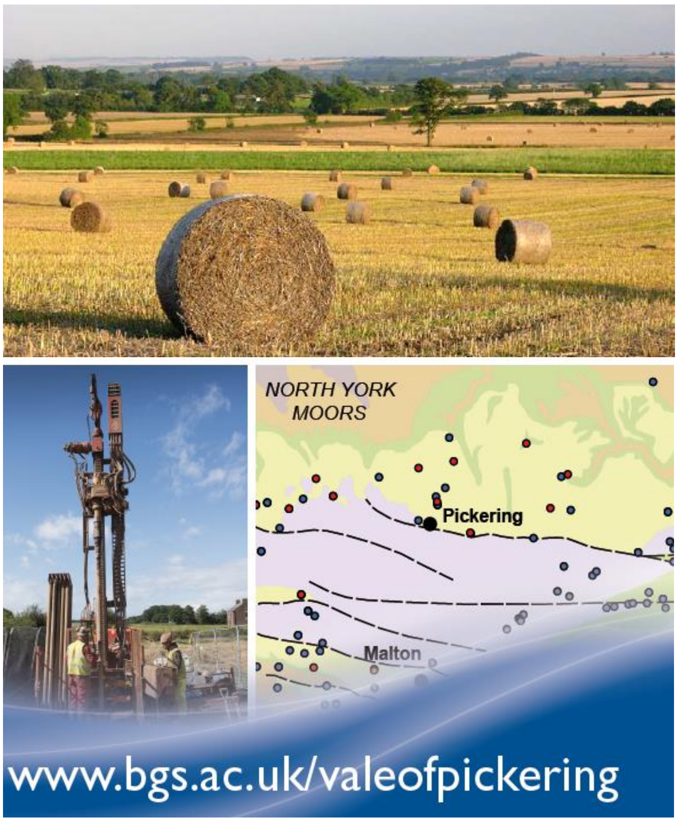
British Geological Survey
NATURAL ENVIRONMENT RESEARCH COUNCIL

Recommendations for Environmental Baseline Monitoring in areas of shale gas development

Groundwater Directorate

Open Report OR/18/043



BRITISH GEOLOGICAL SURVEY

GROUNDWATER DIRECTORATE

OPEN REPORT OR/18/043

Recommendations for Environmental Baseline Monitoring in areas of shale gas development

R.S. Ward, M.O. Rivett, P.L. Smedley, G. Allen, A. Lewis, R.M. Purvis, C.J. Jordan, H. Taylor-Curran, Z. Daraktchieva, B.J. Baptie, A. Horleston, L. Bateson, A. Novellino, D Lowry and R. E. Fisher

Bibliographical reference

R.S. WARD, M.O. RIVETT, P.L. SMEDLEY, G. ALLEN, A. LEWIS, R.M. PURVIS, C.J. JORDAN, Z. DARAKTCHIEVA, B.J. BAPTIE, H. TAYLOR-CURRAN, L. BATESON, A. NOVELLINO, D. LOWRY AND R. E. FISHER. 2020. Recommendations for Environmental Baseline Monitoring in areas of shale gas development. *British Geological Survey Open Report*, OR/18/043. 117 pp.

Keyworth, Nottingham British Geological Survey 2020

BRITISH GEOLOGICAL SURVEY

The full range of our publications is available from BGS shops at Nottingham, Edinburgh, London and Cardiff (Welsh publications only) see contact details below or shop online at www.geologyshop.com

The London Information Office also maintains a reference collection of BGS publications, including maps, for consultation.

We publish an annual catalogue of our maps and other publications; this catalogue is available online or from any of the BGS shops.

The British Geological Survey carries out the geological survey of Great Britain and Northern Ireland (the latter as an agency service for the government of Northern Ireland), and of the surrounding continental shelf, as well as basic research projects. It also undertakes programmes of technical aid in geology in developing countries.

The British Geological Survey is a component body of the Natural Environment Research Council.

British Geological Survey offices

BGS Central Enquiries Desk

Tel 0115 936 3143 Fax 0115 936 3276

email enquiries@bgs.ac.uk

Environmental Science Centre, Keyworth, Nottingham NG12 5GG

Tel 0115 936 3241 Fax 0115 936 3488
email sales@bgs.ac.uk

Murchison House, West Mains Road, Edinburgh EH9 3LA

Tel 0131 667 1000 Fax 0131 668 2683
email scotsales@bgs.ac.uk

Natural History Museum, Cromwell Road, London SW7 5BD

Tel 020 7589 4090 Fax 020 7584 8270
Tel 020 7942 5344/45 email bgslondon@bgs.ac.uk

Columbus House, Greenmeadow Springs, Tongwynlais, Cardiff CF15 7NE

Tel 029 2052 1962 Fax 029 2052 1963

Maclean Building, Crowmarsh Gifford, Wallingford OX10 8BB

Tel 01491 838800 Fax 01491 692345

Geological Survey of Northern Ireland, Department of Enterprise, Trade & Investment, Dundonald House, Upper Newtownards Road, Ballymiscaw, Belfast, BT4 3SB

Tel 028 9038 8462 Fax 028 9038 8461

www.bgs.ac.uk/gsni/

Parent Body

Natural Environment Research Council, Polaris House, North Star Avenue, Swindon SN2 1EU

Tel 01793 411500 Fax 01793 411501
www.nerc.ac.uk

Website www.bgs.ac.uk
Shop online at www.geologyshop.com

Contents

FIGURES

TABLES

Executive Summary

Environmental monitoring plays a key role in risk assessment and management of industrial operations where there is the potential for the release of contaminants to the environment (i.e. air and water) or for structural damage (i.e. seismicity). The shale-gas industry is one such industry. It is also new to the UK and so specific environmental regulation and other controls have been introduced only recently. Associated with this is a need to carry out monitoring to demonstrate that the management measures to minimise the risk to the environment are being effective. While much of the monitoring required is common to other industries and potentially polluting activities, there are a number of requirements specific to shale gas and to what is a new and undeveloped industry.

This report presents recommendations for environmental monitoring associated with shale-gas activities and in particular the monitoring required to inform risk assessment and establish the pre-existing environmental conditions at a site and surrounding area. This baseline monitoring is essential to provide robust data and criteria for detecting any future adverse environmental changes caused by the shale-gas operations. Monitoring is therefore required throughout the lifecycle of a shale gas operation. During this lifecycle, the objectives of the monitoring will change, from baseline characterisation to operational and post-operational monitoring. Monitoring requirements will also change. This report focusses on good practice in baseline monitoring and places it in the context of the longer-term environmental monitoring programme, recognising the need to transition from the baseline condition and to establish criteria for detecting any changes within the regulatory framework.

The core suite of environmental monitoring activities currently required to support regulatory compliance, i.e. meet environmental and other permit conditions, encompasses monitoring of seismicity, water quality (groundwater and surface water) and air quality. Recommendations for each of these are included in this report. Additionally, recommendations for a number of other types of environmental monitoring are included – radon in air, soil gas and ground motion (subsidence/uplift). These are not associated directly with regulatory compliance but can provide information to support interpretation of statutory monitoring results. They are also considered important for public reassurance. Health impacts arising from radon and damage caused by ground motion are both issues of public concern in relation to shale gas.

1. Introduction

This report provides good-practice recommendations for environmental baseline monitoring associated with shale-gas operations. Its purpose is to support the development of effective regulatory and industry monitoring guidance and practice, and associated policy development in the UK. Effective monitoring also plays an important part in reassuring the public of shale gas operations being carried out safely while not putting health or the environment at risk.

The recommendations contained in this report are primarily based on the research findings and experience gained from several British Geological Survey (BGS)/partner-funded and Department for Business, Energy and Industrial Strategy (BEIS)-funded projects. These include national monitoring and survey projects (e.g. Bell et al., 2017), research projects and, more importantly, the targeted baseline studies focused on areas of shale-gas development in Lancashire[1] and the Vale of Pickering, N. Yorkshire[2]. These later studies have been undertaken by an interdisciplinary research consortium led by the BGS and represent the first ever inter-disciplinary environmental baseline studies for shale gas.

The report is cognisant of previous national work and recommendations (including Royal Society/Royal Academy of Engineering, 2012; Public Health England, 2013; UK Task Force on Shale Gas, 2015; UKOOG, 2015; CIWEM, 2016; Environment Agency, 2019a) and likewise informed by the growing body of international literature as well as applicable international standards.

Environmental monitoring plays a central role in risk assessment and management associated with onshore unconventional hydrocarbon development. It needs to be carried out to acquire information both before the start of operations, to establish critically the initial environmental conditions, and during the lifetime of the hydrocarbon operation(s). A key goal of baseline monitoring and supporting site assessment works is to build on and refine the existing site conceptual model and develop a site condition report. The latter documents the nature and condition of the site and surrounding area (including water, air and seismicity) ahead of any industrial development (EA, 2016). Any future impacts from site activities are then determined by monitoring for significant change from the baseline established.

Site-condition reporting will be updated continually based on monitoring throughout operational phases with the intent to demonstrate any significant and unacceptable deterioration in the condition of a site during its lifetime. Where monitoring shows significant deviation from baseline, and other evidence (e.g. known incident occurrence) indicates that site and/or operations pose an increased risk to the environment, action would then be required to investigate and manage those risks and/or impacts.

Monitoring includes measurements undertaken for environmental permit *compliance* and for *assessment* of environmental conditions and operational (industry) performance. Results of the monitoring can also provide *public reassurance*. Compliance refers to the process of ensuring environmental conditions remain within the limits imposed by permit conditions such as any regulatory standard that might apply. Assessment is the process of characterising the environmental conditions (baseline) prior to industry development and then evaluating the significance of any deviation from the baseline, or pre-operational, conditions and attributing it to cause(s). Public reassurance refers to demonstrating that robust, appropriate and trustworthy monitoring is being carried out, and that environmental impacts are not occurring and conditions remain within compliance limits.

Monitoring requirements will vary during the different stages of a shale gas operation to address specific stage monitoring objectives and also respond to evolving understanding and site

[1] http://www.bgs.ac.uk/lancashire

[2] http://www.bgs.ac.uk/valeofpickering

conceptualisation made. Fundamentally though, the baseline monitored condition initially established needs to provide the cornerstone point of reference against which future change in site conditions may be measured across a site's entire lifecycle. It is hence paramount that the baseline condition is defined robustly to allow the detection of significant change from baseline conditions. Baseline monitoring data are thus needed of adequate spatial and temporal resolution and sufficient timeframes to characterise the variability of initial site conditions. They should forensically determine components of the baseline signature due to natural processes versus those derived from existing (and sometimes former) anthropogenic activity that may prove more dynamic.

Throughout the site lifecycle, monitoring will always play a crucial role in not only identifying any influences arising from an operation and the response to any actions taken, but also identify any extraneous changes. The principal monitoring components include:

- definition of monitoring objectives in relation to risk/impact assessment and management
- design of monitoring programme(s) to meet objectives
- installation and management of monitoring infrastructure (including individual stations and/or networks of sensors)
- implementation of monitoring programme (data collection)
- recording of metadata to support the interpretation of monitoring results
- analysis and interpretation of data
- reporting and presentation of data and post-analysis to demonstrate achievement of monitoring objectives

The planning and operation of the monitoring programme should ensure that it is based on delivering an adequate and reliable evidence base to support understanding and management of the risks to the environment and/or human health. This requires the data to be of the correct precision and accuracy, be quality-assured and sufficient for statistical assessment.

The focus of the report is therefore on environmental baseline monitoring before site activities start, but recognising such monitoring will substantially underpin that conducted in the operational and post-operational stages of a site's lifecycle. The following sections present and discuss the recommended approaches for different components of an environmental monitoring programme with the possible range of monitoring activities and their principal objectives summarised in Figure 1. Not all of these are currently associated with regulation and so they may not be formally required as part of the conditions/permissions to operate (e.g. planning, environmental permit) to operate. However, they have been included here because they are considered to be important in providing additional evidence to inform the characterisation of the environmental baseline ahead of onshore shale gas/oil development, and for assessing any future change induced by the associated operations and its environmental significance. This is particularly important at the early stages of industry development in the UK where there is a clear need for evidence to better understand the risks, inform future environmental monitoring priorities and approaches, and address public concerns.

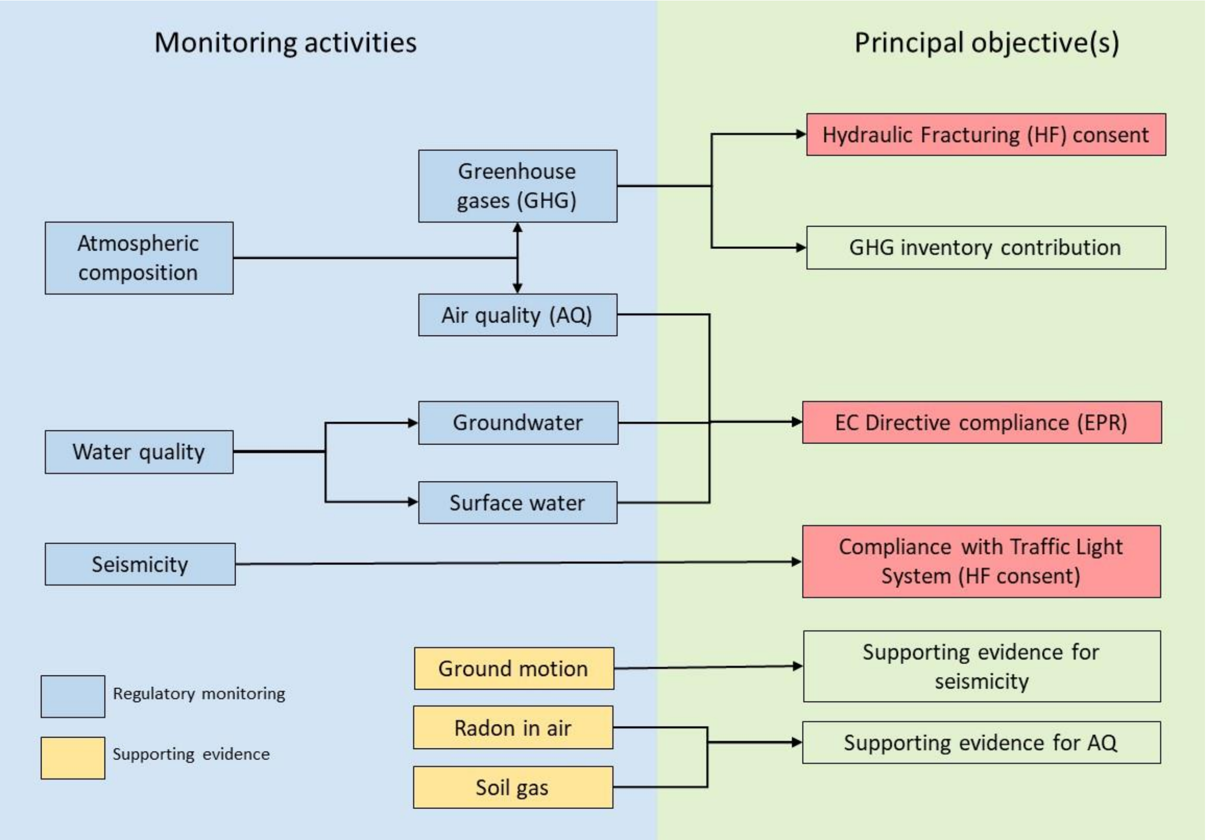


Figure 1. Monitoring activities to establish an environmental baseline for shale gas development and their principal objectives shown within the regulatory context for England.

The report sections cover: seismicity (Section 3), groundwater and surface water (Section 4), atmospheric composition (Section 5), radon in air (Section 6) and soil gases (Section 7) and ground deformation (Section 8). All sections draw from the research findings and experience gained on applying state-of-the-art monitoring approaches on the aforementioned inter-disciplinary baseline studies for shale gas recently undertaken in the UK. The breadth of environmental monitoring required is significant, involves contrasting spatial and temporal scales and involves a broad range of techniques and approaches. Similarly, the availability of real-time data, the ease and costs of data acquisition, processing and interpretation vary significantly.

Within the different components of monitoring, some common questions and challenges arise that this report attempts to address, recognising also that some may be monitoring type or site-specific and perhaps only fully addressed once shale gas operations take place. Questions include:

- What environmental parameters should be monitored? Are there key indicators that could be useful for identifying environmental impact?
- What are the spatial and temporal scales which monitoring should address?
- Is there scope for application of proxy methods (i.e. not necessarily monitoring at each site)?
- What areas of methodological and scientific uncertainty remain to be addressed and minimised?
- What recommendations can be made for detecting change from the baseline during subsequent operational and post-operational stages of a sites lifecycle?

A monitoring programme needs to be designed according to the individual sites environmental setting and by considering the degree of risk that the industrial operation presents to the environment and human health. Fundamental to this is the need to develop a representative, process-based, conceptualisation for the site/area (a 'site conceptual model') that integrates all available information and identifies relevant sources of a possible hazard, possible pathways of exposure and receptors at risk. Environmental monitoring needs to be integrated effectively to the site conceptual model to ensure adequate monitoring of the risks posed. It should be designed to support definition of environmental permit conditions and test critically, and review the site's compliance with these. Consideration also needs to be given to procedures for conveying the information to the public in a transparent and meaningful way.

2. Key principles

2.1 MONITORING PROGRAMME DESIGN

The monitoring programme (Figure 2) that is necessary to underpin environmental understanding of the development site lifecycle is demanding, both in its longevity spanning perhaps two decades, and the breadth of monitoring activity required. Given these demands, effective design of an optimal environmental monitoring programme is paramount, especially of the baseline phase due to its foundational underpinning of later phases. Monitoring approaches selected at outset and many of the individual monitoring point stations established may need to be retained for the completion of a site's lifecycle. Integration of the various monitoring activities is vital to achieving a holistic monitoring approach and allowing informed overall site decision making.

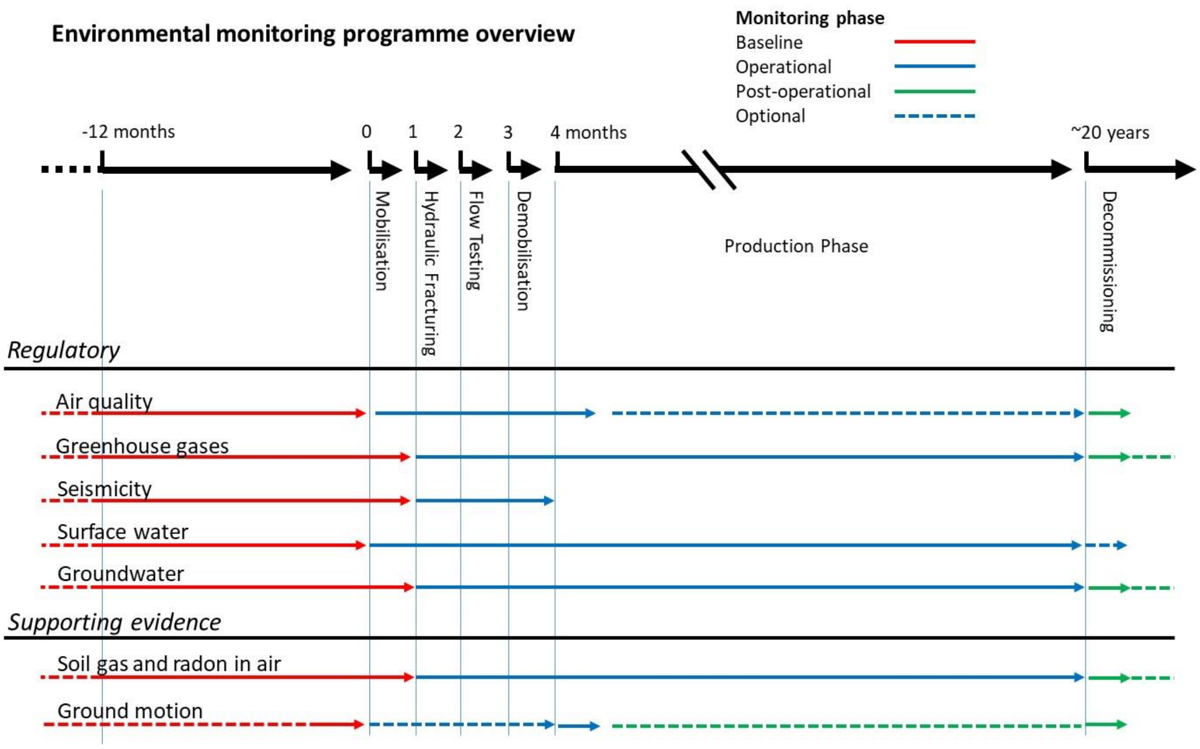


Figure 2. Environmental monitoring programme overview for shale gas development spanning the site lifecycle (solid lines represent recommended minimum period of monitoring and dashed lines indicate useful extended monitoring period).

Appreciation at the outset of the objectives, requirements and logical flow of information and data arising from the monitoring programme is crucial and is conceptualised in the Figure 3 monitoring flowchart. The activities itemised illustrate the logical flow from a foundational site conceptual model of understanding that enables appropriately informed baseline monitoring design and execution from which data arise and are processed to provide a statistical description of the baseline as well as iteration of the site conceptual model understanding. A key output of the baseline period shown is the establishment of change detector indicator values (if these have not already been defined) that determine the change threshold criteria to be taken forward into the operational and subsequent decommissioning phases. Operational phase monitoring and associated data processing to enable testing for change detection with respect to the threshold criteria established at baseline then allows informed decision making on any response actions required. Appreciation of this information flow and its key elements (Figure 3) detail variously referred to later) is foundational to optimal monitoring programme design.

2.2 CONCEPTUAL MODEL DEVELOPMENT

It is important for environmental monitoring of shale-gas operations to be fully integrated and not compartmentalised. The interdisciplinary synergies between the different technical monitoring approaches, data sharing and interpretation and conceptualisation of a site and its surroundings need to be holistic. To support this, a key element is the development of a site conceptual model. A site conceptual model allows the integration of current understanding of the environment and its condition, and provides a framework for identifying information and knowledge gaps and to support design of monitoring programmes. It is well recognised that the development of conceptual models is central to effective decision making and that they should be improved iteratively throughout the monitoring life cycle. The UK Government has published guidelines for risk assessment and management known as '*Green Leaves III*' (Cranfield University/Defra, 2011). This also establishes the importance of conceptual models to support decision making. European guidance to support the implementation of the Water Framework Directive and Groundwater Directive – both of which are relevant to shale gas development in the UK - provides an example of the design and use of conceptual models to support risk assessment for groundwater (EC, 2010).

In all elements of the environmental monitoring described in this report a conceptual model underpins the design of the relevant programme and the interpretation of the data collected. Figure 4 illustrates the importance of an integrated approach to conceptual model development to inform monitoring design.

2.3 BASELINE MONITORING

Baseline monitoring is the period of monitoring before any operational activity starts. It is carried out to define and characterise the 'normal' range of variation in relevant environmental parameters, spot any underlying or natural trends and to enhance the site conceptual model understanding. The frequency and range of monitoring data collected during this period of monitoring need to be sufficient to be able to characterise time-varying and other pre-operational influences on the environment and their contribution to the spatial and temporal characteristics of the baseline. Key elements of the baseline period monitoring are summarised in Figure 3 that develop from site selection, through data collection and processing to the development of a statistical description of the baseline and establishment of change threshold values. The detail of these aspects is covered within the specific monitoring sections that follow.

A broad range of measurements is required to characterise the baseline. In many cases, although valuable geological and hydrogeological area understanding and water and air quality and seismicity datasets may be gathered at desk study allowing preliminary site conceptualisation, detailed and/or relevant characterisation of the site environment and its baseline condition is unlikely to be sufficiently documented to predict the nature of future impacts and associated risks with certainty. Of particular interest in terms of monitoring measurements, are those that indicate introduction or mobilisation of contaminants and possible generation of pathways

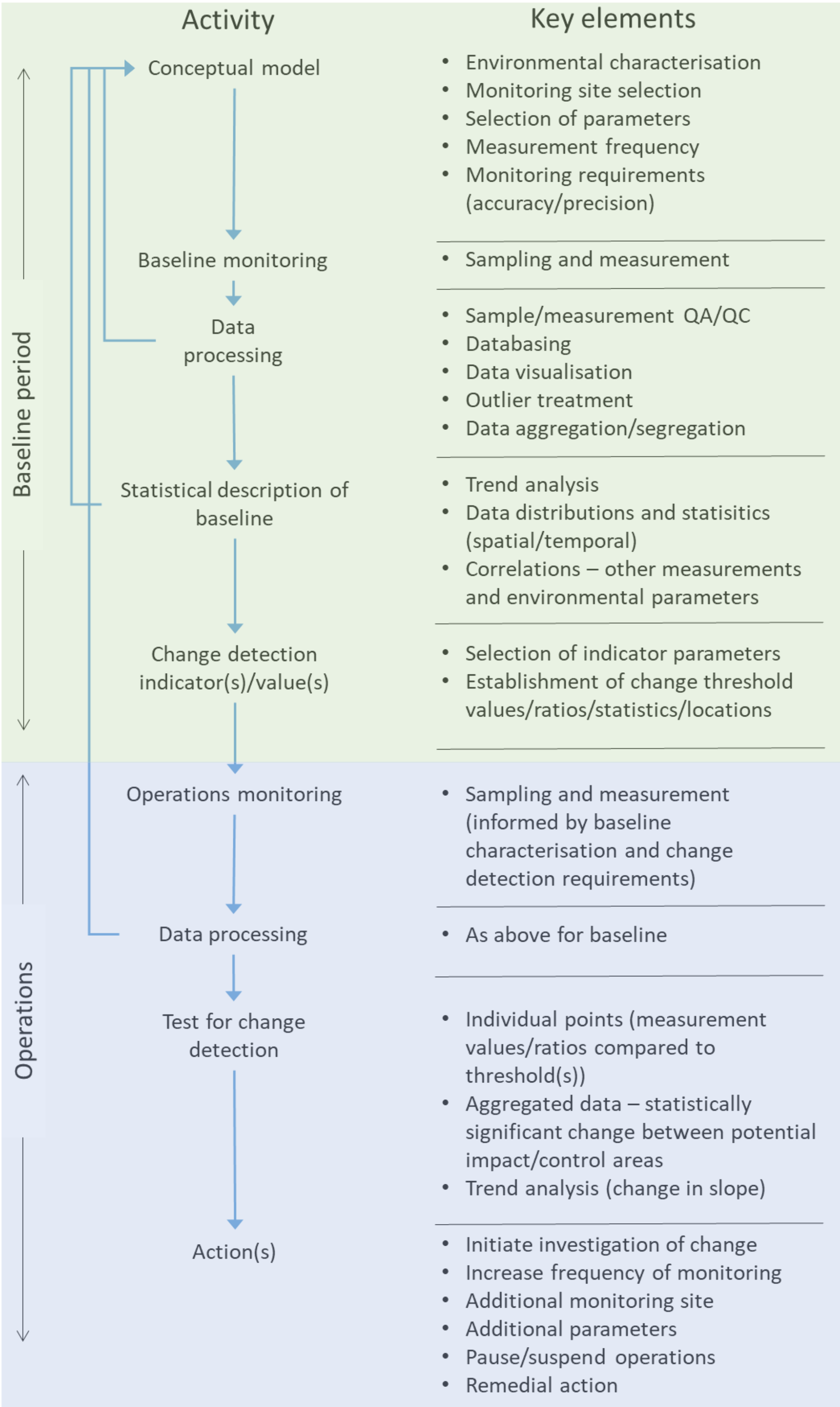


Figure 3. Monitoring flowchart of activities and key elements for a shale gas environmental monitoring programme

during shale-gas development; especially indicators that may provide an early warning of change. In this and other focus areas of monitoring, dedicated baseline monitoring infrastructure are typically required to allow bespoke assessment of the initial site condition, and detection of change.

Whilst establishment of a robust environmental baseline involving a diligent interdisciplinary approach may be perceived costly, such investment should ultimately prove cost-effective. A poorly-characterised baseline condition could, for example, lead to uncertainties in apportionment of subsequent environmental deviations and potential misattribution of cause. Change detection

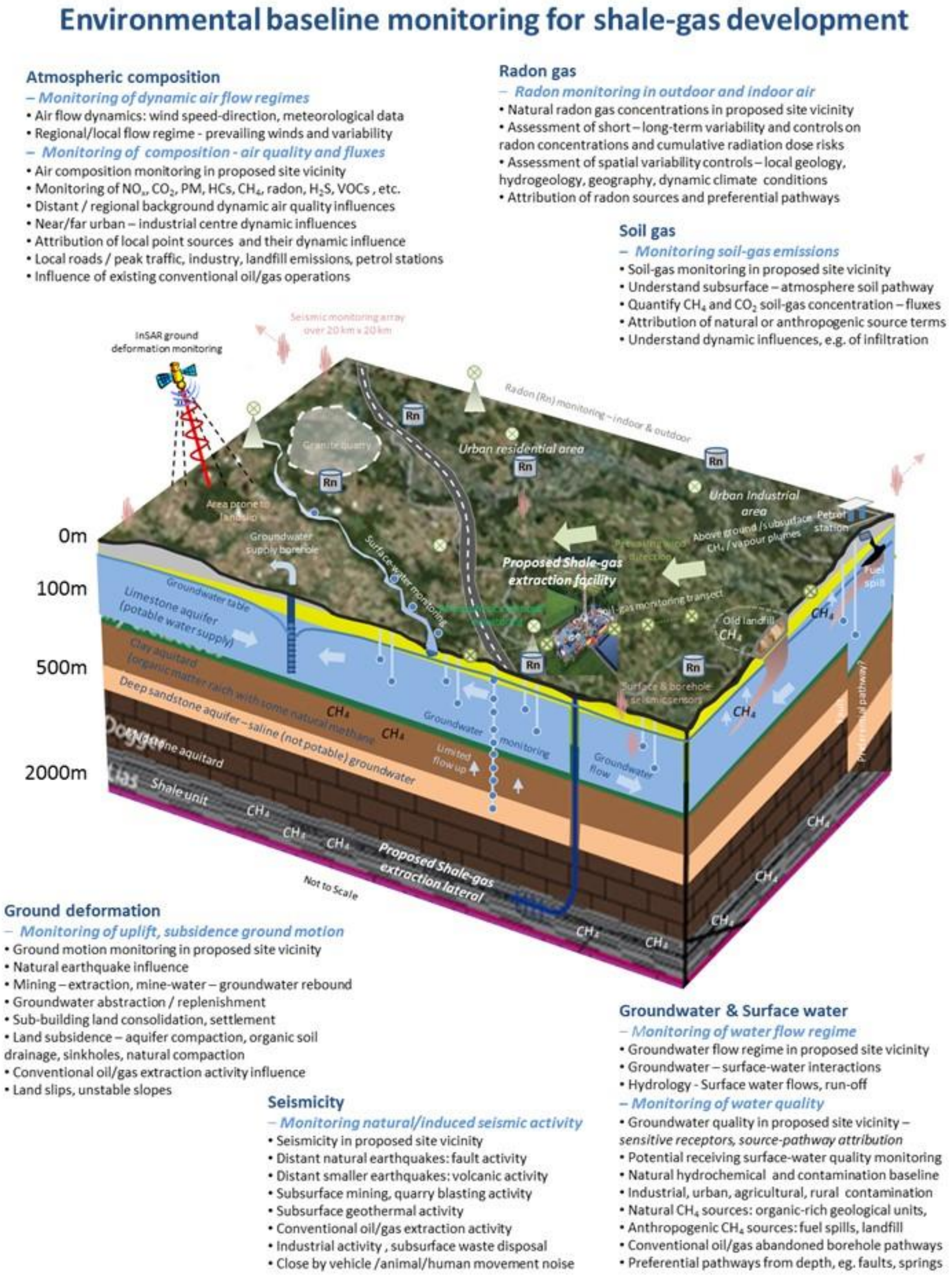


Figure 4. Conceptual model of a shale gas operation and local/regional environment showing key factors influencing environmental quality and monitoring design

Approaches to change detection in the shale gas context have been recently described in detail by the Environment Agency (2019b) for water and air. This includes description of statistical techniques appropriate to the detection of statistically significant changes from baseline conditions (not covered herein). Provided herein and complimentary to the above are summary commentaries on change detection specific to each of the monitoring activity types that aim to be practical and indicative of key issues that include case example illustrations where collected data are available.

Whilst approach detail may vary between the various strands of monitoring activity type, the fundamental essence of approach is similar and common considerations are introduced below. Key elements of the approach to change detection are also summarised in the Figure 3 monitoring flowchart. Fundamentally, tests for change detection implemented in the operational (and post-operational) phases are founded upon the detection indicator (threshold) values output from the baseline phase and involve the comparison of operational phase monitoring data with the statistical description of the baseline to evaluate whether thresholds of significant change are exceeded and a response action required.

2.4.1 Statistical baseline description to underpin change detection

Statistical characterisation of the variation of monitored data collected during the baseline monitoring period fundamentally underpins change detection assessment. Sound statistical principles should be followed to attribute any changes detected correctly. They should provide justification for adjusting sampling/monitoring frequency, analytical suites/methodologies and quality assurance/quality control (QA/QC) procedures. It is important therefore that appropriate statistical methods be adopted for monitoring programme design, monitoring frequency and reliability of measurements. Also, for the evaluation of monitoring data to enable statistical definition of the baseline and identify change and its significance.

To be able to detect change(s) arising from site operations, the pre-existing pattern of variation in a monitored variate, i.e. the baseline, needs to be quantified before any operational activity takes place. Baseline monitoring data are defined as measurements that characterise relevant environmental properties that are unaffected by shale gas development activities. Statistical description of such a baseline may typically comprises not only the definition of various means and variances, but also temporal trend analysis, consideration of spatial variations, and the establishment of correlations and relationships to other measurements and environmental parameters (Figure 3). Together these may provide a robust baseline signature against which any future change may be evaluated.

It should be recognised, however, that the variability in baseline may be complex and attributable to multiple sources or influences. For instance, a baseline largely associated with natural process origins (say natural methane steadily releasing from a geological unit) may have inherent variation due to natural process noise, but could also be subject to frequent, but intermittent spikes of influence from an anthropogenic source component (say dynamic methane emissions from a nearby landfill). Resorting to forensic methods, including the use of isotopic tools or chemical fingerprint signatures, might sometimes prove necessary to determine the provenance(s) of methane (or other chemicals of concern) to better understand the controls upon baseline signature variability. However, it must also be recognised that although baseline variability may be reasonably characterised, controlling factors can still remain elusive.

It should be likewise recognised that variability exists in all baseline (and operational phase) data due to environmental measurements having error. A degree of result uncertainty arises from random fluctuations in the performance of sampling/measurement systems, or any systematic bias introduced by the sampling and measurement systems. It is therefore important to characterise the uncertainty in measurements made during the baseline period and discern the contribution of individual component errors as far as possible.

Treatment of outliers detected in the baseline may be problematic and contentious. Whilst it may be tempting to dismiss the odd elevated ‘high’ (concentration) as ‘not real’ and perhaps say an analytical or simple data transcription error or artefact inadvertently introduced to a sample, outright dismissal is not prudent in that the data may in fact be real and documenting a sporadic occurrence in the baseline of high values actually found in the monitored system. It is important that the occurrence of outliers is recognised otherwise their continued, and perhaps increased occurrence in subsequent operational phases may lead to their erroneous association with shale gas activities.

2.4.1 Change detection post baseline

The primary objective of monitoring conducted during the operational and post operational stages of the site lifecycle is to ensure compliance with environmental permits and other conditions. The detection of change indicates the potential for a breach of permit conditions or unacceptable environmental impact.

For detecting change, data collected after the start of operations can be compared with baseline monitoring data primarily in two ways:

> (1). direct comparison of measurements with the baseline, e.g. by comparing data from individual monitoring points with pre-existing trends or ranges at the same monitoring point; or
>
> (2). by comparison of sets of data relating to two (or more) areas, e.g. for air quality, up and down (prevailing) wind direction of the site, or for radon comparing results from households close to the proposed shale gas site with those from a geographically distant control area with the same radon potential.

To define the frequency, duration and reliability (precision and accuracy) of a measurement, it is necessary to know the requirements for data interpretation. This will be informed by the baseline variability of a monitored parameter and the extent of change from the norm that would indicate an impact (significant deviation). It is important therefore that there is statistical confidence in the baseline and so monitoring needs to be carried out for a sufficient period of time and measurements taken at appropriate intervals. The baseline monitoring programme is therefore likely to be iterative with results continually being used to test and optimise it.

The purpose of operational (post-baseline) monitoring is to provide measurement data with which it is possible to confidently identify deviations from baseline and establish the significance of these with respect to risks posed and regulatory criteria. In change detection it is important to understand and define the uncertainty in measurement and establish a level that is acceptable. This should be informed by a number of factors including, for example, the proximity of the monitoring to sensitive receptors and the nature of the pollutant (source) or other hazard of concern, e.g. seismic activity. The guidance for each monitoring type addresses these and, where appropriate, uses illustrative case studies.

The selection of change detection indicator(s) value(s) or thresholds deemed to indicate a significant change worthy of further investigation and action is far from trivial. In some cases, regulatory standards already exist, e.g. the Traffic Light System for seismicity or Environmental Quality Standards for surface waters, but in other cases thresholds need to be established through a risk-based approach, as is the case for groundwater. In this case recognised risk assessment models such as the Environment Agency’s P20 Hydrogeological Risk Assessment (Environment Agency, 2006) methodology can be used. Tests for change detection may, for instance involve: individual points examining measurement values or ratios compared to threshold criteria; aggregated data whereby there is evaluation of statistically significant change between potential impact and control areas; and, trend analysis examining significant change in slope (Figure 3).

Where changes are identified as significant, further actions are triggered that may comprise suspension of operations or investigation of the cause of the change which may be supported by increased frequency, additional site monitoring or monitoring of additional parameters. The aim of any further investigation should be to extend the evidence base that is used for determining what further management actions are required. Key within the evidence gathering is to be able to correctly attribute the cause of the significant change detected which may or may not relate to shale gas activity and could arise from gradually changing natural processes or other anthropogenic activity changes. It is probable that any increased monitoring triggered may well involve a more forensic approach to provide correct attribution of cause given the actions, if proven to be shale gas-development related, are to suspend operations or implement possibly expensive remedial actions (Figure 3).

2.4 OPTIMAL TRANSITION TO OPERATIONAL-PHASE MONITORING

Monitoring data assessment and interpretation are required at different stages to meet different needs. During the baseline monitoring period they not only allow pre-operational environmental conditions to be characterised, but also should allow development of an optimised operational-phase monitoring plan in terms of parameters, frequencies and locations. It is prudent to ensure that monitoring capability established during the baseline appropriately transitions to and effectively underpins subsequent operational stages (Figure 2).

Due consideration should hence be given to the initial design and any subsequent iteration of baseline monitoring to allow the baseline established to optimally underpin the smooth transition and delivering of later stage monitoring requirements. For instance, the positioning of monitoring should be carefully considered at outset with a view to future requirements. It would be prudent, for example, to locate some baseline groundwater monitoring wells along suspected pathways of migration downstream of proposed key infrastructure localities. That said, the baseline may more confidently establish groundwater flow and potential plume directions that may still require further monitoring borehole installations to optimally monitor facilities in operational phases. The baseline period may likewise provide opportunity to optimise temporal monitoring frequencies and preferred parameter subsets to be measured at later stages.

It should be recognised that transition from the baseline into operational phase monitoring is not 'clear-cut' occurring on a specific date, but rather differs across the various monitoring types (Figure 2). Air quality monitoring, ground motion and surface water (that may be in receipt of direct discharges) each effectively transition from baseline to operational as soon as shale-gas plant mobilisation occurs as potential changes in signature might be expected from baseline from such site activity. This would not be the case for methane in air, soil-gas, seismicity and groundwater monitoring for which any changes in baseline signature may not be applicable until the onset of hydraulic fracturing. Even then, given the generally low flow rates of groundwater coupled with a range of physical, chemical and biological attenuation processes, expression of shale gas development any impacts would be much delayed. Impacts manifesting as changes in groundwater quality may also be slow to affect even relatively close monitoring wells. The early stages of operational phase monitoring of groundwater may hence be expected to continue to display baseline style signatures and delayed detection of shale-gas related problems. This contrasts with essentially instantaneous detection of air quality derogation or seismic events relating to shale-gas activity.

Finally, technology advances continually and so improvements in both precision and accuracy of measurements and new innovations in monitoring methodology are possible, indeed probable with time. Opportunities include for example, developments in capability for collection of automated or continuous (logged) data. Innovation may occur over operational and post-operational timeframes. Clearly, such opportunities should be considered, recognising comparison to baseline data may require some facilitation; for instance, a period of overlap of old and new monitoring technologies.

2.5 REFERENCES

Bell, R.A., Darling, W.G., Ward, R.S., Basava-Reddi, L., Halowka, L., Manamsa, K., O Dochartaigh, B.E. 2017. A baseline survey of dissolved methane in aquifers in Great Britain. Science of the Total Environment, 601-602. 1803-1813. https://doi.org/10.1016/j.scitotenv.2017.05.191.

CIWEM, 2016. Shale Gas and Water 2016. https://www.ciwem.org/policy-reports/shale-gas-and-water.

Cranfield University/Defra, 2011. Green Leaves III. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/69449/pb13670-green-leaves-iii-summary-111107.pdf.

EC, 2010. Common implementation strategy for the Water Framework Directive (2000/60/EC). https://circabc.europa.eu/sd/a/8564a357-0e17-4619-bd76-a54a23fa7885/Guidance%20No%2026%20-%20GW%20risk%20assessment%20and%20conceptual%20models.pdf.

Environment Agency, 2006. Remedial Targets Methodology. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/314317/geho0706bleq-e-e.pdf.

Environment Agency, 2019a. Onshore Oil & Gas Sector Guidance. https://www.gov.uk/guidance/onshore-oil-and-gas-sector-guidance.

Environment Agency, 2019b. Onshore oil and gas monitoring: assessing the statistical significance of change. Environment Agency report SC160020.

Public Health England, 2013. Review of the Potential Public Health Impacts of Exposures to Chemical and Radioactive Pollutants as a Result of the Shale Gas Extraction Process. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/740357/PHE-CRCE-009_3-7-14.pdf.

Royal Society/Royal Academy of Engineering, 2012. Shale gas extraction in the UK: a review of hydraulic fracturing. https://royalsociety.org/~/media/policy/projects/shale-gas-extraction/2012-06-28-shale-gas.pdf.

UKOOG, 2015. Guidelines for the Establishment of Environmental Baselines for UK Onshore Oil and Gas. http://www.ukoog.org.uk/images/ukoog/pdfs/Guidelines_for_the_Establishment_of_Environmental_Baselines_for_UK_Onshore_Oil_and_Gas_Issue_1_January_2015.pdf.

UK Task Force on Shale Gas, 2015. https://www.taskforceonshalegas.uk/reports.

3. Seismicity

3.1 INTRODUCTION

It is well known that anthropogenic activity can result in man-made or "induced" earthquakes. Although such events are generally small in comparison to natural earthquakes, they are often perceptible at the surface and a small number have been quite large with magnitudes greater than 5 MW. Underground mining, deep artificial water reservoirs, oil and gas extraction, geothermal power generation and waste disposal have all resulted in cases of induced seismicity (Davies *et al*., 2013). Such induced events represent a temporary perturbation to the background seismic activity in that region. Since natural earthquake activity is a response to long-term deformation from tectonic processes, such as first order plate motions, the rate of these earthquakes should remain stable when measured over long periods of time, whereas rates of induced earthquakes are likely to vary more strongly with time.

Earthquakes are the result of sudden movement along faults within the Earth that releases stored up elastic strain energy in the form of seismic waves or vibrations that propagate through the Earth and cause the ground surface to shake. The size of any earthquake depends on both the area of the fault that ruptures and also the amount of slip or displacement on the rupture plane. The larger the rupture area and the larger the displacement, the larger the earthquake. The amplitude of the ground vibrations depends on both the size of the earthquake and distance of the observer.

The aim of baseline seismic monitoring in the context of shale gas exploration and production (Majer *et al*., 2012) is to fully characterise background seismic activity in the area of interest by measuring transient ground vibrations in order to help discriminate between naturally occurring seismicity and man-made seismicity resulting from operations such as hydraulic fracturing. This must be established prior to the commencement of any activity that is known to induce earthquakes so that any changes in activity can be robustly identified. Baseline monitoring can also help to identify hidden/unknown active faults that may be affected by industrial operations.

