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UPDATE

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Potential pollution risks of historic landfills in England: Further analysis of climate change impacts

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This article is an update of "Potential pollution risks of historic landfills on low-lying coasts and estuaries".

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Abstract

Five years ago, an article in *WIREs Water* provided the first comprehensive analysis of historic (legacy) landfill sites vulnerable to coastal flooding and erosion at a national scale (England). This update expands upon that article by considering the potential impacts of climate change upon inland historic landfills. Globally, there are hundreds of thousands of landfills that predate modern environmental regulations, and where waste is not isolated from the surrounding environment, but climate change impacts on the pollution risk from historic landfills in freshwater environments has received little attention. Where climate change causes an increase in the frequency and magnitude of fluvial flood events, this will increase leachate generation and the probability of landfill erosion and solid waste release. Where there is increased drought the landfill capping materials may crack, opening up new pollutant pathways, and increasing the risk of solid waste release. Changes to groundwater movement resulting from climate change may open new leachate pathways, and in England alone, thousands of historic landfills are in (groundwater) Source Protection Zones where modern regulations to protect drinking water supplies would not permit their construction. This increased contaminant release from historic landfills in freshwater environments may impact surface and/or groundwater quality and ecological health, increase costs for drinking water monitoring/treatment, or make some abstraction sources unviable. This is especially of concern where receptors are subject to multiple pressures and may cause tipping points to be reached. Further research is warranted into contaminant behavior, receptor vulnerability, historic landfill risk prioritization, and mitigation/remediation methods.

This article is categorized under:
- Engineering Water > Engineering Water
- Science of Water > Water Quality
- Science of Water > Water and Environmental Change
- Water and Life > Stresses and Pressures on Ecosystems

KEYWORDS

contamination, erosion, flooding, hydrology, legacy landfill

1 | INTRODUCTION

Five years ago, an article in Volume 5, Issue 1 of *WIREs Water* provided the first comprehensive analysis of "historic" landfill sites (also known as "old" or "legacy" landfills) vulnerable to coastal flooding and erosion at a national level (Brand et al., 2018). The article reported that there are hundreds of thousands of historic landfills globally, which predate modern waste disposal regulations, typically have no leachate or gas containment, and that have limited or no records kept of the source, type, or volume of waste stored (Brand et al., 2018). Brand et al. (2018) focused on the thousands of historic landfills in England's coastal zone and considered the types of waste and contaminants in the landfills, contaminant pathways, and potential receptors. Coastal landfills were defined as those within an area with 0.5% annual probability of tidal flooding, ignoring the presence of flood defenses. It presented for the first time the growing risk of pollution from historic landfills within the coastal zone as climate change increases the probability of coastal flooding and erosion, which can lead to solid waste and soluble contaminant release to the surrounding environment. It highlighted gaps in knowledge relating to historic landfill contents, changes in contaminant mobility due to marine flooding, legacy pollution in surrounding coastal sediments, mechanisms of solid waste erosion and dispersion, and potential environmental impacts. It identified that coastal management policies are being influenced by the presence of historic landfills, for example, hold-the-line is likely to be chosen at locations where managed realignment would be preferred if landfills were not present. Finally, the article proposed the need for a method to prioritize historic coastal landfills by risk, so that limited coastal management resources can be appropriately targeted.

This update article also considers:

a. The potential impacts of climate change on contaminant release from historic landfills as a result of changes in the freshwater hydrological regime, for example, flooding and drought.
b. The potential magnitude of the risk, by assessing (i) hydrologic connectivity between historic landfills and surface and ground waters, and (ii) the number of designated nature reserves that may be connected to historic landfills via leachate plumes, using England as a case study.

This includes revisiting and updating some aspects of the original article to reflect progress made in addressing the knowledge gaps previously identified. The climate change effects described may apply globally, and equivalent assessments of the magnitude of the risk could be carried out at any regional or national level, providing a powerful tool to aid regulators and policy makers to assess the scale of the problem as a first step towards prioritizing management and/or remediation strategies.

2 | POTENTIAL EFFECTS OF CLIMATE CHANGE ON CONTAMINANT RELEASE FROM HISTORIC LANDFILLS

2.1 | Long-term contaminant release from historic landfills

A wide range of potentially hazardous substances can be found in municipal solid waste including, for example, acids, alkalis and solvents from cleaning products, heavy metals from batteries and electronic equipment, plastics, hydrocarbons from wood preservatives and ash, pesticides, biocides, and pharmaceuticals (Slack et al., 2004, 2005, p. 119; Yu et al., 2020).

The primary pathways for contaminants from historic landfills to reach terrestrial and aquatic environments are the movement of leachate and contamination of ground waters and surface waters, biological uptake of contaminants in the rhizosphere, particularly where capping is absent or poor, and the erosion and transport of solid waste by watercourses (Brand et al., 2018; Du Laing et al., 2009; O'Shea et al., 2018). A secondary pathway is the erosion of contaminated soils and sediments from the leachate attenuation zone (Brand et al., 2018).

Regulatory controls (such as those for the European Union) for modern landfills require leachate containment systems and prohibit them being located where their leachates could adversely affect surface or ground waters (Council Decision, 2003; Council Directive, 1999). However, historic landfills typically do not have leachate containment and as a result, many have hydrological connectivity to surface and/or ground waters, either directly or via leachate plumes (Brand et al., 2018; Gooddy et al., 2014; Propp et al., 2021). Numerous studies have explored leachate composition (e.g., LaGrega et al., 1994; Mukherjee et al., 2014; Robinson, 1995; Robinson, 2007; Robinson et al., 1982; Robinson & Maris, 1979; Ziyang et al., 2009) and leachates contain a wide range of soluble contaminants including metals, which are solubilized in the acetogenic phase of waste degradation, dissolved organic matter, inorganic anions (e.g., sulfate, ammonia) and major elements (e.g., calcium and magnesium), and xenobiotic organic compounds depending on the nature and age of the wastes contained (Kjeldsen et al., 2002). Landfill leachates have also been identified as a pathway for microplastics to reach groundwater (Wan et al., 2022). Of particular concern for historic landfills is the presence of emerging contaminants, especially those not normally monitored in leachates, but which can be persistent and toxic at very low concentrations, including, for example, endocrine disruptors, veterinary medicines, and pharmaceuticals (Yu et al., 2020) and those chemicals whose manufacture, use and disposal has been restricted by recent legislation, but will be present in historic wastes, such as polychlorinated biphenyls (PCBs) and perfluoroalkyl and polyfluoroalkyl substances (PFASs; O'Rourke et al., 2022; Weber et al., 2011).

