Dredged Material Disposal Site Monitoring Round the Coast of England: Results of Sampling (2022-2023)

North Tyne and Souter Point

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Issue Date: March 2023

Cefas Document Control

Submitted to:	Shaun Nicholson - MMO
Date submitted:	31st March 2023
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Approved by and date:	30/03/2023
Version:	V1.0

Version Control History			
Version	Author	Date	Comment
0.1	Stefan Bolam	5th January 2023	Initial draft prior to results
0.2	Stefan Bolam	20th March 2023	Inputting of results
0.3	Stefan Bolam / Charlotte Clarke	28th March 2023	Complete initial draft for internal QC
0.4	Stefan Bolam	31st March 2023	QC'd version submitted to MMO
1.0	Stefan Bolam	13th December 2023	Revised PSA section following query from Claire Mason

Executive Summary

This report presents the scientific findings of, and implications for subsequent monitoring based on the results from, dredged material disposal site monitoring conducted under a Cefas/Marine Management Organisation (MMO) Service Level Agreement (SLA 1.2) project (‘C6794’ hereafter) round the coast of England during 2022-2023 (financial year).

The main aims of this report are:

- to aid the dissemination of the survey results;
- to assess whether observed changes resulting from dredged material disposal are in line with predictions;
- to compare the results with those of previous years (where possible);
- to facilitate our improved understanding of the impacts of dredged material disposal at both a site-specific and a national (i.e. non site-specific) level.

Two disposal sites were targeted for assessment during this period: North Tyne (TY070) and Souter Point (TY081), both located off the northeast coast of England. A single survey was conducted to sample the two sites on 2nd December 2022 aboard the survey vessel *Coastal Surveyor*.

The survey at North Tyne aimed to acquire seabed sediment contaminants data within and around the disposal site to maintain a ‘watching brief’ for the site in accordance with previous recommendations made under this project. Seabed sampling at 12 stations at North Tyne revealed that sediment polycyclic aromatic hydrocarbons (PAHs), organohalogens (OHs) and trace metals displayed concentrations comparable with those observed in previous years for this site. These data indicate that the screening process as part of licencing dredged sediments for marine disposal is currently ensuring that sediments within and surrounding the disposal site in 2022 are within acceptable limits. These results, together with the fact that high tonnages of sediment are annually disposed of to North Tyne, indicate that the site should be considered for subsequent sampling every three to five years, unless a significant change to the disposal campaign materialises at any point.

The stations sampled at Souter Point were targeted to provide data to assess whether the sediment cap covering contaminated dredged material (CDM) disposed as part of a capping trial during 2004 and 2005 was showing any evidence of a breach. Surface sediment samples across nine stations within and one outside the capping region revealed PAHs, OHs and trace metals

all displayed concentrations more typical of material placed as part of subsequent routine maintenance dredged material disposed of to the site as opposed to representing CDM. These data infer that the cap in the locations where sampling was conducted in 2022 remains present and no breach is occurring. Disposal of some of the dredged material licenced for Souter Point should continue to be targeted at the capping region of the disposal site to provide continued assurance that the CDM continues to be maintained below the sediment surface.

The sediment particle size, organic carbon and contaminants concentration data of all stations sampled at both sites in 2022 are presented in this report. Where possible, analyses of temporal trends are presented to understand whether any parameter shows any longer-term trend as a result of the continued use of the site.

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1 Introduction

1.1 Regulation of disposal activity in England

Disposal of waste at sea is strictly regulated through the licensing requirements of the Marine and Coastal Access Act 2009 (MCAA). The MCAA provides the principal statutory means by which the UK complies with EU law, such as the Water Framework Directive (WFD, 2000/60/EC), the Habitats and Species Directive (92/43/EEC), the Wild Birds Directive (79/409/EEC), and international obligations such as under the OSPAR Convention and the London Convention and Protocol (LCLP), in relation to disposals at sea. Following the UK's departure from the EU at the end of 2020, the UK legislation transposing these EU Directives was amended to ensure it operated effectively following the UK's departure.

Pursuant to the OSPAR Convention and LCLP, only certain wastes or other matter are permitted for disposal at sea. During the 1980s and 1990s, the UK phased out sea disposal of most types of waste, including industrial waste and sewage sludge. Since then, dredged material from ports and harbours, and a small amount of fish waste, has been the only type of material routinely licensed for disposal at sea.

The Marine Management Organisation (MMO) regulates, and is responsible for, licensing activities in the marine environment around England, including the disposal of dredged material at sea. The MMO assesses the suitability of dredged material for disposal at sea in line with the OSPAR guidelines for the management of dredged material (OSPAR, 2014). These guidelines provide generic guidance on determining the conditions under which dredged material may (or may not) be deposited at sea and involve the consideration of alternative uses, disposal sites and the suitability of the dredged material for aquatic disposal including the presence and levels of contaminants in the material, along with perceived impacts on any nearby sites of conservation value.

One of the roles of Cefas is to provide scientific advice to the MMO on the suitability of the material for sea disposal at the application stage and, once a licence is granted, to provide technical advice on any monitoring undertaken as a result of licence conditions. Advice on the licensing of dredged material disposal at sea is provided by Cefas' Sustainable Environmental Assessment for Licencing (SEAL) team, work conducted under C6794 helps underpin the scientific rationale for such advice (see Section 1.3).

1.2 Disposal sites around England

There are currently approximately 218 open sites (numerous sites are opened and closed every year) designated for dredged material disposal round the coast of England, not all of which are used in any one year. While the majority of these are located along the coast of the mainland, generally within a few miles of a major port or estuary entrance, a significant number are positioned within estuaries (e.g. the Humber) or on intertidal mudflats as part of beneficial use schemes (Bolam et al., 2006).

Although total quantities vary year to year, approximately 40 Mt (wet weight) are annually disposed to coastal sites around England. Individual quantities licensed may range from a few hundred to several million tonnes, and the nature may vary from soft silts to stiff clay, boulders or even crushed rock according to origin, although the majority consists of finer material (Bolam et al., 2006).

1.3 Overview of Cefas / MMO project C6794 'Monitoring of dredged material disposal sites'

The dredged material disposal site monitoring project C6794, funded by the MMO, falls under a service level agreement (or SLA) between the MMO and Cefas. Operationally, this project represents a continuation of the disposal site monitoring programme SLAB5, which was a component of a former SLA between Defra and Cefas; this SLA formerly ceased at the end of March 2015. C6794 was initiated on 1st April 2015, and, thus, while the project and work planned under this project are termed here under C6794, any reference to its predecessor project is inevitable (i.e. to its survey work, reports or other scientific outputs), and will continue to be referenced herein as SLAB5.

In summary, C6794 provides field evaluations ('baseline' monitoring and 'trouble-shooting' surveys) at dredged material disposal sites around the coast of England. A major component of the project is, therefore, the commissioning of sea-going surveys at targeted disposal sites. Such field evaluations under C6794 are designed to ensure that:

- environmental conditions at newly designated sites are suitable for the commencement of disposal activities;
- predictions for established sites concerning limitations of effects continue to be met; and,
- disposal operations conform with licence conditions.

The outcomes of such surveys contribute, either directly or indirectly, to the licensing process by ensuring that any evidence of unacceptable changes or practices is rapidly communicated and acted upon by the MMO. As such, there are inherently strong links and ongoing discussions between the approaches and findings of this project with the work carried out by Cefas' SEAL team and the licensing team within the MMO. The scientific outcomes of the work undertaken within C6794 are circulated to the Cefas SEAL team and the MMO *via* a number of routes including peer-reviewed publications (including both activity-specific and site-specific findings), reports, direct discussions, and internal and external presentations. The production of this report forms an important element of such scientific communication. The current report, which presents the findings of work undertaken during 2022-23, constitutes the 15th in the series. The previous reports are accessible *via* the Defra website:

https://www.gov.uk/government/publications?departments%5B%5D=centre-for-environment-fisheries-and-aquaculture-science

It is not the purpose of this report to present a detailed appraisal of the processes giving rise to impacts (see Section 1.5) but to encapsulate the essence of the impacts associated with this activity at specific sites targeted within year.

1.4 Sites monitored

To aid with determining which disposal sites should be selected for sampling in any one year, Cefas has derived a tier-based approach that classifies a number of possible issues or environmental concerns that may be associated with dredged material disposal into a risk-based framework (Bolam et al., 2009; Birchenough et al., 2010). The issues that pertain to a disposal site, and where these lie within the tiering system (i.e., their perceived environmental risk) depict where that site lies within the tiered system. This ultimately determines whether that site is considered for sampling during a particular year. It is intended that this approach increases the transparency of the decision-making process regarding disposal site selection for C6794 monitoring, i.e., it establishes a model for site-specific decisions regarding sampling.

A tiered survey design and site assessment system, therefore, facilitates the prioritisation of dredged material disposal sites in terms of the need for, and the scale of, monitoring required at each site. In practice, this method provides a scientifically valid rationale for the assessment of risks associated with relinquished, current, and proposed disposal sites to the surrounding environment and amenities.

Two disposal sites were targeted for Cefas monitoring during 2022-23: North Tyne (TY070) and Souter Point (TY081), both located off the northeast coast of England. These sites were identified following consultation between Cefas' SEAL team and Cefas scientists in several key disciplines (e.g., benthic ecology, sediment contaminants), together with significant involvement from the MMO.

1.5 Aims of this report

This report does not aim to present a critique of the processes leading to observed changes at dredged material disposal sites around the coast of England. Such appraisals are conducted *via* other reporting routes, either *via* discussions with Cefas' SEAL team, presentations and subsequent publications at national and international conferences, and *via* papers in peer-reviewed journals (e.g., Bolam and Whomersley, 2005; Bolam et al., 2006; Birchenough et al., 2006; Bolam, 2014; Bolam et al., 2014a; Rumney et al., 2015; Bolam et al., 2016a; Bolam et al., 2021a). The aims of this report are:

- to present the results of sampling undertaken during 2022-23 under C6794, thereby aiding the dissemination of the findings under this project;
- to indicate whether the results obtained are in line with those expected for each disposal site, or whether subsequent investigations should be conducted;

- where possible, to compare the 2022-23 results with those of previous years to provide a temporal assessment (see Bolam et al., 2009; 2011a; 2012a; 2012b; 2014b; 2015a; 2015b; 2016b; 2017; 2018; 2019; 2020 and 2021b for reports of previous years' monitoring);
- to facilitate our improved understanding of the impacts of dredged material disposal at both a site-specific level and a national level; and,
- to promote the development of scientific (or other) outputs under C6794.

2 Outcomes

2.1 North Tyne (TY070)

2.1.1 Background

The North Tyne dredged material disposal site is a medium-sized, circular sector shaped site located several miles off the entrance to the Tyne Estuary. Sediment disposed of to North Tyne is made up of predominantly silt and sand. In the past, the site has received capital and maintenance dredgings (“capital” in this instance referring to material that has not previously been dredged within the last 10 years, and “maintenance” referring to material that is routinely dredged and disposed), minestone mine-tailings and fly-ash from power stations. An application for the disposal of significant quantities of capital material (up to 1.3 million wet tonnes) from the Tyne navigational channel and deepening of berths was licensed with disposal activity undertaken during 2011. The site annually receives large amounts of material (an average of 234,596 dry tonnes per annum since 2021; Figure 2.1) with recent peaks in 2018 (392,611 dry tonnes) and 2021 (409,841 dry tonnes).

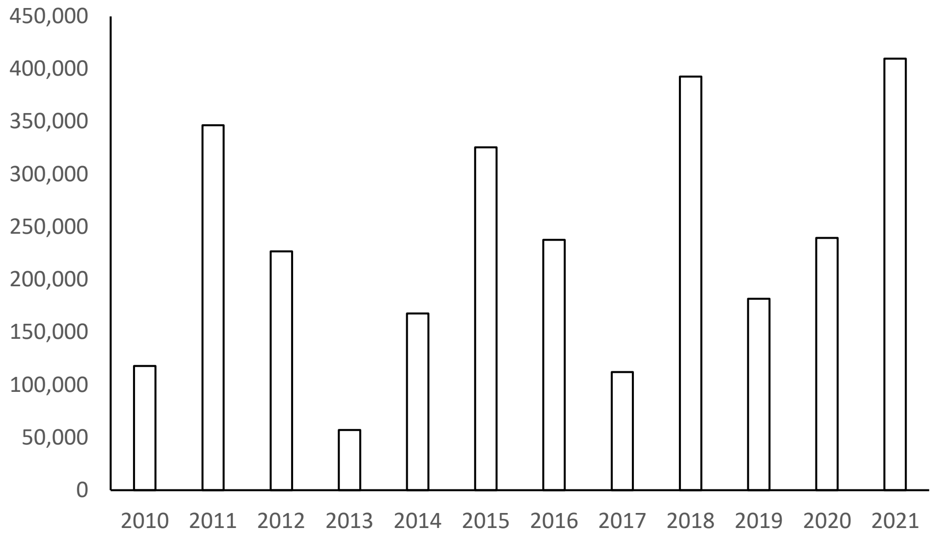


Figure 2.1. Annual amounts (dry tonnes) of dredged material disposed of to North Tyne, 2010-2021 (incl.).

The region's industrial background is associated with dredged material potentially destined for the North Tyne site being relatively contaminated in nature. For example, the mining industry has resulted in elevated levels of heavy metals, and historical ship-building on the Tyne, together with large volumes of shipping traffic in and out of the wharves, have contributed to a legacy of tributyl tin (TBT) and hydrocarbon contamination.

North Tyne was sampled and assessed for sediment contaminants under SLAB5 in 2013 whereupon it was found that all samples exceeded the (effects range low) ERL for low molecular weight (LMW) PAHs (polycyclic aromatic hydrocarbons), and samples from three stations exceeded the effects range median (ERM) for the LMW PAHs and the ERL high molecular weight (HMW) PAHs (Bolam et al., 2015). Regarding trace metals, enrichment relative to regional baseline concentrations was observed for a number of metals, especially Hg, Cd and Zn. The study concluded that future monitoring at North Tyne should continue, focussing primarily on assessing concentrations of PAHs, organohalogens (OHs) and trace metals. Sampling at the site under C6794 was last conducted during 2018 to provide a contemporary assessment of the concentrations of these chemicals within and surrounding the disposal site. The planned sampling at North Tyne in 2022 is in line with the recommendations of previous sampling activities wherein the chemical contaminants of the sediments of the site should be monitored on a regular basis (Bolam et al., 2019).

2.1.2 Survey Design

The seabed grabbing survey for sediment contaminants assessment at North Tyne was designed principally on previous surveys. The site has been sampled on a number of occasions and, to allow comparisons to be made between years, each survey has generally targeted the same stations. Thus, these stations may be regarded to represent a time-series and further sampling at the same stations allows a temporal assessment of the contaminants concentrations. Thus, the 12 stations (four inside and eight outside the disposal site boundary) sampled at North Tyne in December 2022 (Figure 2.2) are those which have been targeted mostly in previous survey campaigns. The grab sampling methods and subsequent sample processing adopted those of previous surveys to maximise the robustness of temporal assessments. A Day grab was deployed to sample the sediments and on retrieval, the surface sediments (top 2-3 cm) were carefully extracted using a stainless-steel spatula and placed in pentane-rinsed glass jars (for PAHs and OHs) or plastic bags (PSA, OCN and metals). A digital photograph image was taken of the grab sample prior to the removal of the surficial sediments. The grab buckets and sampling equipment (stainless-steel spatula) were wiped with 95% pentane between each sampling station. All samples were frozen (-20 degrees) within 24 hours of sampling for storage. The processing of all samples for sediment PSA, OCN, PAHs, OHs and metals were conducted in accordance with Cefas' procedures (see Appendices, further details on methods are available on request) and are in line with those conducted during previous dredged material disposal site surveys.

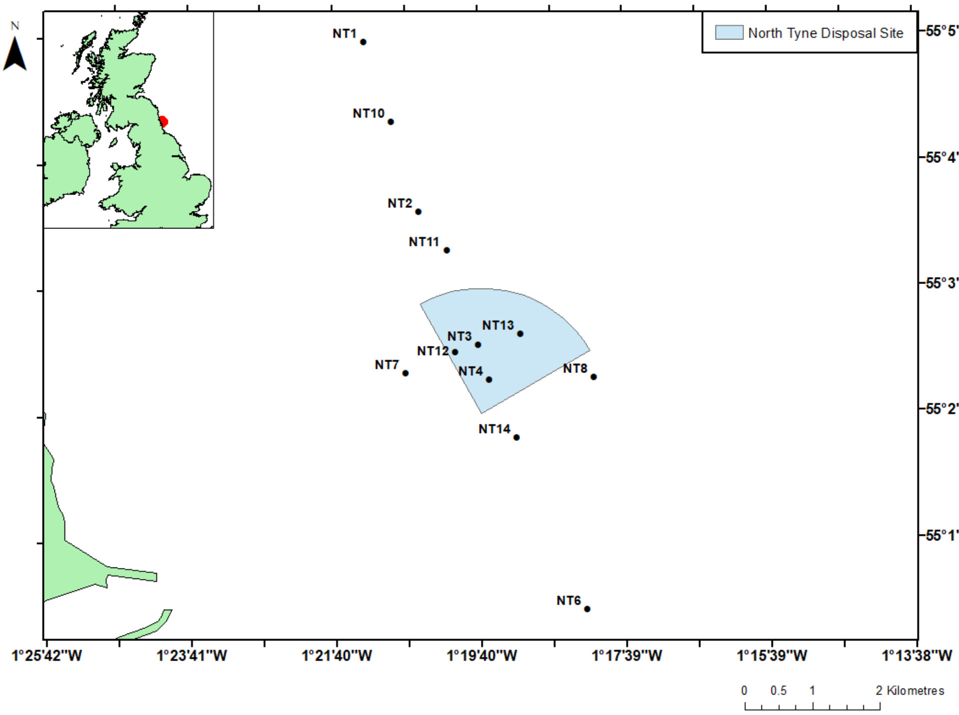


Figure 2.2. Location of the 12 stations sampled at North Tyne, December 2022.

2.1.3 Results

2.1.3.1 *Sediment particle size distribution*

North Tyne sediments are predominantly slightly gravelly muddy sands, with some stations being classed as mixed muddy sandy gravels (Table 2.1), this is based on the data from all temporal samples analysed between 2006 and 2022 (totalling 88 samples). Sediments analysed in 2022 were mainly slightly gravelly muddy sands (four in sediment group NoT1a, and two in NoT1b), with some muddy sandy gravels (four in NoT3a) (Table 2.1).

The spatial variation in the proportional representation of gravel, sand and silt/clay for each sampling station in 2022 as pie charts, and the percentages of silt/clay content, is shown in Figure 2.3. The highest silt/clay content was located immediately south of the disposal site at NT14 (~55%), with the next highest being inside the disposal site at NT4 (~41%). Sampling stations that contained greater than 15% silt/clay content are inside the disposal site at NT3 (~15%); north of the disposal site at NT1 (~27%), at NT2 (~26%) and NT11 (~19%); east of the site at NT7 (~21%) and west of the site NT8 (~17%). The rest of the samples contained < 15% silt/clay content.

Table 2.1. Average sediment descriptions and statistics for each sediment group at North Tyne (2006 to 2022 inclusive). Sediment descriptions and statistics (derived using Gradistat (Blott and Pye, 2001) are presented.

Sediment group	Number of samples	Sample Type	Sediment description	MODE 1 (µm):	MODE 2 (µm):	MODE 3 (µm):
NoT1a	25	Bimodal, Very Poorly Sorted	Slightly Gravelly Muddy Sand	107.5	26.7	
NoT1b	21	Unimodal, Poorly Sorted	Slightly Gravelly Muddy Sand	152.5		
NoT2	21	Unimodal, Poorly Sorted	Gravelly Sand	302.5		
NoT3a	11	Polymodal, Very Poorly Sorted	Muddy Sandy Gravel	2400.0	107.5	215.0
NoT3b	10	Trimodal, Very Poorly Sorted	Muddy Sandy Gravel	26950.0	1700.0	215.0

Sediment group	Gravel (%)	Sand (%)	Silt/clay (%)	Very coarse sand (%)	Coarse sand (%)	Medium sand (%)	Fine sand (%)	Very fine sand (%)
NoT1a	3.62	63.13	33.25	2.31	3.71	9.05	22.12	25.96
NoT1b	2.28	80.06	17.65	1.45	4.41	20.03	36.32	17.85
NoT2	5.13	85.70	9.17	2.60	6.26	34.22	30.75	11.87
NoT3a	38.16	47.55	14.29	11.28	7.90	9.31	10.07	8.99
NoT3b	43.95	46.57	9.49	11.03	7.13	9.25	11.25	7.90

Most of the temporal changes in sediment groups for stations since 2006 occur within the disposal site at NT3 and NT4, as well as immediately north of the site at NT11 (fewer years represented here), and to a lesser extent at NT13 (also within the disposal site); as well as at NT2 (north of the site), while minimal changes are noted at NT1 and NT10 (north of the site), NT12 (just inside the site), NT6 (just south of the site) and for both NT7 and NT8 (just outside of the site)(Table 2.2). This is also the situation observed at NT2 and NT10 (north of the site), while minimal changes are evident at NT1 (furthest north of the site), NT12 (just inside the site), NT6 (just south of the site) and for both NT7 and NT8 (just outside of the site) (Table 2.2). The greater temporal variation in sediment granulometry at stations within or close to the disposal site may perhaps reflect changes in the nature of sediments being disposed, or variations caused by the differences in the proximity of samples to recent disposal events, as indicated in previous years.

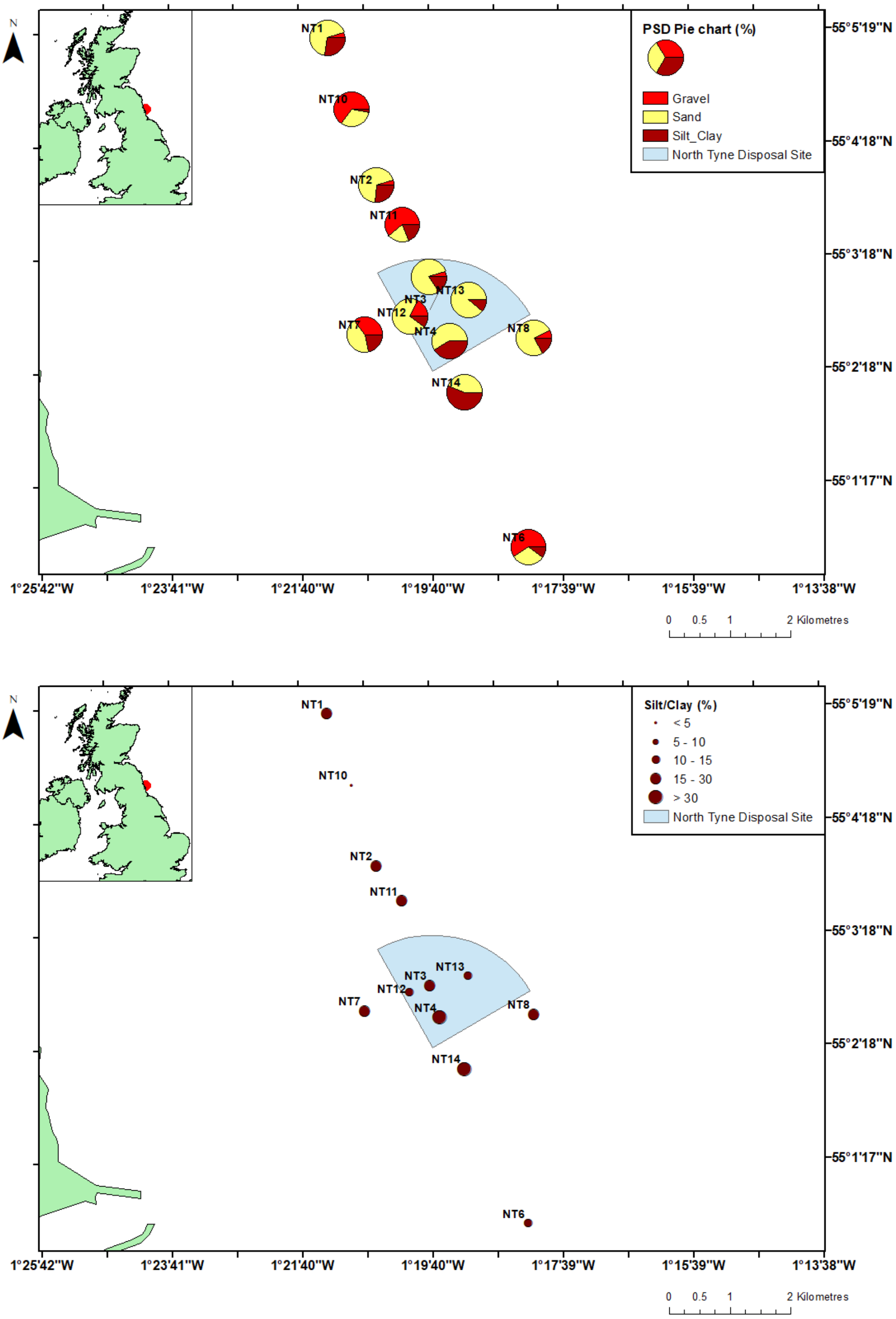


Figure 2.3. Particle size distribution of the sediments sampled at the 12 stations at North Tyne, December 2022. Pie chart of gravel, sand and silt/clay composition (top panel) and % composition of the silt/clay fraction as a bubble map (lower panel).

