An initial study of the utility of some marine M&V methods for subsea CCS:

Bass Strait Case Study
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Executive summary

Offshore Gippsland is widely recognized as one of the most promising CCS sites in Australia, with its proximity of large point sources and promising storage offshore. Storage offshore (at Sleipner and SnØvit) has been successful, but the monitoring techniques there focused on marine seismic and did not investigate effects at the sea bed or in the water column. If monitoring of this type is needed, perhaps because of specific risks or issues of social license, it will be useful for techniques to be available whose benefits and limitations are reasonably well understood by project proponents. This report provides an initial assessment of some proposed techniques for this type of "shallow focussed" monitoring that may be used in offshore Gippsland. The methods considered may be useful for testing for loss of containment, and more probably for defining baselines and checking for some aspects of possible environmental impact. The study will inform the first phase of purchasing of marine monitoring assets, funded under EIF, leading to more detailed investigation of their capabilities. The CO2CRC, as CarbonNet’s research partner, has been allocated $4M for purchase of marine monitoring assets, within the GIPNET segment of its successful EIF bid and this study clarifies how to use the proposed assets.

We have used the existing expertise and databases in CSIRO Oceans and Atmosphere Flagship (O&A) to assess our ability to measure the ingress of stored CO₂ into the water column, and quantify the size and type of loss of containment at the seabed that could be detected. O&A has baseline data and experience, relevant to the likely storage areas, covering tide and current patterns, seasonal and spatial variations in seawater composition, instrumentation, and modelling. An initial CSIRO-funded study collected together the relevant data and identified the opportunities for detailed analysis and synthesis. This report investigates some elements of a model monitoring programme, namely measurement of water chemistry, and acoustic mapping.

Our major conclusions can be summarised as follows.

1. Water sampling, from a combination of fixed moorings and gliders, would be able to monitor a region of order 10 x 10 km² and detect a specific type of leak to the seafloor (point source, 10 kt/yr) anywhere in that area with high confidence and low false alarm rate. Leaks of this size are very unlikely and would probably be detected in other ways much earlier.

2. Natural variability causes changes in water chemistry that are equivalent to much larger releases, so the main application of this type of monitoring would be to characterize the natural background ("baseline") and hence show that environmental impact (in the specific sense of acidification of the water over a large area) was negligible.

3. Monitoring by acoustic methods is extremely sensitive to bubble streams. The background noise level (mainly biota) is very variable, but streams corresponding to only 10 t yr⁻¹ should be readily detectable above environmental clutter with high confidence of low false alarm rates.

4. Seafloor mapping will reveal a multitude of features of unknown provenance or duration, and this type of monitoring would have to be approached with an awareness of the need to control false alarm rates.

5. Field assessment of the technologies described will be a priority for future work along with the collection of baseline datasets for potential storage locations along the Gippsland coast.

Based on this study we propose, using EIF funds, to investigate a model monitoring suite consisting of a small number of fixed moorings to provide reference acoustic, chemical and oceanographic data. These would be backed up by autonomous vehicles (AVs, e.g. Wavegliders) patrolling on a raster pattern, collecting acoustic and chemical data. Investigating this model suite will establish many aspects of real-world feasibility (for example, the robustness of hardware or the use of autonomous vehicles) and should
develop a capacity that could be deployed for collection of appropriate baseline data at a CarbonNet site, if this were to be needed.
## 1 Introduction

Globally, there is growing interest in geological storage of carbon dioxide (CO₂) captured from point sources of pollution and large-scale Carbon Capture and Storage (CCS) demonstration projects are underway in several national jurisdictions (e.g. Australia, UK, Norway, USA, Canada). Offshore Gippsland (Victoria) is widely recognized as one of the most attractive basins for Carbon Capture and Storage (CCS) in Australia, with its proximity of large emission point sources and promising geological storage. The CarbonNet project has been established as one of the national CCS Flagships, and is investigating the potential for a capture and storage network that initially aims at storage of 1 - 5 million tonnes of CO₂ per annum. Successful implementation could be the starting point for the development of a fully commercial-scale CCS network, with significant impact on Victoria’s emissions. Additionally, the Federal Government has released greenhouse gas storage permit areas in Commonwealth waters of Bass Strait and the Victorian Government will release permits in state waters in 2015.

It is a legislative requirement of approvals for CCS activities in Australia that a robust monitoring and verification (M&V) program is implemented. Depending on risk profile, this might include monitoring of specific targets in the marine environment (seabed and water column) in the vicinity of the storage site (DRET 2011 Guidelines for injection and storage of greenhouse gas substances in offshore areas). Designing cost-effective methods for implementing such a monitoring program is an active area of research and off-shelf solutions are still several years away. Overseas examples of offshore storage (at Sleipner and Snøvit) have tested some aspects of monitoring in the marine environment (although monitoring is dominated by reflection seismic imaging and pressure monitoring) but these sites are located in remote and deeper waters (>100m deep) than those under consideration in Bass Strait, thus the techniques trialled at these sites might not be sufficient or practical at the Victorian site.

There is a clear expectation that geological storage sites will be chosen after significant site investigation such that the risk of loss of containment is extremely low. The function of marine monitoring will, therefore, largely be confirming that environmental impact has not occurred. Marine monitoring might be helpful for securing social license when the storage sites are close to land and the area has multiple uses. Gaining legislative and social acceptance of the security of storage locations could be assisted by monitoring programs that can detect putative release signatures and impacts in the marine environment above baseline environmental variability, although the primary demonstration of containment and conformance will come from deep focussed monitoring methods such as reflection seismic. Methods for shallow marine monitoring might require high-precision measurement techniques deploying sensors on a range of platforms that provide adequate spatial and temporal coverage of key parameters. These shallow monitoring concepts will need further development, and close integration with the actual risk profiles of the sites in question.

As significant natural and anthropogenic change in the marine environment is almost certain over the period of storage operations, it is also important for storage operators that changes resulting from other drivers and pressures in multiple-use zones (e.g. climate, fishing, oil and gas operations) are not attributed to CCS. Thus, an effective monitoring program will need to be aware of research into understanding the drivers of environmental variability. One particular consideration is that dissolved CO₂ concentrations in seawater are predicted to increase with rising atmospheric CO₂ levels and that rising global temperatures will exacerbate these effects on seawater pCO₂ and pH. Such ‘acidification’ of seawater is a growing global concern expected to have detrimental impacts on many marine organisms, especially those that build calcium carbonate reefs or shells. Identifying the effects of ‘external’ ocean acidification and differentiating them from the sorts of effect that could characterise local releases is a key area where such research will help build societal confidence in CCS, particularly as CCS provides a mechanism for directly reducing the impacts of ocean acidification. We note that studies of the effects of ocean acidification on marine ecosystems are starting to provide useful information to quantify the possible impacts of CO₂ release from storage sites.
Despite the Bass Strait waters along the Gippsland coast being home to multiple industrial activities, from shipping and offshore oil and gas production to fishing, aquaculture and tourism, little is known regarding baseline variability of the major parameters that would need to be measured to monitor for the signature and impacts of a CO$_2$ release. These signatures may be chemical, acoustic or biological. Dissolved releases will have a chemical signature, gas releases as bubbles will have an acoustic signature but only releases with an environmental impact would be expected to exhibit a biological signature. The relative balance between dissolved and gaseous composition of a release is variable depending on factors such as water temperature, water and gas pressure, water depth and sediment type. In the UK, a recent deliberate seabed CO$_2$ release experiment in coastal waters (QICS; Taylor et al., 2014) showed the seep to manifest as multiple, intermittent bubble streams. The CO$_2$ in the bubbles rapidly dissolved in seawater, creating a dispersing plume containing elevated concentrations of CO$_2$ in the overlying water column. Bubbles in the sediments also left pockmark impressions in the seabed. Such results are consistent with what is known from natural analogues (i.e. natural CO$_2$ seeps). These signatures all have potential for use in detecting loss of containment but understanding their relative balance and range of variability compared to baseline conditions is essential for ensuring they can be used in a cost-effective manner.

CSIRO has undertaken a recent desktop study into the M&V requirements for the marine environment of Bass Strait. This analysed a 6-year time series of sea water CO$_2$ measurements from along the Gippsland coast to understand the range of natural variability and its influence on detection of a CO$_2$ release from an unknown source. Despite seasonal variability of dissolved CO$_2$ being reasonably large, monitoring designs that constrain natural variability were able to detect a theoretical plume within a few weeks. These designs used several optimally-placed fixed moorings. The report concluded that the magnitude of CO$_2$ input that was theoretically detectable above background variability was in the range of 17-100 ktonnes yr$^{-1}$.

In this report we describe the results of a desk-based project to assess the detectability of CO$_2$ sources to the marine environment using both chemical and acoustic methods. We first revisit our assessment of CO$_2$ geochemical detection to determine if the use of mobile platforms can provide gains in the level of detectability. We then consider active acoustic technologies as a complimentary approach to detection of gas sources at the seabed and contrast their sensitivity and practical deployment considerations to those of CO$_2$ measurements. While this study is relevant to near-coastal subsea CO$_2$ storage generally, we specifically consider the possible relevance to the CarbonNet project.

In describing a prototype marine M&V strategy in this report the intent (as stated in the original proposal to ANLEC) is to arrive at sensible decisions about the assets to be purchased as part of the EIF support of marine monitoring. The “prototype” is a straw man, a simple initial model whose function is primarily to arrive at answers as to how much CO$_2$ could be detected in the marine environment by the techniques supported by the assets. While obviously the methods have to be feasible in some zeroth-order sense, at this stage there is no detailed consideration of operational issues, cost, or connection to whatever monitoring plan may eventually be agreed with regulators. The present study provides the initial context for further investigation of these matters, as foreshadowed in a separate detailed proposal for securing the related EIF funds. It is not a proposal for a feasible or necessary monitoring strategy, which can only be discussed once much more detail about the risk assessment for the CarbonNet site is available; in particular, more spatial information will be needed to identify risk-based monitoring targets, and the need for monitoring for environmental impact will have to be clearer.
2 Background to the CarbonNet area of operation

Recent discussions with CarbonNet have given us insight into their preferred locations for storage. These have been narrowed to 3 potential sites in Bass Strait with one selected as the preferred location. Geological integrity of the storage structure has been well assured and no faults have been identified through the location, although there is an existing wellhead. The preferred site is located within the traditional country of the Gunaikurnai people and crosses three jurisdictional boundaries, extending from just landward of the Gippsland coastline, offshore through State waters into Commonwealth waters to a maximum water depth of ~40m. The other two locations lie in Commonwealth waters of around 23-33m depth. We have taken account of these water depths, and the proximity to the shore and other infrastructure, in our consideration of suitable monitoring assets. The preferred site is located close to the Gippsland Lakes Coastal Park and ~20km north-east of the Ninety Mile Beach Marine Park. This region is within the “Twofold Shelf” IMCRA 3.3 bioregion (Marine) (IMCRA 1998). The Twofold Shelf region has exposed coastline with long sandy beaches broken by rocky headlands and numerous coastal lagoons. The subtidal sandy expanses characteristic of this area are recognised as having one of the highest species diversity levels of any place on the planet, with 860 species discovered within 10 m². These shores have moderate tidal range of ~2 m and variable wave energy.

These shallow coastal waters are well-mixed throughout the year due to tidal stirring, thus changes in water properties near the seabed should be reflected throughout the water column which will have advantages for monitoring. Ocean currents in the Bass Strait are largely wind driven and direction is closely aligned with topographic contours, i.e. alongshore. Current meter records near to the preferred site show the current direction to reverse periodically leading to an oscillatory nature in the alongshore flow. The area is also subject to seasonal intrusions of water from the Tasman Sea and these waters have quite different water properties (temperature, salinity, dissolved CO₂) to Bass Strait waters, increasing environmental variability substantially. Lidar mapping of the seabed along this coastline shows water depths to deepen rapidly to 10m depth within a few hundred meters of the coast, thus the majority of the domain under consideration lies within the 10-25m depth range. Seabed features appear to include ripples, reefs and some raised mounds up to ~5m above the seabed to the south of the region. Other features of the site that require further investigation include possible groundwater fluxes at the seabed from a shallow aquifer system.

Bass Strait is home to a diverse and highly endemic marine biota, and supports productive fin and shell fisheries. Many organisms have wide but patchy distributions across Bass Strait, indicative of heterogeneous conditions and diverse microhabitats that support distinctive communities. Primary habitats include:

- soft sediments, which support invertebrates such as polychaetes, pycnogonids, pericarid crustaceans, opisthobranch molluscs, and brachiopods, many of which are undescribed;
- rocky reefs, which support diverse sessile invertebrate assemblages including sponges, bryozoans and gorgonians;
- sandy seafloors, which support macroalgal and seagrass communities.

Key commercial species harvested from eastern Bass Strait include scallops, abalone and southern rock lobster. The Southern and Eastern Scalefish and Shark fisheries also include Bass Strait. The Victorian scallop fishing zone extends for 20 nautical miles from its coast, with the majority of the fishery being conducted on scallop beds accessed from the ports of Lakes Entrance and Welshpool. Southern rock lobster occurs on coastal reefs in the region and abalone is harvested in coastal areas down to depths accessible to divers (~30m). An aquaculture industry based on ranching of brood stock also exists. Recreational fishing is a popular activity within the Gippsland Lakes Coastal Park.

