Legacy iron and steel wastes in the UK: Extent, resource potential, and management futures

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ABSTRACT

The iron and steel industry has a long tradition of bulk reuse of slags for a range of construction applications. Growing interest in recent years has seen slag resource recovery options extend to critical raw material recovery and atmospheric carbon capture. Full scale deployment of such technologies is currently limited in part by absent or partial inventories of slag deposit locations, data on composition, and volume estimates in many jurisdictions. This paper integrates a range of spatial information to compile a database of iron and steel slag deposits in mainland United Kingdom (UK) for the first time and evaluate the associated resource potential. Over 190 million tonnes of legacy iron and steel slag are present across current and former iron and steel working regions of the UK, with particular concentrations in the north west and north east of England, and central Scotland. While significant potential stockpiles of blast furnace and basic oxygen furnace slag could provide up to 0.9 million tonnes of vanadium and a cumulative carbon dioxide capture potential of 57–138 million tonnes, major management challenges for resource recovery are apparent. Over one third are located in close proximity to designated conservation areas which may limit resource recovery. Furthermore, land use analyses show that many of the sites have already been redeveloped for housing (nearly 30% urban cover). Deposits from recent decades in current or recently closed steel-working areas may have the greatest potential for resource recovery where such ambitions could be coupled with site restoration and regeneration efforts.

1. Introduction

Iron and steel making slags are by-products of the production of crude (pig) iron and steel. In the region of 320 to 384 million tonnes (Mt) of blast furnace slags (associated with crude iron production) and between 190 and 280 Mt. of steelmaking slags were produced globally in 2019 (USGS, 2020a). These slags have long been a valuable commodity for construction industries, but emerging resource recovery applications and potential environmental liabilities have been the focus of growing research interest (e.g. Piatak et al., 2015; Gomes et al., 2016).

As a country to experience early industrialisation, the United Kingdom (UK) has a rich history of iron and steel production. Industrial scale iron production began with small crude iron forges of the 15th

century, and later grew to be dominated by the large blast furnaces and Bessemer converters of the 18th and 19th centuries, when UK output accounted for almost half of global iron production (Carr and Taplin, 1963). With increased overseas competition from the middle of the 20th Century, many works were operating at a loss by the 1970s, and widespread closures during the 1980s resulted in a sharp decline in primary iron and steel production to the present day (Goldring and Juckes, 2001).

A result of this long and diverse history is that a significant legacy of iron and steel slag has been disposed of in inland and coastal settings, which in many cases pre-dates the era of strict environmental and waste management legislation (Mayes et al., 2008). As such, records of quantities and composition of disposed slag are sporadic at best (e.g. Harber and Forth, 2001). Using steel production data and typical slag

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mass ratios, Renforth et al. (2011) estimates that between 490 and 640 million tonnes of slag may have been generated in the UK since 1875.

Given that iron and steel slags, which are primarily composed of calcium-silicate phases (Piatak et al., 2015), produce hyperalkaline, oxyanion-rich leachates, concerns have been raised around the effects of waste heaps on the surrounding aquatic environment (Roadcap et al., 2005; Mayes et al., 2008; Hobson et al., 2018). Fugitive dust generation may also result in environmental issues during groundworks at slag disposal areas (Gomes et al., 2016). While such environmental legacies can persist on multi-decadal scales (Riley and Mayes, 2015), iron and steel wastes can also provide possible opportunities for recovering mineral and non-mineral resources. The iron and steel industry has long been an innovator in waste re-use, recycling and other minimisation efforts that would align with modern-day moves towards circular economy thinking (Deutz et al., 2017; Branca et al., 2020). Iron-making slags have been used in road construction since antiquity, for example at the Roman site of Airconium (circa 200C.E.) in Herefordshire, UK (Lee, 1974). Widespread after-use of both blast furnace (BF) slags from crude iron production and basic oxygen furnace (BOF) steel slags developed during peak industrial production in Europe in the late 19th and early 20th Century. Common after-uses included: applications as aggregate in roads, railway ballast and coastal defences for BF slag in particular, and BOF slag after weathering. In addition, BF slag has been, and continues to be, used as a cement substitute in construction applications. In some instances, slags were used in land reclamation efforts, whereby disposal in the tidal zone allowed for seaward expansion of usable land (e.g. at Millom, Cumbria, where reclaimed land was used for mineral railway infrastructure). Slag has also been used in a range of environmental settings such as water treatment filter beds or as an alternative agricultural lime (Das et al., 2019; Lee, 1974; Naidu et al., 2020). These bulk reuses mean that the volumes of modern production slags that are destined for disposal (i.e. landfill) are relatively modest (Branca et al., 2020). However, older production slags generally showed greater variability in composition (Juckes, 2003) and were produced in far greater quantities, primarily due to the low iron content of most UK iron ores (typically 24–30% Fe for bedded phosphoric ironstones of Jurassic and Cretaceous age), compared to imported Fe ores which became dominant from the 1960s (typically 60–70% Fe: Goldring and Juckes, 2001). As such, significant legacy deposits from mid-20th Century workings are apparent in many former iron and steel making districts of the UK (Lee, 1974).

In addition to the established bulk after uses for iron and steel by-products, there has been growing interest in value recovery from steel-making slags in recent years that has extended to incorporate 'Critical Raw Materials' (CRM) and strategically-important elements (e.g. chromium, vanadium, phosphorus, and rare earth elements (REEs)) that can be present in slags in significant quantities (Morita et al., 2002; Ye et al., 2003; Lindvall et al., 2010; Lin et al., 2014; Abhilash et al., 2017). Such interest has largely been focused on laboratory scale recovery tests and have been particularly prominent in the European Union where security of supply issues for CRM have been of strategic importance and investment (e.g. Gomes et al., 2018; Branca et al., 2020). Some studies suggest that it is possible to reuse the slag as secondary aggregate, following the recovery of CRMs (Ye et al., 2003; Naidu et al., 2020; Gomes et al., 2019a).

While studies on resource recovery from legacy slags have been limited, a number of established technologies have been applied to other anthropogenic wastes, such as solvent extraction for REE recovery (Tunsu et al., 2019) and pressurised acidic leaching for vanadium recovery from mine tailings (Zhang et al., 2019). However, the relatively low target metal concentrations within legacy wastes limit the economic feasibility of ex-situ processing (Sapsford et al., 2017). As such, a number of in-situ methods typically used in mining processes have been explored for resource recovery from waste stockpiles, including in-situ leaching (ISL), heap leaching (Petersen, 2016), and bioremediation (e.g. phytoremediation (Mahar et al., 2016)). A key benefit of in-situ

recovery techniques for legacy waste management lies in the minimal disturbance to overlying habitats (Seredkin et al., 2016), though any intervention must be effectively managed to negate potential adverse outcomes (for example, contamination of groundwater by chemical lixiviants).

In parallel the potential benefits of slag weathering for atmospheric carbon dioxide capture have been proposed (Huijgen et al., 2005). The weathering of calcium silicates and oxides in slag generates alkaline (hydroxide-rich) waters which react with atmospheric CO_2 to form stable carbonate minerals (Pullin et al., 2019). Laboratory studies have demonstrated the efficacy of this carbon capture process in a range of conditions, while field studies have shown passive carbonation apparent in soils and drainage waters at slag disposal sites (Renforth et al., 2009; Mayes et al., 2018). Global estimates suggest that steel-making slags could contribute up to 500 Mt. per year of CO_2 removal, which would represent a significant negative emissions technology (Mayes et al., 2018; Renforth, 2019).

