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Risk screening assessment for ranking historic coastal landfills by pollution risk

James H. Brand and Kate L. Spencer

Abstract: Globally there are significant numbers of historic landfills, and in England alone there are over 1200 in low-lying coastal areas. Approximately one-third of these historic coastal landfills are near designated ecological sites, and without intervention, 10% are expected to start eroding within 40 years. Indeed, some sites are already eroding and releasing waste, and erosion is likely to become more common with the anticipated effects of climate change. Mitigating the pollution risk from all historic coastal landfills under threat of erosion would be prohibitively expensive; consequently, it is necessary to understand which sites pose the greatest pollution risk to prioritise management resources. This paper proposes a new risk screening assessment that can support coastal managers in identifying which historic coastal landfills pose the greatest pollution risk at a national scale for minimal cost using existing datasets. The proposed method determines an *overall risk index* for each site by considering the risk of pollution from eroding historic coastal landfills in two stages: the first stage assesses the risk of waste being released (*waste release index*), and the second assesses the risk to various receptors (*pollution index*). The highest risk sites can then be prioritised for further investigation or remediation.

Key words: estuarine and coastal management, risk assessment, historic coastal landfills, coastal vulnerability index, contaminated land management, flood defence management.

1. Introduction

Historically it was common practise to landfill domestic, commercial, and industrial waste in areas considered of limited economic value due to the risk of flooding, such as low-lying estuarine and coastal locations. For example, in England there are over 1200 historic landfills in coastal and estuarine locations that are low-lying and have a high risk of sea flooding (i.e., ≥0.5% annual probability) and (or) erosion if not adequately defended; without intervention, 10% are anticipated to start eroding by 2055 (Brand et al. 2018). The likelihood of these historic coastal landfill sites flooding or eroding is increasing due to climate change effects, such as increased sea level and more frequent extreme weather events, and this may have consequences for pollutant release. Inundation would increase leachate production, but significant dilution in open waters would minimise risk. Solid waste is usually fully contained and isolated from the marine environment by capping materials, and is often protected by flood defences (Brand et al. 2018); however, historic landfills and their defences are increasingly at risk of breaching, because inundation will increase the probability of failure through erosion, piping or excessive seepage (Bujis et al. 2007).

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Consequently, historic waste materials may be released into the coastal zone, and this has already occurred in some locations (Pope et al. 2011). Historic waste may include a wide range of materials that are physically harmful to ecological and public health, such as asbestos and plastics, as well as pathogens, and inorganic and organic contaminants that significantly exceed environmental quality guidelines[1]. This poses a significant challenge to coastal managers as mitigating the risk from all historic coastal landfills is likely to be prohibitively expensive (Weber et al. 2011; Cooper et al. 2013). Therefore, it is essential to prioritise regional and national expenditure by mitigating those sites that pose the greatest pollution risk (Brand et al. 2018).

Risk is typically considered as a function of the probability of something happening and its consequences (Wamsley 2015). There are many factors that may influence the probability that contaminated materials from historic coastal landfill sites are released, including wave exposure, the condition and design standard of any flood defences present, and local coastal erosion rates (Alaska Department of Environmental Conservation 2015). The consequences of pollution occurring are dependent on the vulnerability of the receptors (Wamsley 2015), which can be considered as the probability that the receptors will be affected by hazards or drivers, and is often considered in terms of a dose–response relationship (Gormley et al. 2011). Therefore, the consequences of contaminated materials being released will depend upon the quantity of materials released and their contaminant loads, contaminant bioavailability and mobility, dilution by the receiving waters, and receptor sensitivity to those contaminants. In turn, the quantity of materials released will depend on many of the same factors as the probability of contaminated material release, plus the size of the landfill (i.e., quantity of waste), whether it is divided into structurally stable cells, the mechanical properties of the waste (e.g., waste cohesion), the shape of the landfill (i.e., the proportion of it adjacent to the coast), and how quickly any breach can be repaired (Cooper et al. 2013; Alaska Department of Environmental Conservation 2015).

Combining such diverse data types into a readily understood form that indicates their combined effect can be achieved using index and indicator methods (Ramieri et al. 2011). However, many of these data are not readily available and would require impracticable levels of resources to obtain in countries with large numbers of landfills. Where detailed data are not readily available to assess risk at local, regional, or national scales Rosendahl Appelquist and Balstrøm (2014) propose a three step approach to assessment, where steps 1 and 2 are used for regional or national scale assessments and step 3 is only used for local scale assessments:

Step 1. High level initial screening using remote sensing and existing data to gain a cost-efficient, relatively low accuracy overview of the risk.
Step 2. Field verification of the data used in step 1.
Step 3. Systematic and detailed field investigations for high accuracy, local level assessments of risk hot-spots identified in steps 1 and 2.

This approach has the advantage of reducing expenditure on site investigations and providing a method to prioritise resources when there are multiple sites to manage. It has the disadvantage that existing data may not highlight factors that increase risk (e.g., records may not show that a site has already started to erode).

There have been a number of attempts to apply the index and indictor approaches to the management of landfill sites both on the coast and inland, none of which have been widely

[1]J.H. Brand and K.L. Spencer. In review. Potential contamination of the coastal zone by eroding historic landfills.

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adopted. These typically only consider the risk of pollution when the waste is fully encapsulated, do not consider inundation, and focus on the risk from leachates and gases (e.g., Kumar and Alappat 2005; Sharma et al. 2008; Singh et al. 2009; Okaneya et al. 2013). Where erosion of waste as a pollutant pathway has been considered, methods are too location specific for wide application (Laner et al. 2008, 2009; Neuhold and Nachtnebel 2011; Neuhold 2013; Alaska Department of Environmental Conservation 2015). Hence, a new region-specific method is required for assessing coastal landfills that can be applied in both England and physically similar temperate coastal environments.