The CO2CRC, as CarbonNet’s research partner, has been allocated ~$4M for purchase of marine monitoring assets, within the GIPNET segment of its successful EIF bid. These assets were proposed by CSIRO based on
the world-leading marine monitoring expertise of their Oceans and Atmosphere Flagship. CSIRO is looking to procure, maintain and utilise these assets during their first 5-years of operation.

This study is focussed on assessing existing datasets to clarify options for monitoring strategies and investigating which mix and type of assets (sensors, fixed moorings, manned craft, robotic vessels and lab equipment) is best suited to offshore Gippsland, with the most efficient scheme to deploy them. The study should be relevant to a hypothetical marine monitoring network for environmental assurance in Gippsland, as foreshadowed in the successful EIF bid, and also indicates the type of baseline studies that might be useful.
3 Chemical detection

3.1 Introduction

In this section of the report we are concerned with detecting the chemical effects of the ingress of CO$_2$ into the water column. The approach is to assume a simple, spatially small release at the sea floor and then model the dispersion of the CO$_2$ into the water column. We will further assume that it is possible to survey the area in question with an autonomous, patrolling vessel (e.g. a waveglider, Figure 1) and make regular measurements of the water chemistry. This is an initial assumption but serves to clarify possibilities and poses further relevant research questions. Using available information about the natural variability in the water chemistry, it is possible to estimate the detection probability and false alarm rate of containment losses of various sizes. This approach builds on earlier work that investigated the effectiveness of measurements taken at fixed moorings in the same area that will be modelled in this work (Greenwood et al. 2015). That study assumed the use of three fixed moorings to measure comparable parameters to those we shall use here; releases had to be larger than around 20 ktonnes yr$^{-1}$ to be detected.

![Figure 1. A waveglider being prepared for deployment.](image)

CO$_2$ gas dissolves readily in seawater and a dissolved plume would be the expected fate of a seabed release of CO$_2$ into the marine environment. Dissolved CO$_2$ dissociates into an equilibrium solution of carbonic acid (H$_2$CO$_3$), and the ionic species carbonate (CO$_3^{2-}$) and bicarbonate (HCO$_3^{-}$), together referred to as total dissolved inorganic carbon (DIC). The combined inter-reaction of dissolved CO$_2$, carbonic acid, bicarbonate, and carbonate is often referred to in its entirety as the ‘carbonate system’, which can be defined by four state variables: (1) the CO$_2$ partial pressure ($p$CO$_2$), (2) the concentration of DIC, (3) the total alkalinity (TA), and (4) the pH. Determination of the entire carbonate system can be made from any two of these parameters, although not all pairs give the same accuracy. These variables fluctuate naturally within seawater due to a variety of physical, chemical and biological processes and any addition of CO$_2$ to the water column would also change these variables. Thus, there are a number of variables that could be measured to detect a release of CO$_2$ into seawater. In this study we have focused on DIC because it can be usefully modeled as a conservative tracer, due to it not being influenced by temperature changes. Current sensor technology (as summarised in Section 5, Table 7) would support pH or pCO$_2$ as the preferred variable to measure from autonomous platforms rather than DIC, however, the statistical distribution of
DIC measurements aligns with those of temperature normalized pCO$_2$ and pH, given natural variations in alkalinity, such that the results from this study are applicable to any three of these measurement variables.

The report therefore covers three distinct areas. Firstly, we present the dispersion modelling of the effects of a release to the sea floor, and show the typical signals that a glider could be expected to record. Secondly, we evaluate the natural variations in water and assess how large they would be on the time- and spatial-scales being probed by the glider. Finally we combine this information into an estimate of the detection probability and false alarm rate as a function of release rate.

### 3.2 In situ data

The Japanese Volunteer Observing Ship (VOS) *MV Transfuture 5* has provided surface oceanographic measurements of physical and chemical variables through Bass Strait over the past 5 years. In this study we have used high resolution underway records of surface ocean physical (temperature, T; salinity, S) and chemical (partial pressure of CO$_2$, pCO$_2$) properties, supported by bottle measurements of total dissolved inorganic carbon (DIC) and total alkalinity (TA) collected over the period Oct 2006 to Jul 2011, as reported by Hardman-Mountford et al. (2014). The underway measurements were made using a conductivity-temperature-depth probe (CTD) and a National Institute for Environment Studies model pCO$_2$ system with combined bubble and showerhead-type equilibrator (Murphy et al, 2001). Underway measurements are taken every 10 mins and have a spatial separation of ~5km. Bottle samples for analysis of DIC and TA were collected by crew on board the ship on a daily basis, following the protocols of Dickson et al. (2007), and analysed by the Ocean Carbon Laboratory of the CSIRO Marine and Atmospheric Research Division, Hobart. Total alkalinity was measured by open-cell potentiometric titration and total dissolved inorganic carbon dioxide was measured using a coulometer. The analytical accuracy and precision of both measurements was ± 2 µmol/kg, based on repeat measurements of certified reference seawater standards supplied by Dr A. Dickson, Scripps Institution of Oceanography. Following the approach of Kitidis et al. (2012), the relationship between S and TA from discrete bottle samples was analysed and showed a strong correlation that could be used to predict TA on a continuous basis from continuous underway records of S, as described by Hardman-Mountford et al. (2014).

### 3.3 Model Description

#### 3.3.1 HYDRODYNAMIC PLUME MODEL

A Lagrangian particle-tracking model is used to approximate the advection and dispersion of a small-area (< 0.1% of the storage area) CO$_2$ seep located at the centre of a 20×20 km domain within Bass Strait (Greenwood et al 2014). The seasonal circulation of the Strait is largely wind-driven with currents about 2% in magnitude of the wind speed, and direction closely aligned with topographic contours (Fandry 1982; Sandery and Kämpf 2005). Tidal currents are strong, with lateral tidal excursions of 1-3 km (Fandry, 1982), but their temporal symmetry means that they contribute little to the residual flow. Much greater excursions of water result from prolonged wind-driven alongshore velocities that reverse in direction approximately every 10 days in response to synoptic weather patterns. Tidal stirring ensures that the Strait waters are vertically well-mixed throughout most of the year with little or no stratification (Baines and Fandry 1983). Consequently, the main circulation features are well represented by an assumption of depth-averaged flow (Middleton and Black, 1994; Fandry et al. 1985; Sandery and Kämpf, 2005), which we apply here.

In this approach, the plume mass is divided into particles each of which is subjected to advection by the flow and diffusion represented by a random-walk formulation (Kitanidis, 1994). For example, in the x-direction:

$$ x(t + \Delta t) = x(t) + u \Delta t + (2D\Delta t)^{1/2} \varepsilon $$

(1)
where $x(t)$ is the location at time $t$, $u$ is the advective velocity, $D$ is the diffusion coefficient, and $\epsilon$ is an independent, normally-distributed, variable with zero mean and unit variance. A similar equation can be applied to describe particles in the y-direction, with $x$ replaced by $y$ and $u$ replaced by $v$. Advective velocities ($u$ and $v$) are taken to be positive to the north and east of the model grid and assumed to be spatially, but not temporally, uniform. The horizontal resolution of the model domain is 200 m, and the numerical time step is 50 s. This spatial resolution means that the area of the release is, in effect, modelled as being 200 m x 200 m in extent. A spatially smaller release would result in larger (and more easily detectable) signals than are modelled here, but only near the source. Because of blurring by the diffusion term, the plume does not have to travel far before these fine details of its origin are no longer apparent. Setting the diffusion length scale ($D t$)$^{1/2}$ equal to the model resolution (200 m) gives a time scale $t$ of only two hours, or a distance (at a typical current speed) of less than a kilometer. For larger distances than this, the size of the seep area will not be apparent in the measured concentrations.

To simulate continuous release of CO$_2$ into the model, 5 particles are added to the central grid cell at every time-step. To minimize edge effects, particles are removed from the population only when they have exceeded the model boundaries by > 2 km. The flow field is prescribed from a 120-day time-series of hourly, depth-averaged, $u$ and $v$ velocities collected in Bass Strait, Australia at the location 38S, 148E (Figure 2) between January and May 1991. The hourly velocity data are linearly interpolated onto a 50 s time-grid, and the value of $D$ is kept constant at a value of 5 m$^2$s$^{-1}$ consistent with a plume spatial scale of ~5 km (Okubo, 1971). Particle densities in the model are converted to DIC concentration by assigning equal mass to each of the particles according to the prescribed release rate (i.e. 5 or 10 kt yr$^{-1}$), and taking account of the total volume of one model grid cell (water depth of 20 m).

![Figure 2](image.png)  
**Figure 2.** Location of mooring data (star symbol) and bathymetric contours in the region of Bass Strait, Australia.

### 3.3.2 GLIDER SIMULATION

The modelled plume is sampled to simulate the sort of release signature that would be recorded from a mobile sensor platform, such as a waveglider (Figure 1), instructed to patrol the area. To do this, a 100 km$^2$ rectangular sub-section of the modelled plume is interrogated along a theoretical glider ‘flight path’. It is assumed that the glider moves at a speed of 1 m s$^{-1}$ along a series of 51 north-south parallel tracks, turning at the end of each track, to work its way from one side of the sample area to the other, covering a total distance of 520 km and taking just over 6 days to complete the entire flight. An example flight path is shown in Figure 3; the detailed design of a flight path, taking account of shoreline and no-go areas, is
known (but not trivial) technology and will not be covered here. The north-south tracks in the example are 200 m apart and the plume concentration is recorded once every 200 seconds to provide an overall spatial sampling resolution of 200 m. The plume concentration from a total of 15 flights, consisting of 3 groups of 5, is recorded. Each group of flights has a unique start time relative to the start of the time series of flow data, and each of the 5 flights within a group has a unique starting position relative to the source, to account for the fact that the source location is unknown.

Figure 3. Example of a glider flight path (white line), overlaid on a ‘snap-shot’ of the plume (colour map of DIC concentrations).

3.4 Detection algorithm for glider data

The objective of this section is to estimate the detectability of a release, based on the dispersion modelling described in the last section. The essential aspects of that modelling, from the point of view of detection, are that the glider traverses a number of isolated peaks in DIC, which are quite small spatially (median dimensions of order 1 km) and of variable height. We need to know if some of these peaks will stand out against the variable background level.

3.4.1 LEVELS OF NATURAL VARIABILITY

During 2007-2011, measurements of DIC were made regularly for CSIRO from cargo ships approaching and leaving Melbourne (Appendix A). These ships travelled essentially the same routes over the period (as shown in Figure 4) and measurements were made every 10 minutes, equivalent to approximately 5 km intervals. There is a total of 76 passages in the database.
The most obvious feature of the data is a sharp drop in DIC along the tracks, which corresponds to a front between the cooler Tasman Sea waters and the Bass Strait waters. This front moves to and fro with the seasons and is typically further to the west in the summer. Estimates of the position of the front are shown in Figure 4. Note that these are the positions along the tracks and could be somewhat further east or west closer to shore. The front appears to be to the East of 147.5°E about 85% of the time. In Figure 5, the values of DIC close to 147.5°E are plotted for the inbound and outbound tracks. The seasonality is clear, however with appreciable variations around the Fourier series that has been fitted to the data. This series contains the annual cycle and three harmonics. If the series representation is valid as a long term average, it suggests that waters close to 147.5°E are affected by the passage of the front for about half of each year.

The relevance of the front is that (amongst other properties) the variability in DIC is quite different on either side of the transition between Bass Strait and Tasman Sea waters. As can be seen in Figure 6, to the west of the front there is typically little spatial variation in DIC, whereas to the east the variability is pronounced.
The signature of a release to a glider is a rapid change in DIC across a km or two and so we need to know the natural variability on this spatial scale. From the cargo ship data we can find the distribution of the changes across the shortest sampling interval we have available, namely 5 km. The histogram of the differences between samples 5 km apart is shown in Figure 7, where we have selected data to the west of 147.5°E, that is, in the less variable waters. This gives an estimate of the variability on the scales of interest. For example, in real-world data the off-plume data from a flight could be used to fit a low-spatial-frequency model of the natural background. The fluctuations around this model, on the spatial scales of the plumes, would be approximately as large as the 5-km differences. The standard deviation of the differences is 2.7 DIC units (µmol kg⁻¹), which is close to the expected level of analytical error of a single measurement in this type of measurement. An ensemble of differences between two measurements would be expected to have an error that is sqrt(2) larger than that of a single measurement, so this suggests both that the analytical error is small in these measurements and that environmental variability contributes little to the error. The best-fit to the distribution of the data is a Student’s t-distribution with 3/2 degrees of freedom, which has wider wings but a narrower core than a Gaussian. (Technically, with 3/2 degrees of freedom, the standard deviation is infinite, so the measured standard deviation is sensitive to the outliers.) We will use the fitted t-distribution to generate random noise on the data in a later stage of this analysis.

Figure 6. Under-way data from a sequence of passages early in 2007, showing the movement of the front to the west and the corresponding increase in spatial variability of DIC to the east of the front.

