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REVIEW

Practical implementation of ecosystem monitoring for the ecosystem approach to management

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Summary

1. The implementation of the ecosystem approach means there is a need to monitor an increased range of environmental conditions and ecological components in the marine environment. Many existing monitoring surveys have successfully added tasks or components to an existing monitoring programme while maintaining consistency of time series. This approach is not practical when the immediate data need for a wide range of ecosystem components requires substantial changes to the programme or when collections of different ecological components have conflicting requirements.

2. We propose a more integrated approach aimed at not only assessing change, but simultaneously delivering evidence of the underlying reasons for observed changes. Using principles developed from observational and modelling efforts in the Barents Sea and the wider literature, we distil the essential characteristics an integrated monitoring programme must exhibit. We demonstrate how such an integrated programme can offer substantial operational efficiencies compared to a coordinated approach.

3. Integrated monitoring based on ecosystem processes has significant advantages over the coordinated approach that uses ecosystem states independently and focuses on maximizing precision of each indicator. While integration is needed to address current policy requirements, changes to monitoring risk time-series consistency. However, we explain how such risks can be minimized while at the same time establishing a framework that allows the incorporation of important information from other less flexible data sources to be used in the assessment.

4. *Policy implications*. Process-based integrated monitoring is essential for the ecosystem approach. The focus on ecosystem processes provides the essential elements for future proof efficient management: (i) It provides both unbiased status estimates for reporting requirements and describes the causes of state change. (ii) It minimizes risks to historic time series while coping with changing ecological conditions. (iii) It quantifies ecosystem processes and provides the means to test hypotheses on how different processes interact. (iv) It uses all available information efficiently when used in conjunction with integrated assessments. (v) It is effective due to its adaptability to meet future policy demands and ecosystem requirements while using data in the most efficient manner given these demands.

Key-words: ecosystem modelling, ecosystem variability, integrated monitoring programme, monitoring policy, Marine Strategy Framework Directive, process-based monitoring, status-based monitoring

Introduction

The ecosystem approach to management in the marine environment was born out of the realization that past management focusing on specific human impacts on the ecosystem or individual species (such as commercially exploited fish) had been insufficient to address human impacts on other parts of the system. An approach considering the wide range of indirect impacts was necessary in order to optimize a range of ecosystem services, not just fishing yield, to ensure sustainability (Christensen *et al*. 1996; Mangel & Levin 2005; Crowder *et al*. 2006;

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Longhurst 2006; Murawski 2010). The assessment and management tools underlying the implementation of such a holistic approach need to account for the complexity of ecosystem dynamics that are the result of a multitude of interactions, both between different ecosystem components and between specific components and the environment (Crowder & Norse 2008; Murawski 2010). To support these aspirations, the European Commission has invoked the Marine Strategy Framework Directive (MSFD) designed to assess progress towards maintaining or improving the status of the marine environment (Borja *et al.* 2010). The Common Fisheries Policy (CFP; Jennings & Le Quesne 2012; Qiu & Jones 2013) and Marine Spatial Planning (MSP; Degnbol & Wilson 2008; Katsanevakis *et al.* 2011; Brennan *et al.* 2014) are Commission policies that are interwoven in this legislation. The need for and merits of the ecosystem approach to management have been discussed widely both from a fisheries (Sherman & Duda 1999; : Mangel & Levin 2005; Barange *et al.* 2010; Francis *et al.* 2007) and a marine spatial planning perspective (Plasman 2008; Ray 2010; Katsanevakis *et al.* 2011; Brennan *et al.* 2014). To date, thoughts on implementation have largely focused on the development of indicators. Recent studies compare the merits of specific metrics (e.g. Lyons *et al.* 2010; Borja *et al.* 2011; Ferreira *et al.* 2011; Rice *et al.* 2012) or champion particular MSFD indicators or areas of research (e.g. Bilyard 1987; Snelgrove 1999; Greenstreet *et al.* 2010; Painting *et al.* 2013; Rombouts *et al.* 2013; Modica *et al.* 2014). Looking ahead, there is general agreement that interdisciplinary thinking and thinking across ecosystem components are the key strength of the ecosystem approach and improvements in this area are essential in developing integrated ecosystems assessments in support of ecosystem-based management (Arkema, Abramson & Dewsbury 2006).

Currently, monitoring is being developed based on ecosystem indicators as directed in the legislation (EC, 2008). At the national level and at the regional seas convention, these indicators are developed based on existing monitoring data or in areas where critical data gaps exist on theoretical approaches, for example seabed integrity (Rice *et al.* 2012). Neither considers the full range of ecosystem processes that may help to inform on ecosystem status. This insular development of monitoring can at best lead to operationally coordinated monitoring. At worst, it could provide an inefficient array of disparate monitoring programmes, (Owens 2014), all focused on detecting change in a single ecosystem component.