Following the induced seismicity linked to fluid injection during hydraulic fracturing near Blackpool, UK, in 2011 (De Pater and Baisch, 2011), the UK Department for Energy and Climate Change (DECC, 2013) published a regulatory roadmap outlining regulations for onshore oil and gas (shale gas) exploration in the UK. These regulations contain specific measures for the mitigation of induced seismicity including: avoiding faults during hydraulic fracturing; assessing baseline levels of earthquake activity; monitoring seismic activity during and after fracturing; and, using a 'traffic light' system that controls whether injection can proceed or not, based on that seismic activity. Since existing networks of sensors in the UK are only able to reliably detect and locate earthquakes with magnitudes of 2 or greater, additional monitoring will be required to establish baselines of activity at lower magnitudes.

Local seismic monitoring requires the operation of a network of sensors whose basic purpose is the detection of earthquakes in the area of interest and the determination of accurate locations for these earthquakes (Lee and Stewart, 1981). Continuous monitoring using a well-designed network over a period of time should lead to a catalogue of earthquake activity that is not biased in time and space. However, this is not always a straightforward task, given that the reliable detection and location of earthquakes at small magnitudes is only possible using relatively dense networks of sensors designed to detect and locate these events.

Decades of experience in observational seismology has led to a large body of peer-reviewed literature on seismic monitoring and detection and measurement of earthquakes. Studies of earthquake aftershocks, fault-zone imaging and monitoring of small earthquakes associated with volcanic eruptions are some of the many subjects that can provide useful insights for baseline seismic monitoring. Experience in industries such as the geothermal (e.g. Edwards et al, 2015) and mining industries (Verdon et al, 2017) provides further insights into monitoring induced seismicity. Similarly, recent observations of seismicity related to hydraulic fracturing of unconventional hydrocarbon reservoirs also provides (Schultz et al, 2016; Yoon et al, 2017) also provides essential context on how this can be monitored and better understood. This has resulted in a considerable body of new research on seismicity related to fluid injection. Finally, the underlying theory of both earthquake behaviour in space and time and how seismic waves propagate through the Earth is relatively well understood and provides an essential framework for designing and installing a seismic monitoring network.

In this report we discuss some of the guiding principles for baseline seismic monitoring using a network of seismic sensors. These include; the design and installation of a network of sensors to ensure reliable detection and location of seismic activity in the area of interest; duration of monitoring and its dependence on background earthquake activity rates.

3.2 EXISTING CAPABILITY

The British Geological Survey operates a permanent network of seismic sensors to monitor seismic activity across the UK (Figure 4).Long term earthquake monitoring is required to refine our understanding of the level of seismic hazard in the UK. Although seismic hazard and risk are low by world standards they are not negligible, particularly with respect to potentially hazardous installations and sensitive structures. The monitoring results help in assessment of the level of precautionary measures which should be taken to prevent damage and disruption to new buildings, constructions and installations which otherwise could prove hazardous to the population. The network currently consists of 60 sensors with an average spacing of 50 km. This developed gradually over a period of around thirty years starting in 1969, and grew in size, both in response to specific events, such as the Lleyn Peninsula earthquake in 1984 (Turbitt et al, 1985), and as a result of specific initiatives, such as monitoring North Sea seismicity (Marrow, 1992). has been in place for several decades and is designed to detect all earthquakes with magnitudes of 2 or above throughout the UK, which are usually large enough to be felt by people nearby. Smaller earthquakes may be recorded but not uniformly, owing to irregular distribution of sensors. The density of the existing sensors is lowest in north, central and southeast England.

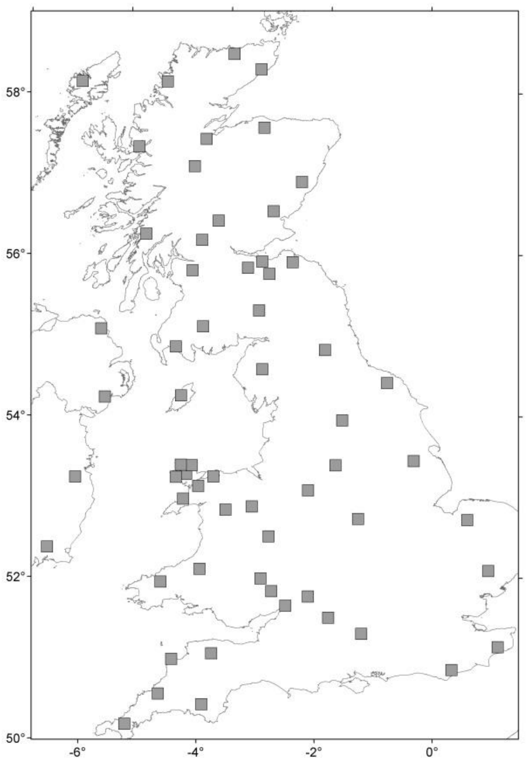


Figure 5. Permanent seismic monitoring stations operated by BGS in the UK along with stations operated by AWE and DIAS (Ireland)

3.3 NETWORK DESIGN AND SITE SELECTION

Detection and location of seismic events across a given area requires a network of seismic sensors. The density of the sensors along with the noise levels at each site control the lowest magnitudes that can be detected reliably. Higher sensor densities will be required to detect and locate lower magnitudes. This is because the signal amplitude is a function of both the magnitude of the earthquake and the distance of the earthquake from the recording position, and decreases with the square of the distance. An event may be undetected because it is too small or too distant, so its signal is indistinguishable from the background noise on the sensors. Also, many detection algorithms require the signal from an event to exceed the background noise level by a certain ratio on a number of sensors for an event to be detected. If the density of the sensor network is low, this will only happen for larger events. The detection of small earthquakes thus requires sensors that are close to the source because the amplitudes of the ground motions are small and are attenuated rapidly within the Earth.

Figure 6 shows ground velocity modelled using a stochastic approach that incorporates earthquake source parameters as well as parameters to characterise path and site effects (Boore, 2003) as a function of the distance from the hypocentre (point of rupture initiation) for earthquakes with magnitudes of -2.0, -1.0, 0.0, 1.0 and 2.0. This provides an indication of possible transient ground motions for earthquakes of these magnitudes, e.g. ground velocities for an earthquake with a magnitude of -1.0 at a distance of 30 km will be around $1x10^{-6}$ cm/s, while those for an earthquake with a magnitude of 0.0 at a distance of 10 km will be around $3x10^{-4}$ cm/s. The former will be below the noise level at almost any surface site. The later will be above the background noise level at all but the noisiest sites, where the background noise may be a result of human or natural noise sources (roads, railways, industrial operations, wind, sea).

This means that the number of sensors in the network depends on: (1) the extent of the area of

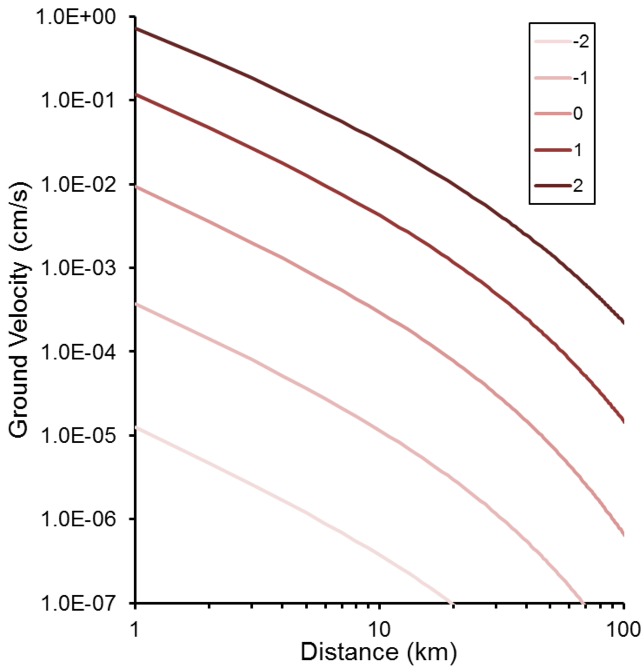


Figure 6. Modelled peak ground velocities (cm/s) for a range of earthquake magnitudes plotted as a function of hypocentral distance (km)

interest; (2) the minimum magnitude of the events to be detected; and (3) the required event location accuracy. It will also depend on the capability of any existing monitoring networks and the completeness of catalogues of seismic activity in the area of interest. Therefore, it is essential to decide what is required before monitoring begins.

3.3.1 The Monitored Area

The monitoring network should be able to provide comprehensive background monitoring over an area that is several times larger than the area of proposed exploitation. We suggest that the area of interest should extend at least 10 km from any possible future hydraulic fracturing operations, so a typical area for baseline monitoring might be 20 km by 20 km. However, this will depend entirely on the extent of the proposed operations. A monitoring network must also extend beyond the limits of the area of interest in order to be able to reliably detect earthquakes that occur close to these limits.

3.3.2 Number and distribution of sensors

Current regulations (BEIS, 2013) mean that hydraulic fracturing operations must be stopped temporarily if there are induced earthquakes with magnitudes of 0.5 ML or greater, therefore it would seem prudent to establish a baseline with a minimum magnitude at this level or less. This will require suitably sensitive monitoring networks to be deployed near sites of interest prior to any hydraulic fracturing operations, since existing regional seismic monitoring networks are not designed or capable of reliable detection of earthquakes with such magnitudes.

The detection capability of any seismic network is a complex function of many factors including the distribution, density and characteristics of individual stations, their local site and noise conditions, as well as processing software and processing strategies. The amplitude of the ground motions caused by any earthquake is a function of both the magnitude of the earthquake and the distance of the earthquake from the recording position. An event may be undetected because it is too small or too distant, so its signal is indistinguishable from the background noise on the seismograph. The detection of small earthquakes thus requires relatively high station densities. The detection threshold Also, many detection algorithms require the signal from an event to exceed the background noise level by a certain ratio on a number of stations for an event to be detected. If the station density is low, this will only happen for larger events.

In order to better understand how detection capability depends on sensor distribution and density, we model this using the amplitude of seismic waves as a function of magnitude and distance (Molhoff et al, 2019). Given the location of a network of sensors, we calculate the minim detectable magnitude across a grid of hypothetical earthquake epicentres as follows:

1. Calculate the distance, R, between the grid point and each station.
2. Calculate the amplitude, A, at each stations for a range of magnitude from the equation for the ML scale.
3. Find the smallest ML value for which the amplitude, A, is greater than three times the background noise for at least three stations.

The amplitude, A, is calculated using the equation for the ML scale (Havskov and Ottemoller ̈ 2010) as follows:

$$ML = \log_{10} A + a \log_{10} R + bR + c$$

where A is the maximum ground displacement amplitude measured with a Wood-Anderson (W-A) seismometer and the parameters a, b, and c are constants representing respectively geometrical spreading, attenuation and the base level which is used to anchor the scale to the original definition by Richter (1935).

For example, Figure 7(a) shows the theoretical detection capability of the eleven station seismic network that was installed around Kirby Misperton in 2015/2016. The contours show the magnitude of earthquake that can be detected at different points across a 40 km by 40 km grid centred on the Kirby Misperton site, where a signal in excess of three times the noise level needs to be recorded on at least three sensors for an earthquake to be detected. The noise levels at each site have the same value of 10 nm, which is representative of average UK daytime ambient noise levels in the 1-20 Hz range. The irregular distribution of the sensors is a result of the distribution of noise sources, the variability in the local geology and logistical constraints such as permissions, and causes some skewing of the detection capability The results suggest that a network of ten sensors with a spacing of a few kilometres should be sufficient for detection of magnitude 0.5 earthquakes across a 10 km by 10 km area. However, further from the centre of the network, only larger magnitudes can be detected, showing that the network must also extend beyond the limits of the area of interest in order to be able to reliably detect earthquakes that occur close to these limits.

Figure 7 (b) shows the theoretical detection capability of the Kirby Misperton network where the background noise level is assumed to be 4 nm and a signal in excess of three times the background noise needs to be recorded at three or more stations in order for an earthquake to be detected. The reduced noise levels show how smaller earthquakes may be detected in more favourable noise conditions.

Reliable estimation of event location and magnitude places additional constraints on network design, since measurements at more stations are needed than for detection alone. In addition,

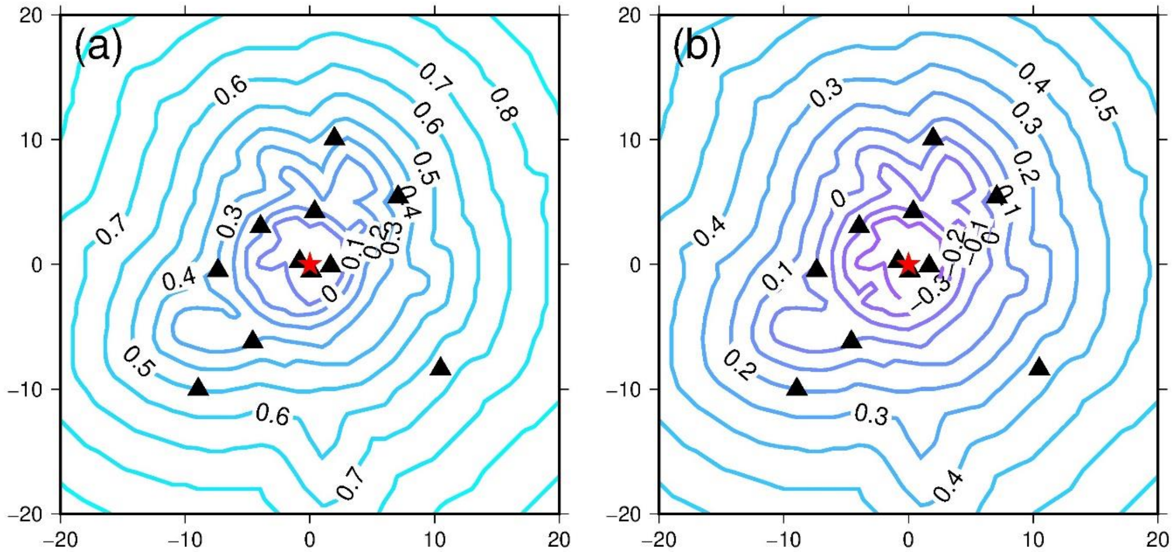


Figure 7. (a) Modelled detection capability of the eleven station seismic network that was installed around Kirby Misperton in 2015/2016. The contours show the magnitude of earthquake that can be detected at different points across a 40 km by 40 km grid centred on the Kirby Misperton site (red star). A signal in excess of three times the background noise needs to be recorded at three or more stations in order for an earthquake to be detected. A background noise level of 10 nm is assumed at all stations. (b) Modelled detection capability for the network of sensors around the Kirby Misperton where a signal in excess of three times the background noise needs to be recorded at three or more stations in order for an earthquake to be detected and a background noise level of 10 nm is assumed at all stations

location errors depend on the distribution and density of the recording stations. These errors may be large if the station density is insufficient, or if the closest stations are far from the earthquake source. For the lowest errors, the source needs to be surrounded by stations. Large errors are likely to limit the capability to discriminate between induced and natural earthquakes. Again, a uniform station density is required to ensure comparable location accuracy across the region of interest, with monitoring stations extending beyond the area of interest.

The measured arrival times of different seismic waves (e.g. P-waves and S-waves) at different points can be used to estimate the location of the seismic event. These depend on the distance from the source and the velocity of the medium, and, in general, will increase with distance from the source. Uncertainties in earthquake locations are dominated by three factors (Pavlis, 1986): (1) errors in the measured arrival times of the observed seismic waves; (2) modelling errors of calculated travel times; and (3), nonlinearity of the earthquake location problem. Measurement errors may arise because it is difficult to clearly identify the arrival time of the seismic phase because the signal is small and cannot clearly be discriminated from the noise. Assuming that the measurement errors are normally distributed, confidence regions may be computed. The size of the confidence regions depends on the variance and is commonly computed using either the F-statistic (e.g. Flinn, 1965) or the $\chi 2$ statistic (Evernden, 1969). The orientation of the error ellipsoid depends on both the number and geometry of the recording stations. For example, a single line of recording stations will result in significantly larger errors in the direction perpendicular to the line than along the line. When designing an experiment, it is important to position recording stations around the expected source location to get good azimuthal coverage. It is also important to have sufficient stations close to the expected location to constrain the depth of the events.

Figure 8 shows the errors in the earthquake source location for a given network geometry The source location is calculated using modelled P-wave arrival times at eight stations (triangles) for a source at a depth of 10 km and offset from the centre of a 100 km by 100 km grid by a distance of 40 km in the X direction and 30 km in the Y direction. The event was located using a probabilistic, non-linear, global-search earthquake location algorithm (Lomax et al, 2009). Gaussian noise, with a mean error of 0.1 seconds was added to the theoretical arrival times. The red dots show density-scatter representing the geometrical properties of the location probability distribution function, where regions with a higher probability of containing the earthquake have

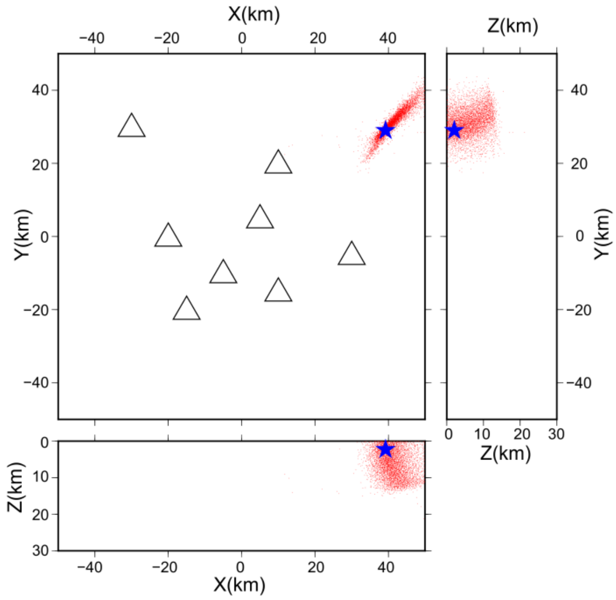


Figure 8. Location calculated for a modelled earthquake at a depth of 10km offset from the centre of a 100 km by 100 km grid, by a distance of 40 km in the X direction and 30 km in the Y direction, using P-wave travel-times for the eight sensors shown by the triangles. The blue star gives the maximum likelihood location. Red dots show the density-scatter in the location probability distribution function

a higher number of samples. The blue star gives the maximum likelihood location. The calculated epicentre is approximately 1 km from the true location; however, the calculated depth of 2 km is significantly different from the true depth. In addition, there is a large scatter in both epicentre and depth. The uncertainty in the epicentre is stretched out in the northeast-southwest direction, as a result of the geometry of the recording stations and the fact that the largest azimuthal gap is greater than 270°. This emphasises the need for a network with uniform station density, extending beyond the location of earthquakes of interest.

Figure 9 shows the calculated location of an event at a depth of 10 km in the centre of a 100 km by 100 km grid. calculated using only the predicted P-wave arrival times from all eight stations. The red dots show density-scatter representing the geometrical properties of the location probability distribution function, where regions with a higher probability of containing the earthquake have a higher number of samples. The blue star gives the maximum likelihood location. The calculated location is less than 100 m from the true location. The location is also well constrained with an epicentre error of only 1-2 km. The depth error is slightly larger, approximately 5 km, but remains well constrained.

3.3.3 Site Selection and Noise Levels

Ambient Earth noise is present in all recordings and can limit the ability to detect and reliably locate small transient signals from earthquakes or other disturbances. The noise levels at individual stations affects data quality and signal-to-noise ratios, so selecting sites where noise levels are low will maximise detection capability. Seismic noise from human activity is often referred to as "cultural noise" and originates primarily from the coupling of traffic and machinery energy into the Earth. This cultural noise propagates mainly as high-frequency

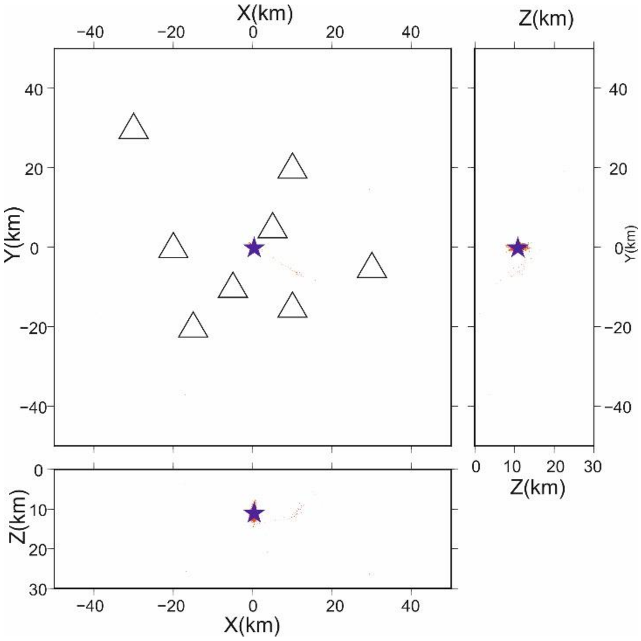


Figure 9. Location calculated for a modelled earthquake at a depth of 10 km at the centre of a 100 km by 100 km grid using P-wave travel-times for the eight sensors shown by the triangles. The blue star gives the maximum likelihood location. Red dots show the density-scatter in the location probability distribution function

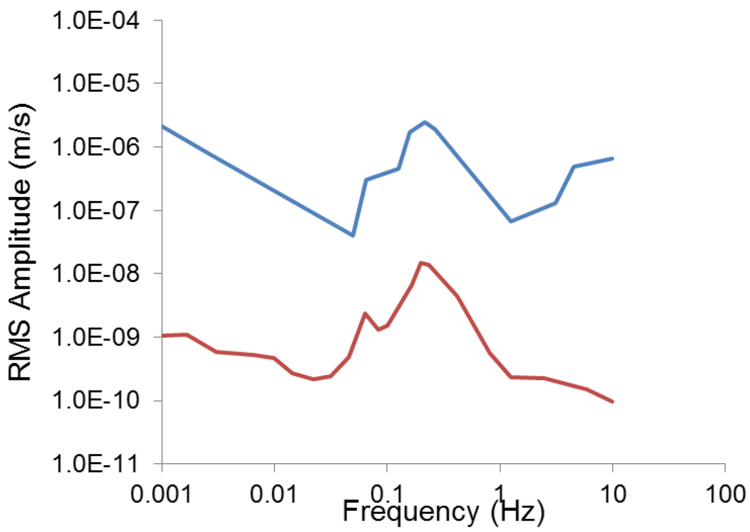


Figure 10. The USGS low noise (red line) and high noise (blue line) models (Peterson, 1993) expressed as RMS amplitudes of ground velocity in a constant relative bandwidth of 2/3 of an octave

surface waves (>1-100 Hz) that attenuate within a few kilometres of the noise source and often shows very strong diurnal variations.

Figure 10 shows the low and high noise values derived from recordings in the United States by Peterson (1993). Noise amplitudes at high noise sites may exceed the ground velocities expected for earthquakes with magnitudes of 0 or less at distances above 10 km.

Site selection should be based on both near-surface geology and the proximity of cultural noise sources such as roads, towns and villages, although logistical constraints are often also a factor. This is a particular problem in very dense arrays, where it can be hard to move away from these sources of noise. The aim of this is to choose a site with good coupling to bedrock and a minimum of cultural noise. Hard, dense rocks which have high seismic velocities are most suitable. Sediments such as clays or poorly consolidated soils, which have a low seismic velocity, act as efficient waveguides for ambient noise from cultural sources. Data recorded on sensors sited on unconsolidated sediments can have low signal to noise ratios, therefore efforts need to be made to avoid deploying in such situations, particularly where cultural noise levels are high.

3.3.4 Installation

Seismic sensors may be installed in a variety of ways, depending on the type of seismometer and the required performance. A comprehensive review of both seismometer installation and site selection can be found in Trnkoczy et al. (2002). Local seismometer networks designed to record high frequency signals from local earthquakes commonly have sensors deployed in shallow pits or buried in postholes. However, in all cases the seismometer should be well coupled to the ground. The following measures can help ensure this:

- Ensure that the sensor is stable, avoiding rough surfaces or surfaces covered with dust or sand.
- Ensure a direct coupling by installing the sensor on bedrock rather than on a buried boulder.
- In unconsolidated sediments it is often effective to directly bury the sensor in a shallow posthole. However, ensure that the instrument is suitable for this.
- Make sure that connecting cables do not exert additional forces on the sensor. Cables should loop round the sensor and be fixed to the ground.

Sensors may also require shielding from thermal effects, air-pressure, magnetic fields, humidity and electromagnetic fields and lightning.

- Insulating covers can be effective for temporary installations, but is only really essential for broadband sensors. Direct burial also provides good thermal insulation.
- Some sensors may be susceptible to water damage. Sensors can be deployed in sealed bags or containers to keep them dry. These should contain desiccant. When situating a pit, drainage should be considered, so that water does not naturally pool around the seismometer.
- Electromagnetic interference caused by strong fields must be avoided. There should be a suitable distance between any signal cables and AC mains power cables if the latter are used. Differential signal transmission should be used where possible and avoid loops of signal cables which might act like pickup coil
- Sensors can be vulnerable to lightning damage. Long analogue cables should be avoided where possible. Placing the sensor on a glass or Perspex plate can also help.

Research has shown that installing sensors in boreholes can significantly improve signal-to-noise ratios, which is critical for both recording of high quality data and the detection and measurement of small earthquakes. For example, Shearer and Orcutt (1987) compared borehole and surface recordings of both seismic refraction shots and earthquakes in the southwest Pacific, finding that the borehole seismometer had significantly better signal-to-noise advantage over the surface instruments. Our experience of baseline seismic monitoring in the Vale of Pickering suggests that noise signal power is 10-20 dB higher on surface sensors compared with borehole sensors at the same location at a shallow depth of 20-30 m. This translates to a reduction in the background noise level by a factor of around 10 in terms of RMS amplitude.

As a result, sensors deployed below the surface in shallow boreholes are likely to offer significantly better performance for background monitoring than surface instruments. Such arrays have become standard practice for the operational phase of many geothermal projects (Majer et al, 2007) and also for microseismic monitoring in the UOG operations (Rutledge et al, 2004). However, boreholes are costly to drill, and the specialist seismic equipment required for them expensive. A borehole deployment can cost up to ten times that of normal surface station.

If installing in a shallow pit or posthole, it is important not to be too close to structures that couple wind noise with the ground. The most obvious example of this is trees, although pylons and even some fences can be problematic if the pit is very close. With trees, a good rule of thumb is that the pit should be at least as far from the trunk as the tree is high.

Animals, such as sheep and cows, can damage equipment, as well as act as a source of seismic noise. Sites in fields used for grazing should be fenced. Finally, the possibility of vandalism should be considered. Sites on public land or near to footpaths should be as inconspicuous as possible.

3.4 MEASUREMENTS

3.4.1 Instrumentation

Earthquake source parameters such as origin time, location and magnitude are commonly determined from high resolution recordings of ground motion as a function of time. Ground displacements in an earthquake magnitude range of -2 to 8 can range from around 10^{-10} to 10^{-1} meters (Bullen and Bolt, 1985), so that sensors with a high dynamic range are needed to capture a range of magnitudes. In addition, sensitivities below typical Earth noise levels are needed to record the smallest detectable events. Modern sensors with high sensitivity and a dynamic range of around 140 dB are recommended. Either seismometers, which measure ground velocity or accelerometers, which measure ground acceleration, may be used for local earthquake monitoring.

Similarly, the digital recording equipment needs to have a high dynamic range and this should be achieved through the use of a 24-bit recording system or better. Digital data must be timestamped using reliable absolute timing measurements so that signals from different sensors can be compared. GPS or similar clocks should be used for this.

The sensor must also provide high sensitivity over a relatively wide range of frequencies. In simple terms, the observed frequencies for earthquake ground motions are largely controlled by the magnitude of the earthquake. Figure 11 shows modelled velocity amplitude spectra for earthquakes with magnitudes of -2 to 2, for a fixed stress drop of 1 MPa. In general, frequency content decreases as magnitude increases, so that earthquakes with magnitudes of around 2 might have a frequency content of around 10 Hz, whereas earthquakes with magnitudes of -1 may have frequencies in excess of 100 Hz. Local microseismicity, with magnitudes of 2.0 or less, will have a higher frequency content than large distant earthquakes from elsewhere in the world, therefore “short period” seismometers, which have a linear response to ground velocity at frequencies of above 1 Hz may be suitable, as well as broadband sensors. High frequency

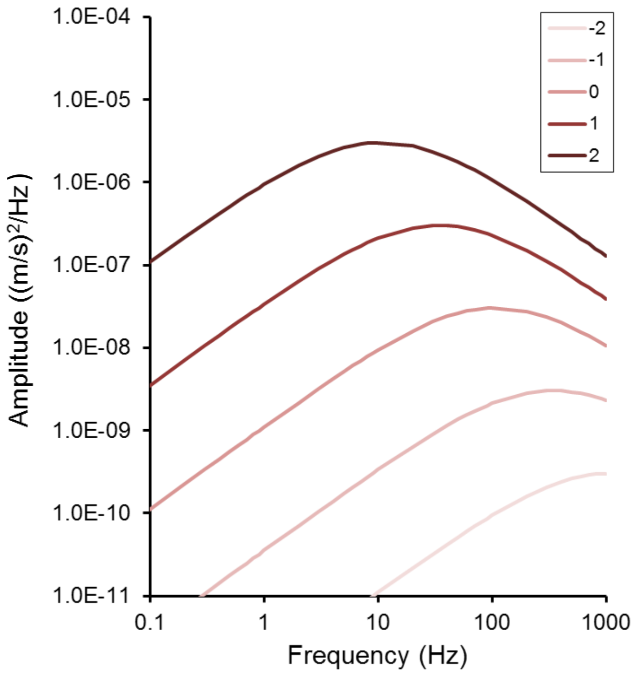


Figure 11. Modelled velocity amplitude spectra for different earthquake magnitudes. Frequency content increases as magnitude decreases

geophones with appropriate characteristics over the expected frequency range may also be suitable. In addition, sample rates in excess of 200 Hz are likely to be needed to reliably record earthquakes with magnitudes of less than 1.

3.4.2 Network Metadata

Seismic sensors produce an analogue output that is proportional to ground motion and digital recording converts this to a number of digital counts for each sample. In order to determine the actual ground motion the calibration information or instrument response of the sensor and recording equipment must be known. Instrument calibration and response data are usually supplied by the manufacturer and can also be tested using a number of methods, allowing the deployed instrumentation to be tested in situ at regular intervals. However, detailed calibration will usually require the equipment to be returned to the manufacturer. The calibration information must be stored in such a way that it can be applied easily to the raw data and must be carefully updated whenever changes to instrumentation are made.

It is vital for the usefulness of a network that full and complete metadata is created and maintained. This should include the location of each site and the serial numbers of all instrumentation present as well as the instrument response data, along with a history of any changes to that instrumentation due, for example, to failure. It is also essential that all archived data contain not only the raw data but also the instrument response information.

3.4.3 Data Completeness

As well as providing high data quality, it is important that the network provide a high degree of data completeness, i.e. there are no significant gaps in recording due to instrumental failure. A completeness of greater than 90% is desirable.

For baseline monitoring studies, data may be recorded locally at each station and collected at regular intervals for subsequent analysis. This, however, can lead to poorer data completeness, as there is no way to know that there is a problem at a station between data collections. It is better if data from individual stations is transmitted to a central recording site using a suitable form of data telemetry. This allows near real-time processing of the data for rapid identification of any events, as well as giving the promptest possible warning of any failures. Telemetry will add to hardware costs but these can be offset by the need for fewer visits to the sites.

Loss of power at a station is the most common reason for data loss. If regular data collection visits are being made then the batteries that power the equipment can be replaced each time. Otherwise, either solar panels can be used or mains power made available. It is worth considering that telemetered sites require much higher power than non-telemetered sites.

3.4.4 Duration of Monitoring

Reliable determination of earthquake activity rates requires a representative sample of events at a range of magnitudes. Therefore, the duration of the background monitoring is likely to depend on both the background earthquake activity rate and also the presence and capability of existing monitoring networks. Earthquake activity rates can vary from place to place, but in a region of homogeneous seismicity, the number of earthquakes above a given magnitude in any sub-region scales with the relative size of the two regions. For example, if a region where seismicity is homogeneous has 1000 earthquakes above a magnitude of zero each year, then a sub-region, whose area is ten times smaller, will have 100 earthquakes above a magnitude of zero each year. This has important implications for baseline monitoring in small regions, particularly where activity rates are low, since the number of earthquakes in a given period of time may be very low, so longer durations of baseline monitoring are required to reliably determine seismicity rates.

In the UK, we record around 17 earthquakes of a tectonic origin with a magnitude of 2.0 or above somewhere in mainland Britain every year and around 500 with a magnitude of 0.5 ML or above. Assuming that seismicity is homogeneous, a 20 km by 20 km sub-region will have an earthquake with a magnitude of 2.0 or above only every 65 years, and three earthquakes with a magnitude of 0.0 or above every two years. As a result, operating a local network for only a year or two is unlikely to contribute significantly to better quantification of seismic activity rates unless it can reliably detect and locate earthquakes with very low magnitudes throughout the region of interest. However, by dense seismic monitoring for one or two years it may be possible to determine if seismic activity rates are significantly different from the national average and to identify seismicity associated with specific fault structures that may be affected by future hydraulic fracturing operations.

Finally, it is important to continue monitoring both during and after any future hydraulic fracturing operations to allow induced events to be discriminated from natural seismicity and ensure adherence to local vibration guidelines.

3.5 DATA PROCESSING, ANALYSIS AND REPORTING

3.5.1 Event Detection

As we have seen, seismic monitoring requires the operation of a network of sensors that continuously record ground motions. However, we are mainly interested in transient seismic events such as earthquakes that are contained in short periods of the continuous data. These events need to be detected and extracted from the continuous recordings, either in real-time or retrospectively and a wide variety of algorithms have been developed for this purpose over many decades of observational seismology. In low noise conditions, this task may be relatively straightforward; however, the presence of noise makes it much more difficult.

Detection algorithms can be divided into two very general types: energy based methods that use some attribute of the signal energy to detect an event; and, pattern matching methods that use the similarity of the entire waveform to detect an event. One of the simplest energy based methods uses the ratio of the amplitude of the signal in a short time window to the amplitude in a longer time window. This is often called the short-term average (STA)/long-term average (LTA) method (Allen, 1982, Baer and Kradolfer, 1987). If this ratio exceeds a given threshold at multiple sites within a time window that is consistent with a seismic source, then an event is detected.

The STA/LTA method requires no prior knowledge of the event waveform or the source, but it may fail or produce many false detections where the signal-to-noise ratio is low, where arrivals are emergent, or where many events occur within a short period of time. This means the method has a low detection sensitivity and may not be able to detect low magnitude events consistently.

When noise levels are very high and signal levels are low then event detection is very difficult. There are essentially two ways to try to improve this situation: (1) better data, e.g. deploying more sensors to improve signal to noise ratios by reducing distances between the sources and receivers; or, (2) the use of more sophisticated detection methods. Pattern matching methods compare the waveforms of known events with other recorded signals, often using waveform correlation (e.g. Schaff and Richards, 2004). If the signals are sufficiently similar, then an event is declared. Such methods have a high sensitivity, but may require some prior knowledge of the expected signals, which may not be available, and can be computationally expensive. Yoon et al (2015) use new data mining algorithms which are very computationally efficient to allow very fast pattern matching. However templates are still needed in order to identify events.

3.5.2 Event Location

An impulsive source of seismic energy can be thought of as a point source in time and space, defined by an origin time (t_0) and hypocentre (x_0, y_0, z_0), respectively. The travel time of a seismic wave propagating away from such a source will depend on the distance from the source and the velocity of the medium, and, in general, will increase with distance from the source. Measured arrival times at different points can be used to estimate the location of the seismic event. The problem of estimating source location from travel time data has been studied extensively in earthquake seismology and numerous algorithms of this type have been developed and are in widespread use. Given observations of arrival times at a number of points we can compute predicted travel times to the same points by assuming a reference velocity model. We can then try to minimise the difference between the observed and modelled travel times and estimate the best fitting location for the event. Although the travel-times are not linearly dependent on the earthquake location, the problem can be linearised by considering only small perturbations from an initial target location. Iterative, linearized methods are based largely on the method of Geiger (1912) and solve the problem using partial derivatives and matrix inversion. These usually converge rapidly unless the data are badly configured or the initial guess is very far away from the mathematically best solution. Nonlinear methods (e.g. Lomax, 2000) solve the earthquake location problem by sampling the full solution space. They have the advantage of obtaining a more complete estimate of uncertainties as compared to the linearized methods and do not rely on the quality of an initial guess.

A minimum of four independent measurements are needed to determine the location of an earthquake. However, the results will have little value due to their uncertainty. There are a number of "rules of thumb" of what is commonly required to obtain well constrained earthquake locations (e.g. Bondar et al, 2004). These include the following:

- At least eight arrival time measurements.
- At least one S-wave arrival time measurement.
- At least one arrival from a station within a focal depth's distance from the epicentre.
- The largest azimuthal gap between two stations should not exceed 180°.

S-wave arrivals within 1.4 focal depth's distance from the epicentre also provide significant constraint on the focal depth (Gomberg et al, 1990). Denser networks will result in better azimuthal coverage and stations and better depth constraint. In order to provide a uniform location capability within a given region, a network of stations must extend beyond the region itself, otherwise, the capability to locate earthquakes at the edges of the monitored region will be compromised.

3.5.3 Magnitude Estimation

Earthquake magnitude is a measure of the amount of energy released during an earthquake and can be determined from the amplitude of the ground motions caused by the earthquake. We also need to know how far away the earthquake was because the amplitude of the seismic waves decreases with distance. The first magnitude scale was developed by Charles Richter in 1935, based on observations of earthquake ground motions in California. Although this magnitude scale is only strictly applicable in California, it has been used all around the world and is commonly referred to as Local Magnitude, M_L. Richter (1935) defined this as

$$M_L = \log_{10}\left(\frac{A}{A_0}\right)$$

where A is the maximum deflection, zero to peak in millimetres registered by the earthquake on a Wood-Anderson seismograph, and A_0 is the deflection produced by a "standard" magnitude zero earthquake at the same distance. The A_0 factor allows observed amplitudes to account for decay between the seismograph and the epicentre of the earthquake. Values for A_0 are given by Richter (1935) to distances of 600 km. A magnitude 3 earthquake was defined as a 1mm displacement at 100km. Although Richter intended his method to be an approximate quantification of earthquake size and his attenuation term, A_0, strictly only applies to California, the formula is still used worldwide today.

Local magnitude is generally only applicable to observations of small to moderate earthquakes at local and regional distances. For larger earthquakes, the scale saturates and at larger distances records are dominated by long period surface waves. All earthquakes in the BGS earthquake catalogue have been assigned a local magnitude (ML) as defined by Richter (1935). Ground motion records are converted to the equivalent Wood-Anderson deflection and the maximum amplitude is measured for each. Ideally, the measurements are made on the two horizontal components of ground motion and then averaged. Ground motion registered at a seismograph varies with site conditions, distance and direction from the earthquake, and the nature of the ray path. Therefore, it is important that the calculated magnitude is an average from a number of recordings at different sites. The resulting errors on magnitudes quoted in the bulletin will normally be less than 0.4 ML.

It is important to note that Richter's local magnitude scale is empirical, and although it was derived from ground motions measured at a range of distances, this did not include any measurements made within a few kilometres of the earthquake source. As a result, it cannot be assumed that it will work at these distances. The very small earthquakes discussed in this report will generally only be recorded at very nearby stations. Recent research has shown that amplitude measurements from epicentral distances of less than 15-20 km considerably overestimate event magnitudes compared to more distant observations (Butcher et al, 2016). Similarly, magnitudes calculated for earthquakes induced by hydraulic fracturing at Preese Hall, Lancashire (Clarke et al., 2014) using ground motions recorded on seismometers distances of a few kilometres away were unrealistically high. Since existing UK regulations (DECC, 2013) require that hydraulic fracturing operations stop if earthquakes with magnitudes of 0.5 ML or greater are induced, reliable estimation of magnitude is essential. The UK local magnitude scale published by Luckett et al. (2019) addresses this and incorporates a correction for near-source observations.

An alternative magnitude measurement that can be applied to earthquakes recorded at local and regional distances is moment magnitude, M_W (Hanks and Kanamori, 1979). This is based on seismic moment, which is related to both the area of the rupture and the displacement on the rupture. M_W can be derived from measurements of seismic moment M_0 that are calculated from the amplitude spectrum of ground displacement records after they have been corrected for source radiation pattern, geometrical spreading and path dependent attenuation (e.g. Edwards et al, 2010). The latter may vary strongly with geology. The calculated seismic moment also depends on the velocity and density of the rocks at the earthquake source depth, so it is important that the velocity depth model is well constrained, otherwise, the seismic moment estimates are likely to be incorrect. Stork et al. (2015) present an assessment of how moment magnitude estimates vary with the method and parameters used to calculate seismic moment. For example, Mw estimates can depend on the length of the measurement window. Given this, moment magnitude is more complicated to determine than local magnitude, although the scale should work at all distances and has the advantage of providing greater insights into the source properties of the earthquake. However, given that the current traffic light system to mitigate induced seismicity is specified in terms of local magnitude, a robust a reliable means of determining local magnitude is essential.

3.5.4 Behaviour in space and time

The relationship between the magnitude and number of earthquakes in a given region and time period generally takes an exponential form that is referred to as the Gutenberg-Richter law (Gutenberg and Richter, 1954), and is commonly expressed as

$$\log_{10} N = a - bM$$

where, N is the number of earthquakes above a given magnitude M. The constant a, is a function of the total number of earthquakes in the sample and is known as the earthquake rate. This is commonly normalised over period of time, such as a year. The constant b gives the proportion of large events to small ones, and is commonly referred to as the b-value. In general, b-values are close to unity. This means that for each unit increase in magnitude, the number of earthquakes reduces tenfold.

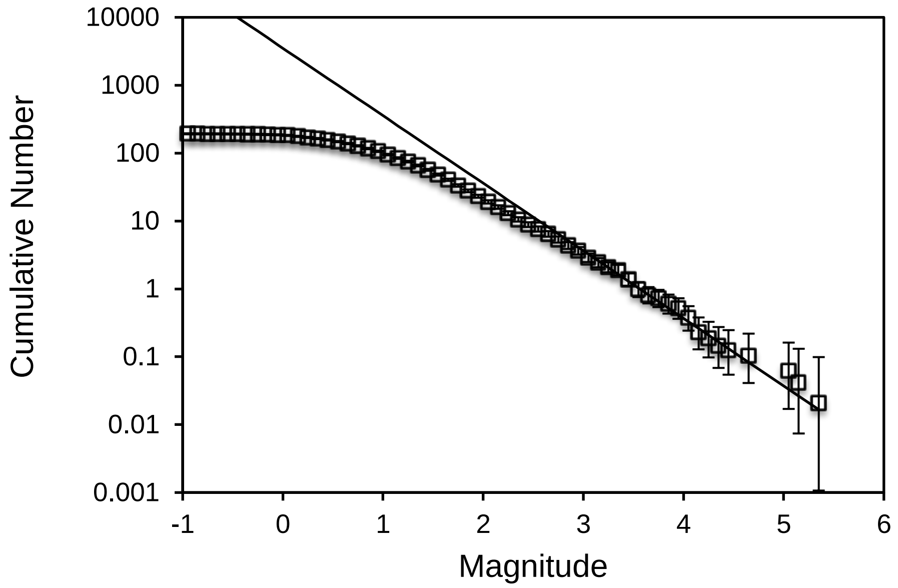


Figure 12. The number of earthquakes above a given magnitude plotted against magnitude. Earthquake data from the British Geological Survey UK Earthquake Catalogue © NERC 2016

Plotting earthquake magnitudes against the logarithm of frequency (Figure 12) gives a straight line, where the slope of the line is the b-value and the rate, a, is the value where the line intersects with a given reference magnitude (often zero). The b-value should be estimated using a maximum likelihood method (e.g. Aki, 1965) rather than a least-squares fit. An observed roll-off in the number of earthquakes at low magnitudes shown by observed data (squares) due to inability of regional seismic networks to detect small earthquakes. This roll-off in the magnitude-frequency relationship at low magnitudes leads to the concept of a completeness magnitude, Mc, which can be defined as the lowest magnitude at which 100% of the earthquakes in a space-time volume are detected (Rydelek and Sacks, 1989). A number of techniques can be used to assess the magnitude of completeness of a seismicity catalogue. See Mignan and Woessner (2012) for a comprehensive review.

