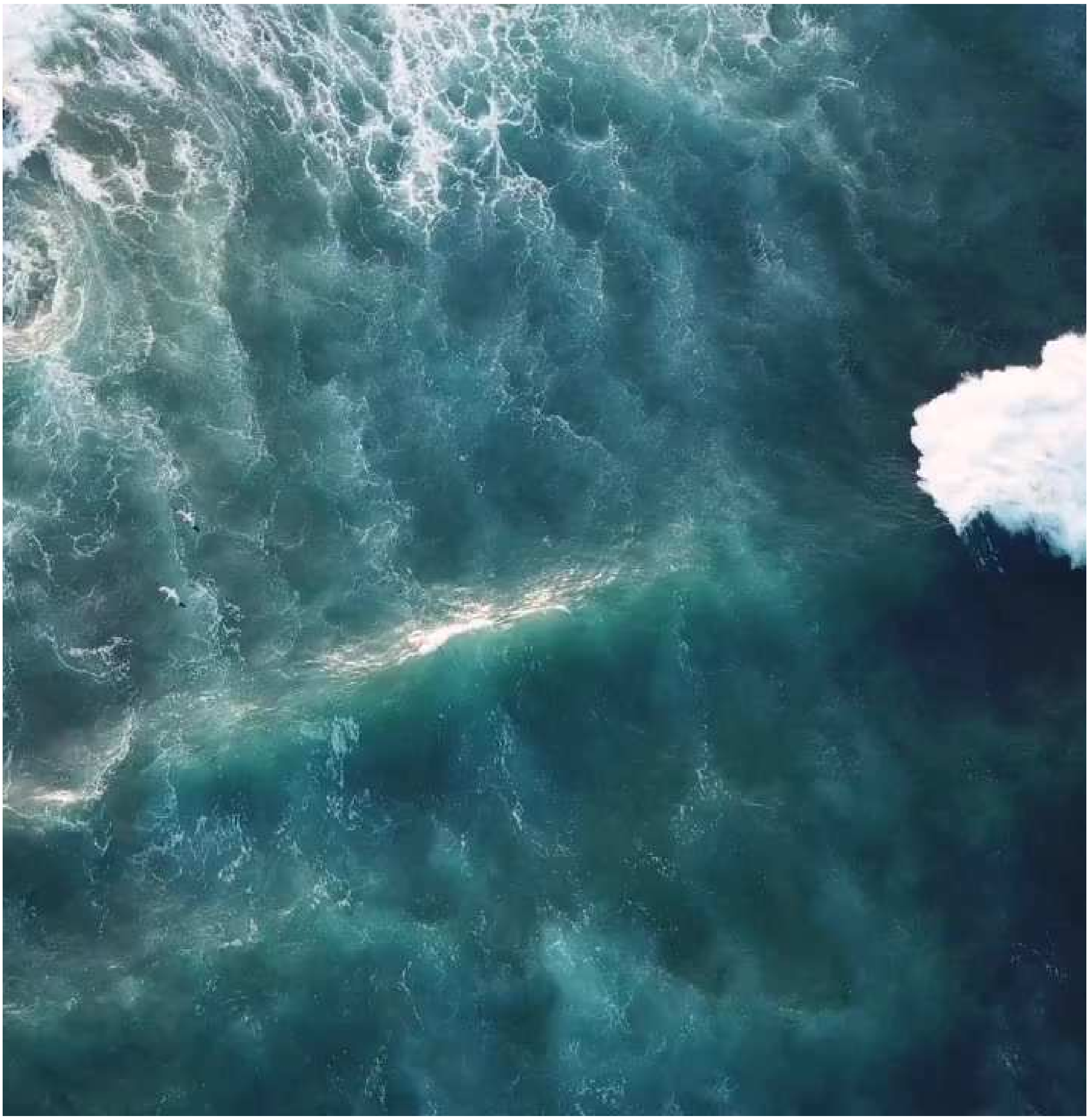
Advancing Monitoring Strategies

Less-invasive methods for detecting change in fish communities in Baltic Sea Marine Protected Areas



Dissertation Constanze Hammerl 2025

Advancing Monitoring Strategies
Less-invasive methods for detecting change in fish communities in Baltic Sea Marine Protected Areas

Dissertation

With the aim of achieving a doctoral degree at the Faculty of Mathematics, Informatics and Natural Sciences Department of Biology of the University of Hamburg Institute of Marine Ecosystem and Fishery Science

Submitted by

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Hamburg, 2025

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STUDY I: Identifying fit-for purpose methods for monitoring fish communities

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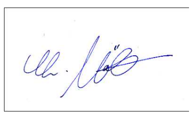
This study has been published in Frontiers in Marine Science (01/2024). The contribution of the authors is as follows: **Constanze Hammerl** developed the concept and methodology of the manuscript, conducted the literature review and analysis and prepared the first draft of the manuscript. Christian Möllmann and Daniel Oesterwind supported the writing process and reviewed the manuscript. Daniel Oesterwind coordinated funding acquisition.

Confirmation of correctness:

Hamburg, 17.04.2025



Prof. Dr. Christian Möllmann

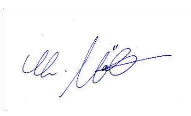
STUDY II: Remote underwater video (RUV) as a non-invasive alternative to bottom trawling for monitoring temperate soft-bottom fish assemblages in Marine protected areas

Authors: Constanze Hammerl, Christina Henseler, Christian Möllmann, Daniel Oesterwind

This study is currently prepared for submission to Regional Studies in Marine Science. The contribution of the authors is as follows: **Constanze Hammerl** developed the concept of this study, collected the data, conducted the analysis and prepared the first draft of the manuscript. Christina Henseler assisted data analysis and writing and reviewed the manuscript. Christian Möllmann and Daniel Oesterwind supported the writing process and reviewed the article. Daniel Oesterwind coordinated the funding acquisition.

Confirmation of Correctness:

Hamburg, 17.04.2025



Prof. Dr. Christian Möllmann

STUDY III: Enhancing gear performance in monitoring temperate fish communities: A comparative analysis of baited vs. unbaited underwater video

Authors: Constanze Hammerl, Christina Henseler, Christian Möllmann, Daniel Oesterwind

This study is a draft that is prepared for submission to Biodiversity and Conservation. The contribution of the authors is as follows: **Constanze Hammerl** developed the concept of this study, collected the data, conducted the analysis and prepared the first draft of the manuscript. Christina Henseler supported data collection, assisted the writing process and reviewed the manuscript. Christian Möllmann and Daniel Oesterwind supported the writing process and reviewed the article. Daniel Oesterwind coordinated the funding acquisition.

Confirmation of Correctness:

Hamburg, 17.04.2025

Prof. Dr. Christian Möllmann

ACKNOWLEDGEMENTS

First of all, I would like to thank my supervisors, Dr. Daniel Oesterwind and Prof. Dr. Christian Möllmann. Oesi, thank you for your time, your positive attitude, your support and feedback, your encouragement and for always reminding me that it is more difficult to fill a blank page than to criticise it. Mölli, thank you for your encouragement, for teaching me to change my focus and especially for sharing your years of experience in scientific writing.

I would also like to thank all my colleagues at the Thünen-Institute of Baltic Sea Fisheries. Particularly, I want to thank Christina Henseler, Kerstin Schöps, Steffen Niemann, Rainer Stechert, Tom Jankiewicz and Matthias Woltering for their help during the fieldwork, and Nakula Plantener for preparing the maps. Christina, thank you for your help during fieldwork, for carrying camera frames across the beach, for your help with statistical questions, for coffee breaks, and also for your emotional support. Kerstin, thank you for your support at sea and especially for the many great conversations. Steffen, thank you for your commitment in building the camera frames, which made the majority of my work possible in the first place.

A heartfelt thank you to Uta and Jens, for not only welcoming me into your home, but for truly giving me a sense of home in Rostock.

Many thanks also to my family and all my friends for your unwavering support and, above all, for your understanding during the past years. Special thanks go to Lena, Alissa and Heike. Thank you, Leni, for always being there for me, even at a distance, encouraging me and always telling me that you are proud of me - I am very proud of our friendship. Alissa, thank you for sharing your knowledge about psychological concepts with me and for always helping me to shift my perspective, your influence has truly shaped the way I see the world today. Thank you, Heike, for your support, especially during the final phase, for your friendship and your constant encouragement.

My special thanks to Lasse. You are my rock. I'm grateful for your partnership, your support in every way, for always trying to free me from my constant overthinking, reminding me not to take everything too seriously, and for putting up with my moods, especially within the last weeks.

SUMMARY

Marine protected areas (MPAs) and in particular no-take zones can contribute to restore ecosystem function and structure, support biodiversity conservation and ecosystem services and may help sustaining fisheries. The area dedicated to marine protection has increased globally and will further increase due to political regulations, such as the EU Biodiversity Strategy. The conservation objectives of MPAs often include the recovery of populations of vulnerable and exploited species, such as fish species or populations. To assess whether conservation objectives are being met and with that ensure effective MPA management, robust monitoring is required. However, traditional monitoring methods, such as bottom trawling, are likely to conflict with conservation objectives of MPAs leading to the increasing importance of non-invasive alternatives.

Therefore, this thesis focuses on the development and evaluation of non-invasive monitoring methods for the detection of changes in sandy-bottom demersal fish communities in MPAs in the German Baltic Sea, a brackish, low diversity environment. Thereby, the particular focus was on the evaluation of video-based methods.

At first a literature review was conducted to identify available monitoring methods based on four criteria as well as highlighting their advantages and disadvantages (Study I). Based on this, a fit-for purpose guide was developed and applied to the case of monitoring Baltic Sea MPAs. The findings indicate, that besides traditional bottom trawling, alternative less-invasive methods, including video-based approaches, could be sufficient for specific research purposes. However, available methods require further development regarding sampling design and standardization of methods, which is crucial for robust monitoring programs and the comparability with established surveys. This study complements the general introduction of this thesis and forms the foundation for the subsequent studies.

Video-based methods have been identified as a potential alternative for the monitoring of demersal fish communities in the Baltic Sea, being one of the methods closest to being non-invasive (Study I). For example, remote underwater video (RUV) has been successfully used in many bioregions on a global scale. To evaluate whether RUV represents a non-invasive alternative to beam trawling, commonly used diversity metrics were compared and sampling effort was assessed. The results indicate that RUV generally exhibits lower precision and requires greater sampling

effort than beam trawling. Moreover, its applicability is limited by low taxonomic resolution and reduced species detectability (Study II).

Recognizing the limitations of RUVs, Study III aimed to enhance the performance by employing baited remote underwater video (BRUVs) and optimizing the sampling design, including the number of stations and deployment duration. The addition of bait led to a clear improvement in performance, as shown by comparisons of commonly used biodiversity metrics derived from RUVs and BRUVs. The optimal sampling design for BRUVs was estimated to involve 33-minute deployments at 20 sampling stations. Nevertheless, low taxonomic resolution remained a limiting factor, primarily due to the specific environmental conditions (e.g., visibility, similar morphological features of fish species) of the Baltic Sea (Study III).

Overall, the findings of this thesis suggests that while RUVs and BRUVs may not be fully suitable for detecting changes in overall fish diversity in low-diversity sandy habitats, they can be effective for investigating specific groups or targeted research objectives. The investigation of other alternatives or the complementary use of methods, such as eDNA sampling, may facilitate the monitoring of these communities. Recommendations for efficient sampling design can be applied in similar ecosystems worldwide. While the findings of this thesis contribute to improving non-invasive monitoring approaches, they also emphasize the importance of pilot studies to optimize sampling strategies for the unique conditions of the Baltic Sea, as insights from other bioregions may not be directly transferable.

ZUSAMMENFASSUNG

Meeresschutzgebiete (Marine Protected Areas, MPAs), insbesondere solche, die keiner Nutzung unterliegen, können zur Wiederherstellung von Ökosystemfunktionen und -strukturen beitragen, die Erhaltung der biologischen Vielfalt und von Ökosystemleistungen unterstützen und zum Erhalt der Fischerei beitragen. Die für den Meeresschutz ausgewiesene Fläche hat weltweit deutlich zugenommen und wird infolge politischer Vorgaben, wie etwa der EU-Biodiversitätsstrategie, weiter steigen. Dabei umfassen, die Schutzziele von Meeresschutzgebieten häufig die Erholung von Populationen bedrohter und ausgebeuteter Arten, wie z.B. Fischarten oder Fischbeständen. Um zu beurteilen, ob die Schutzziele erreicht werden und somit ein effektives Management zu gewährleisten, ist ein robustes Monitoring erforderlich. Die Anwendung traditioneller Standardmethoden zur Datenaufnahmen, wie z.B. der Einsatz von Grundschleppnetzen zur Erfassung der Fischfauna, kann allerdings in Konflikt mit den Schutzzielen stehen. Dadurch entsteht der zunehmende Bedarf an nicht-invasiven Methoden zur Datenerhebung.

Der Fokus dieser Dissertation liegt in der Entwicklung und Evaluierung nicht-invasiver Monitoring Methoden zur Erfassung der Veränderungen in demersalen Fischgemeinschaften in sandigen Habitaten in der Ostsee, einem Brackwassermeer mit geringer Artenvielfalt. Dabei lag ein besonderer Schwerpunkt auf der Evaluierung der Anwendbarkeit von video-basierten Methoden.

Zunächst wurde eine Literaturanalyse durchgeführt, um verfügbare Monitoring Methoden anhand von vier Kriterien zu identifizieren und ihre jeweiligen Vor- und Nachteile herauszuarbeiten. Auf dieser Basis wurde ein Leitfaden entwickelt und auf das Monitoring von Meeresschutzgebieten (MPAs) in der Ostsee angewendet. Die Ergebnisse zeigen, dass neben dem Einsatz von Grundschleppnetzen auch alternative, weniger invasive Methoden – darunter video-basierte Methoden – für bestimmte Fragestellungen geeignet sein können. Allerdings besteht bei den verfügbaren Methoden weiterhin Entwicklungsbedarf hinsichtlich der Probennahme Strategie und der Standardisierung, um robuste Monitoring Programme zu entwickeln und die Vergleichbarkeit mit traditionellen Monitoring Programmen sicherzustellen (Studie I). Diese Studie ergänzt die allgemeine Einleitung und bildet die Grundlage für die nachfolgenden Studien.

Videobasierte Methoden wurden als potenzielle, nicht-invasive Alternative für das Monitoring der demersalen und benthischen Fischgemeinschaften in der Ostsee identifiziert (Studie I). Innerhalb

dieser hat sich der Einsatz von stationären Videoeinheiten (Remote Underwater Video, RUV) bereits in zahlreichen Regionen weltweit bewährt. Zur Bewertung, ob der Einsatz von RUVs eine nicht-invasive Alternative zur Datenaufnahme mit Baumkurren darstellen könnte, wurden gängige Diversitätsmetriken verglichen und der erforderliche Probennahme aufwand analysiert. Die Ergebnisse zeigen, dass RUVs im Vergleich zur Baumkurre in der Regel eine geringere Präzision aufweisen und einen höheren Aufwand bei der Probenahme erfordern. Zudem ist die Anwendbarkeit von RUVs durch eine begrenzte taxonomische Auflösung sowie eine eingeschränkte Erfassung der Diversität limitiert. (Studie II).

Angesichts der begrenzten Effektivität von RUVs hatte Studie III das Ziel, die Effizienz durch den Einsatz von beköderten Unterwasservideoeinheiten (baited remote underwater video, BRUVs) zu verbessern und das Probennahme Design hinsichtlich der Anzahl der Stationen und der Aufnahmezeit zu optimieren. Der Einsatz von Ködern führte zu einer signifikanten Verbesserung, wie der Vergleich etablierter Biodiversitätsindizes zwischen RUVs und BRUVs zeigte. Die optimale Probennahme Strategie für BRUVs wurde auf eine Einsatzdauer von 33 Minuten und 20 Stationen bestimmt. Die geringe taxonomische Auflösung blieb jedoch ein limitierender Faktor, was primär auf die spezifischen Umweltbedingungen der Ostsee zurückzuführen ist (Studie III).

Insgesamt zeigen die Ergebnisse, dass RUVs und BRUVs zwar nicht geeignet sind, Veränderungen in der Diversität der Fischgemeinschaften in sandigen Habitaten mit geringer Diversität zu erfassen, jedoch für die Untersuchung spezifischer Gruppen oder bestimmter Forschungsfragen genutzt werden könnten. Der Einsatz weiterer Methoden (Studie I) sowie der ergänzende Einsatz von eDNA-Analysen, könnten ein Monitoring der Fischdiversität gewährleisten. Die entwickelten Empfehlungen für ein effizientes Probennahme Design lassen sich auf ähnliche Ökosysteme weltweit übertragen. Die Ergebnisse dieser Arbeit leisten einen Beitrag zur Weiterentwicklung nicht-invasiver Monitoring Ansätze und unterstreichen zugleich die Notwendigkeit von Pilotstudien zur Optimierung von Probennahme Strategien, die auf die einzigartigen Bedingungen der Ostsee abgestimmt sind, da Erkenntnisse aus anderen Regionen nicht immer direkt übertragbar sind.

Contents

GENERAL INTRODUCTION

Detecting change and the role of monitoring

Change is a natural process in ecosystems. Ecological change can be either cyclical (e.g. seasonal fluctuations), successional, which refers to a gradual process over time that occurs in response to environmental changes, disturbances or biological interactions, or stochastic/abrupt change that is often observed after extreme events (Hellawell, 1977). Global environmental change has accelerated rapidly in recent decades (Ammar et al., 2021; Chin et al., 2017; Waters et al., 2016) and poses a major challenge to the ability of ecosystems to adapt to these events and disturbances (Ammar et al., 2021; Nyström et al., 2019; Paine et al., 1998; Waters et al., 2016). A major driver of current environmental change is human activity, resulting in regional and global changes in climate, biogeochemical cycles and biodiversity (Ammar et al., 2021; Nyström et al., 2019). Research investigating these changes on marine ecosystems is often based on the concept of a baseline, which aims to describe the ecosystem before human interaction (Atmore et al., 2021). The concept of a baseline in an ecological context is somewhat considered a problematic ecological assumption, as it implies a particular natural state for a given ecosystem (Ammar et al. 2021, Lotze and Worm, 2009, Rodrigues et al. 2019), for which scientifically sound data is unavailable and may results in a shifted baseline. The shifting baseline syndrome was first coined by Daniel Pauly (1995) and describes the phenomenon that collective environmental knowledge and memory evolves with successive generations, leading to a gradual loss of knowledge about the previous state of an ecosystem and thus to different assumptions about the baseline definition in different generations, e.g. generations perceiving a progressively degraded environment as normal (Jackson and Jacquet, 2011). In the context of environmental impact studies, the shifting baseline is relevant as it describes how human-accepted thresholds for environmental conditions are continuously lowered due to progressive degradation, which can affect the interpretation of long-term changes (Soga and Gaston, 2018). This phenomenon can be particularly problematic in the context of marine conservation, as it can influence human perceptions of what constitutes a ‘healthy’ ecosystem. This can lead to long-term changes being incorrectly interpreted and the benefits of conservation measures such as marine protected areas being underestimated (Bohnsack, 2003).

To address these challenges, marine environmental monitoring serves as a fundamental tool for establishing temporal baselines and tracking ecological change, e.g. for the evaluation of the effectiveness of marine protected areas. It involves repeated measurements at the same locations of ecological variables over a period of time, the quantification of anthropogenic activity levels, and the evaluation of ecosystem responses and management effectiveness (Day, 2008). Biological monitoring involves a systematic data collection on species, habitats and/or ecosystem processes (in any combination) to understand their dynamics and obtain information for management. It provides the ability to document natural and anthropogenic environmental impacts and to assess the effectiveness of management measures and therefore, represents a fundamental management tool (Day, 2008; Nygård et al., 2016). In the context of growing anthropogenic pressures (e.g. habitat destruction, overfishing, climate change) monitoring is becoming increasingly important to detect any alterations to the marine ecosystem that might be associated with human activities or resource use. Therefore, monitoring plays a key role in detecting changes in the marine ecosystem. It is recognized by many scientist and resource managers that effective marine management approaches, like the establishment of Marine protected areas (MPAs) cannot be achieved without effective monitoring, evaluation and adaptation (Day, 2008). In this context, it is essential to define the status of these areas, to evaluate whether implemented measures are effective and, where possible, identify potential adverse effects (Davies et al., 2001; Day, 2008).

Since monitoring is an integral tool of marine environmental management and e.g. plays a fundamental part in sustainable fisheries and ecosystems based management (Trenkel et al., 2019) as well as impact assessment, it becomes mandatory by legislation to review environmental directives, like the Marine strategy framework directive (MSFD). The MSFD was introduced to protect the marine ecosystem and biodiversity and to achieve good environmental status (MSFD, 2008/56/EC). Another example highlighting the critical role of monitoring in guiding decision-makers is the monitoring of fish stocks under the legislation of the Common Fisheries Policy. In the Baltic Sea for example, member states conduct the Baltic International trawl survey (BITS) an annual survey to provide fishery-independent data for the commercial, demersal stocks of cod (*Gadus morhua*), flounder (*Platichthys flesus*), plaice (*Pleuronectes platessa*), dab (*Limanda limanda*) and turbot (*Scopthalmus maximus*) (ICES, 2017). Twice a year each member state samples the demersal fish community and collects data on frequency, length, weight and where required sex and maturity and condition at the same time. These surveys are all coordinated by the International Council for the Exploration of the Sea (ICES), which then builds the data base for policy advice.

As human activities in the oceans increase, the current and future monitoring programs will be faced with growing challenges and may need to be adapted accordingly. For instance, the global demand for renewable energy is increasing, driving the expansion of offshore wind energy. This expansion, however, makes many areas inaccessible to conventional monitoring methods due to safety restrictions. At the same time, potential environmental changes caused by offshore wind farms need to be closely monitored in order to detect any impacts (Rezaei et al., 2023). The expansion of marine protected areas is another reason why many areas may become inaccessible for monitoring purposes, but at the same time it is important to measure possible changes in these areas. Besides, the ethical debate especially about the potential impact of especially, methods extracting fish, where it is not mandatory, is growing (Costello et al., 2016; Trenkel et al., 2019). This, together with technological advances that facilitate the development of more efficient, robust and accurate monitoring methods, is leading to a potential shift in the methods used to monitor the marine ecosystem.

Marine Protected Areas

Marine protected areas (MPAs) can be defined in various ways, but they are generally referred to as areas that provide some level of special protection to parts of the ocean for conservation purposes (Blyth-Skyrme et al., 2006; Edgar et al., 2007). The International Union for the Conservation of Nature (IUCN) defines an MPA as "*an area of intertidal or subtidal terrain, including overlying water and associated flora, fauna, and cultural features, reserved by law or other effective means to protect the enclosed environment*" (Humphreys and Clark, 2020; Kelleher and Kenchington, 1992; Morling, 2004).

The establishment of MPAs was driven by the multitude of threats to the marine and coastal environment nowadays and aims to achieve a wide range of objectives (Edgar et al., 2007). The primary goal of most MPAs is the conservation of biological diversity, that can include the protection of threatened or important species, communities or habitats (Edgar et al., 2007; Halpern, 2003; Roberts et al., 2001). Most MPAs are designed to achieve benefits for specific issues or problems, emphasizing the importance of identifying specific goals or expectations of the MPA (Edgar et al., 2007; Laffoley et al., 2019). Consequently, various types of MPAs and different levels of protection exist. Some are designated as ‘no-take zones’ (marine reserves), banning all extractive human activities (e.g. commercial fishing), others function as multi-purpose MPAs that balance conservation objectives with sustainable use, allowing for certain commercial or recreational activities, or are designed to conserve specific species or habitats, or recreational zones such as protected seascapes (Day et al., 2012; Kriegl et al., 2021). The IUCN specifies seven different categories to further define the level of protection. Categories range from strictly protected areas, where “human visitation use and impacts are strictly controlled or limited” over areas that have the aim to protect a specific natural monument or particular species or habitats to areas that “conserve ecosystems and habitats together with associated cultural values and traditional natural resource management” (Day et al. 2018). A complementation of the well-established categorization of MPAS by the IUCN is the MPA Guide (Grorud-Colvert et al., 2021) that defines MPAs as places in the ocean that are protected to conserve biodiversity from avertable threats and categorizes four levels of protection from full protection, where no extractive or destructive activities are permitted to minimal protection allowing for extensive extraction and other high impact activities (Grorud-Colvert et al., 2021). Conservation outcomes of MPAs (e.g. increase in biodiversity, biomass of fish) are directly linked to the level of protection and the stage of establishment, resulting in the assumption that fully protected and actively managed MPAs provide the most effective biodiversity conservation outcomes (Figure 1).

Marine protected areas (MPAs) are widely recognized as an effective conservation tool to protect marine biodiversity and associated ecosystem services from anthropogenic pressures while also supporting the restoration of ecosystem structure and function (Duarte et al., 2020; Gaines et al., 2010; Halpern et al., 2010; Mumby et al., 2006; Palumbi, 2002; Salm et al., 2006). More fundamentally, they can also help build resilience to the impacts of climate change (Laffoley et al., 2019), e.g. by the protection and conservation of seagrass meadows or kelp forests that act as natural carbon sinks (Roberts et al., 2017). A variety of scientific literature has shown that, under specified ecological conditions and management practices MPAs can be of considerable benefit for certain species and habitats occurring within their boundaries (Humphreys and Clark, 2020; Kriegl et al., 2021). MPA effectiveness is commonly defined as the extent to which management actions achieve the objectives of the protected area (Garces et al., 2013; Pomeroy et al., 2004) and the overall success of managing it (Heck et al., 2012;

Hockings et al., 2006; Toropova et al., 2010). MPAs have been shown to improve the biological parameters of species within their limits, e.g. abundance and size of individuals (Edgar et al., 2014; Gaines et al., 2010; Lester et al., 2009). Yet, not all species will benefit from MPA protection, particularly in the case of more mobile or migratory species (Appeldoorn, 2008; Halpern et al., 2010). The effectiveness of MPAs is dependent on different factors like size, the age of the MPA, the level of protection, the enforcement of regulations as well as the connectivity of MPAs (Edgar et al., 2014), putting even more emphasis on an active management of MPAs. Management should include setting goals and objectives from the outset, site selection, zoning, planning and implementing a monitoring and enforcement system and monitoring activities (Kelleher and World Commission on Protected Areas, 1999; Ojeda-Martínez et al., 2009).

While there is widespread agreement in the conservation community about the positive effects of MPAs, their suitability for fisheries management remains controversial, with increasing debates about the effectiveness of this tool (Hilborn, 2018; Kaiser, 2005). A common argument against the effectiveness of MPAs in conserving fish stocks is that MPAs often displace rather than reduce fishing activity (Greenstreet et al., 2009; Jones, 2002), potentially leading to greater ecosystem degradation in unprotected waters (Halpern 2010). Generally, MPAs are a key point of tension as the prohibition of fishing impacts people's livelihoods. MPAs are also generally advertised as a kind of fish bank, since they can theoretically act as fish nurseries (depending on the species) and the biomass from the MPA may spill-over into the surrounding and fished areas (spillover). A study in the Philippines e.g. showed an increase in the outside of the area of about 40% in 5 years. In sustainably managed fisheries, MPAs however, might decrease the fishing yield if more area than necessary is protected from fishing (Kriegl et al., 2021; White et al., 2011, 2010). Yet, threats to the ocean and its biodiversity arise from different directions (Hilborn, 2018) and there is general agreement, that MPAs cannot be a solution to, e.g. global warming or ocean acidification, and must be combined with a range of more "traditional" management approaches (Allison et al., 1998; Edgar et al., 2007; Halpern et al., 2010).

