nets. Long monitoring time series using standardized gear do not exist and hence, it may be difficult to recognize impacts and interactions in food webs without applying statistical gear standardization of the existing data bases.

Coastal monitoring of fish stocks also includes measurements of temperature, salinity, wind speed and direction, and water transparency. The monitoring is to be conducted according to a specific procedure (HELCOM 2006) so that the results can be compared among the areas monitored. The aim of the coastal monitoring is to identify and quantify the effects of anthropogenic discharges and activities in the Baltic Sea, in the context of the natural variations in the system. In addition, the aim is to identify and quantify the changes in the environment as a result of regulatory actions.

The standards of MSFD criteria on GEnS require that the status of coastal fish stocks should be evaluated using the four abovementioned large fish indicators. However, unlike standardized annual survey trawl monitoring of pelagic and demersal fish stocks (sprat, herring, cod), coastal fish stocks do not have such standardized monitoring over the areas that would allow for spatio-temporal comparisons of the stocks status with their implications on food webs. That is, differences in data availability exist between (sub-)regions and statistical standardization of catching efficiencies of survey gear has not been made. Further, the abundance of small-bodied fish (gobies, pipefishes, sand laces, sticklebacks) cannot be evaluated with the present monitoring methods.

In the trap-net fishery, pelagic species (herring, smelt, sprat) are also caught in the coastal areas but with very high annual variation, which of course makes it more difficult to recognize impacts and interactions in food webs. The coastal monitoring methods used are not, however, specifically designed to catch pelagic species.

5.4 Indicators by fish species and groups

Despite limitations in the collection of long time series data from coastal fish stocks, the HELCOM FISH-PRO project has for over a decade followed the state of the coastal fish communities in the Baltic Sea. In 2012 the project harnessed various multivariate tools to find the strongest indicators and to deduce the significance of pressures on these fish indicators.

The fish species abundance indicator in the HELCOM FISH-PRO project has been agreed to focus primarily on perch or flounder, whichever is more characteristic for an area. Abundances of perch and flounder are limited by fishing but also by environmental factors, such as eutrophication, which degrades their spawning areas. Perch is one of the most abundant coastal fish species in the northern Baltic Sea and annual variation in recruitment of perch is very high (up to ten-fold). An important factor governing the recruitment success, growth, and year-class strength of perch is temperature (Böhling *et al.* 1991, Karås and Thoresson 1992, Karås 1996). However, with very abundant and mixing river- and estuary-spawning populations, acidification may overrule temperature that normally has major impact on the year-class formation of sea spawning perch (Hudd et al 1996). That is, local variation in the recruitment, caused by temperature and pH changes in the inner, intermediate and outer archipelago areas should not be confounded (confused) with cascading effects of coastal food webs over the sub-populations and areas.

The distribution of flounder is effectively restricted by low salinity, and its relative abundance is much smaller than that of perch in the northern Baltic Sea. Thus, flounder may not be an appropriate indicator species in the northernmost areas.
The abundance of fish species group indicator follows the abundance of cyprinid fish, non-piscivorous fish or piscivorous fish. Cyprinid fish have been seen as a potential indicator for severely increased eutrophication and an altered species composition of the coastal community, where the abundance of large piscivorous fish species have declined. In these two core group indicators (cyprinids and piscivorous fish) the data originates not only from survey gill-nets but also from commercial catches that have been used to detect trends in abundance. The abundance indicator of cyprinid fish is described in further detail in the by-catch chapter (Chapter 5) because cyprinid catch is mostly by-catch.

In the Archipelago Sea, one of the most valuable piscivorous fish in the professional and recreational fisheries is pike-perch, which is mainly caught with gill-nets. Along with the eutrophication of the coastal waters, the abundance of pike-perch has increased. Also the distribution of pike-perch has extended further north into the Bothnian Bay. During the last two decades, however, the reduced condition factor of pike-perch has caused active public debate but scientific study on the causes of the reduced condition factor has not been made. Thus, we cannot say whether or not the condition factor of pike-perch would be an appropriate indicator of GEnS.

5.5 Indicator recommendation

To address the size and abundance of large fish, and the structure of food webs, there is a need to assess the proportion of selected fish species at the top of food webs. Fishing certainly does change the biomass of piscivorous fish, which may cause cascading effect on benthivorous fish and macrozoobenthos. These cascades are sub-region specific and need to be considered further when using large fish as food web indicators. Indicators need to be further developed, based on the experience in some sub-regions. One such sub-area is the inner Archipelago Sea where the current abundance of large cyprinids is remarkably high, which on the other hand does not contrast food web interactions in the outer Archipelago Sea. For large fish, data are available from fish monitoring surveys and commercial catch statistics but estimating impacts of various pressures requires further consideration within sub-areas.

The abundance indicator of perch suggested by HELCOM FISH-PRO is suitable in the northern Baltic Sea. That is because perch is one of the most abundant fish species in the coastal waters and archipelago areas of the northern Baltic Sea and very long time series data on perch gill-net catch exist. Other major changes and trends in biological parameters (such as growth or L-W) of perch have not been recognized and hence the indicators based on abundance is proposed. Secondly, the abundance indicator of cyprinids is suitable especially in the inner archipelagos where cyprinids have largely hampered pike-perch and perch targeted fishery. Thirdly, the statistical standardization of catching efficiencies of survey gear should be done to allow statistically sound spatio-temporal comparisons between abundance indices of selected fish species (e.g. perch and cyprinids), which in turn would allow analyzing more detailed information on interactions of local food webs.

The current coastal monitoring using gill-nets is adequate but a knowledge gap exists regarding the methods for statistical standardization of various gill-net types and mesh sizes. The lack of monitoring of cyprinid catch and recommendations of cyprinid catch as a potential indicator of GEnS is explained in Chapter 5 ("By-catch and discards") because cyprinid catch is mostly by-catch.

6. By-catch and discards as indicators of change in the population status and food webs

Eero Aro and Pekka Jounela, RKTL

6.1 Introduction

Discarding is a global issue in fisheries. In 1994 the magnitude of the quantities of fish discarded was provided for the first time in an assessment published by FAO. The most recent FAO estimate for discarding is 20 million tonnes, which is about 25% of the reported annual yield from marine fisheries.

Discarding and by-catch of non-targeted fish occur mainly because most fishing gears and fishing practices are not sufficiently selective for the targeted fish sizes and because target species inhabit also areas which are occupied by a wide range of other species. Discarded fish are usually dead or moribund.

Catching and then discarding practices have consequences for example in stock evaluations and fishery management: firstly, the vast majority of specimens caught and then discarded are small and small specimens are sexually immature. This means a reduction of future spawning stock biomass, which at the moment is one of the key parameters in fishery management. Catching small fish reduces the growth potential of the stock and thus reduces the potential yield from a fishery with obvious economic consequences. Usually discarding rates are rough approximations or they are totally unknown. If so, discards represent in some cases a high source of uncertainty regarding the real fishing mortality rates exerted on stocks.

6.2 Definitions of “discarding”

There are several practices in discarding:

a) **Discarding of catch**. Fish or other animals, which have been retained by fishing gear, have been brought on board a fishing vessel or landed and have subsequently and voluntarily been returned to the sea or been disposed of.

b) **“Slipping” of fish.** This almost always occurs with purse seines and hence with pelagic fish, usually mackerel and herring. On some occasions, a catch is made but prior to bringing the fish onto a ship it is discovered that the fish are too small or of poor quality. The whole or part of the catch is then released, dead or moribund, into the sea.

c) **Escaping fish.** As in slipping but also including organisms which escape from fishing gears (usually nets) when the gears are at the surface of the sea immediately prior to being hauled onto a fishing vessel. For example, when a demersal otter trawl is brought to the surface of the sea, the tension in the netting of the gear is reduced and many fish may escape at that time.

Fish that escape through cod-end meshes also count as discarded. In general, escape mortality depends on many factors such as fish species, water temperature, mesh size, rigging, cod-end type and used netting material. In normal fishing conditions the escape mortality of trawl-caught Baltic cod has been estimated to be very low (<3%; Suuronen *et al.* 2005). In contrast, escape mortality of trawl-caught Baltic herring in the northern Baltic Sea has been estimated to be very high (91% and 62% for small and large escapees, respectively; Suuronen *et al.* 1996). The estimated high cod-end mortality, however, is a death percentage of escapees, which does not measure cod-end mortality rate in relation to catch. So, the high cod-end mortality estimates only suggest questionable justification and usefulness of cod-end mesh size management in the herring trawl fishery. That is, cod-end mortality does not provide adequate information on GEnS.

6.3 Reasons for discarding

There are number of reasons for discarding. They are mainly legislative and/or economic or just belong to fishing practices. In many instances the individual reasons operate simultaneously. In a number of cases legislation make discarding compulsory affecting both juvenile and adult specimens. For example European Community legislation implicitly or explicitly requires discarding of fish, molluscs and crustaceans for specimens which are smaller than defined minimum landing sizes, catches in excess of defined percentage compositions of catches taken with of nets of a given mesh size and catches in excess of quotas.

National legislation may also imply or insist on discarding. In some countries national quotas of fish are allocated at national level into sectoral quotas or Individual Transferable Quotas (ITQs). In practice, when a sector or ITQ of the fleet or vessel has taken its quota, and if no further quota becomes available, catches in excess of a sectoral quota or ITQ may not be landed but should be discarded, even if a national quota is not exhausted.

