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abstract

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There are widespread concerns regarding the potential future scarcity of ferrous and non-ferrous mate-rials. However, there are already potentially rich reserves of secondary materials via high ownership of Electrical and Electronic Equipment (EEE) in economically-developed nations. Young people are particu-larly high consumers of EEE, thus university students and campuses may present an opportunity to har-ness this potential. University Distinct Urban Mines (DUM) may be used to exemplify how potential reserves of secondary metals may be exploited, and could contribute to the transition from a linear to a circular economy. This study aimed to evaluate small household appliances (SHA) DUM from a UK uni-versity, with the objectives to identify and quantify student households' SHA ownership, WEEE recycling, stockpiling and discarding habits amongst student households, assess and evaluate the monetary poten-tial of SHA DUM at UK level, and propose methods to exploit DUM for universities in the UK.

To this purpose, a quantitative survey was undertaken to measure students' ownership and discarding behaviour with respect to SHA. The amounts of ferrous and non-ferrous materials were then estimated and converted to monetary values from secondary materials market data to appraise the SHA DUM over-all value. Thirty-five per cent of SHA are discarded in the general refuse. Broken personal care appliances (PCA) tend to be discarded due to hygiene and small size factors. When in working order, SHA tend to be equally reused, recycled or stockpiled. We conclude that a total of 189 tonnes of ferrous and non-ferrous materials were available via discarding or being stockpiled at the University of Southampton. Extrapolated to UK higher education level, discarded and stockpiled SHA represent a potential worth USD 11 million. To initiate DUM exploitation within Higher Education campuses, we suggest improving users' choice architecture by providing collection methods specific to broken SHA.

1. Introduction

1.1. Urban mining

Urban mining is a construct of anthropogenic resources between landfill mining and recycling to integrate secondary material flows and stocks into the Circular Economy (Cossu and Williams, 2015). A Distinct Urban Mine (DUM) involves the spatial and geographical delimitations of different waste categories, mak-ing cities and university campuses ideal prospection grounds due to their delimited geographical area and localised population. The concept of DUMs further advances this notion by segmenting materials such as plastic, ferrous and non-ferrous associated with specific WEEE categories (Ongondo et al., 2015). Similarly to pri-mary material mines, prospecting a DUM involves the aggregation of information about existing stocks and flows (Wallsten et al., 2015). Stocks can be associated with in-use and stockpiled items (EEE); material flows can likewise be associated with reusing, recy-cling and discarding behaviours (WEEE). The concentration of materials within a DUM depends on the products associated for specific EEE/WEEE categories as well as defined ownership levels. In fact, research by Mueller et al. (2015) has shown that anthro-pogenic mines of rare earth metals – effectively metal-specific DUMs - can now have both a higher concentration of such ele-ments and a longer mine life than a current well-established geo-genic mine. However, for a DUM to be viable, there must be reasonable economic prospects for exploiting them (Sun et al., 2015).

DUMs are considered a valid concept to evaluate potential for secondary resources recovery within the anthroposphere and cost-efficient methods need to be implemented to access them (e.g. Ongondo et al., 2015). DUMs are not only defined by their delimited space within the anthroposphere but also by the poten-tial availability of resources for a given type of EEE. Ongondo and Williams (2011a,b) estimated that, for a specific university DUM with approximately 24,000 students in the UK rich in IT and telecommunication equipment, 20 tonnes were currently stock-piled and 87 tonnes would be available within 36 months. Without specifically identifying DUMs, previous authors have estimated the potential of stockpiled WEEE. Milovantseva and Saphores (2013) estimated that 84.1 million televisions were stockpiled in US attics. Ongondo and Williams (2011a,b) evaluated that close to 60 million mobile phones were stockpiled in US and European Higher Educa-tion Institutions.

If a mine is to be exploitable, urban or otherwise, it needs to be economically viable and located within reach of an existing logis-tics network with materials concentration at an optimal level (Zhang and Kleit, 2016). This economic feasibility is defined by the potential revenues after collecting, transporting, separating, processing and recycling materials from WEEE, accounting for the market values of recoverable secondary materials (Sun et al., 2016). These costs are largely driven by the incentives associated with the availability and accessibility of these EEE and WEEE stocks (Krook et al., 2011).

