# The temporal and spatial dynamics of the sublittoral fish community of Kongsfjorden, Spitsbergen


by

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## Abbreviations

AC – Alternating Current

ADCP – Acoustic Doppler Current Profiler

ALK – Age Length Key

ANOVA – Analysis of Variance

ASCII – American Standard Code for Information Interchange

AWI – Alfred Wegener Institute, Helmholtz Centre for Polar and Marine Research

AWIPEV – French - German Arctic Research Base at Ny-Ålesund

COSYNA – Coastal Observing System for Northern and Arctic Seas

CPUE – Catch Per Unit Effort

CSV – Comma-separated Values

CMEMS – Copernicus Marine Environment Monitoring Service

CTD – Conductivity Temperature Depth

DC – Direct Current

EMODnet – European Marine Observation and Data Network

FAIR – Findable Accessible Interoperable Reusable

FTP – File Transfer Protocol

IoT – Internet of Things

IT – Information Technology

JSON – JavaScript Object Notation

LIDAR – Light Detection and Ranging

LTER – Long-Term Ecosystem Research

MDS – Multidimensional scaling

MOSES – Modular Observation Solutions for Earth Systems

NCC – Norwegian Coastal Cod

NEAC – Northeast Arctic Cod

NFDI – German National Research Data Infrastructure

PLC – Programmable Logic Controller

POM – Particulate Organic Matter

RemOS – Remote Optical System

ROI – Regions of Interest

ROV – Remotely Operated Underwater Vehicle

SCUBA – Self-Contained Underwater Breathing Apparatus

SD – Standard Deviation

SL – Standard Length

SME – Small and Medium-sized Wnterprises

SNP – Single-Nucleotide Polymorphism

SSW – Sub Surface Waters

TCP/IP – Transmission Control Protocol/Internet Protocol

TIFF – Tagged Image File Format

UPS – Uninterruptible Power Supply

UV – Ultraviolet

UWO – Underwater Observatory

VLAN – Virtual Local Area Networks

VPR – Video Plankton Recorder

WHOI – Woods Hole Oceanographic Institution

WSC – West Spitsbergen Current

XML – Extensible Markup Language

YOY – Young of Year

# Abstract

The Arctic is one of the areas that is most affected by global climate change (IPCC 2014). As a result of anthropogenic Arctic warming, the Arctic fish community might change, and species from temperate areas are expected to invade (Cheung et al. 2009). In this context, it is critical that "Arctic marine fishes are indispensable to ecosystem structuring and functioning, but they are still beyond credible assessment due to lack of basic biological data" (Christiansen et al. 2014). Especially in the shallow-water zones (3 - 12 m) of Arctic fjord systems only qualitative data on fish community composition are available. In other ecosystems, it was shown that the structured shallow-water zone has special ecological functions (Seitz 2014, Pondella et al. 2015). Therefore, the objective of this study is to increase our scientific knowledge on the fish assemblage of this special area by performing a quantitative first-time assessment of its species composition and abundance as well as the size and age structure of selected species.

As study site, Kongsfjorden (79°N, 12°E) at the west coast of the Svalbard archipelago was chosen. It is one of the best investigated fjords in the Arctic, and the local AWIPEV research base provides one of few sites where this project could be logistically supported. Despite the local infrastructure, access to the field was restricted due to the challenging climatic regime of Kongsfjorden. A thorough risk assessment resulted in the finding that no fishing from small boats can be performed safely during the polar night. Fish assessments were therefore conducted with two complementary methodologies. The first method was seasonal fyke net fishing in June/July and September of the years 2012 to 2014. The second method was a year-round assessment via a stereo-optic imaging system, which was connected to an underwater observatory.

By fyke net fishing 2804 specimens were sampled, representing 12 species and an unknown number of unidentified species of the family *Liparidae*. The most abundant species was the shorthorn sculpin (*Myoxocephalus scorpius*, 74.9 %) followed by Atlantic cod (*Gadus morhua*, 17.2 %) and Arctic staghorn sculpin (*Gymnocanthus tricuspis*, 3.8 %).

We performed age class determination by analyzing the structure of sagittal otoliths of Atlantic cod. The age classes were set with annual increments, starting with age class 0+ for specimens that did not complete their first year of development. In our result, the age classes 0+, 1+, and 2+ are dominating the shallow-water zone. As age class 0+ was not found in June but exclusively in the September sampling campaigns, it is concluded that 0+ specimens immigrate in the shallow-water zone from about August to September. The detected age-length relationships of the sampled specimen were comparable to literature data from the Barents Sea (Brand et al. in draft; Brander 2005).

The stereo-optical assessment of macrofauna by the AWIPEV underwater observatory allowed for the year-round tracking of the macrofauna community regarding their species composition, specific abundance, and length-frequency occurrences. The necessary technologies and operation standards were developed in the framework of the project COSYNA (Coastal Observing System for Northern and Arctic Seas; Baschek et al. 2017). The AWIPEV underwater observatory hosts different sensors for the year-round assessment of hydrographic parameters and macrofauna. The macrofauna assessment was performed by the stereo-optic instrument RemOs1 (Wehkamp & Fischer 2014). Hydrographic parameters were assessed by a Workhorse Sentinel ADCP (Teledyne Marine), a SBE 38 temperature probe (Seabird Electronics Inc.), and a CTD90 Multiparameter probe (Sea & Sun Technology GmbH). The AWIPEV underwater observatory comprises additionally a land-based sensor system, named FerryBox (-4H-JENA engineering GmbH). This flow-through system enables redundant data assessment with sensors that can be serviced without diving campaigns (Fischer et al. 2017 & 2020a). The results of the year-round macrofauna assessment show that the total macrofauna abundance was significantly lower during the polar day than the polar night. The maximum total abundance was observed in February. Furthermore, the growth of a cohort of Gadidae (cod) was tracked from October 2013 to March 2014. Hereby, a growth from 6-12 cm to 9-16 cm standard length (SL) can be shown. Reliable discrimination of cod to species level was not possible by the stereo-optic data assessment.

In Brand & Fischer (2016) data from fyke net catches in the same area and water depth is presented. It shows that 17.23 % of the total catches (n total = 420) were Atlantic cod, while only 0.36 % of the specimen (n total = 10) were identified as polar cod. This suggests that most cod specimens assessed in the shallow-water zone by the stereo-optical instrument were also Atlantic cod. Still, a general absence of polar cod in Kongsfjorden should not be concluded. A report from Fey & Węslawski (2017) shows the presence of polar cod in Kongsfjorden in late September 2013. By bottom-trawl in a water depth of 52 and 134 m, a subsample of 813 specimens of polar cod was taken in this study. The determined age classes of those specimens were 0 - 4+. This indicates vertical species segregation between the shallow-water zone (max. 12 m) and deeper areas of the fjord into separate habitats. The strong presence of Atlantic cod in the shallow-water zone in Kongsfjorden has not been reported before and might be an indicator for the ongoing northward shift of temperate species into the Arctic (Renaud et al. 2012).

The combination of fishing and remote-controlled sampling significantly enhanced the knowledge output of this study. The fishing allowed for precise species identification and tissue sampling for further analysis in compact campaigns. The *in situ* optical instrument allowed for a year-round sampling with a large sampling size. Additionally, the year-round optical assessment had no permanent impact on the local fish community, as no specimens were removed from the field. Thereby, this study demonstrates the potential of remote-controlled sensor operation as a highly valuable technology to track the higher trophic levels of the Arctic ecosystem.

# 1. Introduction

By the end of the 21st century, actual projections predict an increase of global mean surface temperature of 0.3 to 4.8 °C, dependent on future $CO_2$ emissions. In this process, the air temperature in the Arctic will warm more rapidly than the global mean (IPCC 2014). The rapid changes in the Arctic climate regimes within the last decades are caused by anthropogenic global warming and have the potential to shift the spatial distribution of current Arctic species. Additionally, it might lead to an invasion of new species from temperate areas (Cheung et al. 2009). Especially, boreal species, that are adapted to withstand a dynamic hydrographic regime, are likely candidates for a northward shift of their spatial distribution. This expected northward shift of boreal species to the Arctic got known under the term borealization (Fossheim et al. 2015). A boreal generalist in this context is for example the Atlantic Cod (*Gadus morhua*). This generally benthopelagic fish is actually distributed from the Gulf of Biscay up to the Barents Sea (Wienerroither et al. 2011).

The southern Barents Sea is known as an Arctic-Atlantic transition zone, influenced by warm and saline Atlantic water masses, as well as cold Arctic water masses of lower salinity. It is not only impacted by climate change, but also by fishery and other human-induced changes (Johannesen et al. 2012). For the Barents Sea, pelagic species such as capelin (*Mallotus villosus)*, herring (*Clupea harengus*), polar cod (*Boreogadus saida)* as well as demersal species as Atlantic cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus)* are known as ecologically important species. Atlantic cod and herring are boreal species and are known to shift northwards during extended warm periods. In those periods also increased primary and secondary production, as well as an increase in fish production, is reported (Loeng & Drinkwater 2007). For Atlantic cod, Ingvaldsen et al. (2017) report a recent occurrence of Atlantic cod in the deeper waters of the Fram strait, a distinct deviation from its generally benthopelagic lifecycle. It may be linked to avoidance of food competition.

Recent investigation in the physiology of Atlantic cod and polar cod project an increase in competitive strength of Atlantic cod and a decrease for polar cod in conditions as projected for the year 2100 (Kunz 2019). Today higher water temperatures are already linked to a higher survival rate in the early life stages of Atlantic cod (Ottersen et al. 2006). Polar cod, which is associated with cold, sub-zero Arctic water masses, is expected to lose the ice-associated part of its life-cycle, and become restricted to a pelagic distribution during summer. At the same time capelin, which is primarily distributed in Atlantic water masses, may expand north and eastwards in the Barents Sea, with large interannual fluctuations (Hop & Gjøsæter 2013).

Also, by historic reference, a quick northward shift of boreal species as capelin, herring, and Atlantic cod is expected (Rose 2005). The extent of those northward shifts might also be directly associated with the food web. Recent studies between Atlantic cod, haddock, and polar cod revealed less than 40 % dietary overlap (Renaud et al. 2012). However, zooplankton communities are changing with hydrography (Willis et al. 2007), and might therefore influence food availability for the different species individually. The analysis of these expected ecosystem changes and consequences requires thorough knowledge of the ecosystem components and the current and past biotic and abiotic conditions. Such baseline data are crucial for the interpretation and discrimination of natural system variability, temporal dynamics, and longer-lasting changes. This is especially important for the successful modeling of possible future scenarios. In some current studies, Arctic marine fishes are described as “beyond credible assessment due to lack of basic biological data” (Christiansen et al. 2014). This might not be entirely true for commercially important areas, like the Barents Sea, where extensive research is performed for fishery management (Eriksen et al. 2018). But remote areas, as fjord systems in the Arctic, are less in the focus of research. The hydrographic regime of a fjord depends on its location, adjacent water masses, and mixing processes. Fjords on Svalbard specifically show highly structured coastlines with macroalgal beds in hard bottom zones. Such zones are known to contain significantly higher abundances of fish in comparison to soft bottom zones (Pondella et al. 2015; Stephens et al. 2006). The kelp belts themselves are important ecosystems that form a biogenic habitat, which is colonized by a variety of plants and animals (Teagle et al. 2017). Recent studies show a positive relationship between the presence of kelp and fish abundance (Bertocci et al. 2015). A study of the kelp belt food webs in Kongsfjorden showed a remarkable species richness of macrozoobenthos (Paar et al. 2019a). They might thereby act as feeding ground for fish and might also offer structural protection. It can thereby be assumed that the shallow-water zones offer important ecological functions such as feeding grounds, nursery, and shelter from predation as reviewed by Teagle et al. (2017).

The present study was conducted to perform a quantitative assessment of a shallow-water fish community in an Arctic fjord system. The ecological data were gathered to improve the overall understanding of these systems. Kongsfjorden, with its research settlement Ny-Ålesund, was chosen as a study site because it offers a unique infrastructure for year-round research. Facilitated by the research settlement, Kongsfjorden is a site for numerous research efforts in atmospheric, terrestrial, and aquatic research (Hop et al. 2002). The fjord on the west coast of the Svalbard archipelago at 79° N is known to be influenced by Atlantic and Arctic water masses (Cottier et al. 2005). Those water masses originate from the West Spitsbergen Current (WSC) and Coastal Current (CC) in front and on the adjacent shelf. Both currents are showing cyclic changes in regard to temperature and salinity, and a general increase in temperature of 1.2 °C (WSC) and respectively 2 °C (CC) per 60 years (Vesman et al. 2017). Kongsfjorden can be characterized today as a sub-Arctic, glacial fjord located in the Arctic. This combination of factors makes it one of the sites where the effect of hydrographic changes in the Arctic might be observed first. The latest report shows a mean annual increase in water temperature of 0.14 °C/y (Hop et al. 2019).

In the conception of this study, it became clear that a year-round quantitative assessment of the shallow-water fish community by classic fishery methods would result in a challenging endeavor. A risk assessment that considered the special conditions of the polar night resulted in the finding that fishery from small boats requires daylight to establish an acceptable safety regime for this activity. This resulted in a potential time frame for safe fieldwork between April and September. For the remaining seven months of the year, extensive fieldwork is almost impossible due to the missing daylight, the low outside temperatures, drifting icebergs, and potential ice coverage of parts of the fjord system. To compensate for these typical temporal limitations in fieldwork in the polar environment, the use of remote observation technology got integrated into this study.

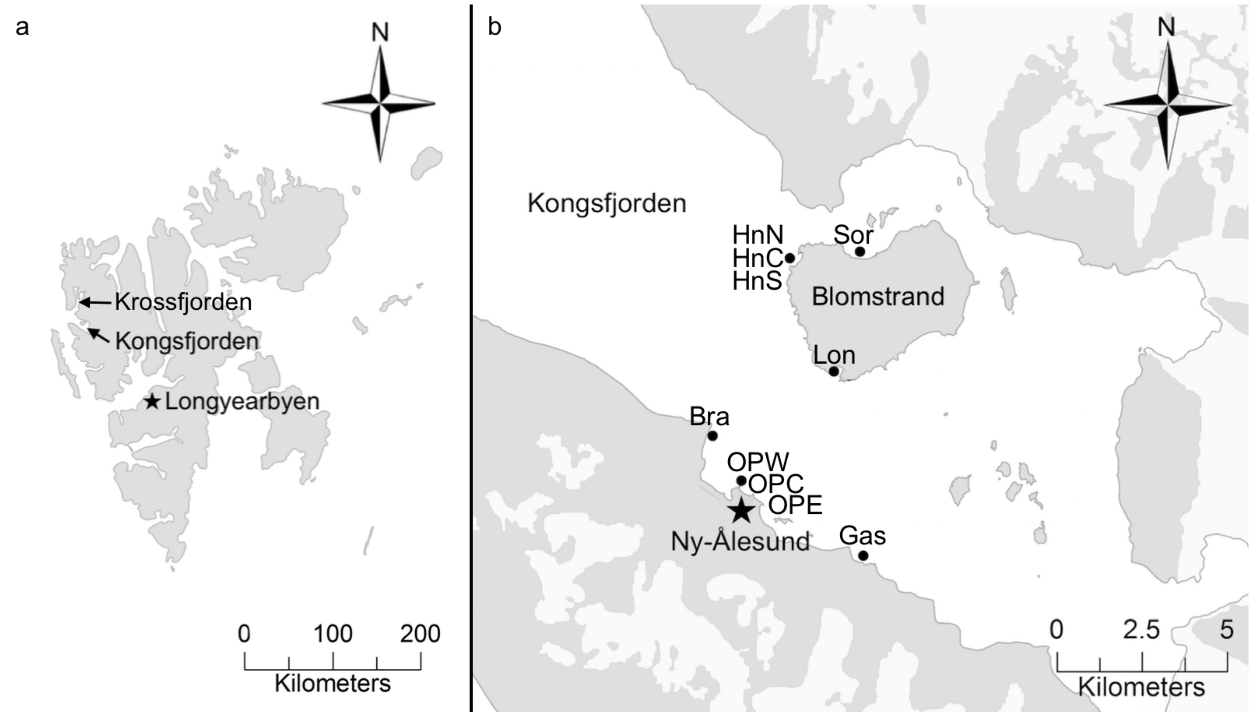


**Figure 1 - Map of Svalbard** a) The Svalbard archipelago with Krossfjorden and Kongsfjorden, as well as the primary settlement Longyearbyen b) Kongsfjorden and the town Ny-Ålesund and the island Blomstrandhalvøya. Glacier surfaces are marked light grey. Abbreviations of sampling sites are Sor - Sørvågen, HnN - Hansneset North, HnC - Hansneset Central, HnS - Hansneset South, Lon - London, Bra - Brandal, OPE - Old Pier East, OPC - Old Pier Central, OPW - Old Pier West, Gas - Gåsebu. The map data was provided by the Norwegian Polar Institute - from Brand & Fischer (2016).

Remote observation is taking place in Kongsfjorden for more than one decade, primarily in the form of different moorings (Hop et al. 2019). They provide valuable data about the hydrographic regime in Kongsfjorden in the polar day and night. Thereby, they allow for the large-scale assessment of open water current patterns and water mass imports in Kongsfjorden. For this study, the application of similar systems deemed not to be feasible as shallow-water installations are subject to be damaged or even destroyed due to drifting ice. Furthermore, our aim was to operate complex optical instruments for fish observation with high requirements in data storage and power supply. Therefore, we decided to install a cable-connected underwater observatory based on the COSYNA standard (Baschek et al. 2017). This observatory technology allowed us to fulfill our demands with respect to continuous energy supply, permanent full operational access to each individual sensor, and practically unlimited storage due to real-time streaming of the data (Fischer et al.

2017). The dimensions and weights of all components were designed to a size that they do not require large research vessels or workboats for deployment. Deployment, maintenance, and recovery of all system components can be performed by scientific divers from small workboats (Fig. 2). Since the deployment in 2012, the system is serviced bi-annual within dive campaigns integrated into the regular dive missions of the Alfred-Wegener-Institute (AWI) in Kongsfjorden. The required diving activity was carried out by the professional guidelines of the German statutory accident insurance (DGUV 2011).

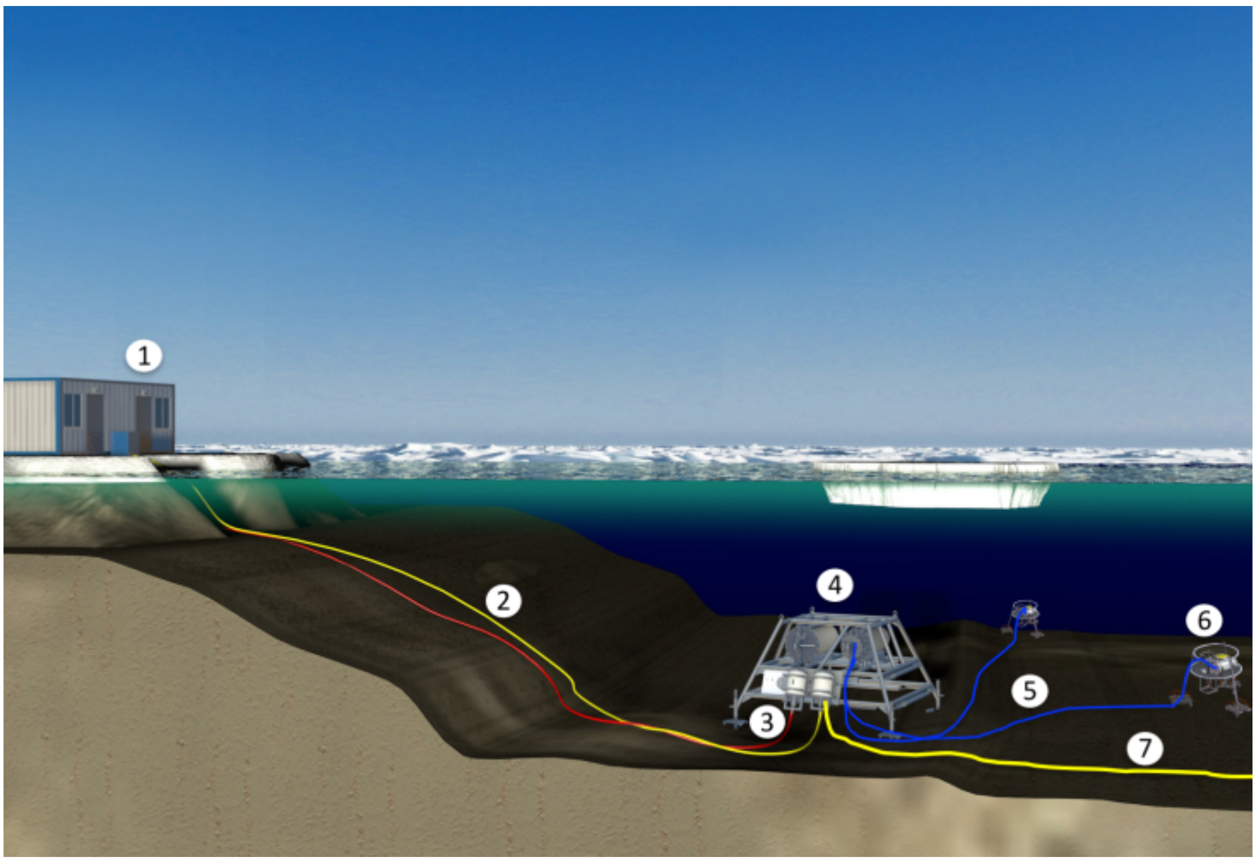


**Fig. 2 - Exemplary underwater node system**, (1) Landstation with power and data connection, (2) Subseacable, (3) Breakoutbox for termination of fiberoptic lines, (4) Underwater Node in Lander, (5) Connection cable to sensor package, (6) Lander with standard sensor package, (7) Second sea cable to next node system - from Fischer et al. (2017).

One of the aims of this study was to provide a first quantitative assessment of the shallow-water fish community in the inner parts of the Kongsfjorden ecosystem. Even though this fjord is one of the best-investigated fjord systems of the Arctic, there is not a single quantitative study available on the shallow-water fish community of this fjord. This study is therefore the first quantitative and year-round study of the shallow-water fish community in Kongsfjorden. The year-round remote observation of hydrography, fish abundance, fish species composition, and size-frequency distributions by the AWIPEV underwater observatory is combined with the results of fishing campaigns during the summer months. The fishing campaigns delivered ground-truthing data and enabled tissue samples of specimens, which were required for advanced analysis. One of those analyses was the determination of length-age relationships by otolith microstructure analysis (Brand et al. in draft).

Furthermore, the fishing campaigns were performed at a total of 10 sites in central Kongsfjorden. By integrating the site of the underwater observatory and additional sites at the southern and the northern shoreline we addressed questions regarding spatial variation in the fish community. By combining the data from fishing campaigns and permanent remote observations the temporal dynamics of the shallow-water community of Kongsfjorden are described.

# 2. Approach

This study combines aspects of biological research and technical development to cope with the listed subjects. The overall question of the study was to increase our knowledge about the fish community in the special habitat that Arctic shallow-water zones represent.

Subject 1:

The year-round operation of a remote-controlled underwater fish observatory in an Arctic fjord system required the development and implementation of sophisticated, sound, and reliable procedures and technologies. The objective for operating this system in "Kongsfjorden" was to gather abundance and morphometric data for the main fish species, as well as associated hydrographic data (Fischer et al. 2017, 2020a).

Subject 2:

To gather ground-truthing data on the local shallow-water fish community a multi-year fishing campaign at different littoral sites of Kongsfjorden was conducted. This assessment was designed to complement the remote data from the underwater observatory with length-weight and length-at-age relationships, as well as additional morphometric and morphological data. Furthermore, this part of the work delivered information and data concerning the spatial variability in the shallow-water fish community of Kongsfjorden (Brand & Fischer 2016).

Subject 3:

The data of the remote observation and fishing campaigns were analyzed in regard to temporal dynamics. The potentials and restrictions of both techniques were evaluated (Brand et al. in draft; Fischer et al. 2017).

The results regarding all subjects are summarized and discussed in section 4.

# 3. Publications

The present thesis includes two first author and two second author publications.

**<u>Overview</u>**

**Publication I**

Authors: Markus Brand, Philipp Fischer

Title: Species composition and abundance of the shallow-water fish community of Kongsfjorden, Svalbard

**Publication II**

Authors: Philipp Fischer, Max Schwanitz, Reiner Loth, Uwe Posner, Markus Brand, and Friedhelm Schröder

Title: First year of practical experiences of the new Arctic AWIPEV-COSYNA cabled Underwater Observatory in Kongsfjorden, Spitsbergen

**Publication III**

Authors: Philipp F. Fischer, Holger Brix, Burkhard Baschek, Alex C. Kraberg, Markus Brand, Boris Cisewski, Rolf Riethmüller, Gisbert Breitbach, Klas O. Möller, Jean-Pierre Gattuso, Samir Alliouane, Willem H. Van De Poll, Rob Witbaard

Title: Operating cabled underwater observatories in rough shelf-sea environments: a technological challenge

**Publication IV**

Authors: Markus Brand, Lisa Spotowitz, Felix Christopher Mark, Jørgen Berge, Jan Marcin Węsławski, and Philipp Friedrich Fischer

Title: Age class composition and growth of Atlantic cod (*Gadus morhua*) in the shallow-water zone of Kongsfjorden, Svalbard

**Cover page - Publication I**

Authors: Markus Brand, Philipp Fischer

Title: Species composition and abundance of the shallow-water fish community of Kongsfjorden, Svalbard

Status: Published 2016
Polar Biology (2016) 39: 2155
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Contributions: I performed the idea, design, and execution of the study with advice from PF. I wrote the manuscript with assistance in the revision by PF.

Polar Biol (2016) 39:2155–2167
DOI 10.1007/s00300-016-2022-y

ORIGINAL PAPER

# Species composition and abundance of the shallow water fish community of Kongsfjorden, Svalbard

**Markus Brand**[1] · **Philipp Fischer**[1]

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**Abstract** Projections show that climate change will lead to structural change in Arctic ecosystems. Studies project the extinction of local species and intense species invasion to the Arctic Ocean. A lack of basic biological data about the Arctic shallow water fish community will make it hard to assess whether these communities will change or not. Baseline studies in combination with future reassessments are needed to establish a basic knowledge about the change of these communities. This study provides a quantitative first time description of the shallow water fish community of Kongsfjorden, Svalbard. The fish assemblage in the depth range from 3 to 12 m was determined with respect to abundance and species composition. Among a total sample size of 2804 specimens, the presence of 12 fish species and one family (Liparidae) was detected. *Myoxocephalus scorpius* (shorthorn sculpin) (74.9 %), *Gadus morhua* (Atlantic cod) (17.2 %), and *Gymnocanthus tricuspis* (Arctic staghorn sculpin) (3.8 %) were identified as the most abundant species across all sampling sites. A significant relationship between algal coverage and fish abundance was detected. Furthermore, we demonstrated a fjord inward increase in biodiversity along the south shore that might be correlated with a change in hydrographic regime.

This article belongs to the special issue on the "Kongsfjorden ecosystem—new views after more than a decade of research", coordinated by Christian Wiencke and Haakon Hop.

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**Keywords** Demersal · Sublittoral · Coastal habitats · Algal belts · Species diversity

## Introduction

It is known that climate change can impact the marine biodiversity through changes in species distribution. Existing projections for marine fish and invertebrates show local species extinctions and intense species invasions into the Arctic Ocean (Cheung et al. 2009). Consequently, Arctic fishes might not just experience a change of hydrographic regime, but may also face new ecological interactions (Fossheim et al. 2015). Due to a lack of basic ecological data about Arctic fishes, the consequence for ecosystem structure and function is unclear (Christiansen et al. 2014). A possible scenario is the retreat of Arctic shelf fish species northwards towards the polar basin and a borealisation of the southern communities (Fossheim et al. 2015). However, a thorough quantitative assessment of Arctic fish communities, to which future investigations can be compared, has hitherto been lacking especially for Arctic shallow water habitats. This study aims to provide this description for the shallow water habitat of Kongsfjorden.

Kongsfjorden on the west coast of Spitsbergen (Fig. 1) has been the focus of ecological research in the Arctic for several decades. Geographically, Kongsfjorden at 79°N is classified as Arctic; however, it is in fact significantly influenced not only by Arctic but also by Atlantic water masses from the Fram Strait (Hop et al. 2002). The 20 km long fjord opens to a shelf system in a westerly direction without a sill and shares this outlet with the more northern Krossfjorden (Cottier et al. 2005). Additionally, an underwater canyon runs from the outlet across the shelf to the

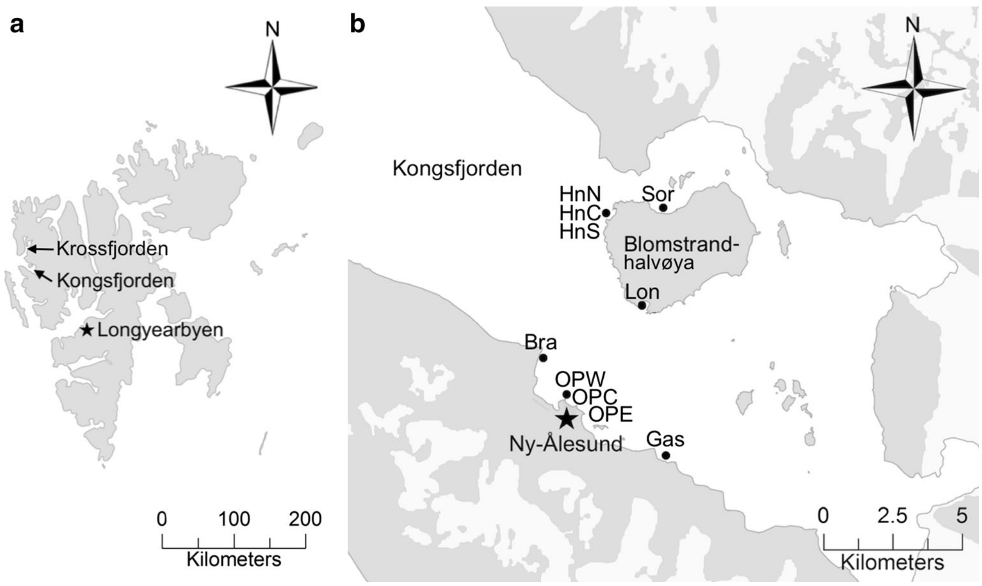



**Fig. 1** **a** Map of the Svalbard archipelago with its main settlement Longyearbyen and the site of study, Kongsfjorden. **b** Map of Kongsfjorden with its settlement Ny-Ålesund and the island Blomstrandhalvøya. Light areas on land indicate glacier surfaces. The sampling sites are marked with abbreviations: Sor—Sørvågen, HnN—Hansneset North, HnC—Hansneset Central, HnS—Hansneset South, Lon—London, Bra—Brandal, OPW—Old Pier West, OPC—Old Pier Central, OPE—Old Pier East, Gas—Gåsebu. The three sites at Hansneset and Old Pier were spaced approximately 100 m from each other. The map data were provided by the Norwegian Polar Institute

continental edge, establishing a connection to the deeper water masses off the shelf that belong to the West Spitsbergen Current. Complex mixing processes between the Arctic shelf water masses, the Atlantic deep water masses, and the highly seasonal fresh water run-off from the inner part of the fjord result in steep environmental gradients from the inner parts of the fjord up to its mouth (Svendsen et al. 2002). Willis et al. (2006) suggest a general counterclockwise circulation pattern in Kongsfjorden for the transport and the mixing of those water masses with an influx of water along the south shore and an efflux along the north shore. The resulting hydrographic gradients and their short- and long-term changes in intensity have the potential to directly influence the pelagic and benthic realms of the fjord. Thereby, the local food web is influenced on a spatial and temporal scale (Stempniewicz et al. 2007).

The complex environmental interactions within the fjord result in a mixed species composition that includes Atlantic and Arctic species in close association. The study of this dynamic habitat and its ecological processes is potentially valuable to enhance the knowledge about the potential influence of hydrographic regime shifts on other Arctic ecosystems. The general differences and characteristics of each ecosystem will have to be carefully reviewed for such comparison, but in the context of a predicted rise of 3.7 °C in mean air temperature in the Arctic over the next 50 years (ACIA 2004; Wong et al. 2014), Arctic marine ecosystems might face similar long-term changes in their hydrographic regimes, e.g., a rise of sea surface temperature and a general decrease of ice cover potentially directly affecting the marine habitat and its fauna. In the outer part of Kongsfjorden, a close relationship between short-term hydrographical regime changes and the zooplankton abundance has already been shown by Willis et al. (2006). In contrast, Renaud et al. (2011) used stable isotope ratios to demonstrate relative little variability in food web structure of Kongsfjorden over the temporal and spatial scale of their study. In turn, Voronkov et al. (2013) showed for hard bottom zoobenthos a general decrease in species richness, frequency of occurrence, mean abundance, and biomass towards the tidal glaciers in inner Kongsfjorden. Hop et al. (2002) highlight the need to better understand the temporal and spatial dynamics of the secondary and tertiary consumers including zooplankton, crustaceans, fishes, birds, and mammals, as well as their interaction with the local hydrographical regime.

Previous fish community assessments in Kongsfjorden have been performed by trawling in the deeper parts of the fjord or by diving observations in the shallow areas (Hop, pers. comm.). Recent studies describe a total of about 30 Arctic, sub-Arctic, and boreal fish species for the entire fjord system. Typical species with an Arctic and sub-Arctic distribution are *Boreogadus saida* (polar cod), *Eumicrotremus derjungini* (leatherfin lumpsucker), *G. tricuspis* (Arctic staghorn sculpin), *Lycodes reticulatus* (Arctic eelpout), *Mallotus villosus* (capelin), *Sebastes mentella* (deepwater redfish), and *Somniosus microcephalus* (Greenland shark). Furthermore, generalist species of the North Atlantic can be found. These are *G. morhua* (Atlantic cod), *M. scorpius* (shorthorn sculpin), *Anarhichas lupus* (Atlantic wolfish), *Leptoclinus maculatus* (daubed shanny), *Hippoglossoides platessoides* (long rough dab), *Leptagonus decagonus* (Atlantic poacher), and the family of Liparidae (snailfishes) (Hop et al. 2002; Renaud et al. 2011). The shallow water habitat of Kongsfjorden has been described by SCUBA surveys as rich in macroalgal belts (Hop et al. 2012). Bartsch et al. (2016) showed for 2012 and 2013 that the maximum density of these kelp belts was found at a depth of 2.5 m, with decreasing densities down to the maximum depth of the study at 15 m. Jørgensen and Gulliksen (2001) described the seabed in depths between 16 and 30 m as highly structured and complex benthic habitat with a mixture of rocky bottom, ice-rafted pebbles and stones. Voronkov et al. (2013) report that the structural complexity of the seafloor is increased by crustose coralline algae, shell hash of molluscs, and barnacles. The maximum abundance and diversity of hard bottom fauna are described by Voronkov et al. (2013) for the depth range of 5–10 m in 1996–1998. The structure rich seafloor has the potential to represent an important habitat for fishes. Especially, the macroalgal belts provide additional structural protection from waves as well as currents, and the infauna is a potential food source (Lippert et al. 2001). The shallow water habitats are commonly considered to be areas of aggregation for the resident fish community due to their function as spawning and nursery grounds (Werner 1977; Werner et al. 1983; Keast 1985; Pierce et al. 1994; Fischer and Eckmann 1997a, b). By feeding in the macroalgal belts, the fish community acts as a secondary producer in the local food web making them potentially important for energy and matter flux. Recent studies of Bartsch et al. (2016) and Paar et al. (2016) show significant changes in the structure of the macroalgal belt at Hansneset and also its macrozoobenthos community between 1996 and 1998 and 2012–2014. In comparison with previous studies, these changes might be caused by an increase of ice-free days over the last decade and a decrease in ice scouring, but the lack of quantitative baseline data for the shallow water fish community in Kongsfjorden makes it impossible to detect potential changes in relation to shifts in habitats and prey. Potential changes in abundance and species composition of the fish community will have direct effects on other compartments of the food web. To close this gap for future studies, the principal goal of this study is to provide the first quantitative overview of the fish species richness and abundance, as well as spatial distribution patterns of demersal fish in the shallow waters of Kongsfjorden and how it relates to habitat complexity. The data presented in this study were sampled in parallel to the field work published in Bartsch et al. (2016) and Paar et al. (2016) in 2012 and 2013 including the sampling site Hansneset and further sites in the central part of Kongsfjorden. While this study provides baseline data of the fish community, further collaborative publications will cover the role of the fish community in the food web and its seasonal dynamics.

## Materials and methods

The study area was the central part of Kongsfjorden, Svalbard (Fig. 1). In 2012, we sampled in week 24–28 and 36–39. In 2013, sampling was performed from week 23 to 30 and 35–40. In 2012, only one sampling site at the southern shoreline (Old Pier Central; Fig. 1b) and one at the shoreline of the island Blomstrandhalvøya (Hansneset South; Fig. 1b) were chosen, because it was unclear how feasible a more intense sampling program would be. The two sampling sites were chosen for coherence with linked projects. The site Hansneset was used for macroalgal research (Wiencke et al. 2004; Fredriksen et al. 2014; Bartsch et al. 2016; Paar et al. 2016), and at the site Old Pier Central, the Kongsfjorden Underwater Observatory was deployed. Additionally, we expected that the counterclockwise water mass circulation in Kongsfjorden (Willis et al. 2006) might influence the fish community on the opposite shorelines of Kongsfjorden.