3.5.5 Outputs

The primary output of any monitoring study should be a comprehensive catalogue of seismic events within the region of interest for the time period that the monitoring network was in place. Such a catalogue should contain at least the source parameters for each event (origin time, latitude, longitude, depth and magnitude) along with the errors in these parameters. Other useful information might include macroseismic information, e.g. was the earthquake felt by people, and if so, at what intensity.

The arrival time or phase data for each event should also be made available as required, so that users can use this to determine new source parameters as required.

Finally, the recorded time series and metadata, both for individual events and the continuous recordings from each site should be made available in an internationally recognised format for data exchange.

The results of the monitoring should be disseminated through the Internet, either through specific web pages or by other means. These should be updated in near real-time if possible to ensure transparency and that data is available for public scrutiny.

3.6 CASE STUDIES

The baseline seismic monitoring and subsequent monitoring of induced seismicity related to early shale gas development in England means that there is now direct experience of the effects of hydraulic fracturing. However, given that experience is limited to only one area of the country, it is too early to consider this as a definitive indication of what might be observed at other sites. A summary of the measured induced seismicity at the Preston New Road shale gas site in Lancashire is provided as a case study, and additionally, two other studies are described to illustrate the establishment of seismic monitoring for detection of low magnitude induced seismicity events triggered by other industrial processes. The experience and recommendations arising from these UK case studies has informed the guidelines contained in this report.

3.6.1 Seismicity induced by hydraulic fracturing operations at Preston New Road, 2018

In 2011, hydraulic fracturing of the first dedicated shale gas well in the UK (Preese Hall 1), near Blackpool, led to felt seismicity that resulted in the suspension of operations and a government enquiry to assess the risk of induced seismicity. Clarke et al (2014) concluded that the seismicity resulted from the interaction of hydraulic fracturing fluids with a previously unmapped fault. Subsequently, the UK Department for Energy and Climate Change (BEIS, 2013) published regulations for onshore oil and gas (shale gas) exploration in the UK that contained specific measures for the mitigation of induced seismicity, including using a ‘traffic light’ system (TLS) to control whether injection can proceed or not, based on that seismic activity (e.g. Bommer et al., 2006). This TLS requires operators to stop hydraulic fracturing if an event with a magnitude of 0.5 ML or above occurs during operations.

In late 2018, hydraulic fracturing of the Carboniferous Bowland Shale was carried out at the Preston New Road 1 site (PNR-1), approximately 4 km south of Preese Hall. Again, operations were accompanied by microseismicity (Clarke et al., 2019).

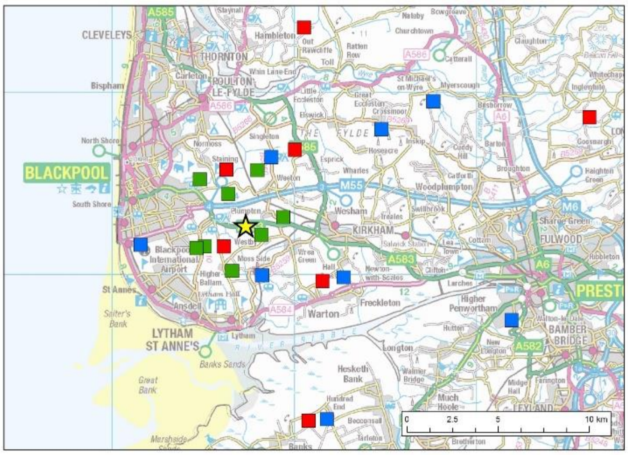


Figure 13. Surface seismic monitoring stations installed at Preston New Road. Monitoring Stations around Preston New Road.Red squares show BGS stations, green squares show stations operated by Cuadrilla Resources, blue squares show stations operated by the University of Liverpool. Contains Ordnance Survey data © Crown copyright and database rights. All rights reserved [2020] Ordnance Survey [100021290 EUL]

A dense network of surface sensors was installed by the British Geological Survey (BGS) (Figure 13) and the operator, Cuadrilla Resources Ltd, to monitor any induced seismicity, partly in order to comply with regulatory requirements. A total of 57 microseismic events were detected in near real-time using a conventional, energy transient detection algorithms (Baptie and Luckett, 2019). 22 of these had magnitudes greater than 0.0 ML, the amber TLS threshold and 7 had magnitudes greater than the TLS limit of 0.5 ML. The largest event had a magnitude of 1.6 ML and was felt by a few people close to the site.

By contrast an array of borehole geophones installed by the operator detected over 38,000 microseismic events during the period of operations, including the largest event of local

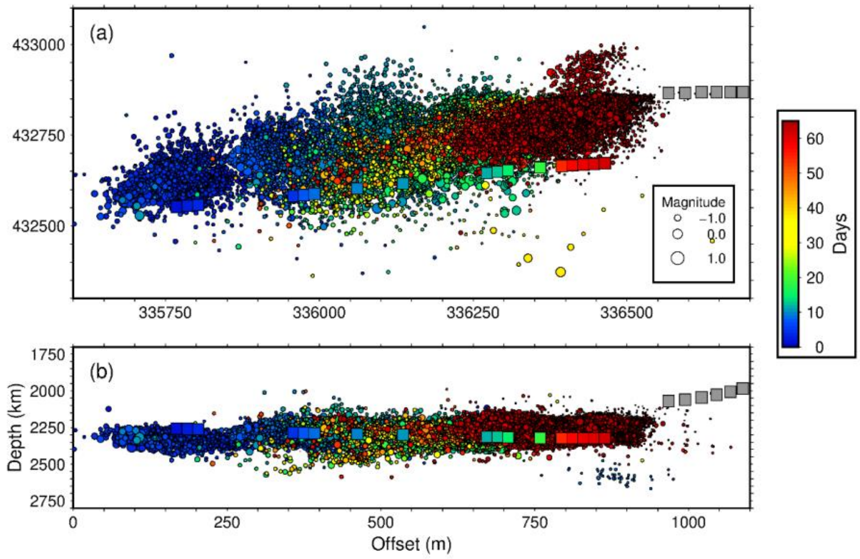


Figure 14. Seismicity as a function of time during operations (red circles). Circles are scaled by magnitude. Blue lines show the cumulative volume of injected fluid during hydraulic fracturing operations. The magenta line shows the cumulative flow-back volume. No hydraulic fracturing was carried out between 3 November and 4 December as flow-back from the well took place. From Baptie and Luckett (2019)

magnitude 1.6. Seven events equal to or greater than magnitude 0.5 occurred, exceeding the UK regulatory threshold that requires operators to halt and pause operations.

The PNR-1z well targets the Bowland shale at a depth of approximately 2,300 m, and runs approximately east-west for 700 m horizontally through the unit. A sliding-sleeve completion method was used, with 41 individual sleeves spaced at intervals of 17.5 m along the well. The hydraulic fracture plan allowed for up to 765 m^3 of fluid per sleeve. A "mini-frac" consisting of a few 10s of m^3 of fluid was pumped prior to each main stage. The sleeves were numbered from 1 to 41 proceeding from the toe (west) to the heel (east) of the well. A total of 16 sleeves were hydraulically fractured with an additional 18 mini-fracs between 16 October 2018 and 17 December 2018. The sleeves were used in the following order: 1, 2, 3, 12, 13, 14, 18, 22, 30, 31, 32, 37, 38, 39, 40 and 41. The average injected volume for each fracture was 234 m^3 and the maximum injected volume was 431 m^3. No hydraulic fracturing was carried out between 3 November and 4 December as flow-back from the well took place.

Locations for all events in the downhole microseismic catalogue are shown in Figure 14. Events are coloured by time and move from west to east corresponding to different stages of hydraulic fracturing in the horizontal well PNR-1z. The locations of the events closely correspond to the positions of the sleeves that were hydraulically fractured (coloured squares in Figure 15). Event depths are around 2280 m, but decrease slightly from around 2300 m at the toe of the well to approximately 2250 m closer to the heel.

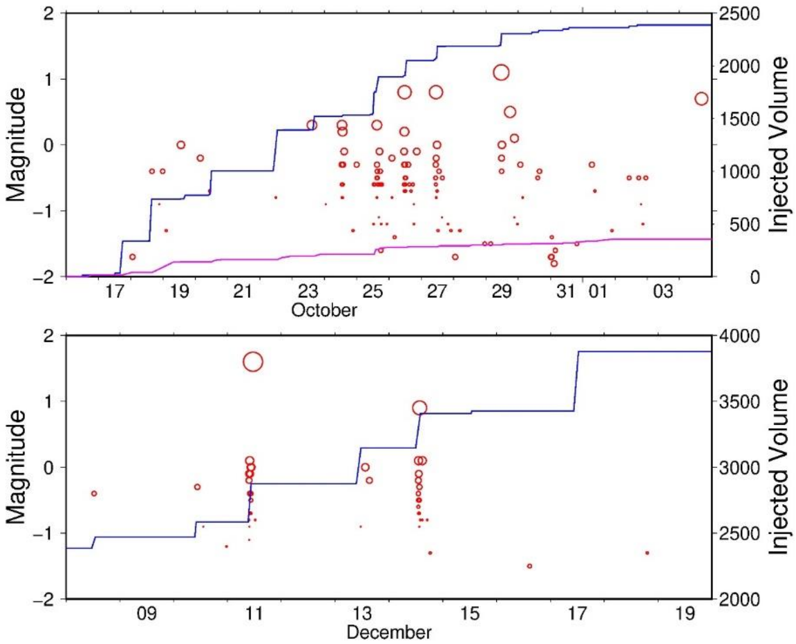


Figure 15. (a) Map of all events in the microseismic catalogue. Events are coloured by time and scaled by magnitude. The coloured squares show the locations of the sleeves that were hydraulically fractured. The squares are coloured using the same colour scale as the events. Axes show British National Grid Eastings and Northings. Grey squares show geophone positions. (b) Depth cross-section showing event depths along an east-west profile

Figure 15 shows seismicity detected using the surface monitoring network as a function of time during operations (red circles) along with the cumulative volume of injected fluid during hydraulic fracturing (blue line) and the cumulative flow-back volume (magenta line). Events are clustered during periods of injection with relatively few events outside these periods, suggesting that activity decays rapidly with time after injection stops. It is clear that most of the seismicity is associated with certain stages or sleeves. For example, sleeves 22, 30, 31 and 32 on 25, 26, 27 and 29 October, all had relatively high levels of detected seismicity. Similarly, sleeves 38, 39 and 40 on 11, 13 and 14 December also have relatively high levels of detected seismicity. These sleeves are all at or closer to the heel (east) end of the horizontal part of the well and all the events with magnitudes greater than 0.5 ML occurred during these hydraulic fracture stages. Conversely, sleeves 1, 2 and 3 on 16, 17 and 18 October at the toe (west) end of the well all have relatively low levels of seismicity, despite similar injected volumes.

Moreover, there appears to be no clear relationship between the volumes of injected fluid during individual hydraulic fracturing stages and either the number or magnitude of events. No seismicity was detected during the stage 41 on 17 December, which had the largest inject volume of 431 m2, while considerable seismicity was observed during a number of mini-fracs when the injected volume was very small, for example sleeve 18 on 24 October.

Magnitudes for events detected by the surface monitoring network were determined from the largest zero-to-peak displacement in nanometres on horizontal component waveforms with a signal-to-noise ratio of greater than 2 that were high-pass filtered at 1.25 Hz. Magnitudes were then calculated using the UK local magnitude (ML) scale of Luckett et al. (2019), which incorporates a correction for near source observations. Between 16 and 28 station magnitudes were measured for each event, with two magnitude measurements for each station, so a large subset of the recording stations is used for all events. The event magnitude is taken as the mean of the magnitudes measured at each station. Individual stations magnitudes typically show a large scatter as a result of source radiation effects and local site effects, however, non-parametric confidence intervals can be estimated using bootstrap resampling.

Figure 16 shows histograms of the mean local magnitudes for each event with a magnitude greater than 0 ML. calculated using 10,000 bootstrap resampling replicates (Baptie and Luckett, 2019). The resulting distributions are approximately normal for most events. The non-

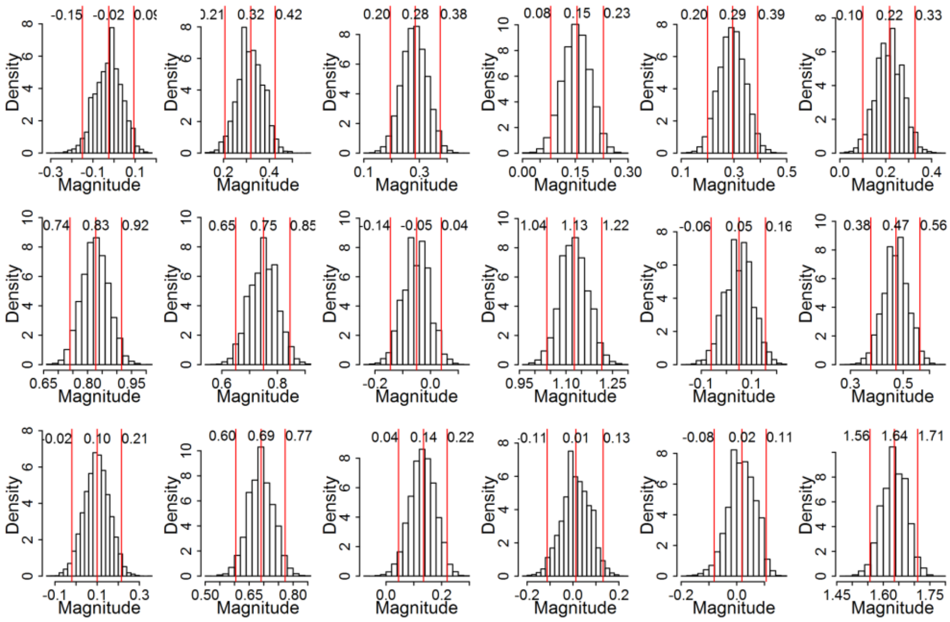


Figure 16. Histograms showing the results of bootstrap resampling of the measured local magnitudes at each station for events with magnitudes greater than 0 ML. Red vertical lines show 95% confidence intervals and the median for each event

parametric 95% confidence limits (red vertical lines in Figure 16) in the mean magnitude for each event are typically ±0.1 ML. However, standard deviations in the observed station magnitudes are significantly greater, varying from approximately 0.2 to 0.25 ML, while the overall spread in the magnitude measurements is typically one magnitude unit. Also the distributions of station magnitudes for each event are often highly skewed suggesting that magnitudes may be strongly influenced by individual station measurements.

Mc for the catalogue of events detected using the surface seismic stations was determined by Baptie and Luckett (2019) using the Goodness-of-Fit test (GFT) (Wiemer and Wyss, 2000), which calculates Mc by comparing the observed frequency magnitude distribution (FMD) with synthetic ones. This gives a magnitude of completeness of -0.6 ± 0.2 ML, where the errors were calculated using bootstrapping. The b-value for the catalogue was estimated using the maximum likelihood method of Aki (1965), which gives a value of 1.029 ± 0.118.

To assess how seismicity rates increase during operations, Baptie and Luckett (2019) compared the frequency magnitude distribution calculated for the Preston New Road events with a frequency magnitude distribution calculated for instrumentally recorded tectonic earthquake activity across the British Isles from 1970 to present (Figure 17). The errors bars show 95% confidence intervals from a $\chi2$ distribution. The numbers of tectonic events are scaled for the time period of operations, 57 days, and for the approximate area of operations (10 km by 10 km). A b-value of close to 1 was calculated for the tectonic event catalogue using a magnitude of completeness of 3.5 ML. This suggests that activity rates increase during the period of operations by a factor of around 100 against the average background activity rate for the UK. The activity will decay to background after the operation stop.

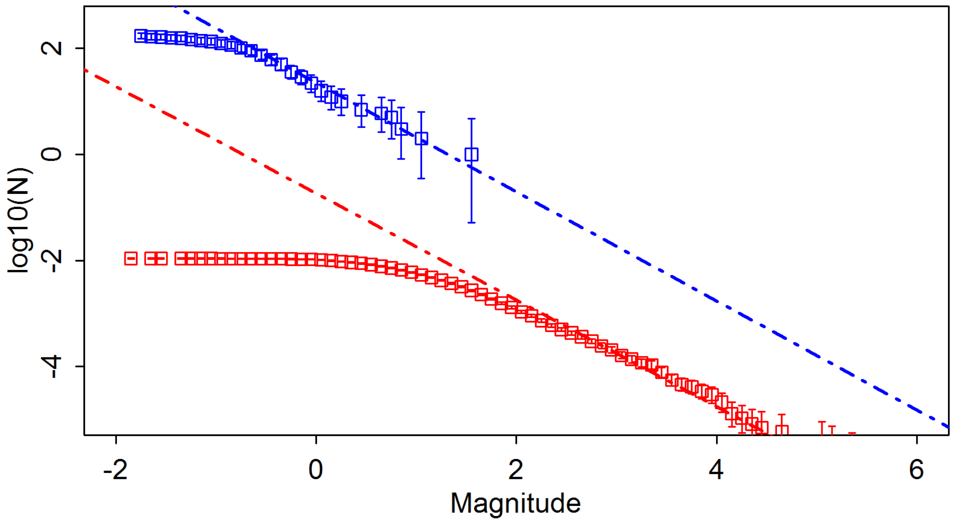


Figure 17. Observed frequency magnitude distributions for the Preston New Road events (blue squares) and instrumentally recorded tectonic earthquakes across the British Isles from 1970 to present (red squares). The tectonic activity data are scaled to a time period of 57 days and for an area of 10 km by 10 km. The blue and red dashed lines show maximum likelihood estimates of the b-value and activity rate for each

3.6.2 The Hot Dry Rock Project, Cornwall, UK

The Hot Dry Rock (HDR) project was a geothermal research project designed to test the feasibility of extracting geothermal energy from the Carmenellis granite in Cornwall by circulating water between deep boreholes (Parker, 1989). The experiments were conducted at Rosemanowes quarry between 1982 and 1991. A historical earthquake study of Cornwall and Devon by Musson (1989) identified some 41 felt seismic disturbances within a 25 km radius of the HDR site in the period 1750 to 1988. The British Geological Survey carried out background seismic monitoring around the site starting in 1981, one year before the start of injection (Walker, 1987), and this continued until the end of the project in 1991.The monitoring network, Figure 18, consisted of three seismometers up to 30 km from the site, a further six seismometers within a 9 km range, and a seismometer at the site itself. The network was considered capable of detecting any earthquake with a magnitude of 0.0 ML or above within approximately 20 km of the HDR site.

Several hundreds of natural background earthquakes were detected across the wider region using this network before, during and after the project, many of which could have otherwise been attributed to the project itself (Figure 18). On 25 February 1981, only a few days after the

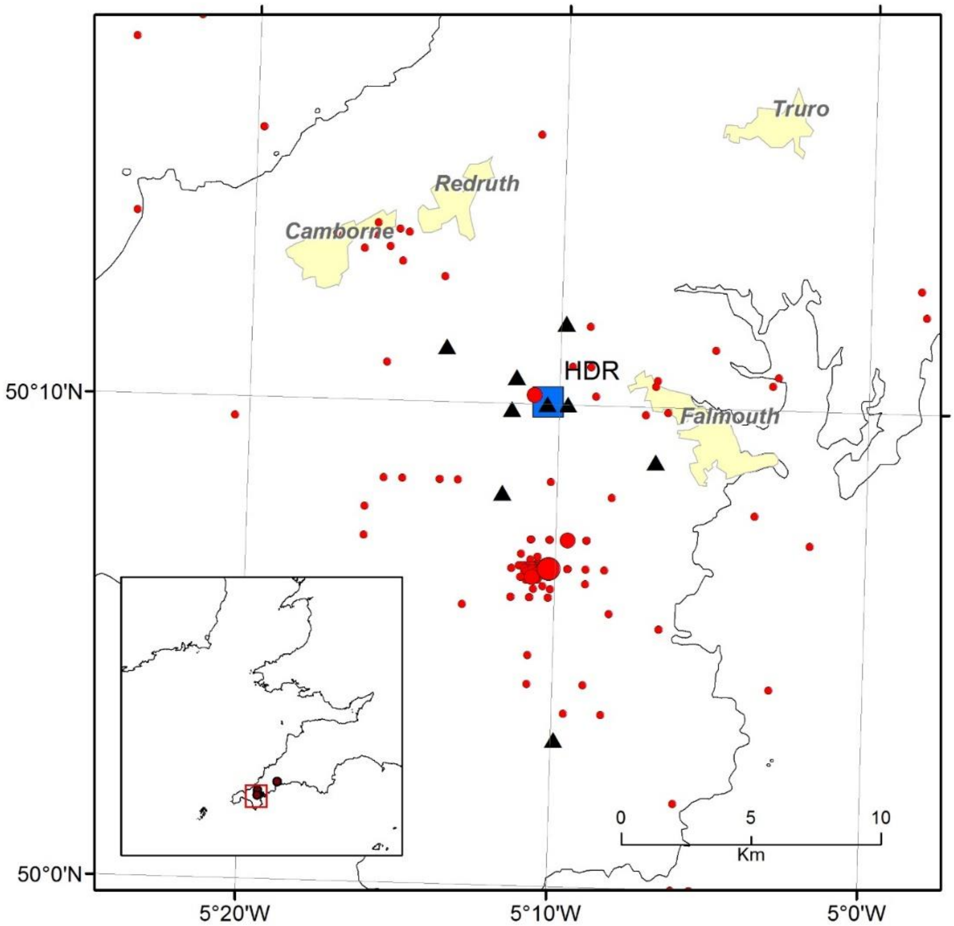


Figure 18. The seismic monitoring network around the HDR site at Rosemanowes (blue square). Stations are shown by black triangles. Seismic activity detected during the operation of the network is shown by red circles

installation of the network, a series of almost 200 earthquakes was detected immediately to the south of the HDR site, around the village of Constantine. These were not caused by operations at the site since they occurred before operations started. The largest had a magnitude of 3.5 ML and was widely felt by people in the area. This activity continued over the following years, with a further magnitude 2.9 ML earthquake on 2 September 1986.

Fluid injection at the site started in 1982, and went on to generate over 11,000 induced events between 1982 and 1987 (Baria and Green, 1990). Most of these were very small, with magnitudes typically ranging from 0.5 to -2 ML, and could only be detected by a separate network of borehole geophones installed at the site. However, a number of these were also detected by the background monitoring network. The largest induced event occurred on 12 July, 1987, had a magnitude of 2.0 ML, and was felt locally.

More generally, seismic monitoring is used widely in the geothermal industry before, during and after operations. For example, during operations, it can be used to image the stimulated volume and effectively manage geothermal reservoirs (Majer et al, 2007). Moreover, the risk of felt earthquakes means that seismic data is also essential for forecasting induced seismicity and mitigating the risk of potentially damaging events. Experience suggests that a reliable measurement of seismicity at low magnitudes (0 to 1 ML) is needed for many geothermal projects to enable active seismic zones to be properly identified. Also, since most geothermal induced seismicity is below magnitudes of about 2.0, it is important to know the baseline level of seismicity at the lower magnitudes.

Majer et al (2012) make the following four recommendations for baseline monitoring in the geothermal industry.

1. Monitoring needs to fully characterise background seismic activity and identify any faults with the potential to be affected by operations, and should not be biased in time or space in the vicinity of the potential geothermal project The duration of the background monitoring may be relatively short (one month) if there is already existing monitoring that can detect small earthquakes with magnitude around 1. If there is no existing monitoring, the duration may need to be extended for as long as six months.

2. High resolution instrumentation will allow induced activity to be modelled and forecast more accurately. As the induced earthquakes may span several orders of magnitude, say from -2 to 4, the monitoring system requires a high dynamic range to ensure that data of sufficient quality is recorded. Also, borehole installations are better than surface sensors as the signal-to-noise ratio is better, and this allows smaller events to be recorded, increasing resolution and location capability. The monitoring network should be able to provide comprehensive background monitoring over an area at least twice as large as the area of geothermal potential.

3. Data processing must provide locations, magnitudes and source mechanisms. A typical geothermal project, consisting of one or two injection wells and several production wells in an area with a diameter of 5 km, will require at least eight monitoring stations distributed over the area of interest.

4. Monitoring should be maintained throughout the injection activity to validate the engineering design of the injection in terms of fluid movement directions, and to guide the operators on optimal injection volumes and rates. This will also allow induced events to be discriminated from natural seismicity and ensure that local vibration guidelines are being followed.

These recommendations are equally valid to the monitoring of seismic activity around areas of shale gas exploration and production.

3.6.3 Mining Induced Earthquakes at New Ollerton

The coalfields of Britain are frequently the source areas of small to moderate earthquakes and tremors in these areas have been reported for at least the last hundred years, for example the Stafford earthquake of 1916 (Davison, 1919). With the growth of instrumental seismic monitoring in the UK in the 1970s many more earthquakes were recorded in mining areas across the UK (Redmayne et al, 1988) and a number of temporary networks of sensors have been deployed to study these events in more detail. This led to the conclusion that these events were related to ongoing mining activity and that these were quite distinct from the natural background seismic activity of the UK.

Between December 2013 and October 2014, over 300 small earthquakes were detected in and around New Ollerton (Verdon et al., 2018). Many of these were felt locally. This is an area with a history of seismic activity related to coal mining and the occurrence of these events coincided with the resumption of deep mining operations at the nearby Thoresby Colliery. This is the most recent example of seismicity associated with deep coal-mining in the UK. A temporary network of seven seismometers was deployed providing some high quality data to allow detailed analysis of these events.

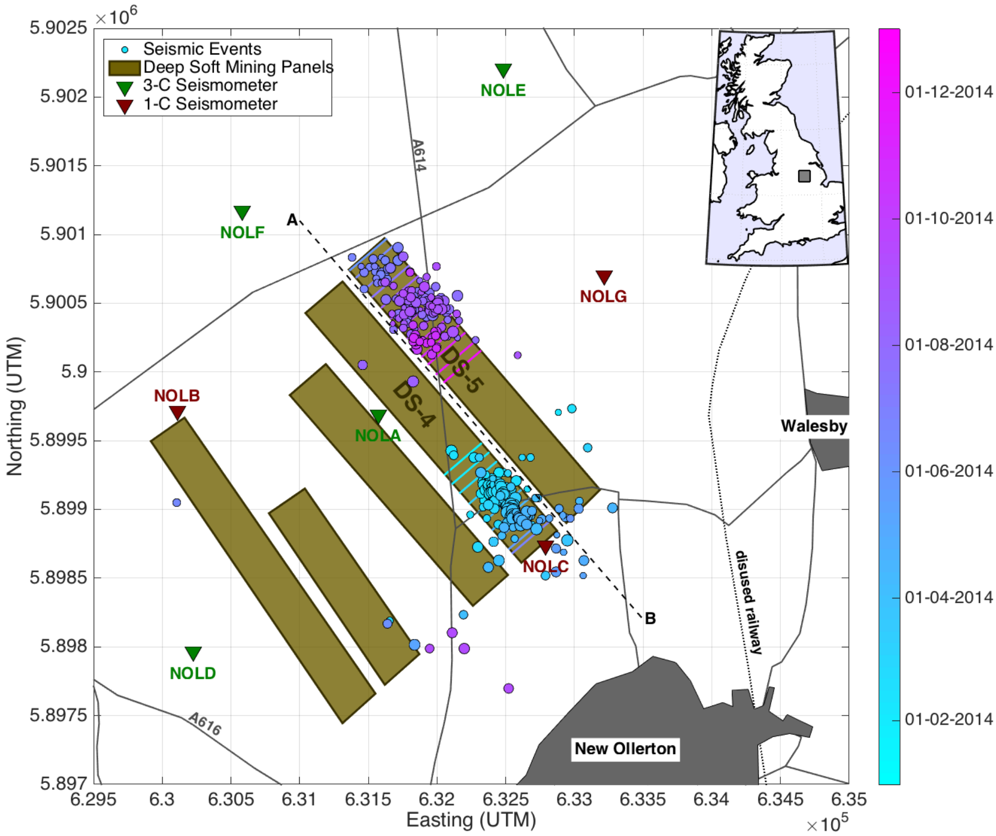


Figure 19. Map of event hypocentres, with events coloured by occurrence date (from Verdon et al 2018). Also shown are the positions of the monitoring network (triangles) and the mining panels (brown rectangles). Panels DS-4 and DS-5 were active during the monitoring period, and the coloured bars running across these panels show the forward movement of the mining faces with time. The position of the cross-section A – B (Figure 10) is marked by the dashed line

Verdon et al (2018) carried out a detailed analysis of the seismicity associated with the mining of the Deep Soft Seam in 2013-2014, calculating precise event locations, comparing locations to the propagation of the mining faces with time, and comparing seismicity rates with the volume of coal extracted from the mine. The calculated event locations are shown in Figure 20, with the event coloured by date. Also shown are the positions of the mining panels and the progress of the mining face with time from the UK Coal Authority mine abandonment plans. Event locations clearly correlate with the position of the face as it moves southeast along panel DS-4, before switching to DS-5 and again following the mining front to the southeast. The monitoring period ceases when the events have propagated approximately half-way along the length of panel DS-5.

3.7 REFERENCES

AKI, K., (1965). Maximum likelihood estimate of b in the formula log N=a-bM and its confidence limits, *Bulletin of the Earthquake Research Institute, University of Tokyo*, 43, 237–239.

ALLEN, R. 1982. Automatic phase pickers: their present use and future prospects. *Bulletin of the Seismological Society of America*. 72, S225–S242.

BAER, M., AND KRADOLFER, U., 1987. An automatic phase picker for local and teleseismic events. *Bulletin of the Seismological Society of America*, 77, 1437-1445.

BAPTIE, B.J. and LUCKETT, R. 2019. Seismicity induced by hydraulic fracturing operations at Preston New Road, Lancashire, 2018. *Proceedings of the Society of Earthquakes and Civil Engineering and Dynamics Conference 2019*, 9-10 September 2019, Greenwich, London.

BARIA, R. AND GREEN, A.S.P. 1990. Microseismics: a key to understanding reservoir growth, IN *Proceedings, Camborne School of Mines International Hot Dry Rock, Conference*, 27-30 June 1989, Robertson Scientific Publications, London, 363-377.

BEIS, 2013. Regulatory Roadmap: Onshore oil and gas exploration in the UK regulation and best practice, https://www.gov.uk/government/publications/regulatory-roadmap-onshore-oil-and-gas-exploration-in-the-uk-regulation-and-best-practice.

BOMMER, J.J. et al., 2006. Control of hazard due to seismicity induced by a hot fractured rock geothermal project. *Engineering Geology*, 83, 287– 306.

BONDAR, I, MYERS S C, ENGDAHL E R AND BERGMAN E A, 2004. Epicentre accuracy based on seismic network criteria. *Geophys. J. Int.*, 156, 483-496.

BOORE D.M., 2003. Simulation of ground motion using the stochastic method. *Pure and Applied Geophysics*, 160, 635–675.

BULLEN, K E AND BOLT B A. 1985. *An Introduction to the Theory of Seismology*. Cambridge University Press, ISBN 0 521 29686 2.

CLARKE H., EISNER L., STYLES P., TURNER P., 2014. Felt seismicity associated with shale gas hydraulic fracturing: The first documented example in Europe, *Geophysical Research Letters*, 41, 8308–8314.

CLARKE H., VERDON J.P., KETTLETY T., BAIRD A.F., KENDALL J.-M., 2019. Real-Time Imaging, Forecasting, and Management of Human-Induced Seismicity at Preston New Road, Lancashire, England, *Seismological Research Letters*, 90 (5): 1902–1915.

BUTCHER, A., LUCKETT, R., VERDON, J., KENDALL, M., BAPTIE, B. AND WOOKEY, J. 2017. Local Magnitude Discrepancies for Near-Event Receivers: Implications for the U.K. Traffic-Light Scheme. *Bulletin of the Seismological Society of America*, 107, doi: 10.1785/0120160225.

DAVIES, R., FOULGER, G., BINDLEY, A. AND STYLES, P. (2013), Induced seismicity and hydraulic fracturing for the recovery of hydrocarbons. *Marine and Petroleum Geology*, 45, 171-185.

DAVISON, C. 1919. The Stafford earthquakes of January 14–15, 1916 and the relations between twin earthquakes of the Midland counties. *Geological Magazine*, Decade VI, 6:, 302–312.

DE PATER, H. AND BAISCH, S., 2011. Geomechanical Study of Bowland Shale Seismicity. Synthesis Report.

EDWARDS B., ALLMANN B., DONAT F. AND CLINTON J. 2010. Automatic computation of moment magnitudes for small earthquakes and the scaling of local to moment magnitude. *Geophysical Journal International*, 183, 407–420.

EDWARDS B., KRAFT T., CAUZZI C., KÄSTLI P. AND S. WIEMER, 2015. Seismic monitoring and analysis of deep geothermal projects in St Gallen and Basel, Switzerland. *Geophysical Journal International*, 201, 1022-1039.

EVERNDEN, J.F., 1969. Precision of epicenters obtained by small numbers of world-wide stations. *Bulletin of the Seismological Society of America*, 59, 1365–1398.

FLINN, E.A., 1965. Confidence regions and error determinations for seismic event location. *Reviews of Geophysics*, 3, 157–185.

GEIGER, L., 1912. Probability method for the determination of earthquake epicenters from the arrival time only (translated from German). *St. Louis University Bulletin*, 8, 56–71.

GUTENBERG, B. AND RICHTER, C.F., 1954. *Seismicity of the Earth and Associated Phenomena*. 2nd edition, Princeton University Press, Princeton, N.J.

GOMBERG J S, SHEDLOCK K M AND ROECKER S W, 1990. The effect of S-Wave arrival times on the accuracy of hypocenter estimation. *Bulletin of the Seismological Society of America*, 80, 1605-1628.

HAVSKOV J. AND OTTEMOLLER L., 2010. *Routine data processing in earthquake seismology*. Springer, London, New York.

LEE, W H K AND STEWART, S W. 1981. *Principles and Applications of Microearthquake Networks*. Academic Press, New York.

LOMAX, A, MICHELINI A. AND CURTIS A. 2009. Earthquake Location, Direct, Global-Search Methods, in *Encyclopedia of Complexity and System Science*, Part 5, Springer, New York, pp. 244

LUCKETT, R, OTTEMÖLLER, L, BUTCHER, A AND BAPTIE, B, 2019. Extending local magnitude ML to short distances, Geophysical Journal International, 216, 2, 1145–1156.

MAJER E.L., BARIA R, STARK M, OATES S., BOMMER J., SMITH B. AND ASANUMA H., 2007. Induced seismicity associated with Enhanced Geothermal Systems. *Geothermics*, 36, 3, 185-2229-2473.

MAJER, E.L., NELSON, J., ROBERTSON-TAIT, A., SAVY J. AND WONG, I. 2012. *Protocol for Induced Seismicity Associated with Enhanced Geothermal Systems*. DOE/EE-0662.

MARROW, P.C., 1992. *Seismic Monitoring of the North Sea*. Health and Safety Executive Offshore Technology Report, OTH-90-323. MIGNAN, A. AND WOESSNER, J., 2012. Estimating the magnitude of completeness for earthquake catalogs. *Community Online Resource for Statistical Seismicity Analysis*, doi:10.5078/corssa-00180805.

Möllhoff M., Bean C.J. and Baptie B., 2019. SN-CAST: Seismic Network Capability Assessment Software Tool for Regional Networks – Examples from Ireland, Journal of Seismology, 23, 493-504doi: 10.1007/s10950-019-09819-0.

PARKER, R.H., 1989. *Hot Dry Rock Geothermal Energy*. Final Report of the Camborne School of Mines Project, Pergamon Press plc, Headington Hill Hall, Oxford, ISBN 0-08-037929-X.

PAVLIS, G., 1986. Appraising earthquake hypocenter locations errors: a complete practical approach for single event locations. *Bulletin of the Seismological Society of America*, 76, 1699–1717.

PETERSON, J. 1993. Observation and modeling of seismic background noise. *U.S. Geological Survey Technical Report*, 93-322

REDMAYNE, D.W., 1988. Mining-induced seismicity in UK coalfields identified on the BGS National Seismograph Network. In *Engineering Geology of Underground Movements*, Geological Society Engineering Geology Special Publication, 5, pp. 405–413

RICHTER C.F., 1935. An instrumental earthquake magnitude scale. Bulletin of the Seismological Society of America, 25, 1–32.

RUTLEDGE, J.T., PHILLIPS, W.S. AND MAYERHOFER, M.J., 2004. Faulting Induced by Forced Fluid Injection and Fluid Flow Forced by Faulting: An Interpretation of Hydraulic-Fracture Microseismicity, Carthage Cotton Valley Gas Field, Texas. *Bulletin of the Seismological Society of America*, 94, 5, 1817–1830.

RYDELEK P.A., SACKS I.S., 1989. Testing the completeness of earthquake catalogues and the hypothesis of self-similarity, *Nature*, 337, 251–253.

SCHAFF, D. P., AND RICHARDS, P.G. 2004. Repeating seismic events in China. *Science*, 303, 1176–1178.

SCHULTZ R., WANG R., GU Y. J., HAUG K., AND ATKINSON, G., 2016. A seismological overview of the induced earthquakes in the Duvernay play near Fox Creek, Alberta. *Journal of Geophysical Research: Solid Earth*, 122, 492–505. https://doi.org/10.1002/2016JB013570.

SHEARER, P.M. AND ORCUTT, J.A., 1987. Surface and Near-Surface Effects on Seismic-Waves - Theory and Borehole Seismometer Results. *Bulletin of the Seismological Society of America*, 77:1168-1196.

STORK, A.L., VERDON, J.P. AND KENDALL, J.-M., 2014. The robustness of seismic moment and magnitudes estimated using spectral analysis. *Geophysical Prospecting*, 62, 1365–2478.

TRNKOCZY, A., BORMANN, P., HANKA, W., HOLCOMB, L. G., NIGBOR, R. L., SHINOHARA, M., SHIOBARA, H. AND SUYEHIRO, K. 2012. Site Selection, Preparation and Installation of Seismic Stations. In *New Manual of Seismological Observatory Practice 2* (ed. P. Bormann), Potsdam, Deutsches GeoForschungsZentrum GFZ, pp. 1—139.

TURBITT T., BARKER E.J., BROWITT C.W.A., HOWELLS M., MARROW P.C., MUSSON R.M.W., NEWMARK R.H., REDMAYNE D.W., WALKER A.B., JACOB A.W.B., RYAN E., WARD V., 1985. The North-Wales Earthquake of 19 July 1984. J. Geol. Soc. (London) 142, 567–571.

VERDON, J.P., KENDALL, J.-M., BUTCHER, A., LUCKETT, R. AND BAPTIE, B., 2017. Seismicity induced by longwall coal mining at the Thoresby Colliery, Nottinghamshire, U.K., Geophysical Journal International, 212 (2).

WALKER, A.B., 1987. Background and induced seismicity monitoring for the HDR Geothermal Programme in Cornwall, *BGS Global Seismology Report,* 323.

WIEMER, S. AND WYSS, M. 2000. Minimum magnitude of complete reporting in earthquake catalogs: examples from Alaska, the western United States, and Japan. *Bulletin of the Seismological Society of America,* 90: 859-869.

YOON, C.E., O'REILLY, O., BERGEN K.J. AND BEROZA, G.C. 2015. Earthquake detection through computationally efficient similarity search. *Science Advances,* 1:e1501057.

YOON C., HUANG Y., ELLSWORTH W.L. and BEROZA G.C., 2017. Seismicity during the initial stages of the Guy-Greenbrier, Arkansas, earthquake sequence. *Journal of Geophysical Research: Solid Earth,* 122, 9253–9274. https://doi.org/10.1002/2017JB014946.

4. Groundwater and surface water

4.1 INTRODUCTION

There are requirements to protect groundwater and surface water from pollution and prevent unsustainable abstraction. European legislation, in particular the Water Framework Directive (2000/60/EC) and the Groundwater Directive (2006/118/EC), provide for the integrated management and environmental protection of groundwater, surface water and associated ecosystems. River flows may often be supported by significant components of groundwater baseflow. Indeed, discharging groundwater may become the dominant component during drier seasons in many catchments underlain by extensive groundwater bodies, i.e. aquifers (Bloomfield et al., 2009). Rivers, lakes or wetlands may form the natural discharge point of groundwater and become the unwitting receptor of discrete plumes of contamination migrating within that groundwater (Freitas et al., 2015).

Should any contamination arise from a shale-gas operation, it is not only the underlying groundwater that may be at risk, but also surface water receptors potentially local or at some distance from a facility. Hence, baseline assessment of sensitive water receptors includes not only groundwater resource - aquifer units deemed at risk, but also associated surface waters to which impacted groundwater could discharge as baseflow as well as (if applicable) any further surface waters that may receive regulated, direct pipe, discharges consented from a shale-gas facility or inadvertent surface run-off leakage. Both groundwater and surface water, quantity and quality, may support sensitive ecosystems.

The Water Framework Directive (2000/60/EC) establishes a series of environmental objectives for groundwater and surface water that must be met. This includes preventing pollution by prohibiting the entry of hazardous substances to water bodies and limiting inputs of non-hazardous pollutants. It also requires that there is no deterioration in quality that would lead to a failure in achieving good chemical, quantitative and surface water ecological status (as defined by a series of tests). Regulations are in place to meet these objectives and one of the principal activities to confirm that these are being effective, and European environmental objectives being met, is through monitoring. The environment agencies carry out statutory monitoring to confirm the status of groundwater and identify any general deterioration in quality and they also require, as appropriate based on risk, those granted an environmental permit to carry out compliance monitoring to demonstrate that their activities are operating within permit conditions and is not causing pollution. This includes shale gas operations.

The objective of baseline groundwater and surface-water monitoring is to provide an evidence base that along with other data and information allows characterisation of the underlying and surrounding groundwater and surface-water systems. This is essential to allow for future comparison against any water (and associated) impacts that might arise, and to support identification of suitable compliance and assessment criteria where appropriate.

The baseline monitoring aims to characterise the surface water or groundwater that could potentially be affected by shale gas development with sufficient confidence to distinguish impacts from contamination via surface spillages, borehole infrastructure leakages or enhanced upflow of deep saline fluids and other contaminants. The monitoring should be designed to target analytes that are indicative of these types of contaminant as well as those best able to demonstrate the baseline condition. The monitoring design, whilst targeting assessment of water quality and potential contamination occurrence, needs to be substantially underpinned by a sound understanding and conceptualisation of the integrated groundwater – surface water flow regime that critically transports chemical solutes or other contaminants present. The water flow system, characterised by surface water and groundwater level and flow data, should hence be monitored alongside water quality to form a truly integrative and holistic baseline monitoring of the water system dynamics present. In combination, this will provide an assessment of spatial variability and trends.

Overall, in operating the monitoring programme the results obtained should be sufficiently representative of groundwater (in the relevant strata) or surface water and not detrimentally influenced by the monitoring point design, sampling methodology, extraneous contamination or analytical limitations.

Additional guidance on development and operation of groundwater and surface-water monitoring programmes is provided in the following British/International standards and monitoring should aim to comply with the best practice identified within these:

- BS EN ISO 5667-3:2012 -Water quality. Sampling. Preservation and handling of water samples;
- BS EN ISO 5667-6:2016 - Water quality. Sampling. Guidance on sampling of rivers and streams;
- BS ISO 5667-11:2009, BS 6068-6.11:2009 - Water quality. Sampling. Guidance on sampling of groundwaters;
- BS EN ISO 5667-14:2016 - Water quality. Sampling. Guidance on quality assurance and quality control of environmental water sampling and handling;
- BS ISO 5667-20:2008 -Water quality. Sampling. Guidance on the use of sampling data for decision making. Compliance with thresholds and classification systems;
- BS ISO 5667-22:2010 - Water quality. Sampling. Guidance on the design and installation of groundwater monitoring points.

