International Journal of Greenhouse Gas Control 109 (2021) 103388

Contents lists available at ScienceDirect

International Journal of Greenhouse Gas Control

journal homepage: www.elsevier.com/locate/ijggc

Efficient marine environmental characterisation to support monitoring of geological CO_2 storage

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ARTICLE INFO

Keywords:
Carbon capture and storage
Marine
Environment
Monitoring
Impact assessment
Baselines
Characterisation
Oceanographic data

ABSTRACT

Carbon capture and storage is key for mitigating greenhouse gas emissions, and offshore geological formations provide vast CO_2 storage potential. Monitoring of sub-seabed CO_2 storage sites requires that anomalies signifying a loss of containment be detected, and if attributed to storage, quantified and their impact assessed. However, monitoring at or above the seabed is only useful if one can reliably differentiate abnormal signals from natural variability. Baseline acquisition is the default option for describing the natural state, however we argue that a comprehensive baseline assessment is likely expensive and time-bound, given the multi-decadal nature of CCS operations and the dynamic heterogeneity of the marine environment. We present an outline of the elements comprising an efficient marine environmental baseline to support offshore monitoring. We demonstrate that many of these elements can be derived from pre-existing and ongoing sources, not necessarily related to CCS project development. We argue that a sufficient baseline can be achieved by identifying key emergent properties of the system rather than assembling an extensive description of the physical, chemical and biological states. Further, that contemporary comparisons between impacted and non-impacted sites are likely to be as valuable as before and after comparisons. However, as these emergent properties may be nuanced between sites and seasons and comparative studies need to be validated by the careful choice of reference site, a site-specific understanding of the scales of heterogeneity will be an invaluable component of a baseline.

1. Introduction

Carbon capture and storage (CCS) is a technology to mitigate emissions from large point-source industries such as cement, iron, steel, chemical production and from power generation. Technology is also currently being developed for capturing CO_2 directly from the atmosphere. In all these cases, captured CO_2 is compressed, transported and stored permanently in suitable deep geological formations. CCS is an essential component of the global climate change mitigation portfolio (e.g. IPCC 2005; IPCC 2014; IEA, 2015), and will be required for at least 13% of total emissions reductions (e.g. approx. 94 $GtCO_2$) required to meet the 2 °C goal (IEA, 2016). Although several methods exist for estimating global geologic storage capacity (Ganjdanesh and Hosseini, 2018; Hosseini et al., 2018; Kearns et al., 2017; Ringrose and Meckel, 2019; van der Meer, 1995), one conclusion is common to all; that there is more than enough storage capacity to receive the needed volumes of CO_2. Whereas storage formations underlie both onshore and offshore areas, Ringrose and Meckel (2019) surmise that the global offshore contains "the most significant gigatonne-scale storage resource" for geological CO_2 storage. Whilst IPCC scenarios include large-scale CCS infrastructures as an essential technology to meet the Paris Agreement goal of "well below 2 °C", public support for future CCS projects is an important feature of the social licence to operate CCS technologies.

Local environmental impact assessment, via monitoring, is one of the criteria on which the public will judge CCS. Different strategies for environmental monitoring must provide enough coverage to detect and

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https://doi.org/10.1016/j.ijggc.2021.103388
Received 17 December 2020; Received in revised form 2 June 2021; Accepted 15 June 2021
Available online 9 July 2021
1750-5836/

attribute CO_2 leaks and, at the same time, foster and build public trust in the safety and integrity of a CCS project; requiring the weighing together of many factors in developing a responsible and transparent environmental monitoring program in a dialogue with relevant stakeholder groups.

In this paper we consider the process of monitoring in the marine environment and discuss an approach to acquiring a sufficient baseline understanding to efficiently underpin such monitoring. Here we use "marine environment" to refer to the upper few meters or so of seabed sediments and the overlying water column, i.e. the zone housing complex ecosystems.

1.1. Monitoring and regulatory requirements

Unlike many industries with long national traditions and diversified national regulation, CCS law and regulations stem from international cooperation, generating national regulation with clear similarities across jurisdictions. In general, law requires demonstration of storage integrity, absence of environmental impact, and accounting of emissions in the unlikely event of leakage. For emissions accounting, and because of the geological variability amongst sites, an emissions factor approach is not suitable for geological CO_2 storage, rather a measurement-based approach is required (Dixon et al., 2015). Specific permitting and related monitoring requirements are closely related to CCS site characterisation and selection, risk and project impact assessments, stakeholder, and public participation, including access to information. The IPCC Guidelines chapter on carbon dioxide capture and geological storage (IPCC, 2005) set the foundation for all global monitoring regulations and can be reduced to the following components as outlined by Dixon and Romanak (2015): 1) site characterisation and identification of potential leakage pathways, 2) assessment of risk of leakage through site characterisation and modelling of CO_2 behaviour, 3) monitoring of CO_2 behaviour during injection and subsequent updating of models, and 4) reporting of CO_2 injected and emissions from storage. Offshore, leakage is defined as CO_2 flux from beneath the seabed into the ocean (with connection between ocean and atmosphere implied). With respect to the environmental portion of these regulations, the methodology requires that a monitoring plan include measurement of background CO_2 fluxes through the seabed as well as any leakage flux that may occur. This activity therefore requires methods that can distinguish between the two types of fluxes, known as "attribution". The resultant protocol for environmental monitoring could be summarized as 1) measurement of background CO_2 concentration, 2) detection of an anomaly, 3) source attribution of that anomaly, and 4) quantification of leakage emissions if attributed to leakage.

CCS is operating in a rapidly changing socio-economic, technological, and physical environment. Adaptive management facilitated by the latest scientific knowledge on the condition and functioning of the marine environment and the management of human activities at sea will be desirable, (Platjouw and Soininen, 2019). In line with adaptive management, regulations on CCS monitoring advocate management that can adapt and incorporate new information as it becomes available. Typically, CCS monitoring plans are not fixed for the whole lifespan of the storage site, but revised to account for changes to the assessed risk of leakage, risks to the environment and human health, new scientific knowledge, and improvements in best available technology (see i.e. the European Union CCS Directive 2009/31/EC art 13 (2)). The specific monitoring requirements within a CCS project are designed in a dialogue-based process between the operator, proposing the monitoring plan, third-party stakeholders, i.e. the general public, fisheries sector etc. partaking in the impact assessment process, and the regulators.

Globally, soft law instruments, or guidelines, recognize CCS as an emissions reduction technology. The IPCC Guidelines for National Greenhouse Gas Inventories, Volume 2, Energy, Chapter 5 (IPCC, 2006, refined IPCC 2019), has inventory methods consistent with the IPCC Special Report on Carbon Dioxide Capture and Storage (IPCC, 2005). The guidelines are foremost aiming for GHG accounting and provide methodologies for estimating and reporting national anthropogenic greenhouse gas sources and sinks. The guidelines (sec. 5.7) state that "the choice of monitoring technologies will need to be made on a site-by-site basis", and as monitoring technologies are advancing rapidly "it would be good practice to keep up to date on new technologies". Dixon and Romanak (2015) state that the methodology of the IPPC Guidelines "has become the basis for all subsequent international regulation and legal guidance for CO_2 geological storage".

The United Nations Framework Convention on Climate Change (1997), the 1997 Kyoto Protocol, Article 12, has been in force since 2005. The Kyoto Protocol is legally binding upon developed countries, but still only includes non-prescriptive commitments, for example ref. Art 3 nr 1 "do not exceed their assigned amounts". The aim of the Protocol relates to GHG accounting and protection of the environment, particular for developing countries. The Kyoto Protocol offers International Emissions Trading, Joint implementation, and the Clean Development Mechanism, rewarding low-carbon projects in developing countries by the creation of carbon credits. In 2011, 'Modalities and Procedures' for CCS were agreed (Decision 10/CMP.7), stating that the monitoring plan shall *"reflect the principles and criteria of international good practice for the monitoring of geological storage sites and consider the range of technologies described in the IPPC Guidelines and other good practice guidance."* Thus, the IPCC guidelines which do not prescribe specific technologies but focus on site-specific monitoring technologies and best available technology, guide the Kyoto Protocol.

Leaving the global arena, regional cooperation has led to legally binding commitments for signatory states, as under the 1992 OSPAR Convention and the 1972 London Protocol. The 1996 monitoring protocol of the London Protocol, amended in 2006 to allow CCS, and the monitoring protocol of OSPAR, (OSPAR Guidelines for Risk Assessment and Management of Storage of CO_2 Streams in Geological Formations, Reference Number: 2007–12, OSPAR 2007) set out, from a legal perspective, mere recommendations, both using the phrase that the monitoring "may include". Read in context they encompass monitoring of sub-seabed geological formations, surrounding geological layers and geological layers above the formation, monitoring to detect migration, monitoring the seafloor and overlaying water to detect leakage, and monitoring seafloor and marine communities (benthic and water column) to detect and measure the effects of leakages on marine organisms.