Table 2.2. Sediment groups for each sample code between 2006 and 2022 inclusive at North Tyne. n = number of samples measured. Only sample codes collected in 2022 shown.

	Year									
Sample code	2006	2007	2008	2009	2010	2011	2012	2013	2018	2022
NT1	NoT1a	NoT1a	NoT1a	NoT1a	NoT1a	NoT1a	NoT1a	NoT1a	NoT1b	NoT1a
NT2	NoT1a	n	n	NoT3a	NoT1a	NoT1a	NoT1a	NoT1a	NoT1b	NoT1a
NT3	NoT1b	NoT2	NoT2	NoT1b	NoT2	NoT1b	NoT2	NoT1a	NoT1b	NoT1b
NT4	NoT1a	NoT2	NoT2	NoT1b	NoT1a	NoT1b	NoT1b	NoT1b	NoT1b	NoT1a
NT6	NoT3b	NoT3a	NoT3b	n	NoT3b	n	n	NoT3b	NoT2	NoT3a
NT7	NoT3b	NoT3b	n	n	NoT3a	n	NoT3a	n	n	NoT3a
NT8	NoT2	NoT2	NoT2	NoT2	NoT1b	NoT1b	NoT3a	NoT2	NoT2	NoT2
NT10	n	n	n	n	n	n	NoT3b	NoT3a	NoT3a	NoT3a
NT11	n	n	n	n	n	n	NoT1a	NoT1b	NoT2	NoT3a
NT12	n	n	n	n	n	n	NoT1b	NoT2	NoT2	NoT2
NT13	n	n	n	n	n	n	NoT2	NoT1a	NoT2	NoT1b
NT14	n	n	n	n	n	n	n	n	n	NoT1a

2.1.3.2 Sediment organic carbon content

Organic carbon values ranged from 1.1 to 7.2 % m/m in the <2mm sediment fraction (Figure 2.4), and from 2.1 to 4.4 % m/m in the <63µm fraction (Figure 2.4). These results are similar to those observed between 2006 and 2022.

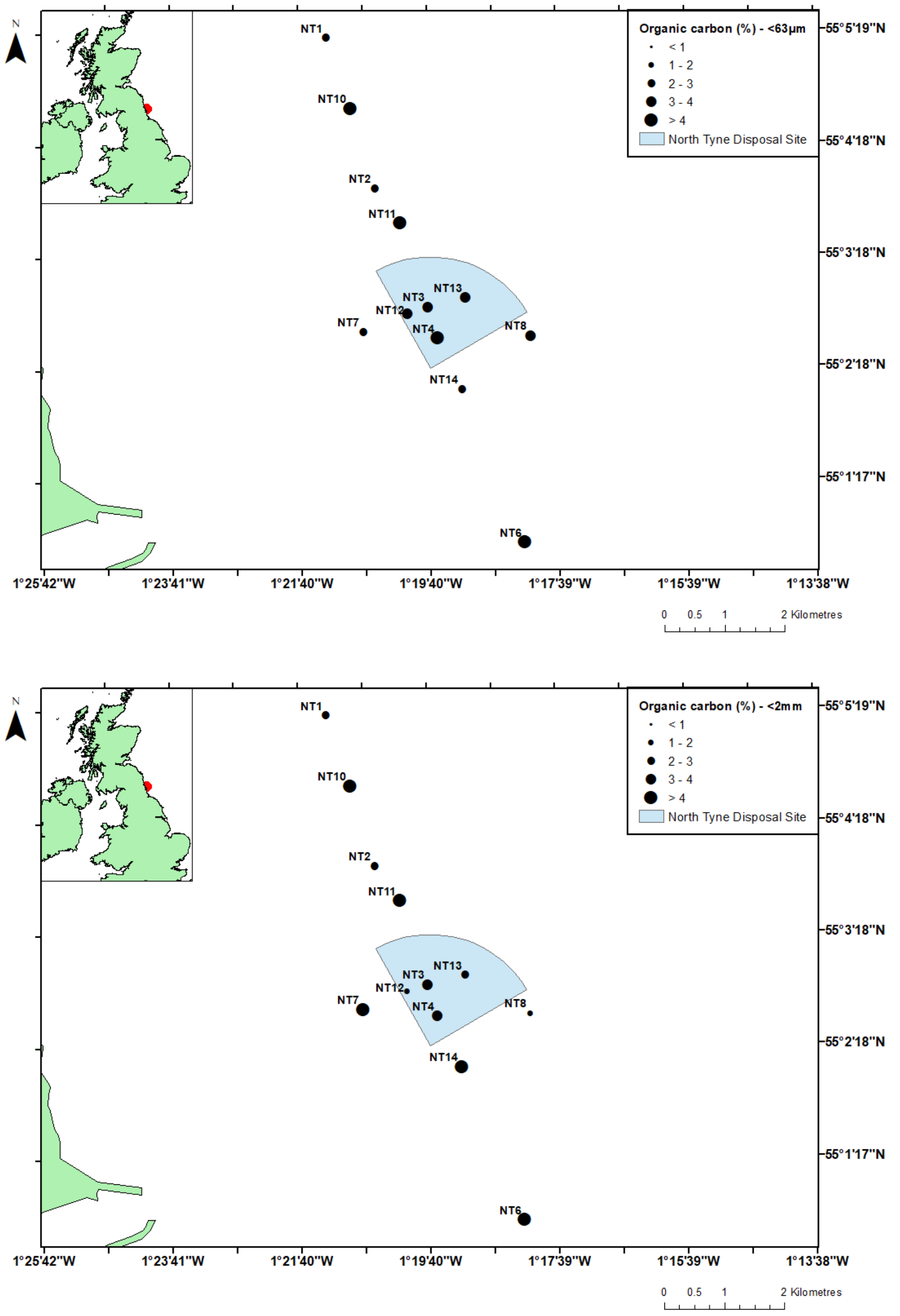


Figure 2.4. Sediments organic carbon content of the sediments sampled at the 12 stations at North Tyne, December 2022. Carbon content of the less than 63 µm fraction (top panel) and on the less than 2 mm fraction (lower panel).

2.1.3.3 Sediment PAHs concentrations

The highest summed PAH concentration at North Tyne in 2022 was 89,400 µg kg^{-1} dry weight (dw) found at NT6 (Figure 2.5), approximately 3 km south southeast of the disposal site (Figure 2.6). NT6 was also sampled in 2013 (52,000 µg kg^{-1} dw) and 2018 (44,100 µg kg^{-1} dw) indicating a notable rise in PAH concentration at this station since 2018. The highest concentration found in 2018 (Bolam et al., 2019) was 66,100 µg kg^{-1} dw (Figure 2.7) at NT15 (which was not sampled in 2022) which is in a very similar geographical location to NT6, being 2 km southeast of the disposal site. This infers that the highest PAH concentrations in this region are generally observed to the southeast of the North Tyne disposal site.

The second highest summed PAH concentration of 2022 was 55,600 µg kg^{-1} dw (Figure 2.5) at NT3 which is within the disposal site boundary (Figure 2.6). This station has been tested nine times since 2007 and this is the highest PAH concentration found to date, more than two times higher than the previous survey 2018 (23,900 µg kg^{-1} dw; Figure 2.7).

The lowest summed PAH concentration in 2022 was 5,030 µg kg^{-1} dw at NT12, also located within the disposal site (Figure 2.5, Figure 2.6). This represents a substantial decline compared with the concentration in 2018 (21,700 µg kg^{-1} dw; Figure 2.7). The lowest summed PAH concentration in 2018 was 8,460 µg kg^{-1} dw at NT2 to the north northwest of the disposal site; in 2022 this increased to 11,500 µg kg^{-1} dw (Figure 2.5). The stations to the north of the disposal site remain among the lowest PAH concentrations of all the stations sampled (Figure 2.6) which is consistent with previous years.

The summed PAH concentrations observed in 2022 at NT4, NT8, NT10, NT11, NT12 and NT13 showed reduced levels compared with 2018 (Figure 2.7). However, stations NT1, NT2, NT3, and NT6, all had elevated concentrations in 2022 compared with 2018. NT14, in close proximity to the southern boundary of the disposal site, was sampled for the first time in 2022 and showed similar levels of PAH to the nearby NT4 located just inside the site (Figure 2.6).

All samples collected at North Tyne during 2022, in harmony with the situation in 2018, exceeded the ERL (effects range low) for low molecular weight (LMW) PAHs. In 2022, sediments from NT3, NT4, NT6, NT7, NT14 all exceeded the ERM (effects range median) for the LMW PAHs. In 2018, sediments from NT10, NT12 and NT13 exceeded the ERM for the LMW but in 2022 the ERM was not exceeded at these stations. In 2022, sediments from NT3, NT4, NT6, NT7, NT13 and NT14 exceeded the ERL for the high molecular weight (HMW) PAHs. In 2018 sediments from NT10, NT12 and NT13 exceeded the ERL for the HMW but in 2022 the ERM was not exceeded at these stations. In 2022 one station, NT3, exceeded the ERM for HMW PAHs which is in slight contrast to 2018 and 2013 where no sampled station exceeded the ERM for HMW PAHs.

Evaluation of the PAH data indicated that the source in all the collected sediment samples from North Tyne was predominantly petrogenic with >68% of the PAH content arising from oil rather than combustion sources, except for site NT3 where 59% of the PAH content arose from combustion sources.

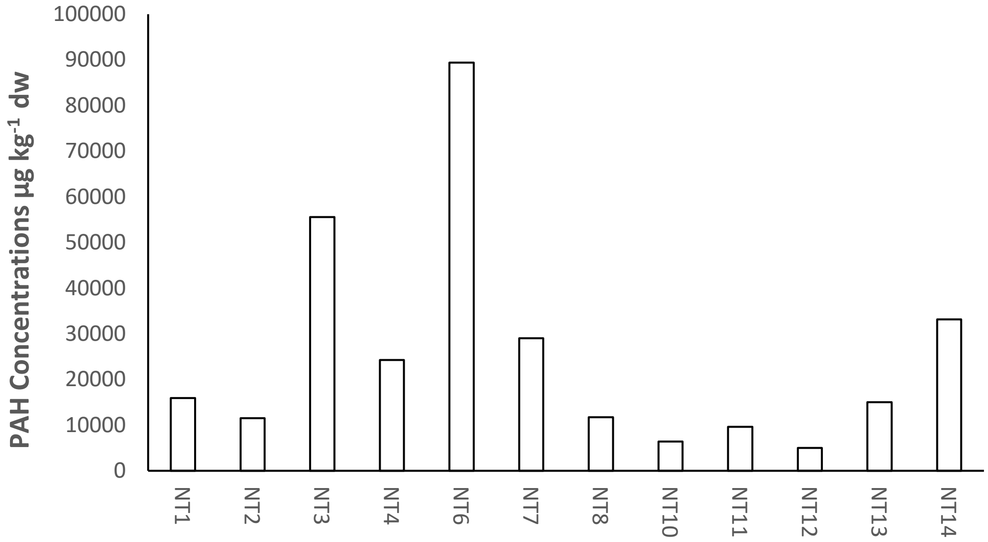


Figure 2.5. Summed PAH concentrations in the sediments sampled at each of the 12 stations at North Tyne, December 2022.

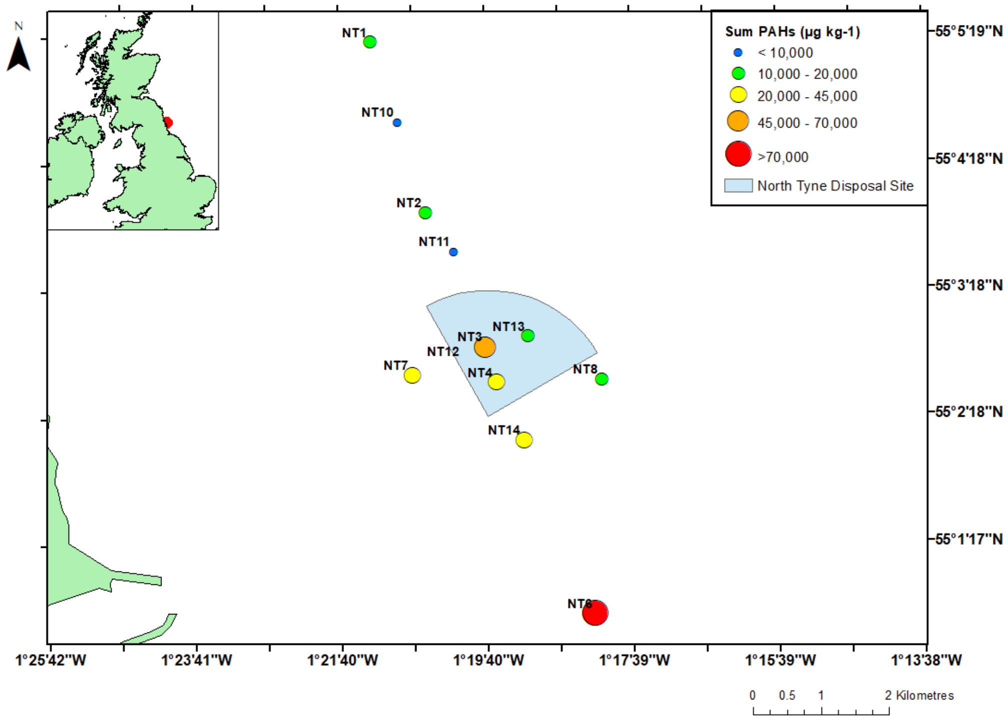


Figure 2.6. Map of summed PAH concentrations in the sediments sampled at each of the 12 stations at North Tyne, December 2022.

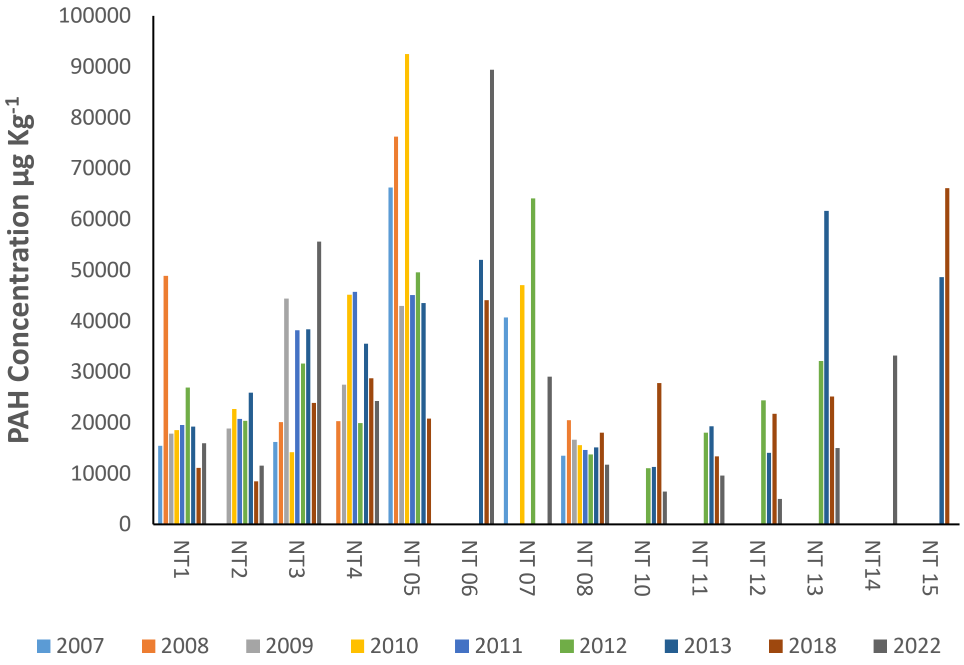


Figure 2.7. Temporal trends in summed PAH concentrations in the sediments sampled at each of the stations at North Tyne, 2007-2022.

2.1.3.4 *Sediment OH concentrations*

At North Tyne, chlorobenzenes (CBs) were detected at all 12 stations sampled (∑ICES7 CBs range 0.70-3.90 µg/kg dw). Concentrations of CBs were lowest at NT13 inside the disposal site, with slightly higher concentrations at NT1, NT2, NT10 to the north and NT08 to the east (Figure 2.8). The highest CB concentration was found at NT4 inside the disposal site (∑ICES7 CBs 3.90 µg/kg dw) with similar concentrations found at NT11 to the north (∑ICES7 CBs 3.70 µg/kg dw) and NT14 to the south (∑ICES7 CBs 3.60 µg/kg dw) of the disposal site, respectively (Figure 2.8).

BDEs (brominated diphenyl ethers) were detected at all stations (∑11 BDEs range 0.47-5.10 µg/kg dw) and, similar to that observed for CBs, the lowest concentrations were at NT13 and NT1, with a low value also at NT3 (Figure 2.9). The highest concentrations of 5.10 and 4.30 µg/kg dw were found at NT4 and NT1 within the disposal site. BDE47 and BDE99 are the dominant congeners present, indicative of the pentaBDE technical mixture, but BDE183 was also detected, suggesting that the octaBDE or decaBDE technical mixture was also in use. Two stations (NT7 and NT12) to the west of the disposal site had much higher levels of BDE183 than other stations (Figure 2.9). Penta and octa technical mixtures are no longer in use, having been banned in the EU since 2004.

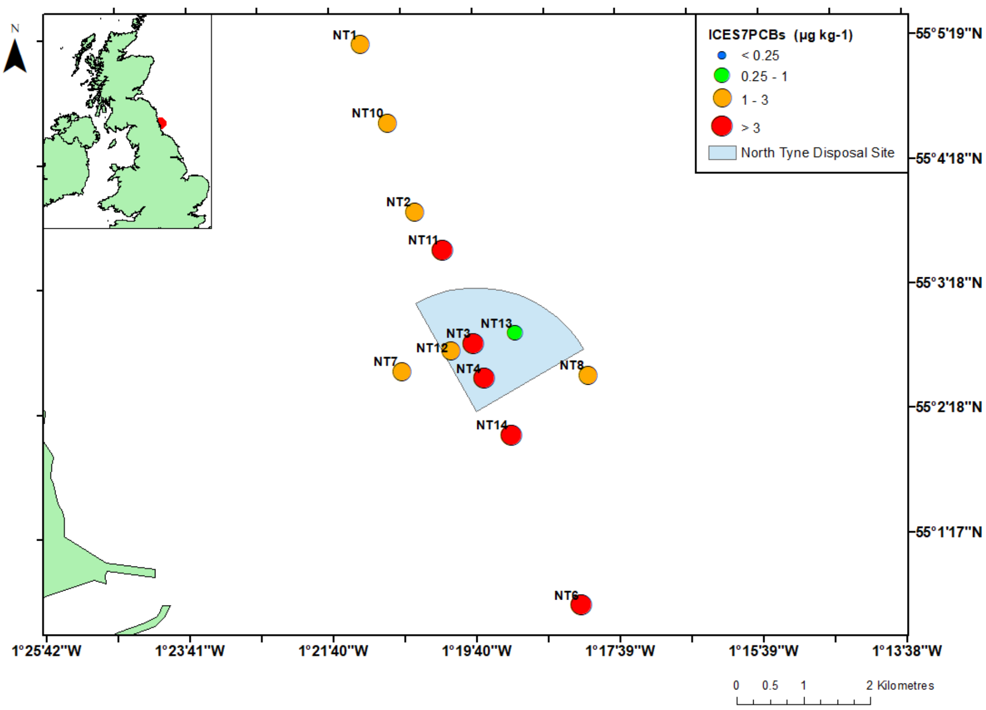


Figure 2.8. ∑ ICES7 CB concentrations for the North Tyne stations, 2022.

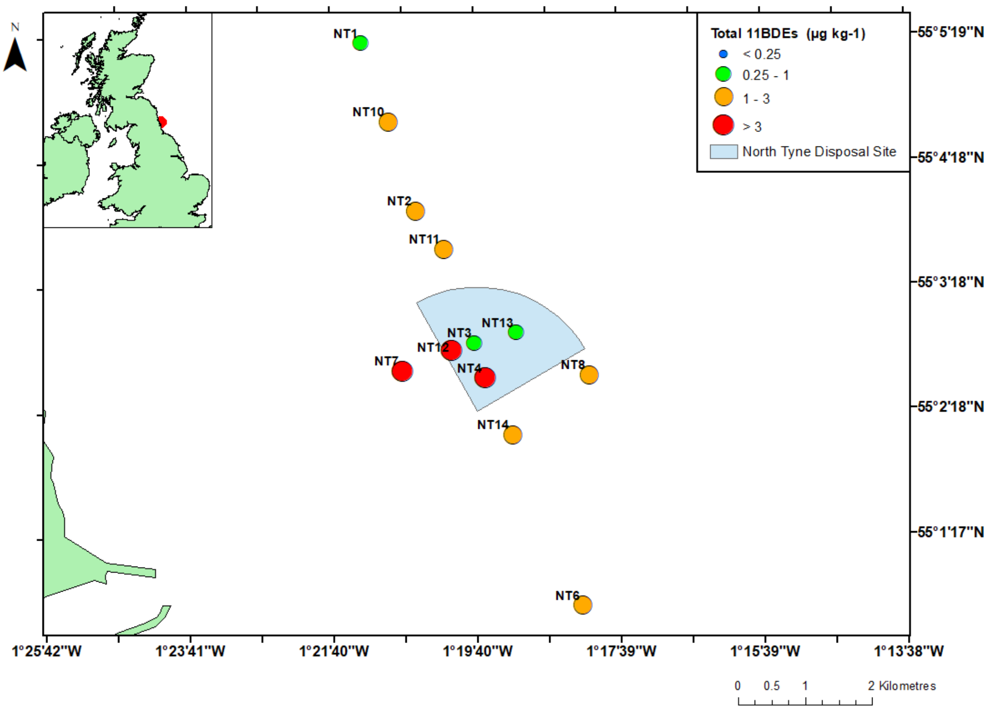


Figure 2.9. ∑ 11 BDEs concentrations for the North Tyne Stations, 2022.

BDE209 was detected at all stations and was at higher concentrations than the other measured OHs (range 5.90-90.00 µg/kg dw; Figure 2.10). When included with the other BDEs, BDE209 made up >61% of the BDEs present (range 61-95%, or 83-95% excluding NT7 and NT12, which had highest levels of BDE183). BDE209 is indicative of the decaBDE technical mixture, which had been in use more recently than the other technical mixtures, although it's use too has been restricted in the EU since 2008. High concentrations of 90.00, 18.90 and 18.80 µg/kg dw were found at NT4, NT3 and NT11, respectively, which are within or just north of the disposal site (Figure 2.10). The next highest concentrations of 17.70 and 17.00 µg/kg dw were found at NT3 and NT14, respectively.

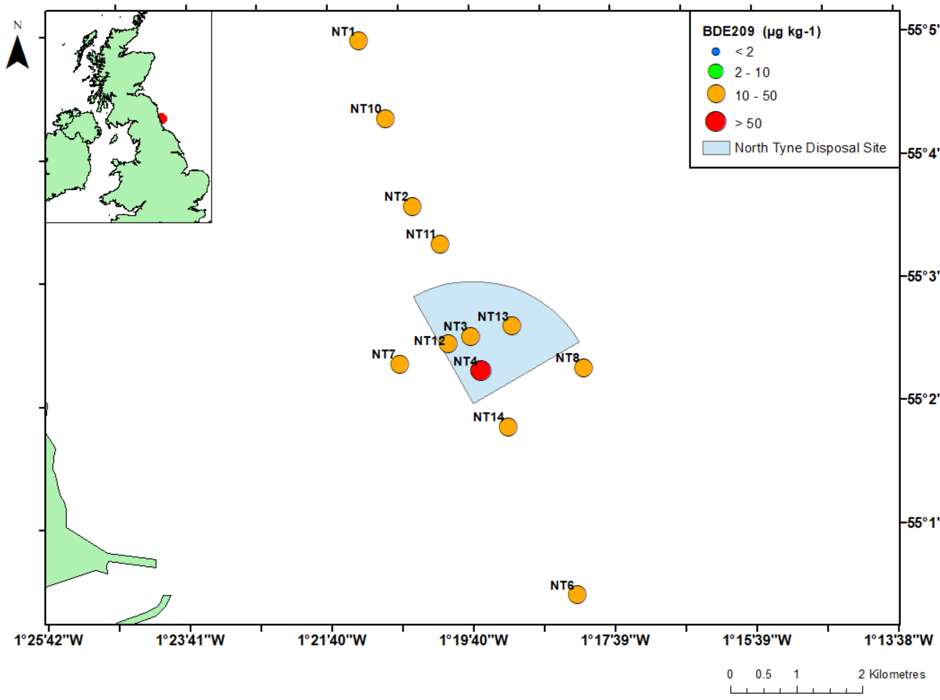


Figure 2.10. BDE209 concentrations for the North Tyne Stations, 2022.

The OCPs HCB, plus DDT and metabolites were detected at all stations (Figure 2.11). $\sum_6$DDTs concentrations ranged from 0.78-3.30 µg/kg dw, with the highest values at NT11 (3.30 µg/kg dw), NT14, NT4 and NT6 (all 2.70 µg/kg dw). The ratio of DDT/metabolites at all stations is much less than 1, suggesting that this sediment has not received fresh inputs of DDT. HCB concentrations ranged from 0.21 – 0.51 µg/kg dw, with the highest values at NT11 (0.51 µg/kg dw) and NT06 (0.49 µg/kg dw). Dieldrin was detected at 6 out of 12 stations (range <0.10-0.44 µg/kg dw), with the highest values at NT04 (0.44 µg/kg dw), NT11 (0.38 µg/kg dw) and NT07 (0.37 µg/kg dw).