WEEE collection events are regularly organised to transform stocks into flows at community levels (WRAP, 2016). Collection events involve householders taking their WEEE to a single location at a specific time. These events are mainly aimed at smaller WEEE, as larger WEEE items are often taken away when a new product is delivered (Directive, 2002/96/EC; Directive, 2011/65/EU). Smaller WEEE tend to be stockpiled, especially if not broken when they are unwanted (Guillard and Pinson, 2012; Ongondo and Williams, 2011a). Personal care appliances (PCA) tend to be more readily discarded than other small WEEE (Darby and Obara, 2005) due in part to a lack of awareness of disposal methods (Timlett and Williams, 2008) and lack of monetary incentives (Ongondo and Williams, 2011b). Given the low residual value indi-vidual items may have, monetary incentives, if implemented, would likely be too low to trigger an intended recycling behaviour (Ariely et al., 2009; Jones et al., 2010).

1.2. Choice architecture

Several millions of mobile and smartphones are stockpiled in the UK (Ongondo and Williams, 2011b). In 2012, the European Commission estimated that 42% of EEE equivalent placed on the market was collected for recycling purposes (Eurostat, 2016). According to behavioural economics theory (Thaler et al., 2014) this situation could be due to a lack of valid alternatives. This would suggest a need to modify users' choice architecture as ‘‘decision-makers don't make choices in a vacuum" (Thaler et al., 2014:428); a choice architect is an organiser who designs a pre-ferred set of alternatives to achieve a desired outcome (op. cit.).

In their approach to choice architecture, Thaler et al. (2014) identify three core principles: defaults or the path to least resis-tance, feedback and errors. Choice architecture, sometimes referred to as ‘‘libertarian paternalism", is the mapping of preferred outcomes and design of alternatives in accordance to these out-comes. Table 1 illustrates choice architecture principles applied to waste management. An example adapted to environmental behaviour is related to utility companies evaluating customers' consumption compared with the local neighbourhood or national consumption average. Choice architects have at their disposal sev-eral principles or methods they can freely adapt to any situation to influence a decision towards a desired outcome.

By transposing the concept of choice architecture into the field of waste management, alternative methods for the collection of small household appliances (SHA) could be proposed. As stated by Darby and Obara (2005), one-size-fits-all solutions are ill-adapted for comprehensive recycling efforts, and there remain challenges to the effective collection of SHA. WEEE tends to be stockpiled, regardless of its broken or unbroken status (e.g. Guillard and Pinson, 2012; Ongondo and Williams, 2011b). A dis-tinction is sometimes made between broken, irreparable and reparable WEEE (e.g. Darby and Obara, 2005), but estimates of WEEE stockpiles do not always make this important distinction (Milovantseva and Saphores, 2013). If WEEE stockpiling is influ-enced by multiple factors, then there should be different methods to convince consumers to destockpile. From a decision-mapping perspective, the decision to stockpile WEEE instead of taking it to a Take-Back Scheme (TBS) or Household Waste Recycling Centre (HWRC) would indicate that the most convenient option is pre-ferred and users are currently unsatisfied with the current alterna-tives offered.

Table 1
Choice architecture applied to urban waste management. Adapted from Thaler et al. (2014).

Factors	Definition	Waste management application
Defaults (or the path to least resistance)	Presented as most preferred alternative with regards to desired outcome	General waste recycling bins with several alternatives (plastic, paper, refuse) placed nearby public vicinities. Most preferred outcomes (plastic and paper) are presented with vibrant colours and closer to users. Least preferred outcome, such as refuse, is further from user, with grey colours and potentially with smaller entry point
Feedback	Information provided to user on action performed	(1) Message informing the equivalent in resources to produce a certain amount of paper or plastic (2) A transparent container for users to evaluate contribution made
Errors	Expected errors in decision-making process	(1) Recyclables disposed of in incorrect recycling bins (2) Sorting and cleaning process performed after bins are emptied