As wastes decompose, precipitation percolates through the landfills and leachate is released; it is often assumed that these soluble contaminants will be flushed out after a few decades and therefore, the longer-term impact of leachate from historic landfills on surface and ground waters is unlikely to be significant (Brand et al., 2018; Brand & Spencer, 2020; Kruempelbeck & Ehrig, 1999). However, there is increasing evidence that soluble contaminant release may persist over longer timescales. For example, ammonium-nitrogen (NH_4-N), produced during the methanogenic stage of organic waste decomposition may be present in leachates at concentrations harmful to freshwater environments for centuries following landfill closure (Gooddy et al., 2014; Hall et al., 2006a, 2006b, 2007; Kjeldsen et al., 2002; Propp et al., 2021; Robinson, 1995). Recent investigations of organic contaminants in leachate have identified elevated concentrations of organophosphate esters and bisphenols up to 60 years following disposal (Propp et al., 2021). Finally, substantial modification to landfill hydrochemistry, for example, due to flooding with marine waters, can solubilize significant metal load (Brand et al., 2018; Brand & Spencer, 2020; Flynn et al., 1984).

This suggests that historic landfills could be long-term sources of soluble metals, ammonia, and emerging contaminants to ground and surface waters and this has received little attention in the literature. It is commonly recognized that landfill leachates may be toxic to flora and fauna, either directly or indirectly as a result of biomagnification/bioaccumulation, eutrophication, and deoxygenation. It is increasingly evident that long-term release of leachate from historic landfills may also result in a significant deterioration in ecological health in proximity to historic landfill sites for centuries (Ausseil et al., 2017; Gooddy et al., 2014; Njue et al., 2012; O'Rourke et al., 2022; O'Shea et al., 2018; Pope et al., 1999).

Recent work has shown that coastal historic landfills are at risk of eroding and releasing solid waste with significant impacts on the coastal environment (e.g., Beaven et al., 2020; Brand et al., 2018; Brand & Spencer, 2019, 2020; Nicholls et al., 2019). Erosion and the physical mobilization of solid wastes during freshwater flooding of historic landfills is rare due to the extreme flow conditions required to erode landfill capping and mobilize solid waste. Therefore, it is often assumed that, once landfilled, solid waste will be permanently retained within the landfill perimeter and isolated from the surrounding environment. However, there are a number of published examples of floodplain/riverbank historic landfills that have eroded and released solid wastes when flooded (Blight & Fourie, 2005; Curtis & Whitney, 2003; Laner et al., 2008; Wille, 2018; Young et al., 2004), and an increasing numbers of unpublished reports, for example, erosion of solid landfill waste into the River Ericht at Blairgowrie, Scotland (Robertson, 2021) and into the Fox River in New Zealand (Department of Conservation, 2019). The likelihood that eroding unmanaged and historic landfills may be significant sources of plastics to the environment has also received recent attention (Yadav et al., 2020). The scarcity of information about historic landfill contents makes this especially concerning. There is potential for eroded solid waste including matrix materials (soil-like waste <10 mm in size), plastics, metal objects, and sharps to physically harm flora and fauna through ingestion, entangling, smothering, or crushing and to degrade receiving surface water quality through increased contaminant, nutrient, and suspended sediment concentrations and reduced dissolved oxygen concentrations (Bolam & Rees, 2003; Derraik, 2002; Pope et al., 2011; Zarfl & Matthies, 2010). Conversely, eroded waste material may provide habitat for macroinvertebrates in water bodies with limited habitat heterogeneity (Wilson et al., 2021). However, the chemical risk to freshwater bodies associated with the erosion of solid wastes has received little attention.

Brand and Spencer (2019) sampled and analyzed solid wastes (matrix, paper, textiles, and wood) from two historic landfills in England, which had received municipal, commercial, and industrial wastes. In Figures 1 and 2, these data are now compared to freshwater sediment quality guidelines (Canadian Council of Ministers of the Environment, 2001) and this suggests that if historic landfills in proximity to freshwater bodies are eroded, contaminant concentrations in a range of released waste materials could have adverse effects on flora and fauna. In addition, solid wastes may also act as disease vectors (Yahaya et al., 2016) and surface and ground waters may provide hydrological pathways for transportation of contaminants to other sensitive environmental receptors. Contaminant data are highly heterogeneous, and there is currently a lack of knowledge regarding the chemical and physical behavior of eroded solid wastes in aquatic environments suggesting this issue deserves greater consideration.