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1 The obvious way to estimate the noise levels on various scales would be via power spectral analysis, but in practice our data series from the cargo ship data typically only contain 10 to 20 samples. Fourier analysis is thus impractical as too much data would be lost to end effects. The spatial differencing approach serves to remove the slower spatial fluctuations which would not affect the detectability of peaks as measured from the glider, and so gives a useful estimate of the relevant noise level.
East of 147.5°E, the region that is more affected by the front, the distribution looks very similar in shape but is appreciably wider, with a standard deviation of 5.6 DIC units. This quantifies what is apparent from inspection of the data, which is that DIC levels are much more variable to the east of the front. This variability is much bigger than can be accounted for by analytical error, and probably reflects the incomplete mixing of two different water masses in this region as well as higher biological production associated with the front.

3.5 Spatial structure of natural variability

As described in the modeling section, from the available data it is possible to construct a variety of “flights” for a glider across the area of surveillance, by varying when the glider starts out (within our window of the flow data that drives the model), and also on where the source is within the area. Some examples of concentrations measured by the simulated glider are in Figure 8. A total of 15 realizations of possible flights is available.

The data recorded by the simulated glider typically take the form of isolated jagged peaks as a function of the along-track distance, as shown in Figure 9. From the point of view of detecting one of the release peaks, what matters is not just the noise level but also the amount of noise on various spatial scales—the spatial correlations of the environmental variability. For the same noise level, noise which is strongly correlated on scales of a few km will have only a small effect on the detectability of a peak compared to noise that is uncorrelated.

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2 We will use the word “flight” to describe the complete to-and-fro sets of traverses of the surveillance area.
The situation is illustrated in Figure 10, which shows uncorrelated (“white”) vs. low-frequency noise with the same standard deviation. Since we only need to see the release peak above its local background, low-frequency noise is better – it may be large, but lifts the peak up on a plateau and does not affect the measurement of peak height very much. White noise (uncorrelated sequential measurements) is a worst case, but may be what applies to our data. As noted, the differences across 5 km are small enough to be completely accounted for by analytical error on each measurement, and these errors would presumably be uncorrelated. However some component of the analytical error might be correlated and related to common conversion errors onto absolute scales. These refinements need further study when more data are available.
To assess the effect of the environmental noise on detection probability, we therefore need to know something about the correlation function of that noise on the spatial scale of the peaks shown in Figure 8 and Figure 9. This spatial scale is 2 – 5 km and a detection algorithm will be affected by correlations in the environmental noise on those scales. Unfortunately the only data we have available are spaced at 5 km intervals, but it is still possible to use these data to make some useful estimates. For data from the west of the front, the scatter in the difference between measurements 5 km apart is close to the nominal analytical error, as previously noted. It follows that there cannot be any significant environmental variability on scales less than 5 km that could be measured above the analytical noise level. The analytical error is (presumably) uncorrelated from one datum to the next, so the situation matches the right panel of Figure 10. For the eastern data, the standard deviation across 5 km is about twice the analytical error. There might thus be correlated noise on scales less than 5 km, resulting in a hybrid of the left and right panels of Figure 10. Since white (uncorrelated) noise is a worst case for detection (for a given noise level), we will assume that east of the front, the situation of the right panel applies, with an error level we will discuss later, but larger than that assigned to the western data.

### 3.6 Detection algorithms

We now consider detection algorithms. Ideally we could use the theory of optimum matched filters, if we knew the power spectra of the signal and noise. For example, one could use a Wiener filter followed by a peak-finding algorithm. Since our knowledge of the spatial structure of the environmental variability is currently incomplete, we will focus on simple algorithms and bounding assumptions. We can obtain pessimistic estimates of detectability by assuming that the noise is white, that is, uncorrelated from one glider measurement to the next, and we know that this noise model cannot be too far from the truth, at least to the west of the front. The method we will use for characterizing the detectability of the loss of containment is conceptually complete, in that it empirically matches the spatial scales of signal and noise. More sophisticated methods would build this matching in a priori, based on fuller background data, but are not essentially different.

A variety of detection strategies are possible: for example, the data from the glider may be assessed in real time as it traverses its area of surveillance, so that it can be programmed to return immediately to areas of suspiciously high DIC. We will test a simpler algorithm, which only assesses the data once the entire area of surveillance has been traversed (which as noted before, takes about 6 days).

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3 This follows from the result that the integral of the power spectrum of a noise process is equal to its variance. This is a corollary of the Wiener-Khintchine theorem.

4 Wiener filters are a standard tool in signal analysis; there is a good Wikipedia article about them.
Clearly the morphology of the release plume is quite complex. The appearance of Figure 8 suggests a simple detection algorithm which uses the spatial information. It simply picks the peak DIC in a flight and calculates the average DIC in some region near that peak in the data. The algorithm’s effectiveness can be evaluated by the following simulation steps.

1. Create a “noisy” flight that includes the effects of measurement error and environmental noise by adding to a flight uncorrelated white noise, drawn from the distribution (solid line) of Figure 7.
2. Locate the peak DIC in this noisy flight and compute the average DIC measured by the glider within some small region around that point.
3. Declare a detection if the average exceeds some threshold value.
4. This part of the simulation, if repeated many times for different realizations of flights and noise, gives the probability of detecting a release if one is indeed present. This is the sensitivity or statistical power. Carrying out a similar procedure when there is no release, only environmental noise in the measurements, gives the false alarm rate.

The amplitude of the release (in tonnes per year) is an adjustable parameter in these calculations.

Picking the size of the averaging region corresponds to a pragmatic matched spatial filter. Trials were made with the radius of the averaging region being 250 m, 500m, 1 km and 2 km. The 2 km performed best, but is at the limit of the size where assuming the background variations to be white noise seems valid.

This naïve algorithm could be improved upon by detailed familiarity of the patterns of currents in the actual area of surveillance to set the size and shape of the averaging area. Figure 11 shows estimates of the probability distributions of the average values (taken over a radius of 2 km from the maximum), both for the ensemble of 15 flights exemplified in Figure 8, and also for the no-release situation (in which all measurements are pure measurement error). These results are calibrated to a release rate of 10,000 tonnes per annum.

\[ \text{Figure 11. The probability density of the near-peak average DIC, for the cases where a 10 kt yr}^{-1} \text{ leak is present (orange) and where there is only measurement error and environmental variability (blue).} \]

The integrals of each of these curves, above a threshold in DIC, gives the false alarm rate (blue curve) and the power for that threshold. A plot of power against false alarm rate is a parametric curve, with the parameter being the chosen threshold; for historical reasons it is sometimes called the “receiver operating characteristic” (ROC) which reveals its origins in radar targeting. Figure 12 shows the ROC for 10 kt yr\(^{-1}\) and 5 kt yr\(^{-1}\) release rate; note these results (shown with solid lines) are for the case where only one flight would be executed before a decision would be taken on whether a release were present or not.

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5 This is a well-known concept in the theory of detections; there is a good Wikipedia article about it.
The ROC diagram is a useful aid to decision-making in detection problems. In the present case, a site operator would typically first set an acceptable false alarm rate. This is the starting point because releases are not expected, and if one seems to be found, a priori it is likely to be a random excursion above the baseline. It is desirable for the false alarm rate to be low; suppose the operator chooses 0.01. This rate implies a threshold in measured DIC above which a release will be suspected; and if no release is present, the threshold would only be exceeded in one flight out of a hundred. The relevant threshold can be calculated by integrating the blue probability distribution in Figure 11. This means that if the situation is as the operator expects (no releases) then the results from glider surveillance will only prompt further investigative action in one flight out of 100.

From Figure 12, the operator can now read off the power that applies at this threshold; the power gives the probability that the threshold will be exceeded if a release is indeed present. If the release is 5 kt yr\(^{-1}\), the power is only 0.25. This means that 75% of the releases at this level will go undetected; they will not be seen above the chosen threshold, and no alarm will be raised. A larger release (10 kt yr\(^{-1}\)) gives a better power, but still only 0.6. The only way to get a better result for the power is either to compromise on the false alarm rate – with its attendant consequences – or to improve the noise level. Since this is fixed by the combination of environmental noise and analytical error, the option exists to combine the results from several flights to reduce the effective noise level.

Extending these results for the combination of data from more than one flight would ideally use a series of sequential modelled flights, as there will be strong correlations in the morphology of the plume over short periods which affect how the signal builds up over the noise. We do not have a long enough modelled time span to do this. A conservative estimate for performance in this case can however be obtained by a small generalization of the notions of false alarm rate and power. Suppose \(p\) is the false alarm rate for one flight, \(q\) is the power, both at some threshold, and we wish to combine \(N\) flights. The quantity \(1-(1-q)^N\) is the probability that the measurement from at least one of the \(N\) flights is above the threshold, if there is a release present, and \(1-(1-p)^N\) is the probability that at least one flight exceeds the threshold when there is no release. This models a situation where a potential release is suspected if one of \(N\) flights showed an anomaly above the set threshold. Because the calculation assumes statistical independence, it is also necessary to assume that the \(N\) flights in question are well spaced in time rather than sequential. The generalised ROC curves for this case are shown in Figure 12 as dashed lines for the case \(N=4\).

We now see the expected improvement; at a stipulated false alarm rate of 0.01, the power for the 10 kt yr\(^{-1}\) case has risen to about 0.95. In principle, further improvement could be obtained by stacking more flights, but it is hazardous to assume this can be done ad infinitum for real data. Four flights, spaced in

![Figure 12](image.png)

**Figure 12.** The ROC is plotted for the cases of 5 kt yr\(^{-1}\) (blue) and 10 kt yr\(^{-1}\), (red). The dashed lines show the ROC from stacking 4 flights.
time, might occupy two or three months and this is close to the amount of front-free time that is available in any given year.

### 3.7 Summary and Discussion

Overall, the impression from the ROC analysis is that a workable detection strategy (acceptable power and false alarm rate) should be possible at around a release rate of 10 kt yr\(^{-1}\), but would be marginal at 5 kt yr\(^{-1}\). These are for observations made in the westerly area, unaffacted by the front. As a check, we note that Figure 6 suggests a noise level of around 3 DIC units, and the peaks in Figure 7 range from 2 to 7 DIC units for the 10 kt yr\(^{-1}\) case; so we would expect this size of release to be detectable, but not convincingly so, unless data were stacked in some way.

The situation east of the front may be less favourable, as the environmental noise seems likely to be at least double the values assumed here for the westerly case. With the worst-case assumption of white noise, this would make release rates of 10 kt yr\(^{-1}\) hard to detect for about half of the year (Figure 7).

A leak of this size, if sustained, would be “large” in the sense that it would exceed the estimates of the leaks that are tolerable from CCS sites while still retaining a climate benefit (e.g. Enting et al. 2008). However such a large leak would surely be “designed out” of any foreseeable storage site. Unless the detection limit can be greatly reduced upon further analysis (especially of analytical error and background variability) it seems that monitoring in the water column is unlikely to be relevant to containment monitoring, although it may have utility in determining baselines and providing episodic environmental impact monitoring.

These results, as emphasized, are for quite simple algorithms, with limited information on background variability, and modelling over a limited time span. More comprehensive baseline data would likely lead to more refined detection algorithms that were tailored to exploit particular features of the dispersion in the area under surveillance. These refined algorithms would be expected to have more sensitivity, although how much more is hard to say without more data on the character of the background.

Our assumption of a large “patrol area” is an initial assumption, since we have no detailed spatial information about what “targets” might have to be monitored. Once this is known, one might consider a strategy where a waveglider moved between a few targets and remained “on station” near each for a while. However an advantage of a spatially-extensive survey area is that it probably would help in better defining the spatial character of the background, which in turn would improve detection thresholds.

The quoted analytical error limits take into account the translation from measured quantities (pH, temperature, etc) to DIC. They also take into account the systematic errors that arise in placing the measurements on an absolute scale. For detection (not quantification) being on an absolute scale is unnecessary and the relevant level of analytical error might be much smaller than quoted, improving detection limits by up to a factor of 10. This requires further study. The quoted limits assume a noise distribution as measured from the ship-of-opportunity data; these include the level of environmental variability on few-km scales. More spatial information is needed about the background variability to refine the calculations.

An important assumption of the dispersion model is that the release is spatially small; given the size of the computational elements in the model, however, a “small” release might extend over many hundreds of square meters and still behave as a point source to the model. Hence, while the limiting rates of a release are large, in terms of fluxes per square meter they need not be. For example, if we assume that a release occurs over one spatial pixel of the model (200 x 200 m\(^2\)) at a rate of 10 kt yr\(^{-1}\) we can convert this (assuming parameters in Table 2, section 4) to a flux of of about 250,000 bubbles s\(^{-1}\) leaving the sea floor (equivalent to 10 sources of 25,000 bubbles s\(^{-1}\)) or 6 bubbles m\(^2\) s\(^{-1}\). This is a very large flux of bubbles and would be trivial to detect by the acoustic methods that are the topic of the next section of this report.