The aim of this paper is to assess the extent and resource potential of legacy iron and steel wastes in Great Britain. Through compilation of a range of physico-chemical and land use databases, the potential future benefits and management challenges for resource recovery at legacy iron and steel waste sites are evaluated. This is timely given (1) the recent technological developments in valorisation of slags to encompass bulk reuse alongside CRM recovery and carbon capture, and (2) the generally poor inventories of legacy waste composition and volume in many jurisdictions (Renforth, 2019; Blasenbauer et al., 2020). Quantification of anthropogenic stocks of legacy waste materials through substance flow analysis is a key aspect of circular economy thinking, and provides the means to effectively guide future waste management efforts (Stanisavljevic and Brunner, 2014).

2. Methods

2.1. Legacy waste identification

Given the absent or unclear documentation of iron and steelmaking slag production and deposition in the early 20th century, location of legacy waste stockpiles was completed through a multifaceted approach to maximise site identification. In the first instance, two extensive databases of historical landfill sites in England and Wales were combined to provide the locations of over 21,000 historical landfills (Environment Agency, 2020; Natural Resources Wales, 2020a). As the historical landfill site databases had the same schema, a direct merge was performed using the 'Merge' tool of ArcMap 10.7.1. Given that the database covered industrial, commercial, and household landfill sites, entries were filtered to identify those most likely to contain iron and steel slag. Only those entries where the landfill license holder name included the term 'steel' or 'iron', or derivations thereof (e.g. steel*works*, iron*works*) were selected. Data filtration was achieved using the 'select by attributes' feature of ArcMap 10.7.1 using SQL query expressions detailed in Table 1.

Presence of slag was assessed through inspection of historical Ordnance Survey (OS) maps and records (Digimap, 2020), through which process 60 were confirmed as iron or steel slag deposits. While this method is likely an underestimate of the entries which contain slag at all, it ensured that those identified consisted primarily of slag, and were not, for example, co-deposited with demolition waste during site redevelopment. For Scotland, where no historical landfill database was available, slag disposal sites were identified through a combination of historical maps and archival research.

Further to this, a geodatabase of historical and current iron and steel works was generated to identify regions where primary iron production occurred. Lists of iron and steel works and their period of operation were collected using online British industrial archaeology repositories (Historic England, 2020a; Historic Wales, 2020; Historic Environment Scotland, 2020; Grace's Guide, 2020), after which the specific locations

Table 1

Secondary spatial datasets used to generate the iron and steel slag database used in this work, and summary of data processing techniques.

Database	Publisher (Source)	Data Processing
Historic Landfill Sites	Environment Agency, Natural Resources Wales (data.gov.uk)	Datasets merged in ArcMap 10.7.1 ('Merge' tool). Filtered by license holder using SQL queries: - "INDUSTRIAL" = 'Yes' - LOWER("LIC_HOLD")LIKE'%steel%' - LOWER("LIC_HOLD")LIKE'%iron%' Resulting features manually validated (e.g. 'steel' present within surname 'Steeley' would be omitted if not also associated with steel works).
1:10,000 and 1:25,000 Scale Geology	British Geological Survey (Digimap)	Regions of 'artificial made ground' extracted within a 1.5 km buffer of former iron and steel works (verified as slag using historical OS maps).
OS Maps (1840s – contemporary)	Ordnance Survey (Digimap)	Used to verify slag heap presence and locations identified during previous procedures. Additional spoil heaps manually digitised in vicinity of former works (especially for Scotland).
OS Terrain 5 DTM	Ordnance Survey (Digimap)	Used to represent the current topography of slag disposal areas during volume calculation analysis.
Land Cover Map (LCM2015)	Centre for Ecology and Hydrology (Digimap)	'Spatial Join' used on slag disposal area shapefiles during land cover assessment.

were obtained through the use of historical OS maps. A buffer of 1.5 km was used around the location of former works to identify areas of 'artificial made ground' in superficial geology datasets (British Geological Survey, 2020), which given their proximity to foundries were potentially likely to contain iron and steel slag. Candidate areas of made ground were confirmed using historical OS maps to identify areas where active waste disposal was synchronous with iron and steel works operation and were cross-checked against other potential waste types (e.g. coal spoil). In instances where made ground was not identified, but OS maps indicated slag heaps were present, these areas were manually digitised from historical maps using ArcMap 10.7.1 and added to the composite shapefile layer prior to volume estimation. Combination of the aforementioned methods resulted in the identification and quantification of 112 discrete areas of slag disposal. A summary of the databases used to produce the final iron and steel slag database and the iron and steel works database is provided in Table 1 alongside a summary of data processing performed.

2.2. Volume estimation

After identifying areas of waste deposition, a 5 m resolution digital terrain model (DTM) was clipped to the boundary of the deposit and used to represent the modern surface topography. The method used to estimate the volume of material deposited varied depending upon the geographical location of wastes, and were distinct to coastal or inland settings. For coastal and estuarine settings, where wastes were heaped upon flat land (e.g. salt marsh, mud flats, beaches), a triangular irregular network (TIN) was generated using the clipped DTM, and the 'surface volume' tool of ArcMap 10.7.1 used to estimate volume using a constant baseline equal to the elevation of the surrounding land (typically 0–2 m). For inland deposits, where the underlying terrain was more complex, the natural ground surface was modelled through digitisation of contour lines and spot height measurements from OS maps dated prior to waste deposition. The difference in elevation between the two modelled surfaces (historical and modern) was used to calculate the volume of material deposited at each site using the 'Cut Fill' tool. Detailed examples of each approach, including assumptions and limitations are provided in Figs. S1 and S2 of the linked Supporting Information.

2.3. Land-use analysis and co-location with designated areas

Further to characterising the volume of slag present, the current land-use of identified slag deposits was characterised, and their locations analysed against areas of cultural and ecological designation (Table S2 of Supporting Information); both of these factors are essential

in influencing the resource recovery approach which may be taken. Using the composite shapefile dataset of legacy slag deposits as a mask, data were extracted from a raster dataset of UK-wide land cover (data collected by the USGS Landsat 5 Thematic Mapper) and summarised using the standard habitat types specified by the Centre for Ecology and Hydrology (2015). A summary of slag deposit locations in relation to areas of cultural and ecological designation was completed at two spatial extents; those directly co-located (as per Crane et al., 2017), and those within a 1 km buffer, given that developments in close proximity to certain designated areas may be subject to additional planning constraints.