With logistical support from the research management company KingsBay and the AWIPEV research station in 2012, we extended the sampling scheme in 2013 to six sampling areas, consisting of three areas at the northern shoreline around Blomstrandhalvøya (Sørvågen, Hansneset and London; Fig. 1b) and three areas at the southern shoreline (Brandal, Old Pier and Gåsebu; Fig. 1b). Starting with week 26 in 2013 one sampling area at each shoreline (Hansneset and Old Pier) was sampled at three subsites (Hansneset North, Hansneset Central and Hansneset South; Old Pier West, Old Pier Central and Old Pier East). The subsites were 50–75 m apart from each other. The triplicate sampling design at the respective middle sites of each shoreline was performed to provide an estimate of the variability within a sampling area. From the beginning of the study up to week 36 of 2012, sampling was performed at each of the two sites using one fyke net [diameter 40 cm, length 90 cm, mesh size 12 mm (bar mesh)], deployed in about 3 m water depth with its mouth set perpendicular to the shoreline and one trammel net (inner/outer mesh size 1/15 cm, length 20 m, height 2 m) deployed from about 5 to 12 m water depth. The distance to the shoreline depended on the slope of the seafloor, and ranged from 5 to 50 m.

During an equipment test in week 35 of 2012, we monitored an interaction of young seals with the trammel nets, indicating potential for entanglement. In response, we decided to replace all trammel nets by sets of three fyke nets per sampling site. One fyke net [diameter 40 cm, length 90 cm, mesh size 12 mm (bar mesh)], was deployed in about 3 m water depth and a second one of the same type in 12 m water depth. Additionally, a double fyke net [diameter 60 cm, length 110 cm, mesh size 12 mm (bar mesh)], connected by a 10 m long, 80 cm high steering net (18 mm bar mesh) was laid out between 5 and 8 m water depth. All nets were laid out in line with their mouths set perpendicular to the shoreline. We also started to place bait (fish muscle tissue) in the fyke nets at 3 and 12 m water depth to increase the fish catches in the solitaire fyke nets. The plan was not to compare between nets with and without bait, but only to do analysis between identical sets of nets at different sampling sites. The sampling design was maintained from this point on for all further samplings in 2012 and in 2013.

The net exposure time was normally 24 h. However, logistic and weather conditions occasionally delayed net recovery to exposure times of up to 96 h. To calculate the effects of a delayed net recovery on the mean catch efficiency, we back-calculated all catches to 24 h exposure time and compared the number of fishes caught per 24 h, per net (CPUE) of the catches with 24 h $\pm$ 3 h exposure time ($N = 566$), 72 $\pm$ 3 h exposure time ($N = 276$) and 96 $\pm$ 3 h exposure time ($N = 130$). Because we had no simultaneous catches for 24, 72 and 96 h at one site and one timepoint to make a real experimental comparison of the effects of exposure time on the CPUE, we included all catches in the analysis to avoid hidden structural site or time effects, which we could not predict. We used three Bonferroni-corrected unpaired two-group Wilcoxon and Mann–Whitney Tests (Wilcoxon 1945; Mann and Whitney 1947), to compare all CPUE values of the three classes with each other. The tests revealed no significant difference between the three exposure time scenarios (24 vs. 72 h, $W = 78{,}512$, $p = 0.90$; 24 vs. 96 h, $W = 33{,}947.5$, $p = 0.16$; 72 vs. 96 h, $W = 17{,}086$, $p = 0.44$). All sampled fishes were identified to species level by morphological characteristics (Able 1990; Węsławski et al. 1990; Muus and Nielsen 1999; Hayward and Ryland 2005), except for fishes of the family Liparidae due to the potential for errors.

Standard length (SL) of all fishes was determined to the nearest 0.5 cm, and specimen was weighed with a precision of 1 g wet weight. For analysing species occurrence over all sampling sites, as well as species-specific size distribution, we used all fishes sampled in 2012 and 2013. Note that the length frequency distribution is only presented for the most abundant species, due to the sample size. For all site comparative analyses, however, we only used the data from week 26 to 40 of 2013, because during this time-period, all sites and subsites were sampled in parallel with an equivalent total fishing effort. Based on the absolute number of fishes caught at the sampling sites in this period, ecological metrics were calculated. Species richness (S) (Colwell 2009) was calculated as the total number of species per site. However, Colwell stated that a comparison of S values among different sites might be biased by significant differences in total abundance among these sites. Because of differences in the sample sizes between our sampling sites, we additionally calculated the rarefied species richness (Raup 1975). It is based on a random subsampling of the total samples to of every site. By this method, a standardised species richness for a sample size of $N = 100$ per site was calculated. Furthermore, the Shannon-Wiener diversity ($H'$) (Shannon 1948) and the associated species evenness ($J = H'/\ln(S)$) (Pielou 1966) were determined to evaluate species richness values independent of the total fish abundances at the different sites. These calculations were performed by the software Diversity (Version 1.6.2; Holland 2010).

An Ad Hoc opportunity allowed us to assess the structure of the seafloor in the six sampling areas from 15 July 2013 to 27 July 2013 by SCUBA diving and visual census. This made it possible to gather information about the habitat complexity of our fishing sites. All SCUBA assessments were carried out according to the German diving standard for scientific diving written in BGR/GUV-R 2112 (DGUV 2011). For these assessments, we used a 55 m long line on a diving reel. It was deployed in parallel to the sets of fyke nets. It started in water of 3 m depth down to the 12 m bathymetric contour. Depending on the slope of the coastline, the line did not extent to a water depth of 12 m, but was never shallower than the lower end of the double fyke net (minimum 8 m). The line had marks every meter and special marks every 10 m to give the observer distance information. A scientific diver followed the line with a head-mounted video camera, keeping a vertical distance of 1.5 m. The seafloor was thereby captured on video for a distance of about 1 m on both sides of the transect line. The videos were analysed in 5 m steps to assess (a) the percentage coverage by algae on an ordinal scale using the classes <25, 25–50, 50–75 and >75 %; (b) the sediment structure using the nominal classifications of sand, gravel, rock, and unidentified. All video analyses were performed repeatedly and independently by five divers. The values of the five divers were integrated by calculation of the modal value to a single value per 1 $\times$ 5 m segment.

To compare the abundance of fish at the different fishing sites, a CPUE value was calculated per site and sampling week. It is the average number of fish caught in one fyke net in 24 h. Because homogeneity of variance and normal distribution were not given, we chose to use the nonparametric Kruskal–Wallis rank sum test (Kruskal and Wallis 1952) for further analysis. First, we grouped the CPUE values according to their sampling sites, and analysed possible differences. Moreover, we grouped the CPUE values based on the location of the sampling site (Blomstrandhalvøya or south shore) and the modal value determined for the algal coverage per site. We repeated this analysis for species diversity, i.e., we determined the total number of species we caught per site and week. We grouped them for the four different analyses according to sampling site, location, and algal coverage. All tests and transformations were performed in R (R Core Team 2015). When significant differences were detected by the Kruskal–Wallis test, we used the Nemenyi tests for multiple comparisons of rank sums for post hoc analysis (Nemenyi 1963). We chose this test as it compensates for family-wise error with Chi squared approximation. For these operations, the R-Package PMCMR (Pohlert 2015) was used. Both analyses were performed with a significance level of 5 %.

To further analyse the inter-site differences in species abundance in more detail, we created a Bray–Curtis resemblance matrix based on the total number of each fish species per site, divided by the total fishing effort per site. The data were square-root transformed to reduce the effect of the highly abundant species. The resemblance matrix was used to create a two-dimensional non-metric multidimensional scaling (MDS) plot (Clarke 1993). Additionally, similarities were analysed by hierarchical cluster analysis using group means (Byrne and Uprichard 2012). These operations were performed with the software Primer 6 (Clarke and Gorley 2006). Bar charts and their metrics were calculated using R (R Core Team 2015) with the additional packages gplots (Warnes et al. 2015), lawstat (Hui et al. 2008), and reshape (Wickham 2007).

## Results

Integrated over all fishing campaigns in 2012 and 2013, a total of 2804 fish and 12 species (including the family Liparidae, which counted as a single group) were caught (Table 1). All species caught by the trammel nets in 2012 were also caught with the fyke nets. The only exception represents the two specimen of *Amblyraja radiata* (thorny skate), which were both caught in trammel nets. The three most common fish species in the catches were *M. scorpius,* with a total contribution of 74.9 %, *G. morhua* with a 17.2 % contribution, and *G. tricuspis* with a 3.8 % contribution. Fish of the family Liparidae represented a total contribution of 2.3 %. All other species including *Anarhichas lupus, Melanogramus aeglefinus* (haddock), *B. saida, Pollachius virens* (saithe), *Lumpenus lampraetiformis* (snake blenny), *A. radiata*, *Clupea harengus* (Atlantic herring), and *Cyclopterus lumpus* (lumpfish) were caught in much lower abundances, with contributions below 1 % per species (Table 1).

The modal value for standard length was 15.0 cm for *M. scorpius*, 15.0 cm for *G. morhua*, 14.5 cm for *G. tricuspis,* and 10.0 cm for Liparidae. (Table 1). Further analysis of the species-specific length data (Fig. 2) revealed a comparatively uniform length–frequency distribution in *M. scorpius*, with a maximum near its arithmetic mean (1st quartile 14 cm, median 16 cm, 3rd Quartile 17.5 cm). *G. morhua* showed a much wider and almost bimodal length–frequency distribution with a first maximum at 7 cm and a second maximum between 15 and 17 cm. A similar bimodal length–frequency distribution was also observed for *G. tricuspis* with a first maximum at 6 cm and a second maximum between 15 and 16 cm.

Based on the dataset of 2013, we compared the different sampling sites in Kongsfjorden (Table 2). The CPUE (average number of fish per net in 24 h) of the sites around Blomstrandhalvøya was 1.11 ($SD = 0.64$), while the sites at the south shore showed a value of 0.78 ($SD = 0.54$) (Fig. 3). A Kruskal–Wallis test showed a significant difference in CPUE between the two coastlines ($\chi^2$ (1, $N = 110$) $= 11.47$, $p < 0.01$). The average number of species counted per sampling week was 2.76 ($SD = 0.77$) for the sampling sites at Blomstrandhalvøya, and 2.25 ($SD = 0.77$) for the sampling sites at the south shore. The Kruskal–Wallis rank sum test reports the two groups as significantly different ($\chi^2$ (1, $N = 110$) $= 10.691$, $p < 0.01$). A site wise comparison of weekly CPUE and number of species could show no statistical significant differences (Fig. 3).

The underwater mapping of the six sampling sites showed distinct differences with respect to algal coverage and sediment structure (Table 3). The grouping of CPUE according to the modal value of algal coverage at the sampling sites shows also differences. The highest average CPUE was detected at the sites with the lowest algal coverage (1.1, $SD = 0.94$), the second highest at the sites of densest algal coverage (0.96, $SD = 0.49$). Statistical analysis reveals a significant difference between the sites of 50–75 % algal coverage and 75–100 % coverage ($\chi^2$ (9, $N = 66$) $= 12.483$, $p < 0.01$). The grouping of the total number of species per week and site according to algal coverage showed the highest species diversity at the site with the highest (75–100 %) and the lowest (25–50 %) algal coverage. The site with the intermediate (50–75 %) algal coverage showed the lowest average number of

**Table 1** Species list of all catches in 2012 and 2013. The total catch is presented for the two fishing set-ups used in this study. The percentage is calculated by the sum of all specimens caught. The minimum, maximum, and modal value of standard length is presented

| Species name | Total catch per net set-up | | Total (%) | Standard length (cm) | |
|---|---|---|---|---|---|
| | Trammel nets, fyke nets 10 June 2012–09 September 2012 | Double fyke nets, fyke nets w. bait 10 September 2012–06 October 2013 | | Range | Modal value |
| *Myoxocephalus scorpius* Linnaeus, 1758 | 557 | 1543 | 74.89 | 5.0–26.5 | 15.0 |
| *Gadus morhua* Linnaeus, 1758 | 63 | 420 | 17.23 | 5.5–82.0 | 15.0 |
| *Gymnocanthus. tricuspis* Reinhardt, 1830 | 22 | 85 | 3.82 | 4.5–20 | 14.5 |
| Liparidae | 4 | 59 | 2.25 | 6.0–19 | 10.0 |
| *Anarhichas lupus* Linnaeus, 1758 | 3 | 10 | 0.46 | 28.0–61.5 | 42.0 |
| *Melanogrammus aeglefinus* Linnaeus, 1758 | 11 | 1 | 0.43 | 6.0–19.0 | 17.5 |
| *Boreogadus saida* Lepechin, 1774 | 0 | 10 | 0.36 | 21.0–30.0 | 25 |
| *Pollachius virens* Linnaeus, 1758 | 0 | 6 | 0.21 | 6.0–16.5 | NA |
| *Lumpenus lampretaeformis* Wallbaum, 1792 | 1 | 4 | 0.18 | 24.0–41.0 | 40.5 |
| *Amblyraja radiata* Donovan, 1808 | 2 | 0 | 0.07 | 54.0–54.0 | 54.0 |
| *Clupea harengus* Linnaeus, 1758 | 0 | 2 | 0.07 | 19.0–19.0 | NA |
| *Cyclopterus lumpus* Linnaeus, 1758 | 0 | 1 | 0.04 | 20.0–20.0 | 20.0 |

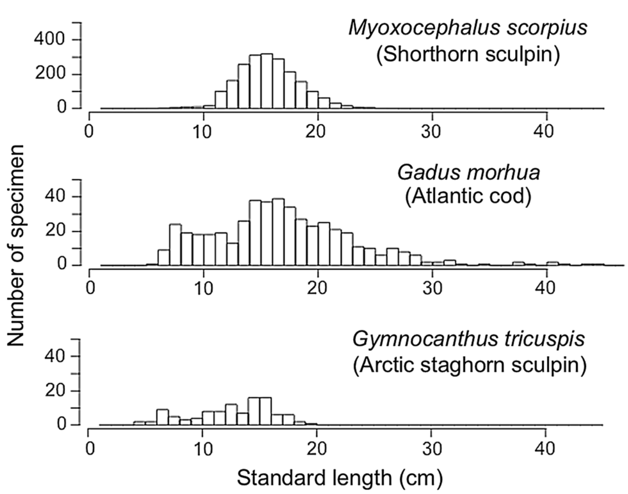



**Fig. 2** Cumulative standard length distributions of the three most abundant fish species, *M. scorpius*, *G. morhua* and *G. tricuspis*

species. Statistical analysis showed a distinct difference between 50–75 % algal coverage and 75–100 % coverage ($\chi^2$ (2, $N = 66$) = 11.576, $p < 0.01$).

The total number of species per site varied between four and seven for the whole sampling period (Table 2). The highest values of species richness (seven species) were found at the sites Hansneset Central, Hansneset South, and London along the shoreline of Blomstrandhalvøya. The sites with the lowest total number of species (four species) were Old Pier East (south shore) and Hansneset North (Blomstrandhalvøya) (Table 2). The calculation of the rarefied species richness for a unified sampling size of $N = 100$ per site showed similar results with the highest values at the sites Hansneset Central (6.3) and London (6.2) and the lowest values at the sites Old Pier East (3.5) and Old Pier West (4.0) (Table 2). The calculated values for Shannon's $H'$ integrate information about species richness and abundance. An apparent gradient was observed along the southern shoreline with the lowest $H'$ value of 0.51 at the site Brandal and the highest value of 1.05 at the innermost sampling site, Gåsebu (Table 2). At the shoreline of Blomstrandhalvøya, $H'$ values between 0.75 and 1.01 were observed with four of five stations having values between 0.95 and 1.01. A comparable west–

**Table 2** Species composition and abundance per sampling site. Additionally ecological indices for each sampling site are presented

| | Sampling site | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Brandal | Old Pier West | Old Pier Central | Old Pier East | Gåsebu | Sørvågen | Hansneset North | Hansneset Central | Hansneset South | London |
| Species composition (%) | | | | | | | | | | |
| *Myoxocephalus scorpius* Linnaeus, 1758 | 88.46 | 75.73 | 74.19 | 80.09 | 60.95 | 67.69 | 72.88 | 60.74 | 64.19 | 68.46 |
| *Gadus morhua* Linnaeus, 1758 | 6.15 | 17.48 | 17.34 | 15.17 | 21.30 | 20.77 | 20.90 | 32.59 | 25.33 | 18.12 |
| *G. tricuspis Reinhard, 1830* | 2.31 | 5.83 | 4.44 | 4.27 | 14.79 | 5.38 | 0.56 | 0.74 | 0.44 | 6.04 |
| Liparidae | 1.54 | 0.49 | 2.42 | 0.00 | 1.78 | 4.62 | 5.65 | 0.74 | 7.42 | 4.70 |
| Other | 1.54 | 0.49 | 1.61 | 0.47 | 1.18 | 1.54 | 0.00 | 5.19 | 2.62 | 2.68 |
| Fishes per net per day ($n \pm$ SD) | 0.49 ± 0.17 | 1.07 ± 0.77 | 0.79 ± 0.37 | 0.98 ± 0.69 | 0.57 ± 0.20 | 1.62 ± 1.10 | 1.12 ± 0.44 | 0.64 ± 0.18 | 1.18 ± 0.33 | 0.98 ± 0.34 |
| Species richness | 6 | 5 | 6 | 4 | 6 | 5 | 4 | 7 | 7 | 7 |
| Rarefied species richness ($N = 100$) | 5.48 | 3.97 | 5.15 | 3.47 | 5.12 | 4.95 | 3.56 | 6.33 | 5.38 | 6.24 |
| Shannon's $H$ | 0.51 | 0.73 | 0.83 | 0.62 | 1.05 | 0.95 | 0.75 | 0.95 | 0.97 | 1.01 |
| Pilou's $J$ | 0.28 | 0.46 | 0.46 | 0.45 | 0.58 | 0.59 | 0.54 | 0.49 | 0.50 | 0.52 |

east trend along the southern shoreline was observed for Pilou's $J'$ (Pielou 1975), a measure of species evenness. The lowest value was determined for the site Brandal (0.28) and highest value at Gåsebu (0.58). Along the north shore, values ranged between 0.49 at Hansneset Central and 0.59 at Sørvågen (Table 2).

The calculated two-dimensional MDS plot shows a Kruskal stress value of 0.09 (Fig. 4). Ordination of sites occurred in groups according to their general geographical locations from west (left) to east (right). An exception are the sites at Hansneset that cluster on the lower right of the plot. Hierarchical cluster analysis of the Bray–Curtis resemblance matrix was used to add further information to the MDS plot. The overall similarity between all sites was determined to be over 77 %. Using a similarity level of >80 %, two clusters could be discriminated (Fig. 4). One contained Hansneset South, Hansneset North, and Hansneset Central. The second contained all other sites, except the site Brandal, which is not part of any cluster. Two subclusters at a similarity level of >90 % could be identified. The first contained the sites London and Sørvågen, and the second contained Old Pier West and Old Pier Central.

## Discussion

This study provides the first systematic field data of the shallow water fish community of this polar ecosystem. According to Hop et al. (2002), such data on the higher trophic levels of shallow water polar ecosystems are completely missing but urgently needed for a basic understanding of the functional responses of Arctic ecosystems to environmental changes like global warming.

Our most striking result is the distinct dominance of the shallow water fish community in Kongsfjorden by the two species *M. scorpius* and *G. morhua,* while other species that were found in previous studies were rare or absent. This is in good agreement with Hop et al. (2002). The apparent lack of *B. saida* in our study, e.g., is most remarkable because previous studies reported high abundances of *B. saida* in the offshore and coastal waters of Kongsfjorden (Haug and Gulliksen 1982). Similarly, we could not detect any *Mallotus villosus, Sebastes sp.* (redfish) or *Hippoglossoides platessoides* (American plaice), which were reported in previous studies (Hop et al. 2002). A key to understand these differences in species occurrence in comparison with previous studies may be the lifestyles of these species. *B. saida* is known to prefer sea ice as habitat for young-of-year (YOY) and older specimen which are often found below sea ice at sub-zero temperatures (Lønne and Gulliksen 1989; Hop and Gjøsæter 2013). The apparent lack of sea ice in Kongsfjorden since the winter of

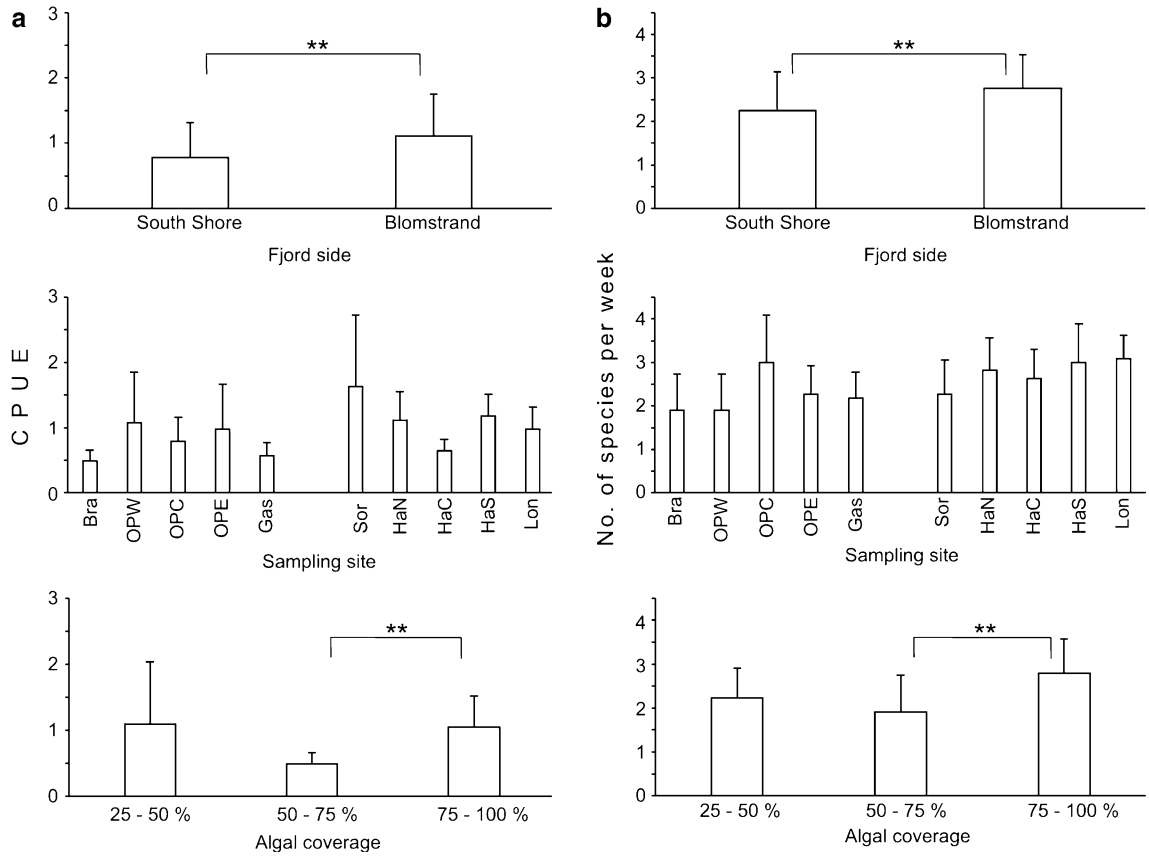



**Fig. 3** **a** Number of fish per net SD in 24 h (CPUE) and **b** Total number of species per week. Values are grouped by their geographical location (*upper panel*), their sampling site (*middle panel*) or the modal value of algal coverage at their respective sampling site (*lower panel*)

2005/2006 might therefore be a factor that explains the low abundances of *B. saida* in Kongsfjorden (Cottier et al. 2007). Additionally, Fischer et al. (2016) reports temperatures of 4 to 8 °C from June to September 2014 at 10 m water depth at the Old Pier in Ny-Ålesund. This clearly exceeds the preferred temperature range of *B. saida* and might additionally explain its low abundances. Also Hop et al. (2002) reports that *B. saida* and *M.* villosus could be found in the pelagic realm of the fjord, which was not sampled in this study. For a similar reason, *H. platessoides* might not be represented in this study. This species prefers deeper waters below 90 m, and it is likely that it is therefore not caught in the shallow waters of Kongsfjorden. In contrast, *M. scorpius* is a classic demersal fish species of shallow water habitats (Lamp 1966). Therefore, the dominance of this species in our fyke net catches is not surprising. Also, young specimen of *G. morhua* are well known to seek protection in structural complex shallow water habitats to avoid predators and cannibalisation by larger specimen, explaining the high abundance of this species with younger age classes in the shallow waters.

Another explanation for the dominance of *M. scorpius* and *G. morhua* in our study might be gear selectivity. It is known that fyke nets are size and species selective (Hubert et al. 2012). Cover-seeking mobile species are reported to be most susceptible to capture by this gear type (Hubert et al. 2012). On the other hand, we minimised size selectivity by small mesh size, and we could not detect a change in species composition with regard to the trammel nets that we used in 2012. The dominance of *M. scorpius* in the shallow areas of Kongsfjorden was also confirmed by continuous diving operations during the last 15 years (Max Schwanitz, pers. com.). Therefore, we doubt that the generally low share of Arctic fishes (<5 %) in our catches in the shallow water system of Kongsfjorden was an artefact.

Parallel to our study Mark (2013) performed trawling catches with RV Heincke in outer Kongsfjorden and also caught *G. morhua* (and the pelagic species *Melanogrammus aeglefinus*) in higher abundances and missed *B. saida*, or *M. villosus* in the catches. Even though this study used a completely different fishing gear, it is most interesting that the species composition was similar to our study with a high dominance of *G. morhua* during 2013. The standard length (SL) of *G. morhua* captured offshore was reported as 5.5–9.5 cm, which was quite below the average length of 15.0 cm in our study. Surveys at the Lofoten in January

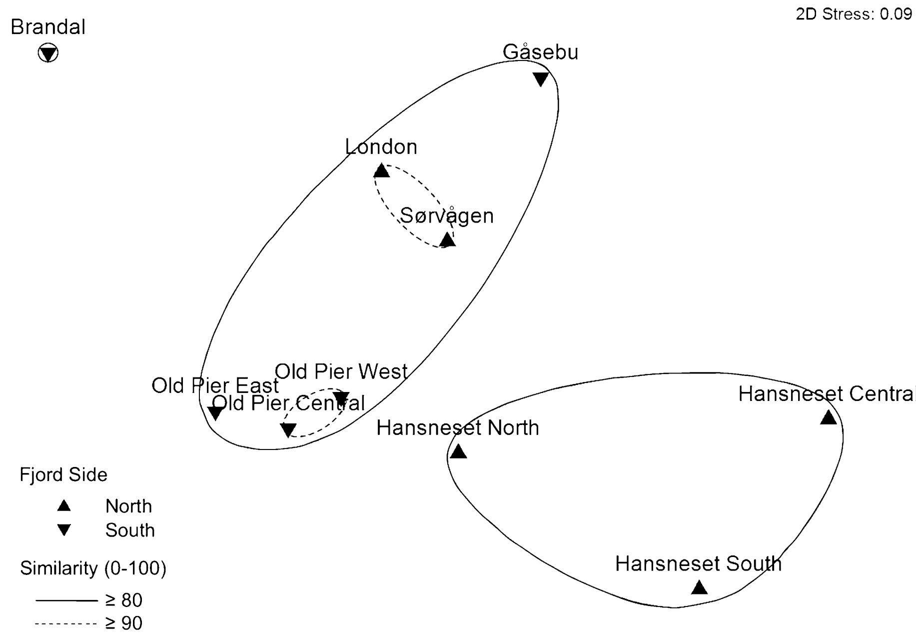
**Table 3** Algal coverage and sediment structure at the sampling sites from 3 to 12 m water depth. The bottom structure was determined by SCUBA video survey and subsequent video analysis

| Distance from shore (m) | Brandal | Old Pier East | Gåsebu | Sørvågen | Hansneset South | London |
|---|---|---|---|---|---|---|
| (a) *Algal coverage* | | | | | | |
| 5 | 1 | 3 | 2 | 4 | 4 | 2 |
| 10 | 1 | 4 | 2 | 2 | 4 | 2 |
| 15 | 2 | 4 | 2 | 4 | 4 | 3 |
| 20 | 3 | 4 | 2 | 2 | 4 | 3 |
| 25 | 3 | 4 | 2 | 4 | 4 | 3 |
| 30 | 2 | 4 | 2 | 3 | 4 | 4 |
| 35 | 3 | 4 | 1 | 2 | 4 | 4 |
| 40 | NA | 4 | 2 | 2 | 4 | 4 |
| 45 | NA | 4 | 2 | 3 | 4 | 4 |
| 50 | NA | 4 | 2 | 4 | 4 | 4 |
| 55 | NA | 4 | 2 | 2 | 4 | 4 |
| Modal value | 3 | 4 | 2 | 2 | 4 | 4 |
| Key | | 1 | 2 | 3 | 4 | |
| | | 0–25 % | 25–50 % | 50–75 % | 75–100 % | |
| (b) *Sediment structure* | | | | | | |
| 5 | 1 | 1 | 1 | 4 | 4 | 1 |
| 10 | 1 | 1 | 1 | 4 | 4 | 1 |
| 15 | 1 | 1 | 1 | 1 | 2 | 1 |
| 20 | 1 | 1 | 1 | 1 | 2 | 1 |
| 25 | 1 | 4 | 1 | 1 | 4 | 1 |
| 30 | 1 | 1 | 1 | 1 | 4 | 1 |
| 35 | 1 | 1 | 1 | 1 | 4 | 4 |
| 40 | NA | 1 | 1 | 1 | 4 | 4 |
| 45 | NA | 2 | 1 | 1 | 4 | 4 |
| 50 | NA | 2 | 1 | 4 | 4 | 4 |
| 55 | NA | 1 | 1 | 4 | 4 | 4 |
| | 1 | 1 | 1 | 1 | 4 | 1 |
| Key | | 1 | 2 | 3 | 4 | |
| | | Sand | Gravel | Rock | Fully covered | |

2002 revealed a mean standard lengths of 12.4 cm at age 1+, 19.9 cm at age 2+ and 31.4 cm at age 3+ (ICES 2005). Assuming a similar growth rate, the majority of Atlantic cod caught in our study belonged to the age classes 1+ and 2+ (Fig. 2) and age-0 fish were almost completely missing. This is interesting with respect to the fact that it is generally assumed that cod in Svalbard belongs to the Arcto-Norwegian cod stock, which is known to spawn along the Norwegian coast from Møre up to East Finnmark (Sarvas and Fevolden 2005; Sundby and Nakken 2008). Assuming that our shallow water specimen and the offshore YOY specimen from the Heincke cruise belonged to the same stock, this would imply a long distance migration of the YOY from offshore towards the Kongsfjorden ecosystem. It is unfortunate that there is to our knowledge no study in a similar area of the Arctic that focuses on the interactions and exchange between the fish communities of the shallow waters, deep waters, and pelagic realms. However, it is generally known that fish of the same size classes spatially separate in different habitats. Especially, smaller fishes often prefer complex shallow water habitats to avoid predation and cannibalism as well as to use these habitats to access suitable food sources. That we did not catch specimen of *G. morhua* with a SL <10 cm in our nets could be explained by the theory that they gain a size of about 10 cm SL before migrating into the shallow water habitat of Kongsfjorden. Another explanation might be that we simply were not able to catch YOY specimen of *G. morhua* due to gear selectivity. The fyke nets and trammel nets we used had mesh sizes of 10 and 12 mm. The body height of *G. morhua* with standard length below 10 cm is expected to be smaller than 10 mm; therefore, they might have passed our nets without any detection.

Our study shows an increase in biodiversity along the south shore of the fjord from the west to the east. The water mass dynamics of Kongsfjorden may be the key to

**Fig. 4** Multivariate analysis (MDS) of total catch per fish species per sampling site, based on square-root transformed data and Bray–Curtis dissimilarity index. Hierarchical cluster analysis was used to identify cluster with a similarity of >80 and >90 of 100




understanding this pattern in species distribution. Willis et al. (2006) showed the existence of a cyclonic and counter-clockwise gyre in the Kongsfjorden. Atlantic and Arctic water masses are mixed at the fjord mouth and are driven along the south shore to the inner parts of the fjord. Along the south shore, fresh water from the glaciers mixes continuously with the imported water masses. This results in a continuous reduction of salinity and a decrease in temperature, forming a gradient towards the inner parts of the fjord. Our study showed an increase of biodiversity and species evenness along this gradient from the westerly to the easterly sampling stations. *M. scorpius,* the most abundant and typically Atlantic species in this study, showed its highest abundance at the westernmost station Brandal, which is most influenced by Atlantic and, thereby, warmer and more saline water masses. The lowest CPUE for *M. scorpius* was found at the most eastern station Gåsebu. This station is most influenced by glacial melt water and, thereby, characterised by lower water temperatures and lower salinities. The reverse trend was found for *G. tricuspis,* a typical Arctic cottid fish species with anti-freezing proteins. Its highest abundance was found at the easternmost stations Gåsebu, its lowest abundances at the westernmost station Brandal. Low salinities and low water temperatures are characteristic for Arctic water masses. Thereby, this hydrographic gradient in Kongsfjorden resembles a small scale Atlantic–Arctic gradient. The reciprocal maximum in CPUE of *M. scorpius* and *G. tricuspis* along the south shore indicates a concurrency between the two species. This hypothesised co-occurrence between *M. scorpius* and *G. tricuspis* is most interesting with respect to the spatial distribution of fish along polar gradients. *M. scorpius* is a generalist with respect to food sources with a fast reproductive cycle, and it can tolerate a wide range of salinities and temperatures (Ennis 1970; Luksenburg and Pedersen 2002). In contrast, G. *tricuspis* is an cold-adapted Arctic species with a smaller range of tolerated hydrography. It is possible that the two species are in competition regarding the same ecological resources such as food, and the differences in physiological adaption in combination with hydrography lead to this difference in small-scale geographical distribution. One factor could be the costs versus the advantages of the anti-freezing proteins of *G. tricuspis.* But it has to be noted that adaptive mechanisms to low temperatures and their metabolic costs are not yet universally accepted (Steffensen 2002). Moreover, the species-specific expression of cold adaptations with respect to seasonality and environmental parameters is still subject of research (di Prisco 2000; Enevoldsen et al. 2003).

On the shoreline of Blomstrandhalvøya, the sites Sørvågen and London that were constantly influenced by the water mass export from inner Kongsfjorden showed the highest abundance of *G. triscuspis*. Also they showed the closest resemblance to Gåsebu with respect to species abundance (Fig. 4). The sampling site Hansneset showed the most distinct pattern in species abundance and diversity that might be driven by its exposed position towards the mouth of the fjord, and a thereby diverse influence of different water masses at this site (Willis et al. 2006).

A further factor that influences the abundance and species diversity of the different sampling sites is the algal coverage. It is known that habitat structure is one of the most important factors for the temporal and spatial distribution of fish abundance (Crowder and Cooper 1982; Johnson and Beaumier 1988; Persson and Eklöv 1995; Fischer 2000, 2004). In this study, we were able to show that the sites with highest species abundance with respect to CPUE were the sites with an algal coverage of 75–100 %, and the sites with the second highest coverage the one with 25–50 %. It can be assumed that the sites with the highest coverage and most structural complexity provided the best protection for fish, but also for prey. Sites with lower structural complexity represented therefore an easier feeding ground for fish, explaining the high CPUE at sites with 25–50 % algal cover. Only one site in this study (Brandal) showed an algal coverage of 50–75 %. The low CPUE at this class of algal coverage might thereby be an site-specific effect. Alternatively, it could be explained by the assumption that the intermediate algal coverage was not optimal for hunting and also not for structural protection. Therefore, it might have shown the least abundance of fish.

For sure, the assessment of this study can only be seen as a snapshot within a limited time scale and a limited number of sampling stations. Nevertheless, a more detailed future study about the influence of algae coverage on the Kongsfjorden shallow water ecosystem is advised. Future replications of this study should integrate a thorough habitat mapping and detailed hydrographic assessments to further analyse the link between fishes and habitat variables in Kongsfjorden. Further investigations on the importance of algal belts in Kongsfjorden as nursing grounds for fish are also warranted, and ongoing long-term observation by automatic camera systems like the Kongsfjorden Underwater Observatory (Fischer et al. 2016), could potentially improve our understanding of the year-round cycles of the fish and macrozoobenthos communities of Kongsfjorden. A publication about the inter-annual variability, assessed by automatic camera systems (Wehkamp and Fischer 2014) and fyke net fishing from 2012 to 2014 is currently in preparation. But also the replication of this study in shallow water systems of other Arctic fjords is necessary to understand if Kongsfjorden shows special characteristics or common trends. It is possible that the results of this study, especially, the dominance of boreal species are connected to changes in the algal belts and the macrozoobenthos community that have been documented by Bartsch et al. (2016) and Paar et al. (2016). To further explore the connection between the sampling sites and the fish community cooperative publications regarding growth rates, nutritional status, and trophic relationships of shallow water fishes at Hansneset are currently in preparation. In combination with future reassessments, this study can be used as a baseline for the detection of borealisation in Kongsfjorden.

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**Compliance with ethical standards**

**Ethical approval** All applicable international, national, and/or institutional guidelines for the care and use of animal were followed.