Description of the "normal" or baseline is part of regional and national legally binding impact assessment and monitoring requirements. In the EU this follows from the EIA Directive 2011/92/EU as amended by 2014/52/EU (European Union, 2014). According to § 4 nr 1 and Annex I nr. 22, CCS storage sites pursuant to the CCS Directive shall be made subject to an impact assessment. Art 5, 1) requires the developer to provide, prior to any development consent, information on the (a) site and (b) the likely significant effects of the project on the environment, … and (f) any additional information specified in Annex IV relevant to the particular project. This is specified in Annex IV as a "description of the relevant aspects of the current state of the environment (baseline) and an outline of the likely evolution thereof without implementation of the project as far as natural changes from the baseline scenario can be assessed with reasonable effort on the basis of the availability of environmental information and scientific knowledge". Further, the CCS Directive Article 7(6) and Article 9(5) requires a monitoring plan submitted to and approved by the competent authority, updated every five years. According to Annex II, (1.1), the monitoring plan shall provide details of the monitoring to be deployed at the main stages of the project, "including baseline, operational and post-closure monitoring". National legislation in EU and EEA member states shall, according to EU law, fulfil these minimum requirements stemming from the EIA and CCS Directives. In accordance with international soft law instruments, the EU Directives also build on site-specific monitoring programs and the principle of best available technology.

Prior to site licensing, the assessment of potential environmental impacts is hypothetical. During a CCS operation and post-closure, the primary rationale for marine environmental monitoring, as an addition to sub-seabed monitoring, is to evaluate the impact if leakage from storage is observed or suspected, including from third party allegations of damage, should they occur. Monitoring a project could give important information to understand the risk of later projects. Secondly, quantification of sediment-water fluxes is essential to determine the return of carbon credits as part of emissions trading schemes (UNFCCC, 2010; Dixon et al., 2009). Monitoring in the marine environment can also play a role in leakage detection and containment assurance per se, as it can be far more sensitive to very small fluxes or anomalies compared with geological monitoring of the storage reservoir and overburden. A third rationale for environmental monitoring may be public assurance, to underpin societal acceptance of CCS as safe and well regulated. Hence monitoring in the marine environment can have relevance to each stage of the monitoring process.

1.2. Challenges bespoke to the marine environment

Different stages of marine environmental monitoring require an assessment of physical and biogeochemical parameters to establish if leakage is occurring, assess the footprint of environmentally impactful changes and quantify any leakage; and of key ecosystem parameters to assess if impact is significant and consistent with the observed biogeochemical changes. In common with terrestrial environments, marine ecosystems are naturally heterogeneous and continually changing. However, the marine system is less accessible and consequently not as well characterized as its terrestrial counterpart, with far more reliance on models to provide spatially and temporally complete data sets. Access to sites perhaps several tens of kilometers offshore and under several tens of meters of water can be expensive, requiring either ship-based operations, or more cost-effective long-range remotely operated autonomous underwater vehicles (Wynn et al., 2014), or semi-permanently deployed landers. Technological solutions are required to ensure that data transition from remotely deployed sensors is reliable and that power and sensor stability do not limit deployment time.

Basin scale fluid fluxes, regional and local processes as well as tidal, diurnal, weather-scale, seasonal and inter-annual drivers all impact the marine system, creating a heterogeneous and dynamic environment characterized by intermittent stratification, frontal features, turbulent transport and mixing, and high mobility of flora and fauna (Cetinic et al., 2015; Barry and Dayton, 1991). Marine processes occur on temporal and spatial scales ranging from climatological and basin via shelf and regional processes, to turbulent microstructures with subsequent dissipation, over seven to eight orders of magnitude. There is no distinct separation of scales; rather they are continuous, so no natural “cut-off” is available. Decadal scale oscillations such as the North Atlantic Oscillation (Hurrell, 1995) or El Niño Southern Oscillation (McPhaden et al., 2006), ongoing modification due to climate warming and, especially in coastal-shelf zones, anthropogenic modification of nutrient inputs via open boundaries, rivers or atmosphere, create significant temporal heterogeneity, potentially leading to regime shifts (Beaugrand, 2004; Clargo et al., 2015; Holt et al., 2018).

CO_2 is a natural and ubiquitous component of the marine system, modulated by many biological and physical processes as well as drawdown from the atmosphere resulting in long term ocean acidification (Caldeira and Wickett, 2003; Raven et al., 2005). Concentrations of CO_2 have significant spatio-temporal heterogeneity expressed at many different scales (Artioli et al., 2014; Friedrich et al., 2012; Hauri et al., 2013). Discriminating between natural variability in CO_2 producing and consuming processes and any changes to the environment specifically due to leakage from CCS, as well as predicting potential footprint, risks and impacts associated with leaks, requires a far greater understanding of the environment than is needed for other industrial activities. CO_2, more properly dissolved inorganic carbon (DIC), equilibrates in sea water as carbonic acid, bicarbonate, and carbonate ions, with speciation controlled by temperature, pressure, and total seawater alkalinity (Zeebe and Wolf-Gladrow, 2001). As such it is difficult to directly measure operationally, with either acidity (pH) or the partial pressure of CO_2 (pCO_2) used as proxies (Dickson, 2011). Temperature varies seasonally and spatially (for example with water depth and latitude); alkalinity scales with salinity and can be impacted by river plumes arising from different terrestrial geologies (Cossarini et al., 2015; Raymond and Cole, 2003). Regional current systems, depending on their origin (for example coastal or oceanic) may differ in temperature and alkalinity. Biological processes, primarily photosynthesis (sunlit ocean) and respiration, and to a lesser extent calcification and carbonate dissolution are responsible for large DIC fluxes with seasonal and spatial variability (Krumins et al., 2013; Snelgrove et al., 2018).

An additional level of complexity is imparted by seabed heterogeneity. Gradients and discrete changes in sediment composition may be associated with changes in calcium carbonate and/or organic matter content, which in turn will affect alkalinity and DIC fluxes at the same scales as the biological heterogeneity. Changes in sediment composition will also modify the character of microbial and infaunal assemblages, and consequently their processes (e.g. bioturbation and nitrification), that also modify respiration and DIC fluxes locally. Heterogeneity associated with harder ground will increase the presence of epifaunal species and the potential for biogenic reef structures that may alter alkalinity via calcification, and alter DIC through respiration. Finally, changes in geomorphology, topography and the presence of substantial biogenic structures will modify the mixing of water, above the sea bed by influencing bed friction, and possibly also within the sediments themselves (Alendal et al., 2005), which further complicates the spatio-temporal heterogeneity of CO_2 and associated chemical derivatives.

Identifying and quantifying this substratum heterogeneity (morphology, composition, sediment dynamics) is decidedly more challenging in the marine environment than in terrestrial settings. The absorption of electromagnetic radiation in water (particularly visible light) restricts photographic observations to close-range operations. Most seabed mapping activities are based on acoustic techniques (echosounders), but so far <20% of the world’s ocean floor has been mapped to a fit for purpose resolution (Wolfl et al., 2019).

These same challenges, in addition to ecological complexity, further hamper assessment of impacts on marine ecosystems. The structure and spatial pattern (patchiness) of biological communities are driven by the physical environment, biological interactions (competition, predation etc.) and ambient disturbances. Boundaries between communities can be sharp, or more gradational, resulting in either patch mosaics of communities, or the development of ‘ecotones’ between distinct assemblages. Similar to the seafloor heterogeneity discussed above, it is not possible to estimate the spatial scale of these patterns without any survey information; models are not yet able to predict such features.

For most seafloor ecosystems, excess CO_2 as result of a CCS leakage will act as a biogeochemical disturbance, influencing the ecosystem structure and complexity (Meadows et al., 2015). The impact will depend on the intensity, duration, and extent of leakage, and occurrence of other stressors in the area (Lessin et al., 2016). How CO_2 is transported, and how rapidly the concentration reduces to normal with distance from the leak site, will determine how large an area will experience a temporal acidification. This will be governed by the anisotropic tidal current, the general flow field in the area, and local turbulence and mixing regime (Ali et al., 2016; Blackford et al., 2020; Enstad et al., 2008).

The complex relationships between species, functional and trophic groups within communities means that the effects of CO_2 leakage cannot be predicted from the reaction of a few individual species and may be different for different ecosystems (Carroll et al., 2014). More complex communities are typically more resilient, at least in terms of ecosystem function: different ecosystem functions are shared between Multiple species, safeguarding the ecosystem if affected by sudden loss of individual species or groups (Gladstone-Gallagher et al., 2019). The presence of potential source populations in the area which can support recolonisation will also influence impact and resilience. Some species or communities, however, are more sensitive and vulnerable to impacts than others, because of their morphology or life history traits (e.g. calcite or aragonite skeleton, slow growth, low fecundity). Their decrease or disappearance will impact the community composition. Such species may be identified for protection (e.g. listed in the Annex I of the EU Habitats Directive, or indicator species for Vulnerable Marine Ecosystems).

A further complication is that leakage through marine sediments may also expel thermogenic or biogenic gas accumulations or lead to release of other potentially toxic substances into the benthos, due to mineral dissolution in the subsurface; metals can be released due to desorption and pH decrease (Ciceri et al., 1992; Kirsch et al., 2014; Sadiq et al., 2003; Wunsch et al., 2014). Advection of precursor fluids, with a different geochemical composition to seawater (e.g. anoxic waters, high salinity) might also have an adverse effect on the ecosystem. Monitoring and impact assessment may need to consider these alternative chemical species and associated impact mechanisms. Other stressors also operate on the system, in particular areas of the seafloor are at risk from transient anoxia (Hennekam et al., 2020) and bottom trawling (Jennings et al., 2012), both of which may severely disrupt benthic communities.