Reasons for discarding may be listed as follows:

1) Economically or legally wrong sized fish
2) Economically or legally wrong fish species
3) Damaged or spoiled fish
4) High grading of (i) economically more valuable fish or (ii) individual quota for later use
5) Exceptionally high recruitment
6) Poor gear selectivity
7) Seasonal ban of a certain fish species in a mixed species fishery

6.4 Discarding according to stocks in the GES-REG area in the Baltic Sea Main Basin and northern Baltic Sea (Sub-divisions 28-30 and 32)

Herring in the Baltic Sea Main Basin and Gulf of Finland (Sub-divisions 25-29+32)

Pelagic stocks in the Baltic Proper and Gulf of Finland (Subdivisions 25-29+32) are mainly taken in pelagic trawl fisheries, the majority of which take herring and sprat simultaneously. The estimates of pelagic catch compositions are mainly based on logbooks and landing declarations, with supplementary sampling of catches. According to WGBFAS (ICES 2012a), estimation of mixed clupeoid catches are considered uncertain and thus, it is difficult to make an accurate estimate on the proportion of herring and sprat in the landings from industrial trawl fisheries with small-meshed trawls. Although information on catch rates of the species is incomplete, discarding of herring at sea is not considered to be a problem for this stock and the amount of discards are minor.

Herring in the Gulf of Riga (Sub-division 28.1)

Herring fishery in the Gulf of Riga is performed by Estonia and Latvia, using both trawls and trap-nets. Herring catches in the Gulf of Riga include the local Gulf herring and the open-sea herring, entering the Gulf of Riga for spawning. Discrimination between the two stocks is based on the different otolith structures due to different feeding conditions and growth of herring in the Gulf of Riga and the Baltic Proper (ICES 2005). The discards of herring in the Gulf of Riga are assumed to be very rare and have not been recorded by observers working on the fishing vessels. Slipping and escaping of catch occur.

Herring in the Bothian Sea (Sub-division 30)

In the Finnish trawl fishery, the same trawls are often used in the pelagic and demersal trawling, and the trawls are classified as pelagic or demersal, depending only on their vertical position in the water column. In 2011 about 51% of the Finnish landings (75 100 tonnes) came with pelagic trawls, 45% with demersal trawls, 4% with trap-nets, and 0.001% with gill-nets. The Swedish part of the fishery has been annually about 5% of the total catches during the last 20 years. In 2011, 52% of the Swedish catch (about 3 400 tonnes) came from demersal trawls, 28% from pelagic trawls, and 20% from gill-nets.

Discarding rate of herring in Sub-division (SD) 30 is very small and of minor importance in the fishery and stock evaluations. It is generally known that some slipping and escaping occur just because of the nature of the main fishery.

Sprat in the northern Baltic Sea (Sub-divisions 28-30 and 32)

In Estonia the information on species composition of mixed sprat/herring catches is obtained from the logbooks, and from the observations of environmental inspectors, regularly visiting the landing sites. Finnish directed sprat fisheries takes place during autumn and early winter in the northern Baltic Sea, when dense sprat schools are available for pelagic single and pair trawling. The fishery is mixed Baltic herring/sprat fishery and usually neither discarding nor sorting take place just because the catch is used mainly for animal fodder. It is well known that some slipping and escaping of herring/sprat occur just because of the nature of the fishery, but the amount of this discarding is neither known nor sampled.

Cod in the eastern Baltic Sea (Sub-divisions 25-32)

Presently cod and cod fishery are missing in the northern Baltic Sea Main Basin. According to official statistics the total catch of cod in Finland and Estonia in Sub-divisions 29 and 32 are just a few tonnes annually and by-catch and discarding of cod is zero. However, if the cod stock in the northern areas will increase, it is expected that total catches, by-catches and discarding will increase as well.

In Latvia trawlers are catching cod mainly with bottom trawls (81% of their catches). The other important fishery is fishing with gill-nets and minor importance is long-lining (approximately 0.5% of catches). Cod fishery has been decreasing in recent years because cod is currently more abundant in the Bornholm Basin (SD 25). The discarding rate has been 2.5–13.6% in 2002-2011. The main reasons for discarding of the eastern Baltic cod have been either an exceeded national Total Allowable Catch (TAC), the catching of fish smaller than the minimum landing size (less than 38 cm), or the too low economic value of caught fish. For example, high recruitment of cod year class 2003 initially increased discarding, since the recruiting fish were too small to be landed.

Flounder in the northern Baltic Sea (Sub-divisions 28-30 and 32)

It is assumed that there are four local flounder stocks/populations in the GES-REG area in the northern Baltic Sea. These are: one in the Irben Strait, including the western part of the Gulf of Riga and up to the Hiiumaa island in Estonia (SD 28 East); one in SD 29 and SD 30; and two stocks in SD 32, one along the Finnish coast (SD 32 North) and one along the Estonian coast (SD 32 South)(Aro 1989). The state of the stocks is, however, unknown and proper estimates of discarding do not exist. Discarding is caused mainly by undersized fish, low quality of the fish and low market demands during off-season.

It is well known that the amount discarded flounder in the demersal cod targeted trawling fishery is very high and very variable depending on fishery and season. Rough estimates suggest that flounder discards are five to ten times higher than landed by-catch of flounder in the cod targeted trawl fishery. Thus discarding estimates are prerequisites for reliable stock assessment of flounder, which is based on catch at age data. In the southern Baltic Sea sampling of discards for flounder in the cod targeted fishery has been initiated, but the discard patterns are so heterogeneous between the fleets, vessels and even individual hauls of the same vessel and trip that a common raising procedure has not been applied.

Salmon and sea-trout in the Baltic Sea Main Basin, Gulf of Finland (Sub-division 32) and Gulf of Bothnia (Sub-divisions 24-31)

The main observed reason for salmon discards in the northern Baltic Sea salmon fisheries are seal damages on adult salmon in coastal trap-nets. The seal-induced salmon catch and gear damages occur mainly in the Gulf of Riga, Gulf of Finland and Gulf of Bothnia. By-catch of young salmon occurs in the Baltic Sea Main Basin and in different types of fisheries, but probably also within pelagic sprat and herring trawl fishery where it is likely to remain unnoticed (e.g. ICES 2012b). Discarding also occurs in the longline salmon fisheries, in terms of mortality among undersized individuals that are released back into the sea. However, data on discards from different fisheries in the Baltic Sea Main Basin are incomplete and fragmentary. Thus, conversion factors have been applied to obtain estimates for the total number of salmon caught, landed and discarded. The magnitude of the present salmon discards and unreported salmon catch is presumed to be highly variable between regions. In general, salmon discards account for 27–50% of the total salmon catch *in numbers*. Some of these conversion factors may well be too low, especially considering the potentially high by-catch of small salmon in the large-scale pelagic trawling fishery (ICES 2012b). So far, however, very little is known regarding the magnitude of discarding. Thus, annual changes in the corresponding catch/discard- multipliers have not been made. One example of poor information is the reported Polish Catch Per Unit Effort (CPUE) data of salmon and sea-trout that have markedly deviated from corresponding CPUE data delivered by other countries fishing with the same gears and in the same salmon fishing areas in the southern Main Basin. This indicates that salmon have been extensively misreported as sea-trout in the Polish salmon fishery. This may be caused by the lower Polish salmon TAC from 1994 onwards.

Overall, discarding of salmon and sea-trout is either local (e.g. seal-induced catch loss in trap-nets) or unknown (e.g. post-smolts in trawl fishing) and hence we were not able to justify how discarding of salmon and sea-trout would provide information on GEnS.

Cyprinids

In the coastal areas of the northern Baltic Sea, a key fish group which is currently affected by human activities both in terms of by-catch and discards is cyprinids. Along with the eutrophication of the coastal waters, the populations of cyprinid fishes (*Cyprinidae*), such as bream, roach and silver bream, have increased dramatically in the coastal areas of the northern Baltic Sea (Figure 6.1).

Large cyprinid by-catch have largely hampered professional, subsistence and recreational gill-net and trap-net fishing that mostly target commercially valuable fish species (e.g. whitefish, perch, pike-perch and sea-trout). Cyprinids may also have had some impact on commercially more valuable fish populations and coastal food webs.

Figure 6.1. *Cyprinid by-catch (tonnes) in the coastal areas of the northern Baltic Sea in 1990, 2000 and 2010. In 2010, the by-catch of cyprinids was 998 tonnes and the catch of other coastal fish species was 3000 tonnes. It is notable that fishing in the northern Baltic Sea mainly targets other species than cyprinids. Hence, un-standardized by-catch/catch proportions very likely underestimate abundance proportions of cyprinids to other fishes.*

To overcome the cyprinid by-catch problem, professional fishers have annually removed approximately 500 tonnes of cyprinids. In recent years the targeted removal of cyprinids has extended to 1000 tonnes. The long term aim of these cyprinid removals are to actively remove nutrients (phosphorous) from the coastal waters. Thus, cyprinid removals by professional fishermen, who mainly use large trap-nets, have become a state-subsidized activity. This is because cyprinid removal has been estimated to be an efficient way to remove nutrients from the coastal area in the northern Baltic Sea. The cyprinid removal and the net nutrient loading in other sectors may be extended towards a wider emission trade system, in which not only fish farming but also e.g. wastewater treatment plants, or agriculture, could buy nutrient removal services from the fishers. The potential future removal of cyprinids as a means of nutrition removal is uncertain but it has been speculated to be in the order of tens of mill. kg annually. Potential (by-)catch numbers and the production potential of current coastal cyprinid populations are, however, approximates at best.

