

Future challenges of coastal landfills exacerbated by sea level rise

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Abstract

In England and Wales, there are at least 1700 coastal landfills in the coastal flood plain and at least 60 threatened by erosion, illustrating a global problem. These landfills are a major issue in shoreline management planning (SMP) which aims to manage the risks associated with flooding and coastal erosion. Where landfills exist, “hold the line” (requiring the building or upgrading of artificial defences to maintain the current shoreline) is often selected as the preferred SMP option, although government funding is not available at present. To investigate these issues in detail, three case-study landfills are used to examine the risks of future flooding and erosion together with potential mitigation options. These cases represent a contrasting range of coastal landfill settings. The study includes consideration of sea-level rise and climate change which exacerbates risks of erosion and flooding of landfills. It is fundamental to recognise that the release of solid waste in coastal zones is a problem with a geological timescale and these problems will not go away if ignored. Future erosion and release of solid waste is found to be more of a threat than flooding and leachate release from landfills. However, while leachate release can be assessed, there is presently a lack of methods to assess the risks from the release of solid waste. Hence, a lack of science constrains the design of remediation options.

1. INTRODUCTION

Coastal flooding and erosion risks around the world (and the UK) are growing due to climatic and human factors (e.g., Wong et al. 2014; Sayers et al. 2015; CCC, 2018). These risks present important challenges to coastal planning. Within the UK there is a strong desire to move to more sustainable shoreline management to address these challenges, whilst balancing public investment and benefits. One major constraint on this goal is the legacy of historic coastal landfills (containing potentially hazardous waste) located along UK coasts. It

is estimated that there are at least 1700 permitted (licensed) and historic landfills in coastal flood and erosion buffer zones in England and Wales alone (Brand et al. 2017; Wadey et al. 2019). These include landfill sites (1) in low-lying floodplains, (2) on eroding cliff tops and (3) immediately behind coastal defence structures (Figure 1).

Shoreline management plans (SMP) in England are produced by Local Authorities in conjunction with the Environment Agency (EA) (DEFRA 2006; Nicholls et al. 2013). SMPs identify the most sustainable approach to managing the risks associated with flooding and coastal erosion. They consider three different epochs: the short-term (0 to 20 years), medium-term (20 to 50 years) and long term (50 to 100 years), when the impact of climate change and sea-level rise (SLR) becomes increasingly important. SMPs cover policies and planning for areas of coast that can either have active defence lines (e.g. a sea wall) or more natural defences such as dunes and beaches. There are four different SMP policy options that can be implemented for each discrete shoreline management unit (EA, 2018): (1) Hold the (existing defence) line (**HTL**): “An aspiration to build or upgrade artificial defences to maintain the current shoreline”, (2) Advance the line (**ATL**): “New defences are built seaward of existing defences”, (3) Managed realignment (**MR**): “Allowing the shoreline to move naturally, but managing the process to direct it in certain areas. This is usually done in low-lying areas, but may occasionally apply to cliffs”, and (4) No active intervention (**NAI**): “There is no planned investment in defending against flooding or erosion, whether or not an artificial defence has existed previously”.

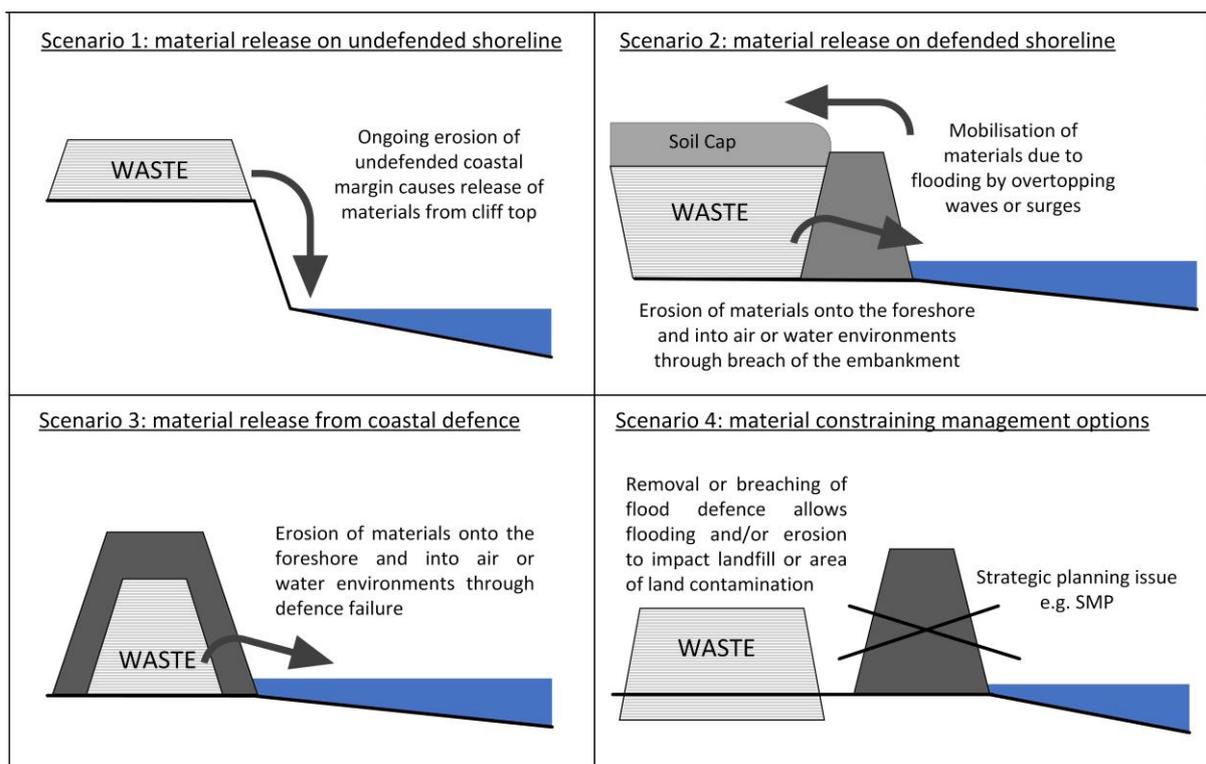


Figure 1. Possible situations (or scenarios) where landfill waste can be released to the sea (adapted from Cooper et al. 2012).

Adopting more sustainable shoreline management would allow coasts to be more dynamic in less developed areas and transition from HTL to MR or NAI SMP policies. However, less interventionist policies need to consider the effects on property and potential loss of habitats, which could include designated sites. The potential release of pollution from historic landfill sites would also need to be managed and often leads to a HTL policy where otherwise MR or NAI might be selected. Note that where HTL is selected in SMPs this is an aspiration, and funding is not guaranteed. In particular, there is no funding presently available to protect

coastal landfills (Wadey et al., 2019; Nicholls et al., 2019).

The aim of this paper is to analyse the implications of historic coastal landfills for shoreline management, including the potential effects of erosion and flooding/leachate under rising sea levels and possible mitigation measures and their limitations. Erosion and the physical removal of landfill, and flooding and the flow of water through or over a landfill are fundamentally different processes as noted below. The research was based on three case study sites in the south of England, covering contrasting landfill situations (e.g. Figure 1). These landfills are all presently or potentially subject to erosion, while two are in the coastal flood plain. They are described in detail below.

2. METHODOLOGY

2.1 Site identification and selection

CIRIA guide C718 (Cooper et al. 2012) provides generic guidance on the identification and management of landfill sites and areas of land contamination located on eroding or low-lying coastlines. The guide identified four main situations (termed scenarios) where wastes or materials could be released from a landfill by erosion or sea flooding (see Figure 1). This guidance was used as a basis to help identify a representative set of potentially vulnerable sites, assess their risks and appraise long term management options. Table 1 shows the factors considered within the site selection process.

Table 1. Criteria for selection of potential Case Study Sites

Criteria	Options considered
What is the dominant coastal hazard?	Flooding, Erosion, Combined
What is the coastal landform?	Low-lying open coast, Cliff, Estuarine
What is the SMP policy option?	Managed realignment (MR) Hold the line (HTL) No active intervention (NAI)
What is the land use setting?	Urban (human, properties, etc.) Rural (environment, agriculture, etc.) Mixed land use
What are the waste and landfill properties?	Known: Hazardous/ Known: non-hazardous Unknown: Potentially hazardous

The main source of information for site selection was provided by the Environment Agency who maintained a number of geographic information system (GIS) based databases (now available from <https://data.gov.uk>). These consider authorised (currently permitted) and historic landfill sites, coastal flood zones, coastal erosion zones, estuary boundaries, SMPs, and extreme sea levels (Batstone et al. 2013). Other data sources included the OMReg (Online Marine Registry) database. These datasets were analysed in a GIS environment. The potential for flooding at the sites in the coastal flood plain was based on bath-tub flood analysis in combination with projected sea levels based on climate changes scenarios. Assessment of the risk of cliff erosion was based historic erosion rates and the application of a simple model of soft-cliff erosion (Walkden and Dickson (2008)). Details of site specific data sources and methods used for coastal erosion and flood analysis are given in Supplementary Information.