In terms of obtaining an adequate characterisation of the non-impacted state to facilitate attribution of anomalies or monitoring for impacts on ecosystems, the specific challenges in the marine environment are associated with the characterisation of community spatial pattern (as discussed above) and composition. Again, without knowledge of species richness and distribution in the area, it can be difficult to determine the optimal number of samples or survey tracks needed to enable statistically robust change detection. With the increasing availability of autonomous vehicles, it becomes less cost-inhibitive to obtain more samples or observations than strictly necessary, which will allow the calculation of the sampling effort needed for, robust change detection (Benoist et al., 2019). How much of that change is the result of natural processes (tidal, diurnal, annual, inter-annual), is still one of the major unknowns in the marine environment, particularly in seafloor communities.

1.3. *Expense and durability of baselines*

Offshore storage thus requires some strategy for "at sea" monitoring, including deep seismic surveys aimed at reservoir, caprock and overburden, and potentially, surveys of shallow sediments and water column for characterization and risk assessment (before injection) and/or for detection, attribution and quantification of leakage, if it were to occur, as well as environmental impact assessment in the event of leakage. Assessing impact, detecting anomalies and quantifying fluxes all require a detailed knowledge of the reference conditions, the "normal"; where the "normal" is a continually cyclic and evolving state for marine ecosystems, CO_2 concentrations and benthic-pelagic fluxes.

Such an understanding is often conceived as a baseline (Beaubien et al., 2015) wherein the pre-operational /pre-injection state of the system is measured for future comparison should leakage be suspected. Marine systems are complex and inherently spatially and temporally variable over multiple scales and supporting the various components of monitoring involves a multi-variate understanding including physical, chemical, and ecological components. Thus, it has been argued that a detailed and expensive multi-variate baseline characterisation, entailing seasonal and inter-annual discrimination is required. However, constructing a baseline that comprehensively documents all necessary system components (physics-chemistry-ecosystem) even over a short period could be prohibitively expensive. For example, estimated costs for marine protected area monitoring are measurable in units of £100, 000 s (JNCC, 2019). The challenge therefore is to assess if we can minimise the need for an expensive and comprehensive spatio-temporal observational baseline, but instead, via the use of process or emergent property techniques coupled with an understanding of the scales of heterogeneity, recommend a methodology by which an efficient and no less effective baseline characterisation can be determined. Here we attempt to describe a robust and tractable approach to characterizing the system, based on describing emergent properties similar to the onshore approach detailed by Romanak et al. (2012) that minimizes the need for expensive bespoke observational programs and utilizing techniques to understand the scales of spatio-temporal heterogeneity.

The term "baseline" has varied connotations but is most often defined in geological CO_2 storage monitoring as an a-priori fixed quantity or state from which a response to perturbation can be assessed. Given that the marine system is highly dynamic and cannot be described by a fixed quantity, from a semantic point of view the term baseline may be misleading, especially in a highly cross-disciplinary, stakeholder involved field such as CCS, where "baselines" might also be used in geological, engineering, regulatory and non-academic contexts. Similarly, "characterisation" can imply a one-time-only spatially resolved assessment (e.g. Romanak et al., 2013) and may also have different connotations within the various disciplines surrounding CCS. Here we use "Baseline" as any data or characteristics of the area that support impact assessments and design of monitoring programs, either through direct use or as support for process modelling in defining anomalies, vulnerabilities, and resilience of the local ecosystem. Further, in this manuscript we have used "leak" to refer to an unplanned release of CO_2 into the water column from storage and "seep" to refer to natural phenomena of gas or fluid exchange from sediments to water column. However, it should be noted that these terms are often used interchangeably.

Risk of leakage from storage is considered to be small, given appropriate geological characterisation of the storage complex. When then is it appropriate to undertake post-injection environmental monitoring, in response to geological anomalies or concurrently with geological monitoring? What does the level of risk justify in terms of monitoring cost, fidelity and the underpinning baselines / characterisation, especially considering the large areas of review for projects which may be of the order of $200 km^2$ (Jenkins, 2020)? Given that marine environments are predicted to change significantly due to climate change over the same time frames required for long term storage monitoring, for how long are particular baselines valid? An environmental assessment is required as part of the permitting process, pre-dating injection of CO_2. Injection may continue for 1–2 decades and some degree of post closure monitoring may also be mandated (Dixon et al., 2015). Obviously the longer any baseline, characterisation or associated data set remains valid, the more cost-effective it is. An awareness or assessment of the "use by" date and the need to update such information is crucial. However in regions like the North Sea which are highly impacted from multiple use (Crain et al., 2008; Gissi et al., 2021; Kannen, 2014) as well as inherently dynamic, an a-priori assessment via observations of state is unlikely to be durable (Bourrin et al., 2015; Cathalot et al., 2010; Cotner, 2000; Mengual et al., 2016; Munguia et al., 2011; Sanchez-Vidal et al., 2012). Longevity may be obtained via combining multi-annual observations into a climatology, however the resulting increase in variability around the mean can lessen the precision of anomaly criteria, leading to inefficient monitoring.

Sensors and monitoring techniques are continually improving, (Jenkins et al., 2015; Jenkins, 2020). However irrespective of sensor fidelity some metric which distinguishes normality from a potential leak scenario, with low error, is always necessary. Such anomaly criteria are essential components of monitoring programs and require a detailed understanding of natural spatio-temporal variability. How can we conceptualise the spatio-temporal variability of the marine environment, to underpin detection and attribution, in a way that has sufficient longevity? A solution is to identify relatively simple metrics or emergent properties of the system which are tightly constrained under normal conditions but would be perturbed by a leak or its consequences, (emergent properties are those that manifest due to interactions between system components rather than individual processes). A classic example is the soil gas stoichiometry method (Beaubien et al., 2013; Romanak et al., 2012) which has been successfully demonstrated in an attribution context (Romanak et al., 2014). Marine equivalents may similarly be based on stoichiometric or multivariate relationships (Botnen et al., 2015; Uchimoto et al., 2018; Nishimura et al., 2018; Omar et al., 2021), or an understanding of (normal) dynamic rates of change in key indicator combinations (for example a rapid change in pH not accompanied by a similarly rapid change in temperature or salinity (Blackford et al., 2017)).

Quantification and impact assessment present perhaps more complex challenges. A common approach to assessing impact in marine systems is the BACI (Before-After-Control-Impact) approach (Green, 1979), whereby multivariate statistical techniques are used to compare the impacted site and a control site both before and after impact. This was used successfully during the QICS controlled release experiment to assess impact and recovery of the seafloor community subjected to a CCS-like leak event (Blackford et al., 2014; Widdicombe et al., 2015). A modified approach, BAG (Before-After-Gradient) replaces the control methodology with a presumed gradient of lessening impact as distance increases from the impact site. Both approaches are suitable where the impact site is known a-priori, as in the case of controlled release experiments, however for CCS we do not know in advance the precise location and time of impact, rendering the "before" component difficult to resolve because of the continuously changing nature of the marine environment. This leaves us with a CI (Control-Impact) or G (Gradient) design, which are established approaches in impact assessment (see Methratta, 2020, and references therein for an in-depth discussion). However, the rigour of CI or G approaches is highly dependant on minimizing all other environmental differences between the impact and control site or along the impact to no impact gradient. Understanding the environmental heterogeneity of the review area is therefore crucial to establish this rigour. A contemporary comparative approach based on comparing impacted sites with neighbouring, ecologically and environmentally similar, non-impacted sites will arguably be more accurate than referencing to a distant historical baseline from a specific seasonal period, and has the advantage of only being required when attribution is confirmed. As well as environmental impact assessment, such an approach could potentially be used to quantify normal background sea-sediment CO_2 fluxes to aid quantification. The legal standing of all such comparative approaches is subject to discussion as none can provide absolutes.

Minimizing the likelihood of false positives, whilst maximizing monitoring sensitivity is key to efficiency and not well served by Ill-fitting anomaly criteria. Emergent properties can be nuanced seasonally and spatially (Fuhrman et al., 2015), for example between water masses of different origin and between low activity (winter) and high activity (spring) periods. There is a cost-benefit analysis between investing in sufficient site characterisation to identify high quality anomaly criteria, reduction of monitoring costs and increased monitoring accuracy. Hence, we argue that knowledge of regional features, drivers, and patterns of spatial and temporal heterogeneity will greatly assist decision making and cost effectiveness of monitoring approaches.

2. Elements of a marine baseline characterisation

In this section we describe the valuable data categories and how each underpins monitoring (2.1) and in 2.2 place each element in a suggested sequential process, describing exemplar common sources of each data type, as summarized in table 1. Whilst some elements are already routinely applied, others are not and may be a valuable addition.