The current non-standardized removal of cyprinids with trap-nets is not an appropriate measure of the actual status of cyprinid stocks. If standardized properly, however, cyprinid removal could potentially be an appropriate indicator of GEnS or regime shifts in coastal food webs along with standardized gill-net monitoring.

The “targeted by-catch” fishing of cyprinids and other under-utilised coastal fish species are nowadays primarily restricted by the insufficient demand of markets for the minced cyprinid products that are aimed for human consumption (mainly fish meal and oil). Therefore, without state subsidization active removal of cyprinids is not commercially viable. Globally there is a growing demand for fish protein and already today the export market is capable of offer prices for underutilized fish species that are close to the profitability limit for fishers. The cost-effectiveness of cyprinid fishing as a means of nutrient removal is currently being assessed in the COEXIST EU-project. The COEXIST assessments also compare direct and indirect costs of cyprinid fishing as compared to other activities in other sectors aiming to reduce nutrient net loads into the northern Baltic Sea.

Coastal piscivorous fish

Discarding of piscivorous fish in the coastal mixed species fishery occurs only in a small scale because piscivorous fish are mainly used for human consumption. Hence the discards are usually attributed only to undersized fish. Coastal piscivorous fish discarding is not monitored and thus, we were unable to recognize effects between discards of coastal piscivorous fish and indicators of GEnS.

6.5 Indicator recommendations

By-catch and discarding of cyprinids could be potential indicators of GEnS in the coastal areas of the northern Baltic Sea. The targeted removals of cyprinids are currently state subsidized and hence, targeted subsidized and non-targeted WFD-driven monitoring practices should be gear standardized jointly with proper statistical tools in order to provide information on GEnS. Regarding other species and species groups in the northern Baltic Sea, we were not able to recognize any relevant links between by-catch, discarding and GEnS.

7. Early warning indicators: zooplankton

Laura Uusitalo, SYKE

7.1 Introduction

Zooplankton has a crucial role in pelagic food web dynamics: it transfers energy from primary production to a form utilizable by fish. Zooplankton is affected by changes in primary production, indicative of eutrophication, and by changes in the structure and abundance of the fish community, indicative of overfishing. Therefore, zooplankton is sandwiched between top-down and bottom-up dynamics, and can potentially yield a lot of information on the state and dynamics of the ecosystem.

Zooplankton consists of animals that live in the water column and drift with the water masses or swim slowly. Some animals are planktonic throughout their whole life cycles, while others have planktonic phases; the latter are called meroplankton. Zooplankton is often divided into the following groups according to the size of the animals (The Baltic Sea Portal, http://www.itameriportaali.fi/en/tietoa/elama/elioryhmat/elaimet/elainplankton/en_GB/elainplankton/):

1) nanoplankton 2-20 µm (flagellates)
2) mikroplankton 20-200 µm (ciliates, small rotifers)
3) mesoplankton 0.2-2 mm (large rotifers, cladocerans, copepods, different meroplanktonic larvae)
4) makroplankton >2 mm (mysids, juvenile fish)
5) megaplankton max. 0.2-2 m, in the Baltic Sea typically 10-15 cm (jellyfish)

Zooplankton research traditionally largely focuses on mesozooplankton due to practical reasons such as its catchability and identifiability. In the northern Baltic Sea, copepods and cladocerans (water fleas) are the largest and most important groups of mesozooplankton. Zooplankton feed on bacterioplankton, phytoplankton, detritus, and other zooplankton. Microphagous zooplankton are those that are able to ingest microscopic particles (Le Fèvre *et al.* 1998). In the northern Baltic Sea, rotifers, some cladocerans, and copepod nauplius stages, are microphagous, whereas adult copepods and predatory cladocerans can feed also on small zooplankton. Zooplankton, in turn, is important food source for fish larvae as well as many planktivorous fish species.

In this chapter, we review the zooplankton-related food web health indicators that have been proposed in the Baltic Sea area (Annex I) and in addition some other zooplankton-related food web indicators that have been proposed in an earlier phase of the HELCOM CORESET project (HELCOM 2012). The level of information available regarding the specifics of these indicators varies, which may affect the level of analysis that can be presented here.

7.2 Review and evaluation of suggested indicators

Biomass of copepods, biomass on microphagous zooplankton, biomass ratio of copepods and mesozooplankton

The biomass of zooplankton components indicators appear in the Marine Strategies of the Baltic Sea countries in different forms: *Biomass of copepods and mesozooplankton* in Denmark, *Biomass of copepod and biomass of microphagous zooplankton* in Lithuania, and *Copepod biomass / mesozooplankton biomass ratio* in Estonia (Annex I). In addition, the HELCOM CORESET interim report (HELCOM 2012) lists the *Biomass of copepods* and *Biomass of microphagous* zooplankton as indicator candidates; however these were in the end not accepted to the core set on indicators (HELCOM 2013a).

Zooplankton biomass correlates positively with phytoplankton biomass and hence with eutrophication; in particular, small-bodied, filter-feeding (microphagous) zooplankters increase with increasing eutrophication (Gliwicz 1969, Pace 1986, Hsieh *et al.* 2011). On the other hand, the large-bodied zooplankters, especially copepods, constitute the best-quality food items for the zooplanktivorous fish (e.g. Cardinale *et al.* 2002, Rönkkönen *et al.* 2004). Rönkkönen *et al.* (2004) reported that in the Gulf of Finland, herring growth correlates positively with the abundance of the marine zooplankton species *Pseudocalanus minutus elongatus*. In addition to this species, the role of many copepods as key species in the Baltic Sea pelagic ecosystem has been established (e.g. Möllmann and Köster 1999, 2002, Hinrichsen *et al.* 2002, Möllmann *et al.* 2003).

The MSFD Task Group 4 report (Rogers *et al.* 2010) specifically states that *"In some cases, when representative species cannot be evaluated, functional groups can be considered."* While no obvious indicator species has been established, copepods, as well as microphagous zooplankters, are strong candidates for relevant functional groups: they are important, functionally distinctive zooplankton groups which have demonstrated links to other trophic levels (phytoplankton and planktivorous fish). The total biomass of copepods, as such, without modifications, describes the availability of good-quality food, and serves well as a food web indicator, while the biomass of microphagous zooplankton is indicative of changes in phytoplankton.

The ratio of copepods to total mesozooplankton biomass, i.e. the share of copepods of the total zooplankton, aims to describe the zooplankton community structure. However, while the ratio may be indicative of the size distribution of the zooplankton community, it does not directly relate to availability of high-quality food for planktivorous fish, as the availability depends largely on the absolute amounts, not of the share, of the large-bodied zooplankton. Therefore the biomass of microphagous zooplankton as well as the biomass ratio of copepods to total mesozooplankton may serve as indicators of the cascading effect of eutrophication in the food web (from nutrients to zooplankton), rather than of the food web effects from primary producers to higher trophic levels.

The Commission Decision as well as the Task Group 4 Report (Rogers *et al.* 2010) specifically address abundance, not biomass. While biomass is normally calculated based on abundance estimates and the average weights of individuals, abundance and biomass are conceptually separate issues that describe somewhat different functions. Despite this, abundance and biomass are tightly correlated, and therefore the zooplankton groups' biomasses would most likely make good indicators also for food web health: a high copepod biomass indicates that zooplanktivorous fish have enough high-quality food, signalling that energy is transferred from primary producers up the food web towards higher trophic levels.

Zooplankton mean size and total abundance

Note that there is conflict of interest: the author of this chapter L. Uusitalo is also one of the authors of this indicator in the HELCOM core set.

This indicator has been developed in the HELCOM zooplankton expert network (ZEN, leader E. Gorokhova), and it appears in the Finnish and Latvian Marine Strategy indicator sets as well as in the HELCOM core set of indicators, albeit under slightly different names (see Annex I). The scientific justifications behind this indicator are largely similar to the copepod and microphagous zooplankton biomass indicators above.

The mean size total stock indicators aim to synthesize these two factors, mean size and abundance of mesozooplankton community, into one indicator: abundant zooplankton with high mean size would indicate good feeding conditions for zooplanktivorous fish as well as high potential grazing on phytoplankton; while other combinations (small total stock or small mean size or both) would indicate limitations in the ability of the zooplankton community to transfer energy from primary production to higher trophic levels (HELCOM 2013b, Gorokhova *et al.* in prep.).

A further complication of interpreting the results arise from the considerable amount of random variation related to the results: the results from reference and non-reference periods have a considerable overlap (MSTS indicator description sheet). This may partly be due to the randomness related to zooplankton sampling; once-a-year sampling may not be sufficient if the aim is to use zooplankton as an early warning indicator, since detecting the changes amidst the random variation requires higher amounts of data. In addition, the indicator author team notes (MSTS indicator description sheet) that the mean sizes are based on biomasses which are estimated using default constant individual weights. Development of automatic measurement methods of zooplankters may help getting more precise assessments in the future.