The geographical region of the study area extends 185km along the south coast of England from Lyme Regis to Shoreham-by-Sea. In total there were over 148 historic landfill sites that intersected the Environment Agency's 1 in 200 year flood level. Under a NAI policy scenario, 26 sites were within the next 20 years erosion buffer zones, increasing to over 60 sites within the next 50-100 years.

In consultation with stakeholders, which included representatives from the UK regulator (Environment Agency), local government and consultants, three case study sites were selected for analysis as detailed in Table 2 and Figure 2.

Table 2. Summary of three case study sites.

Landfill	Spittles Lane	Wicor Cams	Pennington Marshes
Scenario Type (see Figure 1)	Scenario 1	Scenario 2 and 3	Scenario 4
Coastal landform	Cliff	Estuarine	Low-lying open coast
Dominant coastal hazard	Erosion	Combined	Combined
Erosion status today	Actively eroding	Limited erosion	None (defended)
Flood status today	Not applicable	Regular seawater contact	Limited (defended)
SMP policy option	NAI all epochs	HTL all epochs	HTL all epochs
Land use setting	Rural, World Heritage Site, SSSI, SAC	Rural - recreational	Rural – recreational. SPA, SSSI, SAC RAMSAR.
Waste and landfill properties	Mostly non-hazardous	Mostly non-hazardous	Mostly non-hazardous
Location	50°43'50"N 2°55'42"W	50°43'52"N 1°32'49"W	50°50'28"N 1°09'58"W

SAC = Special Areas of Conservation; SPA= Special Protected Areas; SSSI = Site of special scientific interest; RAMSAR = a Wetland of International Importance.

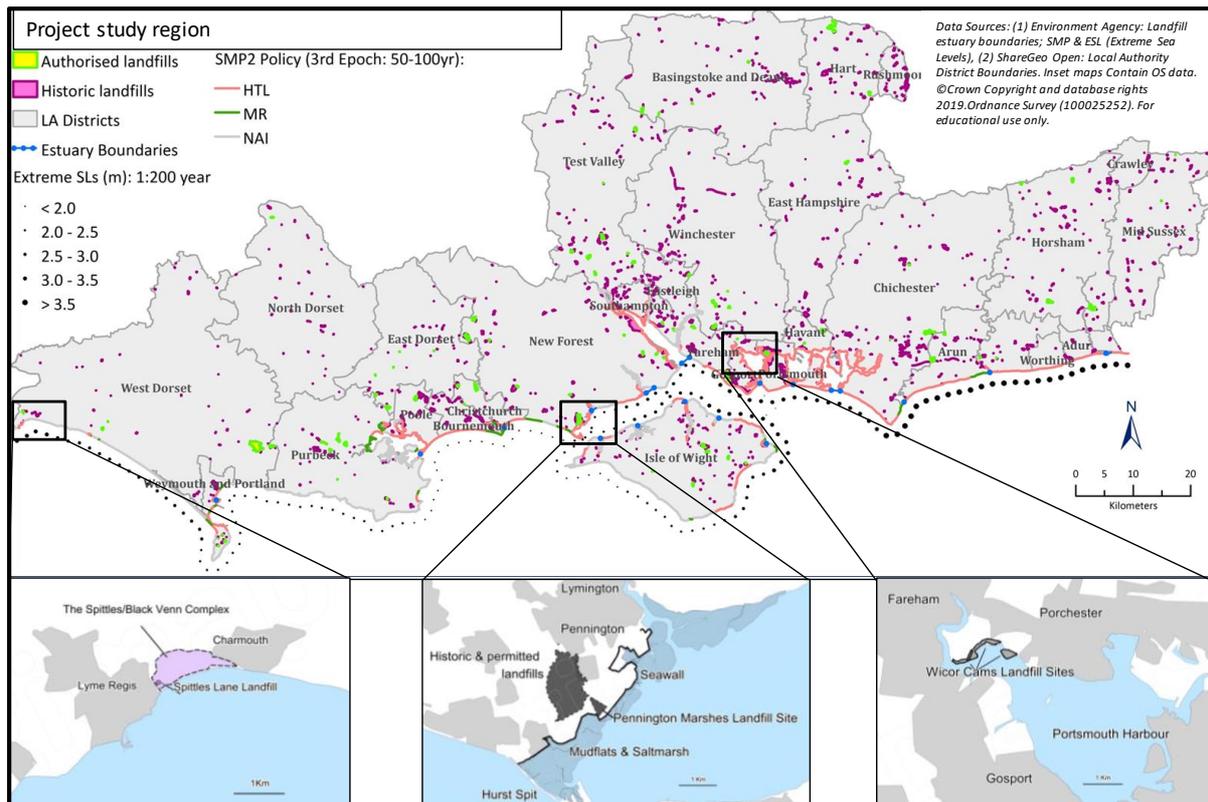


Figure 2. Location of three case study sites.

2.2 Climate change and sea-level rise scenarios

The impact of projected SLR on coastal landfills and SMPs was considered by superimposing extreme water levels with SLR projections onto the surface topography of the case study sites and any surrounding sea defences. Low, medium and high SLR projections for the year 2050 and 2100 are summarised in Table 3; these scenarios are consistent with the recent UKCP18 scenarios. By 2100, SLR is projected to range between 0.2 and 0.75 metres in the study area, with 1.5 metres (H⁺⁺) as a plausible high end scenario. UKCIP09 climate change scenarios (Lowe et al, 2009 and Supplementary Information) suggest increased frequency of wet years for southern England which may result in increased leachate production at all sites.

Table 3. Sea-level rise projections for low, medium and high scenarios for south coast of England (based on Haigh et al., 2009; Lowe et al., 2009).

Time Slice	Sea-Level Rise Scenario	Sea-level Rise (m)	Average annual rise (mm/year)
2015	Base case	0	
2050	Low	0.05	1.4
	Medium	0.15	4.3
	High	0.25	7.2
2100	Low	0.2	2.3
	Medium	0.4	4.7
	High	0.75	8.8
	H ⁺⁺	1.5	17.6

3. LANDFILL CASE STUDY ASSESSMENTS

3.1 Lyme Regis

The Spittles Lane landfill at Lyme Regis (Figures 2 and S1) is perched on an eroding cliff face approximately 50 metres above sea level. The cliff is composed of Lias sediments of Lower Jurassic age, with sub-horizontal layers of mudstones interspersed with harder limestones. It is located within the Spittles-Black Ven landslide complex that represents an unstable and active coastal slope that extends for more than 1 km from Lyme Regis eastwards towards Charmouth. Parts of the landfill have already been undercut (principally during a major landslip in 2008) and have fallen into a tumble zone of waste and natural soils within the landslide complex which feeds waste onto the beach below (Gallois, 2009). The lighter fraction of waste arriving at the beach is being washed away, leaving behind the denser fraction – predominantly metals, glass and masonry / concrete-based waste products.

The landfill site is located within the Dorset and East Devon Coast UNESCO World Heritage Site. More specifically the site lies within a designated SSSI based on the active landslides and geological exposure and is a SAC. This has influenced the SMP policy for the area, with a HTL policy and recently upgraded defences around Lyme Regis town, but NAI for all three epochs at the landfill. The SSSI designation prevents any other SMP policy being adopted for this coastline.

3.1.1 Site characterisation

Inert, industrial, commercial and household wastes were deposited at the site from the 1910s or early 1920s by Lyme Regis Borough Council, and by predecessor local government organisations. West Dorset District Council (WDDC) is now responsible for managing the landfill site and the foreshore below. During the early period of operation, the waste was routinely burned at the site. Bennett (2007) concluded that the site had ceased taking waste by 1978, and identified the early areal extent of the landfill. A re-analysis of aerial photographs used by Bennett (2007) in combination with evidence provided by a local historian indicate that the footprint of the historic tip may originally have covered an area of approximately 1.6 ha, with a maximum dimension perpendicular to the coast of about 130 metres. The area has been reduced to approximately 1.4 ha due to the erosion already discussed, releasing waste materials. Average waste depth is unknown, but maximum average depth is probably less than 3 metres. The volume of waste in the landfill is uncertain, but is estimated to vary from 14,000 m³ to 42,000 m³. Approximately 2,000 to 6,000 m³ of waste has been eroded from the site.

As a consequence of the 2008 landslip that resulted in wastes being deposited on the beach, WDDC commissioned a number of site investigations and reports under Part 2A of the Environmental Protection Act (EPA) (1990). Six rounds of sampling were undertaken between 2009 and 2013 (WPA Consultants, 2010, 2013) and samples were taken from the tumble zone just below the landfill and from the beach. Access to the site is difficult and comprehensive sampling was not undertaken. Descriptions of waste samples included the presence of brick, concrete, pottery, metal, glass, plastic and asbestos (e.g. cement sheets and tiles; chrysotile fibres were identified) in a predominantly soil matrix. Concentrations of certain metals and organic contaminants from soil samples taken from the toe of the landfill are summarised in Table S1. Only two samples of water/leachate taken from seasonal seeps have been analysed (Bennett, 2007) and these results did not identify any indication of leachate contamination (e.g. NH₃-N less than 1 mg/l; Cd <0.01 µg/l; Cr <0.1 µg/l; Pb <0.05 µg/l; Zn <0.5 µg/l).