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**Cover page - Publication II**

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# First year of practical experiences of the new Arctic AWIPEV-COSYNA cabled Underwater Observatory in Kongsfjorden, Spitsbergen

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**Abstract.** A combined year-round assessment of selected oceanographic data and a macrobiotic community assessment was performed from October 2013 to November 2014 in the littoral zone of the Kongsfjorden polar fjord system on the western coast of Svalbard (Norway). State of the art remote controlled cabled underwater observatory technology was used for daily vertical profiles of temperature, salinity, and turbidity together with a stereo-optical assessment of the macrobiotic community, including fish. The results reveal a distinct seasonal cycle in total species abundances, with a significantly higher total abundance and species richness during the polar winter when no light is available underwater compared to the summer months when 24 h light is available. During the winter months, a temporally highly segmented community was observed with respect to species occurrence, with single species dominating the winter community for restricted times. In contrast, the summer community showed an overall lower total abundance as well as a significantly lower number of species. The study clearly demonstrates the high potential of cable connected remote controlled digital sampling devices, especially in remote areas, such as polar fjord systems, with harsh environmental conditions and limited accessibility. A smart combination of such new digital "sampling" methods with classic sampling procedures can provide a possibility to significantly extend the sampling time and frequency, especially in remote and difficult to access areas. This can help to provide a sufficient data density and therefore statistical power for a sound scientific analysis without increasing the invasive sampling pressure in ecologically sensitive environments.

## 1 Introduction

Kongsfjorden (78°55′ N, 11°56′ E) on the western coast of Spitsbergen (Fig. 1) is described as one of the best studied polar fjord systems in the Arctic (Wiencke, 2004). The 20 km long ecosystem opens without a sill in a westerly direction toward the Fram straight (Hop et al., 2002) and is alternatively penetrated by warm saline Atlantic water masses from the West Spitsbergen Current, by cold less saline Arctic water from the East Spitsbergen Current, or a mixture of both (Cottier et al., 2005). This bi-modal hydrographic situation leads to a complex spatio-temporal pattern in the fjord hydrography with an occasionally more Atlantic and in other instances more Arctic characteristic with respect to the water masses, even in the inner fjord system (Svendsen et al., 2002). Due to an increased advection rate of warmer Atlantic water masses in the fjord systems over the last decade (Cottier et al., 2005), the first signs of an overall warming of the fjord system have been observed, with an overall decrease in seasonal ice coverage (Walczowski et al., 2012), signifi-


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**Figure 1.** Spitzbergen with Kongsfjorden (★ in the small inlay panel in the upper left corner) and the location of NyÅlesund in Kongsfjorden (★). Source: Norwegian Polar Institute (2014), 2017.

cant changes in the phytoplankton community (Hegseth and Tverberg, 2013; Willis et al., 2006), changes in the depth distribution of macroalgae in the shallow waters (Bartsch et al., 2016) and in the macrozoobenthos community (Parr at al., 2015), as well as an increase in turbidity due to increased meltwater runoff from the glaciers (Peterson et al., 2002; Bartsch et al., 2016). Although Renaud et al. (2011) and Voronkov et al. (2013) recently started to study the food-chain length, trophic levels, and the main feeding groups in Kongsfjorden, our knowledge of the temporal and spatial dynamics of the higher trophic levels of the food web is still extremely limited (Stempniewicz et al., 2007). Therefore, important knowledge gaps such as a lack of quantitative data on production, abundance of key prey species, and the role of advection in the biological communities in the fjord still exist (Hop et al., 2002).

Such knowledge, however, is mandatory for a better understanding of this polar fjord system and potentially to use it as a model system for future Arctic change scenarios under the pressure of global warming. The most comprehensive review thus far of the occurrence and higher trophic level species in the Kongsfjorden ecosystem has been performed by Hop et al. (2002) and revealed approximately 34 zooplankton taxa, between 29 and 396 macrozoobenthos species, as well as approximately 30 fish species in the fjord system in total, depending on the type of substratum. Most of these data have been sampled during intense summer campaigns with ship-supported sampling methods or by occasional scuba diving operations at different sites of the fjord. Although these datasets are highly valuable, they are mainly restricted to the polar summer when light is available and sampling can be performed on a regular basis. A systematic year-round assessment of the fjord community, especially of the shallow water habitats, which are well known as most important as spawning, hatching, and nursery grounds for juvenile specimens (Fischer and Eckmann, 1997a, b; Werner, 1977), is missing.

Thorough assessments especially of higher tropic levels such as fish and macroinvertebrates are demanding already in northern temperate non-polar waters because of the required logistics, methods, and manpower (Wehkamp and Fischer, 2013a, b, c). In Arctic waters with the even harsher conditions with respect to low winter temperatures, seasonally limited daylight availability and a partial or complete ice coverage, longer-term and year-round assessments especially in shallow coastal areas are almost completely lacking. Furthermore, in several hard bottom fjord systems, such as the Kongsfjorden system, the shallow water areas are relatively inaccessible by trawling with larger vessels due to a complex and highly structured benthic habitat, with a mixture of rocky bottom and ice-rafted pebbles and stones (Jørgenson and Gulliksen, 2001). Therefore, most available studies are temporally restricted to the summer months and the open or deeper water bodies.

In the present study, we present data from a 13-month (October 2013 to November 2014) long hydro-biological survey in the sublittoral zone of the Arctic Kongsfjorden at the southern shoreline close to the research village of NyÅlesund at UMT 8763953° N, 433992° E (Fig. 1). With a 2012 installed cabled underwater observatory (COSYNA@AWIPEV Underwater Observatory – subsequently called UWO), we continuously recorded the main hydrological parameters temperature, salinity, pH, Chl $a$, and turbidity and additionally made a quantitative analysis of the abundance, species occurrence, and (for selected species) length–frequency distribution of the fish and macroinvertebrate taxa. For the latter assessment, a stereo-optical macrobiota observatory called "RemOS1" (Remote Optical System) was used, specifically designed for long-term exposure and assessments of fish and macroinvertebrate communities in shallow water areas (Fischer et al., 2007b). Data acquisition was conducted year-round, remote controlled with a temporal resolution of 1 Hz for the hydrological data and with a stereoscopic imaging frequency of 30 min. Parallel to this study, classic fishing campaigns were performed in 2012, 2013, and 2014 in the months June/July and September in the same area with standard fyke nets to provide ground-truth data for the remotely sampled fish data. These fishing data are published in Brand and Fischer (2016) for the years 2012 and 2013. The data for 2014 will be published together with a comparative analysis of the results of the UWO elsewhere (M. Brand, personal communication, 2016).

The present study aims to demonstrate the high potential of remote controlled sensors to quantitatively assess not only hydrological data such as temperature, current, or plankton community with classical CTD (conductivity–temperature–depth) probes or VPRs (video plankton recorders), but also for the assessment of higher tropic levels such as macroinvertebrates and fish. To the best of our knowledge, there are only a small number of studies and observatories available worldwide that are trying to also assess higher trophic levels with remote controlled optical systems (Aguzzi et al., 2011; Buckland et al., 2005; Fischer et al., 2007b; Wehkamp and Fischer, 2014), and even fewer with regard to quantitative assessments with respect to a specimen's abundances and species-specific length–frequency analysis in an area. Because these technologies will certainly develop and improve over the next years, this study also discusses certain specific requirements and challenges for such systems, especially for shallow water Arctic areas.

## 2 Materials and methods

The UWO was built up in 2012 in the framework of COSYNA (Coastal Observing Systems of the Northern and Arctic Seas). The system comprises a land-based FerryBox system equipped with various hydrographic sensors (Table 1) receiving water from a remote controlled underwater pump station at 11 m water depth. Additionally, a cable connected (fibre-optic and 240 V power) underwater node (Fig. 2) was installed close to the pump station at a 11 m water depth providing power (48 V) and a network (TCP/IP 100 Mbit) connection to additional in situ sensors. To install or exchange sensor equipment at the node system by divers, the node is equipped with four underwater matable power/ethernet connectors and two additional underwater matable power/rs232 connectors.

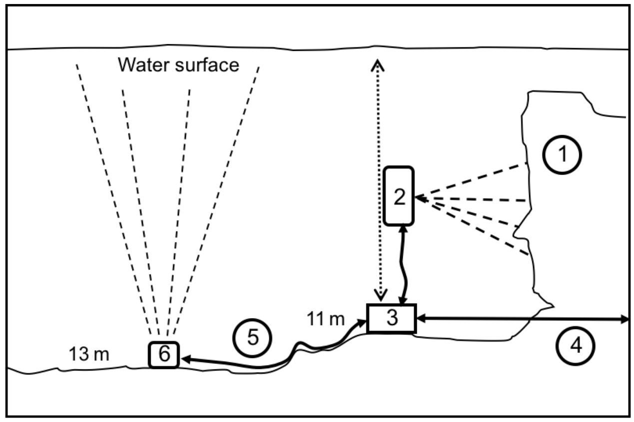


**Figure 2.** Sketch of the underwater installations with the underwater base station and the vertical profiling unit off NyÅlesund. Numbers refer to numbers in the sketch. (1) Steep wall (drop-off) with vertical zonated macrophyte coverage. (2) Vertical profiling sensor carrier with CTD and a stereo-optical imaging device (RemOs1) looking towards the wall. (3) Underwater node with wet-matable plugs. (4) Combined power/fibre-optic cable to land. (5) Combined power/rs232 cable from node to ADCP. (6) ADCP. For details on the single components, see the text.

For the experiment described in this study, the node system was equipped with an upward looking ADCP positioned at 13–15 m water depth (depending on the tide cycle), a SBE38 temperature sensor positioned at 11–13 m water depth (depending on the tide cycle), and a vertical profiling sensor carrier. The profiling sensor carrier was fully remote controlled via the Internet and was operated year-round from October 2013 to November 2014 from Germany. It was equipped with a CTD for the assessment of the main hydrographical parameters and the RemOS1 stereo-optical camera system (Fischer, 2017; Fischer et al., 2007b; Wehkamp and Fischer, 2014) for macrobiota assessments. Using the stereo-optic sensor, we assessed the macrobiota, jellyfish, and fish community along the vertical depth profile from 11 m water depth to the surface with the sensors looking from a distance of about 2.5 m towards a steep wall that reached from 11 m of water depth to 3 m below the mean sea level (Fig. 2). The upper part of the wall was dominated by brown algae of the type of *Alaria esculenta*, the lower part by *Saccharina latissima* and the two red algal species *Phycodryis rubens* and *Ptilota gunneri*. Using the vertical profiling unit, we conducted a 1-year continuous stereo-optical survey of the fish and the macrozoobenthos community in five depth strata (11–9, 9–7, 7–5, 5–3, and 3 m from the water surface). The stereo-optical system and the CTD probe were remotely positioned every day between 11:00 and 13:00 h in one of the five depth layers, with the exact depth being calculated as the distance from the bottom. This means that the effective water depth

**Table 1.** Sensors attached to the COSYNA@AWIPEV UWO at UMT 8763953° N, 433992° E. The FerryBox has its water inlet at a fixed depth of 11 m below mean sea level (http://vannstand.no/index.php/nb/english-section/sea-level-data). The RemOs1 system is profiling from 11 m water depth to the surface (for further descriptions, see the text).

| Sensor carrier | Sensor type | Water depth | Sensor unit manufacturer |
|---|---|---|---|
| FerryBox | Water temperature (°C) | 11 m | SBE45 |
| | Conductivity ($\mathrm{ms\,m^{-1}}$)/salinity (PSU)[1] | | SBE45 |
| | Oxygen (%) | | Anderra |
| | Chl $a$ ($\mathrm{mg\,m^3}$) | | Cyclops |
| | pH | | Meinsberg |
| | Turbidity (FTU) | | Seapoint |
| Underwater node | Current (ADCP Teledyne Workhorse 600 kHz) | 13 m | Teledyne |
| Underwater node | Stereo-optical imaging system RemOs1 | Profiling[2] | Fischer et al. (2007) |
| | Pressure (dbar) | | |
| | Water temperature (°C) | | |
| Underwater node | Conductivity ($\mathrm{ms\,m^{-1}}$)/salinity (PSU)[1] | Profiling[2] | Sea&Sun CTD90 |
| | Oxygen (%) | | |
| | Chl $a$ ($\mathrm{mg\,m^3}$) | | |
| | Turbidity (FTU) | | |

[1] Calculated after actual UNESCO procedures. [2] Between 11 m water depth and the surface.

changed with the tide cycle for max. 1.5 m, but the system itself had a fixed position above the ground (1 m distance from the bottom for the depth stratum 11–9 m, 3 m distance for the depth stratum 9–7 m, 5 m distance for the depth stratum 5–7 m, 7 m distance for the depth stratum 3–5 m, and 9 m distance for the depth stratum 3–0 m). The daily target depths were selected randomly for each week such that all of the depth strata were sampled once per week for 24 h. Missing depths, e.g. because of system or connection problems to the underwater observatory, were repeated on the weekend. The system was positioned for 24 h at the selected depth stratum and made stereoscopic images every 30 min. In parallel, all other in situ and FerryBox sensors recorded with a frequency of 1 Hz. The image pairs and all the hydrographic data were transferred automatically via the Internet to Germany for further daily processing. All hydrographic data were automatically quality controlled by automated procedures, flagged as good, probably good, and bad, and stored at a central data server in Geesthacht, Germany, under an open-access policy at http://codm.hzg.de/codm/. For our study, only the data with the quality flags probably good and good were used. Based on these data, we analysed the temporal succession of the shallow water fish, jellyfish, and macrozoobenthos community in this kelp-dominated shallow water Arctic habitat in Kongsfjorden. Organisms on the stereoscopic images were analysed in a two-step procedure following the routines described in Wehkamp and Fischer (2014). The 48 stereoscopic image pairs of each day were first scanned manually for the presence of organisms. This scanning was performed with image analysis software that presented the left image of the stereoscopic pair for at least 5 s on a 21″ high-resolution computer screen. Only two persons did this basic analysis step over the entire year and thoroughly counterchecked their object findings. During this first step, all the specimens found on an image were counted and pre-classified into the categories fish, jellyfish, appendicularia, pelagic crustacean, benthic crustacean, pteropods, and chaetognats. Organisms that could not be classified into one of these categories were classified as “others”. The analyser (the person who did the analysis) had the possibility of increasing or decreasing the image brightness or of enhancing the contrast by a single mouse click quickly. The possibility of such a rapid preprocessing of the 48 stereoscopic image pairs was revealed to be most important because 48 image pairs were produced every day year-round. This rapid assessment procedure allowed a first analysis of all the images per day within approximately 15 min, so that a quasi-online overview of the actual situation under water in the target area and of the functioning of the monitoring system was achieved within 24 h. With this procedure, problems of the system itself or with the data transfer could be detected fast and could be addressed and solved. With this daily rapid assessment routine, we could achieve an acceptable level of operational stability of the systems with less than 15 unplanned offline days over the entire sampling period of 13 months. Unplanned offline days occurred mainly due to failures in the land-based power support system. During such phases, the underwater part of the system was shut down to avoid hardware damage due to spontaneous and possibly critical voltage fluctuations.

In a second image analysis step, all the images where organisms were detected were rectified, which means that the geometry of the images was corrected to eliminate image distortions due to the lens of the camera. This correction was performed with the "stereo_gui" modified MATLAB routine (Wehkamp and Fischer, 2014). After this step, all the objects that were detected in the first image analysis step were measured (standard length in fish, carapax length in macrocrustacea, and max. dimension in all other organisms) and identified as precisely as possible, i.e. to species level in most fish species except for the two cod species *Boreogadus saida* and *Gadus morhua*, which were not distinguished properly on the images. Furthermore, amphipoda or appendicularia were only identified to the class level.

Because we had a clearly restricted water volume that was assessed by the camera system (volume between the camera and the vertical wall), we calculated the "catch per unit effort" of the system by summarizing all the individuals found on the images per 24 h and depth stratum. These CPUE $\times$ 24 h$^{-1}$ data were used as the basis for all further calculation. We did not recalculate these data on a defined water volume (which is possible) to avoid confounding calculations between benthic organisms living on the two-dimensional bottom or the surface of the algae and planktonic organisms living in the three-dimensional water column.

Length–frequency measurements on the three-dimensional-image pairs were performed pooled for each month for the cod species (mainly *Gadus morhua*), the common sea spiders (*Hyas araneus*), the two main jellyfish species (*Beroe* sp. and *Aglantha digitale*), the appendicularia, and the pteropods (*Clione limacina*). For these species, all the organisms were measured except for the month when more than 200 specimens occurred within 1 month. In this case, only 200 specimens were measured by randomly selecting over the day of the month.

## 3 Results

### 3.1 Habitat description

The Kongsfjorden shallow water ecosystem is characterized by large kelp beds of different species of macroalgae between 0 and approximately 12–15 m water depth (Bartsch et al., 2016). The site where the observatory has been set up is, therefore, characteristic of the fjord habitat and provides a highly diverse habitat with a steep wall completely covered with large macroalgae followed by a sandy to muddy slope that begins at approximately 11 m water depth at the base station of the observatory. The five depth layers covered by the stereo-optical camera system cover the typical vertical gradient of a littoral habitat with a surface near-pelagic habitat (depth range 0–2 m water depth (Fig. 3a), a typical litho-pelagic habitat close to the upper edge of the drop-off (2–4 m water depth (Fig. 3b), the upper drop-off edge between 4 and 6 m water depth) with dense horizontal and vertical macrophyte coverage (Fig. 3c), the vertical wall of the drop-off with overhanging structures and grotto-like crevices (water depth 6–8 m, Fig. 3d) and, finally, the lower edge of the drop-off where the wall goes over in the typical benthic habitat with a gentle slope formed by sand and mud at a depth of around 11 m, decreasing further towards north to the centre of the fjord (Fig. 3e).

The observatory technology allows for daily vertical CTD profiles every noon at approximately 12:00 with a sampling frequency of 1 Hz at a constant profiling speed of 1.5 m per minute from approximately 10 m water depth (depending on the tide) to 1 m below the surface. The FerryBox unity additionally provides complementary hydrographic data from a fixed water depth of 11 m. Figure 4 shows the compiled data for water temperature (°C), salinity (PSU), and turbidity (FTU) from October 2013 to November 2014. The data reveal a distinct seasonal cycle in the water temperature, with the lowest values of approximately $-1.0\,°C$ in the winter months from October to April and the highest temperatures up to approximately 8 °C during the summer months, May to September. Most interestingly, however, are the distinct short-term changes in water temperatures even within the individual seasons. These changes spanned ranges of up to 4 °C within the shortest time periods of a few days both in the summer and in the winter. While the average water temperature, for example, during the middle of December to the end of January was between $-0.5$ and $+0.5\,°C$, the water temperatures then suddenly increased within a few days up to 3 °C and stayed at this comparatively high level until the end of March, when it dropped again to approximately 0.5 °C. In May, the temperatures increased again and reached the highest values of up to 7.7 °C in the surface layers, which indicates a distinct stratification during this time. In July to September, this stratification dissolved, and the water temperatures were almost equally distributed over the water column. Similar temporal patterns were observed also in salinity (Fig. 4), which indicates that the overall patterns in the water temperature in the shallow littoral zone of the fjord system were also significantly determined by a fast (within days) exchange of water masses that brought either colder and lower saline Arctic water or warmer higher saline water masses even to the shallow fjord areas.

Figure 4 shows the seasonal patterns in turbidity over the water columns. The data indicate that the overall turbidity significantly increased during the seasonal cycle, with higher values from July to September and low values during the rest of the year. However, Fig. 4 also shows a longer lasting local and distinct increase in turbidity close to the bottom in May and June. These high turbidity values during this time are confirmed by both systems, the vertical profiling in situ probe as well as the FerryBox unit.

### 3.2 Species community

Figure 5 (upper panel) shows the sum of individual organisms counted on the images per week for the months October 2013 to November 2014. The average values and stan-

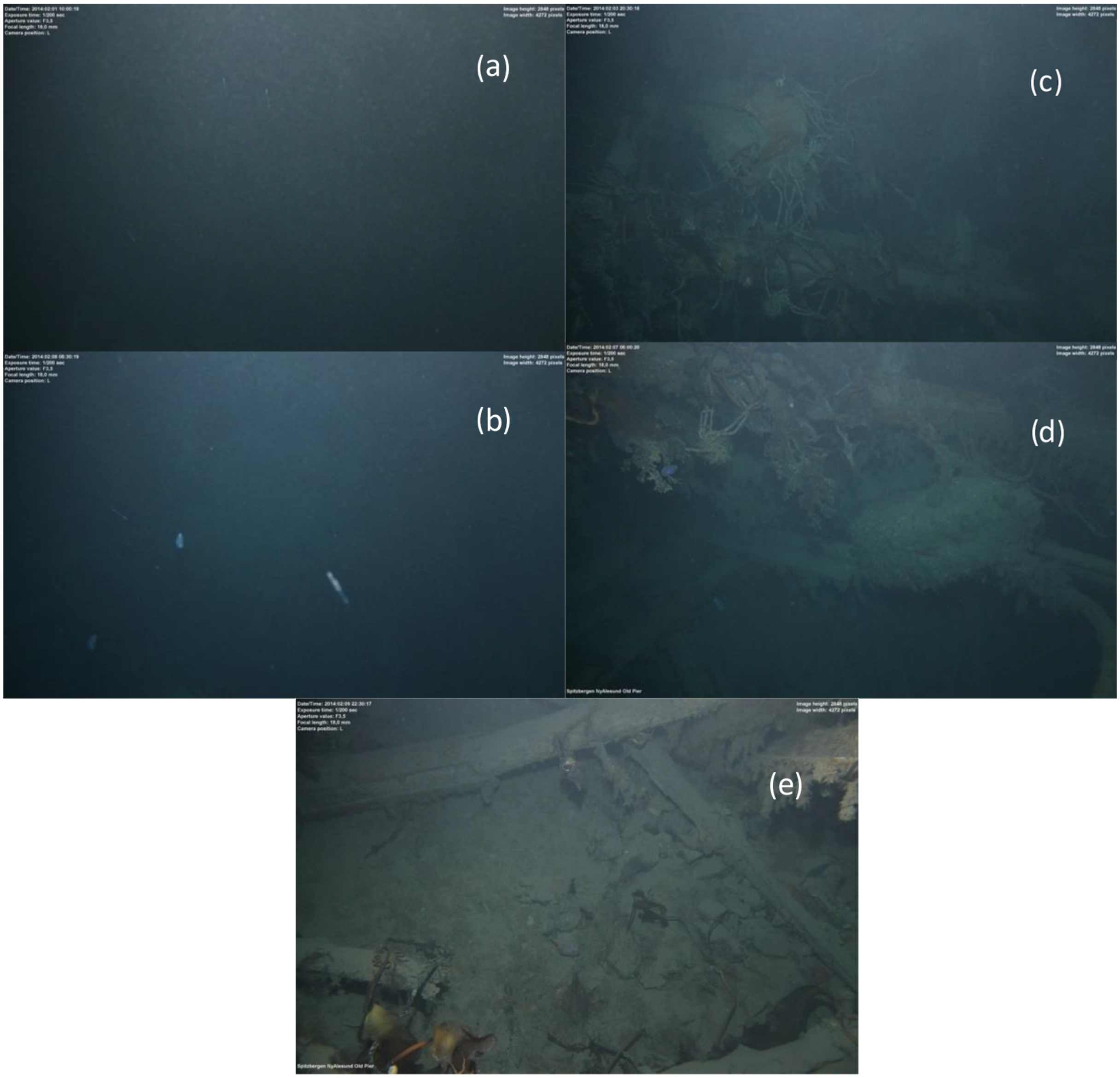


**Figure 3.** **(a, b, c, d, e)** View of the RemOs1 stereo-optical system in the five different depth strata. **(a)** Depth stratum 0–2 m, **(b)** depth stratum 2–4 m, **(c)** depth stratum 4–6 m, **(d)** depth stratum 6–8 m, and depth stratum **(e)** 8–11 m.

dard deviations per month were calculated based on four or five weekly CPUE values depending on how many weeks a month had. The analysis revealed a distinct seasonal cycle with high specimen abundances during the winter months from December to April, lowest values from May to July, and a second smaller peak in August and September. Figure 5 (lower panel) shows the same monthly abundance values but separated by groups of organisms. Ten different groups of organisms were identified over the year, namely, appendicularia, benthic crustacea, birds, chaetognaths, fish, jellyfish, molluscs, pelagic crustaceans, polychaets, and pteropods. From these groups, six occurred in higher abundances, at least during a certain phase of the year (benthic crustacean, fish, jellyfish, appendicularia, chaetognaths, and pteropods).

During the winter–spring peak, benthic crustaceans had the highest share of the total species abundances, followed by jellyfish, pteropods, and fish (Fig. 5, lower panel). In contrast, the summer–autumn peak was almost completely formed by appendicularia and a smaller share of fish.

When analysing the winter–spring phase (December–March) and the summer–autumn phase (August–October) separately and in detail, a strong spatial separation of the winter–spring and summer–autumn communities emerged with respect to the position in the water column (Fig. 6). While the overall share of the winter–spring community was

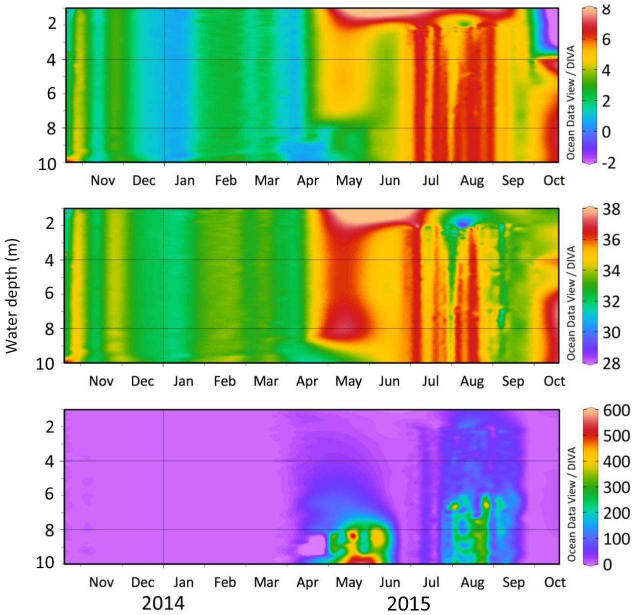


**Figure 4.** Temporal–spatial pattern in water temperature (°C – upper panel), salinity (PSU – central panel), and turbidity (FTU – lower panel) from October 2013 to October 2014 for the depth range 1 to 11 m based on daily vertical CTD profiles from 10 to 1 m and the FerryBox data from 11 m (fixed inlet).

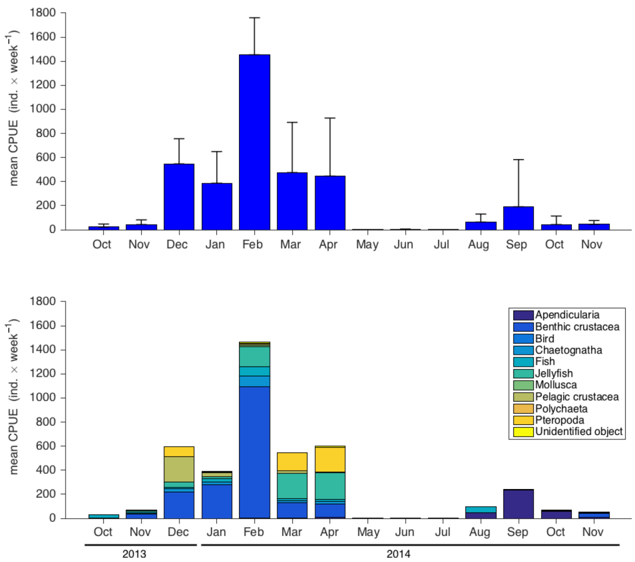


**Figure 5.** Seasonal cycle in total species abundance (upper panel) and species composition (lower panel) pooled per month of the year. For details with respect to "Catch per unit effort", see the text.

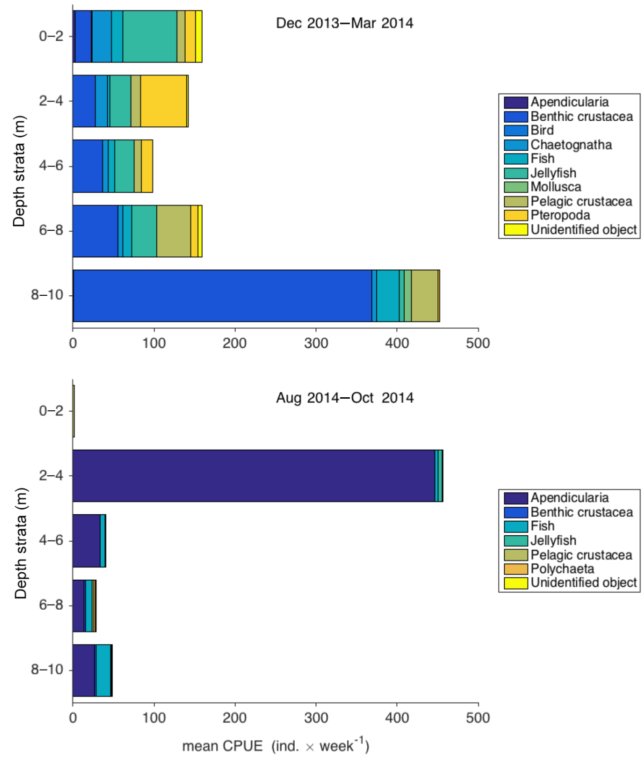


**Figure 6.** Vertical distribution of the different species groups over the water columns. For details with respect to "Catch per unit effort", see the text.

benthic or benthic-associated except for the jellyfish, this benthic-associated community was almost completely missing in the summer and autumn, except for a small share of fish.

Except for appendicularia, all of the other highly abundant species were identified to the species level if possible. Figure 7 shows the species composition of benthic crustaceans (upper panel), fish (middle panel), and jellyfish (lower panel). The analysis revealed that approximately 90 % of the benthic crustaceans identified over the year were made up of a single species, the great spider crab *Hyas araneus* (L.). In addition, hermit crabs (*Paguridae*) were also found occasionally as well as benthic living decapod crustaceans, which most probably belonged to the mysid species *Mysis oculata* (approximately 10 % share). *Hyas araneus*, however, clearly dominated the benthic decapod community, especially in the winter month of February, when a mass invasion of this species was observed in the area.

A similar uniform pattern was observed in fish (Fig. 7 – middle panel); 81 % of the fish on the images were classified as cod of either one of the two species *Gadus morhua* (L.) (50 %) or *Bodeogadus saida* (L.) (31 %). The differentiation of these two species, however, has to be perceived critically because it was based on coloration, which is especially problematic in young specimens. For all the subsequent analyses, we pooled these two fish species and summarized them under "Gadidae".

The most diverse groups over the year were the jellyfish (Fig. 7 – lower panel). A total of nine different species plus one class "unidentified" were found. Integrated over the year, the most dominant jellyfish species (57 %) belonged to the

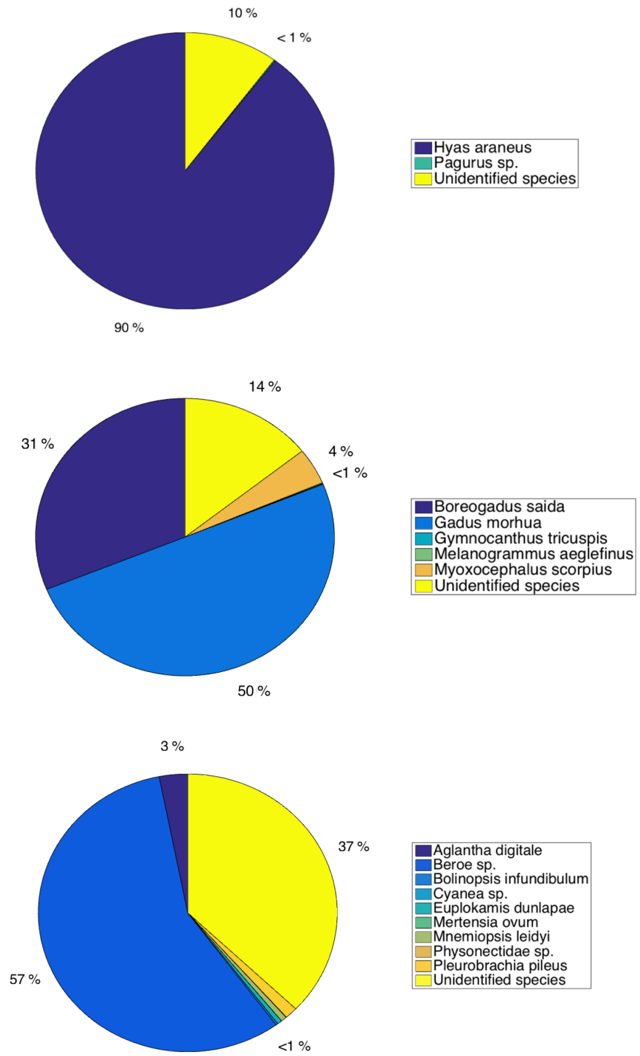


**Figure 7.** Percent distribution of the different species within the different biota groups. For details, see the text.

group *Beroe* sp., followed by *Aglantha digitale* (8 %) and *Pleurobrachia pileus* (5 %). All the other identified species (*Physonectidae* sp., *Mnemiopsis leidyi*, *Mertensia ovum*, *Euplocamis dunlapa*, *Cyanea* sp., *Bolinopsis iunfundibulum*, and *Aglantha digitale*) occurred in abundances with a total share of < 1 %. Unfortunately, 37 % of the jellyfish could not be clearly identified to the species level and, therefore, had to be left unidentified. These species most certainly did not belong to the above-mentioned identified species, which indicates that the jellyfish diversity in this area is even higher.

For the dominant species of the six major biota groups (benthic crustacean, fish, jellyfish, appendicularia, chaetognaths, and pteropods), the body sizes were measured for up to 200 randomly selected specimens per month (if available). In benthic crustaceans, the carapax length from the tip of the rostrum to the end of the telson (in a normal body position) was measured; for fish, the standard length; for jellyfish, the largest body dimension (either longitudinal or transversal); and for chaetognaths and pteropods, the longitudinal body axes were measured. The system allowed for an accuracy in length measurements of approximately 3 % (Wehkamp and Fischer, 2014). Figures 8 to 10 show the size–frequency distributions of the six measured groups per month over the seasonal cycle from October 2013 to November 2014. As the most abundant species during the winter months, November to March, *Hyas araneus* showed an average carapax length of between 50 and 100 mm (Fig. 8 – upper panel) with no temporal trend over the months. However, in November and December 2013, larger animals with a carapax length of up to 180 mm also appeared in the area, which disappeared during the spring and re-appeared again 1 year later in November 2014.

In contrast, in the pooled species group "Gadidae", a clear increase in the average length over the months was observed (Fig. 8 – lower panel). Starting in November 2013, the young-of-the-year (YOY) cohort appeared in the area with an average standard length between 70 and 100 mm. This 2013 cohort stayed in the area until March 2014, when they reached an average length between 100 and 125 mm. After this time, no more cod was observed in the area over the spring and summer until then next YOY cohort appeared for a short time in higher abundances in August 2014 with an average standard length between 40 and 70 mm (mean $\pm$ SD $= 65 \pm 16$ mm). After this time, no more YOY cod could be observed in the shallow area. Instead, larger cod of up to 300 mm were observed sporadically in the shallow waters (Fig. 8 – lower panel, September–October 2014).

All of the other species that occurred in higher abundances in the shallow areas around NyÅlesund belonged to the pelagic community. In jellyfish, the ctenophore *Beroe* sp. made up a major share of the planktonic community and appeared with higher abundances in the winter months, November to April, but with only a few specimens during the summer months. For *Beroe* sp., no temporal size distribution pattern was observed over the months (Fig. 9 – upper panel). The highest abundances were observed in February, with an average size in the longitudinal direction of 45 mm spanning from 10 to 75 mm with average values of $32 \pm 8$ mm (mean $\pm$ SD). Jellyfish occurred with the highest abundances in the shallow-most water layer between 0 and 2 m and in only lower abundance in the water columns between 2 and 8 m. In the deepest water layer close to the bottom, the abundances of *Beroe* sp. were the significantly lowest over the entire water column (LR$\chi^2 = 105$, d$f = 3$, $p < 0.001$).