Indicators should be viewed as the essential link between ecosystem processes and management actions (Degnbol 2005; Dickey-Collas 2014). Ultimately ecosystem responses to pressures need to be understood to manage human activities. Many fisheries' monitoring programmes around the world have been expanded and adapted to include additional ecosystem considerations where these are informative on fisheries' resources (e.g. Kotwicki *et al.* 2012). Such developments can be effective, but progress towards a holistic ecosystem view is slow and unpredictable. The MFSD demands action now (Dickey-Collas 2014) despite suboptimal management in the light of a partial understanding of ecosystem dynamics (Ludwig, Hilborn & Walters 1993). Nonetheless, to assess the multiple interactions amongst indicators and pressures, we must consider how to best collect the data underlying the indicators so that they are comparable at appropriate scales in space and time (Levin 1995; Rose *et al.* 2010).

We argue that unless a step-change towards integrated monitoring is adopted alongside the ecosystem approach to fisheries and marine environmental management, we will make little progress in our ecosystem understanding with the risk that marine resources will be mismanaged. Ecosystem monitoring is a proposed solution, but interpreting the current literature is complicated by a lack of definition of terms. Here, we provide definitions of the different ecosystem monitoring approaches currently being developed (Table 1) and, using illustrative examples, compare the effectiveness of each in terms of practicality and policy benefits. We demonstrate that only truly integrated, rather than coordinated monitoring, can achieve these

Table 1. The current literature on integrated ecosystem monitoring uses the same or similar terms for different activities or designs. To avoid confusion with this loose terminology, we provide definitions to highlight hierarchical improvements in information value from increasingly integrated monitoring

Indicator	Quantitatively defined metric representative of an ecosystem state
Index	Coherent series of indicators evaluating variability in space or time
Time series	Comparably collected set of monitoring data with defined periodicity used to calculate a specific index or indices
Index-based monitoring	Conventional monitoring designed around the purpose of detecting a change in a specific metric through time
Ecosystem monitoring	Monitoring of one or more components of the ecosystem
Coordinated ecosystem monitoring	More efficient ecosystem monitoring by sharing platforms to collect the necessary ecosystem components according to independent sampling designs
Integrated ecosystem monitoring survey	Data collection on more than one ecosystem component, explicitly considering the processes that link the sampled components
Integrated ecosystem monitoring programme	The combination of multi-platform, multi-scale integrated data collection, evaluation of ecosystem status and the monitoring programme

aims and offer the potential for dramatically increasing operational sampling efficiency and analytical efficiency (i.e. power to detect change). Towards this end, we offer principal considerations of how gaps in our ecosystem knowledge can be filled.

Learning from the past

Current monitoring programmes designed to assess change over time put a premium on consistency in an attempt to minimize the effect of factors known to affect the monitored indicator or index in question. We use the example of the iconic cod *Gadus morhua* (Linnaeus, 1758) to illustrate the costs and risks of current monitoring principles. The species and the stock in the North Sea in particular is one of the most intensely studied, both from a population dynamics and an ecological perspective. If current monitoring data in conjunction with the sizeable knowledge of cod behaviour and ecology is insufficient to quantify the effects of ecosystem change on this species, there is little hope for less well-studied species or processes.

OPERATIONAL INFLEXIBILITY

Daily bathymetric movements of cod (Beamish 1966), their use of visual cues to avoid trawls and their potential effects on survey catches (Konstantinov 1964) have long been known to scientists and fishermen. To avoid adding this variability to index-based surveys and with it reducing the power to detect change, catches are standardized to daylight hour. The inevitable consequence, however, is that International Bottom Trawl Survey (IBTS) cod abundance data are comparable only to data collected under the same environmental conditions. Events not under our control such as annual differences in light conditions not due to daily periodicity such as turbidity or cloud cover would equally affect the cod index. The latter effects are likely to be less significant at night, but we are unable to evaluate performance improvements in the absence of night-time collections. Furthermore, Shephard *et al.* (2015) provide examples of conflicting standardizations hindering potentially efficient multiple indicator data collection. Solving the problem by abandoning historic time series and developing new multiple indicator time series with more compatible standardization merely treats the symptom (a specific problematic constraint, i.e. where or when we sample) but not the cause (being methodologically constrained and operationally inflexible).

UNCONTROLLED CHANGES

Where standardization is not conflicting, potentially uncontrollable behavioural or ecosystem changes may still introduce biases into the temporal perspective. Many cod stocks undertake seasonal migrations (Metcalfe 2006; Neat *et al.* 2014). It therefore matters where and when monitoring takes place. Indexed-based fisheries monitoring samples at the same place at the same time of the year to minimize the effect of migration on catchability, ensuring abundance estimates between years are comparable. A number of authors have, however, discovered relative changes in the spatial distribution of cod and other species, either because the composition of substocks has changed or because parts of the population have differentiated in their migration behaviour and/or its timing in response to environmental or ecosystem cues (Brander 1994; Hedger *et al.* 2004; Blanchard *et al.* 2005; Perry *et al.* 2005).