3.2 Wicor Cams

The Wicor Cams case study site (Figure 2 and S2) is located in a low-lying/estuarine environment in the north-west of Portsmouth Harbour in the East Solent. The site, owned by Fareham Borough Council, consists of three historic landfills (together referred to as Wicor Cams) and is surrounded by land mainly used for recreational purposes.

The coastal fringe at Wicor Cams forms a discrete area of open landscape with an eroding frontage, with parts of the low-lying area at Cams Lake containing intertidal mudflats and saltmarshes. There is evidence of an old sea wall around much of the site. Sections of the wall have historically been reinforced with concrete structures (vertical retaining wall, concrete filled sand bags) and other parts have been protected by *ad hoc* placement of large blocks of concrete or stone along the seaward face. However, the sea wall has not been maintained for decades and in places is showing signs of severe degradation and erosion, including locations where it has been completely removed. The core of the wall, although predominantly made of soils, contains some waste materials, including plastics. The adjacent shoreline is of national, European and international conservation interest, and is designated as a SSSI, SPA, and listed as a Ramsar site (FBC 2006).

The SMP option for the site is HTL in the short, medium and long term (50 to 100 years). However, currently there is little formal engineering design or coastal defence reinforcement that is underpinning this policy and slow erosion is occurring. The medium-term and long-term SMP policies for this unit note “Requirements for more detailed study (for management of site to be determined following contaminated land investigations)”. A strategy appraisal carried out for the local authorities stated that the landfill sites were undefended and erosion could present a risk to people and the environment (AECOM 2016).

3.2.1 Site characterisation

The landfill sites occupy a narrow stretch of land following the coastline for approximately 2 km (Figure 2 and S2). The combined overall surface area of all three landfill sites is estimated to be approximately 22 ha. Although there is some ambiguity about the depth of waste, an average value of 4.5m can be estimated from borehole logs, giving an upper estimate of 1 M m³ waste at Wicor Cams.

The sites are believed to have been filled from between 1942 to 1993 with a mixture of commercial, household, industrial and inert wastes by Fareham Borough Council and predecessor local government organisations.

There is no known landfill engineering to the site. Historic Ordnance Survey maps (Digimap) indicate landfilling started in the early 1940s. The superficial geology of the area is recorded as a mixture of sand, silts and clays, and appears to be less than 2 metres deep. The underlying bedrock geology belongs to the Lambeth Group consisting of mottled clay with sand, overlying Chalk.

A limited number of waste and leachate samples have been obtained from the site during a site investigation undertaken in 2013 as part of a controlled waters Qualitative Risk Assessment (QRA), and Water Framework Directive (WFD) assessment (URS, 2013). Soils and sediment samples were collected and analysed for total metal content, polycyclic aromatic hydrocarbons (PAHs), total petroleum hydrocarbons (TPH) and volatile compounds (Table S2). The limited scope of the site investigation does not represent a statistically robust characterisation of the site, but even so there was significant spatial variation in quality.

Ammoniacal nitrogen is a good indicator of leachate, and the degree to which contamination has been flushed out. Four boreholes had NH₄-N concentrations between 27 and 40 mg/l.

Ammonia in the other three boreholes was ~1mg/l with at least two of these associated with elevated chloride levels which may be an indication of previous estuarine water incursion.

3.3 Pennington

The Pennington study site (Figure 2 and S3) is located between Lymington town and the villages of Keyhaven and Milford-on-Sea at the western end of the Solent region in Hampshire County. The study concentrated on Pennington Marshes Landfill, a closed site, owned and managed by Hampshire County Council (HCC) as a coastal nature reserve. The landfill is located on low lying rural land comprising grassland/coastal grazing marshes and brackish lagoons, and is protected from flooding by a defence line consisting of a well-maintained sea wall (between Keyhaven and Lymington) and intertidal mud/saltmarsh. The landfill is located in a sensitive ecological area and is within a designated SPA, a SSSI, a SAC and a RAMSAR site.

The SMP policy is HTL for the short, medium and long term. The main engineered defence system is the sea wall with added protection from Hurst Spit (a 1.6 km long shingle bank to the south-west maintained with nourishment) and the saltmarshes on the seaward side of the sea wall. These are being lost at a rate of 0.3 to 6 metres per year (NFDC, 2004; Colenutt, 2012). The sea wall was upgraded following the breaching of Hurst Spit and the sea wall in 1989 when 184 ha of reclaimed marshland (nature reserve) and 285 ha of agricultural land were flooded with seawater (Wadey, 2013).

3.3.1 Site characterisation

Landfilling at the site was carried out between 1962 and 1969, with a mixture of inert, industrial, commercial, and household waste. The landfill covers an area of 7.5 ha, with maximum surface elevations at ~ 4.1 m AOD (above Ordnance Datum). The volume of waste in the site is estimated to be approximately 160,000 m³. A site investigation reported by Marcus Hodges (1999) included trial pits which established waste depths between 1.9 and 3.7m with a mix of soil, plastic, wood, glass and rags. In some trial pits, a hydrocarbon odour was noticed and in others there was an odour of decomposing waste. Gas analysis did not find methane in any of the trial pits. Leachate was found in the waste material at varying depths across the landfill site, but an average leachate depth of 0.5 m was estimated; this equates to an average leachate level of ~ 0.5 to 0.75 m AOD. The volume of leachate in the site was estimated by Marcus Hodges (1999) to be 5,000 m³. Leachate analysis from trial pits and boreholes monitoring groundwater beneath the landfill showed very low concentrations of heavy metals, and low biochemical oxygen demand (BOD) (<20 mg/l), chemical oxygen demand (COD) (<125 mg/l) and ammonia (<100 mg/l).

The landfill is on marsh land that is typically underlain by a thin (~1m) alluvial clay deposit, overlying a sandy gravel aquifer between 4 and 6 m in thickness. Surrounding marsh land is at an elevation of ~ 0.3 m AOD. Marcus Hodges (1999) indicated that in places beneath the landfill the alluvial clay was absent meaning that the base of the site was sometimes as low as -1.5 m AOD and that the waste was in hydraulic continuity with the underlying sand and gravels.

4. DISCUSSION

4.1 Assessing the current risk

The standard UK approach to assessing potential harm to human health and the environment (HHE) is through a risk based source-pathway-receptor approach. This is the

methodology recommended in CIRIA C718. The potential source is the solid waste in the landfills, its leachate and any hazardous gases and vapours. Potential receptors include humans, the environment (e.g. ecological system / habitat), controlled waters (surface fresh waters, coastal waters and groundwater), the atmosphere and property. The main pathways relate to the release of leachate, that may be exacerbated by flooding, and the potential erosion of solid waste by the sea (e.g. see Figure 1). Other pathways would include the release of hazardous gases and vapours, that might occur in response to flooding.

4.1.1 Current impacts from the release of leachates

None of the three case study sites had any form of leachate control systems. Considering their age the sites can be considered to be at hydraulic equilibrium and any infiltration entering the sites from rainfall will generate leachate which will then migrate into surrounding surface and coastal waters and/or groundwater. Techniques to assess the impacts of water-borne contamination are well-established under legislation and guidance to implement the Water Framework Directive. A full hydrogeological assessment of each site was not undertaken as part of this study, but a number of points are noted.

Firstly, as described above, the leachate in all sites was relatively weak. Secondly, there were no reports from the regulator or local authorities that leachate currently emitting from any of the sites was having any adverse and/or measurable effect on the environment. At both the Spittles Lane and Wicor Cams landfills the main receptor of any leachate discharge is coastal or estuarine waters where large dilution factors come into play (discussed further in 4.2.1).

Although the leachate in the Wicor Cams landfill is considered to be relatively weak for a typical landfill leachate, it did contain levels of contamination that exceeds Environmental Quality Standards (EQS) for waters. URS (2013) compared the leachate quality with the EQS for saline waters. Significant exceedances were seen for chromium, iron, copper and zinc. In the case of chromium, for the highest detected concentration, a minimum 26 fold dilution would be needed to reduce concentrations to the relevant EQS standards. There are also significant exceedances for certain PAHs, including Benzo(g,h,i)perylene, the maximum concentration of which would need a dilution factor of at least 130 to bring concentrations down to EQS standards. Flood risk analysis at the site showed that at current sea levels, inundation of Wicor Cam landfill is limited even during extreme water level events, although seawater can enter the landfill matrix at high tide during storms.

Leachate from the Pennington landfill, which is isolated from coastal waters by the sea wall, potentially seeps into shallow groundwater, the saltwater lagoon to the south of the site, or into the surrounding grazing land. However, no leachate impacts have been reported. Even when the Keyhaven-Lymington sea wall was breached in 1989 (NRA 1990; Wadey, 2013), resulting in extensive flooding of the land surrounding Pennington landfill for a period of up to 3 weeks, the significant environmental damage to the marsh habitat was attributed to saline incursion, not leachate.