The overall aim of this research was to develop a high-level risk screening assessment methodology, focused on the risk to the intertidal zone and tidal waters from eroding historic coastal landfills, which will support coastal managers in allocating limited resources to addressing the sites that pose the greatest pollution risk. The presented risk screening assessment approach has considered the risk of pollution from eroding historic coastal landfills in two stages: the first stage assesses the risk of waste being released, and the second assesses the risk to various receptors.

2. Developing the risk assessment

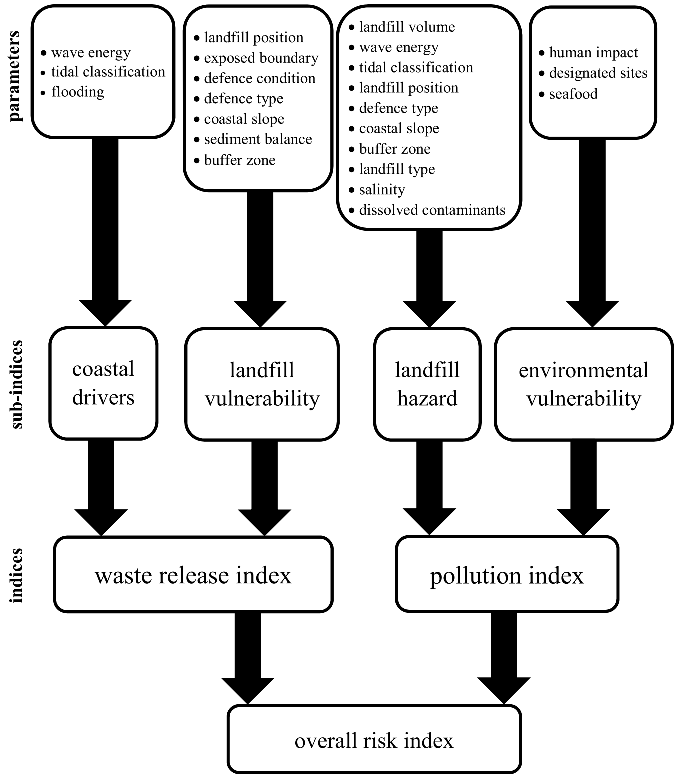
Coastal vulnerability and landfill screening assessments typically provide relative, not absolute, indications of risk by considering parameters that represent the vulnerability of receptors to specific hazards or drivers using the best available datasets (Sayers et al. 2003; Rygel et al. 2006; Kumar et al. 2010; Wamsley 2015). Therefore, to assess the risk of waste being released (hereinafter referred to as the waste release index) this research has identified parameters to represent coastal drivers (e.g., wave action) and landfill vulnerability (i.e., likelihood of the landfill releasing waste). To assess the risk to receptors from eroded waste (hereinafter referred to as the pollution index) parameters have been identified to represent the landfill hazard (representing volumes and toxicity of waste released) and environmental vulnerability (i.e., likelihood of environmental harm from the released waste). The relationship between the sub-indices, indices, and the overall risk index is illustrated in Fig. 1.

The main parameters identified as important for assessing drivers of coastal landfill erosion on low-lying coasts are wave exposure, storm climate, flooding, and tidal range (Laner et al. 2008, 2009; McLaughlin and Cooper 2010; Neuhold and Nachtnebel 2011; Neuhold 2013; Rosendahl Appelquist 2013). There is little variation in mean wind speeds and wind gust speeds around England (Met Office 2016) and, therefore, storm climate was not considered. The main parameters for assessing the vulnerability of coasts to erosion are the coastal geomorphological type, coastal slope, sediment balance, beach width, and vegetated areas (McLaughlin and Cooper 2010; Rosendahl Appelquist 2013; Denner et al. 2015), and landfill assessments also consider the presence or absence of flood defences and the distance from the landfill to mean high water (Alaska Department of Environmental Conservation 2015). Here, features of the landfills were considered in place of the natural geomorphology and underlying geology.

Parameters used in the landfill risk assessments to represent the hazard depend on the overall aim of the specific method and can be summarised as quantities and types of waste parameters, and contaminant concentration parameters (e.g., Laner et al. 2009; Alaska Department of Environmental Conservation 2015). Parameters used in the landfill risk assessments to represent the vulnerability of receptors also depend on the overall aim of the specific method and include water use, the proximity of habitats, and the presence of flora and fauna (including humans) (Cooper et al. 2013).

In the coastal vulnerability and landfill assessments, parameters are assigned relative severity scores to allow both quantitative and qualitative data to be used in the same assessment (Singh et al. 2009; Ramieri et al. 2011; Wamsley 2015). Here, a five-point severity

Fig. 1. Flow chart showing the relationship between the parameters, sub-indices, indices, and overall risk index.



scale for each parameter is used with the highest values indicating the greatest hazards or vulnerabilities of receptors (Table 1) (e.g., Palmer et al. 2011; Gill et al. 2014; Denner et al. 2015). Wherever possible severity scores from existing risk assessment methods have been utilised, but new severity scoring systems are proposed where necessary. How the parameters fit into the overall assessment process is summarised in Fig. 1. Some parameters are included in more than one sub-index, and are therefore counted twice — once in each index — this is because they represent both the likelihood of waste being released and the rate at which waste would be released if a landfill were breached.