Tagging studies (Robichaud & Rose 2004; Righton *et al.* 2007; Tamdrari *et al.* 2012) describe changes in the migration behaviour of cod and can even statistically link these to changes in environmental conditions. The considerable investment in investigating (tagging) and monitoring (IBTS) cod has resulted in short-term qualitative estimates of changes in the distribution of cod populations and long-term quantitative abundance estimates of the cod population that are biased. It is not possible to combine these data sources to derive an unbiased estimate of the trend in cod distribution, because sampling was conducted independently at incomparable spatio-temporal scales.

CORRELATIVE RELATIONSHIPS

Many documented fish stock-recruitment trends are based on correlations between abundance estimates, environmental variables (temperature and NAO index are common examples) and anthropogenic pressures such as fishing effort. However, the majority of the correlations, though statistically significant, may be in fact coincidental (Myers 1998). Larking (cited in Dickson, Pope & Holden 1974) described the situation as: ‘virtually any set of stock-recruit data is sufficiently variable to inspire untestable hypotheses about the effects of trends in environments, especially with the wealth of meteorological and oceanographic data that can be mined for real and fortuitous correlations’. *Vice versa*, some undeniably causal relationships between abundance and exploitation may be masked in correlations. For example, the asymmetry in response rate of cod abundance to increases and decreases of fishing pressure is masked by response lags either through mixing spatially or recruitment dynamics temporally. In the absence of process information provided by integrated monitoring, correlation analyses will undesirably tend to produce more false positives and false negatives than statistical evaluation would suggest.

For ecosystem monitoring to service the needs of the ecosystem approach and ultimately ecosystem-based management, it must causally relate the effects of anthropogenic pressures and environmental variability on the ecosystem and the services it provides to society, while taking account of the complexities in those relationships. Future monitoring must therefore be able to provide the means to test causality (Hjermann *et al.* 2007) and not rely on correlative analyses.

AGGREGATION

Engelhard, Righton & Pinnegar (2014) relate the change in the distribution of the cod stock in the North Sea to a combination of fishing effort and temperature increases. They conclude that the population had shifted northwards in response to climate change and eastwards in response to fishing pressure. To reach these conclusions, cod abundance, temperature and fishing effort were all averaged at the annual North Sea level and smoothed. Both processes remove variance at the finer spatial and temporal scales. Similarly, Tett *et al.* (2013) intentionally smoothed data, explicitly ignoring the variation at the local scale, to relate ecosystem states on the regional scale. Aside from confusing correlations with causality described earlier, averaging or smoothing data further complicates the identification of causal effects and increases the chance of random correlations because it reduces the true variability, potentially to monotonic trends that, by their very nature are statistically correlated. Similarly, aggregation leads to a loss of contrast in both dependent and independent variables. This reduces the ability to detect critical effects of change on ecosystem components and potentially underestimates the magnitude of the impacts.

It is less a criticism of the methods themselves designed to demonstrate changes in time. Rather, we are concerned that the use of such unrelated data collections is inefficient and unsuitable for the application of the ecosystem forcing researchers to address policy questions with inappropriate methods. The inability to investigate ecosystem change at appropriate spatio-temporal scales will underestimate uncertainties in future predictions (Planque, Bellier & Loots 2011) and limit the capacity of managers to respond appropriately to change.

The integrated alternative

The multitude of ecosystem interactions means that it is highly unlikely that indicators are independent of each other. Consequently, information integrated across indicators can be equally, if not more, informative on ecosystem status than the trends in individual indicators. Maximum precision in the measurement of individual indicators does not equate to maximum precision in overall ecosystem status estimates. Maximizing precision means determining the relationship between indicators, for which data must be collected at the appropriate spatial and temporal scales. For existing time series, this is often difficult to achieve because they are standardized by different criteria based on their original purposes (for some examples see Hislop 1996; Reiss *et al.* 2009; Kröncke *et al.* 2011). Standardization is an artefact of an intense focus on precision of a specific aspect of the ecosystem. It has the undesirable consequence that it reduces gains in ecosystem understanding where incompatible standardizations do not allow us to investigate the relationships between different ecosystem components. This means we must replace the standardization paradigm with something else (De Jonge, Elliott & Brauer 2006; Degnbol *et al.* 2006).

In practical terms, it does not mean no standardization, but a controlled flexibility where it is possible to isolate the effects of changes to methods from changes in the ecosystem. This is particularly important where the practices hinder the ability to relate different data types on the appropriate spatial scale.