1.3. Behaviour change

To be exploitable efficiently, a DUM requires end-users to behave in such a manner that it becomes possible to access stock-piles and transform discarding habits into reuse and recycling opportunities (Ongondo et al., 2015; Sunstein and Thaler, 2012). Behaviour change is a complex process that requires the use of var-ious incentives that may be intrinsic and/or extrinsic. Individuals assess potential actions against their consequences and perceived value; the decision-making process is influenced by the ‘‘acquisi-tion, evaluation, execution and interruption of abstract actions” (Balleine et al., 2015:2). However, behaviour is difficult to predict accurately and is an unsteady process (Bouton, 2014). Williams (2015), in an overview of the key determinants of household recy-cling behaviours and the different socio-psychological models that seek to explain them, highlighted three models based on psycho-logical principles: Altruism (Schwartz, 1968), the Theory of Rea-soned Action (TRA; Fishbein and Ajzen, 1975) and the Theory of Planned Behaviour (TPB; Ajzen, 1985). These models are based on intrinsic motivators such as beliefs, attitudes, intentions, social norms, awareness of consequences, ascription of responsibility, personal norms, past behaviour and values, and have been applied extensively by waste management scholars to predict recycling behaviour and develop enablers to sustainably influence recycling behaviour (de Leeuw et al., 2015; Pakpour et al., 2014; Richetin et al., 2010). The review by Schultz et al. (1995) indicated that intrinsic factors such as feedback, removing barriers, goal setting, normative influence, prompts and public commitment, have more influence than extrinsic factors (e.g. rewards) on recycling beha-viour. In addition, Werner (2003) underlines the importance of integrating social and physical contexts when attempting to use intrinsic motivators as means to alter behaviour.

Behaviour change may also be influenced by extrinsic motiva-tors based on monetary incentives, either defined by rewards or penalties such as taxes. Perhaps not surprisingly, households prefer recompense (rewards) over penalties, but may prefer community-based rewards and local taxation rebates over individual rewards (Shaw and Maynard, 2008). Willingness-to-pay (WTP) and pay-as-you-throw (PAYT) studies have demonstrated the difficulties in associating monetary incentives to ‘‘waste” (Afroz et al., 2013; Brown and Johnstone, 2014; Zen and Siwar, 2015).

Enablers to behaviour change advocates do, however, remain available. Timlett and Williams (2008) recommend simple and low cost methods such as public participation in the design of col-lection methods to increase recycling performance. Keramitsoglou and Tsagarakis (2013) recommend that residents should be empowered when taking part in a recycling scheme. Strömberg et al. (2016) agree with this approach as they support trials to dur-ably change behaviour. Chan (1998) and Read (1999) advocate communication campaigns. However, there is a shift towards more personalised messages (Tompson et al., 2015) to alter durably behaviour. This approach is cost-effective and possible with contemporary social media and associated interest groups (Caniato et al., 2014).

1.4. Study aim and objectives

The primary aim of this study was to assess the potential value of a SHA DUM for a university campus. The objectives were to:

- Identify and quantify student households' SHA ownership
- Establish and assess SHA recycling, stockpiling and discarding habits amongst student households
- Assess and evaluate the monetary potential of SHA DUMs for universities in the UK
- Propose methods to exploit DUMs for universities in the UK

2. Methods

Data were collected in two stages. Primary data were collected via a survey to assess SHA ownership. Secondary (monetary) data were subsequently applied to translate SHA available on campus into DUM material financial values.

2.1. Primary data collection and WEEE evaluation

Primary data were collected from University of Southampton (UoS) students via an online questionnaire. The questionnaire was piloted among ten students via email and a focus group of five people was held with respondents answering the questionnaire alongside a researcher. Distribution routes for the questionnaire included student societies as well as the university portal, target-ing a population in excess of 22,000 students. The questionnaire determined how many appliances students owned, how appliances were acquired, whether the appliances were used and, if so, why and how they had been disposed of.

