The Salt Marshes of Cornwall (the Environment of the Camel, Fal, Gannel and Hayle)

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THE SALT MARSHES OF CORNWALL

The environment of the Camel, Fal, Gannel and Hayle estuaries

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PREFACE

Most of this research came from my Ph.D. from the Camborne School of Mines (CSM), which looked at the relationships between flora and mine pollution in Cornwall. Much of the data herein was distributed in various papers, some easy to find, many hidden away in the darkest corners of Cornish files and desks. Really, I thought, what we need is for someone to collect this data together in some easy to find publication. In some ways, that was my Ph.D. However, even then, theses aren’t always that easy to find. That’s when it struck me. By condensing my research into my literature search, I could produce a document useful for ecologists, geologists, naturalists, civil engineers and others interested in the environment of Cornwall, whether for research, general interest or development purposes.

If you’re wondering what didn’t make it into the book for my Ph.D., well, I researched vegetation zonation in relation to mining pollution using canonical correspondence analysis. As well as the geochemical maps presented in this report, I also took samples from specific areas to test the bioavailability of metals in sediments.

Another aspect from my Ph.D. was to test the ability of *Salicornia* to act as a biomonitor - something that’s already achieved with macroalgae (seaweed). And, to put it succinctly, I found that, yes, indeed, the plant did covary with copper and zinc in the sediment.

So, whatever your reasons for interest, I hope you find the book interesting and of use.

ACKNOWLEDGEMENTS

I would like to thank the staff and students of CSM for their interest and support, especially, Drs. Loveday Jenkin, Terri Stoddern, Duncan Pirrie and John Coggan, with a general thank you to all the academics that gave me their time when they could have shown me the door. A special thanks to the fellow Ph.D.'ers from 2000-2005 and the CSM football squad (two wins to one loss in the Bottle Match during my time there).

I would like to thank the Camborne School of Mines Trust Fund for the majority of funding for this research. I would like to thank the following organisations for their support-in-kind: Carrick, Kerrier, Penwith and Restormel District Councils, Cornwall Wildlife Trust, RSPB, the Environmental Records Centre for Cornwall and the Isles of Scilly and the Duchy of Cornwall for providing access and information on the sites for this research.

And, of course, biggest thanks of all to Zeinab for, well, just about everything really.

The Salt Marshes of Cornwall

The environment of the Camel, Fal, Gannel and Hayle estuaries

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1.1 Salt Marsh Formation

Salt marshes have been defined as flat, poorly drained areas of land, subject to periodic or occasional flooding by salt water and are usually covered by a thick mat of grassy halophytic plants (Bates & Jackson, 1980). Estuarine salt marsh develops where sediment becomes deposited upon intertidal mud flats. As surface elevation increases, sediment becomes sufficiently exposed for pioneer plant species to colonise. Further sediment is trapped by plant structures and, combined with vegetation reducing wave action, elevation increases. The reduced affect from the sea allows succession to take place (Williams *et al.*, 1994b; Boorman, 2003). The stabilizing effect on the sediment by vegetation is species specific with vegetation type, cover and height all influencing the accretion rate (Langlois *et al.*, 2003).

If the marsh continues to accumulate sediment with no controlling factors, the marsh would continue to build until there was no effect from the sea. Salt marshes may persist by either transgression (a rise in sea level relative to land) or progradation (sediment progresses seaward and eventually gains enough elevation for succession to begin) (Viles & Spencer, 1995).

Continuation of a salt marsh depends on the relative elevation of the marsh surface to tidal height. Relative elevation is principally determined by four factors (Allen, 2000):

- Space in which salt marsh can develop further (primarily influenced by changes in sea level and tidal range, along with storm events)
- Sediment supply, along with sediment characteristics
- Productivity, particularly subterranean, of halophytes
- Autocompaction, which can provide vertical accommodation space

Short periods of accretion can have a profound effect upon the continuation of the marsh (Boorman, 2003).

1.2 Salt Marsh Dynamics

The salt marsh water table is subject to both temporal and spatial variation (Williams *et al.*, 1994b). In the low marsh region, the water table is at or near the sediment surface. Groundwater levels fluctuate particularly in the higher marsh. Water table is highest during flood conditions where the sediment may become waterlogged. Flooding can be damaging to the salt marsh, as nutrient uptake by vegetation is impaired during flood conditions (Boorman, 2000), whilst toxic substances are able to accumulate (Boorman, 2003). Groundwater levels are determined by frequency of duration of inundation, salt marsh morphology and sediment type - affecting porosity and moisture retention (Williams *et al.*, 1994b). In some marshes, sediment water movement can be significant and impact upon the vegetation by transporting nutrients and organic matter, as well as affecting the oxic status of the marsh (Boorman, 2003).

As well as the regular ebb and flow of tides, plus the input of freshwater from rivers, storm events can be particularly damaging to a salt marsh. Not only can the physical force of increased wave action result in damage to the stability of the marsh (Boorman, 2003) but also salt pulses have been suggested as a critical factor in plant zonation (Brewer & Grace, 1990)

The mixing of river and seawater produces layering of different salinities in estuaries (Forstner & Whittman, 1981; Gerbhardt *et al.*, 2005) exerting a profound effect upon sedimentation and flocculation mechanisms (Kautsky, 1998). Passage from marine to freshwater environments promotes precipitation and flocculation of colloidal material (Gerbhardt *et al.*, 2005). As clay particles from riverine material mixes with the high ionic status of seawater, the potential for cation exchange is high. Clays, having a negative charge, are able to absorb positively-charged particles, such as metals, on to their surface. Positively-charged particles may be exchanged dependent upon the strength of charge, for example, adsorbed calcium ions may be exchanged for sodium. The ion exchange is an attempt for river-borne material to gain chemical equilibrium when moving into a marine environment, otherwise known as halmyrolysis (Andrews *et al.*, 1996). As larger flocs settle out, they may be broken down into smaller particles and re-suspended causing an increase in turbidity (Gerbhardt *et al.*, 2005).

Pollutants cluster in a similar way to natural sediments, often causing flocculation upstream of salt boundaries (Forstner & Whittman, 1981). In the UK, most of the sediment is transported from the marine environment by tidal currents (Boorman, 2003). Sediments are constantly re-suspended through both wind and wave-induced turbulence (Boorman, 2003) notably between the upstream point of an incoming tide and the lower point during ebb flow. The area where most re-suspension occurs is categorised as the turbidity maxima. The exchange between dissolved, bound and particulate matter is elevated in the turbidity maxima (Andrews *et al.*, 1996) resulting in a mid-estuary maximum of dissolved metals (Gerringa *et al.*, 2001). Around 93-95% of the suspended and 20-40% of the dissolved riverine material is deposited worldwide within this zone (Lisitsyn, 1995).

Salt marshes are extremely productive systems, with *Spartina* dominated marshes being equal to most fertile agricultural lands (Cacador *et al.*, 2000), although below ground productivity has been found to be four times that of the aerial portion (Boorman, 2000). However, turnover of organic matter within the marsh is also extremely high as halophytes must use large quantities of energy for use in solute-control by manufacture of osmotically-balancing biochemicals, direct ion-pumping as in salt glands or production of succulent tissue with its high relative respiratory cost (Etherington, 1983). Increased nitrate has been observed to intensify plant growth in salt marshes (Valiela & Teal, 1979, Kiehl *et al*., 1997) and is generally regarded as the critical limiting factor for growth (Tyler, 1971; Levine *et al.*, 1998). Phosphorus has also been identified as a limiting factor in some marshes (Hazelden and Boorman, 1999).

The majority of organic matter and nutrients are provided by tidal influence (Boorman, 2003), though freshwater input can also provide significant nutrients (Dawes, 1981). Despite often being limited by nitrogen and phosphorus, salt marshes have a net release of these compounds. This suggests that, although marshes can act as reservoirs of nutrients (Boorman, 2003), the storage abilities within the marsh are essentially unstable. Organic matter may also be lost from the marsh to the sea with studies on *Spartina* (de la Cruz & Gabriel, 1974) revealing that decay rates are especially rapid. Therefore, upon death (both of plant parts and the plant itself), vegetative material is quickly decomposed and released as a food source to organisms, ranging from microbes to larger animals such as crustaceans and annelids. Soil fauna, microalgae and bacteria further complicate the dynamics of salt marshes by altering sediment accretion and water availability (Boorman, 2003).

1.3 Salt Marsh Zonation

Salt marshes characteristically produce clear vegetation zonation patterns (Williams *et al*., 1994b). Physical factors that influence community zonation include salinity (highly toxic to most plants, halophytes have the ability to either exclude salt or to compartmentalise safely within cells), oxygen availability (tidal inundation reduces root and microbial oxygen availability - salt marsh plants can adapt by either transporting oxygen from aerial portions or through fermentation), hydrogen sulphide (produced in reduced environments and toxic to most plants) and a variety of other factors (a great deal of abiotic variables can contribute, such as temperature, nutrient availability and ionic composition of the sediment) (Hmieleski, 1994). Boorman (2003) suggests that these factors can be summarised as being related to the height of the marsh in relation to local tidal regime. Cantero *et al*. (1998) suggested that salinity is the most responsible for floristic diversity, in that species richness decreased according to salinity. Other research has also indicated that salinity is the most important factor to account for plant zonation (Wilson *et al.*, 1996; Sanchez *et al.*, 1998). Brewer &

Grace (1990) concluded, however, that the general salinity of the marsh was less important in zonation when compared to occasional storm-generated salt pulses.

The reduction of physical stress through increased elevation leads to an increase in species richness (Huckle *et al*., 2000). In the mid to high salt marsh, competition is then believed to be the major zonation factor (Gray, 1992; Pennings & Callaway, 1992; Hacker & Bertness, 1999; Boorman, 2003). A change in species is notable between areas of high physical stress and those where competition is prevalent suggesting tolerance to adverse conditions and competitiveness are mutually exclusive (Grime, 1979). Indeed, Boorman (1966) observed that many salt marsh plants had improved growth when propagated in non-saline habitats. A lack of competitiveness with non-halophytic plants however restricted growth to salt marshes.

In general, the following zones can be commonly observed in UK salt marshes: -

1. *Pioneer* Open communities with one or more of the following – *Spartina* spp., *Salicornia* spp., *Aster tripolium*. Zone covered by all tides except the lowest neap tides.

2. *Low marsh* Generally closed communities with at least *Puccinellia maritima* and *Atriplex portulacoides* as well as the previous species. Zone covered by most tides.

3. *Mid-marsh* Generally closed communities with *Limonium* spp. and/or *Plantago*, as well as the previous species. Zone covered only by spring tides.

4. *High marsh* Generally closed communities with one or more of the following – *Festuca rubra*, *Armeria maritima*, *Elytrigia* spp., as well as the previous species. Zone covered only by highest spring tides. In wetter areas of the high marsh, *Juncus gerardii* and *J. maritimus* are often present as a colony.

5. *Transition zone* Vegetation intermediate between the high marsh and adjoining non-halophytic areas. Zone covered only occasionally by tidal surges during extreme storm events. In the UK, many salt marshes are enclosed by a sea wall, thus limiting the formation of a transition zone.

(Boorman, 2003)

Few findings are available on the relationship between salt marsh zonation and metal pollution. Jenkin (1996) observed that vegetation in Restronguet Creek - an estuary in Cornwall subject to historical mining pollution - was characterised by less diverse, robust vegetation compared with the significantly less-polluted site of the Camel Estuary. *Armeria maritima*, prevalent in Cornish metal-rich estuaries, was hypothesised as a possible indicator of metal pollution. *Armeria* has previously been espoused as an indicator of high levels of copper in bogs (Farago *et al*., 1980).

Smillie (2006) concluded that there were clear divisions in vegetation encountered in the more polluted estuaries of Hayle and Restronguet Creek and Hayle, compared to that of the Gannel and Camel. This research pointed to a successional relationship between *Armeria* and *Plantago*. *Armeria* was considered to be associated with more contaminated sediments than *Plantago*. In marshes of a low to moderate pollution status, *Armeria* was either absent, or in a community with *Plantago* due to the species being unable, or having a limited ability, to compete with non-tolerant species. However, in heavily contaminated estuaries, *Plantago* was restricted in its ability to become established, thus allowing a virtual monoculture of *Armeria* to develop. The high physical stress of estuarine ecosystems tends to lead to relatively low species diversity with large natural fluctuations in abundance (Sanders *et al*., 1994). Subtle toxic effects are, thus, difficult to perceive in such systems due to the lack of response of vegetation.

1.4 Floristic Adaptation in Salt Marshes

Halophytic vegetation is present under extremely stressful conditions, particularly in the low-mid salt marsh. Tidal inundation has been reported as the most important factor when determining salt marsh zonation (Rozema *et al*., 1985). A successional pattern develops from low to high marsh based upon tolerance to salinity, exposure and temperature.

Salinity is a particular hazard in coastal salt marshes. Sodium possesses the power to alter soil structure (Hopkins, 1995). The change in porosity will result in decreased oxygen and hydraulic conductance. Physiological drought may occur as the increased ionisation of the sediment will lower the water potential (Hopkins, 1995). Osmosis cannot occur across an area of low water potential to high, therefore both water and nutrients are difficult for the plant to obtain. Sodium and chloride ions are capable of causing harm by damaging membranes, inhibiting enzyme activity or disrupting metabolic functions (Hopkins, 1995).

Halophytes may be grouped into two classes according to their adaptations to salt stress (Hopkins, 1995). Salt regulators may exclude salt from their roots, therefore not absorbing salt. Other salt regulators may absorb salt but, using salt glands located in their leaves, excrete large quantities, thus reducing any harmful effects (Anderson, 1974; Hansen *et al*., 1976).

Salt accumulators absorb high concentrations into cell vacuoles (Hopkins, 1995). The high ionic concentration lowers the water potential of the plant, thus water can be diffused from the surrounding sediment. Despite the high ionic potential of the cell vacuole, the cytoplasm maintains a low salt concentration (Hopkins, 1995).

Plants incapable of tolerating anything except the smallest amounts of salt are classed as glycophytes. Many plants important in agriculture are part of this classification (Hopkins, 1995).

Oxygen availability is reduced during inundation periods primarily but also in the receding tide when the sediment is still saturated. One response a plant may utilise in such conditions is to transport oxygen from the aerial portion to the root zone via the aerenchyma tissue (Hmieleski, 1994). This can then result in an oxidised rhizosphere quite different to the bulk sediment. Although in terrestrial systems, this oxidised micro-environment may only penetrate a few millimetres, in salt marsh sediments, the oxic environment may be expanded to tens of centimetres (Cacador *et al*., 1996; Madureira *et al*., 1997). The highest concentration of oxygenated sediments through root action is however between 30-100 mm in salt marshes (Caetano & Vale, 2002).

2 HISTORICAL METAL MINING IN CORNWALL

The exploitation of mineral ore has been dated back to the second millennium BC (Gerard, 2000). Cornish tin mining most likely started in the early Bronze Age, with British tin bronzes probably exclusively produced in the southwest of England (Northover, 1982). After a reduction in production in the middle Bronze Age due to “native hostility” (Northover, 1982), the early Iron Age coincided with a substantial increase in production (Northover, 1982). Evidence from contemporary Roman sources suggest that tin from the southwest of England was traded internationally (Gerard, 2000). Tin mining continued through the Roman Period and the Dark Ages with little change in production, probably due to the presence of Spanish mines also providing metal to the Roman Empire (Gerard, 2000). By the 17th Century, east Cornwall had taken over from Devon and western Cornwall as the primary source of tin production in the southwest (Gerard, 2000). The peak of tin and copper production was achieved in the southwest in the late 19th Century, with significant production of arsenic, lead, zinc, tungsten, silver and uranium also reported (Burt, 1998). Tin mining ceased in Cornwall in 1998 (Pirrie *et al.*, 2003).

Streamworking is often the first stage in metal exploitation (Lewis, 1908). This process involves locating mineral grains that have detached from the main lode and eventually transported to a resting place. Using water to wash away lighter sand and clays, the heavier metallic minerals could be isolated (panning). This method involves releasing large volumes of waste into rivers with subsequent transport to estuaries (Gerard, 2000). Excessive siltation of ports meant laws were introduced to curb this form of industrial pollution as far back as the 14th Century (Gerard, 2000). Although this practice became progressively less important, substantial production still existed up until the 18th Century (Gerard, 2000).

Hard rock mining began in the 13th Century, although probably only in very rich lodes (Gerard, 2000). Initially, the ground was mined for tin then for copper and a range of other metals (Pirrie *et al.*, 2003). Openwork mining used opencast quarries to exploit metal lodes. The result of these quarries was narrow elongated valleys, many of which still remain etched into the landscape (Gerard, 2000). Subterranean lodes could be exploited either by lode-backpits (characterised by numerous shallow shafts) or shaft mining (notable for few shafts but at very deep levels) (Gerard, 2000). Drainage initially involved construction of an adit to convey water away from the mine by gravity, however the 16th Century saw the rise of pumping equipment designed to make underground mining more economical (Gerard, 2000). Both of these means of drainage still involved conveying contaminated water to river systems. The result of tailings from both streamworking and mining was rapid and excessive siltation of rivers and estuaries (Pirrie & Camm, 1999).

Mining can adversely affect the environment through land clearance, erosion of spoil tips, hydrological effects, impacts on the ecosystems, disruption of natural and human transport systems, loss of amenity, socio-economic impact, as well as through human health & safety (Merefield, 1995). Deforestation within southwest England has been correlated with smelting practices and use of wood for timber props, safeguarding mines from collapse (Gerard, 2000).

The following sites are presented in this publication: -

- Restronguet Creek - located within the Fal estuary on the south coast of Cornwall (Figure 1), approximately 6.4 km southwest of Truro
- Hayle Estuary – the most south-westerly estuary in the UK (Figure 1). This estuary is divided into two pools, Copperhouse and Lelant.
- Camel Estuary - located in northeast Cornwall (Figure 1), adjacent to the tourist resorts of Padstow, Wadebridge, Polzeath and Rock.
- Gannel Estuary - located to the immediate west of Newquay in the northern portion of Cornwall (Figure 1).

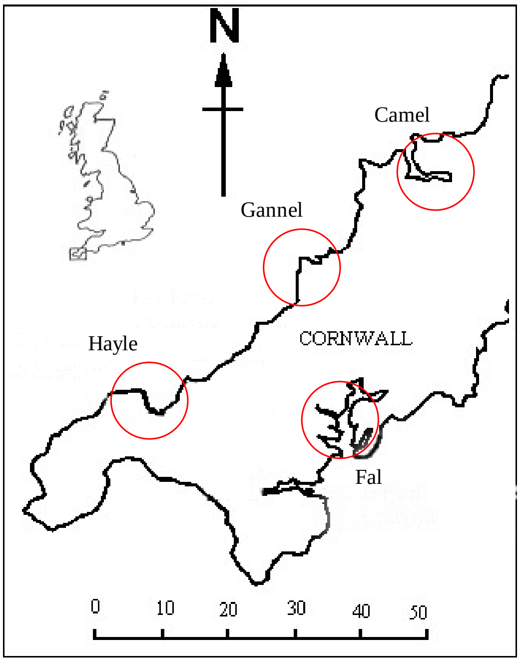

Figure 1 Location of the estuaries in this publication.

A description of the physical and ecological characteristics plus anthropogenic influences, with a particular emphasis on mining activity, is described in the following pages, listed by site. The borders of designated areas were confirmed by up-to-date searches of the websites of the Joint Nature Conservation Committee (JNCC, 2008) and the Multi-Agency Geographic Information for the Countryside (MAGIC, 2008). The position of woodland and trees are taken from the English Nature National Inventory of Woodland and Trees (MAGIC, 2008). Areas described as *'Salt Marsh (sample area)'* in the various figures, refer to the research undertaken by Smillie (2006).

3 RESTRONGUET CREEK

3.1 *Mining History*

Restronguet Creek (Figure 2) has been receiving waste from deep mining for hundreds of years (Younger, 2002). Alluvial tin has, however, been recovered for several thousand years until recently (Bryan & Gibbs, 1983), whilst underground mining occurred in Restronguet Creek during the nineteenth century (Dines, 1956). During the nineteenth century, the catchment area of the Fal Estuary was considered the major world source of copper, tin and arsenic (Bryan & Gibbs, 1983). Siltation of the creek was recorded at 1.8 m during the period 1597 to 1693 (Stapleton & Pethick, 1995), whilst 42 feet of sediment (approximately 13 m) was noted to have accumulated in Restronguet Pool between the years 1698 to 1855 (Hill & MacAlister, 1906). This sedimentation is probably attributable to mining productivity as Stapleton & Pethick (1995) state that there has been no significant change in inter-tidal positioning in the last 100 years. This coincides with a significant reduction in mining activity within the catchment.

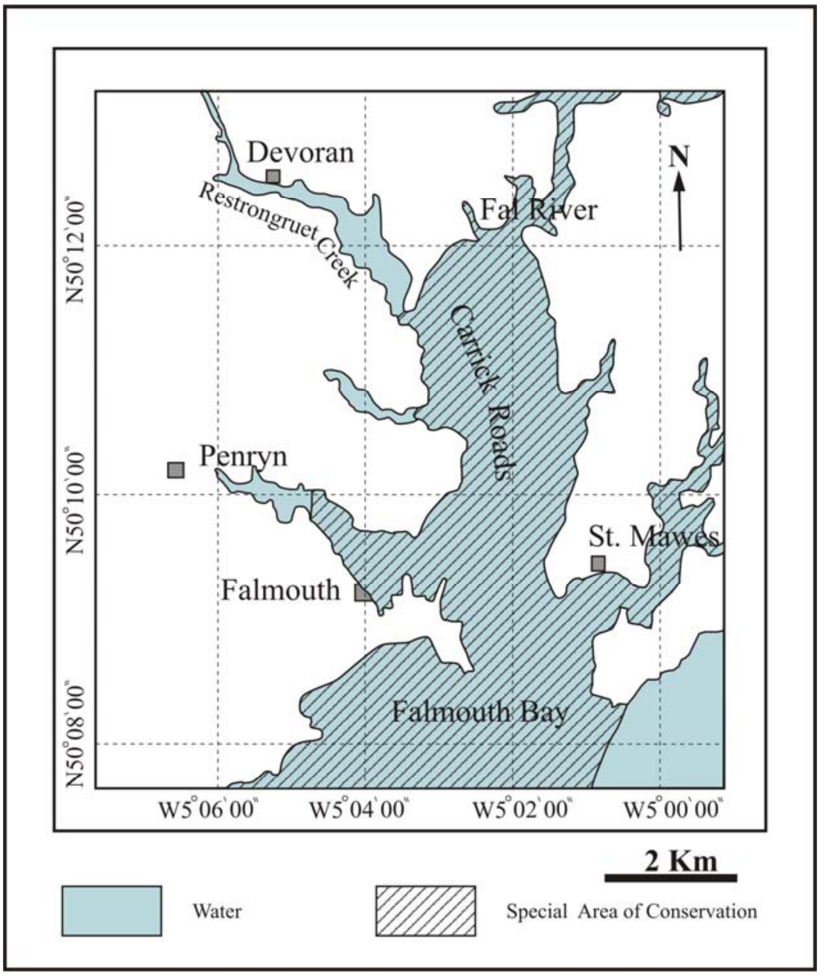

Figure 2 The Fal Estuary showing the location of Restronguet Creek and the limits of the Special Area of Conservation

Restronguet Creek receives the outflow of the Carnon River (Figure 3), which drains the historical copper and tin mining districts served by the County Adit (Bryan & Gibbs, 1983). The Carnon River receives around 50% (dependent upon seasonal variations) of its flow from the County Adit - a tributary which typically has a pH of less than 4 and exceptionally high metal loadings (Environment Agency, 1997a). The drainage from old copper mines provides a source of dissolved copper, whereas drainage from Wheal Jane tin mine is a source of dissolved zinc (Bryan & Gibbs, 1983). Other studies (Hosking & Obial, 1966; Pirrie & Camm, 1999; Pirrie *et al.*, 2003) suggest that historical particulate contamination of the estuary supplies a significant amount of metal through leaching. The result of such extensive mining is that Restronguet Creek has become the most contaminated area within the Fal Estuary (Pirrie *et al.*, 1996).

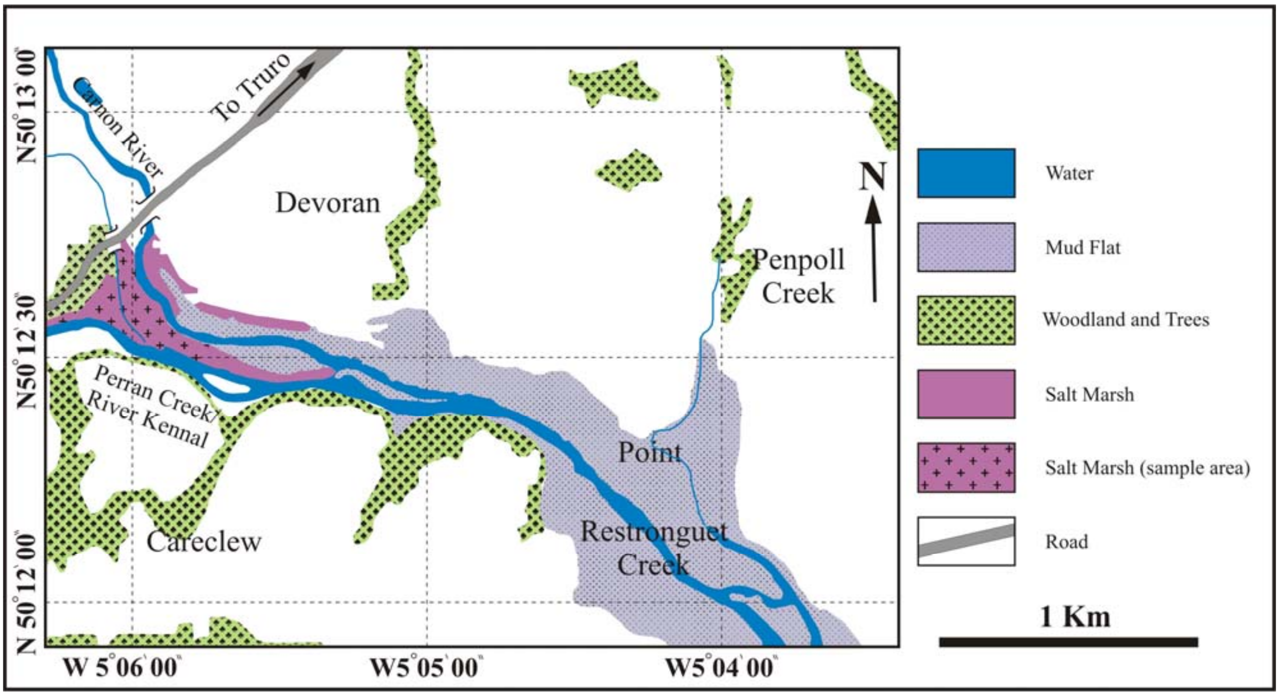

Figure 3 Restronguet Creek

A highly visible pollution incident occurred in 1992. Metal-rich water recorded with a pH of 3.1 (Somerfield *et al.*, 1994a) derived from Wheal Jane, discharged into Restronguet Creek (Banks *et al.*, 1997; Younger, 2002), resulting in a pollution plume. The impact from the incident was reduced due to limited mixing as the dense estuarine water buoyed the contaminated mine water (Somerfield *et al.*, 1994a), resulting in the plume being dispersed seawards.

3.2 Landscape and Geomorphology

The Fal Estuary is an example of a ria - a drowned river system (Pirrie & Camm, 1999) and consists of six tidal rivers and creeks: Penryn and Percuil rivers, plus Restronguet, Mylor, Pill and St. Just creeks (Ratcliffe 1997). The western side of the estuary is characterised by industrial development and modern day settlement, as opposed to the less accessible, more rural eastern side (Countryside Commission, 1996). Most of the farmland associated with the estuary consists of small fields arranged in an irregular pattern, characteristic of anciently enclosed land (Countryside Commission, 1996). Other areas, particularly to the west, consist of more open large-scale farming schemes due to 18th and 19th Century reorganisation of field systems (Countryside Commission, 1996). In places, fields extend to the edge of the shoreline. However, in other areas, the shore is bordered by unimproved grassland or scrub. Within the upper stretches and bordering the creeks, ancient semi-natural broadleaved woodland has developed (Countryside Commission, 1996).