In the IBTS cod abundance example, this means not restricting sampling to the day time only. In sampling around the clock, we are able to distinguish the effects of changing cod abundance from the effects of fishing at different times of day. To identify the diurnal effect in monitoring data requires only that it is orthogonal to any changes in the ecosystem. For example, sampling half day and half night stations in each year will make them orthogonal and the unique contribution of each effect can be estimated. In contrast, sampling 1 year (or habitat) only at night and 1 year (or habitat) only during the day means that the effects (time of day and year) cannot be uniquely identified. To maintain consistency with historic time series, orthogonality can be maintained randomly over time, but purposefully ensuring it allows for more rapid estimation of the effect. The approach can also be applied to effects that are currently not monitored such as ecosystem interactions known to affect distribution of cod including (i) food availability: temperature (Dalpadado *et al.* 2009), (ii) predation: habitat (Gotceitas, Fraser & Brown 1995) and (iii) intraspecific competition (Swain & Wade 1993) which are difficult to account for in index-based survey designs. To identify drivers of faunal patterns, first different ecosystem components need to be sampled on comparable spatio-temporal scales to ensure they are indicative of the same processes. Secondly, the full range and combinations of different states that exist in the ecosystem need to be monitored in a structured way so that the relationship between states can be determined over the full range of conditions. This is the essence of what is understood by 'integrated ecosystem monitoring' in this paper.

By calling for monitoring to change, we are not in any way suggesting that monitoring in the past has been inappropriate given the questions addressed. In index-based monitoring, collecting the covariates (i.e. information on the factors that affect the indicator in question) to maintain the same level of precision increases the sampling burden. So it is not without good reason that index-based monitoring has established itself at a time when the state of only one indicator was thought to be of interest. However, with the advent of the ecosystem approach and the MSFD, it is no longer adequate to merely report on the precise status of a few ecosystem components or indicators. Instead, we need to understand how different ecosystem components interact with each other and their environment and how they change in response to diverse pressures. This requires sampling of a wide range of ecosystem metrics in their own right and not just as covariates to explain variation in an indicator. The current understanding of marine ecosystems, and in particular knowledge of how different ecosystem components are integrated through ecosystem processes operating at different spatio-temporal scales, helps to meet contemporary monitoring objectives efficiently.

Making the ecosystem approach practical

Understanding the relationship between different indicators based on the ecosystem processes that link them is key to integrated monitoring but what does this mean in practice? In its most simplistic form, it means to measure everything, everywhere all the time. This raises real concerns over the feasibility of integrated monitoring due to the magnitude and complexity of the task. But our knowledge about ecosystem function in general (for a summary see Levinton 2001), and in particular the factors that drive the structure of some ecosystems (including those we might wish to monitor), is notable (for example, Barents Sea: Jakobsen & Ozhigin 2011). This knowledge may currently be inadequate to quantify the relationships between different ecosystem components simply because of the way evidence has been collected. Nevertheless, it is sufficient to describe key linkages between ecosystem components in qualitative terms, it identifies major energy pathways in a system, and it describes the causes of variability in ecosystem dynamics. Coupling this understanding with advances in ecosystem modelling provides a clear path towards designing meaningful monitoring for the ecosystem approach.

THE STATE VIEW – SPACE AND TIME (MONITORING WHERE AND WHEN)

The ecosystem state is the result of a large number of cumulative interactions amongst the physical environment, the biota and humans. Jakobsen & Ozhigin (2011) show how much pertinent information on environmental and biological states is available for the Barents Sea. A joint Norwegian and Russian survey in the area since 2004 (see ICES 2012a for a description) has confirmed much of the previous ecosystem understanding of the Barents Sea described by Jakobsen & Ozhigin (2011) based on a compilation of data. A description of a sequence of states of dominant ecosystem components is sufficient to develop monitoring and modelling approaches.

Capelin and herring are the dominant pelagic fish in the Barents Sea ecosystem and use the Norwegian coastal waters as a nursery (Johannesen *et al.* 2012). Both species move northwards with ontogeny but despite a sizeable overlap in the distributions, they maintain a persistent east–west gradient in the proportions of each species. After 1–2 years in the Barents Sea, herring return to the Norwegian Sea while adult capelin remain, annually migrating north to the summer feeding grounds as the ice melts and returning south to overwinter in the warm water ingress from the Atlantic. Barents Sea cod is one of the dominant consumers of the production through capelin and when the capelin stock increases, the cod stock tends to follow. In the eastern Barents Sea, under more typically arctic conditions, polar cod tend to dominate numerically and are the main consumers of zooplankton production (Wienerroither *et al.* 2013), leading to a more trophically compressed system. Looking at the life-history characteristics of these fish and the locations they inhabit (Wienerroither *et al.* 2011, 2013), it is apparent that the environmental characteristics of the water masses (Olsen *et al.* 2010; Johannesen *et al.* 2012) define the spatial and temporal patterns of species distributions. The correlation between demersal species distributions and the water masses suggests that the pelagic processes are the primary determinants structuring both, demersal and pelagic communities in the Barents Sea. It should therefore be possible to design ecosystem monitoring based on the spatio-temporal dynamics of these easily identifiable water masses irrespective of whether the interest is in demersal or pelagic species.