2.1. *What needs to be known and why*

Initial, followed by routine monitoring has the intent of providing assurance that storage is robust, with the ability to identify anomalies with high sensitivity and low false positive rates. If anomalies are detected then sequentially monitoring processes must attribute that anomaly to a storage source, and only if attribution is confirmed, quantify the release rate, and finally assess impact. Impact assessment in itself can have multiple elements; confirmation that an impact is a statistically, spatially or temporally significant divergence from normal biological dynamics and possibly, in the case of dispute, the attribution of that impact mechanistically to the CCS anomaly (for example is the impact consistent with high CO_2 concentrations).

The following set of parameters and variables (what to measure) for environmental monitoring relate to the detection-attribution-quantification-impact continuum (why to measure) as follows (table 1.)

- **Characterisation of the geological storage complex** is a mandatory component of storage regulations which require that environmental monitoring be targeted to areas defined by the risk assessment (Bachu, 2010; Nepveu et al., 2015). With areas of review typically of the order of 200 km^2 (e.g. Dean and Tucker, 2017), and given that the leakage risk is low, a targeted approach to environmental monitoring may be warranted. This approach requires understanding the **geographical distribution of residual risk**. For example the presence of abandoned well bores, pipelines and other infrastructure or other geological features, such as chimneys, pockmarks or fractures alongside hydrodynamic mixing regimes enable the design of more efficient strategies for targeted **detection** monitoring (Alendal, 2017; Hvidevold et al., 2015) and may help to identify ecosystem vulnerabilities. These features are most likely to remain unchanged over time and can be assessed using a one-time characterization.
- **Regional and local current conditions** will govern transport and dilution of tracers in the area. The anisotropic tidal current, with its many constituents can be readily predicted (Davies and Furnes, 1980) . Local regional current systems, e.g. coastal currents, are generally understood and characterized by hindcast models (Tonani et al., 2019). Topographic steering and events, such as storm passages, cause local current variability (Alendal et al., 2005). This temporal variability can be gathered from current metre time series, and high-resolution General Circulation Models (GCMs) could assist in determining spatial and temporal variability and correlations (Gundersen et al., 2021). Such current statistics can define dominating directions for transport, directly assisting in defining **impact** areas and **detection** probability. The current statistics also support simulating ensembles of scenarios with process models, hence further supporting impact area assessment and design of monitoring programs (Alendal, 2017; Blackford et al., 2020; Gundersen et al., 2020; Hvidevold et al., 2016; Hvidevold et al., 2015; Oleynik et al., 2020). This category of information will have longevity and can mostly be achieved through existing data. Additional simulations and current metre sampling could supplement information if necessary.
- **Active natural gas seeps** are common features of marine environments. Often seeps are composed of methane (either biogenic or thermogenic in origin, Oppo et al., 2020), natural CO_2 or a combination of both (McGinnis et al., 2011) and may be associated with deeper oil and gas deposits (Bottner et al., 2020). Seeps may give rise to characteristics: e.g. formation of authigenic carbonates/bacterial mats and other typical fauna, pockmark formation, bubble release or black (sulfidic) sediments (Judd and Hovland, 1992). Passive acoustic methods have shown promise for **detection** and possibly **quantification** of gas bubbles (Berges et al., 2015; Li et al., 2020), by recording the acoustic signature of bubble release and bubble size. Knowledge of the presence or absence of natural biogenic and

Table 1
Relationship between the proposed staged approach to site characterisation (how), data access (where), the relevant variables and parameters (what) and the rationale for the characterisation (why).

	HOW to use in a staged characterisation approach	**WHERE** to access	**WHAT** variables are required	**WHY** it needs to be measured
0.	**Geological characterisation** Definition of storage complex location and infrastructure siting	Operator site assessment Existing geological surveys	Presence and location of geological features that may affect risk. Sediment morphology Carbonate system (DIC, pH, pCO_2) Co-variables (O_2, T, S, nutrients) Natural gas seeps and deposits Acoustic soundscape Marine CO_2 isotopic composition	**Detection:** Monitoring strategy – distribution of sensors **Attribution:** Connecting anomalies to geological features **Impact potential:** Distribution of risk relative to ecological sensitivities
1.	**Regional physical and biochemical characterisation** Use existing datasets to characterise the regional marine system and understand the principal drivers , variability and stressors of the system in terms of physics and biochemistry.	Models derived projections Earth Observation data Observational databases	Spatial and temporal discrimination of: Carbonate system (DIC, pH, pCO_2) Co-variables (O_2, T, S, nutrients) Risks of anoxia Dominant currents and tidal features	**Detection:** Provision of baseline charcterisation enabling definition of high fidelity anomaly criteria and planning of monitoring strategies relative to hydrodynamic features.
2.	**Geophysical and ecological key features, site utilisation.** Desk study to assess key features of the complex: relating to resource use, ecological importance and geophysical features	Observational databases, Scientific literature Marine traffic data Govt. agency databases and reports	Protected species, areas and functions (e.g. spawning ground, MPAs), attributes (e.g. supports fisheries), fishing pressures Occurrence of biogenic sediment gas production including CH_4, CO_2. Sediment type distributions and associated ecosystems	**Impact potential:** Are there ecologically and socially important uses of the area that indicate a greater sensitivity to CCS operations? Identification of ecosystem types and distribution. **Attribution:** Enabling attribution of anomalies to CCS processes via knowledge of natural seepage **Detection:** Will other uses impede monitoring activities?
3.	**Leakage characterisation** High resolution simulation models to simulate hypothetical scenarios, identifying areas that have an enhanced risk of impact Could occur in response to geological or water column anomalies, rather than a-priori, especially if high resolution model systems describing the region already exist.	High resolution hydrodynamic modelling systems Existing models or requiring some model development, depending on location.	Spatial and temporal discrimination of: carbonate system (DIC, pH, pCO_2) for a selection of leak scenarios.	**Detection:** Enables detailled planning and assessment of marine monitoring strategies **Quantification:** Describes relationship between leak rates and observable footprints **Attribution:** Describes relationship between leak location and observable footprints **Impact potential:** Identifying areas that have an enhanced risk of impact
4.	**Focused Surveys** Sampling of hydrodynamics, physics and chemistry to validate models and EO data; observations of more vulnerable habitats and ecosystems; assessment of seeps May not be required if model and observational data is sufficiently evaluated and detailed.	New deployments, of existing oceanographic technology, potentially autonomous.	Carbonate system (DIC, pH, pCO_2) Co-variables (O_2, T, S, nutrients) Spatial and temporal discrimination Protected species, functions (e.g. spawning ground), attributes (e.g. supports fisheries) Natural gas seeps Pore water chemistry	**Impact potential:** Are there any ecologically or socially important uses of the area that indicate a greater sensitivity to CCS operations? Identification of ecosystem types and distribution. **Detection:** Ground truthing/evaluation of synthetic data from models, earth observation and updating of direct observations.

focus

evaluation

geological bubble sources (CO_2 or methane) would reduce the potential for false positives, alongside an understanding of the **acoustic soundscape** of the region. Knowledge of natural seeps would also inform **attribution**, and knowledge of sediment gas deposits may contribute to **quantification.** An understanding of natural gas deposits and the presence of toxic substances such as heavy metals in pore waters that may be expelled as pre cursors to CO_2 leakage may assist both **attribution** and **impact** studies. Whilst on a macro scales seeps and pockmarks can be long-lived, specific flows and acoustic properties are all likely to be transitory over daily time scales.