2.1. Parameter datasets

2.1.1. Wave energy

Wave energy as a driver of erosion influences whether a landfill will breach and the rate at which waste would be released. Wave energy hitting the shoreline depends upon the height of waves, their orientation to shore, wave fetch, and width and vegetation of any buffer zones (Möller and Spencer 2002; Rosendahl Appelquist 2013). Currently, there are limited data available for most of these factors in England. In the absence of wave data, free fetch can be used to classify coasts as protected (waterbody width < 10 km), moderately exposed (10 km < waterbody width < 100 km), or exposed (100 km < waterbody width) (Rosendahl Appelquist 2013). Coastlines with a free fetch >10 km may also be classed as protected if the local geology or wind and wave climate is such that wave action is limited; this is indicated by the presence of saltmarshes (Mangor 2004; Rosendahl Appelquist 2013). Free fetch can easily be determined using most maps and a GIS dataset of saltmarsh extents is available to download (UK Government 2016).

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Table 1. Assigning severity scores to the assessment parameters for each historic coastal landfill.

		Severity score				
Parameter	Measure	1	2	3	4	5
Wave energy[a]	Fetch	<10 km or saltmarsh present	—	10–100 km	—	>100 km
Tidal classification	Tidal range	Macrotidal (>4 m)	—	Mesotidal (2–4 m)	—	Microtidal (<2 m)
Flooding	Predominant RoFRS zone over landfill	Predominantly outside RoFRS	Very low	Low	Medium	High
Landfill position	Landfill boundary to mean high water (m)	>50	>35–50	>20–35	>5–20	≤5
Exposed boundary	Length of landfill boundary facing foreshore (m)	≤500	>500–1000	>1000–2000	>2000–3000	>3000
Defence condition	Flood defence condition grade	1	2	3	4	5
Defence type	—	Hard	Mixed	Soft	Partly undefended	No defence or landfill is defence
Coastal slope[b]	Distance between landfill and 20 m isobath (km)	>4	>3–4	>2–3	>1–2	≤1
Sediment balance	—	Accretion	—	No change	—	Erosion
Buffer zone	Width of saltmarsh (m)	>50	>20–50	>10–20	>0–10	No saltmarsh
Landfill volume	Volume (m^3)	≤500 000	>500 000–1 000 000	>1 000 000–1 500 000	>1 500 000–2 000 000	>2 000 000
Landfill type	NB: for mixed sites choose highest severity score of the types present	Inert	MSW, household or commercial	Industrial	Special waste	Liquid sludge or unknown
Salinity	—	Upstream of oligohaline zone	Oligohaline zone	Mesohaline zone	Polyhaline zone	Downstream of polyhaline zone
Dissolved contaminants	Tidal prism volume (m^3)	>500 000 000 or open coast	>100 000 000–500 000 000	>50 000 000–100 000 000	>5 000 000–50 000 000	≤5 000 000
Human impact	Distance to bathing water catchment (m)	>150	>100–150	>50–100	>0–50	Landfill is in bathing water catchment
Designated sites	Distance to designated site(s) (use highest relevant score)	>250 m upstream and >1 km downstream	≤250 m upstream or ≤1 km downstream	≤250 m upstream or ≤500 m downstream	>0–100 m	Landfill site is within designated site(s)
Seafood	Distance to shellfish/mollusc site (use highest relevant score)	>250 m upstream and >1 km downstream	≤250 m upstream or ≤1 km downstream	≤250 m upstream or ≤500 m downstream	>0–100 m	Landfill site is within shellfish/mollusc site(s)

Note: RoFRS, Risk of Flooding from Rivers and Sea; MSW, municipal solid waste.
[a]Categorisation after Mangor (2004) and Rosendahl Appelquist (2013).
[b]Adapted from Palmer et al. (2011).

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2.1.2. Tidal classification

The tidal range influences how vulnerable coastlines are to wave energy (McLaughlin and Cooper 2010) and flooding (Rosendahl Appelquist 2013). The greater the tidal range the lower the probability that high tide and high waves will coincide, hence the probability of wave-related erosion (McLaughlin and Cooper 2010) and the probability of flooding (Rosendahl Appelquist 2013) are reduced. In addition, wide intertidal zones in which wave energy can dissipate are often present in areas with high tidal ranges (McLaughlin and Cooper 2010). The tidal classification (i.e., whether it is macrotidal, mesotidal, or microtidal) is considered adequate to assess tidal range as a hazard (Davies and Moses 1964; Rosendahl Appelquist 2013) and can be found for all British estuaries in Davidson (1991).

2.1.3. Flooding

Flooding increases the probability of landfills eroding both due to the movement of water over the site (Laner et al. 2008) and because infiltration of high volumes of water can adversely affect the structural integrity of the waste (Blight and Fourie 2005). In addition, the build-up of water pressure behind a flood defence can cause it to fail, exposing waste (Cooper et al. 2013). The GIS dataset Risk of Flooding from Rivers and Sea (UK Government 2016), shows the residual flood zones after mitigation by flood defences broken down into four categories: Very Low (annual probability < 0.1%), Low (0.1% ≤ annual probability < 1%), Medium (1% ≤ annual probability < 3.3%), and High (3.3% ≤ annual probability) and was used to assess flooding as a driver of both erosion and landfill breaching.

2.1.4. Landfill position

The closer the landfill is to mean high water, the greater the risk of it being eroded. There are discrepancies in the position of the high water line between different Ordnance Survey (OS) and Environment Agency (EA) datasets due to different update frequencies and scales used (e.g., Environment Agency 2016; Ordnance Survey 2016). This research used the (mean) High Water line in the OS Boundary-Line dataset (Ordnance Survey 2016) as it was the most recently updated of the large scale datasets (1:10 000) and OS data are used to produce EA datasets (Environment Agency 2016).

2.1.5. Exposed boundary

The length of the landfill boundary exposed to wave impact will also influence the probability of waste being eroded and can be determined by comparing the Historic Landfill Sites National Dataset (Environment Agency 2017) to the High Water line in the OS Boundary-Line dataset (Ordnance Survey 2016).