The Fal Estuary lies within the Cornwall Area of Outstanding Natural Beauty (AONB), however, Restronguet Creek is outside the border of this designation (Environment Agency 1997a).

3.3 Ecology

3.3.1 Ecological Status

The Fal Estuary is recognised as being of European importance through its designation as a Special Area of Conservation (SAC) in the Fal and Helford SAC (Figure 2). The reasons implicated for this designation are (1) exceptionally biologically rich sandbanks, (2) sheltered intertidal mudflats and sandflats, (3) large shallow inlets and bays supporting a rich sediment biota, and (4) typical ria salt marsh vegetation with tidal rivers overhung by fringing trees. Additionally, the presence of the protected species *Rumex rupestris* is also cited as a reason for the designation, although no mention is made of the location of this species (JNCC, 2008).

Restronguet Creek is outside the borders of the SAC but is included within the local designation of an Area of Great Scientific Value (AGSV) – a classification designed to protect and enhance nearby designated sites by providing buffer zones, wildlife corridors and to enable a concentration of resources through emphasizing the most important areas of nature conservation (Environment Agency, 1997a). The occurrence of reduced diversity metal-tolerant communities was noted to be of nature conservation interest by the Environment Agency (1997a).

3.3.2 Floristic Studies

The Salt Marsh Survey of Great Britain (Burd, 1989) describes the salt marsh as being dominated in the upper reaches by a *Salicornia/ Suaeda* assemblage with pockets of *Armeria* bordering the landward and southern reaches of this assemblage. Towards the western edge of the marsh, the site then becomes dominated by swards of *Agrostis, Festuca* and *Armeria* with *Salicornia* present between these stands. Scrub woodland is present towards the south and as an individual copse within the *Agrostis/ Festuca/ Armeria* dominated area.

Figure 4 Restronguet Creek displaying clear vegetation zonation

A survey by the Cornwall Naturalists Trust (1980d) describes the marsh as not very diverse but exhibiting zonation well. The salt marsh sample area is said to be dominated by a *Salicornia/ Spergularia* mix, in contrast to the *Salicornia/ Suaeda* community described by Burd (1989). Distinct communities of dominant *Festuca* and abundant *Armeria* are present towards the upper reaches of the marsh. *Festuca* and *Atriplex* with *Aster* and *Suaeda* dominate the higher ground.

Smillie (2006) confirmed that a *Salicornia – Spergularia* community dominates the pioneer zone in agreement with Cornwall Naturalists Trust (1980d). This matrix is unusual in that it does not appear in the National Vegetation Classification[1] (NVC) of salt marshes (Rodwell, 2000), although there is a *Salicornia* dominated community (SM8). Bounding the tree lined River Kennal is a transitional low marsh assemblage, similar in some ways to NVC SM10 but without *Puccinellia* and with a substantial presence of *Armeria* and *Festuca* – probably in transition to SM16. This zone ends abruptly with the now-vegetated structure of an abandoned ruined bridge.

Bordering the Carnon River is a grassy sward of predominantly *Agrostis-Festuca* comparable with SM16. Within this zone are wetter areas with some surface water (flushes). These flushes are similar to SM8, though they exhibit a greater biodiversity.

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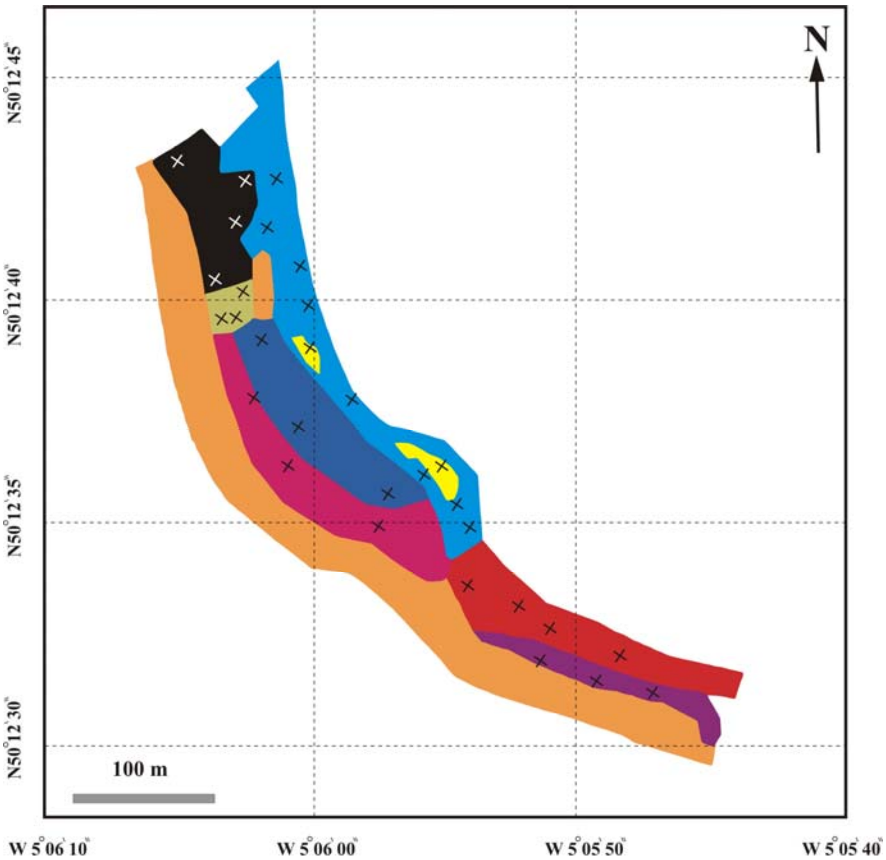
[1] Please refer to Appendix I for a guide to the NVC communities identified in this publication and to Appendix II for the results of the Smillie (2006) survey.

The majority of the remainder of the salt marsh contains a *Salicornia-Spergularia* dominated community. However, an *Armeria* dominated community quite unlike any described in Rodwell (2000) is also present. SM13d is probably the closest fit to this community but without the presence of *Plantago*, *Puccinellia* or *Triglochin*. An isolated stand of mature trees is also present.

Smillie (2006) observed that vegetation zonation cannot be directly related to metal concentration, i.e. zones with similar communities do not necessarily have similar metal concentrations. This does not imply that metal content does not influence vegetation zonation, as the impact of elevated metal concentrations has been reported as causing a number of reactions, such as a reduction in new species arrivals (Zobel *et al.*, 1999), development of metal-tolerant communities in only part of contaminated sites (Lepp *et al.*, 1997) and a reduction in species diversity (Murray *et al.*, 2000).

The study by Smillie (2006) also indicated that the elements most associated with salinity also do not correspond with the vegetation zonation. However, preferential tidal flow may be influential as the ruined bridge may restrict tidal inundation.

According to the Redfield ratio (Redfield, 1963), nitrate is the limiting factor within the survey area of Smillie (2006). Nevertheless, nitrate cannot be directly linked with vegetation zonation.



Zone	Dominant Vegetation	NVC Community
A	*Salicornia/ Spergularia*	SM8 + *Spergularia*
B	*Armeria/ Festuca*	SM10 - *Puccinellia*
C	*Agrostis / Festuca*	SM16
D	*Salicornia* flushes	SM8
E	*Salicornia/ Spergularia*	SM8 + *Spergularia*
F	*Salicornia*	SM8
G	*Armeria*	SM13d - *Puccinellia/ Plantago*
H	*Spergularia/ Salicornia*	*Spergularia* + SM8
I	Trees and shrubs (not surveyed)	N/a

Figure 5 Vegetation zones within Restronguet Creek salt marsh (Smillie, 2006)

Sample points are marked with (X). The survey suggests the zones within Restronguet Creek differ from NVC, particularly with an abundance of *Spergularia* and lack of *Puccinellia* and *Plantago*.

3.3.2.1 Macroalgae

The brown seaweed *Fucus vesiculosus* has been used by the Environment Agency to biomonitor heavy metals within Restronguet Creek. Samples taken in 1991 demonstrated high concentrations of metals (Cu = 150 mg/kg; Zn = 774 mg/kg; Fe = 222 mg/kg – all data expressed as dry weight) reflecting elevated metal concentrations of Restronguet Creek. Following the Wheal Jane Incident, a further increase in metal-tissue concentration was observed (Environment Agency, 1997a) (e.g. March 1992, Cu = 270 mg/kg; Zn = 869 mg/kg; Fe = 2958 mg/kg – all data expressed as dry weight) (Environment Agency, 1997a).

3.3.3 Invertebrate Studies

Invertebrate fauna within Restronguet Creek is limited compared to other Cornish estuaries (Environment Agency, 1997a) but higher than predicted with regard to elevated metal levels (Bryan *et al.,* 1987). The high metal loadings historically associated with the estuary have resulted in Restronguet Creek developing a metallophytic biota significantly different from other estuaries in southwest England (Somerfield *et al.*, 1994a; Warwick, 2001). Not only were species with endemic tolerance to high metal concentrations present but also species that have developed tolerant strains (Bryan *et al.,* 1987, Millward & Grant, 1995). Somerfield *et al.*, 1994a, b noted a spatial variation with copepods related to metal concentration, whilst Millward & Grant (2000) alleged that the distribution of copper-tolerant nematode assemblages exhibited a clear relationship with the level of copper contamination in sediments. An abundance of opportunistic metal-tolerant annelids was also noted (Warwick, 2001).

Biopsies of soft tissues of polychaete worms (Williams *et al.*, 1998) and crustaceans (Pedersen & Lundebye, 1996) revealed elevated concentrations of copper; though in neither of these studies did the organisms reflect the metal concentrations within the sediment. During a nationwide study of British estuaries, Rainbow *et al.* (1989) revealed *Orchestia gammarellus* had elevated tissue concentrations of copper, zinc and cadmium within Restronguet Creek. The study concluded that this organism could be used to biomonitor copper and zinc in British coastal waters. This was in contrast to the elevated copper and zinc concentrations in *Talitrus saltator* and *Talorchestia deshayesii* that was not repeated in other estuaries, making these organisms unsuitable for biomonitoring (Rainbow *et al.*, 1989).

3.4 Geology

The Fal Estuary is underlain by Devonian metasediments, intruded by Carnmenellis Granite to the east and St Austell Granite to the west (Leveridge *et al.*, 1990). The metasediments are comprised of two formations, namely, the Mylor Slate Formation (which outcrops throughout the estuary) and the Portscatho Formation (outcropping to the north and west). The catchment area also drains the Devonian Porthtowan and Bovisland formations (Pirrie *et al.*, 2003).

Within the Devonian metasediments, NE-SW orientated lodes (veins) host significant concentrations of metallic minerals, dominated by an early assemblage of cassiterite, wolframite and arsenopyrite and a later assemblage dominated by copper-zinc-iron-arsenic-lead sulphides (Scrivener & Shepherd, 1998). Younger N-S orientated veins have developed along extensional faults and are dominated by a lead, silver and zinc assemblage (Scrivener & Shepherd, 1998). Subsequent weathering has led to the formation of fluvial cassiterite placer deposits (Camm & Hosking, 1985; Camm, 1999). Hydrothermal alteration and subsequent chemical weathering have lead to extensive kaolinization in the western part of the St Austell Granite (Manning *et al.*, 1996; Bristow, 1998).

3.5 Geochemistry and Mineralogy

Sedimentological and mineralogical studies have reported high concentrations of both metal and arsenic (Hosking & Obial, 1966; Pirrie *et al.*, 1997; Hughes, 1999; Pirrie & Camm, 1999; Pirrie *et al.*, 2003). Previous studies (Bryan & Gibbs, 1983; Somerfield *et al.*, 1994a; Williams *et al.*, 1998; Pedersen & Lundebye, 1996) have assumed the Carnon River to be the prime source of metal contamination. Bryan & Gibbs (1983) report that as the metal-rich Carnon River water enters Restronguet Creek, mixing with seawater takes place. The resultant rise in salinity and pH promotes flocculation, notably of iron, copper and arsenic. These elements may then be removed from the system and deposited as sediment. Manganese, nickel and zinc are not generally removed in this manner. Decontamination is thus more dependent on dilution of freshwater with seawater (Bryan & Gibbs, 1983).

Data from a selection of previous surface sediment studies on Restronguet Creek show a wide variation in metal concentrations, both between and within metals (Table 1).

Millward & Grant (1995) recorded copper concentrations at 1978 mg/kg, however, subsequent research found concentrations to be raised at 2263 mg/kg (Millward & Grant, 2000). These results are however, within the range determined by Bryan *et al.* (1980) at between 1733-2540 mg/kg. Smillie (2006) recorded maximum concentrations at a higher value of 2665 mg/kg, though mean average results were within the range recorded by Bryan *et al.* (1980). Pirrie *et al.* (2003) records maximum copper concentrations significantly higher at 5073 mg/kg.

Zinc concentrations decreased from 7462 mg/kg (Millward & Grant, 1995) to 4231 mg/kg (Millward & Grant, 2000). Pirrie *et al.* (2003) however then recorded a maximum concentration of zinc at 6600 mg/kg. All of these results are substantially higher than the range (1587-3515 mg/kg) recorded by Bryan *et al.* (1980). Smillie (2006) recorded maximum concentrations within these limits but an average concentration of only 1231 mg/kg.

Lead concentration within Restronguet Creek was noted at 400 mg/kg by Millward & Grant (2000). Although no investigation into lead concentrations within sediments was made by these researchers in 1995, this result is in line with the maximum concentration found by Bryan *et al*. (1980) at 396 mg/kg. This is broadly in line with Smillie (2006), who found average concentrations 406 mg/kg. Pirrie *et al.* (2003), however, records over double this concentration at a maximum of 902 mg/kg.

Table 1 Summary of a selection of geochemical data from sediment studies in Restronguet Creek

Author	Cu	Zn	Fe	Mn	Pb	Sn	Ni	As
Smillie, 2006	2665 (694-6436)	1231 (471-3030)	89467 (75880-102830)	1014 (616-1925)	406 (260-555)	3754 (211-6442)	40 (10-77)	3579 (2604-4948)
Pirrie et al., 2003	507-	6600	-	-	902	3400	-	2803
Bryan et al., 1980	2148 (1733-2540)	2701 (1587-3515)	56033 (47500-63000)	475 (401-559	297 (204-396)	1917 (1350-2672)	31 (25-37)	1732 (1076-2520)

Results expressed as mg/kg of dry weight. All results are mean concentrations except (+) indicating maximum concentrations. (-) indicates element not reported. Results also include the range in parenthesis where available.

Although one interpretation of these varying results may be a temporal variation, a more likely explanation would be a spatial dissimilarity, as results from zinc, in particular, do not follow any specific pattern of increase or decrease over time. Although all samples were taken from surface sediments, differences in depth of sampling may result in differing metal concentrations. This would not however account for the discrepancies between the Millward & Grant (1995, 2000) findings, which utilised similar sampling methods. Thus, it appears that as well as seaward differences in metal concentration (Pirrie *et al.*, 2003), there is also a wide variation of metal concentrations within specific sample sites.

The historical deposition of particulate material (Hosking & Obial, 1966; Pirrie & Camm, 1999) has been suggested as a source of long-term metal release (Pirrie *et al.*, 2003). The oxidation of metal sulphides through both physical processes and bioturbation may result in previously unavailable compounds being released to the food web. The importance of these particulates is emphasized by the correlation between sediment geochemistry and mineralogy (Pirrie *et al.*, 2003). Nevertheless, the metals within Restronguet Creek are generally of low bioavailability, with previous studies over-estimating metal availability by using nitric acid digestion – a method also likely to have released sulphidic minerals (Pirrie *et al.*, 2003).

Additional geochemical data are presented in Table 2 from Smillie (2006). These additional elements were analysed to provide data regarding salinity (sodium, potassium, manganese and chlorine) and nutrient status (nitrate is normally the limiting factor for plant growth in estuaries but calcium, sulphur and phosphorus are also common limiting factors). pH and LOF (a measure of organic matter) were analysed due to their importance in metal speciation, whilst moisture content can be a factor limiting the spread of mid to upper marsh species.

Table 2 Further geochemical data from Restronguet Creek from Smillie (2006)

Author	Na	K	Mg	Cl	Ca
Smillie, 2006	5407 (2664-7400)	14237 (9877-17264)	5326 (3840-7020)	2807 (763-5455)	4229 (1278-9088)
NO_3-N	**S**	**P**	**Moisture**	**pH**	**LOF**
9 (0-26)	2737 (1500-9000)	742 (396-1188)	37 (8-85)	6.2* (4.3-7.4)*	80600 (9700-194800

Results expressed as mg/kg of dry weight except (*) as pH units. All results are mean concentrations with the range in parenthesis. n = 30

Spatial geochemical data taken from Smillie (2006) is presented a series of contour maps (Figure 6 to Figure 10) using Surfer software.

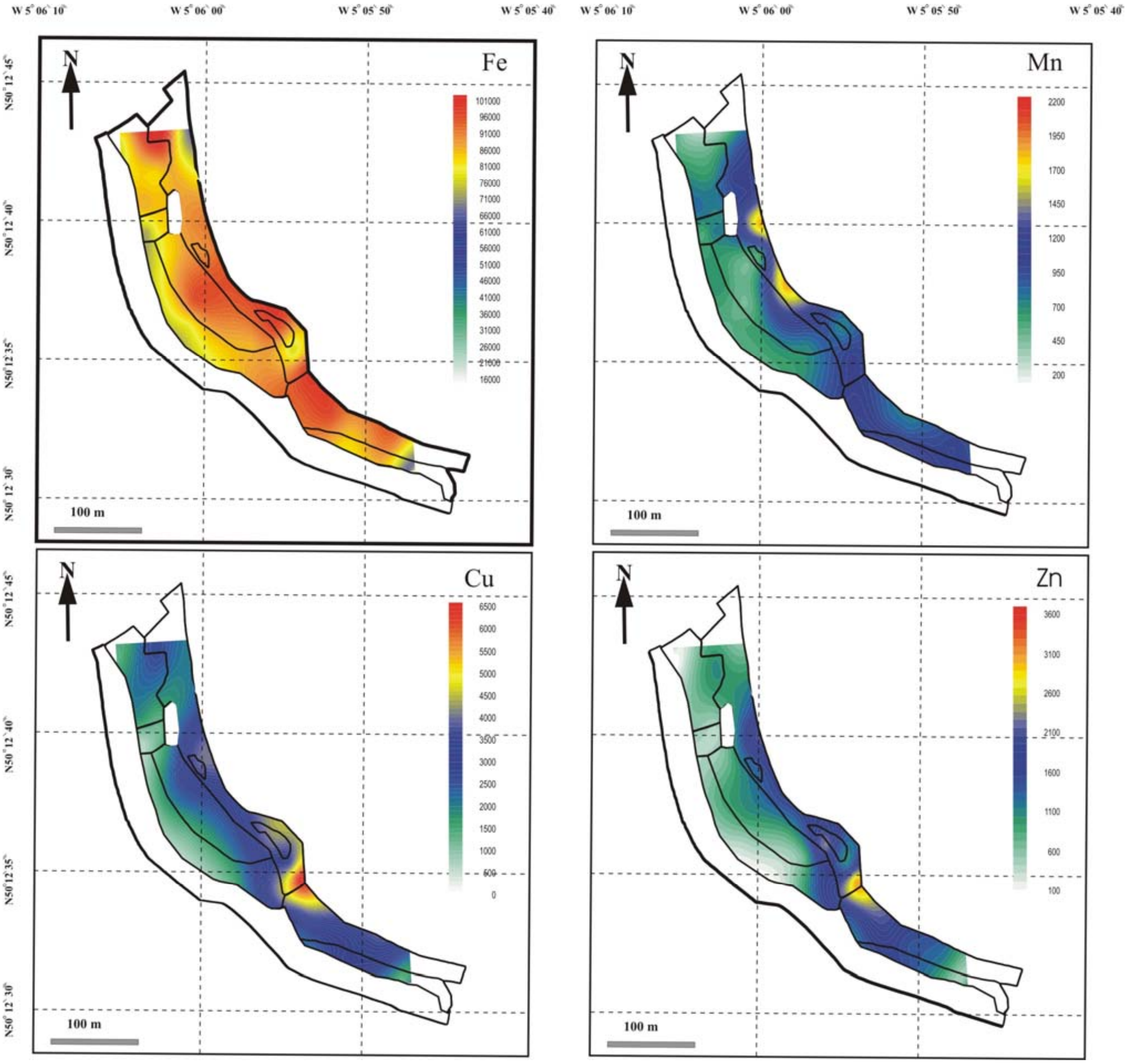


Figure 6 Spatial variation of iron, manganese, copper and zinc in Restronguet Creek

The distribution of iron, copper, zinc and manganese is related somewhat to proximity to the River Carnon (Figure 6). A ‘hot spot’ (i.e. an individual sample where concentrations are unusually high compared with the surroundings) for copper (approaching 6400 mg/kg) and zinc (over 3500 mg/kg) can be seen.

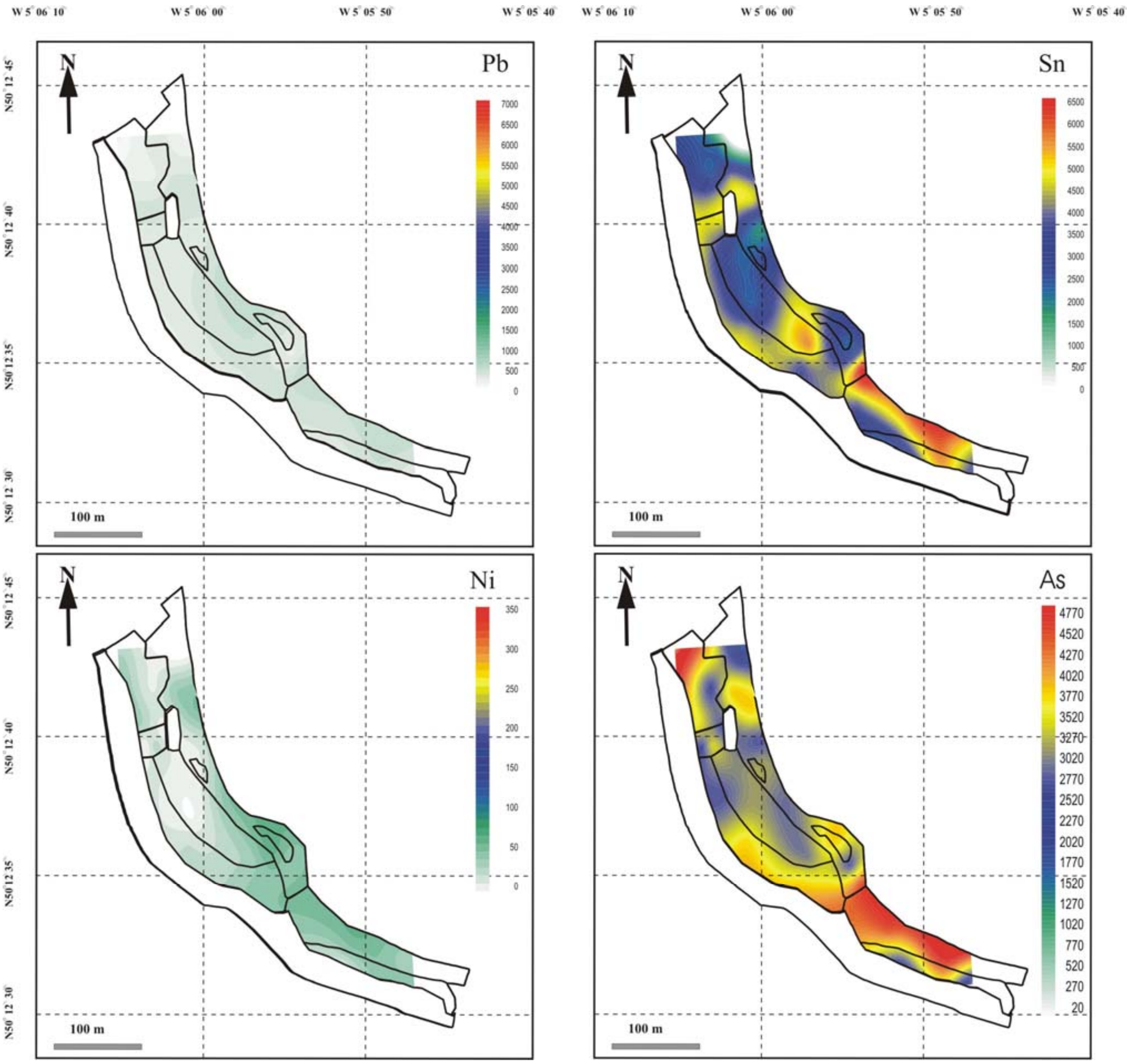


Figure 7 Spatial variations of lead, tin, nickel and arsenic in Restronguet Creek

Neither lead nor tin display much variation throughout the salt marsh of Restronguet Creek. Arsenic is variable throughout the salt marsh, with maximum concentrations being found at the northwest and southeast areas of the sample area. The hot spot recorded for copper and zinc also displays high tin and arsenic.

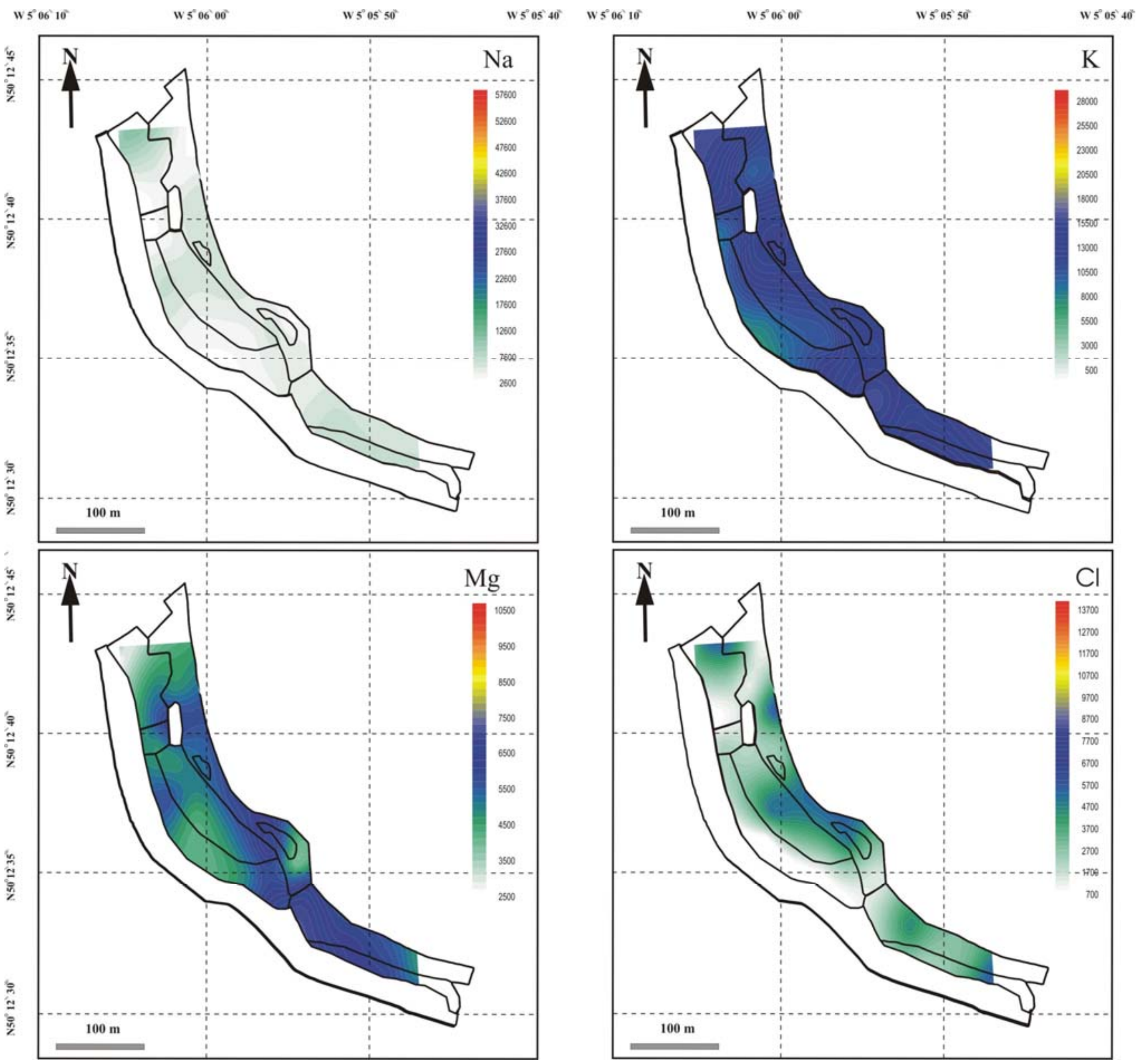


Figure 8 Spatial variations of sodium, potassium, magnesium and chlorine in Restronguet Creek

Both sodium and potassium do not exhibit clear differences over the survey area. Magnesium and chlorine are also fairly even spread, however, there is some evidence of a landward decrease towards the northwest section of the survey area.