Another temporally dominant but more agile species compared to the jellyfish were the chaetognaths. This group also occurred with the highest abundances during the winter months (Fig. 9 – lower panel) and were also completely missing during the polar summer. Compared to the jellyfish, however, which were almost equally distributed over the water column except for the deepest stratum, Chaetognath occurred highly stratified in the water columns, with the high-

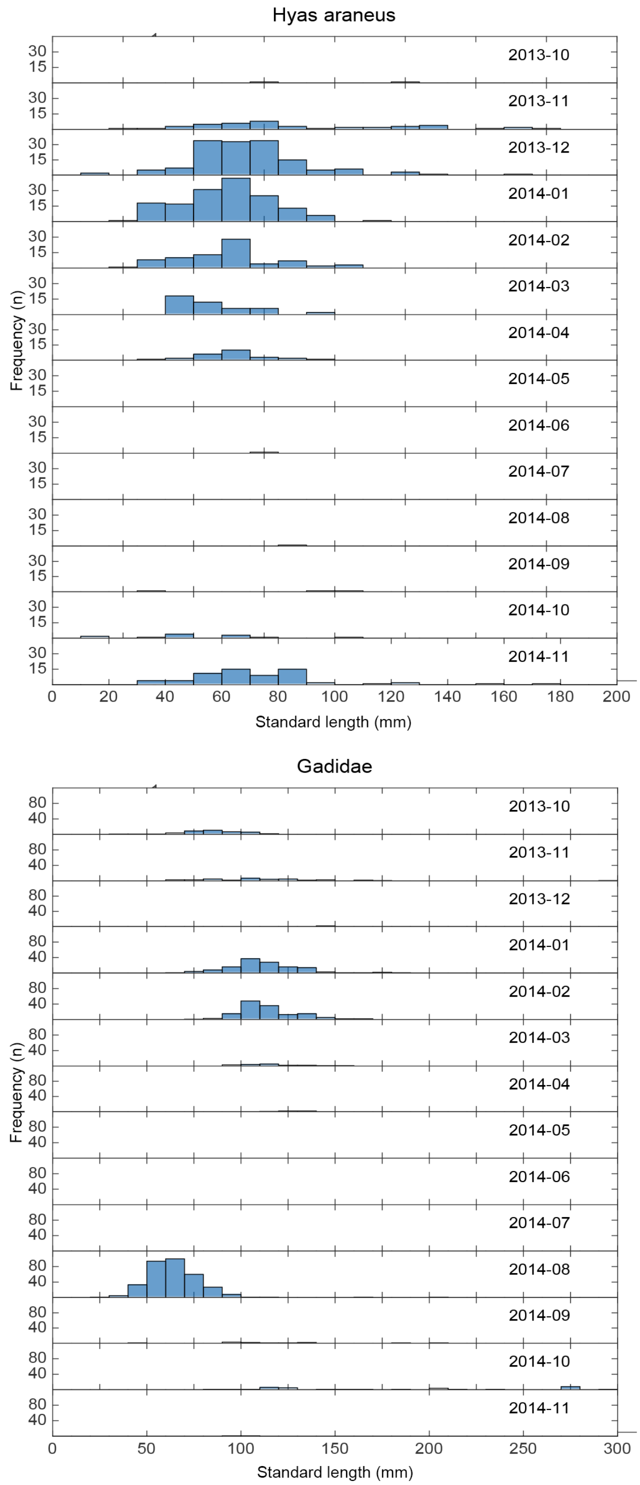


**Figure 8.** Length–frequency distributions of selected species or species groups (see panels) over the seasonal cycle.

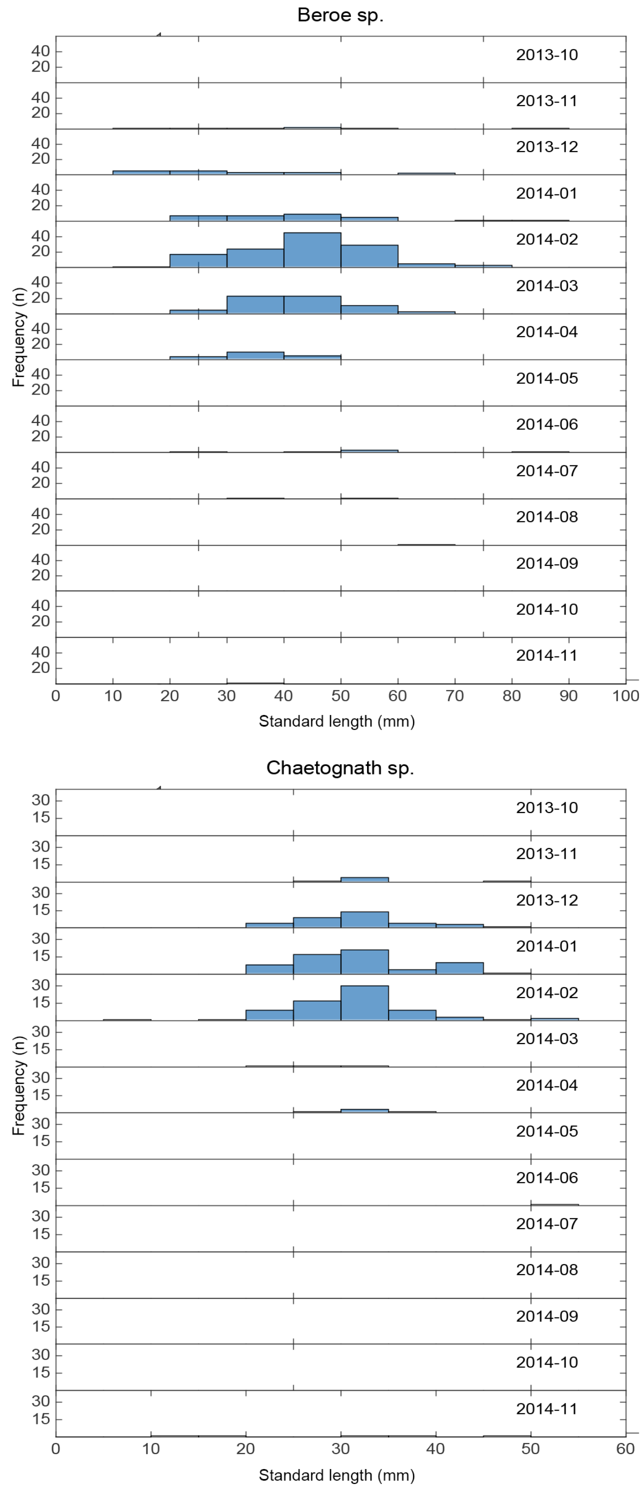


**Figure 9.** Length–frequency distributions of selected species or groups (see panels) over the seasonal cycle.

est abundances in the 2–4 m depth layer; no specimen was found in the surface layer shallow than 2 m, and significantly lower abundances were also found in the deeper water layers (LR$\chi^2 = 490$, d$f = 3$, $p < 0.001$). With lengths between 20 and 50 mm (mean ± SD = 32 ± 8 mm), chaetognaths formed a major part of the pelagic winter community in the shallow areas. A detailed image based on species identification

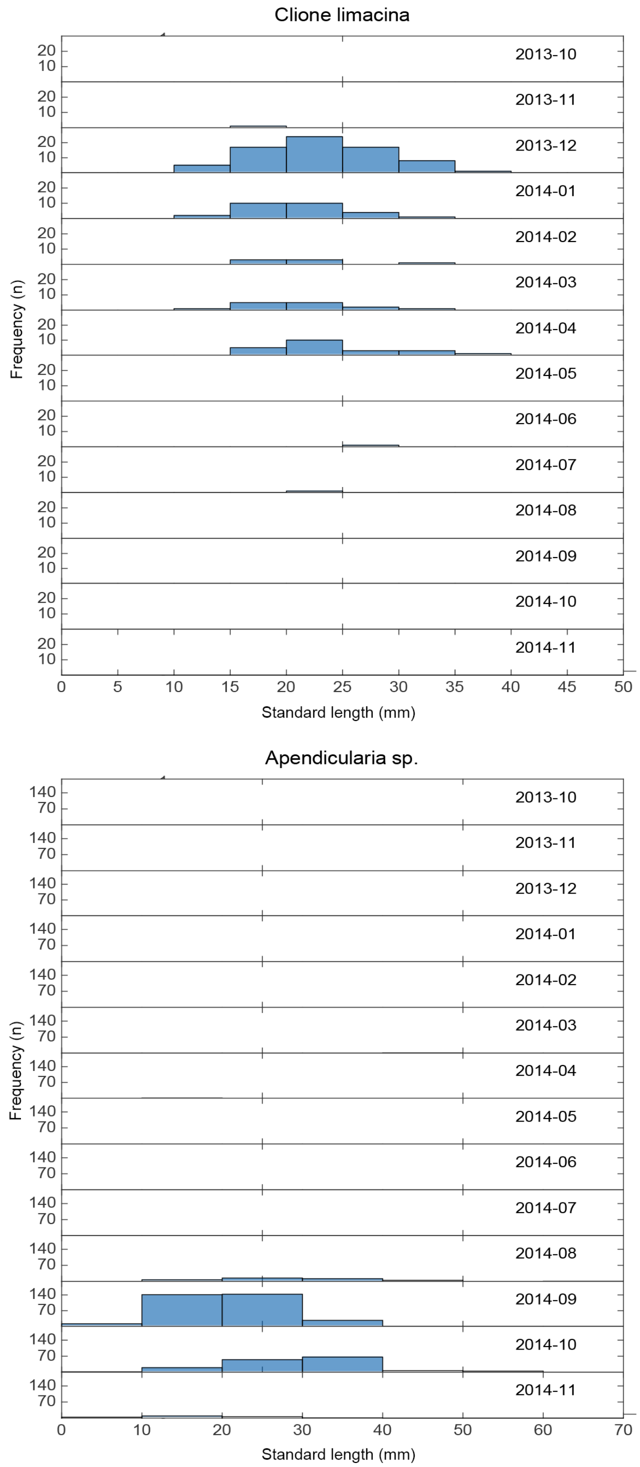


**Figure 10.** Length–frequency distributions of selected species (see panels) over the seasonal cycle.

as well as on the size distribution of the observed chaetognaths suggests that the majority of the observed specimens belong to the species *Parasagitta elegans* (Verrill, 1873).

Temporally, almost synchronized with the chaetognaths, pteropods (Fig. 10 – upper panel) also occurred in the water column and were observed in higher abundances until April. On the images, only *Clione limacina* was observed with body sizes from 10 to 40 mm and a mean size of $23.1 \pm 5.5$ mm (mean $\pm$ SD). Similar to the above-described chaetognaths and jellyfish, *Clione limacine* also occurred highly stratified in the water column, with a peak abundance in the 2–4 m depth layer and significantly lower abundances both in the surface layer and in deeper water strata ($\mathrm{LR}\chi^2 = 143$, $\mathrm{d}f = 4$, $p < 0.001$).

The only species that reached higher abundances not in winter but during the summer months were the appendicularia (Fig. 10 – lower panel). Especially during the months August to October a mass invasion of appendicularia in the upper water columns was observed. As for the other pelagic species, those higher abundances were mainly observed in the 2 to 4 m water layer, while no appendicularia were observed in the uppermost layer close to the surface and significantly lower abundances were observed below 4 m water depth ($\mathrm{LR}\chi^2 = 1039$, $\mathrm{d}f = 3$, $p < 0.001$).

## 4 Discussion

Shallow water areas are well known as important habitats for shallow water fish communities (Reyjol et al., 2005). Due to the often higher structural complexity of shallow coastal waters compared to the deeper parts of the ocean, coastal habitats are often observed as important spawning areas and nursery grounds that form the biological backbone of a diverse and stable benthic and fish community in the associated marine habitats. For the same reason, however, studying higher tropic biota in coastal environments is challenging with regard to a detailed assessment of their temporal and spatial dynamics, especially of mobile communities. The high structural complexity, especially of shallow water hard bottom or reef habitats, often prevents classical ship-supported and space-integrative sampling methods such as trawling or box coring (Brickhill et al., 2005; Fischer et al., 2007a; Wilding et al., 2007). Assessments in these structurally complex environments often require small-scaled and highly specialized "sampling" methodologies often based on optical mapping or imaging technologies operated by divers or ROVs, depending on the water depth. Brickhill et al. (2005), Fischer et al. (2007b), and Wehkamp and Fischer (2014) discussed the potential of such techniques specifically for the assessment of fish–habitat relationships in temperate and boreal habitats such as the southern North Sea. They concluded that in these waters, the comparatively restricted transparency of the water, the lower water temperatures, and the harsher weather conditions often result in only short operation times that result in low numbers of freeze-frame sub-samples taken in most studies, preventing a thorough analysis of the species–habitat relationships due to an insufficiently fine-scale sampling frequency. These limiting factors, especially of diver-operated in situ video technologies, often lead to extremely high variability in organism counts per frame, with too many zero counts, especially when the target organisms are mobile. This leads to a dramatic loss of statistical power in the subsequent data analysis (Brickhill et al., 2005).

These limitations are even more distinct in polar areas where the diver-supported access to the ecosystem is both temporally restricted and extremely expensive. Sampling structurally complex coastal habitats in polar areas is often only possible during a restricted period of time in the polar summer when light is available and the temperatures allow for in situ methods. Therefore, our knowledge of polar shallow water ecosystems and especially their role as nursery and juvenile habitat is extremely restricted. Most of the recent studies (e.g. Hop et al., 2002, 2012; Svendsen et al., 2002) in our addressed study area have been conducted during summer, when the fjord system is accessible by research vessels. Although the summer productive period is of great importance for Arctic ecosystems, several crucial processes (e.g. reproduction) take place during other seasons and especially during the polar winter. During these times, however, almost no information is available in most Arctic fjord systems (Kwasniewski, 2003). Understanding polar ecosystems in the context of global warming and expected or already observed ecosystem changes (Müller et al., 2011; Bartsch et al., 2016) is, however, crucial for thoroughly understanding the ecosystem behaviour in polar areas.

In this study, we do not provide results from experimental work in Kongsfjorden based on discrete studies with a clear short-term ecological hypothesis. In contrast, we provide data from a 1-year long quantitative assessment of hydrographic parameters together with quantitative data on the macrobiota community assessed by a remote controlled cable-connected underwater observatory installed in a typical shallow water habitat in the Kongsfjorden. Using a remote controlled vertical profiling system, we were able to continuously assess temperature, salinity, turbidity, and other hydrographic parameters together with the shallow water macrobiotic community over the entire water column from the benthic over the epi-benthic to the pelagic realm at a high temporal resolution. To our knowledge, this is the first dataset both from Kongsfjorden and from the entire Arctic that reveals such a year-round assessment of the shallow water macrobiotic community together with the quantitative data of the water temperature, salinity, and turbidity and, therefore, allows a deeper insight into the coupling of the seasonal dynamics of the biology and the hydrography compared to pure summer studies. The data reveal a distinct winter community in the fjords' shallow water ecosystem, which by far exceeds the summer community in both abundance and species diversity. Although we have not yet calculated biomass per $m^3$ for the assessed species, our data clearly show that the species abundance and species richness are highest during the polar winter that begins in December when no more light is available under water. During this time, except for the appendicularia, most species, including fish (mainly gadids of the species *Gadus morhua* and *Boerogadus saida*), jellyfish (mainly *Beroe* sp.), chaetognaths (*Parasagitta elegans*), pteropods (*Clione limacina*), and smaller benthic and epi-benthic crustaceans (most possibly *Mysis oculata*, C. Buchholz, personal communication, 2016) invade the shallow water zone and build up highest abundances. During this study, an overall peak abundance was observed in February when the common sea spider *Hyas araneus* clearly dominated the community in numbers and biomass for a short time. Only 1 month later in March, however, *Hyas araneus* almost completely disappeared when fish, jellyfish, and pteropods formed the predominant community with respect to the overall abundances. The "winter" community persisted until April and then almost vanished. The time of the winter community "disappearance" highly corresponds to the increasing availability of light under water. Although sunlight is available at NyÅlesund again already during the middle of March (http://www.awipev.eu/awipev-observatories/current-weather/), the inclination angle of the light is still low until April, so that only a small fraction of the sunlight penetrates the water column (personal observation). However, to really correlate the presence of the "winter community" with the availability of light underwater, discrete measurements of the light intensity and light quality are necessary in the different depth strata to reveal whether light is an ultimate factor in the temporal occurrence of the fjords' shallow water winter community or only a proxy associated with another environmental factory. Our data suggest that especially water temperature may also have a significant influence on the spatio-temporal occurrence of the winter community. Our daily sampled temperature profiles clearly show that water temperature in the shallow water areas of Kongsfjorden can change within short times, even in winter, between $<0$ and up to 4 °C. In particular, the peak abundance in the common sea spider *Hyas araneus* corresponds to the time of higher water temperature during February, and the collapse of the spider abundance occurred when the water temperatures decreased from 4 °C to only approximately 2 °C again. A similar temporal pattern could also be observed in the overall species abundance in April, when a short cold phase in the water temperature occurred. However, these seemingly corresponding changes in the biotic community and the changes in the abiotic environments may also be purely by chance, and we do not know yet whether there are functional relationships between these observations. The permanent installation of the cabled underwater observatory at NyÅlesund allows us to formulate and test such a hypothesis of a persisting shallow water "winter community" in the fjord system as well as the hypothesized controlling or at least affecting abiotic factors.

Our data additionally reveal another distinct community during the summer months when the temperatures increased up to 8 °C in the fjord. Then, appendicularia occurred in higher abundances for a restricted time, i.e. from August to

October, in the shallow water with a peak in abundances in September. In contrast to the winter community, which was mainly benthic or at least benthos-associated, this summer community was almost completely dominated by a single appendicularia species, most certainly belonging to the genus *Oikopleura* sp. (Dahms et al., 2015).

Besides appendicularia, juvenile cod fish were also found in September in the deeper littoral water layers closely associated with benthic habitats. The detailed length–frequency analysis of this cohort reveals that these fish were the YOY offspring of the same year (YOY cohort 2014) with an average standard length of $65 \pm 16$ mm. The data also reveal that these fish seem to stay in the littoral zone (even though the overall abundances strongly decreased over winter) and continuously grow and reach an average standard length of 100 to 125 mm in February–March at age class 1, when they seem to quantitatively leave the shallow water habitats. This outcome indicates a complex migration pattern of YOY cod in this area with a short winter phase in the littoral zone of the fjord system of Spitzbergen and a later migration towards deeper or offshore habitats as adults. Such temporally restricted shallow water phases have been observed already for several other cod species, especially during their juvenile phase (Pihl, 1982). This has been regarded as a juvenile behaviour to prevent predation by older conspecifics in the deeper adult habitats (Ruiz et al., 1993) as well as an improvement in the foraging efficiency of the juveniles during their non-piscivore microzoobenthic benthic feeding phase (Pihl, 1982).

In contrast to the clearly visible seasonal growth pattern in the cod species, no distinct growth could be observed in any of the other species, even in the highly abundant common sea spider, which showed a persisting size range between approximately 50 and 80 mm during all the winter months, except for the month of November in both years, when larger animals between 120 and 180 m were observed in the area, even though in much lower abundances.

As clearly stated before, this study does not provide a singular hypothesis-driven question; instead, it focuses on a basic assessment of the temporal (and with respect to the water column also spatial) pattern in the macrobiota community distribution and possible hydrographic factors that influence the shallow water biota. The results of this study are by far incomplete and only represent a 1-year study at a specific site in the Kongsfjorden ecosystem, which may or may not be representative of the shallow water community of this area. However, the study presents a continuous year-round dataset at a temporal resolution of 1 week, which is, to our knowledge, not available in any other fjord system, and especially not in the Arctic environment, where winter data are missing at almost every level. However, even though the data provide a unique year-round insight into a polar shallow water fjord community, we can assume that the technology used here has a certain bias with respect to species selectivity. Therefore, these data have to be taken with care. For instance, comparing our stereo-optically assessed fish data with data from classical sampling devices in Kongsfjord (Brand and Fischer, 2016; Hop et al., 2002; Renaud et al., 2011) or even with sporadic diver observations (Brand and Fischer, 2016; Hop et al., 2002), it becomes clear that our optical sensors are also species selective. Brand and Fischer (2016) for example reported for the summer month a distinct occurrence of the benthic sculpin *Myoxocephalus scorpius*, a typical temperate and highly camouflaged benthic fish species in fyke-net catches. Although we detected *Myoxocephalus scorpius* during summer also on the stereoscopic images, the overall abundance remained quite low. Unfortunately, the fyke-net catches of Brand and Fischer (2016), as with most other available marine studies of the fjord, are only available for the polar summer months, when our stereo-optical data revealed the lowest overall biota abundance at all. However, taking into account that fyke nets are highly time integrative and catch fish only directly at the bottom, the fyke-net and optical data may be complementary rather than contradictory. In the study of Brand and Fischer (2016), fyke nets with a mesh size of 12 mm and a steering net of 18 mm were used. This type of net gear is highly selective for strictly benthic fish species with a high potential of entanglement, such as sculpins. In contrast, a stereo-optical method is most probably less selective for benthic highly camouflaged fish species and may significantly underestimate fish with these characteristics.

Instead, our overall image assessment procedure was thoroughly performed by two different persons and showed similar results with respect to the quantitative detection of even small benthic mysids. Therefore, we assume that we would have also detected sculpins if available in higher abundances and thus conclude that the quantitative relation of the average abundance between the major fish species found on the images might be more precise, as found in the fyke net catches. This outcome seems to be supported also by the available diver observations in that area, at least during summer. Hop et al. (2002) and Renaud et al. (2011) both reported the cod species *Gadus morhua* as one of the most abundant species in the area, which would be in accordance with our findings. Nevertheless, the comparison of these two methods shows that there is a large uncertainty with respect to the methodological approach that should be used in future studies. Furthermore, our in situ optical methods allow for a low-invasive abundance estimate, for a precise length–frequency analysis of the mapped fish, and also for a continuous year-round assessment of the community. However, it does not allow for further investigations such as stomach content analysis and precise aging based on scale or otolith analysis. If we manage to combine such continuous hydrographic and community observations using cable-connected observatories with classical ground truthing fishing or sampling methods, we may reduce our scientific fishing effort to a limited number of specimens, which are needed for specific detailed analysis such as stomach content and otolith-based aging, and obtain the required more invasive stock abundance and growth data via non-invasive optical methods. These approaches may finally enable the reduction of our fishing effort without losing the required data density and therefore contribute to the increasing scientific demand of a resource conservative science also in fish and community ecology, especially in ecologically sensitive areas such as the polar fjords or marine protected areas.

## Next steps and needs

In addition to the ecological and hydrographical results from the Kongsfjorden ecosystem presented here, the study demonstrates the advantages of permanently operated cabled observatory technology – especially when combined with other research methods in a multidisciplinary approach integrating biology with the understanding of the physical environment. Cabled observatories with continuous power supply and network access allow the use of state of the art IT technology and smart-monitoring approaches under water. These are often not applicable in mooring-based sensor technology because no feedback to the operator is possible and therefore the researcher himself cannot react to specific environmental situations during the measuring process. Furthermore, complex sensor systems like profiling videos or stereo-imaging systems often cannot be operated unsupervised for longer times because the controlling software is either too complex, the power consumption is too high, or the required test and development phases for unsupervised operation of such systems are too long and therefore too expensive. Cabled observatories with permanent access, power supply, and systems control allow even complex sensor systems to be operated for longer periods because in case of failures, the system can give an alert to an operator elsewhere to request remote control and if necessary sensor reset. Based on our experiences with the cabled observatory in Svalbard, we assume that such underwater research facilities, if operated within an international and well-focused research strategy, may significantly promote our knowledge, especially in remote and sensitive areas like the polar regions.

*Data availability.* Supplementary data are available at doi:10.1594/PANGAEA.874141 (Fischer, 2017).

*Competing interests.* The authors declare that they have no conflict of interest.

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# Operating Cabled Underwater Observatories in Rough Shelf-Sea Environments: A Technological Challenge

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Cabled coastal observatories are often seen as future-oriented marine technology that enables science to conduct observational and experimental studies under water year-round, independent of physical accessibility to the target area. Additionally, the availability of (unrestricted) electricity and an Internet connection under water allows the operation of complex experimental setups and sensor systems for longer periods of time, thus creating a kind of laboratory beneath the water. After successful operation for several decades in the terrestrial and atmospheric research field, remote controlled observatory technology finally also enables marine scientists to take advantage of the rapidly developing communication technology. The continuous operation of two cabled observatories in the southern North Sea and off the Svalbard coast since 2012 shows that even highly complex sensor systems, such as stereo-optical cameras, video plankton recorders or systems for measuring the marine carbonate system, can be successfully operated remotely year-round facilitating continuous scientific access to areas that are difficult to reach, such as the polar seas or the North Sea. Experience also shows, however, that the challenges of operating a cabled coastal observatory go far beyond the provision of electricity and network connection under water. In this manuscript, the essential developmental stages of the "COSYNA Shallow Water Underwater Node" system are presented, and the difficulties and solutions that have arisen in the course of operation since 2012 are addressed with regard to technical, organizational and scientific aspects.

**Keywords: coastal cabled observatories, remote sensor operation, data quality, Arctic coasts, North Sea**

## INTRODUCTION

The coastal zone accounts for 14–30% of the primary production in the ocean, 80% of organic matter burial, 90% of sedimentary mineralization, 75–90% of the oceanic sink of suspended river load, and approximately 50% of the deposition of calcium carbonate (Gattuso, 1998). Hydrological conditions in coastal waters change more rapidly compared to the adjacent ocean and may also form the nuclei for seasonal biological patterns, such as spring blooms and subsequent biological production (Harding and Perry, 1997; Cloern and Jassby, 2009). Shallow waters often provide important spawning areas and nursery habitats for marine biota and serve as foraging areas for many fish stocks and mammals (El-Hamad et al., 2009).

Local hydrography in shallow waters is often strongly affected by the specific littoral morphometry and the sediment type (Shalovenkov, 2000), which subsequently affects the biotic community across all trophic levels. Additionally, environmental conditions in coastal waters are significantly affected by atmospheric conditions due to local and regional wind patterns (Savijarvi, 2004) causing complex wave and current patterns as well as temporal and spatial patterns of physical, bio-geochemical and biological parameters (Comin et al., 2004). These often occur over distances and times ranging from millimeters to hundreds of kilometers and from seconds to years.

The study of shallow water coastal environments on a functional level is challenging due to the complexity of the systems themselves. In particular, temperate and polar coastal areas, which are increasingly perceived as vulnerable areas of high interest in the context of climate change, are often characterized by harsh wind conditions, low temperatures or even ice conditions. The North Sea, for example, has average wind speeds of 7–8 m $s^{-1}$, with wind peaks above 6 bft on more than 300 days a year (Ganske et al., 2005).

Such harsh weather conditions significantly reduce the days available for field measurements and oceanographic or biological *in situ* assessments. This restriction of available observation periods based on conventional ship based sampling techniques poses considerable risk of either the inability of resolving existing patterns and relationships in coastal systems or, even worse, of misinterpreting those results. Fixed mooring systems are highly valuable in providing continuous time series data in coastal areas as well (Hop et al., 2019) but require regular ship time for recovery and suffer from the disadvantage that technical problems are only discovered after the deployment phase. Thus, there is a risk of partial or complete data loss due to system failures or even complete mooring loss. Furthermore, mooring systems normally have limited power resources that often restricting sensor types and operation.

Examples of misinterpretation resulting from an insufficient sampling frequency in ecological studies are given in Pearcy et al. (1989) based on the Nyquist theorem (Nyquist, 1928). This risk is even greater in coastal areas than in the open ocean. While excellent models and thorough predictive research capacities are available for blue water systems, the capacities for calculating and predicting functional relationships between oceanographic dynamics and the associated marine biota are rather limited in shallow coastal areas (Androsov et al., 2019). Different "ecosystems" (hard bottom areas, seagrass meadows, and so forth) are often located in the same area but nevertheless act as separate "functional units." Understanding coastal processes and how these ecosystems function therefore often requires an assessment of numerous interacting environmental variables covering all process relevant spatio-temporal time scales.

The technology of cabled coastal underwater observatories has been significantly improved in recent decades (National Research Council, 2003; Hart and Martinez, 2006; Witze, 2013; Favali et al., 2015). Underwater observatories are often designed to provide ground truth data from static reference points over time (Badeck et al., 2004). In contrast to ship based surveys or other mobile observatory platforms such as AUV's and autonomous gliders, cabled underwater observatories, however, cannot provide spatial coverage of a certain area. Together with mobile systems such as Argo floats that are specifically designed to cover extended surface areas but with limited temporal resolution (Levy et al., 2018), cabled underwater observatories can complement an integrated monitoring strategy of a marine region as a Long-Term Ecosystem Research (LTER) reference station and *in situ* lab facility.

Most cabled observatories such as MARS (Monterey Accelerated Research System)[1], VENUS (Victoria Experimental Network Under the Sea) (Dewey et al., 2007), NEPTUNE (North-East Pacific Time-series Undersea Networked Experiments) (Best et al., 2007), ALOHA (Howe et al., 2011; Favali et al., 2015), and LoVe (Godø et al., 2014) have been installed in greater water depth (Best et al., 2016). However, some installations were specially developed for shallow water applications to withstand near-surface conditions and strong hydrodynamic forces. Examples are the cabled observatory "SmartBay" in Galway Bay, Ireland, at 22 m water depth (Cullen et al., 2015)[2], the EMSO-Molène cabled observatory[3] in the Atlantic at 18 m water depth, the EMSO Mediterranean Sea observatories at 20–30 m depth[4], the OBSEA Observatory at 20 m water depth (Del-Rio et al., 2020)[5], and the LEO-15 observatory on the East coast of New Jersey, United States (Forrester et al., 1997).

Although the advantages of permanent underwater observatories are obvious, their operation cannot always be maintained in the long term. For example, the WHOI's PLUTO observatory off Panama was established in 2006, but was partially closed down in 2008. Unfortunately, it is almost impossible to obtain more detailed information about the reasons for the closure of such infrastructures, as negative experiences with new technologies or even the complete failure of systems or projects are rarely reported beyond personal communication. However, a thorough discussion of precisely these failures, pitfalls and drawbacks is particularly important in the case of emerging technologies that are not merely a "flash in the pan," but seem to be developing as new tools that enable major advances in science.

[1] https://www.mbari.org/at-sea/cabled-observatory/
[2] http://www.smartbay.ie
[3] http://www.emso-fr.org/fr/EMSO-Molene/Infrastructure
[4] http://emso.eu/observatories-node/ligurian-sea/
[5] https://obsea.es/

New technologies must also provide a truly sustainable and long-term benefit for science. It is therefore necessary to consider the effort and the risks involved in operating cabled underwater observatories for science (Buck et al., 2019).

In this manuscript, we describe the experiences gained from 7 years of operating of two cabled underwater observatories in the North Sea and Arctic. We present the basic design features of the node systems used, the data handling procedures as well as the design and procedural changes since the systems were commissioned. In the "Materials and Methods" section, we describe the observation sites as well as the technical specifications of the underwater systems developed within the framework of COSYNA (Coastal Observing System for Northern and Arctic Seas) (Baschek et al., 2017) and the two Helmholtz Association projects ACROSS and MOSES (Modular Observation Solutions for Earth Systems). The "Results" section describes the experience with the setups since 2012. Using two scientific examples, the potential of cabled observatory technology, especially for coastal research, is presented together with the problems that have occurred on a hardware, software and conceptual level. In the "Discussion" section, the system optimizations carried out during operation to overcome those hurdles as well as those planned for the next node generation are described. The advantages, disadvantages and risks of operating cabled observatories in coastal research are also discussed.

# MATERIALS AND METHODS

## Study Sites

The two COSYNA Underwater Node Systems are operated at two sites that differ significantly in terms of climatic and hydrodynamic conditions, but exhibit a remarkable similarity in terms of biota composition with respect to the fish and macroinvertebrate species present in both areas (Brand and Fischer, 2016; Wiencke and Hop, 2016). The "COSYNA-Helgoland" observatory (**Figure 1**) is located at 54°11′32.3″ N/7°52′42.2″ E (WGS84), about 500 m north of the island of Helgoland, at a depth of 9.7 m (± 0.9 m SD tidal amplitude), at the AWI (Alfred Wegener Institute) underwater experimental field "Margate" (**Figure 1**)[6] close to the Helgoland roads time series (Wiltshire et al., 2009). The area is particularly characterized by strong hydrodynamic forces with average current velocities of 0.5 m $s^{-1}$ (Fischer et al., 2019a) and dominant M2 and S2 tides, allowing characterization of this area as a hydrodynamically complex ecosystem. Minimum monthly-averaged water temperatures of about 3°C are reached in February and maximum values of about 18°C in August (Wiltshire and Manly, 2004; Fischer et al., 2018a). Another local feature affecting shallow water habitats and permanently installed measurement technology are wind speeds up to 147 km/h (Climate Data Center [CDC], 2019). These strong storms occur primarily in autumn and spring and can lead to "groundswell," where the wave height is greater than the water depth so that the benthic community and technical installations on the seafloor are significantly exposed to strong hydrodynamic forces.

The COSYNA-AWIPEV observatory is located in the Kongsfjorden Arctic fjord system at 78°55′50.37″ N/11°55′12.10″ E (WGS84), at 10 m water depth (± 0.7 m SD tidal amplitude) on the west coast of Spitsbergen (Fischer et al., 2017; **Figure 1**). The site is comparatively sheltered in the inner part of the Kongsfjord, with average tidal currents of 0.1 m $s^{-1}$ (Fischer et al., 2019b). The major threat for any fixed scientific installation in this polar area are freely drifting small and medium sized ice bergs. Until 2006, the fjord was regularly covered by sea ice in winter (Gerland and Renner, 2017). From then on, regular winter ice cover has become rare (Cottier et al., 2007) and closed winter ice cover has no longer been observed since 2009. This is mainly attributed to the increasing warming of the fjord system due to the influence of climate change (Kortsch et al., 2012). This leads to the situation that today, icebergs which are frequently calving from the glaciers inside the fjord are no longer locked by sea-ice but are freely floating in the fjord system reaching the shallow water areas, thus posing a considerable threat to permanently installed measurement systems. With significantly fluctuating minimum winter water temperatures between -1.6 and 0.8°C in February and March, and maximum average water temperatures of more than 6°C in August (Fischer et al., 2018b,c,d,e,f,g,h), there is an on-going discussion as to whether the fjord has exceeded a "tipping point" and will remain permanently ice-free in the future.

A further challenge in terms of continuous operation and regular maintenance of the COSYNA-AWIPEV Underwater Node System is the polar night with a dark phase from November to February and air temperatures below -30°C. This circumstance limits extensive maintenance work under water to the summer months and makes winter operations in the event of system failures a challenge for the participating scientific staff, the scientific divers and the equipment.

## Observatory Layout: Configuration Requirements and Implementation

Both node systems have been developed and operated since 2010 as part of the COSYNA framework (Baschek et al., 2017). They were expanded since then as part of the ACROSS and MOSES projects. The main objective was to develop a cabled underwater node system for shallow water areas between water depths of 5 and 300 m. The system was to withstand the challenging environmental conditions in the North Sea and the Arctic with the requirement that it be continuously operated year round and fully controlled remotely. The weight of a single component should not exceed 1 t, so that it could be deployed with smaller coastal vessels using a standard ship crane. A further requirement was that all single components can be mounted or dismounted individually underwater by divers. An additional major requirement was that scientists must be able to operate a sensor at the node system without familiarity with the back-end software technology. Based on these requirements, two industrial (SME) partners were selected to

[6]https://www.awi.de/en/science/special-groups/scientific-diving/margate.html

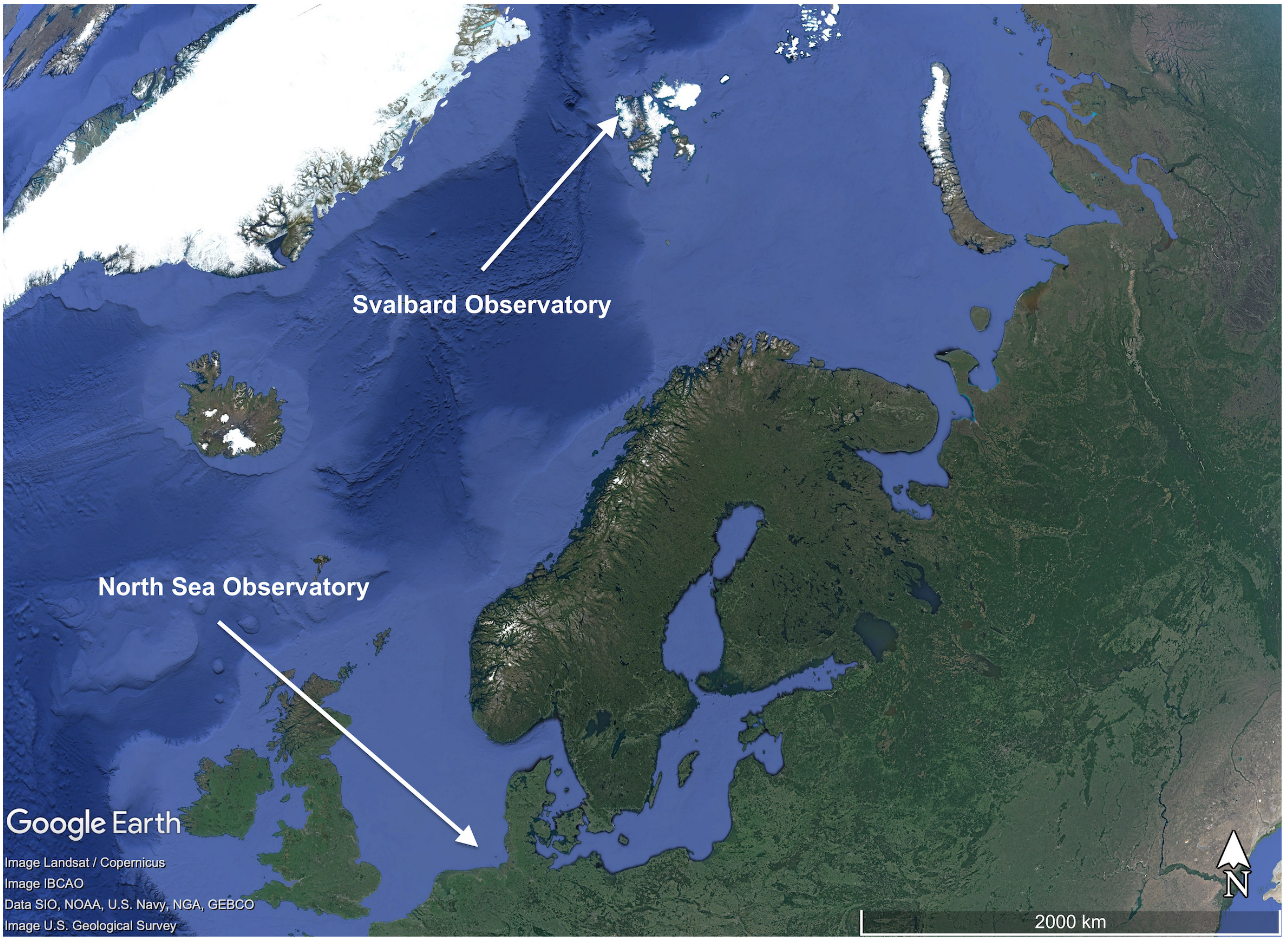


**FIGURE 1 |** Location of the two COSYNA observatories in the southern North Sea (COSYNA-Helgoland) and the Arctic Ocean in the Svalbard archipelago (COSYNA-AWIPEV).

develop a concept for the node hardware and software in a consortium with the participating institutes and to construct a corresponding prototype.