In summary, the indicator has a solid scientific basis and it addresses a relevant aspect of zooplankton as the mediator of energy from primary producers to fish. However, the inherent noise in zooplankton data presents a challenge in setting the GEnS boundaries, as well as evaluating the indicator values year to year. Development of automatic analysis methods may aid by cost-effectively providing larger amounts of data, making the indicator evaluation more robust to random variation. It can be seen to fall into the Commission Decision criterion 4.3 "*Abundance/distribution of key trophic groups*", if 'trophic group' is defined widely so that it covers zooplankton in its entirety. This seems reasonable, as there is contradicting evidence about which specific species would make the best indicators.

Mean zooplankton size

The mean zooplankton size is mentioned in the HELCOM CORESET interim report (HELCOM 2012) as a biodiversity indicator, but no description of the indicator is available. The interim report briefly notes that *"larger zooplankton size indicates a better state of the environment*"; the rationale behind this indicator candidate being the same as in the two above-mentioned indicators: if there are more copepods, the mean size of the zooplankton community members is larger, and if there are more microphagous zooplankters, the mean size is smaller.

Conceptually, this indicator is closely related to the above-mentioned indicator candidates, and in a sense summarizes much of their message. However, strictly speaking, it falls outside the criterion 4.3 "*Abundance/distribution of key trophic groups*", and does not fit any of the other descriptor 4 criteria either.

Zooplankton-phytoplankton biomass ratio

The zooplankton-phytoplankton biomass ratio indicator is mentioned in the HELCOM CORESET interim report (HELCOM 2012), but no detailed description is available. The interim report explains that *"The indicator was meant to follow long-term changes in the biomass ratio of zooplankton and phytoplankton as a response to changes in food web (predation pressure) and eutrophication (hypoxia, nutrient availability). Bias to zooplankton indicates stronger top-down control and hence a better functioning food web (piscivorous fish controlling planktivorous fish, releasing zooplankton from high predation.)"*

The Zooplankton to phytoplankton ratio has been used to indicate the efficiency of conversion from primary to secondary production (phytoplankton to zooplankton) (Richman *et al.* 1990; Stock and Dunne 2010), and to distinguish different levels of eutrophication (Blank *et al.* 2010). On the other hand, planktivory by fish affects the ratio as well, and therefore it can be used to indicate changes in the fish stocks (Jeppesen *et al.* 2011).

Since this indicator focuses on the biomass ratio, it does not directly fall to the Commission Decision criterion 4.3 "*Abundance/distribution of key trophic groups/species*". However, phytoplankton and zooplankton are trophic groups in a broad definition of the term, and their ratio may give valuable information about the energy flow in the low trophic levels of the pelagic ecosystem.

7.3 Setting targets to these indicators

The MSFD (Article 3) defines good environmental status in the following manner: "*'good environmental status' means the environmental status of marine waters where these provide ecologically diverse and dynamic oceans and seas which are clean, healthy and productive within their intrinsic conditions, and the use of the marine environment is at a level that is sustainable […]*" Operationalizing this definition means setting up tools (indices/indicators) to evaluate the environmental state, and setting some targets that indicate conditions in which the ecosystem component in question has not been, or has been just slightly, adversely affected by human activities (Borja *et al.* 2012).

Borja *et al.* (2012) discuss the importance and various methods of setting targets. They identify seven approaches for identifying target levels:

- Pristine areas
- Historical data of pristine era

- Modelling pristine state
- Baseline set in the past
- Baseline set in the present
- Trend target
- Best professional judgement

The Baltic Sea has undergone changes due to eutrophication since the 1950s–1960s (e.g. Hällfors *et al.* 2013 and references therein), to the degree that no pristine areas with which to define targets exist today. We also lack compatible quantitative historical data originating in pristine times. Other methods for target setting have to be applied.

For the MSTS indicator, Gorokhova *et al.* (in prep.; MSTS indicator description sheet) have proposed GEnS boundaries based on reference periods of low chlorophyll *a* concentration (indicative low phytoplankton biomass) and good planktivorous fish growth (indicative of good food availability for fish). These reference periods have been defined based on historical data, and they are expected to describe an environmental state where the ecosystem has functioned well. This approach falls into the "baseline set in the past" category in the above list.

For the rest of the proposed indicators, targets would most likely be set in a similar manner, finding historical data of "healthy and productive" periods and defining the targets accordingly. In the absence of such data or such periods, the other approaches listed in Borja *et al.* (2012) could be applied, such as defining that the indicator value should not alter for the worse from the current state (baseline set in the present) or that it must continually improve (trend target).

7.4 Indicator recommendation

The MSTS indicator is be included into the HELCOM core set of indicators, and will therefore be estimated in the whole Baltic Sea. Therefore the primary effort should be in developing comparable methodology to measure the value of this indicator, as well as comparable criteria to set the targets for different sea areas.

In addition, it should be evaluated whether the other zooplankton indicators would give additional information about the state of the pelagic ecosystem on top of that gained by MSTS indicator.

7.5 Suitability of current monitoring and assessment methods

The Task Group 4 report (Rogers *et al.* 2010) recommends integrated annual values to be used as the temporal unit of all zooplankton variables. All of the reviewed indicators are based on zooplankton and phytoplankton data that is already collected in the Baltic Sea as part of the HELCOM COMBINE programme. The HELCOM CORESET interim report (HELCOM 2012) notes, however, that *"Unfortunately, in some areas, sampling coverage is low and not all sea areas are equally well represented"*. Therefore, some additional effort might be needed in the monitoring, but all in all, the application of these indicators would not impose many additional requirements on existing monitoring programmes.

Zooplankton data is generally “noisy”, i.e. it includes a lot of random variation; a more intensive sampling would decrease the uncertainty around the indicator values. In addition, long time series and data originating from periods of healthy food webs are needed to set the target levels.

The biomass estimates of zooplankton are currently based on abundance counts, from which the total biomass is computed based on average biomasses for each species or genus and, where applicable, sex and developmental state. This approach optimises the use of zooplankton analyst time, as measuring the individuals in addition to counting them would increase the time required to analyse a sample. However, the drawback is that possible changes in the sizes of individuals are not recorded; it is possible that the mean size of individuals within a species and age group varies, and ignoring this may introduce some inaccuracy to the estimates. Introduction of automatic methods for measuring (some of the) individuals would cost-efficiently provide more accurate biomass estimates.

Since all of the zooplankton indicators presented here require only group-level information of the zooplankton abundance, automated sample processing methods, capable of counting, identifying, and measuring the zooplankton individuals (e.g. Bell and Hopcroft 2008, Gislason and Silva 2009, Fernandes *et al.* 2009, MacLeod *et al.* 2010, Di Mauro *et al.* 2011, Manríquez *et al.* 2012, Uusitalo *et al.* in prep.), could be considered also in the Baltic Sea to complement the data produced by trained analysts. This is noted also in the MSTS description sheet: *“Direct measurements by size scanners would be needed.”*

8. Early warning indicators: phytoplankton

Heidi Hällfors and Laura Uusitalo, SYKE

8.1 Introduction

In the early 1900s, the role of phytoplankton as food for higher trophic levels was one of the main motivators for the first large-scale phytoplankton investigations in the Baltic Sea (cf. Kyle 1910, Richardson 2002). At about the same time, the link between polluted coastal waters and the proliferation of certain phytoplankton taxa was recognized (Levander 1908, 1913, cf. Finni *et al.* 2001a), and subsequently such eutrophication and environmental pressure indicators became the focus of phytoplankton indicator development in the Baltic Sea (e.g. Heiskanen *et al.* 2005, Carstensen and Heiskanen 2007, Wasmund *et al.* 2008, Jaanus *et al.* 2009, Hällfors *et al.* 2013a). More recently the scope of phytoplankton indicators has widened ensuing work towards phytoplankton biodiversity indicators (e.g. HELCOM 2012, Uusitalo *et al.* 2013, and references therein), whereas phytoplankton food web indicators are to date still largely lacking (cf. Annex 1).

Primary production constitutes the basis of all food webs, and in the pelagic ecosystem, phytoplankton is responsible for practically all primary production. Considering the aquatic food web on a rough level, primary production (often using ocean colour or chlorophyll *a* as a proxy) is a good predictor of the potential fisheries yield of the world's oceans (Chassot *et al.* 2007, Chassot *et al.* 2010). Also in European seas, among them the Baltic Sea, there is a strong coupling between primary productivity and fisheries production over long time scales (several years to decades; Chassot *et al.* 2007). Further, it has been put forward that a change in the content and ratio of substances exclusively synthesized by primary producers, such as certain essential fatty acids (Ahlgren *et al.* 2005, Litzow *et al.* 2006), pigment precursors (Andersson *et al.* 2003, Van Nieuwerburgh *et al.* 2005), and vitamins (Sylvander *et al.* 2013), may via trophic transfer affect food web health and composition all the way to the pelagic fish communities.