- All monitoring stages require knowledge of the **carbonate system** (CO_2, DIC, or proxies such as pH or pCO_2), as leakage, unless in very shallow water, will enter the dissolved phase rapidly. The range, seasonality, diurnal and mixing-driven variability of the carbonate system are required to set anomaly criteria for biogeochemical **detection** (Blackford et al., 2017), to provide the basis for **quantifying** leakage (Alendal et al., 2017; Gros et al., 2019; Hvidevold et al., 2016) and enable the extent of leak driven chemical perturbation to be assessed, enabling **impact assessment** (Blackford et al., 2020). Carbonate system parameters are highly variable but do confirm to cyclic patterns and are constrained by generally understood processes (Thomas et al., 2005).
- Following Romanak et al. (2012), process based, stoichiometric approaches to aid **detection** and **attribution** are being developed and utilized in marine settings (Botnen et al., 2015; Uchimoto et al., 2018; Omar et al., 2021). Such techniques require a detailed knowledge of **co-variables associated with variations in DIC/CO_2** to distinguish natural variation from leak driven phenomena. These include physical parameters such as temperature and salinity which affect the carbonate chemistry equilibrium (Artioli et al., 2012) and may signal a (natural) physical basis for an anomaly. Chemical parameters, especially O_2, but potentially key nutrients such as nitrate, silicate or phosphate can be used to distinguish a (natural) biological basis for an anomaly based on the dominance of photosynthetic (CO_2, nutrients in, O_2 out) and respiratory (O_2 in, CO_2 out) processes . Emergent properties (system properties that arise from interactions and feedbacks between key processes) tend to produce consistent metrics (for example Redfield stoichiometry or the negatively correlated [O_2] and [CO_2]. However, in aquatic systems there is a degree of divergence from theoretical relationships due to a range of biochemical and physical mixing processes (Vachon et al., 2020). The resulting fuzzy quality of stoichiometric relationships in aquatic systems can degrade the precision of derived anomaly criteria, without additional observations.
- Comparing the **isotopic composition** of sequestered CO_2 with aquatic CO_2, or further the deliberate dosing of sequestered CO_2 with inert tracers has been proposed as a method to aid the **attribution** of detected CO_2 (Myers et al., 2019; Roberts et al., 2017). Isotope analysis was used in the QICS release experiment to demonstrate that injected and seeping CO_2 was identical (Blackford et al., 2014; Lichtschlag et al., 2015). Knowledge of natural (CO_2) isotopic composition which varies in marine environments depending on the origin of water mass and proximity to atmospheric exchanges (Humphreys et al., 2015) and background levels of tracers (if inert tracers were used to label stored CO_2) are key requirements. Isotopic compositions are a reasonably stable property of water masses.
- Assessing ecosystem sensitivity to **impact** by a perturbation can be supported by an understanding of **ecosystem features and functions** coinciding with the storage site. These may either include the presence of marine protected areas (MPAs), protected species and key spawning grounds or the presence of other stressors or agents of disturbance which could cause ecosystem degradation, such as trawling activities (Jennings et al., 2012) or propensity for anoxia (Hennekam et al., 2020). Although quantifying vulnerability of ecosystems is a non-trivial task (Certain et al., 2015; Willaert et al., 2019; Zacharias and Gregr, 2005) considering ecological sensitivities in optimizing monitoring may be important for societal assurance (Mabon et al., 2017), or where the cause of an **impact** is in doubt or contested. Knowledge of the likelihood of other disturbance in the given area would be informative. Ecosystem functions are likely to have a degree of longevity, although specific features may be transitory.
- Confirming and quantifying **impact** requires an assessment of perturbation from immediately prior to the leak event, accounting for the expected seasonal development of the (eco)system during the leak duration. Here an a-priori understanding of community structure and diversity may be useful, but with limited longevity, comparison with contemporary unperturbed neighbouring sites may deliver more certainty than comparison against a pre-injection characterisation (Blackford et al., 2014; Widdicombe et al., 2015). Such an approach is dependant on careful choice of a comparison site which must be demonstrably not exposed to leakage whilst otherwise closely matching in terms of ecosystem properties. To this end an understanding of **hydrodynamic and sediment heterogeneity** would facilitate the choice of comparison sites.

Ecosystems overlying different storage sites will experience very different modes of variability, features and sensitivity., For example, for coastal sites impacted by river plumes, such variability may be driven by terrestrial rainfall patterns and riverine sources of alkalinity along with DIC/CO_2, temperature, salinity and nutrients. Variability tends to be larger in permanently mixed shallow sites (order 30 m of less but very dependant on local hydrodynamics), where sea surface processes directly interact with the sea floor. In deeper stratified sites (for example 50 m plus) variability within the lower water column is likely to be less. Benthic systems are however ubiquitously heterogeneous both in terms of sedimentology, biota, and associated chemistry.

2.2. Process and methodology

In order to maintain a cost effective and useful approach to environmental characterisation, a staged approach is required (Widdicombe et al., 2019). Geological characterisation of the storage complex and surrounding area of review will be carried out as part of the permitting process. This would include a risk assessment of historic infrastructure such as pipelines and legacy wells and geological features such as fractures and chimneys that may indicate locally relatively higher risks of leakage at the seabed, as well as active gas seeps (Dean and Tucker, 2017). Given that this requires deployment at sea of substantial technology, adding basic and routine environmental monitoring at this stage would be far more cost-effective than mounting a separate subsequent sea-going mission. Key observations would be seafloor sediment type and morphology using active acoustics including Multi-Beam Echo Sounder and Side Scan Sonar (Brown et al., 2011; Hogg et al., 2016; Le Bas and Huvenne, 2009) and bottom water column chemistry including pH or pCO_2, O_2 and ideally nutrients, utilizing off the shelf sensors.. The former describes environmental patchiness of the system, whilst the latter quantifies the variability of primary metrics of CO_2 release albeit for a restricted period. Whilst this characterisation is invaluable, we argue that it is insufficient to fully describe the environmental system for the purposes of designing optimal environmental monitoring. Augmenting this initial characterisation with data that is often already in the public domain can address the issues of oceanographic dynamics and heterogeneity outlined above, increasing the value of this initial data and potentially providing sufficient understanding to deliver efficient monitoring, without the need for expensive, long-term, a-priori observational campaigns.

Here, by the "environmental monitoring complex" we are considering the vertical projection of the geological complex or area of review onto the seabed and overlying water column, allowing for any potential lateral movement of gas through the overburden. This latter component might be challenging to estimate. In the very shallow QICS experiment lateral movement of the same order as vertical travel was observed (Cevatoglu et al., 2015), and lateral movement is observed within storage reservoirs (Chadwick and Noy, 2015). In practice lateral movement might be assumed to potentially add 1–2 km in each direction.

Following the initial site characterisation, we propose the following process:

1 Desk study of existing data derived from earth observation, models, and observational biogeochemical databases to characterize the region containing the environmental monitoring complex. Generically, the aim is threefold comprising 1) understanding the regional scale drivers of water mass heterogeneity and dynamics overlying the complex, 2) establishing a spatio-temporally robust understanding of biogeochemical variability, including current conditions, evaluated by observations carried out during site characterisation, to support the identification of anomaly criteria, and 3) identifying data gaps that might require actual site monitoring.
2 Desk study to assess key features of the environmental monitoring complex, including i) other uses such as fishing, with a view to identifying the presence of other stressors that may cause impacts and ii) important species, spawning grounds or marine protected areas (MPAs), isolating any environmental sensitivities that may require some degree of monitoring for assurance.
3 Building on the risk mapping carried out during site characterisation, and any additional sensitivity identified in stage 2, an evaluation of leak scenarios to determine optimal monitoring strategies including the placement of sensors or routing of mobile surveys. Such a model-based analysis can be useful in demonstrating the effectiveness of any given monitoring strategy. This step may be optional, depending on geological risk assessment and morphological considerations.
4 Filling of data gaps with onsite sampling. If the site characterisation and additional analysis outlined above is insufficient to describe the system for subsequent monitoring, focused sampling of sediment, chemistry and ecosystem parameters to evaluate models may be necessary. This will be optional depending on the availability and modernity of existing data.

Note that the regional characterization component has a wider geographic focus than the environmental monitoring complex footprint but defines the overall system within which the area of review exists. Assessing key features and mapping high risk areas are focused on the environmental monitoring complex within the area of review. A significant amount of information can be gleaned from desk based studies of existent information and are therefore low cost. Steps 1–3 are designed to minimize the requirements in step 4. There is an iterative component so that the direct site observations can be used to further evaluate, and quality assure initial data assessments (Table 1).

The following describes each part of the process and illustrates potential data sources.

2.2.1. Use existing datasets to understand and quantify the drivers and variability of the system in terms of physics and biogeochemistry

Earth observation (EO, or satellite derived) data, provides a geographically wide-ranging, high resolution, high repetition view of sea surface properties, primarily sea-surface temperature, sea level height and ocean colour but with a growing range of derived products (Tyler et al., 2016). Decadal scale time-series exist, although historical data has lower resolutions and less sensitivity. There is a high certainty of EO activities continuing with incremental improvements to data quality (National Academies of Sciences, 2018). Whilst EO data only "sees" the sea surface layer, on continental shelves this data is sufficient to determine the principal hydrodynamic features, and primary mixing regimes, via temperature. Indication of biological activity can be derived from ocean colour, which measures the (chlorophyll) pigmentation associated with primary production, which in turn impacts the carbonate chemistry. The primary utility of EO data is to obtain an understanding of the macro-scale processes that influence a region and their inter-annual variability. For example, EO data reveals that the Goldeneye potential storage site sits between two hydrographic current systems, one with an oceanic and the other with a shelf seas origin, the boundary between these current systems is mobile and each has particular physical and biological signatures influencing carbonate chemistry (Fig 1). Knowing the relative position of these features and the variability they induce is key to designing monitoring strategies and interpreting monitoring signals. The cost of accessing EO data, typically via on-line data portals, may depend on national funding models, but often will be free at the point of access.

Hydrodynamic – biogeochemical models are ubiquitous research and operational tools and cover global oceans and many regional seas, the latter typically with horizontal resolutions in the horizontal of 10 km or better, e.g. meso-scale (Holt et al., 2016). The exact specification of such models varies according to the research base, but typically the hydrodynamic part consists of prognostic equations that describe the physical movement of the oceans due to atmospheric forcing, Coriolis driven currents and tidal processes resolved onto a vertically layered grid (Madec et al., 2017). Physical parameters such as temperature and light penetration are used to drive biogeochemical models that describe the cycling of various elements, typically carbon and nutrients through a more or less complex plankton functional-type based food web (e.g. Butenschoen et al., 2016). With ocean acidification emerging as a key research topic, many of these models now include routines that describe carbonate chemistry, including pH, alkalinity and pCO_2 (e.g. Artioli et al., 2012), hence models can directly deliver much of the key information required for site characterisation (Fig. 2). Typically model outputs are saved at daily time intervals for each grid point, however model time steps are usually of the order of a minutes, so higher frequency data can be generated. Model outputs tend to have less small-scale variability than reality, due to the necessary aggregation of processes, water volumes and species into functional groups. Models can only properly describe processes occurring on scales larger than grid resolution, with various methods of parameterizing sub-grid-scale processes such as turbulent decay of energy toward dissipation on submillimeter scales. Regional and local hydrodynamic models, with higher resolution, can resolve smaller features, however, hydrodynamic models resolving processes on scales of 1 km or smaller will be close to breaking the hydrostatic assumption used in ocean models (Marshall et al., 1997). As a consequence, to be able to model topographic steering and especially describe processes with prominent vertical velocity components high resolution models have to be non-hydrostatic (i.e. the hydrostatic pressure assumption is not valid and to find pressure an elliptic equation needs to be solved for each timestep). Such models are computationally much more demanding (Alendal et al., 2005; Berntsen et al., 2006).