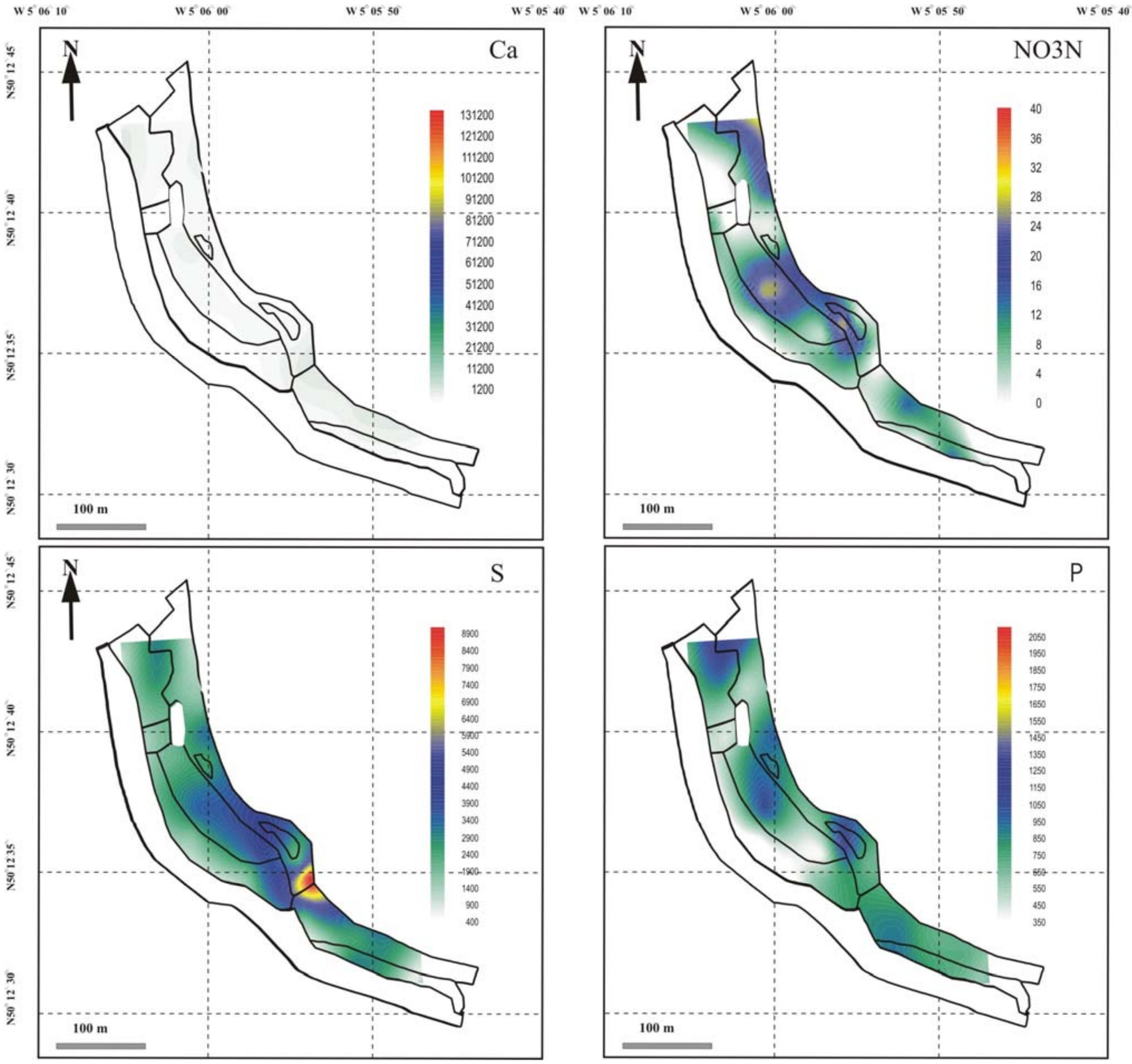


Figure 9 Spatial variations of calcium, nitrate-nitrogen, sulphur and phosphorus in Restronguet Creek

Calcium is reasonably uniformly spread throughout the salt marsh. Although to some extent, nitrate, phosphorus and sulphur are generally higher in the area adjacent to the Carnon River Figure 9), concentrations are variable across the survey area. The hot spot recorded for copper and zinc also displays extremely high levels of sulphur (9000 mg/kg).

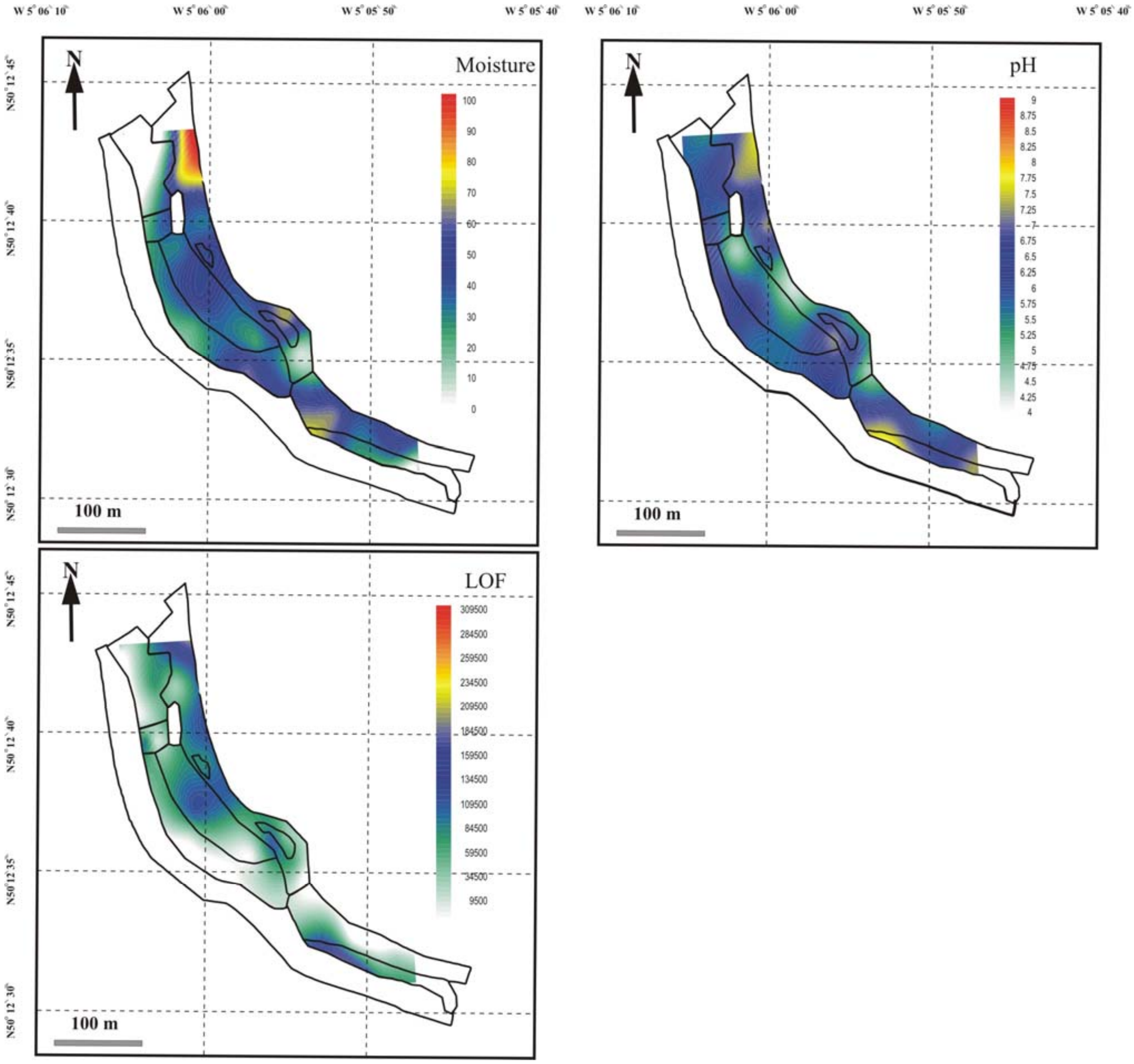


Figure 10 Spatial variations of moisture content, pH and loss-on-fractionation in Restronguet Creek

Moisture content is lowest in the northwest of the sample area, possibly indicative of both distance from tidal influence and the River Carnon. Overall, LOF, moisture and pH are variable throughout the sample area.

4 HAYLE

4.1 *Mining History*

Prior to 1700, metal contamination within the Hayle Estuary (Figure 1) was the result of upstream tin placer mines (Pirrie *et al.*, 1999). From the eighteenth century onwards, Hayle was an important tin and copper smelting area (Pirrie *et al.*, 1999). A copper smelter was established in 1721, with another beginning production in 1758 (Pirrie *et al.*, 1999). A tin smelter ran from around 1816 to 1821 before transferring production to disused copper furnaces at Copperhouse (Pirrie *et al.*, 1999). Another tin smelter ran from 1837 to 1908, with other smelters operating in the nearby area (Pirrie *et al.*, 1999).

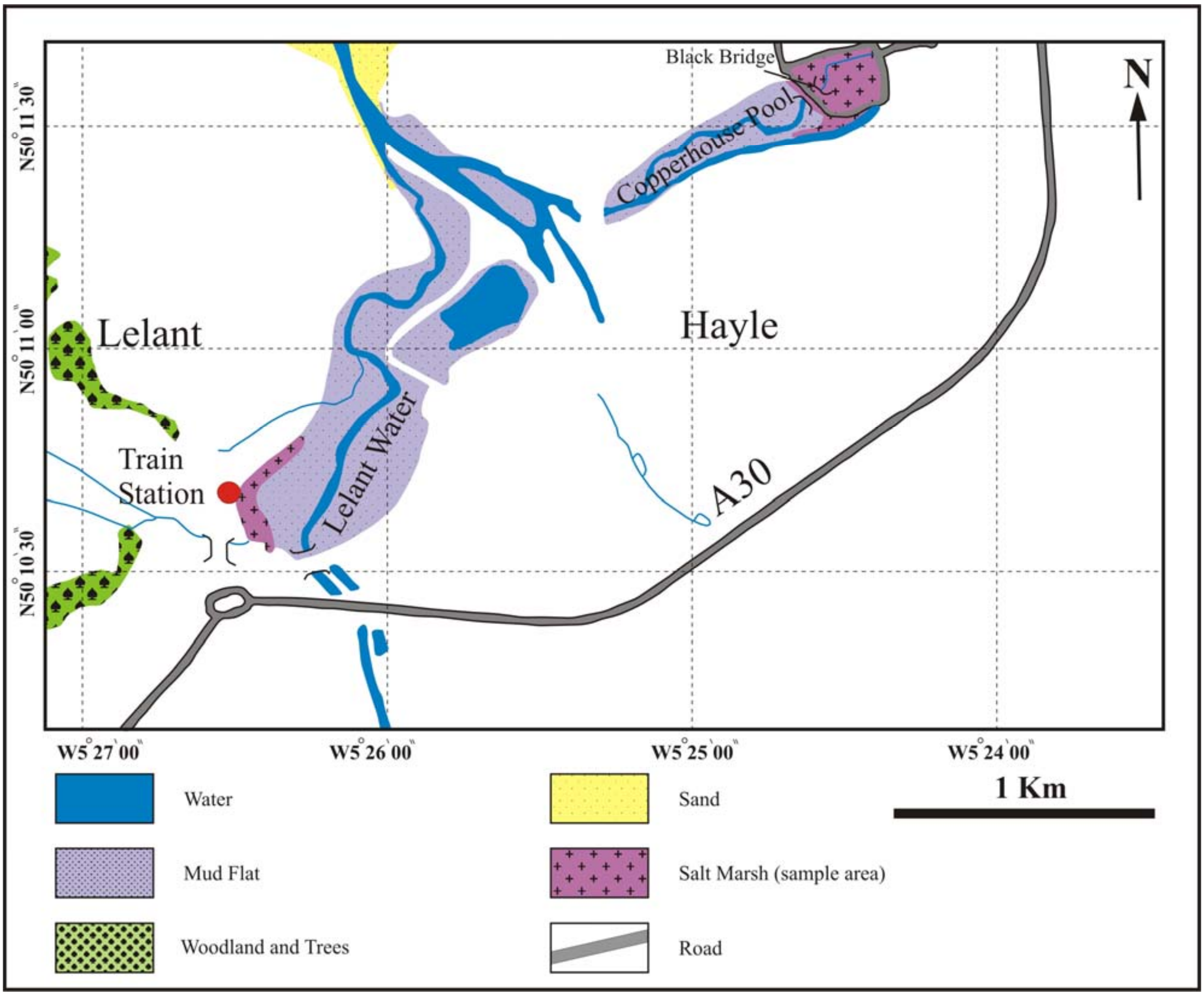


Figure 11 The Hayle Estuary, showing the location of Lelant to the west and Copperhouse to the east

Hardrock mining began initially for copper and then tin - Wheal Alfred Mine being a nearby copper and lead mine (Pirrie *et al.*, 1999). Mine tailings were released directly into local streams and rivers. These rivers drained into Copperhouse and Lelant pools. Both pools became significantly polluted by becoming effective at trapping contaminants (Pirrie *et al.*, 1999). Previous research has confirmed very high metal values within the estuary (Pirrie & Camm, 1999).

Figure 12 Lelant salt marsh showing clear successional patterns

4.2 Landscape and Geomorphology

The Hayle is the most southwest estuary in the UK. The wider surroundings are characterised by fertile lowlands with wooded hedges, with little significant tracks of woodland (Environment Agency, 1997b). The coastline contains a blend of 'heather-clad hilltops with characteristic mining remains interspersed with large sand dune systems and relatively sheltered small fishing ports and coves' (Environment Agency, 1997b).

Hayle is a medium-sized town with the estuary divided into two parts – Lelant Water and Copperhouse Pool (Figure 11). The estuary was formed from the infilled valleys of the Hayle and Angarrack rivers. Two extensive dune systems surround the estuary: in the west, Lelant Sandhills and Hayle Towans in the east. Within the estuary, there are two hundred hectares of intertidal mudflats, sands and salt marsh (Jenkins *et al.*, 1985).

Hayle has been the subject of major estuary modification with the development of sluicing facilities at the western end of Copperhouse Pool, designed to keep the channel to the sea open (Pirrie *et al.*, 1999). The Black Bridge divides the Copperhouse salt marsh – a bridleway constructed of copper slag – though an archway allows exchange of tidal movement and river flow.

Jenkins *et al.* (1985) reported that the Lelant salt marsh increased in size during the 1970s due to siltation in the south-western part of Lelant Water. However, over the past two centuries, the salt marsh has significantly decreased in size (Jenkins *et al.*, 1985).

4.3 Ecology

4.3.1 Ecological Status

The wider environs of Hayle contain a high proportion of Cornish lowland heathland – a European protected habitat. The conservation value is further enhanced where the heathland is associated with maritime sites (Environment Agency, 1997b). Hayle Estuary is an important site for migratory bird species, whilst the contaminated land resulting from past mining operations has led to metal-tolerant invertebrate and lower plants of conservation interest (Environment Agency, 1997b).

The Hayle Estuary and Carrack Gladden Site of Special Scientific Interest (SSSI) protect both the Lelant and Copperhouse salt marshes. Additional protection is afforded by designating the Hayle Estuary and its environs as an AGSV (Environment Agency, 1997b). The salt marshes at Lelant and to the east of Black Bridge at Copperhouse are owned by the Royal Society for the Protection of Birds (RSPB) and managed as a nature reserve.

4.3.2 Floristic Studies

I. Lelant

Salicornia, according to the Salt Marsh Survey of Great Britain (Burd, 1989), dominates the seaward edge of the salt marsh (Figure 13). This gives way to a stand dominated by *Puccinellia*, *Salicornia* and *Armeria*. Directly behind this stand lies a community dominated exclusively by *Armeria*, containing numerous salt pans. The eastern edge of the marsh contains an assemblage of *Festuca, Agrostis, Carex extensa, Juncus gerardii* and *Plantago*. Stands of *J. maritimus* are scattered predominantly in the westward section of the marsh.

Jenkins *et al.* (1985) described Lelant salt marsh as a classic example of salt marsh succession with well-defined zonation. *Salicornia* colonizes the unstable silts towards the mudflats, thus allowing succession by a range of mid-marsh characteristic species. *Puccinellia* is reported as the dominant species in this mid-marsh with a diverse range of associates, including *Aster, Armeria, Plantago, Suaeda* and *Spergularia*. This zone gives way inland to an *Armeria*-dominated community with only a limited number of associates. At the upper-marsh area, a coarse grassland of *Dactylis glomerata, Festuca rubra, Agropyron pungens* and *Agrostis stolonifera* dominates, with stands of *J.*

maritimus, *C. extensa* and *Beta vulgaris*. Additional species include *J. gerardii*, *Triglochin maritima* and *Parapholis strigosa* – a species with limited UK distribution (Jenkins *et al.*, 1985).

Smillie (2006) reported a belt dominated by *Salicornia* with *Plantago maritima* often co-dominant and with many bare sediment patches, behind the mudflats of sparse *Salicornia* and macroalgae (Figure 13). This community is similar to SM8 in the National Vegetation Classification of salt marshes; however, *Plantago* is not reported as ever being co-dominant in this community. It may be this is an SM10 transitional community but with an absence of *Suaeda* plus an abundance of *Plantago*. In the extremities of the assemblage described lies a species-poor *Salicornia*-dominated community. These pioneer zones give way to a community co-dominated by *Armeria* and *Plantago* with frequent salt pans. This assemblage matches the NVC SM13d sub-community, only with an under-representation of *Puccinellia*.

A drainage channel divides the salt marsh. The northern part of the salt marsh, as displayed in Figure 13, contains stands of *Juncus maritimus* within the *Armeria/ Plantago* community and at the landward edge. This is similar to SM15. A small patch of *Juncus gerardii* – an example of SM16b – is located to the rear of the drainage channel.

At the southern part of the marsh, a sward of primarily *Carex extensa* occurs. At the southeastern tip lies a stand of an SM28 community, dominated by a sward of *Elymus repens* grasses. A grassy sward dominated by *Festuca*, close in character to SM16, also occurs nearby. Bordering the southeastern area of the sample site is a community dominated by *Agrostis* that also contains *Elymus*. This resembles the SM28 community, though the dominance of *Agrostis* may be due to the influence of freshwater, present as a river to the south of the survey area.

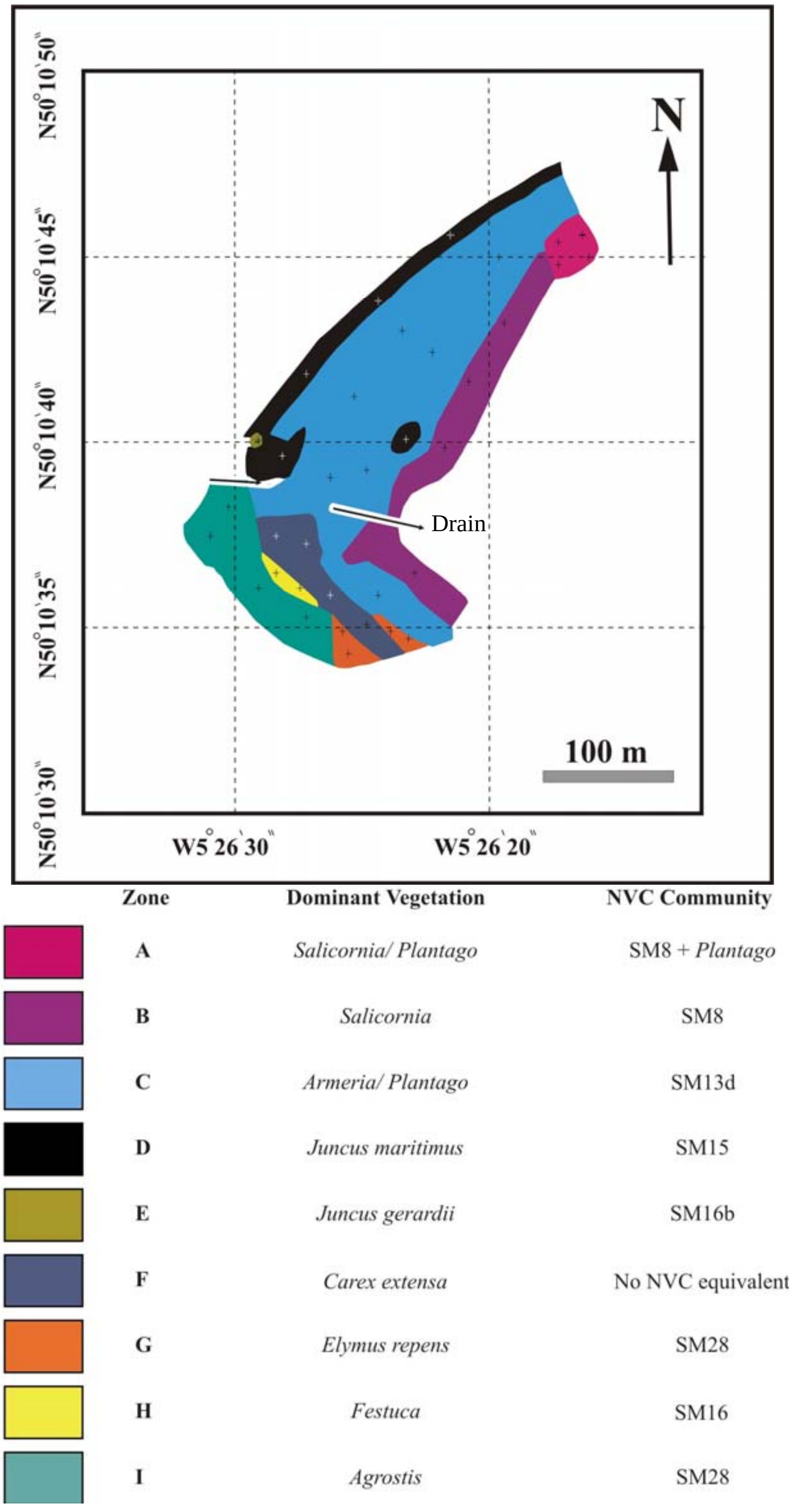


Figure 13 Vegetation zones within Lelant salt marsh (Smillie, 2006)

Sample points are marked with (X). The marsh exhibits succession well, from the pioneer communities in zones A and B to mid-marsh species in Zone C. Upper-marsh/ transition communities are evident towards the southeast.

xAgropogon littoralis was found to be present within the *Agrostis* community. This species has not previously been recorded in Lelant and is of limited distribution in the UK (Fitter & Fitter, 1984).

II. Copperhouse

The Salt Marsh Survey of Great Britain (Burd, 1989) indicates that a large portion of the northern area of the salt marsh is dominated by invading *Phragmites*. This species is present both to the east and west of the Black Bridge (Figure 11).

Figure 14 *Phragmites australis* invading the east and north of Copperhouse marsh

To the west and south of the Black Bridge, *Puccinellia* is dominant with *Inula crithnoides* and *Aster* present. *Armeria* is noted at the extreme south in connection with a disturbed area of the ground. To the east of the Black Bridge, aside from the *Phragmites* dominated community, the area is dominated by an *Armeria/ Spergularia* assemblage interspersed with abundant salt pans. Other occasional plants, such as *Salicornia* and *Puccinellia*, were also recorded.

Figure 15 Copperhouse salt marsh looking towards Lelant and St. Ives

Jenkins *et al.*, (1985) report that to the east of Black Road Bridge, *Puccinellia* and *Armeria* dominate with *Atriplex hastata, Spergularia, Aster* and *C. extensa* present as associates. *J. compressus* – a plant rare in Cornwall and normally associated with less saline conditions – was also recorded. The northern section of the marsh is dominated by *Phragmites australis*, whilst the transitional zone between these two areas contains an assemblage of *J. gerardii, Glaux maritima, C. extensa and F. rubra*. At the time of survey, the flood alleviation scheme was under construction, thus salt marsh was limited. *Armeria* dominated the existing marsh in association with *Spergularia* and *Plantago*. Colonizing the bare mud and sand resulting from the construction were the pioneer species of *Salicornia, Puccinellia* and *Suaeda*. A small, established area of *Spartina anglica* is reported to be immediately to the northwest of Black Bridge.

Figure 16 The Black Bridge

The Salt Marsh Survey of Great Britain (Burd, 1989) reported *Phragmites* to be invading the northern portion of the salt marsh. The survey by Smillie (2006) not only confirms the presence of *Phragmites* but also finds this species to have increased its range, with much of the eastern portion of the marsh now being dominated by *Phragmites* – NVC S4. This is a swamp community (Rodwell, 1995) rather than a salt marsh community. The southern pioneer zone is dominated by *Puccinellia* with, in some areas, *Aster* and *Spergularia* and rare *Inula crithmoides* – probably a rather species poor example of NVC community SM13a. Lining this zone is patchy vegetation dominated by *Armeria.* This appears to be recovering from disturbance, probably from the construction of a surface capable of sustaining vehicles. There is no NVC community matching this description, with SM13d being the closest fit, although without either *Puccinellia* or *Plantago*.

The salt marsh is divided in two by the copper slag-built Black Bridge (Figure 17). To the east of the bridge, *Armeria* and *Spergularia* are the dominant plants, although this zone is interspersed with abundant saltpans. Saltpans are particularly extensive towards the south of this zone. No NVC community adequately describes this zone. Further towards the landward edge of the salt marsh exists a community dominated by *Carex extensa* with frequent *Armeria*. This community was waterlogged at the time of survey and is the only addition to the findings of Burd (1989).