2.4. Waste composition and resource projections

In order to estimate the composition of legacy slag deposits, a composite table was produced through a combination of a UK-focused literature search and results of previous sample analyses by the authors. Data are presented as oxide equivalent concentrations. Resource projections were calculated using the volumes of slag identified during spatial analyses, the recorded furnace types at each location, and the reported density ranges of BF (1150–1440 kg/m³) and BOF (1600–1760 kg/m³) slag (Lee, 1974). Discrimination between BF and BOF slag was possible for 76% of identified slag deposits through archival research, with slag type at remaining sites estimated based on metal production history. Electric Arc Furnace deposits account for a relatively small fraction of slags produced in the UK so are not included in the analysis. Using the mass of each slag type recorded (calculated using the densities above), carbon sequestration potential was calculated using slag carbonation ranges reported by Pullin et al. (2019) under three carbonation scenarios: direct carbonation (296–337 kg CO_2/t) enhanced weathering (422–481 kg CO_2/t), and passive carbonation (513–584 kg CO_2/t). During direct carbonation, dissolved CO_2 (as carbonic acid) is precipitated as solid carbonate minerals at a molar ratio of 1:1, whereas for enhanced weathering this ratio is doubled through formation of aqueous bicarbonate ions. The term passive carbonation relates to the long-term ambient weathering of slags, as previously described for large legacy slag landfills (e.g. Consett, UK; Pullin et al., 2019).

3. Results

3.1. Spatiotemporal extent and volume estimates

A total of 118 iron and steel works which were involved in primary metal production were identified across the UK, of which 70 had at least one associated slag deposit, either in close proximity as heaps, or
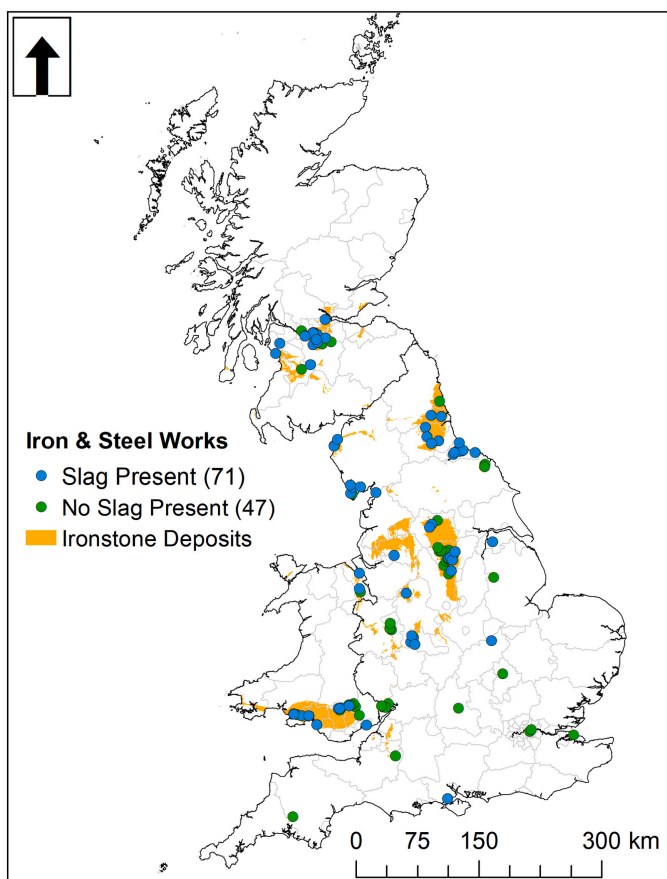
Fig. 1. Locations of historic and current iron and steel works and indication of slag presence presented alongside the location of ironstone bedrock deposits (geological data provided by the British Geological Survey).

Fig. 2. Calculated volumes (million m^3) of identified iron and steel slag deposits by county.

Fig. 3. Duration and intensity of former iron and steelworking operations within UK counties. Grey bars represent duration of a single works, with darker colours indicating a greater number of works active during that time period (lightest grey = 1, black = 16 works (maximum overlap in one region)).

as foundations for site expansion. Works tended to be more densely clustered around regions where ironstone deposits were present in bedrock geology (Fig. 1), resulting in hotspots of works around Lanarkshire, County Durham, South Yorkshire, and South Wales. Further to this, a cluster of works was located on the Furness peninsula (Cumbria), associated with the local high-grade hematite deposits extracted in the area.

Much of the total slag arisings are present in a relatively low number of counties, and counties with clusters of former works contained a higher volume of slag deposits. This was expected given that molten slag was often dumped in the proximity of works before solidifying, and therefore was rarely transported far for disposal. Cumbria, which contained the cluster of works on the Furness Peninsula and north towards Workington (Fig. 1), had the highest calculated volume of slag at 55.4 million m^3, which was in substantially greater than the second-most abundant county, North Lincolnshire, at 17.5 million m^3 (Fig. 2). The national distribution of slag deposits was such that the five most abundant counties accounted for approximately two-thirds of the total volume of slag calculated, with 80% of deposits accounted for within the top ten counties. The temporal distribution of iron and steelworking operations by region is provided in Fig. 3 and are testament to the long history of iron and steelworking in the UK, with large-scale companies forming in the early 17th century. It is important to note that while smaller industries predated those displayed in Fig. 3, output of iron (and therefore slag) was low, and poor records were kept of time of operation. By the turn of the 19th century, the iron and steel industry spread to a greater number of counties, and the intensity of the industry increased throughout the industrial revolution, evidenced by the increased number of active works during this time period. While not
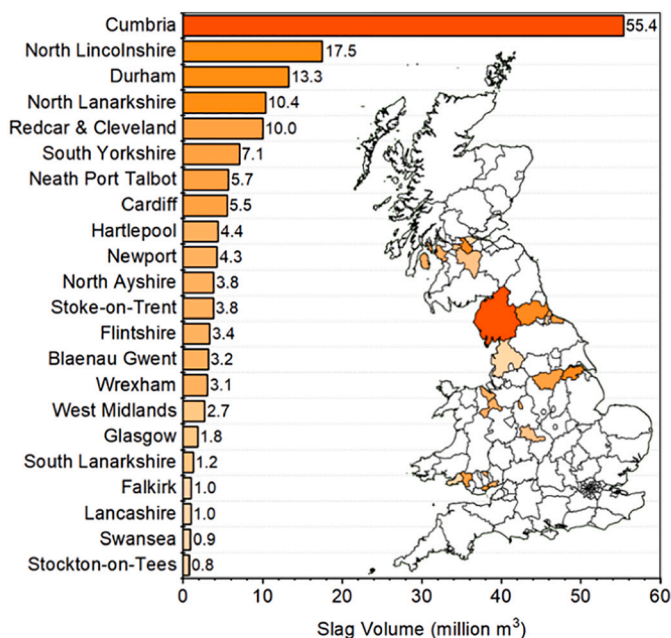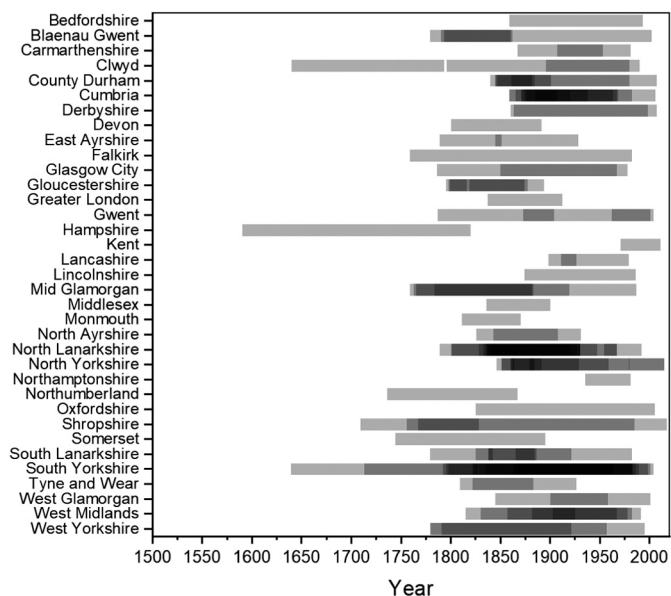
a direct measure for iron and steel output, it was apparent that the counties with a more intense industrial history tended to have higher volumes of slag deposits associated with them (Fig. 2), particularly Cumbria, North Lanarkshire, and South Yorkshire (Fig. 3).