In closing we note that the issue has been phrased so far in terms of the ability of a chemical method to detect a release of CO\(_2\) into the water column. Quite large amounts of CO\(_2\) are needed to cause changes that rise above natural variability. This shows that (except possibly locally to a release site) CO\(_2\)-related environmental impacts of a loss of containment are likely to be small except for very large releases. It also
suggests that, if monitoring of the water column is required to check for environmental impact, it will be important for well-understood baselines to be in place and for there to be a clear understanding and acceptance of the likely false alarm rate if anomalies are interpreted as evidence of release.
4 Acoustic detection and quantification of CO$_2$ gas bubbles in sediment and the water column

4.1 Introduction

The scope of this chapter is a preliminary investigation into the use of acoustic methods to detect and quantify CO$_2$ gas in the seafloor (top ~50 mm) and water column. Hence, a clear focus in this preliminary study is on quantification of the detection capability of some typical acoustic instruments. In this way, informed decisions can be made when planning a monitoring strategy where trade-offs of cost and measurement precision will be required. The region of interest for monitoring is expected to be 10 km in length and extend from 5 to 30 m in seabed depth on an exposed coastline. The seabed sediments on an exposed coast with variable wave energy can be of size class fine sand to sandy gravel (Jones and Davies, 1983). We focus on measuring CO$_2$ gas either in the very top (surficial ~50 mm) layers of the sediment as a bubble layer or as a bubble stream in the water column. A layer of trapped bubbles in the sediments could indicate the presence of a gas release site which is not continuously producing bubble streams (Fonseca et al., 2002). For an active gas source there would be a bubble stream in the water column which we assume here is a continuous stream, although changes in hydrostatic pressure (e.g. tides) have been noted to cause some natural seeps to become intermittent. The bubble streams could have a single source or multiple sources and the mean gas bubble size and rate of ascent is assumed to be as previously measured for a shallow site (Sellami et al., 2015). We assume that larger bubbles (7.5 mm) remain intact for at least 2 m within the water column and do not dissolve over that depth range (Schmidt et al., 2015). Flow rates for detection of bubble streams with mean diameter 7.5 mm (Sellami et al., 2015) were assigned as total release of small (1 tonne of CO$_2$ gas per year) or large (>10 tonnes of CO$_2$ gas per year).
Two acoustic technologies have been assessed: multi-frequency echosounders and multibeam sonars (Figure 14). Multibeam sonars and echosounders send out a pulse of sound and record the backscattered echo as a function of time. Here we have considered currently available technologies including the Kongsberg EM2040 multibeam and single beam, multi-frequency (38, 70, 120, and 200 kHz) Simrad EK60 echosounders. Other instruments such as broadband echosounders, passive acoustics, side scan and sub bottom profilers have not been specifically considered although they are discussed in chapter 5 regarding wider requirements for a monitoring strategy. These wider requirements may involve a site selection survey for geological characterisation that identifies “hot spots” for potential leaks. In this way a sub-bottom profiler would be useful to look at sediment and potential gas pockets deeper than the 100 mm depth investigated here and outside the scope of this investigation.

It is assumed that these instruments would be fitted to a small vessel or an autonomous underwater or surface vehicle (AUV or ASV) surveying in a grid pattern over the area of interest. We have also assumed that a “ground truth” drop camera and remotely operated vehicle (ROV) would be available to inspect release sites should they be discovered (Table 1).
The acoustic ability to detect and quantify CO₂ gas bubbles in sediment (top 100 mm) and the water column has been investigated using forward and inverse approaches. The forward approach involves using backscattering models of the seafloor and water-column at various frequencies to predict the acoustic signal from an assumed gas bubble size and density. An inverse approach is investigated by looking at data and evaluating how to interpret the cause using acoustic, geological or biological knowledge. Applying the inverse approach in the field requires ground truthing (video is normally used although water turbidity can restrict this use - Table 1)
To explore a potential release site requires a survey design that can either be exhaustive or sampling based (Anderson et al. 2007). For seabed mapping it is common to use a multi-beam sonar to create an exhaustive map of the terrain at a defined resolution that is determined by both the instrument configuration and the environment. For seabed and water column classification the echo return from the transmitted pulse (backscatter) is used and this may require alternative sampling designs. Importantly there needs to be consideration of the level of “ground truth” sampling and the temporal variability at diel, tidal, lunar, seasonal or episodic storm event time frames (Figure 15, Anderson et al. 2007). Understanding the effects of temporal variability in seabed classification maps is an area of current research. A typical approach with seabed and water column mapping is to either exhaustively or sub sample the area of interest and identify targets (features) of interest. A range of “truthing” methods can be used to evaluate (classify) the composition of targets and map and quantify their distribution over the region. Finally, for classified gas laden sediments and bubble streams, the composition of the gas and its origin (i.e. biological, natural geological, sequestered CO₂) needs to be evaluated. An ROV to sample the gas would be required if this task were to become necessary.

The following sections summarise the methods and results of the forward and inverse approaches to detecting CO₂ gas bubbles with acoustics.

4.2 Modelling Methods and Results

The ability to detect gas laden sediments and bubble streams can be influenced by the multi-beam or single beam instrument type, its frequency band and the seabed and water column environment. In order to provide a preliminary assessment of the detection capability of the instrument some detection thresholds assumptions have been used based on a limited review of historic data in the region. These estimates of background noise and background biological scattering are reported in the text but need further clarification on their temporal and spatial variability that is outside the scope of this analysis.

Summary of threshold terminology used:

- Background noise is the volume backscatter noise level that above which an acoustic reflection can be easily detected from other non-reflected interference (-70 dB m⁻¹ used in this report).
- Background side lobe noise is the volume backscatter noise level that above which a bubble stream can be detected above the noise interference of side lobe reflections from the seabed (20 dB above the instruments background noise).
- Bubble stream detection threshold is the level of signal above which biological scatter is not normally experienced based on available historic data (-30 dB m⁻¹ used based on historic data of fish schools in the region, need to monitor this background level).
Seabed feature detection threshold of -2 dB based on Kloser et al. (2010).

A survey platform and strategy is needed to use the instruments to detect gaseous sediments and bubble streams. Typically, acoustic surveys are done from vessels steaming in a grid design for seabed mapping where the transect spacing is designed to maximise coverage and detection of targets being surveyed. The spacing of transects is optimised for the detection objectives and for seabed mapping will vary from being spaced at 2.5 to 5 times water depth. For single beam surveys, the spacing of transects is determined by the resolution of the feature and the need to reduce autocorrelation. For large fish schools, transects can be spaced between 0.5 km to several km. We will highlight in the following that to detect bubble streams a grid design would need to be closely spaced and would be best used in combination with the water column backscatter from the multibeam.

The advantage of small vessels for doing the surveys is that they are also needed for the verification of targets and can carry out multiple missions in assessing potential gas releases. For both seabed and water column targets there will be ambiguity about their source and “ground truthing” usually with video is required on weaker, less-defined targets.

4.2.1 GAS-LADEN SEDIMENTS - SEABED SCATTERING MODEL

In this section we explore the detectability of gas-laden sediments that may be associated to a gas release that is not continuous. For seabed scattering the APL_UW (1994) model is used to explore the incidence angle detection of gaseous sediment. This model will be used to explore the detectability of gaseous sediments using known multi-beam sonar equipment at frequencies of 30 kHz, 200 kHz and 400 kHz operating at incidence angles of 0 deg. (normal incidence) to 70 deg. incidence.

The APL-UW (1994) seabed scattering model combines the most dominant dimensionless seabed scattering mechanisms of homogeneous sediment volume scattering coefficient $S_v$ and surface roughness coefficient $S_s$ as a superposition of incoherent scatter to estimate the seabed backscattering strength $S_b$, for incidence angle $\theta$ where:

$$S_b(\theta) = 10\log_{10}[S_v(\theta) + S_s(\theta)] \text{ dB}.$$  

To detect gaseous sediments, the volume scattering of the seabed surface will need to exceed the roughness scattering. Volume scattering from gaseous sediments will depend on the frequency, seabed depth, density and size of gas-bubbles and how close within the sediments to the seabed surface they are. We assume in this work that the gaseous sediment volume scattering is in excess of the un-gaseous sediment volume scattering by 5 dB irrespective of frequency (Fonseca et al., 2002). This equates to gas volumes in the sediment of between 0.03 to 5 %. Actual measurements are needed to define this more closely as it depends on bubble size, sediment type, seabed depth and frequency. Using the APL-UW (1994) model it is possible to explore when volume scattering is in excess of roughness scattering with and without bubbles in the sediments.

Based on the APL-UW (1994) model, the frequency was set to 30, 200 and 400 kHz and the data were separated into the model seabed types with acoustic properties derived from a synthesis of historic physical seabed samples for rock, coarse sand, muddy sand and sandy mud (APL-UW 1994, table 3.2). It is assumed that volume scattering (Sv) is detected over surface roughness and that gas laden volume scattering is assumed to increase by 5 dB and will be detected when $S_v - S_s > -2 \text{ dB}$. These assumptions need to be tested but based on previous habitat mapping work a 2dB detectability is not unreasonable (Kloser et al. 2010)

The model shows that roughness backscattering dominates volume backscattering for rock and coarse sand at all incidence angles from 0° (normal incidence) to 70°and all frequencies (Figure 16). Volume scattering dominates the seabed backscattering for muddy sand and sandy mud at incidence angles greater than normal incidence but is frequency dependent. At low frequencies (30 kHz) the volume backscattering is in excess of roughness scattering from 6° for muddy sand and 11° for sandy mud. At 400 kHz volume
backscattering is dominant between 30° and 62°. When gas-bubbles of volume contribution of 5 dB are added the detection at 400 kHz in sandy mud increases from 15°. Gaseous sediments would not be detected in rock or coarse sand habitat at any of the frequencies if the gaseous bubbles only contributed an extra 5 dB to the backscatter (Figure 16). Hence, if the site has large sediment grains then the detectability of gassy sediments will be low.
Figure 16 Surface roughness backscattering (Ss dB solid) and volume backscattering (Sv dB dotted) at a) 30 kHz, c) 200 kHz and e) 400 kHz for seabed types of rock, coarse sand, muddy sand and sandy mud and incidence angles of 0° to 70°. Decibel difference of volume backscattering and surface roughness backscattering at b) 30 kHz, d) 200 kHz and f) 400 kHz for the 4 seabed types and incidence angles of 0° to 70°. Volume scattering is assumed to be the detected dominant scattering when Sv-Ss is greater than 3 dB (red line). When gaseous bubbles are in the sediment and contribute 5 dB this would be detected over roughness scattering for values higher than the black line.

4.2.2 GAS BUBBLE STREAM- WATER COLUMN MODEL METHODS AND RESULTS

In this section we explore the detectability of continuous release from a gas bubble stream using multi-beam and echo sounder equipment. To investigate the detection of a water column CO₂ gas bubble stream at frequencies from 38 to 400 kHz the Clay and Medwin (1977) bubble scattering model is used. The important inputs to the model are bubble radius, depth and density as well as the expected resonance factor. The model was used to predict the gas-bubble target strength (TS) at 1 kHz to 300 kHz at depth z metres for equivalent spherical radius, $a_s$ from 0.05, 0.1, 0.5, 1, 5 and 10 mm. The target strength $TS_f$ at frequency $f$ in dB form is $TS_f = 10 \times \log_{10}(\sigma_{bs})$ for the back-scattering cross-section, $\sigma_{bs}$, derived from,
\[ \sigma_{bs} = \frac{a_s^2}{\left(\frac{f_{RF}}{f} - 1\right)^2 + \delta_{RF}^2} \text{ m}^2, \]

at damping factor \( \delta_{RF} = \frac{1}{50} \) and \( \frac{1}{7} \). The resonance frequency, \( f_{RF} \), was derived from,

\[ f_{RF} = \frac{1}{2\pi a_s} \left( \frac{3yP_A}{\rho_A} \right)^{0.5} \text{ Hz}, \]

where the ratio of specific heat \( \gamma = 1.4 \), ambient pressure \( P_A = 10^5(1 + 0.1z) \text{ Pa} \) at depth \( z \), \( \rho_A = 1028 \text{ kg m}^{-1} \) and \( a_s \) is the equivalent spherical radius of the gas-bubble volume in metres as described by Clay and Medwin (1977) (p.221).

The results show that small bubbles can be resonant at frequencies less than 150 kHz and produce high backscatter (Figure 17). Hence small bubble streams could effectively look like larger sources depending on the frequency used and bubble size. This resonance backscatter increases as the dampening factor decreases. As many small fishes and zooplankton can be gas bearing, this resonance effect emanating from these organisms could be interpreted as a bubble stream, i.e. a school of fish can look like a bubble stream. Of note is that for larger bubbles with diameter > 5 mm there is no resonance effect in the frequency range of interest because such bubbles produce a high signal over the entire frequency range. Fish in the small to medium size range also have gas bladders and, depending on their schooling behaviour, could also be interpreted as a bubble stream. We have set a -30 dB volume backscattering threshold for detecting a bubble stream. This is equivalent to a jack mackerel fish school (common to this region of estimated -42 dB target strength) with a packing density of 16 fish per cubic metre. This level of packing density and signal strength has not been observed from the data reviewed from the Bass Strait. The -30 dB threshold should be a conservative threshold to ensure that the source is detected with minimal false alarms.
Figure 17 Backscatter (BS) of bubbles of size 0.05 (blue), 0.1 (black), 0.5 (red), 1 (green), 5 (yellow) and 10 (pink) mm at a-b 10 m, c-d 20 m, e-f 30 m for frequency ranges of 1 to 300 kHz with a Q of 50 (a,c,e) and Q of 7 (b,d,f) based on the Clay and Medwin (1977) resonance bubble scattering model.
4.2.3 DETECTION OF BUBBLE STREAM

The expected backscatter for a continuous release of CO$_2$ in the water column is derived for sources of 1 tonne y$^{-1}$, 10 tonnes y$^{-1}$ and 1000 tonnes y$^{-1}$ based on the model results of section 2.2.2. We assume that the median bubble size is 7.5 mm and that the depth is 20 m as observed by Sellami et al. (2015).