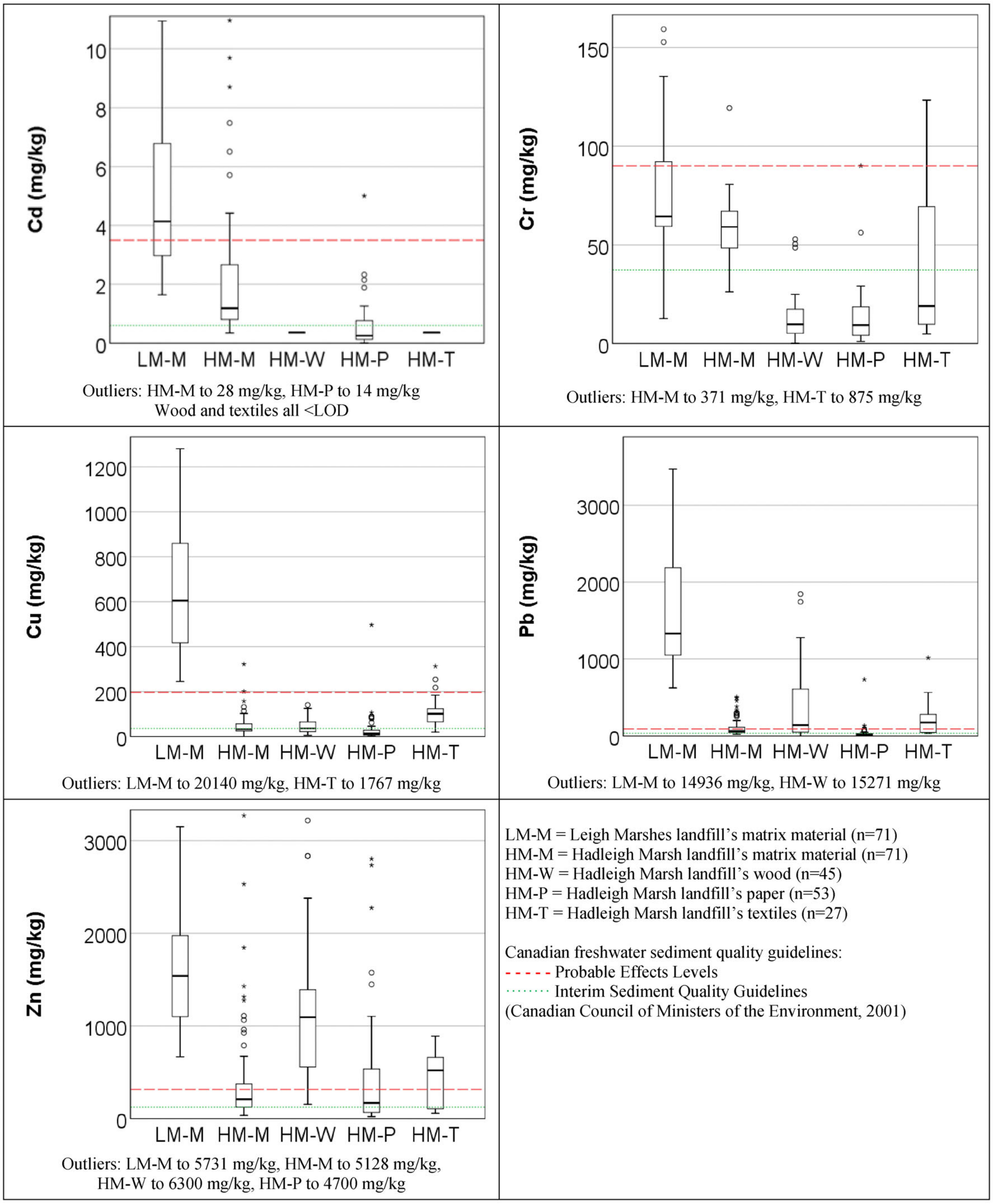

FIGURE 1 A comparison of inorganic contaminant concentrations in waste materials extracted from two historic landfills in England to Canadian freshwater sediment quality guidelines (adapted from Brand, 2017, p. 181; and Brand & Spencer, 2019, p. 288).

2.2 | Potential changes to contaminant release due to climate change impacts

Many historic landfills were located on soils that freely drain to ground waters, soils that are naturally wet due to shallow groundwater, or on floodplains. Such sites are subject to frequent groundwater and/or surface water flooding which will change the hydrological regime and water balance within a landfill. The flooding of landfills is a recognized environmental pollution risk (Arrighi et al., 2018; Laner et al., 2008; Wang et al., 2012). However, changes to in-channel flow conditions could result in erosion and release of solid wastes and this has received considerably less thought (Brand & Spencer, 2019, 2020; Yahaya et al., 2021).