In **Figure 2**, a sketch of the COSYNA underwater node deployment configuration is shown. The system consists of a land station (1), the submarine cable (2), the actual underwater node (4) and a connected lander system (6), which serves as a basic sensor carrier. The system's operational range – that is, the maximum cable length for connecting the land station and a first underwater node system – was defined at 10 km. This maximum distance was constrained by the requirements to reach different areas of sediment types around the designated test area of the first node system, the island of Helgoland in the southern North Sea. The concept, however, includes a range extension of up to 30 km by daisy chaining two further node installations.

The land station (**Figure 2(1)**) comprises one ARGOS 1200 power supply unit for each node[7]. Each unit delivers up to 1000 V and 1.2 A, thus providing an input power of up to 1200 W per node to the sea cable. The supply system is based on direct current (DC), which has a lower voltage loss on longer distances compared to alternating current (AC). Depending on the distance from the land station to the node system, the voltage delivered by the land station can be reduced to prevent the transfer of unnecessarily high voltages via the underwater cable and plugs (see also results section "Underwater Pluggable Cables and Connectors"). This is done, for example at the Svalbard node system, where the distance between the land station and the underwater node is only 200 m. There, the input voltage could be reduced to 250 V without any power limitations for the sensor operation.

As IT infrastructure, a VMware ESXi hypervisor, Version 5.5 was hosted one a local server with local storage (Dell PowerEdge R710, 12C, 96 GB RAM, 2,4TB Raid6 Storage). This early setup was replaced in 2016 by a redundant server infrastructure both at the Helgoland and the Arctic node system. It consists of two VMware ESXi hosts, Version 6.5 (Dell Power Edge R730, 8C, 192GB RAM) and two iSCSI storage units with each 5TB Raid6 storage. Full seamless fault tolerance is

[7]https://tet.industrialpine.de/material/datasheet/de/ARGOS_1200_DATA_de.pdf

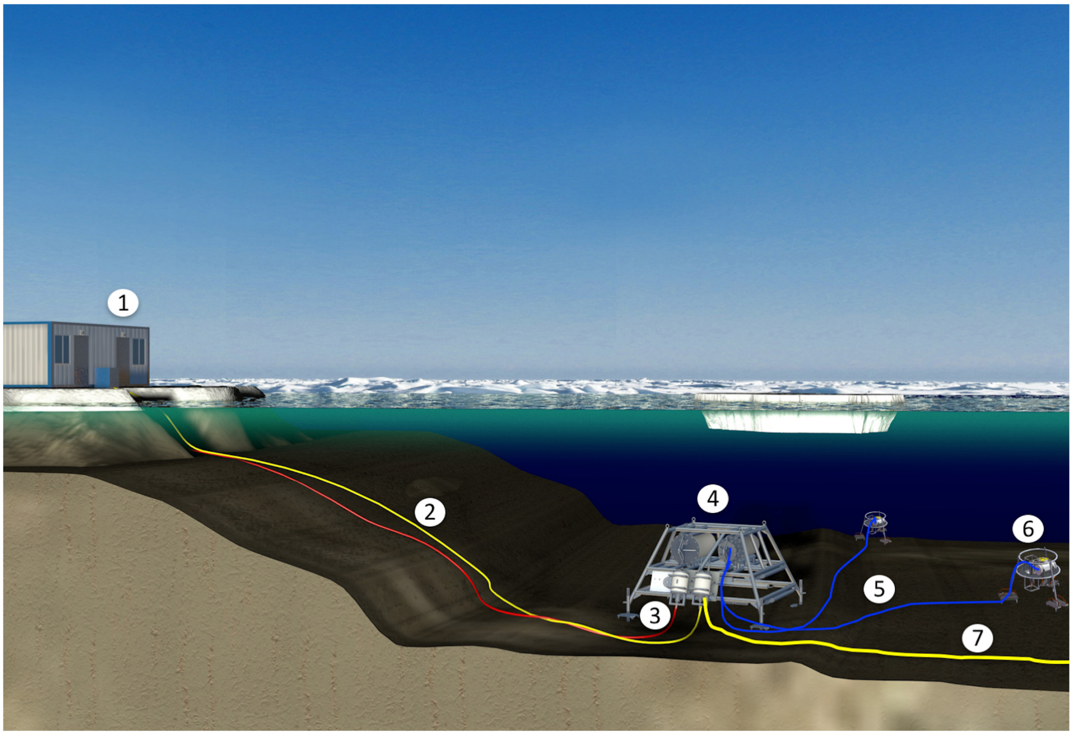


FIGURE 2 | Basic deployment concept of the COSYNA Underwater Node System: **(1)** land station, **(2)** submarine cable (1000V), **(3)** breakout box, **(4)** underwater node, **(5)** Power (48V)/TCP-IP hybrid cable, **(6)** sensor carrier (lander), and **(7)** submarine cable (1000V) to daisy chained second node. The maximum distance from land station to the first node respectively among the daisy chained second and third nodes is 10 km. Maximum water depth is 300 m. See text for a detailed description of the single components.

given this way for the failure of one storage unit or one ESXi host at either site.

The 6-core (6 × 2.5 mm) sea cable (**Figures 2(2)**, **3**) is used together with four single-mode fiber optic lines for data transmission. The cable is reinforced with an aramid sheath and has a copper foil shield with a double wire. The coating is made of polyurethane and the outer diameter is 22 mm. The cable is approved for an operating voltage of 1000 V DC, with a test voltage of 4 kV AC. The cable resistance is 3.3 Ohm/km; the weight is 705 kg $km^{-1}$; and the maximum tension load is 2000 N. The calculated voltage drop is 6.9 V $km^{-1}$ at 1000 V and 1200 W (maximum power transmission). This results in a maximum power drop of up to 207 V at a maximum distance of 30 km from the third node to the land station in the full expansion stage with daisy-chained node systems. For data connectivity, one pair of the fiber optic lines is used to establish a 1000-FX network link to the land station. A further capacity extension by upgrading the fiber optic transceivers is possible.

The submarine cable is connected to the underwater node system at the "breakout box" (**Figure 4**). In this cable termination, the optical fiber connection of the underwater cable is converted into a copper-based data transmission. The incoming 1000 V are converted to 48 V to supply the electronic components in the breakout box. This large-step power conversion was achieved by a special power supply unit from SYKO Type BLG.M. The IT components used are active components with their own IP addresses to communicate with the components and check their function in the event that either no node is connected or an undefined error occurs in the system. The breakout box is constructed of polyethylene (PE) and is approximately weight-neutral in water. An IP-based water intrusion detector is mounted to monitor it.

**Figure 5** shows the complete COSYNA Underwater Node System during operation. The breakout box is connected to the node by two wet-mateable cable connectors: one connecting

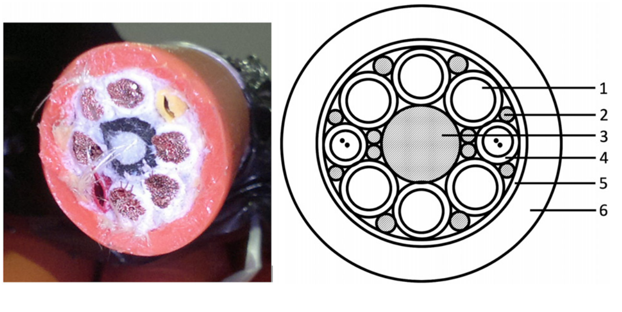


FIGURE 3 | Sea cable (diameter 22 mm) used to connect the land station with the node system (1 = insulated cores, 6.0 $mm^2$; 2 = filler; 3 = GRP fabric; 4 = fiberglass cable single mode; 5 = taping; 6 = outer sheath. For additional details see text). Photo: P. Fischer.

the power (1000 V – type DC), and the other connecting Ethernet (1000BASE-T).

Communication and data transfer from and to the underwater node systems is performed by standard internet protocol TCP/IP. From 2012 to 2015 at the Helgoland systems and until 2016 at the Svalbard system, the land stations of the nodes were connected via IP radio relays over a distance of up to 60 km to the respective national IP network. Even though these connections were sometimes identified as possible bottlenecks for remote node operation, especially in the Arctic, they never restricted the required bandwidth. From 2015 onward at Helgoland (Germany) and from 2016 onward at Svalbard, a cable-based fiber optic connection via the respective national sea cable infrastructure is available for data transfer.

The internal power of the underwater node system and connected sensors is set to 48 V to allow for safe underwater operation by divers when the 1000 V land power supply must be shut down. To keep the system running during intentional (or unintentional) power cuts, an additional battery buffer is installed (**Figure 5(1)**), keeping the system alive for at least 2 h so that divers can safely approach the system under fully operational conditions.

For attaching sensors (or even complex sensor units) to the node, ten underwater mateable connectors are available per underwater node, each providing 100BASE-T ethernet link (max. 1000BASE-T ethernet) for data transfer, and a 48 VDC power supply with a maximum of 200 W per connector (**Figure 6**, right image). IP-based Ethernet connections are used as standard transport protocol. Non-Ethernet sensors can be connected via specific "connector boxes" containing hardware to adapt sensors to serial or USB interfaces (**Figure 6**). The connector boxes have been specially designed and developed for the COSYNA node system. They can be individually configured depending on the sensors to be connected. The boxes are made of POM material (polyoxymethylene) which is commonly used in marine engineering and they are standardized in size. The standard connector box has a diameter of 27 cm and a length of 38 cm and can be equipped with various underwater pluggable connectors in the lid. Connector boxes only differ in the length of the body and not in the lid, so that the lid with the connectors and the wiring can be used with a larger body if additional space is needed. A COSYNA standard "connector box" can take up to six sensors and provides 12, 24, and 48 V as well as standard RS232 and RS485 communication at each of the six ports. For other sensors, a custom configured "connector box" is provided based on the standard input of 48 V and a 100 Mbit Ethernet connection from the node. For all sensor communication via the node to the user, industrial Ethernet-serial/USB converters (Advantech$^{TM}$ EKI 1524 or WUT$^{TM}$ Com-Server Serial/USB) are used.

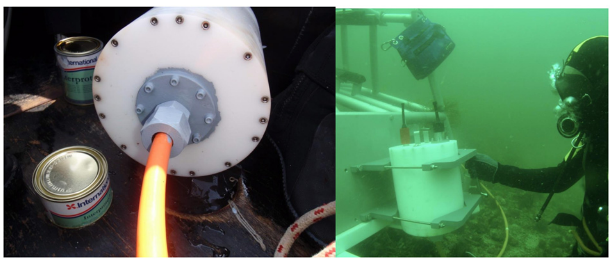

**FIGURE 4 |** Left: Sea cable feedthrough coated with corrosion protection. Right: Breakout box mounted on the frame of the Helgoland underwater node system. Photos: P. Fischer.

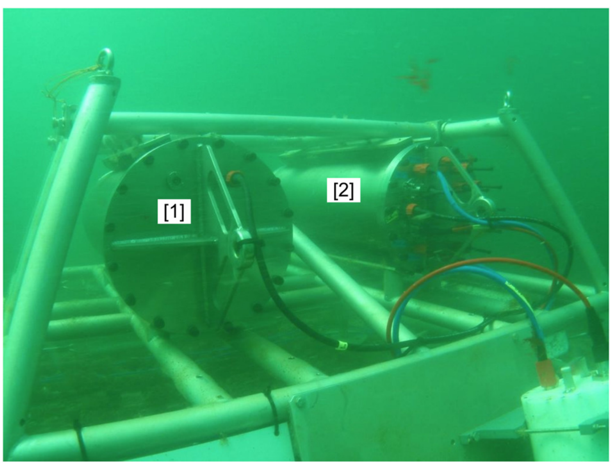


**FIGURE 5 |** Fully operational node system off Helgoland. The left tube (1) houses a battery pack that provides power for 2–6 h, depending on the power consumption of the sensors. The right tube (2) is the node system which is connected to the breakout box by the red 1000 V power line and the blue Ethernet line. On the front panel, ten sensor ports are available, each providing up to 200 W and an Ethernet connection. All cables between the different node components are wet-mateable by divers, except for the sea cable that enters the breakout box from below via a permanent cable feedthrough (see **Figure 4**). Photo: P. Fischer.

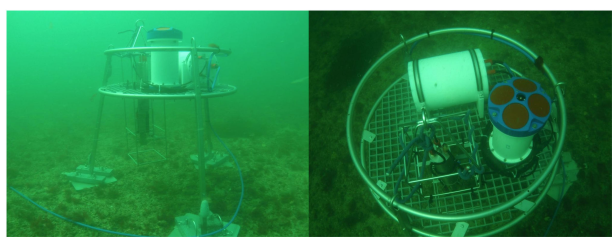

**FIGURE 6 |** Sensor lander with a "connector box" (white PE tube in the right image). The "connector box" is connected to the "breakout box." From there, a serial, USB or any other sensor is connected via the respective communication protocol. The respective Ethernet interface for a sensor is installed in the "breakout box" and connected to the sensor via an underwater mateable subcon plug. The photographs show a COSYNA "standard lander" that is equipped with a Sea&Sun CTD, a Teledyne ADCP and a SeaBird SBE38 temperature sensor. The standard "breakout box" can take up to six sensors and provides 12, 24, and 48 V as well as standard RS232 and RS485 communication at each of the six ports. For other sensors, a custom configured "breakout box" is provided based on the standard input of 48 V and a 100 Mbit Ethernet connection from the node. Photos: P. Fischer.

## Standard Data Provided by the COSYNA Underwater Node System

Each COSYNA Underwater Node System is connected to a standard lander (**Figure 6**, left) carrying a sensor package that measures basic oceanographic parameters (Baschek et al., 2017) continuously year round. It comprises an upward looking ADCP (Teledyne Workhorse 1200 kHz), sensors for pressure, temperature, conductivity, oxygen, chlorophyll-a fluorescence, and turbidity integrated in an extended CTD (Sea&Sun CTD90) and temperature logger (SBE38). All standard oceanographic parameters are publicly available in near realtime (based on the logger after 1 or 24 h) on the COSYNA data portal[8] and the AWI web page[9]. Both data portals offer CSV formatted data for download. The COSYNA data portal offers additional SensorML format via the web service OGC-SOS, the AWI dataportal offers JSON format. Discussions in the scientific community revealed, that most biological oceanographers prefer the CSV format, more standard oriented scientists prefer SensorML or netCDF and data scientists often prefer JSON. Even though the three latter data formats are more efficient with respect to information per data volume, according to our experience it is highly recommended to at least provide one "low-level" data format for download to make data accessible in the context of FAIR also for non-data specialists. On the other hand, CSV formatted data do not fulfill the FAIR criterium of interoperability because CSV files are not *per se* machine readable and linkable. It will certainly require further efforts to implement the FAIR standards for all user groups. An important step in this direction would be the consistent implementation of simple to use import routines for FAIR data formats in the most common spreadsheet programs and the provision of easy-to-use import routines for FAIR data formats in the common script languages such as R or Matlab.

## User Operation of the COSYNA Underwater Node System

The COSYNA system is designed to enable sensor owners to operate their sensors at the underwater node without special knowledge of specific electronics and IT. Nevertheless, the sensor owner must provide basic information about the sensor itself (i.e., the sensor's user manual), about the power requirements of the sensor (voltage and current consumption) and the type of digital communication. The comparatively strict procedure of answering a questionnaire in advance proved to be necessary in the course of integrating the first sensors to avoid misunderstandings between sensor owner and node operator and to avoid malfunctions, or even damage, to the sensor during integration. Based on this information, the physical integration of the sensor is prepared in the lab. There, the user must demonstrate that the sensor will function properly on a computer for at least 24 h with the defined power supply and that the software used for data acquisition (e.g., the original software from the sensor manufacturer) will demonstrate working stability. The final implementation of the sensors in the node system, including the mechanical, electrical and IT integration of the sensor as well as setting up and managing the user access to the sensor, is managed by the COSYNA node consortium, in which the two participating partners – Helmholtz-Zentrum Geesthacht and Alfred Wegener Institute Helmholtz Centre for Polar and Marine Research – are represented.

New sensor integration into the Arctic underwater node system is more extensive than for the North Sea node, as there, sensors are only accessible once or twice a year. In order to ensure the operational stability of these sensors, an integration and *in situ* test operation phase of at least 14 days at the North Sea node has proven to be important to ensure the reliability of the software and hardware components as well as to ensure the capability of complete remote control in terms of power and network. Since the North Sea and Arctic node systems are more or less identical in terms of hardware and network configuration, it is thus ensured that a sensor successfully operated at the North Sea node will also work at the Svalbard node system.

The final access to a sensor or to multiple sensors mounted at either node is established with virtual computers that are set up on the central server. A virtual machine is a software-based individual workstation on which different operating systems (Windows, Linux, Unix) can be installed. Access to the virtual machine is provided through remote login via an open source or a commercial remote login program, whereby the programs "Real-VNC"[10] and "TeamViewer"[11] have become the most popular in the COSYNA consortium. The user has full user rights to install software on his or her workstation to operate a sensor, and each workstation has a standard hard disk size of 500 GB to temporarily store sensor data. This system architecture allows the user to operate a sensor from anywhere in an identical manner and with the same software as used directly in the laboratory without the underwater node system infrastructure.

Data acquisition is important with respect to the software required for continuous sensor operation. For many sensors, only interactive sensor control software is available that requires manual interaction to store data files, read calibration data, or perform other operations. The development of software that allows fully automated operation of sensors, including data storage, is usually costly and requires special programming for each sensor type. Within the framework of the node system development, we developed an alternative way to operate sensors permanently and resiliently without an additional probe-specific software solution. For this purpose, the software "MacroScheduler" was used to code every action a user performs on the screen with a keyboard or mouse into a stable executable program. With this procedure, it has been possible to fully automate any original sensor software thus far. This has the additional advantage that the generated data files can be read with the original software and, if necessary, processed further.

In parallel to the optimization and development of the node hardware, the importance of timely, resilient and, in particular, traceable plausibility and quality checks of the measured data emerged in the course of the operational phase. Especially in

[8]http://codm.hzg.de/codm/

[9]https://www.awi.de/en/science/biosciences/shelf-sea-system-ecology/main-research-focus/cosyna.html

[10]www.realvnc.com

[11]www.teamviewer.com

case of cabled observatories, the tendency and the willingness is great to feed the measured data directly into respective databases and thus to make them immediately available to science and to public stakeholders, especially when the financial support of the systems may depend on this "real-time" data availability. Without reliable and widely automated quality control procedures, there is a considerable risk that unreliable or, in the worst case, false data, e.g., due to sensor failures or sensor drift, may become available and be used by the scientific community or the public. Furthermore, initiatives such as "FAIR" (Wilkinson et al., 2016) address the importance of adequate metadata for each sensor without which it is often not possible to use the data for scientific analyses. In the COSYNA framework, this requirement has been taken into account by checking all oceanographic basic data (see section "Standard Data Provided by the COSYNA Underwater Node System") according to the international standard (SeaDataNet, 2010; Breitbach et al., 2016) prior to their transfer to the corresponding data portals. This ensured that at least impossible or improbable data were clearly marked as "bad" data and therefore could be excluded. In the course of the continuous operation of the two systems, however, it became clear that pure and fully automated plausibility checks, even though internationally accepted, were not sufficient to provide "good" data. We therefore developed a multi-step machine-human procedure to convert probably good data (data which passed the automated flagging routines) into "good" data. The procedure is entirely written in R and uses well published procedures for data de-spiking, data imputation, data cross-validation and visual data inspection and will be published separately. Even though it will never be possible to 100% avoid wrong data in datasets especially from continuous operating observatories, such multi-step machine-human procedure significantly help to minimize the risk of distributing erroneous scientific data and should therefore be always made available together with the respective datasets.

## RESULTS

Similar to moorings or other autonomous sensors, cabled underwater observatories offer the opportunity for temporal high-resolution long-term measurements in areas where it is difficult to perform manual sampling all year round. In addition, automated sensors can form the backbone of intensive measurement campaigns so that discrete sampling, for example, with (costly) research vessels can concentrate on collecting non-automatically measurable variables. In addition to moorings and autonomous sensors, cabled observatories also allow the use of highly complex sensors that need frequent human interaction for reliable operation – even in remote areas where access is limited. At both underwater observatories, in Helgoland and in the Arctic Kongsfjorden fjord system, we successfully operate additional complex stereo-optic sensors and a video-plankton recorder to assess the local fish, macroinvertebrate and plankton community in detail. These sensors provide large datasets of high-resolution images of a certain water volume or benthic area. The images are transferred online directly to Germany, where they are analyzed for total species abundance, species composition, and species-specific length-frequency distributions (Fischer et al., 2007; Wehkamp and Fischer, 2014).

Even though optical systems can be deployed also autonomously, cable connected systems have the advantage of more or less unlimited power supply and storage volume for the images. Furthermore, image analysis is often time consuming especially when no fully automated object detection and measurement algorithms are available, a field of data science in aquatic ecology which is just emerging (Marini et al., 2018). Images are delivered in near-realtime every day and can be analyzed continuously which is often more feasible than analyzing thousands of images after an instrument has been recovered. In addition, 100% autonomous operation over longer time periods is not feasible for such installations. The likelihood is high that such complex systems fail at some point

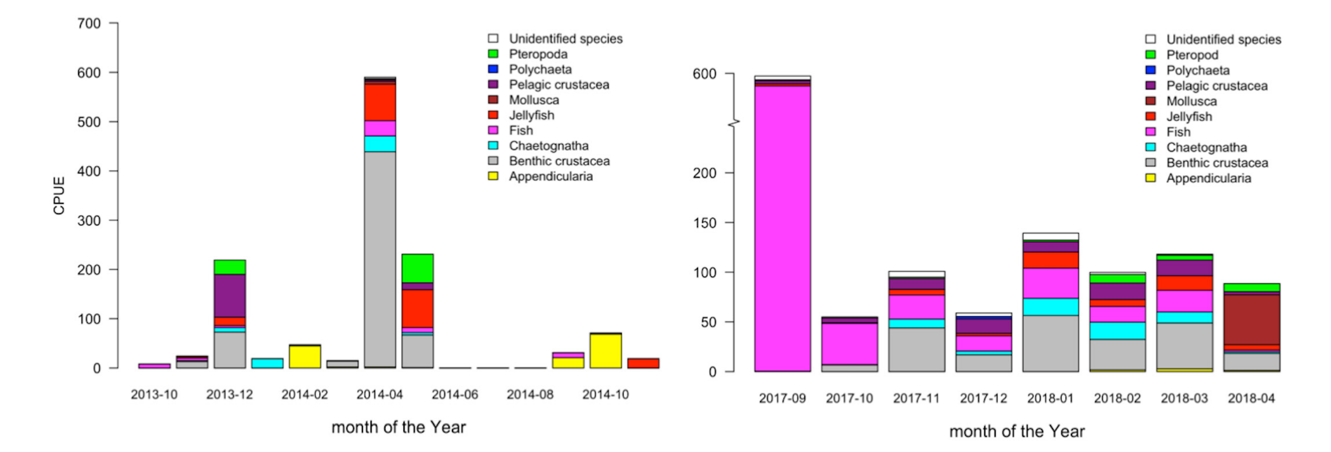


**FIGURE 7 |** Data from the RemOs1 stereoscopic fish observatory attached to the underwater node system in Kongsfjorden. Left panel: Year round survey from October 2013 to November 2014 (modified after, Fischer et al., 2017). CPUE (catch per unit effort) is an arbitrary unit showing the total mean number of specimen counted per week based on 48 stereoscopic image pairs (one image pair every 30 min) summed up per week. Right panel: The same analysis performed for the period September 2017 to April 2018.

and therefore need human interaction for proper operation. Such systems, however, can nevertheless be operated steadily over long periods of time at cabled observatories because they can be continuously monitored with automated routines, and many failures and problems during the operation can be fixed remotely. Furthermore, the samples (e.g., images, videos, acoustic recordings) can be transferred or streamed online to any land-based server, where the samples can be processed and analyzed in real or near-realtime so that not only the functionality of the sensor itself is controlled but also the scientific analysis can be done continuously and concomitantly with the sampling process itself. The latter point in particular is advantageous allowing a rather interactive than static sampling scheme where field campaigns can respond rapidly to signals from the environment, such as the start of the spring bloom or the occurrence of certain species in an area. This is especially advantageous for remote field activities and can make the often costly and labor intensive *in situ* sampling more targeted and efficient.

**Figures 7**, **8** show two examples of such labor-intensive samplings that are impossible to perform without cabled observatory technology. **Figure 7** (left panel) shows a year-round

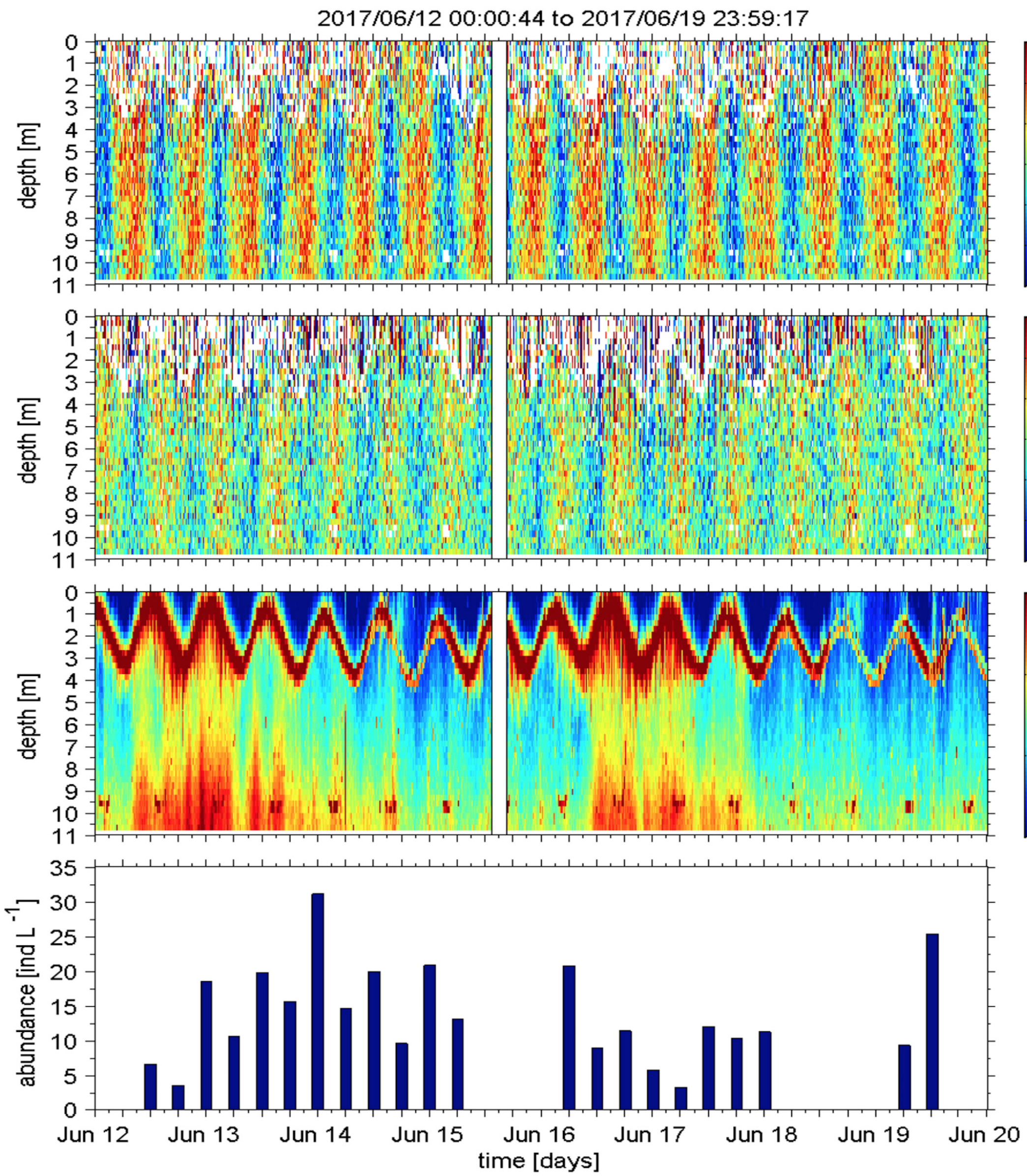


**FIGURE 8 |** An 8-day sample (2017/06/12 00:00 to 2017/06/20 00:00) of horizontal velocity components u and v as well as backscatter estimated from the ADCP, moored about 500 m north of the island of Helgoland at a depth of around 10 m at the AWI underwater experimental field “Margate.” The lowermost panel shows the total abundance of plankton and particles derived from the VPR measurements (no data = underwater maintenance work at the observatory).

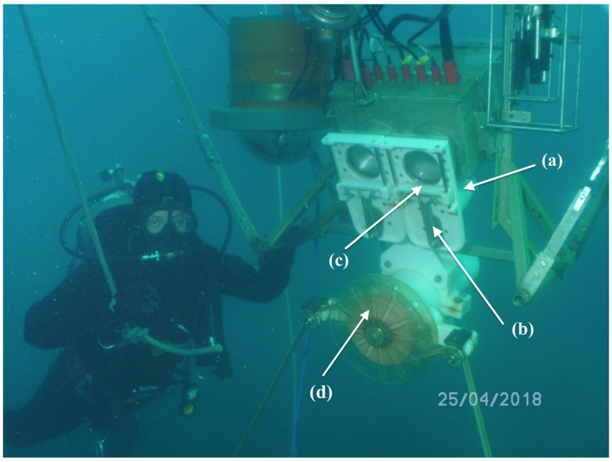


**FIGURE 9 |** Stereo-optical unit in Kongsfjorden with a mechanical cleaning system (see arrow) for the camera lenses. For each lens, the system comprises **(a)** a remotely controlled electrical power unit, **(b)** a connection rod transferring the rotation of the engine to a vertical movement, **(c)** a wiper construction moving up and down to clean the lens systems, and **(d)** remote controlled winch system for vertical profiling and positioning the system in the water column. Photos: P. Fischer.

assessment of the fish and macroinvertebrate community in Svalbard's Kongsfjord in the years 2013 to 2014 using the RemOs1 3D imaging sensors (Fischer et al., 2007).

The profiling optical sensor takes high resolution stereoscopic images with a frequency of one image pair every 30 min and is positioned every week in five different water depths for at least 24 h. Moving the system vertically was done by an in-house designed remote-controlled winch system in combination with a depth sensor (**Figure 9D**) allowing to vertically position the entire system in any depth between the surface and the sea floor. The water column in the littoral zone is thus completely assessed once a week, with 2 days to spare for repeating depth strata that were missed – for example, due to technical problems or poor visibility. The system facilitates measurements of species abundance, species composition and species-specific length frequencies, while providing unique time series over the 24 h diel cycle continuously for 365 days of the year. The observatory enables repeated sampling every year as shown in **Figure 7** (right panel) for the season 2017–2018. This long-term sampling provided the world's first year-round dataset of the littoral fish community in an Arctic fjord system and confirmed the hypothesis that the polar night is rather important for the fish and macroinvertebrate community in very shallow areas. The development and operation of the COSYNA Underwater Node System enabled year-round collection of oceanographic variables together with quantitative data of higher trophic levels in an extremely hostile environment with air temperatures below -30°C and complete darkness during some times of the year. This made completely new insights into the temporal dynamics of this polar shallow water ecosystem possible (for details see Fischer et al. (2017).

**Figure 10** shows a sketch of the remote controlled zooplankton observatory attached to the Helgoland underwater node since 2016. This device is based on the combination of an Acoustic Doppler Current Profiler (ADCP RDI Workhorse Sentinel 1200 kHz, Teledyne RD Instruments USA, Poway, CA, United States) with an underwater imaging system (Video Plankton Recorder, VPR Seascan Inc., United States). The ADCP provides a three-dimensional measurement of the flow field and measures the acoustic backscatter strength, providing continuous high resolution data, for example, to yield precise estimates of timing, velocity and extent of the diel vertical migration of zooplankton communities (Cisewski and Strass, 2016 and **Figure 8**).

The VPR records high-resolution digital images with a frame rate of 15 $s^{-1}$, illuminated by a ring light strobe synchronized with the camera shutter. Four calibrated magnification levels allow the focused imaging of plankton and particles within a size range of 50 μm to several millimeters and thus enable a quantitative optical sampling and size estimate of marine aggregates and fragile species. This includes gelatinous plankton, which is often undetected or underrepresented in the traditional plankton sampling methods (Möller et al., 2012). Both instruments are mounted on the COSYNA node rack in a manner such that one beam of the ADCP (depth cell size 25 cm, sampling interval adjusted to one ping per ensemble with a ping rate of 1 $min^{-1}$) intersects the focal depth of the camera (**Figure 10**) to cover the same volume of water. Plankton and other particle images are automatically extracted from each

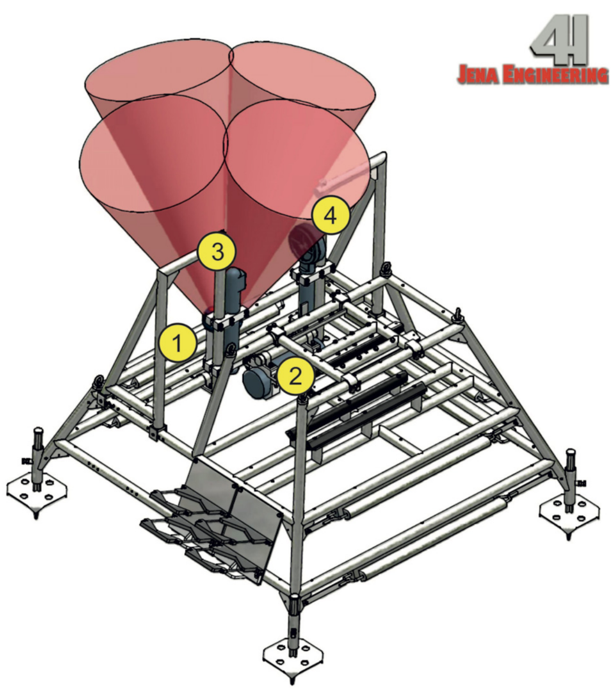


**FIGURE 10 |** Sketch of the Zooplankton Observatory consisting of **(1)** a Workhorse Sentinel ADCP (1200 kHz), **(2)** VPR electronics housing assembly, **(3)** camera housing assembly and **(4)** strobe light housing assembly.

image frame as regions of interest (ROIs) using the Autodeck image analysis software (Seascan Inc., United States), saved to the computer hard drive as TIFF files and immediately tagged using the system's timestamp. This allows later merging with the ADCP and hydrographic parameters.

All images are sent to the land-based server for further processing, where they are classified automatically into taxonomic categories following a method by Hu and Davis (2006). The average power consumption of the entire system including node, sensors and VPR is about 120 W in standard operation mode and the volume of image files from the VPR is app. 20 GB $h^{-1}$. This and the required intermittent human intervention, reprogramming of the system clearly demonstrate that such high-end optical systems cannot reasonably be operated autonomously year-round without cabled observatory technology.

## The Node Hardware

Experience has shown that almost all generic and sensor-specific developments or experimental designs required significantly more time in operation than industrially tested software and hardware. Nevertheless, for some experimental approaches, no off-the-shelf components are available, so that in-house developments are necessary. However, this decision should be carefully examined on a case-by-case basis, as industrial solutions sometimes do exist, which are, however, more expensive initially. For off-the-shelf solutions, however, the financial expenditure is shifted from the investment to the operating expenses. It is important to consider that repairs or adjustments during operation are always associated with the risk of data failure or loss.

In addition to several small changes and optimizations that have occurred over the years during the operation of the nodes, three major problem areas have emerged, each of which has had a lasting effect on the operation of the underwater node during certain phases. These three problem areas were the underwater plug connections, the (non-)availability of essential housekeeping data for error analysis of the system in case of malfunctions as well as the basic software architecture for sensor data processing.

### Underwater Pluggable Cables and Connectors

One of the main features of the COSYNA underwater node is that all individual components – the node, the external battery pack and the sensors – can be exchanged underwater by scientific divers without having to recover the entire system itself. The individual components are therefore connected by cable connections that can be plugged in under water. During the design and construction of the system, special care was therefore taken to ensure that all connectors used were certified by the manufacturer for underwater connection.

During operation, however, it was found that this specification was not fulfilled. Problems arose, in particular, at the main power connection, which delivered up to 980 V to the node. These plugs were officially certified to 1000 V, but failed after only 3 months with a short circuit, although the manufacturer's handling instructions were followed precisely. This stipulated that both the plugs and the sockets, if they are to be plugged in

**FIGURE 11 |** Underwater plug and respective socket of the 1000 V power input circuit after severe damage. Photos: P. Fischer.

under water, must be treated with a thin layer of a special grease supplied by the manufacturer. The analysis of the damage showed that the (+) pins of the 1000 V plug were completely burnt and the jacket of the plug had melted (**Figure 11**), so that sea water had penetrated the plug and led to a massive short circuit on the socket end as well.