In the context of legacy wastes, it is crucial to consider the current land use of the area given that this may be the ultimate control on whether resource recovery technologies can be implemented on-site. The results of land-cover assessment at legacy slag sites show that the majority of old heaps have since been repurposed for urban use, either for residential, industrial, or commercial purposes (Fig. 4). Combining the urban and suburban cover, over 38% of the area of legacy slag deposits is now used for this purpose and is the likely reason for the
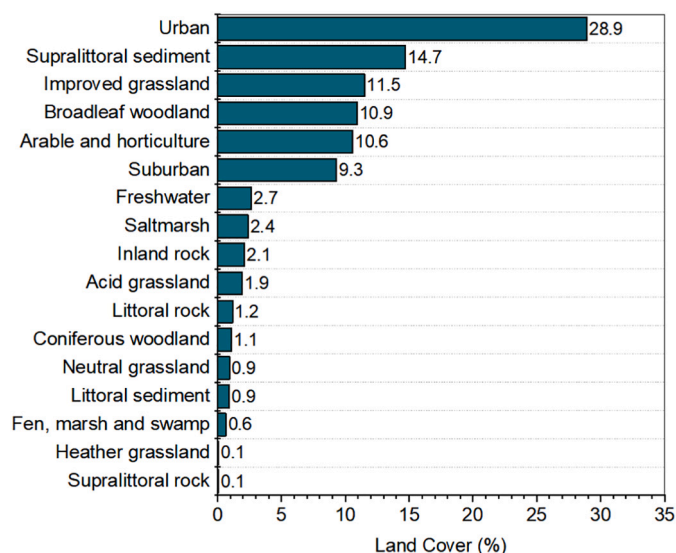
Fig. 4. Current land cover (%) of areas identified as containing iron and steel production slags. Land-use classifications from Centre for Ecology and Hydrology (2015).

lower than expected slag volumes calculated in areas with historically dense steel and iron industries, e.g. South Yorkshire, North Lanarkshire. Second to urban areas, a substantial percentage of slag disposal areas are situated within the littoral and supralittoral zone (total 16.8%), which when combined with saltmarsh habitat reveals that almost a fifth of identified slag disposal areas occur in coastal settings.

In addition to land use, the location of former slag disposal areas in relation to areas of environmental and cultural designations is an important factor in determining future management of the site. A summary of the spatial relationships between designated areas and slag deposits is presented in Table 2 for co-located sites and those within a 1 km buffer region. Slag deposits were directly co-located within all but one ecological designation area included in this study, and were most-present within Sites of Special Scientific Interest (SSSIs; 15 occurrences accounted for by 21 deposits). This is partly due to the extensive number and area of these sites within the UK (26,466 SSSIs, 23,731 km²; Table 2), but it is also the case that slag disposal areas provide opportunities for unique habitats to develop and later become designated because of this. Examples of such cases are presented in Table 3. Further to SSSIs, 8–9% of identified slag deposits were located entirely within other ecological conservation designations (Special Protected Areas, Special Areas of Conservation, and Ramsar Sites). Slag disposal sites were also co-located with areas designated for cultural purposes, with 2.7% of slag sites being located within Areas of Outstanding National Beauty (AONBs), and one slag site associated with a

World Heritage Site (Table 2).

3.2. Waste composition

In order to assess iron and steel slags as a resource, a UK-focused literature review of blast furnace (BF) and basic oxygen furnace (BOF) slags was completed to highlight likely compositions. The paucity of published data for UK slag composition was immediately apparent, with only 7 complete elemental analyses available for BF slag, and 16 for BOF. Despite this, it was possible to consider typical compositions of each. For both slag types, the majority of their composition was accounted for by CaO (both in the region of 40% by weight; Fig. 5). MgO content was fairly consistent between slag types, albeit with slightly lower ranges reported for BOF slags. Key differences in major constituents were the elevated proportion of SiO_2 in BF slags (~36 wt%) compared to BOF (~14 wt%), which appeared to be balanced by the elevated FeO occurrence within BOF slags (22–26 wt%), which was under 1 wt% within BF slags (Fig. 5).

Minor elemental composition of BF slag was typically dominated by SO_3, which ranged from negligible proportions to over 2 wt% by weight of slag. Proportions of other metals within the waste material were generally between 0.25 and 0.6 wt% for oxides of Mn, Ti, K, and Na, with a lower proportion of BaO reported (< 0.25 wt%). Minor constituents of BOF slag were more varied and included elements which are currently warranting attention in regards to recovery, notably P_2O_5 (~1 wt%), V_2O_5 (0.25–0.75 wt%), and Cr_2O_3, which extended to over 3 wt% in some BOF slags.

3.3. Resource projections

Slag volume estimates from spatial analyses were converted to mass estimates using densities of 1150–1140 kg/m³ for BOF and 1600–1760 kg/m³ for BF slag (Lee, 1974), which revealed a national inventory of between 191 and 236 Mt. of slag (Table 4). Coupling these calculated masses with compositional data (Fig. 5) and known slag type it was possible to estimate total resource size of key metals within UK slags. As expected given typical slag composition, Ca, Fe, Si, and Mg oxides were the most abundant resource with mid-range estimates of 85, 71, 25, and 18 Mt., respectively. Despite the limited V data available for BF slag composition, when considering the volume of BOF slag within the UK, between 0.36 and 1.55 Mt. of V_2O_5 is estimated to be present within UK slag wastes. Upper estimates for stockpiles of TiO_2 (1.58 Mt) and Cr_2O_3 (1.26 Mt) were also non-trivial, and of potential interest for resource recovery. Further to elemental resource recovery, iron and steelmaking slags are of value for their carbon sequestration value. Analysis of carbonation rates based upon projected mass/volume estimates indicated that through direct carbonation, UK slag deposits have capacity to sequester between 56.6 and 79.4 Mt. CO_2. Through enhanced weathering processes, this was estimated to potentially

Table 2

Extent of co-location between designated areas (DAs) and identified slag deposits (SSSI – Site of Special Scientific Interest, SPA = Special Protected Area, SAC = Special Area of Conservation, Ramsar = sites protected under the 1971 Convention on Wetlands, AONB = Area of Outstanding Natural Beauty (England/Wales), NSA = National Scenic Area (Scotland; equivalent to AONB), WHS = World Heritage Site, LB = Listed Buildings, * = point data).