On a more detailed level, aquatic food webs are typically complex and the linkages governing top-down regulation (through grazing and predation) and bottom-up (i.e. resource limitation) processes among the different components and trophic levels are hard to resolve. *In this chapter we focus on the bottom-up regulation of the Baltic Sea food web, specifically, on the regulating effect of phytoplankton as food.*

8.2 State of the art of phytoplankton food web indicators

Phytoplankton food web indicator development is in its early stages. The Task Group 4 report on food webs (Rogers *et al.* 2010), while considering phytoplankton a potentially useful early warning food web indicator, and giving some examples of indicators which could be developed further (see

Annex 6 in Rogers *et al.* 2010) recommends however no specific phytoplankton indicators. Consequently, the subsequent Commission Decision (2010/477/EU) which accompanies the EU Marine Strategy Framework Directive (2008/56/EC) (MSFD), also does not specify any particular phytoplankton indicators relating to food webs (see Chapter 1 of the present report), but does address phytoplankton under the food web descriptor 4 in Criterium 4.3. *Abundance/distribution of key trophic groups/species*.

In the three years since the Commission Decision (2010/477/EU), there has on the whole been only minor progress concerning phytoplankton food web indicators in the Baltic Sea area. The HELCOM CORESET project final report lists no ready phytoplankton (biodiversity, hazardous substances or eutrophication) core indicators, pointing out that phytoplankton indicator development turned out to be both time-consuming and scientifically challenging (HELCOM 2013a). HELCOM MONAS noted the need to develop indicators for phytoplankton, and requested experts to further develop them by 2015 (HELCOM 2013a).

The previous, interim, report of the HELCOM CORESET project (HELCOM 2012) lists four phytoplankton related candidate and supplementary indicators; three are relevant in food web indicator development. The first of the three, the *Zooplankton-phytoplankton biomass ratio* is discussed in Chapter 7 of the present report. The second, the *Seasonal succession of phytoplankton groups*, is under development (cf. HELCOM 2012) and proposed in the Estonian Marine Strategy (cf. Annex 1); the third, the *Ratio of diatoms and dinoflagellates*, is proposed in the Finnish Marine Strategy (cf. Annex 1, as the *Diatom/dinoflagellate ratio*). Among the food web indicators identified or proposed in the Marine Strategies of the Member States there are two further phytoplankton-related indicators; the *Cyanophyte/total phytoplankton biomass ratio* and the *Summer phytoplankton biomass* indicators, in the Finnish and Latvian Marine Strategies, respectively (Annex 1). Since the Estonian and Latvian phytoplankton food web indicators are already under development or even operational (cf. Annex 1), we focus in the present report on the indicators deemed to be in most urgent need of attention, i.e. the *Diatom/dinoflagellate ratio* and *Cyanophyte/total phytoplankton biomass ratio* indicators.

8.3 Indicator development: rationale and indicator hypotheses

Background

Results by Suikkanen *et al.* (2013) indicate that recent shifts in the phytoplankton composition toward low-food-quality phytoplankton could together with stressors such as climate warming mean leaner times for mesozooplankton grazers. In line with this, Vehmaa (2012) found that the resilience of copepods against environmental changes depends substantially on the quality of their phytoplankton diet. Since mesozooplankton constitute an important link between primary producers and higher trophic levels in the Baltic Sea food web (e.g. Flinkman 1999), there is an evident and timely need to understand the effects that variation and changes in the phytoplankton biomass and species composition have on the mesozooplankton community. Therefore, the focus of this chapter is on phytoplankton as food for mesozooplanktonic grazers.

However, there is another aspect to phytoplankton as food also. In the northern Baltic Sea, a large part of the spring bloom phytoplankton production sinks to the bottom ungrazed (Lignell *et al.* 1993, Heiskanen and Kononen 1994), providing the main annual input of food to the benthic ecosystem (Kuparinen *et al.* 1984, Tallberg and Heiskanen 1998), and tightly linking the benthic and pelagic food webs. We will touch on this facet also, albeit briefly.

The factors causing variation in phytoplankton species composition (beyond the effects of seasonal succession, temperature, and salinity) are notoriously difficult to determine. This is apparently due to the existence of many interrelated environmental (abiotic and biotic) causative variables, not all of which can at present quantified, as well as due to time-scale problems; the phytoplankton community and environmental parameters are as a rule sampled at the same time whereas the phytoplankton community oftentimes reflects environmental conditions somewhat *prior* to the time of sampling. Since we still do not fully understand the complex cause-and-effect relationships, it has been difficult to identify phytoplankton indicators which respond to environmental pressures in an unambiguous manner (e.g. Carstensen and Heiskanen 2007, Hällfors *et al.* 2013a). The relevant issue in the context of the MSFD is which part of the variation in the phytoplankton community is “academic” and which part has an actual (immediate) effect on (the classification of) the state of the sea. Irrespective of the *causes* for the variations in phytoplankton community composition, the species composition has a documented effect on the *food web* (see below). *Therefore, from the perspective of the food web, we argue that phytoplankton community composition serves as an early warning indicator of the quality and quantity of available food, and therefore the health of the food web and thus ultimately the state of the sea*.

Nutrition of mesozooplankton

Mesozooplankton comprises a variety of different organisms (e.g. Telesh *et al.* 2009), of which copepods, cladocerans and rotifers are the most important in the northern Baltic Sea (cf. Viitasalo 1992a, Viitasalo *et al.* 1994a, Ojaveer *et al.* 1998), and copepods constitute the most intensively investigated group. Many mesozooplankton species, among them most copepods, are omnivorous, feeding on both phytoplankton and microprotozoans, as well as detritus (Turner 1984, Uitto *et al.* 1997). This means that primary producer biomass does not account for their total food availability. Consistently, while distinct bottom-up linkages have been shown for e.g. phytoplankton and predominantly herbivorous copepods (Richardson and Schoeman 2004), the extent to which primary productivity may regulate copepod production varies with copepod species and with environments (Runge 1988). In the southern Baltic Sea, a strong coupling between phytoplankton biomass (as chlorophyll *a* of cells >11 µm) and copepod production and egg production was found, these being highest in connection with phytoplankton blooms (Kiørboe and Nielsen 1994).

While not all carbon flows directly from primary producers to mesozooplankton, the intermediaries, i.e. microprotozoan consumers (mainly comprising ciliates and heterotrophic dinoflagellates), are also important food for mesozooplankton (Calbet and Saiz 2005, Sherr and Sherr 2007, Calbet 2008). Contrary to common expectations (cf. Kivi 1996), recent results revealed that microprotozoans are not always nutritionally superior to phytoplankton as food for copepods (Broglio *et al.* 2003, Tang and Taal 2005). Nor do copepods necessarily positively select for ciliates over phytoplankton as food, even when offered together with low-quality phytoplankton (Siuda and Dam 2010). The relative importance of ciliates and phytoplankton as food varies with the trophic state of the system, with primary producers contributing the majority of the copepod diet in environments characterized by high phytoplankton biomass (Calbet and Saiz 2005), such as the Baltic Sea (cf. Kiørboe and Nielsen 1994, Uitto *et al.* 1997). More specifically, Kiørboe and Nielsen (1994) found that in the southern Baltic Sea, phytoplankton was of great importance in fuelling copepod secondary production during periods of high phytoplankton biomass, and that microprotozoans contributed mainly by maintaining copepod populations during periods of low phytoplankton availability and, likely, by ensuring a nutritionally complete diet (Kiørboe and Nielsen 1994, cf. Tang and Taal 2005). The variations in the importance of herbivory and carnivory seem to be related to the size structure of the phytoplankton assemblage, with herbivory prevailing when large-celled phytoplankton dominate, i.e. in nutrient-rich and high-turbulence conditions such as during the spring bloom (Lignell *et al.* 1993). Correspondingly, predation on microprotozoans becomes more important when small-celled phytoplankton predominate, i.e. in nutrient-poor and low-turbulence situations such as during the post-spring bloom period (Lignell *et al.* 1993).

Mesozooplankton growth and reproduction are affected by both food quantity (Checkley 1980, Koski and Kuosa 1999, Niehoff 2004) and quality (Gulati and DeMott 1997, Koski *et al.* 1998, Schmidt *et al.* 1998, Broglio *et al.* 2003). The quantity of phytoplankton in the northern Baltic Sea varies between years, but on an annual scale two biomass peaks occur; the first, dominated by diatoms and dinoflagellates in spring, and the second, consisting primarily of cyanophytes, in late summer (Hällfors *et al.* 1981, Wasmund and Siegel 2008).

The spring bloom

At the time of the diatom and dinoflagellate spring bloom period in April–May, herbivory is the major nutritional mode for mesozooplankton in the coastal western Gulf of Finland (Lignell *et al.* 1993). Mesozooplankton adults are not yet particularly numerous in the northern Baltic Sea (Viitasalo *et al.* 1994a-b) whereas juvenile stages of the copepods *Acartia* spp. occur abundantly (Viitasalo 1992b, Kivi *et al.* 1993, Lignell *et al.* 1993, Katajisto *et al.* 1998). Although the mesozooplankton community peaks only later in the year (Viitasalo *et al.* 1994a-b, Ojaveer *et al.* 1998) and is thus unable to draw the full benefit from the voluminous spring bloom (Lignell *et al.* 1993), springtime phytoplankton is important in fuelling mesozooplankton reproduction (Peinert *et al.* 1982). Consistently population growth is vigorous with increasing temperatures in spring (Ackefors 1981). Indeed, the spring bloom may affect mesozooplankton growth and reproduction for a fairly long time, particularly in species with slow development or those which can store energy in their tissues (Viitasalo 1994). It has been suggested (cf. Ianora *et al.* 2004, Ask *et al.* 2006) that the time lag between the annual phytoplankton biomass maximum and that of mesozooplankton is caused by the longer generation time copepods (Colebrook 1979), the life cycle of which includes 11 juvenile stages (Katajisto 2006) or alternatively (or in combination with) due to the inhibiting effect of some diatoms on the reproductive success of copepods (Starr *et al.* 1999, Ianora *et al.* 2004).