The manufacturer informed us that this damage could only be the result of improper handling. We modified the manufacturer's handling instructions and filled all sockets under water completely with a syringe filled with the grease recommended by the manufacturer. This alteration extended the operating time of the connectors to almost 9 months. After that period, however, there was another short circuit and the plug and socket were completely destroyed again.

Based on these events, the company commissioned its own investigations into the plugs. They found that the resistance between the individual plug pins was much lower when they were plugged in under water than when they were plugged in on land, regardless of whether they were properly greased or not. The company offered to replace all underwater plugs and cables, worth approximately €45,000. In addition, the manufacturer's instructions for greasing the plugs was updated, the manufacturing process of the plugs themselves was modified and the manufacturers recommendation for the type of grease to be used for underwater mating was changed to a 100% carbon-free product.

### Logging of Housekeeping Data

A second major issue in the operation of the nodes turned out to be incomplete housekeeping data. In the first node version, the input power on land and the output power at the sensor ports were available as housekeeping data and as Boolean information regarding the leak tightness of the underwater housings and the operating temperature of the individual components.

Continuous and largely unattended operation of the system showed that additional housekeeping data is required, particularly in the event of system malfunctions and failures. It turned out during operation that the originally selected variables and their recording frequency were insufficient.

As already mentioned, the most critical components during operation were not the electronic components in the node, but the cable-bound connection between the individual components. The first node generation did not include an explicit infrastructure for a continuous and higher-frequency logging of the undisturbed functionality of cables between the node components. As a result, it was often unclear which component of the system was affected, resulting in unnecessary recovery of all node components or lengthy underwater troubleshooting and testing. Based on this experience, we decided to equip all pluggable cable connections with appropriate sensors for voltage on both ends in order to obtain detailed information on where a possible malfunction is located. The availability of this information significantly accelerates troubleshooting, as defects in cables and connectors can either be detected so early in operation that a problem can be prevented, or malfunctions can be found and corrected more quickly (in case of internal system component failures. In this context, we experienced that in addition to the continuous monitoring of the voltage and current parameters, a continuous monitoring of the residual currents of the power lines is of critical importance. Residual current measurements provide information about the insulation condition of the cables and connectors against the surrounding water. Particularly in the case of the underwater mateable connectors, a slow increase in the residual current indicates a gradual loss of insulation of the connector, e.g., due to the washing out of the insulation grease. This problem can then be solved in time and without potential damage by re-greasing the plug connections under water.

## Node Control Software

The overall power management of the underwater node (switching the individual ports on and off and providing power to the sensor ports) as well as the node monitoring (power consumption and network activity) is realized by Programmable Logic Controllers (PLC) with discrete software. The first prototype node used Siemens Simatic S7 PLC, which was replaced in rebuild by a Beckhoff CX8090 CPU. Both PLC solution were equipped with required analog and digital I/O modules. The remote control and monitoring is realized via a web frontend and IP, and all available information are logged in a SQL database for system monitoring. This frontend has three access levels: "user," "port administrator" and "system administrator." As of now, "users" are allowed to see the status of all ports (i.e., see if a specific port is on or off); "port administrators" are allowed to switch all ports on and off and to change the maximal power (watt) that the individual ports deliver; and "system administrators" have full access to the system, including adding new users with password settings.

This software design proved not to be ideal for an infrastructure used by several independent groups in parallel. In particular, the roles and privileges of the "user" and the "port administrator" were not well designed. Currently, "users" only have read/write access to a port for accessing a sensor and downloading data. In order to switch off a sensor completely, "port administrator" rights are required. "Port administrators," however, cannot be enabled for single ports only, but have access to all ports and extended functions of the node. This leads to the consequence that external users are only assigned the role of "user" and thus cannot switch their own sensor on and off. This is especially problematic with sensors that are not completely developed or automated either in terms of the hardware or the software and therefore frequently must be disconnected from the power supply network in order to reset.

## Software Issues With Respect to Sensor Operation

In the very beginning of node operation, two different scenarios of sensor operation were planned: (a) the operation of sensors for standard parameters by the node consortium itself and (b) the operation of sensors from external partners under the full responsibility of the external users. The external users, in particular, were thought to be fully responsible for their data and, after the initial installation phase, also for the remote sensor operation and monitoring. Both scenarios were adapted based on the experiences of the first year of node operation. Scenario A was initially designed as a type of real-time operation, where the sensor data were to be streamed directly to a central database at the Helmholtz-Zentrum Geesthacht. While the basic principle of this real-time streaming approach works well for our set-up and is still in place, some shortcomings of a pure streaming procedure became apparent. Many sensors do not deliver "to go" data directly from the sensor itself but "raw data," such as voltage, a digital or a binary output. This data must be processed by software using calibration coefficients or conversion algorithms to obtain the target parameters in the correct units. With direct streaming, the raw data (e.g., Volt) are converted by generic software "on the fly" into scientific values which are then directly fed into the database, however without storing the original raw (e.g., Volt) data. This holds a considerable risk in case the calibration files are technically decoupled from the probe and can thus be unintentionally confounded. In 2014, this "on the fly" conversion resulted in almost 2 month data loss from a specific sensor, because the wrong calibration file was used. Because the raw data (Volt) were not stored, the scientific data could not be recalculated with the correct calibration file. To prevent this, it was decided not only to stream the final scientific readings from each probe, but also to store the raw data from each sensor in the highest possible temporal resolution (e.g., in volts at 1 Hz) every hour in single files. This makes it possible, in case of accidental use of the wrong calibration file, to recalculate the data completely afterward. Additionally, it was decided to implement additional security procedures for the data transfer to the respective databases to avoid the transmission of erroneous data in the data portals and to ensure that there is no missing metadata for individual sensors. From 2016 on, the transfer of data into the database itself was obligatorily linked to the availability of a minimum of up-to-date metadata which means that if metadata were missing, no data entry would be possible at all. This strategic upgrade of redundant data acquisition and storage procedure proved to be extremely reliable and allows post-processing of all data in case of a failure in the real-time streaming process occur.

To store raw or scientific data in discrete hourly files, we prefer to run the original program provided with the sensor. This has the advantage that the program can undertake all raw data conversions and usually delivers "readable" ASCII files, which can be used for further processing with standard

**TABLE 1 |** Summary of failures of the underwater nodes in the southern North Sea and in the Arctic.

| System compartment affected | Type of failure in the underwater node system | Frequency of the failure | | Duration of system shutdown in weeks | |
|---|---|---|---|---|---|
| | | North Sea | Arctic | North Sea | Arctic |
| Long distance sea cable connection (1000 V/400 V Helgoland; 400 V/240 V Spitsbergen & GBit fiber optic connection) | External forces | Once in 2018[1] | multiple times until 2016[2] | 8 | 12 |
| | Leakage | Once in 2012[3] | – | 3 | – |
| | Erroneous shut down due to malfunctioning hardware or software | 6–8 times from 2012 and 2013[4] | 2–3 times until 2015[5] | <1 | – |
| | Underwater mateable power plugs | 4 times from 2012–2018[6] | – | 8–12 | – |
| Cable connection between node and sensor units (48 V and GBit copper lines) | Leakage | – | – | – | – |
| | Erroneous shut down due to malfunctioning hardware or software | – | – | – | – |
| | Underwater mateable power/network plugs | 4–5 times from 2012 to 2014[7] | Once in 2014[7] | 2–4 | 12 |

*Shown are the types of risks which led to complete or partial system failure in our nodes from 2012 to 2018; the frequency of occurrence of this type of failure in terms of occurrence over time; and the duration of the system shut down based on the respective failure. Index numbers refer to additional explanations in* ***Appendix****.*

programs. Only very few sensors come with programs that would enable the sensor to run fully automatically for several months and save data files at pre-defined time intervals. We therefore use the macro scripting language "Macro Scheduler" to make these non-scriptable programs fully automatic and remotely controllable. This is done by simulating user interactions in macros, which then can be run in pre-defined time intervals, such as every 60 min or 24 h. This procedure proved to be extremely efficient and reliable, especially when integrating new sensors into the network.

The second sensor operation scenario (sensors operated by external partners under their own full responsibility) more or less failed. Our expectation was that most external groups that asked for the opportunity to operate a sensor at the underwater node were experienced in remote sensor operation and that it would be sufficient for us to provide assistance during sensor integration. This assumption turned out to be unrealistic. Most users approaching us to operate a sensor, either in the North Sea or in the Arctic, are experienced in manual sensor operation and data handling but not in remote controlled automated sensor operation. To remedy this finding, we also applied our internal sensor operation procedures to the external partners. We offered not only to install but also to operate the sensors, utilizing our automation and data saving routines, and most often also using automated data file delivery to any server or e-mail address. It turned out that this "full service" was a better solution for all internal and external partners, often leading to scientific cooperation projects rather than mere infrastructure used by the external partners.

Although the software on sensor control, data transfer and regular node operation developed since 2012 is not so far available in a public repository like GitHub, all scripts and routines are freely available upon request. This is especially true for the complex routines and scripts for data plausibility and quality control, which are mainly written in the script language R (R Core Team, 2017).

# DISCUSSION

Underwater node systems are one of the future technologies that can contribute to real progress in coastal ecological research once their technological development is sufficiently advanced. The possibility of a continuous interactive "presence" in environmentally (e.g., weather-related or geographically) difficult focus regions, such as the polar regions or the North Sea, makes this technology highly valuable for answering Earth system questions (Trowbridge et al., 2019). Cabled underwater observatories should be integrated into larger, networks since the digital connection of the sensors to the Internet is readily possible. The two COSYNA node systems presented here are part of the emerging German Digital Marine Network "MareHub" and the German National Research Data Infrastructure (NFDI) as well as part of the European Jerico 3 network. In addition, COSYNA data are delivered to the European Marine Observation and Data Network (EMODnet) and Copernicus Marine Environment Monitoring Service (CMEMS). However, experiences with operating the node systems described here also show that there are still several technical, conceptual and structural problems that must be overcome in order to improve the use of underwater nodes as a fully operational and stable technology for aquatic research in the future. The most important points concern the power supply, the stability of sensors in continuous operation mode and the handling of large data sets by the scientists themselves – that is, the need for user training.

## Power Related Issues

During the operation of the COSYNA underwater node system from 2012 to 2018, several power related issues emerged, which intermittently hampered the operation of the nodes and the attached sensors considerably. **Table 1** shows a summary of the major power failures during the continuous operation of our observatories. The first two columns compile the sources of failures in the power supply system. Some of the problems listed in **Table 1** occurred only once and could be fixed permanently. The central problem that could not be solved by a single repair and is still virulently occurring in our systems is the issue with the underwater pluggable connections. Although the manufacturer has made some modifications to the installed connector types in response to our damage reports (see section "The Node Hardware"), it must be stated that the connector types used are still only conditionally suitable for long-term use under water and must be maintained at least every 6 months. Even though this can be done under water by scientific divers after some training, the problem is not ultimately resolved, and there is a certain risk that the plugs will show malfunctions even though they are properly maintained. During maintenance, particular care must be taken to clean and degrease the plugs and sockets thoroughly and to fill the socket holes again with 100% carbon-free grease (e.g., Parker SuperOLube). When assembling the plugs under water, it is absolutely crucial that the grease is pressed out of the socket holes during the plugging process and that the grease completely fills the gap between plug and socket. This is necessary to prevent seawater penetrating this gap to avoid, for example, small mussel larvae – which are only few $\mu$m in size – from settling in this space, growing there and slowly pushing the plug out of the socket.

If the procedure described above is followed exactly, it is possible to use medium-priced underwater connectors for shallow water observatories, but with a latent risk of failure. Therefore, to avoid the risk of system failure, industrial plug connections such as GISMA, which are significantly more expensive, however, should be used, especially for voltages above 48 V.

## Sensor Exposure Time

A particular challenge for the longer term operation of underwater nodes is the fact that sensors may not be designed for longer term exposure, i.e., for several months. There are only few sensor systems available which have a manufacturer designed device to prevent biofouling and therefore must be cleaned by hand at regular intervals. Furthermore, probe manufacturers typically do not provide reliable information about the temporal drift behavior of their probes or the recommended maximum duration of a measurement until recalibration is required. Some manufacturers do not even provide accuracy and precision values for their sensors, even if they are properly calibrated. This missing information on data quality of sensors lead to the highly unsatisfactory situation that scientists sometimes have to trust sensor data without being able to estimate data accuracy and without a proper knowledge of the probe's behavior especially during longer time exposure. Because we cannot assume that sensor data, even when a sensor is quite expensive, are correct *per se*, we need a better implementation of validated data quality control routines in aquatic ecological disciplines. Such procedures are already available (see e.g., Ocean Best Practices System Repository)[12] but should be applied as default, e.g., as ready-to-use packages in common software and scripting languages. Until now, automatically generated data are too often not continuously checked for quality from the start and corrected if necessary, but only after several weeks or even months. If no reliable and fully automated control routines are implemented in such a system, errors in the measurements often remain undiscovered for too long and cannot be corrected afterwards. The result is that the data sets must ultimately be discarded.

[12]https://www.oceanbestpractices.org/

### Biofouling

The problem of biofouling is probe and even parameter specific. While temperature and conductivity sensors are less affected, optically or chemically based sensors face the problem of significant accuracy loss as well as potential precision loss after only a short time, especially in warmer temperatures. While our Arctic sensors were normally perfectly stable for months during the Arctic winter when no light was available, in spring and summer, these sensors were overgrown with periphyton within days or weeks. In the Arctic system especially, when daylight returns in spring, periphyton can grow so fast that "soft" anti-biofouling measures such as UV-radiation (MacKenzie et al., 2019) or gentle acid applications on surfaces cannot cope with the growth rates of the biota. In our case, only mechanical hardware cleaning systems such as wipers were effective in preventing sensor overgrowth and uncorrectable data deterioration. Mechanical wipers are, however, not applicable for all sensors and are normally technologically demanding. **Figure 9** shows a mechanical wiper system developed by AWI for a stereo-optical camera system used in our Arctic observatory since 2013. The system's cleaning frequency can be remotely adjusted and removes periphyton mechanically from the windows. The system is quite complex and needs to be fully integrated into the sensor control system itself. However, such a cleaning system can hardly be applied to, for example, commercial multiparameter sensors, where several different sensors are mounted very close to each other. As of yet, there is no overall convincing solution available on the market for such sensors (Delauney et al., 2010; Venkatesan et al., 2017) and most manufacturers simply do not offer "anti-biofouling" systems for this equipment. An emerging technology might be the improved UV-radiation systems, which have recently become available and which rely on modern diode-technology. However, according to Venkatesan et al. (2017), technology has not yet reached a level to avoid biofouling to an extend that the sensor's data quality is not significantly affected when mounted for longer periods of time. Therefore, biofouling remains a major issue in most long-term monitoring projects especially in productive coastal systems.

### Maintenance Frequency

The overall maintenance intensity of the two systems varies depending on the location. The Helgoland system has to be cleaned almost weekly in summer, because biofouling has a strong impact particularly on the optical sensors but also, e.g., on the conductivity sensors. The node system proper (without the sensors) is almost maintenance free and can in principle remain under water for several years, except for electronic and mechanical system failures.

In contrast, the node system in Svalbard is usually completely serviced twice a year. The main reason for this is that system failures are much more difficult to repair than in Helgoland, so that we try to avoid them by more frequent routine maintenance. Furthermore, the mechanical load on the Svalbard system is much higher, especially in spring and summer due to iceberg drift, so that mechanical damage of the system needs to be repaired. Since 2017, the previously fixed scheme of a routine maintenance in spring after the polar night and another routine maintenance in autumn before the polar night has been changed in favor of only a scheduled maintenance in autumn and a second more flexible maintenance phase when it is needed. A maintenance stay in the Arctic is scheduled for 2 weeks on site plus travel time each with a diving team of 3 persons and one or two additional technicians. During this time, the node system is completely recovered, all plugs and cables are carefully checked and individual components are replaced if necessary. For the electronic system components, a replacement interval of 5 years is scheduled, even if the components as such are still fully functional. This is particularly due to the problem of the expensive and time-consuming travel to Spitsbergen. No fixed maintenance intervals are specified for the node system Helgoland, as all maintenance work and repairs can be carried out within a few days due to the easy accessibility.

## Smart Sensor Technology

Another need for future technological development in remote-controlled long term sensor operation is the implementation of modern communication procedures in marine sensors (Martinez et al., 2017; Del-Rio et al., 2018; Lin et al., 2020). Today, even the simplest IT equipment, such as printers, have fully automated reconnection procedures. This is unfortunately not the case in most marine sensors, which often do not have the simplest plug-in connection procedures let alone TCP/IP technology. Significant technological innovations in sensor development are therefore needed to provide smart monitoring technologies with automated error handling procedures if the control software fails (Toma et al., 2013). Also necessary are reliable alerting functions in the event of a contact failure. In addition, we need to implement state-of-the-art IT technology under water that works based on plug and play technology. This includes fully automated transmission, verification, storage and accessibility to sensor metadata and sensor actions, such as deployment or maintenance. The result needs to be a significantly reduced human interaction in sensor operation.

## Housekeeping Data

Closely related to the need for better sensor technology is the need for more comprehensive background information on the status of the node system itself, the so called housekeeping data. The need for continuous recording and storage of such technical data is often only recognized when a problem occurs in the system. Therefore, when systems are fully functional, there is a high potential that the continuous collection of housekeeping data will be disregarded, especially as it does not provide real scientific added value and can be very specific to each system. In the context of the continuous operation of the here described node systems, it turned out to be most efficient if the housekeeping data for the relevant system components are handled identically to the scientific sensor data. We finally decided to feed the housekeeping data into the repository together with the scientific data on the dashboards. This ensures that the housekeeping data receive the same amount of attention as the scientific data and are recognized as important "metadata." A continuous recording of the housekeeping data is also useful because the most critical system failures (i.e., electrical short circuits in submarine cables) develop gradually and can be detected at an early stage when it is still possible to take adequate countermeasures and to plan a timely repair, so that longer system shutdowns can be avoided.

## Software and Conceptual Issues

Further important changes that can only be implemented in the context of future node generations concern the node control software and the general network and software architecture. One important point that proves to be disadvantageous for smooth sensor operation at our node system is the limited rights of external users, who can only communicate with their sensors but not switch their power on and off. This is a particular hindrance when a sensor or the software crashes during the weekend when no node operator is available to reset the sensor. As part of the further development of the node software, we therefore plan to selectively assign "port administrator" rights to external users, so they can switch the power supply of a specific port on and off themselves.

Furthermore, we plan to upgrade the underwater node network architecture to VLAN technology (virtual local area networks) (Wang et al., 2013; Das et al., 2014). This technology allows grouping of selected sensors (e.g., of an external user) into a closed virtual network that is invisible to other external users. This prevents different external users from influencing each other, for example, by accidentally switching off the port or the communication interfaces of another user. The installation and management of separate VLANs requires more time and expertise in early operation, but brings considerable advantages in the long run. It enables, for example, bulk network management by means of professional standard tools for network configuration and maintenance, but also easier forwarding and integration of a certain sensor or VLAN into the IT infrastructure of an external institute. This would significantly simplify remote users' access to their sensors in the node network.

Another problem when integrating external sensors into the COSYNA node infrastructure is the data transfer from the sensor's virtual machine to the sensor owner's IT infrastructure. Although it is almost always possible for a sensor owner to manually copy files from the virtual machine to his/her own institute's drives, automated data transfer requires external access (in the case of our nodes from the COSYNA network) to the owner's own IT infrastructure, such as an FTP data server or a direct data stream service. Experience shows that this is often problematic or even impossible due to different Internet security procedures at the different institutes (see Cragin et al., 2010). In these cases, it actually proved more practicable to use a commercial server provider, such as "Dropbox" or "Google," to which the data was first automatically copied and then retrieved by the external user's institute. In most cases, this rendered read or write access to external institute servers unnecessary.

A further lesson learned in the operation of this node system since 2012 is, that cabled shallow water observatory systems, which are comparatively easy to access, e.g., by divers, can be designed differently than autonomous mooring systems, which have to operate unsupervised over a long period of time. An important advantage of shallow water systems is the possibility to also perform short-term projects with frequently changing sensors in an experimental operation mode. Furthermore, the sensors are fully accessible via remote control at all times and can be even restarted completely in case of a system error. In this case, it is often easier to use the software supplied by the manufacturer of a sensor in terms of the cost-benefit calculation than to program complex special software for remote operation. As a rule, this only makes sense if the respective sensors are planned for long-term use, e.g., to provide relevant oceanographic or biological background information of the area like water temperature, current and light conditions (see section "Standard Data Provided by the COSYNA Underwater Node System") etc. which are often required as auxiliary information for proper data interpretation of experimental setups.

## Data Issues

In addition to the hardware and software changes, which have already been implemented or are planned for future node generations, data processing is also an emerging topic that should receive considerably more attention when dealing with cabled observatory technology.

An important first step with regard to successful operation of automated sensors is the definition of responsibilities for the sensors itself but also for the data (Leonelli, 2016). It should be clarified in advance whether a cooperation partner only needs the node infrastructure and on-site support for installation and maintenance to operate his sensor or whether further support is required for data processing and software engineering for continuous sensor operation. These requirements and the necessary financial expenditures must be made clear in advance to avoid confusion regarding responsibilities during operation, which can also have significant consequences for data quality.

A closely related issue concerns the handling of the continuous data stream. Cabled observatories provide an almost unlimited amount of sensor data that must be quality- controlled, stored, processed and finally published. Even though data processing methods in the area of "big data" have developed significantly in recent years, it cannot be assumed that all sensor owners are able to process a continuous stream of data adequately and reliably in the long term to guarantee data accuracy and reliability (see also Wallis and Borgman, 2011). For this reason, a basic "data policy" was adopted within the framework of COSYNA. Originally, it was planned that each external "sensor owner" would need to handle and process the data files generated by the owned sensor him- or herself and that COSYNA would only take over the data handling in exceptional cases. This method proved to be unsuccessful, with a high risk of data loss for external sensors. Most external users are able to handle individual data files from their sensors, but are overwhelmed when the same data files must be continuously processed. Data files are then often stored unsystematically, locally and without the necessary backups. Based on this experience, the "data policy" for handling external data was changed in such a way that all data, if the sensor owner agrees, are also stored in the corresponding COSYNA databases and are available there in the highest available resolution via a password-protected web interface (Breitbach et al., 2016).

A last major lesson learnt in the course of long-term automated sensor operation at our underwater node systems addresses data management and data verification procedures (Vallejos and Morimoto, 2013). Data verification routines based solely on labor-intensive visual procedures by scientists or technicians are not viable in the long run. This might be possible if an experiment runs only for shorter periods – over, say, 2–3 weeks – but not when multiple complex sensors are online 24 h a day, 365 days a year. Promising steps are undertaken in monitoring systems where near real-time plausibility control procedures are implemented to flag suspicious data (out of range, spikes, stuck values, missing values) automatically (Huang et al., 2016) and send a warning to an operator if too many data were flagged.

However, flagging only addresses the plausibility of the data and is not a comprehensive data quality procedure. When considering data quality, additional parameters must be given for each data point, providing, for example, accuracy and precision of this data point. This means that there is at least an estimate available about the expected maximum possible deviation of a measured value from a real value (accuracy) and additional information on the spread of multiple measurements of the same value (precision) (Menditto et al., 2006). It will be the task of future collaborative projects between engineers, scientists, data managers and statisticians to develop technological and conceptual solutions as well as mathematical procedures for the highly variable coastal seas (Grubbs, 1973). These developments are to provide data in such a way that a scientist using automated sensor data does not only have a single value he/she must trust but a range value identifying, for example, the 90% confidence limits for each measured value. The scientist then is free to decide whether this accuracy is appropriate for his or her scientific application or if he/she must reject this value as too inaccurate or imprecise for the scientific question at hand. Nevertheless, even the best algorithms will not be able to replace a final data check involving human expertise. However, this final check must be automated to the greatest extent possible, for example, using web-applications, which the responsible person can easily access and share online. These applications will assist in deciding whether the data from a certain period are ultimately correct and should be released (or not). This includes supervised online procedures to mark single data points interactively as bad values based on standardized mathematical routines. As long as no robust artificial intelligence procedures are at hand, this will be the only way to detect, for example, wrong calibration constants, gradual sensor drifts, gradual onset of biofouling, and more. Especially for coastal waters, the natural variability of data over long time scales is significant but also hardly predictable by even the most sophisticated mathematical algorithms. Therefore, regular visual inspection in parallel with automated procedures must be considered as a "must have" in the planning and allocation of resources. This must be undertaken in a feasible manner for scientific experts without being too time consuming with respect to the computational effort.

Unfortunately, in marine technology and data management, we do not yet fully use the computational potential of modern interactive data analysis and state-of-the-art data verification technology, even though promising approaches have been developed in recent years within the community[13,14]. These approaches must be consistently developed further in close cooperation between data scientist and ecologists to ensure that they are mathematically/statistically correct and also applicable in natural science without being a data science specialist. Such methods should include in particular sophisticated technologies based on data gap analysis and missing data inclusion as well as intelligent modeling procedures for sensor data prediction. These can be used for online plausibility check procedures, especially in complete data and sensor systems.

## CONCLUSION

Summarizing the experiences in the operation of the cabled COSYNA underwater observatories from 2012 to 2018, several points can be concluded. We need an innovation boost in the field of intelligent underwater sensor technology. This is particularly important in view of global change, since the effects of global change are unfortunately most strongly perceptible in areas such as polar systems, which are only partially accessible due to climatic conditions. The latest research clearly underlines the fact that a deeper and functional understanding of our Earth system is imperative to address the upcoming climatic and anthropogenic challenges for humanity. It also underscores the fact that these challenges cannot be solved separately in individual disciplines, but require an integrated approach across scientific subjects. These areas include natural sciences, engineering, data sciences and informatics. In order to achieve efficient interdisciplinary and transdisciplinary research, more and comprehensive environmental data must be available. In particular, the possibility of data evaluation and data analysis for high-frequency data from fully automatic sensors must also be significantly improved. Data evaluation should not only focus on the computing capabilities in handling large data sets, but on actually gaining scientific insight into Earth systems. In our opinion, this requires two important strategic paths in the planning and operation of automated marine sensor systems. Firstly, this means consistent application of a strict "open source policy" for scientific hardware and software development with the aim that the various disciplines can contribute to technological development and secondly the consistent implementation of the "FAIR" principle (Tanhua et al., 2019) in the field of data science – that is, data must be "findable," "accessible," "interoperable" and "reusable." It will be a great challenge for the next few years to implement measures that work toward this goal on a broad basis, bearing in mind that even the first requirement of keeping data "searchable" has not yet been met in many sensor networks.

[13]https://www.seadatanet.org/Standards/Data-Quality-Control

[14]https://ioos.noaa.gov/project/qartod/

## DATA AVAILABILITY STATEMENT

The datasets generated for this study are available on request to the corresponding author.

## AUTHOR CONTRIBUTIONS

PF and HB coordinated the production of the manuscript. PF, HB, BB, AK, RR, and J-PG wrote the overall text. MB, GB, and SA significantly contributed to the description of the system details and IT infrastructure. BC and KM wrote parts of the "Results" section. WP and RW significantly contributed to the "Discussion" section. All authors contributed to the article and approved the submitted version.

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**Conflict of Interest:** The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest.

The reviewer JB declared a past co-authorship with several of the authors PF and AK to the handling editor.

## APPENDIX

The descriptions and numbers of system failures below refer to **Table 1** in the "Discussion" section.

1. The failure occurred once in October 2018, when the network link to the underwater node became unstable with increasing package losses during operation. About 1 month after these first issues occurred, the connection went down completely. A systematic error search revealed that the sea cable was damaged at the entrance point to the water, where an underwater metal construction was loosened and continuously hit the cable due to the swell of the water. The cable was recovered at this point and could be repaired by a professional company.
2. A failure in the parts of the power connection of the underwater portion of the system occurred several times from 2012 to 2016 due to iceberg damage. All cables and tubes were protected by PE tubes with a wall thickness of 1 cm and a tube diameter of 20 cm buried in the ground to a depth of about 0.5 m up to a water depth of 7–8 m. Nevertheless, the short distance where power cables laid open toward the observatory fundament was affected almost every year until 2016. Cables were either completely ripped to shreds or simply damaged at a single point. In 2016, all non-protected parts of the power cables were additionally coated with flexible and wire-reinforced PVC tubes, with a wall thickness of 8 mm and a diameter of 44 mm. Each single cable was protected with an individual tube, starting from the PE-protection tube at a 7–8 m water depth, ending directly at the fundament of the observatory at a 12 m water depth. No more damages of the underwater cables have occurred after this modification.
3. At the very beginning of the node operation in Helgoland in 2012, a leakage occurred in the underwater terminator of the 1000 V sea cable ("breakout box"), where the node is connected to the sea cable via an underwater mateable power plug (**Figure 4**). The reason for this leakage was a deformation of the PE fabricated cable shell of the central underwater cable at exactly the point where the commercial cable penetration squeezed the cable to ensure water tightness. It finally emerged that the reason for this leakage was an installation error of the cable penetration. Essentially, a specific part was not installed, which should have guaranteed a homogenous squeezing of the seals around the cable.
4. To monitor the main power supply of the node system in the North Sea, independent from the software monitoring, a special hardware fuse system for the 1000 V direct current was installed. Independent from any software, this monitoring system was designed to measure the main power line integrity to the underwater node system up to a distance of 30 km in cable length and would shut down the main power supply in case of a cable failure. Even though the commercial manufacturer of this monitoring system provided evidence that the type of system works properly on land, it never worked properly in our underwater application. The system was modified several times within the first year of operation by the company that manufactured the electrical components of the node power supply, but this never solved the problem of erroneous power line failure messages. This led to a complete shutdown of the system each time. The system's cable monitoring system was finally shut down as a result of too many false error messages that led to unsolved system shutdowns. Because the software-based power monitoring system worked well over the entire period and always shut down the system correctly in case of simulated power failure, it was decided that this software system was more reliable compared to the separate hardware solution.
5. This error occurred only in the Arctic observatory and in the fiber-optic lines. Beginning in 2014, spontaneous but not persistent package losses were observed in the network connections to the node in one fiber-optic line. Therefore, we used only the second fiber-optic line from 2014 onward, working under the assumption that the fiber optic lines of the main sea cable had been damaged. In 2017, the entire network switch and computer infrastructure on the land end was updated and changed from single computers to a redundant server infrastructure, with virtual computers undertaking the data storage and management tasks. During this renewal process, all fiber optic connectors on the land end of the sea cable were cleaned and partly refurbished. From this point on, the second fiber-optic line also worked properly again and did not show any failures since then.
6. The most critical point in the operation of the Helgoland node system is the underwater mateable plug for the primary power supply of the node. We decided to use a standard underwater mateable power plug system in an intermediate price category. This decision was based on a written statement by the manufacturer that the plug system was rated for 1000 V when underwater mated and also because we had positive experiences with this type of plug system in a previous project (Fischer et al., 2007). A first major failure of the plug system occurred only after about 3 months of operation. After recovering the node system, it was apparent that the 2 m long specially manufactured power plug cable connecting the 1000 V line at the terminator box of the sea cable (the breakout box) and the node was melted at the terminator box end and needed to be refurbished (for a detailed description of the damage see section "The Node Hardware").
7. Problems with the cable plug connections also occurred in the first years in particular in the 48V/network hybrid cable technology. After consistent application of the routines and procedures described in section "Underwater Pluggable Cables and Connectors" and Appendix Item 6 when plugging the cables under water, these problems no longer occurred.

**Cover page - Publication IV**

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Contributions: The idea, design, and execution of this study were performed by me with advice from PF. LS prepared otoliths for analysis and acted as a second reader in the analysis. JMW performed the analysis of stomach content samples. The manuscript was written by me, FCM, and BJ added advice for the interpretation of the data. I wrote the manuscript, LS and PF assisted in the revision of the manuscript.

# Age class composition and growth of Atlantic cod (*Gadus morhua*) in the shallow water zone of Kongsfjorden, Svalbard

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Abstract

Atlantic cod (*Gadus morhua*) has been found in Kongsfjorden since the mid-1990s. This glacial fjord can be characterized as sub-Arctic despite its location at 79°N. Its sub-Arctic character is derived from a strong influence of warm Atlantic water masses from the West Spitsbergen Current, a branch of the Norwegian Atlantic Current. The regular catch of juvenile Atlantic cod in Kongsfjorden since 2008 can be seen in the context of an overall northward shift of boreal fish species. In the framework of a quantitative inventory of the shallow water fish community of Kongsfjorden in 2012 - 2014 we detected juvenile specimens of Atlantic cod (N = 730). By otolith microstructure analysis the primary fractions were identified as age class 0+, 1+, and 2+. We show that different cohorts of those specimens express stable growth rates in the polar day and night. By stomach content analysis we show that those specimens primarily feed on benthic food sources. The combination of those observations makes the shallow water zone of Kongsfjorden likely a nursery ground for Atlantic cod.

Keywords:

*Gadus morhua*

Northeast Arctic cod

Fish growth

Svalbard

Kongsfjorden

Climate change

## Introduction

Kongsfjorden at the west coast of Svalbard is characterized as a sub-Arctic, glacial fjord at 79°N. The sub-Arctic character has its origin in the hydrography based on a strong influence of Atlantic water masses in the last decades (Payne & Roesler 2019). Long-term data of Beszczynska-Möller et al. (2012) show that the Arctic is affected by increasing water temperatures originating from the West Spitsbergen Current (WSC), a branch of the Norwegian Atlantic Current.
The Arctic is expected to be one of the focal areas facing climate change-induced temperature increases in the next decades (IPCC 2014). Due to this a northward shift of marine fish species in the northern hemisphere is postulated (Christiansen et al. 2014, Fossheim et al. 2015). In particular, for Atlantic cod (*Gadus morhua*) first signs of such a northward shift have already been reported (Misund et al. 2016). The already sub-Arctic character and the effects of climate change-induced temperature changes make Kongsfjorden an interesting research environment to observe a fish community in the general Arctic realm, but in a sub-Arctic hydrography.
Atlantic cod is found in Kongsfjorden since the mid-1990s, with juveniles regularly caught since 2008 (Berge et al. 2015b). etween 2012 to 2014 quantitative studies on the inventory of the Kongsfjorden shallow-water fish community showed high abundances of specimens between 5 and 20 cm standard length (SL) (Brand & Fischer 2016, Fischer et al. 2017). The shallow-water zones of Kongsfjorden are highly structured environments, especially the hard bottom zones are covered with kelp forests. Those zones are potential nursery areas for Atlantic cod as reviewed in Seitz et al. (2014).
Atlantic cod is well known as one of the most important commercial fish species in the Atlantic region. It is distributed along the continental shelves of the North Atlantic between 40° and 80° of latitude (Neat & Righton 2007, Sundby 2000). Its total thermal niche is reported to range from -1.5 to 19 °C, with a narrower range of 1 to 8 °C during the spawning season (Righton et al. 2010). Ottersen et al. (2014) describe the stock in the Barents Sea and Svalbard, known as Northast Arctic Cod (NEAC), as the largest one at present. This stock, also known as “Skrei” performs annual long-distance migrations between spawning and foraging areas. One foraging area is located in the Barents Sea up to Novaya Zemlya in the East and another one on the western continental shelf of the Svalbard archipelago (Brander 2005). The main spawning area of NEAC is located at the west coast of Norway from Møre in the south to Finnmark in the North, with the main spawning grounds at the Lofoten (Brander 2005, Godø 1984a+b, Sundby & Nakken 2008).
Spawning of NEAC occurs from February to May with the main spawning period in March and April (Brander 2005). Hatching occurs 2-5 weeks after spawning and is followed by a switch to exogenous feeding after 3-6 weeks. A subsequent metamorphosis to the juvenile stage happens 2-3 months after spawning (Ottersen et al. 2014). Suthers & Sundby (1993) could observe post-larval cod in July with a standard length between 25.2 mm in the spawning areas and up to 37 mm offshore of the spawning areas. About 10-40% of the total larval abundance is transported to the west coast of Svalbard with the West Spitsbergen Current (WSC) while the major part (60-90 %) is drifting with the North Atlantic Current (NAC) and is transported to the Barents Sea (Ottersen et al. 1998). A recent modeling study

has shown that the vertical placement of larvae and pelagic juveniles has a significant impact on their body weight and horizontal distribution (Vikebø et al. 2005).

The bottom-settlement of juveniles is known to occur from September to October (Ottersen et al. 2014). From there on the juveniles can be assigned to age class 0+. Typically age class 0+ - 2+ remain in the settlement area and might only perform limited seasonal migrations (Ottersen et al. 1998, Woodhead 1959). Specimens at age class 3+ typically start with migrations in the direction of their later spawning habitats at the west coast of Norway (Ottersen et al. 1998). All Atlantic cod specimens at the Svalbard archipelago and its associated fjord systems are described in the literature as NEAC (Brander 2005).

The aim of this study is to report further details about the life-history of Atlantic cod in the shallow-water zone of Kongsfjorden. The data for this study were assessed during the quantitative inventory of the shallow water zone as described in Brand & Fischer (2016). Otolith-based age determination was used to identify age-length relationships. Thereby, temporal distribution of different age classes, as well as growth rates in different years and seasons, are shown. Furthermore, we present results of stomach content analysis to show potential food sources of this fish population. Hereby, we evaluate if Kongsfjorden is a suitable habitat for the foraging and growth of juvenile Atlantic cod.