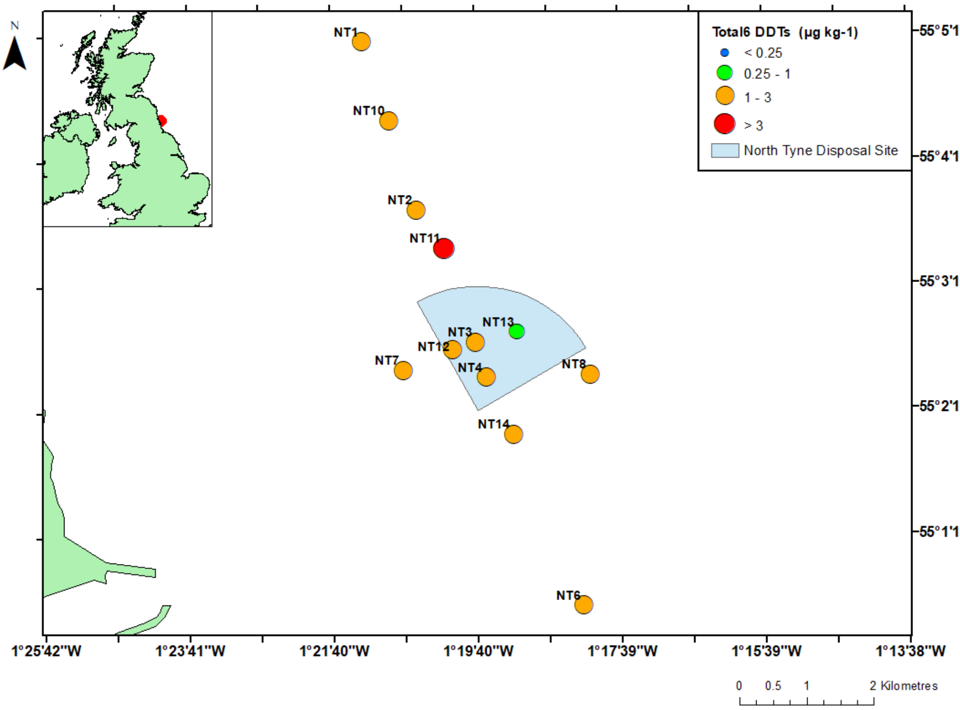


Figure 2.11. Total DDT concentrations for the North Tyne Stations, 2018.

Concentrations of CBs at all stations were all below Cefas Action Level (AL) 1, as was dieldrin. Total DDT concentrations were above Cefas AL 1 at all stations except NT13. No Cefas ALs exist for BDEs including BDE209. According to the OSPAR guidelines, eight of the sampled stations had 'good' environmental status for all ICES 7 CBs and BDEs. The exceptions were NT4, which was above the EACs for BDE99 and BDE209, and NT12, which was above the EACs for CB118 and BDE99.

There are data available to analyse at temporal trends in OHs from 2006 to 2022 (see Table 2.3 to Table 2.6). For ∑ICES 7 CBs, most stations showed a decrease in concentration since the previous sampling campaign at North Tyne in 2018. The exceptions were NT11, NT4 and NT6, which increased. Trends for ∑11 BDEs showed an equal number of samples increasing as that decreasing. The stations that were higher in 2022 than 2018 were NT11, NT12, NT4 and NT6. BDE209 in most stations was lower in 2022 than in 2018, with NT11, NT4 and NT6 again being the stations that increased. ∑3 DDTs concentrations were either very close to what was measured in 2018 or decreased. NT6 showed a very significant order of magnitude decline from the high value recorded in 2018 (Table 2.6).

Table 2.3. Temporal trends (2006-2022) of ∑ICES 7 CBs concentration (in µg/kg dw) at North Tyne.

	∑ICES 7 CBs concentration (in µg/kg dw)									
	2006	2007	2008	2009	2010	2011	2012	2013	2018	2022
NT1	1.54	0.97	2.11	0.93	0.98	1.13	1.37	1.34	1.18	1.05
NT10							2.63	6.62	3.23	1.11
NT2				1.69	1.63	2.66	2.67	4.72	2.96	1.27
NT11							2.89	11.0	1.72	3.70
NT12							17.0	*0.7	2.20	1.92
NT3	1.48	2.03	1.79	4.12	1.63	11.9	3.39	7.80	6.08	3.54
NT4	7.21		*0.7	4.58	11.0	5.84	2.63	4.63	2.88	3.93
NT13							1.61	10.4	1.15	0.71
NT14										3.62
NT8	5.21	2.03	0.81	*0.7	1.12	1.10	1.13	*0.7	1.31	1.11
NT5	2.70	7.59	6.05	3.24	11.9	2.72	3.88	1.37	2.31	
NT15								5.47	4.20	
NT6	2.44	2.54	3.88		2.09			3.36	2.71	3.55
NT7		1.55			1.76		2.24			2.56

*Concentrations represent estimates of concentrations for samples where all ICES 7 congener concentrations were below LODs. Limits of detection for CBs improved between 2013 and 2018, resulting in a step decrease in ∑7 CBs concentration for samples with congeners below LODs.

Table 2.4. Temporal trends (2006-2022) of ∑11 BDEs concentration (in µg/kg dw) at North Tyne.

	∑11 BDEs concentration (in µg/kg dw)									
	2006	2007	2008	2009	2010	2011	2012	2013	2018	2022
NT1	1.56	1.68	1.27	0.95	0.93	1.92	2.45	2.27	0.79	0.81
NT10							2.61	5.09	3.24	1.22
NT2	5.28			1.27	1.84	2.12	2.30	5.74	5.27	1.08
NT11							2.99	1.58	0.99	2.07
NT12							0.67	0.40	1.41	4.26
NT3	1.72	1.54	0.49	2.55	0.52	1.84	1.60	6.80	1.36	0.68
NT4	13.2		0.28	1.27	4.18	7.10	1.48	4.12	1.36	5.15
NT13							0.92	9.54	0.68	0.47
NT14										1.60
NT8	1.86	2.84	1.42	0.74	1.65	1.40	3.98	1.78	1.19	1.26
NT5	2.18	4.49	0.96	5.89	1.31	2.10	2.45	1.23	1.50	
NT15								4.49	2.12	
NT6	7.69	4.12	1.18		1.34			1.61	1.62	2.27
NT7		1.77			0.84		1.98			4.22

Concentrations in italic represent estimates of concentrations for samples where all 11 BDE congener concentrations were below LODs. Limits of detection for BDEs improved between 2007 and 2008 and again between 2013 and 2018, resulting in step decreases in ∑11 BDEs concentration for samples with congeners below LODs.

Table 2.5. Temporal trends (2008-2022) of BDE209 concentration (in µg/kg dw) at North Tyne.

	BDE209 concentration (in µg/kg dw)							
	2008	2009	2010	2011	2012	2013	2018	2022
NT1	104.00	11.50	3.93	7.33	5.56	12.80	8.59	3.92
NT10					16.50	15.70	26.00	12.00
NT2		12.20	12.20	42.90	11.90	69.50	22.90	5.87
NT11					12.50	12.50	10.60	18.80
NT12					7.70	2.43	47.20	6.77
NT3	2.72	48.50	7.91	21.60	35.40	185.00	49.30	18.90
NT4	0.78	36.10	95.50	108.00	38.10	152.00	87.40	90.00
NT13					45.30	206.00	10.20	8.26
NT14								17.00
NT8	8.03	8.95	20.10	7.21	4.91	14.40	10.40	7.76
NT5	6.21	11.90	6.64	10.60	273.00	6.28	22.10	
NT15						12.60	23.00	
NT6	6.15		8.69			12.80	15.80	17.70
NT7			17.50		8.46			7.60

Table 2.6. Temporal trends (2006-2022) of ∑3 DDTs concentration (in µg/kg dw) at North Tyne.

	∑3 DDTs concentration (in µg/kg dw)					
	2006	2007	2008	2010	2018	2022
NT1	0.97	1.54	1.44	0.83	0.71	0.94
NT10					2.16	0.99
NT2	1.56			1.38	1.79	1.11
NT11					1.36	2.86
NT12					2.15	1.05
NT3	0.89	1.83	3.17	1.27	1.03	1.22
NT4	7.20		1.12	6.92	2.25	2.30
NT13					0.84	0.64
NT14						2.36
NT8	0.84	1.59	0.88	1.36	0.96	0.93
NT5	1.12	3.41	3.48	1.25	1.52	
NT15					5.39	
NT6	1.25	2.25	1.84	0.91	83.8	2.34
NT7	1.12	1.66		0.95		1.97

∑DDTs is the sum of 3 chemicals (*p,p'*-DDE, *p,p'*-TDE, *p,p'*-DDT). Also note limits of detection for individual DDTs improved between 2011 and 2021 by a factor of 2-10, depending on chemical.

2.1.3.5 Sediment metals concentrations

Levels of metals enrichment for the sampled stations at North Tyne using the OSPAR BAC and regional baseline values are represented in Figure 2.12 to Figure 2.19. Assessment of the metal enrichment shows either no or slight enrichment for arsenic (As), with no enrichment within disposal site based on OSPAR and baseline assessment methods (a slight reduction in enrichments compared with the previous data from 2018). Stations NT4 and NT13 (both sampled in 2018) are now showing no enrichment, whilst previously were both displaying slight enrichment based on OSPAR and no and slight enrichment respectively based against baseline assessment. There is still a slight enrichment for As along the north of disposal site, while southern stations are displaying no enrichment for As based on both assessment methods. No clear temporal trend was observed from 2006-2022 for As at stations within and outside the disposal site (Figure 2.20).

Chromium (Cr) and nickel (Ni) were slightly enriched with the OSPAR assessment at all stations in 2022 (like that in 2018). Sampling stations generally showed no enrichment when using the baseline approach for both Cr and Ni (NT3 displays slight enrichment in the inner disposal site for Cr by using the baseline assessment method, an increase from no enrichment in 2018). Using the baseline assessment approach, Ni shows no enrichment at all the stations outside the disposal site. In 2018 some slight enrichment was observed at several of these reference stations (NT1, NT2 and NT6, all of which were sampled in this survey displayed no enrichment). Overall, Ni and Cr generally displayed a reduction in concentration enrichments compared with previous surveys based on the baseline assessment method but remained slightly enriched using the OSPAR assessment criterion. This reflects the background levels of these two metals in this region exceeding OSPAR BACs. No clear temporal trend was observed for Ni and Cr from 2006-2022 (Figure 2.20).

Both assessment methods depict the same observations for cadmium (Cd) as the OSPAR and baseline BACS equate to similar values giving identical enrichment values. They indicate mainly moderate Cd enrichment inside disposal site (Figure 2.18). The same pattern was observed during 2018 sampling, with the same stations giving identical enrichments in both years. Outside the North Tyne disposal site there has been a slight reduction in enrichments with the north and the south of the site now displaying no enrichment apart from one station to the south (NT14) showing slight enrichment.

Copper (Cu) displayed slight enrichment at all stations in 2022 for both OPSAR and baseline assessments (Figure 2.14). One station (NT3) in disposal site boundary displayed moderate enrichment, this being a slight increase from 2018 when a slight enrichment was witnessed. Outside the disposal site, the moderate enrichments from 2018 (albeit at some different stations) have now reduced to slight enrichments both north and south of the disposal site. No apparent temporal trend is observed over 2006-2022 dataset (Figure 2.20).

Mercury (Hg) was moderately enriched according to the OSPAR approach for all stations in 2022, except NT3 which displayed high enrichment (Figure 2.15). This reflects a reduction in enrichments compared to 2018, when one high enrichment station within the disposal site was observed, together with high enrichments both north and south of the disposal site (Bolam et al., 2019). However, with the exception of 'slightly enriched' at NT3 inside the disposal site, no enrichment was observed in 2022 based on the baseline assessment method. This again represents an overall reduction in enrichments compared to 2018 using this assessment method.

Lead (Pb) displayed mostly moderate enrichments based on the OPSAR assessment approach, with one highly enriched station inside the disposal site (NT3) (Figure 2.17). This latter station was previously classed as moderately enriched in 2018 using this assessment method (Bolam et al., 2019). Outside the disposal site there was a general decrease in enrichments compared to that observed in 2018, with two stations (NT1, NT7) being slightly enriched when all stations outside the disposal site were previously moderately enriched. When assessing Pb against the regional baseline values, slight or no enrichment was evident in 2022 (Figure 2.17). The disposal site stations showed a similar enrichment pattern compared to those in 2018, and a general trend of decreasing enrichment ratio values for those outside the disposal site.

A similar enrichment pattern has been observed with zinc (Zn) for the stations inside the disposal site for both the OSPAR and baseline assessments methods compared to 2018. Some reductions in enrichments outside the disposal site has been observed compared to the 2018 data, this is more evident for the baseline assessment which displays no enrichments for Zn outside the disposal site in any of the stations sampled in 2022 (Figure 2.19).

In conclusion, the metals concentrations tend to be much higher than the OSPAR BAC values especially for Hg, and Pb and to a lesser extent Zn. When assessing those concentrations against the proposed baseline values which were derived considering regional variability, enrichment is still observed, but to a reduced level. Cd (and to a lesser extent Zn) had a noticeably higher enrichment inside the disposal site compared to outside, reference stations. Enrichment values for all other trace metals were comparable between stations inside and outside the disposal site. Meanwhile, arsenic displays lower values inside the disposal site compared to outside, although the associated enrichments are only slight. In general, there has been a reduction in enrichments observed outside the disposal site and no significant change to enrichment levels observed within disposal site for trace metals. Despite this, there has been no evident temporal trend observed with any metals either inside or outside the disposal site (Figure 2.20).

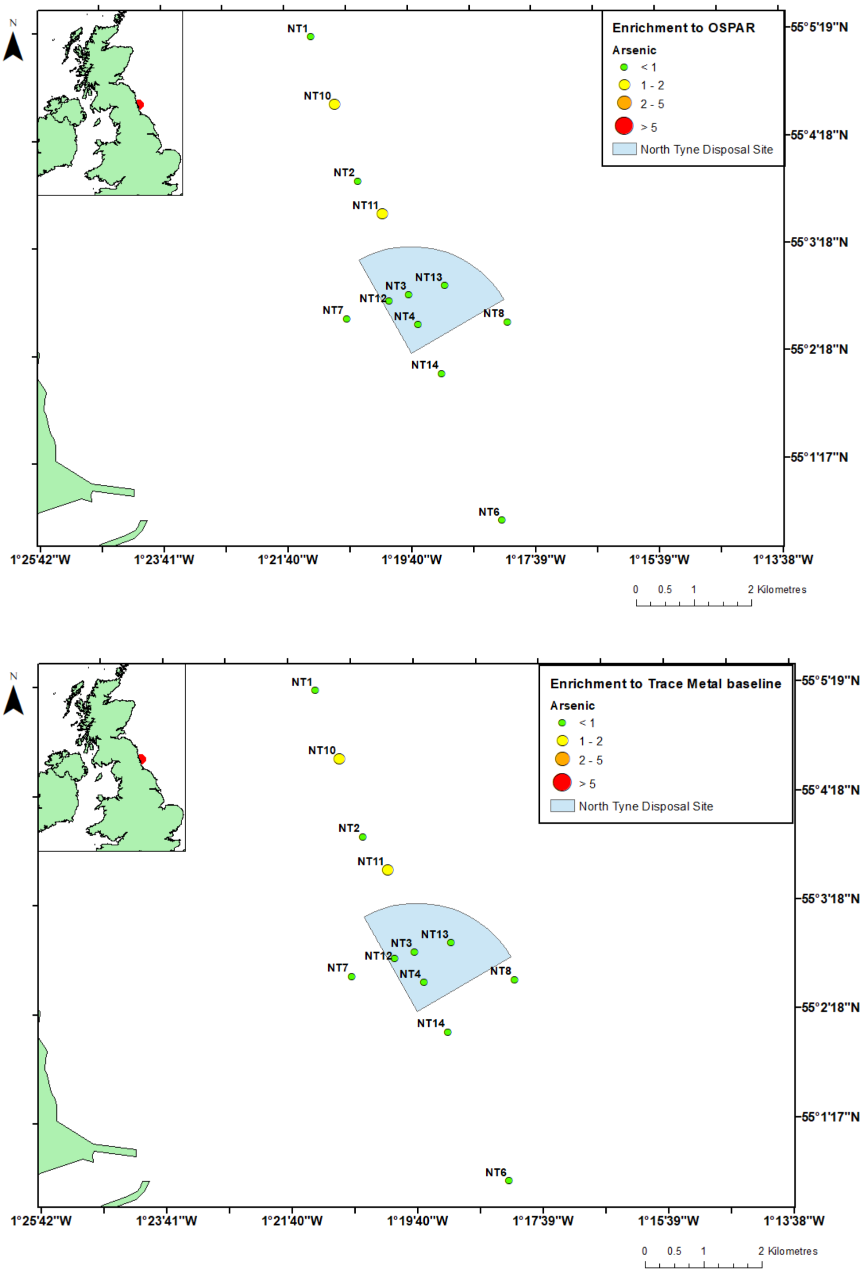


Figure 2.12. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for arsenic at North Tyne, December 2022.

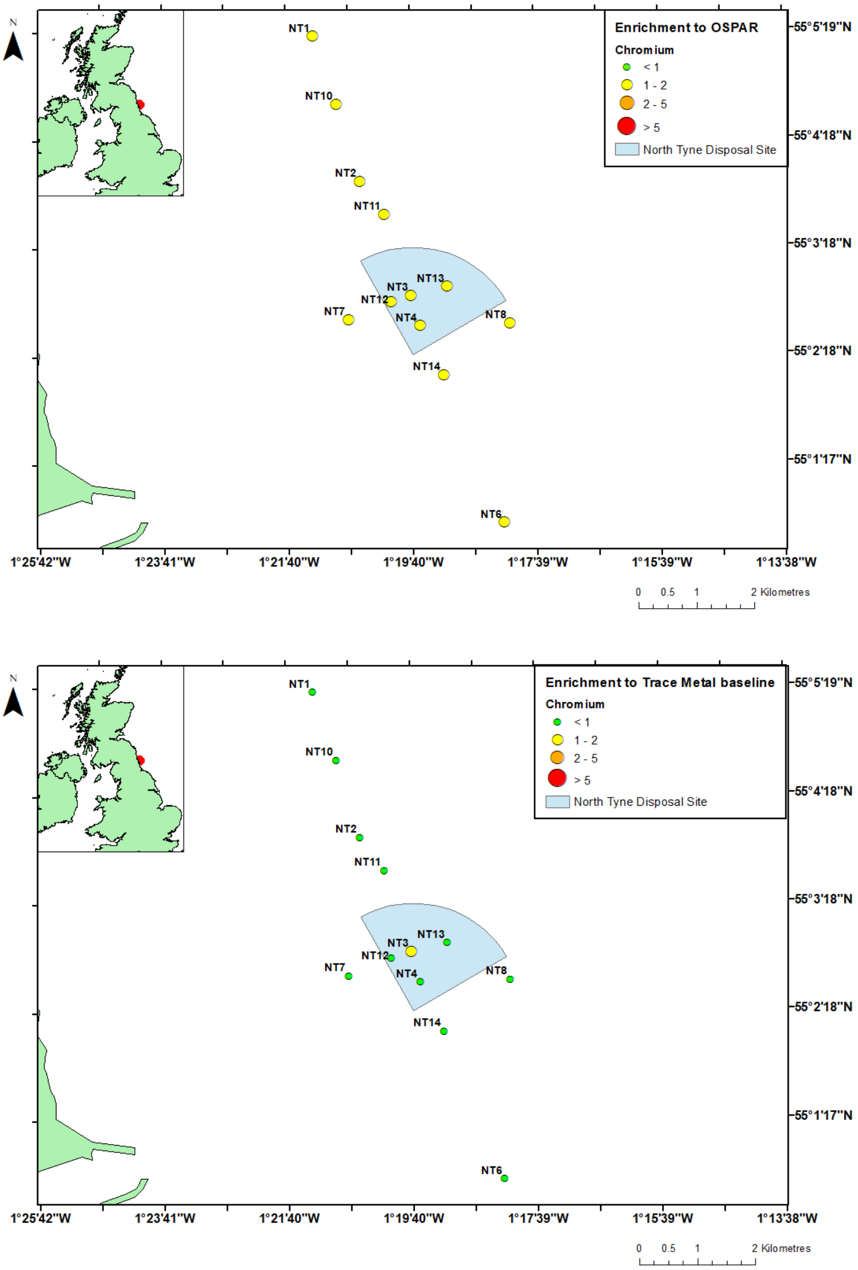
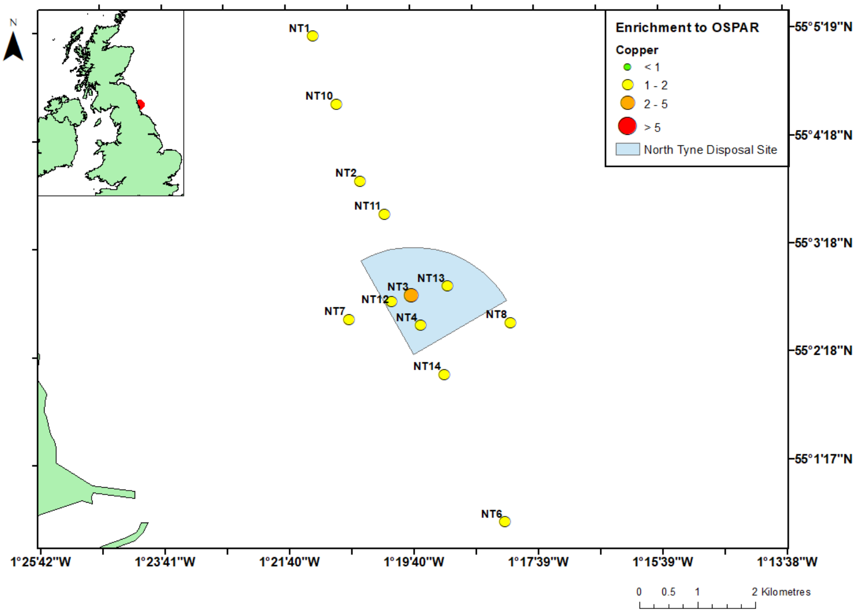


Figure 2.13. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for chromium at North Tyne, December 2022.



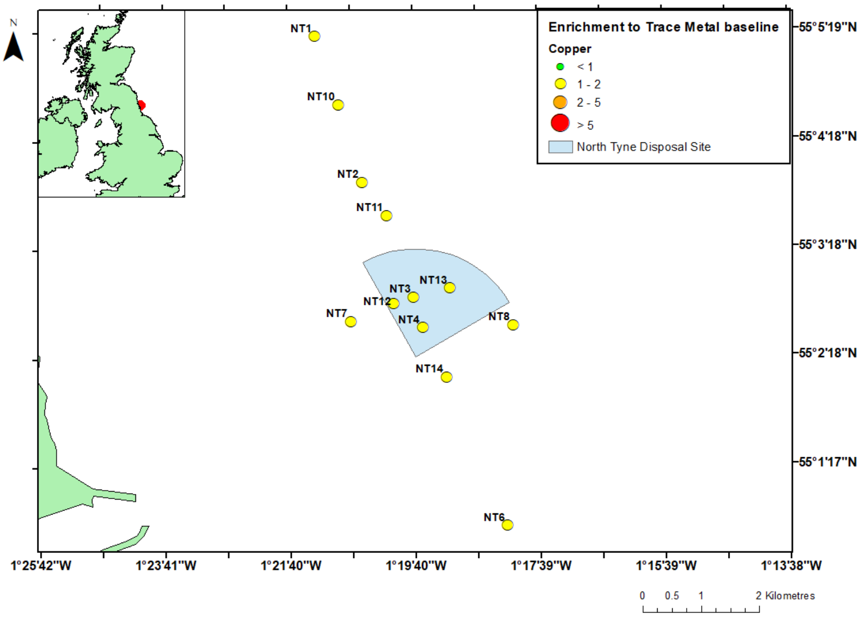


Figure 2.14. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for copper at North Tyne, December 2022.