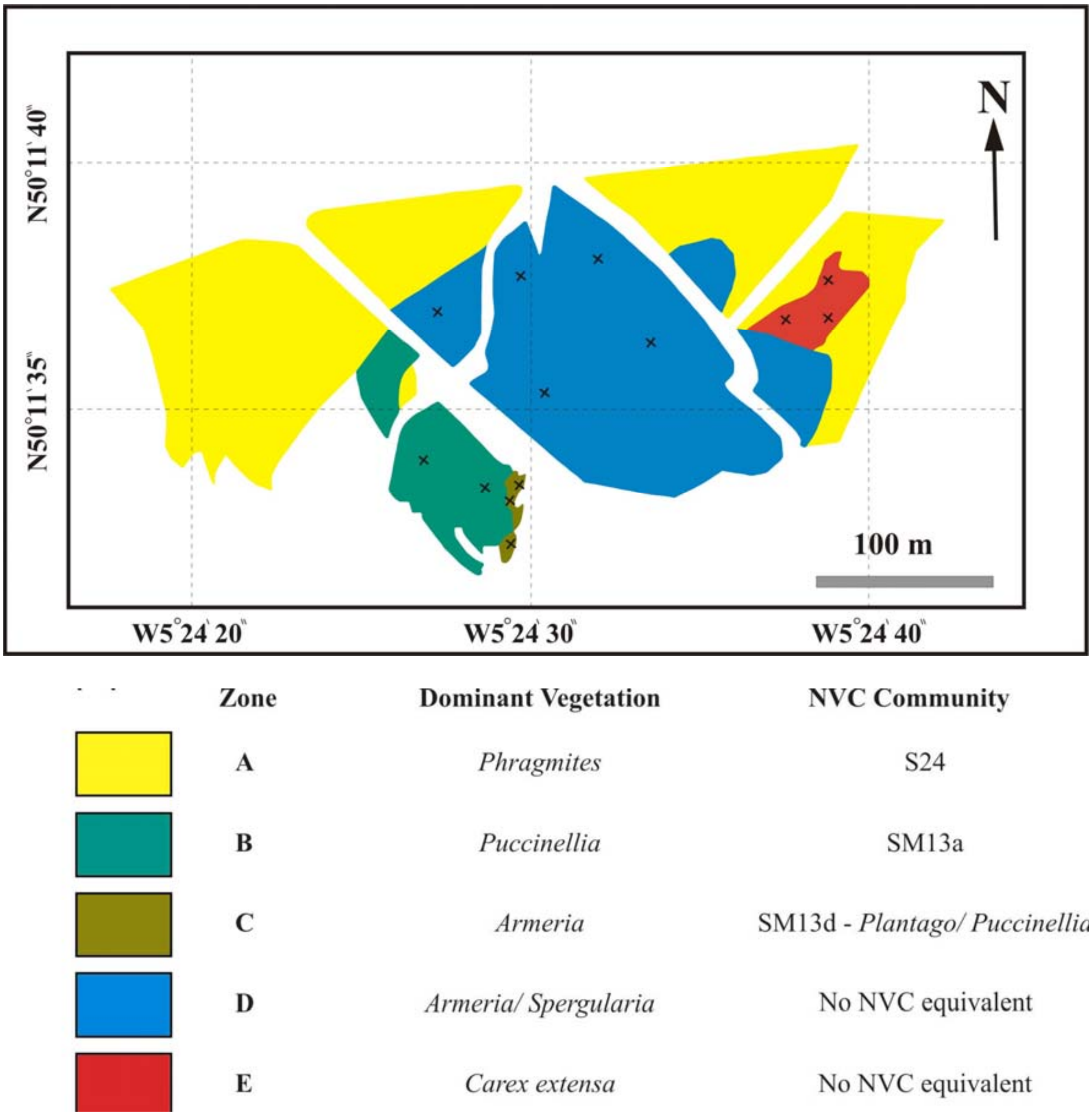


Figure 17 Vegetation zones within Copperhouse salt marsh (Smillie, 2006)

Jenkin (1985) lists a number of plants not recorded by Smillie (2006), notably *Spartina*, *Juncus compressus* and *J. gerardii*. *Plantago* and *Puccinellia* are also reported to be more abundant to the east of the Black Bridge. The survey by Jenkin (1985) took place during the construction of the flood alleviation scheme. This scheme may therefore have led to a loss of biodiversity within Copperhouse.

Figure 18 Copperhouse salt marsh east of the Black Bridge

4.3.3 Invertebrate Studies

Jenkins *et al.* (1985) described *Nereis diversicolor* as the dominant species within the Hayle Estuary, with abundant *Corophium volutator* and *T. saltator* – all three of which contribute the main food source for bird life. *Arenicola marina* is found occasionally in Lelant Water but only rarely in Copperhouse Pool. *Ligea oceanica, Carcinus maerus, T. saltator* and *Anurida maritima* are all common within the mudflats of the Hayle Estuary, although no surveys had been published regarding the fauna of the sandbars and beaches at the mouth of the Estuary (Jenkins *et al.*, 1985).

Asellus meridianus within the Hayle Estuary was found to be tolerant to both copper and lead (Brown, 1976). This is in comparison to *A. meridianus* within the Gannel Estuary, which was revealed to be only tolerant to lead and not copper.

4.4 Geology

The borders of the catchment of the Hayle Estuary comprises of Carmenellis Granite (Environment Agency, 1997b). This encloses the dominant Devonian Gramscatho Group, which is overlain in the vicinity of the Hayle Estuary by Quaternary marine and estuarine alluvium (Goode & Taylor, 1988). Dominating the mouth of the estuary are Quaternary aeolian sand dunes, known locally as towans. A belt of hydrothermal metalliferous tin-copper mineralisation crosses the catchment (Environment Agency, 1997b).

4.5 Geochemistry and Mineralogy

I. Lelant

Geochemical investigations within Lelant salt marsh are limited. Turner (2000) examined the oxic sediment (no depth given) from the intertidal zone of a number of estuaries associated with granite related polymetallic vein mineralisation (Fal, Hayle, Helford and Tavy - a tidal tributary of the Tamar Estuary), as well as estuaries receiving metal contamination from current industrial and municipal sources (Clyde, Dee, Forth, Loughor, Mersey and Poole Harbour). The Hayle Estuary was found to be the most concentrated in copper, zinc, iron and lead as well as having the highest concentration of manganese within the mineralised estuaries.

Yim (1976) recovered two cores of depths 915 mm and 1120 mm from Lelant, recording exceptionally high concentrations of tin, arsenic, copper, lead, tungsten and zinc. These concentrations were correlated with pollution from past mining activity, though sample mixing was believed to compromise the results. Results from Yim (1976) showed a greater range in metal concentrations than recorded by Smillie (2006), in that maximum metal concentrations are significantly higher, whilst minimum concentrations are always lower. Mean concentrations are also higher in Yim (1976) than in Smillie (2006). Yim (1976) states that Core 2 has maximum metal concentrations below 255 mm, whilst Core 3 records the highest metal concentration below 660 mm (although this is metal dependant, with Yim (1976) correlating metal enrichment with different stages of mining history). Hence, metals are most enriched in Core 3 of Yim (1976) below the sampling depth of Smillie (2006) – sampling depth of 500 mm. Core 3 is also the most enriched core, thus increasing the mean concentration above that of Smillie (2006). Below the areas of highest metal concentration, Yim (1976) records sediments with comparatively low metal concentrations (Core 2 below 510 mm and Core 3 below 1120 mm). These were correlated as being associated with small-scale mining activity (Yim, 1976) and account for the lower minimum concentrations found within Yim (1976) compared with Smillie (2006). With regard to the upper 500 mm, copper, iron and zinc are recorded as being concentrated within the upper surface sediment of Core 3 by Yim (1976), though not within Core 2. Manganese is however, generally concentrated in the sediments below the surface, whilst lead displays little variation between the cores sampled by Yim (1976).

Turner (2000) recorded concentrations of copper (948 mg/kg), zinc (1650 mg/kg) and lead (301 mg/kg) above the maximum concentration of Smillie (2006) (Cu = 823 mg/kg; Zn = 469 mg/kg; Pb = 201 mg/kg). Manganese (381 mg/kg) and iron (13600 mg/kg) concentrations however, were below that recorded by Smillie (2006) (Fe = 73150 mg/kg; Mn = 1001 mg/kg). Turner (2000) sampled at the south of Lelant Water rather than the salt marsh in the west. Spatial differences are likely to account for some of the differences in metal concentrations recorded. Also, although no sampling depth is given by Turner (2000), mention is made that 'samples were manipulated with a plastic spatula' rather than coring, suggesting sampling occurred only at the very surface. The difference in sampling depths between Smillie (2006) and Turner (2000) is likely to also yield dissimilar results, as Yim (1976) recorded downcore variations between cores 2 and 3.

Table 3 Summary of a selection of geochemical data from sediment studies in Lelant

Author	Cu	Zn	Fe	Mn	Pb	Sn	Ni	As
Smillie, 2006	613 (344-823)	253 (117-469)	60993 (34790-73150)	639 (231-1001)	130 (73-201)	2658 (991-3822)	30 (10-66)	316 (126-680)
Turner, 2000	948	1650	13600	381	301	-	-	-
Yim, 1976	1186 (68-2760)	816 (80-3450)	40059 (3840-77760)	1676 (48-1080)	263 (20-600)	4601 (50-10000)	-	632 (12-1920)

Results expressed as mg/kg of dry weight. All results are mean concentrations except (+) indicating maximum concentrations. (-) indicates element not reported. Results also include the range in parenthesis where available.

Additional geochemistry results are presented in Table 4.

Table 4 Further geochemical data from Lelant (Smillie, 2006)

Author	Na	K	Mg	Cl	Ca
Smillie, 2006	10938 (3404-56980)	21541 (15521-26560)	6657 (2520-8580)	4182 (1044-13490)	4821 (2272-8236)
NO_3-N	**S**	**P**	**Moisture**	**pH**	**LOF**
17 (0-38)	1543 (400-3900)	1419 (924-1848)	52 (18-81)	6.8* (6-7.5)*	124277 (26500-324100)

Results expressed as mg/kg of dry weight except (*) as pH units. All results are mean concentrations with the range in parenthesis. n = 35

Spatial geochemical data from Smillie (2006) is presented a series of contour maps (Figure 19 to Figure 23) using Surfer software.

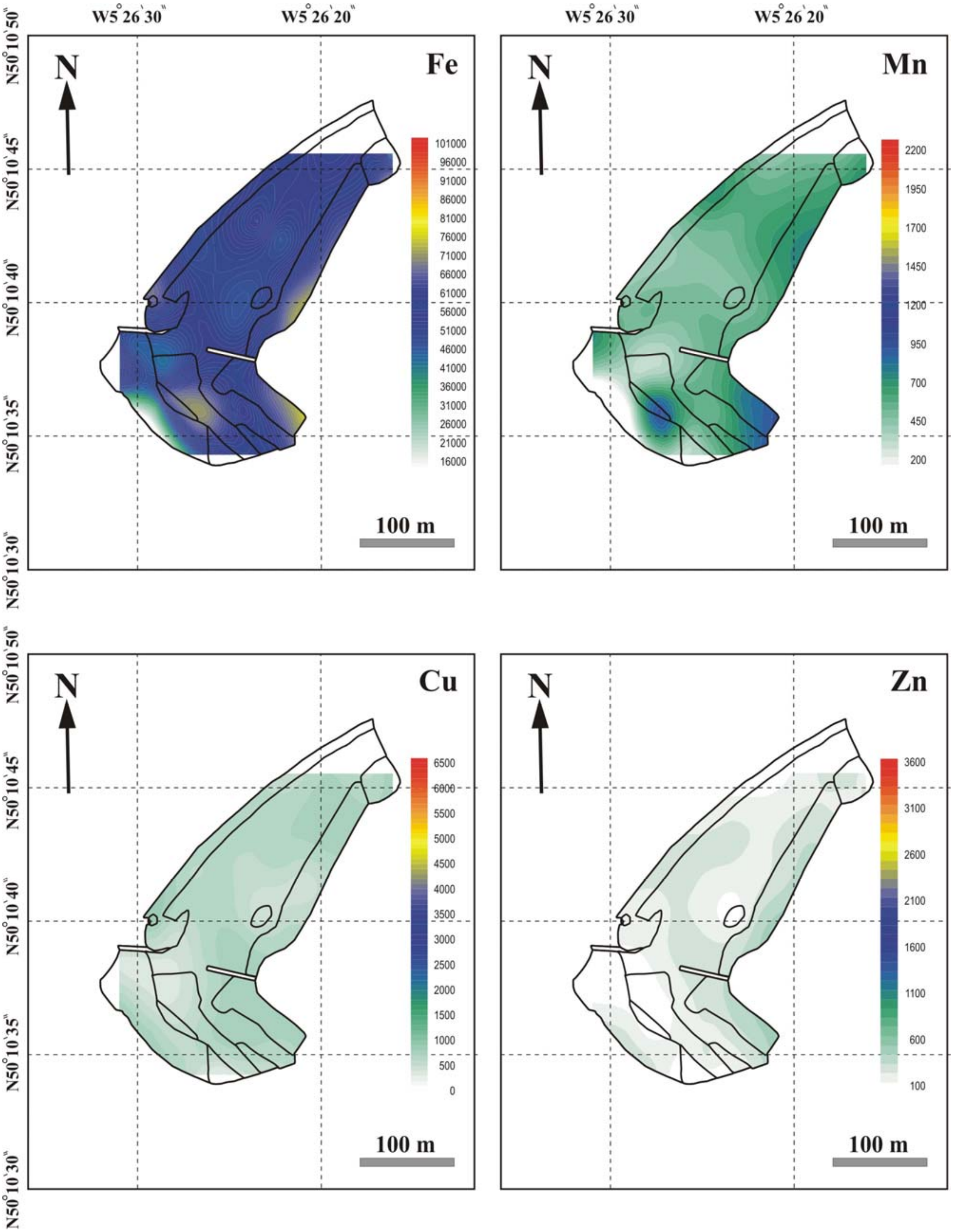


Figure 19 Spatial variation of iron, manganese, copper and zinc in Lelant

Although metal concentrations appear to be fairly uniformly spread throughout the marsh, zinc is distinctly lower in the *J. maritimus* dominated community (Figure 19).

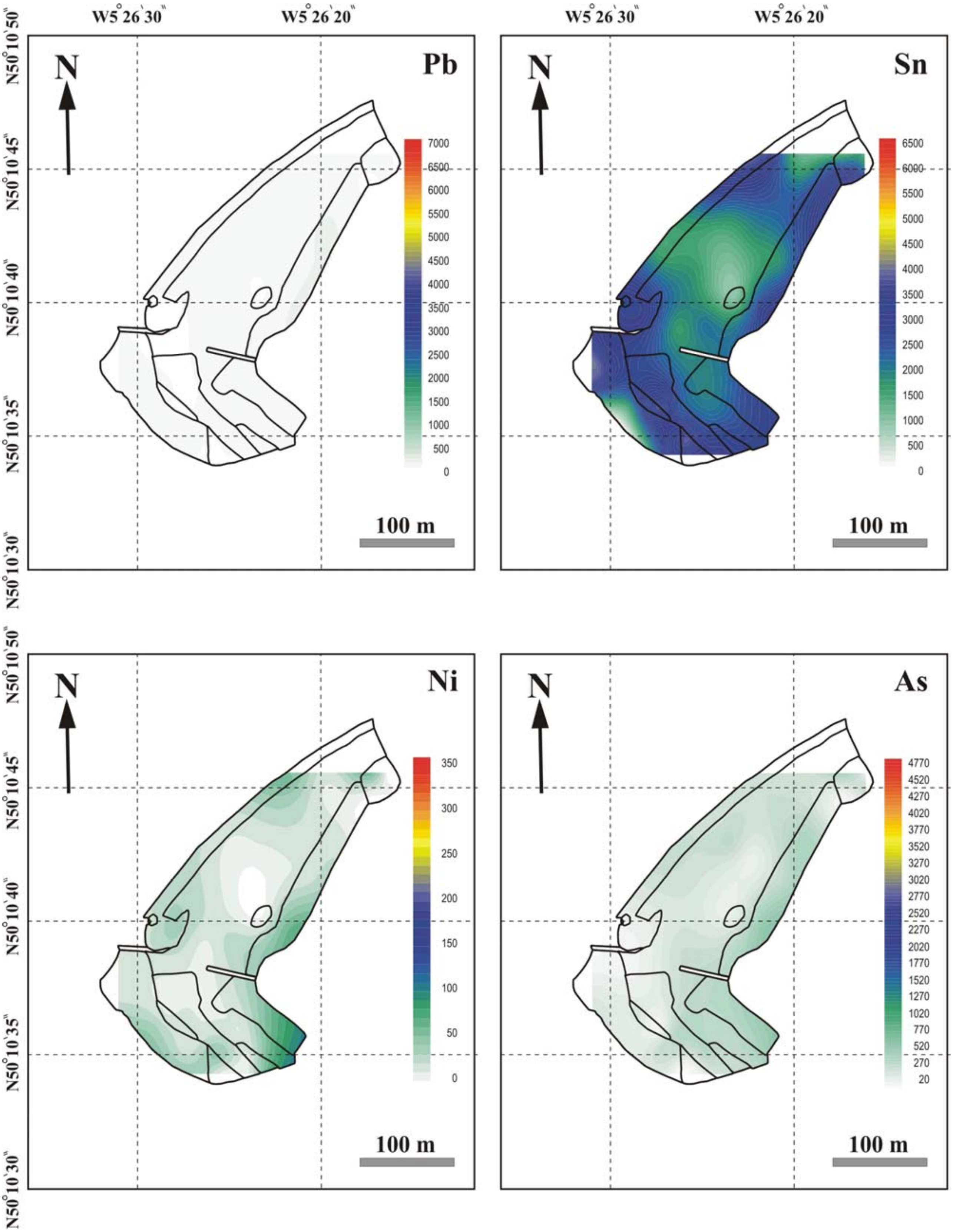


Figure 20 Spatial variations of lead, tin, nickel and arsenic in Lelant

Lead, nickel and arsenic are all relatively low, compared with the other marshes examined by Smillie (2006). Tin, however, is moderately enriched.

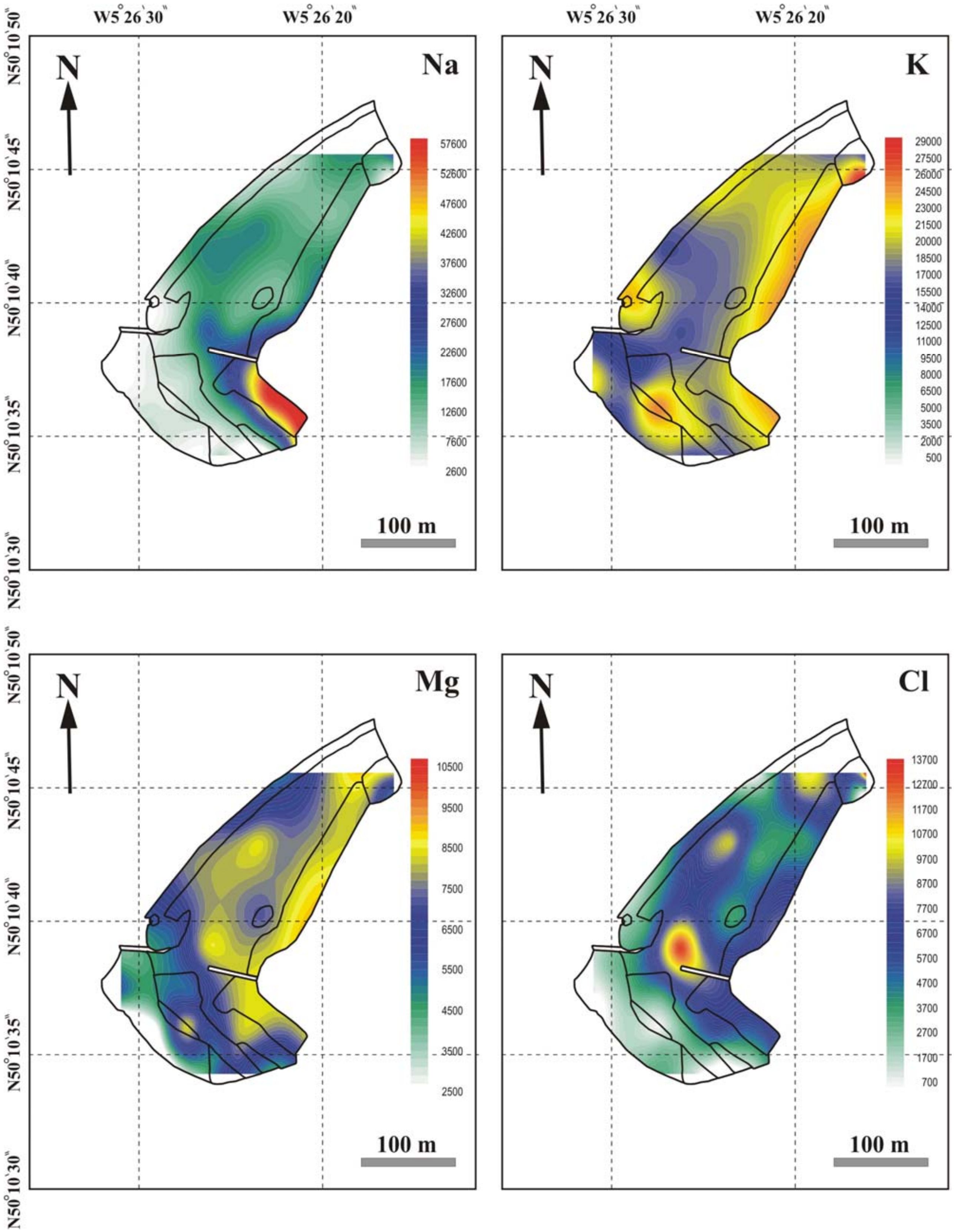


Figure 21 Spatial variations of sodium, potassium, magnesium and chlorine in Lelant

There is evidence of a landward decrease in salinity (Figure 21), although this change is neither uniform nor constant. Nevertheless, the upper-marsh areas are often of lower concentrations of elements often associated with salinity than that of the mid to upper-marsh. The drain and low-to-mid marsh surrounds are associated with high concentrations of chlorine, sodium and magnesium (Figure 21), probably indicating the direction favoured by flow during high tides, as opposed to the taller vegetation in the high marsh to the south.

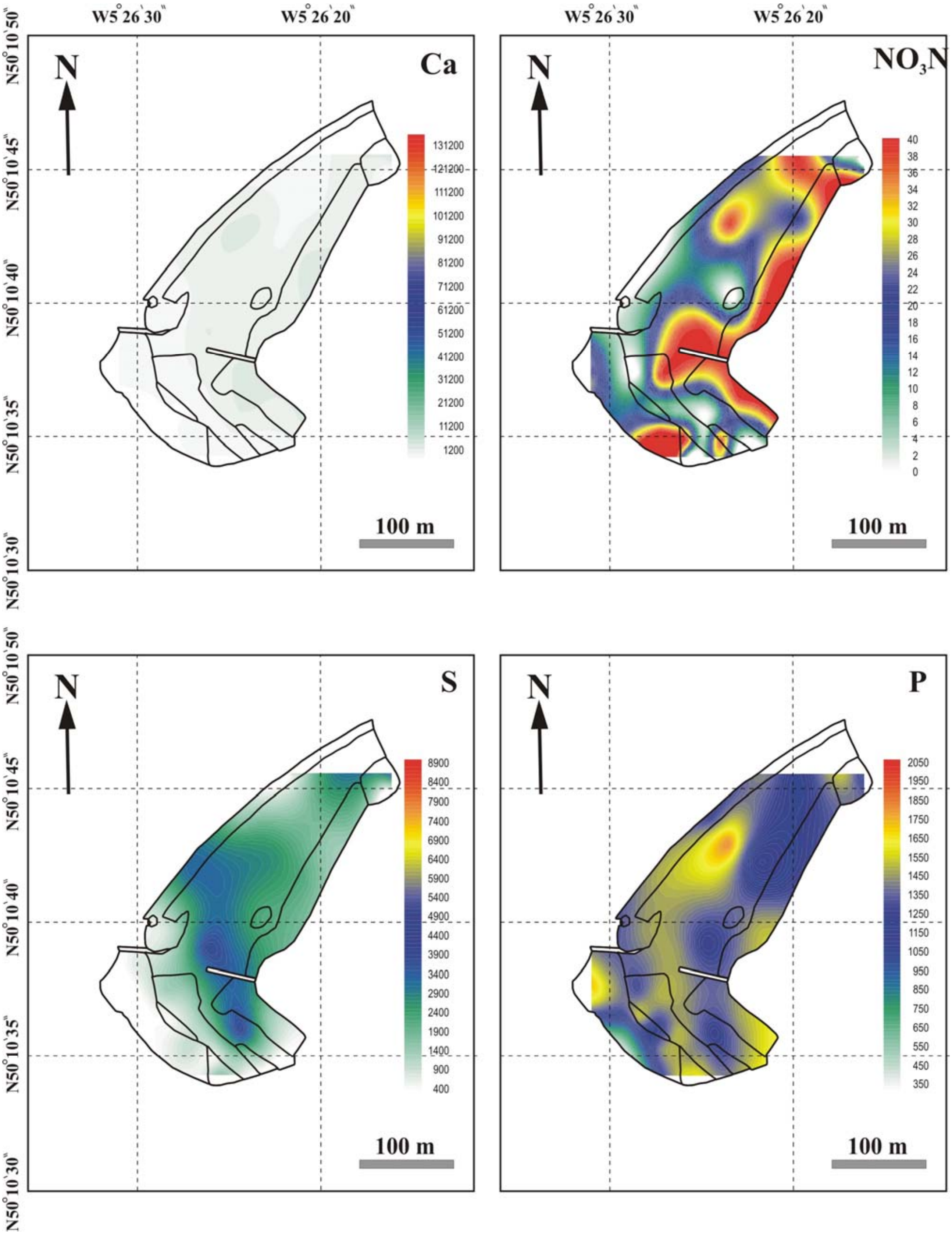


Figure 22 Spatial variations of calcium, nitrate-nitrogen, sulphur and phosphorus in Lelant

There is some evidence of a landward decrease in nitrate concentration (Figure 22), though this decrease is not constant. Although not uniform, *J. maritimus* tends to be associated with very low concentrations of nitrate (Figure 22).

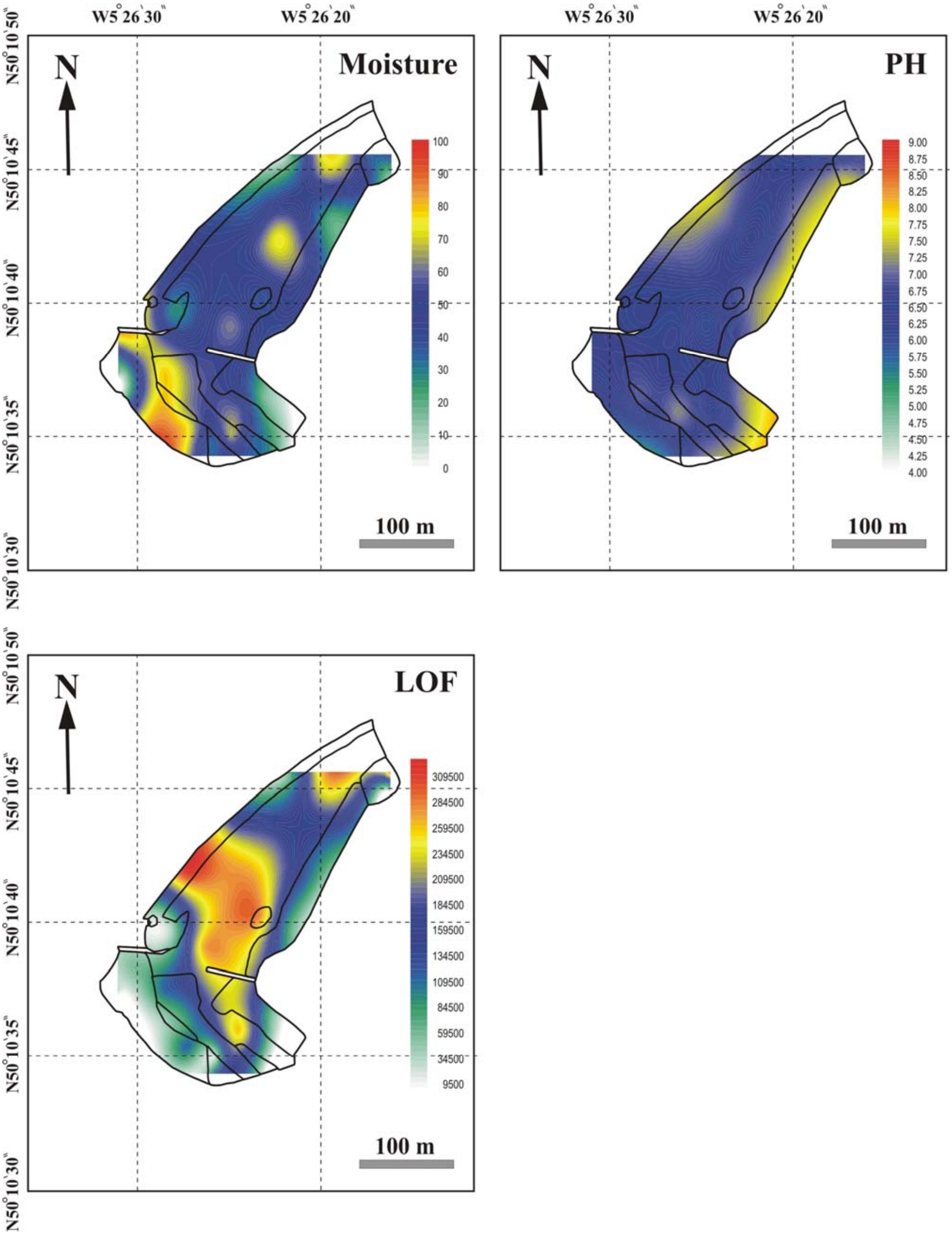


Figure 23 Spatial variations of moisture content, pH and loss-on-fractionation in Lelant

Relatively high pH tends to be associated with the pioneer communities (Figure 23). The upper marsh tends to have a high moisture content, though this is not constant. LOF is, to some extent, associated with the drainage channel and spill over, as seen in Figure 13. The high marsh communities tend to be low in terms of LOF (Figure 23).