2.1.6. Defence condition and defence type

The likelihood of historic coastal landfills eroding and releasing waste is linked to whether there are effective flood defences present. The probability of flood defences breaching is linked to the probability of them overtopping and coastal erosion, which are already accounted for within the assessment, and their current state of repair and type (Bujis et al. 2007; Scott Wilson 2008; Environment Agency 2010*b*), which are recorded in the EA's Spatial Flood Defences GIS dataset (UK Government 2016).

2.1.7. Coastal slope

The shallower the coastal slope (below mean high water) the lower the rate of coastal erosion (Palmer et al. 2011). The Portal for Bathymetry online map (European Marine Observation and Data Network 2016) depth profile function was used to approximate

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distances between landfills and the 20 m isobaths as a proxy for coastal slope (after Palmer et al. 2011).

2.1.8. Sediment balance

There is a paucity of national scale erosion and accretion (rate) mapping for England; however, Shoreline Management Plans (SMPs) exist for the entire coast and include data indicating whether areas are eroding or accruing sediment (e.g., Royal Haskoning 2009; Environment Agency 2010*a*). Where the plans provide more than one erosion scenario (e.g., No Active Intervention (NAI) and With Present Management (WPM)), WPM data were used to determine the value of the sediment balance parameter as they account for any artificial sediment recharge that may be taking place.

2.1.9. Buffer zone

The presence of vegetated saltmarshes can significantly attenuate the impact of waves upon flood defences, dissipating up to half of the wave energy in the first 10–20 m of saltmarsh surface, reducing the risk of defences being overtopped or breached (Möller and Spencer 2002; Committee on Climate Change 2013). A GIS dataset of saltmarsh extent was used to determine the average width of saltmarsh in front of the landfill (UK Government 2016).

2.1.10. Landfill volume

Existing landfill risk ranking methods (Laner et al. 2008, 2009; Neuhold and Nachtnebel 2011; Neuhold 2013; Alaska Department of Environmental Conservation 2015) determine the hazard posed by assuming the entire landfill will erode, as saturated waste is known to be mechanically unstable (Blight and Fourie 2005; Liang et al. 2015). The area of historic landfill sites can be determined from the Historic Landfill Sites National Dataset (Environment Agency 2017) and GIS mapping software (e.g., ArcMap); however, the dataset does not provide information on waste volumes. Waste volume data for some sites can be obtained from local authorities (see Supplementary Material, Table S1[2]), elsewhere volume can be estimated by comparing historic records of site topography to the present topography or using monitoring well depths (where present) in conjunction with the landfill's area.

However, it seems unlikely that entire landfills would erode, as waste is often deposited in discrete cells, where the walls are more resilient to erosion, and breaches in flood defences are likely to be quickly repaired before all of the waste is released. Therefore, to assess the magnitude of the hazard from eroded waste materials, consideration also needs to be given to how quickly waste materials are likely to erode as well as how much waste is present in total. Hence, parameters that are proxies for the erosion rate (i.e., wave energy, tidal classification, landfill position, defence type, coastal slope, and buffer zones) are included in the landfill hazard sub-index as well as the coastal drivers and landfill vulnerability sub-indices.

2.1.11. Landfill type

The Historic Landfill Sites National Dataset (Environment Agency 2017) provides an indication of whether sites contain inert, industrial, commercial, household, special waste, liquid sludge, or if the type of waste is unknown. Just 37% of historic coastal landfill sites contain only a single waste type, 45% of the sites contain a mixture of waste types in unknown proportions, and 18% of the sites have no record of the waste received. The range of materials and contaminant concentrations in each waste type vary depending on when

[2]Supplementary material is available with the article through the journal Web site at http://nrcresearchpress.com/doi/suppl/10.1139/anc-2018-0001.

waste was deposited (Parfitt 2009; Quaghebeur et al. 2013), but only 44% of England's historic coastal landfills have both the opening and closing dates recorded.

Even where waste types and operating periods are known, material types and contaminant concentrations are highly variable (e.g., metal concentrations can vary by up to four orders of magnitude between and within sites[1]). Contaminant concentrations, speciation and behaviour are site specific and vary at the micro-scale, and, therefore, representative sampling is challenging and impracticable for a regional or national scale screening assessment[1] (Neuhold 2013).

The maximum permissible (leachable) concentrations of contaminants in materials being landfilled vary with the landfill site type (e.g., sites that are permitted to take hazardous waste (also known as special waste) are allowed maximum (leachable) concentrations of mercury 200 times higher, and of chromium 140 times higher, than inert sites (Council Decision 2003)). Therefore, for this research the site type is used as a proxy for ranking the severity of the hazard from contaminants in the waste (Singh et al. 2009; Alaska Department of Environmental Conservation 2015). The severity increases in the order: inert < municipal solid waste (including household waste) or commercial < industrial < special waste (Council Decision 2003; Singh et al. 2009; Alaska Department of Environmental Conservation 2015; NetRegs n.d.). Liquid sludges contain chemical wastes, sewage sludge, and industrial wastewater mixed with municipal solid waste (Environment Agency 2013), but no information could be found to indicate how hazardous they are in relation to other waste types. However, as historic coastal landfill sites typically pre-date regulations controlling which chemicals are disposed of (Brand et al. 2018), liquid sludge landfills may contain chemical wastes that would not be accepted at modern landfills for special waste, therefore, in the absence of better data they have been assigned the highest hazard rating. Landfills where the contents are classified as unknown have also been assigned the highest hazard rating as they may contain liquid sludge.