4.2 MONITORING SITE SELECTION

Monitoring site selection should be informed by a conceptual geological/hydrogeological model (CM) of the study area which should aim to identify all potential contaminant sources, pathways and receptors as well as the prevailing hydrogeological and surface-water conditions. It is important to identify sources of contaminant that could give rise to background signatures of the types of contaminants frequently associated with shale gas operations. For instance, methane arising from natural sources or other anthropogenic (non-shale-gas development) activities should be included in the conceptualisation. Collation of data to contribute to the CM should incorporate information on lithology, texture and permeability of represented strata, locations and orientations of known and suspected faults and data on well constructions (water well/pump depths, screen intervals) as well as construction of the proposed hydrocarbon infrastructure and its key source terms or principal activities posing a risk. Water usage (actual and potential) and water quality are also a key criterion for determining vulnerability of local/regional water bodies as is surface drainage from a shale gas site.

Monitoring design should incorporate sites likely to be representative of potential impact from operations as well as a comparable number of likely non-impacted sites. These, together with decisions on numbers of sites required for the proposed network, should be informed by the CM. Monitoring sites should therefore include all water courses and aquifers deemed potentially at risk and of environmental significance, e.g. sufficiently shallow to be accessible and of a quality to be usable (e.g. a cut-off of around 400 m depth has been advocated for this purpose by UK-TAG, 2011). For groundwater, consideration should also be given to monitoring in deeper geological formations with significant permeability which may act as pathways for contaminant migration.

Monitoring design should also take into account the longer-term objectives. Specific design objectives relating to groundwater monitoring points that need to be addressed include ability to both measure an accurate water level or pressure (‘piezometric’) level of groundwater and be recorded relative to a recognised datum to determine groundwater flow rates and directions and/or to enable an appropriate sample to be obtained from the surrounding stratum containing groundwater. This requires an appropriate design standard to be adopted for the environmental conditions encountered.

As a minimum:

- Using the CM, a sufficient number of groundwater monitoring points in the control area (up groundwater gradient ('upstream') of the proposed industrial operation) should be identified to characterise adequately the groundwater/surface water conditions taking into account the range of land uses, potential contaminant sources and the complexity of the hydrogeological conditions. For example, if there are multiple aquifers upgradient, each of these should have adequate monitoring to define the spatial and temporal variability in water quality, levels and flows, including exchange flows between units.
- Monitoring points should be identified and/or installed close to the proposed operational site considering both the area of surface-based hydrocarbon infrastructure and the area overlying the sub-surface infrastructure footprint, e.g. well laterals. The depth, location of monitoring points and spacing should be informed by the CM and the proposed design and operation of the hydrocarbon well infrastructure. The use of existing boreholes for sampling should only be considered if their locations are suitable, construction (e.g. well screen interval) and geological details are known, purging and sampling protocols are adequate to underpin reasonable understanding of sample provenance, and the data obtained are relevant to the objectives.
- Monitoring is required down groundwater gradient ('downstream' in terms of groundwater flow) of the site, particularly along suspected potential pathways between key site infrastructure locations (if known) and environmental receptors. The CM model should be used to identify potential source-pathway-receptor combinations. This is a priority monitoring area for an operational site recognising that any inadvertent release of site contamination may lead to groundwater contaminant plumes developing and expanding down groundwater gradient (pathway). It is therefore necessary to ensure during baseline monitoring this pathway area is well monitored with multiple monitoring points. Within the operational phase, monitoring point densities will need to be sufficient in this area to safeguard receptors, recognising the probable small scale of any groundwater contaminant plumes arising and also the uncertainty in plume direction of transport (flow).
- Deeper aquifers should be identified and included within the conceptual model. They may in themselves be receptors, e.g. have current or potential future use, or may represent significant pathways for contaminant migration, where the contaminants might originate from either the target hydrocarbon formation or from other formations where the integrity of the installed infrastructure is inadequate. Monitoring of these deeper units should be included in the both baseline monitoring and operational monitoring. Sufficient monitoring of these aquifers is required within, and downgradient of, the sub-surface footprint of the site's infrastructure. As well as water quality, it is important to measure hydraulic head (water level) and compare with near-surface aquifers to determine the groundwater flow regime and in particular the potential for upward flows and flow pathways to potential receptors.

For surface-water monitoring, monitoring points should be established on each significant water course within the area of the sub-surface footprint of the site at locations both upstream and a short distance downstream of the site. Within this monitored area, surface waters that may potentially be at future risk of contamination from baseflow from groundwater discharges should be identified and monitored. Monitoring points should also be considered for wetlands and any standing water bodies such as lakes identified at possible future risk. Establishing the actual and possible connectivity of both shallow and deeper groundwater systems to surface-water bodies, the ‘groundwater–surface-water interaction’ (GSI), is seen as a priority in the baseline period in that identifying pathways of concern will improve the conceptual model and may allow future rationalisation and optimisation of surface water monitoring programmes.

For groundwater, sites might include pre-existing boreholes and wells as well as springs, including deep-source springs. Consideration should be given to the sampling zone of the site being considered, e.g. the screened interval of boreholes and capture zone of springs. It is essential that the borehole completion details are known for any site being considered and that it meets the objectives of the monitoring programme, e.g. can yield a representative sample from a known geological formation. Often pre-existing boreholes will not be adequate. In cases of boreholes with long screens, consideration should be given to the thickness of aquifer being represented by the sample as this, and other construction details, will affect the sampling methodology (BS ISO 5667-11, 2009). Groundwater-quality data from potential receptor (public-supply) boreholes which are typically long-screen and would be presumably at distance, may still provide useful information and should not be overlooked.

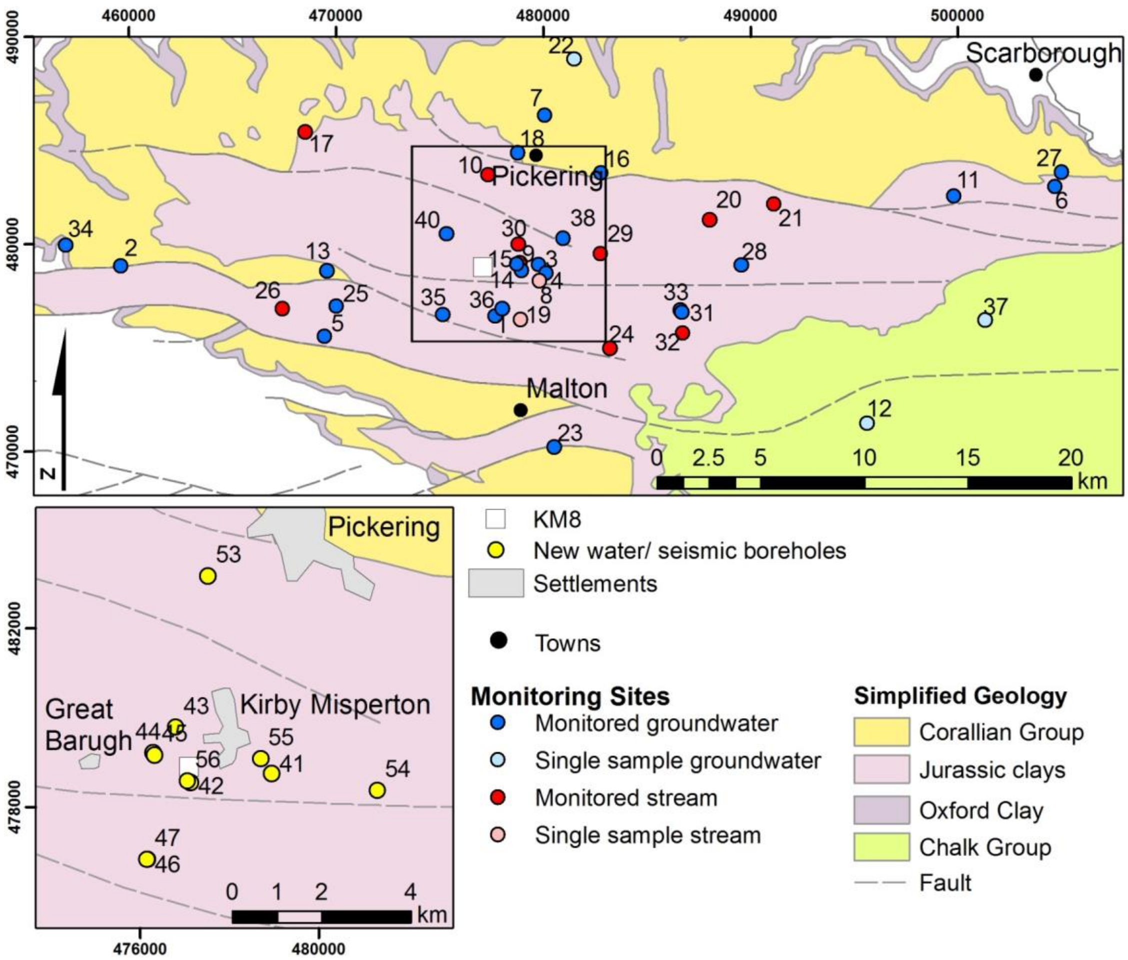
In most situations, new bespoke boreholes are expected to be needed to fulfil the monitoring requirements. These should be constructed according to objectives of the monitoring and informed by the local hydrogeological conditions. They should be screened (open section for sampling) as appropriate and taking into account BS ISO 5667-22 (2010). Borehole records should be documented to provide information used to update the CM. Borehole records should also be lodged with the British Geological Survey (Water Resources Act 1991 and Water (Scotland) Act 1946).

Where the hydrogeological system is stratified, e.g. multi-layered, and monitoring is required at different depths, consideration should be given to installation of multi-level samplers (MLS) (Chapman et al., 2014). These allow sampling of/measurement at discrete depths within the sub-surface at a single location. The advantages of manufactured multilevel samplers over multiple completions include: fewer drilled holes, reduced drilling and installation costs, reduced site disturbance, minimisation of purge water volumes, reduced waste handling and disposal costs. Although the MLS construction materials are more expensive than materials used for individual borehole completion, MLSs become cost-effective where sampling is required at four or more discrete horizons as a result of lower drilling costs (CL:AIRE, 2002). The data obtained are also likely to be of better quality and MLSs provide greater potential as an early-warning system for detecting contaminant release(s) at depth and informing on the conceptualisation of deep to shallow system contaminant migration potential.

In all cases, site selection will be influenced by logistical considerations such as site access and site safety and compromises will inevitably be needed for these factors. Justifications for site selection and operation should be recorded quality assurance purposes. An example is provided in the case study describing the water quality network site selection in the Vale of Pickering.

CASE STUDY Water-quality network site selection – Vale of Pickering, North Yorkshire

Selection of sites for monitoring in the Vale of Pickering around the Kirby Misperton proposed shale-gas site (KMA) has been determined by factors including location and distribution of recognised aquifers, locations of identified faults, groundwater flow patterns, use of aquifers for water supply, land use and practical considerations (site suitability, access and safety). This information has informed the development of the conceptual model for the area. The network of groundwater sites includes sites from a proximal shallow aquifer (Quaternary lacustrine and/or Jurassic clay) used for local private supply, and a more distal principal aquifer (Jurassic Corallian limestone), used for regional public supply as well as private supply. This limestone aquifer also occurs at depth (>200 m) below the KMA site but is not exploited due to depth and salinity. The Cretaceous Chalk aquifer is not included in the network as it is more remote from the KMA site. Surface waters in the monitoring network comprise low-order streams (tributaries of the River Derwent). For both surface water and groundwater, sites have been selected both within and outwith the area of potential impact, as defined by the hydrogeological conceptual model, and with representatives from both recognised aquifers. The network comprises 25 sites for groundwater (in roughly equal proportions of potential impact/non-impact) and 10 sites for surface water (5 of each). Groundwater sites in this network include suitable privately-owned, pre-existing water-supply boreholes. However, as locations of pre-existing groundwater sites were not always ideal in terms of monitoring objectives, 10 additional water monitoring boreholes were drilled at strategic locations (within 1 km of KMA, within the shallow aquifer upgradient ('upstream') north of KM8, and within the deep Jurassic limestone aquifer in the centre of the Vale). These have been incorporated in the monitoring network.



4.3 MEASUREMENTS

4.3.1 Selection of chemical analytes and measurement parameters

Identification of the analytes that require measurement and/or monitoring should be informed by the CM, existing knowledge of the prevailing groundwater quality and the substances and potential pollutants that will be used in the industrial operations and/or that may be mobilised by them. These may be defined in an environmental permit if one has already been issued but should also include others where they are required for environmental baseline characterisation and to support interpretation of data collected during/after operations.

Measurements will include observational and physical measurements (such as water levels), physicochemical indicators (e.g. field-determined parameters such as pH, redox potential (Eh), Electrical Conductance (EC)), major ions (providing general characterisation, informing CM development, analytical quality control and assurance), selected trace elements (indicators of change/pollutants), trace organic compounds including hydrocarbons and frack compounds (indicators of anthropogenic/industry impact) and naturally occurring radioactive materials (NORM; indicators of change/sub-surface pollutants). The parameters identified in Table 1 should be included as a minimum. This analyte suite incorporates general indicators of pre-development conditions, indicators of environmental change/impact and indicators of health exposure. In the latter case, not all health-impacting analytes (e.g. analytes contained within the relevant Water Supply (Water Quality) Regulations) are included and the selection of analytes with this objective should be risk-based and informed by the CM.

Where multi-element analytical techniques are used (e.g. ICP-MS, ion chromatography), extra data acquired as part of the analytical process should be stored (and reported) alongside the essential analytes.

It is essential for water levels to be measured in the network of groundwater monitoring boreholes. This allows lateral groundwater flow direction and changes over time to be determined. Measurement of levels (hydraulic heads) in boreholes completed at different depths, or in MLS, can also allow vertical hydraulic gradients to be measured. In both cases this is important for interpreting the water-quality data and for identifying potential flow paths for contaminant movement from deep to shallow systems (or vice versa). In combination with other hydrogeological parameters, e.g. hydraulic conductivity of the rocks, the water-level data can also be used to estimate groundwater velocities and hence the movement of contaminants, recognising most contaminants migrate at (substantially) lower velocities than groundwater. In a similar way, data on surface-water flows is important for understanding variation in surface-water quality and the spatial variation of groundwater baseflow contributing to those flows.

4.3.2 Sampling, measurement and analytical methods

Choice of appropriate sampling, measurement and analytical methodology should be defined by analyte stability, reliability of measurement and required limits of detection. Table 1 indicates the analytes that are most unstable and require measurement on-site. In the case of groundwater, these should be measured as close to the wellhead (the point that groundwater reaches the surface) as possible and preferably using in-line techniques to avoid exposure to air.

For surface water (rivers and streams), the sample should be collected directly from mid depth within the water column. Insertion of probes directly into the surface water body may be appropriate in order to obtain a representative observation if conditions are safe to do so (BS ISO 5667-6 (2014)). For lakes and wetlands the sampling point should be as close to the outlet as possible (BS ISO 5667-4 (2016)).

Table 1. Baseline water-monitoring analytes.

Analyte	Min. reporting value (limit of quantification)	Unit	Comment	Justification
Temperature		°C	On-site measurement	Heat from deep sources, purging QA
pH		pH units	On-site measurement	Rapid response to fluid mixing, purging QA
Electrical Conductance (EC)		µS/cm		Detecting salinity, purging QA
Dissolved oxygen		mg/L or % sat	On-site measurement	Defining redox conditions, purging QA
Redox potential (Eh)		mV	On-site measurement	Defining redox conditions, purging QA
Ammonium (as NH_4)	0.05	mg/L		Defining redox conditions
Nitrate	0.5 (as NO_3)	mg/L		Defining redox conditions, major non-shale-gas-related pollutant indicator
Dissolved Organic Carbon (DOC)	0.2	mg/L		Identifying hydrocarbons
Calcium	1	mg/L		Defining major characteristics of water
Magnesium	1	mg/L		Defining major characteristics of water
Sodium	1	mg/L		Defining major characteristics of water
Potassium	1	mg/L		Defining major characteristics of water
Total alkalinity	5 (as HCO_3)	mg/L		Defining major characteristics of water
Sulphate (as SO_4)	3	mg/L		Defining major characteristics of water
Chloride	1	mg/L		Defining major characteristics of water
Iron	20	µg/L		Defining redox conditions
Manganese	50	µg/L		Defining redox conditions
Arsenic	1	µg/L		Defining redox conditions; health-related
Boron	100	µg/L		Defining redox conditions; assessing saline fluids
Strontium	5	µg/L		Defining redox conditions; assessing saline fluids
Dissolved methane	5	µg/L		Hydrocarbon contamination and/or organic degradation
Total Petroleum Hydrocarbons (TPH)	10	µg/L		Hydrocarbon contamination
BTEX compounds	5 (each)	µg/L		Assessing hydrocarbon leakage
Water level (groundwater)	0.5	cm	On-site measurement	Groundwater flow and perturbations due to gas pressures
Any diagnostic compounds identified in the hydraulic fracture fluid				Pollutant indicator

For groundwater and the sampling of monitoring wells or boreholes, a diversity of approaches (protocols) is possible with a variety of water removal (purging) options used prior to collection of a 'representative' groundwater sample. These may be categorised under (McMillan et al., 2018):

- 'Zero/minimal-purge' protocols using grab or passive (diffusion-based samplers) that remove no or a very small amount of water volume prior to sampling – effectively obtaining a grab sample of the ambient flow regime at the point within the well screen sampled;
- 'Low-flow' (low-stress) protocols that purge (remove) and sample at low flow rates until indicator parameter stabilisation occurs, and may involve low to moderate volumes of water being extracted to achieve this condition;
- 'Multiple well-volume purging' (fixed-volume-purge) protocols based upon a specified number of well volumes being purged prior to sampling.

Indicator parameters include temperature, EC and Eh. Whilst the selection of one approach may be preferred for various reasons for a given monitoring point(s) (e.g., cost effectiveness, time efficiency, perceived technical appropriateness for the monitoring well – hydrogeological scenario), a key principle to follow is that once a protocol is determined for a monitoring point, it should be repeated on each round of sampling to enable data comparison over time (BS ISO 5667-11 (2009).

For determination of trace metals, dissolved quantities constitute more representative analyses of water chemistry than total quantities. Membrane filtration (to at least 0.45 µm, preferably less) is appropriate. This removes particulate matter of larger dimension and reduces the quantity of micro-organisms that can modify the water chemistry after collection. With filtration, consideration should be given to analytes that are relevant to health assessment as total and dissolved quantities might give differing indications of exposure.

Depending on the range of analytes to be measured, a range of sampling equipment may be required. For example different types of sampling equipment may be needed for dissolved gases than for stable inorganic parameters. The advantages and disadvantages of each need to be considered in the context of the monitoring objectives and the chosen approach justified and recorded. The same approach should be used for each sampling event (round) to ensure comparable results.

Of particular importance is the sampling for unstable and/or volatile parameters, e.g. volatile organic compounds (VOCs). Appropriate sample preservation methods should be used to minimise the loss or transformation of analytes between sampling and analysis. BS ISO 5667-11 (2009) provides guidance on sample preservation.

For dissolved gases and other volatile substances, there is a significant potential for loss of the parameter being sampled if the correct sampling method(s) and sample storage procedures are followed. When sampling from depth, there may be significant pressure difference between the sample point and the ground surface. This can lead to rapid degassing and hence loss of sample integrity. Therefore, sampling should be carried out using low flow sampling methods and without exposure of the groundwater to air. Sample containers should also be completely filled with no headspace.

Choice of sample containers, preservation, handling, storage and holding times also needs to be considered carefully to reduce the risk of sample deterioration between the time of sampling and analysis. The analytical laboratory(s) should be consulted to determine the optimum procedure. This should be recorded and followed for each sampling event.

For most analytes given in Table 1, laboratory analysis is the more reliable and cost-effective approach. This requires consideration of sample preservation before analysis. Inorganic constituents of water require preservation using mineral acid; care is needed to use preservative of sufficient purity to avoid contamination above acceptable limits. Samples should be maintained in cool and dark conditions (e.g. refrigerator) before analysis to minimise degradation. Other stability considerations include laboratory holding times and decay rates of radioactive analytes (e.g. radon, half-life 3.8 days).

Limit of quantification (LOQ) will be a key criterion for choice of laboratory analytical method. Appropriate limits of quantification for important analytes are given in Table 1. If a limit is specified that is too high, then the dataset may contain only values reported as being below this level. These data will then be of very limited value, for example in providing early warning of change and/or identifying factors contributing to change.

For all analytes measured, it is essential that the following be reported: analyte measured and how reported (e.g. alkalinity definition), unit of measurement, detection/quantification limit, analytical method, reporting laboratory, analytical uncertainty (accuracy, precision) and associated QA metadata (e.g. data for certified reference materials). Assurance of data quality is also supported by reporting of analytical charge imbalances (ionic balance).

All water sampling should be carried out by competent individuals with sufficient skills, training and experience to conduct the task. Analysis of samples should be conducted by competent laboratories operating suitable QA procedures (e.g. as demonstrated by ISO 17025 accreditation). Accreditation credentials and scope need to be stated alongside the metadata listed above. It is recognised that not all methods may be accredited and in these cases, the method should be validated and documented. Data recording should include chain-of-custody and should be sufficiently detailed to enable a full audit trail.

4.3.3 Monitoring frequency and duration

For previously undeveloped and/or unmonitored areas, baseline monitoring programmes should be initiated *at least* one year in advance of any operational development to characterise the baseline over differing seasons. However, if it has not been possible to characterise the baseline adequately within this period, e.g. to establish statistical trends, then the duration and/or frequency of measurement will need to be increased.

The monitoring frequency for groundwater and surface water should be informed and determined by the site-specific geo-environmental conditions, the sensitivity of the identified receptors, any reliable historical monitoring data that might exist and conditions specified in the environmental permit (although these should also be based on the aforementioned criteria).

Different monitoring frequencies may also apply for different analytes depending on the objectives to which they apply. For analytes that are used to characterise the baseline variability in water quality, the frequency should at least be sufficient to characterise the seasonal variation. This will require monthly or at least quarterly monitoring depending on the hydrogeological/environmental setting and its response to influencing environmental factors. Where shorter-term influence might be expected, such as tidal influences, a period of more frequent monitoring may be needed, at least for a period of time. Recommended minimum criteria are shown in Table 2. Monitoring frequency should be reviewed and modified to reflect the nature of the operational activities once these are initiated. This might result in a monitoring plan requiring different frequencies of monitoring at different sites depending on their proximity to the operations and also sensitivity of receptors. This will also be the case if unexpected changes in water quality are detected.

In the absence of information to support an alternative strategy, at least 12 sets of data should be obtained from each monitoring point spread out over a period of 12 months as part of baseline characterisation.

For more specialist analytes beyond the list outlined in Table 1, e.g. stable carbon isotopic analysis of methane, less frequent monitoring (or even single measurements) may be adequate as this information is considered to be supporting data to aid interpretation of the chemical monitoring data collected during the baseline period and subsequently during the operational phase. For example, if high concentrations of dissolved methane are measured (e.g. >1 mg/L), analysis of the stable carbon ($\delta^{13}C$) composition of the methane can provide information to support the identification of the origin of the methane. Repeat measurement of this analyte is usually not necessary unless anomalous events occur, e.g. increased concentrations or upward trends are observed.

Monitoring frequency should be reviewed and modified to reflect the nature of the operational activities once these are initiated. This might result in a monitoring plan requiring different frequencies of monitoring at different sites depending on their proximity to the operations and also sensitivity of receptors. This will also be the case if unexpected changes in water quality are detected.

Table 2. Recommended minimum baseline monitoring frequency for groundwater and surface water in relation to different influencing and risk factors (note: will not necessarily apply to all parameters)

Relevant factor	**Suggested *minimum* frequency of measurement**
Groundwater	
Seasonal/no seasonal influence Tidal influence	Monthly (for minimum of 12 months with subsequent reduction to quarterly if monitoring extends beyond 12 months) Hourly monitoring of key salinity indicator analytes (for minimum of 3 days) and then monthly along with other parameters (for minimum of 12 months, with subsequent reduction to quarterly if monitoring extends beyond 12 months).
Surface water	
Seasonal/no seasonal influence Tidal influence	Monthly Hourly monitoring of key salinity indicator analytes (for minimum of 3 days) and then monthly along with other parameters.

4.4 DATA ANALYSIS AND CHANGE DETECTION

The design of the groundwater and surface-water monitoring networks will have taken into account the proposed shale-gas operations at a site, the need for characterising the baseline, ongoing monitoring and the overall objectives of the monitoring. A key objective is to allow comparisons to be made between the measurements taken before the operations (baseline period) and those during (and after) operations for the purposes of change detection once sufficient data have been collected.

It is essential that the monitoring data are collected in a way that allows identification of changes in groundwater/surface water that arise from the shale-gas operations (permitted activity) as well as demonstration that no change is occurring. Where changes do occur they need to be identified as early as possible to allow the most appropriate management action to be taken, e.g. to avoid a pollution incident. This is both important for regulatory compliance and public reassurance. Earlier sections of this chapter have set out a recommended approach to designing and implementing of a monitoring programme and the collection of data for this purpose. If the recommendations are implemented effectively, the data collected should be sufficient for change detection.

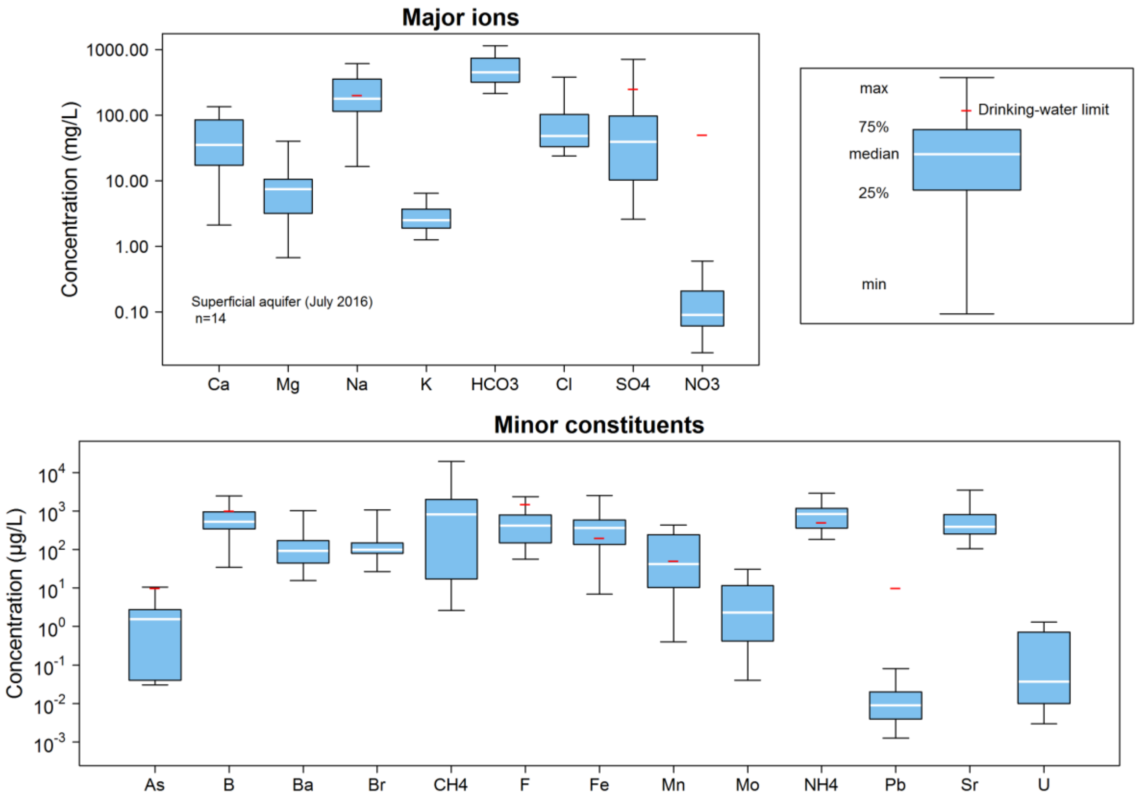
Reporting of the results of sampling and/or monitoring should be carried out with a frequency depending on the design of the programme. Quarterly reporting should be considered the default but any new data should be reviewed to identify any significant differences from previous data. It is not necessary to report the methods and protocols on each occasion but these should be reported early in the programme and then revised if any changes to procedures are made. Each data report should make reference to the methods report and confirm that the data presented were collected by the specified methods. Any differences should be recorded. Evaluation of the acquired monitoring data should be used to optimise and refine the monitoring strategy for future rounds both in the baseline and operational phases.

CASE STUDY

Water data visualisation – Vale of Pickering, North Yorkshire

Representation of water-quality data for the baseline assessment can take many forms, including maps, profiles, box plots and cumulative-probability plots. Some example box plots for the Vale of Pickering are depicted below.

The plots show the measured range of concentrations for several analytes measured in the aquifer at multiple monitoring sites (14) sampled in July 2016 along with a range of statistics. Plotting the data in this way enables a rapid assessment of the data, highlighting any unusual features and if water quality standards (or other triggers or thresholds) are also shown, as can be seen in the plots below, then any exceedances and their significance can be seen. Box plots are versatile and allow monitoring data to be presented in different ways, e.g. site by site, by sample round, by season, by year.



Reporting should include an evaluation/update of groundwater and/or surface water quality in terms of spatial and temporal variation. Suitable visualisation approaches include use of maps, profiles, box plots, cumulative-probability plots (see case study – water data visualisation).

Processes controlling chemistry and chemical variation should be assessed and likely sources of any contaminants evaluated. Recommendations for any further investigation to resolve uncertainties should be described. Also included should be an assessment of uncertainty in data results or interpretation and a critique of data adequacy for achieving the stated objectives.

A challenge for groundwater and surface-water quality data is that frequency of measurement is generally low relative to other environmental monitoring such as air quality or seismicity. This is because of the technical and logistical difficulties of monitoring water quality. Whilst some analytes (but only a limited range) can be measured in situ, the sensors used for this are subject to drift and variable performance because of the environments in which they are operating. This reduces the reliability of the in-situ monitoring data, and so the results of samples collected and analysed in an accredited laboratory provide the best quality data for reliable change detection. However, the in-situ monitoring can provide a useful indicator dataset that, because of its higher frequency measurement, could highlight a change that requires further investigation.

The recommended approach for water quality change detection is to use the laboratory analysed data as the primary data set because of its reliability in terms of QA/QC. The time when change detection is required is after operations have started; in order to detect change using statistical methods, there needs to be a sufficient amount of data to carry out an analysis. This presents a challenge when the frequency of data collection is low and, as is often the case, when natural variability in observed values is high.

The nature of the activity and the multiple potential pollutant sources and migration pathway combinations means that a multiple-lines-of-evidence approach to change detection is recommended; one that considers point, spatial and temporal change detection. More specifically these are:

- Comparison of analyte values measured at a monitoring point during the operational period with data from the same site during the baseline period;
- Comparison of parameter values measured at a monitoring point during the operational period with the space-time mean of a group of sites within a pre-defined area of potential impact (API) (see below);
- Identification of temporal trends in indicator analytes that are different from the baseline period.

4.4.1 Baseline data analysis

The starting point for assessment is the processing of the data for the baseline period in order to provide the benchmark(s) against which future measurements or measurement statistics can be compared to identify change.

Analysis should be performed on results from individual monitoring sites and aggregated data for a group of sites. The selection of a group of sites should be informed on the conceptual model (hydrogeological/catchment setting) and the nature and 'footprint' of the shale-gas infrastructure and operations. One group should include monitoring sites that are in close proximity to the operations and immediately down the hydraulic gradient. This group is known as API (Area of Potential Impact) group. A second group of sites should be identified that are remote from the site and/or upgradient. This is the control area group. A key requirement is that the control group of sites should be monitoring the same hydrogeological system (aquifer) as the API group of sites, so that the two groups of sites will be subject to the same, as far as possible, non-shale gas regional (natural and anthropogenic) influences on water quality.

For each monitoring point/group of sites and for the selected parameters (environmental permit defined), a range of statistics should be calculated. This includes (as a minimum):

a) Number of measurements (n)
b) Units of measurement

c) Minimum concentration/parameter
d) Maximum concentration/parameter
e) Mean concentration/parameter
f) Median (P50) concentration/parameter

If the 25th- and 75th-percentiles are also calculated, box and whisker plots can be produced (see water data visualisation case study).

It is recommended that the maximum concentration for the baseline period (comprising at least 12 measurements) is taken as the value indicative of the upper baseline for substances that are naturally occurring.[3]. For synthetic substances, this approach is not generally appropriate, as they should not be present in groundwater or surface water. Therefore, for synthetic substances, the analytical level of quantification (LOQ) should be used in the case where the synthetic analyte has been measured and found not to be present under baseline conditions. If there is a measurable background of the substance, then the maximum measured values should be used.

Experience has shown that baseline measurements indicate complex patterns of variation for significant numbers of parameters, with mean parameter values and standard deviations varying greatly from site to site. The time series from individual sites also can include large fluctuations that do not necessarily appear to follow regular seasonal or temporal trends, nor occur at the same time at different sites. Such behaviour is inconsistent with many statistical tests of differences between sets of measurements. To address this a statistical model can be constructed for the baseline variation of each of the indicator parameters. This model can be used to determine the degree of variation that could be expected in the data from the operational period if no underlying change has occurred. Models should be developed for both the API and control group of sites.

To develop the baseline statistical model the data need to be standardised so that the measurements from each site are comparable (Ward et al, 2019). Where parameter values have a highly skewed (i.e. asymmetric) distribution, as is often the case, a log-transform is required to reduce this skew so that the data are more consistent with a normal distribution. The data for each site in the group, e.g. API, are then standardised by subtracting the mean for that site and dividing by the standard deviation.

The most basic model assumes that the standardised data are drawn from a normal distribution with mean zero and unit variance. However, this may not be the case and so variograms of the standardised data need to be calculated and inspected to see if any spatial and/or temporal correlation is evident. Where it is, an appropriate correlation term needs to be added to the model. Similarly, if the mean of the baseline data varies with time an appropriate term needs to be added to the model if required. A statistical test can be used to confirm that any additional term improves the model before it is finally accepted.

The final baseline model can then be used to determine the expected mean standardised measurement value across the group of sites for a future round of sampling and the specified confidence limits for this mean (95% is recommended).

Trend assessment should also be undertaken to determine whether there is i) any seasonality in the data, and ii) any underlying trend. As a minimum, 12 measurements distributed over a year, i.e. monthly, are required for trend assessment.

Recognised statistical methods should be used for trend analysis and be applicable to the available data. Water-quality data possess unique characteristics that require specialist approaches to statistical testing. The data often have asymmetric or non-normal distributions and will therefore require non-parametric statistical methods where no assumptions are required

[3] UK Technical Advisory Group on the Water Framework Directive Paper 11b(i) - Groundwater Chemical Classification for the purposes of the Water Framework Directive and the Groundwater Directive

about the underlying data distribution. Methods that can be used include Sen's method (Sen, 1968) and Seasonal Kendall (Hirsch and Slack, 1984) where there is evidence of seasonality. Both of these methods are robust and will allow for some missing data in the time series and are not badly affected by outliers in the data series. These methods are also used for WFD trend assessment in the UK (UKTAG, 2012)[4]. To be consistent with WFD trend assessment, trends should be significant at the 80% confidence level.

Where a data series has a significant number of values that are reported as below the level of quantification (LOQ) care should be taken when calculating summary statistics and undertaking trend assessment. A commonly used approach is to replace values reported as below the LOQ with a value equal to half of the reported LOQ, although more sophisticated methods such as Kaplan-Meier or ROS (Regression on Order Statistics) approaches are appropriate (Helsel and Hirsch, 2002). Where more the 80% of the measurement values are below the LOQ, trend assessment should not be carried out and only the statistics a) to d) (identified above) reported.

If a statistically significant trend is identified and the data set has a significant number of censored data (but <80%), additional trend assessment can be undertaken by substituting the LOQ values by, firstly a value equal to the LOQ, and secondly, by zero. This further analysis will allow investigation of the effect of the censored data on any trend identified.

4.4.2 Change detection

(a). Individual measurements

Following the initiation of shale-gas operations, measurement values for each monitoring site and for each parameter identified in the environmental permit monitoring plan (and any others selected as indicators of change) should be compared to:

i. For naturally occurring substances (or analytes), the respective maximum value from the baseline analysis;
ii. For synthetic substances, the LOQ value or else maximum value if the substance was detected routinely during the baseline period.

If the measured value exceeds the maximum or LOQ value then this could indicate a change arising from shale-gas activities, and further investigation is required. This should include in the first instance:

i. Checking the results from other sites for any other exceedances, including outside the API,
ii. Comparison with regulatory surface-water environmental quality standard (EQS) values or groundwater (WFD) threshold values/quality standards as appropriate.

If the change indicates an impact arising from shale gas operations, the appropriate reporting/notification procedure should be followed and in a severe case, pollution incident protocols followed.

(b). Groups of monitoring sites (API)

Following the initiation of shale-gas operations, measurement values for each monitoring site in a sampling round should be standardised using the empirical site means and standard deviations from the baseline model (see Section 4.4.1).

The mean of these standardized data across the API should then be calculated for the sampling round. The corresponding confidence bands are then constructed using the baseline model parameters. A comparison of this mean with the mean from the basleine model can then be used

[4] UK Technical Advisory Group on the Water Framework Directive - Groundwater Trend Assessment

to indicate whether a deviation or change might have occurred. In his case further investigation should be carried out. This sshoul dinlciude in the first instance:

i. Checking the results from other sites for any other exceedances, including outside the API,
ii. Comparison with regulatory surface-water environmental quality standard (EQS) values or groundwater (WFD) threshold values/quality standards as appropriate.

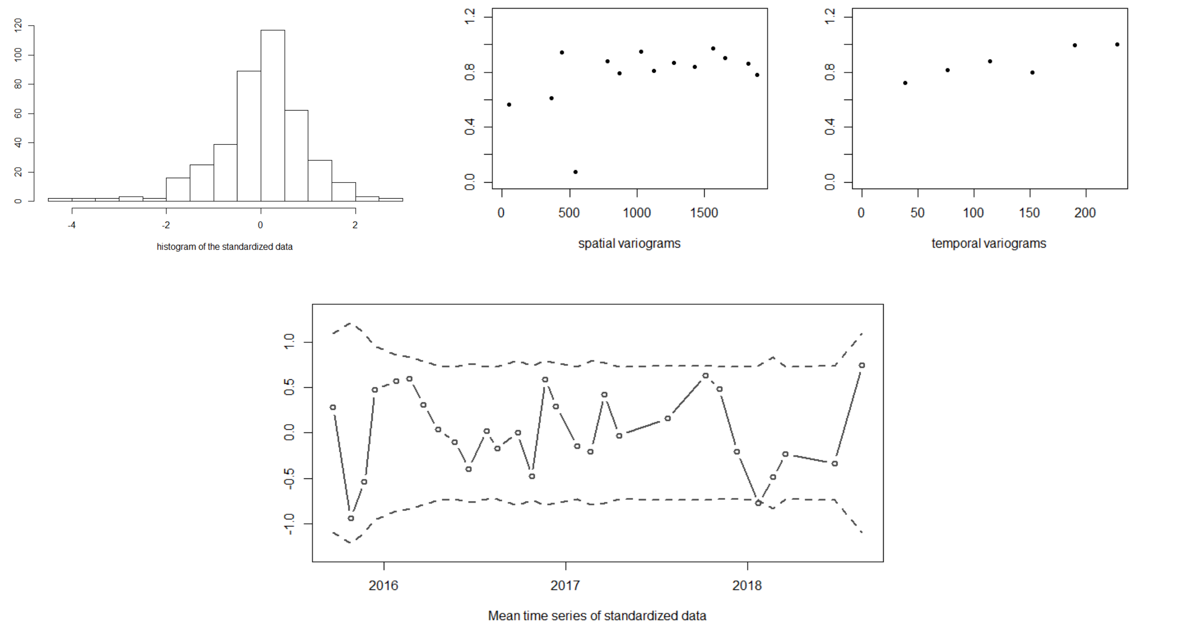
CASE STUDY

Water monitoring: detecting change – Vale of Pickering

Evaluation of water-quality data in the context of hydrocarbon exploration activities requires robust objective protocols to determine whether any differences observed between measurements carried out under baseline compared to operational conditions are larger than those that could have resulted wholly due to the underlying variability of the measurements. Observations indicate that baseline water-quality measurements can show complex patterns of variation that require a statistical model of the baseline variation of each analyte. Using Vale of Pickering groundwater-quality data, models have been produced to determine the degree of variation expected from the operational period if no underlying change has occurred. This can be compared to actual variation in the operational phase to determine their correspondence.

Figures below show a histogram of log-transformed standardised data (subtracting the mean for a given site and dividing by the standard deviation) for methane in groundwater from 20 selected monitoring sites, with empirical spatial and temporal variograms (middle/right panel). These reveal no distinctive patterns or correlations. A linear mixed-effect model with temporal random effect was fitted. The model was tested against a simple intercept model and the likelihood ratio test suggests that the temporal random effect is significant ($p<0.05$). The mean standardised time series (2015–2018) and the adjusted 95% confidence bands are also shown. The approach assumes that the model reflects accurately the baseline variation and does not account for any uncertainty in estimating the model.

Monitoring data collected during shale-gas operations can be compared to the baseline model by first standardising the data (as above) and then calculating the standardised mean. If this mean falls outside the 95% confidence bands for the baseline model, an expected change has been identified and further investigation should take place,



(c). Trend assessment

Determining whether there is a statistically significant upward (or downward) trend in analyte values requires a sufficient time series of measurements. Section 1.5.1 outlines the minimum data requirements and the recommended approaches for statistical trend assessment. This means that at least one year's worth of monitoring data is required. A significant trend in terms of triggering further investigation is one that indicated a change of $\geq$ 10% from any trend observed during the baseline period.

It is possible that a change will start to be observed before sufficient operational monitoring data have been acquired to carry out statistical trend analysis. Therefore, it is recommended that time-series data are plotted regularly and a visual inspection made to identify any emerging trends in the data in the API that are different from those seen during the baseline period and/or in the control areas.

If a trend is suspected, further investigation is required. This should include increasing the frequency of monitoring to confirm any trends with greater confidence, and initiating other management actions.

Further information on assessing the statistical significance of changes in water quality data can found in an Environment Agency report (EA, 2019). This report also covers air quality but it is also relevant to other 'concentration' type monitoring data.

4.5 REFERENCES

Bloomfield, J P, Allen, D J, and Griffiths, K J. 2009. Examining geological controls on baseflow index (BFI) using regression analysis: An illustration from the Thames Basin, UK. Journal of Hydrology, 373, 164–176

BS EN ISO 5667-3:2012 -Water quality. Sampling. Preservation and handling of water samples;

BS EN ISO 5667-3:2012 -Water quality. Sampling. Guidance on sampling from lakes, natural and man-made;

BS EN ISO 5667-6:2016 - Water quality. Sampling. Guidance on sampling of rivers and streams;

BS ISO 5667-11:2009, BS 6068-6.11:2009 - Water quality. Sampling. Guidance on sampling of groundwaters;

BS EN ISO 5667-14:2016 - Water quality. Sampling. Guidance on quality assurance and quality control of environmental water sampling and handling;

BS ISO 5667-20:2008 -Water quality. Sampling. Guidance on the use of sampling data for decision making. Compliance with thresholds and classification systems.

BS ISO 5667-22:2010 - Water quality. Sampling. Guidance on the design and installation of groundwater monitoring points;

CL:AIRE, 2002 Technical Bulletin TB2 – Multi-level sampling systems - http://www.claire.co.uk/component/phocadownload/category/17-technical-bulletins?download=46:technicalbulletin02

Chapman, S., Parker, B. L., Cherry, J., Munn, J., Malenica, A., Ingleton, R., Jiang, Y., Padusenko, G., Piersol, J. 2015. Hybrid multilevel system for monitoring groundwater flow and agricultural impacts in fractured sedimentary bedrock. Groundwater Monitoring & Remediation, 35(1), 55-67. http://dx.doi.org/10.1016/10.1111/gwmr.12084.

Environment Agency. 2019. Onshore oil and gas monitoring: assessing the statistical significance of change. Environment Agency report SC160020.

Freitas, J.G., Rivett, M.O., Roche, R.S., Tellam, J.H., Durrant, M., Walker, C., 2015. Heterogeneous hyporheic zone dechlorination of a TCE groundwater plume discharging to an urban river reach. Science of the Total Environment 505, 236–252, http://dx.doi.org/10.1016/j.scitotenv.2014.09.083

Helsel, D.R., Hirsch, R.M. 2002. Statistical Methods in Water Resources. Techniques of Water-Resources Investigations of the United States Geological Survey, Chapter A3, USGS.