II. Copperhouse

Copperhouse has been the subject of a number of geochemical studies. Healy (1995) recorded variable zinc (up to 3100 mg/kg) and iron (peaking at 80000 mg/kg) concentrations on an 800 mm core, with copper and lead averaging at around 4000 mg/kg and 2000 mg/kg respectively. The results are high, compared with findings of previous research on Copperhouse stream sediments (Merefield, 1993). A 1120 mm core from Copperhouse Pool (Yim, 1976) and a later study of the oxic sediment by Smillie (2006) found iron was actually higher in concentration, though results from copper, zinc and lead were all considerably less.

Yim (1976) noted that the wide variation in sediment-metals range was a product of layering of sediments of different metal concentrations. This layering was correlated with different stages in commercial metal production. The core collected from Copperhouse by Yim (1976) sampled to a depth of below 1500 mm, revealed a pulse of abundant metals between 600 and 1220 mm depth. Thus Yim (1976) encountered maximum values of certain metals (copper, iron, lead, tin and arsenic) in an area not sampled by Smillie (2006), which sampled to a depth of 500 mm. Zinc was observed to be more abundant at a depth between 355 to 455 mm, thus explaining the similar range and mean concentrations encountered with Smillie (2006). Certain mean results from Smillie (2006) (copper, iron, zinc and manganese) are higher than the results reported by Yim (1976), as, below 1220 mm, concentrations are comparatively low, thus diluting the mean concentration of the core. Manganese was significantly higher within Smillie (2006) when compared to Yim (1976). This is likely to be due to spatial differences, although other reasons, such as the construction of the flood alleviation scheme at Copperhouse, cannot be discounted.

Table 5 Summary of a selection of geochemical data from sediment studies in Copperhouse

Author	Cu	Zn	Fe	Mn	Pb	Sn	Ni	As
Smillie, 2006	1259 (384-2322)	1341 (494-2439)	69472 (49630-90580)	1208 (770-2233)	337 (173-488)	2563 (987-4749)	61 (17-132)	1252 (628-2690)
Yim, 1976	1029 (270-2600)	1223 (560-2290)	46985 (17760-96000)	836 (312-1512)	388 (40-850)	3431 (850-10000)	-	1551 (240-4080)

Results expressed as mg/kg of dry weight. All results are mean concentrations except (+) indicating maximum concentrations. (-) indicates element not reported. Results also include the range in parenthesis.

Additional geochemistry data from Smillie (2006) is presented in Table 6.

Table 6 Further geochemical data from Copperhouse from Smillie (2006)

Na	K	Mg	Cl	Ca	NO_3-N	S	P	Moisture	pH	LOF
8641 (4366-13394)	20712 (14525-28718)	8672 (5220-10500)	3313 (1584-5748)	14713 (2840-82928)	16 (0-16)	1985 (800-3400)	1198 (572-2068)	61 (18-88)	6.8* (6-7.5)*	61 (18-88

Results expressed as mg/kg of dry weight except (*) as pH units. All results are mean concentrations with the range in parenthesis. n = 13

In examining 3 m cores from Copperhouse Pool, Pirrie *et al.* (1999) found abundant diagenetic copper sulphides plus the first reported occurrence of arsenic sulphides in a Cornish Estuary. The formation of copper and arsenic sulphides necessitates different environmental requirements, with iron being a crucial factor to growth of chalcopyrite whilst inhibiting the development of the diagenetic arsenic sulphides. A variety of geochemical microenvironments within estuary sediments are thus proposed to explain the occurrence of both minerals.

Bryan *et al*. (1980) studied sediment metal concentrations in the Hayle Estuary, however, the location of sample points is unclear.

Spatial geochemical data from Smillie (2006) is presented a series of contour maps (Figure 24 to Figure 28) using Surfer software.

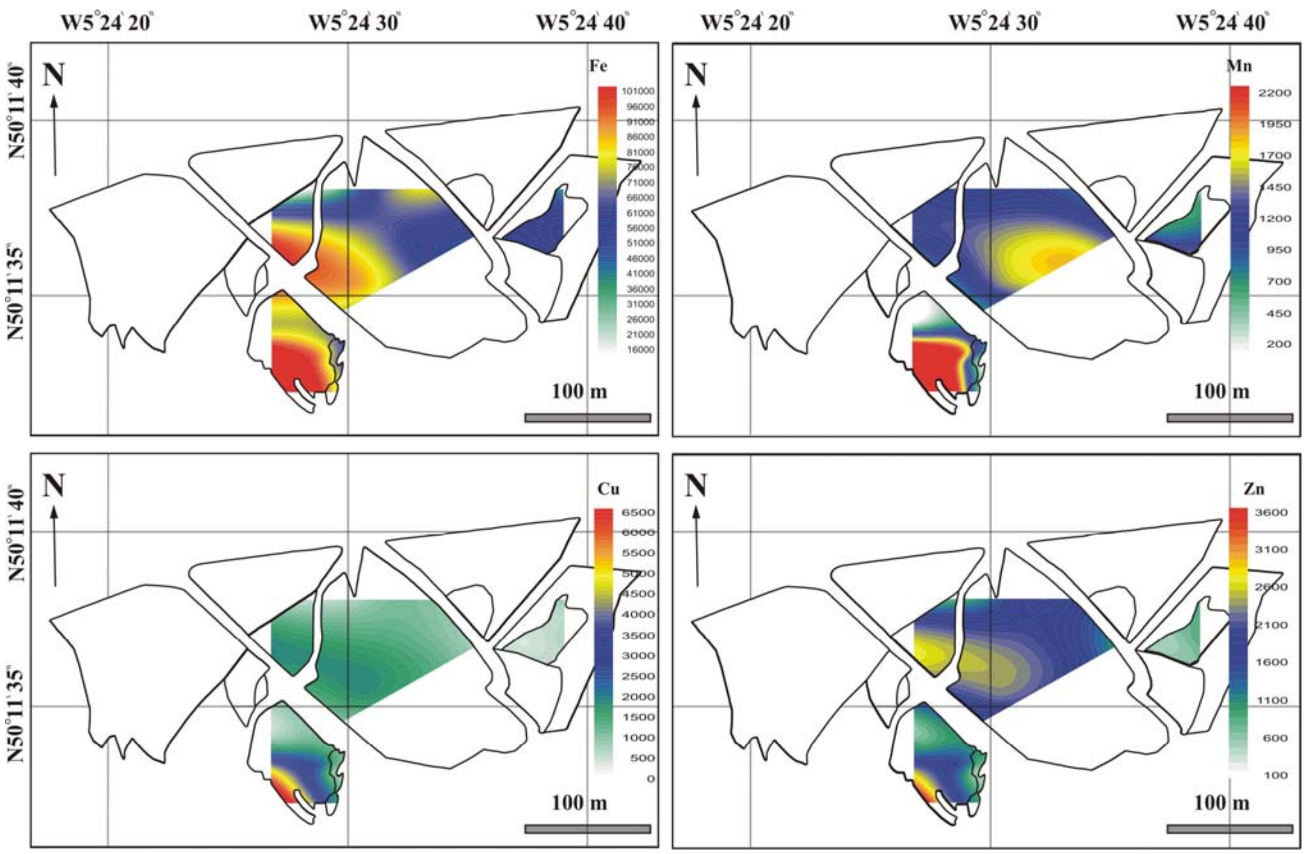


Figure 24 Spatial variation of iron, manganese, copper and zinc in Copperhouse

Iron, copper, zinc and manganese are all elevated in the salt marsh at the seaward end (Figure 24). Both iron and zinc are also elevated around the underpass of the Black Bridge (Figure 11). However, other areas of this zone have relatively lower metal concentrations.

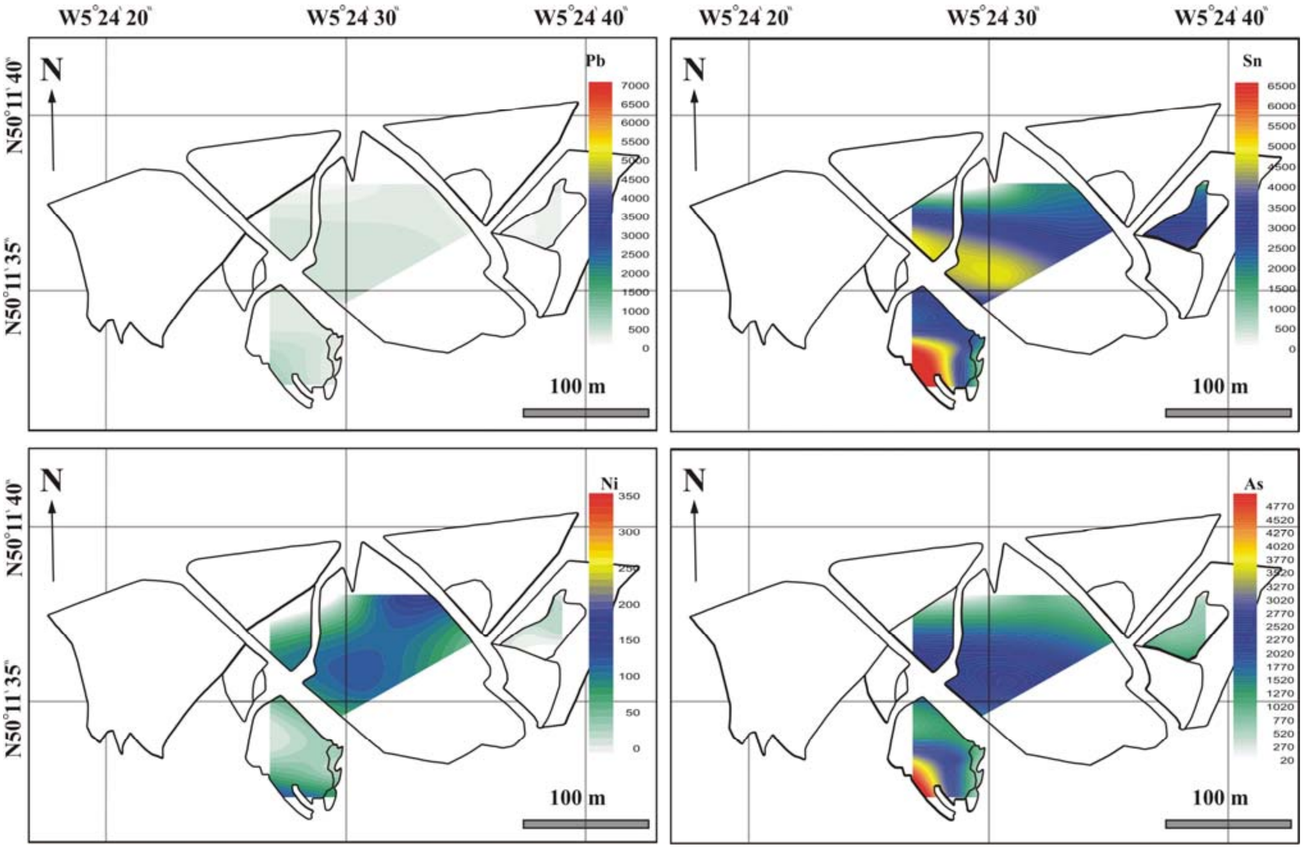


Figure 25 Spatial variations of lead, tin, nickel and arsenic in Copperhouse

As with iron, copper, zinc and manganese, both arsenic and tin are elevated towards the seaward end of the salt marsh, as is lead to a lesser extent. Tin concentrations in the pioneer community to the west are especially variable, ranging from less than 1000 mg/kg to over 4750 mg/kg.

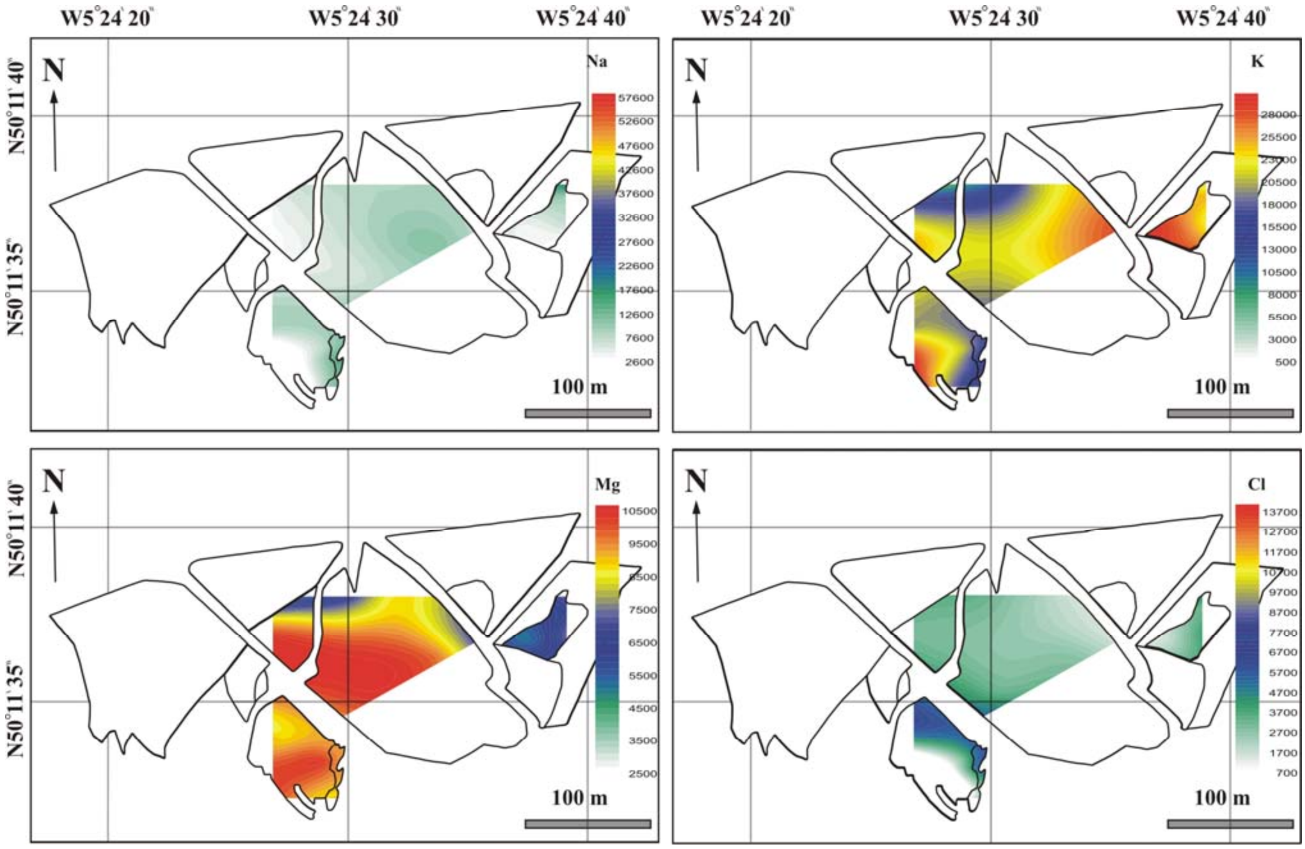


Figure 26 Spatial variations of sodium, potassium, magnesium and chlorine in Copperhouse

Only magnesium follows a basic linear trend of high to low from seaward to landward, allowing for limited fluctuation. Chlorine and sodium have only limited variation within the sample area. Potassium, however, fluctuates considerably throughout the salt marsh.

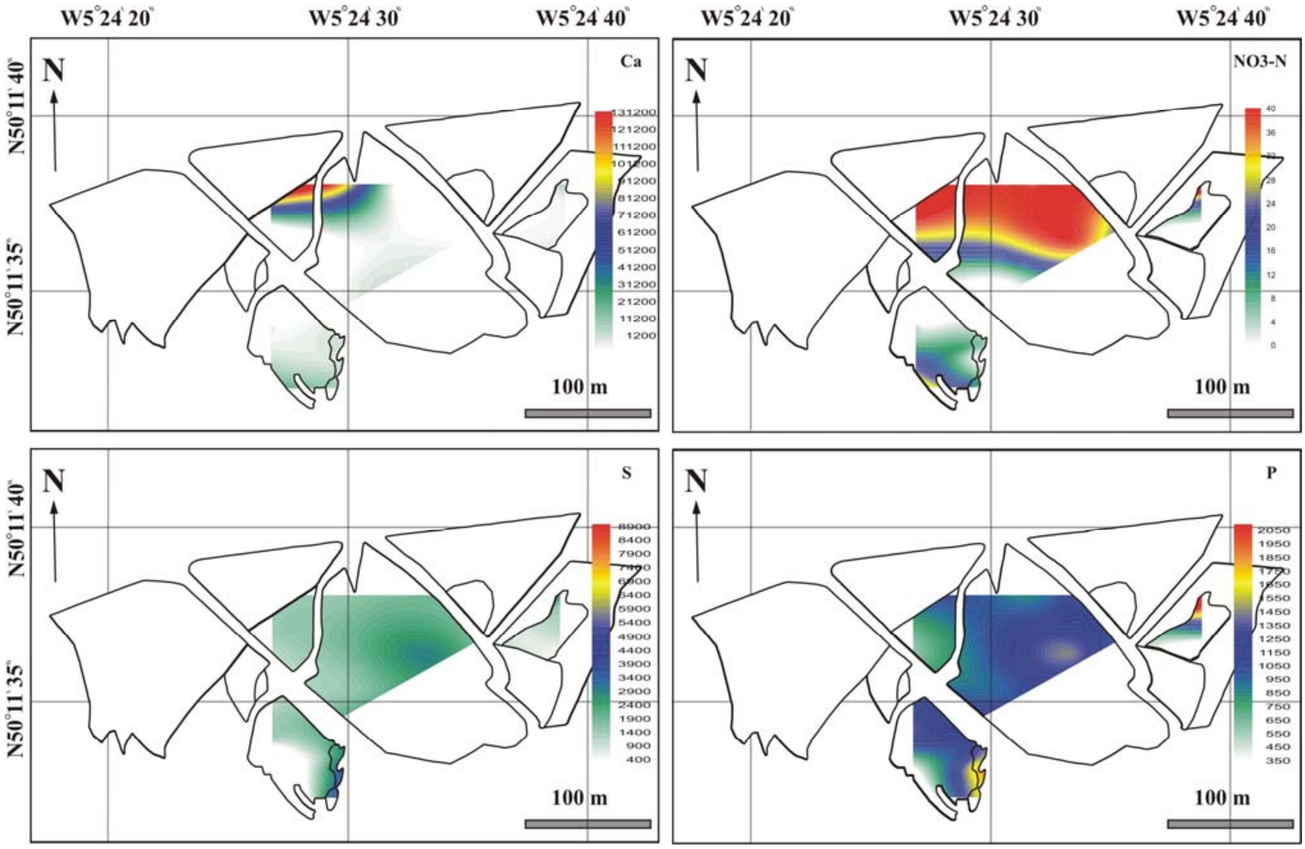


Figure 27 Spatial variations of calcium, nitrate-nitrogen, sulphur and phosphorus in Copperhouse

Calcium is reasonably uniformly spread throughout the salt marsh, as is sulphur, although east of the Black Bridge does display elevated concentrations compared with the rest of the marsh (Figure 27). Though phosphorus is seen to be highest in the easternmost section of the survey area, this actually reflects elevated concentrations within a single sample point. Thus, it is unclear whether this is merely a hot spot or a true reflection of high phosphorus towards the east. As with sulphur, high concentrations of phosphorus are associated with the east of the Black Bridge.

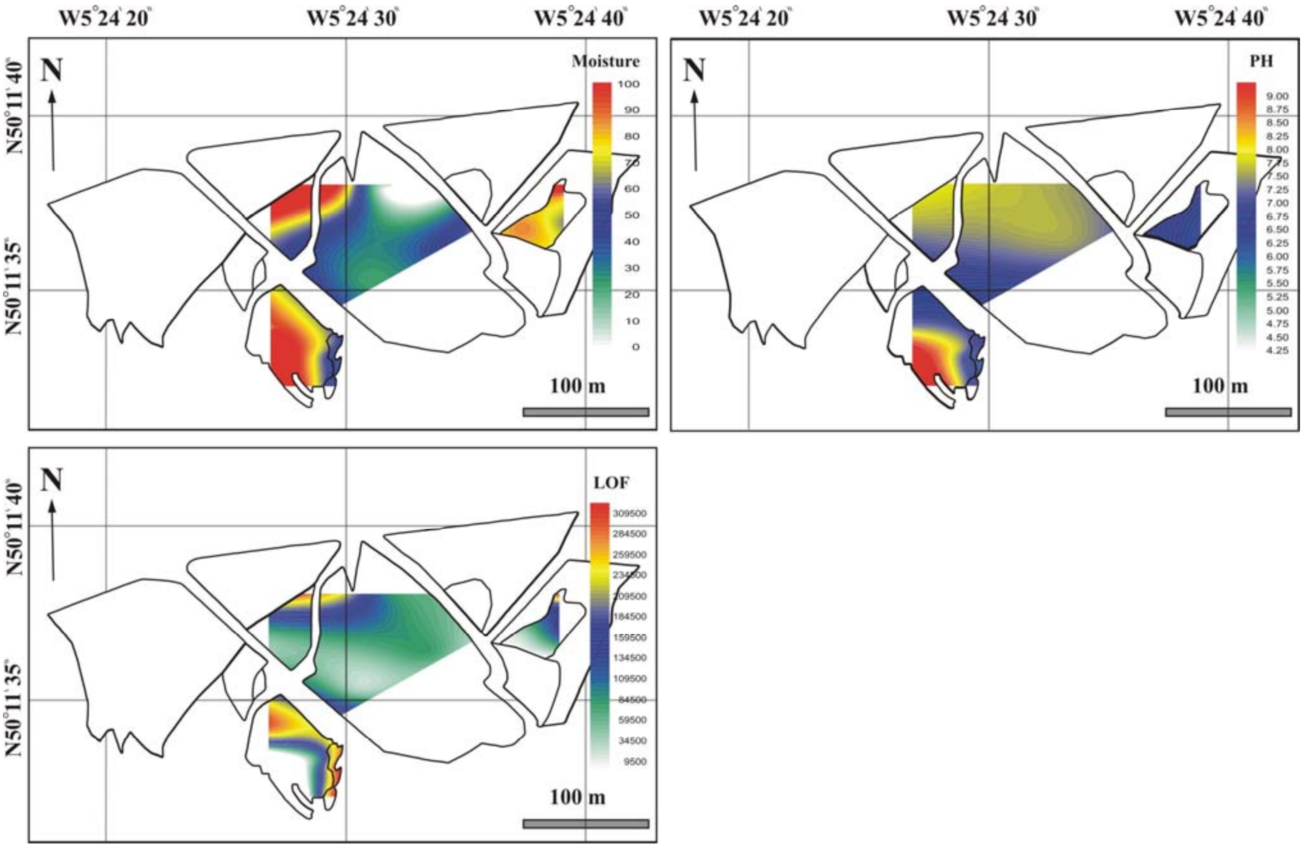


Figure 28 Spatial variations of moisture content, pH and loss-on-fractionation in Copperhouse

Moisture content is particularly high towards the westward end of the marsh, probably indicative of proximity to the sea (Figure 28). The high moisture content in the north of the sample area, to the east of the Black Bridge, may be a function of water flow through the underpass. The extreme eastern samples were waterlogged at the time, thus accounting for the high moisture level. Although, pH appears to vary considerably (Figure 28), the range of variation is only 1.5 pH units, therefore, the entire survey area is close to neutral. LOF is particularly high close to the seaward edge of the survey area (Figure 28). There is also some indication of increasing LOF in the north and east.

5 CAMEL

5.1 *Mining History*

Extensive archaeological evidence of medieval streamworkings is associated with a minor mineral vein on Bodmin Moor (Gerard, 2000). Selwood *et al.* (1998) suggests that this implies the vein was previously more significant. Pirrie *et al.* (2000) however found no evidence of waste from streamworking appearing in cores from a study of sedimentology within the Camel Estuary.

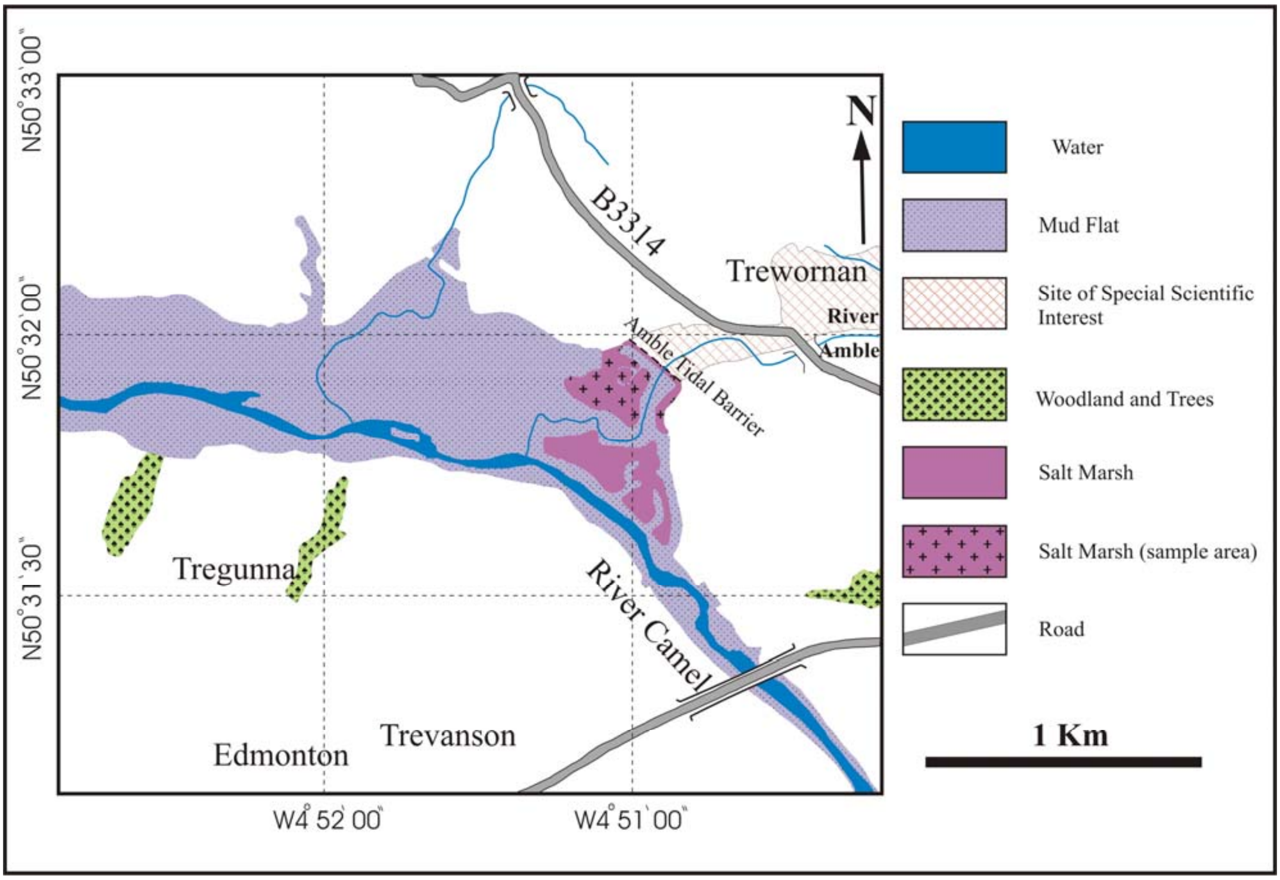


Figure 29 The Camel Estuary

St Endellion, associated with the River Amble to the north of the estuary, was an important site for antimony production in southwest England (Selwood *et al.*, 1998). Pirrie *et al.* (2000) found little evidence of contamination from this area within the sediment profile of the estuary. The major mines of Wheal Prosper and Mulberry, which worked tin stockwork deposits, were located to the south of the catchment area for the Camel Estuary (Pirrie *et al.*, 2000). Ore from Mulberry was also processed within the catchment area (Dines, 1956). Contaminant pulses probably associated with the release of tailings from the closure of these mines were evident in the sedimentological record (Pirrie *et al.*, 2000).