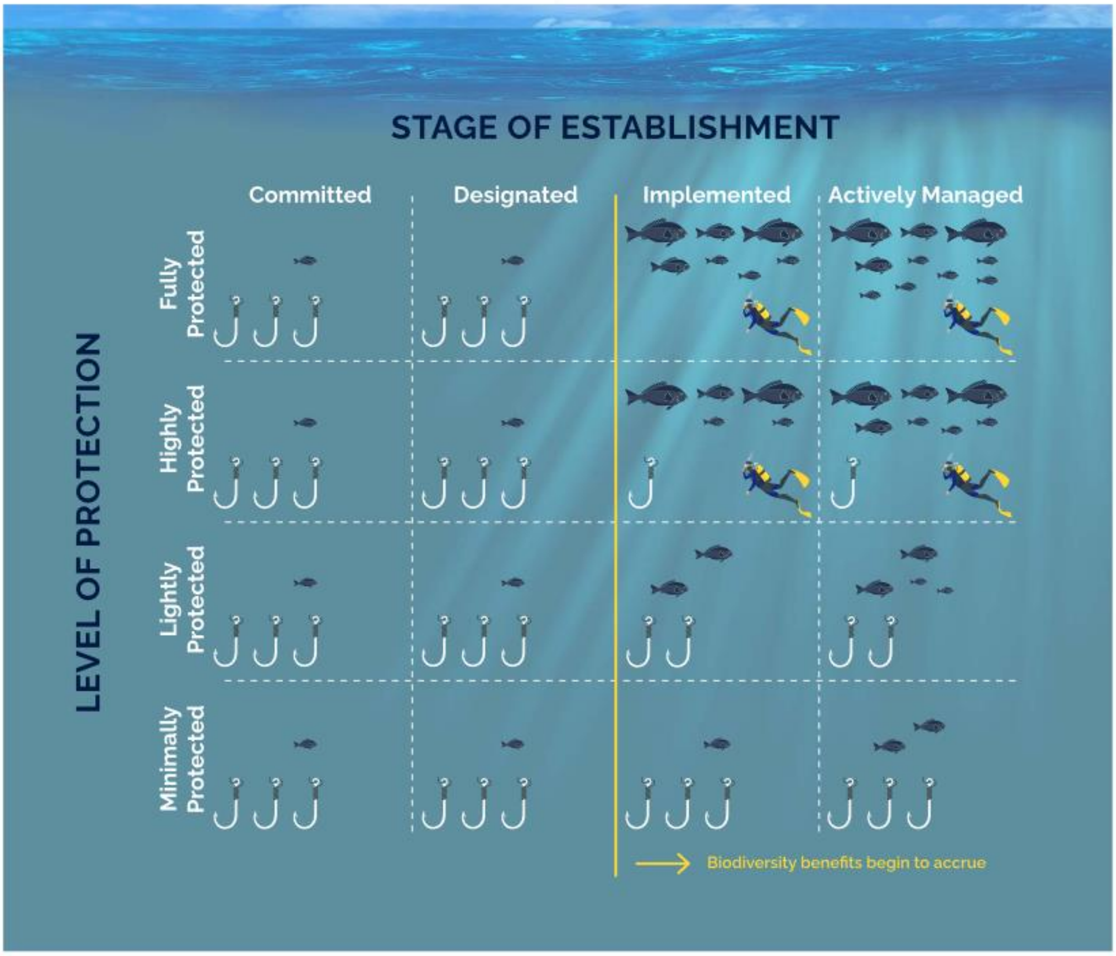


Figure 1: Matrix based on the Level of protection and stage of establishment of MPAS. Hooks indicate extractive use (e.g. commercial fishing), divers represent recreational, traditional and cultural use, fish indicate biodiversity outcomes (from: Grorud-Colvert et al., 2021. The MPA Guide: A framework to achieve global goals for the ocean. Science, 373(6560), eabf0861, doi: 10.1126/science.abf0861. Reprinted with permission from AAAS.)

Nevertheless, the area dedicated to marine protection and conservation has been increased significantly (Laffoley et al., 2019) and is likely to further increase in the coming years. The Aichi Biodiversity Targets of the Convention on Biological Diversity call for protection of at least 30% of the world's oceans by 2030. This target has been adopted by the European Union (EU) Biodiversity Strategy, which aims to protect 30 % of the exclusive economic zones (EEZs) of EU Member States, 10 % of which are to be under special protection by 2030 (EU, 2021). In the EU 8% of the sea areas are already under protection by the Natura 2000 network. A special network of Marine protected areas, which includes the 'sites of community interest' (SCI) or 'special protection areas' (SPA) (Zoppi, 2018). This network has been established by the European Union under the Birds and Habitats directives to protect important species or habitats (Evans, 2012). Six of those MPAs were implemented in the EEZ of the German North Sea and Baltic Sea in 2017 and have a total size of 10.392 km^2 (BfN, 2020). In the German EEZ of the Baltic Sea the three MPAs 'Fehmarnbelt', 'Kadetrinne' and 'Pommersche Bucht-Rönnebank' cover around 51% (2.472km2) of the entire German EEZ in the Baltic Sea (Figure 2). The area 'Pommersche Bucht-Rönnebank' is further divided into the SCA 'western Rönnebank', SCA 'Adlergrund', SCA 'Pommersche Bucht and Odra bank' and the SPA 'Pommersche Bucht'. Protected features vary in all areas, yet harbour porpoises, reefs and harbour seals are protected in all areas. A further objective in the Fehmarnbelt and Odra bank areas is the protection of sandbanks and gravel coarse sand and shell grounds in the Fehmarnbelt alone. In these areas, all activities that could lead to the destruction, damage or alteration of the protected goods are prohibited. This includes the construction of artificial islands, installations and structures, the introduction of dredged material, marine aquaculture, the introduction of animals or plants of non-native species and, in certain areas, recreational fishing. These restrictions do not apply to air traffic, shipping, military use under international law and scientific marine research. In addition, measures for the administration of the nature reserve and for the fulfilment of public tasks such as hazard prevention, law enforcement, marine surveying, fisheries control and disaster control are excluded. A recent development is, that since December 2024, the use of mobile bottom contacting fishing gears for any purpose has been prohibited (EU, 2024).

The Baltic Sea

The Baltic Sea is the largest brackish water system globally. It is a semi-enclosed, low-diversity, non-tidal and relatively shallow sea (Snoeijs-Leijonmalm et al., 2017). The water in the Baltic Sea is generally more turbid compared to oceanic waters, making the photic layer narrower as in the oceans (Helcom, 2023). The Baltic Sea is further characterized by a strong horizontal and vertical salinity gradient. Salinities decrease from about 20 in the western part to approximately five in the north-east (Ojaveer et al., 2010) and is typically higher on the seafloor compared to the sea surface (Meier et al., 2022; Snoeijs-Leijonmalm and Andrén, 2017; Zweifel et al., 2009). The brackish characteristic and decreasing salinity is driven by variable saltwater-inflows of marine water from the Kattegat through the shallow and narrow Danish straits and the constant freshwater inflow of 200 river-runoffs framing the Baltic Sea (Snoeijs-Leijonmalm and Andrén, 2017). The strong salinity gradient in the Baltic Sea creates a corresponding gradient in biodiversity, resulting in a relatively low overall number of species (about 5000) compared to other seas. This number decreases from the west to the north-east, consequently, the Baltic Sea is characterized by both low species and low functional diversity (Ojaveer et al., 2010). The Baltic Sea inhabits marine as well as freshwater species along clear geographical patterns (Snoeijs-Leijonmalm and Andrén, 2017), many of these living close to the edge of their salinity tolerance, making them very sensitive to any change in environmental conditions. This has caused species to be genetically distinct from their relatives in other areas, showing two species to be endemic to the Baltic Sea the narrow wrack (*Fucus radicans*; (Bergström et al., 2005)) and the Baltic flounder (*Platichthys solemdali;* (Momigliano et al., 2018)).

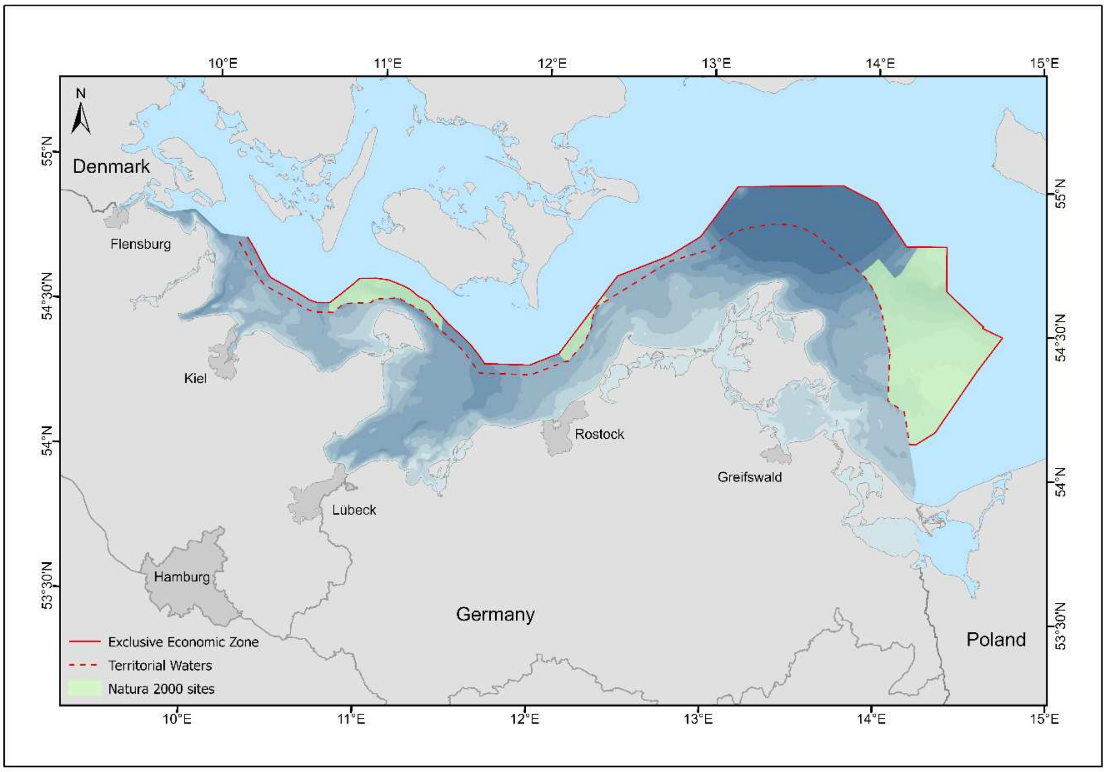


Figure 2: Map of the German Baltic Sea, showing territorial waters (dashed red line), the Exclusive Economic Zone (EEZ), and designated Natura 2000 sites (green).

The Baltic Sea is a sea with a long history of anthropogenic use (Reusch et al., 2018) and is currently suffering due to multiple stressors like eutrophication, deoxygenation, acidification, warming and overfishing (Reusch et al., 2018). In the last 40 years the Baltic Sea has warmed the most of all marginal seas globally (Belkin, 2009). Since the 1980s annual mean sea surface temperature has increased around 1-2°C (Belkin, 2009). Further, the Baltic Sea is significantly influenced by surrounding landmasses and particularly affected by nutrient inputs leading to eutrophication (Reusch et al., 2018; Savchuk, 2018; Snoeijs-Leijonmalm and Andrén, 2017; Wulff et al., 1990). More challenges arise by the impact of hazardous substances, marine litter, non-indigenous species and underwater noise. Due to its low biological complexity and the impact of multiple stressors the Baltic Sea is seen as a perfect model region to study the functioning of an ecosystem by the impact of superimposed stressors and the effectiveness of mitigation measures (Reusch et al., 2018), like e.g. the exclusion of mobile bottom contacting fishing gears from Marine Protected Areas (MPAs).

Motivation and outline of the thesis

The overall aim of this dissertation was the evaluation of current available alternative and less-invasive monitoring methods for the detection of changes in demersal fish communities, particularly in Marine protected areas of the Baltic Sea. The dissertation will contribute to the advancement of less-invasive and non-invasive monitoring approaches by identifying suitable methods, evaluating the comparability of underwater video techniques to existing baselines, quantifying potential biases, limitations, gaps and developing an efficient sampling strategy.

Study I aimed to identify the limits and opportunities of monitoring methods that are already available and present a guide that can support the choice for a fit-for purpose method. This was done by a literature review where me and my co-authors evaluated both traditional and modern methods used for monitoring fish stocks and communities based on four criteria: the sampling strategy used, the type of information provided, the target species and the target habitat along with an evaluation of their respective advantages and disadvantages (**STUDY I**: Identifying fit-for purpose methods for monitoring fish communities).

Among the potential non-invasive methods identified in Study I, remote underwater video (RUV) emerged as the method that is closest to being non-invasive and simultaneously allows the collection of abundance data. Study II aimed to evaluate whether (RUV) represent a non-invasive alternative to beam trawling by computing commonly used fish diversity metrics and the sampling effort required to detect a comparable level of family richness (**STUDY II**: Remote underwater video (RUV) as a non-invasive alternative to bottom trawling for monitoring temperate soft-bottom fish assemblages in Marine Protected Areas).

Study III aimed to investigate biases and limitations of baited remote underwater video (BRUV) compared to RUV along with the evaluation of an optimal sampling design in terms of the number of stations and the optimal deployment time needed to detect a certain level of family richness (**STUDY III**: Enhancing gear performance in monitoring temperate fish communities: A comparative analysis of baited vs. unbaited underwater video).

STUDIES OF THE THESIS

STUDY I

Hammerl, C., Möllmann C., Oesterwind D. 2024. Identifying fit-for purpose methods for monitoring fish communities. *Frontiers in Marine Science*

STUDY II

Hammerl, C., Henseler C., Möllmann C., Oesterwind D. Remote underwater video (RUV) as a non-invasive alternative to bottom trawling for monitoring temperate soft-bottom fish assemblages in Marine Protected Areas. *To be submitted to Regional Studies in Marine Science in April*

STUDY III

Hammerl, C., Henseler C., Möllmann C., Oesterwind D. 2025. Enhancing gear performance for monitoring temperate fish communities: A comparative analysis of baited vs. unbaited underwater video. *Manuscript in Preparation*

STUDY I

Identifying fit-for purpose methods for monitoring fish communities.

Published in Frontiers in Marine Science

Title: Identifying fit-for purpose methods for monitoring fish communities

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Abstract

Scientific monitoring is a fundamental basis of scientific advice. Among others, monitoring aims at contributing towards understanding the influence of anthropogenic use (e.g. fisheries), the health of a stock and individuum and effectiveness of management and conservation measures (e.g. MPAs) Monitoring of demersal and benthic fish communities is often based on invasive methods like bottom trawling, however in some cases less invasive methods might be available. The need for developing alternative and less invasive monitoring methods is supported by an increasing number of Marine Protected Areas and Windfarms where traditional methods such as trawls cannot be deployed due to conservational or technical and safety reasons. To support the development of new monitoring concepts we conducted a literature review to identify limits and opportunities of methods that are already available. Furthermore, we present a fit-for purpose guide that can help identifying the appropriate method for individual purposes. We defined eight different methods which were analyzed using four different criteria and listed their advantages and disadvantages. We further apply this guide to monitoring in Marine Protected Areas in the Baltic Sea as a case study, indicating that besides traditional bottom trawling, alternative and less invasive methods could be sufficient for specific research purposes. We therefore, encourage scientists and managers to consider alternative data collection methods to minimize environmental impact of scientific sampling. However, our results also indicate that most of the methods still need further refinement especially regarding sampling design, standardization of methods and comparability with established survey methods

Keywords: Underwater video, passive gears, hydro acoustics, Marine protected areas, eDNA, Offshore Wind Farm, sustainable fisheries, impact assessment

Introduction

Monitoring is often understood as a part or form of resource assessment, research or baseline surveys (Day, 2008). It can provide a context for marine science and has enabled the development of a science based understanding of marine ecosystems and human impacts affecting it (Bean et al., 2017). Monitoring hence serves a strategic purpose in the decision-making process and supports to establish and compare the baseline status of an ecosystem component in relation to a management target as well as to improve the relevance, efficiency and effectiveness of policy decisions (Bean et al., 2017). Further, it is an integral tool of marine environmental management and e.g. plays a fundamental part in sustainable fisheries and ecosystem based management (Trenkel et al., 2019) as well as impact assessment. Therefore, monitoring is not only question or target driven by the interests of scientists, it furthermore becomes mandatory by the legislation to review environmental directives. Beside the provision of information about 'target species' (dedicated monitoring) it can provide additional information about non-target species (non-dedicated monitoring), although the design for the latter is often not optimal. However, an example for a legislate monitoring is the monitoring within the framework of the Marine Strategy Framework Directive (MSFD), which was introduced to protect the marine ecosystem and biodiversity and to achieve good environmental status (MSFD, 2008/56/EC).

However, current monitoring approaches are strongly impacted by an increased interest in protecting and reserving nature, biodiversity and ecosystems, which will result in the establishment of more protected areas (EU, 2021), and by the increasing anthropogenic use of our seas. Besides recreational use, the seas e.g. become more important for the production of emission-free energy, which results in the expansion of offshore wind farms (OWFs) (Stelzenmüller et al., 2022). On the one side, monitoring becomes necessary in this context to i) review the achievements towards the goal of a good environmental status and the effectiveness of marine protected areas and ii) to understand the impacts of OWFs on the ecosystem. On the other side, especially in the context of monitoring in fisheries research the establishment of OWFs and MPAs can become challenging, since monitoring targeting demersal and benthic species is often conducted by scientific bottom trawling. While, scientific bottom trawl surveys are primary conducted to provide fishery-independent data to support fish stock assessment, scientific bottom trawling is more recently also used for multidisciplinary monitoring of ecosystems (Maureaud et al., 2021).

However, in the above-mentioned areas it is i) questionable whether the use of invasive monitoring methods such as bottom trawling are still in line with the conservation objectives (e.g. in MPAs with a benthic conservation target), and ii) will be prohibited in some parts (e.g. MPAs) or will be impossible due to technical and/or safety circumstances (e.g. in OWF). These potential `no fishing areas' may become a blind spot for fisheries research if no other alternative monitoring method is available to collect necessary information (Haase et al., 2023). On the other hand, MPAs are conservation measures and may help in sustaining fisheries (Kriegl et al., 2021) and hence their efficacy needs to be monitored and understood, as well as the impacts of OWFs on the marine ecosystem and associated fish communities that are not yet fully understood (Abramic et al., 2022; Lindeboom et al., 2011). In consequence, there is a need for reconsidering fisheries monitoring and the consideration of alternative and less invasive monitoring approaches. Towards this goal, we here present a literature review that allowed us to design a fit-for-purpose guide to support the decision-making process in identifying ways for a less invasive monitoring in fisheries research and impact assessment. The developed guide can serve as a starting point for further investigations and analysis of additional methods not covered in this review. We further, demonstrate the application of the guide to the case of Baltic Sea MPAs because solutions are currently needed for MPAs where bottom-towed fishing will most likely be excluded in the near future.

Literature search

We conducted a literature review evaluating both traditional and modern methods used for monitoring fish stocks and communities. We distinguished traditional methods such as active and passive fishing gears from "modern" approaches such as hydroacoustics, underwater video footage and molecular methods. We conducted the literature search using the database Google Scholar (https://scholar.google.de/) applying terms related to monitoring and sampling of fish communities/assemblages in the marine environment (see supplementary material, Table 1). Based on an initial selection of publications, further publications were added to the analysis applying a snowball system. We incorporated eight distinct methods in the analysis: trawling, passive fishing gear, stationary video, towed video, split-beam echosounder, acoustic camera, acoustic tagging and eDNA. All methods were analysed according to four criteria: the sampling strategy used, the type of information provided, the target species and the target habitat along with an evaluation of their respective advantages and disadvantages (see Table 1 and Table 2). In total we initially skimmed 179 articles for information about different monitoring methods of which 86 were considered relevant for this review. An article was considered relevant if it provided a review of different methods or one method, mainly focused on the marine environment and/ or contained comparisons of different methods or addressed the context of Marine protected areas and fisheries closures.

Traditional methods

Active gears

Active fishing gear includes among others pelagic- or bottom trawls, seine nets and dredges. We focus here on bottom towed gear, especially bottom trawling since we mainly target monitoring of benthic and demersal fish communities. Bottom towed gears include beam trawls, otter trawls and dredges but also other more specific gears. The tradition of scientific bottom trawling dates back to the 1900s and is still probably the most commonly used method for monitoring demersal fish species on continental shelfs and slopes (Garces et al., 2006; Maureaud et al., 2021; Trenkel et al., 2019). Considering only standardised surveys with otter trawls, there are about 95 surveys investigating the demersal marine fauna of continental shelves and slopes worldwide (Maureaud et al., 2021). In Europe, multiple surveys are conducted with bottom trawls, for example the ICES (International Council for the Exploration of the Sea) coordinated Beam Trawl Survey (BITS), the International Bottom Trawl Survey (IBTS) or the Baltic International Trawl Survey (BITS) (for more details regarding the surveys see ICES.dk). Many of these surveys have been performed continuously over years and produce long time series and therefore provide a unique opportunity to track species range shifts and improve the assessment of biodiversity under global change (Maureaud et al., 2021). Standard large-scale scientific bottom trawl surveys support both fish stock assessments and the identification of spatio-temporal changes of fish species and communities in the shelf areas worldwide (Trenkel et al., 2019). Regular bottom trawl surveys mainly vary with respect to the width, height and mesh sizes of the gears used depending on the target species and driven by the scientific objective.

The main advantages of scientific bottom trawling are that due to their extractive nature they are more precise and simpler in identifying species compared to some non-extractive methods. Furthermore, the extrapolation to abundance per area is easier because the area sampled is approximately known by towing time and speed as well as the dimension of the fishing gear. Another advantage is the wide variety of important and necessary information that is provided by individual fish like weight, length, sex, maturity, age and other condition parameters that are essential for fisheries management. The major disadvantage of bottom trawling are potential unwanted effects on the ecosystem especially on habitats and organisms associated with the seafloor (Johnson et al., 2015; Oberle et al., 2016). Even though the impact of scientific bottom trawling is negligible compared to commercial fishing with bottom-towed fishing gears, it might still be desirable to minimize these potential impacts (Trenkel et al., 2019), especially in ecosystems with very sensitive benthic components and a risk of bycatch of e.g. threatened species. Moreover, large-scale fisheries independent monitoring programs are time-consuming and involve expensive surveys on research vessels (Biber, 2011; Maureaud et al., 2021). Another drawback is that trawls also have a certain selectivity (e.g. mesh size, towing speed, ground gear) and can be more sensitive towards smaller and slow-moving fish (Murphy & Jenkins 2010, Cotê & Perrow, 2006). Furthermore, bottom trawling is not possible everywhere. Many habitats, especially highly structured habitats, pose too high a risk for trawling, as nets could be damaged or even lost. A relatively recent problem arises from the exclusion of bottom trawling in additional areas, such as wind parks and MPAs. This can lead to a non-representative survey of the respective target species and impact assessments are not possible with this traditional fishing methods due to conservational (in MPAs) or technical and safety (bottom constructions) reasons.

Passive gears

Passive gears for monitoring fish stocks such as traps, gillnets as well as hook and line rely on an active encounter by the fish (Côté and Perrow, 2006). As a consequence, passive gears can only provide relative abundance data and surveys require high level of standardisation. Similar to bottom trawling, passive gears are extractive methods, which allows for more precise identification of fish species compared to other, non-extractive methods. Fish traps are commonly applied to determine the abundance of a number of fish and invertebrate species (Jones et al., 2003; Recksiek et al., 1991; Rudershausen et al., 2010; Wells et al., 2008). In the

south-eastern United States, for example, the assessment and management of black sea bass (*Centropristis striata*) relies partly on fishery-independent trap data (SEFIS-Survey) (Bacheler et al., 2013b; Southeast Data Assessment and Review, 2011). Trapping is considered to catch most species largely independent of environmental conditions. Traps can be further distinguished into pot gears, fyke nets and trapping barriers (Côté and Perrow, 2006). Traps can also be used for estimating fish densities and movement patterns (Murphy and Jenkins, 2010) and usually provide reliable CPUE data if design and size are standardised (Côté and Perrow, 2006). They are applicable in various habitats and depths (Bacheler et al., 2013b; Beentjes, 2019; Harvey et al., 2012; Thrush et al., 2002; Wakefield et al., 2013) and are therefore, useful for the detection of fish inhabiting structural complex habitats (Wells et al., 2008). Traps can also be placed along transects at a set distance (Kaunda-Arara and Rose, 2004; McClanahan and Mangi, 2000) and can fish unattended (Bacheler et al., 2013b). A great advantage of trap fishing is that fish remain in their natural habitat and can often be released alive (Miller, 1990), if information on age, maturity and sex is not required. However, a determination of the sampled area is often impossible and the influence of environmental parameters (e.g. currents) and bait plume on gear performance are hardly quantifiable (Stoner, 2004).
Hook- and line fishing is a kind of remote surface sampling and used to offset the behaviour of fish to divers (Willis et al., 2000) and to catch large predatory fish that occur in low densities (Côté and Perrow, 2006). Hooks are generally baited appropriately for the target species (Murphy and Jenkins, 2010), making hook- and line fishing highly selective and therefore unsuitable for monitoring entire fish communities (Côté and Perrow, 2006). However, the method can generate CPUE statistics and has been used e.g. to produce area estimates of relative densities of snapper and blue cod (Côté and Perrow, 2006; Willis et al., 2000). Hook- and line fishing is furthermore relatively cheap but may present a significant risk to marine birds or other groups (Côté and Perrow, 2006). Additionally, a major drawback of hook-and line fishing is the significant damage and stress to which the fish are exposed which often results in the mortality of the individuals (Côté and Perrow, 2006; Murphy and Jenkins, 2010)

Gillnets are mostly used to catch mobile fish species and are considered to be the most selective nets (Côté and Perrow, 2006). Mesh size can be adjusted to the target species, so that non-target species bycatch can be reduced effectively. In the case of multi-species surveys, different mesh sizes can be deployed across the net or less selective nets like trammel nets can be used. Gillnets are relatively cheap and long lasting and catches can be reliably expressed as CPUE (Côté and Perrow, 2006). However, estimating the sampled area is not possible and removing caught fish and other organic or inorganic material out of nets can be time consuming. Furthermore, gillnets pose a risk of catching marine mammals and birds in some areas (Côté and Perrow, 2006). Additionally, gillnets can cause high mortality of the fish caught, especially if nets are set over a long period of time (Côté and Perrow, 2006).

Modern Methods

Hydroacoustics

Hydroacoustic methods are based on the sound transmission properties in water (Tessier et al., 2016). Thereby, sound is generated by the conversion of a pulse of electricity produced by a transmitter. The sound then travels through the water column until it encounters an object and the echo is returned to the boat, where it is picked up by a transducer (Côté and Perrow, 2006). By that, echoes can provide information on the size and number of fishes and their distribution (Boswell et al., 2007; Côté and Perrow, 2006; Johnston et al., 2006). The areas of application are diverse and range from characterising habitats to surveying the spatial and temporal movements of large mobile fish species (Murphy and Jenkins, 2010). Depending on the objective target different methodologies can be applied. In terms of detection of fish the common methodology is a split-beam echosounder, but also acoustic cameras, acoustic tagging and passive acoustics can be used to monitor fish assemblages (Murphy and Jenkins, 2010). Passive acoustics differs from other methods since the sound is not actively generated by a transmitter. Instead, underwater sounds are recorded using underwater microphones. In this way, ambient sounds, like sound coming from animals can be recorded. In the following, we focus on split-beam echosounder, acoustic cameras and acoustic tagging and will not discuss passive acoustics.

Table 1: Summary of traditional techniques used for monitoring fish communities.

Observational method	Categories	Sampling strategy	Type of information	Target species	Target Habitat	Advantages	Disadvantages
Trawling	Otter trawl Beam trawl dredge	Towed transects	Species id, abundance, length, weight, physiological data (sex, condition)	Gear-specific	Sandy, soft or muddy bottoms	• Long tradition in a lot of international surveys • common technique for sampling demersal species • provides relative abundance estimates • can detect small, cryptic and burrowing species • can't be used in structurally complex habitats	• destructive • only provides • semiquantitative data biased towards smaller and slow-moving species
Passive fishing	Traps Vertical hook and line fishing angling Gill nets	Sets of passive fishing gears set in a transect and soaked for a specific amount of time	Species id, relative abundance, lengths	Gear-specific	All habitats	• Usable in structurally complex habitats • accurate size measurements • no post-survey laboratory analysis	• Inability to define sampling area • No information about influence of environmental parameters (currents, bait plume) • difficult to quantify • high selectivity • damage and stress to fish often results in death of the individuals • Interspecific competition may cause bias

Split-beam echosounder

Split-beam echosounders are probably the most common method of hydroacoustic monitoring of fisheries resources and are used in a variety of traditional surveys. They are commonly used in monitoring pelagic fish stocks, like the Baltic International Acoustic Survey (BIAS) or the Baltic Acoustic Spring Survey (BASS). Other application areas include deep waters off the continental shelf (Johnston et al., 2006), shallow estuaries (Boswell et al., 2007) and mangroves (Krumme and Saint-Paul, 2003; Murphy and Jenkins, 2010). They commonly use horizontal and vertical beams to record the movement of fish and post-processing echo integration software to measure recorded biomasses. Advantages of this method are the ability of estimating absolute population estimates, the ability to map large areas quickly in a non-destructive way (Johnston et al., 2006) and immediately retrieve results. However, the exact taxonomic identification and individual fish length has to be proved by ground-truthing, which requires additional fishing hauls. Also split-beam echosounders are unable to detect species closer than 2m off the seafloor (Johnston et al., 2006), and therefore, are not appropriate for demersal fish species.