2.1.12. Salinity

Metal release from waste is significantly higher in saline waters compared to freshwaters (Brand 2017) and is included as a parameter by using salinity zones determined using the Joint Nature Conservation Council's (JNCC) Variable Salinity Areas dataset (McBreen et al. 2011).

2.1.13. Dissolved contaminants

There are insufficient data to determine the release of contaminants into the water column by leaching (Brand 2017; Brand et al. 2018). However, it is possible to rank the hazard posed by leached contaminants using the amount of waste eroded and waste type as proxies for the maximum mass of contaminants that could leach, and by considering dilution in the receiving waters. Here, the tidal prism was used as a proxy for the total effective volume of water and was calculated using the average tidal range from SMPs and the estuary's transitional area recorded in the Water Framework Directive (WFD) transitional and coastal waterbodies cycle 2 dataset (UK Government 2016). For estuaries large enough to be split into multiple transitional zones in the WFD dataset, only the zone adjacent to the landfill was considered. This potentially overestimates the dilution of contaminants for landfill sites that are on tributaries that are not considered independently in the WFD dataset. However, this level of accuracy in determining dilution was considered appropriate given the uncertainty associated with the concentrations of contaminants in the waste and their mobility.

2.1.14. Human impact

In the intertidal zone, humans are most likely to come into contact with any eroded waste or released contaminants during recreational use of beaches. The distances between

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Table 2. Minimum and maximum possible sub-indices scores (before normalisation).

Sub-index	Minimum possible score	Maximum possible score
Coastal drivers	3	15
Landfill vulnerability	7	35
Landfill hazard	10	50
Environmental vulnerability	3	15

historic coastal landfills and bathing water catchments shown in the EA's areas affecting bathing waters dataset (UK Government 2016) were used as a proxy for the quantities of solid waste materials and dissolved contaminants that humans may come into contact with. This was based on the assumption that the greater the distance from the source of the waste, the greater the dispersion of the waste and dilution of the contaminants.

2.1.15. Designated sites

There exists a multitude of environmentally designated sites around England. The availability of GIS datasets for those highlighted as being vulnerable to contaminants from historic coastal landfills by Cooper et al. (2013), and others that fall within the coastal (flood) zone are shown in Supplementary Material Table S2[2]. For the purposes of this assessment heritage coasts were also treated as designated sites. Designated sites upstream as well as downstream of the landfills were included to account for tidal movement of contaminants.

2.1.16. Seafood

Seaweed, crustaceans, other shellfish, and fish may be harvested from the intertidal zone and tidal waters for human consumption. Only GIS datasets relating to shellfish waters were available for the assessment: Cefas's Classified Bivalve Mollusc Harvesting Areas GIS dataset (O. Morgan, personal communication, email, 2 November 2015) and the Shellfish Waters GIS dataset (Defra 2016). Similar to assessing the vulnerability of human receptors, distances between these areas and historic coastal landfill sites were used as a proxy for the quantities of solid waste materials and dissolved contaminants that may reach these areas.

2.2. Calculation of the sub-indices, waste release and pollution indices, and overall risk index

A summation method was used to combine the severity scores to determine the values of the sub-indices (Ramieri et al. 2011; Khouakhi et al. 2013; Musekiwa et al. 2015)

$$(1)\quad \text{subindex} = \sum \text{severity scores}$$

No weightings were directly applied to individual parameters within the sub-indices. Where a different number of parameters are used for each of the sub-indices, normalising each sub-index value to a percentage allows them to be combined into the overall risk index without any one sub-index dominating the overall risk score (McLaughlin and Cooper 2010). Therefore, the four sub-indices were normalised to percentages using (after McLaughlin and Cooper 2010)

$$(2)\quad \text{normalised subindex} = \frac{\sum \text{severity scores} - \text{min. possible score}}{\text{max. possible score} - \text{min. possible score}} \times 100$$

and values from Table 2, before being combined into the waste release index and pollution index using

$$(3)\quad \text{waste release index} = \frac{\text{normalised coastal drivers} + \text{normalised landfill vulnerability}}{2}$$

and

Table 3. Screening assessment test site histories.

Name and landfill database reference No.[a]	Operating period[b]	Type[c]	Volume[b] (m^3)	Flood defences[a]
Common Road EAHLD01226	1970–1993	Household, commercial, and industrial	450 000	Partly defended
Hadleigh Marsh EAHLD01181	1980–1987	Household and commercial	500 000	Landfill is the flood defence
Leigh Marshes EAHLD00531	1955–1967	Household, commercial, and industrial	800 000	Yes
Martins Farm North EAHLD01246	1960–1995	Household, commercial, and industrial	1 400 000	Yes
Martins Farm South EAHLD01241	1985–1995	Household, commercial, and industrial	1 200 000	Yes
Newlands EAHLD01178	1954–1989	Household, commercial, and industrial	1 000 000	Landfill is the flood defence
Park Drive EAHLD01739	1974–1994	Household, commercial, and industrial	800 000	Landfill is the flood defence
Sea Wall EAHLD01228	1988–1991	Household, commercial, and industrial	275 000	Landfill is the flood defence

[a]GIS datasets from UK Government (2016).
[b]Site records (A. Brown, personal communication, email, 26 October 2015), except Leigh Marshes operating period from Environment Agency (2013) and volume estimated using GIS data from UK Government (2016) and trial pit depths recorded in a report by Halcrow Group Ltd. (2012).
[c]Environment Agency (2013).