Bogstad, Hauge & Ulltang (1997) developed a spatio-temporal raster for modelling pelagic ecosystem interactions in the Barents Sea. The raster was based on the life-history stages and migration patterns of ecologically dominant fish species and the environmental and ecological interactions that determine their dynamics. The raster would equally well describe the distribution of many of the less dominant species, including those examined by Wienerroither *et al.* (2011, 2013). This is because the less dominant processes operating at smaller spatial scales are invariably correlated or nested within dominant hydrological or ecological processes operating at large spatial scales.

Defining areas or times in which environmental conditions and biological states are comparable (i.e. temporal or spatial strata) greatly simplifies the monitoring approach because we do not have to sample everywhere all the time. Using a common stratification is efficient because it is irrespective of the ecosystem descriptors or indicators to be assessed and will provide time series of standardized ecosystem information in line with the MSFD reporting requirements. At the same time, commonality in the balance of states infers commonality of processes within strata, allowing ecosystem processes to be quantified even when they are poorly understood at present. Given the common scales of variability shared by ecosystem processes, Stommel's (1963) work on monitoring efficiency can be extended from the single to the multi-metric case. This reduces the complexity of the monitoring design and the need to *a priori* define the metrics in relation to the design with only a small loss of precision. The generalization is necessary, because monitoring, unlike an experiment has to both provide information on changes over time of specific metrics as well as be adaptable to address future questions.

THE PROCESS VIEW – DYNAMICS AND VARIABILITY (MONITORING WHY AND HOW)

Ecosystem processes, comprising complex material cycles and flows of energy, link the biotic and abiotic elements of ecosystems. The state of the ecosystem is the integral of these processes over time. Any particular state variable in the system is characterized by the balance between the upstream and downstream process rates (e.g. recruitment and mortality for fish populations). Changes in this balance will result in a change of state but state alone is not indicative of a process. However, by monitoring states over time, it is possible to gain an ecosystem process view and to understand the important processes affecting ecosystem state. This principle and the importance of linking the data through modelling have been demonstrated by ecosystem research in the Barents Sea and the Baltic Sea over the last 20 years.

Hjermann *et al.* (2007) examined the ecosystem processes that link capelin, herring and cod. They developed a simple statistical multi-species model, based on the most recent and more detailed time-series data, to predict cod recruitment. They then extended their model back in time, substituting abundance of species about which no information exists with a function of the abundance of species for which long time-series data are available. The innovative approach considered the value of the different information sources so that recent highly temporally resolved samples were more influential in some structural parts of their model, whereas the historical lower resolution data provided the long-term view. In this way, time series can be maintained throughout the transition from species-specific to ecosystem monitoring. This maintains existing time series of indicators where necessary, including fish, eutrophication etc., while ensuring that monitoring remains flexible to meet future demands of policy, and responsive to advances in ecosystem understanding.

Loeng & Drinkwater (2007) qualitatively summarized the links between environment and organisms and the trophic dynamics of the Barents Sea and Norwegian Sea ecosystem in terms of predation and competition amongst dominant fish species. Rather than regarding the ecosystem as static or in equilibrium, they took the variation in ecosystem dynamics and its known or hypothesized causes into consideration. Their ecosystem overview is notable for the realization that it is the variation in processes that helps their significance to be understood. The understanding is more qualitative than quantitative because it relies on outputs from different models dealing with different units and feedback loops that do not function across model boundaries (Travers *et al.* 2007). Working in the Baltic Sea, Köster *et al.* (2001) made significant strides towards a more quantitative integrated ecosystem understanding. They showed how both ecological and environmental processes combine to create recruitment variation in Baltic Sea cod. Importantly, they demonstrated that the effects are not additive so that the same pressure can have opposing effects under contrasting conditions. Building on this advanced ecosystem understanding, Möllmann *et al.* (2008) developed a conceptual model for trophic pathways in the Baltic Sea, considering anthropogenic, ecological and environmental effects and their interactions simultaneously.