As a consequence of the low zooplankton grazing pressure, a large part of the spring bloom phytoplankton production sinks to the bottom ungrazed (Lignell *et al.* 1993, Heiskanen and Kononen

1994). How great a fraction sediments to the bottom depends on the species composition; diatoms and their resting spores sink, as do dinoflagellate resting cysts, while the bulk of the dinoflagellate population disintegrates in the water column producing slowly settling phytodetritus (Heiskanen 1993, Heiskanen and Kononen 1994, Tamelander and Heiskanen 2004). Thus it seems plausible that if the spring bloom is dominated by dinoflagellates, a larger fraction of the organic matter produced during the spring bloom remains in the water column for the heterotrophic plankton community to consume during summer, than if the bloom is formed mainly by diatoms (cf. Heiskanen and Kononen 1994).

An increase in the dinoflagellate to diatom ratio has been reported from different areas of the Baltic Sea, particularly for the spring bloom period (Hällfors *et al.* 2013a and references therein). Recent evidence bespeaks a cyclic variation of the diatom to dinoflagellate ratio, with alternating dominance periods lasting ca 10 years (Wasmund *et al.* 2011 and references therein). The plausible answers to the alternating dominance seem to be linked to a combination of environmental conditions and the intrinsic characteristics of the organisms themselves (Hällfors *et al.* 2013a and references therein), however, definitive explanations as to which conditions favour one group over the other are lacking; as is knowledge on the potential consequences for the Baltic Sea food web.

The late summer bloom

In the northern Baltic Sea, cyanophytes recurrently form extensive blooms (Kahru *et al.* 2007). These mass occurrences occur predominantly in late summer, at the time of the highest mesozooplankton biomass and production (Viitasalo 1992a, Viitasalo *et al.* 1994b, Ojaveer *et al.* 1998). As in spring, herbivory is throughout the summer the main nutritional mode for mesozooplankton, with the ratio of microprotozoans in the diet increasing towards the end of the season (Uitto *et al.* 1997). The species dominating the late summer cyanophyte bloom are the filamentous *Aphanizomenon flos-aquae, Nodularia spumigena* and *Anabaena/Dolichospermum* spp. (Laamanen and Kuosa 2005), which are variously grazed by mesozooplankton (e.g. Sellner *et al.* 1994, 1996, Koski *et al.* 2002, Kozlowsky-Suzuki *et al.* 2003, 2007). Little biomass sink to the bottom since most of the cyanophytes decompose already in the water column (Kuparinen *et al.* 1984, Heiskanen and Kononen 1994, Tallberg and Heiskanen 1998), and the remaining organic matter stays in the water column for the heterotrophic plankton community to consume.

Although filamentous cyanophytes were important members of the phytoplankton community already in the early 1900s (Hällfors *et al.* 2013a), the occurrence and intensity of cyanophyte blooms (Poutanen and Nikkilä 2001) and *Aphanizomenon flos-aquae* and *Nodularia spumigena* in particular (Finni *et al.* 2001b) increased with the progressing eutrophication during the course of the 20th century. As with changes in the springtime diatom to dinoflagellate ratio, the effects of variation in the summertime cyanophyte occurrences on the food web are largely unknown. However, recent result suggest that changes in the summertime phytoplankton community composition may have contributed to a shift in the grazer community toward smaller-sized mesozooplankton, a development considered unfavourable for higher trophic levels (cf. Suikkanen *et al.* 2013).

Food quality

The quality of phytoplankton as food for mesozooplankton is affected by cell (or colony) size and morphology as well as the biochemical properties of the cell, e.g. the amino acid, vitamin, sugar, fatty acid, mineral, and toxin content (Koski *et al.* 1998). Differences in the presence and concentrations of these components are partly phytoplankton species (even strain; Md Amin *et al.* 2011) specific and partly related to the physiological state of the cell, thus varying with phytoplankton growth rate and cell age (Koski *et al.* 1998). Also the physical and chemical environment, i.e. light and nutrient availability, affects phytoplankton cell physiology (Koski *et al.* 1998). While a mixed phytoplankton diet is considered superior to a uniform diet (Kleppel 1993, Bonnet and Carlotti 2001, Li *et al.* 2008), different copepod species react differently to the same phytoplankton food (Engström *et al.* 2000, Md Amin *et al.* 2011). Consequently the determining of the nutritional value of a phytoplankton community for the mesozooplankton consumers at any given time is challenging. *However, the prevailing view is that in general diatoms and cyanophytes are "bad" food, whereas dinoflagellates are "good".*

The diatom/dinoflagellate ratio food web indicators

Diatoms and dinoflagellates as food for mesozooplankton

In a global perspective, diatoms were traditionally considered the phytoplankton group that provided most of the food sustaining the marine food web (Paffenhöfer *et al.* 2005), whereas dinoflagellates were not deemed important in copepod diets (Kleppel 1993). However, in the 1990s this view was challenged when it was repeatedly shown that copepod reproduction was disturbed or even inhibited by diatoms (Ban *et al.* 1997, Ianora *et al.* 2003, Ianora and Miralto 2010).

Both experimental and field studies have shown that high-density populations of diatoms often cause reproductive failure in copepods (reviewed by Ianora and Miralto 2010). In their seminal study, Ban *et al.* (1997) demonstrated that inhibitory effects of diatoms on copepod reproduction are a widespread phenomenon, potentially influencing copepod recruitment patterns and marine food web energy flows (Ban *et al.* 1997). Subsequently, extensive research has been conducted to investigate the effects of diatoms on copepod reproduction (e.g. Dutz *et al.* 2008). Despite this, the ecological significance and operative mechanisms of the detrimental "diatom effect" (*sensu* Halsband-Lenk *et al.* 2005) remain largely unresolved; the drawing of conclusions in part confused by the fact that diatoms are demonstrably *not* always and consistently inhibitory (Koski *et al.* 2008, Jónasdóttir *et al.* 2011).

Explanations for the diatom effect have been sought in diatom toxicity, diatom nutritional deficiency, and inefficient assimilation of essential compounds from diatoms in the zooplankton digestive tract (Ianora and Miralto 2010). As a group, diatoms contain less carbon (Menden-Deuer and Lessard 2000), lipids, carbohydrates and proteins (Hitchcock 1982) per cell volume than dinoflagellates; diatoms may also lack some bioactive or essential compounds (Dutz *et al.* 2008). Furthermore, some diatoms contain inhibitory compounds such as short-chained polyunsaturated aldehydes (PUAs), fatty acid hydroperoxides (FAHs) and non-volatile oxylipins (Barreiro *et al.* 2011).

Hitherto unknown diatom toxins may also be involved and it cannot at this point be excluded that multiple factors are at work simultaneously (Dutz *et al.* 2008).

The chemical defence displayed by diatoms is antiproliferative, i.e. it primarily affects the progeny (rather than the exposed individual; Ianora and Miralto 2010). This effect may serve to relieve grazing pressure by sabotaging subsequent generations of grazers, thus allowing diatom populations to persist in conditions where grazing pressure would otherwise have caused the population to decline (Ianora and Miralto 2010). Inhibition of copepod egg hatching success is diatom density-dependent (Chaudron *et al.* 1996, Starr *et al.* 1999, Kang and Poulet 2000); with increasing diatom concentrations, deleterious effects increase and are induced faster (Starr *et al.* 1999). Mixed diets dilute, but do not delete, the inhibitory effects of diatoms on copepod recruitment (Kang and Poulet 2000, Turner *et al.* 2001).

Few field studies on the effects of diatoms on mesozooplankton exist from the northern Baltic Sea. However, in a study from the Gulf of Bothnia, during the spring bloom when diatoms accounted for approximately 50% of total biomass (irrespective of chlorophyll *a* concentration or total phytoplankton biomass), hatching frequency of the copepod *Eurytemora affinis* was significantly lower than after the spring bloom, when ciliates dominated (Ask *et al.* 2006). There was a significant negative correlation between hatching frequency and total diatom biomass, as well as hatching frequency and the biomass of diatoms belonging to the genus *Chaetoceros* (Ask *et al.* 2006).

Since the investigations on mesozooplankton diets started in earnest, dinoflagellates have been found to be good quality and preferred food of copepods (Kleppel *et al.* 1991, references in Koski *et al.* 1998). In line with this were the results of Vehmaa and co-workers (2011), who performed experiments on a spring bloom diatom and dinoflagellate from the northern Baltic Sea. They found that egg production of the copepod *Acartia bifilosa* was greater on a diet of the dinoflagellate *Scrippsiella hangoei* alone as well as on a mixed (50:50) diet on *S. hangoei* and the diatom *Skeletonema marinoi*, than on *S. marinoi* alone, or on the cryptophyte *Rhodomonas salina* (Vehmaa *et al.* 2011).