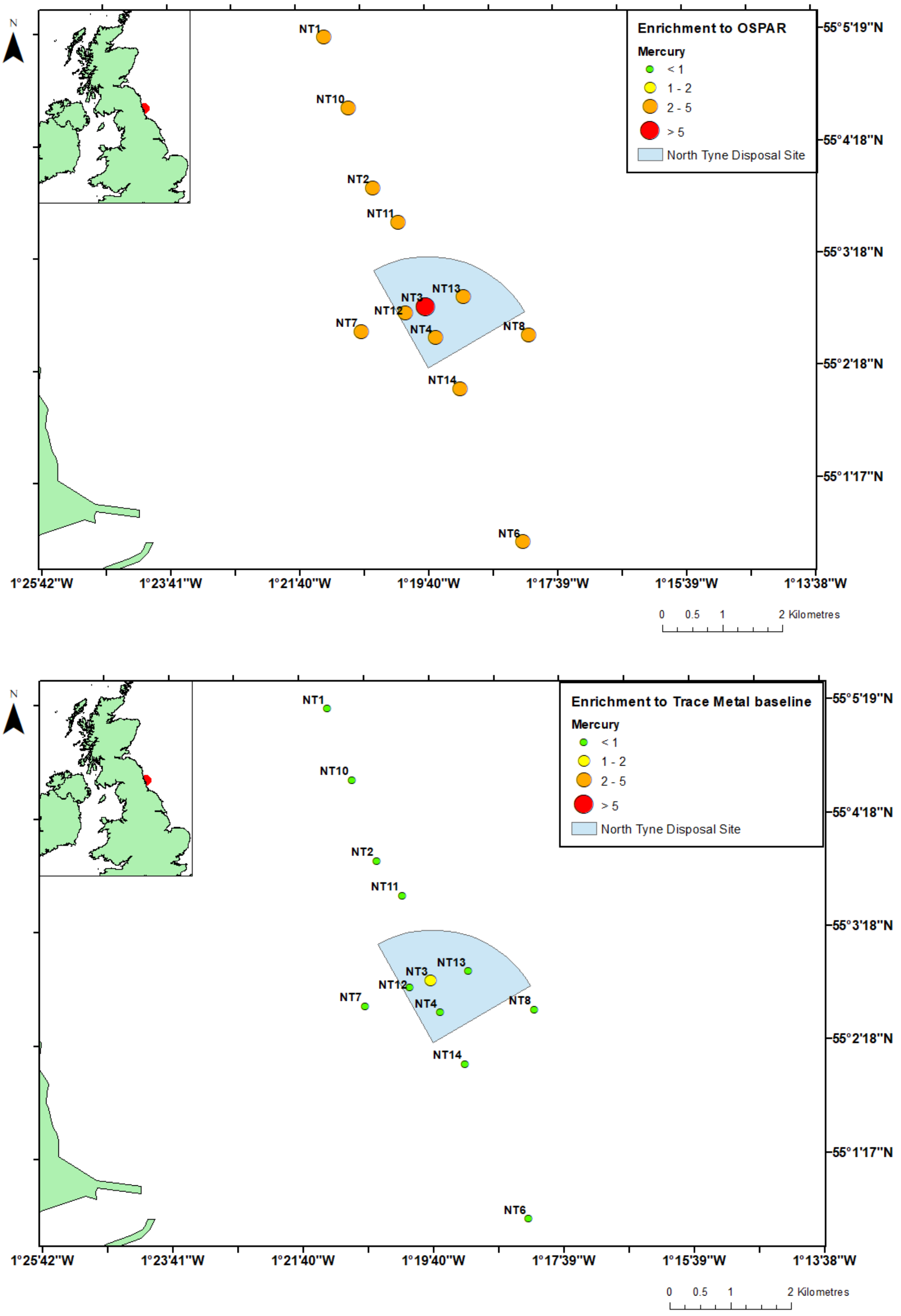


Figure 2.15. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for mercury at North Tyne, December 2022.

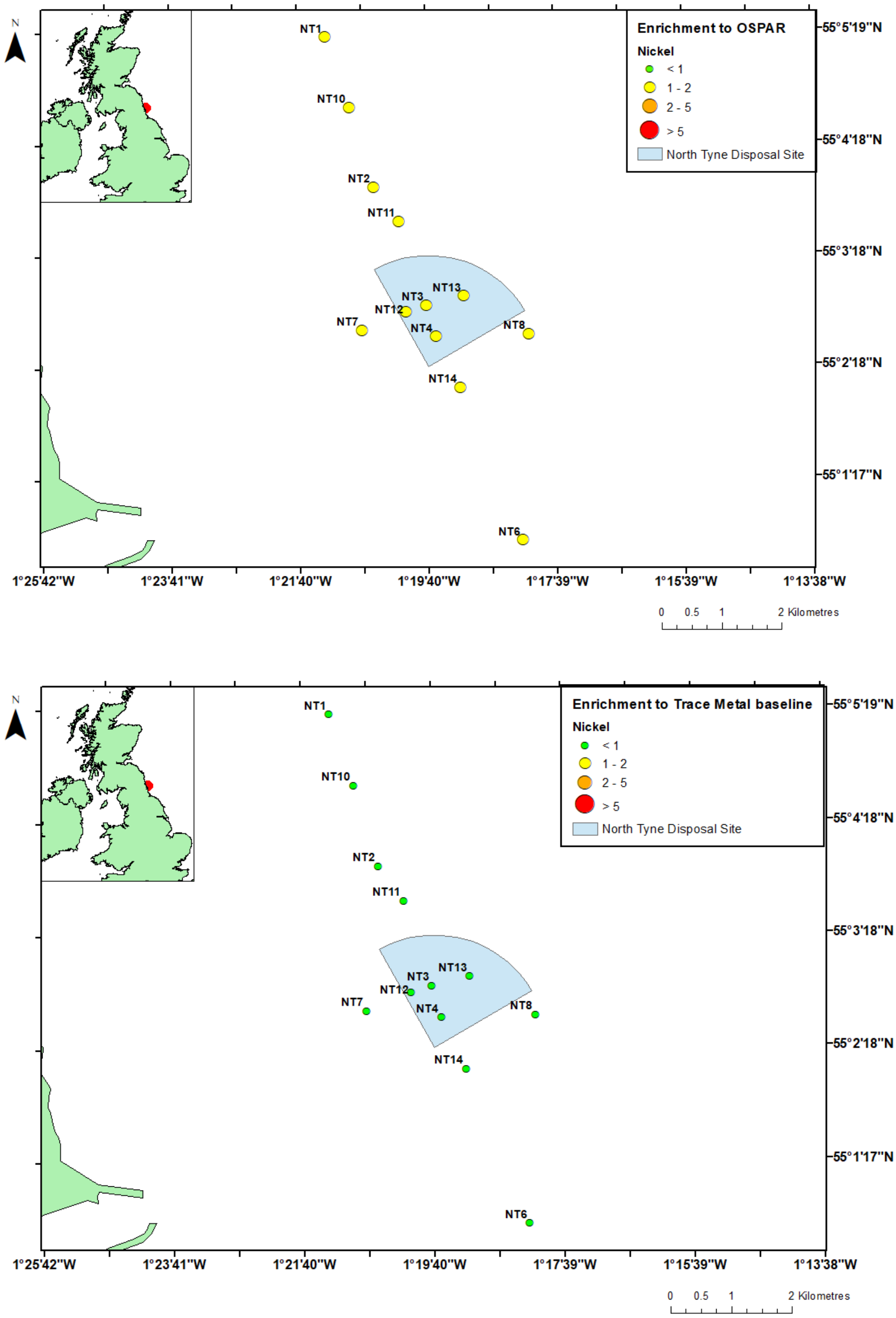


Figure 2.16. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for nickel at North Tyne, December 2022.

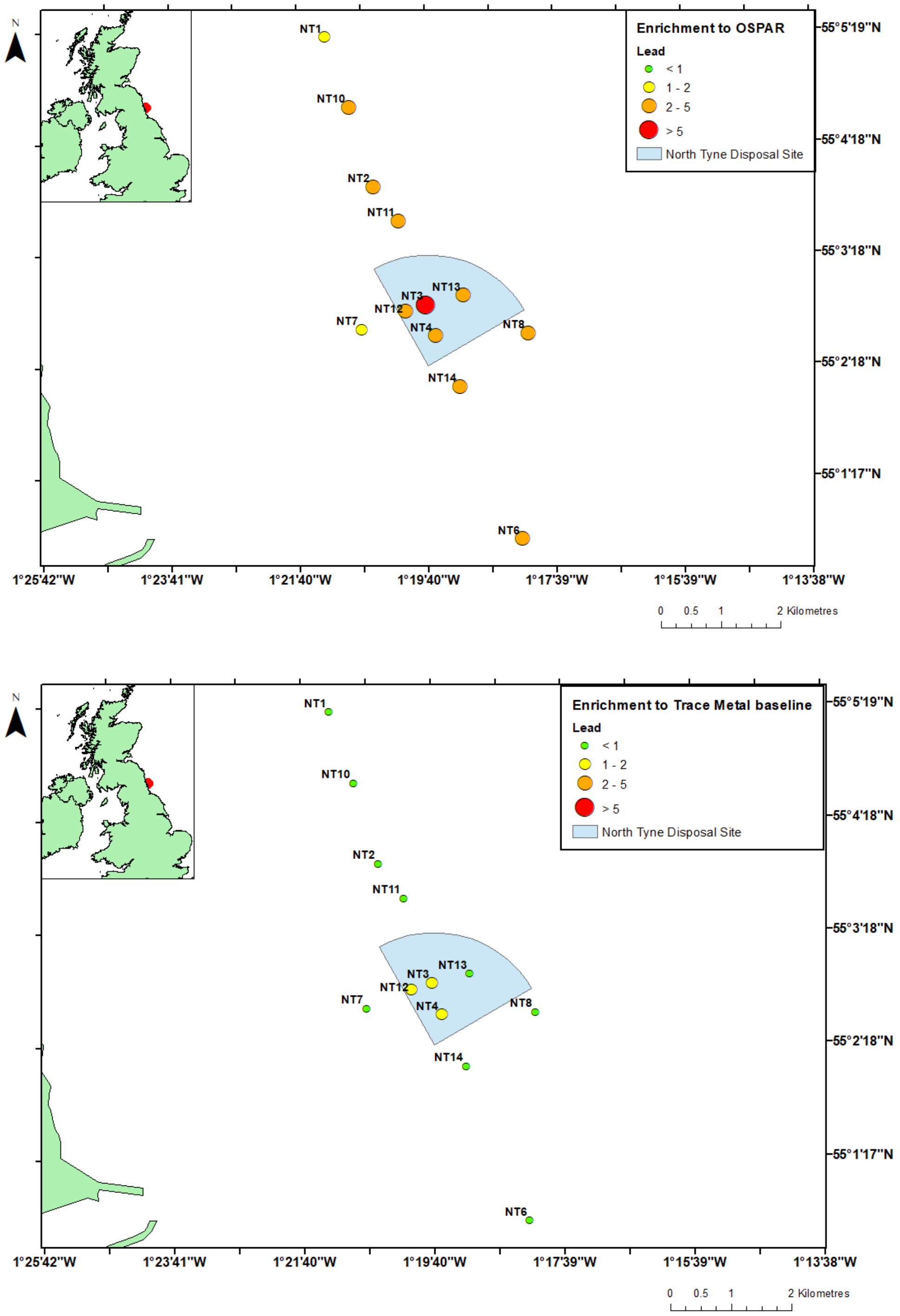


Figure 2.17. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for lead at North Tyne, December 2022.

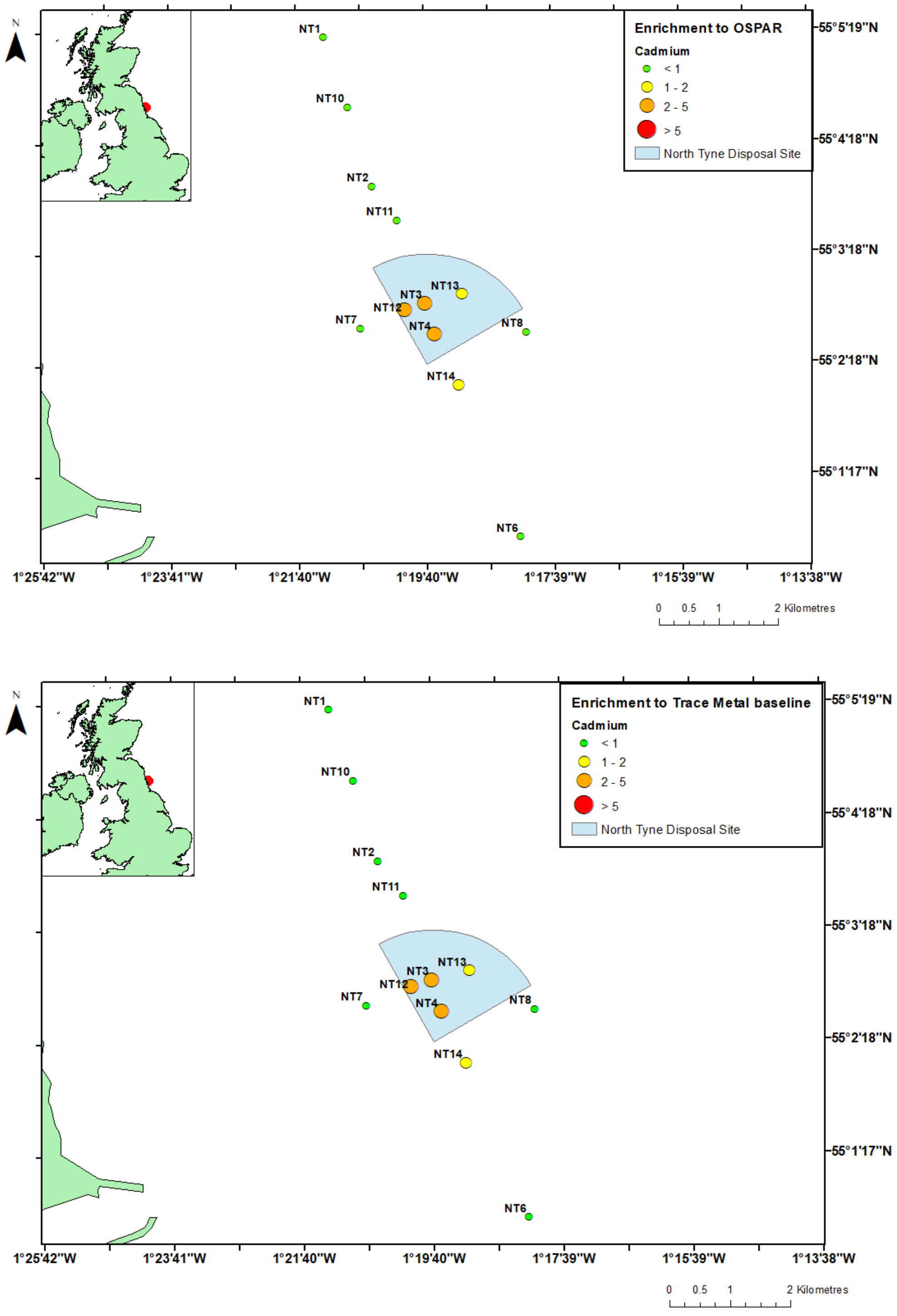


Figure 2.18. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for cadmium at North Tyne, December 2022.

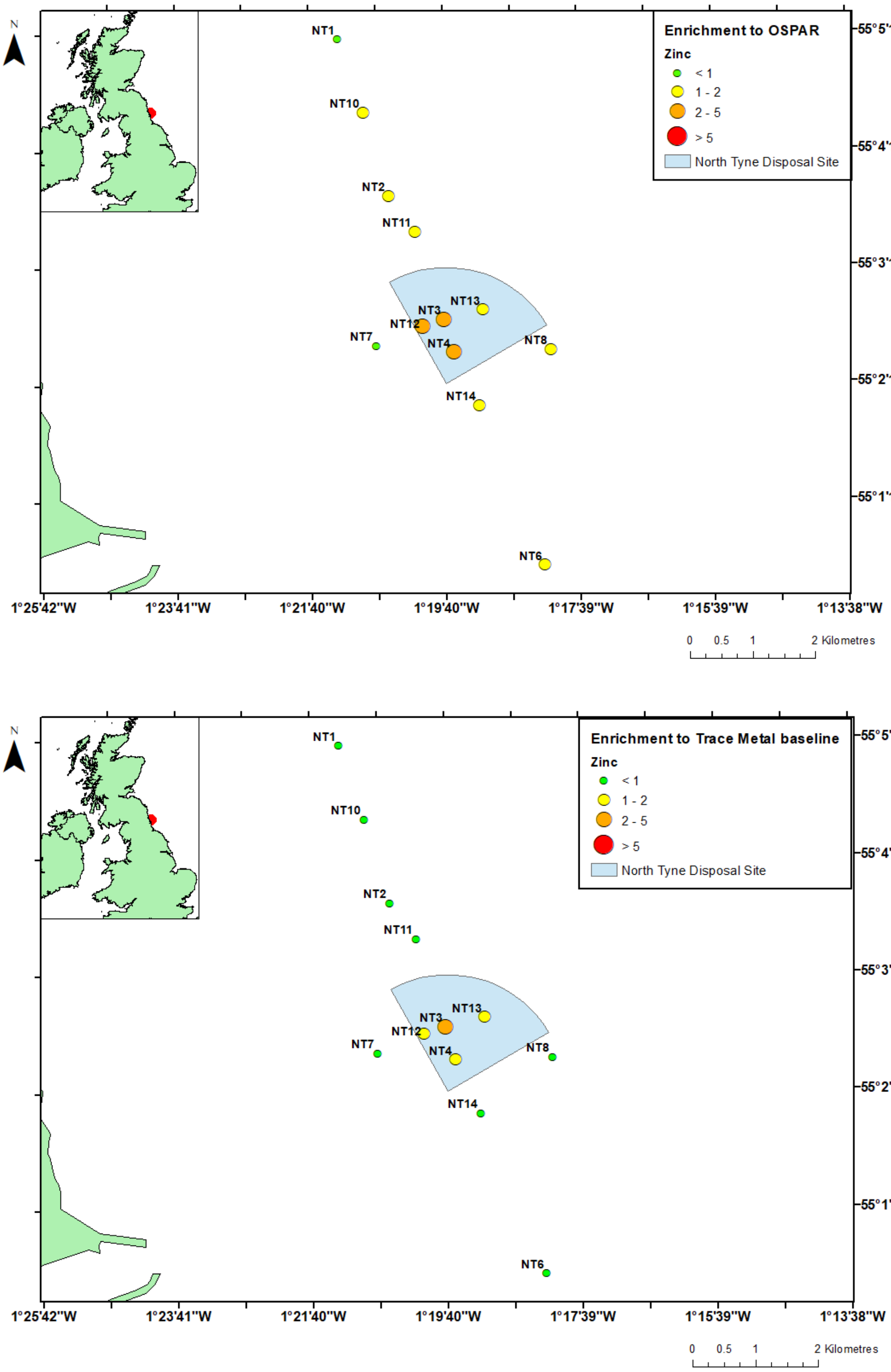
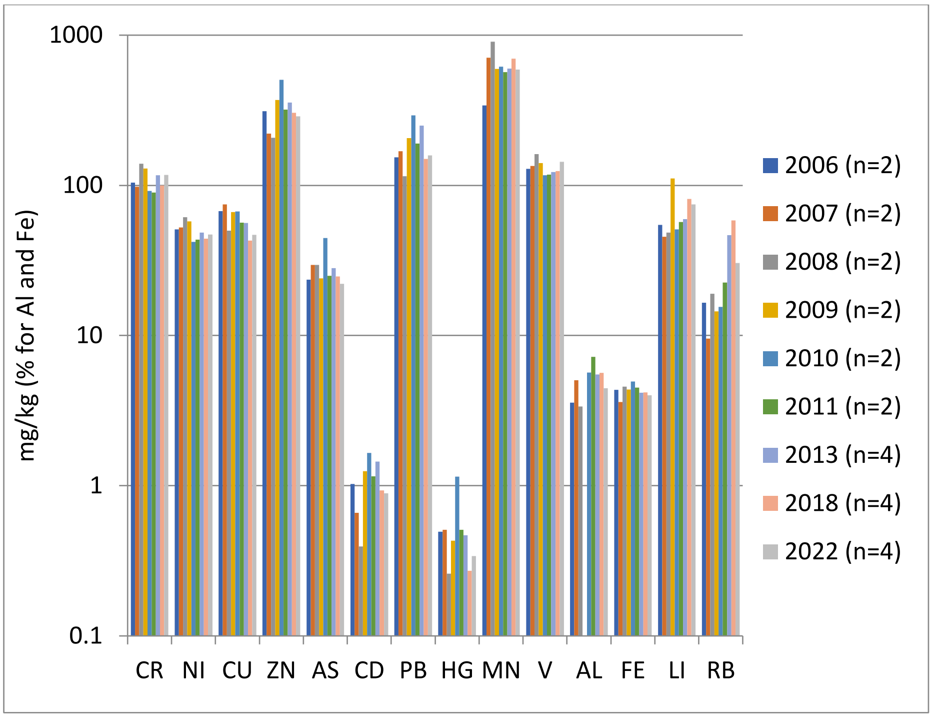


Figure 2.19. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for zinc at North Tyne, December 2022.



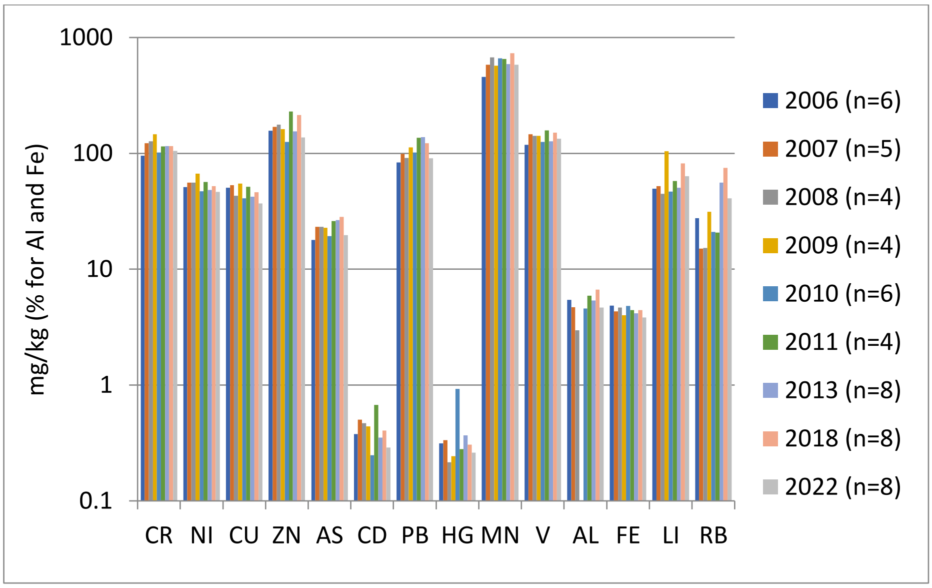


Figure 2.20. Temporal trends in metals concentrations (mg kg^{-1}; log scale) for stations sampled inside (upper panel) and outside (lower panel) the North Tyne disposal site. The numbers of stations sampled each year is given as ‘n’.

2.2 Souter Point

2.2.1 Background

The Souter Point disposal site is a comparatively large, trapezoid-shaped site located approximately 6 km from the coast of northeast England. The site, which has a maximum depth of 46 m, receives reasonably large quantities of maintenance material on an annual basis. Since 2010, mean annual disposals amount to 210,970 dry tonnes, with peaks during 2015 (331,627 dry tonnes) and 2021 (333,177 dry tonnes) (Figure 2.21). The sediments within the vicinity of Souter Point disposal generally muddy sands, however, these vary to a large extent following dredged material disposal and in response to its earlier history of receiving solid industrial wastes or other (unregulated) discharges further inshore. Tidal currents in the vicinity of the disposal site are moderate in strength and run generally parallel with the coastline, with a net residual drift southwards, at least in surface waters.

Between December 2004 and April 2005, a trial level bottom-capping project was undertaken within the centre of the disposal site. The Port of Tyne disposed 60,000 m^3 of contaminated dredged material (CDM), which was covered with 100,000 m^3 of silt and around 60,000 m^3 of sand. On placement of the silt, around 80% was siphoned off to leave a 1.5 m cap: 90,000 m^3 of sand was subsequently placed on top. Further material was later deposited to ensure isolation of the CDM. During this time the maintenance dredged material from the Tyne was disposed of to the nearby North Tyne disposal site.

Following this trial capping project, there have been ongoing concerns regarding the integrity of the cap, specifically related to cap thickness. Previous monitoring of this site under the auspices of C6794 has included a number of techniques to determine the temporal changes in bathymetry (using multibeam acoustic techniques) to assess areas of sediment/cap erosion, together with Nioz core sampling to allow acquisition of samples of the top 30 cm for sediment contaminant determination. Sediment Profiling Imagery (SPI) techniques have also been used to allow in situ visual descriptions of the sediment profiles and the presence of faunal activity (burrows, tubes, cavities). In 2012, a more intensive sampling programme was conducted under SLAB5, wherein 27 stations were sampled with a vibrocorer and the various sediment layers then analysed to determine their physical and contaminants characteristics (Bolam et al., 2014b). The results obtained indicated that while the integrity of the cap remained, cap thickness was reduced to 15-18 cm in areas where time-series bathymetric data had indicated sediment erosion.

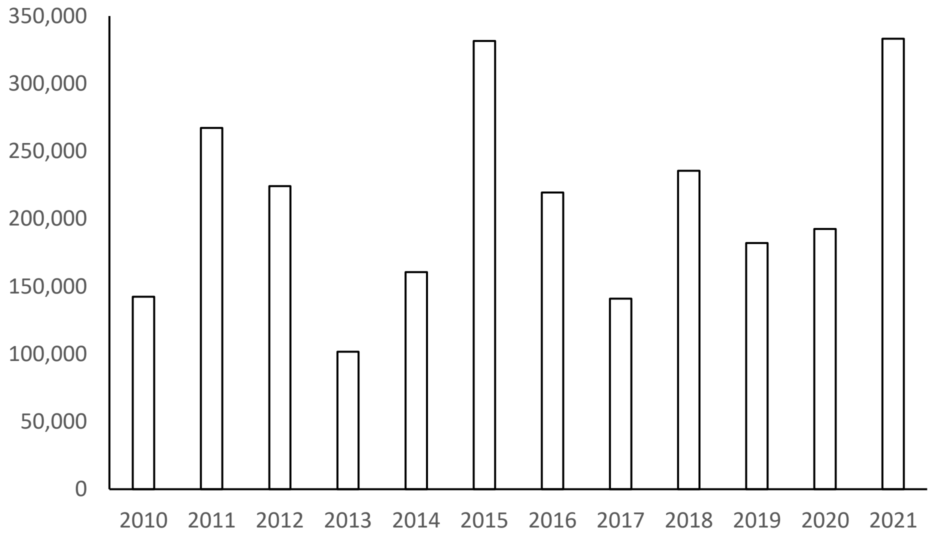


Figure 2.21. Annual amounts (dry tonnes) of dredged material disposed of to Souter Point, 2010-2021 (incl.).