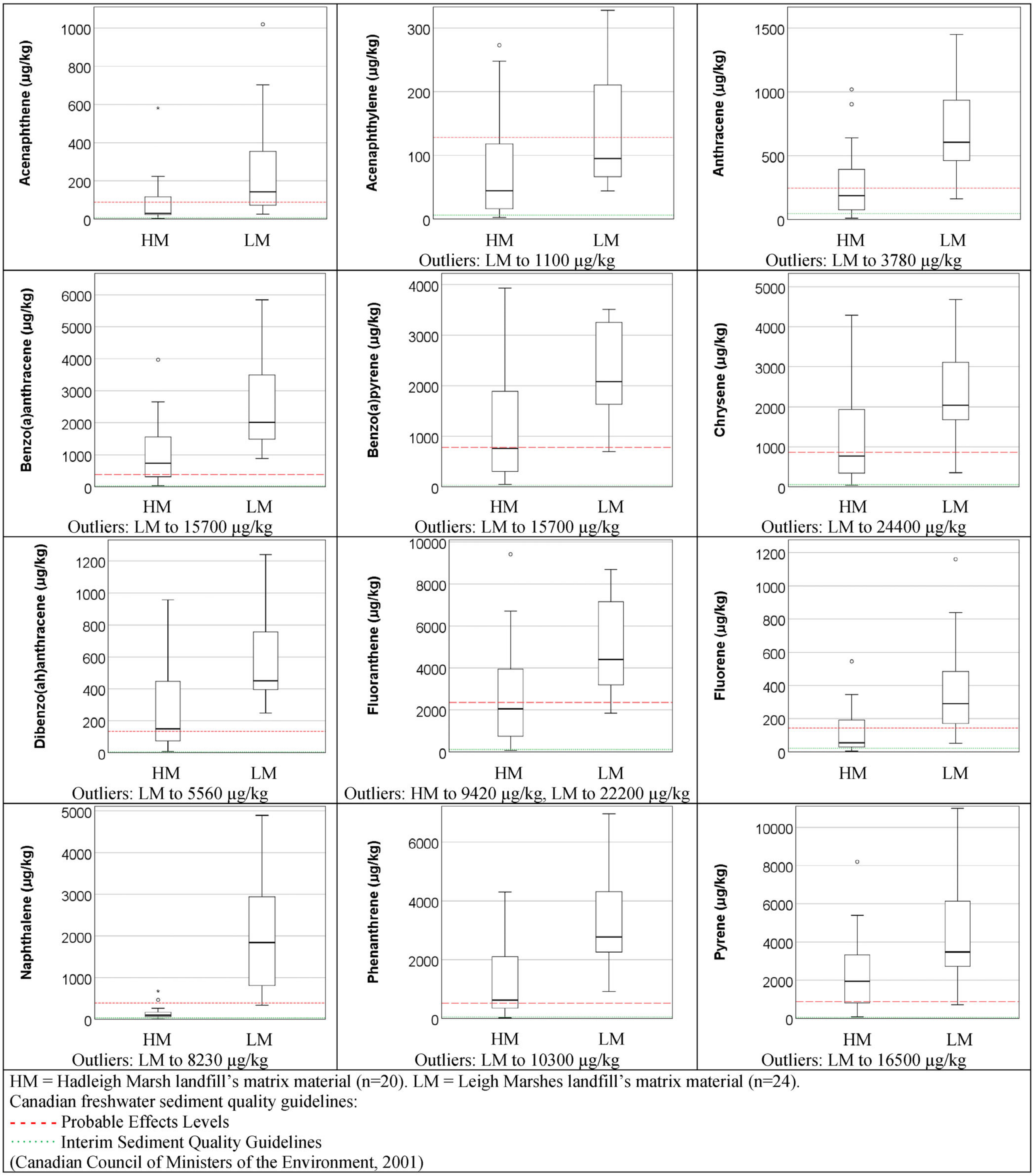

FIGURE 2 A comparison of organic contaminant concentrations in matrix materials extracted from two historic landfills in England to Canadian freshwater sediment quality guidelines (adapted from Brand, 2017, p. 185; and Brand & Spencer, 2019, p. 289).

Historic landfills are extremely vulnerable to climate change related hazards. The main effects are from changes in precipitation, storms, and water balance, for example, increased flooding or drought (Schneider et al., 2017; Yahaya et al., 2016, 2021; Young et al., 2004). Projections for the detailed impacts of climate change on the hydrological cycle are uncertain and highly variable on national, regional, and global scales (IPCC, 2013). Therefore, this article considers the general potential impacts on historic landfills of increased drought and/or increased flooding due to predicted changes in precipitation patterns, increased storminess, and rising sea levels. Some regions may be impacted by both of these climate change effects. For example, UK climate models predict warmer, wetter winters with an increase in the frequency and magnitude of flood events as well as hotter, drier summers with an increase in drought frequency; this is likely to vary across the country (IPCC, 2012; Lane & Kay, 2021).

2.2.1 | Increased flooding

Any increase in the intensity, frequency, duration, and magnitude of precipitation, and concomitant fluvial/groundwater flooding is likely to result in an increase in the number of historic landfills becoming subject to inundation as water levels increase. The area of direct runoff within catchments may be increased by a wetter climate, increasing the potential for contaminants to reach surface waters from locations that would previously have drained wholly via ground waters or been hydrologically isolated (Murdoch et al., 2000).

Similarly, sea level rise and, in some regions, increases in storm events will increase the frequency of coastal landfill flooding. Some landfills may be subject to multiple forms of flood risk and flooding has the potential to influence contaminant release. Rising groundwater or surface flooding of a landfill will increase its moisture content, the depth of saturated waste, and leachate generation and release. This may increase waste decomposition rates, landfill gas production, and leachate flux, mobilizing organic and inorganic contaminants that can be discharged downstream or enter groundwaters (Bagchi, 1994; Flynn et al., 1984; Yahaya et al., 2016). This has been observed in monsoon climates and following river flooding, resulting in increased release of leachates, soluble metals, and landfill gas (e.g., Schneider et al., 2017). Conversely, in coastal environments, increases in salinity may inhibit anaerobic bacteria and slow waste decomposition and gas production (Alkaabi et al., 2009; Ogata et al., 2016) although the solubility of metals may increase with seawater inundation (Brand & Spencer, 2020).

Changing hydrological cycles may mobilize contaminants and microorganisms, and alter surface and groundwater interactions, potentially creating new or removing existing contaminant pathways, potentially bypassing normal attenuation processes, and extreme flows can increase contaminant dispersion, but may also increase contaminant dilution improving water quality (Bloomfield et al., 2006; Goderniaux et al., 2009; Lipczynska-Kochany, 2018; Musacchio et al., 2021; Visser et al., 2012).

If the landfill becomes saturated this increases waste instability and pore pressure, which can damage the structural integrity of any landfill capping, increasing the likelihood of erosion (Flynn et al., 1984; Green et al., 2011; Schneider et al., 2017). The erosion of landfills, and the subsequent release of solid waste materials, is likely to increase in regions that are projected to realize an increase in the frequency and magnitude of extreme fluvial flood events (IPCC, 2013). However, the likelihood and magnitude of erosion will depend on the combination of hydrological and geomorphic conditions in the river channel/floodplain, the presence of any landfill engineering, and the mechanical response of the waste to structural failure of the capping materials. For example, in Canada, long-duration, moderate-magnitude flood events combined with a sandy substrate were predicted to present the greatest erosion risk (Curtis & Whitney, 2003).

2.2.2 | Increased drought

In many parts of the world, there is anticipated to be a reduction in available surface and groundwater freshwater resources as a result of changes in precipitation patterns, increased evapotranspiration, and increased abstraction for irrigation and drinking water (Bloomfield et al., 2006; Green et al., 2011; Kløve et al., 2014; Lipczynska-Kochany, 2018). This may result in changes to surface and groundwater interactions, reduced leachate production, and reduced dilution of leachates that reach surface and groundwaters due to reductions in surface water runoff and baseflow (Green, 2016; Murphy et al., 2018). In some instances, surface dispersal of landfill contaminants leached into watercourses may decrease as flow rates reduce, as increased residence times allow aquatic plants to utilize nutrients and contaminated sediments settle out, but the sediments may then become a secondary source of pollution (Delpla et al., 2009; Murdoch et al., 2000). Anticipated changes in the seasonality and rate of recharge, increased abstraction to replace surface water supplies, changes in sea level and the water table level could all result in changes in groundwater flow that may change the dispersal rate, distance, or direction of leached contaminants (Green et al., 2011; Larocque et al., 2019; Retter et al., 2021; Visser et al., 2012). Drying of flood embankments increases the likelihood of them cracking and then failing in future floods, increasing the risk of landfills being inundated (Sentenac et al., 2013). Drying of landfill capping materials increases the likelihood of them cracking and opening up new pathways for leachates and gases to escape and surface waters to enter (Bloomfield et al., 2006). It also risks damaging the landfill's structural integrity, increasing the likelihood of it eroding and releasing solid waste materials (Umana et al., 2016). The likelihood of this will vary depending upon the materials used in the construction of the landfills and the hydrological and geomorphic conditions where they are located.