Meso scale model systems are typically run in decadal scale hindcast (e.g. Ciavatta et al., 2016), operational forecast (e.g. O'Dea et al., 2012) and climate prediction modes (e.g. Artioli et al., 2014; Holt et al., 2018), using observed atmospheric and riverine data to drive hindcasts and climate projections to drive forecasts, addressing the "expected evolution" aspect required of environmental impact assessments. Operational marine modelling has advanced in recent years, using data assimilation to reduce errors in model simulations. EO products such as temperature and more recently ocean colour (Ciavatta et al., 2018) are used for assimilation, and operational products are often available via on-line data portals. Whilst models deliver the precise products required for site characterisation in a CCS context at excellent spatio-temporal resolution, model skill must always be considered. Evaluation against observations is commonly used to demonstrate model quality (Edwards et al., 2012). Prognostic model skill is usually greatest at the meso scale (weekly to seasonal, 10 km) (Edwards et al., 2012) except for short term forecasts derived from operational models, however models do generally describe ranges, dynamic patterns and co-variation well, implying value for specifying CCS baselines. Model skill is also better for physical

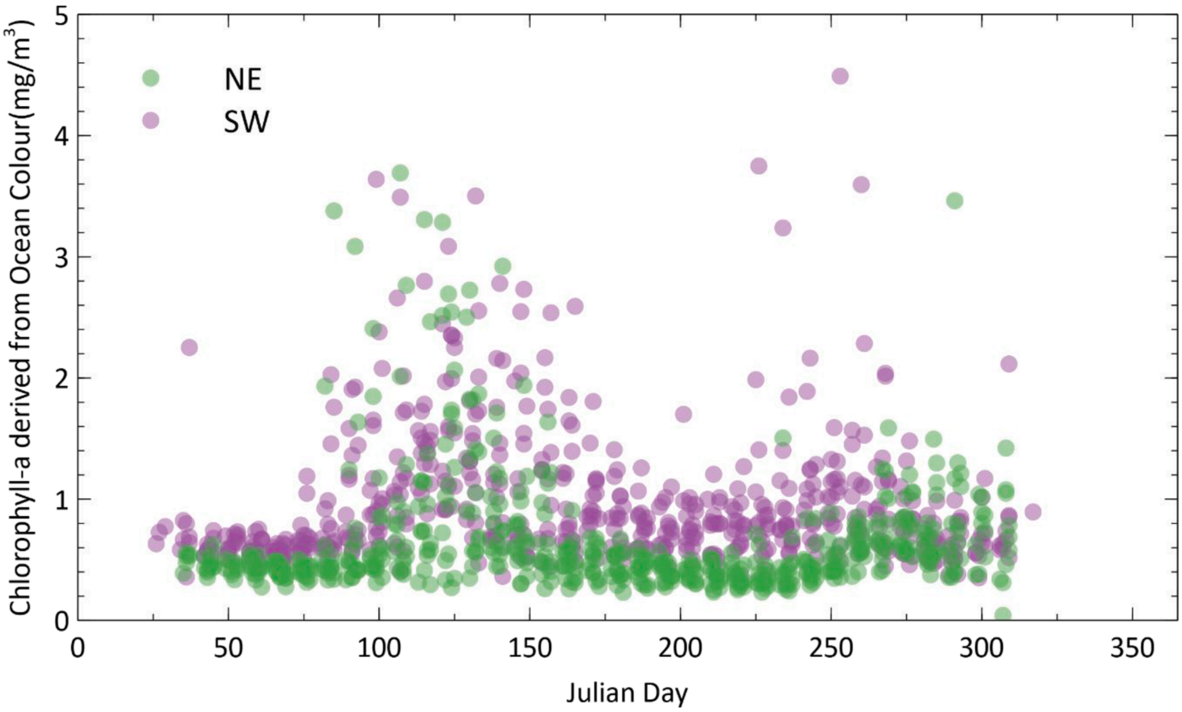

Fig. 1. Contrasting sea surface chlorophyll-a (derived from ocean colour) for region centred 50 km to the SW (Coastal Fair-Isle current influenced) and 50 km to the NE (Atlantic current influenced) of the Goldeneye storage site, data aggregated over the period 1997–2017.

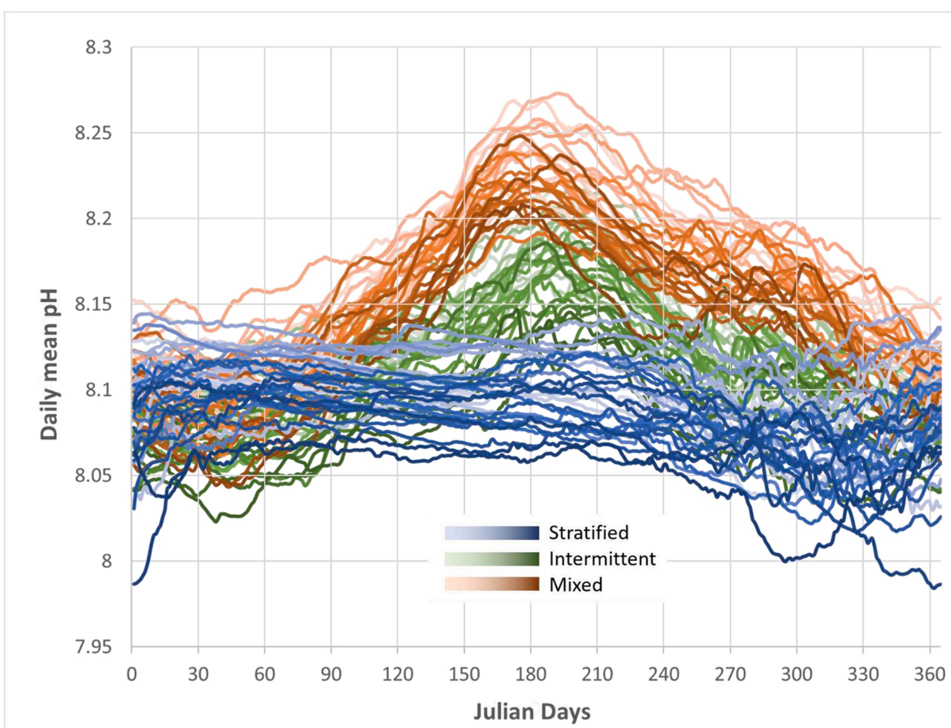

Fig. 2. Example of data extracted from a 30 year simulation of the North Sea using the ERSEM-NEMO model, showing superimposed annual cycles of pH for three hydrodynamically contrasting potential locations for storage sites. Lighter colours represent earlier years, revealing the downward trend in pH.

and biogeochemical components and decreases for biological and higher trophic level components (Rose et al., 2010). Hence models are not particularly useful for describing biological populations in the context of impact analysis due in part that the granularity of biological components is too coarse (e.g. functional groups) to be informative.

Observational databases curate key parameters obtained from scientific cruises, ships of opportunity (SOO), autonomous deployments and fixed observing platforms; data is multivariate, multi-resolution and acquired over diverse timescales with varying levels of completeness. Long term geographically complete datasets are very rare and tend to be associated with fixed sampling stations, for example the Bermuda Atlantic Time-series Study (Steinberg et al., 2001) and the Western

Channel Observatory (Smyth et al., 2015). Cruise based data can have good spatial coverage but is often limited to one-off cruises and has poor repeatability. Data obtained via SOO are limited in parameters, however pCO_2 is commonly measured and whilst restricted to commercial ship routes, there is near global coverage and collated archiving e.g. SOCAT (Bakker et al., 2016), (fig 3.) comprising of high frequency data, albeit of the surface waters only. Data from multiple sources are collected in national and international databases and are rigorously quality controlled. Such data sets also contain information on biological species and other features relevant to understanding sensitivity. Observational data sets are routinely used to evaluate and develop simulation models and ground truth EO algorithms. Exemplar data sources are listed in table 2.

New opportunities to increase the stock of relevant observational data, for example arising from recent initiatives to improve ocean monitoring such as the UN Decade of Ocean Science (Claudet et al., 2020) or the Global Ocean Acidification Observing Network (Tilbrook et al., 2019), could potentially provide significant relevant data acquisition.

Outcomes of this activity are:

- Understanding of the scales of variability in time and space that influence the environmental monitoring complex, including current conditions.
- Spatially and temporally resolved understanding of the variability of carbonate chemistry in the region (pH, pCO_2) and co-variables such as O_2 and nutrients.
- An assessment and quantification of optimal anomaly criteria for chemical monitoring with low error, for example see (Ali et al., 2016; Blackford et al., 2017).