The need for conceptual models that help organize complex information on system components and interactions is obvious. Collecting information on ecosystem components independently of the processes that connect them can only work if by chance the spatio-temporal scales of these collections are commensurate with the scales at which key ecosystem processes operate. Experience with the Barents Sea and the Baltic Sea ecosystem has been useful to the management of key ecosystem components because it is based on sound conceptual models that articulate key system components and their interactions at scales relevant to management. However, attempts to apply these principles to modelling of the North Sea ecosystem have not progressed beyond the lower trophic levels (Moll & Radach 2003; Gibson, Atkinson & Gordon 2006; Travers *et al.* 2007). Greater ecosystem complexity is frequently cited for the slow progress (Anderson 2005), but this appears to be only part of the problem. North Sea modelling work has been based on highly standardized time series which are spatially and temporally incompatible and hence require aggregation. Consequently, there usually is insufficient contrast in the aggregated data to appropriately quantify key processes. Inevitably, investigation of high-profile topics such as regime shifts or the effects of temperature and fishing activity on cod (for example Kempf, Floeter & Temming 2009; Engelhard, Righton & Pinnegar 2014) and the wider ecosystem (Kröncke 2011; Tett *et al.* 2013) have reverted to large-scale correlative analyses that cannot address the important question as to why observed ecosystem changes occurred or what can be done to minimize the impacts at scales relevant to management. A consideration of the underlying processes can fulfil those aspirations (Roff & Evans 2002).

THE PROCESS VIEW – PRIORITIZATION OF PROCESSES AND SAMPLES (MONITORING WHAT AND WHAT WITH)

Knowledge of the spatial distribution of multiple states can be used to infer commonality of processes so that monitoring becomes efficient. Ecosystem modelling informs on how to assess states to maximize gains in understanding ecosystem processes. However, the conclusions still leave us sampling everything. How do we decide what to measure and how?

Not all ecosystem processes are equally important. By focusing on processes that quantitatively dominate the energy flow through the system, we can capture the vital signs of the ecosystem. For example, primary productivity (of either photic or chemical origins) forms the basis of all ecosystems. Understanding how much is being produced and how this production is influenced by the timing and magnitude of the ice melt in the Barents Sea (Slagstad & Wassmann 1996) or the extent of stratification and the amount of suspended particulates in the North Sea (Tett *et al.* 1993) is essential to understanding those specific ecosystems. The relative rates of dissipation of this production through the ecosystem are also highly informative regarding the importance of subsequent ecosystem processes. When most of the primary production is consumed by the pelagic components, benthic components perform a smaller role in structuring the ecosystem. In such instances, it is unlikely that the role of benthic systems can be understood without understanding pelagic systems. In contrast, systems that are top-down-structured will need to more heavily rely on estimating the variation in the consumption of top predators and the pathways by which the primary production reaches them.

The links between some processes are more obvious than others. For example, the link between phytoplankton productivity and zooplankton productivity is stronger and more obvious than the link to demersal productivity (Snelgrove 1999; Renaud *et al.* 2008). Understanding the process sequence and how processes interact helps to strategically identify a subset of processes that inform on, or contribute to, the estimation of those processes that are not intensely monitored. A balance of monitored processes across the system maximizes the cumulative information samples provide and reduces the risk of failing to detect changes in the system. Model interpretations can and should be included in the estimation of states to further reducing monitoring requirements in cases where processes or sequences of processes can be modelled reliably and quantitatively. Complete replacement of monitoring with modelling, however, is not advisable, as modelled estimates are only ever as good as the models themselves. Unusual and unlikely events invariably lead to new and improved ecosystem understanding. Deep Sea vents, for example, would not have been discovered without sampling, since then current models predicted that conventional energy inputs were insufficient to maintain benthic communities (Roff, Taylor & Laughren 2003) and those models would not have changed without the discovery.

Every sampling methodology has a unique combination of costs, operational characteristics and contributions to the estimation of ecosystem processes. To determine the most effective use of monitoring resources, we must address the following questions: What processes does the sample inform on? How important is this process in the context of the ecosystem under consideration? How much does it cost? How do we balance or trade-off the collection of one type of information over another? What cheaper, less restrictive or more informative alternatives exist for assessing a particular process? For example, in benthic monitoring we might consider whether sediment particle size data are necessary to our ecosystem understanding beyond the classification of predominant habitat types. At the subregional level, particle size distribution appears to be spatially more heterogeneous and temporally more stable than the associated community structures (Barents Sea: Künitzer *et al.* 1992; Carroll *et al.* 2008; Renaud *et al.* 2008; North Sea: Clark & Frid 2001; Kröncke *et al.* 2011). McBreen *et al.* (2008) consider sediment particle size as insufficient to describe benthic communities in the Irish Sea and suggest that sediment types are correlated with other environmental variables known to structure benthic communities. Are those environmental variables, including tidal currents and wind-driven disturbance more ecologically relevant, easier and more cost-effective (i.e. not requiring a vessel to collect samples) to measure? Assuming particle size information is still required, should we use Hamon grabs, NIOZ corers or dredges to collect this information? Corers maintain the structure of the sample so can also inform on geochemical processes in the sediments. The Hamon grab can be deployed in coarser sediments and has a fixed sample volume which means it provides more consistent additional information on infaunal communities. Dredges are the quickest to deploy and are most robust, but provide only an unquantified portion of the sediment and infauna. Evaluating trade-offs between indicator requirements by assessing what is necessary, as opposed to ideal, but maximizing the benefit for other indicators makes monitoring efficient.