2.2.3 | Compounding factors

The consequences of climate change effects on receptors will depend upon their susceptibility to change; additional contaminant inputs from landfills alone may be within receptors' buffering capacities, but combined with other climate change impacts critical thresholds or "tipping points" may be reached, where receptors are significantly affected (Barnard et al., 2021; Murdoch et al., 2000). Hence, here we give an overview of some of the other surface and groundwater quality stressors that additional contaminant inputs from landfills may exacerbate, although the interactions are extremely complex and are likely to vary geographically, with water quality potentially being degraded in some locations and improved in others depending on the initial conditions and rate of climate change (Murdoch et al., 2000).

Increases in temperatures are likely to result in decreases in soil organic matter and increased dissolved organic matter as biodegradation increases, as well as desorption of metals and organic contaminants from humic substances as a result of changes to microbial transformations, which may increase contaminant concentrations in watercourses when the land floods (Bloomfield et al., 2006; Delpla et al., 2009; Lipczynska-Kochany, 2018). Additionally, increased surface water temperature is anticipated to increase eutrophication by increasing the growing season and biological activity where there is sufficient oxygen and nutrients, but where oxygen is insufficient anoxia could occur (Murdoch et al., 2000). Increased biological activity and chemical transformations may increase bioaccumulation of toxins in flora and fauna, improving water quality, but potentially adversely affecting the food web (Murdoch et al., 2000; Retter et al., 2021). Decreased residence times during high flows will decrease chemical and biological transformations, which may offset the benefits to water quality of dilution due to increased volume (Murdoch et al., 2000).

Reductions in river flows during drought periods are likely to result in reduced water quality where wastewater effluents are subject to reduced dilution (Kløve et al., 2014). However, increased residence times may allow aquatic plants to utilize nutrients and contaminated sediments to settle out improving water quality, but the sediments may then become a secondary source of pollution and increased surface water temperatures will increase cycling of contaminants between sediments and the water column (Delpla et al., 2009; Murdoch et al., 2000). Crack formation in soils because of drought may create pathways for contaminants such as pesticides to enter groundwater (Bloomfield et al., 2006). Drying of soils can also create hard surfaces, which results in faster surface water runoff when the drought ends, and drying and rewetting cycles increase decomposition of organic matter and can cause pulses of high concentrations of nutrients and contaminants in runoff into watercourses (Bloomfield et al., 2006; Delpla et al., 2009; Murdoch et al., 2000; Whitehead et al., 2009).

Groundwater quality is anticipated to alter due to changes in the physical and chemical properties of groundwater and subsurface biogeochemical reactions, and changes in hydrogeological connections to contaminated areas. These changes result from increased subsurface temperatures, saline intrusion in coastal areas, and changes in spatio-temporal precipitation patterns (Green et al., 2011; Retter et al., 2021). Microbiome and fauna communities in groundwater are at risk from increased biological agent (e.g., viruses and pathogens), nutrient, and chemical inputs due to agriculture and wastewater discharge, potentially impacting groundwater food webs (Delpla et al., 2009; Retter et al., 2021). It is difficult to quantify how climate change may affect these stressors, but microbiome and fauna communities in aquifers are anticipated to change in composition which will affect their ecosystem functions, such as nitrogen and carbon cycling, and nutrient loads are expected to increase, which in some regions may exacerbate existing issues with groundwater nitrate concentrations exceeding drinking water limits (Delpla et al., 2009; Retter et al., 2021; Stuart et al., 2011).

These potential adverse changes to water quality may affect drinking water abstraction, potentially requiring increased water quality monitoring and modification to treatment infrastructure leading to increased production costs or the loss of abstraction sites (Delpla et al., 2009; Lipczynska-Kochany, 2018). Any additional contaminant and nutrient inputs from historic landfills resulting from climate change effects may further increase the treatment costs and the risk of abstraction sites becoming unviable, and increase the adverse impacts upon flora and fauna, potentially resulting in tipping points being reached.

3 | ASSESSING THE VULNERABILITY OF SURFACE AND GROUNDWATERS AND DESIGNATED NATURE RESERVES

While there have been some national scale assessments of potential contaminant emissions resulting from historic landfills flooding or eroding (e.g. Laner et al., 2009; Neuhold, 2013) such assessments rarely consider the likely receptors, yet this is essential to regional and national planning and adaptation to climate change (Arrighi et al., 2018; Brand et al., 2018; Brand & Spencer, 2018). Determining the scale of the potential risk of historic landfills to controlled waters and other sensitive ecological sites requires an understanding of the likelihood of hydrological connectivity between historic landfills and water bodies, the probability of erosion, the magnitude of the chemical hazard, and vulnerability of receptors. In our original article (Brand et al., 2018), we assessed the vulnerability of environmental receptors by considering their proximity to historic coastal landfills using England as a case study to demonstrate the scale of the issue. Here, we expand that assessment to consider the vulnerability of surface and groundwaters, and designated nature reserves, by also considering their proximity to England's inland historic landfills.