Not all results are as straightforward, however. Depending on which reproduction parameter is measured (e.g. mesozooplankton egg production, egg hatching success, RNA:DNA ratio), a dinoflagellate-dominated community is not always consistently better food than a community dominated by diatoms (Vehmaa *et al.* 2012). An ultimate example of dinoflagellates as poor food is the toxic dinoflagellate *Alexandrium ostenfeldii*, which has proven ill, even mortal, effects on the copepods *Acartia bifilosa* and *Eurytemora affinis* (Sopanen *et al.* 2011). However, *A. ostenfeldii* occurs very sparsely and infrequently the during the spring bloom period (Hällfors *et al.* 2013b) becoming abundant only later in summer (Hakanen *et al.* 2012), as do most toxin producing (cf. Hällfors 2007) dinoflagellates in the northern Baltic Sea (Hällfors *et al.* 20013b).

Indicator #1: Diatom/dinoflagellate ratio in reference to mesozooplankton

While acknowledging that there are conflicting results regarding the quality of diatoms and dinoflagellates as food, we suggest that in the northern Baltic Sea the species composition of the spring bloom, i.e. diatom or dinoflagellate dominance is relevant for mesozooplankton for a number of reasons:

- The spring bloom period is the period of the highest planktic primary production on an annual scale in the northern Baltic Sea (Hällfors *et al.* 1981, Wasmund and Siegel 2008).
- Many diatom species are poor food due to their (to date unspecified) inhibitive effects (Ianora *et al.* 2003, Ask *et al.* 2006, Ianora and Miralto 2010).
- At the demise of the spring bloom diatoms sink out of the water column, while dinoflagellates do not (Heiskanen 1993, Heiskanen and Kononen 1994, Tamelander and Heiskanen 2004) and remain available as potential food.
- The organic matter from the decaying spring bloom dinoflagellates feeds the microbial loop, which ultimately feeds the omnivorous and carnivorous members of the mesozooplankton community. Members of the microbial loop may be even more suitable food for the spring and early summer mesozooplankton community than the spring bloom dinoflagellates themselves (cf. Noji *et al.* 1986).

The above points serve as rationale for **our first diatom/dinoflagellate ratio food web indicator development hypothesis**, i.e.; *in the northern Baltic Sea, in terms of providing more food of a higher quality for mesozooplankton, a dinoflagellate-dominated spring bloom community is more favourable, than a diatom-dominated spring bloom of equal biomass*.

Diatoms and dinoflagellates as food for the zoobenthos

While this chapter mainly focuses on the *pelagic* food web, phytoplankton also serves as food source for *benthic* fauna. In the northern Baltic Sea, sedimentation of organic matter following the spring bloom provides the main annual input of food to the benthic ecosystem (Kuparinen *et al.* 1984, Tallberg and Heiskanen 1998), and the species composition of the spring bloom would seem to have obvious consequences for the benthic community. As described above, if the spring bloom is dominated by diatoms, a larger fraction of the organic matter produced during the spring bloom sediments to the bottom for the benthic community to consume, than if the bloom is formed mainly by dinoflagellates (cf. Heiskanen 1993, Heiskanen and Kononen 1994, Tamelander and Heiskanen 2004). The input of high-quality food in the form of spring bloom diatoms is crucial for meiozoobenthos growth (Nascimento *et al.* 2009). The late summer bloom of cyanophytes seems to be of less importance for the benthic community, as most of the cyanophytes decompose already in the water column (Kuparinen *et al.* 1984, Heiskanen and Kononen 1994, Tallberg and Heiskanen 1998). Furthermore, it has been shown that settled blooms of the filamentous cyanophyte *Nodularia spumigena*, although utilized by the zoobenthos (Nascimento *et al.* 2008) are nutritionally inferior food (Karlson *et al.* 2008, Nascimento *et al.* 2009).

Indicator #2: Diatom/dinoflagellate ratio in reference to zoobenthos

The above serves as rationale for **our second (and tentative) diatom/dinoflagellate ratio food web indicator development hypothesis**, i.e.; *in the northern Baltic Sea, in terms of providing more food* *for the zoobenthos, a diatom-dominated spring bloom community is more favourable, than a dinoflagellate-dominated spring bloom of equal biomass.*

The cyanophyte/total phytoplankton biomass ratio food web indicator

Cyanophytes as food for mesozooplankton

Cyanophytes have usually been considered poor or even harmful food for mesozooplankton (de Bernardi and Giussani 1990, Sellner *et al.* 1994, Martin-Creuzburg *et al.* 2008). Since the peak of the annual mesozooplankton cycle in the northern Baltic Sea coincides with the period of cyanophyte blooms, the effect of cyanophytes on mesozooplankton reproduction, growth and survival has repeatedly been investigated in the area (e.g. Koski 1999, Engström-Öst 2002, Karjalainen 2005).

The negative effects of cyanophytes may be caused by poor manageability, meaning that cyanophytes mechanically interfere with the food gathering process of certain mesozooplankton species (Infante and Abella 1985, Haney 1987). Additional negative effects may be caused by cyanophyte toxicity (DeMott *et al.* 1991, Koski *et al.* 1999, Kozlowsky-Suzuki *et al.* 2003). Furthermore, the nutritional value of some cyanophytes is considered inferior due to the absence or low concentrations of essential compounds such as sterols (Martin-Creuzburg *et al.* 2008) and long-chain polyunsaturated fatty acids (PUFAs; see references in Koski *et al.* 1999 and Schmidt and Jónasdóttir 1997). Finally, some cyanophytes may simply constitute poorly digestible food (Lampert 1987).

Different mesozooplankton species react differently to cyanophytes as food (de Bernardi and Giussani 1990). In the Baltic Sea, mesozooplankton consume (or do not consume) cyanophytes at various rates (Sellner *et al.* 1994, 1996, Koski *et al.* 2002, Engström *et al.* 2000, Kozlowsky-Suzuki *et al.* 2003, 2007, Karjalainen *et al.* 2007). For example, Sellner and co-workers (1994) found that the cladoceran *Bosmina longispina maritima* ingested markedly more *Nodularia spumigena* than the copepods *Acartia bifilosa* and *Eurytemora affinis*, which ingested negligible amounts of the cyanophyte. In experimental studies, some copepod species ingested filamentous cyanophytes such as *N. spumigena* (Koski *et al.* 2002, Kozlowsky-Suzuki *et al.* 2003), particularly if offered as sole food (Engström *et al.* 2000, Lehtiniemi *et al.* 2002). However, copepods are able to, and have been found to, avoid ingesting non-toxic and/or toxic strains of filamentous cyanophytes (Koski *et al.* 1999, Engström *et al.* 2000).

The degree to which mechanical interference inhibits mesozooplankton is not entirely clear (cf. Lampert 1987, de Bernardi and Giussani 1990), but seems to vary with feeding strategy (Lampert 1987). Lampert (1987) suggests that copepods, rotifers and certain cladocerans such as some *Bosmina* species are less affected by mechanical interference than most cladocerans. Copepods are selective feeders, they are able to choose to either ingest or reject food particles (e.g. Keppel 1993), and *Bosmina* performs both passive filtering and active capturing of food particles (DeMott and Kerfoot 1982 in Lampert 1987), whereas most cladocerans are filter feeders. However, Lehtiniemi *et al.* (2002) suggest that difficult manageability of cyanophyte filaments contributed to the low feeding by the copepod *Eurytemora affinis* on *Aphanizomenon flos-aquae* and *Nodularia spumigena*.

Whereas the chemical defence of diatoms mainly affects the next generation of mesozooplankton (Ianora and Miralto 2010), toxic cyanophytes may cause both lowered reproductive success (Koski *et al.* 1999, Kozlowsky-Suzuki *et al.* 2003) and adult mortality (Koski *et al.* 1999). Hatching success has been found to decrease even when copepods avoid feeding on the cyanophytes (Koski *et al.*1999), indicating that the mere presence of cyanophytes may be harmful to mesozooplankton. Indeed, it has been shown that copepods may accumulate dissolved toxins directly from the water, and not only through ingestion (Karjalainen *et al.* 2003, Sopanen *et al.* 2009). However, it is not always clear whether the negative effects are caused by the toxicity or the poor nutritional value of cyanophytes (cf. Sellner *et al.* 1994, 1996). Koski *et al.* (1999) found that the copepod *Eurytemora affinis* failed to reproduce on a diet of *Nodularia* sp. irrespective of its toxicity, indicating that the filamentous cyanophyte was lacking in terms of nutritional quality.

The suitability of cyanophytes as mesozooplankton food has repeatedly been investigated with contrasting and inconclusive results. Cyanophytes are not always just bad food for mesozooplankton; indeed, their biochemical composition may serve to supplement the copepod diet (Schmidt and Jónasdóttir 1997). Moreover, cyanophyte aggregates host a variety of associated organisms, which are suitable food for mesozooplankton (Karjalainen *et al.* 2007); in addition to which organic matter from the decaying late summer cyanophyte blooms fuels the microbial loop, which ultimately feeds the omnivorous and carnivorous members of the mesozooplankton community (cf. Engström-Öst *et al.* 2002). Like the cyanophyte-aggregate associated organisms, members of the microbial loop are likely more suitable food for the mesozooplankton community than the cyanophytes themselves. Finally, copepods have been found to feed, survive and produce eggs in a plankton community dominated by toxic cyanophytes (Koski *et al.* 2002).