<table>
<thead>
<tr>
<th>Variable</th>
<th>Units</th>
</tr>
</thead>
<tbody>
<tr>
<td>CO$_2$ density</td>
<td>1.9 kg m$^{-3}$</td>
</tr>
<tr>
<td>bubble size</td>
<td>0.75 cm</td>
</tr>
<tr>
<td>volume</td>
<td>0.220893 ml</td>
</tr>
<tr>
<td>ascent rate</td>
<td>0.3 m s$^{-1}$</td>
</tr>
<tr>
<td>number of sources</td>
<td>1</td>
</tr>
<tr>
<td>depth</td>
<td>20 m</td>
</tr>
<tr>
<td>pressure</td>
<td>3 atm</td>
</tr>
<tr>
<td>Pulse length</td>
<td>0.5 ms</td>
</tr>
<tr>
<td>Sound speed</td>
<td>1500 m.s$^{-1}$</td>
</tr>
<tr>
<td>Target strength</td>
<td>-50 dB re m$^2$</td>
</tr>
</tbody>
</table>

Table 2 Input parameters for the expected backscatter from continuous release

Based on input parameters, the continuous flow rate for a 1 and 10 tonne y$^{-1}$ release is 25 and 252 bubbles s$^{-1}$, respectively. This results in an acoustic volume backscatter of -35 dB and -25 dB respectively. If the release was 1000 tonnes y$^{-1}$ from a single continuous source, the flow rate would be 25,183 bubbles s$^{-1}$ and the backscatter -5 dB. This is well above the expected background volume reverberation noise of -70 dB m$^{-1}$ for the depths and frequencies considered in this chapter (see section 4.2.4). For a low false alarm rate where the source signal could be easily distinguished from other biological signals a -30 dB threshold has been used (Figure 18). Based on this threshold a single continuous bubble stream release site could be detected if it was greater than 3 tonnes CO$_2$ per year. If there are ten release sites of equal flow then the detection capability would be 30 tonnes CO$_2$ per year (Figure 18).

Figure 18 A) The expected number of 7.5 mm bubbles per metre in a continuous stream for a release rate of 1 to 10,000 tonnes of CO$_2$ per year and B) the expected volume reverberation based on inputs from for 1, 10 and 50
sources within the survey area. A release volume reverberation detection threshold of -30 dB m$^{-2}$ is shown. A release is expected to be easily detected above this biological scatter threshold with a low false alarm rate.
4.3 Examples from multi-beam and echo sounder instruments

In this section we investigate acoustic data from shallow depths (5 to 50 m) and evaluate how to interpret the cause, i.e. the inverse approach. To illustrate the challenge addressed by the inverse approach we show two examples of data collected with a multi-beam sonar and single-frequency echo sounder. These examples are hypothetical and are designed to illustrate the issues of data collection and interpretation. Actual data from a gas release site is required to do this more rigorously.

4.3.1 SEABED BATHYMETRY AND BACKSCATTER FROM A MULTI-BEAM EXAMPLE

The multi-beam sonar has three data sources of interest being the seabed bathymetry, seabed backscatter (reflectivity) and water column backscatter. We illustrate the potential for CO₂ source detection for each of these data types from data collected with an EM2040 multi-beam in shallow water over hard and soft substrate (Figure 19). The site was surveyed from 3 to 28 m covering 0.25 km² in 2 hours with a swath width of 4 times water depth. The bathymetry (Figure 19 a) highlights the fine scale topography of the region and, together with the backscatter referenced to 40° incidence angle, enables inferences of the seabed type to be made (Figure 19 c). Seabed backscatter referenced to 40° incidence angle is a processing method that removes the seabed backscatter incidence angle and allows for maps to be created that enable easy visualisation of distinct seabed features from the backscatter intensity alone (Kloser and Keith, 2013; Kloser et al., 2010). Within the shallow, high backscatter (> -27 dB) reef area, detection of gaseous sediments would not be possible as we have demonstrated with the model where the roughness scattering would exceed the volume scattering. However, in the deeper, softer substrate (sandy mud), where the surrounding backscatter is ~ -39 dB, elevated backscatter features of ~ -27 dB are apparent. One feature, a circular depression, could be indicative of gaseous sediments and would trigger a need for a sampling event (for example, using video) (Figure 19 a and c). Hence, if the sediment is soft, “gaseous sediments” can be detected with a multi-beam sonar. The combined backscatter and bathymetry image highlights the complexity of the reef region (left of image).

In summary the example shows that for soft sediment where the surrounding backscatter is low it is easier to detect the potential of gas laden sediments but these features need to be “ground truths” with video and other physical sampling (e.g. grabs and sediment profile cameras) to verify. Inclusion of a sub-bottom profiler in this survey would provide more evidence of the release-detection potential of particular sites. After detection of gas laden sediments, the cause would need to be ascertained, which may or may not be attributed to CO₂ release.
Figure 19 Example of data collected from the EM2040 multi-beam operating at 300 kHz and 1.3° nominal resolution over a mixture of rock and sand habitat for a) bathymetry, b) raw backscatter, c) backscatter referenced to 40° BS40 and d) backscatter overlay on the bathymetry, 1 m grid, 5m contours and 0.5 km scale bar. Note the higher backscatter (dark) indicative of sediment bubbles or hard substrate in the circular depression that would need to be “ground truthed” with video and other sampling gear.

4.3.2 WATER COLUMN BACKSCATTER FROM A MULTIBEAM EXAMPLE

The multi-beam sonar also has water column capability and in principle this should provide 100% coverage of the water column signal above the seabed when doing an exhaustive survey of the seabed bathymetry
and backscatter. Figure 20 shows that, due to how the instrument forms its sampling beams, the water-column echoes are contaminated with between-beam side lobes. This contamination decreases the detection capability of bubble streams of the instrument for bubble streams that are not directly underneath the survey path.

**Figure 20** Highlight of the feature in the centre right of figure 3 suspected of being gassy sediment. The a) bathymetry (centre right in Figure 3) shaded by backscatter at 0.2 m resolution, 1 m contours, 0.1 km scale bar highlights the feature. Using the multi-beam water column data it can be seen that there is a fish school in the centre of the 0.4 m deep depression and that the side lobes mask echoes in the water column off the central beam.

### 4.3.3 WATER COLUMN BACKSCATTER FROM A SINGLE BEAM EXAMPLE

In contrast to the high side lobes of the multi-beam a single beam echo sounder has very low side lobe contamination and lower noise threshold. Figure 21 highlights a key problem for detecting low flow releases where in this instance a small fish school located close to the seafloor of volume backscatter less than -34 dB could be interpreted as a gas bubble stream. This feature may need to be verified either using temporal sampling (e.g. resample the feature periodically) or video/sonar “truthing”. The use of an underway current measurement system (i.e. Acoustic Doppler Current Profiler, ADCP) would also assist to interpret the non-vertical features. In this report we have set the bubble stream threshold to -30 dB to ensure a low false alarm rate where biological features can be more easily separated (Figure 18). Alternatively if more instruments (side scan sonars and sub-bottom profilers) and expertise (geological and biological) are available for the surveys then potential features could be fully assessed at the time.
4.4 Summary and Conclusions

Based on our preliminary analysis and our experience with the instruments and application in field surveys we make the following general conclusions about the ability of acoustic instruments to detect gaseous sediments or continuous gas bubble streams from single or multiple sources at flow rates of 1 tonne year\(^{-1}\) or 10 tonnes year\(^{-1}\) contained within the near shore 5-30 m water depth (Table 3). Flow rates greater than 10 tonnes year\(^{-1}\) and of a single source would be easier to detect.

<table>
<thead>
<tr>
<th>Situation</th>
<th>low flow 1 tonne y(^{-1})</th>
<th>high flow 10 tonnes y(^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>single source</td>
<td>multiple sources</td>
</tr>
<tr>
<td><strong>Pros</strong></td>
<td>Can detect with instruments, depending on the biological background.</td>
<td>Can detect with instruments, depending on the biological background acoustic signal. Many sources increases the chances of finding a low level source within the low noise portion of the acoustic sampling volume.</td>
</tr>
<tr>
<td><strong>Cons</strong></td>
<td>A single source requires 100% coverage to find. If flow is not continuous then sampling design will require a temporal component for detection.</td>
<td>Due to the low level of flow, it may be necessary to verify a number of false targets (biological derived acoustic signals), e.g. with camera equipment.</td>
</tr>
</tbody>
</table>

Table 3 Summary of acoustic instruments to detect low flow and high flow continuous CO\(_2\) gas bubble streams within a confined survey area.

Given the situations outlined in Table 3 the specific attributes of multi-beam and single beam echo sounders are outlined in Table 4.
<table>
<thead>
<tr>
<th>Instrument</th>
<th>Flow/sites</th>
<th>Method</th>
<th>Pros</th>
<th>Cons</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seabed designed multi-beam</td>
<td>low flow (&lt;1 tonne per year)</td>
<td>Using the water column backscatter from the instrument to detect bubble streams provides potential to see gas emission sources with bathymetry and changes in seabed backscatter.</td>
<td>Possibility of detection for a single source.</td>
<td>Potential for many false targets to identify for a single source, with high noise from side beams making this a visual inspection task.</td>
</tr>
<tr>
<td>Seabed designed multi-beam</td>
<td>high flow (&gt;10 tonne per year)</td>
<td>Using the water column backscatter from the instrument to detect bubble streams provides potential to see gas emission sources with bathymetry and changes in seabed backscatter.</td>
<td>Water column backscatter should easily detect the bubble stream from single and potentially multiple (10-50) sources. Should be few false alarms at this rate.</td>
<td>As the number of sources increase (&gt;100) there would be difficulty in detecting all sources due to side lobe interference.</td>
</tr>
<tr>
<td>Single beam echo sounder</td>
<td>low flow (&lt;1 tonne per year)</td>
<td>Using water column backscatter, can detect a single source with low false alarm rate. Due to limited beam width, an effective sampling strategy will be needed to detect a source.</td>
<td>High detection capability, able to detect if there are multiple sources with the right sampling design</td>
<td>Due to low spatial coverage and need for a sampling design may miss a single source due to insufficient coverage. Multiple sources could provide many false targets due to low signal levels.</td>
</tr>
<tr>
<td>Single beam echo sounder</td>
<td>high flow (&gt;10 tonne per year)</td>
<td>Using water column backscatter, can detect a single source with low false alarm rate due to other biological sources. Due to limited beam width, an effective sampling strategy will be needed to detect a source.</td>
<td>High detection capability able to detect if there are multiple sources with the right sampling design</td>
<td>Due to low sampling may miss a single source due to insufficient coverage. Due to coverage, will not detect all sources but sampling strategy may estimate the number with a sampling variance.</td>
</tr>
</tbody>
</table>

Table 4 Attributes of multi-beam and echo sounders to detecting gasy sediments and bubble streams

To detect gaseous sediments using backscatter from multi-beam sonar equipment at 30 kHz, 200 kHz or 400 kHz requires the volume backscattering component to be in excess of the roughness scattering backscatter component. The APL-UW model predicts that gaseous sediment cannot be detected in rock and coarse sand sediment types as roughness backscatter dominates the volume backscattering for all incidence angles from 0 to 70°. In muddy sand habitat the gaseous sediment can be detected at a wide range of incidence angles depending on the frequency. At higher frequencies the volume scattering decreases and gaseous sediments cannot be detected over as wide range of incidence angles. We estimate that the volume fraction of gas that can be detected is between 0.03 to 5 % of the near surface sediment. Actual measurements are needed to define this more closely as it depends on bubble size, sediment type, seabed depth and frequency.

A survey platform and strategy is needed to use the instruments identified in Table 4 to detect gaseous sediments and bubble streams. Typically acoustic surveys are done from vessels steaming in a grid design for seabed mapping where the transect spacing is designed to maximise coverage and detection of targets being surveyed. Alternatively other platforms such as autonomous underwater vehicles or wave gliders can be deployed (Table 5). The advantage of small vessels are that they are also needed for the verification of targets and can carry out multiple missions in assessing potential gas releases.
Depending on the technique, grid spacings are usually a few times the water depth. This means that acoustic surveys would need to be focussed on relatively small areas and hence good information is needed on the likely location of monitoring targets.

<table>
<thead>
<tr>
<th>Deployment method</th>
<th>Pros</th>
<th>Cons</th>
</tr>
</thead>
<tbody>
<tr>
<td>Small vessel</td>
<td>Flexible for multiple operations of surveying and sampling; can supply adequate power for long survey missions; relatively high speed</td>
<td>Weather dependent for data quality</td>
</tr>
<tr>
<td>Autonomous underwater vehicle</td>
<td>Can survey a fixed track autonomously; good for repeat operations</td>
<td>Usually still need a small vessel and has limited power and range. May need to use side scan sonar to detect bubble streams because of shallow depth.</td>
</tr>
<tr>
<td>Wave glider</td>
<td>Autonomous operations, potentially good for repeat operations.</td>
<td>There has been some deployment of acoustics on wave gliders, however, adjusting the power balance for active acoustic systems will require work to optimise deployment time frame. Comments re depth apply as above.</td>
</tr>
</tbody>
</table>

Table 5 Comparison of survey platforms to sample a region using acoustic instruments.
5 Prototype Monitoring Strategy

5.1 Introduction

Developing fit-for-purpose marine monitoring and verification (M&V) strategies for offshore CCS requires a clear articulation of objectives. Blackford et al. (2014) framed these objectives in a four-component hierarchy: detection, attribution, quantification and impact. We modify this framework here recognising that activities focused on monitoring the properties of a seabed CO₂ release have different considerations from those focused on monitoring for environmental impact, thus detection, attribution and quantification are objectives for both sets of activity, as summarised below.