Acoustic camera

The dual frequency identification sonar (DIDSON) produces near video-quality images of fishes by using sound (Holmes et al., 2006; Murphy and Jenkins, 2010). DIDSON can detect fish at ranges up to 15m from the camera at high frequency setting and 40 m distance at low frequency settings (Holmes et al., 2006; Murphy and Jenkins, 2010). Types of data include abundance estimates, length data and fish behaviour (Moursund et al., 2003; Rose et al., 2005). The acoustic camera can be used in turbid and low light waters, and thereby, provide a great alternative to underwater visual census (UVC) or underwater video. It has already been proven to be an effective tool for fisheries stock assessments (Moursund et al., 2003). However, the application of DIDSON is limited to small-scale studies, and habitat structure can block the beams (Holmes et al., 2006; Murphy and Jenkins, 2010). Further, it only provides low taxonomic resolution, is expensive, and image analysis is relatively time-consuming (Holmes et al., 2006; Murphy and Jenkins, 2010).

Acoustic tagging

Acoustic tagging, besides other form of tagging such as radio frequency (RFID) tagging or satellite tagging is mostly used for studies on larger, mobile species and in association with MPA monitoring. Areas of research include homing behaviour, movement and activity patterns, migration and use of space (Egli and Babcock, 2004; Lowe et al., 2003; Meyer et al., 2007). The type of acoustic tag used depends on the exact research objective. For short-term but fine scale movement data, active tracking of the acoustically marked animal is carried out with the help of directional hydrophones and acoustic receivers (Afonso et al., 2008; Murphy and Jenkins, 2010; Zeller, 1998). Residency and movement pattern data can be provided by a passive monitoring system using radio acoustic positioning telemetry (RAPT) buoy systems. Acoustically tagged fish within the RAPT system range are detected by acoustic receivers on buoys which communicate with a base station by radio signals or cable (Jorgensen et al., 2006).

Long-term and large-scale studies use independent acoustic receivers to record movement patterns of acoustically tagged fish (Murphy and Jenkins, 2010; Starr et al., 2005). Thereby, acoustic receivers record time, date and identity of the tagged fish. Acoustic tagging techniques provide a great method to investigate spatial and temporal movement patterns of various species without recapture, which is required for standard tagging. However, due to high costs associated with tags, sample sizes are usually small, reducing statistical power (Jorgensen et al., 2006; Zeller, 1999). Additionally, the lifespan of tags is limited. Furthermore, fish need to have reached a certain size in order to be tagged. Further, fish must be extracted prior to tagging using conventional methods and therefore cannot be considered isolated from other methods. Tagging, consequently, always involves the use of any of the passive or active methods, which is the focus of our manuscript, which needs to be considered when selecting a tagging method.

Underwater video

The development of underwater video technology can be traced back to the middle of the 20th century (Barnes, 1952). Technological developments since then have led to an enormous improvement in these methods and thus also to an increasing use of camera techniques in marine research (Mallet and Pelletier, 2014). The fields of application, as well as the technical details of the different methods vary greatly and offer a large pool of possibilities for non-destructive sampling methods. Video methods in general offer several advantages but also disadvantages compared to conventional methods like trawling. Video methods are by their design non-invasive and non-extractive and therefore have very little impact on the marine environment and can be applied in almost every habitat. They generate large datasets with extensive spatial coverage over a wide depth range (Letessier et al., 2015). In addition, video techniques can be applied in habitats where the use of other methods is not possible or prohibited. Thus, video techniques are more suitable for highly structured habitats (Starr et al., 2016) and possibly provide an option for use in offshore wind farms (OWFs) or wave farms (Sheehan et al., 2010). Video techniques require less scientific staff and can also be used from smaller vessels (Sheehan et al., 2010). Further, species selectivity is considered lower for video techniques (Bacheler et al., 2013a; Christiansen et al., 2020; Harvey et al., 2012), and the probability of detecting rare species is considered to be higher (Goetze et al., 2019; Langlois et al., 2020). However, systematic selectivity studies for video-techniques are still seldom as well as studies focussing on effective sampling design. Furthermore, analysis of video-material can be rather time-consuming, which might be the biggest challenge for future research. Another major limitation is the dependency of video-recording on relatively good visibility in the water.

Table 2: Overview of different stationary video observation categories, subcategories and references.

Category	subcategory	References
RUV	H-RUV	(Fedra and Machan, 1979; Jan et al., 2007)
RUV	Rotating	(Aguzzi et al., 2011; Pelletier et al., 2021a, 2021b, 2012)
RUV	360°	(Mallet et al., 2021)
BRUV	H-BRUV	(Ellis and DeMartini, 1995)
BRUV	V-BRUV	(Priede and Merrett, 1996; Willis et al., 2000; Willis and Babcock, 2000)
BRUV	360°	(Schobernd et al., 2014; Wells et al., 2008)
BRUV	Stereo	(e.g. Harvey et al., 2012; Langlois et al., 2015; Letessier et al., 2015)
BRUV	Rotating stereo	(Starr et al., 2016)

Stationary video

A stationary unit can be anything, and hardware configuration can range from small tripods to large and heavy metal slides. They are categorised as remote underwater video (RUV) and baited remote underwater video (BRUV). Within these categories, there are wide variations in hardware and technical details, most of which are applicable to both categories (Table 2). First of all, a distinction can be made between the camera orientation, which can be either horizontally (H-RUV, H-BRUV) or vertically (V-RUV, V-BRUV) aligned (Ellis and DeMartini, 1995; Fedra and Machan, 1979; Jan et al., 2007; Priede and Merrett, 1996; Willis et al., 2000; Willis and Babcock, 2000). A horizontal orientation of the cameras is the most common, providing a greater field of view and reducing the risk of the system affecting behaviour. Vertical alignment, on the other hand, offers the advantage of a simple calculation of the field of view as well as the possibility to easily perform length measurements. However, to our knowledge no studies focussing on fish using V-RUV have been conducted. Some studies also focus on a 360° view, which can be achieved either by using multiple cameras in an array, 360° cameras or rotating systems (Aguzzi et al., 2011; Mallet et al., 2021; Pelletier et al., 2021b, 2021a, 2012; Schobernd et al., 2014; Starr et al., 2016; Wells et al., 2008). Many of these techniques can be used not only with a single camera but also with a stereo camera, allowing more accurate length and field of view calculations (Harvey et al., 2002). This variety of technical setups can be applied to both RUVS and BRUVS. The precision of the monitoring differs from individual setups and purposes of the programme. Generally, studies using stationary systems assess species richness, relative abundance and length. Most studies use the relative abundance measure of MaxN (Priede and Merrett, 1996), which describes the maximum number of individuals of one species in a single frame. Video based methods, especially BRUV-systems have become very popular (De Vos et al., 2014; Whitmarsh et al., 2017). Besides the general advantages video-based methods offer, stationary systems have the advantage over towed video systems that fish are not scared by noise or movement of the system. They also have the potential to be used by non-scientific staff (Mallet et al., 2014) offering the integration into citizen science programs. Additionally, they can be deployed over long time periods offering the possibility to observe different activity patterns influenced by diurnal variation. Since stationary systems similar to passive gears are dependent on the activity of the fish, calculation of total abundances is not possible, observation duration is rather long and zero counts are common. Though the use of bait can attract a larger number of animals, it can introduce bias towards predatory and scavenger fish whose presence may deter other species. Further, the effect of bait and bait plume remains rather unknown (Mallet et al., 2014). In addition to the general disadvantages of video-based systems, the above-mentioned limitations and the typically poor visibility in temperate waters result in a seldom use of stationary systems in temperate regions.

Towed video

Towed camera systems can be either sledges that lightly touch the bottom or towed bodies that hover just above the seabed. Most of the above-named technical details for stationary systems are simultaneously applicable to towed systems. They can be either oriented vertically or horizontally, or use a combination of both (Trobbiani et al., 2018). Towed systems can also be equipped with either a single or a stereo camera, but to our best knowledge there has not been a device using 360° cameras. Towed systems are used for the detection of benthic invertebrates, habitat classification and/or detection of fish and are e.g. used in standardized surveys for anglerfish (*Lophius* spp.) (McIntyre et al., 2013). They can be used in various depths and are often not limited by habitat structure. Comparisons of beam trawls, diver transects and towed video for estimating the abundance of juvenile flatfishes have shown equal performance of the different methods (Spencer et al., 2005), making towed video an adequate non-destructive sampling alternative, which is inexpensive and simple to operate (McIntyre et al., 2015, 2013; Spencer et al., 2005). These systems are unrestricted to tow duration and efficient to record large areas of the seafloor quickly (McIntyre et al., 2015; Sheehan et al., 2010). However, a main disadvantage of towed systems may be due to the stability of the device, as towed camera systems usually have difficulties maintaining a constant height above the bottom (McIntyre et al., 2015). Consequently, difficulties arise in determining the exact transect size and complex evaluations of inclination and distance from the device are required. Other disadvantages and limitations might be the influence of environmental conditions on towing speed or visibility (McIntyre et al., 2013) or bias caused by fish behavioural responses to the towed video system (Stoner et al., 2008).

Molecular methods

Environmental DNA (eDNA) is a relatively new and promising non-invasive tool of molecular methods for marine environmental monitoring (Garlapati et al., 2019; Hansen et al., 2018; Maureaud et al., 2021; Salter et al., 2019). It is based on the principle that DNA is continuously released from organisms in the environment (Hansen et al., 2018). It describes the process of collecting DNA from different environmental samples (such as water, sediment, etc.) with the objective of obtaining information about biodiversity (Garlapati et al., 2019). Commonly water samples are first filtrated, followed by the extraction and preservation of DNA. eDNA is then extracted from the filter, sequenced, detected and then analysed (see Garlapati et al., 2019). eDNA analysis can be based on two different approaches, depending on the aim of the study. The species-specific approach is based on eDNA barcoding, while the multispecies approach is based on eDNA metabarcoding (Garlapati et al., 2019; Hansen et al., 2018). eDNA offers a cost-efficient, quick and sensitive method (Garlapati et al., 2019; Hansen et al., 2018) to detect invasive species (Gold et al., 2021). Thereby, it can potentially detect the presence of fish in bottom waters and is regarded to be especially promising for studying rare marine species, which are hard to detect by traditional methods (Afzali et al., 2021; Garlapati et al., 2019; Maureaud et al., 2021; Russo et al., 2020; Salter et al., 2019). For instance, studies in the West Antarctic Peninsula revealed signatures of benthic invertebrates, endemic fishes and king crabs (Cowart et al., 2018). Also studies in the Baltic Sea proved eDNA to be a successful tool in detecting fish communities (Sigsgaard et al., 2017; Thomsen et al., 2012). And species-specific analysis indicate that eDNA concentrations correlate with biomass and abundance (Doi et al., 2015; Hansen et al., 2018; Maruyama et al., 2014; Takahara et al., 2012).

However, abundance estimation is usually problematic, as concentrations of eDNA are influenced by many factors. First, organismal production rates, release of DNA, can differ between species, size and metabolic rates. Therefore, studies need to investigate the influence of environmental factors on metabolic rate and release of particles for different species and sizes. Second, degradation rates of eDNA particles can vary between ecosystems and areas most likely related to other environmental processes. Studies on the influence of abiotic and biotic factors on degradation have shown that temperature, solar radiation and pH-concentration cause large variations in the persistence of eDNA particles (Hansen et al., 2018; Strickler et al., 2015). Third, the physical transport in order to determine the exact origin of the eDNA is largely unknown, which is another challenging issue in terms of eDNA analysis (Hansen et al., 2018). Currently fish is the most studied group with the eDNA approach and there is already good concordance between eDNA surveys and traditional survey methods in terms of species detection (Hansen et al., 2018; Maruyama et al., 2014; O'Donnell et al., 2017; Salter et al., 2019). However, eDNA results are still seldomly used in ecosystem management, as the effective use of eDNA tools would require further addressing of the above-mentioned uncertainties that exist in data elucidation (Garlapati et al., 2019). A major problem arises with the false positives and false negatives and the reasons for those have to be investigated further (Garlapati et al., 2019).

Table 1: Summary of modern techniques used for monitoring fish communities.

Observational method	Categories	Sampling strategy	Type of information	Target species	Target habitat	Advantages	Disadvantages
Video	BRUV RUV 360 ° BRUV /RUV Stereo BRUV/RUV	Sets of 3-4 stationary devices set in a transect and soaked for a specific amount of time	behavioural studies, species composition, relative abundance, lenghts	diverse	Not habitat specific, commonly used in benthic habitats	• Non-invasive • Easy and fast deployment • no requirement of expert staff • survey large areas • almost no depth and time limitation • video-annotation • enables annotators to work collaboratively	• Time consuming video analysis • double counting possible • dependent on visibility • bait can cause bias towards specific species • calculation of total abundance not possible
Towed video	Video sledge Towed body	Transect lines	Species-id, counts, densities, lengths, fish behaviour	Fish and benthic invertebrates	Not habitat specific, commonly used in benthic habitats	• Non-invasive • inexpensive • simple to operate • unrestricted to tow duration • mostly no influence on fish behaviour • can provide behavioural data • cheaper compared to ROVs • ideal for sampling large areas quickly •cost-effective • low crewing requirements • can be applied from small vessels or boats	• accurate species-id can be difficult • risk of double counting • depends on visibility and light • time-consuming video analysis • dependent on the stability of the platform • bias can be introduced by fish behaviour
Split-beam-echosounder	-		Biomass, total abundance	Pelagic fish	Pelagic	• Ability to survey large areas quickly • Non-destructive gear is portable easy to use	• Needs ground truthing for species-id • can't be used <2m off the ground
Acoustic Camera	-		Abundance, length, behaviour		Not habitat specific	• Not dependent on visibility or turbidity • non-destructive	• Habitat structure can block beams • limited to small-scale studies • low taxonomic resolution • expensive and time-consuming analysis
Acoustic Tagging	-	Tagging of target species	Actitvity patterns, movement patterns	Larger, mobile species	Not habitat specific	• no recapture necessary	• expensive
eDNA	-	Water or sediment samples	Species-id	Marine live in general	Not habitat specific	• non-invasive • high taxonomic resolution • cost-efficient	• many uncertainties regarding, transport and concentration of particles

Fit-for purpose monitoring – how to choose

Monitoring is a complex issue and several factors need to be considered when deciding on the appropriate monitoring concept. A monitoring concept should be designed based on the questions to be answered and data requirements, on habitat characteristics, target species and resource availability (Henseler and Oesterwind, 2023) as well as conservation management decisions and technical restrictions. This also means that in cases where extractive methods are not absolutely necessary to answer the specific question (e.g. documentation of abundance & biodiversity), the use of alternative less invasive methods can be considered. Based on this review we identified criteria, which we suggest are fundamental in order to choose the appropriate method for monitoring (Figure 1). It should be noted here, that the respective classification of methods is referring to various criteria and is not always appropriate for all subcategories of the individual methods. In this context, it is important to emphasise that none of the described methods can provide true species richness or abundance/biomass information (Hansen et al., 2018). This is due to the specific selectivity and catchability of the individual method, which is why it is crucial to estimate selectivity patterns for each type of method within the development of monitoring programs (Christiansen et al., 2020). Additionally, combining different methods can be effective in reducing method specific bias and can increase the range and detail of data from spatial surveys (Murphy and Jenkins, 2010; Watson et al., 2005; Willis et al., 2000). Another issue that should be considered is the required workload. Despite certain methods might be less expensive and are less labour intensive in the field, post-fieldwork workload can increase compared to traditional methods. This applies in particular to hydroacoustic and video-based methods, as post-processing of data can be very time-consuming. However, with the rapid development of artificial intelligence-based methods, the workload for the processing of these data could soon be reduced.

Figure 1: Different target criteria and attribution of methods. The first row refers to criteria related to the target parameters, second row refers to criteria related to target species/groups and habitats, third row refers to criteria related to resources needed. In case not all methods belonging to one observational category are appropriate for the respective target, individual methods are listed.

Baltic case study

The Baltic Sea is a semi-enclosed, low-diversity brackish sea with a long history of anthropogenic use and currently suffers from multiple stressors like eutrophication, deoxygenation, acidification, warming and overfishing (Reusch et al., 2018). There is an increasing demand of Offshore Windfarms with a planned increase in the long term from 31 OWFs with a spatial expansion of around 500 km² to 149 OWFs with an approximate 5-fold expansion (Haase et al., 2023). Currently fishing for commercial use and fisheries research is prohibited (Haase et al., 2023) within OWFs. Besides the drastic expansion of OWFs in the Baltic Sea, marine protected areas (MPAs) will likely be expanded so that 30 % of the EEZ are under protection and 10 % of those under special protection by 2030 (EU, 2021). A special case of MPAs are the ‘sites of community interest’ (SCI) and ‘special protection areas’ (SPA) which form the Natura2000 network. In the German EEZ of the Baltic Sea six of those MPAs were implemented in 2017 with different management options. At least in parts of the MPAs commercial fishing with mobile bottom contacting gears will be excluded most likely in 2024, to protect reef and sandbank structures. Since it is not finally clarified whether traditional monitoring methods like bottom trawling are still viable in MPAs and are no longer an option in OWFs, the areas where traditional mobile bottom contacting methods have been used decrease while the area of "black boxes" in fisheries science increase. But to gain knowledge about the impact on the fish community is of high importance because some sampling stations of the ICES coordinated Baltic International Trawl Survey (BITS), which aims to provide data on demersal commercial species, in fact collide with OWFs in the Baltic Sea (Haase et al., 2023) (Figure 2) and become "black boxes" for fisheries management. Therefore, adaptive methods to collect data are required to be able to i) detect changes in fish communities occurring in these areas and ii) provide information for fisheries management (ICES, 2023) if necessary.

To do so, data is required to evaluate relative changes in fish species diversity, abundance and biomass, as well as in food web, when assuming that the benthic fauna is changing after the exclusion of mobile bottom contacting gears or due to reef effects due to OWPs, in order to address the condition of fishes and fish communities (see Figure 1). However, in order to assess changes in species diversity a multi-species approach might be required. Furthermore, the respective method should be applicable in multiple and structural complex habitats since Baltic Sea MPAs are characterised by gravel and coarse, stony reefs or sandy bottoms. Therefore, a combination of different methods might be the best solution for a future monitoring strategy in MPAs and OWFs.

Common methods for monitoring MPAs include underwater video census (UVC), video-based observations, passive fishing gear or hydroacoustic methods. However, as mentioned above, most of them are often used in tropical reef areas, where conditions differ from those in the Baltic Sea. UVC methods for example require the engagement of divers, which in return results in depth and time limitations. Further, visibility and light conditions are often better in tropical and reef areas compared to temperate deeper areas, which results in problems using underwater video. In the Baltic Sea, for example, increased primary production results in a reduction of light penetration during blooming periods (Hopkins, 2000). However, since video-based methods have successfully been used for studies in Baltic Sea cobble and artificial reef areas and sand habitat (Rhodes et al., 2020; Wilms et al., 2021), they can be considered as a promising alternative. Since MPAs cannot be isolated from many activities and impacts outside their boundaries (Halpern et al., 2010), changes in MPAs are more likely expected to occur in less-mobile or less-migratory species (Pilyugin et al., 2016), especially in the Baltic MPAs which are relatively small. Less-mobile species are often associated with a benthic lifestyle, which further minimises the pool of appropriate methods. Therefore, most hydroacoustic methods are not applicable here (see 4.1). However, acoustic cameras still provide a good opportunity for collecting abundance data in a relatively large spatial frame. Monitoring MPAs often aims to provide data on species diversity and follows a multi-species approach. Therefore, high taxonomic-resolution is needed, which might require the use of complementary methods like eDNA or extractive methods like potting or gill-netting. Methods used in studies evaluating the effect of OWFs on fish communities include trawling, seine and gill netting, angling, hydroacoustics, underwater video, dredging and UVC (Andersson et al., 2007). Since, the use of mobile bottom contacting gears is prohibited in OWFs in the German EEZ dredging, trawling and seine netting will most likely not be possible. Further, the above-mentioned data requirements and constraints for MPAs will most likely also apply to the case of OWFs. Consequently, methods that might be fit for purpose for monitoring fish communities in MPAs or OWFs include passive fishing (e.g. gill netting, angling), underwater video (towed or stationary), eDNA or hydroacoustic methods (acoustic tagging, acoustic camera). However, none of the above-mentioned methods will be able to show the complete picture. Due to gear specific catchability and selectivity, biases of each method have to be assessed beforehand in order to develop an appropriate monitoring strategy. Pre-investigations on the use of alternative methods, especially video stations and eDNA, have so far only been able to detect a few species. However, eDNA in particular was able to record species that were not found in the catches of the 2m or 3m beam trawl (Hammerl, unpublished). Overall, the sampling effort for the alternative methods was low and a direct comparison with the beam trawl catches was not always possible. The number of species observed will most likely increase with increased effort. Beside the testing of alternative monitoring methods, monitoring design and optimal effort will be developed.

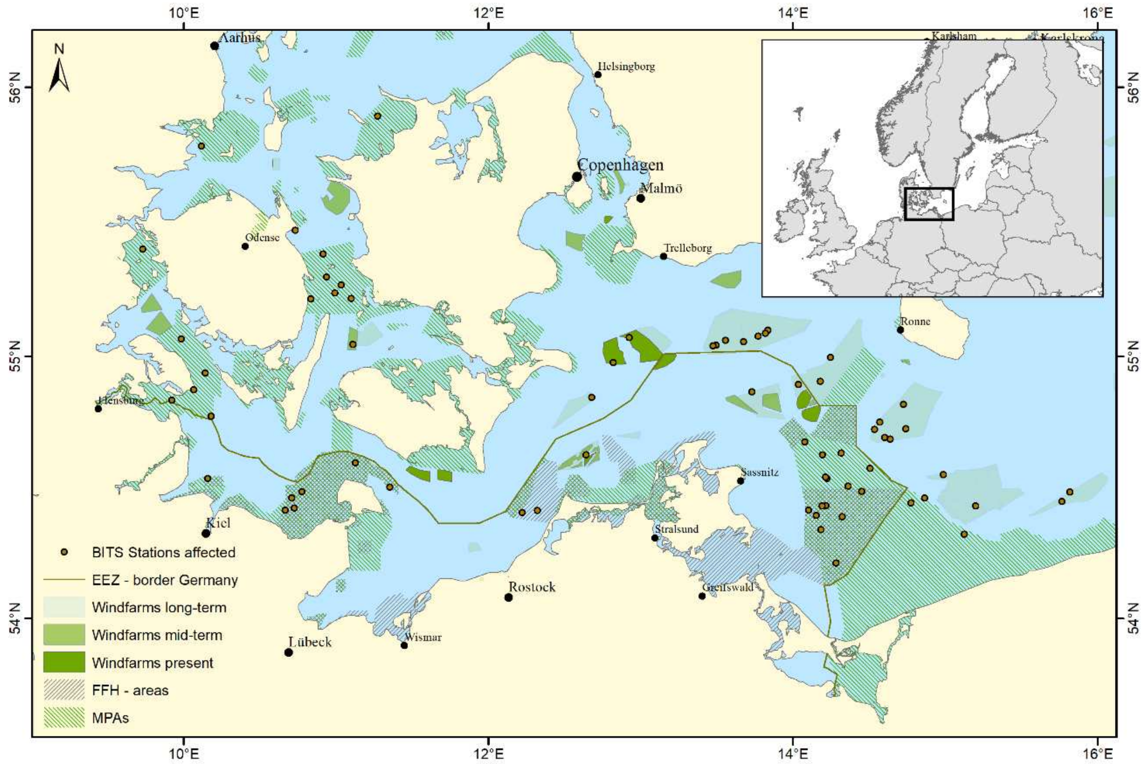

Figure 2: Western Baltic Sea including FFH–areas in- and outside of the German EEZ and coastal waters, Offshore Windfarms (OWFs) of different scenarios (defined according to Stelzenmüller et al. 2022) and affected stations of the Baltic International Trawl Survey (BITS).

Conclusion

Our study revealed that while new sampling methods are already accessible in certain cases, they often require further refinement and development and provide less information compared to traditional bottom trawling. However, in some cases the less available data can be sufficient to address specific research questions. Since all methods have diverse and perspective-dependent advantages and disadvantages, identification of a suitable method can be challenging. The use of non-or less invasive monitoring methods in our case study is still very limited. Designing such a monitoring can be complex, especially in terms of standardization, sampling design and effort. Information about gear performance, effort quantification, catchability and selectivity of different gears have to be collected to design a cost-efficient and statistically robust monitoring program for marine protected areas where mobile bottom trawling will likely be excluded. Additionally, integrating the here discussed methods into existing long-time series requires calibration experiments to ensure the quality and the right interpretation of the data. Moreover, the results show that bottom trawling is still needed, because there is currently no alternative sampling method for the data requirements of EU fisheries management for example. Data on age structure and reproductive capacity cannot be generated without extracting the fish and often requires the dissection of the fish. However, it becomes clear, that due to the current and future restrictions in fisheries data collection methods, alternative data collection methods become increasingly important.

Author Contributions

CH developed the conceptual frame, designed figure 1 and wrote the first draft of the manuscript. DO and CM contributed substantially to the conceptualization and revised the manuscript. All authors approved the manuscript for publication.

Funding

This work was carried out as part of the DAM pilot missions "MGF-Ostsee" and “MGF-Ostsee II” funded by the German Federal Ministry of Education and Research (funding codes 03F0848E/03F0937E).

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.

Acknowledgements

We thank everyone involved in ICYMARE for the organization of the special issue and funding of the publishing costs. We also thank Nakula Plantener for providing help with figures and designing figure 2., as well Christopher Zimmermann for his valuable comments during the writing process.

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Supplementary material

Table 3 : Combination of Keywords Used for Literature Search. One term from Level 1 and Level 2 keywords was consistently included and used in various combinations. All combinations were then paired with optional keywords.

Keyword Level 1	Keyword Level 2	Optional keywords
Monitoring/Sampling	Fish/fish assemblages/ fish communities	Marine Protected Areas Passive fishing Underwater video Acoustic methods/hydroacoustics eDNA

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STUDY II

Remote underwater video (RUV) as a non-invasive alternative to bottom trawling for monitoring temperate soft-bottom fish assemblages in Marine Protected Areas

To be submitted to Regional Studies in Marine Science in April

Title: Remote underwater video (RUV) as a non-invasive alternative to bottom trawling for monitoring temperate soft-bottom fish assemblages in Marine Protected Areas

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Abstract

Traditional monitoring of demersal fish communities is often based on invasive methods, such as bottom trawling. These methods can have detrimental impacts on the ecosystem by, e.g., disturbing the sediment and its associated fauna and flora, and might not be in line with management measures of marine protected areas (MPAs). A regular monitoring of MPAs is, however, crucial to understand whether management measures, like the prohibition of certain types of fishing or other activities, are effective. Remote underwater video (RUV) is a non-invasive alternative for monitoring fish used successfully in many bioregions on a global scale. Yet, in the Baltic Sea, a brackish sea with overall low fish diversity and medium visibility, video-based methods have rarely been tested with regard to monitoring fish communities, especially in sandy habitats. To fill this gap, we tested whether RUV represents a non-invasive alternative to beam trawling by means of computing commonly used fish diversity metrics, including species richness and abundance, as well as evenness. We furthermore compared the ability of beam trawling and RUV to detect richness as a function of sampling effort. Monitoring with RUVs generally resulted in diversity indices lower than derived by beam trawls. Consequently, a higher sampling effort was required for RUVs to obtain the same level of richness compared to beam trawls. While, RUVs seem to offer a non-invasive alternative to conventional methods in many bioregions, their effectiveness in monitoring demersal fish communities in Baltic Sea MPAs may be limited by species detection and effort required.