McMillan, L.A., Rivett, M.O., Wealthall, G.P., Zeeb, P., Dumble, P., 2018. Monitoring well utility in a heterogeneous DNAPL source zone area: insights from proximal multilevel sampler wells and sampling capture-zone modelling. Journal of Contaminant Hydrology, 210, 15-30. https://doi.org/10.1016/j.jconhyd.2018.02.001

UK-TAG, 2011. Defining & Reporting on Groundwater Bodies. UK Technical Advisory Group on the Water Framework Directive. https://www.wfduk.org.

Ward et al. (2019). Environmental Monitoring - Phase 4 Final report. Report of the British Geological Survey, OR/19/044.

5. Atmospheric composition monitoring

5.1 INTRODUCTION

There is a range of different potential air pollution and climate impacts arising from emissions associated with shale gas extraction activities. These have been well documented in a US context in recent papers, e.g. Edwards et al. (2014). This section describes the scientific and regulatory context to emissions thought to be associated with hydraulic fracturing and then defines what is meant by an environmental baseline in terms of atmospheric composition. It then provides some recommendations for good practice in the establishment of baseline environmental conditions.

5.1.1 Air pollutants

Short-lived air pollutant emissions can arise from site infrastructure (e.g. particulate matter (PM) and gaseous pollutants such as NO_x from generators and traffic movements), fugitive emission of volatiles from condensates, extracted gases and flow back (e.g. light non-methane hydrocarbons, radon, hydrogen sulphide (H_2S), benzene and malodorous compounds). There are also distributed emissions associated with the wider life-cycle such as road transport and supply chain (primarily nitrogen oxides (NO_x) and PM), and downstream gas distribution and end-use. Air pollution impacts linked to shale gas extraction in some locations in the US have led to localized exceedances of safe exposure thresholds for concentrations of VOCs such as benzene, and regional elevations in tropospheric ozone (O_3) due to photochemical production downwind. In some cases unconventional gas extraction has led to non-compliance with air quality standards in locations that had not previously breached US standards (Edwards et al, 2014).

The 2008 European ambient air quality directive (2008/50/EC) sets legally-binding limits for concentrations in outdoor air for both particulate matter (PM_{10} and $PM_{2.5}$) and nitrogen dioxide (NO_2), which are associated with a range of acute and chronic health conditions such as cardiovascular and respiratory illnesses. There are also limits for known carcinogens and air-toxics such as H_2S, benzene and 1,3-butadiene. The UK also has short and long-term exposure thresholds for ozone concentration. These thresholds are explained further in the National Air Quality Objectives [1]. It should be noted that some air quality pollutants can be enhanced before hydraulic fracturing and shale gas extraction begin due to site and well preparation work. Elevated concentrations of PM and NO_2 will be observed as a result of the movement of equipment (plant) on to the site, the operation of generators and increased vehicle movements in preparation for operations to start. Such emissions should therefore be considered to be a direct consequence of shale gas operations and not a component of any prior local baseline. It is important that monitoring includes, and differentiates, the different periods of activity on a site.

5.1.2 Greenhouse gases

The class of impacts relating to greenhouse gases concern both direct CO_2 emission (i.e. combustion and end-use) and controlled, or fugitive, emissions of CH_4 associated with extracted gas (including flowback), gas storage, and possible geological seeps (via groundwater, including well annulus pathways) induced by drilling and/or hydraulic fracturing. This latter impact, which concerns geological pathways, remains contentious and uncertain, with limited evidence to-date from US case studies, yet it also represents possibly the most difficult pathway of fugitive emission to monitor and quantify, and/or mitigate if it arises. Soil gas monitoring may offer an important tool from which to assess this and a soil gas monitoring programme can provide supporting evidence to support interpretation. Recommendations for soil gas monitoring are in Section 7. There has been a clear increasing trend in global methane concentration since 2006, so there is intense scientific interest in understanding the components of the global methane budget – including the relative importance of the oil and gas industry, e.g. Allen (2016).

The potential contribution of shale-gas-related greenhouse gases to the UK emissions inventory was discussed in the DECC report on *Potential greenhouse gas emissions associated with shale gas extraction and use* (Mackay and Stone, 2013). The report concluded that the potential and anticipated impacts of shale-gas-related emissions in the UK remain uncertain. Currently there is substantial uncertainty over possible emissions at the scale of a single installation, and there exist a range of possible scenarios for the sector's expansion trajectory. Importantly, the UK experiences markedly different atmospheric conditions (e.g. wind speed, solar insolation, boundary layer, background atmospheric composition etc) to typical major shale gas plays currently in production in the US. Perhaps as importantly, the UK may be expected to have different exposure profiles (compared with the US) due to proximity to operational sites and potentially greater proximity of communities to operational sites.

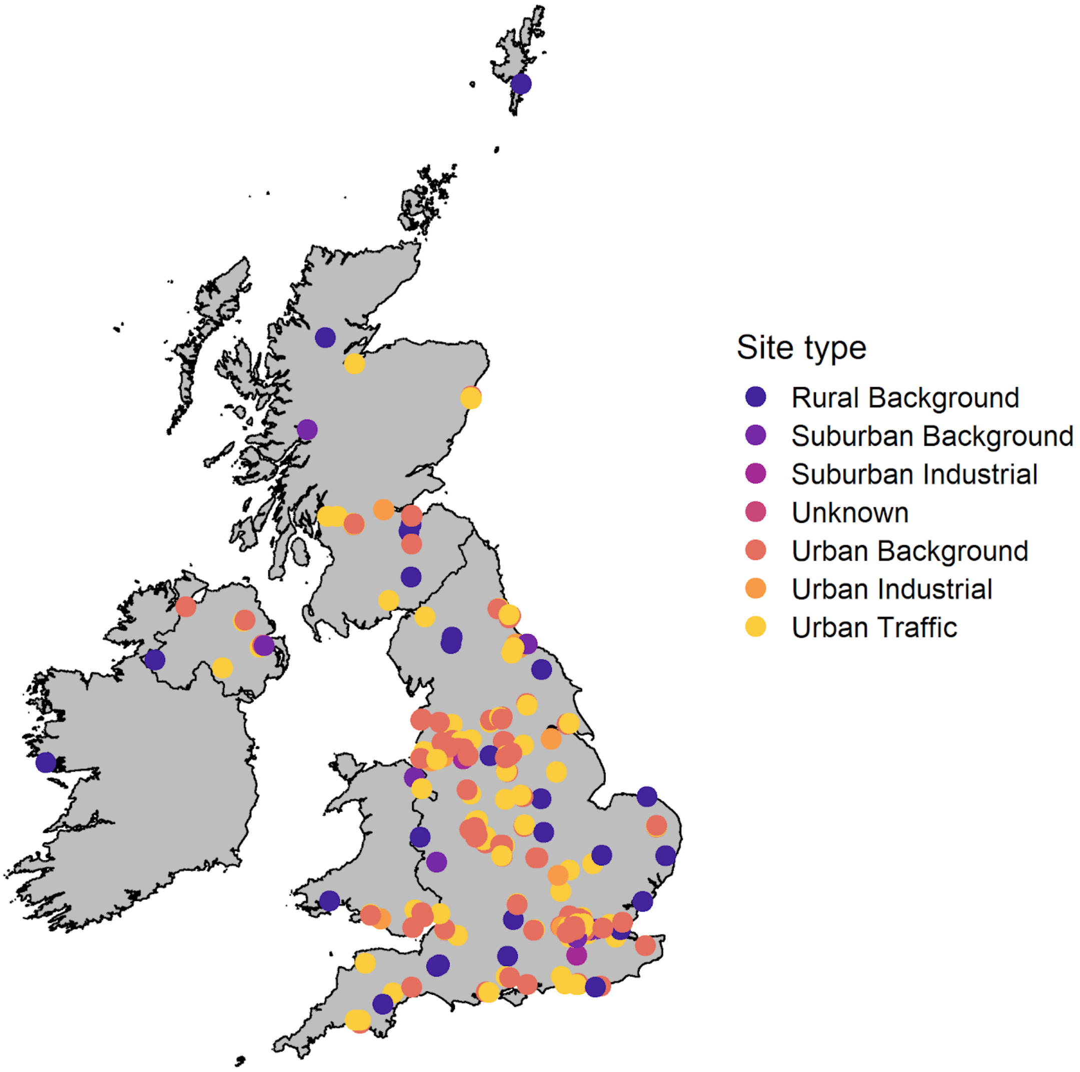


Figure 20. Map of the current UK AURN (Automatic Urban and Rural Network) air-quality network, coloured by site type ©University of Manchester, 2020

There is some notable existing monitoring infrastructure that can augment site-specific monitoring and aid in characterising the regional background in atmospheric composition. The AURN (Automatic Urban and Rural Network) air-quality network is the UK's largest automatic monitoring network and is used for compliance reporting against European air-quality directives. There are currently 127 AURN sites (Figure 20) in operation, measuring a range of parameters including carbon monoxide (CO), NO_x, PM, O_3, and sulphur dioxide (SO_2). Not all of these sites measure all of these parameters, as the specification of each depends on location and classification (e.g. roadside, urban, sub-urban, rural and industrial). However, there can be large distances between sites, especially in rural areas which are typically of most interest to the shale gas industry. Therefore, existing networks, while useful for regional purposes and for long term trend analysis, cannot be interpolated to derive meaningful data for local baselines fit for exposure monitoring.

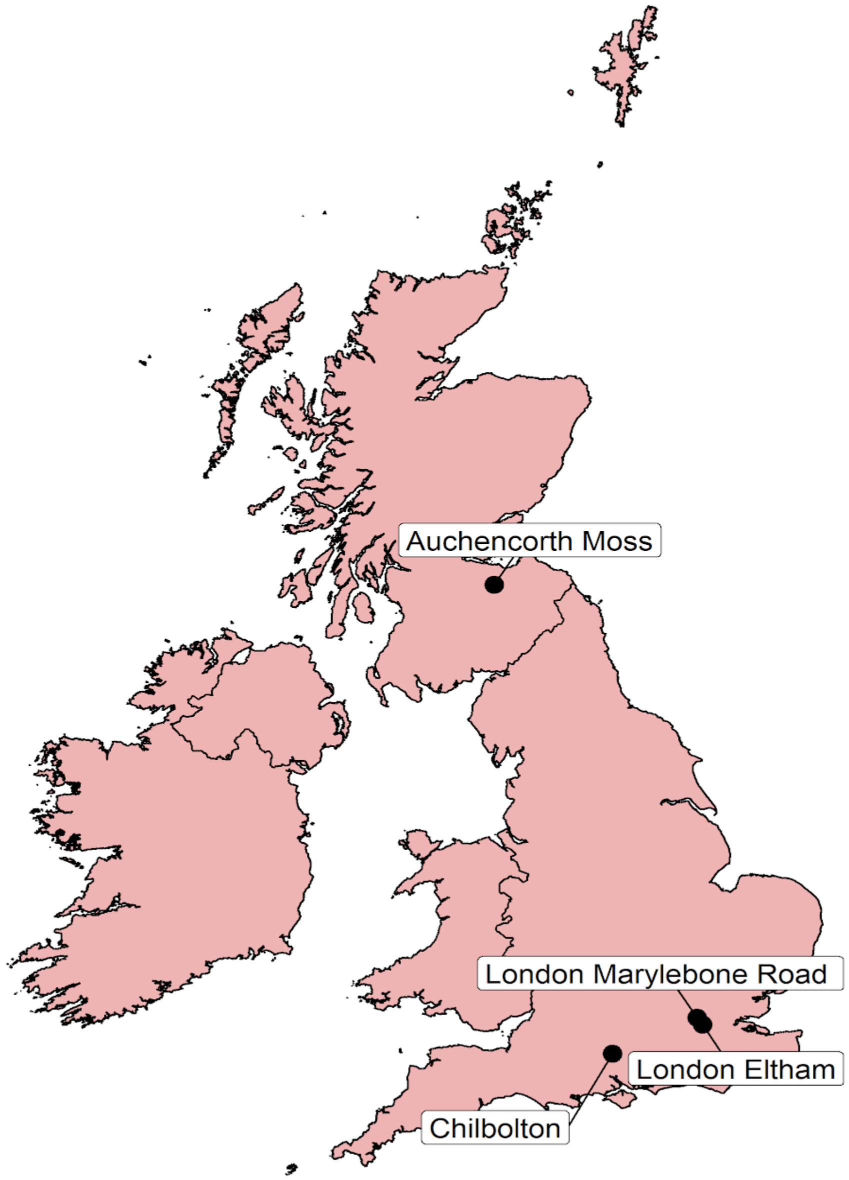


Figure 21. Map of the current UK Hydrocarbon network. ©University of Manchester, 2020

Two networks exist for measurements of hydrocarbons in the UK. The first is the Automatic Hydrocarbon Network, which currently measures hourly concentrations of a range of hydrocarbons at four sites in the UK. The sites are Harwell, London; Eltham, London; Marylebone, London; and Auchencorth Moss in Scotland, which represents the most rural site.

The non- automatic hydrocarbon network consists of 37 sites where benzene is sampled by adsorption tube and analysed to yield concentration data at a later date in the laboratory. These sites are all co-located with existing AURN sites (Figure 21).

The Deriving Emissions for Climate Change (DECC) tall-tower greenhouse-gas (GHG) network (used for national GHG inventory validation) represents measurements in more rural locations by virtue of their installation high on large telecommunications masts outside of urban settings. However, their number (currently just six across the UK) and siting (e.g. the North West currently lacks such a station for example) is not sufficient for local baseline measurements and characterisation, and does not provide data at the local ground level from which to characterise future near-field impacts.

The following sections describe recommendations on the best available practice in the application of baseline atmospheric composition monitoring, defined here as the statistical characterisation of the prevailing conditions in any local area, or at a site, prior to the commencement of shale gas operational activity (including any preparatory activity). The intention of any atmospheric baseline monitoring is that a statistically and locally representative data set sampled at a site will serve usefully as a comparison with any future operation observations in order to attempt to apportion incremental impact to local atmospheric composition should this arise, or to confirm that no change has arisen. Further, this baseline characterisation can inform on the pre-existing local and regional background in terms of near- and far-field sources of pollution and enable future changes in these unrelated air pollution sources to be identified.

5.2 SITE SELECTION AND LOGISTICS

The assessment of the impacts of local sources of pollution on local and regional receptor environments requires a local monitoring approach. A shale-gas site may (based on US experience) represent a potentially large point source of pollutant gases and particulate matter. Given that the public-health issues typically concern local (<10 km distance from source) and regional (<100 km) exposure, this leads to the conclusion that site-specific baseline monitoring will be necessary to obtain meaningful statistics at the local scale of interest, especially concerning air quality.

For baseline monitoring, a single monitoring site positioned near to a planned shale site (see later) can be sufficient to collect data from which to derive a locally-representative statistical climatology. However, this must not be confused with potential monitoring requirements for assessment of operational activities, which must be assessed separately, and optimally may require a spatial network of monitoring to best capture emissions over various wind directions. Furthermore, direct attribution of site-specific emissions may require a simultaneous upwind measurement to rule out extraneous sources of pollution further upwind to an operational site. As the cost of high-precision atmospheric monitoring equipment and operation is non-trivial, the number of operational monitoring sites may require a compromise between the cost of multi-site operation, and the idealised sampling of all emissions from an operational site. Where such a compromise is deemed necessary, it is important to select sampling locations that best capture emissions. For example, monitoring sites placed downwind with respect to the dominant winds at a planned shale gas facility can optimize the sampling time and measurements directly attributable to on-site activities and emissions.

The siting of a baseline monitoring station (or stations) must be guided to optimise the sampling of a locally representative baseline over a wide (and typical) range of meteorological conditions. A further consideration should be that measurement instrumentation is sited appropriately with respect to its role in meeting potential operational monitoring requirements, i.e. be near to sites of planned shale-gas extraction.

Table 3. Parameters and appropriate measurement techniques required for baseline assessment at any monitoring station.

Measurement	**Technique**	**Temporal sampling/ integration**	**Dynamic Range**	**Measurement precision - 1 σ rms (at 1 minute integration)**
NO, NO_2, NO_x	Chemiluminescence with photolytic converter	1 minute	0 - 20000 ppb	0.25 ppb
O_3	Photometric Ozone	1 minute	0 - 200 ppm	0.25 ppb
PM (1, 2.5, 4, 10 size fractions)	Optical Light scattering	1 minute	0 - 10000 $\mu g/m^3$ (PM size fractions 0-20000 particles/cm^3 (particle count)	0.1 $\mu g/m^3$
CH_4	Infrared spectroscopy	1 minute	0.01-100 ppm	1.3 ppb
CO_2	Infrared spectroscopy	1 minute	0.1-2000 ppm	0.03 ppm
Speciated non-methane hydrocarbons	Gas Chromatography	*1 hour	0 - 10 ppb	#<5%
Meteorological data (wind speed, wind direction, Temperature, pressure, Relative Humidity)	Automatic weather station	1 minute	-50-100 ^0C 800-1100 hPa 0-100% RH 0-60 ms^{-1}	0.1 ^0C, 0.5 hPa 0.8% +/-2% (at 12 ms^{-1})

*At current sites NMHC canister collection is weekly but ideally this would be continuous hourly sampling. Alternatively, new instrumentation is now available to measure ethane at 1 minute intervals.

A baseline monitoring station should be ideally placed between 100 m and 500 m downwind of a planned operational site, in a radial direction that would best capture background airmasses that pass over the area. Here, a downwind direction is defined with respect to the most common wind direction assessed from historical meteorological data for the site of interest. Local data should be used to ensure the measurement site is positioned in a place that will be influenced by mixed air impacted by sources from the future operational site. The Met Office has over 200 monitoring stations in the UK with historical wind information. This range in distance (100 m to 500 m) allows time for emissions to partially mix in the air downwind, allowing emission plumes to be sampled more readily, whilst ensuring that dilution does not negate their detectability using instruments described in Table 3. Put simply, a baseline measurement station positioned too close to an operational site may not be optimal for subsequent operational sampling, as it may not observe emitted plumes that have not had time to mix over a wide angle; while monitoring further away may mean that plumes are too diluted to discern a signal within measurement uncertainty. An assessment of optimised monitoring location(s) might be facilitated by dispersion modelling of plumes for a range of simulated emission fluxes, to establish limits of detection as a function of distance downwind. Monitoring locations should avoid places where a shale-site signal could be obscured by nearby confounding emission sources. This is because the superimposition of these on shale-site plumes can potentially make any shale signal undetectable or difficult to attribute - until/unless that signal is elevated to levels that clearly exceed the high baseline. It should also be recognised that the baseline can become out-of-date as a result of significant changes in nearby emission sources, and if this occurs the confidence in attributing any future change to activities at the shale gas site could be reduced.

These siting criteria serve to optimise efficient baseline measurements prior to site activity, and facilitate future operational measurement. By way of example for the baseline measurement site at Little Plumpton, Lancashire, operated as part of the BGS-led baseline project (Shaw et al, 2019), the monitoring site was positioned ~400 m directly to the east of the proposed well pad. This position was chosen because of the dominance of westerly winds at this location (common for many areas of the UK), and therefore its downwind position relative to the operational site, to ensure that the site could transition to an operational monitoring site once shale gas operations started..

It should be acknowledged that an atmospheric monitoring station, or stations, may have practical and logistical issues due to site security and safety and the availability of suitable space and land ownership. In addition, there is a need for continuous power (ideally from a mains supply). The use of a generator for power would manifest a non-baseline source of emissions, which would likely adversely affect measurements; and should therefore be avoided. Solar and small wind turbine power solutions, with suitable battery provision, may be considered; however, care must be taken to ensure that power provision is continuous and that adequate warning is given of any projected power shortage such that steps can be taken to prevent disruption to data collection (e.g. site visits to install fully charged batteries or alternative power supply).

5.3 MEASUREMENTS

The development and operation of a shale gas site consists of different operational stages, each of which has the potential for emissions to the atmosphere. The main stages during development of a site are shown in Table 4. When measuring near to any emission source, it is important to have a relatively high measurement frequency to yield useful information on source strength and chemical transformations (e.g. NO, NO_2 and O_3 (ozone)) (see case study – *methane measurements at different frequencies*). Therefore, integration and sampling at 1-minute intervals (or more frequently) is required to capture transient source features and to characterise plume morphology, in turn facilitating apportionment and flux quantification. 1-minute intervals are especially useful for CH_4, NO, NO_2, O_3, PM; additionally, ethane (C_2H_6) would be a good marker for the shale gas industry specifically. A report from the Air Quality Expert Group in 2015 on Evidential Value of Defra Air Quality Compliance Monitoring recommended that this resolution for air-pollution data would also allow new and innovative use of the measurements to support a range of science and policy needs. This is a higher temporal resolution than the AURN networks, which currently report at 15-minute intervals. High temporal resolution allows the establishment of the frequency and duration of short-term episodic pollution events, down to the minute timescale, and in combination with weather data identify the geographic regions that are potentially contributing.

Table 4. List of potential atmospheric pollutants associated with different stages of development of a shale gas site.

Stage	Source of emission	Potential pollutants
Well drilling and completion	Dust	PM
	Diesel generators	PM, NMHC, NO_X
	Traffic	PM, NMHC, NO_X
	Chemical processing	O_3
	Fugitive	NMHC, H_2S, CH_4
Pre-operational/mobilisation phase	Dust	PM
	Diesel generators	PM, NMHC, NO_X
	Traffic	PM, NMHC, NO_X
	Chemical processing	O_3
Hydraulic Fracturing	Dust	PM
	Diesel generators	PM, NMHC, NO_X
	Traffic	PM, NMHC, NO_X
	Chemical processing	O_3
	Fugitive	NMHC, H_2S, CH_4
Well Production	Fugitive	
		NMHC, H_2S, CH_4
Well decommissioning and site restoration	Traffic	PM, NMHC, NO_X
	Chemical processing	O_3
	Fugitive	
		NMHC, H_2S, CH_4

It is also important to highlight that 1-minute data still has some limitations, although these can generally be overcome by averaging over a longer time period. For example: (i) 1-minute data emphasise "fluctuations" in dispersion due to turbulence that can make it hard to interpret individual 1-minute values; (ii) it might be impractical to analyse individual 1-minute values at a site, and unnecessary because most short-term site emissions are likely to last for longer than 1-minute - so collating over a longer period might be more practical; (iii) 1-minute is not a usual averaging time for short-term human health purposes - which usually focus on 15-minutes or 1-hour.

Measurement at the minute-average timescale enables a much more informative data analysis linking short-term variability in wind speed and direction to atmospheric composition when compared to the regulatory requirement (2008 Ambient Air Quality Directive (2008/50/EC)) which sets legally binding limits and target values for a minimum of hourly averaging. In the case of the instruments listed in Table 3, measuring on a 15-minute or 1-hour averaged basis would involve the same instrumentation, so there is no cost advantage in making slower, less time-resolved, measurements. Calibration and maintenance would also still be carried out at the same frequency so there would not be any resource implications of higher time resolution data. Flux-related calculations also require measurements with high time resolution, minute average and preferably faster.

Table 3 lists the parameters and appropriate measurement techniques required for assessment of baseline conditions at any monitoring station. The measurement techniques represent those known to be commensurate with the precision and dynamic range requirements needed to establish appropriate data quality for baseline ambient background air. Such high-precision measurement is very different to typical requirements for on-site monitoring in the context of natural gas leak detection and repair (LDAR), where much cheaper and more portable methods such as open-path tunable diode laser (TDL) handheld instrumentation may suffice. While instruments such as TDL are highly efficient in detecting and pin-pointing sources of fugitive emission, and wearable sensors can serve to safeguard personal safety, they do not facilitate the detection and characterisation of emissions everywhere on a site at all times, nor facilitate the calculation of an emission flux and assessment of off-site receptor relationships. The guidance in Table 3 concerning dynamic range (limit of detection to maximum range of measurement linearity) captures the typical extremities expected in ambient measurements during baseline assessment, while the measurement precision (for the recommended 1-minute integration period) represents the data resolution required to usefully detect typical changes in ambient air associated with changes in air mass or source inputs. Instrument precision, a measure of the white noise of sampled data and hence an indicator of the resolution of data should not be confused with instrumental accuracy (a measure of the instrumental drift with respect to a reference standard).

5.3.1 Miniaturised sensors

Currently (as of 2019) many different types of miniaturised sensors for the measurement of atmospheric air pollutants are available. Such sensors are defined as devices that purport to make autonomous observations of multiple pollutant parameters at a lower capital cost than laboratory-grade analytical equipment, with costs spanning a range from £100 up to £10,000 per observing location (see Lewis et al (2016)). However, there are notable limitations and performance issues with such sensors. Recent research has found that sensor data can be unreliable as they can react inconsistently to a given input and respond as much to humidity, temperature, or other atmospheric gases as to the pollutants being targeted (Lewis et al, 2016). At this time they are not recommended to be used for atmospheric baseline monitoring, which requires the precision of the techniques listed in Table 3 for detecting small temporal concentration fluctuations.

Data quality objectives defined by GAW-VOC measurement guidelines (Report 204) are currently set at 20%, but recent work from the ACTRIS-VOC community describes the need for 5% uncertainty targets to be reached in order to observe decadal trends in VOCs.

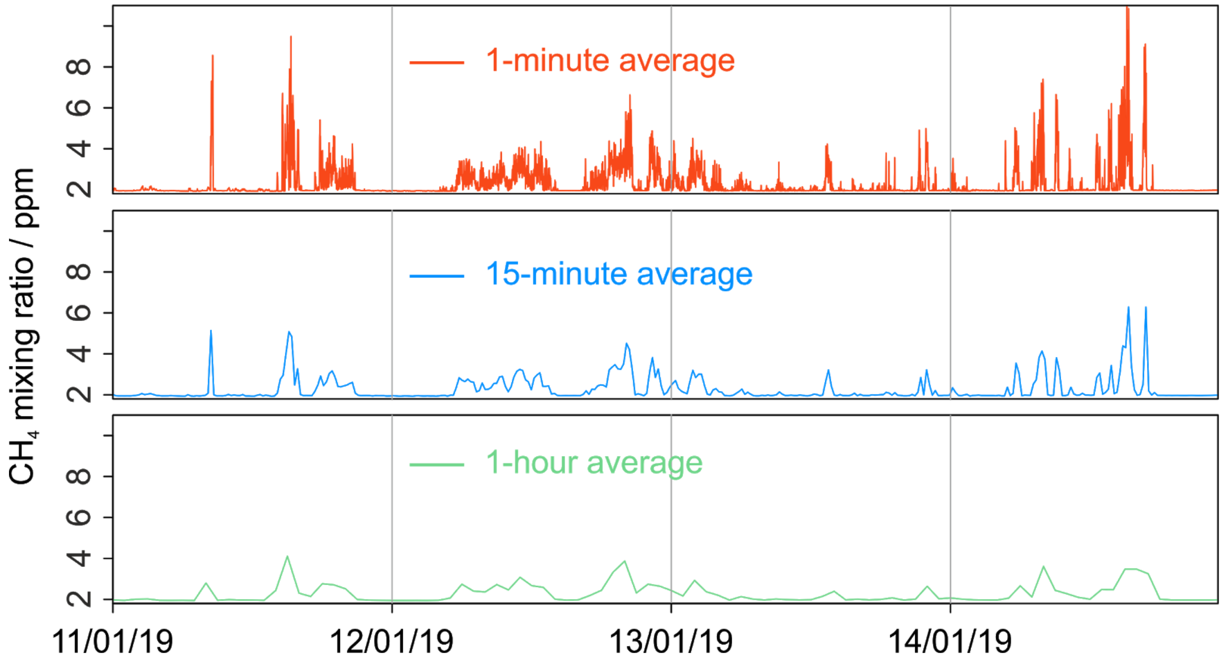
It is important to note that currently benzene is the only VOC routinely measured by DEFRA to assess compliance with UK concentration thresholds for air quality. To assess the impact of shale gas activities it is advisable to measure additional speciated hydrocarbons such as ethane and propane, along with methane, to establish characterisable emission factors for the industry. More recently, Helmig et al. (2016) used ethane data from global surface networks to show the decline observed between 2005 and 2010 has reversed and calculate a yearly increase of ethane in the Northern Hemisphere of 0.42 ± 0.19 Tg yr^{-1} between 2009 and 2014. North

American oil and natural gas development is suggested to be the primary source of these emissions.

Whilst a minimum 12-month period of continuous baseline measurement is considered necessary to establish an environmental baseline, a longer period of monitoring is recommended to ensure that intra-annual variability can be assessed. This includes meteorological parameters, air quality and greenhouse gas concentrations. Assessment of inter-annual variability is required since there is considerable seasonal dependence in the prevailing meteorology, atmospheric reactivity and the formation of secondary pollutants. Year round measurements capture appropriately the diurnal, weekly, and seasonal patterns. As an example, ozone has a broad seasonal cycle (see case study on seasonal differences) peaking in the spring months, and with occasional very high episodes in summer. These are typically anti-correlated with CO_2 concentration, which dips in summer months due to biospheric respiration. Wind speed and direction, monitored at the measurement station, is used to inform on the direction and proximity of near-field pre-existing sources in the baseline and also serve to deconvolve the role of long-range inputs, this is done by looking at air mass history and concentration ratios.

Case study- Methane measurements at 1 minute, 1 hour, and daily frequencies

The figure below shows CH_4 mixing ratio data for the period 11th – 14th January 2019. Different averaging scenarios have been applied to the data to simulate the impact that a reduced frequency of measurements would have on the data. Whilst the enhancements in CH_4 mixing ratio are visible in the 1-hour average data, much of the temporal resolution is lost. This plot clearly demonstrates the advantages that higher frequency, 1-minute measurements have over lower frequency measurements.



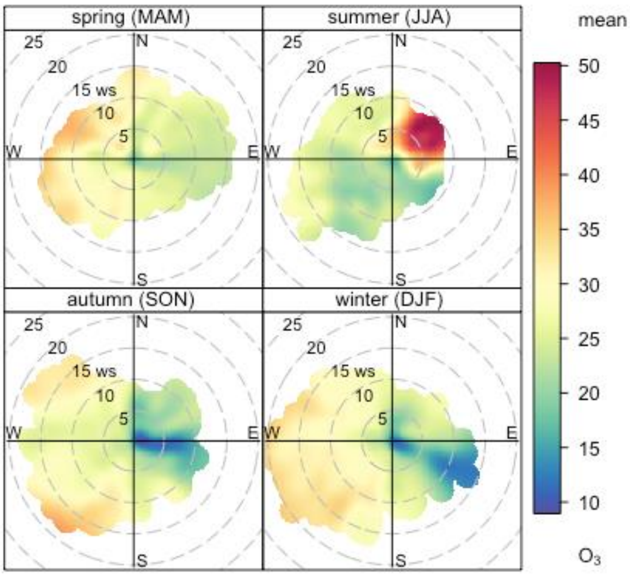
CH_4 mixing ratios from 11th – 14th January 2019. The top panel shows 1-minute averaged data. The bottom panels show mixing ratio values with a lower frequency of measurements (15-minute and 1-hour). ©University of Manchester, 2020

Provision of metadata is essential to allow end-users to properly assess, interpret, understand and use a dataset. Observational metadata should include details on how (with the instrument or technique), where, with what accuracy, and by whom the data was collected by (contact details). It should also include the corresponding measurement uncertainties, such as instrument precision, calibration and traceability (see below) as well as information on known sources. Such metadata must be prepared for each measurement reported.

Case study - Seasonal differences

The time of year can have an effect on air quality baseline measurements. Some air quality parameters will be higher or lower at certain times of the year due to meteorology and source differences. It is important seasonality is taken into account for interpretations of air quality to avoid the impacts of new activities (such as shale gas) being masked or wrongly attributed.

The figure below shows polar plots for the seasonal cycle of ozone at a site close to the west coast of northern England, with concentrations (colour scale, in ppb), wind direction (radial bearing) and wind speed (radial distance from source in m/s). A polar plot combines meteorological measurements with concentrations to show how they vary with wind speed and direction. These have then been split to show the seasonal cycle.



Polar plot showing the seasonal cycle of ozone measurements (in ppb) at a site near the west coast of the UK in 2016. © University of York, 2020.

Ozone (O_3) concentrations are highest in the summer at low wind speeds. In the summer, during periods of high temperature and anticyclonic weather conditions, ozone can also increase due to photochemical production. In some regions of the UK, in these conditions, ozone can be measured at concentrations greater than 100 ppb for short periods. High time resolution of the dataset to establish the frequency and duration of short-term episodic pollution events, down to the minute timescale and in combination with meteorological data, is required to identify the geographic regions that are potentially contributing.

There are also elevated O_3 levels when wind speeds are high and from the west. This is likely due to the annual O_3 peak in the northern hemispheric and north Atlantic Ocean, and the impact of efficient long-range transport of this air to the measurement site. Elevated O_3 is indicative of a matured air mass as it is not a primary emission, and is instead produced through photochemical reactions within the air mass.

The lower O_3 levels from the east and south-east are particularly notable during the autumn and winter may indicate ozone titration due to NO_x emissions from a nearby dairy farm.

A good quality-assurance and quality-control (QA/QC) regime for data provision, which covers all aspects of measurement, including equipment evaluation, site operation, site maintenance and calibration, data review and ratification, is required. All calibrations must be traceable through an unbroken chain to established international metrological standards. Regular site visits (at least monthly) and remote monitoring (at least weekly) should be conducted to perform checks on instrumental performance in terms of accuracy, precision and response time, as well as calibration against traceable reference standards. A detailed list of calibrations and checks is given in Table 5, all are available commercially.

5.4 DATA PROCESSING

There are a range of open source tools available to derive basic statistics for air quality and climate data, including statistical variability within daily, weekly and seasonal timescales. There will be considerable seasonal difference in baseline air pollution at potential sites driven by meteorological factors on large scales, and it is essential that the monitoring period sufficiently captures this change.

One of the tools available for data processing is the open source resource OPEN_AIR (http://www.openair-project.org/). This is a collection of tools for the analysis of air pollution data. All plots in this guidance report are produced by OPEN_AIR.

The Openair project is fit for purpose for atmospheric baseline interpretation by providing:

- a free, open-source set of tools available to everyone;
- a range of existing techniques and developing new ones for the analysis of air pollution data;
- the statistical/data analysis software R as a platform – a powerful, open-source programming language ideal for insightful data analysis;
- an easy method for carrying out sophisticated analyses quickly, in an interactive and reproducible way;
- opportunity for the air quality community to use and help further develop these tools.

Data statistics relevant to atmospheric baselines are the mean, standard deviation, 5th and 95th percentiles, and maximum and minimum concentrations for measurements grouped by:

- The full baseline period (≥12 months);
- Wind direction (in 16 or more compass sectors);
- Time of day;
- Day of week;
- Month;
- Meteorological season.

It is also recommended to construct polar plots (see case study – seasonal differences) to diagnose existing near-field emission sources, and to examine relationships (e.g. correlations and anti-correlations over large concentration ranges) between different tracers to facilitate source-type characterisation and long-range airmass history. Bivariate plots use wind speed and direction coloured by pollutant mixing ratio to help reveal source locations. It can also be useful to plot the ratio of concentrations for two pollutants concentrations on a polar plot. For example, the ratio of NO to NO_2 can suggest if a situation has either (i) mostly "fresh" nitrogen oxides from nearby combustion, or (ii) mostly "aged" nitrogen oxides from more distant sources with a higher proportion of NO_2 as there has been more opportunity to convert NO to NO_2.

Such statistics, when interpreted over and within a 12-month (or greater) period of baseline sampling, provide for a direct comparison with future analogous observations during operational phases to examine any change in the background over time. This facilitates the careful apportionment of any increment to specific nearby activity if it arises. However, care must be taken to establish the unique site-specific shelf-life of the baseline used for operational comparison in order to remove or otherwise account for any significant extraneous (non-target) changes that may manifest, e.g. the installation or removal of other significant near-field emission sources in the period between the establishment of the baseline and later sampling.

Examples of such changes may be nearby (non shale gas) new industry or land use, agricultural practice, or changes in the road fleet etc.

Table 5. Calibration and maintenance procedures recommended for atmospheric composition monitoring

Parameter	Calibration and maintenance procedure	Field or lab calibration
NO and NO_2	Traceable calibration cylinders from the National Physical Laboratory (NPL). Monthly checks of analyser accuracy, precision convertor efficiency.	NO - field NO_2 - laboratory
Ozone	Six monthly calibration in the field by a calibration unit links to a primary UV photometric standard that is itself calibrated against a certified national source annually at the National Physical Laboratory.	Laboratory
Particulate matter	Six monthly calibration in the field by a monodust (CalDust), monthly maintenance checks	Field
NMHCS (Non Methane Hydrocarbons)	Calibration of NMHCs is performed by reference to an NPL (National Physics Laboratory) ozone precursor mix. This calibration scale has been adopted by the (Global Atmosphere Watch) GAW-VOC network and hence the measurements of NMHCs made by this instrument are directly comparable to those made by all of the WMO-GAW (World Meteorological Organisation - Global Atmosphere Watch) global observatories. Calibrations are performed each month or more frequently if field deployment allows. A long-term data set of the response of the instrument is held and regularly updated to ensure that the instrument responses do not change and to highlight any issues with stability of components within the gas standards used.	Laboratory
CH_4	Six-monthly traceability to WMO - compliant reference gas standards, e.g. as currently provided by NOAA (US National Oceanic and Atmospheric Administration), and EMPA-Switzerland across a calibration range between 1.5 to 2.5 ppm to establish linearity. Response times must also be established using calibrant gas pulses at inlet. Monthly checks of inlet obstruction.	Field
CO_2	Six-monthly traceability to (WMO-compliant reference gas standards, e.g. as currently provided by NOAA, and EMPA-Switzerland (Swiss Federal Laboratories for Materials Science and Technology) across a calibration range between 1.5 to 2.5 ppm to establish linearity. Response times must also be established using calibrant gas pulses at inlet. Monthly checks of inlet obstruction.	Field

As a shale gas site transitions into operational activities, it may be necessary to look at the monitoring data in different ways or use different monitoring methods to detect changes. As budget constraints might mean that there is only one monitoring station, its efficacy would be reduced if the wind direction was not the dominant wind direction for the period of operation, i.e. if the wind was not blowing predominantly from the shale gas site towards the monitoring point. It is also useful to have measurements both upwind and downwind of an activity to positively ascertain that the source is actually the shale gas site. This could otherwise be achieved by using mobile measurements that are repeated periodically from before operations start and during the period of operations at the site.

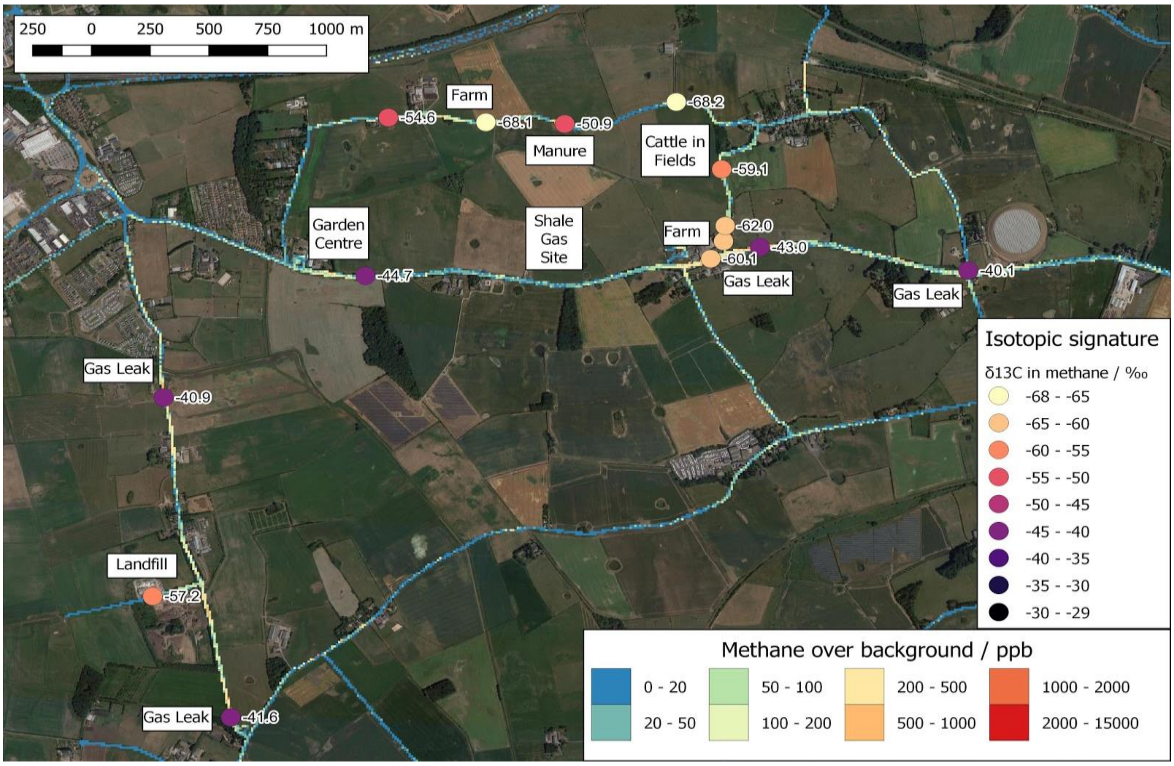
Case study – Characterisation of local methane emissions

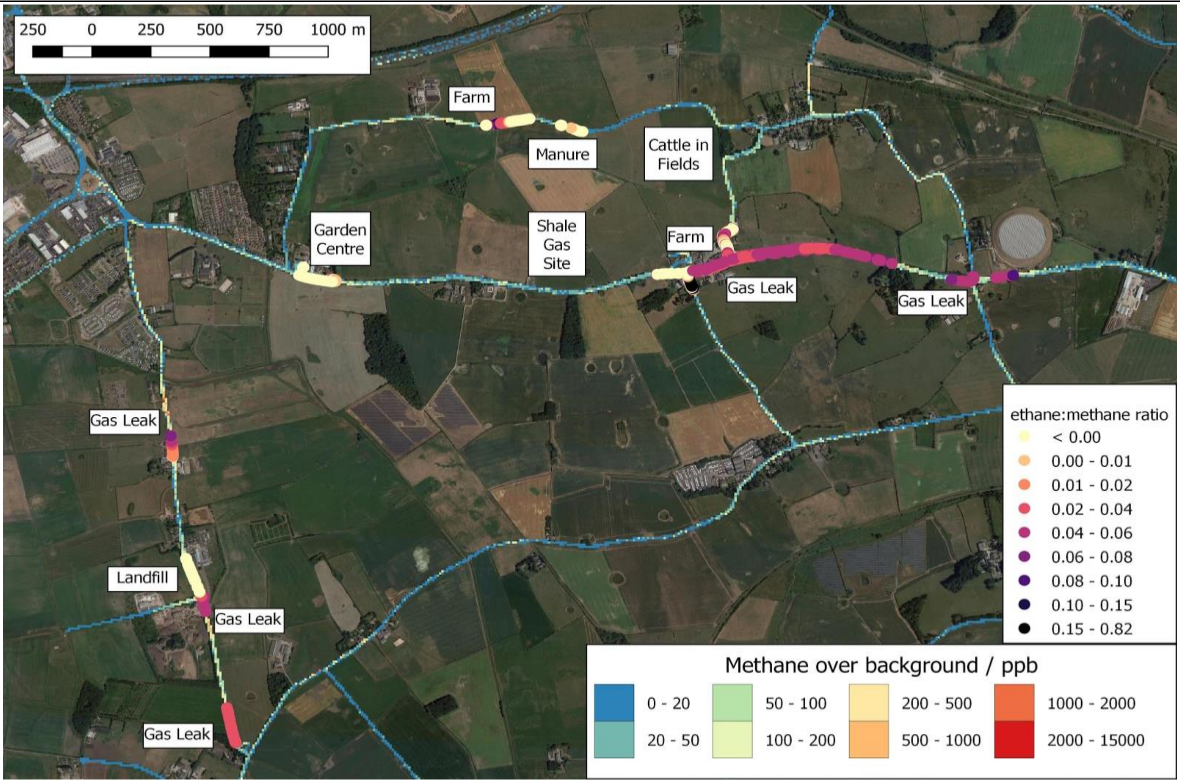
Cavity enhanced spectroscopic analysers measuring methane and ethane at 1 Hz were installed in a car with an air inlet on the roof, with a linked GPS system to record location. Air samples were collected in Flexfoil bags for laboratory analysis of methane $\delta^{13}C$ by high precision isotope ratio mass spectrometry. The figure gives an example of mean methane concentration elevations for a site in the NW of England prior to shale gas extraction. Measurements from 18 surveys carried out over 2 years are averaged here. Isotopic signatures of the sources are calculated using Miller-Tans analysis (Miller and Tans, 2003). Methane emissions from cattle, gas leaks and landfills can thus be distinguished isotopically or with ethane:methane ratios as shown below in the two figures.