Subsequent sampling under C6794 during 2016 at Souter Point provided data to determine whether there was evidence of a breach in the cap. This was conducted by assessing the contaminants concentrations of the surface sediments over the capped area, focussing more intensively on the regions previously identified as those of greatest risk of cap breach. The data acquired during 2016 did not provide any evidence that a breach of the cap above the CDM was present at that time. However, it was acknowledged that this did not totally exclude the potential for breaches in other parts of the cap which were not sampled, and the published report recommended that subsequent monitoring of cap thickness, or detecting whether a cap breach had occurred, should be conducted. In 2022, Cefas sampling focussed on the continuation of previous sampling campaigns of acquiring data to assess whether there is evidence of a cap breach. Ten stations were sampled by a grab and the surficial sediments analysed for a suite of contaminants.

2.2.2 Survey Design

The stations selected for sampling in 2022 were those where the highest concentrations of contaminants were witnessed and/or in areas where cap integrity was regarded as being most vulnerable following the intensive coring and acoustic survey conducted at Souter Point in 2012 (Bolam et al., 2014b). Thus, most of the 10 stations targeted were located in the area where CDM was disposed (Figure 2.22). These stations were sampled, and all samples subsequently treated and analysed, in the same manner as those taken at North Tyne (see section 2.1.2 of this report)

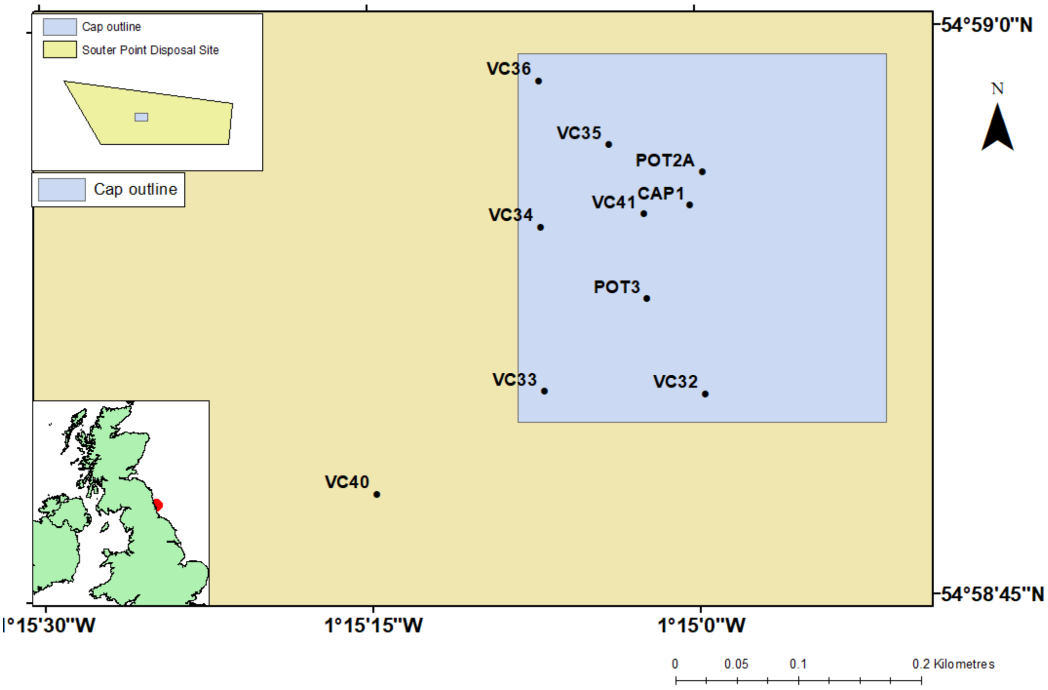


Figure 2.22. Location of the 10 stations sampled at Souter Point, December 2022.

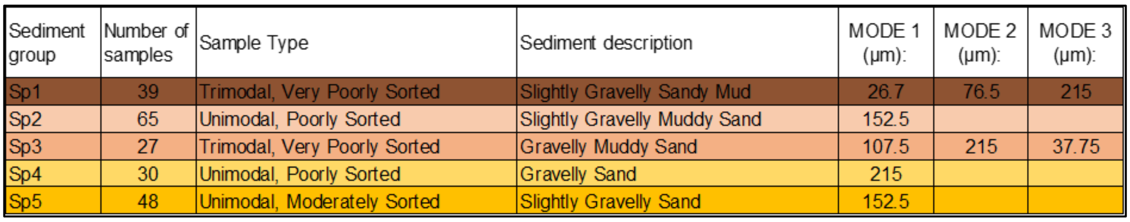
2.2.3 Results

2.2.3.1 Sediment particle size distribution

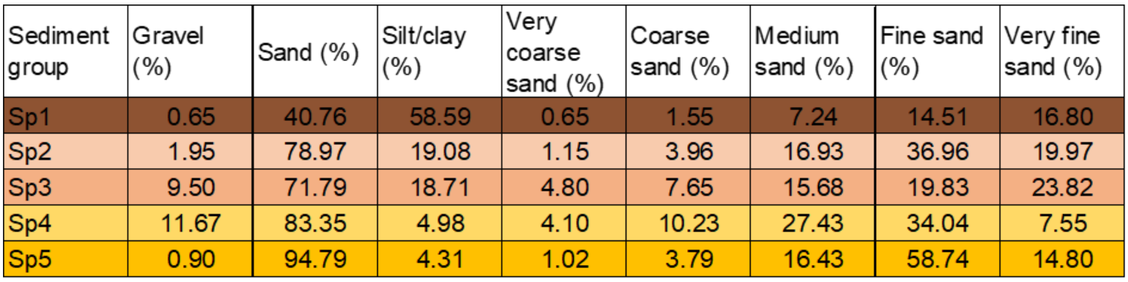
Sediments were predominantly muddy sands and sands across the Souter Point sampling stations, with some sandy muds, gravelly sands and gravelly muddy sands (Table 2.10), based on all the data collected at this site between 2005 and 2022 (and all core slices collected in 2012; Bolam et al., 2014b) totalling 209 samples. Sediments analysed in 2022 were mainly slightly gravelly sands (five samples in sediment group Sp5), gravelly muddy sands (four samples in Sp2) and one gravelly sandy mud sample (Sp1) at VC36 (Table 2.7).

The spatial variation in the proportional representation of gravel, sand and silt/clay for each 2022 sampling station, together with the percentages of silt/clay content, is shown in Figure 2.23. The highest silt/clay content was located at VC36 (~78%), with the next highest being at VC40 (~24%), while all other samples contained less than ~16% silt/clay content.

Table 2.7. Average sediment descriptions and statistics for each sediment group at Souter Point (2005 to 2022 inclusive). Sediment descriptions and statistics (derived using Gradistat (Blott and Pye, 2001) are presented.



Sediment group	Number of samples	Sample Type	Sediment description	MODE 1 (µm):	MODE 2 (µm):	MODE 3 (µm):
Sp1	39	Trimodal, Very Poorly Sorted	Slightly Gravelly Sandy Mud	26.7	76.5	215
Sp2	65	Unimodal, Poorly Sorted	Slightly Gravelly Muddy Sand	152.5		
Sp3	27	Trimodal, Very Poorly Sorted	Gravelly Muddy Sand	107.5	215	37.75
Sp4	30	Unimodal, Poorly Sorted	Gravelly Sand	215		
Sp5	48	Unimodal, Moderately Sorted	Slightly Gravelly Sand	152.5		



Sediment group	Gravel (%)	Sand (%)	Silt/clay (%)	Very coarse sand (%)	Coarse sand (%)	Medium sand (%)	Fine sand (%)	Very fine sand (%)
Sp1	0.65	40.76	58.59	0.65	1.55	7.24	14.51	16.80
Sp2	1.95	78.97	19.08	1.15	3.96	16.93	36.96	19.97
Sp3	9.50	71.79	18.71	4.80	7.65	15.68	19.83	23.82
Sp4	11.67	83.35	4.98	4.10	10.23	27.43	34.04	7.55
Sp5	0.90	94.79	4.31	1.02	3.79	16.43	58.74	14.80

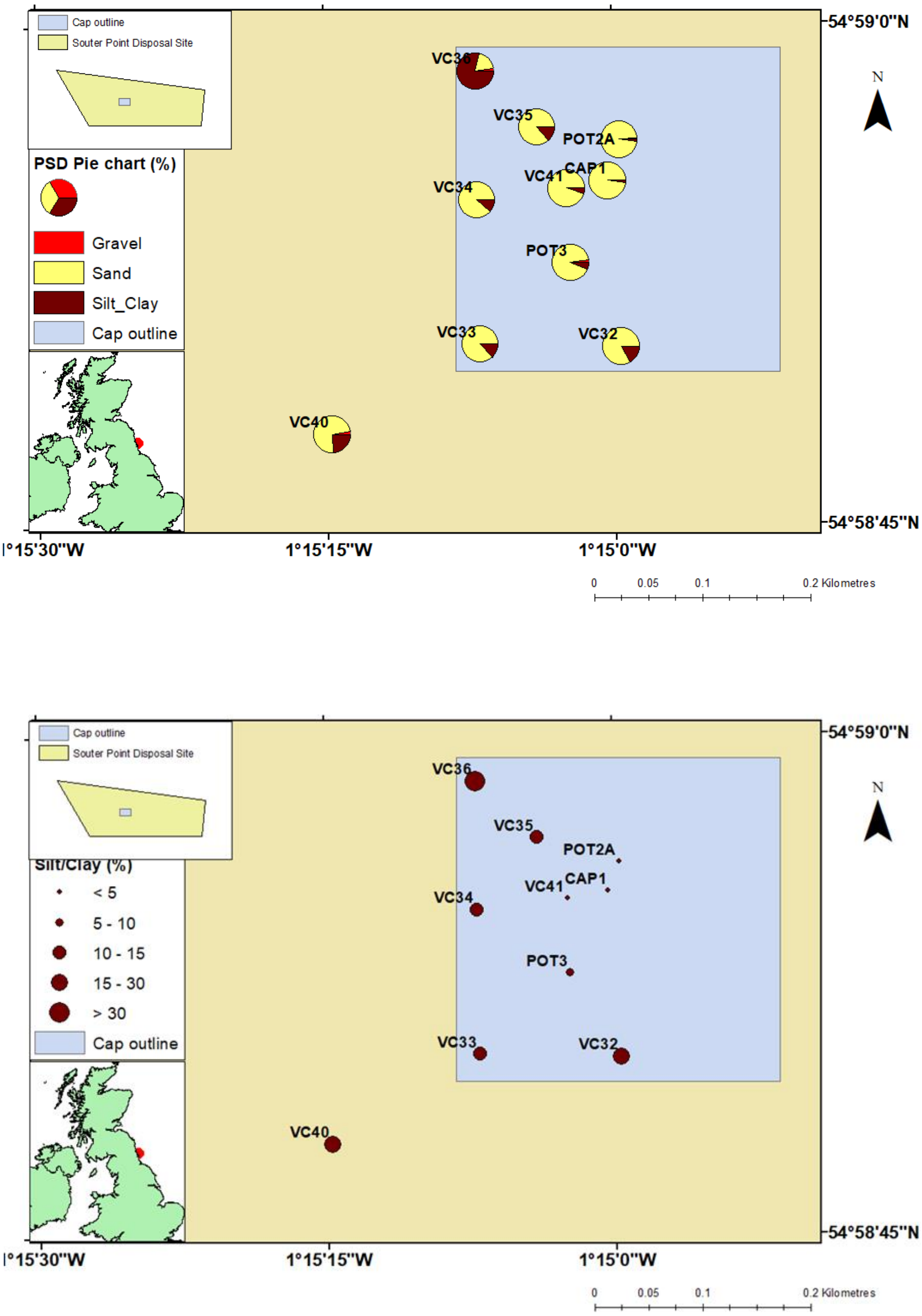


Figure 2.23. Particle size distribution of the sediments sampled at the 10 stations at Souter Point, December 2022. Pie chart of gravel, sand and silt/clay composition (top panel) and % composition of the silt/clay fraction as a bubble map (lower panel).

The temporal changes in sediment groups for sampling stations since 2005 are presented in Table 2.11 (for sample codes collected in 2022 only). Minimal changes have occurred at most site sampled except at VC36, which is much muddier than when previously sampled, and to a lesser extent VC32, VC33 and VC35. Note, care should be taken when comparing 2012 and 2016 as these samples were taken using a vibrocorer with samples taken as slices according to sediment horizons.

Table 2.8. Sediment groups for each sample code between 2005 and 2022 inclusive at Souter Point for sample codes collected in 2022 only.

Sample code	Year									
	2005	2006	2007	2008	2009	2010	2011	2012	2016	2022
CAP1	Sp5	Sp5	Sp4	Sp4	Sp4	Sp4	Sp4	Sp4	Sp5	Sp5
POT2A								Sp4		Sp5
POT3								Sp4		Sp5
VC32								Sp5	Sp5	Sp2
VC33								Sp5		Sp2
VC34								Sp2		Sp5
VC35								Sp4		Sp2
VC36								Sp5	Sp2	Sp1
VC40								Sp1	Sp2	Sp2
VC41								Sp4		Sp5

2.2.3.2 Sediment organic carbon content

Sediment organic carbon values (in the <2mm sediment fraction) range from 0.59 to 5.6% and those in the <63µm sediment fraction range from 2.91 to 4.76% (Figure 2.24). These are similar values to those observed in previous years.

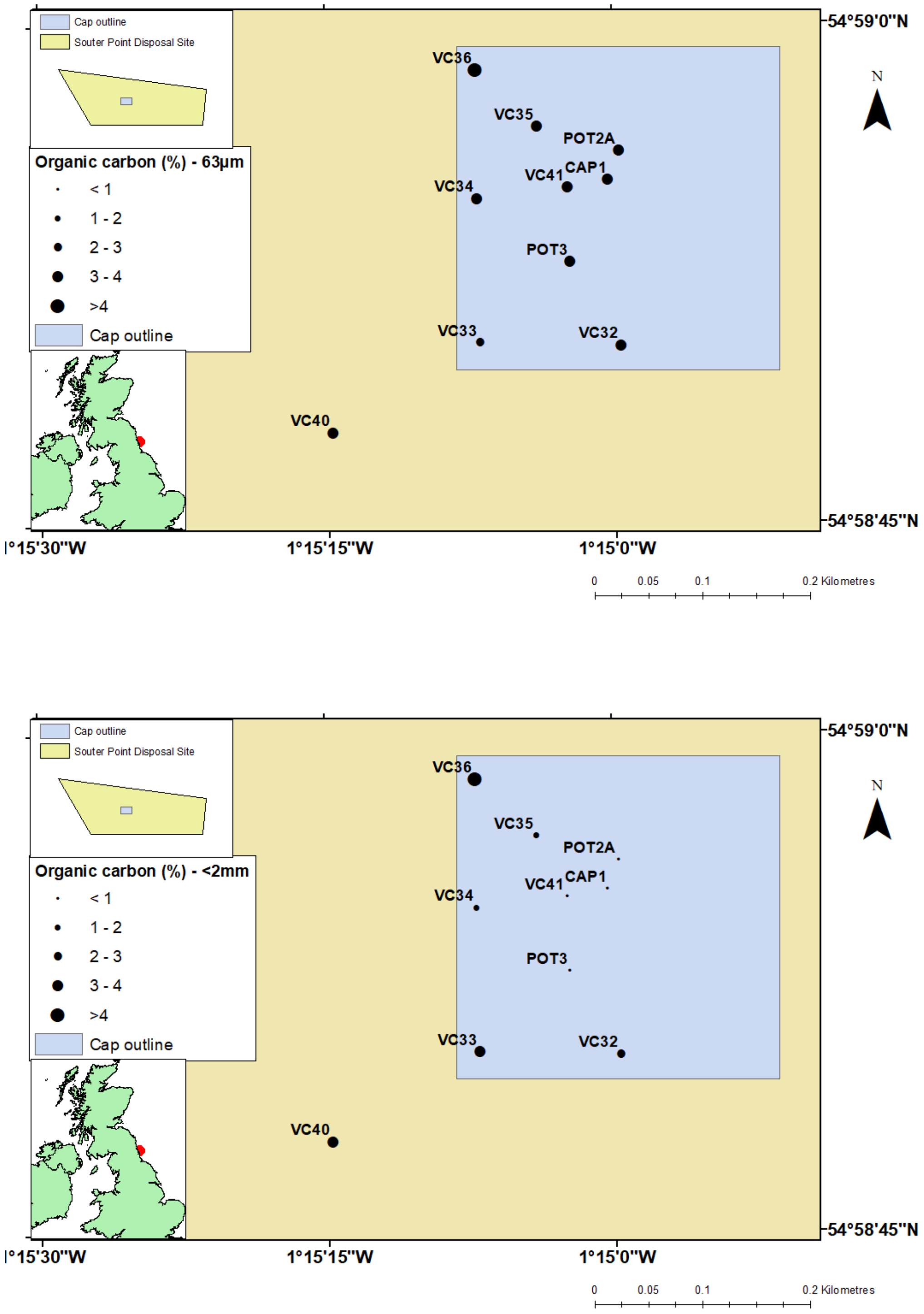


Figure 2.24. Sediments organic carbon content of the sediments sampled at the 10 stations at Souter Point, December 2022. Carbon content of the less than 63 µm fraction (top panel) and on the less than 2 mm fraction (lower panel).

2.2.3.3 Sediment PAHs concentrations

The CDM disposed of, and subsequently capped, in 2004 originated from three different dredged areas, which makes direct comparisons of summed PAH concentrations found at Souter Point with source concentrations difficult. However, the overall average summed PAH concentration for the CDM was 50,994 µg kg^{-1} dw, with a percentage deviation of 66%. Using this indicative value, we can make observations of the integrity of this capping exercise where applicable.

The highest summed PAH concentration in 2022 was 37,700 µg kg^{-1} dw found at site VC36 (Figure 2.25) located to the north western corner of the capped area (Figure 2.26). This represents a decline in PAH concentration at this station since 2016 (46,600 µg kg^{-1} dw) when this site also had the highest concentration found (Bolam et al., 2017). The second highest summed PAH concentration in 2022 was 34,700 µg kg^{-1} dw found at site VC40 (Figure 2.25) located approximately 125 m south west of the cap boundary (Figure 2.26). This represents an increase in PAH concentration at this site since 2016 (23,800 µg kg^{-1} dw)

The lowest sum PAH concentration in 2022 was 5,150 µg kg^{-1} dw found at VC41 which is located approximately 50 m northwest of the centre of the cap. Three other stations were sampled in this central cap region all had comparably low values (CAP1 7,130 µg kg^{-1} dw, POT2A 7,830 µg kg^{-1} dw , POT3 9,950 µg kg^{-1} dw) compared with locations to the edge or outside of the capped area. However, in 2016 CAP1 had a much higher concentration of 25,800 µg kg^{-1} dw but in the seven separate sampling events since 2007 the concentrations at this site have varied widely (4,200 µg kg^{-1} dw lowest in2010 to 62,000 µg kg^{-1} dw highest in 2009).

Comparisons of 2022 summed PAH concentrations to previous years can only be made at 4 sites (CAP1, VC32, VC36, VC40) and all represent comparable or lower values in 2022 except for VC40 as detailed above. The rest of the stations sampled in 2022 were within the cap boundary and seem to indicate the lowest levels are found in the centre of the capped region, indicating the cap is intact in this area, with increasing PAH concentrations towards the edge of the cap. In summary, the summed PAH concentrations sampled at Souter Point in 2022 do not indicate there is a breach of the cap integrity at any of the stations sampled. The concentrations are generally in line with previous surveys which were notably below those indicative of the CDM.

In 2022, the LMW ERL was exceeded at all sites and the ERM was exceeded at VC33, VC36 and VC40. These are the sites towards the edge or outside of the cap.

The HMW ERL was exceeded at VC32, VC33, VC36 and VC40 and, as with the LMW ERL, these are the stations towards the edge or outside of the cap. No stations exceeded the ERM for the HMW PAHs in 2022.

Evaluation of the PAH data indicated that the source in all the sediment samples was predominantly petrogenic with >68% of the PAH content arising from oil rather than combustion sources, which is in harmony with what was found in 2016.

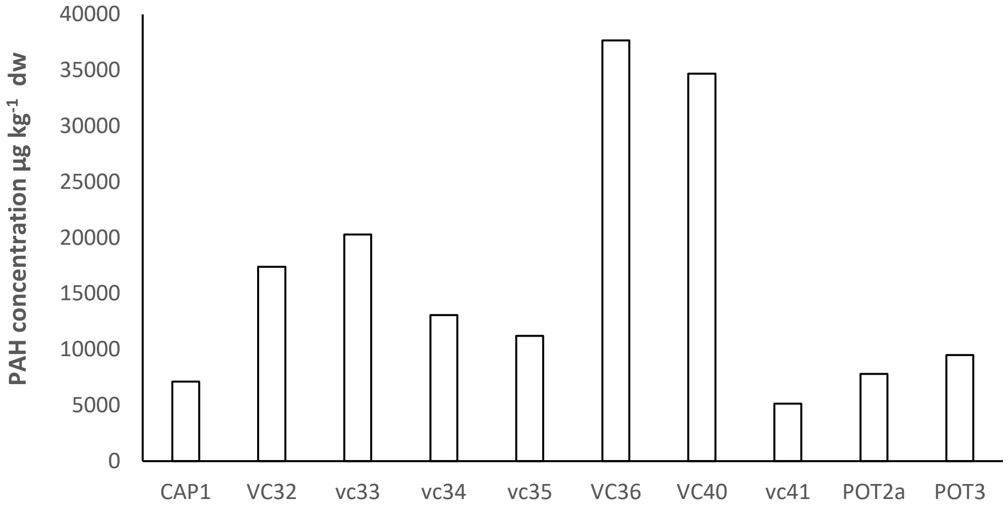


Figure 2.25. Summed PAH concentrations of the sediments from the stations sampled at Souter Point, December 2022.

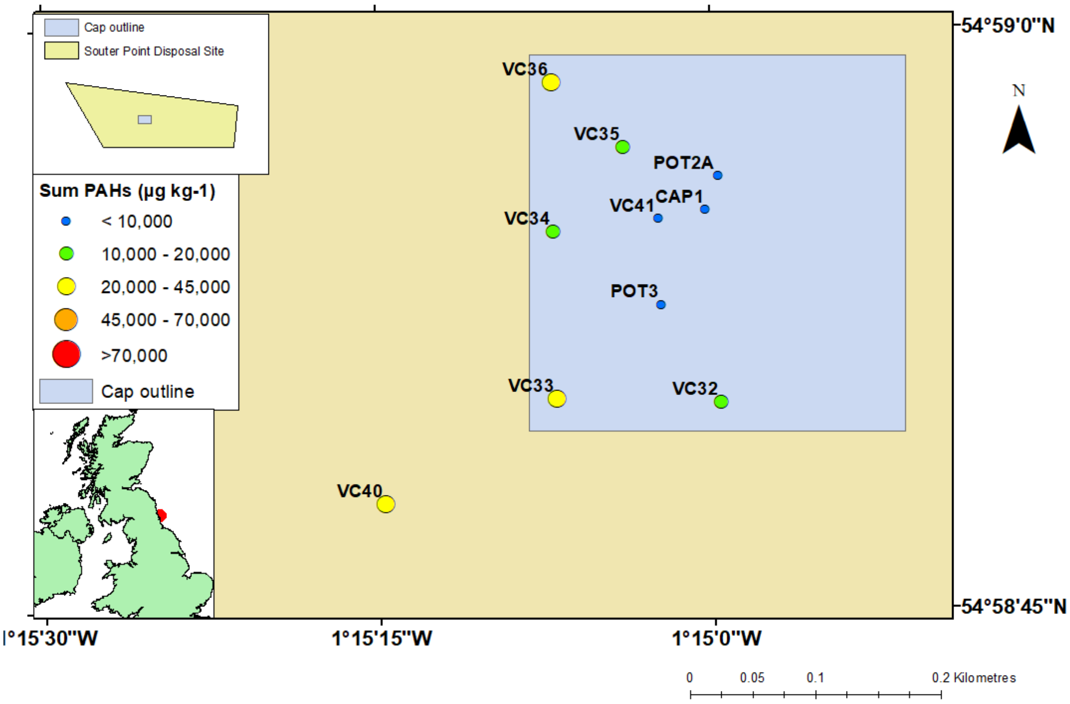


Figure 2.26. Map of summed PAH concentrations of the sediments from the stations sampled at Souter Point, December 2022.

2.2.3.4 Sediment OH concentrations

CBs were detected at all stations (∑ICES7 CBs range 0.18-5.9 µg/kg dw), with concentrations being lowest at POT2A and POT3 (0.18 and 0.22 µg/kg dw, respectively; Figure 2.27). Highest ∑ICES 7 concentrations of 5.9 and 4.2 µg/kg dw were found at VC36 and VC40 respectively (Figure 2.27).