Keywords: remote underwater video stations, RUV, trawling, marine protected areas, fish communities

Introduction

Marine protected areas (MPAs) are extensively used as a management tool to conserve ecosystems that are threatened by increased human impacts (Bayley et al., 2019). They can provide benefits for biodiversity conservation and fisheries (Edgar et al., 2007). In recent years, the number of MPAs has increased considerably (Kriegl et al., 2021) as a result of calls from the Convention on Biological Diversity to increase protected areas up to 30% by 2030. The European Union ratified the goal of rendering 30% of water regions into marine protected areas by 2030, among these 10% under strict protection (EU, 2021). A special case of European MPAs are the `Sites of Community Interest´ (SCI) and `Special Protection Areas´ (SPA) which are represented in the Natura2000 network (Zoppi, 2018). Six of those MPAs were implemented in the German Exclusive Economic Zone (EEZ) of the Baltic Sea in 2017 and are presently, at least in part, protected from mobile bottom contacting gear (European Commission and STECF, 2023). This management measure was implemented to protect reef and sandbank structures. To prove the effectiveness of such measures and, thus, ensure successful conservation management, effective monitoring and evaluation are crucial (Dunham et al., 2020).

Monitoring approaches targeting (demersal) fish communities are typically based on bottom trawling (Wells et al., 2008; McIntyre et al., 2013). For example, in SCI and SPA of the western Baltic Sea, 3 m-beam trawls are currently used for monitoring the demersal fish community. Yet, bottom trawling can be destructive to the marine environment and can have significant impacts on the seabed, as well as adverse effects on organisms associated with the seafloor (Johnson et al., 2015). Alternative non-invasive monitoring approaches are required for areas that are implemented to protect biodiversity (Edgar and Barrett, 1997). Consequently, non-destructive and non-extractive methods, such as underwater video, are becoming increasingly relevant (Collie, 2000; Murphy and Jenkins, 2010; Sciberras et al., 2018; Nalmpanti et al., 2023).

In the last decades, video-based methods have gained popularity in marine science due to the diverse implementation options and technological advances (Mallet and Pelletier, 2014; Nalmpanti et al., 2023). Especially the use of stationary systems, like remote underwater video (RUV) or baited remote underwater video (BRUV), has increased in the last years, particularly in Australian waters (Bailey et al., 2006; Mallet and Pelletier, 2014; Whitmarsh et al., 2017;

Harvey et al., 2021; Nalmpanti et al., 2023). Yet, video-based approaches are also becoming more important in other parts of the world, for example, to support stock assessment of sea bass in the Southeast Atlantic (Bacheler et al., 2022). Studies have shown that species selectivity by BRUVs is often lower compared to conventional methods (Harvey et al., 2012; Bacheler et al., 2013; Christiansen et al., 2020), but that detection probability of rare species might be higher (Goetze et al., 2019; Langlois et al., 2020). Among video-systems, stationary systems have the benefit that fish are not deterred by the system's movement or noise and that they can be deployed over extended periods providing insights into fish activity patterns influenced by diurnal variations (Mallet and Pelletier, 2014; Murphy and Jenkins, 2010). Moreover, stationary systems can also be applied by non-scientific staff enabling the usage in citizen science projects (Cappo et al., 2003; De Vos et al., 2014; Murphy and Jenkins, 2010). While the non-extractive nature of RUVs is a major benefit for their application in protected areas, they require good visibility in the water column and provide only relative abundance estimates and the analysis of video material is time-consuming (Mallet and Pelletier, 2014; Shah Esmaeili et al., 2021). Nevertheless, nowadays RUV systems are commonly used to evaluate MPA effectiveness and have been able to detect early signs of success, such as increases in fish body size in areas where fishing is prohibited (Jaco and Steele, 2020), making them a promising alternative for the monitoring of fish communities, also in the Baltic Sea. The Baltic Sea however, is a unique ecosystem with special characteristics, that differ considerably from those of ecosystems where video-based methods are typically applied. The brackish conditions associated with low species and functional diversity with numerous morphologically similar species, as well as the relatively shallow water depth and poor visibility as a consequence of relatively turbid waters (Ojaveer et al. 2010), may limit the applicability of video-based methods in this ecosystem.

A general requirement when replacing a traditional sampling method with a new one, such as using non-invasive video-based monitoring methods instead of bottom trawling, is to compare the catch efficiency of both methods to ensure that the results are not biased by differences in catchability (Holmes et al., 2013). This becomes especially important, when the new method is to be applied in bioregions, where it has not been evaluated. Yet, most existing studies, evaluating the performance of RUVs, do not compare optical methods with conventional bottom trawling or restrict their comparisons to passive fishing methods alone. Moreover, research in this context mostly focusses on structured seabed habitats, such as reefs or seagrass meadows (Wells et al., 2008; Unsworth et al., 2014; French et al., 2021), where the biodiversity and abundance of individuals is often higher compared to less structured habitats, such as sandbanks (Gomes et al., 2024).

To fill this gap, our study tested whether RUVs can provide scientifically sound information to evaluate the effectiveness of the exclusion of mobile bottom contacting gear from MPAs in the Baltic Sea by comparing the performance of RUVs to the previously applied 3m beam trawl. To do so, the fish community was sampled in two MPAs in the Baltic Sea, and commonly applied biodiversity measures were computed and compared between the two methods. Furthermore, the sampling efficiency of the two methods was evaluated with regard to the effort required to obtain a certain species richness. The results of this study demonstrate that while RUVs can provide a non-invasive alternative to conventional methods in many bioregions, their efficacy in monitoring benthic and demersal fish communities in Baltic Sea MPAs may be constrained by their limitation in species detection and the effort required. Thus, findings of this study can help to identify fit-for purpose methods for the evaluation of MPA effectiveness in habitats with a low structural complexity and diversity, and, therefore, support the development of an urgently needed monitoring strategy based on non-invasive methods for protected areas in the Baltic Sea.

Materials & Methods

Study Area and Sampling Design

Fish were sampled in two distinct areas (Fehmarnbelt and Odra bank), each situated in a Marine Protected Area (MPA) within the German Exclusive Economic Zone (EEZ) of the western Baltic Sea (Fig.1.). Both MPAs were implemented in 2017 and are part of the Natura2000 network (BfN, 2020). The MPA Fehmarnbelt, covering an area of 27.992 ha, is located north of the German island “Fehmarn” and characterized by sandy habitats and stone reefs at depths of 17 to 35 m and salinities between 17 and 23. For this study, we chose an area in the western part of the MPA (size: 400 ha) which is mainly characterized by sandy and muddy sediments. The MPA Odra bank, with a total size of 110.115 ha, is situated east of the German island “Rügen” and characterized by sandbanks with medium to fine sand and a maximum depth of 20 m, as well as low salinities of 5 to 8. Sampling took place in the eastern section of the MPA (Figure 1). Earlier investigations revealed significant differences in species diversity and abundances between Fehmarn and the Odra bank, likely driven by decreasing salinity gradients from the west towards the east in the Baltic Sea (Reusch et al., 2018).

All sampling was conducted in May and June 2023 aboard the fishery research vessel FFS Solea. The fish community was recorded/sampled using four unbaited remote underwater video stations (RUVs) and a 3m-beam trawl with a mesh size of 20 mm in the codend and a height of 0.45 m. The design of the RUVs follows the recommendations of Langlois et al. (2020), ensuring that RUVs are as compact and handy as possible. RUVs consist of a stainless-steel frame with dimensions of 110 cm width and 60 cm height. Two cameras (GoPro Hero8) were mounted onto the frame with a distance of 50 cm and a 5° convergence angle, positioned approximately 25 cm above the seafloor. An additional light (bigblue VL4600PB) was installed diagonally above each camera at the upper edges of the frame. Each light (n = 2 in total) had a beam angle of 120° and was used with a luminous flux of 2100 lumens. To provide better stability, weights were attached to the bottom of the frame. Each individual frame was connected to a floating line that was attached to two buoys to mark the position of the RUV on the sea surface and to facilitate retrieval.

The sampling design was arranged according to a grid scheme, which covered the complete study area, but allowed the sampling stations to be treated independently (Figure 1). We followed a similar strategy like Cappo et al. (2004), where one sampling station within each study area consisted of four RUVs and one beam trawl haul. The four RUVs were set in a row according to the sampling grid with a distance of about 300 m and were retrieved after approximately 60 minutes, to ensure that at least 45 min video material could be analysed. After RUV retrieval, one 15-minute beam trawl haul was conducted along the transect of the four RUVs with an average speed of 3 kn starting at the location where the first RUV had been retrieved to ensure a maximum time span between camera deployment and trawling. The catches were processed directly on board, fish were identified to the lowest possible taxonomic level, counted and weighed. Due to the failure of individual RUV stations along the transect line as a result of poor visibility, the intended pairwise comparisons between RUVs and beam trawls at transect level were not feasible. Instead, the analysis was switched to a more comprehensive comparison between the two methods as a whole, where biodiversity indices from both methods were compared directly, without the originally planned paired comparison.

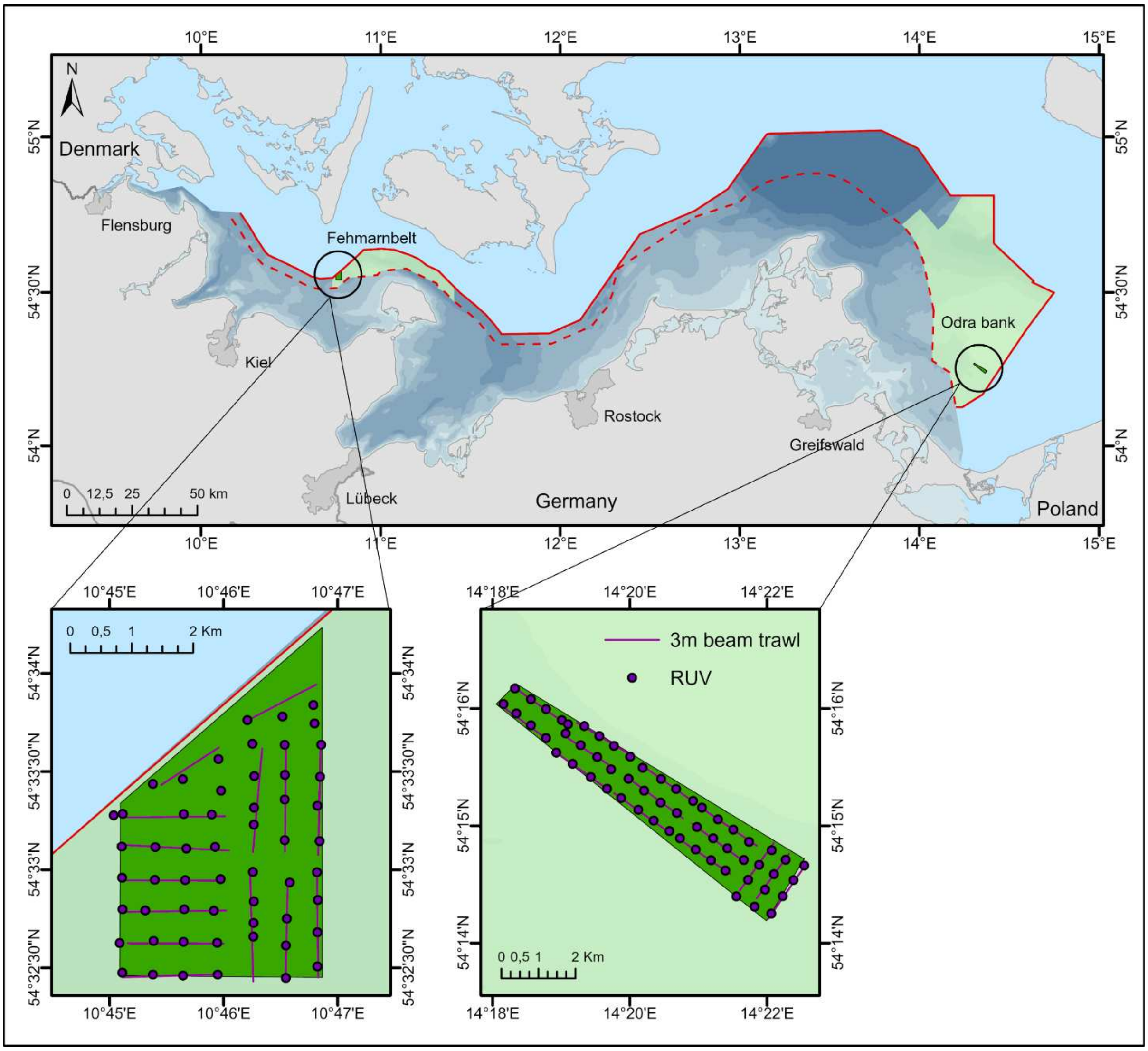

Figure 1: Top figure: Location of the two Marine Protected Areas (Fehmarnbelt (west) and Odra Bank (east)) in the Western Baltic Sea, indicated by black circles. The solid red line represents the boundary of the German Exclusive Economic Zone (EEZ), while the dashed red line marks the extent of German territorial waters. Bottom figures: Detailed layout of studied areas (left: Fehmarn, right: Odra bank) and sampling design. Violet dots represent RUV stations, violet lines indicate trawl transects.

Data analysis

A total of 118 stations were considered in the analysis, including 28 trawl transects, 14 in each area, and 90 RUV stations, of which 56 were carried out in the Odra bank and 34 in the Fehmarnbelt. All videos were analyzed using the software Eventmeasure from SeaGIS (www.seagis.com.au). Prior to analysis, all videos were skimmed through and checked for sufficient visibility to analyse the full 45 min. Due to visibility constraints or failure of light, twelve RUV stations from the Fehmarnbelt area (n=46) could not be analysed, resulting in a total of 34 usable video recordings for this area. Analysis of recordings started three minutes after the RUV had reached the seabed to allow sediment to settle. All fish were identified to the lowest possible taxonomic level. As a measure of relative abundance, we used MaxN, a conservative measure that describes the maximum number of individuals of a species in any one frame within a video (Cappo et al. 2003). The MaxN measure avoids pseudo-replication caused by individuals that swim in and out of the camera's field of view (FOV) (Willis et al. 2003, de Vos et al. 2014).

To assess how the diversity of fish communities differs between fishing methods, we computed several diversity indices, that are commonly applied to describe fish communities. This included abundance, the Shannon index (Shannon, 1948) evenness (Pielou, 1966) and the number of families observed with each method. We chose the Shannon Index since it provides a more comprehensive representation of biodiversity, as it does not weight rare species as strongly and is more sensitive to the introduction of new species compared to other indices. Evenness was computed to assess how equally species are distributed across samples. For the calculation of the indices, all species were pooled to family level, since identification to species level was rarely possible from obtained video data, and diversity measures were standardized by effort for each sampling method. RUV data was standardized to a soaking time of 45 min, while data of the 3m-beam trawl was standardized to a towing duration of 15 min at a speed of 3 kn. Species present in fewer than two samples across the entire dataset were excluded from the analyses (n=7), since they were not considered to be caught representatively by either of the methods**.**

Indices were statistically tested for differences between the two fishing methods and the two areas using generalized linear models (GLM) either assuming a gamma or tweedie distribution for continuous count data and to account for zero-inflation, with a log-link function. The interaction of method and gear was included in the models to assess whether the efficiency of the methods varies with the area. Since some video stations only comprised one family resulting in zero values for the Shannon Index, we replaced zeros with 0.0001 to ensure the model functioned properly and to retain all observations (Aston et al., 2024). Model assumptions were verified by plotting residuals against fitted values. F-tests were performed to assess the significance of the factors and their interaction. The model was specified as follows:

$Y \sim Gamma(\mu, \theta)$

$\log(\mu) = intercept + \beta_1 \cdot method + \beta_2 \cdot area + \beta_3 \cdot (method \times area)$

Where Y represents the response variable (different indices) at each station, following a gamma or tweedie distribution. The predictor variables include method, area and their interaction (method × area). Model parameters β_1, β_2 represent the effect of the predictor variables ‘method’ and ‘area’ on the response variable, while β_3 represents the effect of the interaction of ‘method’ and ‘area’ on the response variable.

To assess total family richness acquired with each fishing method as a function of sampling effort (number of samples), we generated rarefaction/extrapolation curves for both, the number of samples as well as the number of individuals, for each method following Chao and Jost (2012). This method allows to examine the amount of effort necessary to acquire the same level of biodiversity, i.e., number of species or, in our case, number of families.

To evaluate whether the different methods are comparably efficient in detecting common families, the standardized abundances of the four most common families, (1) *Pleuronectidae*, (2) *Gobiidae*, (3) *Stichaeidae* and (4) *Psychrolutidae*, were compared between fishing methods using the same models described above. Since *Stichaeidae* and *Psychrolutidae* only occurred at either of the areas or were caught with one of the methods, the interaction term of method and area was not included in these models. An interaction was not included either in the model for *Gobiidae* since sample size for RUVs in Fehmarnbelt was too small. Additionally, we calculated the frequency of occurrence within samples of each family by method and area to evaluate the effectiveness of RUVs in detecting specific families.

All statistical analyses were performed in the open-source software R, version 4.3.0 (R Core Team, 2023) using the packages car (Fox and Weisberg, 2019), DHARMa (Hartig, 2024), effects (Fox, 2003), iNEXT (Hsieh et al., 2024), lsmeans (Lenth, 2016), statmod (Dunn and Smyth, 2018), tidyverse (Wickham et al., 2019) and vegan (Oksanen et al., 2024).

Results

Comparison of diversity indices

In our study, we explored the difference of the performance of beam trawls and RUVs in sampling fish communities in Baltic Sea MPAs by comparing different diversity measures, as well as the sampling effort required to detect family richness between the two gear types. Significant differences in the ability to describe the fish community were detected between RUVs and beam trawls with regard to the diversity measures computed. RUV and beam trawls captured a total of 1749 individuals of 23 species belonging to 13 families across both areas (Fehmarn and Odra bank). RUV stations observed 308 individuals of seven species in four families and beam trawls captured 1441 individuals of 19 species in 13 families, in total. The average number of individuals was about 15 times higher in samples collected with beam trawls and the average family richness was about 3 times higher compared to RUV samples considering both sampling areas (mean abundance ± SD: Beam trawl = 42.9 ± 32; RUV = 3.4 ± 0.6; mean family richness ± SD: Beam trawl= 4.3 ± 1.5; RUV = 1.5 ± 0.3).

GLMs showed that family richness, total abundance and the Shannon Index were significantly higher in samples taken with beam trawls compared to RUV samples in both areas (Figure 2, Table 1). Thus, fish samples derived with beam trawls were characterized by a higher abundance and diversity compared to samples obtained with RUVs at the same location (mean abundance: Beam trawl (Fehmarnbelt) = 74 ± 20.9; Beam trawl (Odra bank) = 11.86 ± 3.3; RUV (Fehmarnbelt) = 2.6 ± 0.5; RUV (Odra bank) = 3.9 ± 0.5; mean family richness Beam trawl (Fehmarnbelt) = 5.7 ± 1.4; Beam trawl (Odra bank) = 2.9 ± 0.7; RUV (Fehmarnbelt) = 1.8 ± 0.3; RUV (Odra bank) = 1.3 ± 0.2). Shannon indices obtained with the two methods were smaller than one, indicating a general low diversity of fish communities (Figure 2) (mean Shannon Index: Beam trawl (Fehmarnbelt) = 0.86 ± 0.8; Beam trawl (Odra bank) = 0.71 ± 0.7; RUV (Fehmarnbelt) = 0.32 ± 0.2; RUV (Odra bank) = 0.14 ± 0.07). Evenness was significantly higher in both areas in RUV samples, indicating a more uniform distribution of families derived by the video-based approach (Figure 2d) (mean Shannon Index: Beam trawl (Fehmarnbelt) = 0.47 ± 0.06; Beam trawl (Odra bank) = 0.64 ± 0.09; RUV (Fehmarnbelt) = 0.92 ± 0.11; RUV (Odra bank) = 0.83 ± 0.11). Contrary, low evenness for beam trawl samples points towards a dominance of certain families in the samples. The impact of method on fish abundance was highly significant ($F=595.92$, $p <0.001$). Similarly, method had a significant effect on Family richness and Shannon index as well as evenness. While GLMs revealed a significant effect of method on all diversity measures, as shown above, the effect of area varied for the different taxonomic metrics (Table 1). A significant effect of area was observed for Family richness, Shannon index, with a weaker but still significant effect on evenness, while there was no significant effect of area on fish abundance.

The significance of the interaction of method and area suggests that the efficiency of the two methods differed between the two areas in representing total fish abundance, richness and evenness, although, beam trawls depict significantly higher abundances compared to RUVS in both areas (i.e. significant interaction term: method*area; Table 1).

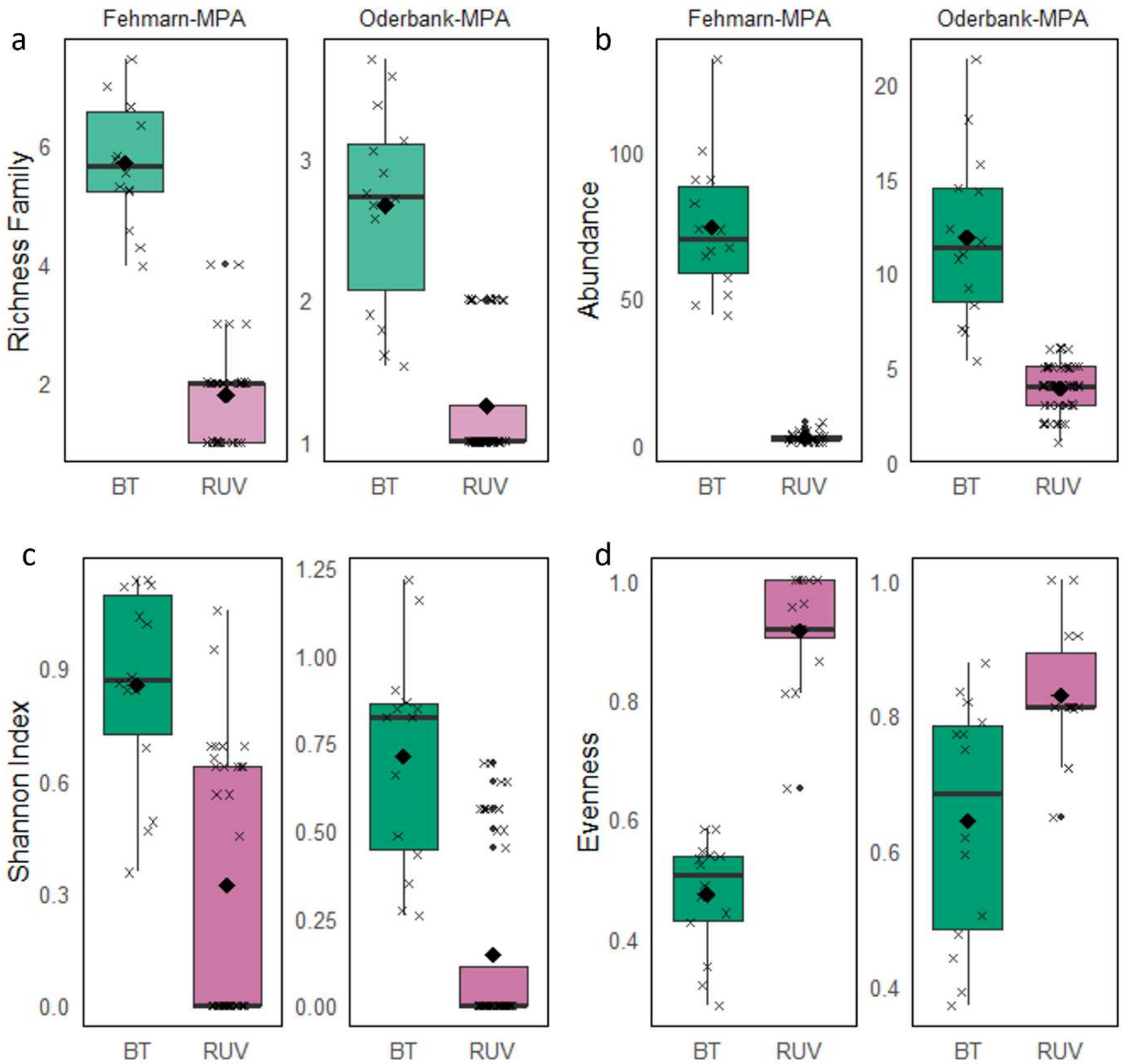

Figure 2: Comparison of indices (a: Family richness, b: Abundance, c: Shannon Index, d: Evenness) between the two fishing methods (BT: 3m beam trawl; RUV: remote underwater video) in the two study areas (Fehmarn and Odra bank). Individual values of each sampling station are represented by "x", mean values by diamonds, and outliers by dots. The median is indicated by the line within each box, and the box boundaries represent the interquartile range. Whiskers extend to the minimum and maximum values within 1.5 times the interquartile range from the box limits, representing the range without outliers.

Table 1: Generalized Linear Model results regarding the comparison of indices and abundances of the four most frequent fish families between fishing methods and study areas. Bold values indicate significant differences between fishing methods at the 0.05 level.F = value of the F-statistic; df =degrees of freedom, p = p-value showing statistical significance.

Variable	$F_{(df=1)}$	p-value
Abundance		
Method	595.92	**<0.001**
Area	1.24	0.27
Method:Area	128.77	**<0.001**
Richness (Family level)		
Method	174.14	**<0.001**
Area	41.66	**<0.001**
Method:Area	4.33	**0.039**
Shannon Index		
Method	24.93	**<0.001**
Area	7.59	**0.006**
Method:Area	0.01	0.942
Evenness		
Method	85.67	**<0.001**
Area	3.90	0.05
Method:Area	16.88	**<0.001**
Abundance: Pleuronectidae		
Method	696.96	**<0.001**
Area	93.20	**<0.001**
Method:Area	29.16	**<0.001**
Abundance: Gobiidae		
Method	6.50	**0.013**
Abundance: Stichaeidae		
Method	16.22	**<0.001**
Abundance: Psychrolutidae		
Method	5.29	**0.03**

Family richness as a function of sampling effort

To compare the sampling efficiency of the two methods, family richness obtained for each gear was analysed in relation to sampling effort. Size based rarefaction/extrapolation curves show that both methods provide different estimates of fish diversity (Figure 3a, b). Beam trawls recorded a family richness more than twice as large as the one recorded with RUVs and required about half as many samples to reach the maximum number of families compared to RUVs (Figure 3a). Increasing the number of samples beyond 40 or observed individuals beyond 100 does not result in higher recorded family richness in RUVs. In contrast, richness detected with beam trawls reached saturation after about 10 samples or 500 individuals. RUVs performed similar in both areas in representing family richness, saturating after about 15 samples or after about 100 individuals observed, while the number of families detected with beam trawls in the Fehmarnbelt saturated after approximately six samples or 500 individuals, and only after about 30 samples or around 400 individuals in the Odra bank (Figure 3b).

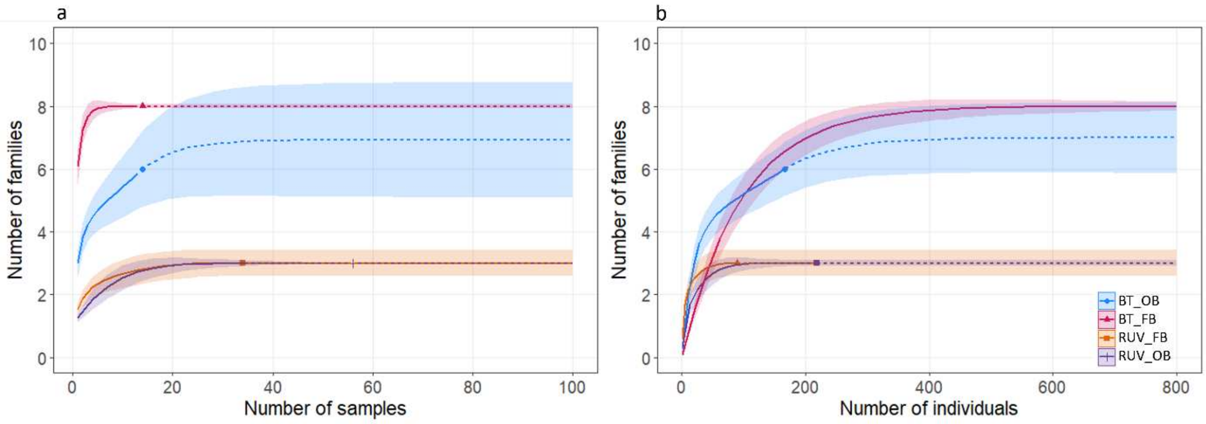

Figure 3: Size-based rarefaction/extrapolation curves (a) based on the number of samples and (b) based on the number of individuals. Blue lines with dots represent the 3m-beam trawl in the Odra bank (BT_OB), red lines with triangles refer to the 3m-beam trawl in the Fehmarnbelt (BT_FB). Orange lines with squares present RUV in the Fehmarnbelt (RUV_FB) and violet lines with crosses refer to RUV in the Odra bank (RUV_OB). Solid lines are interpolated, dashed lines are extrapolated. Shaded areas represent 95% confidence intervals.