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$$(4)\quad \text{pollution index} = \frac{\text{normalised landfill hazard} + \text{normalised environmental vulnerability}}{2}$$

respectively. The overall risk index was then calculated using

$$(5)\quad \text{overall risk index} = \frac{\text{waste release index} + \text{pollution index}}{2}$$

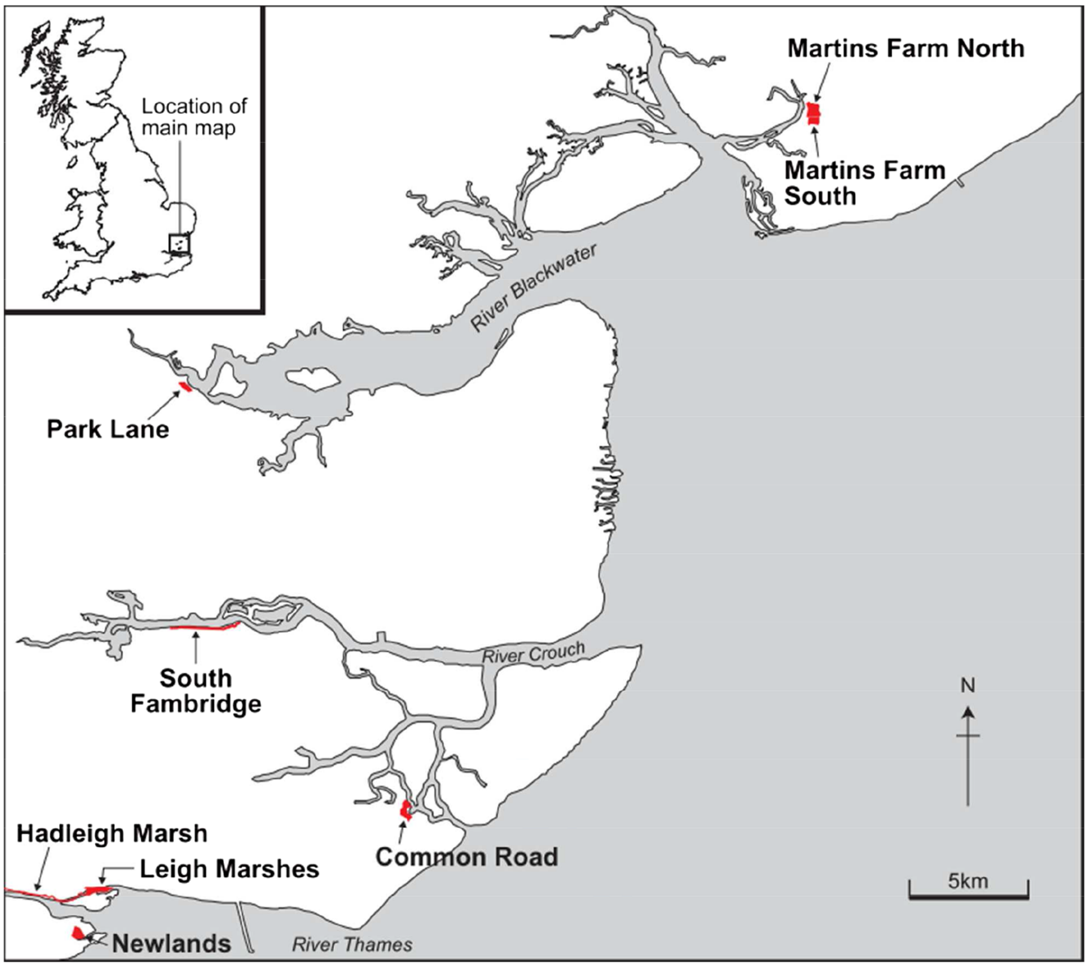
All three indices have value ranges from 0 to 100.

3. Testing the risk screening assessment methodology

3.1. Study site selection

Eight historic coastal landfills were selected for testing the screening assessment methodology (Table 3 and Fig. 2). The landfills are distributed over four estuaries in southeast England, but some are adjacent to each other, which allows testing of the method for sensitivity to changes in factors, such as the distance between the landfill and mean high water. All of the landfills were chosen from the same region as the method needs to distinguish risk at a local level in the case that remediation funds are allocated locally. As adjacent sites could be affected by the same extreme event, Martins Farm North and Martins Farm South were used to test whether the risk ranking would be affected if sites in close proximity were subject to a joint assessment as well as individual assessments. To test the effect of giving individual weightings to parameters, the analysis was done twice, once with the default method given and once with double weighting applied to the unique landfill hazard

Fig. 2. Map showing the locations of the eight historic coastal landfills selected for testing the risk screening assessment (created using data © Environment Agency copyright and (or) database right 2017. All rights reserved. Contains information © Local Authorities. © Crown copyright and database rights 2004 Ordnance Survey 100024198).



parameters (landfill volume, landfill type, salinity and dissolved contaminant; i.e., those parameter scores were multiplied by two, and the landfill hazard normalisation calculation adjusted accordingly by using a minimum possible score of 14 and a maximum possible score of 70).

3.2. Results

The parameter severity scores, sub-indices' values, and indices' values are shown in Table 4 for the default methodology with no individual parameter weightings applied. Of the sites tested, Hadleigh Marsh had the highest overall risk index, with a value of 53.5, and Martins Farm South had the lowest overall risk index, with a value of 40.9. The results of the analysis with double weighting applied to the unique landfill hazard parameters are shown in Table 5, the combined Martins Farm sites moved one place higher and Martins Farm North moved two places higher in the ranking under this methodology.

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Table 4. Results of assessing the test sites; sites shown left to right from highest to lowest overall risk index value — no weightings given to individual parameters.