The figures show an area around the Preston New Road shale gas site mapped using the Royal Holloway mobile laboratory for mixing ratios of CH_4 and/or and C_2H_6.

Top Figure: at points of known elevated methane along the roads, as indicated by higher methane over background values, multiple spot samples were taken for isotopic analysis over multiple surveys. The isotopic signature for $\delta^{13}CH_4$ is calculated through Miller-Tans analysis and exact result shown alongside each marker. Darker colours for isotopes represent more thermogenic sources, and lighter colours more biogenic sources.

Bottom figure: points where CH_4 is elevated by 200ppb (above background) for more than 10 continuous measurements have their C_2H_6:CH_4 ratio calculated. Darker colours for C_2H_6:CH_4 represent more thermogenic sources, and lighter colours more biogenic sources. ©RHUL, 2020.





Mobile surveys of methane using cavity enhanced spectroscopic analysers (e.g. Zazzeri et al., 2015) are recommended to identify the locations of pre-existing emissions in the area (during baseline characterisation). These are ideally carried out under different wind conditions and during different seasons. Identified emission plumes can be characterised by ethane:methane ratios or carbon isotopic composition ($\delta^{13}C$) for improved source attribution in areas where there are multiple co-located methane sources. Typically methane from biogenic emissions (e.g. ruminants, waste) is depleted in ^{13}C and has no co-emitted ethane whereas thermogenic emissions (e.g. UK gas leaks) are relatively enriched in ^{13}C and have co-emitted ethane and consequently higher ethane:methane ratios. An example of this is shown in case study - Characterisation of local methane emissions. Appropriate instrumentation and calibration of the methane analyser should be made using the recommendations in Table 4 and Table 5.

5.5 CHANGE DETECTION

To evaluate the impact of a new activity (such as unconventional shale gas extraction) on the environment, it is important to characterise that environment before, during and after the target activity. Change detection is defined as any statistically-significant change in environmental parameters that occurs after the conclusion of the baseline period (i.e. when operational, or pre-operational activity commences), relative to the environment characterised during the baseline period. Such changes may need to be assessed on short timescales (e.g. transient emissions such as venting and flaring) and long timescales (e.g. on-site generators and chronic fugitive emissions). Care must be taken before positively associating any detected change with the targeted activity as there could be extraneous sources that may need to be characterised and discounted.

Change detection can manifest itself in short-term events, such as through major leaks or intentionally vented pollutants, which could result in measurements that exceed typical baseline conditions over a short period of time (hours or days). Alternatively, change detection can manifest itself in long-term monitoring, through incremental changes in the average, or typical range, of baseline conditions (weeks, months, and seasons). It is therefore essential that supporting information (meta data) on site activities is collated to enable effective interpretation of any changes. As discussed in previous sections, long-term monitoring of the baseline environment yields statistics for each environmental parameter that need to be evaluated to determine the expected range across different time periods, be that time-of-day, day-of-week, month-of-year, or season (e.g. Shaw et al., 2019).

The exact thresholds at which the typical range is defined for a particular parameter must be interpreted based upon the measurement context. Using CH_4 as an example, the maximum recorded CH_4 mixing ratio, over a sufficiently long period, may not be representative of typical values, but may instead be due to a transitory event such as the close proximity of a ruminant animal. Similar momentary events could feasibly occur for many other atmospheric constituents; a tractor passing by the instrument inlet would have an unanticipated impact on the measurements of NO_x or PM that would not be truly representative of the wider environmental background. In a similar manner, the minimum measured value may also be a poor representation of the true lowest typical range in the environment. Rare meteorological events which draw pristine air from the Arctic free troposphere may result in the measurement of minimum values that are outside the typical range of values. Using, for example, the 1st and 99th percentile values from the baseline measurements may therefore provide a more suitable representation of typical local upper and lower environmental limits, thereby providing an expected range within which the majority of measurements should fall. Measured values that fall outside of this range during operational monitoring may warrant investigation as to their cause, as they may indicate the detection of an extreme event. Establishing a catalogue of potential and actual emission per-existing sources and their characteristics is important for developing and refining the conceptual model, informing the operation of the monitoring and interpretation of data.

Once a typical range has been quantified from the baseline dataset, a set of threshold criteria can be developed to yield an algorithm for quick and easy short-term change detection. This algorithm can combine multiple threshold parameters to aid in source identification. An example of an algorithm, for the identification of non-combusted methane emissions from a hydraulic fracturing site, is given in Figure 22 (Shaw et al., 2019).

The algorithm in Figure 22 sets a precedent for calculating baseline thresholds from the baseline dataset, as well as using those values to identify changes that are likely to be associated with hydraulic fracturing and associated operations. It uses a number of parameters to do so:

1. Wind direction: Data are limited to include only those where the air sampled had passed over the shale gas extraction facility. In the case of Preston New Road and Kirby Misperton, where the monitoring stations are to the east of the facility, the data was limited to westerly winds i.e. those between 225^{o} and 315^{o}.
2. CH_4 mixing ratio: This provides a cursory assessment of CH_4 mixing ratio, relative to baseline conditions. If the measurement exceeds that 99th percentile value recorded during the baseline, it is flagged as anomalous.
3. CH_4 mixing ratio and wind speed: This parameter takes into account the measured wind speed. Periods with low wind speed (below 2 m s^{-1}) may be associated with the accumulation of pollutants as the air mass as the air mass stagnates. Incorporating wind speed into the algorithm removes periods of stagnation, where high CH_4 mixing ratios may be observed regardless of outside influence. If the measurement exceeds that 99th percentile value recorded during the baseline, it is flagged as anomalous.
4. CH_4:CO_2 ratio: This ratio can provide an indication of the age of the sampled air mass. Higher ratios would indicate a more local source of non-combusted CH_4, which has not been oxidised to CO_2. If the measurement exceeds that 99th percentile value recorded during the baseline, it is flagged as anomalous.

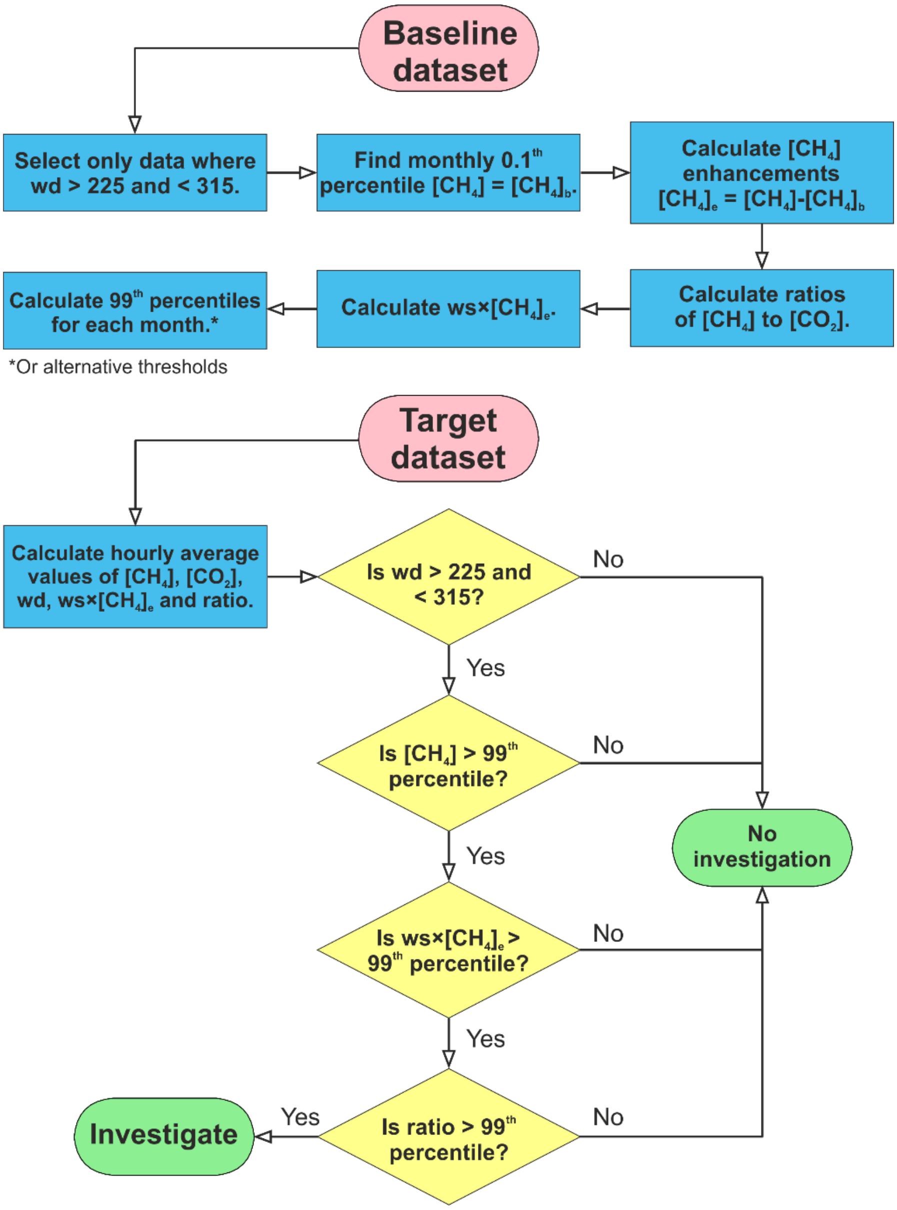


Figure 22. Algorithm for change detection. Key to abbreviations: $[CH_4]_b$ = 0.1th percentile $[CH_4]$, $[CH_4]_e$ = $[CH_4]$ enhancement = $[CH_4]$ – $[CH_4]_b$, wd = wind direction, ws = wind speed. Wind directions between 225° and 315° incorporate all wind directions that can be considered to be westerly winds (i.e. 270° ± 45°)

In theory, the combination of these four parameters should aid in positive identification of CH_4 venting. Any single parameter on its own may be expected to flag 1% of "typical" data, whereas combining multiple 99th percentile thresholds should reduce the number of false positives. When applied to a two-year baseline period (1st February 2016 to 31st January 2018) for the two shale gas sites in England (Preston New Road (PNR) and Kirby Misperton (KM)), nine one-hour periods at PNR and seven one-hour periods at KM were flagged by this algorithm. This corresponds to approximately 0.05% of the data, or 10 in every 17,500 hours. The flagged periods at PNR were generally associated with extremely low wind speeds (<1 ms), or with rapidly changing meteorological conditions. The flagged periods at KM were confirmed (by the operators) to be associated with emissions from the conventional well-head located nearby to the monitoring station. This in itself, validates this form of algorithm, for the detection of cold vented CH_4.

This algorithm was also applied to operational data recorded at PNR after exploratory hydraulic fracturing operations commenced in October 2018. Periods in which CH_4 mixing ratios exceeded the baseline thresholds were flagged by the algorithm, as shown for a series of emissions events in December and January 2019 in Figure 23. The periods highlighted in red in Figure 23 were confirmed by the operator to coincide with nitrogen lift operations undertaken by Cuadrilla (Allen et al., 2019). These operations resulted in the emission of non-combusted CH_4 into the atmosphere that was measured by the monitoring station due to the favourable meteorological conditions at the time, i.e. wind blowing from the site to the monitoring station.

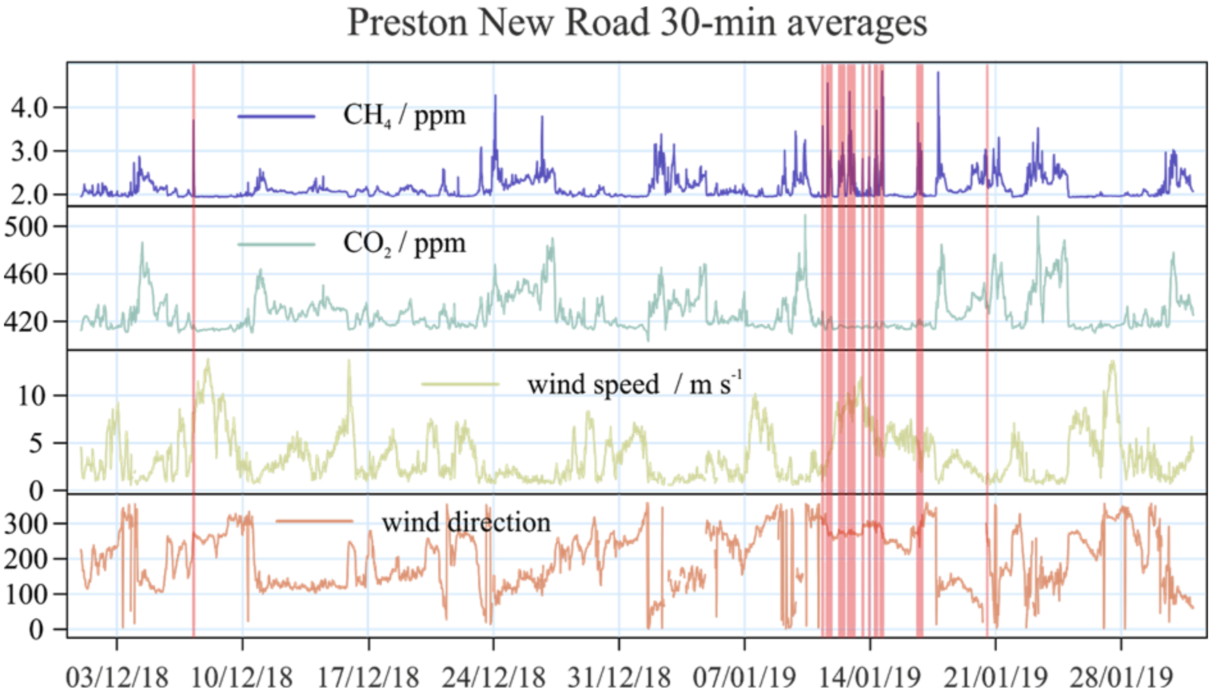


Figure 23. 30-minute averaged CH_4 and CO_2 mixing ratios, wind speeds, and wind direction at the Preston New Road monitoring station for the period 1st December 2018 to 31st January 2019. The red highlighted areas represent hourly periods which exceeded the threshold criteria for the identification of excursions from the baseline conditions. ©University of Manchester, 2020

It should be noted that, as the algorithm can only be applied to data under westerly wind conditions, the emission event recorded in January 2019 would have been missed had the wind direction been from the north, east or south. This is an obvious short-coming of operating a single monitoring station; emissions due to operational activity will only be measured during a limited set of meteorological conditions.

The application of the algorithm in Figure 22 can aid in change detection for short-term events, such as major leaks or intentional venting. More long-term change detection (e.g. from a small leak, or incremental activity) requires continuous monitoring and the assessment of long-term data. In this case, statistical averages (mean, median, percentile values) measured during operational activity would need to be compared against the statistical averages recorded during the baseline. This would be performed over different time-periods (hour-of-day, day-of-week, month, season etc.) to identify any small, incremental change in the environment measured after the conclusion of the baseline period. Care would have to be taken during interpretation of this data, as any change detected may not be solely due to operational activity (e.g. if another industry had moved to the area or emission form an existing activity changed).

In summary, change detection takes two forms. The first representing short-term but large excursions in an environmental parameter due to a major leak or intentional venting – this is detected by testing operational data against an algorithm to evaluate threshold exceedances. The second is more subtle, and requires long-term monitoring at the site b to detect trends in the environmental climatology, relative to that measured during the baseline, over time.

5.6 REFERENCES

Allen, G., Rebalancing the global methane budget, Nature, 2016.

Allen et al., Methane enhancements detected at Little Plumpton air monitoring site, *British Geological Survey*, http://www.bgs.ac.uk//downloads/start.cfm?id=3510, 2019

Edwards et al, High winter ozone pollution from carbonyl photolysis in an oil and gas basin, Nature, 2014

Helmig et al, Reversal of global atmospheric ethane and propane trends largely due to US oil and natural gas production, Nature Geoscience Letter, 2016

Lewis et al, Evaluating the performance of low cost chemical sensors for air pollution research, Faraday Discussions, 2016.

MacKay and Stone, Potential Greenhouse Gas Emissions Associated with Shale Gas Extraction and Use, DECC Report, 2013.

Miller, J. B. and Tans, P. P. Calculating isotopic fractionation from atmospheric measurements at various scales, Tellus, 55, 207– 214, 2003.

Purvis et al, Effects of 'pre-fracking' operations on ambient air quality at a shale gas exploration site in rural North Yorkshire, England, Science of the Total Environment, 2019

Shaw et al, A baseline of atmospheric greenhouse gases around prospective UK shale gas sites. Sci. Tot. Env. 684, 1-13. https://doi.org/10.1016/j.scitotenv.2019.05.266. 2019

Ward, R. S., P. L. Smedley, G. Allen, B. J. Baptie, M. R. Cave, Z. Daraktchieva, R. Fisher, D. Hawthorn, D. G. Jones, A. C. Lewis, D. Lowry, R. Luckett, B. P. Marchant, R. M. Purvis & S. Wilde. 2018. *Environmental baseline monitoring : Phase III final report (2017-2018)*. In *British Geological Survey Open Report*.

Ward, R. S., P. L. Smedley, G. Allen, B. J. Baptie, Z. Daraktchieva, A. Horleston, D. G. Jones, C. J. Jordan, A. Lewis, D. Lowry, R. M. Purvis & M. O. Rivett. 2017. *Environmental Baseline Monitoring Project. Phase II, final report*. In *British Geological Survey, 163pp. (OR/17/049)*.

Zazzeri, G., D. Lowry, R. Fisher, J. France, M. Lanoisellé and E.G. Nisbet (2015). Plume mapping and isotopic characterisation of anthropogenic methane sources. Atmospheric Environment. 110. 10.1016/j.atmosenv.2015.03.029.

6. Radon monitoring

6.1 INTRODUCTION

Radon is the largest source of radiation exposure for most of the UK population (AGIR, 2009b) and is the second highest cause of lung cancer after smoking (Darby et al, 2005). Since radon is a gas, it has much greater mobility than other radionuclides in the uranium radioactive decay chain. Radon and its radioactive decay products are present in indoor and outdoor air throughout the UK.

The release of radon from rocks and soils is determined largely by the types of minerals in which uranium occurs. Radon can more easily transit through porous rocks and soils and can escape into fractures and openings. It is also soluble in water and so can be transported with groundwater. In most situations, human exposure to radon arises from its release from geological material in the upper few metres of the Earth's surface. Radon migration to the surface is controlled by the transmission characteristics of rocks and soils and the nature of carrier fluids, including groundwater. Prolonged radiation exposure from the inhalation of radon decay products results in an increase in lung cancer risk especially in smokers and ex-smokers.

In 2014 Public Health England (PHE) published a report (PHE-CRCE-009) on the potential health impacts of shale gas in the UK. The report identified that radon is likely to be present in shale gas and released to the environment as a result of its exploitation. A number of exposure pathways were identified as leading to potential, but limited radiation exposure. These pathways included the de-gassing of radon from drilling returns, flow-back fluids, produced water, contaminated groundwater and from the natural gas stream. The aim of baseline radon monitoring associated with shale gas activities is to establish a database of measurements of indoor and outdoor radon levels and their statistical distribution, prior to the start of shale gas operations, which can then be compared with the equivalent distribution measured once relevant industrial activities have started.

A programme of baseline radon monitoring should be focused on determining the long-term average concentration of radon in outdoor and indoor air, rather than the presence or size of short term fluctuations. Indoor radon concentrations exhibit diurnal, monthly and seasonal variation (Miles and Algar, 1988). Short-term variations in local concentrations occur because of changes in local factors, such as weather and seasonal conditions, building occupancy, ventilation and heating cycles. However, these short term variations are generally not significant in terms of potential radiation exposure since it is the exposure integrated over many years that results in cumulative radiation dose and hence risk. The programme includes a limited element of time dependent radon measurements that potentially offer some additional insight into shorter timescale variations.

Since background concentrations are partially dependent on changing conditions e.g. weather, there is variation throughout the year that can be used to predict the annual average for indoor testing (Daraktchieva, 2017). There is also a year on year variation. Consequently, the observed distributions will not be identical, but the monitoring programme must be sufficient to identify whether the shale gas activities can be associated with any significant change in the local background distribution. Radon measurements in outdoor air and in homes were recommended, in order to assess the baseline and provide evidence on radon distributions before shale gas extraction commenced. It is therefore recommended that both indoor and outdoor radon monitoring is carried out. Each has different measurement challenges that must be addressed. An overview of the recommended approach is shown in Figure 24.

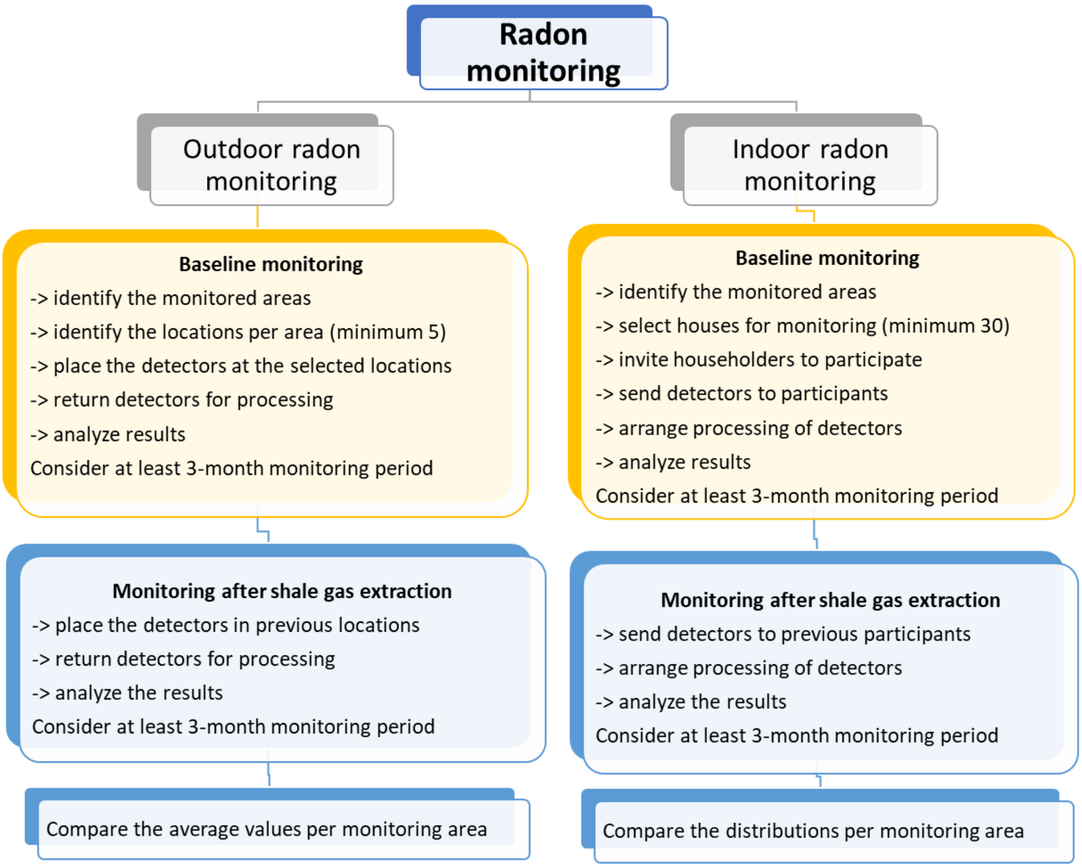


Figure 24. Overview of baseline radon monitoring. ©PHE, 2020

6.2 SITE SELECTION AND LOGISTICS

The area around the proposed shale gas extraction site should be assessed for existing information on indoor and outdoor radon levels. Baseline monitoring of radon levels should include inspections of local radon levels in the area under exploration as well as in the surrounding area. The likely probability of radon in homes being in excess of the Action Level should be assessed using the UKradon website[5]. That online resource uses the existing radon risk map created by PHE and BGS from over 400,000 indoor radon measurements, made over many years, and the BGS digital geological dataset. While that provides an appropriate basis for risk assessment in UK homes and workplaces, it does not provide the level of detailed site-specific information that is required for the present project.

An area in close proximity to the shale gas site should be selected for monitoring prior to operations starting and also a control area which should be situated at a reasonable distance (ca 10 km) from the first area and away from any potential influence from the operational site. Both areas should have similar radon potential and ideally similar geological and environmental characteristics. Radon Affected Areas are areas where at least 1% of the homes are expected to have radon levels at or above the UK Action Level of 200 Bq/m^3. If the extraction site is close to a radon Affected Area one or more nearby areas with elevated radon potential should additionally be included in the baseline monitoring programme to assess any difference in the measurement range and thus highlight existing elevated radon levels that are not associated with the shale gas activities.

Representative radon monitoring should be achieved by a good spatial coverage of the sampling area. Sampling areas should be determined to include the shale-gas extraction site and control sites. For outdoor radon monitoring enough sampling points should be installed to provide good

[5] http://www.ukradon.org

coverage of the monitored area, out to a suitable distance (no more than a few km) from the extraction site and control. For indoor radon monitoring sampling points should be determined to give statistically significant results for each area. Typically, this will require results from at least 30 homes.

6.2.1 Outdoor radon

Since a concern about radon gas relates to possible fugitive emissions from operational shale gas sites, it is appropriate to place sets of detectors at locations around the site(s) of interest with locations at approximately equal radial distances from the site and distributed at approximately regular angular intervals. To represent exposure that the public might receive, and to allow for placement in a suitable number of locations, these detectors should be placed within 1–2 km of the site. The number of detectors in each area should be sufficient to allow statistical analysis. As a minimum, there should be at least five (5) detectors in each area.

Outdoor radon monitoring locations should be in the open air but discreet, to avoid vandalism, interference and loss (particularly in urban areas), reasonably accessible to support safe and efficient location and retrieval, and at approximately 1.5 m height above the local ground that (i.e. corresponding to the point above the ground at which potential exposure occurs).

Some radon monitors are sensitive to the outdoor environment, especially passive etch track detectors that are used extensively in the UK (Wasikiewicz, 2017). Passive etched track detectors are passive detectors that are using plastic as the detector material and can been used for a long term exposure. There is therefore a need to ensure that detectors are unaffected by, or suitably protected from, the effects of rain, strong sunlight and extremes of temperature. Detectors should therefore be packaged in waterproof, light-tight, radon-gas and air permeable containers.

Radon detectors are themselves harmless and would not present a risk to anyone who encounters them. However, where special types or configurations of detectors are used and placed in the outdoor environment where members of the public may encounter them by accident, it would be important to ensure that they do not present a health and safety risk and are suitably labelled to indicate their purpose.

6.2.2 Indoor radon

Indoor radon concentrations display diurnal, monthly and seasonal variations (Miles and Algar, 1988). Radon ingress into buildings is caused by a small pressure difference between the inside of the house and outside which is caused by several factors including soil permeability, wind direction and temperature difference between indoors and outdoors (Nazaroff and Nero, 1988). Indoor radon levels vary by more than a factor of 1,000, with the lowest being similar to outdoor levels. Geology is the single largest source of variation but not the only one. Therefore long-term measurement of the average radon concentration is the most suitable technique for identifying radon levels above the UK Action Level (200 Bq/m^3). Placement of passive radon detectors at homes should follow standard protocol for deployment and collection of detectors (Daraktchieva et al. 2018).

The aims of undertaking a local baseline of indoor radon levels are twofold: firstly, to provide one or more local distributions that can be compared with equivalent results made when shale-gas activities have started; secondly, where the locality has areas of elevated radon potential, to demonstrate clearly, before shale-gas extraction occurs, that this is the case in some local homes.

The selection of areas for monitoring is the same for outdoor radon with a minimum number of 30 households per area. A recommended approach for selecting homes in an area is to use random sampling of addresses from a validated source such as the Royal Mail Postal Address File or similar product that supports filtering of non-domestic addresses. A significant degree of over-sampling is likely to be required since surveys of this type tend to generate only a minority uptake. It is generally not appropriate to recruit candidate addresses by asking for volunteers since this potentially adds unintended bias, such as people outside the targeted area, social bias, or multiple requests from the same address. In less-densely-populated areas, it may be necessary to issue invites to all of the dwellings in an area in order to secure sufficient results to yield a useful statistical distribution.

6.3 MEASUREMENTS

The recommended approach for both indoor and outdoor radon measurement is to use passive radon detectors. The instruments used should meet the criteria outline in the PHE Validation Scheme (Daraktchieva et al., 2018).

6.3.1 Outdoor radon

Outdoor radon levels vary according to a number of parameters including local geology, geography and weather but are generally very low across the UK, below 10 Bq/m^3 (Wrixon et al., 1988). This is close to the limit of detection for most practical radon measurement systems with most passive detectors having a limit of detection of around 5 Bq/m^3. At these low levels, individual measurements have large uncertainties which can be reduced by taking multiple samples (detectors) at each location and, subject to maintaining measurement quality, using extended monitoring periods. As a minimum, to provide adequate experimental statistics (signal to background ratio), monitoring should be for at least three months. However, 12 months is recommended to provide better statistics and a more representative annual average concentration.

6.3.2 Indoor radon

As with outdoor radon measurement the recommendation is to use passive detectors. Indoor radon concentrations exhibit diurnal, monthly and seasonal variation (Miles and Algar, 1988), therefore at least 12 months of testing is strongly recommended with 3 months as an absolute minimum. If the locality includes areas of elevated radon potential, the indoor measurement programme may identify some homes that have radon at levels where remedial action is recommended. This is achieved by comparing suitable indoor measurements, seasonally corrected where appropriate, with the radon Action Level which is expressed as an annual average concentration. Indoor measurements should follow the UK validation scheme (Daraktchieva et al., 2018).

6.4 DATA ANALYSIS AND REPORTING

Passive detectors used for radon monitoring should be analysed by an accredited laboratory after their collection or return. They should be processed as soon as possible to avoid additional exposure from indoor radon.

Standard calibration procedures and quality controls should be used including regular blind tests, compliance with the PHE Validation scheme (Daraktchieva et al., 2018) and inter-comparisons of passive radon detector performance (Howarth, 2014).

Data should be analysed taking into account the calibration parameters, reference radon sources and instrumentation, the length of the measurement and measurement uncertainty. For indoor radon monitoring additional uncertainties should be taken into account regarding the seasonal correction factors, occupancy factors and reported duration of the measurements.

Indoor radon concentrations are generally log-normally distributed (Gunby et al., 1993 and Daraktchieva et al., 2014). Statistical analysis of baseline data should therefore determine parameters of local radon distributions taking into account the log-normality of radon data.

There is both statistical and measurement uncertainty (uncertainty of reference instruments) in radon results (ICRU Report 88, 2012). Statistical uncertainty can be reduced by increasing of the size of the sample and the sampling time. Measurement uncertainty should be estimated for each instrument or monitoring device according to its specification and taken into account. Data should be assessed and evaluated regularly to avoid errors and misinterpretations.

Results can be reported using different methods such as reports and graphs. While the health protection aspect of existing radon in homes is not the primary aim of a baseline monitoring programme, it is appropriate to communicate results for individual homes to the householder, including comparison with the Action Level and identifying whether action to reduce radon levels is recommended. Since the indoor measurements relate to private dwellings, it is not appropriate to identify publicly the radon level in specific homes. Ranges and statistical approaches are appropriate and should be chosen to preclude identification of individual homes. Since radon is a known lung carcinogen, it would also be appropriate to provide access to further information sources and support for those householders whose results are of concern to them. Extensive information is provided by national authority on radon - Public Health England at www.ukradon.org.

Indoor radon levels can be strongly affected by changes in the use, occupancy and fabric of the home and material changes to the property. It is therefore appropriate to ask householders to complete a relevant questionnaire when they participate in baseline indoor radon monitoring programmes. If the aim is to measure the same homes in two phases (i.e. before and during shale-gas activities), it is important to understand if significant actions have been undertaken that might have changed indoor radon levels. In cases where homes with high radon levels were identified, this might include remediation action to reduce radon levels. It may be necessary to exclude some homes where such changes are known to have been made.

6.5 CASE STUDY

Radon concentrations have been measured in homes in two locations within the Vale of Pickering, North Yorkshire to characterise baseline conditions. Detectors were placed in volunteer households for a period of three months and then analysed by PHE's laboratory. This was repeated so that 12 months of data were collected, i.e. 4 x 3 months.

The locations were chosen because one is within a Radon Affected Area and the other is an area that is not radon affected (Figure 25). A Radon Affected Area (RAA) is where domestic properties are expected to have at least a 1% probability of exceeding the Radon Action Level (200 Bq m^{-3} annual average). RAAs are identified to support those who have to make decisions about testing properties for radon and to support radon prevention requirements in building regulations. RAAs are identified jointly by PHE and BGS through the use of PHE indoor domestic radon measurements and BGS digital geological data. This reflects the evidence that local geology is a significant but not the only determinant of the indoor radon level in a building. The current map that identifies RAAs in England and Wales was derived using over 400,000 radon measurements.

Results from the four 3-month back-to-back tests in homes are presented in Table 6. The annual average radon concentrations were calculated employing seasonal correction factors as outlined in PHE Validation scheme (Daraktchieva et al, 2018). Distribution parameters assuming log-normality show that homes in Kirby Misperton and Little Barugh are situated in areas with low radon potential while Pickering is situated in an area with elevated radon potential. Local radon distributions for the four 3-month tests in homes in Kirby Misperton/Little Barugh, and in Pickering are compared in Figure 26.

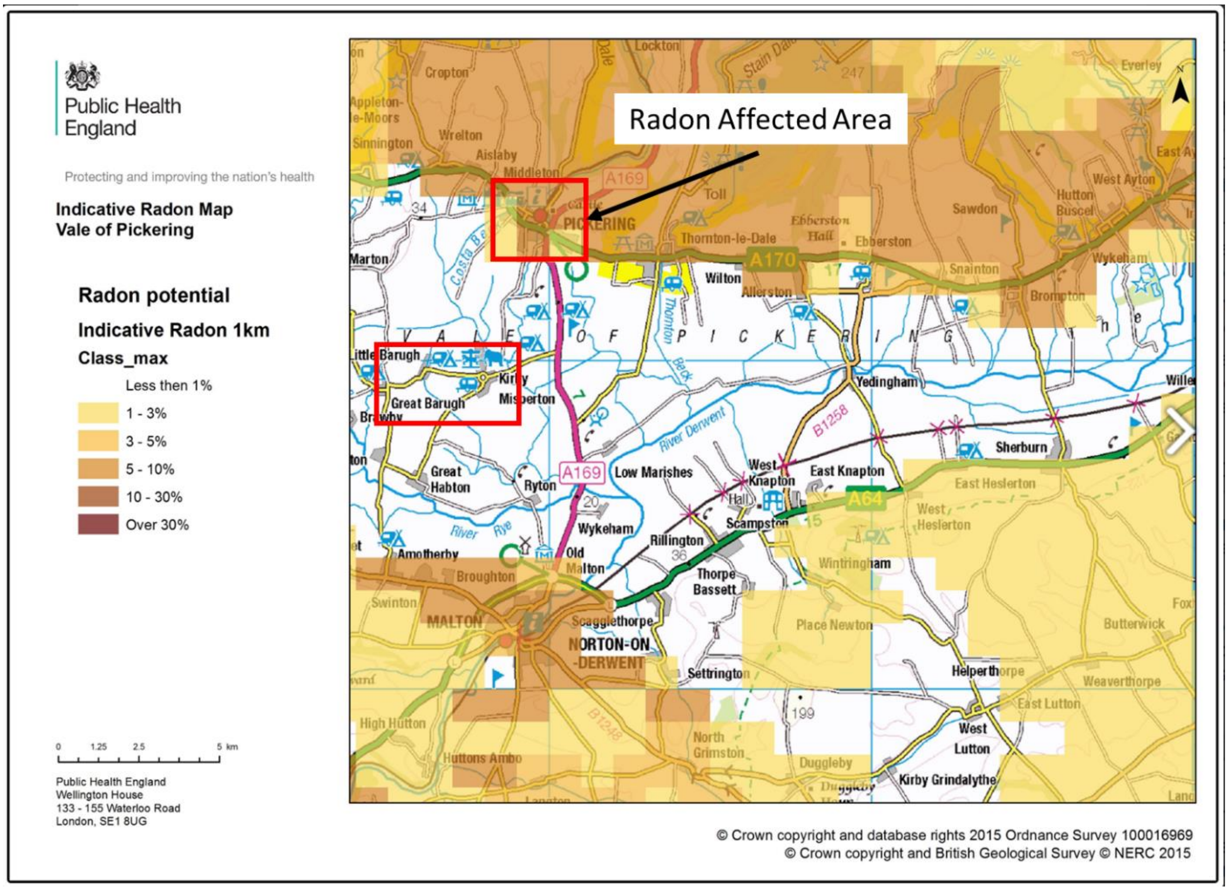


Figure 25. Indicative radon map for the Vale of Pickering, North Yorkshire. The red boxes indicate the areas in which radon detectors were located. © PHE, 2020

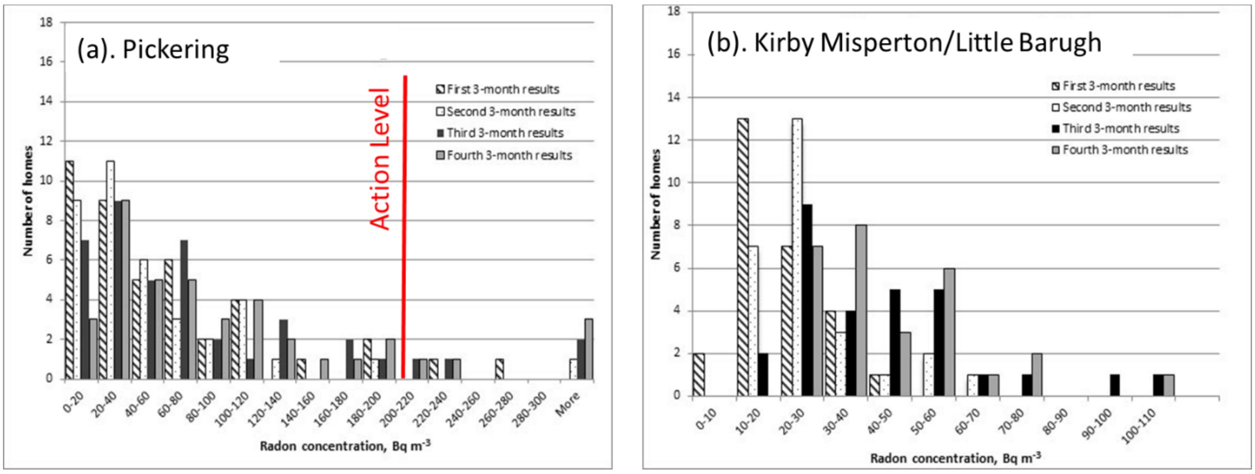


Figure 26. Indoor radon concentrations over a 12 month period in (a). Pickering (Radon Affected Area) and (b) Kirby Misperton/Little Barugh (not radon affected). ©PHE, 2020

Table 6. Range and distribution of indoor radon measurements.

Area (number of homes)	First 3-month results (Dec 15-March 16), Bq m^{-3}			Second 3-month results (Apr –June16), Bq m^{-3}			Third 3-month results (July-Sep 16), Bq m^{-3}			Fourth 3-month results (Sep-Dec 16), Bq m^{-3}		
	Range	GM	GSD	Range	GM	GSD	Range	GM	GSD	Range	GM	GSD
Kirby Misperton and Little Barugh (27/27/29/28)	9 - 40	18	1.5	13 – 70	25	1.5	16 - 110	37	1.6	20-100	41	1.5
Pickering (42/38/41/40)	6 - 270	40	2.7	9 – 450	44	2.6	13-460	56	2.6	17-620	71	2.5

Concentrations in the Pickering area range from 0 - 600 Bq m^{-3} and in Kirby Misperton/Little Barugh area from 0 – 100 Bq m^{-3}. The observed differences reflect the naturally elevated radon potential in the Pickering area compared to the centre of the vale which is not radon affected. The UK Action Level is 200 Bq m^{-3} and, based on the studies carried out by PHE, a proportion of homes in radon affected areas (as indicated in Figure 25) would be expected to exceed this level.

The case study hence demonstrates the need for undertaking baseline monitoring for radon ahead of any shale gas development (Daraktchieva et al., 2017). It is important that sufficient data are generated in the baseline monitoring of radon-affected and not radon-affected areas in the locality to establish the variability of radon concentrations against which any future changes may be evaluated.

6.6 CHANGE DETECTION

Identifying a potential change in outdoor or indoor radon levels following the commencement of shale gas operations is not a simple process. Radon concentrations vary over short and long timescales, including year-by-year variations. Within any local area, indoor radon levels generally follow a log-normal distribution. Within any property the radon concentration may be altered by changes in how the property is used, ventilated, heated, occupied, etc – its indoor living environment. The methodology to be adopted for identifying changes in radon levels after the start of shale gas operations should consider the following:

- The comparison will need to recognise that radon levels vary from year to year. This has been observed for indoor and outdoor radon as part of the baseline monitoring programme. It may be appropriate to use the locally observed year-by-year variations, together with other evidence from the literature, to identify minimum levels of variation that are likely to be observable.
- Comparison for the purposes of identifying change may benefit from looking at changes in the statistical distribution of local radon concentrations in selected areas, if individual properties can be safely assumed to be largely unchanged over the period in terms of structure and “indoor living environment”.
- In seeking to identify changes in radon, it would be important to aim to measure the same locations and properties with the same techniques to minimise the number of variables to be considered.
- It may be necessary to exclude some properties or locations if there is clear evidence other events, not related to shale gas, have occurred that might be the cause of a change in radon concentration.
- The indoor and outdoor baseline radon monitoring programme has included control locations that are located at some distance from the proposed shale gas site but within the same part of the country. Results from these locations may provide useful evidence about the local consistency in year-to-year variations and potentially differences in changes between locations close to the site and difference at the control sites.
- For outdoor monitoring detectors need to be placed if possible in the same monitoring locations as during the baseline monitoring, and preferably for the same duration. The average radon levels per area should be calculated and compared with its baseline values using standard statistical tests. There is a year to year variation of outdoor radon which should be taken into account in the data analysis. An annual variation of outdoor radon levels was measured for the Vale of Pickering- the first and third year results were about 3 times higher than the second year results (Ward et al., 2018). Therefore, in order to consider the changes in outdoor radon levels during fracking as significant these should be much higher than the baseline levels, i.e. exceeding the maximum observed at any time during the baseline period.

- For the indoor radon monitoring the same homes selected for the baseline phase need to be monitored during the fracking phase. The sample of homes per area needs to be sufficient to ensure a robust statistical analysis. Radon distributions before and after fracking in each area need to be compared using appropriate statistical tests. Any significant changes in indoor radon levels attributed to fracking have to take into account the well-known year to year variation of radon concentrations of up to 50 % (Darby et al., 1988 and Hunter et al., 2005).

6.7 REFERENCES

AGIR (2009b), Radon and public health. Report of the independent Advisory Group on Ionising Radiation. Chilton, Doc. HPA, RCE-11,1-240.

Darby, S. C., Whitley, E., Silcocks, P., Thakrar, B., Green, B. M. R., Lomas, P., Miles, J. C. H., Reeves, G., Fearn, T., Doll, R. Risk of lung cancer associated with residential radon exposure in south-west England: a case–control study. British Journal of Cancer 78, 394–408 (1988).

Darby S, Hill D, Auvinen A, Barros-Dios J M, Baysson H, Bochicchio F. et al. (2005) Radon in homes and risk of lung cancer: collaborative analysis of individual data from 13 European case-control studies. BMJ, 330 :223.

Daraktchieva Z, Miles J C H and McColl N (2014) Radon, the lognormal distribution and deviation from it. J. Radiol. Prot. 34, 1–8.

Daraktchieva Z (2017). New correction factors based on seasonal variability of outdoor temperature for estimating annual radon concentrations in UK, Rad. Prot. Dosimetry 175-1: (65-74).