Designation	Type	Total		DAs co-located with slag				DAs within 1 km of slag		Slag deposits within DAs	
		Count	Area (km²)	Count	%	Area (km²)	%	Count	Area (km²)	Count	%
SSSIs	Ecological	26,466	23,731	15	0.1	506	2.1	54	681	21	18.8
SPA	Ecological	259	37,388	4	1.5	1069	2.9	5	1136	10	8.9
SAC	Ecological	576	41,938	4	0.7	1510	3.6	8	2215	9	8.0
Ramsar	Ecological	152	7951	4	2.6	589	7.4	5	653	9	8.0
AONB/NSA	Cultural	80	34,364	2	2.5	2061	6	3	2184	3	2.7
National Park	Cultural	15	23,142	0	0	0	0	0	0	0	0
WHS	Cultural	24	869	1	4.2	449	51.7	2	466	1	0.9
LB	Cultural	12,446	NA	5	0.04	NA	NA	26	NA	1	0.9

Table 3

Examples of natural environment conservation designations co-located with slag disposal sites in the UK. Superscripts indicate geographic importance of designations: [1] = local; [2] = national, [3] = European, [4] = international. Information from Ash et al. (1994); Natural England (2020); Natural Resources Wales (2020b), Roberts (2019) and unpublished data of the authors.

Site, location	Designation	Site description and communities/species/features of interest
Brymbo Fossil Forest, Wrexham	SSSI[2]	Internationally important plant fossil assemblage in 14 m bed of exposed Upper Carboniferous Coal Measures. In situ fossil lycophytes and large stands of fossilised *Calimites* (Roberts, 2019)
Carnforth Slag Bank, Lancashire	Local Nature Reserve (LNR)[1], SAC[3], Ramsar[4]	Calcareous grassland on slag heaps which form a protective barrier to partially reclaimed landward saltmarsh. Lapwing (*Vanellus vanellus*), ringed plover (*Charadrius hiaticula*), oystercatcher (*Haematopus ostralegus*), marsh harrier (*Circus aeruginosus*).
Coatham Marsh, North Yorkshire	LNR[1], National Nature Reserve[2], SPA[3], Ramsar[4].	Reedbeds, open water and grassland on complex of slag heaps and medieval salt diggings. Northern marsh orchid (*Dactylorhiza purpurella*), bee orchid (*Ophyrs apifera*). Redshank (*Tringa tetanus*), reed warbler (*Acrocephalus scirpaceus*), reed bunting (*Emberiza schoeniclus*), water rail (*Rallus aquaticus*).
Kirkless Nature Reserve, Wigan, Lancashire	LNR[1]	Locally important calcareous grassland similar to coastal dune communities on former iron and steel slag (Ash et al., 1994). Common broomrape (*Orobanche minor*); southern marsh orchid (*Dactylorhiza praetemissa*).
Maryport Harbour, Cumbria	SSSI[2]	Ruderal plant communities on coal washings, spoil, and iron slag. Northerly most site in UK for purple or yarrow broomrape (*Orobanche purpurea*); pyramidal orchid (*Anacamptis pyramidalis*).
Millom Ironworks, Borwick Rails and Askham Slag Banks, Cumbria	LNR[1], SSSI[2], SPA[3], SAC[3], Ramsar[4].	Extensive slag banks on estuarine margin with calcareous grassland and ponds. Natterjack toad (*Epidalea calamita*). Orchid-rich grassland with northern marsh orchid (*Dactylorhiza purpurella*) and bee orchid (*Ophyrs apifera*). Skylark (*Alauda arvensis*).
Sawcliffe Hill / Crosby Warren, Lincolnshire	LNR[1]	Community woodland and calcareous grassland on steel slag. Common lizard (*Zootoca vivipara*). Grayling (*Hipparchia semele*).
Shotton Lagoons and Reedbeds, Flintshire, Wales	SSSI[2]	Reedbeds and lagoons formed in part by land reclamation for adjacent steel and industrial plants. Nationally scarce white mullein (*Verbascum lychnitis*) grows on slag heaps; variegated horsetail (*Equisetum variegatum*) present in swamp. Breeding population of common tern (*Sterna hirundo*).

increase to 137.6 Mt. CO_2. While yielding lower carbonation potentials, a passive carbonation approach has potential to remove up to 2.9 Mt. CO_2 (Table 5).

4. Discussion

4.1. Spatiotemporal patterns and land use

The total extent of iron and steel slag identified in Fig. 1 and Table 4 suggested that approximately 38% of iron and steelmaking slag remains in landfill, based on previous estimates of total slag production between 1875 and 2008 in the UK (Renforth et al., 2011). The regional extent of slag stockpiles is explained by a number of factors, including age of deposits, iron ore sources, industrial intensity, and reclamation efforts. UK iron ore was used in up to 50% of iron and steel production until around 1970, when technological advances in steelmaking, reduced import costs and ore discoveries led to the dominance of higher iron content imported ores (Goldring and Juckes, 2001). Slag production would have been relatively higher at sites using UK ores given the lower Fe content (~20 wt%) compared to imported material (~60 wt% Fe; Lee, 1974). Slag production peaked in the UK in 1967 at 11.7 million tonnes per year (Lee, 1974), with a consistent decline thereafter as a result of steel mill closures and improvements in process efficiency (Juckes, 2003). The effects of nationalisation during this period are also reflected in Fig. 3; evidenced by a reduced number of works in operation as smaller foundries were consolidated to larger production facilities capable of achieving higher output. Therefore, sites operating at peak capacity in the 1960s and 1970s are likely a key driver of volumes estimated in Fig. 2 (e.g. Millom, Barrow and Workington in Cumbria; Scunthorpe in North Lincolnshire; Ravenscraig in Scotland; Llanwern and Cardiff in South Wales: Goldring and Juckes, 2001; Fig. 3).

The low phosphorus content of BF slags associated with hematite ores (Juckes, 2002) may in part explain the extensive Cumbrian stockpiles. Hematite ore derived slags were prone to fragmentation (a process known as 'falling') which limited potential use as aggregate (Juckes, 2002; Yildirim and Prezzi, 2011). Phosphorus, along with magnesium (also present in low concentrations in Cumbrian slags due to distance from local sources of dolomite: BGS, 2006) is considered a stabiliser of dicalcium silicate, and low concentrations of Mg and P lead to the relatively higher CaO content and instability of Cumbrian slags (Juckes, 2002). Issues of water pollution when Cumbrian slags were reused as road ballast (e.g. on the A66 trunk road) have also been documented (e.g. Lamming, 1986) which may have limited reuse in these locations. At other coastal iron and steelworking locations, there was extensive reuse of slags in construction applications and land reclamation (Lee, 1974). At Redcar in northeast England, there was also widespread sea disposal of slag via boat ("for a shilling a tonne": Lee, 1974, p6), which limits the extent of onshore deposits in the national figures (Fig. 2). At some sites where slag was initially disposed of in opencast voids (associated with ironstone or coal extraction) determining volumes can be problematic, as is the case at Scunthorpe, Lincolnshire (Fig. 1). Here, in the region of 15 million tonnes is estimated to be disposed of within former opencast Jurassic ironstone workings, which as part of planning permission need to reach original contour with slag infill (Deutz et al., 2017). Such disposal settings may pose additional engineering constraints for potential slag reuse (e.g. sub-water table operations).