<table>
<thead>
<tr>
<th>High-level goal</th>
<th>Monitoring for a seabed CO₂ release</th>
<th>Monitoring for environmental impact</th>
</tr>
</thead>
<tbody>
<tr>
<td>Objectives</td>
<td>Detection of anomalies indicative of CO₂ release</td>
<td>Detection of changes within the environment</td>
</tr>
<tr>
<td></td>
<td>Attribution of the anomalies to a source of released CO₂</td>
<td>Attribution of the environmental changes to CCS or non-CCS related activities</td>
</tr>
<tr>
<td></td>
<td>Quantification of the magnitude of the release, either as a rate or a mass.</td>
<td>Quantification of the extent of the impact</td>
</tr>
</tbody>
</table>

Table 6 Objectives for monitoring seabed CO₂ release and environmental impact

This study has reported analyses informing the high-level goal of monitoring for a seabed CO₂ release, and its associated impact on water chemistry with a specific focus on detection and attribution. For detection to be cost effective, monitoring approaches must be able to clearly identify target signals with low false alarm rates, thus also addressing attribution. While the approaches we have considered may also have use for quantification, these are not explored further in this report.

If storage reservoirs are chosen with good geological integrity, the chance of a loss of containment is extremely low. However, non-leakage related environmental change in offshore coastal environments is inevitable during the long period of CO₂ storage and is highly likely even within the first decade of storage. Thus, the attribution of environmental impacts will be a necessary requirement of M&V even in the absence of a release. A monitoring strategy for loss of containment has to be based in rigorous risk assessment, so that it is clear what one is looking for. Otherwise, it is difficult to claim that it has not been seen. Monitoring to check for environmental impact (or the lack of it) is logically different: there may well be no impact even if there is a loss of containment. This is because impact is a function of biological response (as well as regulatory definition) and the effects of a release may be below a biological (or regulatory) threshold.

In this section we make recommendations on the technologies we consider important for implementing M&V in shallow, near-coastal Australian waters, drawing on the studies reported in previous sections and CSIRO’s expertise in marine instrumentation and monitoring. Two categories of monitoring methodology have been considered in earlier sections: geochemical and acoustic. For each of these we now consider the types of sensor available, the range of deployment platforms and ancillary data sources required for processing and interpretation of these methodologies. We also consider a third category of biological
monitoring approaches in this section. A summary mapping of these methodologies to monitoring objectives is given in Table 7.

<table>
<thead>
<tr>
<th></th>
<th>Geochemical</th>
<th>Acoustic</th>
<th>Biological</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seabed release</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Environmental impact</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Magnitude of release detectable (t y⁻¹)</td>
<td>10,000</td>
<td>&gt;1-10</td>
<td>NA</td>
</tr>
<tr>
<td>False alarm rate</td>
<td>1%</td>
<td>low at &lt;10 t y⁻¹ single source</td>
<td>?</td>
</tr>
<tr>
<td>Spatial resolution</td>
<td>100s-10s m</td>
<td>10s-0.1s m</td>
<td>?</td>
</tr>
</tbody>
</table>

Table 7 Relevance of different categories of methodology to monitoring objectives

A key consideration for both monitoring goals is the need for high quality baseline data that goes beyond the immediate variables of interest. Understanding relationships between seabed sediment composition and biogenic bubble signatures or variability in seabed bathymetry induced by seasonal, tidal and weather-related effects will help reduce false alarm rates from acoustic anomalies. Furthermore, understanding variability in surface and subsurface currents, wave fields, temperature, salinity, primary production and respiration (biological carbon uptake and release), optical characteristics and other oceanographic variables in the immediate vicinity of a storage site will be essential for interpreting and attributing anomalies. For example, unravelling relationships between oceanographic water masses, primary production (which can be monitored using optical satellites) and carbon dynamics will be fundamental for attributing changes to non-CCS related drivers.

5.2 Technical methodologies and sensors

5.2.1 GEOCHEMICAL APPROACHES

Measuring CO₂ in the water column is likely to be an effective method of identifying ingress of large amounts of CO₂ into the water column because it will mainly dissolve in the overlying seawater before any bubbles reach the sea surface. The oscillatory nature of currents in the region means that a fluid plume can be detected over a wide area. The results of Chapter 3 show that minimising the rate of detectable ingress requires state-of-the-art accuracy and high measurement frequency; achieving this will require a combination of instrument types. Hardman-Mountford et al. (2014) considered available sensor types for measuring CO₂ related variables in seawater and a summary is reproduced in Table 8. Moored and underway surface pCO₂ systems have demonstrated high accuracy due to their self-calibration against standard gases but repeat sampling is of the order of 10-30 mins which may not be sufficient to adequately resolve a plume. Submersible ISFET (ion-sensitive field-effect transistor) pH sensors, such as the Satlantic SeaFET, provide stable pH measurements in seawater for prolonged deployments and at high accuracy. They can measure on the order of seconds rather than minutes, providing high-frequency measurement
A combination of pCO$_2$ and pH sensors will provide both highest accuracy and highest frequency of measurements giving the highest probability of detecting a plume in the water column. CSIRO has much experience with these sensors and maintains the only maintenance and calibration facility in the Southern Hemisphere for in situ sensors of sea temperature, salinity, pH (ISFET), pCO$_2$, and dissolved oxygen.

Although CO$_2$ release is highly improbable, the most likely locations to experience ingress are wellheads and pipelines. In these locations, near-field changes in the pH of seawater near the seabed relative to the surface would likely be observable. Therefore, establishing measurements around these locations would be highly valuable for ensuring the security of these features. Any disused wellheads within the proposed storage area would be an obvious target for seabed monitoring. Other wells may be drilled as monitoring wells for the geological storage and these too would be obvious sites for deployment of seabed pH sensors.

SeapHOx systems are small systems that measure, salinity, temperature, oxygen and pH and are suitable for this application. The SeapHOx is not commercially available, but CSIRO have a number of the systems through partnership with the Scripps Institution of Oceanography who developed the technology. The salinity and temperature measurements are required to finalise pH data and with oxygen are useful for identifying seasonal and regional changes in water masses that could result in pH variability that is unrelated to CO$_2$ release. The sensors can be deployed on the seafloor in depths from 1-60m and additional sensors (e.g. methane) can be added as required. Deployment alongside surface pCO$_2$ and pH monitoring will allow for referencing of these measurements to each other to assist with data quality control. CSIRO has considerable experience in the use of these sensor packages from shallow rocky reefs off Tasmania, to under sea-ice conditions on the Antarctic shelf.

Geochemical detection is highly relevant to demonstrating that there is no environmental impact from CCS activities, i.e. there is no detectable change in seawater CO$_2$ levels or changes are smaller than levels of natural variability to which the ecosystems are already adapted. Given the low probability of loss of containment, this is most likely to be of use in attributing any identified changes in sensitive environments to non-CCS sources. As such, seabed monitoring systems would be useful for establishing baselines at locations deemed vulnerable to the impacts of CO$_2$ release, e.g. rocky reefs or scallop beds, enabling verification that environmental change in these regions is or is not related to CO$_2$ ingress.

If an anomaly were to be detected by chemical methods, an attribution step would be necessary. Attribution of measured changes in CO$_2$ / pH from sensors deployed in the water, will likely be assisted by measurements of natural tracers, with benefits realised through reductions in false alarm rates and better understanding of the drivers of baseline variability. Any CO$_2$ ingress would likely perturb the ratio of gas concentrations in seawater as well as the ratios of natural isotopes and the concentrations of mobilised tracers. Therefore, it is important to assess high-precision measurement capabilities to identify robust, distinct signatures for different sources of CO$_2$ anomalies (e.g. groundwater seepage). Cavity ring down spectrometry allows the measurement of a number of tracers including the stable isotopic composition of dissolved CO$_2$ ($\delta^{13}$C of CO$_2$), the oxygen isotopic signature of water ($\delta^{18}$O of H$_2$O) and dissolved methane. The $\delta^{13}$C of CO$_2$ is useful for distinguishing fossil fuel derived CO$_2$ from background seawater CO$_2$ due to the different isotopic signatures. The $\delta^{18}$O of H$_2$O, dissolved methane and radon (measured by radioactive decay) are potential indicators of porewater or submarine groundwater discharge that may be linked to natural variability or seepage. Such measurements would require a range of sensors and equipment that are not suitable for field deployment but could be used underway on periodic ship-based surveys and for laboratory analysis of collected water samples.

Such ship and lab-based activities are also fundamental for obtaining calibration data and maintenance of field deployed sensors. For example, the measurement of total alkalinity (potentiometric titration) and total dissolved CO$_2$ (coulometry) are needed for calibration of in situ pH and pCO$_2$ sensors as well as to monitor for dissolution in sedimentary carbonates.
<table>
<thead>
<tr>
<th>CARBONATE SYSTEM STATE VARIABLE</th>
<th>METHOD</th>
<th>EXAMPLES OF PLATFORMS</th>
<th>SURFACE / SUBSURFACE</th>
<th>DEPLOYMENT TIMEFRAME</th>
<th>ACCURACY</th>
<th>RANGE</th>
<th>RESPONSE TIME</th>
<th>EXAMPLES OF INSTRUMENT TYPES</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>pCO₂</strong></td>
<td>IR spectroscopy / mechanical equilibration (continuous)</td>
<td>Research vessel, wave glider, fixed moorings</td>
<td>Surface</td>
<td>Months to years</td>
<td>0.5 ppm</td>
<td>0-20,000 ppm</td>
<td>minutes</td>
<td>General Oceanics 8050-pCO₂; PML-Dartcom Live-pCO₂</td>
</tr>
<tr>
<td></td>
<td>IR spectroscopy / membrane equilibration (continuous)</td>
<td>Fixed moorings, ROV, Research vessel</td>
<td>Subsurface</td>
<td>Weeks to months</td>
<td>From 1ppm to +/- 1%</td>
<td>0-1000 ppm</td>
<td>minutes</td>
<td>Pro-Oceanus CO₂-Pro; Contros Hydroc/CO₂</td>
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<tr>
<td></td>
<td>Spectrophotometric dye / membrane equilibration (continuous)</td>
<td>Fixed moorings, Research vessel</td>
<td>Subsurface</td>
<td>Weeks to months</td>
<td>3 ppm</td>
<td>150-700 ppm</td>
<td>minutes</td>
<td>Sunburst sensors SAMI2-CO₂ / AFT-CO₂</td>
</tr>
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<td><strong>pH</strong></td>
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<td>Fixed moorings, Research vessel</td>
<td>Subsurface</td>
<td>Weeks to months</td>
<td>+/- 0.01</td>
<td>7-9 pH units</td>
<td>minutes</td>
<td>Sunburst sensors SAMI2-pH / AFT-pH</td>
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<tr>
<td></td>
<td>Solid state (continuous)</td>
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<td>Subsurface</td>
<td>Weeks to months</td>
<td>+/- 0.01</td>
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<td>seconds</td>
<td>Satlantic SeaFET</td>
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<td></td>
<td>Glass electrode (continuous)</td>
<td>Research vessel</td>
<td>Subsurface</td>
<td>na</td>
<td>+/- 0.1</td>
<td>2-12 pH units</td>
<td>seconds</td>
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<td>na</td>
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<td>seconds</td>
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<td>na</td>
<td>+/- 0.001</td>
<td>na</td>
<td>minutes</td>
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</tr>
<tr>
<td><strong>TA</strong></td>
<td>Gran titration</td>
<td>Research vessel / laboratory</td>
<td>na</td>
<td>na</td>
<td>+/- 2 µmol/kg</td>
<td>na</td>
<td>10's minutes</td>
<td>Metrohm Titrand; Marianda VINDTA 3S/3C;</td>
</tr>
</tbody>
</table>

Table 8 Technical information for a variety of carbonate system measurement methods. Source: Hardman-Mountford et al. (2014).
5.2.2 ACOUSTIC APPROACHES

Bubbles in seabed sediments and (in the case of a larger release) in bubble streams emanating from the seabed are known signatures of subsea gas release and monitoring for them would be expected as part of any monitoring program. Acoustic sensors are extremely sensitive to small gas fluxes from the seabed (as demonstrated in Section 4) and provide powerful methods for (a) detecting the presence of bubbles from a release, both in the water column and in sediments and (b) for mapping the shape and acoustic properties of the seabed to determine the major habitat types. For large releases (single source $> 10$ tonnes y$^{-1}$ continuous), detection of bubble streams should be achievable with low false alarm rates. This detectability decreases as the flow decreases or the number of sources increases. For lower level releases, a good understanding of baseline variability in features, such as biologically-generated pockmarks and bubbles associated with burrowing organisms, plankton, zooplankton and fish, is essential for increasing confidence in bubble monitoring approaches and reducing false alarm rates for CCS sources. Well-tuned acoustic monitoring could therefore be important for providing early detection of seabed release and for pinpointing sources of ingress. Understanding the sensitivities of acoustic processes for detecting seabed anomalies above baseline variability should, therefore, be a key component of an M&V strategy, e.g. through gas release experiments at existing well-characterised locations.