A useful measure of how well flushed and stabilised (in terms of the potential to release soluble contaminants) a waste is likely to be is the Liquid-Solid ratio (LS), which is the ratio of clean water that has passed through a unit dry mass of waste. The LS is derived from leaching experiments used in waste acceptance criteria column tests (ISO/TS 21268–3), but can usefully be applied to the field (see Supplementary Information). A LS value of 10 indicates very high degrees of flushing and clean-up, whereas values significantly less than 0.5 are typical of most modern “engineered landfills” (Nordic Council of Ministers, 2016, Yasutaka et al., 2017). The LS values calculated for the three case study sites is shown in Table 4, with the high calculated values supporting the evidence of relatively weak leachate reported above. Although a high LS for a site does not guarantee that the leachate will be weak (the test should not replace proper site investigation and/or an understanding of

potential geochemical interactions) it is useful tool to characterise historic landfills.

Table 4 Liquid-Solid (LS) ratios of case study sites.

Site	Average waste depth	Dry density*	Average Infiltration**	Assumed average minimum duration of infiltration	LS ratio
	m	t/m ³	m/yr	years	
Spittles Lane	1-3	0.8	0.325-0.45	40	5 - 22
Wicor Cams	1.5-5	0.8	0.325-0.45	35	3 -13
Pennington Marshes	2	0.8	0.325-0.45	47	9 -13

* Assumed values

** Based on average effective rainfall for region (EA, 2008)

4.1.2 Current impacts from the release of solid waste

Waste from Spittles Lane landfill is currently eroding into the sea: potentially as much as 6,000 m³ of waste has been eroded from the landfill, although not all of this will have reached the beach yet. There are also some wastes exposed around Wicor Cams landfill, due to limited erosion of the sea wall with possible release of waste. The landfill at Pennington is protected by the sea wall and there are currently no pathways for release of solid waste.

The Spittles Lane landfill has been subjected to a number of site investigations aimed at assessing its risk to HHE. More specifically concentrations of various solid waste contaminants were compared against “soil guideline values” (SGVs). SGVs are *guidelines on the level of long-term human exposure to individual chemicals in soil that, unless stated otherwise, are tolerable or pose a minimal risk to human health. They represent “trigger values” – indicators to a risk assessor that soil concentrations above this level may pose a possibility of significant harm to human health* (Environment Agency, 2009). Bennett (2007) indicated that the only contaminants of potential concern were poly-aromatic hydrocarbons (PAHs), nickel and arsenic for which the concentrations were close to SGVs for residential land-use with potential for plant uptake’ (for example, consumption of wild produce e.g. blackberries). The subsequent contaminated land assessment reports (e.g. WPA Consultants, 2010, 2013) indicated that lead concentrations were greater than SGVs in beach and landslip toe samples, and arsenic and nickel exceeded SGVs in some of the landslip toe samples. Polyaromatic hydrocarbons also exceeded guidance values in beach and toe samples. The 2013 report also identified lead as exceeding assessment criteria in soil samples from the toe, but not from the beach; benzo(a)pyrene and asbestos were found in both sites.

Pope et al. (2011) studied the concentration of metals in sediments and aquatic life in the vicinity of the Lyme Regis 2008 landslip, and attempted to link results to the presence of waste deposited on the beach from the landfill. Although elevated concentrations of certain metals in sediments and intertidal biota were found, the evidence that this was directly caused by the waste is weak; a geogenic source from the local eroding geology seems plausible.

While the impact of plastics in the marine environment is receiving considerable attention (e.g. Gregory, 2009; Erikson et al, 2013; Zalasiewicz et al, 2016; Alimba and Faggio, 2019), there are currently no methods for assessing the direct impact of other landfill waste streams entering the sea. However, a useful comparison can be made with established international guidelines that control the disposal of solid matter to sea. The London Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter, was updated by the London Protocol (2009). The London Convention and Protocol (LCP) allows, under certain conditions, dumping of materials such as dredged material, inert inorganic geological material and organic material of natural origin. The LCP does not cover the erosion of waste into the sea, but nevertheless creates a useful starting point for the consideration of the acceptability or otherwise of the direct impact of different waste streams entering the sea. For the types of wastes that may be dumped at sea, the LCP requires that Action Levels for a wide range of contaminants are set to help define the acceptability of dumping (MMO 2015). Samples below Action Level 1 (cAL1) are generally considered acceptable for disposal at sea, pending other considerations such as physical suitability for the disposal site and potential beneficial uses. Sediments above Action Level 2 (cAL2) are considered unacceptable for uncontrolled disposal at sea without special handling and containment.

A summary of the metals analysis of soil/ waste samples taken from Spittles Lane and Wicor Cams landfills are provided in Tables S1 and S2. These are also compared to the UK's cAL2 values. At Spittles Lane (Table S1) the maximum recorded concentration of all metals other than arsenic and chromium were higher than cAL2 levels. When average concentrations are considered it is only lead that exceeds the respective cAL2 level of 500 mg/kg. At Wicor Cams (Table S2), none of the average metal concentrations exceeded cAL2 values, although maximum recorded concentrations of copper, lead and zinc exceed the cAL2 levels. There are no cAL2 action level standards for PAHs, but URS (2013) states that PAH concentrations in borehole samples were elevated at "probable effect levels" (PEL) set in Canadian sediment quality guidelines (CCME, 2001). At Spittles Lane, Benzo(a)pyrene concentrations exceeded PEL in samples from the beach and landslip toe.

A further comparison (Tables S1 and S2) of the contamination levels found in the soil/waste samples of the case study sites can be made with the normal background concentrations (NBCs) of contaminants found in English soils (Johnson et al. 2012). A distinction is made between soils arising from different domains: e.g. urban and non-urban (principal) areas, and in areas underlain by mineralised bedrock that are known to be rich in the contaminant of concern.

Johnson et al. (2012) reports NBCs for six metals and Benzo[a]pyrene. The average metal concentrations of all waste samples from both Spittles Lane and Wicor Cams landfills was lower than the maximum NBC from a mineralised domain, although peak concentrations (other than for As and, at Wicor Cams, Ni) were well in excess of NBC values. The average benzo[a]pyrene concentration from Spittles Lane was likewise lower than its NBC from an urban area.

The above comparisons do not endorse release of waste to the marine environment, but form a starting point for the development of proper impact assessments for waste release, for which new science will need to be developed.

4.2 Impact of climate change and sea-level rise

4.2.1 Future impacts from the release of leachates

Spittles Lane landfill is located at an elevation of ~50 m AOD and is not at any risk of direct flooding from projected sea level rises. The results from bath-tub flood analysis of the Pennington Marshes and Wicor Cams landfill sites is shown in Figures 3 and 4 (method

described in Supplement Information).

By 2100 under an extreme high-end SLR (H++ scenario) major parts of Wicor Cams are at risk of flooding regularly (Figure 3). With high SLR and a 1 in 200 year extreme water event 20,000m² of Wicor Hard would be inundated, with sea levels during this event being approximately 2 metres above existing leachate levels in Wicor Hard. Assuming the inundation resulted in an increase in leachate levels of 2 metres in the whole of Wicor Hard (6.65ha) and a drainable porosity of 15%, then this could result in a maximum of ~20,000 m³ of additional leachate being created. As a worse case if it is assumed that the entire volume of leachate were released over one tidal cycle (in reality leachate would be released over several days to weeks) flow in the upper reaches of Portsmouth Harbour around Wicor Cams would result in a 100 fold dilution of the leachate. Based on concentrations in leachate measured in 2013 (URS, 2013), and assuming zero background concentration of a contaminant in Portsmouth Harbour, this level of dilution would be sufficient to reduce metal concentrations, and most organic contaminant levels (apart from benzo(g,h,i)perylene), below EQS threshold values for saline water (URS, 2013). It is likely that leachate concentrations would be further decreased by 2100 due to dilution of leachate by rainwater and inundation events in the interim period. We did not consider the effect of saline waters which may increase the release of metal contaminants from waste compared to freshwater (Brand, 2017).

Analysis by Kebede (2009) showed that the Pennington sea wall is currently at risk of breaching in a 1 in 50 year storm, rising to 1 in 5 years by 2055 with medium sea level rise. Assuming that background leachate levels are at 0.75 m AOD, then a spring tide in 2050 under a medium sea level rise scenario could reach 1.13 m AOD. With a landfill surface area of 7.5 ha and a drainable porosity for the waste of 15%, potentially 4,300 m³ of leachate could be produced from Pennington landfill. Full inundation of the site, which could occur in 2100 during a 1 in 50 year storm under a high sea level rise scenario (Figure 4), could result in ~15,000 m³ of leachate being created.