2.2.2. Desk study to assess key features and uses of the environmental monitoring complex

Key properties, including eutrophication status (Skogen et al., 2014) and acoustic signatures (Syrjälä et al., 2020), and uses such as fishing (Jennings et al., 2012), windfarms (Gusatu et al., 2020), marine protected areas (MPAs) (European Environmental Agency, 2015), wrecks (Olsvik et al., 2011), and dumping (Carton and Jagusiewicz, 2009) impact marine environments. Such features and uses are not independent and can act as stressors, multi-stressors or provide co-benefits for the system (e.g. Ashley et al., 2014; Balazy et al., 2019; Rouse et al., 2018; Slavik et al., 2019; van den Burg et al., 2017; van der Molen et al., 2018).

Three aspects are pertinent to CCS monitoring. Features and uses that may cause impact to the system are relevant to attribution, for example the presence of bottom trawling, or the potential for eutrophication driven anoxia. The presence of key or protected species or spawning grounds and MPAs are relevant to impact or sensitivity analysis and may modify monitoring strategies accordingly. Monitoring itself may have impacts on ecosystems, for example use of active acoustics impacting cetaceans (Nowacek et al., 2013). The availability of key feature information is variable and more often delivered by national agencies and databases. For example, information is often collated by different governmental agencies which cover waters under national jurisdictions, see table 3 for exemplar sources.

Periodically, reviews of regional marine systems are published, for example the CMEMS Ocean State Reports (e.g. von Schuckmann et al., 2020 which provides a comprehensive state of the art assessment of the state of the global ocean and European Regional Seas including spatial and temporal trends of key variables and marine uses for EU waters). These reports are designed for both the scientific community, policy, and stakeholder use.

Outcomes of this step are:

- Knowledge of the potential for other stressors to impact sea floor communities and arising recommendations for monitoring practice. For example, anoxia risk suggests the use of O_2-CO_2 stoichiometry for detection-attribution.
- Knowledge of sensitive ecosystems or species and consideration of monitoring practice, for example minimizing interference from active acoustics on mammal populations or adjusting the placement of monitoring to provide assurance that key species are not impacted.

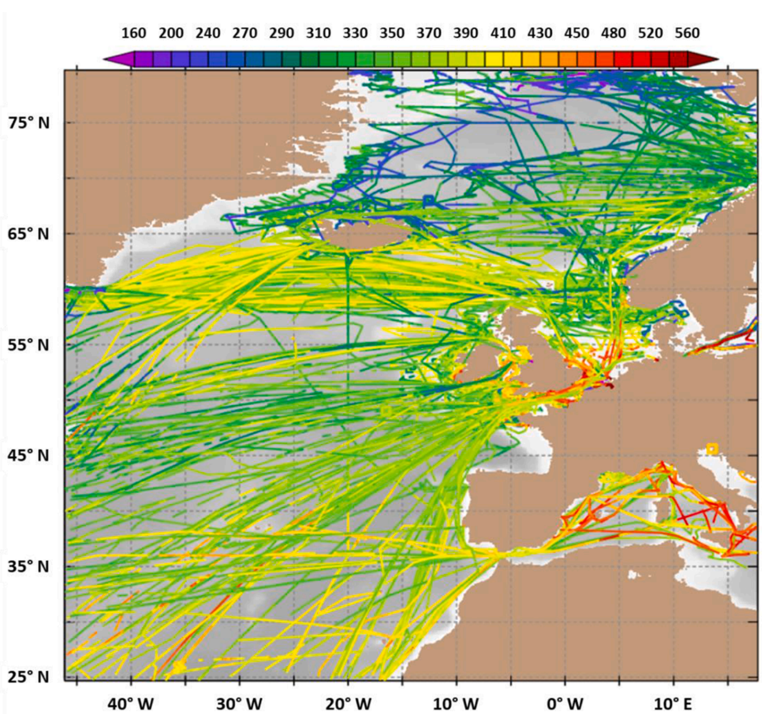

Fig. 3. Example data extraction from SOCAT, showing fCO2 data coverage in the North East Atlantic for the period 01/01/2010 – 24/11/20, extracted on 24/11/20 (Bakker et al., 2016).

Table 2
Example sources of characterisation data.

	Example Source	Variables	Horizontal resolution	Vertical resolution	Temporal resolution
Earth Observation data	NASA Earth Observing System Data and Information System (EOSDIS) https://earthdata.nasa.gov/earth-observation-data European Space Agency data portal. http://www.esa.int/Applications/Observing_the_Earth/How_to_access_data NERC Earth Observation Data Acquisition and Analysis Service NEODAAS, http://www.neodaas.ac.uk/	Sea surface temperature Sea surface colour Primary production (derived)	varies, of order km scale	~	1–5 days
Hydrodynamic-biogeochemical models	Copernicus Marine Environment Monitoring Service (CMEMS), http://marine.copernicus.eu/ Institute data portals e.g.https://portal.ecosystem-modelling.pml.ac.uk/	Carbonate system (CO_2: DIC, pH, pCO_2) Nutrients Temperature Salinity Oxygen PFT Biomass	1–10km	<1–10 m On shelf	Daily with sub hourly potential
Oceanographic databases	British Oceanographic Data Centre, BODC. https://www.bodc.ac.uk/ International Council for the Exploration of the Sea (ICES) https://ices.dk/marine-data/Pages/default.aspx The Surface Ocean CO_2 Atlas (SOCAT) https://www.socat.info/ Oak Ridge National Laboratory Distributed Active Archive Centre (ORNL DAAC) for Biogeochemical Dynamics https://daac.ornl.gov/about/ Pangaea. https://pangaea.de/ European Marine Observation and Data Network, EMODNET https://www.emodnet.eu/en	Carbonate system (CO_2: DIC, pH, pCO_2) Nutrients Temperature Salinity Oxygen PFT Biomass	Varies, may relate to fixed instrumentation, series of oceanographic stations or continuous underway data	Varies, of order 1–10 ms on shelf	Varies, instantaneous or periodic means

Table 3
Example sources of key feature information.

Marine Protected Areas, Sea Mammals	Joint Nature Conservation Committee, JNCC https://jncc.gov.uk/our-work/marine-protected-area-mapper/
Fisheries, trawling pressures, spawning areas	The International Council for the Exploration of the Sea (ICES) https://www.ices.dk/data/Pages/default.aspx Regional Fisheries Management Organisations https://ec.europa.eu/fisheries/cfp/international/rfmo/ OSPAR https://odims.ospar.org/ centre for Environment, Fisheries and Aquaculture Science, CEFAS https://www.cefas.co.uk/data-and-publications/fishdac/ Inshore fisheries and conservation authorities http://www.association-ifca.org.uk/ European Marine Observation and Data Network, EMODNET https://www.emodnet.eu/en
Protected species lists	Marine Management Organisationhttps://www.gov.uk/government/publications/protected-marine-species International Union for Conservation of Nature ICUN https://www.iucn.org/
Marine Traffic	Vessel Monitoring System (VMS) / Automatic Identification System (AIS) https://www.marinetraffic.com/
Summary Reports	EU Copernicus https://marine.copernicus.eu/science-learning/ocean-state-report/

- Knowledge of other physical infrastructures that may affect monitoring deployments.

2.2.3. Assessment to define areas with enhanced risks and detection challenges to inform monitoring strategy

Apart from the location of pipelines and injection well heads, features that could indicate a higher risk of leakage are existing relic wells, geological chimney structures and pockmarks, geological discontinuities (faults) and outcropping of storage bearing layers. Identification and mapping of such features will form part of for geological site characterisation, prior to permitting. Given the potential expense of long-term detection monitoring, an assessment of monitoring strategies that accounts for possible elevated risks within the environmental monitoring complex and in particular the likely dispersion patterns of leaking CO_2 as driven by dominant in-situ mixing, accumulation zones and topographic steering of currents may significantly increase the cost-effectiveness of a monitoring programme, by quantifying potential detection targets and identifying optimal sensor placement (Fig. 4).