Receptor vulnerability has been assessed at a national scale for England using the Historic Landfill Sites National Dataset, comprising digitized boundaries of known historic landfills, a unique Historic Landfill Database Reference Number, and (where known) data such as site operator names, site addresses, waste types, and opening and closing dates (Environment Agency, 2021d). This dataset is regularly updated and at the time of writing, the Environment Agency had identified 19,717 historic landfills in England. Using ESRI ArcMap, this dataset was compared to the Environment Agency's Historic flood map (Environment Agency, 2021c), Flood map for planning (rivers and sea)—Flood Zone 3 (Environment Agency, 2021a), Flood map for planning (rivers and sea) areas benefiting from defences (Environment Agency, 2021b), Statutory main river map (Environment Agency, 2020b), Nitrate Vulnerable Zones (NVZ) 2021 designations map (Environment Agency, 2020a), Water Framework Directive groundwater bodies cycle 2 map (Environment Agency, 2020c), Source Protection Zones [merged] (Environment Agency, 2021e) and Natural England's Priority river habitat—rivers and Priority river habitat—headwater areas maps (Natural England, 2020a, 2020b). The National Soil Resources Institute's NATMAP Soilscapes map and accompanying guidance (Cranfield University, 2022; Cranfield University (NSRI), 2022) record where soils drain to surface water, groundwaters, or both, and were used to determine where leachates from historic landfills would drain to in this analysis. All datasets used in this review paper were correct on April 1, 2021, except the Flood map for planning (rivers and sea) areas benefiting from defences, which was correct on December 11, 2021, and NATMAP Soilscapes, which was correct on June 19, 2022.

To explore whether there is potential hydrological connectivity between landfill waste and surface/groundwaters, we identified how many landfills over-lapped with a water body or areas at flood risk, or where the landfill boundary was within a 25 m buffer of these water bodies/flood risk zones. A buffer of 25 m was included as rivers are only shown as center lines, historic landfill boundaries may not be exact (Environment Agency, 2015), and to allow for leachate plumes, although this is likely to be a conservative estimate of the plume size for many landfills. The water bodies examined included statutory main rivers (water courses England's Environment Agency has legal powers to do maintenance, improvement, and construction works on) and priority river (or headwater) habitat (naturally operating streams and rivers operating, free from significant anthropogenic impact) and Water Framework Directive (WFD) Groundwater Bodies. While areas at flood risk were considered by identifying Flood Zone 3, which is the Environment Agency's 'best estimate of the areas of land at risk of flooding, when the presence of flood defences are ignored and covers land with a 1 in 100 (1%) or greater chance of flooding each year from rivers; or with a 1 in 200 (0.5%) or greater chance of flooding each year from the sea' (Environment Agency, 2021a). This provides a measure of the number of sites at risk of inundation if defenses fail. Areas that had been historically flooded were also identified. However, historic flood maps do not capture all flood events, do not include surface water flooding, and generally do not record flooding that occurred before 1946. Therefore, this number is likely to be an underestimate.

In England, 2636 (13.4%) historic landfills are within 25 m of the center of a statutory main river, and 667 (3.4%) are within 25 m of the center of a priority river (or headwater) habitat, (NB these figures should not be summed as the datasets overlap). At least 6451 (32.7%) of England's historic landfills are in areas with soils that freely drain to inland surface waters or with naturally wet soils that drain to inland surface waters, and only 97 of these are recorded as having leachate controls in place. Flood Zone 3 contains 4767 (24.2%) of England's historic landfills, the vast majority of these (82.8%) are not currently protected by flood defenses. An additional 966 (4.9%) historic landfills are within 25 m of Flood Zone 3. Excluding those within the combined fluvial and tidal flood zone, 3774 (79.2%) of the 4767 historic landfills in Flood Zone 3 are in fluvial flood zones, suggesting fluvial flooding of historic landfills may pose a greater risk than coastal flooding. Within areas that are recorded as having historically flooded there are 2096 (10.6%) historic landfills and an additional 386 (2%) are within a 25 m buffer zone.

NVZs are areas that drain into surface and/or ground waters and are designated as being at risk from agricultural nitrate pollution (Defra, 2016). Despite 503 (83.8%) of England's NVZs containing at least one historic landfill, the risk/contribution from landfill leachates which may contain ammoniacal nitrogen is not considered in the designation process. In total, 10,756 (54.6%) of England's historic landfills are in NVZs and an additional 107 (0.5%) are within a 25 m buffer around NVZs. At least 90 (79.6%) of the 113 NVZs designated for groundwater protection have one or more historic landfills in areas with soils that freely drain to groundwater or with naturally wet soils that drain to groundwater. At least 212 (49.4%) of the 429 NVZs designated for surface water protection and 25 (43.1%) of the 58 NVZs designated as having eutrophic waters have one or more historic landfills in areas with soils that freely drain to inland surface waters or with naturally wet soils that drain to inland surface waters. Therefore, historic landfills could in the long-term pose an additional nitrogen risk, with potential impacts on drinking water abstraction and eutrophication of watercourses.

Also of concern, is the potential proximity of sites to groundwater bodies with 91.0% (17,951) of England's historic landfills in areas with Water Framework Directive (WFD) Groundwater Bodies, with an additional 0.2% (40) within a 25 m buffer. Virtually all (269 of 271 or 99.3%) WFD Groundwater Bodies in England are beneath at least one historic landfill. Groundwater sources that supply drinking water are of particular concern and 18.8% (3700) of England's historic landfills are in (groundwater) Source Protection Zone (SPZ) 3 (total catchment zone), only 47 of these are recorded as having leachate controls in place. There is at least one historic landfill in 50.8% (216 of 425) of the SPZ total catchment zones. While 25.4% (939) of the historic landfills in SPZ total catchment zones are believed to only contain inert waste, which is considered unlikely to have any negative effects upon the water environment, 50.1% (1855) contain industrial, commercial, household, special, and/or liquid sludge waste and would not be permitted in (groundwater) SPZs under current regulations to protect drinking water supplies. The remaining 24.5% (906) contain unknown waste types and, therefore, unknown risk to groundwater.