5.2 Landscape and Geomorphology

Much of the surrounding land is agricultural, although quarries are infrequently scattered along the edge (Cornwall Naturalists Trust, 1980a). Camelford and Bodmin form the major urban concentrations in the area (Environment Agency, 1997c).

Figure 30 The Camel Estuary looking towards Wadebridge

Bronze Age remains are common on higher ground, whilst Iron Age cliff forts survive in many places along the rocky coastline (Environment Agency, 1997c). The Medieval Period is most responsible for shaping the landscape of the Camel Region, with open moorland being enclosed by hedges. Many of these tree-topped Cornish hedges still line a number of the fields and roads (Environment Agency, 1997c). The close proximity of the arable land to the estuary means that only a small strip of semi-improved vegetation lines the shoreline (Cornwall Naturalists Trust, 1980a).

Although woodland is rare, many of the steep valleys are heavily wooded (Environment Agency, 1997c). The Camel River passes through moorland and woodland in its upper reaches before discharging into a scenic estuary (Environment Agency, 1997c).

Much of the coastline is designated as an AONB. Key features being long open views to the sea, contrasting with wooded thickets and creeks within valleys. The medieval field systems, narrow-laned dead-end paths and small towns and hamlets are also reasons for this designation (AONB page). Most of the estuary has gently sloping sides, excepting the steeply sided Brea Hill and Cant Hill (Cornwall Naturalists Trust, 1980a).

The Amble Marshes, adjacent to the salt marsh sample area, are archaic lowland grazing marshes (MAGIC, 2008). These are separated from the salt marsh by ploughed, reseeded fields used for livestock and arable farming, then, immediately bordering the salt marsh is the Amble Tidal Barrier. The Barrier was constructed by means of an earth bank, being protected on the seaward side by concrete reinforcements (Cornwall Naturalists Trust, 1980a).

5.3 Ecology

5.3.1 Ecological Status

The Atlantic Ocean creates extreme conditions, limiting the number and structure of biota. Of the habitats present, maritime grassland, heathland and stunted woodland is considered of high biological value (Environment Agency, 1997c). A section of the Camel Estuary is designated a Voluntary Marine Wildlife Area, where seabirds breed on the rocky cliffs, whilst grey seals inhabit sea caves (Environment Agency, 1997c).

The River Camel SAC houses important populations of *Cottus gobio* and *Lutra lutra*, plus the Priority Habitats of European dry heaths, old *Quercus petrea* woods with *Ilex* and *Blechnum* and alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (JNCC, 2008). The boundary of this habitat is, however, outside the salt marsh sample area.

Along the floodplain of the River Amble (Figure 29) lies Amble Marshes SSSI. The site is notable for its variety of habitats, in particular a number of grasslands, including lowland grazing marsh, along with open water, scrub and marginal habitats (English Nature, 1986). Amble Marshes is particularly important for its ornithological interest, this significance being increased by the close proximity to the Camel Estuary (English Nature, 1986). The boundary of the SSSI ends at the tidal barrier (Figure 29) and, therefore, does not encompass the adjacent salt marsh.

The Camel Estuary is afforded limited protection through the local AGSV designation, most notable for linking a number of SSSIs and the Voluntary Marine Wildlife Area at Polzeath.

5.3.2 Floristic Studies

Burd (1989) reports a large area of *Spartina*-dominated community at the northwest area of the salt marsh by the Amble Tidal Barrier (Figure 29). Elsewhere, *Puccinellia* dominates the salt marsh with *Festuca* and *Triglochin* in scattered patches. To the southeast lie two distinct swards: the upper dominated by *Scirpus*, the lower dominated by *Phragmites*.

Figure 31 Stand of *Phragmites australis* at the north-east of the Camel salt marsh

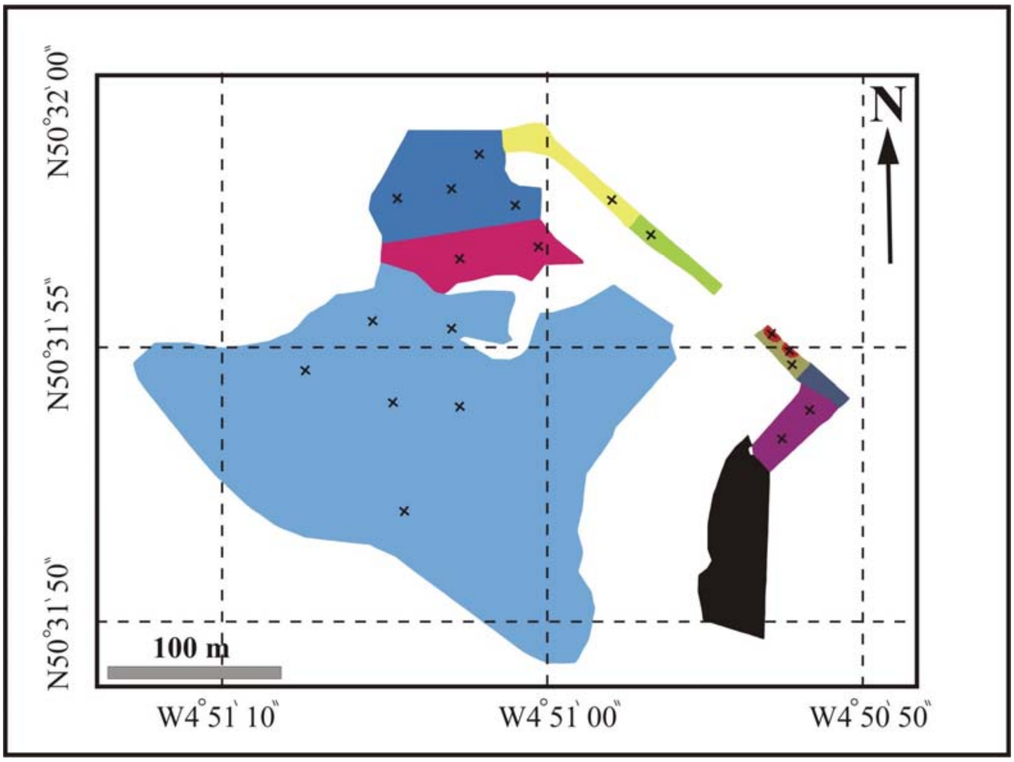
The Cornwall Naturalists Trust (1980a) describes the vegetation within the salt marsh as moderately diverse. The building of a dam across the north of the River Amble in 1963 has resulted in the marsh extending through the invasion of *Spartina*. This species was not reported during a 1974 survey by Coastal Ecology Research Station (Cornwall Naturalists Trust, 1980a).

Figure 32 Large stand of *Spartina*

Smillie (2006) reports a large section of the pioneer zone is dominated by *Spartina anglica* with abundant *Puccinellia* and *Aster,* although neither species is constant (Figure 34). This is indicative of a species poor SM6, as categorized by the NVC system (see APPENDIX I). *Halimione* is present at the edge of the many creeks within this zone.

Figure 33 *Halimione* lined creeks within the *Spartina* marsh

Behind the *Spartina* and across the marsh in general, a mix of *Puccinellia*, *Aster* and *Plantago* is present. This is similar to the transitional low marsh community, SM10. *Scirpus maritimus*, similar to NVC S21b, forms two clumps within this community. A high marsh assemblage consisting of a grassy sward of *Agrostis* and *Festuca* succeeds. This is similar to NVC SM16 (see APPENDIX I), although *Glaux maritima* is conspicuous by its absence. At the northeastern edge of the map, as shown in Figure 34, lies a stand of *Phragmites*. An *Aster*-dominated community similar to NVC SM12 is present in the east. SM12 is described in Rodwell (2000) as having *Salicornia* as a constant species however, here, no such association was recorded.



Zone	Dominant Vegetation	NVC Community
A	*Spartina*	SM6
B	*Puccinellia/ Aster/ Plantago*	SM10
C	*Agrostis / Festuca*	SM16
D	*Puccinellia/ Aster/ Plantago*	SM10
E	*Agrostis / Festuca*	SM16
F	*Puccinellia/ Aster/ Plantago*	SM10
G	*Scirpus*	S21b
H	*Aster*	SM12
I	*Puccinellia/ Aster/ Plantago*	SM10
J	*Phragmites*	N/a

Figure 34 Vegetation zones within the Camel salt marsh (Smillie, 2006)

Spartina is an invasive non-native species to the UK. Historical vegetation maps and descriptions suggest this species is extending into the mudflats of the Camel Estuary rather than displacing native salt marsh species (Smillie, 2006).

5.4 Geology

The catchment of the Camel Estuary is primarily underlain by Upper and Middle Devonian Slates (Environment Agency, 1997c) composed mainly of interbedded sandstones and slates with subordinate limestones and volcanic tuffs, plus balsatic lavas in the Harbour Cove Slate and the Trevose Slate formation (Selwood *et al*., 1998). The Trevose Slate and Harbour Cove Slate Formations contain the most important mineral veins in the catchment.

The rivers Amble and Allen drain the Harbour Cove Slate Formation, Polzeath Slate Formation, Trevose Slate Formation and the Tredorn Slate Formation, whilst the River Camel drains the western margin of the Bodmin Moor Granite and overlying head deposits, before flowing south then west across the Trevose Slate Formation (Selwood *et al*., 1998). Smaller rivers drain the Staddon Grit Formation, Bedruthan Formation and the Trevose Slate Formation to the south of the estuary.

5.5 Geochemistry and Mineralogy

Pirrie *et al.* (2000) examined a number of cores from the Camel Estuary, with a core of 600 mm depth being extracted from the salt marsh sample area adjacent to the Amble Marsh Tidal Barrier. Tin value was low at the base of cores but rapidly increased to around 7500 mg/kg before gradually decreasing towards the upper portion of the core. Zinc and copper exhibited similar concentrations and distributions, with both peaking at approximately 300 mg/kg. These metals also exhibit some correlation with the distribution of tin, although actual loadings are far less variable. Lead concentrations are of a comparatively low concentration (typically less than 50 mg/kg) but do mimic variations with zinc (Pirrie *et al.*, 2000). The maximum concentrations of metals recorded by Pirrie *et al.* (2000) within Core 5 are all within the range measured by Smillie (2006). Only arsenic is of a higher concentration (maximum 283 mg/kg compared with 242 mg/kg). The bioavailability of the metals is likely to be low, as the elements exist primarily as geochemically stable minerals (Pirrie *et al.*, 2000).

Bryan *et al.* (1980) recorded surface sediment at 50 to 165 mg/kg for copper, 33 to 100 mg/kg for arsenic and 330 to 1000 mg/kg for tin, although no specific sampling points were given. The results from Smillie (2006) suggest metal concentration is raised compared to Bryan *et al.* (1980). The focus, however, for Bryan *et al.* (1980) was concerned with animals living in inter-tidal mudflats, rather than vegetated salt marshes. Vegetated areas have been correlated with higher concentrations of metals compared with unvegetated areas (Doyle & Otte, 1997).

Table 7 Summary of a selection of geochemical data from sediment studies in the Camel Estuary

Author	Cu	Zn	Fe	Mn	Pb	Sn	Ni	As
Smillie, 2006	128 (77-267)	223 (172-280)	38869 (15960-47350)	592 (385-1078)	71 (44-159)	376 (128-940)	45 (15-88)	88 (33-242)
Pirrie et al., 2000 Core 5	241+	207+	-	-	53+	842+	-	283+
Bryan et al., 1980	63 (45-80)	138 (60-215)	19500 (11400-27600)	414 (275-552)	47 (30-64)	741*	21 (28-14)	55*

Results expressed as mg/kg of dry weight. All results are mean concentrations except (+) indicating maximum concentrations. (-) indicates element not reported. Results also include the range in parenthesis where available. (*) denotes only one sample taken.

Additional geochemistry data is presented in Table 8.

Table 8 Further geochemical data from the Camel Estuary from Smillie (2006)

Author	Na	K	Mg	Cl	Ca
Smillie, 2006	8759 (3922-12654)	22414 (17679-26311)	9011 (7620-10080)	3783 (1936-6804)	29468 (3621-89815)
NO_3-N	**S**	**P**	**Moisture**	**pH**	**LOF**
3.61 (0-31.47)	1832 (900-2800)	980 (356-1408)	54 (27-96)	7.9* (7.1-8.6)*	164632 (83000-234600)

Results expressed as mg/kg of dry weight except (*) as pH units. All results are mean concentrations with the range in parenthesis. n = 22

Spatial geochemical data from Smillie (2006) is presented a series of contour maps (Figure 35 to Figure 39) using Surfer software.

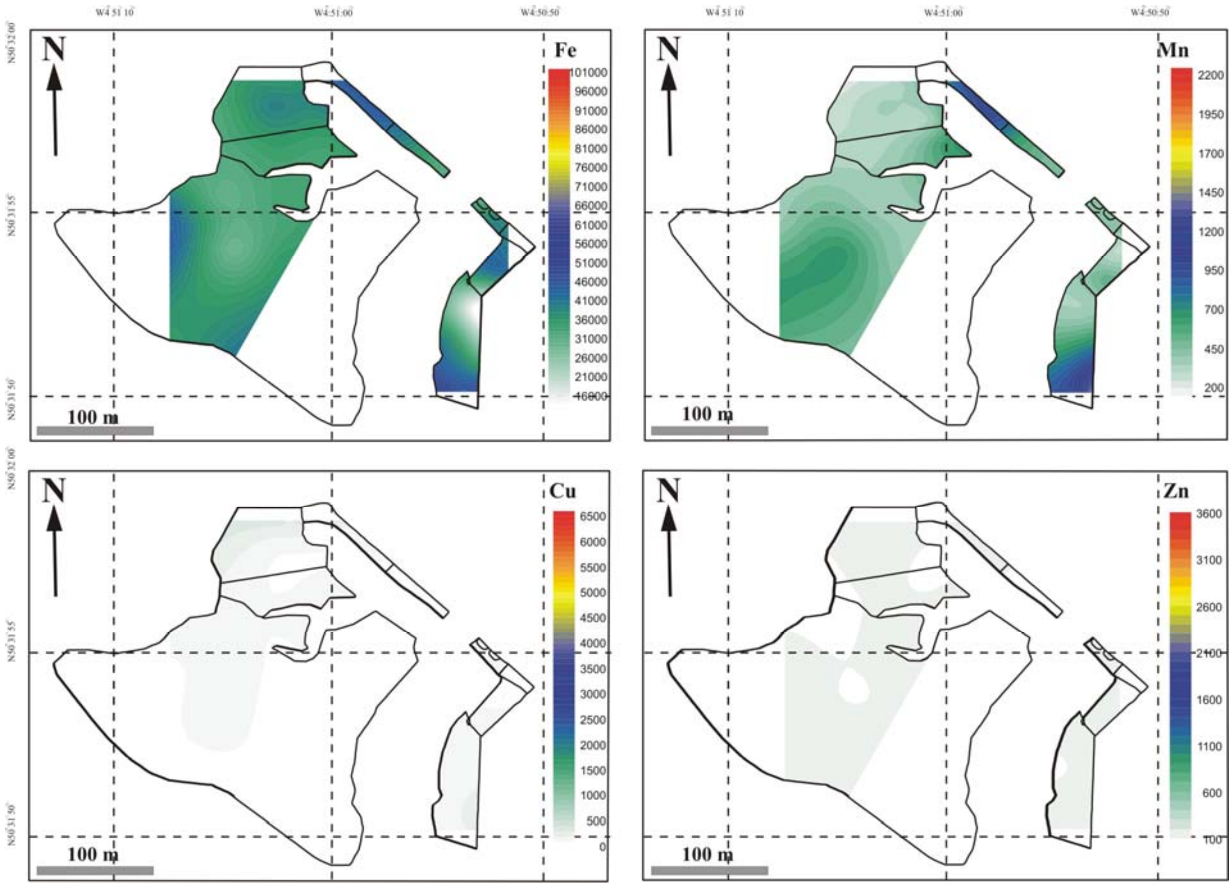


Figure 35 Spatial variation of iron, manganese, copper and zinc in the Camel

The concentrations of metals in comparison to other salt marshes in this publication are limited.

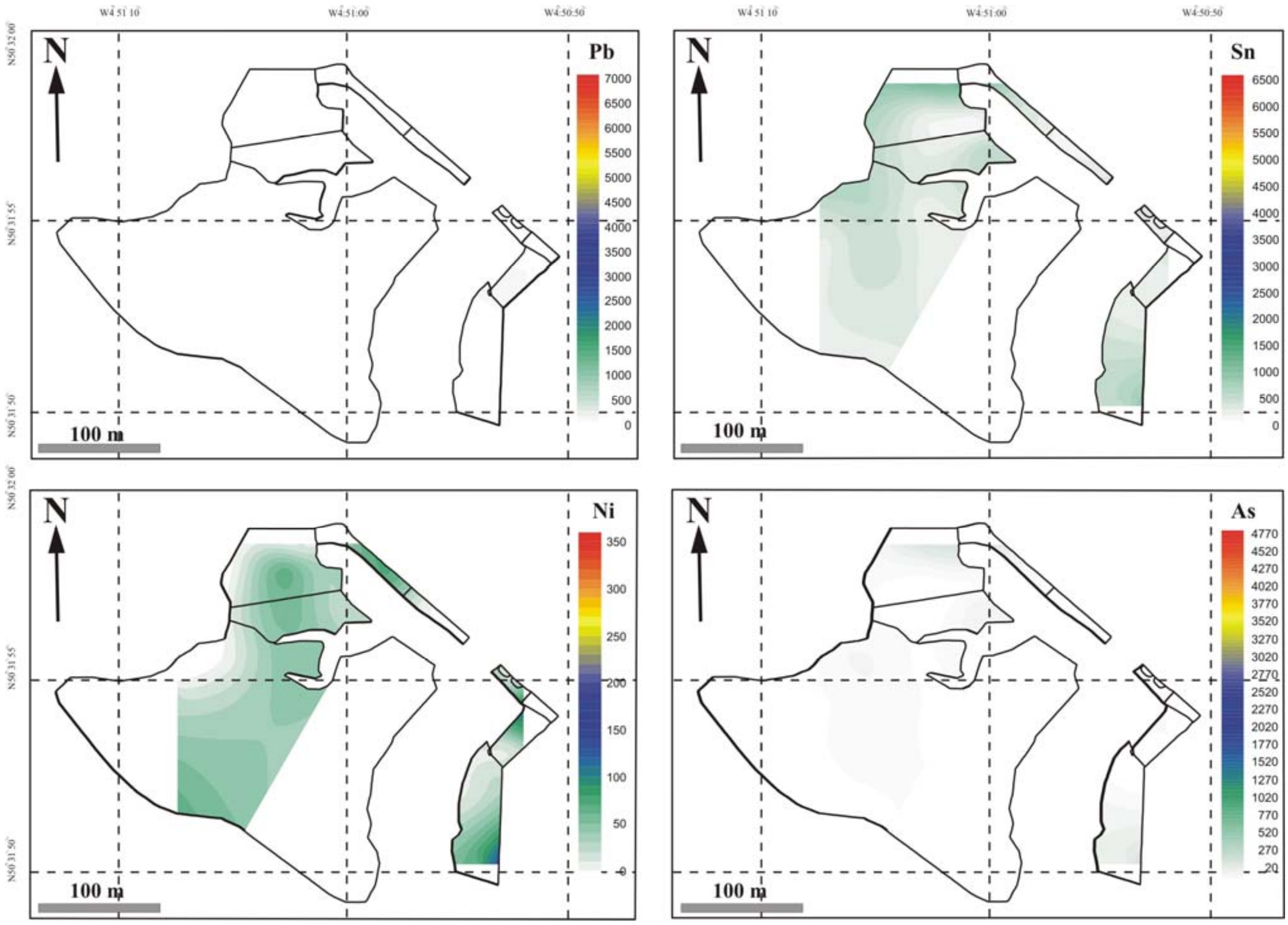


Figure 36 Spatial variations of lead, tin, nickel and arsenic in the Camel

Lead is extremely impoverished compared to the other salt marshes of interest. Lead, tin, nickel and arsenic are relatively uniformly spread throughout the marsh.

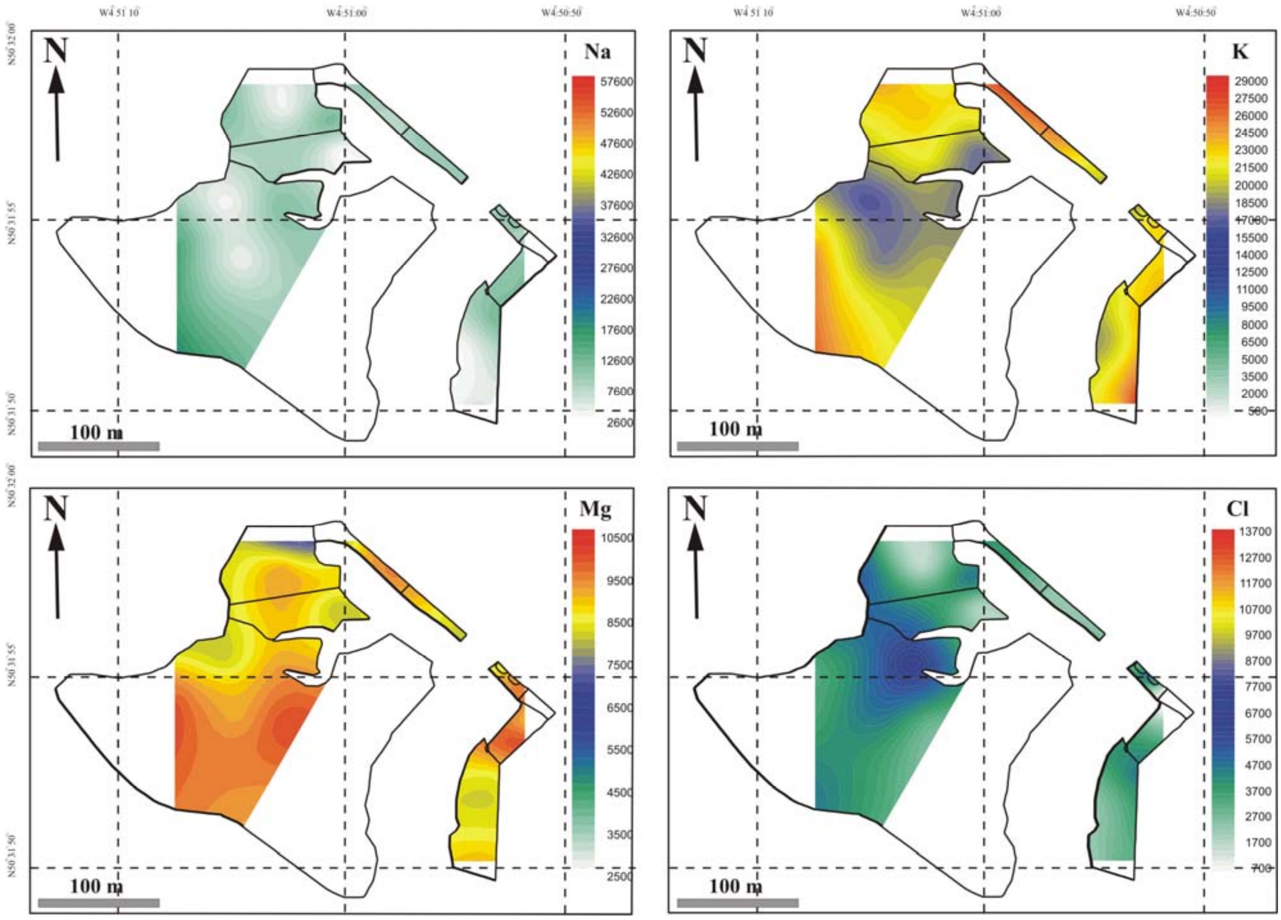


Figure 37 Spatial variations of sodium, potassium, magnesium and chlorine in the Camel

The pioneer *Spartina* zone possesses the highest concentrations of both potassium and chlorine. Equally, this zone also exhibits the highest concentration of magnesium.

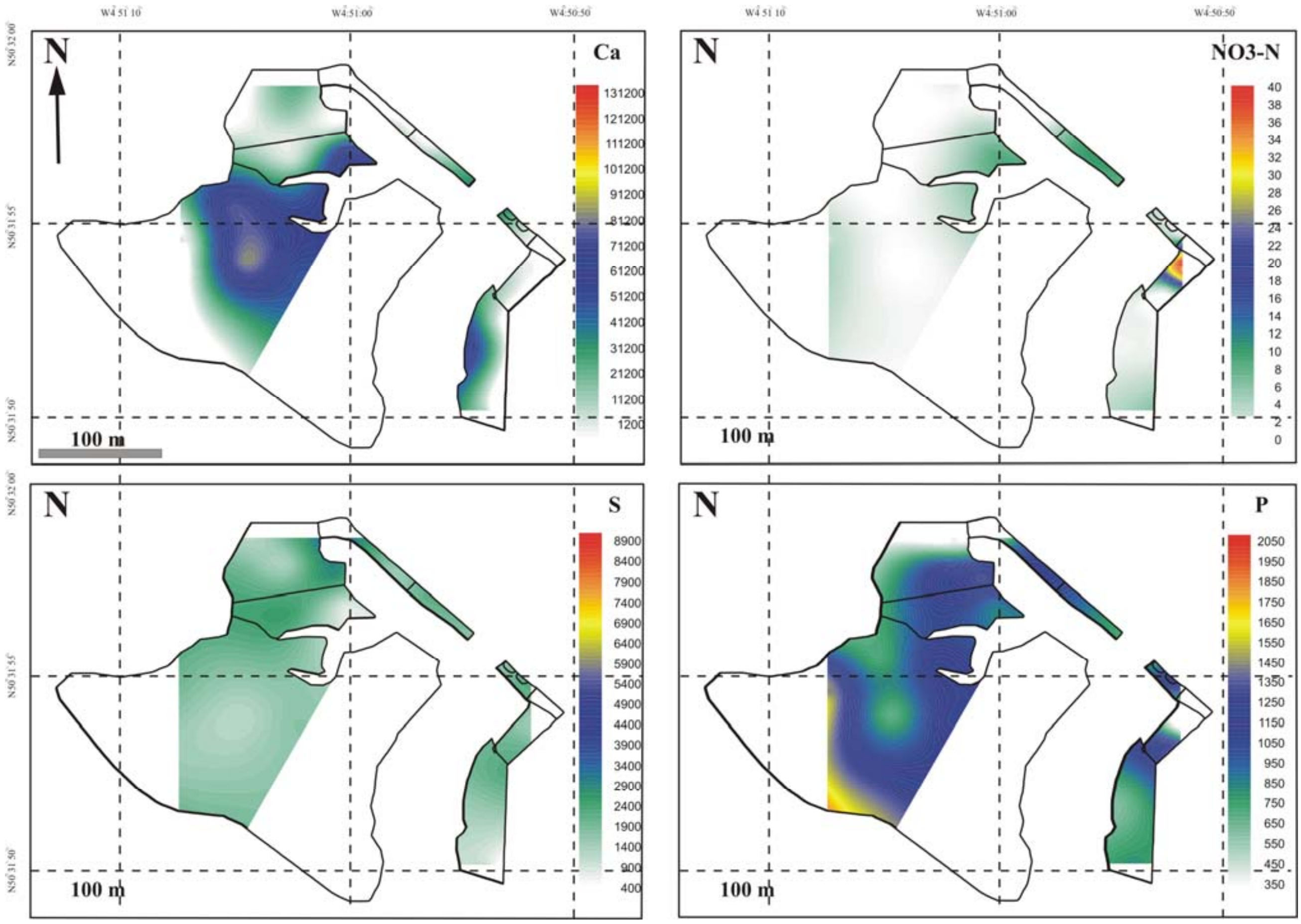


Figure 38 Spatial variations of calcium, nitrate-nitrogen, sulphur and phosphorus in the Camel

Calcium and phosphorus are particularly concentrated within the *Spartina* community. The low range of nitrate-nitrogen concentrations suggests that this compound in the limiting factor for growth within the marsh (Figure 38).