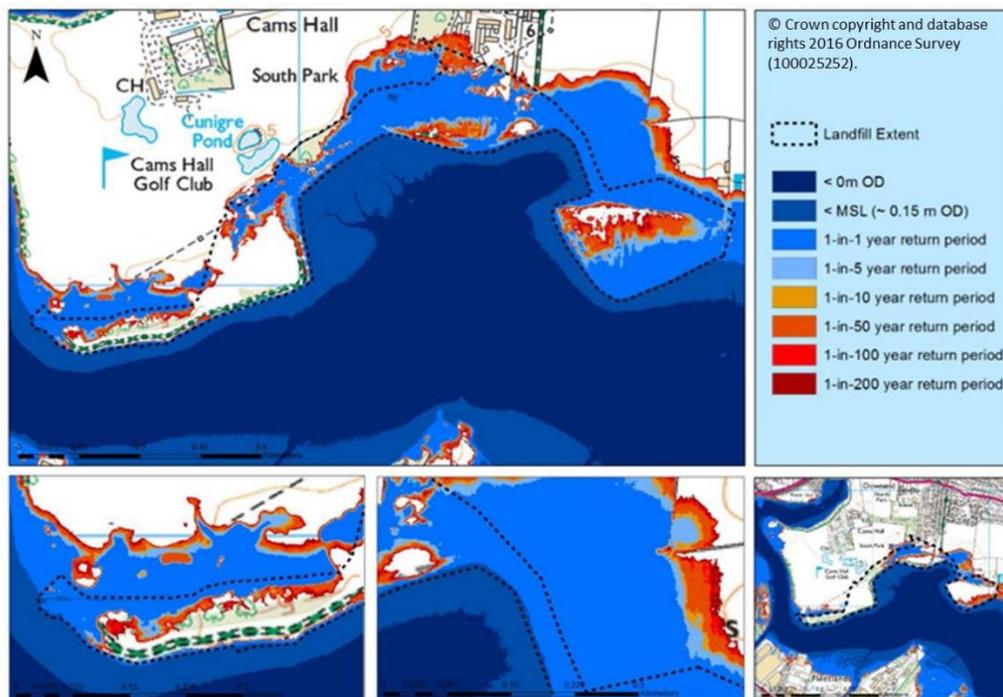


Figure 3. Potential flood zones (by return period) at Wicor Cams under the H++ SLR scenario in 2100 in the absence of defences. (MSL = mean sea level)

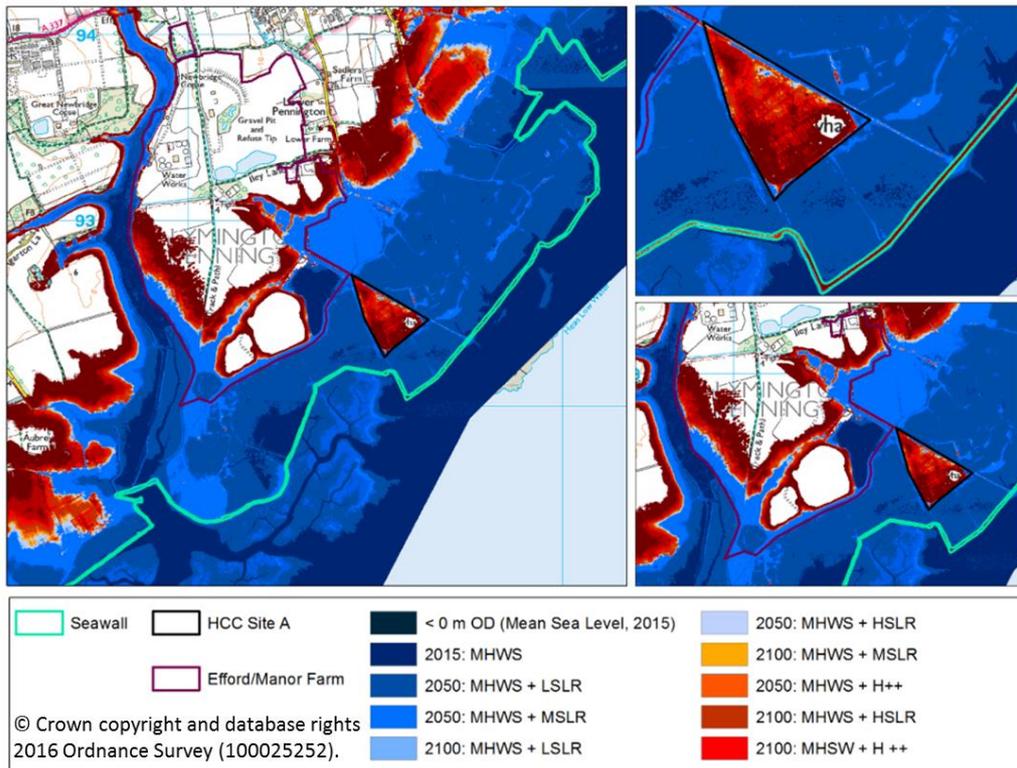


Figure 4. Indicative 1-in-50 year flood zones in the current day, 2050 and 2100 under projected low (LSLR), medium (MSLR), high (HSLR) and extreme high (H++) sea level rise in the absence of defences at Pennington. (MHWS = mean high water springs)

4.2.2 Future impacts from the release of solid waste

Once placed in a landfill, the longevity of solid wastes should be considered to be over geological timescales (Brunner, 2013); potential impacts resulting from the release of toxic solid wastes (e.g. asbestos, and lead and other metals (for example in electronic waste such as cathode ray tubes)) may persist for millenia or longer.

Extreme weather events, which include periods of increased rainfall, are predicted to become widely more frequent in response to global warming (Murphy et al. 2009). This, combined with increased annual rainfall, may result in increased pore water pressures in the Spittles Lane landfill and surrounding area, increasing the possibility that the landslip complex will be more active.

Application of Walkden and Dickson (2008) (Supplementary Information) allows an assessment of increases in rates of cliff erosion at Spittles Lane in response to SLR. Table 5 summarises a range of indicative erosion rates under different SLR scenarios and estimates of different historical erosion rates.

The maximum distance between the front of the Spittles Lane landfill at the cliff face and its landward extent is approximately 130 metres and it is almost certain that without intervention the whole of the landfill will be eroded into the sea. Assuming there is no change in the historical rates of erosion of 0.3 m/yr, the whole landfill is predicted to have been undercut with waste entering the tumble zone by the year 2450. Using higher historical rates (Table 5) of 0.8 m/yr and high SLR projection then much faster erosion rates are predicted. Consequently, the timeframe over which the whole landfill will be undercut and enter the tumble zone is estimated to be 50 to 500 years. Erosion will be episodic in nature and could include large failures such as the 2008 landslide.

Table 5: Indicative rates of erosion under sea-level rise at Spittles Lane, Lyme Regis.

Historic Rate of Erosion (m/yr)	Source	Sea-Level Rise Scenario							
		Low		Medium		High		H++	
		Potential Future Rate of Erosion (m/yr)*							
		2050	2100	2050	2100	2050	2100	2050	2100
0.3	Brunsdon (1996)	0.3	0.4	0.5	0.5	0.6	0.7	0.8	1.0
0.8	HPR (2000)	0.7	1.0	1.2	1.3	1.6	1.8	2.2	2.6
3	Brunsdon (1996)	2.6	3.8	4.6	4.9	5.9	6.9	8.3	9.7

*Based on Walkden and Dickson (2008)

At Wicor Cams the Environment Agency identifies three erosional buffer zones for the short (2005–2025), medium (2025–2055) and long (2055–2105) term in the area. These are 1.8-8 m, 10-20m and 40 m + respectively. Lower and upper potential rates of erosion (0.2 and 0.4 m/year) were derived from these projections. However, there was no evidence of chronic erosion during the site visit and topographic profile data from 2007 to 2011 from the Channel Coastal Observatory also showed no significant change suggesting the shoreline would appear to be stable or eroding slowly. The minimum rate (0.2 m/yr) derived from the Environment Agency erosion zones appears to be much higher than the current rate of change at this site.

At Pennington, rapid erosion of the saltmarshes in front of the sea wall is already occurring with total loss on some frontages. Loss of the protective saltmarsh may result in increased wave attack on the sea wall resulting in damage and increasing the risk that the wall will be breached (Kebede, 2009) potentially leading to erosion of the landfill.

4.2.3 Future impacts from the release of gases

No landfill gas emissions have been reported for any of the three case study sites, indicating that the organic waste in the sites are largely stabilised. However, inundation in the site may cause activation of organic material and/or release of toxic gases as localised areas of hazardous materials are flooded.

4.3 Landfill management options and SMPs

4.3.1 Lyme Regis

At the Spittles Lane landfill the SMP policy is NAI. This will intentionally allow the slope on which the landfill is located to continue to fail. The slope failure mechanisms are complex and difficult to predict, but the history of slope failures indicates that occasional major events dominate. Over the next 100 years or so it is highly likely that the whole landfill will have been entrained in the landslip or have reached the beach.

The nature and age of wastes taken at the site suggest that the quality of any leachate now being generated will be relatively dilute and will have minimal impact on the marine environment, and this is certainly less an issue than the direct release of solid waste. If the landfill was not affected by an SSSI designation a HTL policy to stop the ongoing erosion of the landfill into the sea could be considered. However, the designation constrains SMPs in this instance, so attention turns to alternative approaches to manage the landfill.