Since the Arctic coastal ecosystem is assumed to face dramatic changes during the next decades due to rising seawater temperatures and the establishment of non-Arctic species (Fossheim et al. 2015), these data may provide a valuable snapshot for comparison with past and future studies.

## Material and Methods

### Sampling

Sampling was conducted in the) in the years 2012, 2013, and 2014. Per year two sampling campaigns were conducted respectively in June and September. In June 2012, the sampling started with two locations, one at the southern shoreline (Fig. 1, OPC = Old Pier Central) and one on the shoreline of Blomstrand island (HnS = Hansneset South). At each site, one fyke net (diameter 40 cm, length 90 cm, mesh-size 12 mm (bar mesh)) was deployed in 3 m water depth together with a trammel net (inner/outer mesh size 1/15 cm, length 20 m, height 2 m) in 5-12 m water depth. Deployment was performed perpendicular to the shoreline. The deployment aimed to be continuous with recovery every 24h during the sampling campaign. The 24h interval was extended to up to 48 hours if bad weather conditions did not allow recovery.

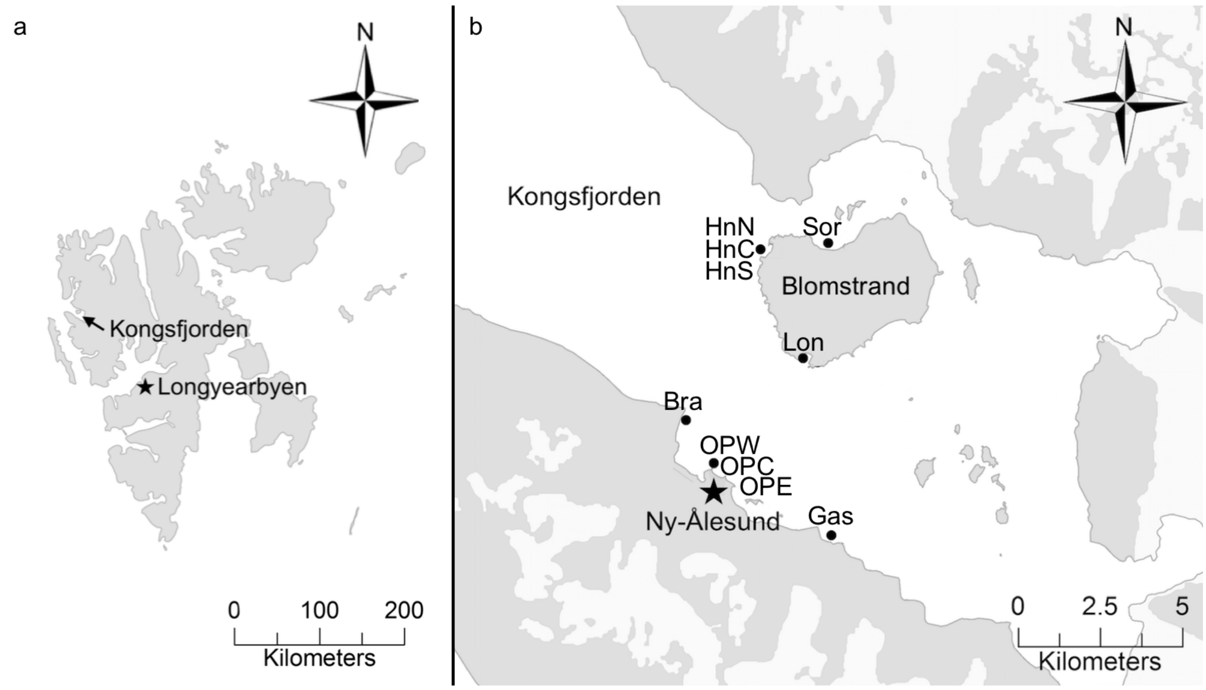


**Fig. 1 - Map of Kongsfjorden.** (a) the Svalbard archipelago with its primary settlement Longyearbyen the study site, Kongsfjorden. (b) Kongsfjorden with the settlement Ny-Ålesund and its island Blomstrandhalvøya. Light areas on land represent glacier surfaces. The sampling sites are marked as follows: Sor - Sørvågen, HnN - Hansneset North, HnC - Hansneset Central, HnS - Hansneset South, Lon - London, Bra - Brandal, OPE - Old Pier East, OPC - Old Pier Central, OPW - Old Pier West, Gas - Gåsebu. At the locations Hansneset and Old Pier, three sampling sites were spaced 100 m apart in a perpendicular orientation to the coastline. The map data was provided by the Norwegian Polar Institute (from Brand & Fischer 2016).

The September 2012 campaign, however, showed problematic interactions between young seals and the trammel nets. To avoid harm to wildlife, the use of trammel nets was stopped and fyke nets were used exclusively. The new configuration comprised the mentioned fyke net in 3 m water depth, complemented by a double fyke net (diameter 60 cm, length 110 cm, mesh-size 12 mm (bar mesh)) in 5-8 m water depth and a third fyke net (diameter 40 cm, length 90 cm, mesh-size 12 mm (bar mesh)) in 12 m water depth. To enhance the efficiency of the fyke nets, each net was baited with fish tissue and the fyke nets were connected by an 80 cm high steering net (18 mm bar mesh) perpendicular to the shoreline. This set of nets was the new standard configuration and was used for all further sampling. Species-level identification of gadoid specimens was performed on the basis of morphological traits using the methods proposed by Hayward and Ryland (2009) and Klekowski & Węsławski (1990). The primary features for distinction were the structure of the lateral line, the coloration of the ventral side, and the protruding upper or lower jaw. In the laboratory, all sampled fish were measured for standard length (SL) and wet weight (WW). Integrated overall samplings, a total of 730 Atlantic cod were caught. For a listing of all other species that were caught in the campaigns of 2012 and 2013, see Brand & Fischer (2016). In the lab, the sagittal otoliths (left and right) were extracted, cleaned in distilled water, and stored dry for later analysis. Furthermore, stomach content samples were taken.
After evaluation of the sampling campaigns of 2012, the sampling campaigns of 2013 and 2014 were extended to 5 sites at Blomstrand island and 5 sites on the south shore (Fig. 1). The exposure time of the nets was planned to be by standard 24 h. Due to logistical and weather constraints, the exposure was extended up to a maximum of 96 h. Due to a generally low saturation of the fyke nets, this extended exposure time was deemed feasible. As a metric for fish abundance, catch per unit effort (CPUE) is used to normalize fish catch against the different exposure times. The CPUE represents the number of fish per net per 24 h exposure time. No effect of the different exposure times on the CPUE could be detected by previous analysis in Brand & Fischer (2016). All quantitative analysis in this study is using CPUE values of the years 2013 and 2014, where identical sampling strategies and gear was used (App. Tab. 1). For comparison of CPUE among years, seasons, and sampling sites an ANOVA based on rank-converted data was used (Bortz 1985). A post hoc TUKEY-HSD test was utilized for further analysis of the results (Tukey 1949). Data of 2012 is only used for qualitative analysis, due to the differences in sampling strategy. The analysis of length-frequency distribution of samples from 2012, 2013, and 2014 gives no reason to assume that the use of different sampling gears affected the sampling in regard to standard length distribution (Fig. 2).

**Otolith analysis**

We performed otolith structure analysis on sagittal otoliths to be able to assign age classes to standard lengths reliably. For this purpose, we chose the full set of otoliths from our two sampling campaigns in 2013, as this campaign showed the widest spectrum in standard lengths. All otoliths sampled in 2013 were chosen for processing, independent of the standard length of the specimens. An overview of the total sample size and analyzed samples is given in the appendix in Tab. 2. By standard, the left sagittal otolith was used for analysis. If it was deformed, missing, or unusable, the right otolith was used instead. For processing, the otoliths were embedded in epoxy resin and ground laterally to the core as described in Stevenson and Campana (1992). Two independent readers determined the age of the fish using a binocular microscope (Zeiss Stereo Discovery V8 with a magnification of 8x) using reflected light on a dark field. After completion of the double-blind analysis by two independent readers, the datasets were checked and compared for differences in age determination. Samples with differences were read a second time. If the mismatch was persistent the sample was excluded from further analysis. The results are visualized in Fig. 3.

**Age length keys**

Specimens that were not aged by otolith microstructure analysis were assigned to an age class according to an age length key (ALK). The key is based on the performed otolith structure analysis was calculated with steps to the full centimeter of standard length. Due to the fish growth between June and September separate ALK were calculated for these two seasons of 2013. If overlaps between age classes occurred at certain standard lengths the specimens were distributed proportionally to one of two age classes. The full method is presented in (Ogle, 2016). As processing software R (R Development Core Team 2014) was used in accordance with the presented method. As the standard length-frequency are comparable in between all June samplings campaigns, and all September sampling campaigns of 2012-2014 (Fig. 2), the respective ALK was also applied to the specimens of 2012 and 2013.

**Growth rates**

For the calculation of intra- (June to September) and interannual (September to June) growth rates hypothetical cohorts of fish were tracked over multiple sampling campaigns. The data therefore are taken from the average SL per age class and sampling campaign. The number of days between the middle of successive samplings campaigns was determined (App. Tab. 3) and the change in SL per age class normalized to growth per day. s an additional growth parameter, the global length to weight relationship was calculated. Therefore, all specimens (N = 725) were pooled, and a regression applied (Fig. 7).

**Food sources**

The stomach content of specimens was sampled and stored in Formalin (4 %). A subset of 47 stomach content samples of the campaigns in 2013 was analyzed for the presence of different food items. A determination of the items to the lowest possible taxonomic level was executed by an expert taxonomist by eye. We processed the resulting data to show the presence of certain categories of food per sampling season and age class. The food items are grouped in the categories “benthic”, “demersal”, “pelagic” and “fish tissue”. The two most common items per group are shown in Table 4, while the remaining items per category are shown cumulatively.

## Results

### Comparison of spatial and temporal differences in species abundance

In between the sampling campaigns of 2013 and 2014, no significant difference in regard to the overall catch per unit effort (CPUE) could be detected (Tukey-HSD, $diff_{(2014-2013)} = 2.58$, $p = 0.35$). Also no significant difference in CPUE between the sampling sites at the South shore and the sampling sites along the shoreline of Blomstrand could be detected (Tukey-HSD, $diff_{(Southern\ shore-Northern\ shore)} = 4.95$, $p = 0.11$). Significant differences in CPUE were detected between the sampling campaigns in June and September (Tukey-HSD, $diff_{(September-June)} = 9.8$, $p = 0.002$). The observed abundance was generally higher in September than in June (App. Tab. 1).

### Comparison of length frequency distribution

In Fig. 2 the length-frequency distribution of Atlantic cod (*Gadus morhua*) caught from 2012 to 2014 is illustrated. In June 2012, 2013 and 2014 peaks in frequency distribution are recognizable around 12 cm standard length (SL). In September 2012, 2013, and 2014 these peaks are also recognizable at around 17 cm SL. In June 2013 a second peak around 20 cm SL is observable, which can also be found in September 2013 at approx. 23 cm SL. Exclusively in the September sampling campaigns, a peak at approx. 8 cm SL can be observed.

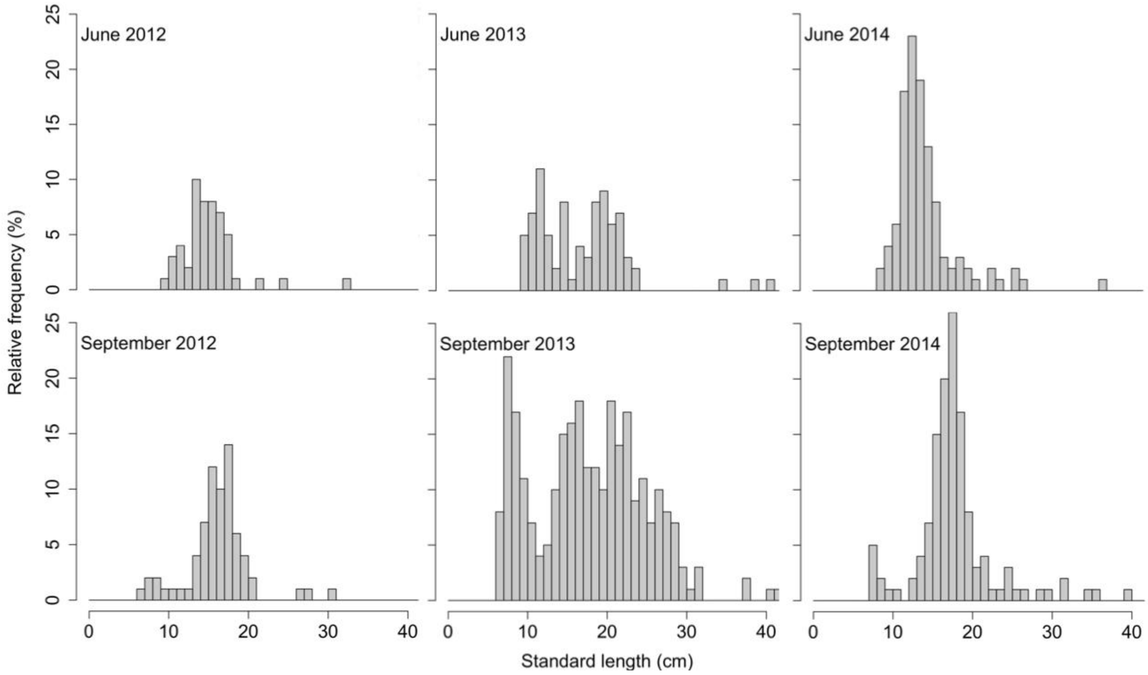


**Fig. 2 - Length frequency distribution in percent per sampling campaign**. The graphs are grouped per sampling season (upper panels = June, lower panels = September) and sampling years (left panels = 2012, middle panels = 2013, right panels = 2014).

**Results of age class determination by otolith analysis**

For June 2013 a total of 58 specimens were analyzed regarding otolith structure. The lowest detected age class was 1+ (n=30, 12.0 cm SL ± 1.8 cm SD). The second age class in significant numbers was 2+ (n=25, 20.1 cm SL ± 1.7 cm SD). Furthermore, a small number of specimens of age class 4+ were detected (n=3, 38.2 cm SL ± 0.5 cm SD). For September 2013 a total of 94 samples were analyzed. The lowest detected age class was 0+ (n=40, 8.6 cm SL ± 1.4 cm SD), followed by age class 1+ (n=29, 15.8 cm SL ± 2.4 cm SD) and age class 2+ (n=18, 24.9 cm SL ± 3.2 cm SD). Low numbers of higher age classes were additionally detected. Those were age class 4+ (n=4, 39.6 cm SL ± 2.2 cm SD), age class 5+ (n=2, 48.8 cm SL ± 5.3 cm SD) and age class 8+ (n=1, 82.0 cm SL).

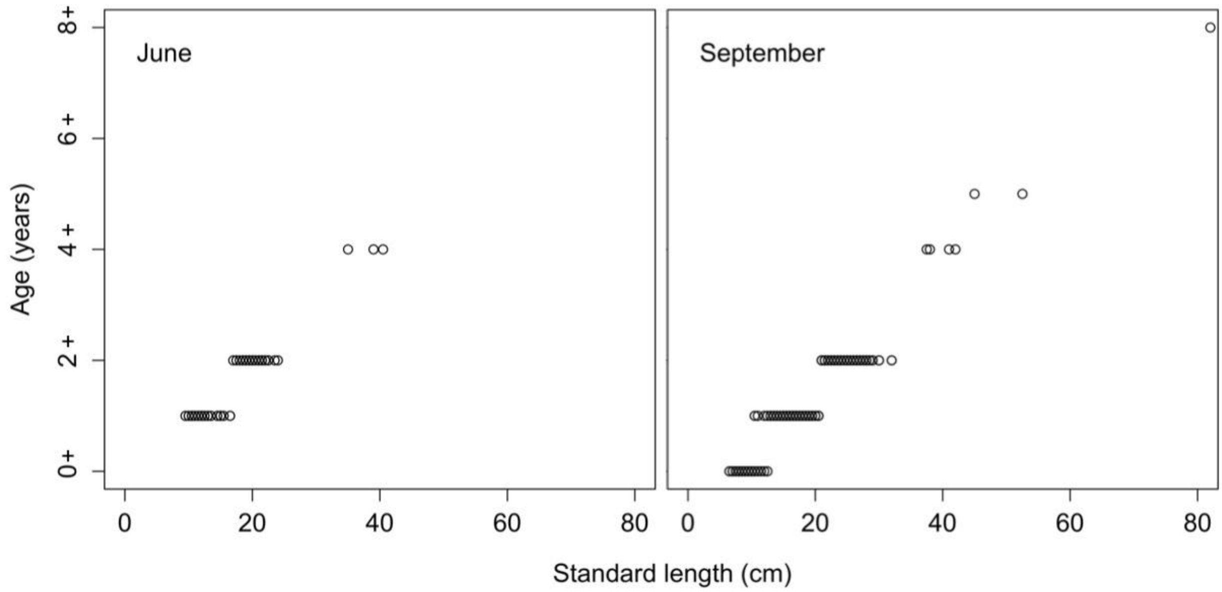


**Fig. 3 - Age length relationship based on otolith analysis.** Every mark represents one specimen. Left image: Specimens from June 2013 (n = 58). Right image: Specimens from September 2013 (n = 94).

**Comparison of age class distribution in between all sampling campaigns**

The application of the calculated age length keys (ALK) to the full data set of all Atlantic cod (N = 725), allows a comparison of age class distribution between all sampling campaigns. It shows that age class 1+ represents the dominant fraction of all specimens in all sampling campaigns (Fig. 4). Furthermore, it is common between all sampling campaigns that age class 0+ could only be detected in the September sampling campaigns. Specimens of age classes > 2+ represent over all campaigns a total of 3.6 %. In comparison of the June sampling campaigns, it is noteworthy that the share of age class 2+ specimens in 2013 (46.43 %) is clearly elevated in comparison to 2012 (23.08 %) and 2014 (13.51 %). This observation is also persistent for the September sampling campaign of 2013 where age class 2+ specimens represent 35.82 %, a distinctively higher amount than in 2012 (4.17 %) and 2014 (14.52 %). The September campaign of 2013 also shows the highest share of age class 0+ specimen (23.40 %) which is more than double the amount of 2012 (9.72 %) and 2014 (7.26 %).

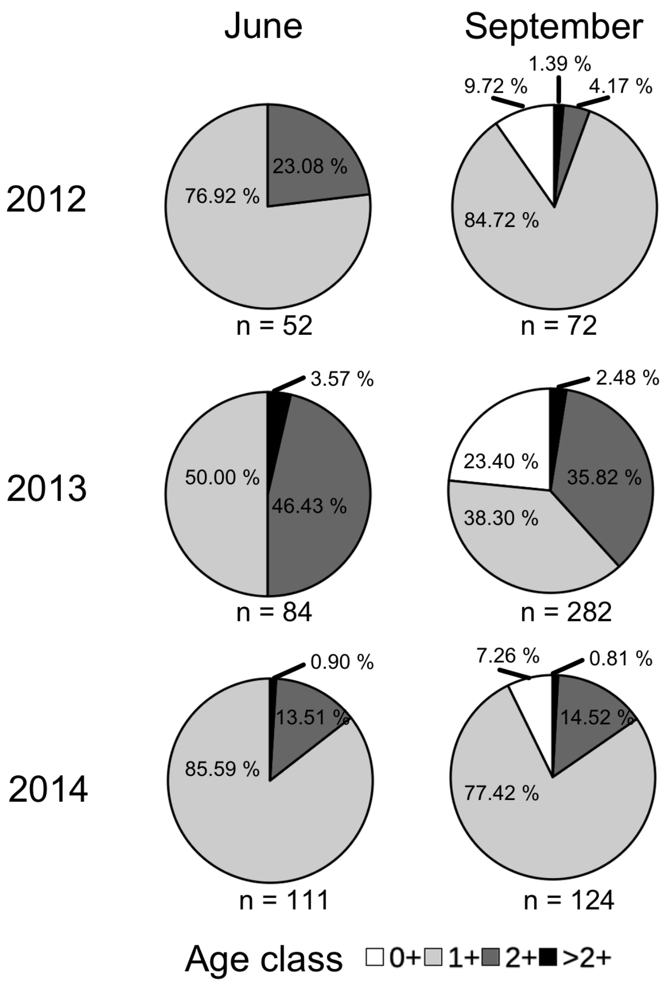


**Fig. 4 Share of age classes per sampling campaign.** The figure is based on the application of age length keys to specimens without age determination by otolith microstructure analysis.

**Comparison of standard length per age class in between sampling years**

A comparison of age at length data from otolith analysis and the application of the ALK shows no major deviations in between both techniques (Fig. 5). Fish of age class 1+ in June 2013 show by otolith analysis an SL of 12 ± 1.8 cm SD, while the ALK gives an SL 12.2 ± 2 cm SD. The same ALK applied to 2012 and 2014 results in 15.4 cm SL ± 1.94 cm SD, respectively 13.01 cm SL ± 1.7 cm SD.

The same in age class 2+ shows for otolith analysis 20.1 cm SL ± 1.7 cm SD, and by application of ALK for 2013 16.8 cm SL ± 3.7 cm SD. The respective SL for 2012 is 21.6 ± 0.8 cm SD and for 2014 21.1 ± 0.8 cm SD. A similar variation in SL derived from otolith analysis and ALK is detected for the September sampling campaigns.


**Fig. 5 Length at age class for the dominant age classes 0+, 1+, and 2+.** The values in the center are based on otolith analysis, values on the left (June campaigns) and right (September campaigns) are based on the age length keys.

**Fish growth per cohort and between seasons**

The average SL per sampling campaign and age class can be used to track hypothetical cohorts of fish (based on their year of spawning) over multiple sampling seasons. The cohort of 2011 could be tracked over 4 sampling campaigns from age class 1+ up to age class 2+. The cohort of 2012 from age class 0+ to 2+, and the cohort of 2013 from age class 0+ to 1+ (Fig. 6). Linear regression reveals an average growth rate of 0.206 mm SL/d for the 2011 cohort, 0.217 mm SL/d for the 2012 cohort, and 0.211 mm SL/d for the 2013 cohort. By differentiation between growth over the summer months (June to September) and growth over the winter months (September to June) differences in growth speed were revealed (App. Tab. 3). The growth in summer months was higher (n=5, 0.37 - 0.70 mm SL/d) than in winter months (n=4, 0.12 - 0.16 mm SL/d). The length-weight relationship of Atlantic cod (N = 725, 5.5 to 82.0 cm SL) was determined as $W_{(wet)} = 0.007379 * L_{(sd)}^{3.145}$ ($r^2 = 0.9912$; Fig. 7).

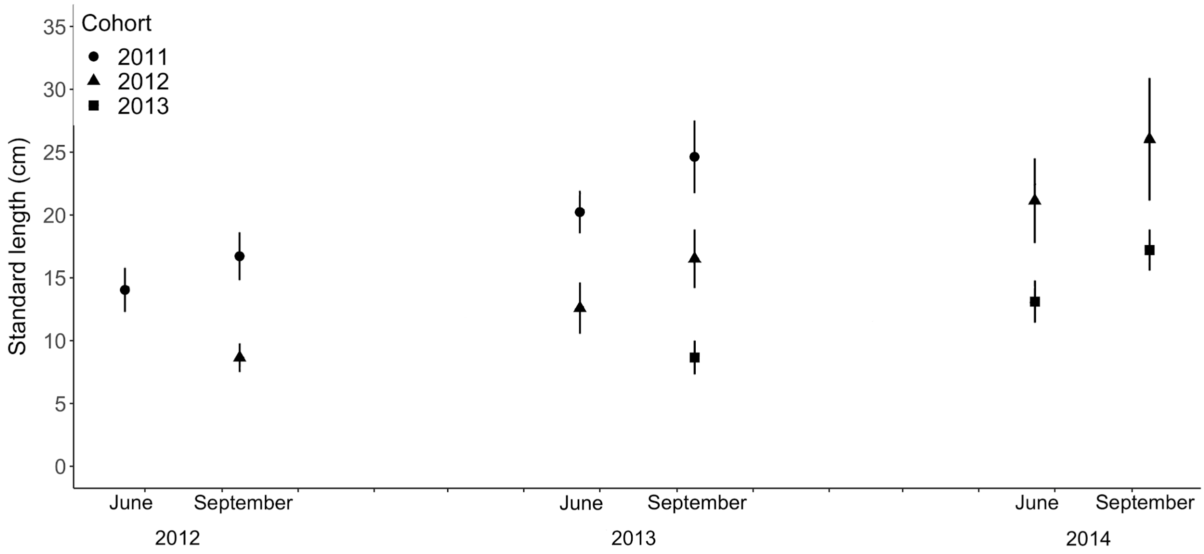


**Fig. 6 Growth of the three cohorts** of specimen, spawned in 2011, 2012 and 2013 over the following years. The growth rate of the cohort 2011 is given by y = 0.0206x - 830.2 ($r^2$ = 0.918), of 2012 by y = 0.0217x - 883.3 ($r^2$ = 0.933) and of 2013 by y = 0.0211x - 867.2 ($r^2$ = 0.927).

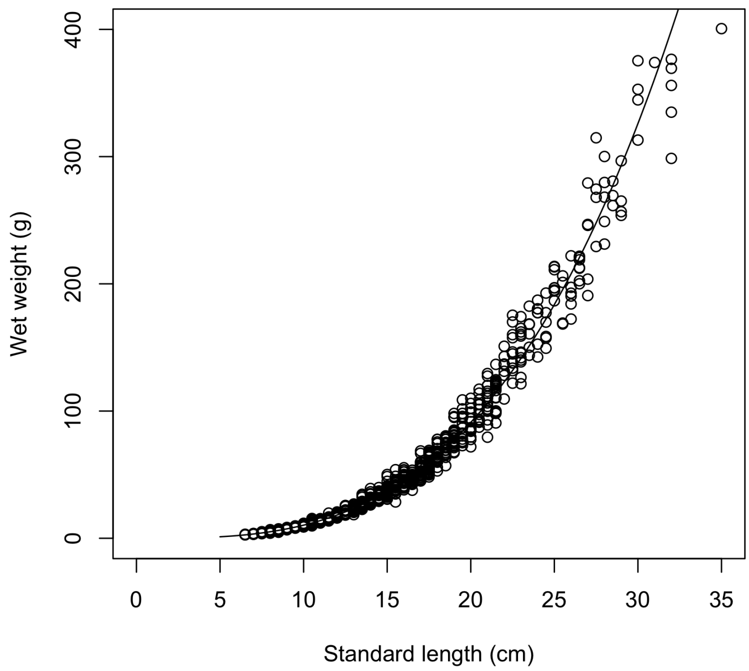


**Fig. 7 Wet-weight to standard length relationship** based on all Atlantic cod sample in this study (N = 730, $r^2$ = 0.9912)

**Stomach content analysis**

Stomach content analysis of 47 samples revealed a total of 35 different types of food items. These food items were categorized into benthic organisms (n = 14), demersal organisms (n = 14), pelagic organisms (n = 4), and the singular item fish tissue. In 97.9 % of all samples, amphipods were present with *Ischyrocerus spp* and *Anonyx sarsi* in the highest frequency. Further benthic items were contained in 66 % of all samples whereas *Caprella septentrionalis* and Harpacticoida were most abundant. Prey of the category pelagic was present in 29.8 % of all samples *Calanus spp.* and *Thysanoessa inermis* were represented in highest frequency. Fish tissue was in 8.5 % of all samples (App. Tab. 4). A more detailed view reveals that fish was only found in September and only in age class 1+ (14.3 %) and age class 2+ (37.5 %). Items of the category benthic and demersal are found in all age classes and at all sampling campaigns.

**Discussion**

This study shows that the shallow water zone of Kongsfjorden is dominated by Atlantic cod of age class 0+,1+, and 2+. For the age class determination in this study, we otolith analysis, which has established itself as one of the standard tools in fish ecology. It provided us the possibility to create age length keys (ALK) to assign an age class to specimens without otolith samples. Due to the differences in length class distribution (Fig. 2) we decided that separate keys for the June and September sampling campaigns were required, as the maxima in standard length (SL) distribution were shifted in between the sampling campaigns, due to fish growth within the different age classes. In between the June and in between the September sampling campaigns of the different sampling years no significant shifts in the maxima of the SL distribution are recognizable. This indicates no significant differences in fish growth and thereby age length relationship (Fig. 2). Thereby, an application of the ALK from June and September 2013 on the corresponding sampling campaigns of 2012 and 2014 is . This us to increase the sample size for the following age-based analysis.

A factor that likely has an influence on the data quality of this study is gear selectivity, as every fishing gear has selectivity. We chose trammel nets and later fyke nets with 12 mm bar mesh size. By keeping the fyke net mesh size constant in all sampling campaigns, we minimized differences in qualitative sampling results. The comparison of SL distribution between 2012, 2013, and 2014 shows no qualitative difference regarding (Fig. 2). As both sampling gears had a bar mesh size of 12 mm it is likely that specimens of Atlantic cod with a height of less than 12 mm were under sampled. The smallest sampled specimens in this study show a body height of around 10 mm with an SL of 6.5 cm. In Mark (2013) the presence of Atlantic cod from 5.5 to 9.5 cm SL is shown for Forlandsundet and the mouth of Kongsfjorden in August 2013. Also, the presence of Gadidae from 4.0 to 10.0 cm SL is reported at the Old Pier in Ny-Ålesund for August 2014 (Fischer et al. 2017). It is thereby possible that age class 0+ specimens with body heights of less than 12 mm and an SL of less than 7 cm are underrepresented in this study. In consequence, also the average SL shown for age class 0+ specimens is likely elevated, as the smallest specimens might not have been sampled. As the gear selectivity has a systematic character the comparative aspects within this study are not affected. By year-round observation via the Kongsfjorden underwater observatory no specimens >6.5 cm SL could be detected before August (Fischer et al. 2017). The absence of age class 0+ specimen in all June sampling campaigns in this study is thereby unlikely an artifact of gear selectivity, but rather caused by migration of specimens towards the shallow water zone. Based on the reported spawning period of Arctic Cod (NEAC) from February to early May (Brander 2005), and a bottom-settlement at an age of 5-6 months (Ottersen et al. 2014), it seems plausible that the observed age class 0+ specimen origin from the spawning grounds of NEAC.

The stomach content analysis in this study shows that pelagic food sources (primarily *Calanus spp.* and *Thysanoessa inermis)* were found than benthic and demersal food resources as e.g. Amphipods (App. Tab. 4). This supports the assumption that Atlantic cod is using kelp forests and subtidal soft bottoms as nursery areas, as reviewed in Seitz et al. (2014). Hereby, the kelp forests in the depth strata between 2.5 m down to 15 m (Bartsch et al. 2016), might fulfill a dual function. It can provide a feeding ground for benthic organisms that are prey for Atlantic cod (Norderhaug et al. 2005). For Isfjorden Renaud et al. (2015) showed that most taxa of the benthos feed on a broad mixture of particulate organic matter (POM) and macroalgal detritus. During the polar night, the infauna of the decaying kelp beds of Kongsfjorden might thereby be an important energy and food resource for Atlantic cod. This coincides with the observation of Berge et al. (2015b) of feeding activity by Atlantic cod and a high abundance of fauna associated with *Saccharina latissima* during the polar night. This supports the hypothesis that the polar night is not a time of biological quiescence (Berge et al 2015a). However it can be expected that due to lower water temperatures during the polar night, the growth rate is reduced in comparison to the polar day. Its potential second function was shown in an experimental approach in Gotceitas et al. (1995). that juvenile Atlantic cod use kelp forests as structural protection to avoid active predators. Fish size was noticed as an important factor, because if fish exceed a certain size, it hinders their roaming through the kelp forest. Depending on the structure and density of the kelp forest this might facilitate an age class separation as we see in the current study. Here we see that age class 0+ to 2+ as dominant fractions in the shallow water zone. Specimens of age classes >2+ were sampled in very low abundance, this indicates that those specimens might have shifted their habitat afterwards. This is in accordance with the report of Ottersen et al. (1998) that after settlement the fish do not undertake large seasonal movements in their first two years. After this period, it is reported for Atlantic cod in the Barents Sea that an onsetting horizontal migration movement is connected toward a shift to a fish-based diet apelin (*Mallotus villosus*) is one of the primary food sources (Brander 2005). In the study, it was noticeable that specimens of age class 4+ and above were sampled only in the year 2013 (Fig. 2+3). Also, the age class 2+ was stronger represented in 2013 (Fig. 4). The reason is unclear, one potential explanation might be differences in the hydrographic regime between 2013 and the other sampling years. We analyzed data from the AWIPEV underwater observatory located at 11 m water depth at the sampling site “Old Pier Central” (Fischer et al. 2017). It showed that the water temperature in the littoral zone was not significantly different between the years 2012, 2013 and 2014 (Tukey-HSD, $diff_{(2013-2012)} = 0.46$, $p = 0.81$; $diff_{(2014-2012)} = 1.36$, $p = 0.24$; $diff_{(2014-2013)} = 0.91$, $p = 0.49$; data from (Fischer et al. 2018a, b, c). In contrast the subsurface waters (SSW) of Kongsfjorden are reported by Payne & Roesler (2019) to have shown lower temperatures in 2013 than in 2012 and 2014.

As Atlantic cod prefers higher temperatures at higher age classes (Nakken & Raknes 1987), the colder temperatures in the SSW of the fjord in 2013 might have resulted in avoidance by older specimens. The result might have been the mixing of different age classes in the shallow-water zone (Fig. 2). However, the presence of age classes >4+ in Kongsfjorden indicates the possibility that those specimens might not undertake horizontal migrations as the specimens in the Barents Sea. It suggests a rather vertical separation between different age classes with adult Atlantic cod at the bottom, as reported by Mark (2013) at Forlandsundet and the mouth of Kongsfjorden. Such a vertical separation is also known from Atlantic cod at the Norwegian Coastline, known as Norwegian Coastal Cod (NCC). This species with a non-migratory lifestyle is known shows a settlement of juvenile specimens in shallow waters of coastal areas and fjords (Løken et al. 1994). After the completion of age class 2+, specimens can be found in deeper waters of up to 500 m (Bakketeig & Bakketeig 2018). In a current study by Andrade et al. (2020) it is hypothesized that Atlantic cod of the NEAC population has established themselves in Isfjorden and Kongsfjorden. For specimens in Isfjorden it is suggested that they perform limited local movement as

NCC. Recent investigations by SNP genotyping show Atlantic cod specimens with markers characteristic for CC in Kongsfjorden (L. Spotowitz, pers. comm., 07.01.2021). An origin of these markers could be genetic introgression and admixture between NCC and NEAC, as suggested by Dahle et al. (2018). NCC and NEAC are spawning at some locations in mid and northern Norway in proximity, and eggs and larvae are thereby subject to the same process of transport and spreading (Brander 2005). An alternative explanation for genetic markers of NCC in Kongsfjorden might be that eggs and larvae of NCC are also transported via the Norway Coastal Current and West Spitsbergen Current (WSC) towards Svalbard. Over the last decade the change in hydrographic regime led to generally rising water temperatures in the area (Spielhagen et al., 2011). Especially in Kongsfjorden hydrographic conditions and sea-ice show a high inter-annual variability and increasing water temperatures as well as advection of seawater from the WSC are important abiotic factors influencing food availability (Hegseth et al. 2019; Hop et al. 2002, 2019). This might open an ecological window of opportunity for Atlantic cod to establish a permanent non-migratory population. During the last Arctic warm period from 1920-1940 a report from Iversen (1934) refers to Atlantic cod in the spawning stage at a bank of Isfjorden and around Bear sland. He also reports age class 0+ specimen at Grønfjorden on Svalbard and mentions that sporadic spawning seemed to occur close to Isfjorden and in the Bear sland area. Yet, he stressed that the biggest number of Atlantic cod in Svalbard waters had to be associated with the spawning grounds off the coast of Norway.

Unfortunately, the question about the origin, and the complete lifecycle of Atlantic cod in Kongsfjorden cannot be answered by this study. It could be shown that the shallow water zone of a fjord in the Arctic can provide a nursery and foraging habitat for Atlantic cod, enabling growth rates comparable to those in the Barents Sea as described in Brander (2005). For a better understanding of the current state of the Atlantic cod population in Svalbard waters it seems worthwhile to investigate the origins of the specimen in the fjords. For this purpose, genetic and otolith analyses can be used to gather a more detailed understanding. Furthermore, a year-round monitoring of local fish populations seems to be advisable to gather precise data regarding their temporal variability, and their reaction to fluctuations in the hydrographic regime. Automated underwater observatories with hydrographic sensors and camera systems can deliver a valuable contribution. Sampling campaigns in the shallow water and deep water zones should be coordinated for a holistic assessment. Recent research shows that the northward expansion of Atlantic cod might also affect the Greenland shelf (Strand et al. 2017). An expansion of such a research effort to waters in Greenland might give valuable early insights about this process.