Much of the existing monitoring data does not readily lend itself to answering these questions quantitatively, but we do have enough conceptual knowledge to rank the major ecosystem processes. Adequate sampling methodologies to quantify these processes are also in place. Initially, the ranking of importance is unlikely to be uniform across all disciplines. However, a recurrent quantitative evaluation of the monitoring data as part of the ecosystem assessment process will provide the necessary information to achieve future consensus regarding the importance of processes. Effects of optimizing the monitoring can be isolated from future changes in the ecosystem because the monitoring programme is flexible. In consequence, impacts of poorly informed decisions are reduced compared to monitoring programmes that rely entirely on consistency for assessing change.

COORDINATED VS. INTEGRATED MONITORING: LEARNING FROM EXPERIENCE IN THE CELTIC SEA

We provide an example in relation to a biodiversity indicator to illustrate and contrast coordinated vs. integrated ecosystem monitoring. Phytoplankton abundance and communities vary at fine spatio-temporal scales (fronts and seasonal, respectively) and sample analysis to species level is costly. Standard vertical ring nets provide information consistent with the taxonomic and spatial resolution of historic information. Automated techniques such as flow cytometry (size and pigment composition of individual cells) on a continuous pump system provide a higher spatial resolution and reduce costs but the information is of low taxonomic resolution. This may be sufficient for the purpose of evaluating food chain effects but it does not support biodiversity descriptors that rely on species-level information. Moreover, pump systems are restricted to intake at a certain water depth and plankton vertical distribution is strongly affected by turbulence which varies with weather conditions and cannot be standardized for. The coordinated approach then simply applies the two sampling approaches in parallel and addresses biodiversity and food chains independently, accepting the variance in each and then relates the variable indicators at an aggregate scale. Conversely, the integrated approach accounts for both in a common model of phytoplankton dynamics, using weather conditions as a proxy covariate for turbulence. The value of a pumped sample increases with an increase in turbulence from a biodiversity perspective, suggesting the collection of costly vertical ring net samples can be decreased at times or in areas of high turbulence. Additionally, it can be shown that changes in the plankton composition at a specific depth, which are not due to changes in turbulence, indicate spatial changes in species composition. Using the low cost continuous method to maximize information content in the ring net samples through real-time evaluation, even when the method has little direct benefit in calm conditions, illustrates the benefit of integration. Deploying the ring net when continuous sampling indicates a change in community differentiates integration from co-collection and post-sampling data collation in that it uses the ecological and sampling processes to maximize efficiency.

In integrated monitoring programmes, such information can be linked to other information including satellite chlorophyll data and turbulence estimates from hydrodynamic models, combining the higher spatial and temporal resolution of the satellite data with the higher specificity of the survey data to develop a holistic view of plankton dynamics. A consistent time series of indicators can be developed irrespective of weather conditions. Continuous Plankton Recorder (CPR) data that lacks the spatial resolution outside specific routes or fixed platform collections at key points in the system could enhance the temporal and spatial accuracy of the species information without biasing the results when spatial hydrographic changes affect these samples.

Existing time series can be used to improve the model of plankton dynamics by tuning hind casts in observing system simulation experiments style exercises while simultaneously ensuring consistency with historic data and assessments. The same models can be used to investigate different management options where effects such as nutrient concentrations are known to affect status. The outputs from such evaluations can indicate the spatial and temporal resolutions at which changes are orthogonal, that is the same changes occur in space and time. This informs on the value of specific samples in relation to their costs and allows for improved efficiency of future sampling. Where biases exist, for example observed phytoplankton abundance is consistently lower than expected from model outputs, it is possible to test formally for the most likely cause. Are variations in turbidity in coastal waters a suitable alternative to the current model (difference between satellite and *in situ* chlorophyll) or is grazing by herbivorous zooplankton a more likely explanation for the low production? If an improved model is warranted, the relative importance of samples is likely to change and this can be incorporated seamlessly into decisions on future monitoring with minimal risk to the consistency of time series.