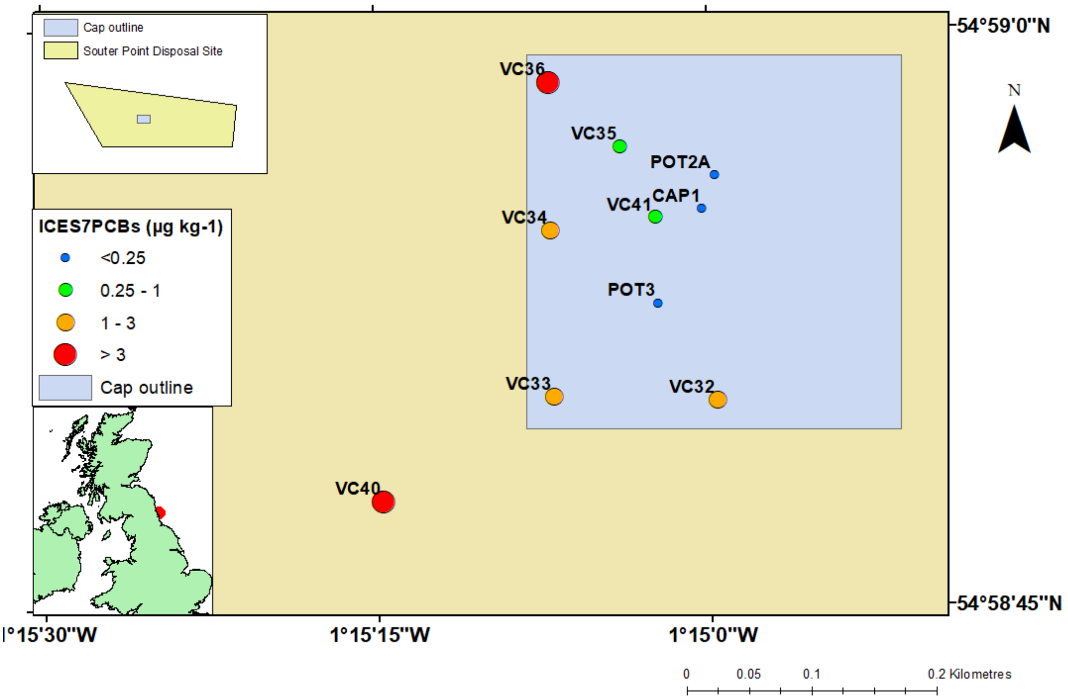


Figure 2.27. ∑ ICES7 CB concentrations for the Souter Point stations sampled in December 2022.

BDEs were detected at all stations, ranging between 0.16 and 3.80 µg/kg dw. The highest concentration was found at VC34, with the next highest value of 2.40 µg/kg dw at VC40 (Figure 2.28). The lowest (∑11 BDEs concentration of 0.16 µg/kg dw) was observed at POT2A, with comparably low concentrations (0.18 and 0.19 µg/kg dw) witnessed at POT3 and VC41, respectively (Figure 2.28). BDE47 and BDE99 are the dominant congeners present, indicative of the pentaBDE technical mixture, but BDE183 was also detected at six of the 10 stations, suggesting that the octaBDE or decaBDE technical mixture had also been in use.

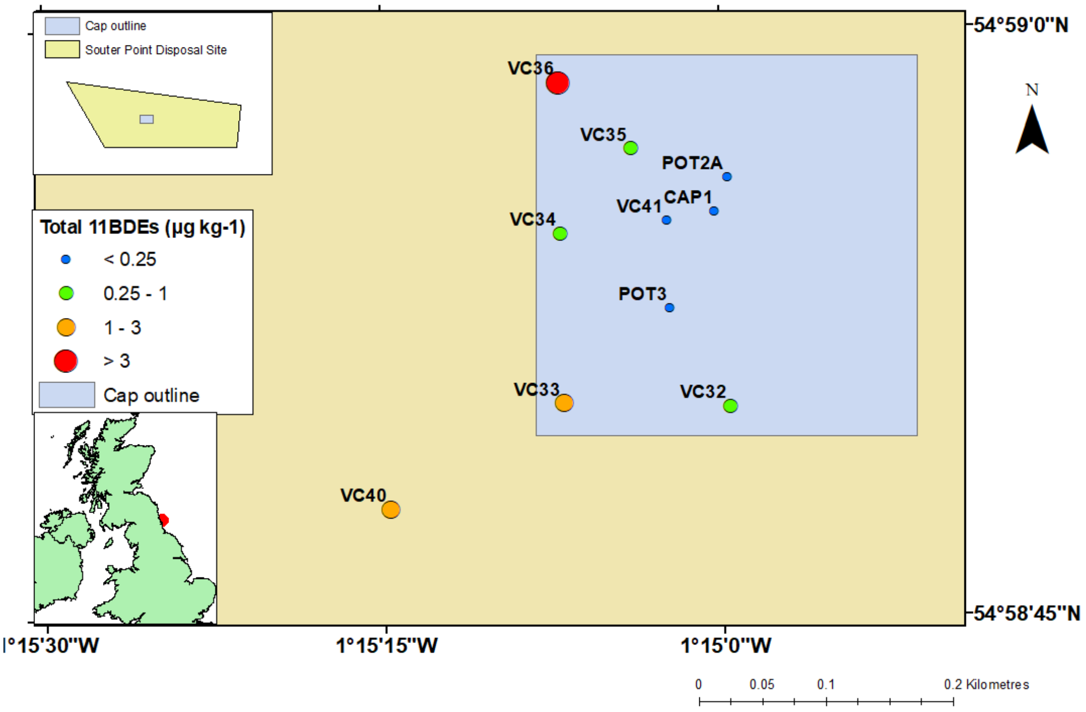


Figure 2.28. ∑ 11 BDE concentrations for the Souter Point stations sampled in December 2022.

BDE209 was detected at all stations and at higher concentrations (range 0.6 to 254.0 µg/kg dw; Figure 2.29) than the other measured organohalogens. When included with the other BDEs, BDE209 represented >77% of the BDEs present (range 77-99%). BDE209 is indicative of the decaBDE technical mixture, which had been in use more recently than the other technical mixtures, although it's use too has now been restricted in the EU since 2008. The highest concentration of 254.0 µg/kg dw was detected at VC36, with 32.6 µg/kg dw at VC33 and 23.8 µg/kg dw at VC40. Other stations were all in the range 0.6-5.4 µg/kg dw, with the lowest concentrations sampled at VC41 and POT3.

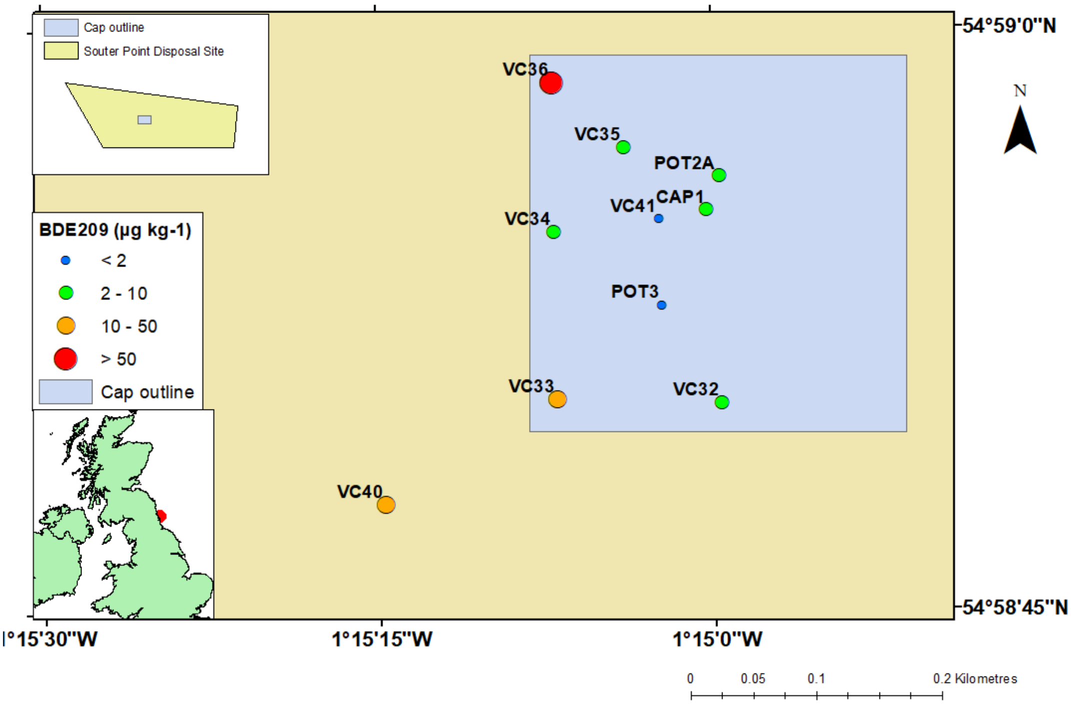


Figure 2.29. BDE 209 concentrations for the Souter Point stations sampled in December 2022.

The OCPs (organochlorine pesticides) HCB, plus DDT and metabolites were detected at all stations. ∑ 6DDTs concentrations ranged from 0.23-6.00 µg/kg dw, with the highest values at VC36 (6.00 µg/kg dw), VC40 (2.20 µg/kg dw) and VC33 (1.70 µg/kg dw) (Figure 2.30). The ratio of DDT to metabolites at all stations is much less than 1, suggesting that these sediments have not received fresh inputs of DDT. HCB concentrations ranged from 0.04 – 0.78 µg/kg dw, with the highest values at VC36 (0.78 µg/kg dw) and VC40 (0.34 µg/kg dw). Dieldrin was detected at three of the 10 stations (range <0.10-0.48 µg/kg dw), with the highest values at VC36 (0.48 µg/kg dw), VC40 (0.44 µg/kg dw) and VC33 (0.20 µg/kg dw).

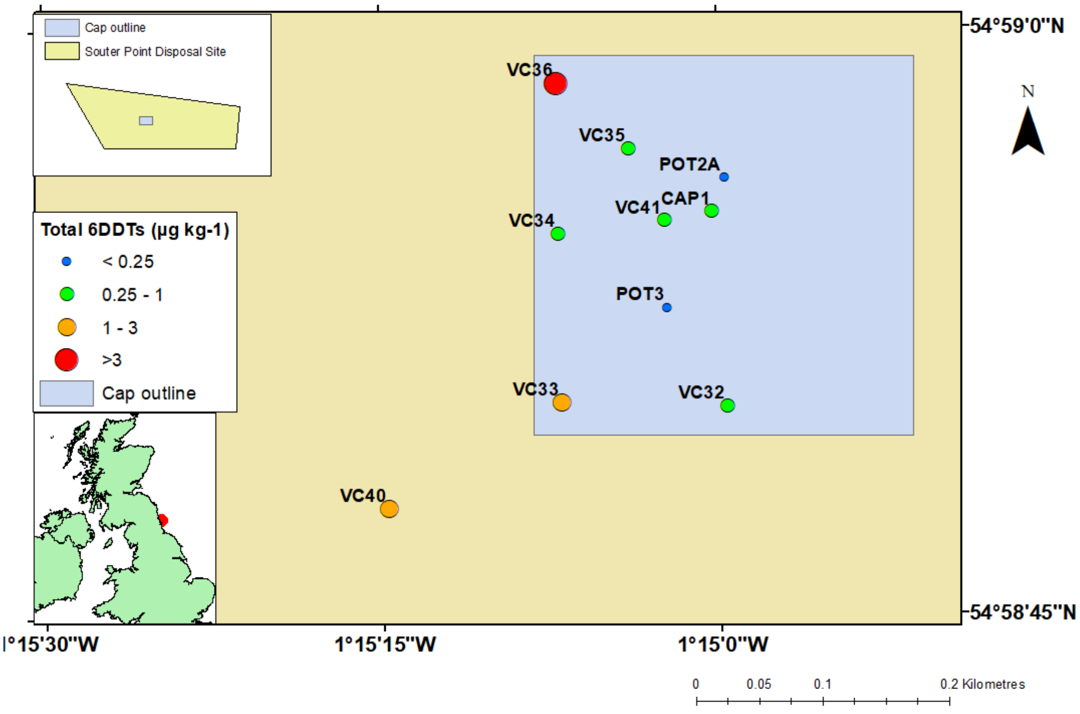


Figure 2.30. Total DDT concentrations for the Souter Point stations sampled in December 2022.

Concentrations of CBs and dieldrin at all stations were all below Cefas AL1. ∑ 6DDTs concentrations were above Cefas AL 1 at three out of the 10 stations. No Cefas ALs exist for BDEs including BDE209. According to the OSPAR guidelines, most stations had 'good' environmental status for all ICES 7 CBs and BDEs. The exception was VC36 which was above the EAC (environmental assessment concentration) for BDE209.

In 2012, vibrocore samples were collected under the auspices of SLAB5, and samples from below the cap were analysed for contaminants to obtain a 'fingerprint' that could be used to identify if the cap had been breached (Bolam et al., 2014b). The resulting PCB data allowed the sediment from each core slice to be matched to the material source from where it was originally dredged (Table 2.9). Similarly, it was also possible to relate the levels in the capping material to its source. Although information on levels of PBDEs and OCPs was not obtained from the source material at the time of dredging, this could be inferred from the results obtained from these cores (Table 2.9). Using these data, it is possible to compare the results from the 2022 survey with these 'fingerprints'. This assessment reveals that all of the stations sampled in 2022 have concentrations typical of the capping material (Table 2.9 to Table 2.11). Even the station VC36, which had the highest levels of contaminants, had ∑ICES 7 CBs, ∑25 CBs, ∑11 BDEs, HCB and *p,p'*-DDE that matched the concentrations in the capping material. Only the BDE209 concentration for VC36 was above this range, but without the elevated presence of the other compounds, this is likely to have originated from more recent dredged material disposed of to the site.

There are sufficient data to allow temporal trends of contaminants at Souter Point to be assessed over the period 2005 to 2022, although only CAP1 was sampled at these trend stations in this 2022 (see Table 2.12 to Table 2.15). At CAP1, levels of ∑ICES 7 CBs, ∑11 BDEs and ∑3 DDTs in 2022 were at the lower end of what had been measured previously. BDE209 concentration was in the mid-range of what had been measured previously and approximately half the concentration that was measured there in 2016.

Table 2.9. Concentration (in µg/kg dw) of PCBs in the Souter Point Capping Survey vibrocore samples collected in 2012 (Bolam et al., 2014b).

	Core layer (in cm)	Concentration (in µg/kg dw)	
		∑ICES 7 CBs	∑25 CBs
POT4A	158-234	324	576
POT3	94-123	249	518
POT1A	59-72	133	326
VC34	112-130	141	270
POT2A	150-230	128	249
CAP1	158-210	99.4	210
VC41	128-230	80.2	161
VC35	140-197	60.9	125
VC42	62-67	52.2	112
POT5	31-39	36.2	72.6
SPI2	0-7	5.43	11.0
VC40	0-28	7.6	15.3
Wallsend*		6-324	11-630
Swan Hunter*		141	277
Neptune Yard*		70-73	150-151
Capping material*		0-12	0-26

*From 2004 survey of dredge material that was to be disposed of in the capping project.

Table 2.10. Concentration (in µg/kg dw) of contaminants in the Souter Point Capping Survey vibrocore samples collected in 2012.

	Core layer (in cm)	Concentration (in µg/kg dw)			
		∑11 BDEs	BDE209	HCB	*p,p'*-DDE
POT4A	158-234	15.9	128.0	2.10	5.70
POT3	94-123	9.9	74.3	1.80	5.06
POT1A	59-72	17.9	103.0	3.15	6.38
VC34	112-130	12.2	116.0	1.57	5.21
POT2A	150-230	16.1	224.0	2.26	3.64
CAP1	158-210	10.7	96.2	2.09	4.75
VC41	128-230	18.2	143.0	2.11	5.30
VC35	140-197	34.7	303.0	0.89	2.95
VC42	62-67	19.3	140.	1.13	4.22
POT5	31-39	10.2	136.0	1.21	1.95
SPI2	0-7	5.7	54.2	0.59	0.84
VC40	0-28	10.7	105.0	0.74	1.20

Table 2.11. Concentration (in µg/kg dw) of contaminants in the Souter Point samples collected in 2022.

	Concentration (in µg/kg dw)					
	∑ICES 7 CBs	25 CBs	∑11 BDEs	BDE209	HCB	*p,p'*-DDE
POT3	0.22	0.43	0.18	0.87	0.05	*0.01*
VC34	1.75	3.06	0.46	5.05	0.11	0.14
POT2A	0.18	0.36	0.16	2.61	0.04	0.03
CAP1	0.24	0.45	0.22	5.44	0.07	0.06
VC41	0.29	0.52	0.19	0.62	0.05	0.02
VC35	0.77	1.35	0.83	4.91	0.12	0.16
VC36	5.90	12.0	3.77	254.00	0.78	1.46
VC33	1.81	3.34	1.22	32.60	0.30	0.46
VC40	4.18	8.03	2.36	23.80	0.34	0.48
VC32	1.04	1.83	0.71	4.45	0.24	0.23

Table 2.12. Temporal trends (2005-2022) of ∑ICES 7 CBs concentration (in µg/kg dw) at Souter Point.

	∑ICES 7 CBs concentration (in µg/kg dw)							
	2005	2006	2007	2008	2009	2010	2016	2022
TC2		6.20	2.58		1.54	1.47	1.91	
CAP4	3.70	3.60	2.35	3.39	3.11	1.50	1.45	
CAP2	0.83	1.01	*0.70	*0.70	7.23	2.96	4.65	
CAP1	1.10	0.84	*0.70	0.96	1.88	*0.70	1.24	0.24
CAP5	1.10	0.86	*0.70	3.22	1.11	0.80	2.56	
CAP9	4.97	2.91	2.00	2.84	3.25	3.13	2.53	
CAP7	1.34	1.12	2.23	1.51	1.24		1.44	
TC3	0.96	1.19	*0.70	*0.70	0.96	*0.70	0.84	
TC4	1.17	1.14	*0.70	2.62	1.30	1.09	0.86	
SPI15						56.00		
SPI16						3.61		

*Concentrations represent estimates of concentrations for samples where all ICES 7 congener concentrations were below LODs. Limits of detection for CBs improved between 2010 and 2016, resulting in a step decrease in ∑7 CBs concentration for samples with congeners below LODs.

Table 2.13. Temporal trends (2005-2022) of ∑11 BDEs concentration (in µg/kg dw) at Souter Point.

	∑11 BDEs concentration (in µg/kg dw)							
	2005	2006	2007	2008	2009	2010	2016	2022
TC2	3.95	8.30	2.94		1.36	2.16	2.59	
CAP4	5.84	2.15	2.57	1.37	1.18	1.95	1.25	
CAP2	1.11	1.01	0.79	0.41	1.55	1.24	0.98	
CAP1	0.83	0.80	0.96	0.26	0.62	0.28	0.75	0.22
CAP5	1.13	1.07	0.91	0.41	0.54	0.88	1.75	
CAP9	3.77	12.6	5.92	1.02	1.92	1.95	1.35	
CAP7	1.94	2.37	3.96	0.85	0.80		1.34	
TC3	1.45	5.96	1.36	0.55	0.73	1.45	0.90	
TC4	1.78	8.59	1.51	0.82	0.86	1.34	0.76	
SPI15						1.28		
SPI16						2.75		

Concentrations in italic represent estimates of concentrations for samples where all 11 BDE congener concentrations were below LODs. Limits of detection for BDEs improved between 2007 and 2008 and again between 2010 and 2016, resulting in step decreases in ∑11 BDEs concentration for samples with congeners below LODs.

Table 2.14. Temporal trends (2008-2016) of BDE209 concentration (in µg/kg dw) at Souter Point.

	BDE209 concentration (in µg/kg dw)					
	2008	2009	2010	2011	2016	2022
TC2		9.00	6.68		26.4	
CAP4	3.37	3.88	7.63		12.2	
CAP2	0.90	49.2	17.8	6.36	58.2	
CAP1	0.77	7.49	0.89	2.95	9.79	5.44
CAP5	2.75	12.0	3.79	6.35	18.1	
CAP9	4.08	13.6	5.08	11.9	13.8	
CAP7	3.72	3.92			56.3	
TC3				6.14	1.59	
TC4				5.68	1.69	

Table 2.15. Temporal trends (2006-2022) of ∑3 DDTs concentration (in µg/kg dw) at Souter Point.

	∑3 DDTs concentration (in µg/kg dw)					
	2006	2007	2008	2010	2016	2022
TC2		1.05		1.03	1.68	
CAP4	0.95	0.59	28.6	1.02	2.41	
CAP2	0.56	0.63	0.57	10.5	1.34	
CAP1	0.49	0.59	0.54	*0.30*	0.83	0.25
CAP5	0.87	0.56	1.35	1.01	1.77	
CAP9	1.55	1.69	1.10	4.10	1.63	
CAP7	0.82	1.96	0.74		1.24	
TC3	1.18	0.82	0.53	1.95	2.08	
TC4	1.11	0.77	1.79	0.80	0.79	

∑DDTs is the sum of 3 chemicals (*p,p'*-DDE, *p,p'*-TDE, *p,p'*-DDT). Also note limits of detection for individual DDTs improved between 2010 and 2016 by a factor of 2-10, depending on chemical.

2.2.3.5 Sediment metals concentrations

Levels of metals enrichment for the 10 stations sampled at Souter Point in 2022, based on both the OSPAR BAC and regional baseline values, are represented in Figure 2.31 to Figure 2.38. When assessing the level of metal concentrations for the Souter Point data, findings from both approaches show no enrichment of arsenic (As) for the most stations and a shift from some slight enrichment in 2016 (Bolam et al., 2017) to no enrichment (Figure 2.31). Arsenic was previously present in low concentrations and has appeared to decrease slightly in 2022, suggesting no breach of arsenic contamination from within the capped sediment.

Cadmium (Cd) displays either slight or no enrichment for most stations monitored within the capped area for both assessment methods (Figure 2.37), which is in line with previous observations from 2016. Chromium (Cr) displayed either no or slight enrichments by both assessment methods (Figure 2.32) and was not a significant change from the 2016 study (no breach of Cr; Bolam et al., 2017). Similarly, nickel (Ni) showed either no or slight enrichments in 2022 (Figure 2.35) which is in line with observations from 2016, also suggesting that the CDM is not exposed.

Zinc (Zn) displayed slight enrichments when assessed against OSPAR levels with moderate enrichment at two stations (one in capped region and one outside capped region; Figure 2.38). This is a slight increase but not necessarily indicative of a breach of the cap's integrity. Assessing the stations against regional baseline levels show a reduced enrichment relative to the OSPAR approach within the capped region to no enrichments (from previously being slightly enriched in 2016). Overall, this reflects a reduction in zinc concentrations.

Comparing the 2016 data to the 2022 data for lead (Pb) reveals that Pb has reduced enrichments, with OSPAR assessment approach going from moderate to slight (for most stations within the capped zone; Figure 2.36). Similarly, the baseline assessment method for Pb shows a shift from slight to no enrichments. This suggests no recent breaching of contamination from within the capped zone. Mercury (Hg) displays mainly moderate enrichments (with a few highly enriched stations) *via* the OSPAR assessment approach, and mainly no enrichments (with a few slightly enriched points) for the regional baseline assessment method (Figure 2.34). These data for Pb in 2022 are in line with those observed in 2016 (Bolam et al., 2017).

Since most of the proposed regional assessment concentration baseline values are higher than the current OSPAR BACs values, using OSPAR BACs as an approach to assess metal enrichment in this region could be misleading for some metals, in particular Cu, Zn, Hg and Pb. The proposed baselines give a more accurate level of enrichment for the Tyne/Tees region. Generally, levels of Cu, Zn, As, Cd, Pb and Hg are higher at stations within the disposal site than those situated outside the disposal site. In summary, the trace metals data reveal no evidence of a breach in the cap integrity.