Extensive reworking and landscaping of slag disposal areas has been apparent at many UK steelworking districts closed in the latter half of the twentieth century. A range of former tip areas are now urbanised (38.2%; Fig. 4) with a range of features including housing estates and parks (e.g. Consett, County Durham; Ravenscraig, Scotland; Askham-in-Furness, Cumbria; Llanwern, Wales), industrial units and leisure facilities (e.g. racing circuits at Corby, Northamptonshire and Carnforth, Lancashire) as part of major redevelopment schemes (e.g. Hudson and Sadler, 1987; Stone, 2002). At major urban steelworks such as those in Sheffield, Rotherham, and Birmingham there are few slag heaps remaining given the demand for redevelopment of brownfield sites. Some sites are also used for renewables developments (e.g. onshore wind power at Maryport, Cumbria) which is a growing area of interest for marginal and brownfield sites in other jurisdictions (e.g. Milbrandt et al., 2014). Such redevelopment obviously limits scope for additional mineral resource recovery, but the viable land uses generated on slag disposal areas showcase a range of options for post-closure economic activities. Some bulk recovery of BF slag from disposal areas continues at a small number of sites such as Templeborough (South Yorkshire)
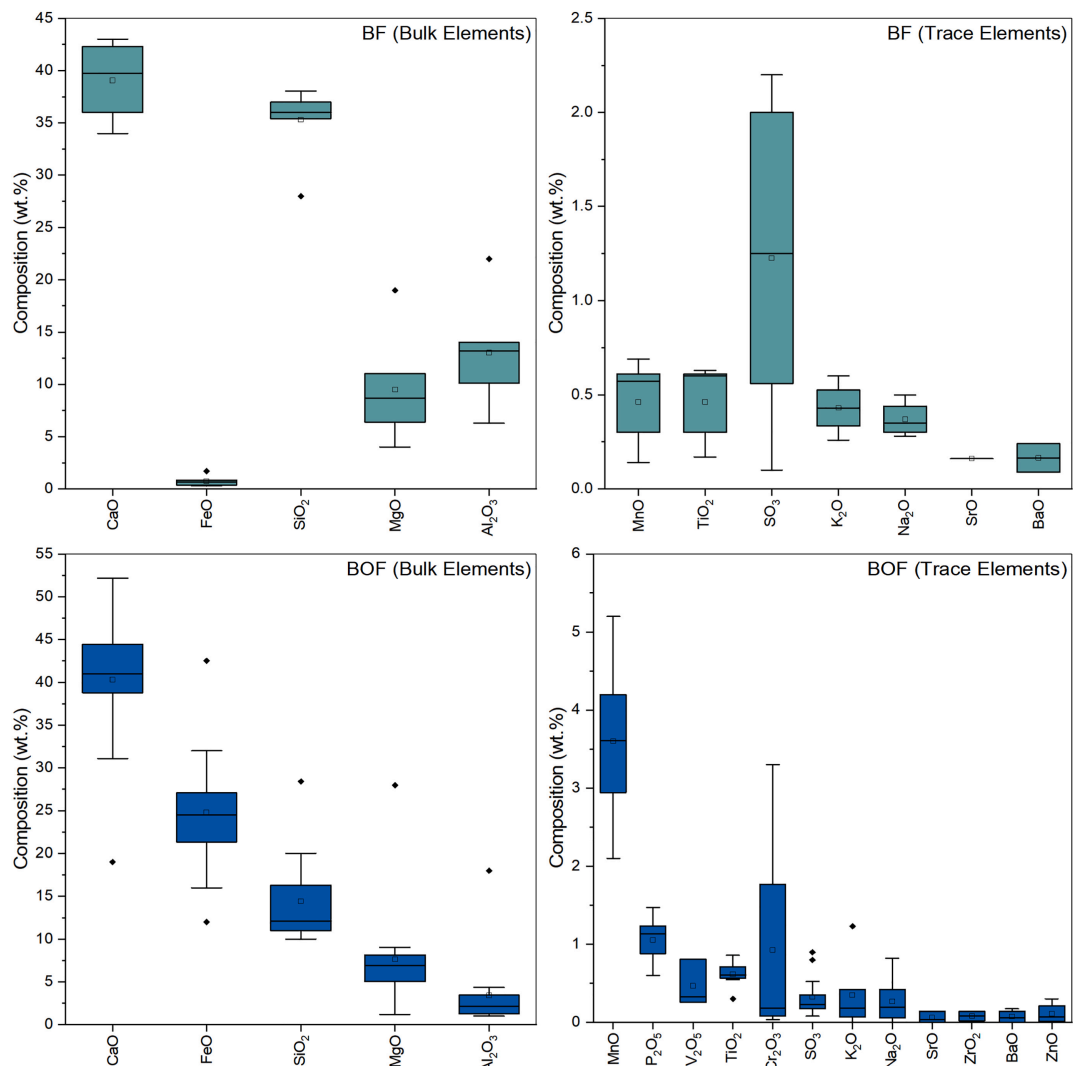
Fig. 5. Composition of UK blast furnace (BF, *number of observations = 7*) and basic oxygen furnace (BOF, *n = 16*) slags. Data collected through literature review and from unpublished data of the authors. Further detail of composition and regional breakdown provided in Table S1 in the Supporting Information.

and Barrow-in-Furness (Cumbria) for aggregate and cement applications. Land cover analysis indicated that of the 'unimproved' habitat types (i.e. not developed or managed), supralittoral sediment and broadleaf woodland were most prevalent (14.7% and 10.9%, respectively; Fig. 4). Such land cover may pose constraints to waste management options, for example erosion rates at coastal sites may prohibit major infrastructure, or established ecological communities within

broadleaf woodland may limit options for in-situ leaching through concerns related to disruption. Furthermore, it may be the case that slag deposits in the coastal zone provide defence benefits for sensitive habitats (e.g. saltmarsh, Fig. 4) or existing infrastructure, and any potential for resource recovery may be less of a priority. Such considerations are difficult to encapsulate during a national-scale study and would require detailed case-by-case exploration at site level before

Table 4
Masses of elements present within iron and steel slag deposits of the UK (converted from volume estimates (Fig. S1 and S2) using the review of UK slag type composition for slag heaps with calculable volumes (Table S1)).

	Slag	CaO	FeO	SiO_2	MgO	MnO	Al_2O_3	P_2O_5	V_2O_5	TiO_2	Cr_2O_3
	Mt	*Mt*	*Mt*	*Mt*	*Mt*	*Mt*	*Mt*	*Mt*	*Mt*	*Mt*	*Mt*
Low	191.1	61.3	49.1	10.7	7.0	3.5	0.2	0.8	0.4	0.4	0.5
Mid	213.4	85.1	70.6	25.3	18.1	7.5	2.4	2.0	1.3	0.9	0.7
High	235.6	103.8	87.0	50.8	47.2	15.0	4.8	2.8	1.6	1.6	1.3

	SO_3	K_2O	Na_2O	SrO	ZrO_2	BaO	NiO	CuO	ZnO	CoO	PbO
	Mt	*Mt*	*Mt*	*Kt*	*Kt*	*Kt*	*Kt*	*Kt*	*Kt*	*Kt*	*Kt*
Low	0.2	0.3	0.2	63.0	8.8	9.2	0.0	0.0	0.0	0.4	0.0
Mid	0.3	0.3	0.3	85.1	47.1	25.5	18.6	5.2	3.9	2.2	1.3
High	0.6	0.4	0.4	218.9	891.7	43.3	81.1	12.6	8.1	4.1	2.7

Table 5

Projected carbonation potential of quantified slag deposits under three carbonation regimes (rates of CO_2 sequestration from Pullin et al. (2019)).