Comparison of community composition between methods

To evaluate how the two methods differed in displaying fish community composition, abundances of the most frequent families were compared between Beam trawls and RUV samples.

The most frequently recorded species in the two methods belong to the families *Gadidae*, *Gobiidae*, *Pleuronectidae*, *Psychrolutidae* and *Stichaeidae*. *Pleuronectidae* represented the most abundant family in the Fehmarnbelt, and the second most abundant in the Odra bank (see supplements Table S.1 & Figure 4), and were detected in every single beam trawl and RUV sample in the Fehmarnbelt. The second most prevalent family in the Fehmarnbelt area was *Stichaeidae*, represented exclusively by the snake blenny (*Lumpenus lampretaeformis*). This species was detected in approximately 90% of beam trawl samples and in 45% of RUV samples. Individuals of *Gadidae,* predominantly juvenile cod (*Gadus morhua*), were observed in all beam trawl transects in the Fehmarnbelt, whereas only a single individual was found at the Odra bank and none were detected with RUVs in either of the two areas. Beam trawls in the Fehmarnbelt detected some individuals of *Psychrolutidae, Soleidae and Zoarcidae*, while RUVs failed to record these families in this area. *Gobiidae* were by far the most abundant family in the Odra bank and were detected in all RUV samples, while they were absent in about

15% of the beam trawl samples. Notably, *Pleuronectidae* were present in only 13% of RUV samples in the Odra bank, while all beam trawl samples comprised individuals of this family. *Psychrolutidae* were present in about 60% of the beam trawl samples, but only in 13% of the RUV samples. Since RUVs failed to detect individuals of several families, including *Gadidae*, *Labridae*, *Scophthalmidae*, *Soleidaea* and *Zoarcidae*, comparisons of abundance were limited to the four families identified by both methods (Figure 5).

GLMs show that abundances of *Pleuronectidae* differ significantly between the two methods with higher values in beam trawl samples compared to RUV samples and overall abundances being significantly higher in the Fehmarnbelt (Table 1, Figure 5a). Vice versa, abundance of *Gobiidae* was significantly higher in RUV samples collected at the Odra bank compared to beam trawl samples, while the opposite pattern was observed in the Fehmarnbelt, where abundance of *Gobiidae* was lower in RUV samples (Figure 5b). *Stichaeidae* only occurred in the Fehmarnbelt, so that abundance could only be compared between methods in this area. Here, abundances were significantly higher in beam trawl samples than in RUV samples (Figure 5c). RUVs failed to detect individuals of *Psychrolutidae* in the Fehmarnbelt. On the Odra bank, abundances of this family were significantly higher in beam trawl samples compared to RUV samples (Figure 5d).

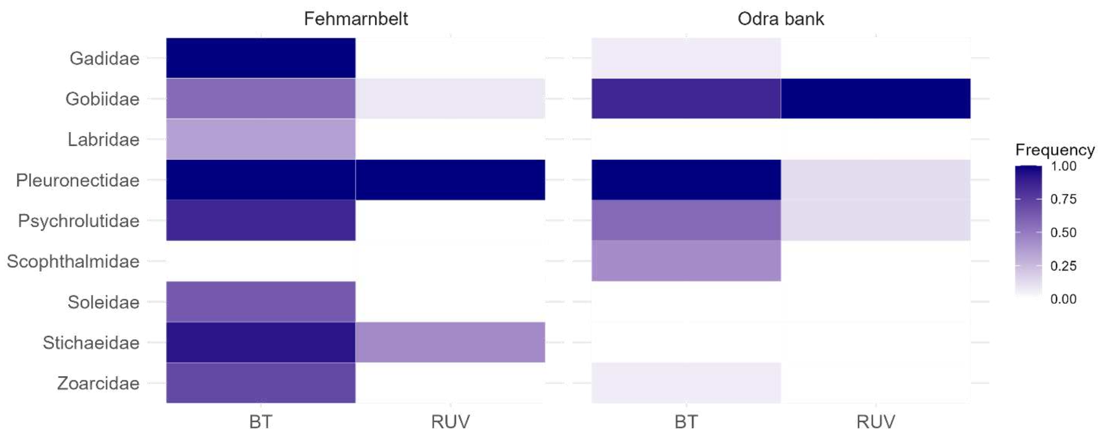

Figure 4: Heatmap of the frequency of occurrence of different fish families in the two areas compared between fishing methods (BT = Beam trawl, RUV = Remotely operated underwater video). Colors represent frequency of occurrence of each family relative to overall abundance in samples.

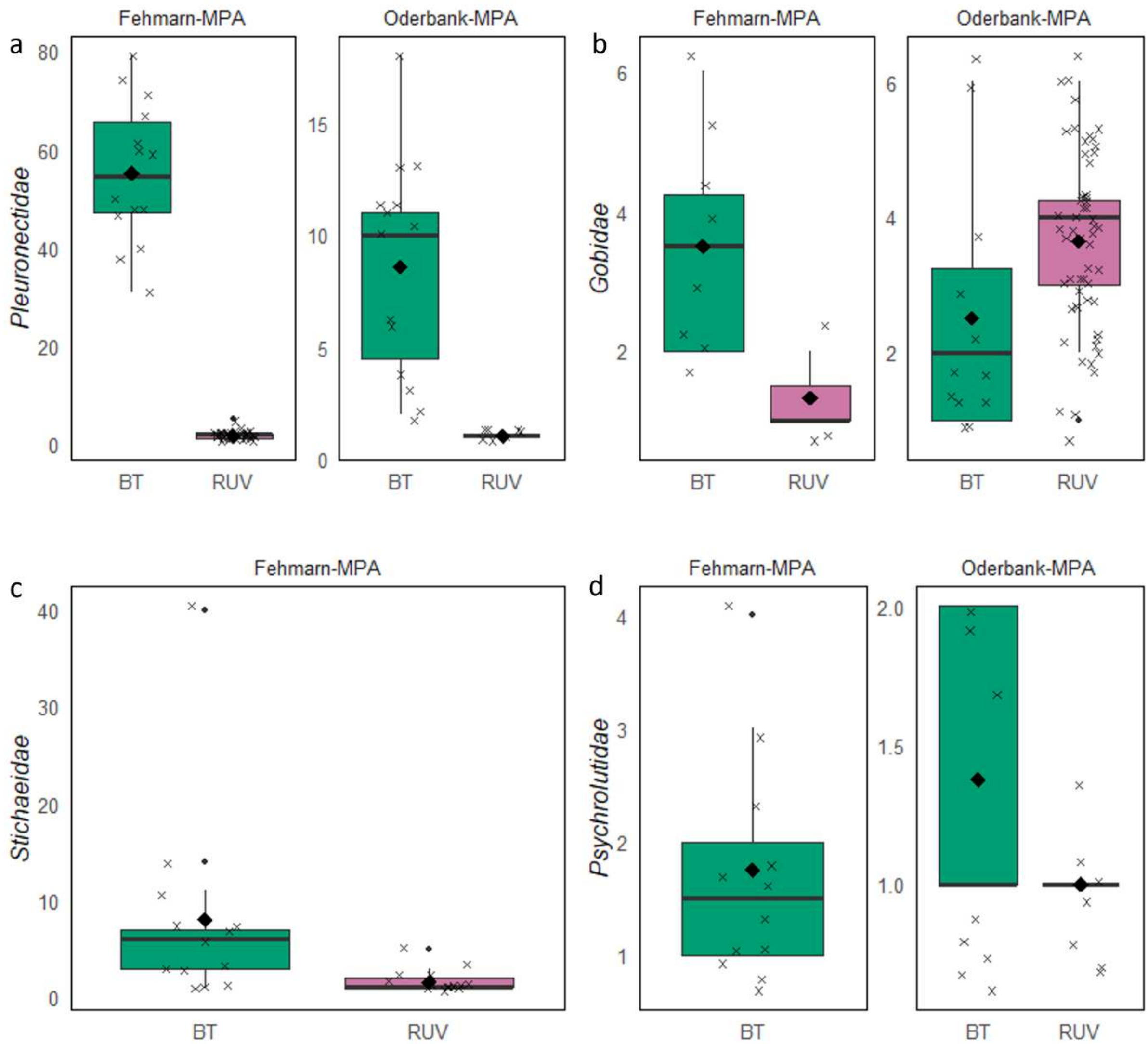

Figure 5: Comparison of abundances of (a) *Pleuronectidae*, (b) *Gobiidae,* (c) *Stichaeidae* and (d) *Psychrolutidae* between fishing methods (BT: 3m beam trawl; RUV: remote underwater video) within the different study areas. Y- axes are scaled differently for the two areas. Details as in Fig.2.

Discussion

The establishment of marine protected areas (MPAs) worldwide requires less invasive monitoring of fish communities within these MPAs. All survey methods are limited in their capacity to accurately represent fish community composition due to gear specific selectivity and detection biases (De Vos et al., 2014). Therefore, the aim of this study was to investigate whether monitoring with remote underwater video (RUV) can provide scientifically sound information for monitoring sandy bottom demersal fish communities, specifically in the western Baltic Sea, by comparing the performance of RUVs to the previously applied 3m beam trawl. Yet, the special characteristics, like low species and functional diversity, similarity in fish species morphotypes, as well as poor visibility present in the Baltic See (Zweifel et al., 2009), pose a challenge to the development of video-based approaches.

Comparability of sampled diversity and abundance

The differing operational modes of beam trawls (active) and RUVs (passive) likely result in a larger effective sampling area for beam trawls, indicating that the two methods are not directly comparable. In this context, data from beam trawl hauls serve as a reference for the expected diversity and abundance. The sampled fish community included typical demersal fish species of the western Baltic Sea, such as cod, gobies, various flatfish species, eelpouts, and snake blennies (Rau et al., 2019). However, the overall low catches and detection rates, alongside low diversity, indicate a lower catchability of the beam trawl compared to other commonly used trawls. Otter trawls cover a larger sampling area, have a higher horizontal and vertical opening and can be towed with a higher speed, making these more suitable to capture species with higher swimming capacity and the ability to move upwards from the bottom, resulting in higher detected biomass and number of species (Farriols et al., 2024).

Our results show that Richness, Abundance and the Shannon Index were significantly lower in RUVs compared to beam trawls, which aligns with findings from comparisons of beach seines and BRUVs in tropical sandy habitats (Shah Esmaeili et al., 2021). In contrast, our results differ from comparisons between shrimp trawls and BRUVs in tropical structured habitats, where richness was found to be higher in BRUV samples during the daytime (Cappo et al., 2004). Generally, structured habitats support higher fish density and diversity, which could be the main driver of these differences, indicating that fish diversity is particularly underestimated by RUVs in sandy habitats. Fish density is rather patchy on sandy bottoms (Mallet et al., 2021) further increasing the effect of the limited detection range of RUVs. Since individuals can only be detected when they move into the field of view, may resulting in a failure to observe stationary species with a restricted radius of movement, even though they are present in the sampling area.

Yet, higher evenness detected with RUVs demonstrates that the distribution of species is more even in RUV samples, while certain species seem to be dominating in beam trawl catches, mostly driven by high abundances of *Pleuronectidae* in beam trawl samples. Since beam trawls are active fishing gears that sweep a specific area and do not depend on the fish to move actively to be caught, they are capable of capturing non-mobile species (Cappo et al., 2004; Morrison and Carbines, 2006), like flatfish or gobies, which can lead to a greater detected diversity. However, this can also result in the overrepresentation of dominant species, like flatfish. Additionally, these different operation modes probably result in a larger sampled area by beam trawls, making direct comparisons of abundances difficult. Overall, we found an underrepresentation of abundances when using RUVs. One possible approach to improve comparability of abundances between the two methods would be to aggregate several RUVs along a transect (Cappo et al., 2004) assuming that a specific number of RUVs deployed along a transect line is representative of one fishing haul. However, this requires that all recorded videos along a transect can be analysed, which was not the case in this study, although the study design had been planned accordingly (see Materials and Methods).

There is some correlation in the frequency of occurrence between the two methods for flatfish (dab and plaice) and snake blennies in the Fehmarnbelt and for gobies on the Odra bank. As an example, flatfish were the most abundant group in the Fehmarnbelt, a pattern observed with both methods, although they were observed in much lower abundances than in the beam trawl catches. This could indicate that the frequency of occurrence might be a better indicator of relative changes in the abundance of a group.

Yet, some uncertainties remain, such as why flatfish were not frequently recorded by RUVs on the Odra bank, although they were abundant in beam trawl samples. Vice versa, the situation is reversed for gobies *(Pomatoschistus spp.)*, which were detected in greater numbers at the Odra bank using RUV than using the beam trawls, but only very rarely using RUVs in the Fehmarnbelt. Some species of flatfish and gobies, like plaice, dab or sand gobies, that occur in both areas could only frequently be detected with RUVs in either of the areas. This highlights that the performance of RUVs in detecting certain species also varies between areas. Since both flatfish as well as gobies are associated with a benthic lifestyle and rather low mobility, one possible explanation for this pattern is that detection probability of less-mobile species is linked to their abundance. This may explain why flatfish were less frequently detected with RUVs on the Odra bank, where fish abundance is generally lower (Figure 2b). Thus, the idea that frequency of occurrence might be an indicator of abundance, is supported by the observed area dependent differences in detection. Yet, the difference between abundances of gobies between the two areas in RUV samples could also be due to the higher abundance of flatfish and snake blennies that might deter individuals of gobies from entering the field of view. Further, the undersampling of small and cryptic species, especially gobies or blennies in RUVs is often reported (Mallet et al., 2021; Mallet and Pelletier, 2014), but somehow in conflict with our results for gobies on the Odra bank and blennies in the Fehmarnbelt. Another factor that could contribute to differences in the detection frequency of species between areas is the water depth and thus the incidence of natural light. In the deeper Fehmarnbelt, the influence of the artificial light source on the behavior of fish may be greater and fish, especially flatfish, might be attracted towards the light source (Trenkel et al., 2004). Studies evaluating the effect of different lights have found the use of light to increase abundances (Fitzpatrick et al., 2013; Harvey et al., 2012). Furthermore, visibility in the Fehmarnbelt is much lower due to strong currents, potentially decreasing the detectability of fish in video footage (Bacheler et al., 2022) and can result in smaller and cryptic fish such as gobies being overlooked.

Comparison of sampling effort

Overall estimated sampling effort was lower for beam trawl samples compared to RUVs, but varied between the two areas, reaching highest level of richness after approximately six (Fehmarn) or 30 (Odra bank) samples. Whereas RUVs performed similar in the two areas, reaching highest level of richness after 15 samples. In this study, rarefaction/extrapolation curves showed that the estimated effort required by RUVs to capture about half of the number of fish families detected by the beam trawl is about twice as high compared to the beam trawl. This is consistent with results of Cappo et al. (2004) reporting that video methods consistently detect fewer species compared to trawls as more samples are collected. Additionally, rarefaction curves indicate that even if the number of samples or individuals would be increased four times, richness would not increase for either of the methods. This implies that an increase in sampling effort for RUVs will not result in an increase in the observed richness nor approach towards the observed richness detected by the beam trawls used in this study. A similar relationship was found for the sampling efficiency of BRUVs and seine nets in tropical sandy habitats, where seine nets were able to detect a three times higher richness than BRUVs with approximately equal sampling effort (Shah Esmaeili et al., 2021). These findings demonstrate that a higher effort would be required when using RUVs to be able to detect only a fraction of the richness captured by active methods. In contrast to RUVs, the required effort for beam trawls varied between the two areas. The higher effort needed at the Odra bank (saturation after 30 samples) is likely driven by the unbalanced composition of the fishing hauls at the Odra bank, with two families (*Gadidae* and *Zoarcidae*) occurring in only a few hauls. Whereas, in the Fehmarnbelt all families were recorded relatively consistently within the beam trawl hauls (see Figure 4). This also highlights the potential for false negatives generated by the beam trawl and emphasizes that each method carries its own bias, making it essential to select the appropriate method based on the research question being addressed as well as habitat characteristics and target species (Hammerl et al., 2024; Henseler and Oesterwind, 2023).

Potential applications and limitations of RUVs

Video methods have some advantages over mobile bottom contacting gears, like their non-extractive nature, that they can be applied in almost every habitat (Priede and Merrett, 1996; Starr et al., 2016) and the ability to collect information about habitat type and complexity and fish behavior (Schobernd et al., 2014; Bacheler et al., 2022). However, major drawbacks of RUVs include their limited ability to identify individuals up to species level (Shah Esmaeili et al., 2021) and the time that is needed to process the videos (Spencer et al., 2005; Harvey et al., 2012; Holmes et al., 2013). The time-consuming analysis of video material is exacerbated by the fact that RUVs require twice the effort to record only half of the fish richness detected by beam trawls. However, modern solutions to process videos with artificial intelligence that automatically count and identify species are likely to tackle the amount of effort needed for video analysis in the future. Currently, various solutions for automated image analysis using deep learning methods are under development (Saleh et al., 2022; Siddiqui et al., 2018).

Especially, distinction between species within the family *Gobiidae (*e.g., sand goby and painted goby) and within the family of *Pleuronectidae* (e.g., plaice and dab) depends largely on detailed morphological features that RUVs cannot detect so far. Extractive methods like trawls have the advantage that identification on the basis of morphological features is feasible and individuals can be identified by experts in the laboratory in case of uncertainty, especially when it comes to small and cryptic species (Cappo et al., 2004; Unsworth et al., 2014; Shah Esmaeili et al., 2021). Further, beam trawls can provide biological information, such as age and condition (otoliths, reproduction), population structure and genetic diversity (DNA) and allow estimating biomass, which represents a significant component needed to conduct traditional stock assessment (Bacheler et al., 2022; Hammerl et al., 2024). Extractive methods can also support feeding ecology studies of fish, which require the analysis of stomach contents and tissue samples (e.g. muscle tissue, for stable isotope analysis). As changes in the structure and function of the benthic food web are expected to occur in areas, where for example mobile bottom trawling is prohibited, this information might be essential to be able to evaluate the effectiveness of these management measures in MPAs (De La Vega et al., 2023).

In this study, we were able to identify clear limitations of RUVs and show that they can only provide a limited snapshot of species diversity in the Baltic Sea. However, our results also showed quite consistent detections for the three most abundant families, implying that RUVs might be a suitable method for specific research questions or particular for some species/families. All survey methods are limited in their ability to capture biodiversity (De Vos et al., 2014; Henseler and Oesterwind, 2023; Watson et al., 2005), but fish biodiversity detected using RUVs was so little, that it is unlikely to be a viable method for biodiversity assessments in Baltic Sea MPAs. To incorporate RUVs into the monitoring of specific groups, further research is still needed to detect changes and interpret the data correctly. In such cases, a calibration phase, pairing video and traditional sampling gears, like the beam trawl used here, probably also among different seasons, would be beneficial to get a better understanding of RUV data and be able to make right indications from comparisons between RUVs and traditional sampling gears. Performance of RUVs might also depend on the habitat type sampled, as well as prevailing abundances and diversity, as our results imply. Soft bottom habitats are generally difficult to sample due to the scatteredness of individuals over large areas (Mallet et al., 2021). Therefore, RUV performance might be higher in more structured habitats or seagrass meadows with a shallower water depth, that are often characterized by higher densities and species diversity. Even in ecosystems with very low diversity, like the Baltic Sea, RUV has been used to monitor Baltic Sea cobble and artificial reefs (Rhodes et al., 2020; Wilms et al., 2021).

Video methods, especially baited RUVs are commonly highlighted as a complementary method to traditionally used invasive methods, because they are able to detect species that cannot be recorded using other methods, such as highly mobile predatory species (Cappo et al., 2004; Shah Esmaeili et al., 2021; Bacheler et al., 2022). In this study, RUVs were unable to detect any additional species that were not caught with the beam trawl. One approach to increase diversity detection and extend sample coverage might be the use of bait to attract more species and more individuals to the field of view from a wider area (Cappo et al., 2003; Harvey et al., 2007; Shah Esmaeili et al., 2021; Stoner et al., 2008). Yet, there can be many biases introduced by the use of bait, like alterations in species interaction (Birt et al., 2021;

Harvey et al., 2007; McLean et al., 2016; Shah Esmaeili et al., 2021) or a bias towards the detection of predatory species, resulting in avoidance behavior of other species (Cappo et al., 2004; Harvey et al., 2007; Willis et al., 2003; Willis and Babcock, 2000). For this reason, future studies should compare the use of baited and unbaited videos with regard to examining the diversity of fish communities in the Baltic Sea or other ecosystems that are dominated by similar communities. The combination of video and other traditional and less-invasive methods, like traps could be another opportunity to increase the detectability of important species (Bacheler et al., 2022). Another approach might be the combined use of RUVs and the genetic analysis of water samples (environmental DNA; eDNA), which could also improve species identification. This approach has already been proven to provide a wider picture of fish communities in Western Australia (Stat et al., 2019; Bacheler et al., 2022). Initial results of a study investigating the applicability of eDNA as a method to assess biodiversity in our study areas yielded promising results, where a total of 24 species were detected, five of which found exclusively with eDNA analyses (Piontek et al, in prep.). Yet, there are still many uncertainties regarding eDNA like the transport and concentration of particles that contain DNA, false positives and negatives, as well as the estimation of abundances, which requires more research on the influence of environmental factors on metabolic rates and release of particles (Strickler et al., 2015; Hansen et al., 2018; Hammerl et al., 2024).

The results of this study demonstrate that while RUVs can provide a non-invasive alternative to conventional methods in many bioregions, their efficacy in monitoring benthic and demersal fish communities in Baltic Sea sandy habitats may be constrained by their limitation in species detection and the effort required. For future applications, complementary methods or technological improvements (e.g., improved camera technology, use of additional attractors) may need to be considered to improve the performance of RUVs to be applicable in monitoring low diversity ecosystems dominated by sandy habitats.

Acknowledgements

We thank the captain and crew of the FRV “Solea” for support in data collection. We also thank Nakula Plantener for providing help with figures (Figure 1), as well as Wolf Christian Lewin for discussions on statistical analysis. This study was carried out as part of the DAM pilot missions “MGF-Ostsee” and “MGF-Ostsee II” funded by the German Federal Ministry of Education and Research (funding codes: 03F0848E/03F0937E).

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Supplementary Material

Table S 1: Number of individuals (N_{fish}) and frequency of occurrence ($N_{transects}$) displaying fish families for RUVs and beam trawls pooled across fishing methods for the Fehmarnbelt and the Odra bank.

Species	Family	Fehmarn N_{fish} ($n_{transects}$ = 48)	Odra bank N_{fish} ($n_{transects}$ = 70)
Ammodytes spp.	*Ammodytidae*	6 (1)	-
Arnoglossus laterna	*Bothidae*	1 (1)	-
Gadus morhua; Merlangius merlangus	*Gadidae*	47 (14)	1 (1)
Aphia minuta, Neogobius melanostomus, Pomatoschistus minutus, Pomatoschistus spp.	*Gobiidae*	34 (13)	235 (68)
Ctenolabrus rupestris	*Labridae*	6 (5)	-
Pholis gunnellus	*Pholidae*	2 (2)	-
Hippoglossoides platessoides; Limanda limanda; Platichthys flesus; Pleuronectes platessa	*Pleuronectidae*	836 (48)	126 (21)
Myoxocephalus scorpius	*Psychrolutidae*	19 (12)	17 (15)
Scophthalmus maximus	*Scopthalmidae*	-	6 (6)
Solea solea	*Soleidae*	33 (9)	-
Lumpenus lampretaeformis	*Stichaeidae*	128 (28)	-
Syngnathus typhle	*Syngnathidae*	1 (1)	-
Zoarces viviparus	*Zoarcidae*	28 (10)	1 (1)

STUDY III

Enhancing gear performance for monitoring temperate fish communities: A comparative analysis of baited vs. unbaited underwater video.

Manuscript in preparation

Title: Enhancing gear performance for monitoring temperate fish communities: A comparative analysis of baited vs. unbaited underwater video

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Abstract

To comply with management measures such as the prohibition of extractive fishing, which is progressively implemented in some MPAs, non-invasive and non-extractive methods are required for monitoring demersal fish communities. Baited remote underwater video (BRUV) have increasingly been used to survey fish assemblages on a global scale. Studies investigating biases and standardization of BRUVs are still lacking in areas with low biodiversity. Yet, methods used in long-term monitoring programs should be standardized in advance, especially when planned to be applied in a different bioregion or habitat to account for biases and limitations. We therefore, investigated bait induced biases and determined the optimal deployment time and sampling effort by comparing remotely operated underwater video (RUV) cameras with and without bait in the Baltic Sea, focusing on their effectiveness in assessing species diversity in soft-bottom fish communities with low species richness. We found a clear effect of bait, with higher family richness, abundance and Shannon index, while evenness remained largely similar between methods. Further, our results show that BRUVs yielded a higher number of species with a lower number of samples and a shorter deployment period compared to RUVs. Our results further illustrate, the limitation of BRUVs in biodiversity studies in this region, and that further technological improvement or the integration of complementary methods is required for a more complete documentation of the demersal fish community. Yet, our findings show that the use of bait significantly enhances the efficiency of detecting family richness and abundance and could provide a sufficient method for specific groups or to address specific research questions.

Keywords: Marine protected areas, Baltic Sea, RUV, BRUV, fish communities

Introduction

Marine protected areas are an emerging tool for the conservation of marine ecosystems that are increasingly threatened by human impacts (Bayley et al., 2019; Hoyt, 2014). To prove the effectiveness of implemented management measures, such as the prohibition of commercial fishing, as well as to assess whether biodiversity conservation objectives have been achieved, effective monitoring and evaluation of MPAs is crucial (De Vos et al., 2014; Dunham et al., 2020; Willis et al., 2003). Conservation objectives of MPAs often include the recovery of populations of vulnerable and exploited species, like fish species or fish populations (De Vos et al., 2014; Dorman et al., 2012; Willis et al., 2003). Monitoring approaches targeting bottom associated fish communities are typically based on mobile bottom trawling (McIntyre et al., 2013; Wells et al., 2008). Yet, the use of extractive sampling techniques such as bottom trawling is often not in line with the conservation objectives of MPAs and therefore problematic (Lipej et al., 2003; Watson et al., 2005; Willis et al., 2000; Willis and Anderson, 2003). This highlights the need for non-invasive monitoring methods. In recent years the usage of non-destructive and non-extractive methods, such as underwater video, have gained more importance (Collie, 2000; Murphy and Jenkins, 2010; Nalmpanti et al., 2023; Sciberras et al., 2018). Especially, stationary systems like remote underwater video (RUV) and baited remote underwater video (BRUV) have increasingly been used as alternatives to conventional methods to assess fish assemblages (Langlois et al., 2020; McLean et al., 2016; Skinner et al., 2020; Williams et al., 2019). These methods have shown to yield representative samples of temperate and tropical demersal fish assemblages, can be used to sample across a variety of habitats and depths and are by their design non-destructive (Cappo et al., 2004, 2003; Harvey et al., 2007; Langlois et al., 2020; Murphy and Jenkins, 2010; Watson et al., 2005). Additionally, video methods are considered less species-selective (Bacheler et al., 2013; Christiansen et al., 2020; Harvey et al., 2012) and can have a higher probability of detecting rare species (Goetze et al., 2019; Langlois et al., 2020).

Similar to passive fishing gears like e.g., traps stationary video systems depend on the activity of the fish, resulting in rather long observation duration and high number of zero counts (Mallet and Pelletier, 2014). These problems might be overcome by the use of bait (Dorman et al., 2012; Stobart et al., 2007). However, the use of bait as an attractant can introduce inherent biases by altering fish behavior, potentially leading to the underrepresentation of certain species (Bernard and Götz, 2012; Watson et al., 2005) and thereby may reduce the ability to detect changes in the fish community (Dorman et al., 2012). Research comparing the effectiveness of various monitoring methods commonly conclude that no single technique is able to measure changes in fish assemblages without introducing its own biases (De Vos et al., 2014; Watson et al., 2010, 2005; Willis and Babcock, 2000). Therefore, it is crucial to understand how a certain methodology influences the result and consider which method may yield the most accurate result for the ecological questions posed, when designing a sampling program (Dorman et al., 2012; Watson et al., 2010). Consequently, biases and limitations of new methods have to be examined beforehand and methodology needs to be standardized and adapted. This becomes particularly important for comparisons across studies and regions and when the method should be implemented in long-term-monitoring programs (Birt et al., 2021). BRUVs have been put forward as a novel standardized and non-extractive approach to estimate the relative abundance and diversity of demersal fishes (Dorman et al., 2012). Yet, the biases introduced by the use of bait and the resulting odour plume are still largely unknown (Mallet and Pelletier, 2014). Studies examining differences in species diversity and composition of warm-temperate and temperate reef fish between BRUV and RUV samples have shown that BRUVs detected greater species richness and a higher number of individuals. Additionally, BRUVs were more effective at recording rarer and larger predatory fish but were less successful in detecting small and cryptic species (Bernard and Götz, 2012; Harvey et al., 2007; Watson et al., 2005). In temperate bioregions with low species and functional diversity, like the Baltic Sea, studies focusing on the effects introduced by bait are still lacking.