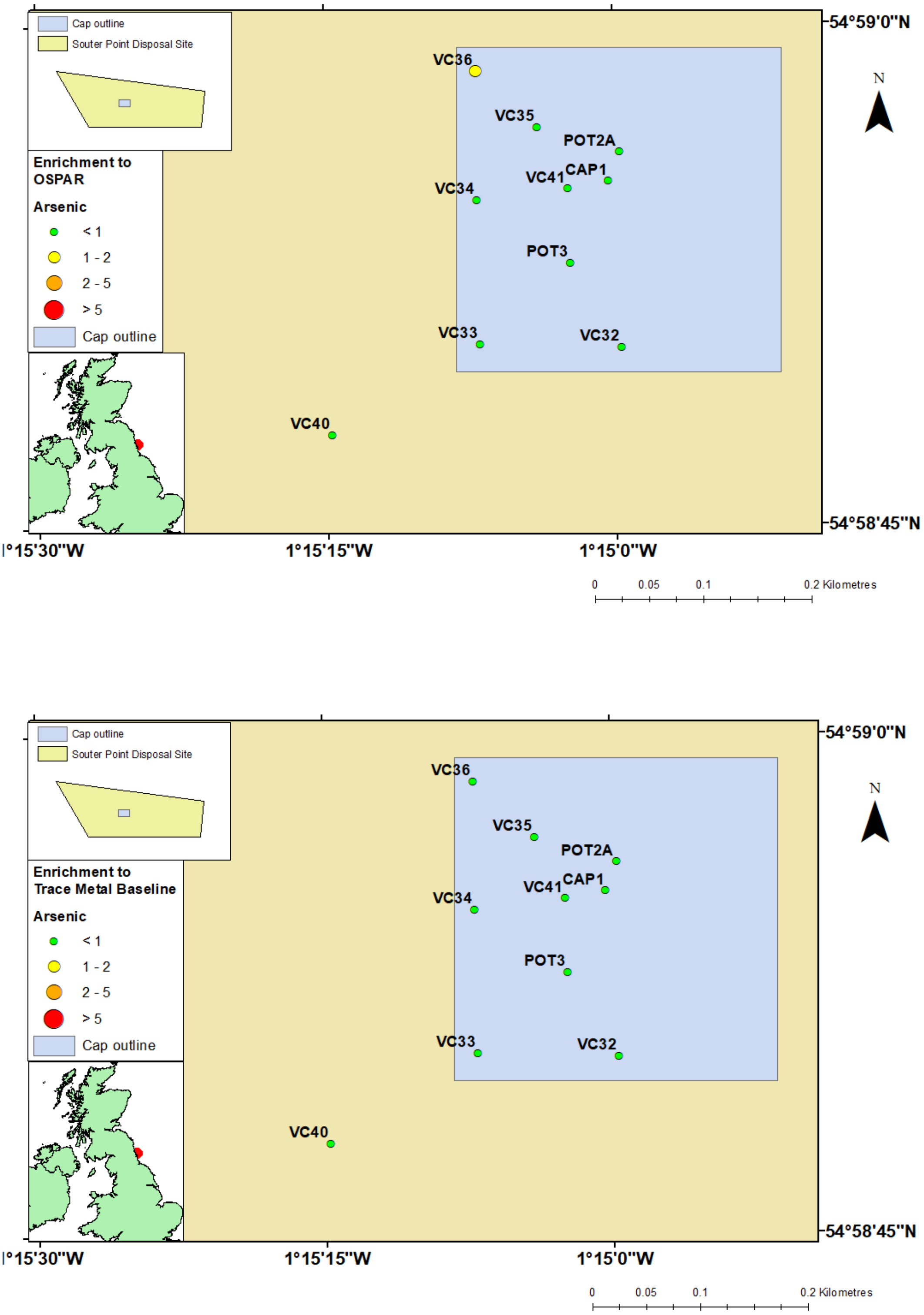


Figure 2.31. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for arsenic at Souter Point, December 2022.

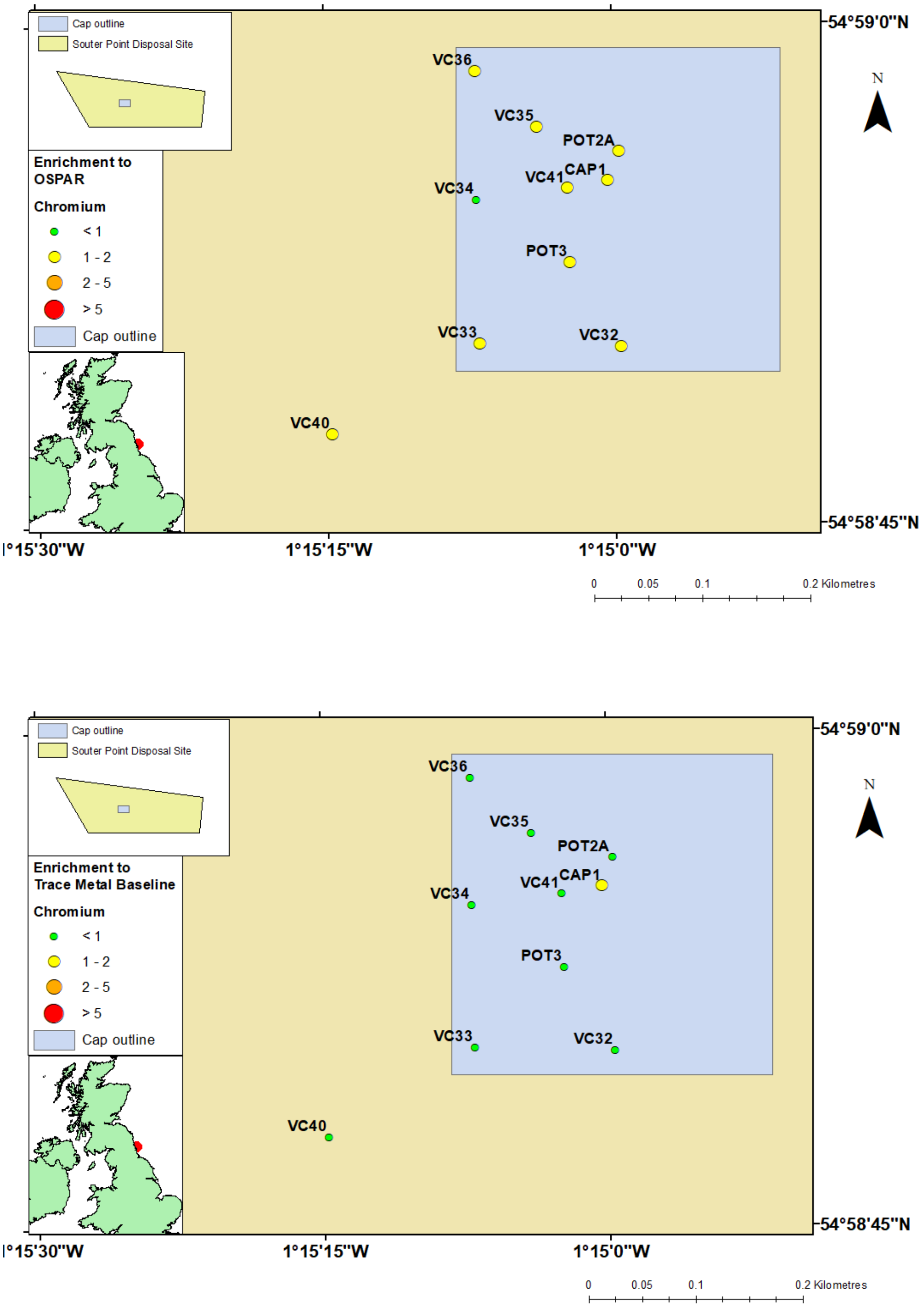
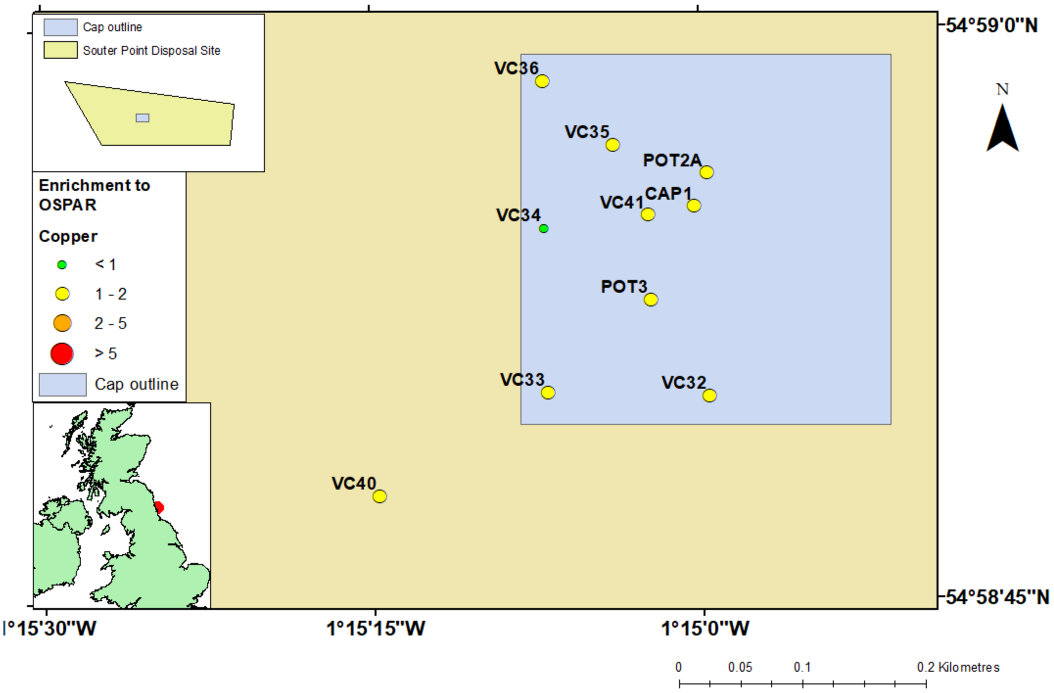


Figure 2.32. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for chromium at Souter Point, December 2022.



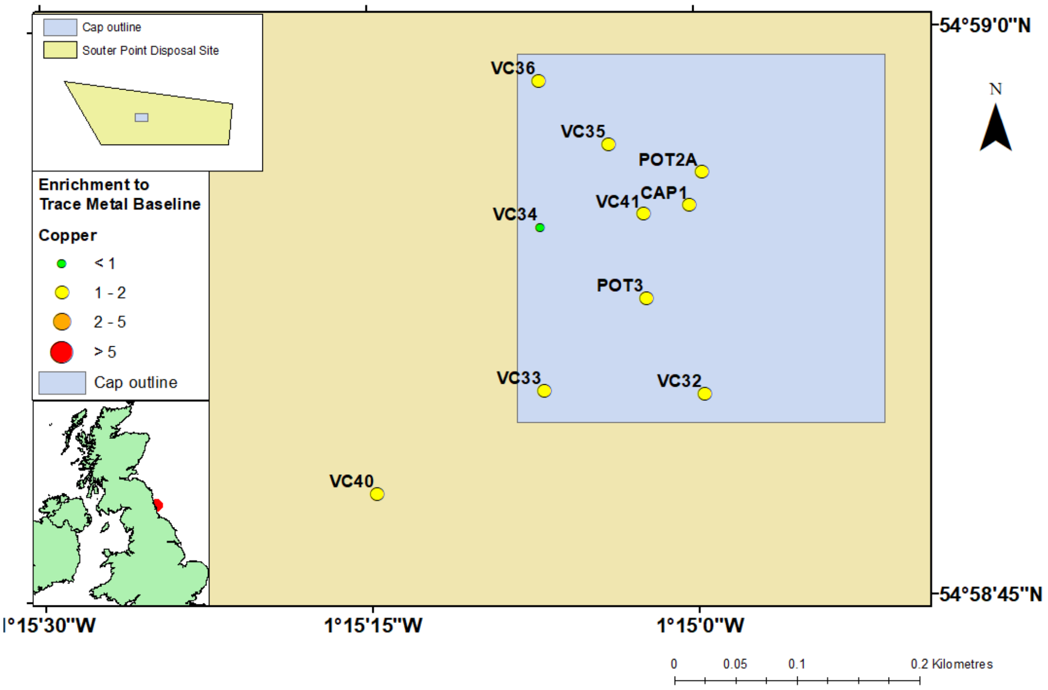


Figure 2.33. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for copper at Souter Point, December 2022.

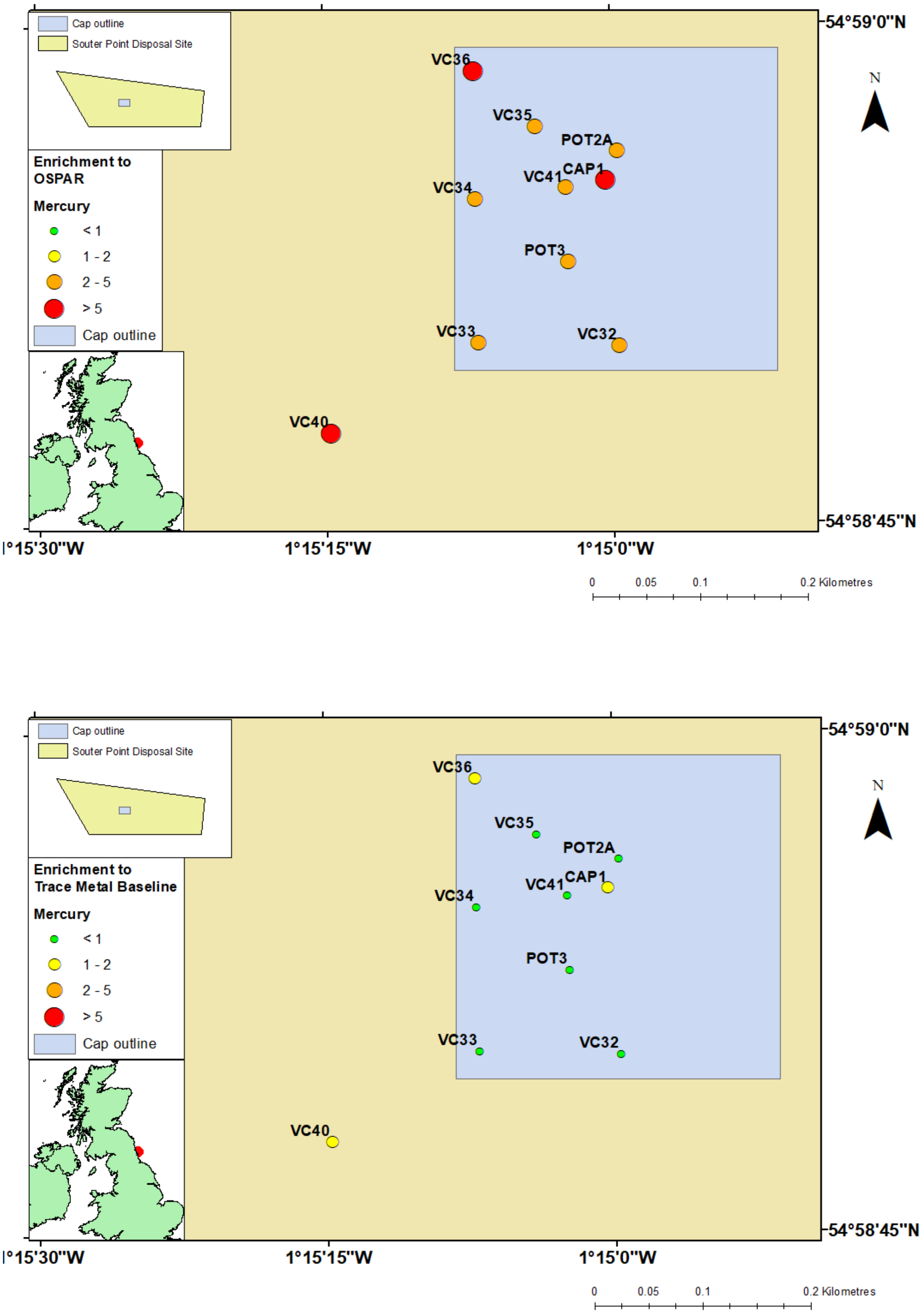


Figure 2.34. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for mercury at Souter Point, December 2022.

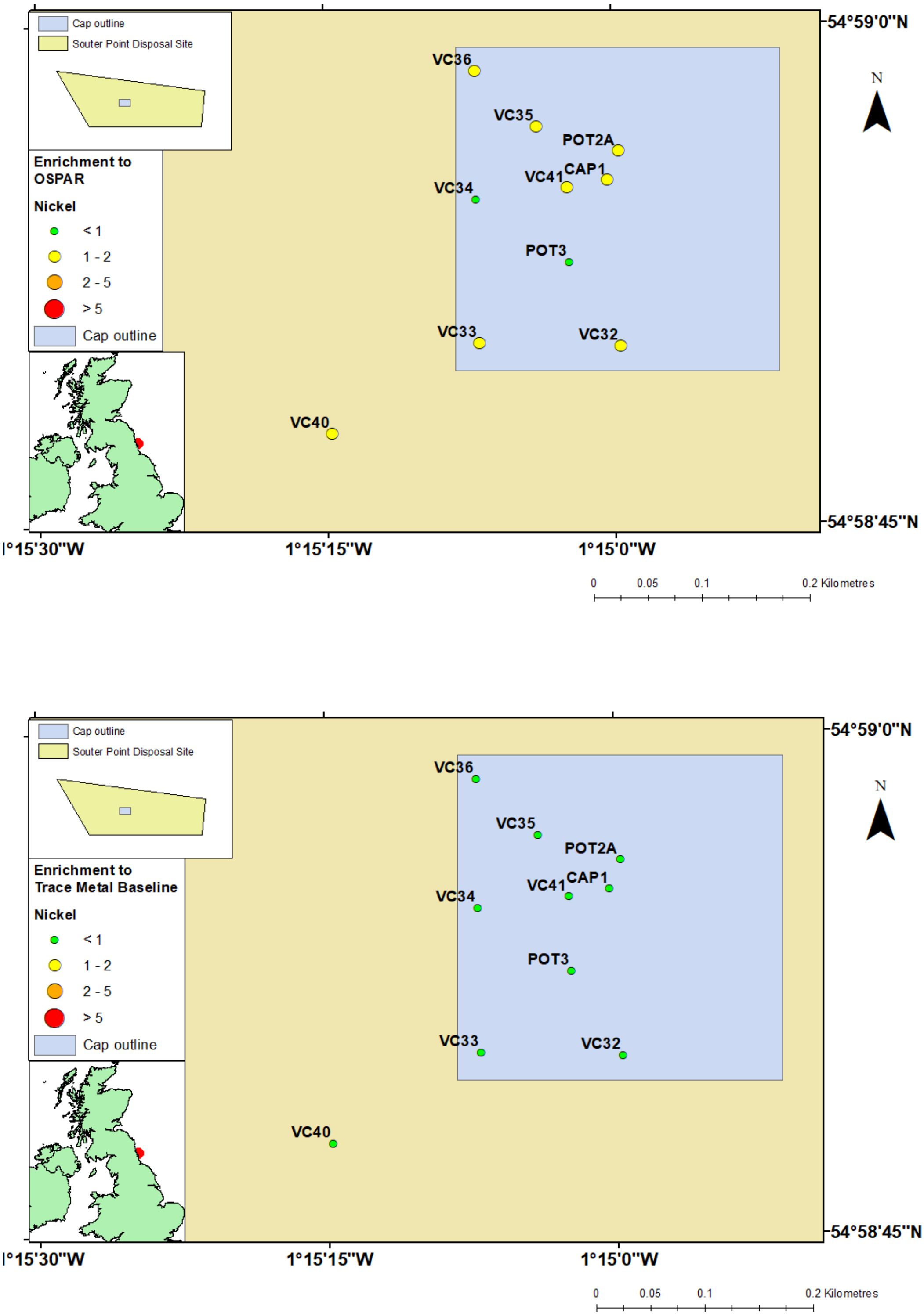
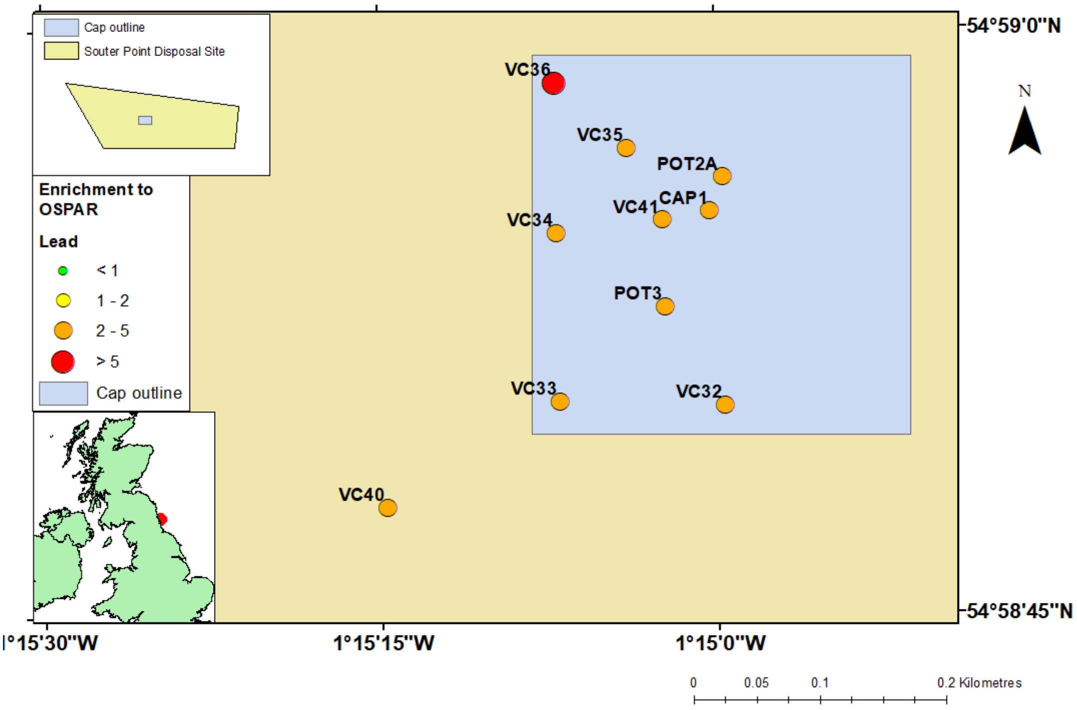


Figure 2.35. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for nickel at Souter Point, December 2022.



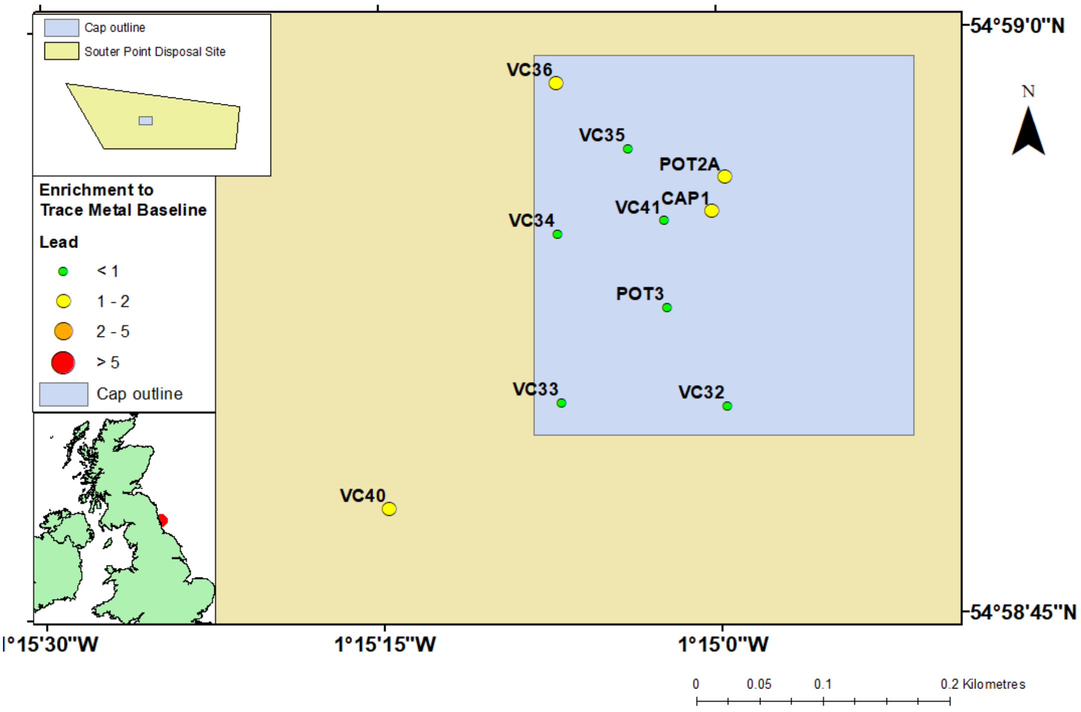


Figure 2.36. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for lead at Souter Point, December 2022.

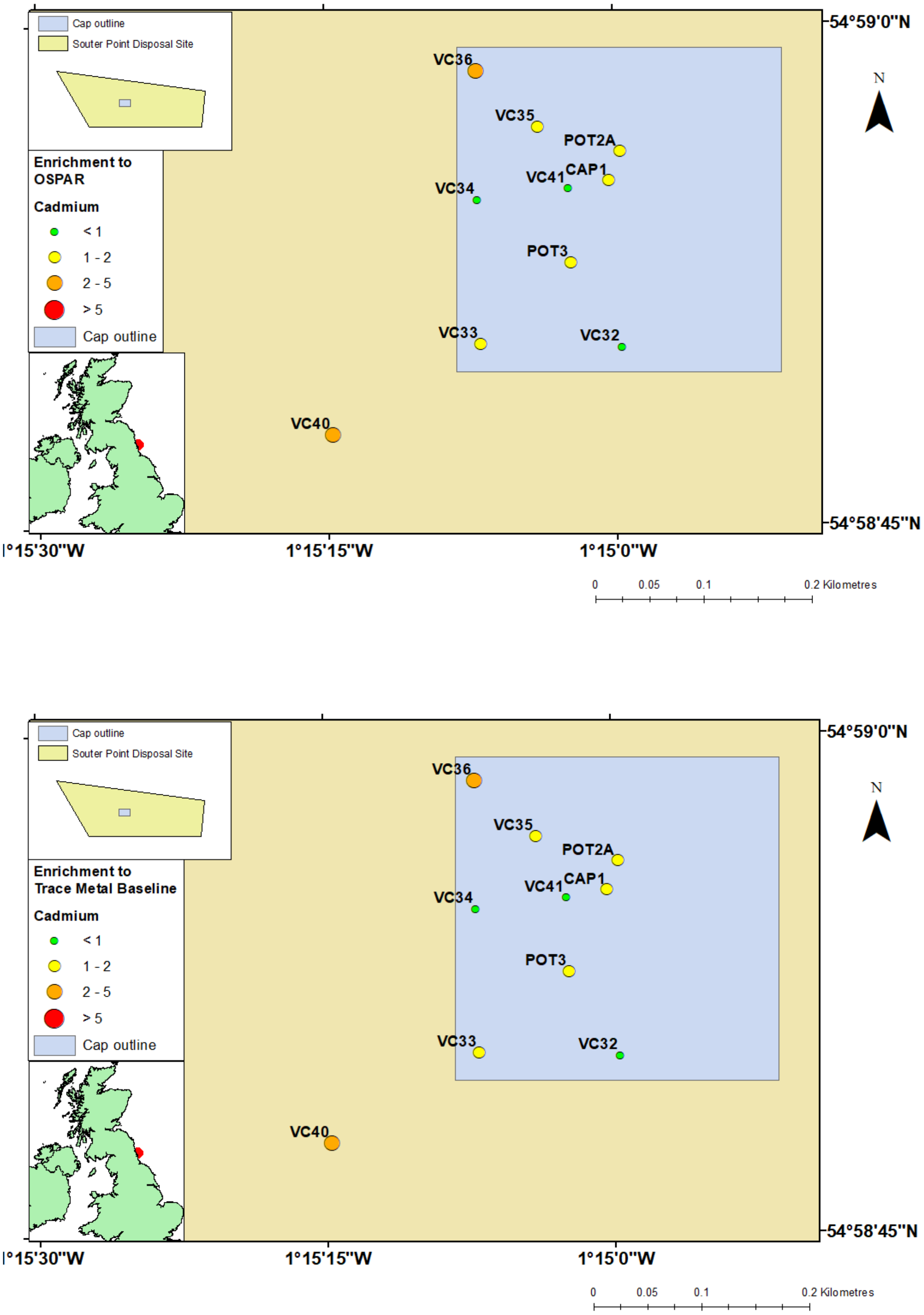


Figure 2.37. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for cadmium at Souter Point, December 2022.