Parameter being scored	Hadleigh Marsh	Sea Wall	Common Road	Martins Farm, both sites combined	Leigh Marshes	Newlands	Martins Farm North	Park Drive	Martins Farm South
Wave energy	1	1	1	1	1	1	1	1	1
Tidal classification	1	1	1	1	1	1	1	1	1
Flooding	2	3	1	1	3	1	1	1	1
Landfill position	5	5	5	4	5	5	4	4	1
Exposed boundary length	5	5	3	2	3	3	2	1	1
Defence condition	3	3	3	4	3	3	4	2	3
Defence type	5	5	4	3	2	5	3	5	3
Coastal slope	4	1	1	1	4	5	1	5	1
Sediment balance	5	5	3	5	3	5	5	3	5
Buffer zone	4	5	5	5	5	1	5	4	5
Landfill volume	1	1	1	5	2	2	3	2	3
Landfill type	2	3	3	3	3	3	3	3	3
Salinity	4	5	5	3	4	4	3	4	3
Dissolved contaminant	1	2	2	4	1	1	4	2	4
Human impact	5	1	5	5	5	5	5	5	5
Designated sites	5	5	5	5	5	5	5	5	5
Seafood	2	5	5	4	2	3	4	1	3
Coastal drivers sub-index	4	5	3	3	5	3	3	3	3
Landfill vulnerability sub-index	31	29	24	24	25	27	24	24	19
Landfill hazard sub-index	28	29	28	30	28	28	28	31	25
Environmental vulnerability sub-index	12	11	15	14	12	13	14	11	13
Normalised coastal drivers sub-index	8.3	16.7	0	0	16.7	0	0	0	0
Normalised landfill vulnerability sub-index	85.7	78.6	60.7	60.7	64.3	71.4	60.7	60.7	42.9
Normalised landfill hazard sub-index	45.0	47.5	45.0	50.0	45.0	45.0	45.0	52.5	37.5
Normalised environmental vulnerability sub-index	75.0	66.7	100	91.7	75.0	83.3	91.7	66.7	83.3
Waste release index	47.0	47.6	30.4	30.4	40.5	35.7	30.4	30.4	21.4
Pollution index	60.0	57.1	72.5	70.8	60.0	64.2	68.3	59.6	60.4
Overall risk index	53.5	52.4	51.4	50.6	50.2	49.9	49.3	45.0	40.9

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Table 5. Results of assessing the test sites; sites shown left to right from highest to lowest overall risk index value — unique landfill hazard parameters double weighted.

Parameter being scored	Hadleigh Marsh	Sea wall	Martins Farm, both sites combined	Common Road	Martins Farm North	Leigh Marshes	Newlands	Park Drive	Martins Farm South
Coastal drivers sub-index	4	5	3	3	3	5	3	3	3
Landfill vulnerability sub-index	31	29	24	24	24	25	27	24	19
Landfill hazard sub-index	36	40	45	39	41	38	38	42	38
Environmental vulnerability sub-index	12	11	14	15	14	12	13	11	13
Normalised coastal drivers sub-index	8.3	16.7	0	0	0	16.7	0	0	0
Normalised landfill vulnerability sub-index	85.7	78.6	60.7	60.7	60.7	64.3	71.4	60.7	42.9
Normalised landfill hazard sub-index	39.3	46.4	55.4	44.6	48.2	42.9	42.9	50.0	42.9
Normalised environmental vulnerability sub-index	75.0	66.7	91.7	100.0	91.7	75.0	83.3	66.7	83.3
Waste release index	47.0	47.6	30.4	30.4	30.4	40.5	35.7	30.4	21.4
Pollution index	57.1	56.5	73.5	72.3	69.9	58.9	63.1	58.3	63.1
Overall risk index	52.1	52.1	51.9	51.3	50.1	49.7	49.4	44.3	42.3

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4. Discussion

The coastal drivers sub-index ranked Leigh Marshes and Sea Wall in South Fambridge as the sites potentially subjected to the greatest drivers of erosion, followed by Hadleigh Marsh. However, the landfill vulnerability sub-index indicated that Leigh Marshes is better protected from the coastal drivers than Sea Wall and Hadleigh Marsh, which reflects the fact that it has a much shorter length of boundary facing mean high water and is separated from the estuary by a flood defence. In contrast, Sea Wall and Hadleigh Marsh are both waste-filled flood embankments with several kilometres of exposed boundary. The waste release index, which combines the coastal drivers and landfill vulnerability sub-indices, indicated the two flood embankments (Sea Wall in South Fambridge and Hadleigh Marsh) are the two most likely test sites to release solid waste to the environment, reflecting their exposure to their estuaries, having no flood defences separating them from the water, and having very long boundaries adjacent to mean high water, increasing the probability that at least part of the landfill sites will breach.

In contrast, the two waste-filled flood embankments were ranked low in the range of pollution index values suggesting that, if waste erodes from them, they are likely to cause comparatively less pollution than the other sites tested. This reflects the relatively small volumes of waste in the two flood embankments, combined with the high levels of dilution at Hadleigh Marsh landfill site and the absence of bathing water catchments in the estuary at the Sea Wall in South Fambridge landfill site. However, the two waste-filled flood embankments had the two highest overall risk index values reflecting that, for the test sites, the range of waste release index values (range = 26.2) is greater than the range of pollution index values (range = 15.4) and therefore the waste release index has greater influence in determining the overall risk index ranking of the test sites. The limited range of pollution index values reflects the very similar waste contents and ecological environments of the eight sites, and the greater range of waste release index values reflects the greater range of vulnerabilities of the landfill sites to coastal drivers, particularly differences in defences and the lengths of their boundaries.