The current management plan enacted by the local authority is to identify and remove any wastes as and when material reaches the beach. An alternative strategy would be to remove the landfill. The volumes of material involved are not excessive, but there would be

major technical challenges in removing material from the unstable slope without triggering further landslides.

4.3.2 Wicor Cams

At Wicor Cams there is a HTL policy, but this is not currently under-pinned by any detailed plans for new defences, reflecting a lack of funding. The historical sea wall that runs along most of the frontage has deteriorated over time, with evidence of erosion along some sections. However, because the site is in the upper reaches of Portsmouth Harbour where wave energy is limited, erosion appears to have been minimal and is less than the values suggested by the EA buffer zones. The ad hoc placement of blocks of concrete and stone does appear to have had some limited effect in restricting erosion. Notwithstanding these limited rates of erosion, without further works it is predicted that in the long term there will be erosion of the waste into the marine environment. Current rates of erosion could double if annual SLR increased to the maximum rate of ~ 8mm/year (Table 3). However, it would take many centuries of uncontrolled erosion for the whole landfill to be removed. There is currently little justification for major engineering works to protect the whole of the coastal frontage along Wicor Cams, but there are some shorter sections where a more targeted approach is appropriate. Monitoring and periodic assessment of any erosional change is recommended.

With an estimated 1 million m³ of waste material in place along a frontage exceeding 2km, removal of the landfill is currently not an economically viable option to eliminate the pollution source. The cost of landfill tax in the UK (standard rate = £91.35/tonne for year from April 2019) is a major cost impediment to any such scheme (see Supplementary Information). A landfill mining/remediation strategy that removed the fractions of waste which contained an unacceptable hazard to the marine environment and left behind materials which could be allowed to erode is worthy of further exploration. However, in the absence of any methodology to assess the impact of solid waste eroding into the marine environment, there is no technical basis to support this approach, nor alter the SMP HTL policy.

4.3.3 Pennington

Pennington is particularly susceptible to coastal flooding and highly vulnerable to increase in water levels and wave erosion, including the dramatic loss of saltmarsh. Hurst Spit and the Isle of Wight provide a shield for the shoreline at the study region, with wave climate predominantly controlled by south-westerly winds and locally-generated waves (Wadey, 2013). In addition, the ~1km earth and grassed embankment (between the lee of Hurst Spit and Keyhaven) and the 8.1 km sea wall are the first lines of hard coastal defence structures against flooding from the sea. It was designed considering that it was fronted by a certain width of saltmarshes when it was rebuilt and raised in height following breaching due to the 1989/1990 winter storms (NRA 1990). Its crest was raised by up to 0.4–0.5m (the highest being raised at the northeast end to accommodate wave run-up) to provide approximately a 25 year standard of protection (Wadey, 2013). The top elevation of the sea wall varies in different sections along its length from about 2.4 to 3 m AOD.

With the loss of saltmarsh on the seaward side of the sea wall, together with the predicted increase in average sea levels, a HTL policy will require costly ongoing upgrade of the wall into the future. This needs to consider the sea levels in Table 3, plus increasing wave heights as water depths increase and overtopping is likely to increase.

A change to NAI or MR policies in this locality has significant potential advantages, but is being partly constrained by the presence of the landfill site. The main longterm problem with the landfill relates to erosion rather than flooding. Consequently, removal (and/or landfill mining) of the waste material is seen as a viable and cost-effective solution (see Supplementary Information) within the context of potential savings that could be made to ongoing sea defence works. If the landfill was removed a case could be made to relocate

the sea wall further inland, which would both result in a shorter length to maintain (~ 5.5 km saving) and lower height requirements due to increasing elevation moving inland. A move towards a MR policy would require a rebalancing of the existing habitats in the locality which would need to be carefully managed and is likely to be a major cost component of any scheme. The inter-tidal habitats in the region, particularly the saltmarsh, have been considered a key aspect of the defence system. However, Gardiner et al. (2007) indicates that under a MR policy the predominant habitat will be mudflats as the elevation of new intertidal areas is too low to be colonised by saltmarsh plants. Additionally, designated coastal grazing marsh and saline lagoons in the area, which should be maintained to comply with the Habitats Directive, will be lost. Relocation is an option as these are largely artificially managed habitats, but opportunities are restricted within the case study site due to competition from urban and industrial land use.

5. CONCLUSIONS

This paper explores the issues raised by erosion and flooding of coastal landfills based on detailed case studies of three contrasting landfills on the south coast of England, including varying sea-level rise projections. The findings are transferable and have relevance to coastal landfills around the world.

For the three case study sites considered, future flooding of the landfills was assessed as being a limited problem in comparison to erosion partly because leachate is already dilute and in the case of Pennington and Wicor Cams landfills, dilution by tidal waters would be significant. The application of a simple Liquid-Solid ratio calculation to coastal landfills is a potential tool that provides an indication of leachate quality and how well the landfill has previously been flushed. However, not all historic landfills in coastal or estuarine environments may contain leachate that is as dilute as in these two case study sites.

The longevity of solid waste in landfills is geological and the hazards associated with certain solid wastes may not diminish over time unlike leachate strength. This contributes to a significant conclusion of the study that the potential erosion of old landfills in coastal zones is more significant and a greater problem to solve than the issue of flooding, which will worsen with sea-level rise. While erosion is an episodic hazard, as shown at Spittles Lane any landfill that is allowed to erode will ultimately be completely removed releasing all its contents to the environment. The assessment of waterborne pollution is established under the Water Framework Directive, however protocols for assessing solid waste eroding from landfills need to be developed.

There are assessment protocols for contaminated soils deposited to sea (London Protocol) and there is considerable work by marine scientists considering the impact of plastics in the marine environment, but this detailed work needs to be extended to the full range of wastes within landfill. The development of risk assessments for solid waste will require improved methods to sample and characterise waste in landfills in a way that relates to the potential impact of different categories of waste if released into the marine environment.

The costs of relocating landfill wastes away from coastal zones to less vulnerable locations are potentially high, especially in situations where landfill tax is a significant component. However, relocation of two of the landfills studied was considered worthy of further consideration, although significant funding would be required. Removal of Spittles Lane landfill is the only solution to prevent the remaining $42,000\text{ m}^3$ of waste eroding over the next 50 to 500 years. The costs associated with removal of the $160,000\text{ m}^3$ of waste in the Pennington Marshes landfill might be justified by savings achieved by relocating existing sea defences inland. Removal of the Wicor Cams landfill could not be justified. As an alternative to protection, there may be potential to adopt landfill mining or remediation leaving some

fractions of waste *in situ*, but the science-based evidence for this approach is not yet available. In conclusion, managing coastal landfills over the next century (and beyond) poses a significant challenge to SMPs and our scientific tools to analyse these problems require significant enhancement.

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Supplementary Information for **Future challenges of coastal landfills exacerbated by sea level rise**

Data sources consulted for the case study sites

Primary information sources consulted included: historical and contemporary Ordnance Survey maps from the 1870s to date (Digimap); Aerial photographs (various sources); Digital Elevation Mapping (DEM) sourced from the Environment Agency; and area (e.g. SSSI) and priority habitat (e.g. mudflats) designations, sourced from the Department of Food and Rural Affairs (Defra). Literature sources included published journal papers, site investigation, consultant and contaminated land assessment reports, land ownership records, planning permissions, borehole logs and geological/hydrogeological maps and SMPs. Where relevant, site specific literature sources are referenced in the results section.