To address this gap, our study compared baited and unbaited RUVs to assess (i) the effect of bait on the observed diversity of fish assemblages and (ii) differences in sampling efficiency, aiming to identify the optimal deployment duration and number of samples required.

Our results show that the use of bait significantly enhances the efficiency of detecting family richness and abundance and could provide a sufficient method for specific groups or to address specific research questions in areas characterized by sandy bottoms, with low species- and trophic diversity. Yet, a major limitation that persists is the inability to identify individuals up to species level due to the specific conditions in the Baltic Sea. To effectively apply BRUVs in biodiversity studies focusing on temperate, soft-bottom, low diversity fish assemblages, further technological improvement or the integration of complementary methods is required.

Material and Methods

Study area and Sampling design

The Baltic Sea is a large brackish, semi-enclosed sea with salinity decreasing from the west to the east (Andersen et al., 2015). It is further, characterized by relatively shallow depths and a generally low species and functional diversity (Ojaveer et al., 2010).

Sampling was carried out on four days during the last two weeks of August 2025 close to Warnemünde in the Mecklenburg Bay, situated in the Western Baltic Sea (Figure 1). The sampling site was characterized by sandy sediments and depths between 13 and15 m.

RUVs and BRUVs setup, hereafter referred to as treatments, consisted of a stainless-steel frame with dimensions of 110 cm width and 60 cm height, on the bottom of the frame we attached a 1.5 m long stainless-steel bar with a bait canister (see supplementary material: Figure S1). Two cameras (GoProHero8) were mounted on the frame at a distance of about 25 cm to the seafloor, a 5° convergence angle and a 50 cm distance from one another. Additionally, two lights (bigblue VL4600PB) were installed diagonally above each of the cameras at the upper edges of the frame. These lights provide a beam angle of 120° and a luminous flux of 2100 lumens. Individual frames were connected to a floating line that was attached to a buoy to facilitate retrieval and to mark the position of the camera frame on the sea surface. RUVS were deployed with the bait arm and canister to account for any distortions that might be introduced by the presence of the arm and the bait canister itself. Baited RUVS were equipped with about 200g of Clupeid Mix (Hering and Sprat), which were roughly chopped into 3 – 5 cm pieces and crushed. A mix of herring and sprat was selected, as these species are prevalent in the Baltic Sea and, similar to pilchards, are oily fish known for their effectiveness as bait in attracting target species (Whitmarsh et al., 2017).

Sampling was organized in a grid scheme, where individual sampling stations were placed 300 m apart from one another, allowing the sampling stations to be treated independently (Cappo et al., 2004; Ellis and DeMartini, 1995) (Figure 1b). RUVs and BRUVs were soaked for about 60 minutes to assure analysis of 45 minutes of video material and still allowing time for sediment to settle after the frame hit the seafloor. At each station, RUVs were deployed and retrieved before BRUVs were deployed at the same location, with a break of 20 to 30 minutes between deployments. Additionally, temperature and salinity data were collected within the sampling area each day, both before the first deployment and after the completion of the last station.

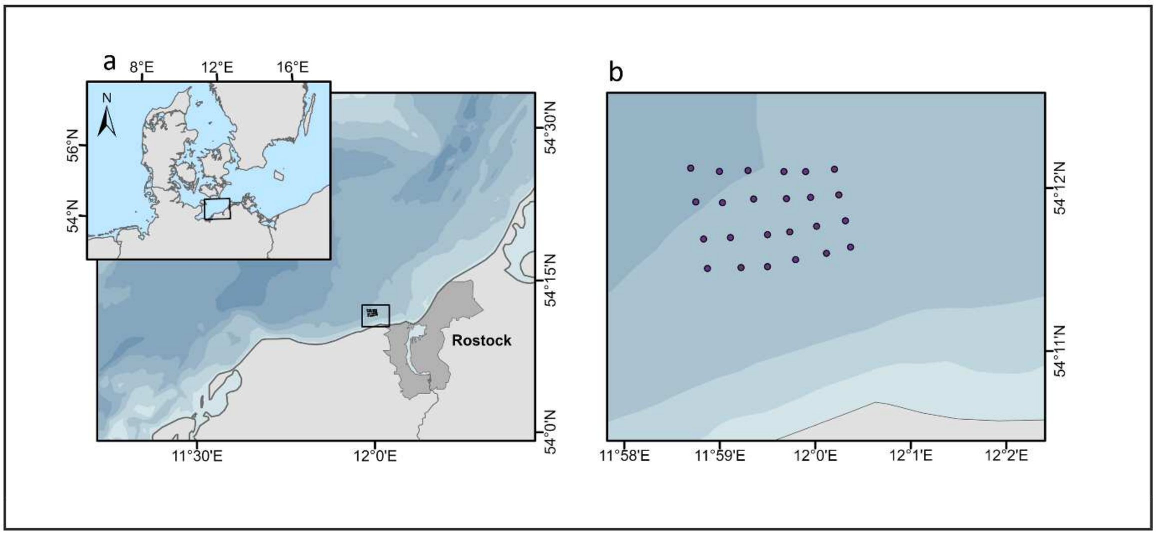

Figure 1: **a**: Location of the study area in the Western Baltic Sea. **b**: Detailed layout of study area and sampling design. Violet dots represent sampling stations.

Data analysis

A total of 48 stations was completed over four days. However, due to poor visibility and tilting of camera frames at some stations, resulting in upwards facing cameras, only 35 stations could be considered in the analysis, including 19 BRUV stations and 16 RUV stations. All videos were analyzed for a duration of 45 minutes after allowing sediment to settle for three minutes. Analysis was conducted using the software Eventmeasure from SeaGIS (www.seagis.com.au). All fish were identified to the lowest possible taxonomic level. As a measure of abundance, we used the commonly applied metric (MaxN), a conservative measure that describes the maximum number of individuals of a species in any one frame within a video (Cappo et al., 2003). The MaxN metric avoids pseudo-replication caused by individuals swimming in and out of the camera's field of view (De Vos et al., 2014; Willis et al., 2003).

To assess differences in biodiversity and species composition between the two treatments, we computed different biodiversity measures. Biodiversity measures included abundance as well as the following indices: family richness, Shannon index and Pielou's evenness (calculated with the vegan package in R; (Oksanen et al., 2024)). Family richness was used in the analysis, as reliable identification to species level was not possible for the majority of observed individuals. We chose the Shannon index since it provides a more comprehensive representation of biodiversity compared to e.g., the Simpson index, as it does not weight rare species as strongly and is more sensitive to the introduction of new species. Evenness was computed to assess how equally species are distributed within samples.

To analyse the effect of bait on the different indices, we applied generalized linear models (GLM). Model fit was evaluated for Poisson and Gamma distributions, since both distributions are appropriate to model count and continuous data. Model fit and distribution were verified by plotting residuals against fitted values. The best model fit was found for models assuming a gamma distribution with a log link function for all models, except for Shannon diversity, where a poisson distribution with a log-link function was the most favorable model. F-tests were performed to assess the significance of the factors. The model was specified as follows:

$Y_i = \exp(\beta_0 + \beta_1 \cdot \text{method}_i) + \epsilon_i$

Where Y_i represents the response variable (family richness, Shannon Index, evenness) at each station, method_i is the predictor variable, the parameter β_0 is the intercept, representing family richness for the baited method, β_1 is the coefficient associated with the effect of method and ϵ_i the error term.

To evaluate whether RUVs and BRUVs differ in their ability to detect the abundance of individual fish families, we compared the family-level abundances between treatments using the same models described above. Differences in species composition were visualized by calculating the relative proportion of each fish family at each station.

The relationship between soak time and the cumulated number of families observed was modelled using species accumulation curves. Thereby, the accumulation of newly observed families was quantified at five-minute intervals until the 45 min of soak time elapsed. The accumulation curve was generated by estimating the mean for each interval across stations, applying a resampling approach with 1000 bootstrap iterations. A non-linear regression model with a logistic growth function was then fitted to the data. We tested the fit of exponential models and logistic growth models, including the Michaelis-Menten and von Bertalanffy model, with the von Bertalanffy model providing the best fit. Model fit was evaluated using Akaike Information Criterion AIC (Akaike, 1974) and by visually assessing how the model fitted the actual data. The model is described by the following equation:

$y = L\infty * (1 - e - k(t - t0))$

Where y is the predicted cumulated family richness at a given time interval, $L\infty$ is the predicted value for the maximum number of families, k is the growth coefficient, t the time interval and t0 the theoretical starting time. To examine whether deployment duration influences the occurrence of specific groups, we calculated mean counts per time interval for *Pleuronectidae* (flatfish) and *Gobiidae* (gobies) as these were the most frequently detected families. To determine family richness by each method as a function of sampling effort (number of sampled stations) we generated species accumulation curves based on rarefaction using the specaccum and fitspeccacum functions from the vegan package in R (Oksanen et al., 2024).

All statistical analyses were performed in the open-source software R, version 4.3.0 (R Core Team, 2023) using the packages car (Fox and Weisberg, 2019), DHARMa (Hartig, 2024), tidyverse (Wickham et al., 2019), zoo (Zeileis et al., 2004) and vegan (Oksanen et al., 2024).

Results

Comparison of detected diversity

In our study we explored the difference in the performance of unbaited and baited RUVs by comparing different biodiversity indices, abundance of fish species, fish species composition and observed family richness by functions of sampling effort. A total of 277 individuals belonging to four families were recorded by 19 BRUVs and 16 RUVs. BRUVs observed 218 individuals of four families, whereas RUVs detected 59 individuals of two families.

Bait had a significant positive effect on abundance, family richness and Shannon diversity (Table 1, Figure 2), with the average number of individuals being about two times higher in samples collected with BRUVs (mean abundance ± SD: BRUV = 6.28 ±0.39; RUV =2.71 ± 0.22) and the average family richness being about 1.5 times higher compared to RUVS (mean family richness ± SD: BRUV= 2.96 ± 0.08; RUV = 1.97 ± 0.07), while Shannon index was found to be 1.6 times higher in BRUV samples compared to RUV samples (mean Shannon ± SD: BRUV= 0.95 ± 0.13; RUV = 0.57 ± 0.14). Evenness was found to be slightly lower in samples taken with RUVs, there was however, no statistically significant effect of bait on evenness, indicating that samples obtained by both methods were equally balanced (Table 1).

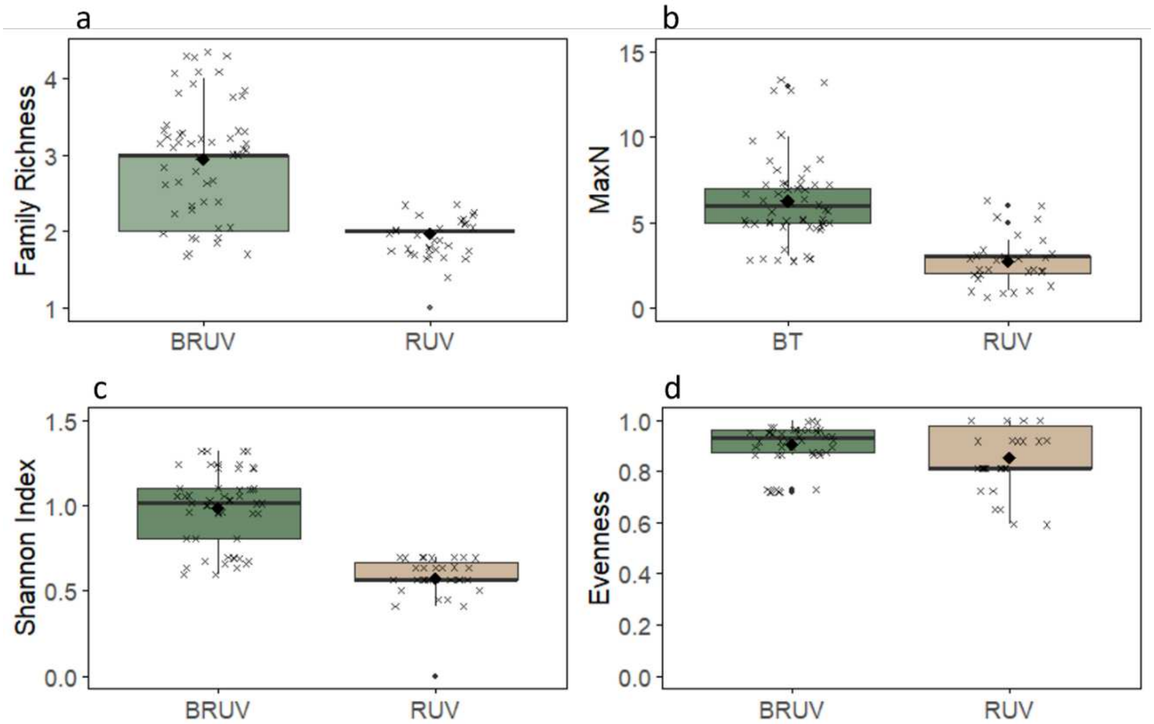

Figure 2: Comparison of indices (a: Richness Family, b: Abundance, c: Shannon Index, d: Evenness) between the two treatments (unbaited /RUV and baited/BRUV). Individual values of each sampling station are marked with "x", mean values with diamonds, and outliers with dots. The median is indicated by the line within each box, and the box boundaries represent the interquartile range. Whiskers extend to the minimum and maximum values within 1.5 times the interquartile range from the box limits, representing the range without outliers.

In a next step we compared the capture efficiency of both methods on a family basis. Flatfish (*Pleuronectidae)* represented the highest proportion in BRUV samples, whereas the highest proportion in RUVs is represented by gobies (*Gobiidae)* (Figure 3, see supplementary material: Figure S2). RUVs only observed individuals belonging to the families *Pleuronectidae* and *Gobiidae,* while BRUVs were also able to detect individuals of the families *Gadidae* (gadoids), predominantly juvenile cod (*Gadus morhua*), and *Scophthalmidae* (Figure 3) as well as some individuals that could not be identified due to visibility constraints. GLMs show abundance of Flatfish and gobies to be significantly higher in BRUV samples compared to RUV samples (

Table 1). Although taxonomic resolution in this study was in most cases limited to the family level, the families observed typically comprise species such as plaice (*Pleuronectes platessa*), dab (*Limanda limanda*), sand goby (*Pomatoschistus minutus*), and cod (*Gadus morhua*) (see supplementary material: Table S1 for a full list).

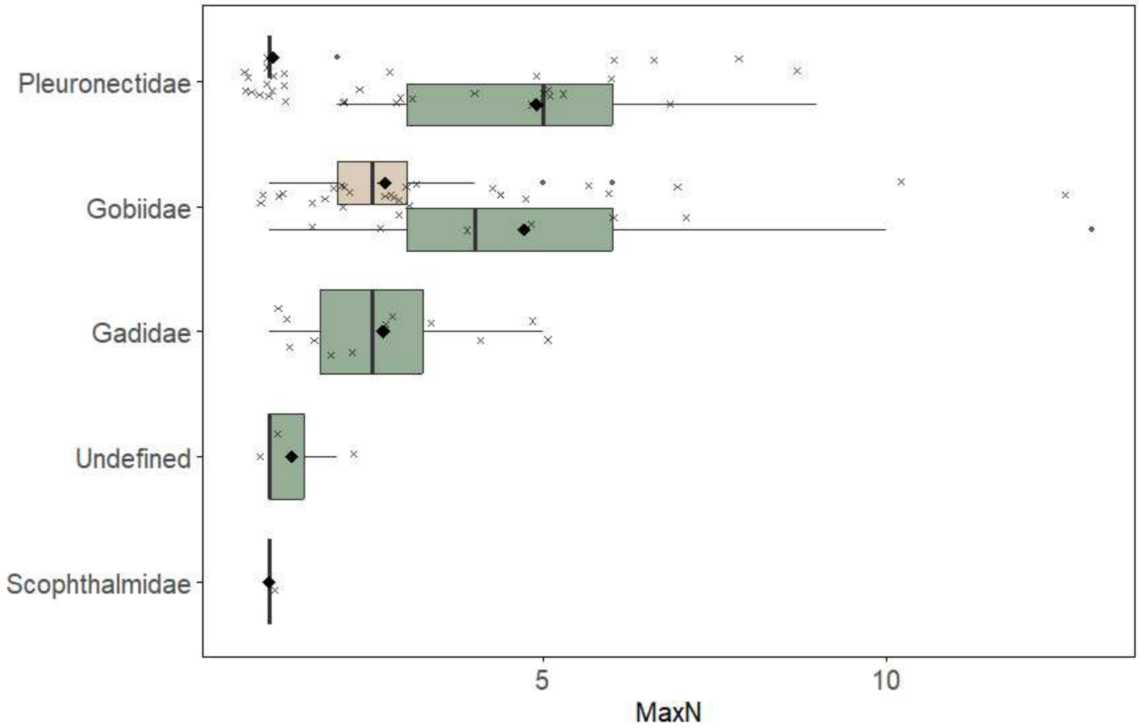

Figure 3: Comparison of abundances (MaxN) of all observed families between the two treatments (BRUV: green; RUV: lightbrown). Individual values of each sampling station are marked with “x”, mean values with diamonds, and outliers with dots. The median is indicated by the line within each box, and the box boundaries represent the interquartile range. Whiskers extend to the minimum and maximum values within 1.5 times the interquartile range from the box limits, representing the range without outliers.

Table 1: Generalized Linear Model results regarding the comparison of indices and abundances between the two treatments. Bold values indicate significant differences between fishing methods at the 0.05 level. F = value of the F-statistic; df =degrees of freedom, p = p-value showing statistical significance.

Variable	F (df=1)	P
Abundance	60.29	<**0.001**
Richness (Family Level)	80.35	**<0.001**
Shannon Index	85.40	**<0.001**
Evenness	4.69	>0.05
Pleuronectidae abundance	175.31	**<0.001**
Gobiidae abundance	7.486	**<0.01**

Optimal Deployment time and number of stations

Following the comparison of biodiversity metrics obtained with both treatments, we compared different means of effort between baited and unbaited RUVs in order to evaluate optimal deployment time and optimal number of samples required.

Comparison of diversity accumulation patterns show clear disparities in the sampling efficiency of BRUVs and RUVs (Figure 4). The higher initial value of the BRUVs curve indicates that bait rapidly attracts individuals from different families, resulting in BRUVs yielding more than double the richness compared to RUVs. BRUVs observed more than half of the family richness after less than five samples, but saturation has not been reached after 19 samples (Figure 4a). Sample size required to estimate maximum species richness using BRUVs is 20 based on the predictions of species accumulation models.

Family accumulation curves as a function of deployment time also reveal rather different efficiency of both methods in the detection of family richness (Figure 4b). BRUVs yielded a higher asymptotic richness, with an estimated maximum of 2.6 family counts after a deployment time of 33 minutes, as predicted by the non- linear regression models with a von Bertalanffy relationship. Based on the model RUVs reach saturation faster compared to BRUVs, after only 17 minutes, with an estimated maximum of 1.9 families. Accordingly, RUVs observed 75% of mean family richness obtained with BRUVs in 17 minutes. However, BRUVs detected 55% of the mean family-level richness within just five minutes and reached 75% after ten minutes of deployment.

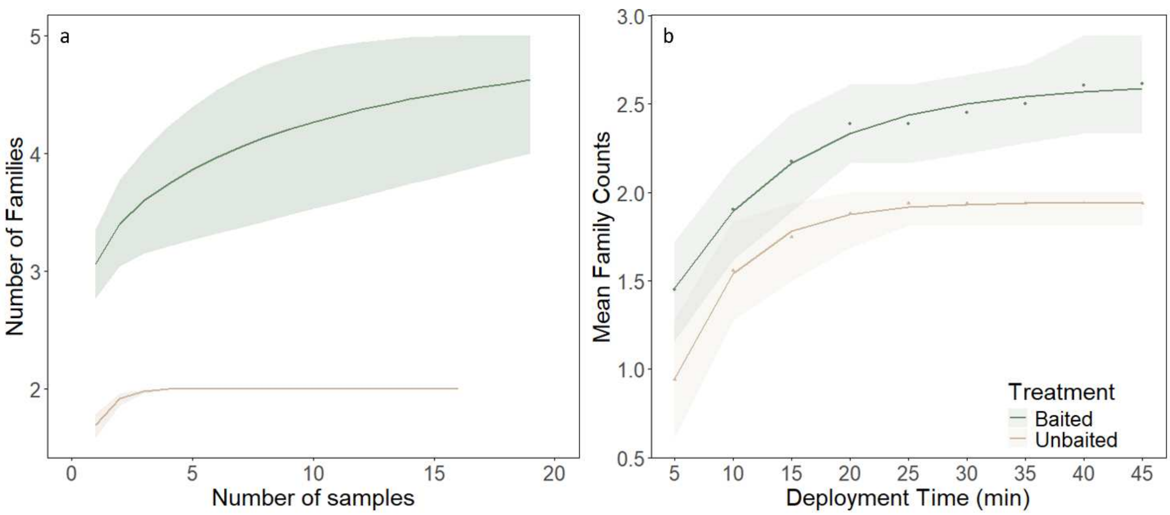

Figure 4: **a:** Size-based rarefaction curves based on the number of samples for BRUVs (green) and RUVS (brown). Shaded areas represent 95% confidence intervals. **b**: Accumulation of mean number of families per station in five min intervals up to 45 min as derived by the resampling approach (dots) and by non-linear regression models based on a Von Bertalanffy relationship (solid lines). Shaded areas represent 95% confidence intervals.

Discussion

A previous study comparing the performance of RUVs and beam trawls to assess fish assemblages in the western Baltic Sea revealed that RUVs efficiency in sampling fish diversity is rather limited by low species detection and relatively high amount of effort required (Hammerl et al., in prep). The use of bait in underwater video studies can increase the number of species and reduce the effort needed, but is also known to introduce bias towards predatory and scavenger species. Objectives of our study were therefore, to compare baited and unbaited RUVs in their ability to assess the diversity of fish assemblages as well as to compare different means of effort.

Effect of bait on detected biodiversity

Our findings show that the use of bait significantly enhances the efficiency of detecting family richness and abundance, suggesting that some limitations of RUVs can be mitigated by the use of bait. This aligns with findings from several other studies reporting increased abundance and richness associated with the use of bait. (Aston et al., 2022; Bernard and Götz, 2012; Dorman et al., 2012; Harvey et al., 2007; Watson et al., 2005). It is likely, that this effect is caused by the attraction of the bait and the possible extension of the sampled area by the spatial distribution of the bait plume (Cappo et al., 2003; Dorman et al., 2012). The attraction to the bait might be species-specific and is often associated with the increase of scavenger and predatory species, especially elasmobranchs (Aston et al., 2024; Dorman et al., 2012; Harvey et al., 2007). Thereby, the occurrence of large predatory species might result in a displacement of lower trophic species (Harvey et al., 2007; Jones et al., 2003; Shah Esmaeili et al., 2021; Willis et al., 2003). This could lead to the underrepresentation of non-aggressive or lower tropic level species in baited samples (Malcolm et al., 2007; Stoner et al., 2008, 2007). We observed a decline in the occurrence of gobies in BRUV samples with increasing deployment duration, possibly due to the increasing occurrence of flatfish during the recording time, indicating a deterrence effect (see supplementary material: Figure S3 and S4). Additionally, gadoids and in very low abundances *Scopthalmidae* have only been detected with BRUVs, both comprise piscivores and benthivores species (Henseler and Oesterwind, 2023), encouraging the clear effect of bait on the detection of these groups.

In comparable studies abundances of carnivorous and piscivorous fish were found to increase while counts of herbivorous species remained rather stable between baited and unbaited RUVs (Harvey et al., 2007). We did not observe any differences in the composition of trophic groups (e.g., piscivorous, benthivorous species) between RUV and BRUV samples, but generally abundances increased in BRUV samples. Yet, only one of the potential species (within detected families) is known to be planktivorous (*Aphia minuta*) (see supplementary material: Table S1). Additionally, abundances of small cryptic fish such as gobies, most likely sand gobies, which are benthivorous (Henseler and Oesterwind, 2023), were significantly higher in BRUV samples, suggesting that bait had a more general attraction effect in our study. However, in our study, BRUVs failed to detect several demersal fish species previously recorded in the study area using 3 m beam trawls (e.g., snake blennies, eelpouts, sand eels, shorthorn sculpins). Nonetheless, BRUVs were effective in sampling specific groups that typically occur in the Baltic Sea, like flatfish, gadoids, and gobies. Inferring that, even if BRUVs failed to detect the same richness as traditional sampling methods, they may still be sufficient to provide data on specific groups of interest and to address specific research questions. Notably, flatfish and gobies were consistently recorded in abundances comparable to or higher than those observed in sandy habitats in the North Sea (Howarth et al., 2015).

One approach to further increase detected richness could be the use of different bait types, such as pilchards that are commonly used (e.g., Dorman et al., 2012; Harvey et al., 2007). However, different species exhibit different reactions, that depend upon a variety of stimuli, such as feeding motivation, search patterns, activity, schooling behavior, chemosensory ability as well as the response time of the individual (Bailey and Priede, 2002; Dorman et al., 2012; Stoner et al., 2007). Thus, different bait types may result in different conclusions about the assessed fish community, making it necessary to examine the effects of different bait types in case they are modified in the course of a long-term monitoring or to further increase the efficiency of the method. For example, using a falafel mix as bait type was found to increase the number of small cryptic fish that are often overseen in other studies using bait (Dorman et al., 2012; Harvey et al., 2007; Mallet et al., 2021; Mallet and Pelletier, 2014). To test whether the use of other bait types like cat food or pilchards instead of clupeid-mix could further increase the efficiency of BRUVs in order to assess biodiversity metrics in the Baltic Sea could be an interesting future study.

Although, the use of bait usually enhances species identification by attracting individuals closer to the camera (Dorman et al., 2012; Harvey et al., 2007), species identification remains limited in the Baltic Sea due to the combination of similar morphotypes of different species, e.g., dab and plaice, and poor visibility. This is a persistent limitation, reducing the potential of video-based approaches for precise assessments of fish biodiversity.

Optimal sampling design

To determine optimal sampling design for the assessment of temperate, low diversity, soft bottom fish assemblages and to compare efficiency of both methods we investigated the optimal deployment time and number of samples required to assess family richness.