Daraktchieva Z, Wasikiewicz J M, Howarth C B and Bradley E J (2017) Radon baseline monitoring around a potential shale gas development site in Yorkhire, England. Radiat Prot Dosimetry 177(1-2), 140-143.

Daraktchieva Z, Howarth C B, Gooding T D, Bradley E J and Hutt N (2018) Validation Scheme for Organisations Making Measurements of Radon in UK Buildings: 2018 Revision, PHE-CRCE-040

Gunby J A, Darby S C, Miles J H C, Green B M R, Cox D R (1993) Factors affecting indoor radon concentrations in the United Kingdom. Health Physics 64 (1), 2-12

ICRU Report 88 (2012). Measurement and Reporting of Radon Exposures, Journal of the ICRU Vol 12 No 2 , Oxford University Press 2015.

Howarth C B (2018) Results of the 2016 PHE Intercomparison of Passive Radon Detectors, PHE-CRCE-042.

Hunter, N., Howarth, C. B., Miles, J. C. H., Muirhead, C. R. Year-to-year variations in radon levels in a sample of UK houses with the same occupants. Seventh International Symposium on the Natural Radiation Environment(NRE-VII). In: Radioactivity in the Environment Book Series, vol. 7. Elsevier, 438-447 (2005).

Kibble A, Cabianca T, Daraktchieva Z, Gooding T, Smithard J, Kowalczyk G, McColl N P, Singh M, Mitchem L, Lamb P, Vardoulakis S and Kamanyire R (2014) Review of the Potential Public Health Impacts of Exposures to Chemical and Radioactive Pollutants as a Result of the Shale Gas Extraction Process, Chilton, PHE-CRCE-009

Miles J C H and Algar R A (1988) Variations in radon-222 concentrations. Journal of Radiological Protection 8 (2), 103-106.

Nazaroff, W. W. and Nero (1988)., A. V. Radon and its decay products in indoor air. John Wiley and Sons Inc., New York, NY.

R.S. Ward, P.L. Smedley, G. Allen, B.J. Baptie, M.R. Cave, Z. Daraktchieva, R. Fisher, D. Hawthorn, D.G. Jones, A.C. Lewis, D. Lowry, R. Luckett, B.P. Marchant, R.M. Purvis, S. Wilde, Environmental baseline monitoring: phase III final report (2017–2018), British Geological Survey Open Report (2018) WA Google Scholar.

Wasikiewicz J M (20017) Impact of Environmental Factors on PADC Radon Detector Sensitivity During Long Term Storage. Radiation Physics and Chemistry 142,

Wrixon, A. D., Green, B. M. R., Lomas, P. R., Miles, J. C. H., Cliff, K. D., Francis, E. A., Driscoll, C. M. H., James, A. C., O'Riordan, M. C. Natural radiation exposure in UK dwellings. London: Her Majesty's Stationery Office, Report NRPB-R190 (1988).

7. Soil Gas

7.1 INTRODUCTION

There is no specific regulatory requirement in the UK to monitor soil gas in relation to hydrocarbon exploration and exploitation. However, the unconventional gas sector in the UK is an emerging industry, which has yet to be fully established, and any fugitive emissions from related sub-surface activities that pass through the soil have the potential to affect air quality (Section 5), indoor radon (Rn) levels (Section 6) and near surface ecosystems. Regulations do apply to other activities, such as the geological storage of CO_2 as part of carbon capture and storage (CCS), and for landfill sites. The CCS regulations are set out in European Directives (European Union, 2009a, b) and require site operators to monitor for possible movement of gas out of the storage complex. There is also a stipulation that any emissions to atmosphere must be quantified for greenhouse gas accounting purposes. Landfill operators need to monitor methane (CH_4) emissions through the cap of a landfill as part of demonstrating compliance with the Landfill Directive and other legislation, and to quantify the total emissions from the areas measured (Environment Agency, 2010).

Guidance on ground gas monitoring is given in a number of national and international standards including BS8576 ‘Guidance on investigation for ground gas, permanent gases and VOCs’ (British Standards Institution, 2013); BS8485 ‘Code of practice for protective measures for methane and CO_2 ground gases in new buildings’ (British Standards Institution, 2015) and ISO/DIS 18400-204 ‘Soil quality sampling – guidance on sampling of soil gas’ (ISO, 2017). The UK guidance builds on a body of work carried out in the 1990s following the Abbeystead tunnel and Loscoe gas explosions (Appleton et al., 1995; Crowhurst and Manchester, 1993; Hooker and Bannon, 1993; O'Riordan and Milloy, 1995; Sizer et al., 1996; Wilson et al., 2007). More recent advice is also given in the Ground Gas Handbook (Wilson et al., 2009) and on the risks of hazardous gases when drilling near coal (The Coal Authority et al., 2012). A useful review of near surface gas methods is provided by Klusman (2011).

Although there is no specific current requirement for soil gas monitoring for shale gas development it has the potential to identify leakage, arising from operations in the subsurface, which finds a pathway to the surface. Possible pathways are wellbores, fractures faults and permeable geological material. In the case of wells, it needs to be borne in mind that, if leakage did occur, it may not be confined to a well casing failure, but could be via the surrounding annulus if the well is poorly sealed. Should gas escape from the casing, or the well annulus, it could follow a higher-permeability pathway, which might lead to it reaching the surface some distance from the well head (e.g. Allison, 2001). Thus, it might not be detected by wellhead monitoring.

Understanding the pre-existing ground gas conditions is essential to provide the baseline against which any change during/following shale gas operations can be measured. There are a growing number of examples where a good baseline dataset has been important for identifying leaks and/or resolving accusation of impact being caused by industry. For example, landowners near the Weyburn enhanced oil recovery and CO_2 storage site in Canada alleged that high CO_2 values in the soil gas on their property were the result of leakage from the site. However, it could be shown from baseline measurements that the gas concentrations were mostly within the range for that time of year and subsequent investigations (see case study) demonstrated typical seasonal variations and showed that the CO_2 was of shallow biogenic origin (Beaubien et al., 2013; Romanak et al., 2014; Sherk et al., 2011; Trium Inc. and Chemistry Matters, 2011).

Baseline measurements provide context on the natural composition of the soil gas and its variability, and identification of any pre-existing anthropogenic inputs. Certain types of natural soil, such as alluvium and peat may be associated, for example, with methane generation. Both CH_4 and CO_2 can be produced from landfill and sewage sludge. Mine workings, especially coal mines, can be associated with gas emissions including CH_4, CO_2, CO and N_2. Existing oil and gas facilities and gas pipelines are also potential sources of gas.

The main gases of concern for baseline soil gas monitoring in relation to shale gas development are CH_4 and CO_2, along with volatile organic compounds and other trace components. Methane is potentially harmful because it is a flammable gas that can form explosive mixtures, or contribute to photochemical air pollution. CO_2 is potentially toxic to humans and animals. Both CH_4 and CO_2 are significant greenhouse gases and can act as asphyxiants in confined spaces. In the soil environment, CH_4 is microbially oxidised to CO_2, which may lower the risks associated with CH_4 emission (Topp and Pattey, 1997). Emissions of CH_4 or CO_2 from the sub-surface are not likely to be of direct concern for human health through combustion/explosion or toxicity/asphyxiation except in very specific cases related, most likely, to leaks from infrastructure. When there is very little air movement, leaking gases can accumulate in confined spaces or near the ground surface where CO_2, being heavier than air, has a propensity to collect in depressions or excavations. Other gases might accompany CH_4 and CO_2, but are likely to be at lower concentrations, for example other light hydrocarbons, such as ethane and propane, H_2S, N_2 or Rn.

To ensure representative data, the monitoring strategy needs to consider both the spatial and temporal variability of the soil gas. Monitoring can therefore be divided into survey and continuous modes of operation. Surveys provide spatial coverage and, through repetition, address temporal (mostly seasonal) changes. Continuous monitoring largely addresses the temporal variability, typically at a specific location, although some instruments can also provide a degree of spatial coverage. There are inevitably trade-offs between surveys, which can provide spatial coverage in a narrow time window, and continuous monitoring, which can only monitor a restricted area. A balance needs to be reached between the two based on an assessment of the leakage risks, with continuous monitoring at higher risk locations and surveys to cover the wider areas of lower risk where predicting the location of low probability events is difficult.

The specification of baseline monitoring, including the techniques used and the overall strategy, will have site-specific elements, within an overall framework that can be considered to be more generic.

7.2 SITE SELECTION/SURVEY AREA

To define the area to be monitored by both survey and continuous measurements, the pathways for potential surface emissions need to be considered for the site. They may be both geological, for example pre-existing faults and fractures, or permeable strata overlying the target shale formation, and those that are of an anthropogenic nature such as buried infrastructure including gas pipelines, boreholes and wells. Faults can be identified from existing geological maps and 3D models or those developed from exploration data acquired during shale gas projects e.g. 3D seismic surveys. Active faulting might also become apparent from baseline seismic monitoring (Section 3) or ground deformation studies (Section 8).

Site selection therefore needs to take account of the near-surface geology, both bedrock and superficial, and its modification, as well as the surface traces of any mapped faults. It also needs to consider the range of surface environments in terms of soil type (related to surface geology) and land use. Baseline planning should also account for any existing potential sources of ground gas, or near-ground gas emissions including landfills, current or former mine workings, especially coal mines, gas compressor stations and agricultural activity. Consideration should also be given to any sensitive receptors such as protected habitats or population centres bearing in mind local conditions of topography, prevailing wind directions etc.

The information outlined above needs to be incorporated into the site conceptual model alongside information relevant to other environmental monitoring.

As well as these spatial influences, baseline soil-gas monitoring needs to consider temporal variability. In particular the migration of soil gas is sensitive to the water contents of the soils and hence infiltrating water fronts that may impede the advection of soil gas and diffusion of (trace) contaminants (diffusion coefficients of VOCs in the gas phase are orders of magnitude higher than the aqueous phase) (Rivett et al., 2011). Thus, diurnal effects, specific events such as changes in atmospheric pressure, and seasonal variability need to be evaluated through monitoring.

The choice of monitoring sites will also, almost certainly, be governed by pragmatic, mostly logistical considerations. These include permission to access land from the landowner/tenant, health and safety requirements, avoiding interference with other activities, provision of power for continuous monitoring instruments and having (or being able to create) a secure location for long-term monitoring where equipment is not likely to be damaged or removed. Equipment needs to be inconspicuous if close to public areas or footpaths and requires protection from farm and wild animals. This is likely to mean fenced enclosures on farm land and protection of cables against rodent damage. Access to mains power is preferable, and more straightforward, than the use of batteries backed up by solar panels and/or fuel cells.

The area to be covered by surveys and sites for long-term monitoring will need to take account of the characteristics of each shale gas development. The overall area should cover all the wells being used for the project, including those for hydraulic fracturing and gas production and monitoring. It should cover the surface footprint above any laterals drilled from each wellhead, plus the likely fracture zone around those wells, and take account of any abandoned or decommissioned wells in that footprint that might provide gas migration pathways.

7.3 MEASUREMENTS

Gases that need to be measured include: CH_4, CO_2 (which could be produced from CH_4 oxidation in the shallow sub-surface or could be a significant component of the gas produced), O_2 and N_2 (useful in helping determine the source of CH_4 and CO_2. For an initial assessment, N_2 can be assumed to make up the balance of the gas once CO_2 and O_2 have been determined). Rn and He are useful as possible tracers of existing gas migration pathways. Other light hydrocarbon gases and trace gases such as H_2S can also be included. These may help to define the source of gas.

It is important to measure the flux of the main gases (CO_2 and CH_4) as well as their concentration. High concentration values do not necessarily indicate a significant source of gas. They can be caused by ground conditions, for example waterlogged or frozen surface layers that prevent escape of gas to the atmosphere. The natural range of CO_2 in soil can exceed 10% even without such enhancement. On the other hand, the coincidence of higher concentrations and fluxes indicates a significant flow of gas from the soil.

Instrument precision requirements depend on the type of measurement and the gas being measured. Since ambient levels of CH_4 in the atmosphere and soil are much lower than CO_2 (less than 2 ppm for CH_4 compared to about 400 ppm for CO_2 in the atmosphere on average) much greater sensitivity is needed for CH_4 even for screening-type soil gas measurements. In order to identify anomalies in field soil gas measurements, relatively low-accuracy portable instruments may be adequate for CO_2, O_2 etc (e.g. accuracy may be around ±0.5% (vol) over the range 0-60% CO_2). However, sensitivity of 1 ppm or less is needed for CH_4 and other trace gases such as H_2S. Much greater precision and accuracy are needed, most likely from laboratory measurements, to back up field results and allow full source attribution. Typically these sensitivities should be better than 1 ppm for CO_2 and 10 ppb for CH_4.

Baseline measurements seek to define the pre-existing background and its variability in space and time ahead of any shale gas development. From this, a strategy needs to be developed to identify potentially anomalous readings and how their origin can be established. This may be obvious in some cases but less clear in others. Continuous monitoring should be considered at higher risk locations. Monitoring at sites of future/proposed wellheads could overlap with atmospheric monitoring, although this could usefully include the measurement of gas fluxes at the ground surface on site to help distinguish these from external airborne emission sources. Carbon dioxide flux can be measured using automated accumulation chamber systems, where up to 16 (or more) individual chambers can be deployed, or through the use of eddy covariance techniques for quantifying soil gas flux (e.g. Figure 27). Chamber methods typically provide data on an hourly cycle whereas eddy covariance requires data collection usually at 10 Hz. Eddy covariance is an atmospheric technique that provides a CO_2 flux for a larger footprint, which varies with the wind speed and direction. Three dimensional wind speed and direction, pressure, temperature and relative humidity are measured simultaneously with CO_2 concentration to enable flux to be calculated. Scanning laser methods have shown promise for continuous monitoring of larger areas (100–200 m across) with a single instrument and have successfully located leakage points and made reasonable estimates of the flux of gas emitted (Hirst et al., 2013; Levine et al., 2016). Such instruments typically acquire data at 1 Hz. In this case, the scanning laser was used to determine near surface CO_2 concentrations across the KM-8 well pad in North Yorkshire.

Telemetry of data is very useful for unsupervised continuous monitoring equipment. It enables data to be processed and evaluated shortly after acquisition, so that any higher values are spotted quickly and any instrumental errors identified soon after they occur enabling downtime to be minimised.

Figure 27. Continuous monitoring using automated flux chambers (foreground) and eddy covariance (on tripod in centre)

Spatial variability requires survey measurements. Mobile techniques, for example using off-road vehicles (e.g. Jones et al., 2009) or Unmanned Aerial Vehicles (UAVs; e.g. de Vries and Bernardo, 2011; Neumann et al., 2013) offer the most comprehensive detailed coverage for near-ground atmospheric monitoring. However, leakage can be rapidly dispersed in the atmosphere so measurements need to be close to the ground surface and are typically made at less than a few metres height for UAVs and less than 0.5 m for ground vehicles (e.g. Figure 28). Point soil gas and flux measurements (e.g. Figure 29) avoid such atmospheric dispersion but cannot cover large areas as quickly or with such a high density of observations.

Figure 28. Measurement of CH_4 and CO_2 close to the ground using mobile open path lasers

The duration and frequency of baseline soil gas monitoring needs to be adequate to cover seasonal changes. This would suggest a minimum duration of 1 year for continuous monitoring and spring, summer and autumn survey repeats. It should be borne in mind that longer term baselines have shown significant year-on-year variability (Beaubien et al., 2013) such that a single year may not necessarily be representative. Indeed, for CCS, soil gas baselines collected over 3 years have been suggested (Schlömer et al., 2013) although there is not general agreement on this.

Soil gas measurements need to be made at sufficient depth to minimise gas exchange with the atmosphere and thereby atmospheric dilution of the gas concentrations. In the UK this typically means below a depth of about 60 cm (Ball et al., 1991). Closer to 1 m depth (or even greater) is preferred. Gas exchange depends on the permeability of the soil and measurements need to be above the water table, which, in the UK, can be relatively close to the surface and thus limits the sampling depth. It also needs to be ensured that atmospheric air is not reaching the sampled depth along the annulus of the probe used. This can be achieved with simple, small diameter soil gas probes or, equally, with more bulky commercial soil gas sampling equipment. Monitoring of ground gas is also often carried out in shallow boreholes. These can be used for survey or continuous measurements. The boreholes can also be designed to allow sampling at different depths to create vertical gas profiles. However boreholes are more costly to prepare, which limits their effective coverage, and their construction quality or design, or conditions created within them or externally, can give rise to spurious higher gas concentrations by creating artificial pathways for gas to migrate into pore spaces, headspaces or sample collection systems (Card and Wilson, 2012).

Soil gas concentrations can be measured either directly in the field, using a portable analyser with an in-built pump drawing gas from the soil probe (Figure 29), or a sample can be taken from the probe into an evacuated container for laboratory (or field laboratory) analysis. It is good practice to keep the sample storage period to a minimum. Rapid field measurements, even of lower precision, can provide a useful check on sample container integrity; lower than expected laboratory values can indicate that the container integrity has been compromised. Laboratory instruments in general provide analyses of higher sensitivity and precision and for a wider range of gases. However, portable field equipment is becoming available that comes close to matching laboratory performance albeit usually for a more limited range of gases. The advantage of field measurements is that data are available immediately and any anomalous values can be investigated further during the same visit or sampled for more detailed investigation, without the need for a return field visit.

Figure 29 Measurement of soil gas concentrations (CH_4, CO_2, O_2, H_2S etc.) using a portable survey meter (left) and CO_2 and CH_4 flux using a survey accumulation chamber (right)

Soil gas survey flux measurements are most typically made using small accumulation chambers, which can be linked to gas analysers, giving simultaneous measurement of CO_2 and CH_4 fluxes (Figure 29, right). When making flux measurements, disturbance of the soil surface and vegetation needs to be minimised. Chambers should be equipped with a pressure equalisation mechanism to prevent them restricting the natural flux during the measurement.

The advantages and limitations of different monitoring approaches are set out in Table 7.

7.4 DATA QA/QC

Measuring and monitoring equipment should be controlled within documented quality control policies and procedures. Metadata is needed on equipment specifications, their location and maintenance or updating carried out. Instruments should be calibrated regularly against certified gas mixtures; field instruments should be QC checked before and after fieldwork. Typically this involves a zero and a span step, the former usually being made using nitrogen and the latter with the gas or gases being measured within their normal operational range. In practice for soil gas that means around 2 ppm for CH_4 and 2% for CO_2. Laboratory analyses should be conducted using ISO 17025/UKAS-accredited methods where possible, with appropriate use of blanks, replicates, certified reference materials and other laboratory standards. Reputable laboratories will usually apply the QA methods and principles required for UKAS accreditation even if the specific method used is not itself accredited.

The use of the eddy covariance technique strictly requires certain conditions to be met, such as the terrain being horizontal and uniform (Burba and Anderson, 2010). These can be difficult to achieve in practice at locations chosen for continuous monitoring (e.g. around wellheads) and the implications of any departures from the ideal conditions need to be considered carefully and properly documented before using this method.

Table 7. Comparison of different near surface monitoring approaches.

Monitoring technique	Advantages	Limitations
Soil gas concentration measurements with field portable equipment	Easy and rapid, relatively low cost equipment (higher cost, higher precision equipment is available) Instant results allow follow up of any anomalies	Point data only at one moment in time. Limit to number of points that can be measured so spacing may need to be wide to cover a large area. Therefore leaks could be missed. Low cost equipment gives lower precision. Better precision at higher cost.
Soil gas concentration measurements with laboratory equipment	Higher precision More gas species (including isotopes) can be measured	Higher cost Longer turnaround in getting results More laborious Point data (as above)
Survey flux chamber measurements	Quick and easy Direct measure of flux	Limited number of gases can be directly measured (e.g. CO_2, CH_4, H_2S). Other gases only via indirect sampling and analysis Point data (as above)
Mobile ground vehicle surveys	Can cover larger areas with greater density of measurements Some sensitive, high precision equipment available Measurements close to ground surface minimise atmospheric dispersion	May need closely spaced traverses to detect small leaks Some areas may be inaccessible to ground vehicles Limited temporal coverage Higher cost equipment compared with some soil gas techniques
Unmanned Aerial Vehicle surveys	More rapid, larger area coverage May be able to fly over areas inaccessible to ground vehicles	New approach, not extensively tested Limited temporal coverage Relatively high cost UAV permitting and safe operation may limit use in some areas May need 2-3 people for safe operation
Automated soil gas monitoring stations	Continuous data Multiple probes possible Can view data remotely	Limited spatial coverage Moderately expensive compared with low-cost soil gas survey equipment
Automated flux chambers	As above	As above. Measures flux over a small surface area
Eddy covariance	Continuous data Larger measured footprint (typically 50 x height of sensors) Calculation of flux	Footprint varies with wind speed and direction Requires uniform surface roughness which may not exist at shale gas site
Scanning lasers	Continuous data Larger area can be covered (up to a few hundred metres across) Possible to locate leak and estimate flux	Moderately expensive Complex data processing

7.5 DATA PROCESSING, ANALYSIS AND REPORTING

Established statistical methods should be used to summarise the data and for the statistical classification of results and the identification of possible anomalies. Box and whisker plots (e.g. Figure 30) are a useful way to summarise results, which can be classified by location, land use, geology or other factors for comparative assessment. Normal probability plots can also be used to examine statistical data distributions and identify potentially different populations or anomalous values within a dataset. This information can then be used to classify data for use with appropriate software for spatial plotting and data visualisation. GIS software or other packages designed for mapping spatial data can then be used (e.g. Figure 30). Continuous flux measurements can be processed using validated software from equipment providers or using open-source code (e.g. EdiRe for eddy covariance measurements).

Interpretation of the data is likely to require the use of ancillary information on parameters that are known to control soil gas and flux, such as rainfall, soil moisture, pressure, temperature (atmospheric and soil), wind speed and direction (e.g. Hinkle, 1994; Schlömer et al., 2014). Most should be recorded as part of the atmospheric monitoring package (Section 5). The exception is soil moisture, which needs to be included, along with all the other ancillary measurements where soil gas monitoring occurs away from other atmospheric monitoring.

A key element of any monitoring will be attributing the source of any gas anomalies detected. Ratios of CO_2 to O_2 and N_2 have been shown to be effective in distinguishing near-surface biological CO_2 from that leaking from depth or produced by oxidation of CH_4 (see case study; Beaubien et al., 2013; Romanak et al., 2012). The presence of other hydrocarbon gases (e.g. ethane, butane etc.) may be diagnostic of deep gas escape (Klusman, 1993; Tedesco, 1995), as might coincident anomalies of gases carried by the deep CH_4 or CO_2, such as Rn or He (e.g. Baubron et al., 2002). The ratios of CH_4 and higher hydrocarbon gases can be diagnostic of the source, for example if the composition of a gas reservoir has been well characterised. Other possible approaches to source attribution include the use of isotopes, including stable C and O isotopes (Giustini et al., 2013; Hakala, 2014; Humez et al., 2016; Mayer et al., 2015; Sherwood et al., 2016), radiocarbon (e.g. Trium Inc. and Chemistry Matters, 2011) and noble gas isotopes (Giustini et al., 2013; Hunt et al., 2012; Mackintosh and Ballentine, 2012). C and O stable isotopes can help to distinguish biogenic and thermogenic CH_4 and CO_2, but do not always give unambiguous results. This is also the case with most methods especially where migration has occurred over great distances. A range of process may operate to modify the initial composition or characteristics of the gas (e.g. dilution, oxid4202ation, retardation etc) and so care needs to be taken when applying these methods. The recommendation is not to rely on a single method but apply a range of methods.

Radiocarbon measurements enable modern biogenic sources to be distinguished from fossil gases of geological origin older than about 30,000 years, the latter having little or no remaining radiocarbon because of its half-life of 5,730 years. Noble gases are non-reactive and isotopes of different species are formed in different ways allowing a variety of processes/sources to be evaluated, for example He isotopes can shed light on deep earth inputs whilst Ne isotopes can help understand atmospheric influences.

Source attribution is unlikely to be needed routinely during baseline or operational monitoring, but rather used to help understand pre-existing soil gas occurrences and to identify the source of any anomalies identified during operations.

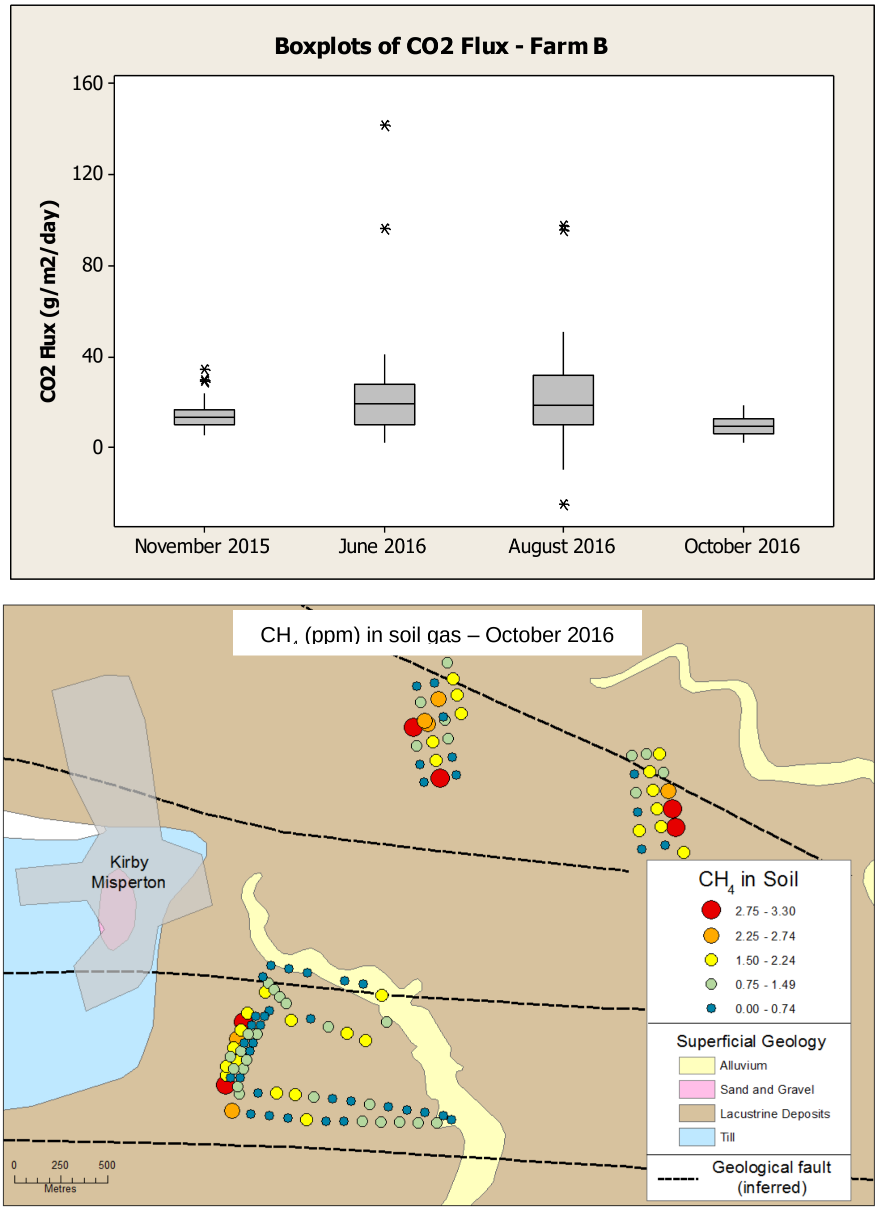


Figure 30 Examples of baseline data presentation, against which data collected during a subsequent operational phase can be assessed. Box and whisker plot of CO_2 flux values from different surveys (top) showing seasonal variations. Spatial variable symbol size plot showing classified CH_4 concentrations from a single survey. Grey zone denotes the Kirby Misperton urban area

7.6 CASE STUDY – USE OF GAS RATIOS AND ISOTOPES TO DETERMINE SOURCE

A landowner close to the Weyburn CO_2 enhanced oil recovery and geological storage project alleged that high CO_2 concentrations in the soil on their property were the result of leakage of the injected CO_2 from depth. This was investigated by a number of groups using gas ratios and stable and radiogenic isotopes. The need to undertake a detailed and costly forensic analysis of the gases was as a result of the lack of a good baseline. Had a baseline been available an earlier and more definitive diagnosis might have been possible. Following investigations, the higher CO_2 values were demonstrated to be of biogenic origin (Beaubien et al., 2013; Romanak et al., 2014; Sherk et al., 2011; Trium Inc. and Chemistry Matters, 2011) from their CO_2/O_2 and CO_2/N_2 ratios (e.g. Figure 31). Radiocarbon analysis showed the gas to be near 100% modern carbon, consistent with a recent biological origin, whereas the injected CO_2 had no modern carbon (Figure 32). Noble gas isotope data supported these conclusions although stable C isotopes were inconclusive in this case (Figure 32). It may be possible to also exploit contrasts in signatures in a shale gas context, providing the isotopic signatures of deep and shallow sources can be distinguished. This will be dependent on the origin of produced shale gas, or being able to 'fingerprint' a sample in advance.

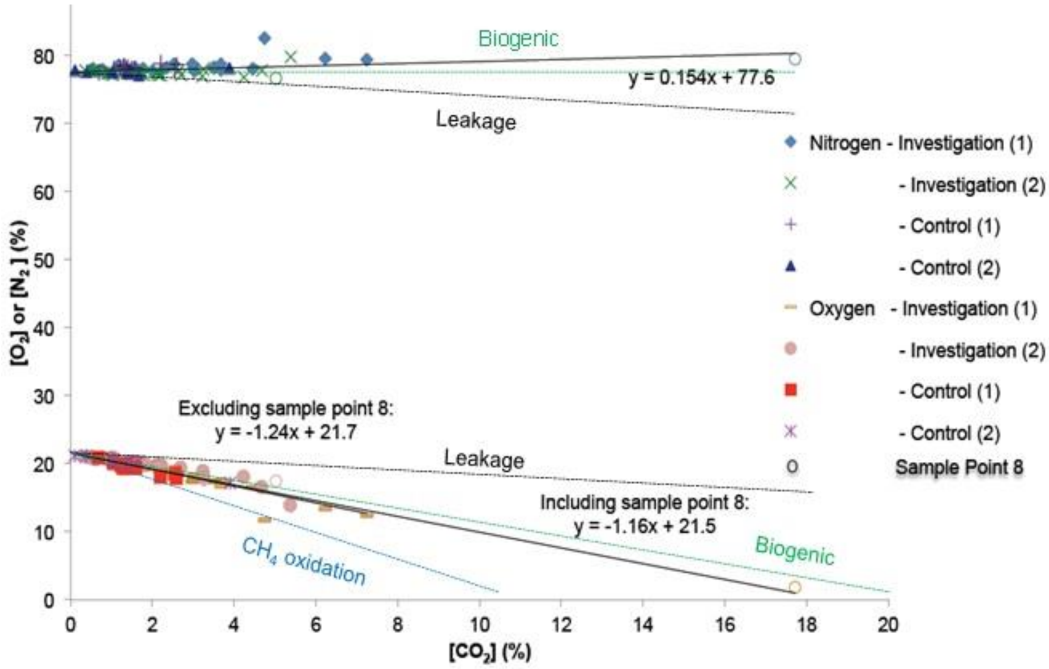


Figure 31. Use of gas ratios to refute an allegation of leakage of deep injected CO_2 at the Weyburn project, Canada. Most of the higher values fall near the perfect biogenic lines. Scatter below the biogenic line for CO/O_2 and above the line for CO_2/N_2 can be attributed to dissolution of CO_2 in soil pore water. (Data from Trium Inc. and Chemistry Matters, 2011)

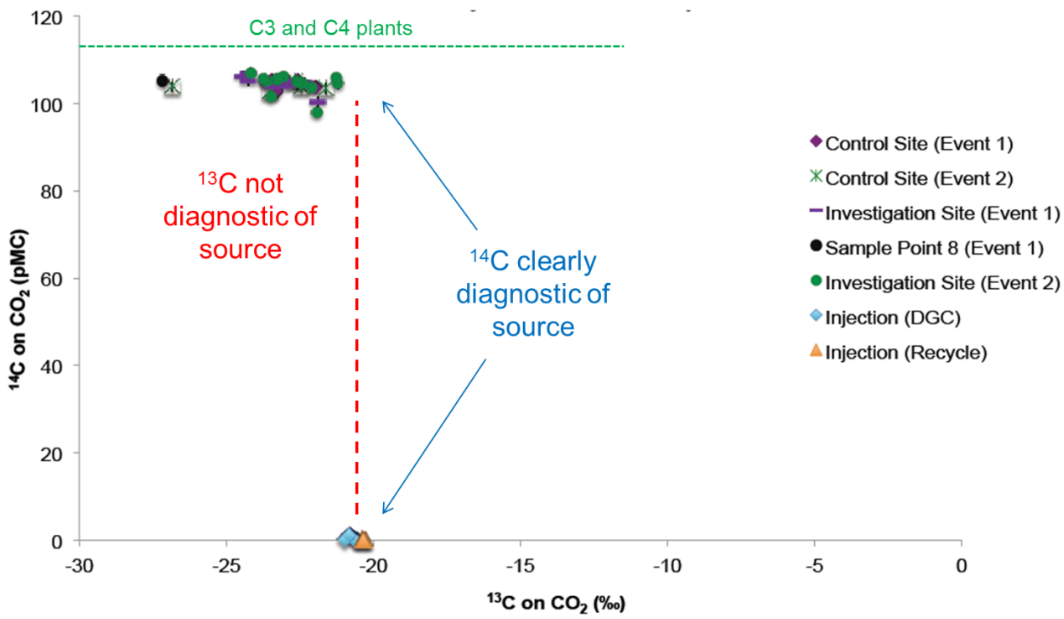


Figure 32. Radiocarbon (^{14}C) clearly distinguishes the gas at the site of alleged leakage from that injected at Weyburn whereas stable C isotopes (^{13}C) are not diagnostic in this case since there is overlap between modern plant signatures and values for the injected CO_2 (Data from Trium Inc. and Chemistry Matters, 2011)

7.7 CHANGE DETECTION

Ultimately the purpose of collecting baseline soil gas data is to use it to identify, assess, and ideally attribute any change in soil gas characteristics (or to support investigations of changes in atmospheric or groundwater characteristics) arising after shale gas operations begin. Examples are included at appropriate points in earlier sections, but are summarised here.

If change is suspected, i.e. concentrations exceed the maximum for the baseline period within the monitoring area, then identifying and assessing whether the change is attributable to shale gas operations requires collection and evaluation of sufficient new data for comparison against the baseline data set. Surveys will need to include both soil gas and flux (as coincident change may be diagnostic of gas leaking to the surface), and gas samples collected for additional parameters (e.g. light hydrocarbons, stable and, possibly, radioisotopes etc) to attempt to apportion the source of the anomaly. Depth profiles should also be considered. Given the limited amount of soil gas data that might be available for the baseline period, a weight of evidence approach should be used to attribute the observed change to one or more sources.

Where an anomaly is suspected but not located, a wide area survey (e.g. mobile open path laser, possibly UAV) can be rapidly deployed, and will allow large scale screening. This would then be followed by focussed point measurements to determine spatial extent and additional parameters, possibly with the installation/repositioning of continuous monitoring equipment if this is feasible.

Since change may also be suspected or detected outside the continuous monitoring and routine baseline monitoring area because of the complex nature of gas migration to the surface, surveying will need to expand beyond the original baseline monitoring area. Drawing on baseline data to assess the anomaly would probably not be appropriate, and a process based approach (e.g. gas compositions and isotopic ratios) combined with evidence from atmospheric and groundwater monitoring is likely to be more robust.

Finally, there may not be any obvious surface manifestation of change following the start of operations. Nonetheless, repeat routine surveys are recommended to increase the likelihood of any change being detected and assessed early. It is also, of course, prudent to continue routine monitoring throughout the operational phase whether a change is evident or not.

7.8 REFERENCES

ALLISON, M L. 2001. Hutchinson, Kansas: A Geologic Detective Story. *Geotimes*, Vol. October 2001.

APPLETON, J D, HOOKER, P J, AND SMITH, N J P. 1995. Methane, carbon dioxide and oil seeps from natural sources and mining areas: characteristics, extent and relevance to planning and development in Great Britain. *British Geological Survey Technical Report*, WP/95/1 (Keyworth, Nottingham).

BALL, T K, CAMERON, D G, COLMAN, T B, AND ROBERTS, P D. 1991. Behavior of Radon in the Geological Environment - a Review. *Quarterly Journal of Engineering Geology*, Vol. 24, 169-182.

BAUBRON, J C, RIGO, A, AND TOUTAIN, J P. 2002. Soil gas profiles as a tool to characterise active tectonic areas: the Jaut Pass example (Pyrenees, France). *Earth and Planetary Science Letters*, Vol. 196, 69-81.

BEAUBIEN, S E, JONES, D G, GAL, F, BARKWITH, A K A P, BRAIBANT, G, BAUBRON, J C, CIOTOLI, G, GRAZIANI, S, LISTER, T R, LOMBARDI, S, MICHEL, K, QUATTROCCHI, F, AND STRUTT, M H. 2013. Monitoring of near-surface gas geochemistry at the Weyburn, Canada, CO_2-EOR site, 2001–2011. *International Journal of Greenhouse Gas Control*, Vol. 16, Supplement 1, S236-S262.

BRITISH STANDARDS INSTITUTION. 2013. Guidance on investigations for ground gas. Permanent gases and Volatile Organic Compounds (VOCs), BSI Standards Limited. BS8576.

BRITISH STANDARDS INSTITUTION. 2015. Code of practice for the design of protective measures for methane and carbon dioxide ground gases for new buildings, BSI Standards Limited. BS8485.

BURBA, G, AND ANDERSON, D. 2010. *A Brief Practical Guide to Eddy Covariance Flux Measurements: Principles and Workflow Examples for Scientific and Industrial Applications.* (Lincoln, USA: LI-COR.) ISBN 978-0615430133

CARD, G, AND WILSON, S. 2012. A pragmatic approach to ground gas risk assessment for the 21st Century. *CL:AIRE Research Bulletin*, Vol. RB17, 1-11.

CROWHURST, D, AND MANCHESTER, S J. 1993. The Measurement of Methane and other Gases from the Ground. *CIRIA Report*, 131.

DE VRIES, D F H, AND BERNARDO, C H. 2011. Spatial and temporal variability in atmospheric CO_2 measurements. *Energy Procedia*, Vol. 4, 5573-5578.

ENVIRONMENT AGENCY. 2010. Guidance on monitoring landfill gas surface emissions. *Environment Agency Report*, LFTGN 07 (Bristol).

EUROPEAN UNION. 2009a. Directive 2009/29/EC of the European Parliament and of the Council of 23 April 2009 amending Directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading scheme of the Community (Text with EEA relevance).

EUROPEAN UNION. 2009b. Directive 2009/31/EC of the European Parliament and of the Council of 23 April 2009 on the geological storage of carbon dioxide and amending Council Directive 85/337/EEC, European Parliament and Council Directives 2000/60/EC, 2001/80/EC, 2004/35/EC, 2006/12/EC, 2008/1/EC and Regulation (EC) No 1013/2006 (Text with EEA relevance).

GIUSTINI, F, BLESSING, M, BRILLI, M, LOMBARDI, S, VOLTATTORNI, N, AND WIDORY, D. 2013. Determining the origin of carbon dioxide and methane in the gaseous emissions of the San Vittorino plain (Central Italy) by means of stable isotopes and noble gas analysis. *Applied Geochemistry*, Vol. 34, 90-101.

HAKALA, J A. 2014. Use of stable isotopes to identify sources of methane in Appalachian Basin shallow groundwaters: a review. *Environmental Science: Processes & Impacts*, Vol. 16, 2080-2086.

HINKLE, M E. 1994. Environmental-Conditions Affecting Concentrations of He, CO_2, O_2 and N_2 in Soil Gases. *Applied Geochemistry,* Vol. 9, 53-63.

HIRST, B, JONATHAN, P, GONZÁLEZ DEL CUETO, F, RANDELL, D, AND KOSUT, O. 2013. Locating and quantifying gas emission sources using remotely obtained concentration data. *Atmospheric Environment*, Vol. 74, 141-158.

HOOKER, P J, AND BANNON, M P. 1993. Methane: its occurrence and hazards in construction. *CIRIA Report*, 130 (London).

HUMEZ, P, MAYER, B, ING, J, NIGHTINGALE, M, BECKER, V, KINGSTON, A, AKBILGIC, O, AND TAYLOR, S. 2016. Occurrence and origin of methane in groundwater in Alberta (Canada): Gas geochemical and isotopic approaches. *Science of The Total Environment*, Vol. 541, 1253-1268.

HUNT, A G, DARRAH, T H, AND POREDA, R J. 2012. Determining the source and genetic fingerprint of natural gases using noble gas geochemistry: A northern Appalachian Basin case study. *Bulletin of the American Association of Petroleum Geologists*, Vol. 96, 1785-1811.

ISO. 2017. Soil quality -- Sampling -- Part 204: Guidance on sampling of soil gas. ISO 18400-204:2017.

JONES, D G, BARLOW, T, BEAUBIEN, S E, CIOTOLI, G, LISTER, T R, LOMBARDI, S, MAY, F, MÖLLER, I, PEARCE, J M, AND SHAW, R A. 2009. New and established techniques for surface gas monitoring at onshore CO_2 storage sites. *Energy Procedia*, Vol. 1, 2127-2134.

KLUSMAN, R W. 1993. *Soil gas and related methods for natural resource exploration*. (Chichester: J. Wiley & Sons.)

KLUSMAN, R W. 2011. Comparison of surface and near-surface geochemical methods for detection of gas microseepage from carbon dioxide sequestration. *International Journal of Greenhouse Gas Control*, Vol. 5, 1369-1392.

LEVINE, Z H, PINTAR, A L, DOBLER, J T, BLUME, N, BRAUN, M, ZACCHEO, T S, AND PERNINI, T G. 2016. The detection of carbon dioxide leaks using quasi-tomographic laser absorption spectroscopy measurements in variable wind. *Atmospheric Measurement Techniques*, Vol. 9, 1627-1636.

MACKINTOSH, S J, AND BALLENTINE, C J. 2012. Using $^3He/^4He$ isotope ratios to identify the source of deep reservoir contributions to shallow fluids and soil gas. *Chemical Geology*, Vol. 304–305, 142-150.

MAYER, B, HUMEZ, P, BECKER, V, DALKHAA, C, ROCK, L, MYRTTINEN, A, AND BARTH, J A C. 2015. Assessing the usefulness of the isotopic composition of CO_2 for leakage monitoring at CO_2 storage sites: A review. *International Journal of Greenhouse Gas Control*, Vol. 37, 46-60.

NEUMANN, P P, ASADI, S, BENNETTS, V H, LILIENTHAL, A J, AND BARTHOLMAI, M. 2013. Monitoring of CCS Areas using Micro Unmanned Aerial Vehicles (MUAVs). *Energy Procedia*, Vol. 37, 4182-4190.

O'RIORDAN, N, AND MILLOY, C. 1995. Risk assessment for methane and other gases from the ground. *CIRIA Report*, 152.

RIVETT, M O, WEALTHALL, G P, DEARDEN, R A, AND MCALARY, T A. 2011. Review of unsaturated-zone transport and attenuation of volatile organic compound (VOC) plumes leached from shallow source zones. . *Journal of Contaminant Hydrology*,, Vol. 123, 130–156.

ROMANAK, K D, BENNETT, P C, YANG, C, AND HOVORKA, S D. 2012. Process-based approach to CO_2 leakage detection by vadose zone gas monitoring at geologic CO_2 storage sites. *Geophysical Research Letters*, Vol. 39, L15405.

ROMANAK, K D, WOLAVER, B, YANG, C, SHERK, G W, DALE, J, DOBECK, L M, AND SPANGLER, L H. 2014. Process-based soil gas leakage assessment at the Kerr Farm: Comparison of results to leakage proxies at ZERT and Mt. Etna. *International Journal of Greenhouse Gas Control*, Vol. 30, 42-57.

SCHLÖMER, S, FURCHE, M, DUMKE, I, POGGENBURG, J, BAHR, A, SEEGER, C, VIDAL, A, AND FABER, E. 2013. A review of continuous soil gas monitoring related to CCS – Technical advances and lessons learned. *Applied Geochemistry*, Vol. 30, 148-160.

SCHLÖMER, S, MÖLLER, I, AND FURCHE, M. 2014. Baseline soil gas measurements as part of a monitoring concept above a projected CO2 injection formation—A case study from Northern Germany. *International Journal of Greenhouse Gas Control*, Vol. 20, 57-72.