The inclusion of some parameters within more than one sub-index means that the sub-indices are not fully independent of each other and the duplicated parameters have greater influence upon the overall risk index (see Supplementary Material — Sensitivity analysis of the risk screening assessment[2]). In addition, the greater number of parameters in the landfill vulnerability sub-index compared to the coastal drivers sub-index means the waste release index and overall risk index are more sensitive to changes in the coastal drivers sub-index parameters than the landfill vulnerability sub-index parameters. Similarly, the greater number of parameters in the landfill hazard sub-index compared to the environmental vulnerability sub-index means the pollution index and overall risk index are more sensitive to changes in the environmental vulnerability sub-index parameters than the landfill hazard sub-index parameters. It could be argued that the vulnerability of receptors is more important in determining pollution risk than the chemical content of the material released from an eroding landfill site because waste material has the potential to physically and chemically alter the coastal or estuarine environment if eroded, but studies of the impact of landfill debris on the marine environment are limited (Pope et al. 2011). These potential issues highlight that further consideration is needed to determine which parameters, if any, are in reality more significant in determining the overall risk and, hence, whether weightings should be applied to increase their influence on the final risk rankings. To demonstrate the importance of weighting individual parameters, testing the application of a double weighting to the unique landfill hazard parameters increased the risk ranking of the combined Martins Farm sites one place and Martins Farm North two places.

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Weightings must be specific to the combination of parameters and indices being used; therefore, to determine weightings with any useful level of accuracy would require input from experts in coastal processes, landfill engineering stability and contamination, and ecology. In addition to weighting of parameters, it may also be appropriate to include a distinction between different types of ecological sites to ensure that those most difficult to replace or rehabilitate are given priority when considering which landfill sites to remediate first. The necessary consultations were beyond the scope of this research, but should be considered in any future developments of the risk screening assessment. The consultations should also consider whether sites in close proximity should be subject to a joint assessment as well as individual assessments in case there is an event of sufficient magnitude to breach multiple sites (e.g., a storm surge) and, if so, what the minimum separation between sites should be before they are only assessed independently. To demonstrate the importance of this, considering Martins Farm North and Martins Farm South in combination ranked them fourth by overall risk index compared to sixth and eighth when only individual sites were considered.

The value of the overall risk index can range from 0 to 100 under the proposed scoring system. As there are over 1200 (currently known) historic coastal landfills to be ranked (Brand et al. 2018), there will be multiple landfill sites with similar overall risk index values. If a series of overall risk index value thresholds were set to provide categories of risk (e.g., very high, high, moderate, low, and very low), then this would mitigate the issue of having multiple sites with the same or similar index values. Note a zero risk category is deliberately not included as there is always a residual risk of a site eroding and causing pollution (Neuhold and Nachtnebel 2011). A categorical risk approach would also have the advantage of allowing the end-user greater discretion in determining the order in which sites are considered for further investigation and (or) remedial action, which would better support management of limited budgets. For example, if all sites in a risk category are given the same priority for remediation, rather than using the overall risk score to rank them individually, it would allow multiple sites with low remediation costs to be addressed instead of a single site within the same category that has a higher overall risk score and a higher remediation cost. However, such categories cannot be implemented until a much greater number of sites have been assessed to provide a benchmark of the levels at which such fixed thresholds should be set.

To undertake a national-scale risk screening assessment using this methodology would be relatively straightforward. The majority of parameter scores can be determined from data tables in the literature, or from GIS datasets either by reading data tables, creating buffer zones, or directly measuring distances. Only the sediment balance and dissolved contaminant scores require searching documents (e.g., SMPs) for data. Based on the test it is estimated that it would take approximately 3 months to assess the circa 1200 sites around the coast of England, but this timescale could be significantly reduced using efficiencies in data collection (e.g., tidal prism would not need to be calculated for each individual site and some scores could be automatically calculated in GIS software). If the assessment were divided by SMP area then it would take <1 week per SMP (before efficiencies) and could easily be integrated into one of the periodic SMP updates.

5. Conclusion

A new risk screening assessment method has been proposed that can support coastal managers in identifying which historic coastal landfill sites pose the greatest pollution risk at a national scale for minimal cost using existing datasets. The highest risk sites can then be prioritised for further investigation, including ground-truthing, or remedial works as appropriate. The risk screening assessment provides a snapshot of the current highest risk

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sites and should be updated as the underlying datasets are modified to reflect changes to factors such as site condition (e.g., due to maintenance works) or flood extent (e.g., due to climate change related sea level rise or changes to defences).

Prior to the assessment being implemented nationally, consultations should be carried out with experts in coastal processes, landfill engineering stability and contamination, and ecology to ensure the parameter severity scores, sub-indices, and indices calculations are appropriately weighted to reflect their contribution to the overall risk of historic coastal landfill sites eroding and causing pollution. These also need to be agreed upon with appropriate regulators. Currently, parameters representing the total landfill volume and contaminant concentrations in the waste have the lowest influence on the overall risk score, and parameters representing the probability of waste being released, the rate at which it will be released, and the vulnerability of receptors are of much greater importance in determining the overall risk score. This suggests the uncertainty and incompleteness of the data representing the landfill volumes and contaminant concentrations in waste are not a major obstacle to assessing the risk of pollution from historic coastal landfill sites, and that resources should not be expended on attempting to improve the accuracy of these parameter datasets, particularly given the difficulties of obtaining representative contaminant data and the high costs involved[1]. However, the importance of the landfill volume and contaminant concentrations in the waste in determining the overall risk score may increase once weightings have been added to the risk screening assessment parameters and indices.

Testing the risk screening assessment, by applying it to eight historic coastal landfills in southeast England, found that despite their relatively small volumes, the only two waste-filled flood embankments screened (Hadleigh Marsh and Sea Wall in South Fambridge) pose the greatest overall risk of pollution. This is due to their relatively high exposure to drivers of coastal erosion and vulnerability to erosion, which means they are more likely to breach than the other sites screened and, if breached, are likely to release waste at a greater rate than most other sites screened. This means that these two sites should be given priority for expenditure on further investigation and (or) remedial actions ahead of the other six sites screened.

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