To deliver this, simulation models, similar to those described in 2.2.1, but often with much-higher resolution and less biogeochemical complexity have been used to simulate ensembles of a wide variety of leak scenarios. Shelf scale models with resolutions approaching 1 *km* (Ali et al., 2016; Phelps et al., 2015) have been applied as have sub-regional model domains whose resolution can be as fine as 1 *m* (Blackford et al., 2013; Dewar et al., 2015). The latter allow very small release rates to be tested which generally pose the highest detection challenge. High resolution models also allow multiphase simulations, including the dynamics of bubble plumes and the subsequent formation of the dissolved phase (Dewar et al., 2013; Gros et al., 2019). Collectively these models can assess areas that might be more prone to receiving impact and determine and evaluate optimal monitoring strategies using both fixed and mobile platforms (Alendal et al., 2017;

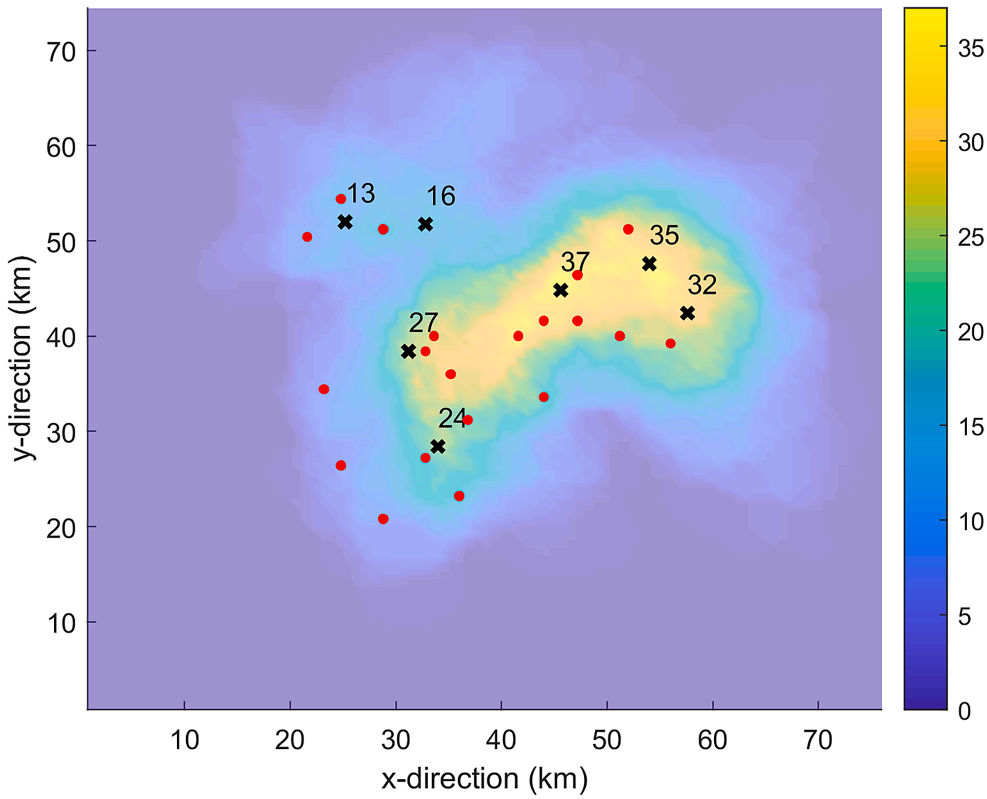

Fig. 4. An example of optimal sensor placement for 100 simulated and putative leaks from 20 different locations (red dots) with a fixed duration and a constant flux. The colour-code corresponds to the number of intersecting sets, black crosses to the optimal sensor positions. The numbers indicate how many leaks were detected by each sensor.

Greenwood et al., 2015; Hvidevold et al., 2016; Oleynik et al., 2020). Such high resolution model systems are becoming more common in oceanographic research, based partially on the development of variable grid systems such as FVCOM (Qi et al., 2009), which allow for very high resolution at specified points in a model domain, and less resolution at the periphery, ideal for leakage simulation. Such models however can have a high computational cost and may need bespoke development for specific storage sites, imposing some degree of financial cost. Efforts to develop high resolution simulators are ongoing and promise an ability to simulate large ensembles of leak scenarios efficiently, significantly reducing computational cost.

Outcomes of this step are:

- An understanding of the vulnerability of key assets to identified leakage scenarios
- Quantification of precise, low error, anomaly criteria
- Optimisation of monitoring deployment strategies, maximising detection whilst minimising cost.

2.2.4. Focused sampling of sedimentary, chemical, and biological features

There are many established and recently developed methodologies suitable for geomorphological, chemical, and biological sampling, which are detailed elsewhere (Woodall et al., 2018). In brief, the categories of direct observations consist of

- Confirmation of sediment features
- Focused sampling of physical parameters and water column and sediment chemistry.
- Focused observations of biological vulnerabilities

Assessing which direct observations are necessary a-priori is challenging and will depend on the quality and quantity of existent data available. For example, if a regional model has a recent comprehensive evaluation that demonstrates skill, especially for reanalysis or operational forecasts using data assimilation, the model data may be considered fit for purpose. However, CCS monitoring often requires knowledge of highly resolved spatial and temporal chemistry, for which observational data (and therefore model evaluation) is often lacking. A restricted deployment sampling carbonate chemistry, temperature, and salinity, using AUV or platform mounted commercially available sensors for a relatively short time period will often be a valuable addition. Confirmation of sediment features such as pockmarks and biogenic methane seeps can be achieved using acoustic mapping techniques such as side-scan sonar or multibeam echosounder mounted on ships or deployed on autonomous underwater vehicles or towed from surface boats. These techniques have the advantage of being able to survey large areas of seabed in a relatively short time, especially with shipboard systems. If no specific biological features requiring protection are revealed by desk studies, it might be that additional habitat surveys are deemed unnecessary. Alternatively, some information on habitats can be derived from acoustic surveys which indicate seabed roughness and sediment type. If necessary, community type can be estimated using ground-truthing, potentially via grab sampling or use of video recording coupled with machine learning interpretation. Sediment properties (particle size, chemical composition, presence of infauna) can be confirmed with a few well-targeted cores.

Outcomes of this activity are:

- Gap-filling, evaluation and confirmation of information retrieved from prior steps.

3. Conclusions

In this manuscript we present an alternative approach to undertaking detailed observational campaigns to provide baselines for marine monitoring. We argue that the wholesale (long term and geographically complete) collection of chemical, acoustic, and ecological data to enable detection, attribution, quantification, and impact studies in the marine environment is inefficient. The high level of spatio-temporal variability of the marine system, the implied need to consider long-term and geographically wide-ranging surveys and the time-bound nature of such information in an ever-changing environment renders defining a marine baseline in this way a high-cost and effort-intensive activity. Individual observational datasets will never be sufficient to characterize the system because they are time bound and geographically specific. But the combination of site-specific observations, relevant historical observations and temporally, geographically and biogeochemically complete system models can provide sufficient a-priori information to allow environmental impact assessments and environmental monitoring to proceed efficiently, in particular for detection and attribution. Site specific observations could potentially be gathered via judicious addition of necessary geological surveys (for either reservoir characterisation or storage assessment) with short-term, high-frequency lower water column sampling of carbonate and associated variables using ship deployed underway vehicles or landers. Alternatively, shore based autonomous vehicle deployments, lasting a few weeks would not require additional expensive ship time. Seabed sediment and morphology mapping and underway high frequency sampling of near seabed carbonate chemistry, temperature and salinity would be the primary requirements, enabling skill assessment of models and the subsequent derivation of anomaly criteria and monitoring strategies.

Quantification and impact assessment will however require reference to baseline or "normal" values. As baseline surveys undertaken prior to the commencement of storage may predate a requirement to assess flows or impacts by several years, they are unlikely to represent the contemporary set of environmental circumstances and seasonality. Consequently we argue that post attribution comparison with similar nearby non-impacted areas that are environmentally and seasonally close analogues, using for example Control – Impact or Gradient based multivariate analytical techniques (Arvanitidis et al., 2009; Methratta, 2020; Somerfield and Clarke, 2013) will provide a more reliable analysis, as demonstrated in a CCS context (Blackford et al., 2014; Widdicombe et al., 2015).

The necessary a-priori environmental information required to develop an environmental monitoring plan can be summarized as:

- Information about the background variability of key parameters relevant to detection etc., such that error minimized anomaly criteria can be identified
- An understanding of risk distribution and hydrodynamic mixing to identify specific vulnerabilities and optimize monitoring strategies
- An understanding of meso scale heterogeneity to enable comparative studies (contrasting impacted vs. non-impacted areas).

Whilst at this point, we don't see sufficient rewards of initial high cost and high effort activities for environmental monitoring, this could change if public support hinges on over-designed monitoring. Given the present need to build momentum for CCS projects to achieve the Paris Agreement, we have outlined arguments for using existing data from public sources. We hope this contribution adds to the public dialogue with stakeholders regarding a transparent and responsible design of environmental monitoring programs.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This work was primarily supported by funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No.654462 (STEMM-CCS) and was additionally supported by: the Norwegian Research Council (CLIMIT project no 254711, BayMoDe); the ACTOM Act on Offshore Monitoring project, (Accelerating CCS Technologies, Horizon2020 Project No 294766) with financial contributions made from the UK Department for Business, Energy and Industrial Strategy (BEIS); the NERC single centre national capability programme – Climate Linked Atlantic Sector Science (NE/R015953/1) and The Research Council of Norway.

Data for fig 1. was provided by the NERC Earth Observation Data Acquisition and Analysis Service and were processed by NEODAAS from ESA Ocean Colour CCI version 5.0 L3 data. The image of underway pCO_2 data coverage, (fig 3.), is courtesy of 'The Surface Ocean CO_2 Atlas (SOCAT) which is an international effort, endorsed by the International Ocean Carbon Coordination Project (IOCCP), the Surface Ocean Lower Atmosphere Study (SOLAS) and the Integrated Marine Biogeochemistry and Ecosystem Research program (IMBER), to deliver a uniformly quality-controlled surface ocean CO_2 database. The many researchers and funding agencies responsible for the collection of data and quality control are acknowledged.

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