We found that optimal deployment time and number of samples varied between the two methods. BRUVs did yield a higher number of species with a lower number of samples and a shorter deployment period, indicating that the use of bait increases the efficiency of stationary underwater video systems. Optimal deployment duration for BRUVs was estimated to be 33 minutes. This is in line with studies from tropical and subtropical as well as warm-temperate locations, where optimal deployment time is suggested to be between 30 and 49 minutes (Birt et al., 2021; De Vos et al., 2014). Another study that was conducted in a warm-temperate region however, estimated optimal deployment time to be at least 60 minutes (Birt et al., 2021), while other studies that were conducted in temperate locations found no difference in the assessed fish assemblage between deployment durations of 30 and 60 minutes (Harasti et al., 2015). A study that estimated optimal deployment time for unbaited stationary underwater video for the assessment of tropical soft bottom fish assemblages found that 95% of the theoretical species richness was observed within 17 minutes (Mallet et al., 2021), which is comparable to the estimated deployment time to reach the maximum of the mean family richness for RUVs in our study. These findings support the assumption that a combination of habitat type and fish density is a key driver of the variability in optimal deployment times (Gladstone et al., 2012). Another interesting observation is that the majority of species enters the field of view within the first ten minutes of the deployment period independently of the treatement. This observation again is in line with results from other studies, where most of the species and individuals occur in the first 15 minutes of deployment duration (Birt et al., 2021). Based on individuals our results reveal a different pattern, where abundance of flatfish in BRUV samples increased during the deployment period and was highest towards the end of the recording, while abundance in RUV samples was relatively stable across the deployment period (see supplementary material: Figure S3). Vice versa, we observed a different pattern for the abundance of gobies in observations taken with BRUVs, where highest abundances have been recorded after about ten to 15 minutes and decrease towards the end of the video recording, while abundances where again relatively stable in RUV observations, yet still increasing from five to ten minutes (see supplementary material: Figure S4). This inverse pattern, in which the number of flatfish increases while the number of gobies decreases, supports the assumption that the presence of species belonging to higher trophic levels can lead to the displacement of lower trophic species (Cappo et al., 2004; Farnsworth et al., 2007; Harvey et al., 2007; Jones et al., 2003; Shah Esmaeili et al., 2021; Stoner et al., 2008; Watson et al., 2010; Willis et al., 2003). The absence from other families, commonly observed in the area could also be related to deployment time, since the probability of a species to be observed is dependent on its home range, mobility, general abundance of the species and the level of attraction to the bait (Birt et al., 2021). Species with greater home ranges and mobility might take longer to arrive in the field of view (Birt et al., 2021; Gore et al., 2020). Yet, species associated with high mobility and great home ranges are often larger predatory species. Other demersal species known to be common in the area based on previous investigations with 3m beam trawls, which did not occur in any of the recordings, include individuals from the families *Agonidae*, *Psychrolutidae* and *Triglidae*. To our best knowledge, there is no research on home range and mobility behaviour of these families. Still, the most common species in the studied area belong to the groups of flatfishes, gobies or gadoids, which were represented in BRUV recordings, suggesting that a deployment time of about 33 minutes might be adequate for the detection of these groups. Yet, optimal deployment time is not only influenced by fish behavior, but also by the extent of the bait plume. The substratum's topographic complexity, wave action and strength and direction of bottom currents all influence bait plume dispersal (Birt et al., 2021; Cappo et al., 2007, 2003; Priede and Merrett, 1996). Consequently, the area that is influenced by the bait plume might increase with strengths of currents. It is similarly, considered that a longer deployment duration will result in a larger potential area sampled (Birt et al., 2021; Taylor et al., 2013). A larger sampling area could also be covered by increasing the number of sampling stations. Consequently, the choice of strategy may favor either longer video recordings at a limited number of stations or shorter recordings distributed across a greater number of stations.

We detected the same pattern observed for the comparison of optimal deployment time between methods when comparing the number of samples required to detect theoretical family richness. With the same number of samples BRUVs yielded about double the richness compared to RUVs. Yet, family richness in BRUVs is estimated not being fully covered by the number of samples, suggesting that more species/families would occur, with increased sample size. This pattern is likely driven by the single occurrence of an individual from the family *Scophthalmida*e (*Scophthalmus maximus*), which may be skewing the richness estimates. A study investigating optimal number of samples for the assessment of tropical demersal soft bottom fish species found 80% of the theoretical species richness being observed after 17.9 stations/km^2 (Mallet et al., 2021), which is comparable to our finding that most families have been detected after about 19 samples in an area that covers approximately 1km^2.

Limitations and conclusion

One important component of long-term monitoring programs that assess fish assemblages are length distributions and biomass estimates (Tessier et al., 2016). Accurate and precise species-specific length measurements require the use of stereo-video cameras (Dorman et al., 2012; Harvey et al., 2007). To acquire representative length distributions a certain number of individuals needs to be measured. We did not account optimal sampling design in terms of deployment time and number of samples to derive representative length distributions in our study. Since this would require in-situ before and after calibration of the stereo-video pair, which was not possible for this study due to logistical and technical reasons. Therefore, the logistical effort that is required to enable pre- and post-calibration of stereo-video can be a limitation (Whitmarsh et al., 2017) for the use of BRUVs in monitoring programs that aim to detect biomass and length distributions of fish assemblages. Calibration of the cameras requires good visibility, constraining the possibilities to complete calibration in free water, which is why calibrations in our case can only be conducted in a pool with the assistance of divers. When stereo systems are applied over a course of several days or weeks and discrepancies between pre- and post-calibration measurements arise, it is almost impossible to comprehend which videos are still usable for length measurements. However, optimal deployment time might need to be adjusted in order to compile species-specific length data that is representative of the community.

A major limitation of BRUVs and RUVs is the inability to identify individuals up to species level due to the specific conditions in the Baltic Sea, like poor visibility and low species- and functional diversity. To apply these methods in biodiversity studies in this region, further technological improvement or the integration of complementary methods is required. Yet, our findings show that the use of bait significantly enhances the efficiency of detecting family richness and abundance and could provide a sufficient method for specific groups or to address specific research questions, e.g., assessing relative changes in the abundance of flatfish, in areas characterized by sandy bottoms, with low species- and trophic diversity.

Acknowledgements

We thank Rainer Stechert, Tom Jankiewicz and Matthias Woltering for their support in data collection. We also thank Nakula Plantener for providing help with figures (Figure 1). This study was carried out as part of the DAM pilot missions “MGF-Ostsee” and “MGF-Ostsee II” funded by the German Federal Ministry of Education and Research (funding codes: 03F0848E/03F0937E).

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Supplementary Material

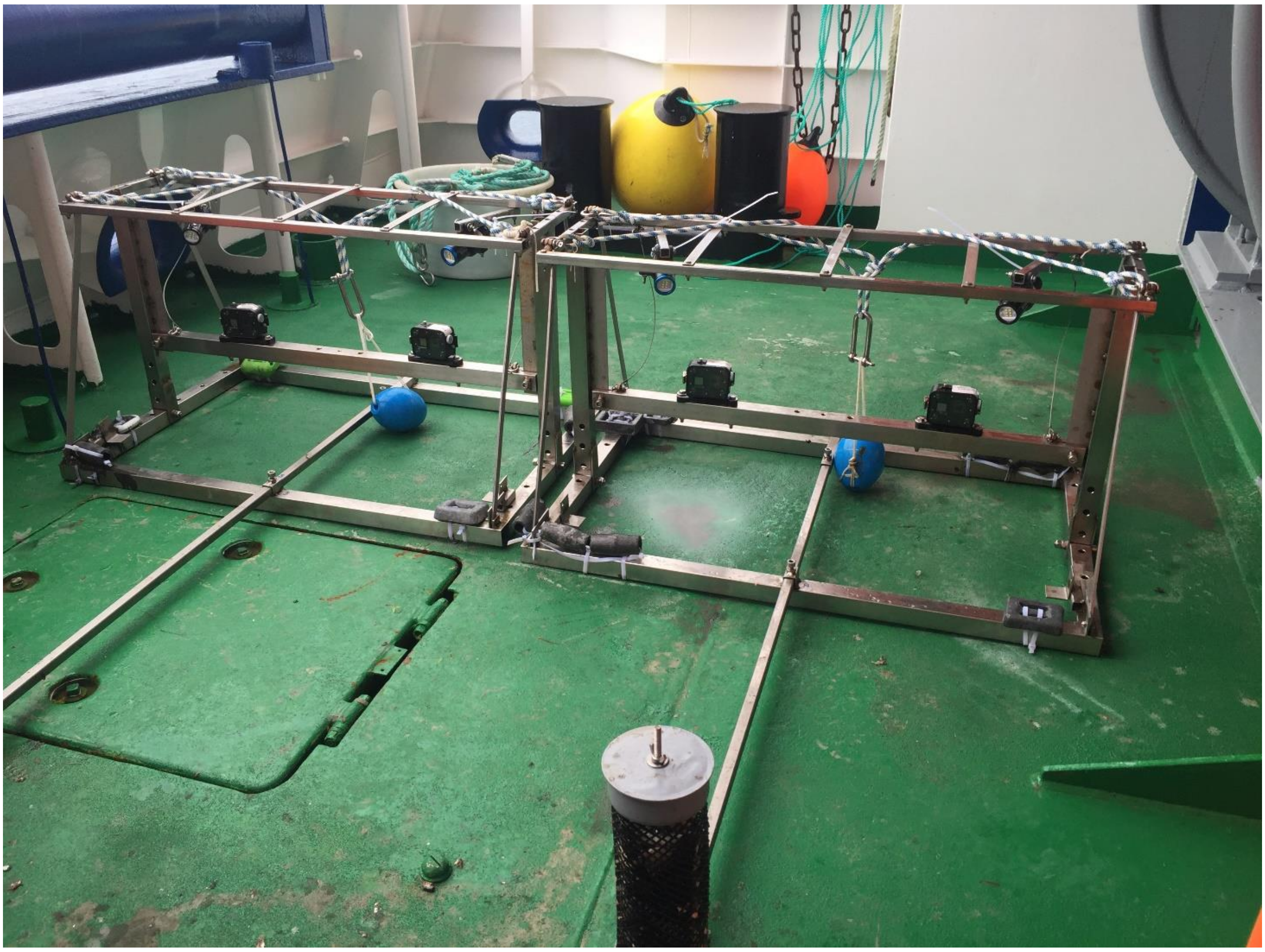

Figure S1: Picture of the camera frames.

Table S2: Potential species (within detected families) occurring in the area, with associated habitat, behavior, and diet.

Species	**Common name**	**Family**	**Habitat & behaviour**	**Diet**
Aphia minuta	Transparent goby	*Gobiidae*	Pelagic, schooling	planktivorous
Gadus morhua	Atlantic cod	*Gadidae*	Benthopelagic & schooling	piscivorous and benthivorous
Gobius niger	Black goby	*Gobiidae*	Demersal, non-schooling	piscivorous
Hippoglossoides platessoides	American plaice	*Pleuronectidae*	-	piscivorous & benthivorous
Limanda limanda	Dab	*Pleuronectidae*	Demersal, non-schooling	piscivorous & benthivorous
Merlangius merlangus	Whiting	*Gadidae*	Benthopelagic, schooling	piscivorous & benthivorous
Neogobius melanostomus	Round goby	*Gobiidae*	Demersal, non-schooling	piscivorous & benthivorous
Platichthys flesus	European flounder	*Pleuronectidae*	Demersal, nocturnal and burrowing, non-schooling	piscivorous & benthivorous
Pleuronectes platessa	European plaice	*Pleuronectidae*	Demersal, non-schooling, resident intertidal species with homing behavior	piscivorous
Pomatoschistus minutus	Sand goby	*Gobiidae*	Demersal, sometimes in schools, with small home ranges	benthivorous
Scophthalmus maximus	Turbot	*Scophtahlmidae*	sluggish	-

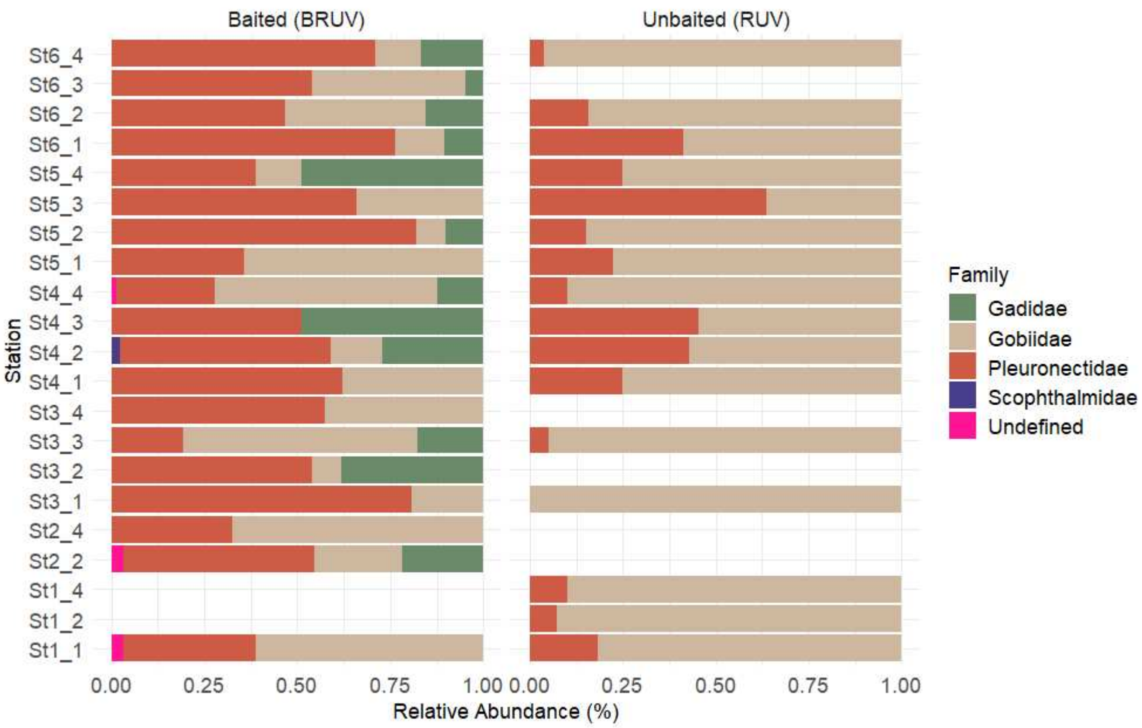

Figure S2: Species composition plots by station for both treatments.

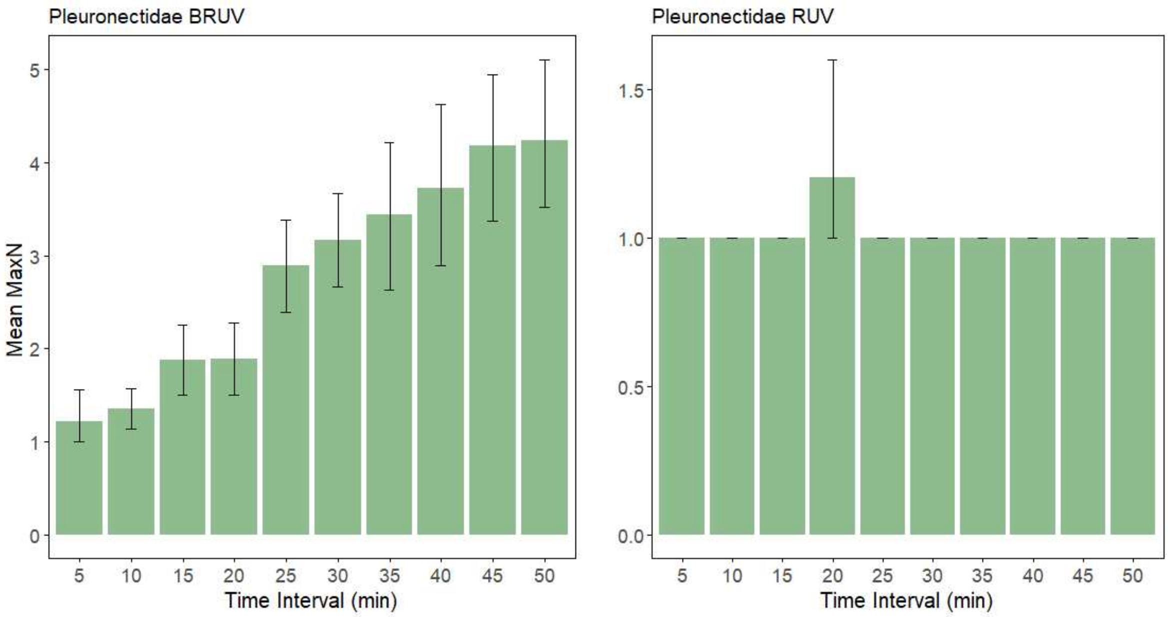

Figure S3: Mean MaxN of *Pleuronectidae* per time interval for BRUVs and RUVs.

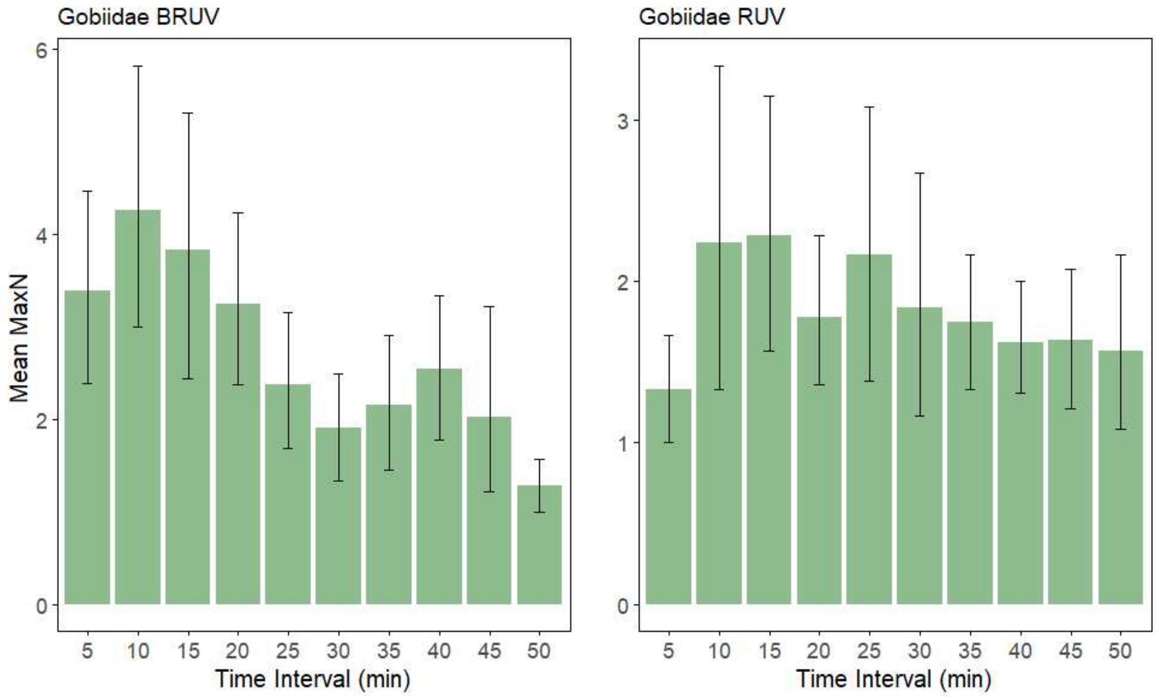

Figure S4: Mean MaxN of *Gobiidae* per time interval for BRUVs and RUVs

GENERAL DISCUSSION

The global extent of strictly protected areas requires alternative strategies to ensure that these areas do not become a blind spot for impact assessment as well as for the monitoring of fish communities. In the case of the Baltic Sea this need is highlighted by the recent exclusion of fishing with mobile bottom contacting gear from parts of MPAs. For this reason, the overall aim of this dissertation was to evaluate currently available alternative and less-invasive monitoring methods for the detection of changes in demersal fish communities, particularly within Marine protected areas of the Baltic Sea. Additionally, the findings of this dissertation project may support the development of less-invasive methods suitable for monitoring of OWFs where traditional methods like trawls are excluded due to safety reasons.

To date, the probably most commonly applied method for the monitoring of bottom dwelling fish species are bottom trawls (Garces et al., 2006; Maureaud et al., 2021; Trenkel et al., 2019), but passive gears like traps (Bacheler et al., 2013) or gillnets for e.g., coastal fish (Helcom, 2019a) are also used for this purpose. Yet, there is ongoing development in the application of non-invasive monitoring strategies like various techniques that are based on underwater video, acoustics or molecular methods. Studies that evaluate these methods for monitoring fish communities are mostly located in tropical, warm or warm- temperate regions and often focus on reef fish. Implications of these studies are probably not applicable to the special characteristics of the Baltic Sea, like low species- and functional diversity as well as poor visibility. Furthermore, the MPAs that are the focus of this work are typically characterized by sandy bottoms, where the distribution of fish is often patchier, and the diversity and density of colonization are lower compared to more structured habitats such as reefs. Therefore, the application of emerging non-invasive methods has to be evaluated for the specific purpose of monitoring demersal fish communities in Baltic Sea soft-bottom habitats. The structure of the General Discussion of the thesis centers around the detection of ecological change and associated challenges, with the focus of the method-evaluation being on the comparability of relative baselines, factors influencing the performance of underwater video-based methods in the Baltic Sea and the sampling effort required. In addition, I reflect on the limitations of the method and the studies presented to identify further research gaps that could be addressed in the future.

Summary of key findings

Identification of fit-for purpose methods

This literature review aimed to identify the limits and opportunities of monitoring methods that are already available. From this meta-analysis we developed a fit-for purpose guide that can support the identification of an appropriate method for the individual research question or purpose. For the monitoring of fish communities, we defined eight different methods, which were analyzed based on four criteria: target parameters, target species, target habitat and the resources required. Additionally, we listed the advantages and disadvantages of each method. In a next step, we applied this guide on the case of Baltic Sea Marine protected areas, suggesting that besides traditional bottom trawling, alternative and less-invasive methods could be sufficient for the monitoring of demersal fish communities in these areas. These methods include passive fishing (e.g., gill netting, angling), underwater video (towed or stationary), eDNA or hydroacoustic methods (acoustic tagging, acoustic camera). Yet, compared to traditional bottom trawling, all of these methods provide less information and need further development, particularly in terms of standardization. It is further argued that, since no single method is able to measure changes in fish communities accurately and precisely without introducing its own biases, these biases have to be assessed beforehand. Another important finding of this study is, that to date there is no complete alternative for data collection in the context of the EU fisheries management. For this purpose, extractive methods are required in order to collect data on age structure and reproductive capacity. The methods evaluated in this dissertation project were chosen based on this literature review, which serves as its foundation and complements the introduction.

Remote underwater video as an alternative to bottom trawling

One of the identified suitable methods for the monitoring of demersal fish communities in Baltic Sea MPAs is underwater video. Remote underwater video systems (RUVs) are among these methods the closest to being non-invasive and were successfully used in many bioregions on a global scale. Yet, conditions in the Baltic Sea differ significantly from the regions where these methods have been previously applied. We therefore, investigated whether RUVs can represent a non-invasive alternative for monitoring demersal fish communities in sandy habitats in the western Baltic Sea. Here, we compared different biodiversity measures derived with a 3m-beam trawl and RUVs as well as the ability to detect family richness as a function of sampling effort. Our findings show that while RUVs generally result in diversity indices lower than derived with 3m-beam trawls and required a higher effort to detect the same level of family richness, they could be sufficient to derive information on specific species. However, their efficacy in monitoring benthic and demersal fish communities in Baltic Sea MPAs might be constrained by their limitation in species identification and the effort required. To improve the performance of RUVs, complementary methods or technological advances are needed to establish them as an applicable method for monitoring low diversity ecosystems dominated by sandy habitats.

Comparative analysis of baited and unbaited underwater video

Remote underwater video systems were found to provide very limited data on fish diversity (Study II). In addition, the restrictions on species identification and the high effort required may constrain their applicability for monitoring fish communities in the Baltic Sea. One approach to increase the efficacy of remote underwater video is the use of bait to attract fish to the field of view. Yet, the use of bait is associated with the introduction of bias towards specific species. Methods that are to be used in long-term monitoring programs require standardization and, in this context, an evaluation of biases. We therefore, aimed to quantify biases and evaluate the optimal number of samples and the optimal deployment time required to detect a certain level of diversity by a comparison of baited and unbaited remote underwater video stations. Here, we found that the efficiency of underwater video stations can be increased by the use of bait. All evaluated biodiversity indices were higher in BRUV samples compared to RUV samples and the effort required to reach the same level of richness was lower for BRUV samples. BRUVs still did not yield a comparable detection of diversity like e.g., beam trawls in this area, however, they were found to increase the abundances of specific species, indicating that they could be a sufficient method for the assessment of specific groups. The major limitation of difficulties in identifying individuals up to species level remains, suggesting that these methods should either be applied with a complementary method or should be further refined in terms of video resolution of the cameras.

Detecting ecological change – methodological considerations

Detecting ecological changes associated with environmental impacts and quantifying the effectiveness of environmental management actions requires good experimental design and representative sampling methods (Andrew and Mapstone, 1999; Harasti et al., 2015; Underwood, 1993). Thereby, a monitoring concept should be designed based on the questions to be answered, the data requirements, habitat characteristics, the target species and the resource availability (Henseler and Oesterwind, 2023) as well as conservation management decisions and technical restrictions (Study I). Furthermore, the standardization of methods used in monitoring programs is essential to accurately detect changes in comparison to baseline assessments. Thus, integrating a new method into an ongoing monitoring program is challenging because the baseline for detecting change is affected by the assumptions inherent to the previously used method. This is particularly relevant as species- and size selectivity patterns vary among gear types due to gear design and capture method (Bosch et al., 2017; Christiansen et al., 2020; Fraser et al., 2007; Huse, 2000; Rotherham et al., 2007; Wells et al., 2008). Thus, no single method provides the capacity to accurately represent fish community composition (Bosch et al., 2017; De Vos et al., 2014). For assessing MPA effectiveness a so called Before-After Control Impact (BACI) approach is often applied, since it is capable of disentangling protection effects from natural variability or external influences (Green, 1979). However, the BACI approach is based on relative comparisons made before and after implementing an intervention. Generally, this means that any alteration to the monitoring strategy complicates the interpretation of the before-and-after condition or may even render the baseline meaningless. This further, emphasizes the importance to quantify biases and limitations as well as to standardize methods used in monitoring. To account for the gear-specific selectivity in the first place and aim to produce a more comprehensive picture of the demersal fish communities inhabiting the investigated area, additional samples were collected with a traditionally used otter trawl (TV-520/BITS trawl) close to the study areas. This trawl type is commonly applied in scientific monitoring of commercial fish stocks in the Baltic Sea and is the standard gear for the Baltic International Trawl Survey (BITS). Comparing the observed diversity between both methods demonstrates, that even methods that rely on the same principle (e.g., active, bottom towed gear) can present rather different fractions of the fish community (Figure 3). Which is consistent with findings from the Mediterranean, where the comparison of the Jennings Beam trawl (Jennings et al., 1999) and the traditionally used otter trawl for data collection in stock assessment also found that the otter trawl yielded higher estimates of fish abundance and biomass and also provided different information on the fish community (Farriols et al., 2024). It is rather logical, that otter trawls result in higher detected biomass and number of species since they cover a larger sampling area, have a higher horizontal and vertical opening and can be towed at a higher speed, making these more suitable to capture species with higher swimming capacity and the ability to move upwards from the bottom (Farriols et al., 2024).

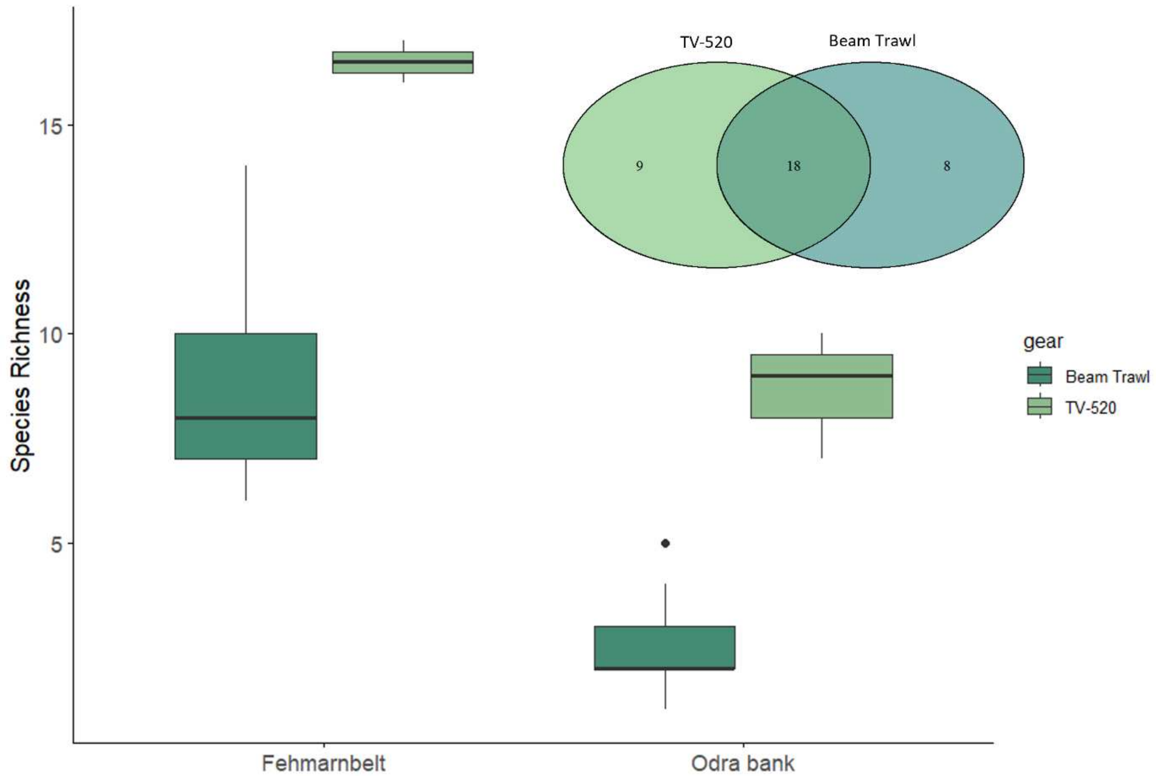

Figure 3: Boxplot of detected species richness and a Venn diagram showing the number of species detected using the Beam Trawl (dark green) and the TV-520 (light green) in the two study areas (Fehmarnbelt and Odra Bank).