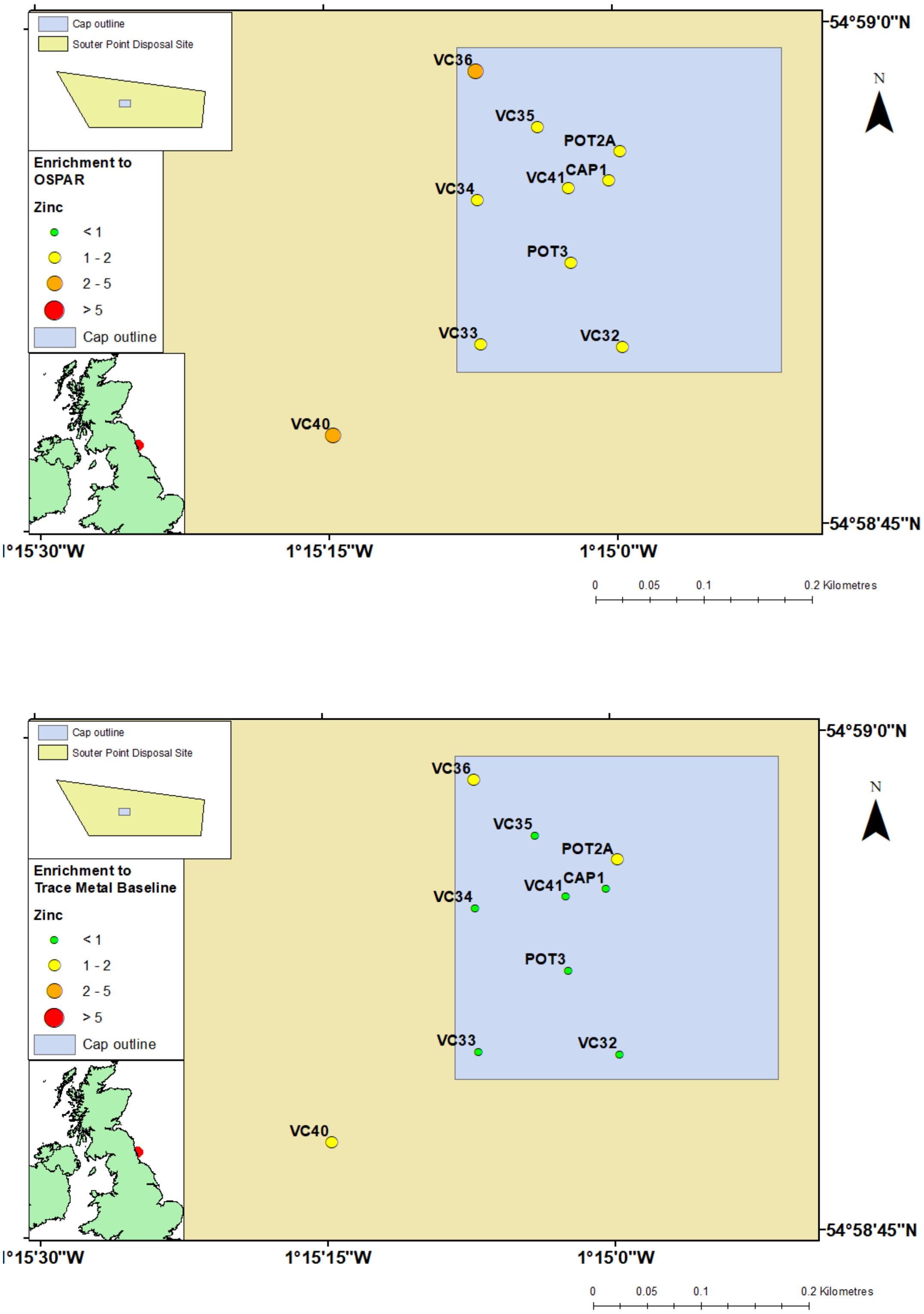


Figure 2.38. Enrichment to OSPAR BACs (upper panel) and regional baseline values (lower panel) for zinc at Souter Point, December 2022.

3 References

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Appendices

4 Appendix 1: Polycyclic aromatic hydrocarbons (PAHs) assessment methods

4.1 Sample extraction

Sediment samples, collected in glass jars, were frozen immediately after collection and not defrosted until required for analysis. Each homogenized wet sediment sample was extracted using alkaline saponification followed by liquid/liquid extraction. A sample of sediment was taken for a total solids determination as all results are reported on a dry weight (dw) basis. The sample extract was then passed through an alumina chromatography column in order to remove polar compounds, concentrated to 1 ml and sealed in a vial. A suite of alkylated and parent PAH were then determined using coupled gas chromatography/mass spectrometry (GC/MS). Quantification was by means of deuterated internal standards added prior to digestion, with analytical quality control samples being run within each sample batch. Full details can be found in Kelly et al. (2000).

4.2 Method used for assessment

Cefas currently has action level limits for contaminants such as trace elements and PCBs but none currently exist for PAHs. Reviews of what has been investigated in other countries has indicated that the most promising of the currently available co-occurrence methods is the Effects Range Low/Effects Range Median (ERL/ERM) methodology which is founded on a large database of sediment toxicity and benthic community information (Long et al., 1998).

The ERL/ERM methodology derives SQGs representing, respectively, the 10th and 50th percentiles of the effects dataset and can be derived for individual PAH compounds. In a regulatory context, where SQGs are to be used as informal (non-regulatory) benchmarks to aid in the interpretation of sediment chemistry (Long et al., 1998), this becomes complicated where a large number for individual PAH are determined, as is usually the case. This has led to separate ERL/ERM derived SQGs being set for “Low molecular weight PAHs” and “High molecular weight PAHs”. In this context, LMW PAHs include 2- and 3-ring PAH compounds: naphthalene; monomethyl naphthalenes; acenaphthene; acenaphthylene; fluorene; phenanthrene; and, anthracene. HMW PAHs include the 4- and 5-ring PAH compounds: fluoranthene; pyrene; benz[*a*]anthracene; chrysene; benzo[*a*]pyrene; and, dibenz[*a,h*]anthracene. Although a wider suite of PAHs is routinely determined for both licensing and monitoring purposes, these can be considered as toxicity markers for the PAH as a whole. The ERL and ERM concentrations applied are given in Table 4.1.

Table 4.1. ERL and ERM concentrations for LMW and HMW PAHs in sediments. The limits for LMW PAH are lower than those for HMW PAH as they carry a higher acute toxicity.

PAH compounds	ERL (µg kg^{-1} dw)	ERM (µg kg^{-1} dw)
LMW PAH	552	3,160
HMW PAH	1,700	9,600

Appendix 2: Organohalogens (OHs) assessment methods

4.3 Methods

Full details of the analytical methodology are given in Bersuder et al. (2020).

4.3.1 Sample extraction

Sediment samples were air dried and sieved (<2mm) in a controlled environment. 10 g of dried sediment were mixed with sodium sulphate, transferred to a glass Soxhlet thimble and topped with 1 cm of sodium sulphate. A solution containing $^{13}C_{12}$-labelled ICES7 PCBs ($^{13}C_{12}$-CB28, $^{13}C_{12}$-CB52, $^{13}C_{12}$-CB101, $^{13}C_{12}$-CB118, $^{13}C_{12}$-CB138, $^{13}C_{12}$-CB153 and $^{13}C_{12}$-CB180), labelled OCPs (d_6-alpha-HCH, d_6-gamma-HCH, $^{13}C_6$-HCB and $^{13}C_{12}$-p,p’-DDT), fluorinated-BDEs (F-BDE69 and F-BDE160), and $^{13}C_{12}$-labelled BDE209 was added as recovery standard to all samples prior to the extraction step. Samples were extracted over a 6 h period using 50:50 *iso*-hexane:acetone, with an average of 9 - 10 cycles h^{-1}. Sulphur residues were removed at this stage with copper filings.

4.3.2 Sample extract clean-up

An aliquot of the Soxhlet extract was cleaned up and using alumina (5% deactivated) columns. The eluate contained polychlorinated biphenyls (PCBs), organochlorine pesticides (OCPs) and polybrominated diphenylethers (PBDEs).

4.3.3 Analysis of PCBs and OCPs by GC-MS/MS

After addition of internal standard CB53, CB112 and CB200, PCB and OCP concentrations were determined with an Agilent 7890A GC coupled with 7000 QQQ-MS/MS in positive electron impact mode (ESI+). The separation of analytes was performed using two 25.0 m × 200 µm, 0.33-µm-film-thickness DB-5 capillary columns (J&W) with a backflush system installed. The carrier gas and collision gas were helium (1.4ml/min) and nitrogen (1.5ml/min), respectively. The initial oven temperature was 90°C, held for 2.00min, then increased to 165°C at 15°C/min, to 285°C at 2°C/min, to 310°C at 40°C/min and finally held for 10 min, with the column backflush instigated when the oven reached 285°C (total run time 71.7 mins). The injector temperature, ion source and quadrupole temperatures were 270°C, 280°C and 150°C, respectively. A 1-µl extract was injected in pulsed-splitless mode with a purge time of 2 min.

4.3.4 Analysis of PBDEs by GC-MS/MS

After addition of internal standard CB200, PBDE concentrations were determined with a Shimadzu 2010plus GC with TQ8050 QQQ-MS/MS in positive electron impact mode (ESI+). The separation of analytes was performed on a 15.0 m × 250 µm, 0.15-µm-film-thickness Rtx-1614 capillary column (Restek). The carrier gas was helium (1.28ml/min) and the collision gas was argon. The initial oven temperature was 120°C, held for 1.00min, then increased to 275°C at 15°C/min, to 300°C at 50°C/min, and finally held for 5 min. The injector temperature and source temperature was 340°C and 230°C, respectively. A 1-µl extract was injected in pulsed-splitless mode with a purge time of 2 min.

4.3.5 Analysis of BDE209 by GC-MS

BDE209 concentrations were determined with an Agilent 6890Plus GC with 5975C MS in NCI mode. The separation of analytes was performed on a 15.0 m x 250 µm, 0.1-µm-film-thickness DB-1 capillary column (J&W). The carrier gas was helium (1.3ml/min constant flow, average velocity 59 cm/s) and the reagent gas was methane (40 psi). The initial oven temperature was 90°C, held for 1.00min, then increased to 200°C at 25°C/min, to 295°C at 10°C/min, and finally held for 20 min. The injector temperature and detector temperature was 250°C and 200°C, respectively. A 1-µl extract was injected in pulsed splitless mode with a 20psi pulse until 1 min and a purge time of 2 min.

4.3.6 Quantitation methods

The identification of PCBs and OCPs was based on the retention time of individual standards in the calibration mixtures. Quantitation was performed using internal standards and 9 calibration levels (range 0.1 – 200 ng/ml). The combined PCB and OCP standard solutions contained the following 41 compounds in iso-octane: Hexachlorobenzene; hexachlorobutadiene, alpha-HCH, beta-HCH, gamma-HCH, *p,p'*-DDE, *p,p'*-TDE, *p,p'*-DDT, *o,p'*-DDE, *o,p'*-TDE, *o,p'*-DDT, dieldrin, heptachlor, heptachlor epoxide, endosulfan-I, endosulfan-II, endosulfan sulfate; IUPAC CB101; IUPAC CB105; IUPAC CB110; IUPAC CB118; IUPAC CB128; IUPAC CB138; IUPAC CB141; IUPAC CB149; IUPAC CB151; IUPAC CB153; IUPAC CB156; IUPAC CB158; IUPAC CB170; IUPAC CB18; IUPAC CB180; IUPAC CB183; IUPAC CB187; IUPAC CB194; IUPAC CB28; IUPAC CB31; IUPAC CB44; IUPAC CB47; IUPAC CB49; IUPAC CB52; IUPAC CB66. Concentrations were corrected for the recovery of the labelled recovery standards.

Quantitation for PBDEs was performed using internal standards and 10 calibration levels (range 0.05 – 100 ng/ml). The PBDE standard solutions contained the following 11 compounds in iso-octane: IUPAC BDE17; IUPAC BDE28; IUPAC BDE47; IUPAC BDE66; IUPAC BDE100; IUPAC BDE99; IUPAC BDE85; IUPAC BDE154; IUPAC BDE153; IUPAC BDE138; IUPAC BDE183; plus an additional 13 compounds: IUPAC BDE3; IUPAC BDE7; IUPAC BDE15; IUPAC BDE49; IUPAC BDE71; IUPAC BDE77; IUPAC BDE119; IUPAC BDE126; IUPAC BDE156; IUPAC BDE184; IUPAC BDE191; IUPAC BDE196; IUPAC BDE197; together with the internal standard IUPAC CB200 and recovery standards F-BDE69 and F-BDE-160. Concentrations were corrected for the recovery of the F-BDE recovery standards.

Quantitation of BDE209 was performed using an internal standard and 7 calibration levels (range 0.5 – 500 ng/ml). The BDE209 standard solutions contained IUPAC BDE209 in iso-octane, plus an additional 3 compounds IUPAC BDE206; IUPAC BDE207; IUPAC BDE208; together with the internal standard $^{13}C_{12}$- labelled IUPAC BDE209.

4.3.7 Quality assurance/ quality control procedures

AQC procedures included reagents purification, method blanks, and use of control charts created from repeated analysis of the NIST-1944 Certified Reference Material (CRM) and Quasimeme CEMP-245 materials.

4.4 Method used for assessment

PCB, OCP and BDE concentrations were determined in the sediments and reported on a dry weight basis. The ∑ICES 7 CBs (CB28, CB52, CB118, CB153, CB138, CB170, CB183), and the sum of all 25 measured CBs (∑CBs) were calculated, together with ∑DDTs (*p,p'*-DDE, *p,p'*-TDE, *p,p'*-DDT, *o,p'*-DDE, *o,p'*-TDE, *o,p'*-DDT). Where individual congener concentrations were below the limit of detection (LOD) of 0.02 µg/kg, a value of half the LOD was inserted for calculation of summed concentrations. The ∑11 BDEs were calculated. Where individual congener concentrations were below the LOD of 0.02 µg/kg, a value of half the LOD was inserted for calculation of summed concentrations. The congener patterns were evaluated, with BDE183 a marker constituent of the octa-BDE technical mix, and the other BDEs constituents of the penta-BDE technical mix. Additionally, BDE209 ("Deca BDE") concentrations were calculated. Where BDE209 concentrations were below the LOD of 0.1 µg/kg, a value of half the LOD was inserted.

The Total Organic Carbon (TOC) content in the <2 mm fraction determined at the sampling stations was used to additionally calculate the contaminant concentration normalised to 2.5% TOC content.

Concentrations of PCBs and OCPs in the sediment were compared with various action limits, to investigate whether any adverse effects in benthic biota were likely to expected as a consequence of their presence. There are no action limits available to compare PBDE concentrations with at the present, although some were recently proposed. Concentrations are expressed on a dry weight basis unless otherwise stated.

The current Cefas action limits for dredge disposal are: PCBs Action level 1 if ∑ICES7 CBs > 10 µg/kg or ∑CBs > 20 µg/kg and action levels 2 if ∑CBs > 200 µg/kg; OCPs Action level 1 if ∑DDTs > 1 µg/kg, dieldrin > 1 µg/kg, no Action level 2 for either ∑DDTs or dieldrin. Concentrations are expressed on a dry weight basis.

OSPAR in Charting Progress2 (CP2) have set criteria for Background Assessment Concentrations (BAC) and Environmental Assessment Concentrations (EAC) for the ICES7 CBs in sediments (see Table 2). Concentrations are expressed in µg/kg dry weight normalised to 2.5% organic carbon. Concentrations below BACs would be considered to have high environmental status.

Concentrations significantly below EACs could be considered to have good environmental status and those above, bad environmental status.

Table 4.2. OSPAR assessment criteria for CBs in sediment from CP2.

Sediment (µg/kg dry weight, normalised to 2.5% TOC)		
Compound	**BAC**	**EAC**
CB28	0.22	1.70
CB52	0.12	2.70
CB101	0.14	3.00
CB118	0.17	0.60
CB138	0.15	7.90
CB153	0.19	40.00
CB180	0.10	12.00

OSPAR MIME have recently adopted the Canadian FEQG (Federal Environmental Quality Guidelines) levels as EAC results for PBDEs, and also calculated BAC values. These thresholds are shown in Table 4.3.

Table 4.3. OSPAR assessment criteria for BDEs in sediment from Canadian FEQGs.

Sediment (µg/kg dry weight, normalised to 2.5% TOC)		
Compound	**BAC**	**EAC**
BDE28	0.05	110.00
BDE47	0.05	97.50
BDE66	0.05	97.50
BDE85	0.05	1.00
BDE99	0.05	1.00
BDE100	0.05	1.00
BDE153	0.05	1100.00
BDE154	0.05	1100.00
BDE183	0.05	14000.00
BDE209	0.05	47.50

Appendix 3: Trace metals assessment methods

4.5 Introduction

4.5.1 Methodology

The sediment samples were collected in plastic bags and were frozen immediately after collection. The samples were collected for PSA and metal analyses on the <63µm fraction. Details on obtaining the <63µm sediment fraction can be found in the Particle Size Analysis technical report.

The sample is digested in a mixture of hydrofluoric, hydrochloric and nitric acids using enclosed vessel microwave, the digest is made up in 1% nitric acid and further diluted prior to analysis by Inductively Coupled Plasma-Mass Spectrometry (ICP-MS) and Inductively Coupled Plasma-Mass Spectroscopy (ICP-AES). Quantification of Al, As, Cd, Cr, Cu, Fe, Li, Mn, Ni, Pb, Rb and Zn is done using external calibration with Indium as internal standard. A certified reference material is run within each sample batch for quality control. Results are reported in mg kg^{-1} (ppm).

Some samples contained insufficient silt/clay (<63µm) for trace metals determination. Values lower than the detection limits were omitted from the dataset. This is because several detection limits were higher than real values, causing spurious elevations.

4.6 Numerical assessments

In order to assess relative level of trace metal contamination for a sampled station, enrichment to a baseline is required. This report presents two assessment methods; (i) comparisons with OSPAR Background Assessment Concentrations (BACs) as in previous reports, and (ii) comparisons with regional baseline concentrations.

Enrichment ratio is defined as:

$$\frac{\text{Metal raw value}}{\text{OSPAR BAC or proposed baseline value}}$$

Enrichment is arbitrary defined in four levels:

0-1: no enrichment

1-2: slight enrichment

2-5: moderate enrichment

>5: high enrichment

4.6.1 OSPAR BACs

OSPAR (BACs) are defined for Clean Seas Environment Monitoring Programme (CSEMP) assessment to determine temporal trends in concentrations (OSPAR, 2008). They are derived from Background Concentrations (BCs) which are based on concentrations recorded in ‘pristine’ areas.

As trace metal concentrations used for in this study were measured on the fine fraction of sediment (<63 µm), normalisation to some extent has already been completed, therefore all enrichment calculation were based on the raw data.

4.6.2 Regional baselines

The above-mentioned Background Concentrations (BCs) are based on concentrations recorded in ‘pristine’ areas. There is only one set of values assigned by OSPAR for the whole North Atlantic (http://www.ospar.org). However, trace metal concentrations are known to show regional variation in the UK, largely related to the variable geology around the coast and historical industrial activity in the early 19th Century which has caused localised elevated levels (Ridgeway et al, 2003; Rowlatt and Lovell, 1994). Therefore, for assessing enrichments at disposal sites, Cefas have developed regional baselines utilising various spatial datasets around England and Wales. Recently, an extensive study was carried out on 8 regions defined in the Clean Seas and Environment Programme (CSEMP) (Figure 4.1) and the proposed metals baselines concentration derived from this study have additionally been used in this report as a validation tool to i/ compare with OSPAR BACs values and ii/ to assess the credibility of using those proposed baselines values instead of the OSPAR BACs values when studying for metals enrichment. The proposed baselines for the areas are given in Table 4.4, along with the corresponding OSPAR BACs values for each metal (OSPAR, 2006).

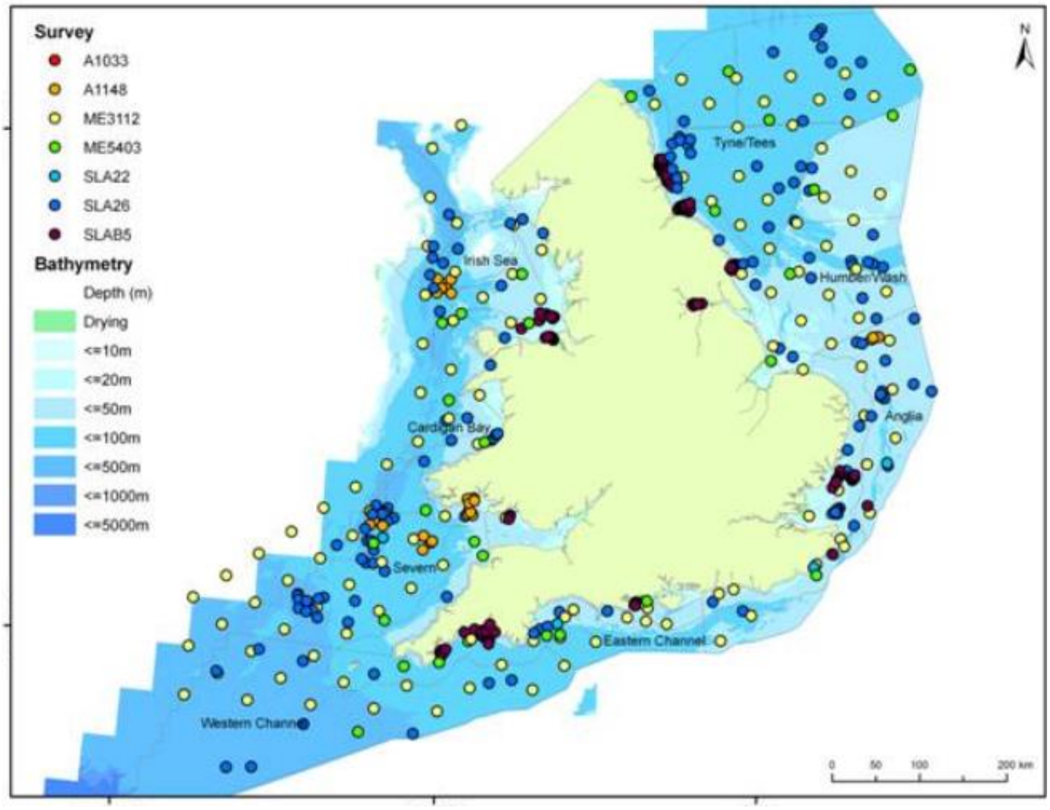

Figure 4.1. Location of stations sampled to provide metals data as part of the regional baseline approach.

Table 4.4. OSPAR BACs with proposed baselines for regions covered in disposal site assessment in 2011.

	As (mg/kg)	Cd (mg/kg)	Cr (mg/kg)	Cu (mg/kg)	Hg (mg/kg)	Ni (mg/kg)	Pb (mg/kg)	Zn (mg/kg)
Anglia	33	0.20	115	40	0.16	56	46	130
Cardigan Bay	26	0.29	103	26	0.12	44	73	145
Eastern Channel	23	0.18	90	26	0.12	31	45	107
Humber Wash	30	0.17	109	31	0.21	44	67	129
Irish Sea	21	0.29	115	38	0.43	47	77	240
Severn	21	0.20	81	27	0.10	36	47	135
Tyne/Tees	27	0.31	135	29	0.35	55	131	171
West Channel	34	0.19	105	72	0.77	50	108	153
OSPAC BAC	25	0.31	81	27	0.07	36	38	122

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