Detailed site maps



Figure S1: Location of Spittles Lane Landfill, near Lyme Regis

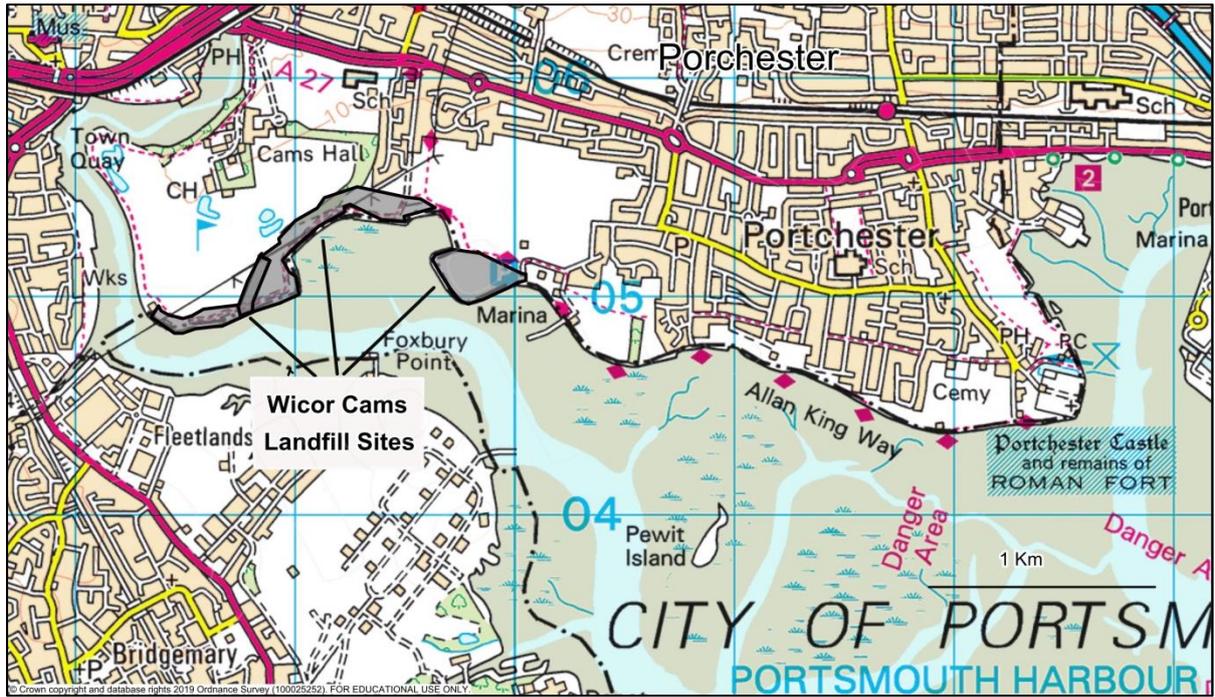


Figure S2. Location of Wicor Cams landfills, near Fareham

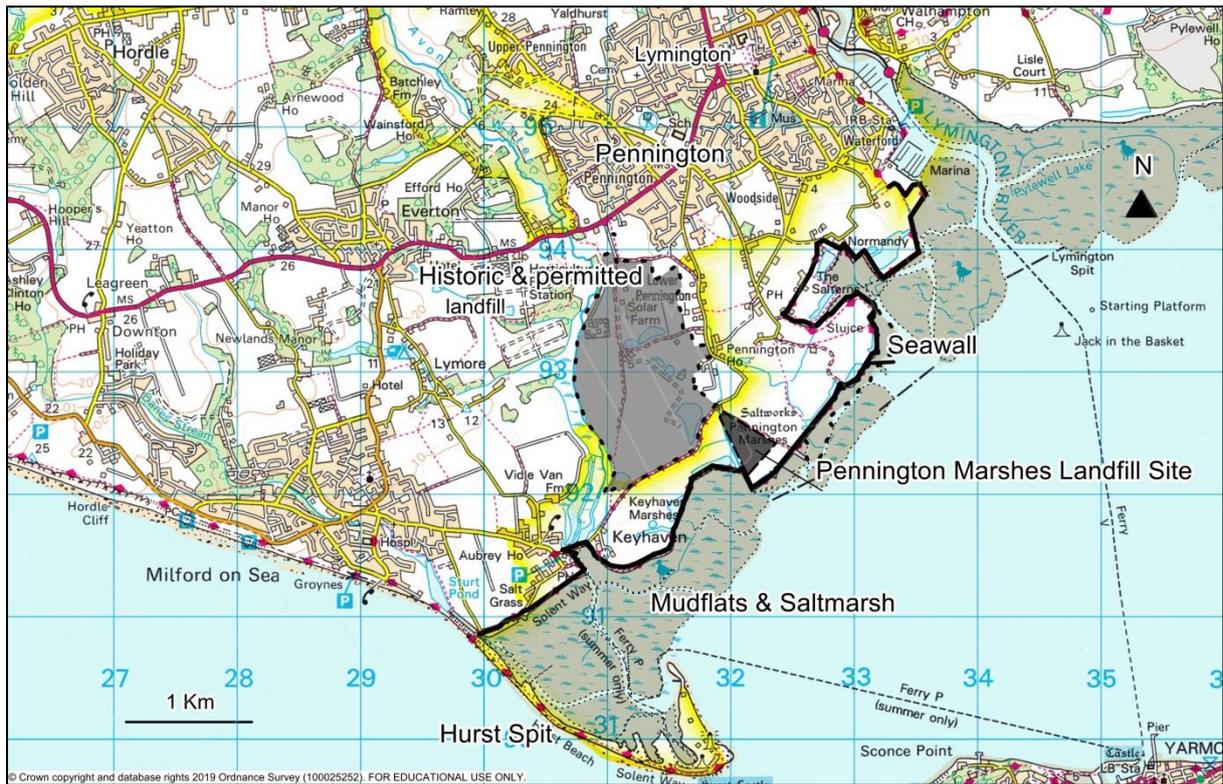


Figure S3. Location of Pennington Marshes landfill, near Lymington. Note the large historic and permitted landfills to the north of this site.

Climate change and sea-level rise scenarios

The UKCIP02 climate change scenarios (Hulme et al. 2002) give estimates of future potential change to mean precipitation and temperature for 50km x 50 km grid cells covering the UK. Winter precipitation is projected to increase across the UK, both in the short- and long-term, with the biggest relative changes in the south and east. By the 2080s, winter precipitation on the West Dorset coast is projected to increase by 10–15% under the Low Emissions scenario and 20–25% under the High Emissions scenario. Extreme winter precipitation will also become more frequent. For example, by the 2080s, winter daily precipitation intensities with a 2-year return period experienced on the West Dorset coast could become 10–15% heavier under the High Emissions scenario. The UKCP09 scenarios (Murphy et al., 2009) show similar trends. This will result in increased rainfall induced leachate production for all sites.

Sea levels have risen in the study area based on analysis of tide gauges at about 1.2 to 1.8 mm/yr over the last few decades (Haigh et al., 2009). Lowe et al (2009) provides low, medium and high sea-level rise projections around the UK coast to 2100, and a high H++ scenario which is plausible, but unlikely, and hence worth considering in this kind of analysis. These scenarios were adapted for the study area and are summarised in Table 3. By 2100, sea-level rise is estimated to rise between 0.2 and 0.75 metres compared to present water levels under the low to high scenarios. Under the H++ conditions, a 1.5 metres rise is considered as a plausible high-end scenario.

Coastal erosion (Pennington Marshes and Wicor Cams sites)

The analysis of potential coastal erosion used publicly available data as well as values and observations in published literature (see above). Erosion was assessed using the EA predictions for the short, medium and long term, and compared to evidence from Ordnance Survey mapping over the last 100 years together with topographic profiles of the landfill site and fronting shoreline. To assess the potential for the release of waste from the landfill site over time, lower and higher rates of erosion were calculated.

Cliff Erosion (Spittles Lane Landfill site)

One impact of SLR is an increase in sea cliff erosion rates, which is applicable to our case study site at Lyme Regis. A simple model of soft-cliff erosion is used to examine the impact of rising sea levels on the rate of erosion of the landfill site. Walkden and Dickson (2008) (equation 1) gives a relationship between historic and future rates of erosion (R_1 , R_2), and historic and future rates of SLR (S_1 , S_2). Historic rates of erosion are taken from the literature.

$$R_2 = \sqrt{R_1 \left(\frac{S_1}{S_2} \right)} \quad (1)$$

Brunsdon (1996) quotes average erosion rates for the basal Lias of between 0.3 and 3.0 m per year. A report by HPR Ltd (2000) reported in Bennett (2007) quotes recession rates of 0.5, 0.8 and 1.3 m per year for areas of East Cliff prior to the construction of the defences.

There are, however, a number of limitations to the application of this equation. Firstly, the fronting beach to the cliffs must be assumed to be relatively narrow and volumetrically small. Secondly, the final 'future' rate of erosion predicted does not represent the actual (transient) rate that will occur, but rather the new equilibrium rate that the system is adapting towards. At the Spittles/Black Ven complex, erosion is episodic and long periods of relative stability may be punctuated by short periods of high instability resulting in landslides and cliff failure. This is expected to continue into the future.

Flood analysis (Pennington Marshes and Wicor Cams sites)

The exposure of the Pennington Marshes and Wicor Cams landfill sites to potential flooding under different still water level scenarios was assessed using a simple bath-tub flood analysis. In the bath-tub method, areas which lie below the current and projected still water levels will be flooded if they are hydraulically connected to the source of flooding. The topographic data (LiDAR DEM) was re-classified in ArcGIS to indicate the areas with an elevation below the still water levels predicted for each time-slice under low, medium, high and extreme high (H++) sea-level rise. These were then assessed for hydraulic connectivity and edited accordingly to remove any areas which were incorrectly classified.

Table S1. Organic and inorganic contaminants in waste/soil samples* from Spittles Lane Landfill, near Lyme Regis, Dorset

	As mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Ni mg/kg	Pb mg/kg	Zn mg/kg	PAH µg/kg	Benzo[a]pyrene µg/kg
Average	14.8	3.4	42.0	213.1	104.5	613.7	509.3	26,400	1,500
STD	12.4	7.8	26.7	310.6	141.6	697.5	392.2	30,700	2,200
Max	62	40	129	1300	790	3351	1500	9,500	11,400
UK action level 2 ¹	100	5	400	400	200	500	800		
Canadian PEL guidelines ²	41.6	4.2	160	108	42.8	112	271	10,000	763
NBC Principal ³	32	1	n/a	62	42	180			500
NBC mineralised ³	290	17	n/a	340	230	2400			n/a
NBC urban ³	437	2.1	n/a	190		820			1800

* average values based on samples taken: March 2009; June 2009; Feb 2010; May 2012 and Sep 2013. (WPA Consultants Ltd, 2010 and 2013).