Active acoustic approaches include multi- and single beam, multi-frequency systems described in section 4 for detecting bubble signatures, pinpointing bubble plumes and estimating the composition and size of the bubbles (to quantify dissolution rates). A sub-bottom profiler (high-frequency echo sounder) can be used to detect bubbles within sediments that have not reached the seabed. Underwater sonar (e.g. mounted on a ROV or AUV platform) can be used to validate and image any bubble plumes and determine their source (biological, natural seep, CCS ingress). Furthermore, a multi-beam sonar can be used to map the seabed to provide a baseline of habitat type and variability. Low cost moored passive acoustic recorders can be used to monitor suspected gas release sites to determine episodic source behaviour. Near shore application of passive acoustics will need to be investigated due to the high and variable background noise.

To verify bubble stream acoustic targets, optical methods are best and can be done with both drop cameras or, for finer and better quantitative analysis, optics mounted on a remotely operated vehicle. Using a remotely operated vehicle it would be possible to determine bubble sizes and flow rates from the video and imaging sonar as well as taking samples of the gas released for analysis.

5.2.3 BIOLOGICAL APPROACHES.

Monitoring marine organisms may be required by regulators and could contribute to social licence. It can potentially provide early indication of CO$_2$ release through rapid changes in community composition and stress responses, especially within the microbial community. Without examining the need or usefulness of these methods, we comment here on practical aspects were they to be required.

Establishing good environmental baselines for habitat types and key species using a combination of remote (towed video, multibeam swath, ROV) and direct (dive surveys, net, sled and grab sampling) approaches is costly and time consuming. CSIRO has expertise in these activities and undertakes them routinely for a range of clients, from Federal and State Government agencies to offshore industry developments. In the context of CCS, larger releases are likely to be rapidly detected using a combination of chemical and acoustic methods but low level chronic impacts would more likely manifest through subtle ecosystem changes that may be better detected using state-of-the-art molecular meta-barcoding approaches that can sample across the whole ecosystem, from microbes to higher trophic levels like fish, and provide an integrated view of perturbations. Analysis of flux in gene expression profiles (meta-transcriptomics) can also indicate what the likely drivers of change may be. These approaches have matured substantially over the past few years and now provide a rapid and cost-effective solution to environmental monitoring. Given their time and cost efficiencies when compared to traditional biological monitoring approaches, we
recommend trialling these new molecular approaches alongside traditional methods to determine their effectiveness in providing ecosystem baselines and detecting / attributing change.

A key aspect of attributing observed biological changes to CO₂ ingress is understanding what a particular impact may look like. To this end, it may also be advantageous to undertake growth experiments on target commercial calcifying species (e.g. abalone, mussels, oysters, scallops) within the vicinity of the storage location, looking at both larval and adult life stages. The outputs of these experiments would provide critical quantitative information on the scale of ingress that would be needed to cause significant damage to these commercially-valuable species. The findings may effectively be applied to provide a cost effective method for monitoring potential impacts to populations of these species going forward. Experimental chambers have been specifically designed as undersea laboratories for undertaking such experiments within the marine environment, thereby allowing only selected variables to be changed relative to background conditions. The designs for these chambers are freely available under creative commons licensing and could be assembled for use in Australian waters.

5.3 Platform considerations

While it is inevitable that any marine monitoring will require boat-based activity, a range of alternative platforms are available that may provide cost-efficient alternatives that can significantly reduce reliance on boat-based sampling.

5.3.1 AUTONOMOUS AND REMOTELY-OPERATED MOBILE PLATFORMS

The results we have described from modelling a CO₂ release plume show that higher spatial and temporal resolution in sampling can substantially improve the probability of detection by (a) monitoring over a wider area and (b) reducing the level of background variability to a minimum by constraining its spatial structure (in geological terms, the variogram of natural fluctuations). Autonomous platforms are a potentially cost effective solution for increasing spatial coverage of high frequency measurements without needing multiple fixed platforms and minimising requirements for fuel and labour intensive boat-based approaches. These would be well suited to identify releases emanating from an unknown source such as a pipeline rupture or seepage through geological features.

A particular design that has been proven for undertaking seawater CO₂ monitoring is that of the Liquid Robotics’ Wavegliders. These platforms are driven by wave motion and can operate autonomously for months, providing coverage across a site. The Wavegliders can be directed remotely to target anomalies in CO₂ that may indicate a release. In this study we have simulated the chemical signals we would expect to retrieve from a Waveglider traversing the modelled CO₂ plume and our results show them to be more effective than fixed moorings in detecting plume-related changes in seawater CO₂. For nearshore sites there would be constraints, as Wavegliders are limited to operation in water deeper than ~5m (operational draft of 4m, equates to ~300m offshore along the Gippsland coast) but they represent proven technologies for offshore industry applications in coastal waters. They have also undertaken transects across the Pacific Ocean (USA-Australia) and in the Arctic Ocean, demonstrating their capability for long-term deployment under a variety of conditions. Wavegliders also have capacity to carry acoustic sensors, although power demands would need to be balanced against other requirements.

The regulatory environment for autonomous vehicles, both on- and off-shore, is set to develop rapidly and, given the commercial pressure to use autonomous vehicles of various kinds, a workable set of regulations is likely to emerge. At the present time, there are no regulatory restrictions on using these platforms offshore in Australia as they are not categorised as marine vessels.

Submersible autonomous underwater vehicles (AUVs) and remotely operated vehicles (ROVs) are another category of mobile platform to which sensor systems can be added. Underwater gliders, such as the Slocum and Seaglider, can survey wide areas. ROVs are generally used for more localised activities but can be usefully employed to target particular subsea anomaly features, such as high acoustic scattering targets suggestive of bubble streams, using a combination of sonar and video systems.
5.3.2 FIXED MOORINGS

Fixed moorings are structures from which sensors can be mounted that are anchored to the seabed. The sensors can be attached on the seabed at the base of the mooring or to cables extending vertically from the mooring to sample throughout the water column. Some moorings have a surface expression, such as a buoy but others do not. The choice of whether to have a surface expression depends on both the cost and operational considerations, for example, a need for real-time communications. CSIRO maintains a large number of moorings around the Australian coast as part of the Australian Integrated Marine Observing System (IMOS, www.imos.org.au), including those measuring physical properties of major current systems (temperature, salinity, velocity) and water quality variables, passive acoustic moorings and ocean acidification (carbon) moorings. Figure 22 shows the range of mooring designs used and variety of depths for the IMOS National Reference Stations.

Fixed moorings (Figure 22 and Figure 23) provide an excellent approach for obtaining long-term baseline measurements at specific locations, thus are an important platform to consider in an M&V strategy. They are also important as reference points in the monitoring system, allowing mobile measurements to be cross-calibrated with these fixed-point measurements in order to separate spatial and temporal variability and improve interpretation of data. The locations targeted can be based on their representativeness of a wider system or their proximity to specific monitoring targets, for example, higher risk features such as old well heads. If deployment of sensors is needed in waters shallower than 5m (i.e. within ~300m of the coast off Gippsland), fixed moorings present the most realistic option for monitoring as these waters are not easily monitored by mobile platforms. The alongshore oscillatory nature of currents along the Gippsland coast means that water masses are moved large distances and mooring locations can be chosen to be representative of water column properties across a wider region, increasing both the value of baseline data and the possibility of encountering a CO₂ plume in the unlikely case of a release.
For CCS monitoring, retrievable carbon mooring systems could provide real-time, high frequency, highest accuracy seawater pCO₂ at the sea surface and cross-referenced pH data throughout the water column and at the seabed. These moorings can also measure atmospheric pCO₂ to monitor for CO₂ gas anomalies that may reach the ocean surface before fully dissolving in these shallower waters, or be transported offshore from terrestrial CO₂ sources. Such atmospheric pCO₂ measurements would likely need to be referenced against land-based atmospheric monitoring that can account more fully for such CO₂ anomalies, e.g. through isotopic composition, to insure against unreasonable false alarm rates. However, depending upon where the moorings were sited, they may have the advantage of measuring against a much cleaner background signal than land-based atmospheric monitoring. CSIRO has extensive experience with such moorings, with systems installed in water depths ranging from 4m to 100m on the Great Barrier Reef, off the west coast of Kangaroo Island and along the east coast of Tasmania. Passive acoustic systems (hydrophones) can be added to fixed moorings as listening stations for bubble noise, although these would be unlikely to be effective in very shallow waters close to significant surf action.
Fixed moorings can also accommodate a variety of other sensors required for ancillary marine system measurements and high-quality baseline data (e.g. CTD and thermistors for temperature and salinity, AWAC for current velocities and wave heights, optical sensors for phytoplankton carbon uptake), as well as for geological monitoring (e.g. ocean bottom seismometers). In contrast, the intermittently sampled baseline CO₂ data used in this study, obtained from a ship-of-opportunity, does not resolve the highest and lowest frequency variability in the dynamics of the region and lacks adequate contextual information to attribute naturally-occurring anomalies.

5.3.3 OPERATIONAL CONCEPT

Clearly a marine monitoring strategy needs to be built upon a detailed risk assessment, and will involve detailed discussion with regulators and other interested groups. Hitherto unused regulations will have to be tested and given operational content. None of these elements is yet in place, and so in outlining an operational concept, we are only outlining an effective integration of the techniques and hardware we have investigated in this report. Whether this particular concept will be needed remains to be seen; apart from risk and regulation, there are a number of purely technical challenges to be overcome to implement the systems we now describe. Systems of this type have not been assembled anywhere, although of course there are analogues in other areas of marine research. The concept we now state is therefore tentative but illustrative of how the equipment in Table 8 might be integrated and (eventually) used as a system.

Our concept involves the following elements:
1) Patrolling wavegliders (number TBD but probably two, for redundancy). These would be outfitted with automated systems for regular high-frequency sampling of water chemistry, and would also make acoustic surveys of the water column and sea bed to establish spatial and temporal background variability for false alarm threshold knowledge. These data would be telemetered back to base in real time and analysed in near-real-time. Patrol areas of hundreds of square kilometres, or smaller if required, seem feasible in usefully short times (a week or two). Data streams from these measurements – particularly the imaging – will have to be processed largely automatically and pattern-recognition algorithms will need to be developed. Waveglider activity can be limited to within particular seabed lease areas as required by operational reasons.

2) Fixed moorings. Probably one located at or near a wellhead, and one to provide a control some distance from the areas at risk (and possibly inshore in waters too shallow for wavegliders to patrol). These would be instrumented with chemical samples, acoustic imagers, and sensors for oceanographic data such as currents and salinity. The moorings may also host ocean-bottom seismometers. The function of the moorings would be (a) to provide intensive surveillance of highest-risk sites, (b) ground-truth data from wavegliders, (c) provide oceanographic data for inversion of chemical measurements to locate sources, (d) provide reference points for the reconstruction of the spatial structure of environmental fluctuations in water chemistry, to quantify the sensitivity of the system to CO$_2$ ingress, (e) provide temporal background levels in physical, chemical and acoustic parameters to assist in false alarm detection capabilities.

3) Follow-up investigations. Should anomalies be detected (bubble streams, gassy sediments, new seabed features, anomalous water chemistry) follow-up investigations by manned craft will be needed. A combination of boat-based sampling and ROVs can be used for deploying a range of groundtruthing technologies including lowering underwater video cameras and introducing probes for pH and oxygen into the target environment, obtaining grab samples of seabed sediments and water samples for hydrochemistry.

4) Additional baseline investigation. Boat-based sampling will be required on top of the waveglider and fixed mooring measurements to develop a baseline data set against which to monitor for anomalies. We estimate this may involve monthly surveys in the vicinity of the storage site for the first year of monitoring followed by bimonthly surveys for the following two years to understand seasonal and interannual variability. Following this, a lower frequency of surveys may be feasible, for example quarterly surveys every 3rd year. The surveys would map bathymetry and seabed features using multibeam sonar and single beam echo sounders as well as undertaking grab and towed video sampling for sediments and benthic fauna. Baited remote underwater video (BRUV) systems would be deployed to monitor fish populations in a non-intrusive manner. Water column samples would be taken for baseline measurements of hydrochemistry (including isotope ratios), production and plankton to determine vulnerable seasons for eggs and larvae of marine organisms.

5) Biological monitoring. Approaches for biological monitoring have not been investigated in this study but are mentioned nonetheless as an important monitoring tool. As well as the baseline sampling described under point 4) above, ecogenomic approaches that can rapidly and sensitively assess for changes in biodiversity or ecosystem function (e.g. enhanced CO$_2$ metabolism) would be compared with traditional labour-intensive approaches (e.g. video or microscopic analysis) to calibrate the different methods and assess the utility of ecogenomics for routine assessment of biological samples (from sediment or water column sampling). For particular commercially-sensitive shellfish species, it may be feasible to culture control populations in the vicinity of higher-risk sites, such as legacy wellheads.