Carbonation method	Carbonation rate (kg CO_2/t slag)	Cumulative CO_2 uptake potential
Direct Carbonation	296–337	56.6–79.4 Mt
Enhanced Weathering	422–584	80.7–137.6 Mt
Passive Carbonation	0.007–12.1	1.3 Kt–2.9 Mt

management decisions are finalised.

4.2. Conservation designations

A large proportion of the legacy iron and steel sites identified in Fig. 1 are in close proximity or overlap with designated areas of conservation importance (Fig. 4; Table 2). The majority of these designations relate to biodiversity, but in some cases relate to cultural assets (e.g. built environment designations such as Maryport, Cumbria) or geodiversity of strata that were mined on or adjacent to steelworks (e.g. Brymbo, Wales in Table 3). The ecological significance of calcareous, alkaline low nutrient status substrates from alkaline slags has been long acknowledged (e.g. Gemmell, 1974; Bradshaw and Chadwick, 1980; Dobson et al., 1997), with substrates lending themselves to a low, open sward and floristic diversity (Table 3). In some cases, extreme initial pH (> 11) requires weathering to lower pH to a range suitable for plant growth (Gemmell, 1975), while the introduction of native species and amendments (e.g. organic materials or urea) can overcome the nutrient deficiency issues (notably P) and accelerate revegetation (Ash et al., 1994). Over time, some of these legacy sites have received designation (and subsequent management custody from conservation groups) specifically for plant communities present on slag heaps (e.g. regionally-rare communities where local bedrock would ordinarily not give rise to such soil types: e.g. Kirkless, Table 3; Ash et al., 1994). In many cases, the slag disposal adds topographic variability and helps create a mosaic of habitats (e.g. grassland on thin soils on heaps alongside damper, dune slack communities at margins: e.g. Coatham Marsh, Table 3). This is particularly apparent in coastal settings and it is here where nature conservation designations most prominently overlap with slag disposal areas (Table 2). In part this is due to the higher density of conservation designations in coastal areas, particularly those protecting migratory birds such as Special Protected Areas (SPAs) under the EU Birds Directive (EU, 2009).

In most of the cases listed in Table 3, the slag banks are contributing habitats to much broader designations (e.g. those around the Duddon Estuary and Morecambe Bay in north-west England), but specific communities and species of interest can be concentrated on disposal areas (e.g. Natterjack Toads at Millom, NW England: Table 3). At some disposal sites, the slag banks themselves offer physical protection to areas of saltmarsh that would otherwise be exposed to storm surges, for example Carnforth, Lancashire (Table 3) where heaps were initially placed as a coastal barrier as part of land reclamation efforts.

The cultural importance of a region may also determine the suitability of waste stockpiles for resource recovery through limitations on site development associated with Areas of Outstanding Natural Beauty (AONBs) and World Heritage Sites. In this case, one slag deposit was located within the buffer zone of Hadrian's Wall on the England-Scotland border (Table 2). While World Heritage Site designation alone does not bring additional statutory controls (Historic England, 2020b), limitations to disturbance and development may be introduced their common co-location with other designations (e.g. AONBs, where developments must conserve or enhance natural beauty (UK Government, 2018)).

Conservation designations for both the natural and built environment are common at a range of legacy waste sites and have been

detailed for former mine workings in particular (e.g. Crane et al., 2017; Lucarini et al., 2020). Some authorities have made progress in incorporating such issues into management planning at legacy waste sites (e.g. Younger and Wolkersdorfer, 2004; Johnston, 2004; Jarvis and Mayes, 2012). This is something that should be embedded in planning processes, be it for remediation or resource recovery, at iron and steel disposal sites also given there may be opportunities to direct restoration efforts at recently closed steel mills to ecological communities of high conservation value (e.g. Ash et al., 1994). With the case of the former Brymbo Steelworks in Wales (Table 3), the geodiversity that became apparent during site reclamation became the centrepiece for an exemplar regeneration programme at the former steelworks (Roberts, 2019). Improved ecological data availability for legacy steel and iron waste sites is of importance in formulating management strategies. While previous work has focused on terrestrial settings (e.g. Gemmell, 1975; Ash et al., 1994), there is a paucity of ecological data on coastal and intertidal disposal sites.

4.3. Resource potential

4.3.1. Metal and mineral resources

The composition of BF and BOF slags for UK deposits is largely consistent with global and national reviews (e.g. Naidu et al., 2020; Proctor et al., 2000; Piatak et al., 2015). Fe recovery has long been practiced from BOF slags usually via magnetic separation (given the relatively higher Fe content compared to BF slags; Lee, 1974; Fig. 5), and the data highlight considerable quantities of Fe associated predominantly with sites operating at peak in the 1960s and 1970s (Fig. 3). Beyond major mineral recovery and bulk recovery of BF and BOF slags for widely established applications predominantly in construction, emerging applications for Critical Raw Material recovery from iron and steel slags have been developed in recent years (Lindvall et al., 2010; Naidu et al., 2020) as has been the case for a range of other alkaline residues (e.g. Gomes et al., 2016; Ujaczki et al., 2018).

Although regional-scale resolution in data is sparse for UK slags (Table S1 in Supporting Information), typical P_2O_5 concentrations of 0.6–1.5 wt% are present in slags derived from Jurassic and Cretaceous ironstones (Juckes, 2003). Significant quantities of P are therefore likely present in the extensive deposits of North Lincolnshire, County Durham and Redcar and Cleveland (Fig. 2; Table 4). Deposits in Cumbria derived from hematite ores generally have far lower P content (P_2O_5 < 0.05%: Juckes, 2002) and would unlikely be suitable targets for recovery.

Vanadium was added to the European Union list of Critical Raw Materials in 2015 due in part to its use in emerging energy storage technologies and high-grade steel applications (Watt et al., 2018). Vanadium recovery from slags has been investigated for both improving the viability of downstream bulk reuse of slag with lower environmental risks once V is removed, and studies assessing V recovery methods (Lindvall et al., 2010; Gomes et al., 2017). Vanadium concentrations within slags have not been widely reported for UK deposits (Fig. 5, Table S1), but modern production slags are reasonably well characterised at a small number of steelworks (e.g. Hobson et al., 2017). These data suggest between 0.2 and 0.9 million tonnes of V within legacy deposits which equates favourably with annual global production of 73,000 t in 2019 (USGS, 2020b), albeit with major caveats on viability and efficiency of recovery technologies. Improved data availability would improve certainty in these estimates given the wide variations in V content that would be anticipated based on iron ore sources (Juckes, 2002).