The catch efficiency and selectivity of a bottom towed sampling gear is not only influenced by fish behavior (e.g swimming speed, herding behavior, net-avoidance) but also by the sweep length (Engas and Godo, 1989; Fraser et al., 2007), mesh size (Suuronen and Millar, 1992), net spread (Rose and Nunnallee, 1998), trawl speed and duration (Ehrich and Stransky, 2001), and the size and type of the trawl ground gear (Engas and Godo, 1989; Walsh et al., 1992) but it can also be influenced by environmental factors like water clarity or light intensity (Buijse et al., 1992). Consequently, the catching efficiency of each sampling gear will be different and can only ever provide a gear-specific perception of the community being sampled (Bosch et al., 2017). The fact that even the beam trawl does not provide a comprehensive documentation of the fish community highlights the importance of relative 'before' and 'after' comparisons and the associated standardization of sampling methods. In order to reliably detect changes, it is therefore essential to understand the factors influencing catchability and limitations of the newly integrated method compared to the previously used method.

Evaluation of less-invasive methods

A summary of available less-invasive methods, together with a guide to selecting the most appropriate method for a specific purpose, is provided in Study I. Based on the findings of this study, potential alternative methods for monitoring demersal fish communities in MPAs in the Baltic Sea have been identified. Study II and Study III of this dissertation are focused on stationary underwater video methods as they have been considered viable alternatives or complementary approaches to many extractive techniques, such as longlines (Ellis and DeMartini, 1995), demersal trawls (Cappo et al., 2004; Morrison and Carbines, 2006; Priede and Merrett, 1996), beach seines (Shah Esmaeili et al., 2021) traps (Bacheler et al., 2022, 2013b; Bacheler and Shertzer, 2014; Wells et al., 2008) and angling (Willis et al., 2000) and have widely been applied for change detection in MPAs (Figure 4) (e.g. Bornt et al., 2015; Harasti et al., 2015; Heyns-Veale et al., 2016; Hill et al., 2014; Howarth et al., 2015; Parker et al., 2016; Stobart et al., 2015). In addition, these methods are among the closest to being non-invasive and allow an estimate of relative abundance, which is, at least to date, not readily possible with approaches such as eDNA sampling. However, these methods have only rarely been used in Europe (Figure 4) and their application has mainly been focused on reef habitats, which are associated with rather high levels of diversity (Cappo et al., 2004; Mallet et al., 2021; Whitmarsh et al., 2017). Their use in temperate sandy habitats remains uncommon (Figure 4), impeding comparisons between existing studies and the results presented here and further highlighting the distinctiveness of the results presented here.

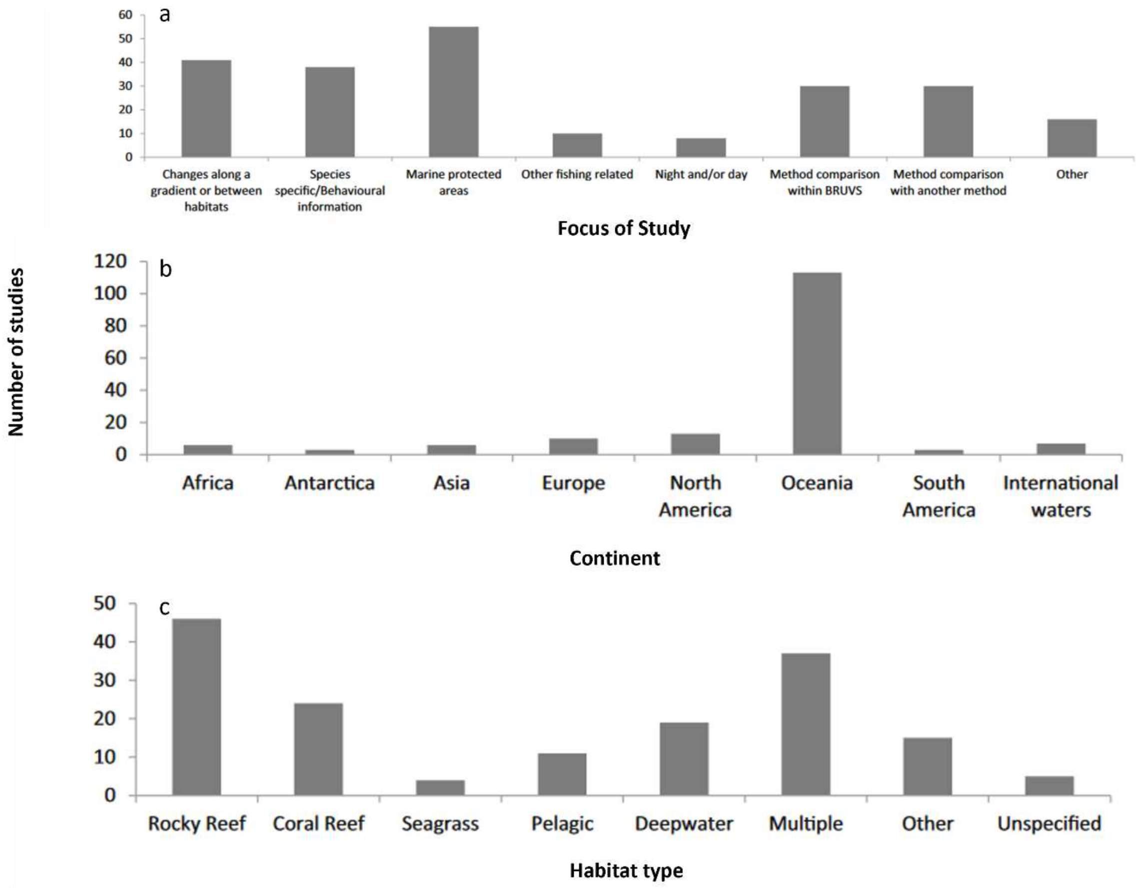

Figure 4: The number of studies using either RUVs or BRUVs for different (a) study focus, (b) continents, and (c) habitat types (Whitmarsh et al., 2017, modified). **a** the category "other" includes all studies that did not fit in any other category (e.g. artificial reefs). **c** "Multiple" includes studies were more than one habitat type was studied, "Other" includes sand habitats and deep water.

Previous studies comparing either baited remote underwater video or remote underwater video with active methods (trawling, seining) found that fish community composition differed between the two methods and active methods generally yielded higher richness (Cappo et al., 2004; French et al., 2019; Shah Esmaeili et al., 2021). Which is consistent with findings of STUDY II and STUDY III that show that the recorded fish diversity and abundance are not comparable with the existing baseline (based on catches with 3 m beam trawl). Most studies using underwater video report of detecting larger and highly mobile and schooling species (e.g., Cappo et al., 2004; French et al., 2019; Shah Esmaeili et al., 2021), but generally fail to detect smaller bodied and cryptic species (Cappo et al., 2004; Whitmarsh et al., 2017). Findings of STUDY II and STUDY III revealed an opposing trend, as neither RUVs nor BRUVs were successful in detecting additional species that could not be detected with beam trawls, but were able to detect smaller bodied and cryptic species, like gobies or blennies. The partly contrasting results may be explained by the special conditions prevalent in the Baltic Sea and the difference in investigated habitat types, which included subtropical, tropical structured, and sandy habitats compared to temperate sandy habitats in the Baltic Sea. The combination of low abundance and patchy distribution of individuals in sandy habitats can make the use of passive methods, such as RUVs, challenging, as their detection efficiency depends on the activity of the fish. Previous applications of RUVs and BRUVs in the Baltic Sea have been limited to the monitoring of fish assemblages in artificial boulder reefs. These investigations revealed the presence of 30 marine fish species, with gadoids, flatfish, and gobies comprising the predominant groups (Wilms et al., 2021). This aligns with our observations in STUDY II and STUDY III, where flatfish, gobies and in BRUVs also gadoids were identified as the predominant groups. Notably, these studies also demonstrated the efficacy of RUVs and BRUVs in detecting small and cryptic species, such as gobies. Additionally, BRUV sampling in a temperate sandy habitat revealed gadoids as the primary fish group, followed by flatfish and gobies although they were detected in considerably lower abundances (Howarth et al., 2015). Given the consistency of these identified main groups with beam trawl catches, RUVs or BRUVs appear suitable for addressing research questions related to these taxa, yet with considerable constraints regarding taxonomic resolution.

Despite the influence of habitat structure and climatic zone, detection frequency of the described fish groups was also affected by other factors. In particular, the detection frequency of flatfish and gobies exhibited variability across study areas (STUDY II). This variability was likely driven by disparities in salinity, reduced species diversity, and lower fish abundance, rather than sediment structure, as both study sites were characterized by sandy bottoms. This could hint to differences in fish density between sampled areas and the random reliability on an individual passing the cameras field of view, suggesting that frequency of occurrence may serve as a proxy for changes in relative abundance (STUDY II). Another factor that could influence the detectability and detection frequency is the higher attraction to the light in slightly deeper waters. Artificial light stimuli are known to influence fish behavior and have been employed in fishing to attract individuals and increase commercial catches (Nguyen and Winger, 2019; Tomiyasu et al., 2022). However, the effectiveness of light attraction varies depending on the wavelength and the sensitivity of individual species towards particular wavelengths (Fitzpatrick et al., 2013; E.S. Harvey et al., 2012; Marchesan et al., 2005). Studies investigating the effects of different wavelengths (red, white, blue) on species detectability have produced mixed results. Fitzpatrick et al. (2013) found that red light attracted the highest abundance of nocturnal reef fish, particularly non-commercial species. As red-light wavelengths are believed to fall outside the visual sensitivity range of most fish species, they concluded that red light did not alter the behavior of nocturnal fish, whereas white and blue light did. Harvey et al. (2012) reported that white light was more effective in capturing a greater number of individuals and fish species during the day. To my best knowledge, no studies have investigated individual responses to the interaction between light and water depth, but findings from Harvey et al. (2012) as well as the use of artificial light in commercial fishing, suggest that white light can act as an attractant under dark conditions. However, the behaviour of gobies and flatfish species towards artificial light sources has not been evaluated so far. Understanding how these species may respond to different light conditions may be beneficial to interpret abundance data and avoid potential biases.

A fundamental factor influencing both efficiency and detection frequency is the usage of bait. Bait is commonly employed to attract a greater number of species and individuals to the field of view, thereby enhancing the efficiency of remote underwater video methods and expanding the sampled area. In order to increase the performance and enhance the efficiency of remote video methods in the Baltic Sea we tested whether baited RUVs and BRUVs provide different estimates of richness and abundance by a paired comparison of BRUVs and RUVs (STUDY III). Our findings reveal the same pattern like comparable studies (Aston et al., 2024; Bernard and Götz, 2012; Dorman et al., 2012; Harvey et al., 2007; Watson et al., 2005b), that the performance and efficiency of the method can be increased by the use of bait. However, in contrast to other studies using BRUVs, we still yielded far lower richness than is known from prior catches using bottom otter trawls in the study area (Figure 3), suggesting that neither RUVs nor BRUVs can provide a comprehensive inventory of fish diversity in sandy bottoms in the Baltic Sea. The higher abundance of flatfish observed with BRUVs compared to unbaited RUVs (STUDY III) is consistent with findings from other studies reporting increased numbers of piscivorous and benthivorous fish species in baited samples (Bernard and Götz, 2012; Harvey et al., 2007). This was somewhat expected, as bait is known to attract species with piscivorous and benthivorous feeding habits. In this study, the flatfish group mainly includes dab, plaice and possibly flounder (as known from trawl catches). While flounder and dab are both piscivorous and benthivorous, plaice is predominantly benthivorous (Henseler and Oesterwind, 2023). The attraction of benthivores and piscivores fish by bait is further emphasized by the presence of gadoids in baited samples which are however absent in unbaited samples. Sampling more individuals of a species can be advantageous as it decreases the variance of individual species sampled and can minimize false negatives, which can improve the statistical power to detect differences between samples or groups (e.g., in- and outside an MPA) (Harvey et al., 2007; Watson et al., 2005b; Willis and Babcock, 2000). Additionally, the higher number of individuals of one species/group can result in higher precision of length measurements using stereo-systems (Harvey et al., 2002b, 2002a; Harvey and Shortis, 1995). Generally, the detection of flatfish using remote video systems has rarely been reported, Cappo (2004) did not observe one single flatfish in BRUV samples while they were abundant in trawl samples. While flatfish have been observed with BRUVs in temperate sandy habitats, their abundances are generally much lower compared to gadoids (Howarth et al., 2015). This is in contrast, to the findings presented here, where flatfish are the most abundant group observed within RUV and BRUV samples and was somehow unexpected since these species are characterized by a sedentary lifestyle and are rather slow-moving (Cappo et al., 2004). However, the findings of Study II and Study III suggest that due to the high frequency of occurrence as well as quite balanced occurrence between samples RUVs or BRUVs may be a suitable method to estimate changes in flatfish abundances, yet the use of bait is advantageous.

Sampling effort

There is a considerable variation in the applied effort used in studies investigating remote underwater video methods (baited or unbaited), with replicate numbers varying between one and >20 per sampling station (Whitmarsh et al., 2017), sometimes resulting in the analysis of over 50 video recordings per year for the evaluation of MPA effectives (Blampied et al., 2022; Walsh et al., 2017). This variability is further impeding comparability between studies (Whitmarsh et al., 2017). Rarely do researchers provide information about the effort that might be required to detect a certain level of richness or answer a specific research question. Further, investigating optimal sampling design in terms of number of stations and deployment time can increase the sampling efficiency by reducing per-sample data processing time and can thereby significantly alleviate one of the main bottlenecks of underwater video methods—the intensive post-sampling effort required for video analysis (Misa et al., 2016). Reductions in effort can also help minimise the environmental impact of monitoring but data quality should not be compromised (Trenkel et al., 2019).

Species accumulation curves were used to determine the number of stations where no additional species detections would be expected based on the models (Study II and Study III). Overall, family richness detected by RUVs remained relatively low, even after hundreds of samples taken, models predictions suggest that no additional species/family would be detected. This pattern is consistent with results from comparisons of beach seines and BRUVs on sandy beaches in Brazil, where species richness detected by BRUVs saturated after approximately 50 samples, whereas the estimated richness detected by seine nets was predicted to further increase even after 150 samples taken (Shah Esmaeili et al., 2021). The estimated number of sampling stations required for RUVs, where no new species were expected was reached relatively quickly, with no additional species detected after approximately four samples (sufficient for a sampled area of approximately 1 km^2). This finding is consistent with results from Mallet et al. (2021), who estimated that 6.8 stations per km^2 are sufficient to detect 80% of the species richness on soft bottom habitats in New Caledonia. In contrast, BRUV observations were predicted to stop detecting new species after about 20 samples. BRUVs yielded approximately twice the species richness of RUVs after only two samples, suggesting that the effort required to detect the same level of richness is reduced when using bait. Previous studies have assessed the statistical power of different sample sizes to detect a 50% change in fish assemblage composition and found no difference between BRUVs and RUVs for detecting reef fish (Schramm et al., 2020), while BRUVs had greater statistical power than RUVs for equivalent sample sizes in seagrass habitats (French et al., 2021).

Regarding optimal deployment time, indications from STUDY III were similar to those of other studies that have investigated this aspect (Birt et al., 2021; De Vos et al., 2014b; Mallet et al., 2021). In temperate rocky reefs and estuarine seagrass and soft bottom habitats deployment times of 30 minutes provided a reliable and representative estimate of fish diversity and abundance (Birt et al., 2021; Gladstone et al., 2012; Harasti et al., 2015), while 33 minutes were estimated to be sufficient for sandy bottoms in the Baltic Sea (STUDY III). Yet two other studies in warm temperate and temperate reefs report optimal deployment times of 29 minutes for species richness but 48 /49 minutes to observe species at their maximum abundance (Bernard and Götz, 2012; De Vos et al., 2014). This could be due to so-called "secondary attraction" (Taylor et al., 2013) which describes the attraction of predatory species over time as a result of the aggregation of smaller fish species (Harasti et al., 2015). Additionally, the effect of bait dispersal from BRUVs must be considered, as the dispersion of the bait plume may increase over time, potentially attracting individuals from a wider area (Taylor et al., 2013). This is supported by the shorter deployment times required when using unbaited RUVs, where 35 minutes were found to be sufficient to detect 95% of the species at their maximum abundance (Bernard and Götz, 2012). Regarding species richness (family richness), the optimal deployment time to observe 95% of the expected richness using unbaited RUVs was estimated to be between 17 and 20 minutes, which is consistent with results of Study III (Bernard and Götz, 2012; Mallet et al., 2021). For conservative reasons, a 60-minute deployment duration remains commonly recommended and used (Birt et al., 2021; Whitmarsh et al., 2017). However, longer deployment times increase boat time, person hours in the field and post analysis time, thereby increasing overall sampling costs (Gladstone et al., 2012; Harasti et al., 2015). This highlights the importance of selecting optimal deployment durations based on the specific research objectives as well as the available resources.

Questions that remain open are whether specific species or groups that did not occur within the sampled stations or deployment time would likely to be observed by extending deployment time or increasing the number of samples, like predicted by the models. However, these models are based on the available data, which means that the probability of detecting additional species depends on the patterns observed in the samples already collected. Therefore, these models may be limited by the original sampling design and may not fully capture species that are rare or not observed in the sampling effort. In order to account for these uncertainties, further experiments with a longer deployment time and a larger number of samples would be required.

The evaluation of optimal sampling effort, in terms of both the number of stations and the optimal deployment time, is a significant contribution to the standardization of video-based methods for the application in Baltic Sea soft-bottom habitats. The findings indicate that even baited RUVs are limited in the detection of species richness by constraints in species identification as well as general low detection of diversity. Yet, BRUVs may allow the data collection for abundance estimates for specific groups like flatfish or gobies. Since half of the species richness was detected after just five BRUV samples, and flatfish and gobies were present in nearly all samples, the sampling effort could be further reduced by ensuring deployment times of at least 33 minutes. This could further reduce the effort required. The findings also indicate that the required effort aligns with that of other studies conducted in temperate regions. Therefore, the estimated optimal deployment time (33 minutes) and sample size (20) can support the development of monitoring programs using remote underwater video methods, particularly in regions and habitats with similar characteristics.

Limitations of Applicability within different marine areas

A notable constraint associated with the application of RUVs and BRUVs in Baltic Sea soft-bottom habitats is the inability to identify individuals to species level in most samples (STUDY II & STUDY III). This contrasts with international findings, where species identification is less problematic (e.g., Malcolm et al., 2007). According to Whitmarsh et al. (2017), no study reported more than 40% of individuals being unidentified at the species level. A combination of the large overlap in morphological characteristics of the species found in the Baltic Sea and the already low diversity is probably responsible for this issue. STUDY II and STUDY III focus on areas that are dominated by flatfish, especially dab (*Limanda limanda*) and plaice (*Pleuronectes platessa*). These species are mainly distinguished by skin structure, coloration, and the shape of the lateral line system. All of these characteristics are difficult to discern in video footage and require very good visibility. Smaller individuals, in particular, are almost impossible to differentiate.

The generally low species richness found, combined with the difficulty of species identification, suggests that RUVs and BRUVs are not well suited for studies requiring detailed fish diversity inventories, especially in sandy habitats. Further technological advances, such as improved camera technology, using different or additional attractors such as light (see above), sound or reflectors, offer a potential solution to increase the number of species detected as well as improving species identification. The use of other attractors like sound recordings of bait fish and metallic reflectors together with bait have shown to increase abundance of pelagic fish in mid-water baited RUVs (Rees et al., 2015). However, the use of different or additional attractors has not been evaluated for demersal fish assemblages. This suggests that future research to improve the efficiency of remote underwater video methods should investigate the applicability of additional attractors, such as different light colors, reflectors, sound or other bait types. To overcome low taxonomic resolution as well as low species detection the combination of underwater-video methods and other non-invasive methods could be of interest. The genetic analysis of water samples (environmental DNA; eDNA) in combination with remote underwater video methods has already proven to provide a wider picture of fish communities in Western Australia (Stat et al., 2019). A study investigating the applicability of eDNA as a method to assess fish diversity in our study areas yielded promising results, where a total of 25 species were detected, five of which found exclusively with eDNA analysis (Piontek et al., in prep,). Yet, the five species that were only detected by eDNA are either pelagic species and therefore, are unlikely to be caught with beam trawls (*Belone belone*, *Salmo trutta*, *Scomber scombrus*) or species that have not been identified up to species level from beam trawl samples (*Ammodytes marinus*, *Pomatoschistus pictus*). Additionally, a comparison of sampling effort indicated that eDNA analysis of four bottom water samples (2-3.5 L) yielded comparable species presence/absence data to 13 beam trawl samples. However, the application of eDNA as a monitoring tool is still evolving, presenting several uncertainties concerning transport and concentration of DNA particles. The potential for false negatives positives and inaccurate species identification due to limitations in genetic databases also persists. Moreover, to accurately detect changes in fish assemblage composition, abundance data is essential, thereby demonstrating the inadequacy of relying solely on presence-absence information and restricting the current use of eDNA as an independent method.

In addition to data on diversity and relative abundance, it may be desirable to investigate other metrics, such as length measurements. Body size measurements can facilitate the delineation of life stages and allow for conversion to fish biomass via established length weight relationships for specific species and thereby offers the potential to provide estimates of biomass, population and recruitment dynamics, and fecundity (Harvey et al., 2002a; Langlois et al., 2020). Regarding the detection of change in MPAs assessing these metrics would be particularly valuable since an increase in individual fish size and biomass is often observed in the context of protected areas (Watson et al., 2009). Additionally, these metrics are of high importance for data supporting fisheries management. Within this dissertation project, efforts to optimize and standardize the sampling design for BRUVs and RUVs have not considered optimizing the design for representative length measurements, due to the failure of pre- and post-calibration of stereo-video systems. Future investigations should therefore, also focus on the evaluation of adequate sampling design (no of samples, soak times) for representative length measurements acquired from remote underwater video footage. This could be achieved by accumulating lengths per species over time using the same approach as presented in STUDY III for diversity data. Nevertheless, studies addressing optimal sampling design, particularly for length measurements, remain limited, with existing studies primarily focusing on species richness and individual fish species abundances (Birt et al., 2021; Harasti et al., 2015).

Acquiring accurate length measurements by video footage often requires stereo-measurements, which, however, involve greater logistical effort due to the required calibration of the stereo system. Calibration adds to both preparation efforts and analysis time. Due to calibration issues, length measurements could not be conducted in any of the studies presented here. Ideally, calibration should take place before and after the gear is used, making planning for calibration crucial. Calibration attempts in open water near to the beach failed multiple times due to unsuitable weather conditions, even small wave action could reduce visibility to the point where calibration was impossible. Calibration in pools requires diver assistance, adding further logistical complexity and coordination with pool operators and diving teams. Since pre- and post-calibration are necessary to ensure no changes in camera positioning during deployment and retrieval, the logistical effort increases. Another challenge is that the gear must be deployed multiple times (about 10-20 times per cruise), and any discrepancies between pre- and post-calibration measurements render all data meaningless, as determining when the camera position may have been altered is not possible.

There are also other logistical challenges, such as the reliability of ‘good’ weather conditions. Visibility in the Baltic Sea is already problematic due to the narrower photic layer and the relatively shallow water depth, whereby even small wave action can disturb the seabed and further reduce visibility. Weather conditions that may still be ‘good enough’ for the use of other methods may already be ‘too poor’ for the use of underwater video. The problem of poor visibility is further exacerbated by the rise in algae blooms, which are driven by climate change and eutrophication in the Baltic Sea (Helcom, 2019b). However, to compare relative baseline values, researchers depend on repeated measurements over time. It is crucial that these measurements are taken under the same conditions, including the same time (season), location, and environmental factors.

Implications for MPA Monitoring

Based on the findings presented here there is currently no alternative method that can provide the same level of information compared to scientific trawling methods. However, specific methods might be able to answer certain research questions related to the exclusion of mobile bottom contacting gear from MPAs. As we know from other bioregions, the exclusion of mobile bottom contacting gear is likely to increase species diversity, abundance, biomass or individual fish size (Edgar et al., 2014; Gaines et al., 2010; Laffoley et al., 2019; Lester et al., 2009). Together, results from Study II and Study III, investigating remote underwater video methods have shown that these methods can provide information about specific groups and potentially are capable of detecting relative changes in abundances of specific groups (e.g. flatfish) followed by the exclusion of mobile bottom contacting gears from MPAs, with the use of bait potentially reducing effort and increasing the number of detected families. These methods may be suitable for flatfish, snake blennies, gobies and potentially gadoids. However, limitations in species identification restrict the ability of RUVs and BRUVs to generate comprehensive inventories of fish diversity, suggesting that these methods alone may be insufficient for detecting changes in overall fish diversity within MPAs. To enable a more precise quantification of methodological differences, support the development of calibration models, and enhance long-term data comparability as well as comparability with existing baselines, a long-term comparative study integrating RUVs/BRUVs with beam trawls in unprotected areas would be highly beneficial. Additionally, the use of stereo-video would enable length measurements and biomass estimates for the aforementioned groups. Nevertheless, responses to protection can vary among species and areas, with some experiencing negative effects (Lester et al., 2009). Achieving the most comprehensive possible representation of individual ecosystem components (e.g., fish communities) would thus provide a robust foundation for assessing ecological changes following the exclusion of mobile bottom-contacting gear from MPAs. However, such data is often unavailable. Accordingly, further research is required to develop methods that can either complement or surpass BRUVs in terms of species detection and identification, age structure analysis, reproductive assessment, and stomach content examination in nutrient ecology studies. Potential approaches include the complementary use of traps, environmental DNA (eDNA), or passive acoustic monitoring (see Study I).

Finally, it is important to emphasize that pilot studies should precede applying any method, especially those still being under development (McGeady et al., 2023; Whitmarsh et al., 2017). Overall, the results clearly show that a globally established method such as RUV/BRUV is limited in its application for monitoring demersal soft-bottom fish communities in Baltic Sea MPAs. Hence, further optimization efforts are needed, either through technical improvements, changes in sampling design or the use of complementary methods. However, adequate financial resources must be allocated to these efforts. It would also be beneficial to define responsibility for monitoring in order to ensure its long-term continuity. In addition, the identification of indicator species could be helpful for the development of an efficient monitoring approach that effectively detects ecological changes.

CONCLUSION

This dissertation provides a baseline for the development of less invasive monitoring methods for demersal fish communities with a particular focus on remote underwater video methods for soft-bottom habitats characterized by low species and trophic diversity.

The findings suggest that remote underwater video methods (baited and unbaited) are not readily applicable for detecting changes in overall fish diversity in low diversity sandy-bottom habitats. Yet, baited remote underwater video methods may be effective for investigating specific groups (flatfish, gobies) or targeted research objectives (e.g., change in abundance). A major limitation is the low taxonomic resolution, yielding data solely on family level. To enhance the detection of changes in demersal soft-bottom fish communities in Baltic Sea MPAs, further technological development is recommended, as well as the investigation of other alternatives or the use of complementary methods, such as eDNA sampling, to facilitate the monitoring of demersal fish communities in areas where the use of traditional extractive methods is not permitted. A key contribution of this work is the development of a fit-for-purpose guide to support the integration or development of other methods. Additionally, recommendations for an efficient sampling design have been derived, which can inform underwater video monitoring not only in the Baltic Sea but also in similar ecosystems worldwide.

Overall, this thesis underscores the specific requirements for designing a robust monitoring strategy for demersal fish communities in Baltic Sea MPAs featuring soft-bottom habitats. While the findings contribute to improving non-invasive monitoring approaches, they also emphasize the importance of pilot studies to optimize sampling strategies for the unique conditions of the Baltic Sea, as insights from other bioregions may not be directly transferable.

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FUNDING

This dissertation was carried out as part of the DAM pilot missions “MGF-Ostsee” and “MGF-Ostsee II” funded by the German Federal Ministry of Education and Research (funding codes: 03F0848E/03F0937E).

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