FROM MONITORING TO A MONITORING PROGRAMME

Ecosystem monitoring improves the ability to detect change by maximizing contrast, prioritizing the major ecosystem pathways and permiting us to determine the variability in ecosystem processes. Our understanding of ecosystem processes is critical to ecosystem monitoring because it forms the basis of ecosystem models that can be used to evaluate future scenarios of change, be that ecosystem- or impact-based. A meaningful and efficient ecosystem monitoring programme is therefore much more than the collection of data (Arkema, Abramson & Dewsbury 2006; Plasman 2008). It is a common framework for all activities associated with ensuring responsible use of natural resources. Where data are formally integrated, changes to the programme affect all other aspects of monitoring. If CPR methodology was to change in our phytoplankton example above, or lose cooperation with one of their vessels of opportunity, the impacts would cascade through the monitoring system and change the way data would be collected under the new circumstances. Such changes have to be evaluated carefully in the wider ecosystem monitoring context. The trade-off of integration and greater efficiency is the loss of autonomy of different monitoring efforts, and this requires better communication and greater flexibility amongst monitoring actors than is currently the case. For some governments, greater top-down influence on monitoring activities, changes in monitoring institutions and/or greater cooperation between centres in polycentric management systems will have to be found to implement a coherent integrated monitoring programme.

Closing arguments and reality

We have presented a theoretically based monitoring design framework, and shown how it can be implemented practically and efficiently. A successful integrated ecosystem monitoring programme is fundamental for meeting needs of a range of stakeholders including ecosystem assessment and policy makers. An integrated monitoring programme starts with the ecosystem as a whole and defines increasingly homogenous subdivisions in that ecosystem in space and time. The specific consideration of processes that discriminate between subdivisions enables the monitoring programme to quantify and compare the interactions between processes and the relative importance of each in driving the variation in states. The established relationships reduce the unexplained variability in future monitoring, suggesting that monitoring effort can be reduced without loss of precision.

Various monitoring programmes exist within the EU, many of which are not under the direct control of either the EU or the national authorities responsible for marine ecosystem monitoring (polycentric governance). This and the desire to maintain time series has resulted in a largely agglomerative or bottom-up approach, advocating supplementing or complementing current monitoring programmes (coordination rather than integration) where there are gaps based on the MSFD reporting requirements (as proposed by WGISUR (ICES 2012b) and the Joint Monitoring Program (Shephard *et al.* 2015)). Agglomerative evaluation is complementary to our integrative approach, in the sense that the bottom–up approach defines the scope of the possible, while the top–down approach offers guidance on how to make appropriate choices within that scope to maximize the utility of the monitoring programme. Relying on one without the other is unlikely to succeed given the current situation in Europe.

We stress the importance of linking different data types in order to determine not only ecosystem change, but also its causes. The need for monitoring data capable of performing this function is also recognized in the coordinated approach. Using the example of North Sea benthos data, Shephard *et al.* (2015), demonstrate that it is possible to combine independently collected information into an ecosystem view citing Reiss *et al.* (2009) as having reconciled the design differences between the North Sea benthos surveys in 1986, 2000 and the North Sea IBTS. This is true, but out of the total 1379 samples collected only 490 (35%) were used in the comparison (Reiss *et al.* 2009). Kröncke *et al.* (2011) found that of the 1349 stations sampled only 156 (11%) could be related because the monitoring programmes lacked the necessary oversight to value the importance of assessing the relationships between ecosystem components. The North Sea Benthos Survey was never designed to be used in monitoring, but it illustrates that without integration between sampling programmes, it will be more costly, if not impossible, to quantify cause–effect relationships underlying environmental change. Practically it is not possible or desirable to design surveys that integrate all ecosystem components, instead we need to design surveys to maximize the integration of all available data sources. Therefore, the choice of what, where and when to sample should be driven as much by other available data as it is by the need for specific data.

Similarly, the integrated approach needs to recognize that historic time series put future data into context. We demonstrated that there are modelling approaches that perform this function but these need to be considered when developing future monitoring as the ability to maintain time-series consistency is inversely proportional to the magnitude of the change in monitoring. We also acknowledge that not all monitoring is designed to serve only the ecosystem approach; some monitoring programmes will continue to need to focus on detecting change (e.g. contaminants in seafood). The truly integrated monitoring programme we advocate provides flexibility to accommodate these needs while also addressing changing policy requirements.

The knowledge exists to start developing integrated monitoring programmes. The potential benefits for moving towards real integration (Bricker & Ruggiero 1998; Arkema, Abramson & Dewsbury 2006; Bennett & McGinnis 2008; Parrott & Meyer 2012; Murdoch, McHale & Baron 2014) outweigh the residual risks to time series that are less suited for meeting contemporary monitoring objectives. We need to avoid developing new independent time series identified as gaps in current monitoring which would make future integration problematic. Most importantly, we must define the 'common understanding' between the different types of monitoring stakeholders (comprehensiveness and participation), improve collaboration between scientists across different disciplines (cooperation) and ensure policy commitment to integrated monitoring (long termism: bracketed terms defined by Stojanovic, Ballinger & Lalwani (2004) as the important factors for success in integrated management). Only then can we accelerate the currently small gains towards full implementation of the ecosystem approach (Pitcher *et al.* 2009) by delivering the monitoring that bridges the knowledge gaps precluding better management.

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Data accessibility

Data have not been archived because this article does not contain data.

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