Chromium recovery from steel slags has been demonstrated via high temperature processes such as alkali roasting (e.g. Kim et al., 2015). Again, Cr data are relatively sparse for UK slags, but the overall resource potential was between 0.3 and 0.9 million tonnes in the identified iron and steel slag deposits. While chromium was recently removed from CRM lists in the EU, due to a relative increase in security of

supply, its economic importance for use in high grade steel alloys remains (European Commission, 2017). Modest quantities of Co and Ni may also be present in iron and steel slags, although data availability for UK deposits is poor and should therefore be treated with caution (Table 4, Table S1). Furthermore, Co and Ni recovery technologies from slags have only been demonstrated for smelter slags with concentrations two-to-three orders of magnitude higher than those documented here (e.g. Jones et al., 2002). It is therefore likely that recovery of these metals from legacy slags, while technically feasible, is likely to be more costly than production from virgin ores where concentrations range up to several wt% and recovery processes are already established (Jeong et al., 2018; Yusifova et al., 2019).

4.3.2. Carbon capture potential

The carbon capture potential of slag is an area of burgeoning interest (e.g. Pullin et al., 2019; Renforth, 2019) with Table 5 considering various scenarios for reaping their mineral carbonation potential. Estimates for total carbonation consider the carbon capture potential based on cation mineral content of slags as per Pullin et al. (2019). Realisation of such potential would likely require comminution of material and heat / pressure, or enriched CO_2 streams to achieve the upper range of quoted carbonation rates (e.g. Gomes et al., 2016). While these upper cumulative estimates of 79.4 million tonnes of CO_2 (Table 5) and an additional 1–2 million tonnes CO_2/yr from contemporary production through carbonation of slag show non-trivial values compared to UK targets for atmospheric carbon capture (130 million tonnes removed per annum by 2050: UK Government, 2019), such re-processing could cause considerable disturbance and could be difficult to justify in many locations given environmental sensitivities and economic value from current land uses (Fig. 4). The 'passive' carbonation estimates consider terrestrial scenarios associated with in-situ slag weathering and carbon uptake as detailed at the Consett slag heaps in County Durham, UK (Renforth et al., 2009; Mayes et al., 2018; Pullin et al., 2019). These studies have shown that a very modest carbon capture potential is realised at typical slag depositories due to low atmospheric CO_2 diffusion through the heaps and armouring of slags by secondary minerals which minimises reactivity of Ca-silicate phases (Hobson et al., 2017; Stewart et al., 2018). These estimates for passive carbonation (1300 t to 2.9 million tonnes) offer a useful comparison between carbon capture that is likely already being realised under ambient disposal conditions (e.g. Pullin et al., 2019) and the significant scope for improving this with direct carbonation (Table 5). Approaches to enhance such passive approaches have been suggested such as emplacement of slags in shallow weathering heaps with active water management to minimise surface armouring, which may also limit any longer-term leachate generation (Pullin et al., 2019). Further enhancement of the carbon capture potential could be realised in coastal settings where captured inorganic carbon is likely stored as bicarbonate (HCO_3^-) in drainage waters which effectively doubles carbon capture potential compared to direct mineral carbonation (a term known as 'enhanced weathering': Renforth and Henderson, 2017). The extensive coastal slag deposits of south Wales, north east England and Cumbria would be particular candidates for assessing this potential, although ecological considerations and coastal defence functions need investigating (Table 3). The Cumbria deposits cover around 25 km of coastline where slags are emplaced below the historical mean high water mark, while the relatively high calcium content of the BF slags derived from local hematite ores (Juckes, 2002) may increase carbon capture potential beyond the average figures quoted here.

4.4. Conclusions and management implications

The compiled data presented here highlight the significant quantities of legacy slag deposits present in the UK. A range of potential opportunities for resource recovery exist from bulk reuse, carbon capture, ecological enhancement, and CRM/REE recovery (e.g. Abhilash

et al., 2017; Gomes et al., 2019a; Naidu et al., 2020). Improvements to characterisation of slag deposits would be crucial to underpin such efforts, particularly for CRM and REE content, for which data are sparse at present for UK legacy slags. Furthermore, more detailed site-by-site estimates of volume from higher resolution topographic data, coupled with geophysical surveys (e.g. Mayes et al., 2018) could constrain the slag heap volume estimates and would be recommended at sites showing the most promise for resource recovery.

The data presented highlight some of the potential obstacles to slag valorisation such as current land uses and conservation designations. The former includes a range of activities, with urban developments dominant and sites where slag forms important coastal defence assets. The high proportion of slag disposal areas co-located (21.4%) or in close proximity (51.8%) to conservation designations may pose conflicting issues for potential bulk reuse, given risk of environmental damage during groundworks. Although these issues are not insurmountable, they would need to be carefully considered in any approaches to rework slag deposits. However, given the abundance of sites where spontaneous recovery has led to the development of sites of conservation value, ecological enhancement should be considered a potential resource of slag depositories and may complement some resource recovery options (e.g. weathering of slag for carbon capture and CRM recovery prior to landscaping and development of calcareous grassland: e.g. Ash et al., 1994; Pullin et al., 2019). Furthermore, where environmental liabilities are present at slag disposal sites, then ecological approaches to remediation, such as wetlands (e.g. Gomes et al., 2019b), could further enhance environmental quality.

Given the potential physical and likely legislative constraints for resource recovery at many slag disposal sites in the UK, valorisation efforts are likely best targeted at active and recently decommissioned steelworks and sites (e.g. Redcar, Port Talbot, Llanwern and Scunthorpe; Table S3) where regulatory barriers (e.g. planning constraints) to reworking materials would be relatively modest. At these locations, efforts to fully integrate the multifarious resource recovery opportunities (e.g. bulk reuse, CRM, Fe, carbon capture, ecological enhancement) into operating or closure plans should be a priority. This could both offset some of the costs of site remediation (e.g. Sapsford et al., 2019) and also extend viable economic activities at the sites. Such developments would likely require government intervention to (1) incentivise resource recovery (e.g. for using secondary sources of minerals instead of primary sources), (2) facilitate regulatory approval of novel and emerging approaches to valorisation (e.g. new carbon capture approaches), and (3) ensure legislation around waste classifications are flexible to permit bulk reuse after such recovery efforts where safe to do so (e.g. Deutz et al., 2017). As many initiatives develop internationally to improve inventories of mineral and non-mineral resources in anthropogenic deposits (e.g. Blasenbauer et al., 2020), this study highlights the importance of simultaneous assessment of potential management challenges to assist in assessing feasibility of waste valorisation.

CRediT authorship contribution statement

Alex L. Riley: Conceptualization, Methodology, Formal analysis, Investigation, Data curation, Writing - original draft, Visualization. **John M. MacDonald:** Investigation, Validation, Writing - review & editing. **Ian T. Burke:** Conceptualization, Writing - review & editing. **Phil Renforth:** Conceptualization, Writing - review & editing, Funding acquisition. **Adam P. Jarvis:** Writing - review & editing, Funding acquisition. **Karen A. Hudson-Edwards:** Writing - review & editing, Funding acquisition. **Jessica McKie:** Conceptualization, Writing - review & editing. **William M. Mayes:** Conceptualization, Methodology, Validation, Investigation, Writing - original draft, Funding acquisition.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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