Indicator #3: Cyanophyte/total phytoplankton biomass ratio in reference to mesozooplankton

While acknowledging that there are inconsistent results regarding the properties of cyanophytes as food, we suggest that in the northern Baltic Sea the species composition of the late summer period, i.e. whether or not cyanophytes dominate the phytoplankton community, is relevant for mesozooplankton for a number of reasons:

- The late summer cyanophyte bloom period is the period of the second highest planktic primary production on an annual scale in the northern Baltic Sea (Hällfors *et al.* 1981, Wasmund and Siegel 2008).
- Several factors contribute to making filamentous cyanophytes unsuitable food: their toxicity, which causes rejection of the food (Koski *et al.* 1999) and thus possibly delays the finding of appropriate food; their morphology, which may cause interference with feeding (Infante and Abella 1985, Haney 1987, Lehtiniemi *et al.* 2002); and their low nutritional value (Koski *et al.* 1999).
- At least one of the dominant species, *Nodularia spumigena*, inhibits reproduction and causes mortality due to its low nutritional value and toxicity (e.g. Koski *et al.* 1999).

The above points serve as rationale for **our cyanophyte/total phytoplankton biomass ratio food web indicator development hypothesis**, i.e.; *in the northern Baltic Sea, in terms of providing more food of a higher quality for mesozooplankton, a late summer phytoplankton community dominated by phytoplankton groups other than cyanophytes is more favourable, than a cyanophyte-dominated community of equal biomass*.

8.4 Indicator recommendation

While acknowledging challenges brought on by unavoidable generalizations regarding the properties of phytoplankton taxa as food, as well as the discrepancies between the timing of the annual phytoplankton and mesozooplankton biomass maxima, we conclude that the theoretical background, which bases on solid scientific research, justifies the further development of the *Diatom/dinoflagellate ratio* and *Cyanophyte/total phytoplankton biomass ratio* food web indicators. *We recommend the further development, testing and evaluation of the following three phytoplankton food web indicators, which aim to describe the quality and quantity of food available for the consumers of phytoplankton*:

- *Indicator #1: Diatom/dinoflagellate ratio in reference to mesozooplankton*
- *Indicator #2: Diatom/dinoflagellate ratio in reference to zoobenthos*
- *Indicator #3: Cyanophyte/total phytoplankton biomass ratio in reference to mesozooplankton*

Next steps: important considerations, indicator testing and setting targets for indicators

The ratio of certain phytoplankton groups (as compared to each other or the whole phytoplankton community) aims to describe the phytoplankton community composition in terms of food quality. However, a ratio does not directly reveal the actual availability of food, as the availability depends largely on absolute amounts, not relative amounts, of high-quality phytoplankton. The *Zooplankton mean size and total abundance* indicator (reviewed in Chapter 7 of the present report) aims to combine the two facets, i.e. food quality and food quantity in one indicator. Applying the same approach to the suggested three phytoplankton food web indicators, our indicator hypotheses (see above) result in the assumptions schematically illustrated in Figure 8.1: a high phytoplankton biomass in combination with a high ratio of nutritionally beneficial phytoplankton would indicate favourable feeding conditions for the consumers of phytoplankton, whereas the other combinations infer intermediate or poor feeding conditions.

We anticipate that extensive data exploration and testing will be required in order to determine the applicability of the suggested indicators. Among important considerations is whether phytoplankton, mesozooplankton and zoobenthos should be investigated in terms of biomass or carbon (or something else). Another major consideration is the delimitation of the datasets. For example, in indicators #1 *Diatom/dinoflagellate ratio in reference to mesozooplankton* and #3 *Cyanophyte/total phytoplankton biomass ratio in reference to mesozooplankton* it may be necessary to separate herbivorous mesozooplankton from omnivorous and carnivorous (cf. Richardson and Schoeman 2004). On the other hand, while not all carbon flows directly from primary producers to mesozooplankton, the intermediaries, i.e. microprotozoans, are important food for mesozooplankton (Calbet and Saiz 2005, Sherr and Sherr 2007, Calbet 2008), wherefore exclusion of other than strictly herbivorous species is perhaps not requisite. Regarding indicators #1 *Diatom/dinoflagellate ratio in reference to mesozooplankton* and #2 *Diatom/dinoflagellate ratio in reference to zoobenthos*, it must be tested whether spring bloom biomass should be represented by the total phytoplankton biomass, or by the combined diatom and dinoflagellate biomass (excluding all other taxa). Regarding indicator #3 *Cyanophyte/total phytoplankton biomass ratio in reference to mesozooplankton*, it will be necessary to test different combinations of taxa to uncover the best descriptor of food quality, whether it is e.g. the ratio of the common bloom-forming filamentous cyanophytes *Aphanizomenon flos-aquae, Nodularia spumigena*, *Anabaena/Dolichospermum* spp. to the rest of the phytoplankton community, or the ratio of all cyanophytes (the whole group) to the remaining phytoplankton community.

The requisite next steps are therefore testing the indicators using available long-term data and, providing the indicators prove viable, setting the Good Environmental Status target levels based on the results of the data analyses.

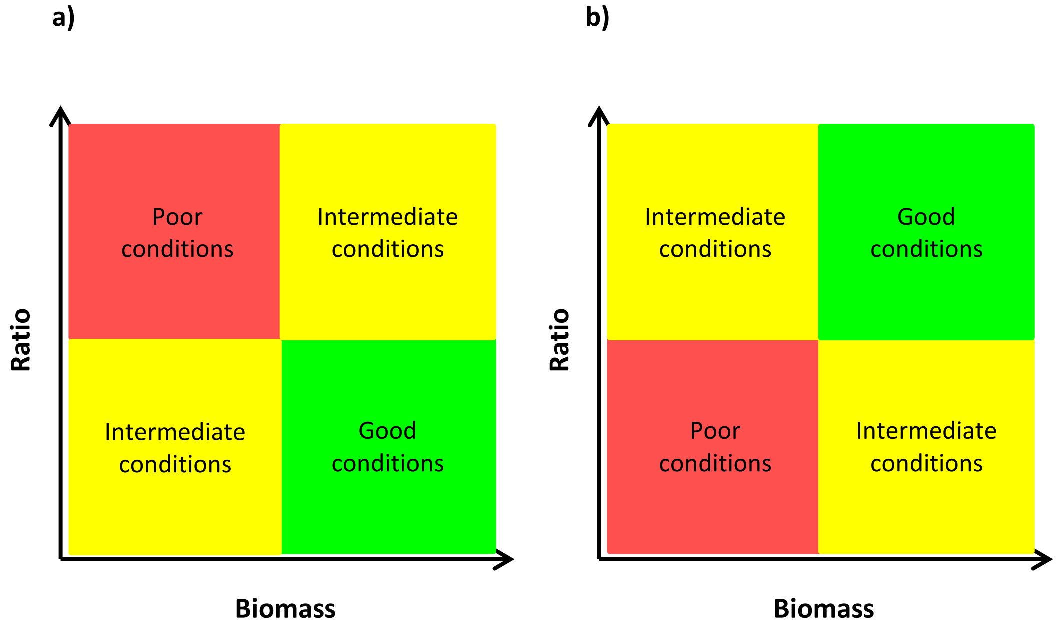

Figure 8.1. *Schematic illustrations of the phytoplankton early warning food web indicator approach.* ***a)*** *Good nutritional conditions for mesozooplankton are indicated by indicator #1 when a low ratio of diatoms in a high spring bloom biomass occurs, and by indicator #3 when a low ratio of cyanophytes together with high total phytoplankton biomass prevail.* ***b)*** *Good nutritional conditions for zoobenthos are indicated by indicator #2 when a high ratio of diatoms together with a high spring bloom biomass occurs.*

8.5 Suitability of current monitoring strategies

The HELCOM Programme for monitoring of eutrophication and its effects in the Baltic Sea (HELCOM 2006) includes the monitoring of phytoplankton, mesozooplankton and zoobenthos species composition and biomass. The suitability of current monitoring strategies for the purposes of the three suggested phytoplankton food web indicators is difficult to evaluate prior to indicator testing. Provisionally it can be said, that the rigorous testing and application of the suggested indicators in all likelihood requires intensive phytoplankton data from the spring bloom period and the late summer cyanophyte bloom, which in the northern Baltic Sea occur in April–May and July–August, respectively. A frequent (twice-a-week or weekly) sampling of a lower number of stations is recommended over infrequent sampling (twice-a-month or monthly) of a higher number of stations (see discussion on spatial distribution of samples in Hällfors 2013). The areas covered by the indicators should be delimited in such a way that no strong salinity and climatological gradients occur, i.e. the indicators should be tested separately for different sea areas in the Baltic Sea. In summary, the assessment of the sufficiency and temporal and spatial compatibility of the extant phytoplankton, mesozooplankton and zoobenthos monitoring and data is pending subject to data exploration and testing.

Acknowledgements to Chapter 8

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Annex 1.

The catalogue of food web indicators of HELCOM (2013) and those Baltic Sea EU Member States whose indicators were available. Provided as a separate spread sheet file ***Annex 1 food web indicator table.xlsx***

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