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**Appendix**

**Table 1 Overview of Catch per Unit effort (CPUE) per sampling site and season.** A CPUE value of 1 represents one fish in 24 h exposure time in one fyke net.

| | | Sampling site | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Year** | **Season** | **South shore** | | | | | **Blomstrand (North)** | | | | |
| | | Brandal | Old Pier West | Old Pier Central | Old Pier East | Gasebu | London | Hansneset South | Hansneset Central | Hansneset North | Sorvagen |
| 2013 | June | 0.0325 | 0.0739 | 0.1154 | 0.0106 | 0.0146 | 0.0303 | 0.1775 | 0.1254 | 0.094 | 0.0464 |
| | September | 0.0724 | 0.2381 | 0.2552 | 0.2636 | 0.2841 | 0.2033 | 0.3012 | 0.2936 | 0.2507 | 0.1885 |
| 2014 | June | 0.0484 | 0.4355 | 0.1690 | 0.4194 | 0.3069 | 0.0969 | 0.0969 | 0.0339 | 0 | 0.1777 |
| | September | 0.5217 | 0.3745 | 0.1250 | 0.0971 | 0.4444 | 0.3785 | 0.0648 | 0.0195 | 0.0542 | 0.2381 |

**Table 2 Overview of otolith analysis** in regard to samples per season and results of otolith analysis.

| Year | Season | Total number of specimen | Specimen with sampled otoliths | Successfully analyzed otoliths | Number of specimens by age-class, based on otolith analysis | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | 0+ | 1+ | 2+ | 3+ | 4+ | 5+ | 6+ | 7+ | 8+ |
| 2012 | June | 52 | 52 | - | - | - | - | - | - | - | - | - | - |
| | September | 72 | 71 | - | - | - | - | - | - | | - | - | - |
| 2013 | June | 84 | 65 | 58 | 0 | 30 | 25 | 0 | 3 | 0 | 0 | 0 | 0 |
| | September | 282 | 109 | 94 | 40 | 29 | 18 | 0 | 4 | 2 | 0 | 0 | 1 |
| 2014 | June | 111 | 82 | - | - | - | - | - | - | - | - | - | - |
| | September | 129 | 124 | - | - | - | - | - | - | - | - | - | - |

**Table 3 Growth rate per season and age class** in mm SL/day.

| Time intervall | | | Growth rate in mm SL/d | | |
|---|---|---|---|---|---|
| **Start** | **End** | **Duration (d)** | **Age class 0+** | **Age class 1+** | **Age class 2+** |
| June 2012 | Sept. 2012 | 75 | NA | 0.37 | NA |
| Sept. 2012 | June 2013 | 286 | 0.13 | 0.12 | NA |
| June 2013 | Sept. 2013 | 75 | NA | 0.53 | 0.60 |
| Sept. 2013 | June 2014 | 288 | 0.16 | 0.13 | NA |
| June 2014 | Sept. 2014 | 81 | NA | 0.49 | 0.70 |

**Table 4 Results of stomach content analysis per age class and season.** Percentage of stomach contents that contain given food item.

| | Atlantic cod | | Benthic (%) | | | Amphipods (%) | | | Pelagic (%) | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Season** | **Age class** | **No. of specimen** | ***Caprella septentrionalis*** | ***Harpacticoida* n. det.** | **Other** | ***Ischyrocerus spp*** | ***Anonyx sarsi*** | ***Amphipoda spp*** | ***Calanus spp* remains** | ***Thysanoessa inermis*** | **Other** | **Fish tissue** |
| | 0+ | - | - | - | - | - | - | - | - | - | - | - |
| June | 1+ | 10 | 10.0 | 0.0 | 20.0 | 70.0 | 0 | 60.0 | 10.0 | 20.0 | 0.0 | 0 |
| | 2+ | 10 | 10.0 | 20.0 | 60.0 | 40.0 | 0 | 60.0 | 10.0 | 10.0 | 10.0 | 0 |
| | 0+ | 12 | 33.3 | 25.0 | 16.7 | 16.7 | 0.0 | 50.0 | 16.7 | 0.0 | 0.0 | 0 |
| September | 1+ | 7 | 28.6 | 42.9 | 42.9 | 14.3 | 14.3 | 85.7 | 0.0 | 14.3 | 0.0 | 14.3 |
| | 2+ | 8 | 12.5 | 0 | 12.5 | 12.5 | 25.0 | 50.0 | 0.0 | 12.5 | 12.5 | 37.5 |
| | **All** | **47** | **19.1** | **17.0** | **29.8** | **31.9** | **6.4** | **59.6** | **10.6** | **14.9** | **4.3** | **8.5** |
| | | | | **66.0** | | | **97.9** | | | **29.8** | | **8.5** |

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## Declarations

### Conflict of Interest Statement

The authors declare that the submitted work was carried out in the absence of any personal, professional or financial relationships that could potentially be construed as a conflict of interest.

### Ethics approval

The local regulations related to fauna harvest on Svalbard exclude saltwater fish, except for salmonids. Additionally, an application for the research project submitted to the local governor (Sysselmannen) was approved by his authority in accordance with local legislation. All work was performed in accordance with the act on animal welfare and by accepted research methods. We were also in contact with NARA (Norwegian Animal Research Authority) and they confirmed that for our sampling no additional approval had to be applied for.

### Consent to participate

Not applicable

### Consent for publication

Not applicable

### Availability of data and material

Not applicable

### Code availability

Not applicable

### Author Contributions Statement

The idea, design, and execution of this study were performed by MB with advice from PF. LS prepared otoliths for analysis and acted as a second reader in the analysis. JMW performed the analysis of stomach content samples. The manuscript was written by MB, FCM, and BJ added advice for the interpretation of the data. MB wrote the manuscript, LS and PF assisted in the revision of the manuscript.

# 4. Summary & Discussion

This study demonstrates that the combination of a state-of-the-art underwater observatory and traditional fish sampling campaigns can be a highly effective tool in the assessment of a complex, remote, and hard-to-assess ecosystem. In the first publication of this thesis (Brand & Fischer 2016), the results of a first-time quantitative study of the fish composition in the shallow-water zone of Kongsfjorden are presented. This initial classic fish survey was performed from 2012 to 2013 mainly with fyke-nets in water depth between 3 to 12 m at ten different sampling sites in central Kongsfjorden. Five sites were situated along the southern shoreline and five on the northern part at Blomstrandhalvøya. The study aimed to provide primary ground-truthing fishery data for the actual situation of the shallow-water fish community. It forms, together with literature data, an initial approach for a better quantitative understanding of the shallow-water fish community in this Arctic area. Using fyke-nets, we sampled a total of 2804 specimens and identified 12 species plus one family (*Liparidae*) of fish. As the dominant species across all sampling sites, we determined shorthorn sculpin (*Myoxocephalus scorpius)*, representing 74.9 % of all caught specimens. The second most abundant specimen was Atlantic cod (*Gadus morhua,* 17.2 %) and after that Arctic staghorn sculpin (*Gymnocanthus tricuspis*, 3.8 %). The dominance of two boreal species and the comparatively low numbers of Arctic species shows that the shallow-water zone of central Kongsfjorden is clearly dominated by boreal species. In parallel to our study further areas of the fjord were sampled by trawling for other studies. Fey & Węsławski (2017) report in bottom trawl samples in inner Kongsfjorden significant numbers of polar cod (*Boreogadus saida).* Also, Berge et al. (2015) were able to sample polar cod from 2013 to 2015 during the polar night. This year-round presence of this Arctic species in Kongsfjorden indicates a clear separation between different compartments/habitats in the fjord, potentially based on hydrographic aspects. Polar cod is known to be pelagic and sea-ice associated. The sampling of specimens close to the Kongsbreen glacier by Fey & Węsławski (2017) would fit into the concept that those fish prefer cold and low saline water masses. Additionally, it is known that these zones are nutrient and thereby zooplankton rich.

A multivariate analysis (MDS analysis) of species-specific fish abundance was performed to analyze the similarities/dissimilarities between the sampling sites with respect to its fish communities (Brand & Fischer 2016). The analysis revealed a grouping into three significant clusters (Fig. 3) of sampling sites which correspond mostly to the counterclockwise water mass movement in Kongsfjorden (Willis et al. 2006). Cluster 1 includes the single sampling site Brandal at the south shore. This site is the south-westernmost station in the fjord, located closest to the inlet. Water masses that enter Kongsfjorden pass by this station first. All sites in central Kongsfjorden could be located in cluster 2, except the three westernmost sampling sites at the northern shoreline around Blomstrandhalvøya. The sites in cluster 2 are characterized by soft bottom substratum with only low slopes mixed with larger stones and intermittent *Laminaria* forests. Along with those sites in central Kongsfjorden, the water masses are mixing with glacial runoff from the shore and tidewater glaciers. Cluster 3, and thereby the westernmost sites at Blomstrandhalvøya is located on the exposed west shore of Blomstrand island. It is characterized by rock, partially in the form of steep slopes. Those rocks are overgrown with macroalgae (Bartsch et al. 2016). According to the counterclockwise circulation patterns, as suggested by Willis et al. (2006), water masses exit central Kongsfjorden at this point and flow fjord outward along the north shore.

This analysis furthermore revealed that, with respect to fish abundance and species composition, the fish community close to the position of the underwater observatory is similar to the shallow-water sites in central Kongsfjorden (Cluster 2). The underwater observatory is positioned in front of the Old Pier in Ny-Ålesund at 11 m of water depth. It enabled the continuous year-round assessment of the fish community together with the main hydrographic parameters at the site. After the initial prototype operation from June 2012 to September 2013, the first continuous year-round operation could be achieved from October 2013 to November 2014. During this operation temperatures between -0.5 °C in December 2013 up to 7.7 °C in May 2014 were recorded (Fischer et al. 2017).

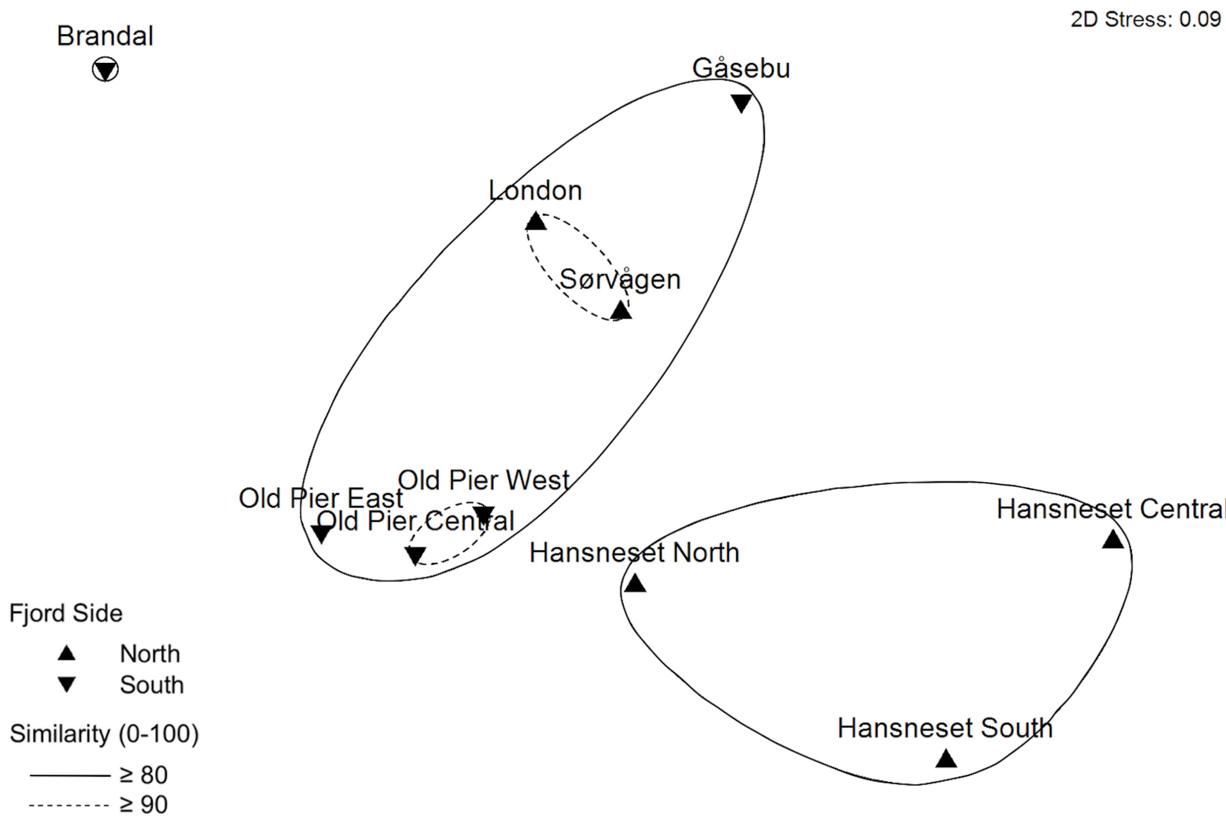


**Fig. 3 - Multivariate analysis (MDS)** of total catch per fish species per sampling site, based on square-root transformed data and Bray-Curtis dissimilarity index. Hierarchical cluster analysis was used to identify sites with a similarity of > 80 and > 90 of 100 - from Brand & Fischer (2016).

In contrast to the fyke-net sampling campaigns, the *in situ* optical sampling classified 81 % of all fish as Gadidae. Due to the results of the fyke net samplings, it can be assumed that those were mostly Atlantic cod and polar cod.

At the underwater observatory, the first Gadidae were detected in October 2013 with standard lengths between 6 and 12 cm, which corresponds to the lower size spectrum of *Gadus morhua* assessed by the classical fishing methods described in Brand & Fischer (2016). While the observation period of the fishing campaigns was limited, the use of the stereo-optic allowed to track this cohort of fish during the winter months towards spring. In this period, a steady increase in standard length could be shown. The observation ended in March 2014, when the specimen reached a total standard length between 9 and 16 cm. The system detected the next age-0 cohort of cod specimens in August 2014, with standard lengths between 3 and 9 cm.

The permanent presence of juvenile Atlantic cod indicates that the shallow-water habitat is a refuge area for juvenile fish, as also demonstrated for other ecosystems (Ruiz et al. 1993). The absence of specimens of age class 3+ also indicates that fish of this age class leaves the shallow area. This can be connected to size-dependent prey-predator interaction, and also due to a diet shift from invertebrates to fish (Dalpadado & Bogstad 2004). The integration of the shallow-water zones in future research is important to assess the holistic life cycle of species like Atlantic cod. A future repetitive study at all depth strata of the fjord system has the potential to assess the full extent of ongoing climate-induced ecosystem change.

One of the central challenges of such a combined study is the comparison of the results of different sampling schemes. One of the most remarkable differences in this study is that the stereo-optical system identified 4 % of the fish as shorthorn sculpin*,* while the fyke net fishing identified 75 % as this species (Brand & Fischer 2016).

To explain these differences both sampling strategies have to be discussed. One major difference is that the bottom standing fyke nets were permanently deployed at the same position on the ground, while the stereo-optical system was sampling at a total of 5 depth strata. Of those 5 depth strata, the bottom was only one, while the other four assessed the rather pelagic realm in front of the Old Pier. This might also explain why the majority of Atlantic cod specimens detected with the camera system belonged to the 0+ or 1+ cohort. Larger specimens, as detected in low abundances by net sampling were missing (Brand & Fischer 2016). Those larger specimens are known for a demersal lifestyle and would therefore be expected close to the bottom. It seems thereby that benthic and demersal fish species are underestimated in the assessments of the camera system, and pelagic species might be overestimated. At the same time, the fyke nets have most likely underestimated pelagic fish species, and the chosen bar mesh size of 12 mm made them less effective for specimens with heights of less than 10 mm, as in example 0+ specimen of Atlantic cod.

A further factor influencing the results of net catches is catchability due to body parameters. Benthic species, with generally larger body diameter, that are moving along the bottom are more likely to be caught by bottom standing nets. Additional parameters are morphological traits as e.g. spines on the gill cover of the shorthorn sculpin that lead to easier entanglement in nets.

A further aspect might be that small-scale distribution patterns, driven by habitat characteristics, might have a significant effect. The ground in front of the underwater observatory is characterized by sand, with small algal coverage, while the vertical wall surfaces are overgrown completely by algae (Fischer et al. 2017). It is known that kelp beds have a macrofauna community (Lippert et al. 2001; Paar et al. 2019a+b), which might be an attractive food source. In Brand & Fischer (2016) it could also be shown that algal coverage had a significant effect on fish abundance. Assuming a strong habitat-fish relationship with respect to algal coverage, small scale variation of this coverage might affect the results. Therefore, the small distance between fyke nets and the observatory of approx. 20 - 30 m, might have had a significant effect on the results.

Also, the general sampling strategy of the underwater observatory and fyke nets differ in detail. The underwater observatory shot one picture pair every 1800 sec. A specimen had to be in the observation zone at this exact moment. The results of the fyke nets are less influenced by this aspect, as every specimen who entered a net was likely to remain inside. The fyke nets might be influenced over time by their content. As all specimens remain alive inside, they might act as bait and attract further predatory specimens. A total of 4375 fish specimens were sampled and 682 stomach contents analyzed. In 43 specimens fish or its fragments could be found. Of those cases, 8 stomach contents were heavier than 10 g. It is unlikely to catch specimens of smaller weight by the utilized fyke nets.

The development of a sound methodology for comparing the state-of-the-art non-invasive sampling method “stereoscopic imaging” with classic net sampling methods with respect to abundance and species selectivity deems to be an important project. Therefore, an interesting experiment might be to deploy an online stereo-optic video observatory statically close to the kelp belts. In combination with another fyke net campaign during the polar day, a complete 24 h data assessment via fyke net and optical systems might be possible. Hereby, also the mouth of a fyke net could be surveyed optically. It might furthermore reveal exciting insights into the interaction between fish and the kelp belts. It might be worthwhile to switch the assessment of the underwater cameras from time-selective to time-integrative because for time-selective sampling the choice of sampling frequency can significantly influence a study's results. The choice of a too low sampling frequency for the assessment of a natural phenomenon of higher frequency can lead to an incorrect impression of the natural phenomenon (Nyquist 1928; Shannon 1948). A way for an integrative sampling with an underwater observatory might be the recording of permanent stereoscopic video, instead of stereoscopic picture pairs. The arising challenge here would be the analysis of 24 h of video per day. Therefore, the development of new workflows for automated image processing, and especially object recognition, is necessary. Also, this methodology is only viable during the polar day. A permanent artificial illumination during the polar night might affect the assessment of fish as well as crustacean, as a phototactic reaction to visual stimuli is known for many species (Guthrie et al. 1993; Warrant et al. 2006). Recent investigations on the artificial attraction of Northern krill showed that 530 nm (green light), as well as white broadband light (450-750 nm), are equally attractive light sources. The same study also showed that those light sources have a slightly repulsive effect on Atlantic cod, but suggests that an attracted swarm of krill overcomes this repulsion (Utne-Palm et al. 2018). In consequence, a permanent illumination of a video based study might severely affect its results. In the current study, the phototactic effect was minimized by using a single light pulse of a flashlight every 30 min. In a previous study, this light exposure showed no significant effect on the abundance of fish around a stationary target (Fischer et al. 2007). A further factor that has to be considered is that an underwater observation system represents an artificial substrate. This substrate is gradually settled by sessile organisms and is potentially able to provide structural protection. In an environment with low structural complexity, this might bias the observations themselves. In an environment with high structural complexity, the effect is likely neglectable as the observation system is not standing out from the general environment. As the observation site in this study has a high general complexity this effect is likely neglectable in this study.

Despite the discussed sources of bias, the year-round data assessment revealed significant differences between summer-autumn (August-October) and winter-spring (December-March) communities in regard to total abundances and species richness. The summer-autumn community was by abundance dominated by Appendicularia in 2 - 4m water depth, while the winter-spring community was dominated by benthic crustaceans. Of those 90 % were identified as great spider crabs (*Hyas araneus*) (Fischer et al. 2017). The massed occurrence of the great spider crab started in November and ended in April, thereby it seems related to the polar night. It might be connected to food sources in the kelp belts of the shallow-water zone. Additionally, it is known that ovigerous females of the great spider crab release their larvae between late February and early April (Walther et al. 2010). This corresponds with the onset of the spring bloom in April to May (Hegseth & Tverberg 2013). This observation in the spider crab might be transferable to fish, which might also access the food resources in the kelp belt during the polar night. The continuous growth of Atlantic cod in the polar night, as presented in Brand et al. (in draft), indicates that the necessary food resources are available.

This shows that the general perception of the polar night as a period of low overall energy, and thereby low activity, seems not to be true. Berge et al. (2015) also showed that, when primary production is almost stopped during the polar night, the trophic interactions and metabolic rates remained high for most of their examined consumers also without daylight. It was concluded that those consumers are sustaining their activity on stored reserves or alternative food sources. Most specimens of Atlantic cod, which were sampled by Berge et al. (2015) during the polar night, had an at least partially filled stomach. This enables growth, also in the absence of light. Our study seems to confirm this assumption (Brand et al. in draft). It additionally shows the food sources of Atlantic cod of age-class 0+ to 2+ in the shallow water zone comprises from different ecological niches. The major fraction of organism comes from the kelp forests, with amphipods as the most abundant group. Amphipods are known to use the structured kelp forests as habitat. On the other hand, pelagic copepods (Calanus spp.) and euphausiids (*Thysanoessa inermis*) were also found in the stomach content of the specimen. Both are not known as typical inhabitants of the shallow water zone, but rather deeper zones in the water column. This leaves the option that Atlantic cod moves out of the algae belts into deeper zones of the fjord for feeding, or that copepods and euphausiids enter the shallow water zone. An explanation therefore would be a vertical migration of both species.

To provide ground-truth data for fish growth aside from the increase in standard length (SL) measured by the stereo-optical system we used otolith-microstructure-based age class analysis to calculate growth rates for Atlantic cod of age class 0+ to 2+ caught by fyke-net fishing. It could be shown that growth rates between September and June were 0.13 to 0.16 mm SL/day, while growth from June to September was in between 0.37 to 0.70 mm SL/day.

The growth rates of Atlantic cod are significantly influenced by temperature and food availability. The higher water temperatures in the summer months, in combination with good food availability, enables rapid metabolism and faster growth. The continuous growth in the winter months shows the general availability of food to sustain this process. The average standard lengths for North East Arctic Cod (NEAC) in the Barents Sea reported by Brander (2005) are 12.4 cm for age class 1+, and 19.9 cm for age class 2+. Between 2012 and 2014, we could detect in Kongsfjorden standard lengths of 12.6 - 14.0 cm for age class 1+ and 17.2 - 20.2 cm for age class 2+. This shows that the shallow-water environment of Kongsfjorden enables a similar growth regime as the Barents Sea. The determination of further parameters as e.g. mortality in further studies will allow a better description of the shallow-water zone of Kongsfjorden as a nursing ground.

The first fyke-net sampling of age class 0+ specimens of Atlantic cod was from 2012 - 2014 each in September. The specimen had an average SL of 8.5 - 8.7 ± 1.1 - 1.4 cm SD (Brand & Fischer 2016). The earliest detection of gadoid specimens of a similar size class by the underwater observatory was August (Fischer et al. 2017).

The arrival of age class 0+ specimens of Atlantic cod in either August or September is in its tempo-spatial aspect in accordance with the transport of eggs and larvae from the spawning grounds of NEAC at the Lofoten and off Møre in Norway (Godø 1984a,b). After their spawning from March to April, these specimens of NEAC are known to be transported with the Norway Coastal Current and the West Spitsbergen Current (WSC) to Svalbard. In the process of transport, the specimens are split between the west coast of Svalbard and the Barents Sea. The larvae are known to settle down to a demersal lifestyle at the end of their transport (Wienerroither et al. 2011). Another stock of Atlantic cod is spawning close to the spawning sites of the NEAC. This stock is classified as Norwegian Coastal Cod (NCC). It is reported to be a local, non-migratory cod stock that has its habitat along fjords and islands along the Norwegian coast. No large-scale horizontal migration movements are reported for this stock (Brander 2005). The spawning of this stock happens in local fjords along the Norwegian coast. Recent investigations in the population genetic structure of NCC suggests that genetic introgression and admixture between NCC and NEAC might happen (Dahle et al. 2018). NCC eggs that might drift out of those fjords might also be transported with the WSC as it happens for NEAC. Recent investigations by SNP (Single-nucleotide polymorphism) genotyping show Atlantic cod specimens with markers characteristic for NCC in Kongsfjorden (L. Spotowitz, pers. comm., 07.01.2021).

For NEAC as well as NCC segregation mechanisms are reported that separate juvenile fish from larger specimens. Such segregation seems also to happen in Kongsfjorden, where we could identify age class 0+, 1+ and 2+ as most abundant in the shallow-water zone. Specimens of age class 3+ with standard lengths of over 30 cm were rarely sampled at all. The reported segregation mechanism for NEAC is connected to a large-scale horizontal migration movement. This migration is known to start at the age of 3+ and is associated with food, especially capelin (*Mallotus villosus),* and movement towards the southern spawning grounds. The extent of this migration pattern is varying but is generally increasing with age (Johansen et al. 2013). The first full migrations that reach the spawning grounds can be observed from the age class of 6+ onwards. For NCC at the Norwegian coastline, it is reported that specimens of 3+ and older are also leaving the shallow-water zones. They migrate into deeper waters of up to 500 m (Bakketeig & Bakketeig 2018).

For NEAC as well as NCC the segregation between age groups is also connected to a shift from an invertebrate to a more fish-based diet. Therefore, this segregation also reduces the amount of cannibalism within a stock. In summary, the absence of 3+ specimens in the shallow-water zone of Kongsfjorden can be well explained and is independent of the stock affiliation of the specimen.

In the context of historical reports about fluctuations in abundance of Atlantic cod around Svalbard, the question of the origin of specimens was already raised. In the last reported Arctic warm period from 1920-1940, the presence of Atlantic cod increased in general around Svalbard. Around Bear Island and for the bank off Isfjorden in Svalbard specimens in the spawning stage are reported by Iversen (1934). Additionally, in September 1923 the catch of age group 0+ specimen of 3,5 - 6,0 cm was reported for Grønfjorden. Due to

the distribution of those 0+ specimens, Iversen (1934) brought up the assumption that not all that specimen originated from the known NEAC spawning grounds near the Lofoten.
With the current warming of the Arctic, a similar situation might recur. In this context, it seems worthwhile to reinvestigate if specimens of Atlantic cod at Svalbard belong exclusively to the NEAC stock, with active seasonal migrations between Svalbard and Norway. An existing parallel process might be a seasonal passive drift input of non-migratory NCC towards Svalbard. Continuous input of NCC with no migratory tendencies might facilitate the establishment of local spawning stock of Atlantic cod in the warming fjords on Svalbard. Kongsfjorden, which is strongly influenced by warm Atlantic water masses, might be one of the first fjords where local spawning might occur.
The question of stock affiliation of Atlantic cod specimens was unfortunately not raised during the conception of the study and cannot be answered within the current project. In future sampling studies, those questions might be answered by genetic assessments.
The combination of this genetic analysis with an ongoing operation of the underwater observatory might significantly contribute to a better functional understanding of polar coastal processes, especially in transition zones like Svalbard.
The development, maintenance, and operation of the underwater observatory itself became an integrated part of this study. In the framework of the project COSYNA (Baschek et al. 2017) the AWIPEV underwater observatory, as well as its sister system on Helgoland started operation simultaneously in 2012. The objective of both systems was to enable long-term, high-frequency, real-time observation in two shallow-water zones in which fieldwork is limited due to their harsh weather conditions. Weather dependence, in combination with logistic limitations as e.g., limited research time on vessels, often leads to small datasets. These datasets open up room for the misinterpretation of hydrological and biological processes. Moorings have proven to be a valuable tool for permanent observation of large-scale hydrological conditions, for example in Kongsfjorden (Hop et al. 2019). As in every autonomous system, the challenges in long-term monitoring system operation are system resilience with respect to the sensors, data transfer, storage limitations, as well as a stable long-term power supply (Fischer et al. 2020a). The cabled underwater observatories of the COSYNA consortium were built with these challenges in mind and belong to the comparatively

smaller number of cabled observatories for shallow-water use. Therefore, they were optimized by design and setup to withstand heavy swells in the North Sea (Ganske et al. 2005) and drift ice in the Arctic. To cope with this continuous mechanical threat the general strategy was to provide the lowest possible resistance against drift ice. The system was further optimized after its first contact with drift ice. One of those changes was that the profiling sensor unit was originally attached to the bottom with a winch line and had two additional guidelines for stabilization. Those guidelines were each attached to a foundation concrete block on the bottom and a heavy-duty fender approx. 1 m below the water column. While the sensor system was protected against drift ice while in its bottom position, the guidelines came in contact with drift ice. The concept assumed that the lines would simply give way to drift ice and return to their original position afterward. Unfortunately, we could observe that the lines got caught by ice and were displaced permanently out of position, including their foundation block. We hardened the system against further events of this kind by removing the guiding ropes and thereby changing to a freely floating sensor system. The system is now stabilized by two winch lines that end in one common foundation. This gives the system the option to hide from larger drift ice by positioning itself fully on the bottom. Furthermore, the online character of the system enables the real-time detection of damages and allows prompt scheduling of repairs. This avoids unrecognized periods of downtime and therefore unnecessary data loss. Important for a prompt and easy repair of the system is its modular design. It enables service and exchange of all parts of the system from small workboats by SCUBA divers. In combination, these design features have proven essential to maintaining high system uptimes by avoidance and quick elimination of system failures.

The modular design also allows easy integration of off-the-shelf sensors to the underwater node system. An integrative part for this purpose are connector boxes which adapt the sensor interface to ethernet standard and provide the required power supply to the individual sensor. The sensors with their specialized connector boxes are interchangeable between the systems on Helgoland and at AWIPEV. This enabled a sophisticated evaluation process for the integration of new sensors at the AWIPEV underwater node. In this integration, new sensors were tested in succession in the lab, in a test pool, and afterward on the underwater node system on Helgoland. If all integration tests were successful, they were shipped to AWIPEV and deployed there. By establishing this procedure, expedition time could be effectively reduced, and the reliability of sensor operation was enhanced. This validation process was introduced with the REMOS stereo-optic system and later applied to other sensor systems (Wehkamp & Fischer 2014; Fischer et al. 2017).

Further operational aspects arose during the operation of both observatories. It became clear that the conception and the management of the IT systems is a central aspect of the operation of an underwater observatory. It was therefore migrated from a single self-managed server towards redundant server installations managed by the AWI IT department. A second key aspect was the installation of uninterruptible power supply systems (UPS) for those servers. Short-term power fluctuations can shutdown servers and result in multiple hours of administrative work for rebooting all systems. Those power fluctuations are not uncommon if the power source is a diesel generator, as at the AWIPEV base, or is transported over long distances as on Helgoland. By providing redundant power supply to redundant server clusters we furthermore gained the ability to service and exchange every component of the IT infrastructure without causing downtime in the data assessment.

A further field for optimization are marine sensors, and especially their proprietary software packages. Those are optimized for two use cases. The first one is the short-term online operation, which gets mostly unstable if not terminated within a certain time interval. The second is the one-time initialization of a sensor for independent long-term-operation. Both use cases do not represent the requirement for automated long term online operation (Fischer 2020b). To enable long-term operation, scripting tools such as Macro Scheduler (MJT Net Ltd) are used to operate those generic software packages. While this requires less effort than the complete reverse engineering of the proprietary software, the development of automatic data assessment and error handling routines is shifted to the scientist operating this sensor.

This could be avoided if scientific sensors would start to use recent communication protocols and standardized data formats. Currently, most sensors use RS232 datagrams, which are interpreted by proprietary software. The sensors are hereby also not providing any information regarding their state, calibration, and the expected data quality.

Current developments in the framework of the Internet of Things (IoT) and Industry 4.0 are mostly neglected so far. Drivers of those developments are smart data connectivity and standardized interfaces. Scientific sensors should adapt to this modern standard and carry their own metadata as identification, parameters, calibration, and predicted accuracy and precision of observation for easier management. Their connectivity should be realized via TCP/IP (Cerf et al. 1974) and their output should be in a standardized machine-readable form as e.g., XML (Bray et al. 2008). In combination with current network discovery protocols, this would allow for automated integration of sensors into networks. In commercially used sensor networks (e.g. surveillance cameras) protocols for automated discovery, setup, and integration are standard. This kind of automatization and standardization is the key to the cost-efficient and reliable commercial operation of sensors. In the current situation in science, sensor specialists must spend significant amounts of time at every sensor deployment and maintenance to ensure correct operation. A situation that would not be imaginable in commercial operations due to cost factors and the interruptions in data assessment. Especially for the creation of long-term datasets in science, reliable long-term operation is also important, and interruptions need to be reduced to the minimum. Optimization of current sensor technology is therefore of utter importance (Fischer 2020b).

The management and quality control of recorded sensor data have proven to be another significant point. An automatic data acquisition also requires automatic quality control to ensure the validity of the recorded data. Sensors might be subject to drift, e.g. due to electrochemical aging or biofouling. A cascade of quality checks, e.g. a comparison to other sensors nearby and a comparison to upper and lower plausible limits were established to ensure data quality. Still, the currently established protocols cannot cover all situations, and manual verification of sensor data before its final publication is required. In the future, machine-learning systems might be able to be trained on the specific sensor data. They might support the sensor operator in this task and predict required sensor maintenance before faulty data occurs (Namuduri et al. 2020). The quality control procedures used in the Svalbard underwater observatory project e.g. for the dataset 2019 can be downloaded at Fischer et al. (2019).

Also, machine learning algorithms for raw data processing might reduce the workload of scientists. For example, the stereo-optic camera system RemOS produces in year-round operation 8784 pairs of pictures, which have to be processed manually as described in Wehkamp & Fischer (2014). In this processing, an observer has to mark every specimen on an image pair in the first step. It follows the manual determination of species and the manual marking of the specimen's physical boundaries on both picture pairs. By recognition of these points, the software calculates the dimensions of the specimen. This procedure creates high-quality data but is also elaborate and work intensive. Also, the system is like all manual analysis open to observer bias. An ongoing topic is therefore the optimization of the system with automatic object detection. Current research in the automotive industry shows that also their 3D object detection by stereoscopic methods is evaluated as an alternative to technologies as LIDAR (Chen et al. 2016). Further research here might lead to practical algorithms that might be transferable for analyzing RemOS's stereoscopic pictures. A second more challenging aspect is the classification of objects. As it can be expected that all algorithms will be refined over the next years, all raw data must be stored for future reanalysis. In this context, the results of manual image processing are of critical importance. As they represent good training data for future machine learning algorithms.

While the technological development of underwater sensor networks is still ongoing, the large potential of their application in the observation of remote environments as Arctic fjord ecosystems is obvious. The completed study shows that the combination of permanent online remote observation with campaign-based ground-truthing gives valuable insights into an ecosystem that is otherwise temporarily inaccessible by many means. By comparing observations of the polar day and polar night insights into the dynamics of the shallow-water ecosystem could be gathered.

# 5. Conclusion

This study shows that boreal species dominate the shallow-water fish community of Kongsfjorden. This result raises the question if this state is something new or if a similar species community would have been detected at investigations 10, 50, or even 100 years ago? In dependence on the answer to this question, we might see either a stable ecosystem within its natural variability, or we see a snapshot of a transition phase in an ongoing process of borealization.

To be unable to answer this question for one of the most investigated fjords of the Arctic shows the existing gap in our knowledge of Arctic fish ecosystems. This first quantitative study on the shallow-water fish community in the Kongsfjorden ecosystem cannot answer on its own if this fish community is changing. However, it may stimulate further hybrid studies using classical methods together with new IT-supported remote-controlled observation methods. Hereby, the still large observation gaps in this area, especially during the polar night, could be further reduced. Therefore, this PhD study hopefully provides the first point in time to which future studies of a similar kind can refer to. Additionally, we established the Kongsfjorden underwater observatory, which is in operation for now over eight years. In this timeframe, tremendous amounts of hydrographical and stereo-optical data were assessed and are ongoingly processed. Processed and quality-controlled data are published continuously for public use (Fischer et al. 2018a,b, 2019). As a result of this, the data are available for future studies that require quality-controlled long-term data for their work.

The year-round data assessment enabled us to show significant differences between summer-autumn (August-October) and winter-spring (December-March) communities in regard to total abundances and species richness (Fischer et al. 2017). The continuous growth of Atlantic cod in the polar night, as presented in Brand et al. (in draft) shows that the necessary food resources are available. Kelp beds in the shallow-water zone might play a role in the storage of energy, and provide this energy to higher levels of the food web during the polar night.

To gather further knowledge about this ecosystem, a continuous observation of the shallow-water region of Kongsfjorden, especially during the polar night, is required. The remotely operated underwater observatories' technology is promising for a detailed and cost-efficient long-term monitoring of these remote areas. A further advantage is that observatories are minimally invasive to the ecosystems they are monitoring. For example, the assessment of fish by the stereo-optical instrument in this study has not removed any fish from its natural environment. For responsible research, this is a great advantage, especially for research in protected marine zones. Combined sampling campaigns with the long-term observation by underwater observatories in combination with ground-truthing data assessments using classic sampling strategies for validation appear to be a promising strategy for future campaigns.

For this study, it has been shown that field stations, as in this case the AWIPEV base in Ny-Ålesund, are crucial for the research community as enablers of research. Without the station's services, the necessary fieldwork for fish sampling and the operation of the underwater observatories in its current form would not be possible. As a “crystallization point” for research, the AWIPEV base provides a platform for atmospheric, terrestrial, and increasing marine research. With the underwater observatory deployment in 2012, the use of remotely operated instruments is now a common nominator in all areas of research. While the primary workload on the operation of these sensors is located at the remote sensor operator, specific tasks have to be performed on-site. This relates especially to the preservation of critical infrastructure as, for example, power and network infrastructure. A sufficient amount of well-trained permanent station crew is crucial to guarantee continuous sensor operation in this extreme environment.

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