¹ MMO (2015)

² CCME, (2001)

³ Normal background (soil) concentration. Johnson et al. (2012).

Table S2. Organic and inorganic contaminants in waste/soil samples from Wicor Cams landfill, Fareham

	As mg/kg	Cd mg/kg	Cr mg/kg	Cu mg/kg	Ni mg/kg	Pb ¹ mg/kg	Zn mg/kg	PAH µg/kg	Total petroleum hydrocarbons (TPH) ² µg/kg
Average	12.9	0.7	23.2	104.9	21.1	173.6	223.1	31,512	369,552
STD	7.3	1.2	8.8	209.8	10.4	171.7	282.1	62,608	454,791
Max	38.2	4.26	45	923	47.6	677	1020	255,000	1,480,000
UK action level 2 ³	100	5	400	400	200	500	800	n/a	n/a
Canadian PEL guidelines ⁴	41.6	4.2	160	108	42.8	112	271	10,000	n/a
NBC Principal ⁵	32	1	n/a	62	42	180	n/a	n/a	n/a
NBC mineralised ⁵	290	17	n/a	340	230	2400	n/a	n/a	n/a
NBC urban ⁵	437	2.1	n/a	190		820	n/a	n/a	n/a

¹ Excluding one anomalous value of 24,300 mg/kg; n/a = not available

² Total Aliphatics & Aromatic hydrocarbons:C5-35

³ MMO (2015)

⁴ CCME, 2001

⁵ Normal background (soil) concentration. Johnson et al. (2012).

Removal of waste from Wicor Cams and Pennington Marshes landfills

The SMP policy for the areas including Wicor Cams and Pennington Marshes landfills is HTL for the next three epochs. Nevertheless, the costs to remove the site were estimated. Costs included excavation, transport and disposal to an alternative landfill, health and safety and environmental control measures and landfill tax. Costs for characterisation of the waste prior to excavation are included. There are many uncertainties with the cost analysis, not least because the values chosen against the various categories are mostly estimates and are not based on a detailed analysis of costs. However, the analysis does give an indication of the magnitude of potential costs. Planning permission would be required to remove waste from the landfill, and this would include a consideration of the impact of lorry and other environmental nuisances on the local community. The Environment Agency would also need to issue a permit to cover the operation.

Total removal of the Wicor Cams Tip landfill sites to an unspecified landfill, assumed to be within 80 km driving distance. The disposal of the waste to landfill is currently liable to landfill tax charges, and waste characterisation would need to be made to determine whether the waste is subject to the full tax charges (£91.35.40/tonne – rate applicable at time of costing (2019)) or at the lower rate for inactive waste (£2.90/tonne). This could cost in the region of £149M if all removed materials that were landfilled attracted the top rate of landfill tax (Table S3). In this scenario over two thirds of the costs (73%) are accounted for by landfill tax. If it is assumed that only 30% of the removed and re-landfilled material attracted the higher rate of landfill tax, the total remediation cost is halved to around £75M. Given the large cost of excavation and disposal, total removal of the landfills from Wicor Cams is not considered financially viable.

An estimate of the cost of removing the waste from Pennington Marshes landfill was made, again assuming that the waste will be transported to an unspecified landfill site 80 km away. The total cost for removal of the Pennington Marshes landfill would be approximately £23M if all removed materials attracted the top rate of landfill tax (Table S3). In this scenario, landfill tax would amount to 77% of the total costs. If it is assumed that only 30% of the removed and re-landfilled material attracted the higher rate of landfill tax, the total remediation cost is halved to around £11M.

Excavation and in-situ treatment of the landfilled waste with recovery of some of the material for use on site or in shoreline defences may be possible, but a comprehensive waste characterisation would be needed to determine the feasibility and cost of this approach. An end of waste protocol would also probably be required to cover the nature of recovered materials.

Table S3. Potential costs to remove waste from Wicor Cams and Pennington landfills to an alternative landfill at 80 km driving distance.

Landfill	Wicor Cams		Pennington	
% attracting higher band of landfill tax	30	100	30	100
Estimated Volume of Waste (m ³)	1,000,000	1,000,000	160,000	160,000
Estimated mass of waste (Bulk density assumed 1.2t/m ³)	1,200,000	1,200,000	192,000	192,000
Excavation Costs (£2/tonne)	2,400,000	2,400,000	384,000	384,000
H&S environmental control measures (£5/tonne)	6,000,000	6,000,000	960,000	960,000
Transport Costs (£6/tonne)	7,274,000	7,274,000	1,164,000	1,164,000
Disposal Costs (£15/tonne)	24,000,000	24,000,000	2,880,000	2,880,000
Landfill Tax (£91.35/tonne)*	32,886,000	109,620,000	5,262,000	17,539,000
Landfill tax - lower rate (£2.90/tonne)*	2,436,000	0	390,000	0
Total costs (£)	74,996,000	149,294,000	11,040,000	22,927,000

* rate applicable at time of costing (2019)

An approach for calculating LS ratios of historic landfills

The liquid to solid ratio (LS) is a measure used in waste acceptance leaching tests that is the ratio of the amount of liquid (normally deionised water) that has been brought into contact with a dry mass of solid waste. LS values are normally reported as volume of water in litres divided by dry mass in kg. CEN/EN 14405 is the European standard for the leaching behaviour of inorganic and non-volatile organic substances in up-flow percolation tests through granular wastes. Waste in a column (typically ~0.3 m high and with a diameter of 0.05 to 0.1m) is pre-saturated with water before an up-flow leaching test is undertaken. Fractions of eluate are collected at various LS ratios between LS = 0.1 to 10 l/kg over a period of approximately 21 days. There are no imposed controls over the pH of the eluate. Concentrations of dissolved substances in the eluates are reported against LS.

In principle an average LS for a historic landfill can be calculated based on the total dry mass of waste in the landfill and the volume of leachate that has been produced over its whole history. A number of major assumptions and estimation of parameters is required to derive a landfill's LS, and there is no direct equivalence with a formal CEN leaching test. Nevertheless an approximate value of a landfill's LS is likely to provide a useful indication of how well a landfill has been flushed. Of critical importance are the average depth of the landfill (a shallow landfill will generate higher LS values in comparison to a deep landfill) and the average infiltration rate. Landfill depth will vary across the site, and so the LS will vary depending on location. Waste depth was assumed for each study site based on available data.

The dry mass of waste in a landfill can be estimated from the volume of landfilled waste and an average dry density. The average density of waste in a landfill will be dependent on many factors including waste composition, compaction and depth of burial (e.g. Beaven et al 2011). Landfill bulk densities have been reported to vary from between ~0.6 and 1.5 tonnes/m³ for landfills containing municipal solid wastes and up to ~1.8 tonnes/m³ for inert landfills. The equivalent dry density range is between 0.4 and 1.2 tonnes/m³ for MSW containing landfills and 1.5 tonnes/m³ for inert sites. Lack of compaction, shallow landfill depths, and the preponderance of low density waste components (e.g. plastics) favour low waste densities. Waste compaction, increased landfill depths (especially over 20 metres) and high proportions of soil-like materials lead to higher waste densities. Although all three of the case study sites were shallow (<5 metres on average) the evidence is that they all contained a reasonable proportion of soil-like materials and consequently average bulk and dry densities of 1.2 and 0.8 tonnes/m³ are assumed for all sites.

There is no direct measurement of the volume of leachate that has been “leached” from each site, so an estimation is based on average regional effective rainfall in the location of the case study sites. Environment Agency (2008) contains maps of average summer and winter effective rainfall for England and Wales. All three case study sites are in areas that received between 201 and 300mm of winter effective rainfall and between 126 and 150 mm of summer effective rainfall. Assuming that regional effective rainfall is a reasonable proxy for infiltration at each of the case-study sites, then the volume of leachate production at each site can be based on between ~325 and 450 mm infiltration per year.

$$LS = \frac{\text{Volume of leachate}}{\text{Dry mass of solid}} = \frac{\text{Annual Infiltration} \times \text{Years} \times \text{Area}}{\text{Average waste depth} \times \text{Dry density} \times \text{Area}} = \frac{\text{Annual Infiltration} \times \text{Years}}{\text{Average waste depth} \times \text{Dry density}}$$

Table S3. LS ratios of case study sites (same as Table 4)

Site	Average waste depth	Dry density*	Average Infiltration**	Assumed average minimum duration of infiltration	LS ratio
	m	t/m ³	m/yr	years	
Spittles Lane	1-3	0.8	0.325-0.45	40	5 - 22
Wicor Cams	1.5-5	0.8	0.325-0.45	35	3 - 13
Pennington Marshes	2	0.8	0.325-0.45	47	9 - 13

* Assumed values

** Based on average effective rainfall for region (EA, 2008).

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