In practice, the vulnerability of groundwater and SPZs will depend on the soil type and depth separating them from the historic landfills, and the permeability of the underlying geology, with the risk of pollution being highest where soils are free-draining and the water table is shallow (Armstrong et al., 2004). Virtually all (263 of 271 or 97.0%) WFD Groundwater Bodies in England are beneath at least one historic landfill that is in an area with soils that freely drain to groundwater, with an additional 1.5% (4 of 271) are in areas with naturally wet soils that drain to groundwater. With 39.3% (167 of 425) of the SPZ total catchment zones containing historic landfills with industrial, commercial, household, special, liquid sludge, and/or unknown waste in areas that have soils that freely drain to groundwater or have naturally wet soils that drain to groundwater, and a further 4.0% (17 of 425) with historic landfills only containing inert waste, on the same soil types. Furthermore, it is likely there are additional historic landfill sites created from disused quarries where the bottom of the waste is below the water table, potentially providing a direct hydrological pathway for leaching contaminants depending on the permeability of the underlying geology (Abiriga et al., 2021).

Further analysis was undertaken using ESRI ArcMap to determine the proximity of historic landfills to other sensitive environmental receptors that may be affected by leachate plumes (specifically areas legally designated as nature reserves; Tables 1 and 2). Over 10% of England's historic landfills are in or within 25 m of designated nature reserves. Over 16% of England's Special Protection Areas (SPAs) are on or within 25 m of historic landfills as are 22% of England's local nature reserves. However, it should be noted that the high numbers of nature reserves in close proximity to historic landfills is partly because the creation of habitat and recreational areas has frequently been implemented on historic landfill as a low-cost, sustainable management strategy (Bardos et al., 2020).

Given the understanding that historic landfills will continue to release leachate over long timescales, this preliminary analysis suggests there is significant potential for contaminants from historic landfills in England to reach designated ecological sites, surface, and particularly ground waters through hydrological pathways. While it is typically not possible to directly link water quality deterioration in water bodies to historic landfills due to the diffuse nature of the pollution source and the wide-ranging chemical contaminants present, it is likely that historic landfills will impact sensitive environmental receptors, including surface and groundwaters. This potential will increase if projected increases in flooding frequency and extent due to climate change transpire. There are thousands of undefended historic landfills on floodplains, and if anticipated climate change driven increases in flood frequency and magnitude occur this will also increase the risk of them eroding and releasing solid waste into water bodies. To fully assess the pollution risk would require data that are not currently available, for example, susceptibility and buffering capacity of the receptors, erosion rates, and

TABLE 1 Number of historic landfills in or within 25 m of designated nature reserves in England.

Site type	Number of landfills in sensitive sites	Number of landfills in or within 25 m of sensitive sites
Local Nature Reserve[a]	562 (2.9%)	811 (4.1%)
National Nature Reserve[b]	49 (0.2%)	82 (0.4%)
Ramsar[c]	224 (1.1%)	346 (1.8%)
Special Areas of Conservation (SAC)[d]	223 (1.1%)	338 (1.7%)
Special Protection Area (SPA)[e]	320 (1.6%)	491 (2.5%)
Site of Special Scientific Interest (SSSI)[f]	884 (4.5%)	1318 (6.7%)
Any of the above[g]	1381 (7.0%)	2033 (10.3%)

Note: Percentages based on England having 19,717 historic landfills.
[a]Natural England (2021a).
[b]Natural England (2021b).
[c]Natural England (2020c).
[d]Natural England (2020d).
[e]Natural England (2021d).
[f]Natural England (2021c).
[g]Note some sites have multiple designations.

TABLE 2 Number of designated nature reserves on or within 25 m of historic landfills in England.

Site type	Number of sites on historic landfills	Number of sites on or within 25 m of historic landfills
Local Nature Reserve	408 (16.1%)	559 (22.0%)
National Nature Reserve	40 (4.3%)	63 (6.8%)
Ramsar	104 (8.1%)	145 (11.2%)
Special Areas of Conservation (SAC)	116 (6.1%)	159 (8.3%)
Special Protection Area (SPA)	146 (12.2%)	194 (16.2%)
Site of Special Scientific Interest (SSSI)	589 (6.2%)	817 (8.5%)

contaminant loads (Brand & Spencer, 2019; Murdoch et al., 2000). Furthermore, the presence of thousands of historic landfills in SPZs, which would not be permitted for modern landfills under current regulations, suggests further investigation is warranted into the potential future impacts on drinking water supplies as climate change effects alter the release and dispersion of contaminants.

The issue demonstrated here of surface and groundwaters becoming increasingly vulnerable to contamination from historic landfills is likely to be broadly representative of other countries, although the magnitude of risk will vary depending upon the types and vulnerability of receptors and their proximity to historic landfill sites, which will influence contaminant concentrations reaching them. Although the density of surface waterbodies varies significantly between countries, humans have historically settled where there is easy access to surface waters (Fang & Jawitz, 2019), and it was historically common practice globally to use floodplains as landfills with minimal/no leachate management or record keeping (Brand et al., 2018; Fang & Jawitz, 2019). However, countries with larger catchments and waterbodies may have reduced risk because of increased dilution (Macklin et al., 2023). In recent decades, populations have moved away from surface waters as infrastructure projects have allowed easier access to groundwater and pumping of water supplies over large distances (Fang & Jawitz, 2019), but few countries have defined buffering distances between landfills and abstraction points (Ya et al., 2019). Globally there have already been many reports of contaminants from landfills being detected in aquifers (e.g., Abiriga et al., 2021; Han et al., 2016; Ludvigsen et al., 1999), for example, in the United States, approximately 75% of landfills have polluted adjacent waterbodies (Xiang et al., 2019).

4 | PRIORITIZING RESOURCES FOR REMEDIATION OF HISTORIC LANDFILLS

Mitigation options for managing the risk of pollution from historic landfills include removing the source of the risk, for example, excavation and relocation of waste, breaking pathways to receptors, for example, construction of flood defenses, or removing the receptors, for example, excluding people from contaminated sites (Cooper et al., 2013). All of these are likely to be prohibitively expensive for managing large numbers of historic landfill sites, and in most cases, removing the receptor is unlikely to be practicable when managing the risk to waterbodies and designated ecological sites. There is increasing interest in landfill mining, where scarce resources are recovered from historic landfills for reuse. As yet, few sites have been identified as having the potential to be profitable; however, even where it is not profitable, landfill mining may allow the offsetting of some landfill remediation costs (e.g. Ford et al., 2013; Wagland et al., 2019; Winterstetter et al., 2015).