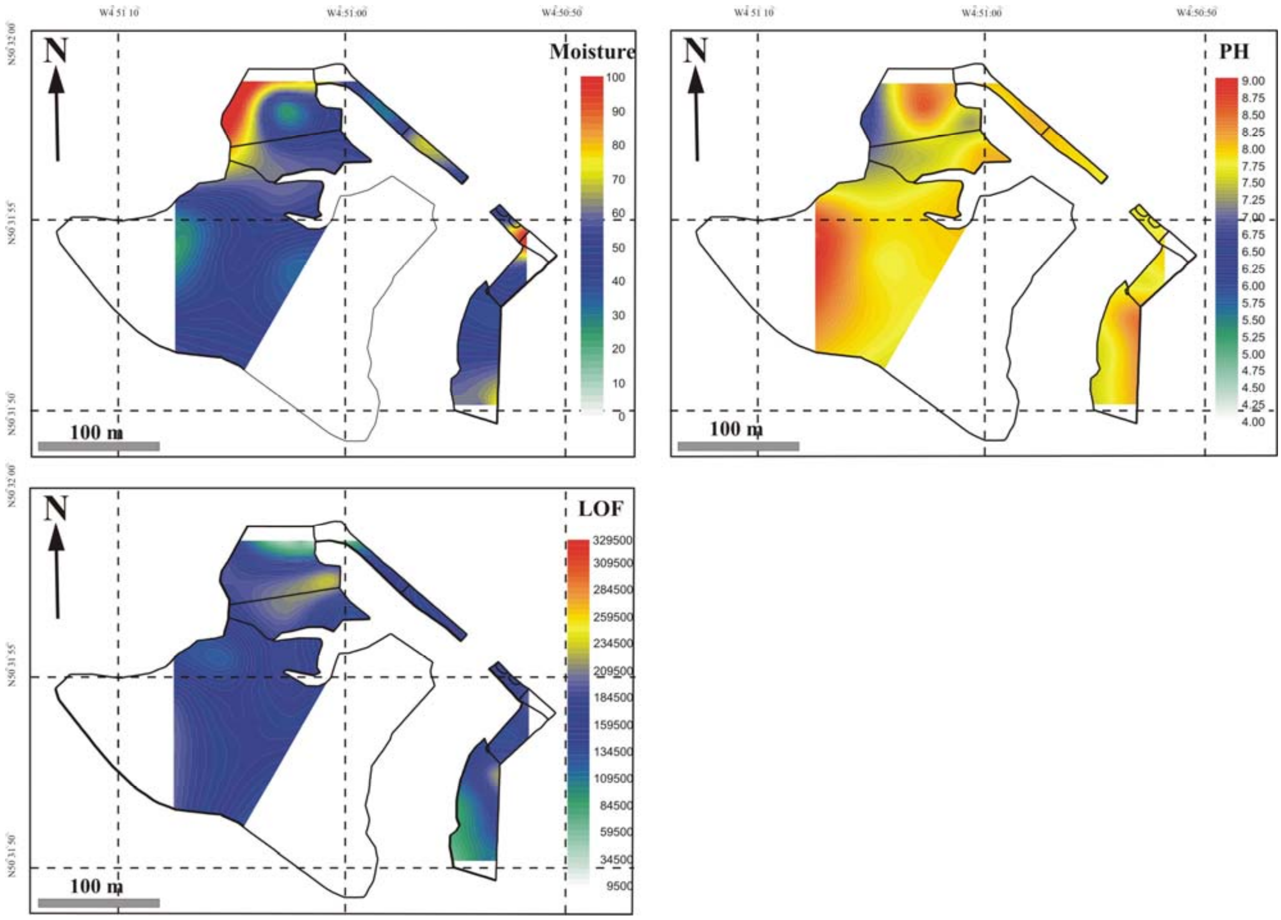


Figure 39 Spatial variations of moisture content, pH and loss-on-fractionation in the Camel

pH is particularly high in the Camel, compared to the other marshes in this study (Figure 39), however, results are still around neutral. Both LOF and moisture content are reasonably uniformly spread throughout the marsh, bar individual hot spots.

6 GANNEL

6.1 Mining History

The Gannel (Figure 40) drains an important area for lead, zinc and silver mining. Indeed the upper reaches of the estuary contain the most lead-contaminated sediments for any estuary in Cornwall or Devon, at least (Pirrie *et al.*, 2000). The most significant mines within the catchment include East Wheal Rose, which operated from 1845 to 1885 (Dines, 1956), with production peaking in 1850 (Reid & Scrivenor, 1906). This mine principally produced lead ore with significant amounts of silver, plus lesser amounts of copper and zinc ore (Dines, 1956). Other mines in this area include Wheal Constance, Cargoll, South Cargoll and New Cargoll, which principally worked lead, zinc and silver lodes, along with small amounts of copper ore between 1845 and 1892 (Dines, 1956).

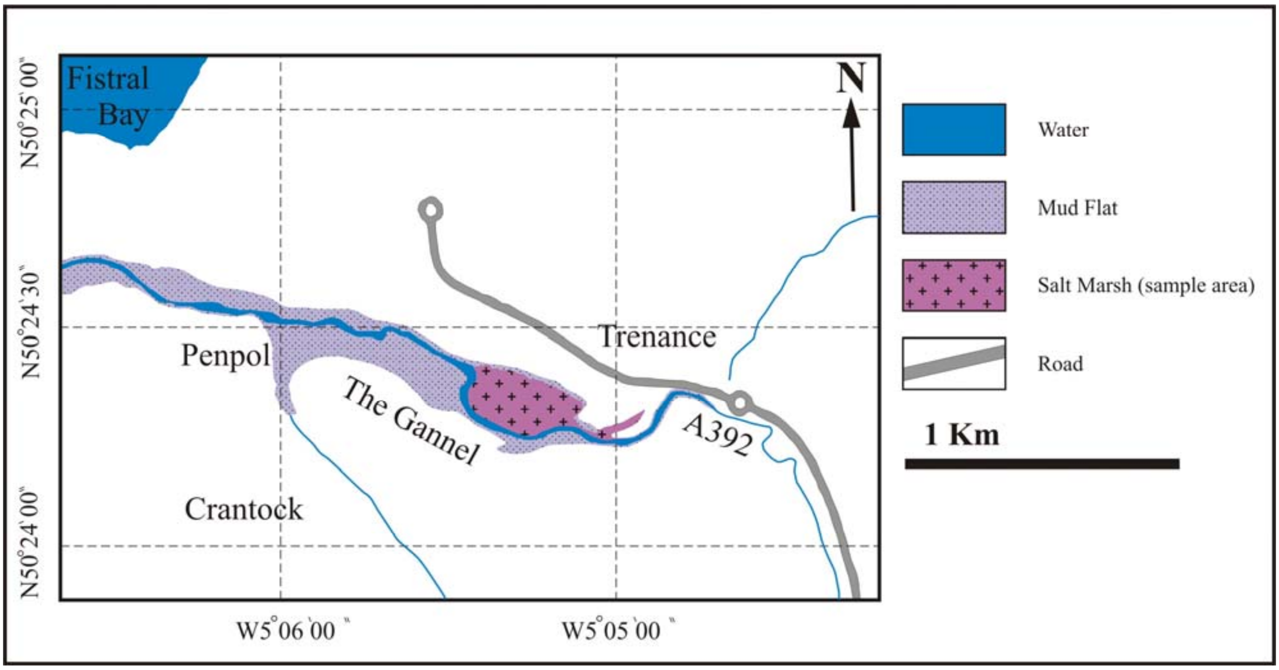


Figure 40 The Gannel Estuary

Reid & Scrivenor (1906) reported that siltation derived from mining activity, combined with marine sand, had silted up the estuary so that it "could no longer be used for shipping". Pirrie *et al*. (2000) reported that a significant release of mine waste is most likely associated with the release of tailings following the closure of mines in the late 19th Century.

6.2 Landscape and Geomorphology

Wind-blown sands dominate much of the Gannel Estuary. Most of these sands have stabilised, however, mobile dunes exist towards the mouth of the Estuary (Ford, 1992). The banks of the

Gannel Estuary are designated a Special Area of Great Landscape Value (SAGLV). This designation occurs when an area is considered to be equivalent in value to an AONB in a local context (Environment Agency, 1997c).

Figure 41 The Gannel looking west towards the open sea

The estuary stretches for 70.6 ha with much of the adjacent land being used for residential and agricultural purposes (Cornwall Naturalists Trust, 1980b). Livestock farming is the principal farming activity within the wider catchment (Environment Agency, 1997c). The estuary receives heavy public pressure due to the close proximity of Newquay (Ford, 1992).

6.3 Ecology

6.3.1 Ecological Status

The Gannel Estuary is not protected by either statutory or non-statutory (such as AGSV) conservation designations. The catchment does drain the Newlyn Downs SSSI, however, this site is protected due to the occurrence of lowland wet heath and is not likely to impact the estuary.

The abandoned coastal lead mines now house important bat colonies (Environment Agency, 1997c).

6.3.2 Floristic Studies

The salt marsh of the Gannel Estuary is relatively small (20.25 ha), but well developed in the upper-reaches of the estuary (Burd, 1989). The vegetation, however, does not exhibit the clear zonation pattern normally associated with salt marsh communities (Cornwall Naturalists Trust, 1980b). Pressures on the salt marsh include fishing, bait digging, boat mooring and walking, however, vegetation appears little affected by these activities (Cornwall Naturalists Trust, 1980b).

The Salt Marsh Survey of Great Britain (Burd, 1989) describes the largest portion of salt marsh (Figure 40) with *Salicornia* as the pioneer species, sparsely colonizing the mud flats at the edge of the salt marsh. This gives way to a zone of *Puccinellia*, rapidly succeeding to an area of *Halimione* with *Puccinellia* clumps. A mix of *Armeria* and *Halimione* dominates the main portion of the salt marsh. This is bordered to the south by a line of *Halimione* and to the north by sparse *J. maritimus* with *Armeria* and *Festuca*. To the west of the zone of *J. maritimus* lays a community of *Armeria* and *Festuca*, with *J. gerardii* immediately to the north. The *J. maritimus* zone succeeds to the *Armeria/ Halimione* complex to the east (Burd, 1989).

Figure 42 *Puccinellia* dominated marsh succeeding to *Halimione* by the River Gannel

Ford (1992) records that the salt marsh is predominantly composed of *Salicornia* and *H. portulaciodes*, with *Armeria maritima* and *Festuca rubra* present away from the river. Stands of *Juncus* species, *Scirpus maritimus* and *C. extensa* are noted towards the road to the north.

A survey by the Cornwall Naturalists Trust (1980b) describes the sample area (Figure 40) as having *Salicornia* as a pioneer species with *Armeria* and *Halimione* present as a community upon bare patches, with the more developed marsh as being dominated by *Armeria* and *Festuca* with *Aster* and *Cochlearia* also present. Both nationally rare (*J. compressus, Schoenoplectus tabernaemontani* and *Apium graveolens*) and county rare species (*H. portulaciodes, Ranunculus sceleratus, C. anglica* and *Oenanthe lachenlii*) were present, however, no mention is made of the location of these species.

Figure 43 *Puccinellia* and *Cochlearia*

Smillie (2006) reports a patchy *Puccinellia* dominated community is found both on the pioneering area of the main portion of the salt marsh and as a couple of small islands (Figure 44). This zone is probably best represented by the NVC community SM13a. A *Halimione* dominated community representative of SM14 is present particularly lining the River Gannel to the south and also as part of a small island to the west. There is, however, an under-representation of *Puccinellia* in this community, except in the pioneer are to the west where this community grades into a species-poor *Puccinellia* dominated SM14 NVC community. The largest community in the salt marsh is primarily dominated by *Armeria* with *Plantago* constant, occasionally existing at co-dominant coverage. This community approximately matches to the NVC sub-community SM13d, although *Puccinellia* is, on the whole, substantially under-represented. Located within SM13d community are two *Juncus spp*. communities: one dominated by *Juncus gerardii* (comparable with SM16b) and the other dominated by *Juncus maritimus* (comparable with SM18a) with rare *Parapholis strigosa*. Towards the landward end of exists a community dominated by *Plantago*, with rare *Armeria*. *Parapholis strigosa* is rare to occasional in this zone. This matches a species-poor version of the NVC SM13d sub-community.

A number of nationally rare (*J. compressus*, *Schoenoplectus tabernaemontani* and *Apium graveolens*) and county rare species (*Ranunculus sceleratus, C. anglica* and *Oenanthe lachenlii*) recorded by the Cornwall Naturalists Trust (1980b) were not found by Smillie (2006). The location of these species was not recorded and so it may be possible that they occur outside of the survey area of the research by Smillie (2006).

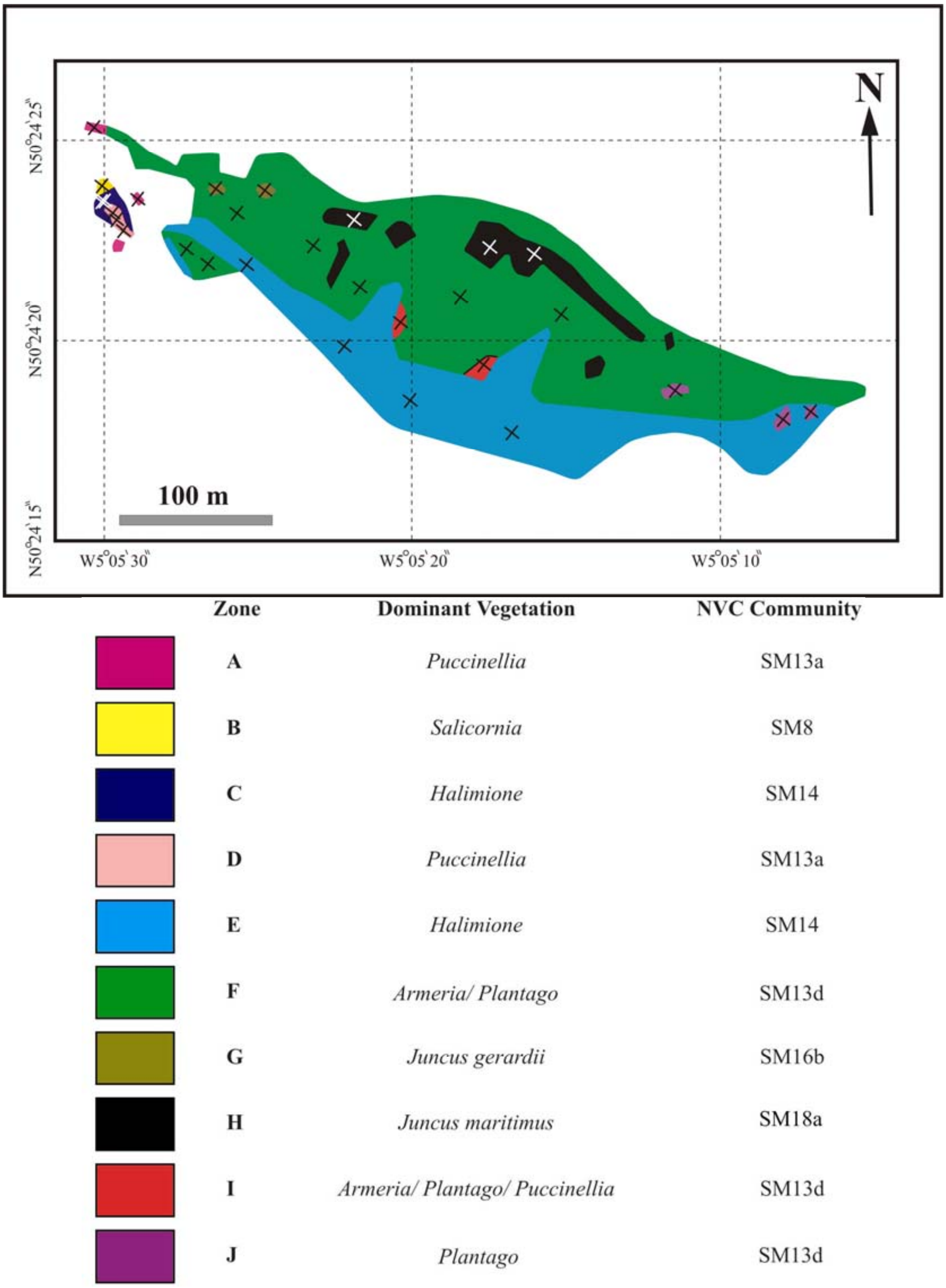


Figure 44 Vegetation zones within the Gannel salt marsh (Smillie, 2006)

Sample points are marked with (X). Zone E borders the River Gannel to the south. The low marsh and pioneer communities (zones A, B and D) are more prevalent to the east, corresponding with the seaward end of the marsh.

6.3.3 Invertebrate Studies

In a study on the use of amphipods to biomonitor metal concentrations within British coastal waters, Rainbow *et al.* (1989) discovered elevated copper concentrations within *T. saltator* and *T.*

deshayesii. Zinc, however, was not enriched within tissues of the organisms in the Gannel Estuary. Comparison with other estuaries made these invertebrates unreliable for biomonitoring purposes. Copper and zinc concentrations within *O. gammarellus* did, however, reflect concentrations within coastal waters, thereby making *O. gammarellus* a suitable organism for biomonitoring of copper and zinc in British coastal waters (Rainbow *et al.,* 1989).

Biopsies of the polychaete *N. diversicolor* and the deposit-feeding bivalves *Scrobicularia plana* and *Macoma balthica* revealed high concentrations of metals within tissues (Bryan *et al.,* 1980). Pirrie *et al.* (2000) suggested this reflected the bioavailability of metals in the Gannel Estuary.

Brown (1976) revealed that the isopod, *A. meridianus*, within the Gannel Estuary, was tolerant to lead but not copper. Tolerance was found to be inherited through the use of breeding studies. This contrasts with findings from the Hayle Estuary, where the species was found to be tolerant to both copper and lead.

6.4 Geology

The catchment area of the Gannel is underlain by undifferentiated Devonian Meadfoot beds composed of slates and thin limestones along with the Porthtowan Formation to the south (Pirrie *et al.*, 2000). The most significant mineralisation is the lead-zinc-silver deposits around Newlyn Downs (Pirrie *et al.*, 2000).

6.5 Geochemistry and Mineralogy

Geochemical research on the Gannel Estuary is limited, although sediment research by Bryan *et al.* (1980) did report exceptionally high lead concentrations (around 2000 mg/kg) within surface sediments. This result is also reflected in research from Thornton *et al*. (1986), which reported sediment lead concentrations of 2670 mg/kg of lead, as well as elevated zinc levels (2995 mg/kg). Copper concentrations (313 mg/kg), however, were comparable with levels found within the Camel Estuary (Pirrie *et al.*, 2000).

Pirrie *et al.* (2000) sampled the salt marsh of the Gannel Estuary using cores of less than 1 m. Again, the Gannel Estuary was found to be enriched in lead (typically between 500 and 1000 mg/kg) and zinc (commonly around 600 mg/kg). These metals were found to correlate positively with depth. One core recorded values of over 8500 mg/kg of lead and 1600 mg/kg of zinc between 150 and 350 mm below the sediment surface. This section also displayed elevated copper (300 mg/kg) and arsenic (520 mg/kg) concentrations, though typically these metals are below 100 mg/kg.

Tin values were of a relatively low concentration (below 150 mg/kg) throughout the cores. No strong correlation with copper, tin or arsenic was found with lead or zinc.

Smillie (2006) also found lead to be particularly high within the salt marsh of the Gannel, although extremely variable (188 to 6877 mg/kg). Indeed, large ranges in all metal concentrations were found to be a feature of the marsh. For example, copper ranged from 42 to 530 mg/kg, zinc had a minimum value of 238 mg/kg and a maximum of 3543 mg/kg, whilst tin ranged from 79 to 553 mg/kg. The variations in metal concentrations are probably partially attributable to sampling technique, for example, Smillie (2006) sampled the bulk sediment to 500 mm, whilst Pirrie *et al.* (2000) split cores into 50 mm downcore depth intervals. Pirrie *et al.* (2000) also describes metal concentrations varying spatially, as well as downcore.

Table 9 Summary of a selection of geochemical data from sediment studies in the Gannel Estuary

Author	Cu	Zn	Fe	Mn	Pb	Sn	Ni	As
Smillie, 2006	211 (42-530)	731 (238-3543)	42700 (19320-71610)	892 (385-1540)	1861 (188-6877)	285 (79-553)	41 (10-86)	201 (24-559)
Pirrie et al., 2000 Cores 4, 5 and 6	411+	3500+	-	-	16000+	200+	-	975+
Bryan et al., 1980	137 (91-217)	664 (357-1215)	29200 (26800-33200)	1067 (922-1160)	977 (376-2175)	428 (305-550)	33 (25-49)	174 (115-233)

Results expressed as mg/kg of dry weight. All results are mean concentrations except (+) indicating maximum concentrations. (-) indicates element not reported. Results also include the range in parenthesis where available.

The mineralogy reflects the geochemistry, being dominated by abundant lead and zinc phases. Dominant minerals are galena, sphalerite and a mineral complex identified as probably plumbogummite (Pirrie *et al.*, 2000). Abundant minerals are iron oxides/ carbonates, detrital pyrite and zircon, whilst cassiterite, barite, arsenopyrite, loelignite, monazite, xenotime, apatite, rutile and ilmenite are less common (Pirrie *et al.*, 2000). Diagenetic minerals, such as framboidal pyrite and diagenetic lead phases, are abundant within the lead and zinc enriched horizons of the salt marsh (Pirrie *et al.*, 2000). This suggests that lead, zinc and copper are dominated by the more mobile phases, thus making these elements potentially bioavailable (Pirrie *et al.*, 2000).

Additional geochemistry data from Smillie (2006) is presented in Table 10.

Table 10 Further geochemical data from the Gannel Estuary from Smillie (2006)

Author	Na	K	Mg	Cl	Ca
Smillie, 2006	11477 (3700-20794)	19565 (11122-23904)	8191 (6840-9840)	6560 (2088-13490)	31296 (2343-138734)
NO_3-N	**S**	**P**	**Moisture**	**pH**	**LOF**
5.35 (0-38.46)	2166 (900-3900)	889 (296-1276)	56 (32-92)	7.5 (6.8-7.3)	183576 (106800-265700)

Results expressed as mg/kg of dry weight except (*) as pH units. All results are mean concentrations with the range in parenthesis.

Spatial geochemical data is presented a series of contour maps (Figure 45 to Figure 49) using Surfer software (Smillie, 2006).

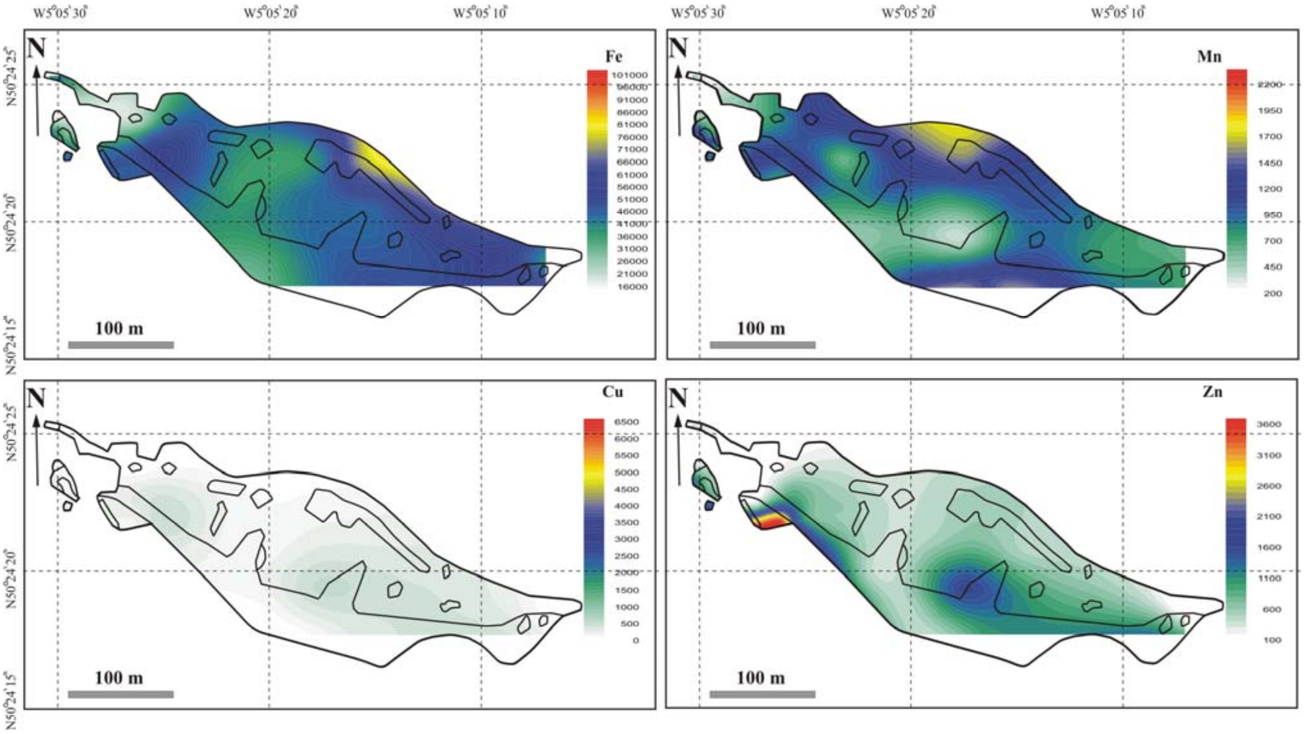


Figure 45 Spatial variation of iron, manganese, copper and zinc in the Gannel

Iron, copper, zinc and manganese are well distributed marsh (Figure 45). A hot spot for zinc (1644 mg/kg) is present within the centre of the marsh, though this is based on results from a single sample.

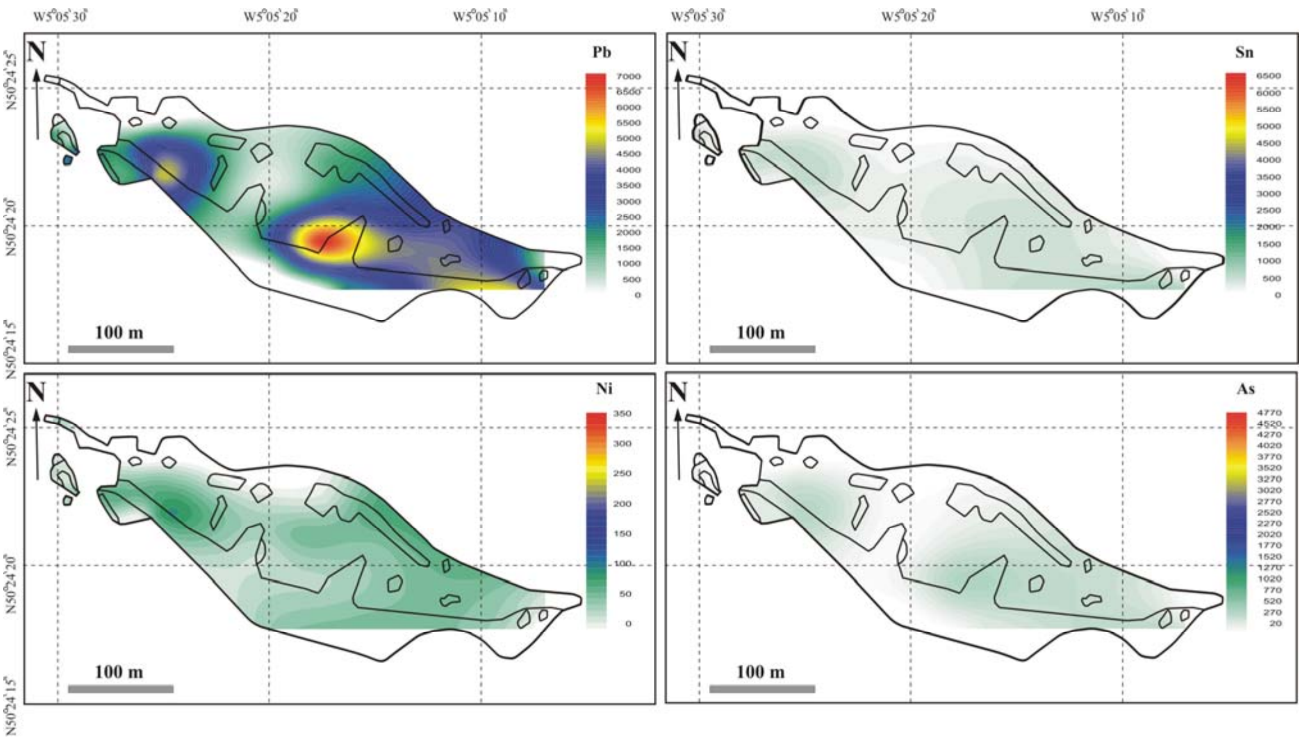


Figure 46 Spatial variations of lead, tin, nickel and arsenic in the Gannel

As with iron, copper, zinc and manganese, Figure 46 shows that lead, nickel, tin and arsenic are well distributed throughout the Gannel Estuary. The hot spot noted for zinc also contains relatively elevated concentrations of lead (6877 mg/kg).