SHERK, G W, ROMANAK, K D, DALE, J, GILFILLAN, S M V, HASZELDINE, R S, RINGLER, E S, WOLAVER, B D, AND YANG, C. 2011. The Kerr Investigation: Final Report. Findings of the investigation into the impact of CO_2 on the Kerr Property. *International Performance Assessment Centre for geologic storage of Carbon Dioxide (IPAC-CO2)* (Regina, Saskatchewan, Canada).

SHERWOOD, O A, ROGERS, J D, LACKEY, G, BURKE, T L, OSBORN, S G, AND RYAN, J N. 2016. Groundwater methane in relation to oil and gas development and shallow coal seams in the Denver-Julesburg Basin of Colorado. *Proceedings of the National Academy of Sciences*.

SIZER, K, CREEDY, D, AND SCEAL, J. 1996. Methane and other gases from disused coal mines: the planning response. *The Stationary Office Department of the Environment Technical Report*, Research Contract PECD 7/1/445 (London).

TEDESCO, S A. 1995. *Surface geochemistry in petroleum exploration*. (New York: Chapman and Hall.)

THE COAL AUTHORITY, HEALTH AND SAFETY EXECUTIVE, BRITISH DRILLING ASSOCIATION, FEDERATION OF PILING SPECIALISTS, AND THE ASSOCIATION OF GEOTECHNICAL AND GEOENVIRONMENTAL SPECIALISTS. 2012. Guidance on Managing the Risk of Hazardous Gases when Drilling or Piling Near Coal.

TOPP, E, AND PATTEY, E. 1997. Soils as sources and sinks for atmospheric methane *Canadian Journal of Soil Science*, Vol. 77, 167–178.

TRIUM INC. AND CHEMISTRY MATTERS. 2011. Site Assessment (SW-30-05-13-W2M), near Weyburn, Saskatchewan. *Trium Environmental Inc. and Chemistry Matters, Project Number: T10-167-CEN*.

WILSON, S, CARD, G, AND HAINES, S. 2009. *Ground Gas Handbook*. (Whittles Publishing.) ISBN 9-781904-445685

WILSON, S, OLIVER, S, MALLETT, H, HUTCHINGS, H, AND CARD, G. 2007 Assessing risks posed by hazardous ground gases to buildings. *CIRIA Report*, 665 (London).

8. Ground Motion (subsidence/uplift)

8.1 INTRODUCTION

It is currently unproven whether there is potential for shale gas operations at depth to cause a long-term change in surface elevation (subsidence or uplift), i.e. ground motion. Conventional oil and gas operations have on rare occasions been shown to result in subsidence above compacting oil and gas reserves (Geertsma 1973) and a recent study suggests that surface uplift in eastern Texas was due to fluid injection, which was distinguished using satellite remote sensing (Shirzaei et al., 2016). These studies do not imply that shale gas operations at depth will cause ground motion. The lack of information on ground motion in shale gas operation areas was noted by Dost et al (2013) in relation to the Groningen area following the seismic activity in the gas field, stating that "no reliable local ground motion measurements are available to constrain the ground motion" at that locality. It is imperative to undertake objective and authoritative monitoring of the ground surface at operation sites and surrounding regions (a) to determine if there are any impacts on the ground surface and (b) to reassure stakeholders (including the public) that appropriate independent monitoring of all potential environmental impacts is being undertaken.

The key monitoring question is whether shale gas operations alter the site or surrounding region. It should not be assumed that an area is stable prior to shale gas operations. When considering a monitoring system, it is important to account for the dynamic nature of the earth's surface i.e. there may be some pre-existing displacement due to either natural or induced factors. Examples of pre-existing natural ground motion include landsliding and clay shrink/swell, while underground mining and groundwater abstraction are examples of anthropogenic activity that may cause ground motion. It is necessary to characterise any pre-existing ground motion so that potential shale gas related motion can be resolved from them, and therefore a baseline survey is required to determine the pre-existing conditions of the site including displacement such as upwards motion (uplift), downwards motion (subsidence) or horizontal / lateral motion. Furthermore, in this context, the term 'ground motion' does not refer to seismicity, which is the frequency, intensity and distribution of earthquakes (induced or otherwise) in an area.

The specific objectives of a ground motion analysis are to:

- characterise whether the ground surface was stable or moving in the past;
- confirm the current ground motion status;
- characterise any motion identified e.g. average velocity and temporal trends;
- identify the most likely geological causes of discrete areas of motion, where/if motion is measured;
- provide a body of impartial information to inform the regulators and other stakeholders of the ground motion situation.

The strategy proposed in this guidance document for identifying and monitoring the ground motion situation is to utilise radar satellite imaging techniques, as opposed to installation of in situ sensors. Interferometric Synthetic Aperture Radar (InSAR) from orbiting satellites can be used in a non-invasive way to determine the status of the ground surface motion with millimetre precision. The technique may be applied to determine objectively if shale gas operations have altered the environmental conditions of the ground surface. Archive satellite data acquired from 1992 onwards can provide a baseline of the ground motion situation prior to shale gas operations while currently-orbiting satellites can be used to monitor the present-day situation. In situ sensors including Global Navigation Satellite System (GNSS) such as the U.S. GPS or Russian GLONASS can provide data on ground conditions at a point in space, but clearly it is not possible to 'go back in time' and install such receivers at a site, which is why InSAR is proposed as the preferred technique for a baseline survey.

Satellite imagery acquired during operations can be used to identify if there are any changes to the pre-existing ground motion situation that has been assessed by a baseline survey. Current satellite imagery can be integrated with (and validated by) GNSS data if appropriate instruments are installed on site.

Authoritative and impartial ground motion monitoring is vital in relation to shale gas operations. Quantitative and precise data regarding the motion or stability of the ground can inform regulators and stakeholders about the environmental situation and potential anthropogenic / induced impacts. Seismic events have been linked to hydraulic fracturing operations in the UK at Preese Hall (de Pater & Maisch, 2011) and in the U.S. (Ellsworth 2013; Holland 2011; Kim 2013). The public perception, as noted at a series of public engagement events in both Lancashire and Yorkshire, is that induced seismic activity will result in ground motion or vice versa. Quantitative measurements of ground motion, both historic and current, are required to confirm any surface displacement and potentially to allay public concerns regarding the impacts to the surface environment and the structures built upon or within it.

8.2 INSAR DESCRIPTION

Synthetic Aperture Radar (SAR) is an active microwave imaging system that can penetrate clouds and operate at night time. By measuring the phase difference between satellite images it is possible to measure sequential changes of the Earth's surface with millimetric accuracy and metric resolution (Pepe and Calò, 2017). Processing a stack of images acquired over a particular time period can provide an average of ground motion as well as a time series showing if the point or distributed scatterer has moved relative to the previous and subsequent images. The InSAR measurements are generally described in terms of Line of Sight (LOS) from the satellite or as absolute motion (vertical / horizontal displacement). The motion that is measured does not take account of ground acceleration, i.e. peak ground acceleration (PGA).

The InSAR process has been refined since early applications over 25 years ago (e.g. Massonnet et al., 1993) to include techniques such as Persistent Scatterer Interferometry (PSI) (Ferretti at al., 2001), Small Baseline Subset (SBAS) (Bernardino et al., 2002), SqueeSAR (Ferretti et al., 2011), Intermittent SBAS (ISBAS) (Sowter et al., 2016) and RapidSAR (Spaans and Hooper, 2016).

InSAR is an appropriate technology for precisely monitoring surface motion in shale gas baseline monitoring studies because archive radar data (acquired since 1992) can be utilised (where available) to analyse regions where in situ GNSS/EDM/tiltmeter data are not available historically. Furthermore, even where historic and current in situ data are available, InSAR studies can provide a more regional picture than the interpolated point coverage derived from traditional techniques such as GNSS stations. Ideally, the remote and in situ methods should be integrated because they provide complimentary information at a range of scales e.g. an array of tiltmeters can provide information on local micro-motion in comparison to the more regional picture provided by InSAR.

InSAR can provide millimetric measurements of surface ground motion from satellite platforms such as ENVISAT, ERS1&2, RADARSAT, Sentinel-1A/B, TerraSAR-X and COSMO-SkyMed. Raw data from the European Space Agency (ESA) satellites (e.g. Sentinel-1A and B) are free and there is now good coverage of data over the UK. There is a cost associated with obtaining data from commercial satellites such as TerraSAR-X, and coverage of the UK is not complete. Both ESA and commercial satellite data generally come in raw form and need expert processing and interpretation before they are usable.

The technology has been validated by BGS in projects such as TerraFirma and used in projects including PanGeo, SubCoast and EVOSS to develop and demonstrate viable services (e.g. Jordan et al 2011 & Jordan et al 2013). InSAR has also been successfully used in CO_2 sequestration monitoring projects in locations such as In Salah where Mathieson (2010) stated that "*perhaps the most valuable, and initially surprising, monitoring method so far has been the use of satellite based Interferometric synthetic aperture radar (InSAR) to detect subtle ground deformation*".

It is suspected that InSAR has not yet been applied to shale gas operations primarily due to the challenge of gaining results in non-urban vegetated areas; however newly-developed methods such as ISBAS, SqueeSAR and RapidSAR are addressing this limitation (Gee et al., 2016).

8.2.1 Comparison with in situ ground monitoring systems

Site-specific ongoing ground motion monitoring can be undertaken using a dense network of GNSS stations and/or tiltmeters and/or continuous total station surveys. Geodetic (GNSS and total station) and geotechnical sensors (tiltmeters) are mature technologies that can measure ground motion at a point in space or between two or more points. Geotechnical sensors, located at the surface or down boreholes are used to measure non-georeferenced displacements or movements. Geodetic measuring devices record georeferenced displacements or movements in 1, 2, 3 or 4 dimensions; this group includes GNSS. GNSS is the generic term for a constellation of satellites that provide geospatial positioning. There are various forms of GNSS including the two operational systems; GPS (the U.S. Global Positioning System) and GLONASS (the Russian Global Navigation Satellite System) along with developmental systems such as the European Union's Galileo system and the Chinese Beidou. In situ sensors and monitoring approaches require a period of baseline recording in order to provide a comparison with ongoing and post-shale gas operations.

The use of integrated GNSS and tiltmeters (either in isolation or integrated at a site) is common practice for ground motion monitoring in many applications including volcanology (e.g. Hawaii - http://hvo.wr.usgs.gov/kilauea/update/deformation.php) and CO2 storage monitoring (e.g. In Salah - Mathieson et al., 2010). Furthermore, Fisher and Warpinski (2011) published a summary of US microseismic and tiltmeter data in shales based on the Barnett, Woodford, Marcellus and Eagle Ford shales, noting that a surface array of tiltmeters located on the ground surface can be used to measure the deformation pattern and determine some details of the fracture orientation. Tiltmeters can also be installed downhole, with Fisher and Warpinski (2011) concluding that they can be used to measure the height of the hydraulic fracture when installed near the treatment well with an array sufficiently long enough to span the fractured interval's thickness. Typically, between 15 and 100 tiltmeters will be placed on the ground around the well. It is worth noting that the studies referenced above did not include a baseline monitoring component. Table 7 provides a guide to the advantages and limitations of remote and in situ systems for ground motion monitoring.

The InSAR process does not specifically require calibration or validation with in situ sensors such as GNSS. Nevertheless, GNSS stations were employed during the BGS baseline monitoring of Lancashire in order to provide an extra level of assurance that the process is fit-for-purpose for monitoring shale gas operations. InSAR active and passive reflectors can be installed on site to increase the number of persistent scatterers, if deemed appropriate.

8.3 METHODOLOGY

Several actions are required in order to effectively undertake a monitoring programme of ground motion conditions using InSAR techniques; these are illustrated in Figure 33. The actions describe both a baseline study and continuous ('current') monitoring of the ground motion conditions of a region using InSAR technologies. These actions form a general suggested set of recommended steps for baseline monitoring of ground motion using InSAR technologies:

Table 8. Comparison of remote and in situ ground surface motion monitoring systems

Monitoring Technique	Advantages	Limitations
InSAR	Measurements are made remotely (non-invasive) Retrospective measurements can be made using historic data to gain a baseline prior to operations. Imagery can cover a large area simultaneously. Entire deformation field can be imaged, rather than isolated points.	Conventional techniques have difficulty in vegetated areas. Rapid motion (greater than the satellite detected phase difference) cannot be measured. Temporal and spatial resolution is limited by satellite set up and orbital parameters. Affected by steep topography (shown not be an issue in most of the UK).
GNSS	High precision. Does not require line of sight between benchmarks. Continuous site can operate without frequent human interaction.	Potentially difficult and expensive to install in remote or difficult to access areas. Equipment can be stolen / vandalised / damaged. Sampling of deformation field is limited to individual points; several points are required. Requires at least 4 satellites in view simultaneously. No historic baseline if sensors not installed prior to operations.
Tiltmeters	High precision. Does not require line of sight between benchmarks. Continuous site can operate without frequent human interaction.	Equipment can be stolen / vandalised / damaged. Sampling of deformation field is limited to individual points. Complex installation (e.g. in boreholes) – several tiltmeters are required. No historic baseline if sensors not installed prior to operations.
Total Stations	High precision. Continuous sites can operate without frequent human interaction.	Requires line of sight between benchmarks. Generally they are operated manually, requiring repeat site visits to operate the system. No historic baseline if sensors not installed prior to operations.

1. Conduct a catalogue search of satellite radar data to confirm that suitable stacks of images are available for the study area in order to mitigate the atmospheric noise. If a suitable stack of archive data is not available then InSAR monitoring cannot be undertaken for that time period. If a suitable stack of imagery is not available for current monitoring (e.g. using Sentinel-1A) then consider the acquisition parameters of the satellite and the length of time required to obtain a suitable stack, and revisit the archive in due course to monitor image acquisition progress.
2. Download the stack of image datasets covering the geographic area and the time period(s) of interest.
3. Process the imagery for the region using InSAR technique(s) that are appropriate for the landcover types to ensure (as much as possible) that suitable results are obtained for the region of interest accounting for factors such as whether the area is urban or rural or a

combination of both. The results must display LOS motions and may extend to absolute motion if appropriate data are available.

4. Ensure that the outputs from the InSAR processing match the quality required e.g:
    a. Suitable density of spatial coverage in the area of interest
    b. Suitable temporal coverage in the area of interest
    c. Assess output statistics to gauge if the results are fit-for-purpose
5. Interpretation of the InSAR outputs. This is a key stage because the outputs from the InSAR image processing are dependent on the quality of the interpretation. There are two fundamental components.
6. Ensure that interpretation is undertaken by sufficiently-experienced personnel. For shale gas applications the interpretation should be done by geoscientists experienced in compiling and integrating geoscientific information (noted below).
7. The interpretation is reliant upon access to a comprehensive range of geoscience data – these should be considered mandatory:
    a. Bedrock geology (incl. faults)
    b. Surficial geology (incl. compressible ground)
    c. Historic mining information / plans
    d. Seismic records
    e. Groundwater abstraction records
    f. Borehole records
    g. Geohazard information (e.g. landslides and shrink/swell)

    and ancillary data – these should also be considered mandatory:

    h. Landcover information
    i. Current and historic topographic maps
    j. Aerial photography
    k. Digital elevation models
    l. Digital terrain models.
8. Provide an impartial report on the ground motion conditions within the time period and geographic area covered by the InSAR processing. The report should outline if there are discrete zones of uplift or subsidence and should be accompanied by interpretations of the most probable causes of the motion. The InSAR results must be made available in formats readable and understandable by stakeholders along with statements outlining potential limitations of the information.

Additionally, it is vital to take into account the fact that InSAR techniques utilise large volumes of raw image data; they produce significant volumes of intermediate data, and the outputs invariably produce large raster / map files. Monitoring projects using these techniques must include appropriate data management protocols relating to data and product storage and management / distribution.

8.4 DATA HANDLING

Baseline monitoring is achieved over a wide region using archive and current radar data from satellites such as ERS-1/2, ENVISAT and Sentinel-1A/B. The data acquisition and selection process involves the following steps (explained in more detail below):

- Review quantity and quality of radar images covering the area of interest
- Identification of GNSS sites in operation when the images were acquired
- Assessment of the terrain in terms of its terrain, landcover and the preferred InSAR technique(s) to employ.

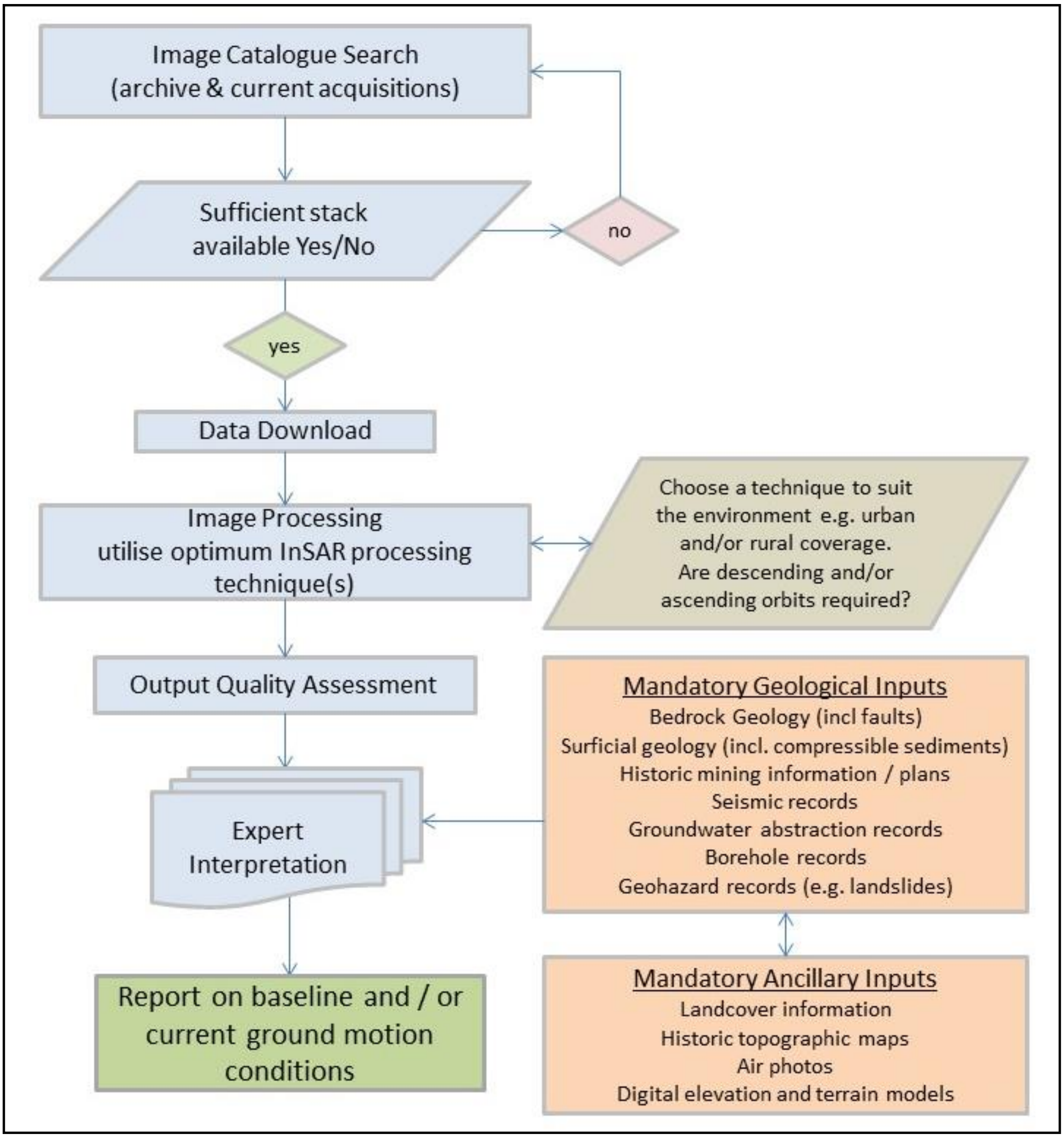


Figure 33. Actions and inputs required in order to effectively undertake a ground motion InSAR study

The first step in an assessment is to search the catalogues for archive data covering the area of interest. Each time the radar satellite passes overhead it captures an image of the terrain. There are several radar satellites currently in orbit that acquire imagery that can be processed for interferometry, including TerraSAR-X, COSMO-SkyMed and Sentinel-1. Additionally there are satellites that have acquired large archives of imagery over the UK, but which are no longer operating, such as ENVISAT and ERS-1/2. The European Space Agency's (ESA) C-band archive provides the most complete database of radar data for the UK, providing consistent stacks of historic ERS-1/2 (from 1991 to 2001) and ENVISAT data (from 2002 to 2010). These archives are vitally important because they can be used to 'go back in time' to 1992 (when they first started operating) and to create a baseline of ground motion prior to shale gas operations.

A full Sentinel-1 image covers ~250 km in range and ~180 km in azimuth (e.g. Figure 34). The Sentinel-1 A and B constellation capture an image of the same location of the UK every 6 days. Multiple images of the same location over a period of time (called a stack) can be processed to provide a time-series showing the relative motion of the terrain at each overpass, and therefore it can be determined if the ground is moving or stationary. Sufficiently long and populated stacks of radar imagery are required to generate a complete picture of ground motion over an area of interest. Greater numbers of images in a stack result in higher accuracy of ground motion, atmospheric phase components and height errors when processing with multi-interferogram methods such as PSI (Persistent Scatterer Interferometry) and SBAS (Small BAseline Subset) (e.g. Berardino et al., 2002 and Ferretti et al., 2001). It has been observed that at least ~15-20 images of the same acquisition geometry (i.e. same mode, orbit and track) are required to undertake a multi-interferogram InSAR analysis (e.g. Crosetto et al., 2010), and the quality of the results improves when the number of images in the stack increases.

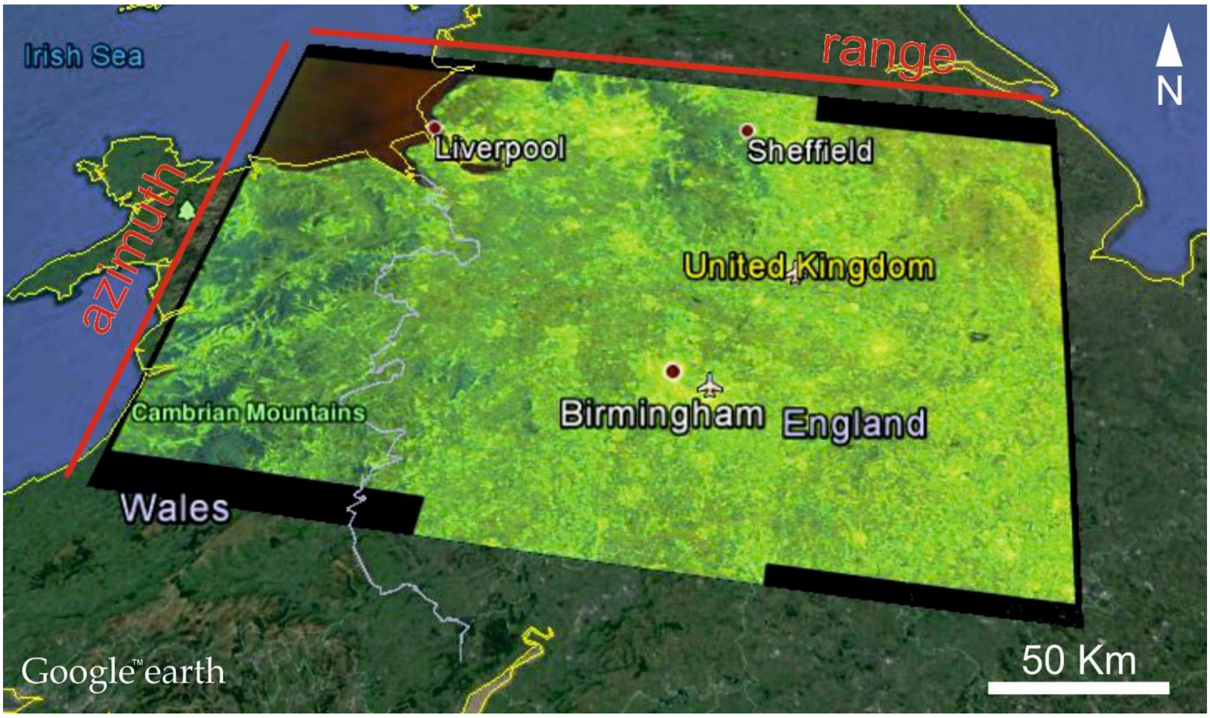


Figure 34. Example coverage available with the Interferometric Wide Swath Sentinel-1A satellite image stack

It is also beneficial to identify if GNSS stations exist so that they can be used to validate the InSAR results. In the UK, the BIGF (UKRI/BGS British Isles continuous GNSS Facility) provides archived RINEX data from GPS and GLONASS satellites, from a high density network of ~160 continuously recording stations, sited throughout mainland Britain, Northern Ireland and the Republic of Ireland.

Radar satellites are sideways looking and are therefore prone to geometric distortions when viewing the Earth; for example the presence of radar layover prevents the application of InSAR. Moreover, in areas of high relief there can be radar shadows, making some areas invisible to the sensor. However, Cigna et al (2014) reported that with the ERS and ENVISAT LOS, only ~1.0–1.4% of Great Britain is potentially affected by shadow and layover in ascending or descending mode. Combining ascending and descending modes brings the area affected down to ~0.02–0.04%, bearing in mind that distortions in hilly areas can be compensated for using either ascending or descending orbits. This indicates that the vast majority of the landmass can be monitored. Sentinel-1 data have a similar LOS incident angle and ground track angle to ERS and ENVISAT, therefore it is expected that the same proportion of the UK landmass could be monitored.

The existence of persistent scatterers must be accounted for when applying InSAR techniques. A persistent scatterer is a location on the ground that maintains coherence through several radar images and identified based on the scatterer amplitude value over time. Persistent scatterers are required for point-based InSAR analyses i.e. they can be compared to the prism / mirror that total stations use for their readings. The ability of surface targets to operate as persistent scatterers is related to ground properties such as geometry and land cover. Using the CORINE Landcover database Cigna et al (2014) calculated the likelihood of identifying persistent scatterers for different landcover types. They identify the “significant control of landcover on the potential for PS methods to identify scatterers, with particularly critical evidence for rural and grassland regions where only a few radar targets per square kilometre can be extracted and monitored via multi-interferogram processing.” The paper highlights that persistent scatterer InSAR using archive ERS1/2 or ENVISAT data is not capable of providing suitable coverage in rural areas and is likely to result in insufficiently dense networks of monitoring targets, with little possibility to obtain full understanding of ground motion. Therefore, the guidance is to employ multi-look techniques such as SqueeSAR™, ISBAS or RapidSAR that improve the capacity to deliver sufficient results over non-urban areas where shale gas operations are most likely to occur, whilst noting that these approaches also gain coherence in urban areas.

8.5 ANALYSIS METHODS

The baseline ground motion monitoring programme should be undertaken using archive satellite data where sufficient and appropriate stacks of data have already been captured by the satellites. A longer baseline provides a better chance of detecting and interpreting the causes of pre-shale gas ground motion as some deformation may be slow onset, while there may also be seasonal trends (such as shrink/swell) that need to be resolved. InSAR is a non-invasive method and images are selected that cover the site as well as the surrounding region. The baseline data can be validated by GNSS stations if they were operational within the image scene while the archive data were collected. This would also enable the InSAR data to be processed to absolute motion rather than a relative Line of Sight (LOS) motion. Pre-existing networks of GNSS receiving stations include the BGS/NERC British Isles continuous GNSS facility (BIGF) and the Ordnance Survey base station network (OS Net).

The measurements derived from the analysis include average velocity over the period of the satellite images (which could be several years) and a time series indicating the velocity of each point/pixel for each image. All measurements are relative to a stable reference point. It is important to ensure this reference point is within at least 20km of the shale gas site so that the data relates specifically to motion within that area. The average ground motion results are portrayed in digital datasets that are colour-coded to show speed of motion (uplift in blues and subsidence in reds) and degree of stability (for example with the range of 2 mm subsidence to 2 mm uplift per year considered stable, according to the standard deviation of the measurements).

The digital InSAR datasets should be displayed in map format, which must include a legend that explains the range of motion associated with each colour unit, a scale bar and a map projection grid (e.g. Figure 35). This type of map provides a general indication of the motion and can be misleading and even misrepresentative. Time-series plots for specific locations should also be included to highlight detailed motion within the average, ideally correlated with nearby GNSS data, if available (Figure 36). The average map and time series data should be accompanied by an interpretation that explains the motion and its causes.

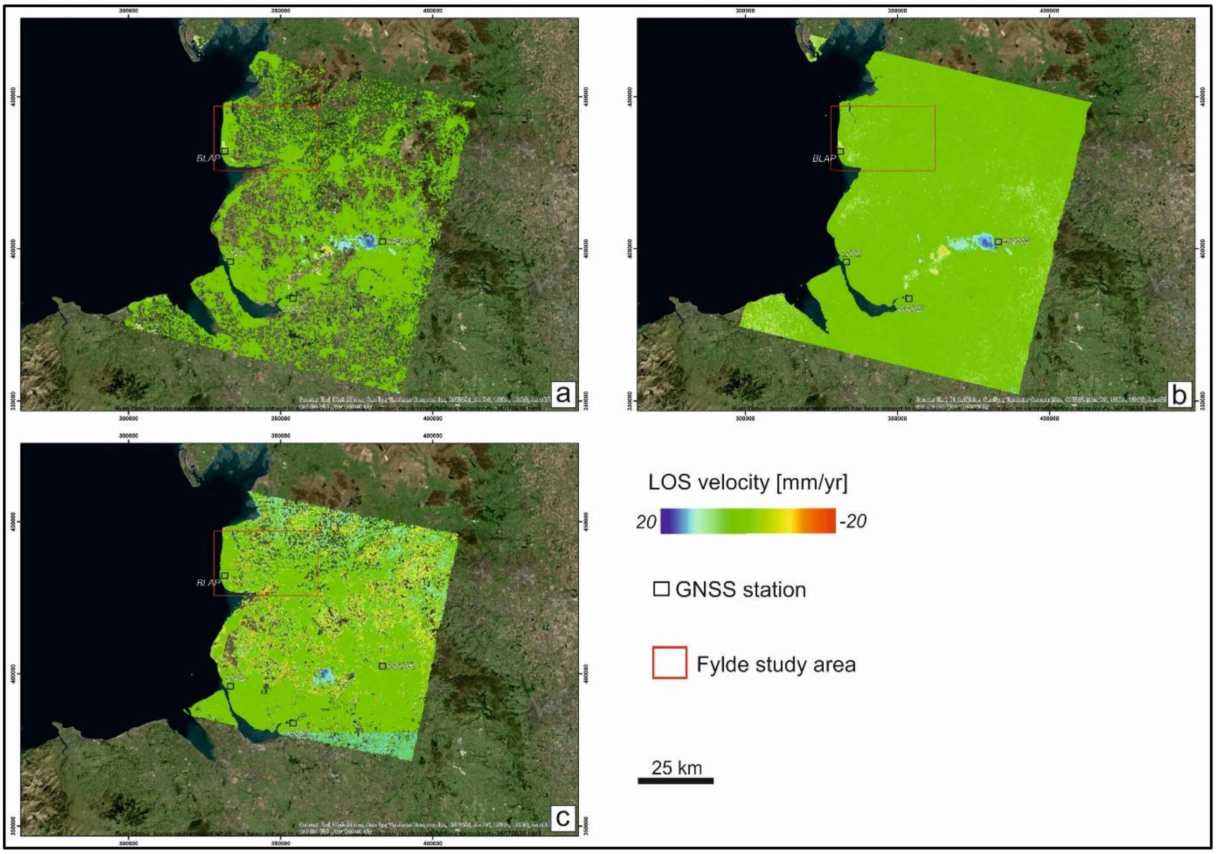


Figure 35. Representation of InSAR average velocity results; an example from Lancashire showing uplift (positive velocity) and subsidence (negative velocity). [A] 1992-2000 ERS SBAS InSAR average annual velocities. [B] 1992-2000 ERS ISBAS InSAR average annual velocities. [C] 2015-2019 Sentinel-1 RapidSAR Rural InSAR average velocity. Red box indicates Fylde study area

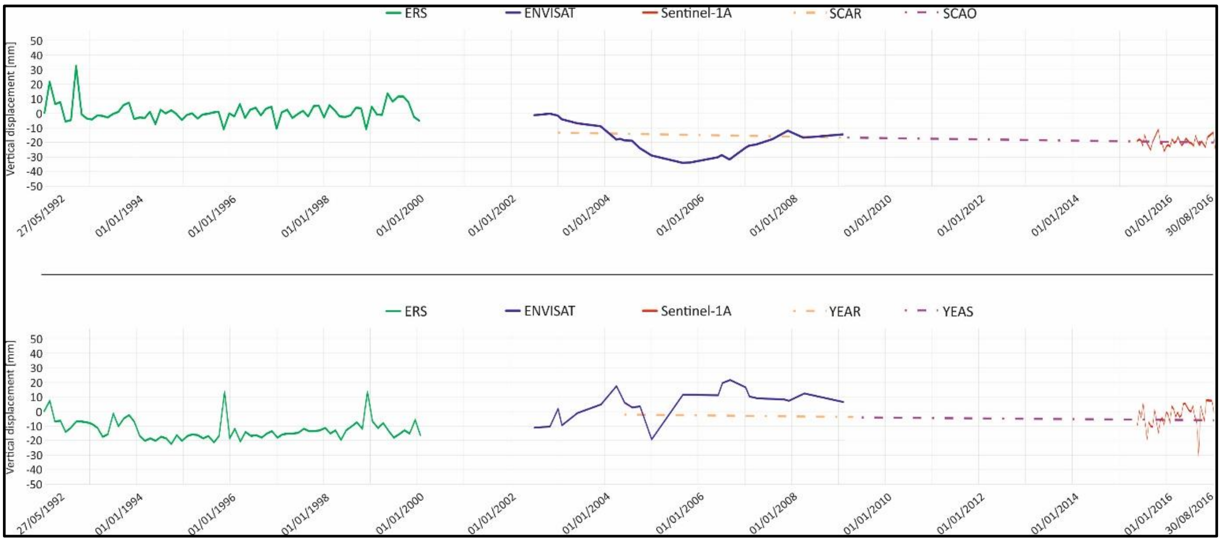


Figure 36. Example of non-linear time series for selected ISBAS points. The solid lines represent the ISBAS non-linear vertical displacements for different acquisitions and the dotted lines represent the GNSS linear and vertical displacements. It is worth noting that the InSAR time series reported were generated considering a linear displacement velocity in the temporal gaps between the ENVISAT and Sentinel-1A datasets. From Ward et al. (2017)

8.6 RECOMMENDATIONS FOR GROUND MOTION CHANGE DETECTION WITH INSAR

The InSAR process is recommended for ground motion detection and monitoring of site locations and surrounding regions. It is non-invasive and incorporates the ability to use archive data starting from 1992. Where GNSS data are available (even if only for part of the time period) they should be used to validate the InSAR results.

When it is known that shale gas operations are planned at a site, the options are i) utilise remote sensing (InSAR), ii) install a network of geodetic and/or geotechnical sensors, and iii) to use a combination of remote and in situ monitoring. Ongoing ground deformation monitoring would be undertaken using satellites that are in current orbit and collecting appropriate radar images e.g. Sentinel-1A/B or TerraSAR-X. The Sentinel-1 constellation collects data every 6 days over the UK. An automatic processing system should be utilised to deliver InSAR results in near-real-time i.e. each month or more regularly, where appropriate. A reference point should be selected within ~20 km of the shale gas site.

It is necessary to have at least 20 satellite images in a stack in order to achieve precise results, however with such a relatively low number account must be taken for atmospheric effects when interpreting the results. With 20 or more images, there is still a requirement to account for quality of processing. The processing result should be accompanied by:

- List of each image used in the process, including the satellite, sensor, acquisition date and geometry.
- Description of the processing chain e.g. coregistration, interferogram formation, point selection (threshold), phase unwrapping and identification of the reference point.
- Details of standard error of the velocities for each point/pixel
- Data Analysis

The output from the InSAR processing chain provides two sets of data i) average velocity over the time period of the satellite image acquisition and ii) time series that indicates the relative motion at the time when each image was acquired. These outputs illustrate the status of the ground i.e. whether it is stable or moving, but do not explain *why* the ground is moving (or stable). The ‘expert interpretation’ workflow in Figure 33 illustrates the geological analysis component that seeks to explain the most likely cause(s) of the motion. Interpretation of the InSAR results must be undertaken by experienced geoscientists who have access to a series of mandatory inputs; it is considered unviable to isolate or explain the causes and dynamics of ground motion without these inputs:

- Bedrock geology (incl faults)
- Surficial geology (incl. compressible ground)
- Historic mining information / plans
- Seismic records
- Groundwater abstraction records
- Borehole records
- Geohazard information (e.g. landslides and shrink/swell)
- Landcover information
- Historic topographic maps
- Aerial photography
- Digital elevation models
- Digital terrain models.

The interpretation is predominantly a manual process and is a derivation of expert elucidation. It is not considered possible to associate statistical analyses to the interpretation process yet but it is recommended that the organisation undertaking the interpretation utilises a cross-check where results are validated by another (in-house) expert. Statistical methods, machine learning and artificial intelligence are being developed and tested to automate the interpretation process and should be integrated where possible in the future.

Published interpretation procedures (e.g. Bateson et al, 2012 and Notti et al., 2015) should also be considered and followed, as appropriate. The interpretation should be accompanied by an explanation of the process or a reference to the publication that defines the procedure that was followed.

8.7 ACKNOWLEDGEMENTS

ERS-1 data were obtained via a Category-1 ESA project ID:13543 '*Enhancing landslide research and monitoring capability in Great Britain using C-band satellite SAR imagery and change detection, InSAR and Persistent Scatterers techniques'*. Sentinel-1 data have been obtained via the Copernicus Program of ESA. The authors would like to acknowledge ©Geomatic Ventures Ltd and SatSense Ltd. who processed the ISBAS and RapidSAR data respectively.

8.8 REFERENCES

Bateson L., Cuevas M., Crosetto M., Cigna., F., Schijf M., Evans J. (2012). Enabling access to geological information in support of GMES; Pangeo Deliverable 3.5 Production Manual Version 1.3. pp120 http://www.pangeoproject.eu/sites/default/files/pangeo_other/D3.5-PanGeo-Production-Manual-v1.3.pdf

Berardino, P., Fornaro, G., Lanari, R., Sansosti, E. (2002). A new algorithm for surface deformation monitoring based on small baseline differential SAR interferograms. IEEE Transactions on Geoscience and Remote Sensing, 40, 2375–2383.

Cigna, F., Bateson, L., Jordan, C J., Dashwood, C. (2014). Simulating SAR geometric distortions and predicting Persistent Scatterer densities for ERS-1/2 and ENVISAT C-band SAR and InSAR applications: nationwide feasibility assessment to monitor the landmass of Great Britain with SAR imagery. Remote Sensing of Environment, 152, 441–466, doi: 10.1016/j.rse.2014.06.025.

http://www.sciencedirect.com/science/article/pii/S0034425714002454

Crosetto, M., Monserrat, O., Iglesias, R., Crippa, B. (2010). Persistent scatterer interferometry: Potential, limits and initial C- and X-band comparison. Photogrammetric Engineering and Remote Sensing, 76, 1061–1069.

Dost, B., Caccavale M., van Eck T., Kraaijpoel D. (2013) Report on the expected PGV and PGA values for induced earthquakes in the Groningen area. KNMI December 2013.26pp

Ellsworth, W.L. (2013). Injection-Induced Earthquakes. Science, 341(6142). Available at: http://www1.gly.bris.ac.uk/bumps/secure/POTW/Ellsworth_2013.pdf

Ferretti, A., Fumagalli, A., Novali, F., Prati, C., Rocca, F., Rucci, A. (2011). A new algorithm for processing interferometric data-stacks: SqueeSAR. IEEE Transactions on Geoscience and Remote Sensing, 49(9), 3460-3470.

Ferretti, A., Prati, C., Rocca, F (2001). Permanent scatterers in SAR interferometry. IEEE Transactions on Geoscience and Remote Sensing, 39, 8–20

Gee D., Sowter A., Novellino A., Marsh S., Gluyas J. (2016) Monitoring land motion due to natural gas extraction: validation of the intermittent SBAS (ISBAS) DInSAR algorithm over gas fields of North Holland, the Netherlands. Marine and Petroleum Geology, 77 pp.1338-1354.

Geertsma J. (1973) Land subsidence above compacting oil and gas reservoirs. Journal of Petroleum Technology Vol 25 Issue 6. https://doi.org/10.2118/3730-PA

Holland, A.A. (2011). Examination of Possibly Induced Seismicity from Hydraulic Fracturing in the Eola Field, Garvin County, Oklahoma, Available at: http://www.ogs.ou.edu/pubsscanned/openfile/OF1_2011.pdf

Jordan C. J., Aldiss D., Bateson L. B., Cigna F., Garcia-Bajo M., Jones L., Jordan H., Turner P. (2013) GMES-service for assessing and monitoring subsidence hazards in coastal lowland areas around Europe. European Integration Methodology. SubCoast D3.5.6. British Geological Survey, 85pp. http://www.subcoast.eu/alfresco/d/d/workspace/SpacesStore/da5914bc-f15d-4993-8d98-06869670c85f/D3.5.2_D3.5.3%20European%20Integration%20v1_FINAL.pdf

Jordan C J, Bateson L, Glanfield G (2011) PANGEO: Enabling Access to Geological Information in Support of GMES; D3.1 Survey Team Requirements and Recommendations V2. *British Geological Survey Commissioned Report CR/11/117*. http://nora.nerc.ac.uk/19292/

Kim W. -Y., (2013). Induced seismicity associated with fluid injection into a deep well in Youngstown, Ohio. Journal of Geophysical Research: Solid Earth, 118(7), pp.3506–3518

Massonnet, D., Rossi, M., Carmona, C., Adragna, F., Peltzer, G., Feigl, K., Rabaute, T. (1993). The displacement field of the Landers earthquake mapped by radar interferometry. Nature 364, 138–142.

Mathieson, A., Midgley, J., Wright, I., Saoula, N., Ringrose P. (2010). In Salah CO2 Storage JIP: CO2 sequestration monitoring and verification technologies applied at Krechba, Algeria. Energy Procedia 00 (2010) 1063-00

Notti, D., Calò, F., Cigna, F., Manunta, M., Herrera, G., Berti, M., Meisina, C., Tapete, D., Zucca, F. (2015). A user-oriented methodology for DInSAR time series analysis and interpretation: Landslides and subsidence case studies. Pure and Applied Geophysics, 172(11), 3081-3105.

de Pater C.J., Baisch, D.S., (2011). Geomechanical Study of Bowland Shale Seismicity, Lichfield. Available at: http://www.cuadrillaresources.com/wp-content/uploads/2012/02/Geomechanical-Study-of-Bowland-Shale-Seismicity_02-11-11.pdf

Pepe, A. and Calò, F., (2017). A review of interferometric synthetic aperture RADAR (InSAR) multi-track approaches for the retrieval of earth's surface displacements. Applied Sciences, 7(12), p.1264. https://doi.org/10.3390/app7121264

Shirzaei M, Ellsworth W L, Tiampo K F, Gonzales P J,Manga M (2016) Surface uplift and time-dependent seismic hazard due to fluid injection in eastern Texas, Science 353 (6306), 1416-1419. Doi: 10.1126/science.aag0262

Sowter, A., Che Amat, M., Cigna, F., Marsh, S., Athab, A., Alshammari, L. (2016). Mexico City land subsidence in 2014–2015 with Sentinel-1 IW TOPS: first results using the Intermittent SBAS (ISBAS) technique. Int. J. Appl. Earth Obs. 52, 230–242.

Spaans, K. and Hooper, A. (2016). InSAR processing for volcano monitoring and other near-real time applications. Journal of Geophysical Research: Solid Earth, 121(4), 2947-2960. https://doi.org/10.1002/2015JB012752

Wilson, M. P., Davies, R. J., Foulger, G. R., Julian, B. R., Styles, P., Gluyas, J. G., Almond, S. (2015). Anthropogenic earthquakes in the UK: A national baseline prior to shale exploitation.