In the original article, we proposed that the large numbers of historic coastal landfills that pose a pollution risk, the limited resources available to investigate them, and the prohibitively high costs of investigation and mitigation, means there is a requirement for a method to prioritize expenditure based on which historic coastal landfills present the greatest risk (Alaska Department of Environmental Conservation, 2008; Brand et al., 2018; Weber et al., 2011). The "Risk screening assessment for ranking historic coastal landfills by pollution risk" proposed by Brand and Spencer (2018) meets this need. However, data presented here suggests there is also a requirement for a method to prioritize expenditure based on which historic landfills pose the greatest pollution risk in freshwater environments. Brand and Spencer (2018) found existing methods for prioritizing landfill sites by risk in freshwater environments typically concentrate on the risk from leachates or gasses under routine operating conditions, i.e. the landfills are not inundated or eroding. Where flooding is considered as a pathway, often the sensitivity of receptors is not considered, and none of the methods identified consider the risk from solid waste materials eroding in freshwater environments. Some methods designed for prioritization of coastal landfills consider flooding, erosion of solid waste, and receptors, but use some parameters that are not applicable to noncoastal sites (e.g., Alaska Department of Environmental Conservation, 2015; Brand & Spencer, 2018). None of these methods intrinsically consider how the risk may change under different climate change scenarios, although some parameters could be varied to test these without needing to change the underlying methodology, for example, including additional landfills in the assessment by increasing the flood extents.

Hence, to support resource management, there is a need to adapt an existing method or develop a new approach to risk assess landfills in freshwater as well as coastal environments, which considers surface and groundwater flooding and erosion as contaminant pathways in addition to leachate movement, and considers potential climate change effects. In the interim, prioritization of landfills by risk could be provisionally assessed using different existing proposed methods for different receptor types, although not all pathways are considered by existing methods. Inland, given global concerns about water security (Vörösmarty et al., 2010), the risk to (groundwater) Source Protection Zones from historic landfills is of particular concern, and Singh et al. (2009) proposed a prioritization method that can be used to assess risk to drinking water abstraction points. In the coastal environment, contamination from eroding waste is likely to pose the greatest pollution risk and Brand and Spencer (2018) proposed a prioritization method that can be used to prioritize historic coastal landfills.

To optimize existing and future risk assessment methods would require improved data relating to contaminants, especially those of emerging concern, and their release, mobilization, and dispersion in surface and groundwaters, as well as the susceptibility and buffering capacity of receptors, and erosion rates of landfills in freshwater and coastal environments.

5 | CONCLUSION

Data regarding wastes received and/or contamination levels are often limited for historic landfills, but they can be long-term sources of metals, microplastics, ammonia, and emerging contaminants to ground and surface waters, with the primary pathways being leachate movement and the erosion of contaminated solid waste materials. In England alone, nearly 4000 historic landfills fall within areas where there is greater than 1% annual probability of fluvial flooding and/or 0.5% greater annual probability of flooding from the sea and do not have flood defenses. The majority of these are on fluvial floodplains, where dilution is likely to be lower than coastal floodplains, suggesting fluvial flooding of landfills may be a greater pollution risk than coastal flooding. Nearly 11,000 historic landfills fall within NVZs, but despite the potential for landfill leachate to contain ammoniacal nitrogen, historic landfills are not considered in the NVZ designation process. Around 3700 historic landfills are in (groundwater) Source Protection Zones and the majority of these would not be permitted in these locations under current regulations to protect drinking water supplies. This is unlikely to pose a public health risk due to the extensive regulatory testing drinking water undergoes, but may increase water monitoring/treatment costs or make some sources unviable in the future. This is of particular concern as groundwater sources are expected to become increasingly important resources as climate change increases the frequency of droughts in some areas. These issues are likely to be broadly representative of other countries as globally it was historically common practice to construct landfills near surface and ground waters and on floodplains with minimal/no leachate management.

There is clearly already a huge risk of historic landfills causing a significant deterioration in ecological health of surface and ground waters and, subject to regional variations, this risk is likely to increase with climate change. Where climate change is anticipated to increase the frequency and magnitude of flood events, this will increase leachate generation from landfills already in floodplains and is likely to expand floodplains to reach additional landfills. Associated increases in fluvial flow velocities will increase the probability of landfills eroding and releasing solid waste materials. Where anticipated hotter, drier summers manifest it will increase the chances of landfill capping materials cracking, opening up new pathways for leachates and gases, and increasing the risk of solid waste materials eroding. The possibility of landfills eroding in freshwater environments and the subsequent impacts on freshwater bodies has received little attention until now, but there is evidence there could be significant adverse effects on sediment dwelling organisms, particularly when compounded by other climate change impacts upon flora and fauna. This is an area that warrants further research.

Although the risks associated with historic coastal landfills due to climate change have recently been gaining recognition within regulatory authorities, there is still a general lack of recognition of the risks associated with historic landfills in freshwater environments due to climate change. Consequently, there is a lack of coordinated effort to fully understand and mitigate the risk. Given the potential magnitude of the issue, lack of understanding of what the sites contain, high costs of remediation, and limited funding available to manage them, it is proposed a new method for prioritizing the sites by risk is required, along with guidance on the remediation strategies available to site managers.

AUTHOR CONTRIBUTIONS

James H. Brand: Conceptualization (equal); data curation (lead); formal analysis (lead); writing – original draft (lead); writing – review and editing (equal). **Kate L. Spencer:** Conceptualization (equal); data curation (supporting); formal analysis (supporting); writing – original draft (supporting); writing – review and editing (equal).

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CONFLICT OF INTEREST STATEMENT

James H. Brand now works for the Environment Agency. The Environment Agency had no involvement in the writing of this paper.

DATA AVAILABILITY STATEMENT

Data sharing is not applicable to this article as no new data were created or analyzed in this study.

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RELATED WIREs ARTICLE

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FURTHER READING

Interactive historic landfill maps for England are available at https://jameshbrand.com/landfill-maps/.

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