6) The monitoring systems will be backed up from shore facilities including calibration laboratories and maintenance workshops. There will also need to be a base station to receive and interpret data, organize follow-up if needed, report to regulators, and so on.
<table>
<thead>
<tr>
<th>Equipment</th>
<th>Deployment</th>
<th>Application</th>
<th>Examples of instruments</th>
</tr>
</thead>
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<td>Seabed multibeam sonar</td>
<td>Boat</td>
<td>Baseline seabed and habitat mapping and boat-based monitoring surveys</td>
<td>Kongsberg EM2040</td>
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<td>Multibeam sonar head for mounting on an ROV</td>
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<td>Bottom mounted active sonar</td>
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<td>Sub bottom profiler and side scan sonars</td>
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<td>Side scan sonar</td>
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<td>Klein/Simrad</td>
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<td>Hydrophones</td>
<td>AV, Fixed moorings</td>
<td>Cost effective passive acoustic option for listening for bubbles (not near-shore)</td>
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<td>Moored thermistors, sound velocity probes and CTD</td>
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<td>Dissolved oxygen optodes</td>
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<td>Attribution of CO₂ anomalies and reduction in false-alarm.</td>
<td>Aanderaa 4330 or Seabird SBE63</td>
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<tr>
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<td>Integrated approach to bottom measurements of pH, dissolved oxygen, salinity and temperature for baseline monitoring and attribution.</td>
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<td>Cavity ring-down gas analysers for pCO₂, CH₄, δ¹³C, δ¹⁸O</td>
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<td>Radon detector</td>
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<td><strong>Platforms</strong></td>
<td>Autonomous platforms (AVs)</td>
<td>Extended unmanned monitoring operations carrying a payload of different sensors described above</td>
<td>Liquid Robotics Waveglider</td>
</tr>
<tr>
<td>Type</td>
<td>Description</td>
<td>Example</td>
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<td>-------------------------------------------</td>
<td>-----------------------------------------------------------</td>
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<tr>
<td>Fixed moorings</td>
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<td>Remotely Operated Vehicles (ROVs)</td>
<td>For targeted investigation of seabed features</td>
<td>Phantom ROVs</td>
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</table>

Table 9 Types and examples of equipment and relevant platforms that would be used for the applications described in Chapter 5.
6 Discussion and Conclusions

In this report we have undertaken a desk-based evaluation of the sensitivity of two types of technology for detecting anomalous targets related to CO₂ intrusion into the marine environment. For geochemical sensors that measure changes in the seawater carbonate system, the anomalous targets would be elevated concentrations of CO₂ in seawater (either elevated pCO₂ and DIC or decreased pH). For active acoustic sensors that measure the acoustic backscatter (echo) from reflective surfaces in the water column or in seabed sediments, the anomalous targets would be bubbles. While experience of both deliberate CO₂ release and natural analogues, such as geological seeps, indicate that both geochemical and acoustic signals are associated with CO₂ intrusion into the marine environment, the challenge of fit-for-purpose M&V lies in detecting these signals above background levels of variability with low false-alarm rate.

6.1 Detection by geochemical sensors

There are many background factors contributing to CO₂ variability in the marine environment. These can be broadly categorised into the biological factors of production, respiration and calcification; geological factors related to carbonate formation and dissolution as well as natural geological seeps; physical mixing of different water masses; terrestrial sources of organic matter coming from rivers, run-off or groundwater and uptake of anthropogenic emissions from the atmosphere. Understanding which of these factors are dominant within a particular region, their magnitude and the timescales over which they vary is important for assessing the detectability of a new source of CO₂ to the marine environment.

In this report we have built upon our previous work that explored the behaviour of a chemical plume originating from a CO₂ release to the marine environment along the Gippsland coast. This previous work showed there was a high probability of encountering the plume in these near-coastal, well mixed waters given the right configuration of fixed sensors (Greenwood et al. 2014). The report of Hardman-Mountford et al. (2014) also analysed the variance structure of a time series of seawater CO₂ data from Bass Strait and concluded that the minimum detectable source of CO₂ into Bass Strait waters would be in the range 17-100 kt yr⁻¹ dependent on season, position of the Bass Strait front and configuration of monitoring locations. Using the spiky temporal structure that would characterise a release of CO₂ into the environment gave a detection time of 2-3 weeks, thus giving a total release mass of 1-6 kt prior to detection. In this study we have re-explored the same background dataset in the context of using mobile platforms and have applied a simple detection algorithm that picks the peak CO₂ measurement in a flight and contrasts it in relation to the average CO₂ measurement from surrounding measurements. By reducing the variance structure through differencing of sequential measurements and ‘stacking flights’ (i.e. accumulating the data from multiple survey lines) from mobile wavegliders patrolling the area of interest, we were able to reduce our estimation of the size of CO₂ release that could be detected above background variability to 10 kt yr⁻¹ at better than 95% confidence level while achieving a low false alarm rate of 1%. To stack 4 waveglider patrols of 6 days each would take 24 days minimum although this time frame could be reduced by using multiple wavegliders. Thus, allowing for a maximum detection time of 1 month would give a total release of ~0.8 kt. While this is a small number, the implied rate is high and would probably be detected first by other methods e.g. bubbles.

Limitations of the approach are firstly, that it depends on a tightly constrained variance structure to the data that we have been able to achieve during locations and seasons to the west of the Bass Strait front. Further work is required to understand the correlation structure of variance in CO₂ data to the east of the front to understand if there is correlated structure on scales less than 5km or whether the uncorrelated ‘white noise’ assumption can hold. Nonetheless, the standard deviation at the 5km scale in this eastern region is double that of the westerly case, suggesting release rates of 10 kt yr⁻¹ would be hard to detect for about half of the year. Furthermore, these results are for quite a simple algorithm, with limited information...
on background variability, and modelling over a limited time span. More comprehensive baseline data would probably lead to more refined detection algorithms that were tailored to exploit particular features of the dispersion in the area under surveillance. These refined algorithms would be expected to have more sensitivity, although how much more is hard to say without further study and more field data.

While the geochemical analysis undertaken in this report has focused on direct sensing of bulk CO\textsubscript{2} in the ocean due to the availability of existing datasets, other geochemical approaches exist that may improve detectability beyond the theoretical limits we have derived. Oxygen optodes are a mature sensor technology that can provide robust long-term measurements of dissolved oxygen in seawater. A release of CO\textsubscript{2} into the marine environment will likely perturb the ratio of CO\textsubscript{2}/O\textsubscript{2} in a different way to naturally occurring processes such as photosynthesis and respiration. Thus, a combined use of oxygen and CO\textsubscript{2} measurements may reduce the false-alarm rate and improve detectability of anomalous CO\textsubscript{2} signals. Other geochemical approaches include measurement of naturally-occurring isotopes in CO\textsubscript{2}, or of other tracers for geological sources (e.g. methane) or groundwater sources (e.g. radon). Off-the-shelf sensors are available for such measurements but further work is required to assess their effectiveness in helping detect and attribute CO\textsubscript{2} releases to the marine environment as part of an M&V strategy.

Finally, we note that the issue has been phrased so far in terms of the ability of a geochemical method to detect release into the water column. This work has shown that a substantial mass of CO\textsubscript{2} is needed to cause changes that rise above natural variability. This suggests that (except possibly locally to a release site), CO\textsubscript{2}-related environmental impacts of any ingress are likely to be small, except for a very large release. It also highlights that, if monitoring of the water column is required to check for environmental impact, it will be important for well-understood baselines to be in place and for there to be a clear understanding and acceptance of the likely false alarm rate if anomalies are interpreted as evidence of a release.

### 6.2 Detection using active acoustic instruments

Whereas geochemical approaches measure CO\textsubscript{2} in the marine environment more-or-less directly, acoustic methods take both a direct and indirect approach looking for signatures in acoustic backscatter (i.e. echoes) that correspond to known properties of bubbles. As with geochemical variability, there are many factors that can produce bubbles and bubble-like signatures in the marine environment. For example, respiration of biota living in seabed sediments, photosynthesis of benthic primary producers, organic matter breakdown in sediments, surf-zone processes and natural seepage are sources of bubbles, while organisms such as zooplankton and fish can contain gas that appears acoustically to look like bubbles. Detection of a CO\textsubscript{2} release above background noise will vary considerably between different habitats that have different acoustic properties as well as different levels of background-noise producers. Strategies to help differentiate between these different targets include choice of instrument (multi-beam vs. single beam) and acoustic frequency band, spatio-temporal configuration of survey design (e.g. ‘watching’ a target for a time period to see if it is mobile like a fish school or more constant like a bubble source, although note that bubble-sources can also be intermittent over e.g. tidal time scales) and use of drop-cameras for visual ground-truthing of targets.

While baseline data was not available to test the level of background noise along the Gippsland coast, existing classifications of the dominant sediment types, together with knowledge of typical background noise signatures from other environments, was used to make an expert assessment of the likelihood of acoustic technologies to detect a release of CO\textsubscript{2} as either a bubble stream in the water column or a cluster of bubbles in seabed sediments. Assuming a ‘typical’ average bubble-size of 7.5 mm, the assessment suggests that there is a medium probability of detecting a low-flow release of 1 tonne yr\textsuperscript{-1} and a high probability of detecting a larger 10 tonnes yr\textsuperscript{-1} release. These flow-rates are 3 orders of magnitude lower than the scale of release detectable using geochemical methods. Other considerations are the trade-off between sensitivity and coverage and whether there are multiple sources or just a single source, as summarised in Table 9.
The choice of platform for active acoustic approaches is an area that requires further investigation. The spatial coverage required to survey an area relies on a mobile platform. Traditionally such work has been undertaken from small boats but more recently echo sounders have been deployed on wavegliders for short term applications. The feasibility of using such autonomous platforms for longer-term acoustic surveys while satisfying the power demands of active acoustic sensors will require further investigation. Passive acoustic hydrophones may also be worth considering due to their ability to listen for bubbles and lower power demands. Ground-truthing will be a necessary part of reducing false alarm rates from use of active acoustics so consideration of how best to deploy drop cameras or other ground-truthing technologies is another outstanding question. ROVs may also be a useful technology to consider here, although autonomous rather than just remotely-operated platforms would potentially be more cost-effective owing to the lower requirement for human intervention.

The conclusion of our assessment of acoustic technologies is that they are a promising methodology for detecting a gas release to the marine environment and, as such, should be investigated further. Priority areas for investigation would include: (1) collection of baseline data for Bass Strait in order to quantify background noise signatures and assess likely false-alarm rates; (2) investigation of technological requirements for deploying acoustic sensors on autonomous mobile platforms such as wavegliders; (3) field testing of acoustic technologies and sampling strategies to detect ‘unknown’ sources, either through a small bubble release experiment or against natural seeps with intermittent flow.

6.3 Conclusions

In summary we can draw the following conclusions from the studies reported here:

1. Both geochemical and acoustic methods are capable of detecting signatures associated with CO\(_2\) release, however, the scale of release that can be detected by the two approaches varies by 3 orders of magnitude with acoustic approaches looking significantly more sensitive. Caveats remain around quantifying the false alarm rate for acoustic approaches. The extent to which a release is characterised by bubbles compared to a dissolved chemical plume is variable and uncertain, thus both approaches are required.
   a. Chemical water sampling, from a combination of fixed moorings and gliders, would be able to monitor a region of order 10 x 10 km\(^2\) and detect a CO\(_2\) release anywhere in that area of 10 kt yr\(^{-1}\) with high confidence and low false alarm rate.
   b. Monitoring by acoustic methods is extremely sensitive to bubble streams. The background noise level (mainly biota) is very variable, but streams corresponding to only 10 t yr\(^{-1}\) should be readily detectable above environmental clutter.

2. The relative insensitivity of geochemical approaches is largely due to the large range of natural variability in seawater CO\(_2\) concentrations and a substantial mass of CO\(_2\) would be needed to cause changes that rise above natural variability. This suggests that (except possibly locally to a release site), CO\(_2\)-related environmental impacts of any ingress are likely to be small, except for a very large release. Thus, monitoring of the ocean chemistry at this level could also be an effective way of demonstrating that there is not any wider spatial scale environmental impact.

3. While our studies have focused on detection of CO\(_2\) ingress, monitoring for environmental impacts and attribution of both anomalous sources and environmental change are also important components of a fit-for-purpose M&V strategy. Furthermore, biological monitoring approaches may provide cost-effective means to help achieve some of these monitoring goals.

4. Various technologies have been identified that could contribute to aspects of M&V for subsea CCS. Several of these are proven technologies that can underpin monitoring requirements, others require further investigation to determine their feasibility and cost-effectiveness for use in operational monitoring. Field assessment of the technologies described should be a priority for
future work and many of these will be available either through the list of EIF-funded assets or from CSIRO’s marine equipment pool.

5. All environmental monitoring approaches, whether for anomalous sources or environmental impact, require well-understood baselines to be in place and for there to be a clear understanding and acceptance of the likely false alarm rate if anomalies are interpreted as evidence of a release. Well-understood baselines require at least an understanding of variability throughout the annual cycle, such that a minimum of 3 years data are required (to provide a variance estimate between different years). Furthermore, the design of a baseline monitoring program should seek to quantify the variance structure of the environment across all relevant scales of variability. Priorities for baseline monitoring in Gippsland are for acoustic and biological data broadly and for chemical data in the nearshore region and over spatial scales of less than 5km.
References


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