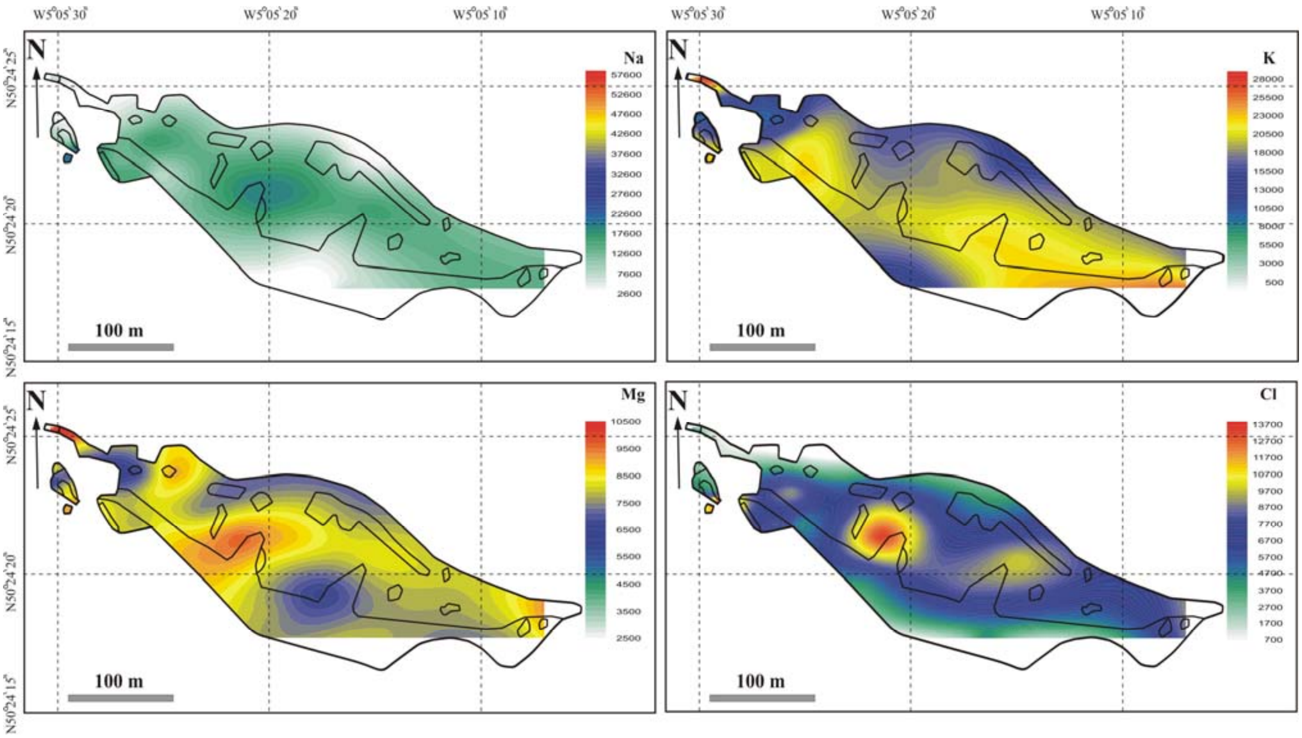


Figure 47 Spatial variations of sodium, potassium, magnesium and chlorine in the Gannel

There is some, though limited, evidence of a reduction in salinity at the south of the sample area, possibly due to the river that runs to the south (Figure 47), as well as to the north, probably due to distance from the sea and the presence of a slope.

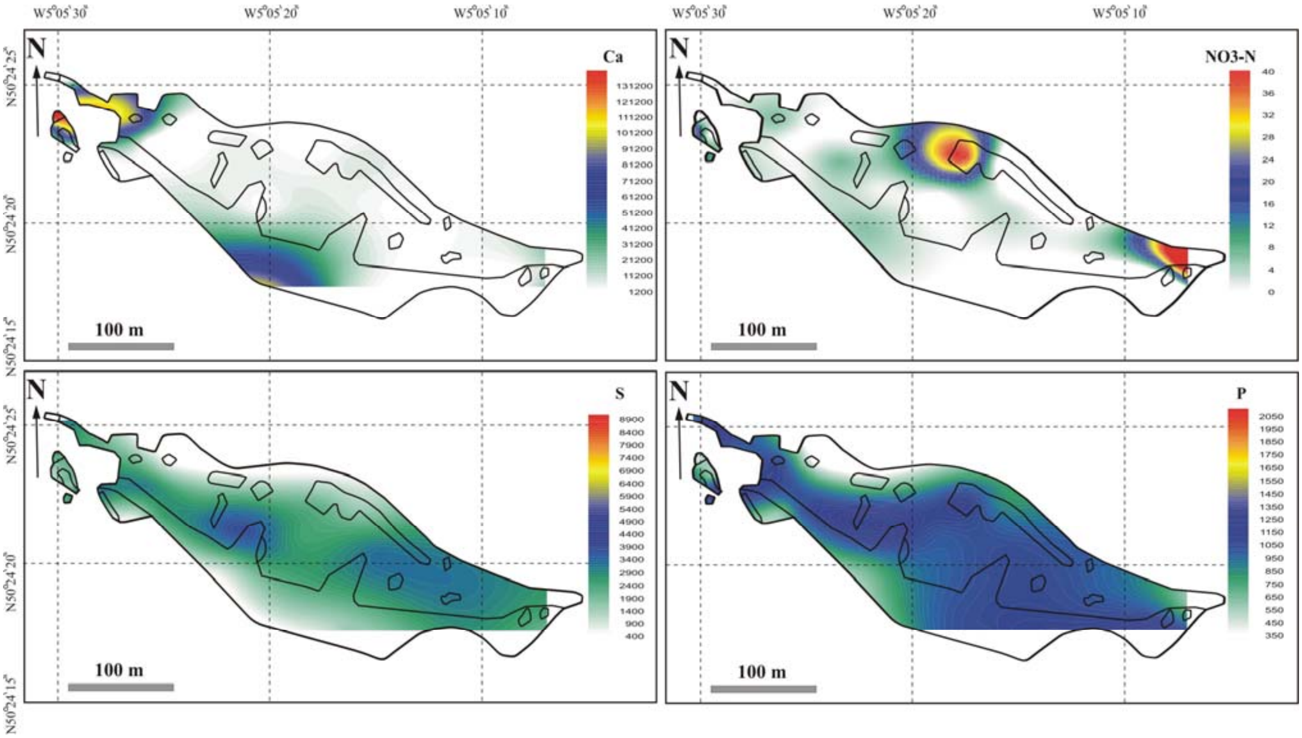


Figure 48 Spatial variations of calcium, nitrate-nitrogen, sulphur and phosphorus in the Gannel

Calcium is particularly high at the west (Figure 48), possibly indicative of the remnants of shells, these zones being closest to the sea. Nitrate, phosphorus and sulphur are well spread over the salt marsh, although a nitrate hot spot (38.46 mg/kg) is found in the north.

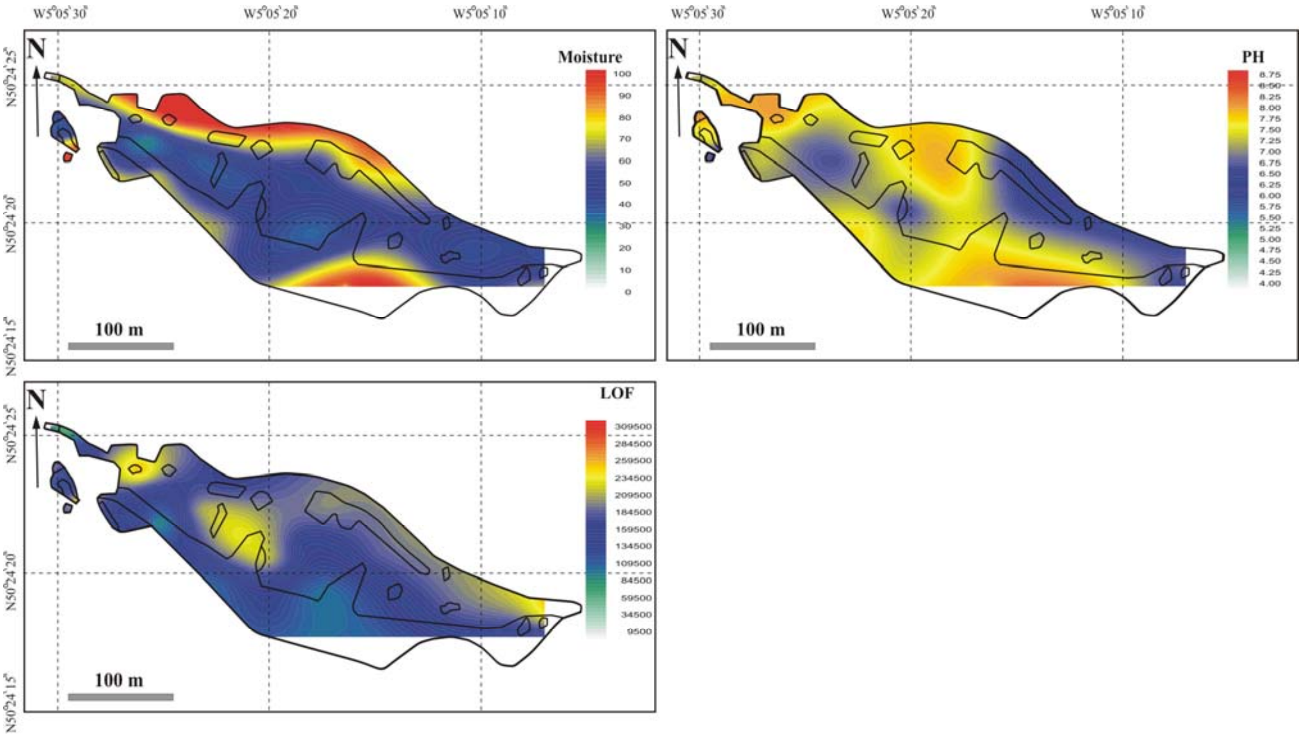


Figure 49 Spatial variations of moisture content, pH and loss-on-fractionation in the Gannel

Relatively high moisture content is linked with the north of the sampling area (Figure 49), associated with the *Juncus* communities (Figure 44). pH, although variable, is always around neutral, generally between 7 and 8 pH units. The highest pH is recorded towards the seaward end of the marsh, to the west (Figure 49).

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APPENDIX I

NVC communities

NVC communities found within the salt marshes examined by Smillie (2006). Descriptions are largely taken, or adapted, from Rodwell, 1991 *et seq.*: -

SM6 (*Spartina anglica* salt marsh community)

Spartina always dominates as scattered tussocks, coalescing clumps or a continuous sward up to 1 m height. The community is species-poor though the associates are somewhat varied. *Puccinellia maritima* and annual *Salicornias* occur frequently and may account for up to 50% cover.

SM8 (Annual *Salicornia* salt marsh community)

Comprises of ephemeral stands of *Salicornia*, usually with no other species. Scattered associates include *Puccinellia*, *Suaeda* and *Spartina*, with other species only occasional.

SM10 (Transitional low-marsh vegetation with *Puccinellia maritima*, annual *Salicornia* species and *Suaeda maritima*)

Generally species-poor but always dominated by varying proportions of *Puccinellia*, *Salicornia* and *Suaeda*. *Aster* may also be frequent, though never abundant.

SM12 (Rayed *Aster tripolium* on salt marsh)

Most commonly encountered within saline areas with some freshwater influence, such as in brackish ditches behind sea walls, with *Spartina* and *Puccinellia* as associates. Also abundant within periodically flooded saline muds in inland salt marshes, with *Spergularia* and *Puccinellia* as associates. The ecology of this community is not yet fully understood.

SM13 (*Puccinellia maritima* salt marsh community)

Usually occurs as a species-poor closed grassland but may be present across a wide range of marsh zones, from pioneer to high marsh herb-dominated stands. *Triglochin*, *Plantago* and *Armeria* are the most common associates. **SM13a** represents the most species poor aspect of this community, where *Puccinellia* is constant and dominant throughout. The **SM13d** occurs where these three associates become abundant and constant with *Puccinellia* contributing far less frequently than in the typical community.

SM14 (*Halimione portulacoides* salt marsh community)

This is a closed species-poor community, dominated by the distinctive bushy *Halimione* canopy, with *Puccinellia* constant. *Suaeda* may also be frequent.

SM15 (*Juncus maritimus-Triglochin maritima* salt marsh community)

Tall tussocks of *J. maritimus* overwhelmingly dominate, with *Triglochin* and *Plantago* constant in small amounts. Species such as *Puccinellia*, *Aster, Glaux* and *Armeria* can occur throughout. Differs from SM18 through the lack of grass species and *Juncus gerardii* present.

SM16 (*Festuca rubra* salt marsh community)

Commonly dominated by mixtures of *Festuca* and *Agrostis* with a variety of herbaceous associates, such as *Plantago, Glaux, Armeria* and *Triglochin. J. gerardii* is usually an associate. **SM16a** sub-community is dominated *J. gerardii* and is rarely larger than 2-3 m diameter at most. *Plantago, Glaux, Armeria* and *Triglochin* are constant, though rarely abundant.

SM18 (*Juncus maritimus* salt marsh community)

Usually dominated by tall clumps of *J. maritimus,* with an understory of *Festuca, Agrostis, Glaux* and *J. gerardii*. May also be associated with mesotrophic grassland and disturbed ground flora.

SM21 (*Suaeda vera-Limonium binervosum* salt marsh community)

An open community dominated by scattered bushes of *Suaeda vera* and *Halimione*, with *Puccinellia* and *Limonium* constant and abundant, plus smaller amounts of *Armeria*. *Suaeda maritima* is fairly frequent throughout.

SM28 (*Elymus repens* salt marsh community)

Tall *Elymus* dominates this tall sward with other grass species, *Festuca* and *Agrostis* also present. Beneath the sward is usually sprawling *Atriplex*, or, on more open ground, *Potentilla anserina*. *Oenanthe lachenalii*, *Sonchus arvensis*, *Rumex crispus* and *Cirsium arvense* are occasional and often give a scruffy appearance to the vegetation and tussocks of *J. gerardii* or *Festuca* may be locally prominent.

S21b (*Scirpus maritimus* swamp-*Atriplex prostata* sub-community)

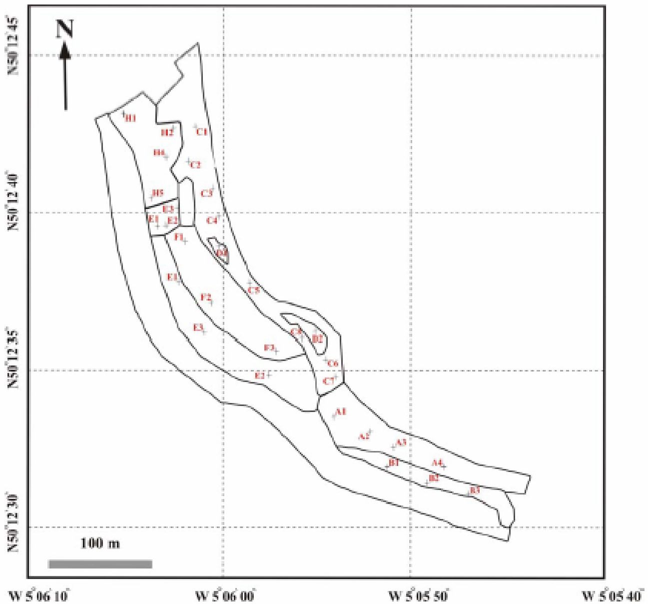
Dominated by tall *Scirpus* almost to the exclusion of all other vegetation. This particular sub-community is associated with salt marshes – S21 being more commonly associated with lowland wetlands. *Atriplex* is a constant associate as are salt marsh plants, though species may vary widely.

As well as the above communities, the salt marshes examined also contained the **S4** *Phragmites australis* swamp. This community is recognisable by the monoculture of *Phragmites*; other vegetation contributing typically no more than 5% cover.

APPENDIX II

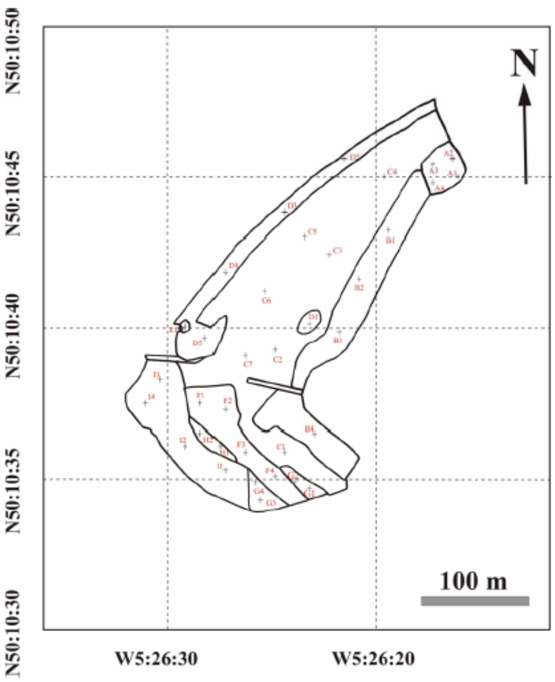
Domin Vegetation Data

Table 11 NVC survey of Restronguet Creek



Sample	*Armeria maritima*	*Salicornia Spp.*	*Suaeda maritima*	*Spergularia media*	*Atriplex hastata*	*Agrostis stolonifera*	*Festuca rubra*	*Bare*
A1		7		4				8
A2		7		4				8
A3		7		4				8
A4		8		5				8
B1	8	4	4		1	3	5	
B2	9	4	4	3			5	3
B3	9	4	3	3				4
C1	5						10	
C2	4					4	10	
C3	7	5	2		1	4	7	
C4	4				5	5	7	
C5	7	2			5	4	8	
C6	2				2	4	10	
C7	4				2	4	10	
C8	7	2	2		3	4	7	
D1	2	7		5	3		7	4
D2		8		5	5	5		
E1	2	5		5				7
E2		6		6				7
E3		6		6				5
F1	3	7		5		2		7
F2	2	7		4		3		8
F3	4	5		5				8
G1	6	4						8
G2	5	4		3				8
G3	5	4	2		2		2	8
H1		4		6			2	8
H2		5		6			2	8
H3	2	3		6			2	8
H4		5		6	1	2		8

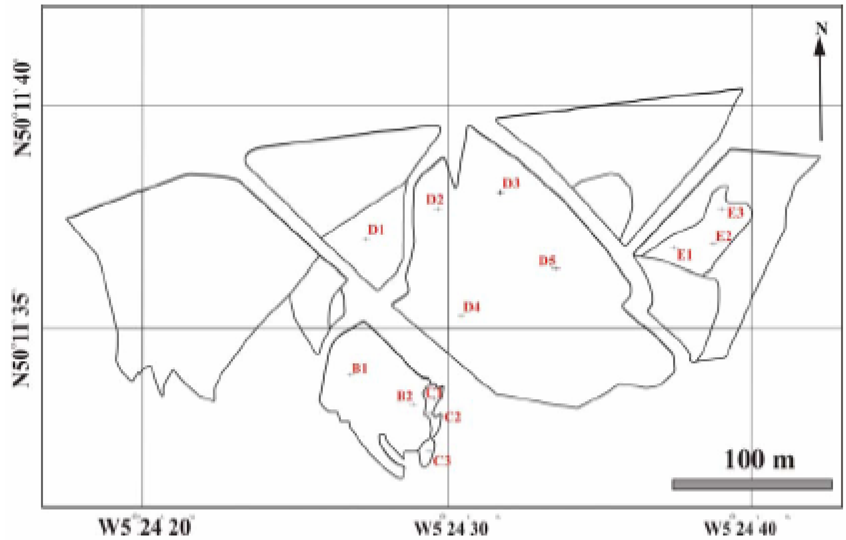
Table 12 NVC survey of Lelant



Sample	*Armeria maritima*	*Salicornia Spp.*	*Atriplex hastata*	*Agrostis stolonifera*	*Festuca rubra*	*Bare*	*Juncus maritimus*
A1	3	9				4	
A2	4	7				7	
A3	2	7				7	
A4	1	7				7	
B1		8				7	
B2		7				7	
B3		7				7	
B4		6				8	
C1	9	1				4	
C2	7						
C3	7					1	
C4	7	1				4	
C5	5	2					
C6	8						
C7	8	2				1	
D1							10
D2							10
D3							10
D4			2				10
D5							10
E1				2	4		4
F1	2	3				4	
F2	3				2	2	
F3	2			1			
F4	2		3			2	
G1			1		10		
G2				1	10		
G3					10		
G4					10		
H1				10	4		
H2				10	4		
I1				2			
I2				3			
I3					7		3
I4				4			4

Sample	*Plantago maritima*	*Aster tripolium*	*Cochlearia officinalis*	*Elymus repens*	*Carex extensa*	*Triglochin maritima*	*Juncus gerardii*	*Puccinellia maritima*
A1	4							
A2	5							
A3	7							2
A4	7			2				2
B1								
B2								
B3								
B4								
C1	5							
C2	7	4						
C3	7	4	2					
C4	7	2	2					
C5	8	4	3					
C6	8	4	3	3				
C7	5	3		2				
D1								
D2								
D3								
D4								
D5								
E1	3	2			2		10	
F1	4	2	4		10	1		
F2	4	3	4		10			
F3	4	3	4	1	10			
F4	4		4	3	10			
G1	4	3			1			
G2	4	2						
G3	4	3						
G4	4	3			2			
H1				3				
H2				4				
I1		1	1	10				
I2				10				
I3				8				
I4				9				

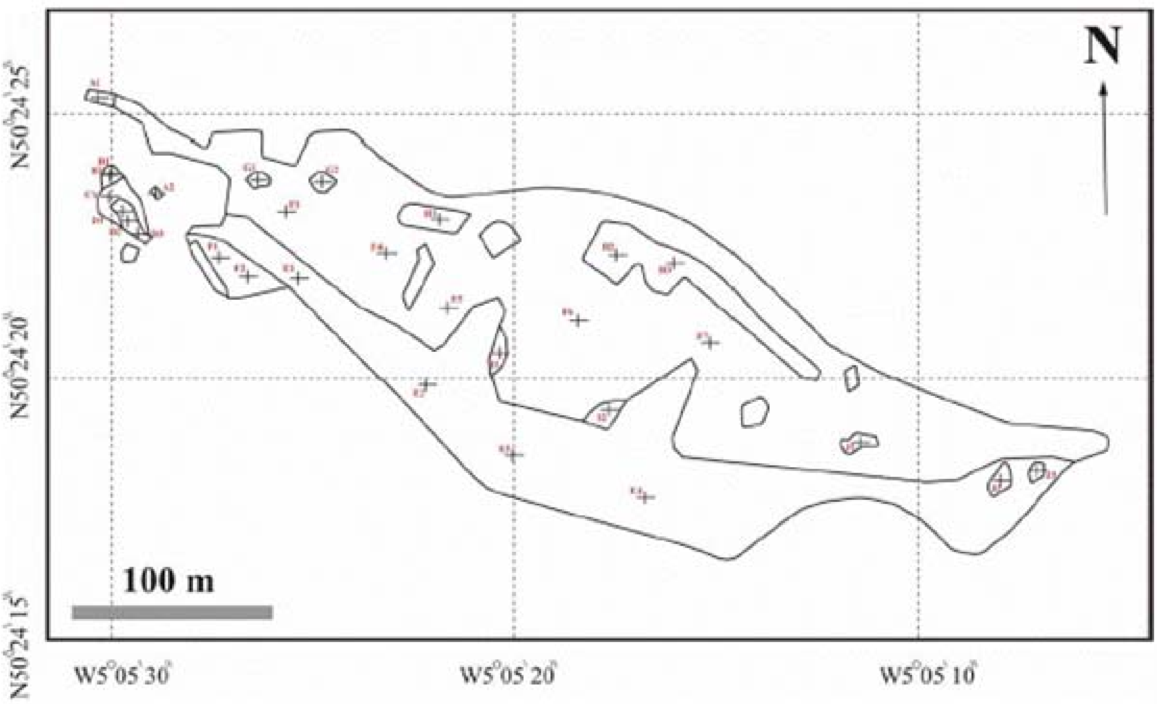
Table 13 NVC survey of Copperhouse



Sample	*Armeria maritima*	*Spergularia media*	*Agrostis stolonifera*	*Festuca rubra*	*Bare*
B1		4			4
B2		4			1
C1	10				
C2	10				
C3	10			4	3
D1	5	7			3
D2	7	4			4
D3	8	5			5
D4	8	5			4
D5	8	5			5
E1	6	4	4		4
E2	8				2
E3	5	5			

Sample	*Juncus maritimus*	*Plantago maritima*	*Aster tripolium*	*Carex extensa*	*Puccinellia maritima*
B1			4		10
B2			4		10
C1		2	3		
C2		4	3		
C3			4		
D1					7
D2		7		4	
D3				2	
D4					3
D5				4	
E1		2		8	
E2	3	4		5	
E3		4		8	

Table 14 NVC survey of Gannel



Sample	*Armeria maritima*	*Salicornia Spp.*	*Spergularia media*	*Festuca rubra*	Bare	*Juncus maritimus*	*Plantago maritima*
A1		3			6		
A2		3			6		
B1		7			8		
B2		7			8		
C1		2			2		
D1		2	2				
D2							
D3							
E1							1
E2			2				5
E3							
E4	2		4				4
F1	7						7
F2	7						6
F3	8					1	4
F4	8						6
F5	10						4
F6	7						7
F7	6						8
G1	2		1				4
G2	2						3
H1						7	6
H2						7	6
H3				4		7	5
I1	8		1				6
I2							8
J1						2	10
J2				1			8
J3	1						10

Sample	*Aster tripolium*	*Cochlearia officinalis*	*Carex extensa*	*Triglochin maritima*	*Juncus gerardii*	*Puccinellia maritima*	*Halimione portulacoides*
A1				4		8	
A2	2			4		8	
B1							
B2							
C1							10
D1	3					10	4
D2						10	4
D3						10	4
E1							10
E2				3		1	9
E3							10
E4		2		4		2	8
F1	4	2				2	
F2	2	1		4			
F3	5	3		3			
F4	3	1	4	2			
F5	4	2		4			
F6	5	4		2			
F7	4						
G1	4	1		4	8		
G2	4	3		7	7		
H1	2	4		5			
H2	3	5					1
H3	5	4	1	4			
I1	4	4				5	
I2	4	4				7	
I3		3		2			
J1	4	4		3			1
J2	4	4		2			
J3	4	4				5	