Data were then categorised into two subsets of SHA: personal care appliances (PCA) and small kitchen appliances (SKA) (Table 2). The study focused on SHA that (1) are commonly owned by univer-

Table 2
Small household appliances considered in the study, subdivided into personal care appliances and small kitchen appliances.

Small household appliances subset	Appliances in subset
Personal care appliances (PCA)	Electrical hair removal appliances Hair dryers Hair styling appliances Electric toothbrushes
Small kitchen appliances (SKA)	Kettles Toasters Sandwich makers/grills Food blenders/mixers Other – i.e. Bread makers, rice cookers Irons (Non-kitchen appliance)

sity students and light enough to be easily disposed of by students, and (2) have, due to their size, high potential for storage. Conse-quently, other household appliances not intended for use in the kitchen or bathroom or that are not commonly owned by students were not taken into account (i.e. carpet sweepers, vacuum clean-ers, appliances for sewing, knitting weaving and other processing for textile, fryers, grinders, electric knives, clocks, watches, weigh-ing scales). To increase the response rate, a prize draw was offered to participants completing the survey with £100, £50 and £25 supermarket gift cards offered.

2.2. Secondary data collection and DUM assessment

Secondary data were collected online from a large UK retailer (Tesco, 2016) to obtain estimates of the amount of plastic, ferrous and non-ferrous materials in each of the small household appli-ances identified in the survey. This retailer was selected due to the large amount of data available online for items' weights (excluding packaging). Ten data were obtained for each product, with a total of 90 data points collected across all observed small household appliances considered (Table 2). The secondary data col-lection was completed after the primary data collection was over, therefore information regarding buying habits was available and key products identified. Preliminary results indicated students were more attracted by cheaper products. As a consequence, the cheapest products were selected on the retailer website for retrie-val of product weight data.

Data from the Waste and Resources Action Programme (WRAP, 2012) were obtained to evaluate the amount of SHA by material composition. Minor metals such as zinc or nickel were excluded as they comprised less than 2% of SHA product weight. In terms of weight, plastics, ferrous and non-ferrous (aluminium and cop-per) materials were the most prevalent. According to Martinho et al. (2012:1) small WEEE plastics can be composed of ''more than 15 different types of engineering plastics''. To add complexity, each SHA is composed of different plastics and each plastic has a differ-ent polymer composition. Furthermore, plastic residual value is marginal compared to the value of metals. WRAP (2016), estimates that prices for PET and HDPE plastic from bottles range from USD 13 to USD 40 per tonne. By comparison, ferrous materials approx-imate USD 250 and copper in excess of USD 4700 per tonne (LME, 2016). SHA plastic composition is beyond the scope of this study and, due the marginal monetary value of SHA plastics, the study thus focused on SHA plastic appraisals by weight.

Using data on ownership levels associated with each product and their average composition, results were aggregated into two larger categories: SKA and PCA (Table 2). Each category was further sub-divided into ferrous, aluminium and copper metals. Financial data were accessed from the world's largest market for industrial metals trading London Metal Exchange (LME). To minimise price

Table 3

volatility, average prices per tonne at three months forward were considered as of the 11th of July 2016 for ferrous metals.

2.3. Statistical analysis

Two statistical analyses were conducted, one for primary data and a second for secondary data on PCA and SKA weight data.

2.3.1. Primary data analysis

For the questionnaire data, non-parametric tests were applied between groups and within groups according to the number of variables of interest. Kruskal-Wallis tests were used between-groups for more than three variables. Wilcoxon signed-ranked tests were used within-groups for two variables. Friedman's ANOVA was used within groups as well but for more than three variables. These tests were carried out for ordinal data. For categor-ical variables, for example when two variables are of interest, Chi-square tests are preferred.

Chi-square tests were carried out for association between age, sex, home/overseas students and the purchasing factors, owner-ship, and disposal routes. Further Chi-square analysis was carried out for the condition of an appliance cf. disposal route and the dis-posal route of SHA cf. general recycling. Friedman's analysis of vari-ance and Kruskall-Wallis tests were used to assess for significance and agreement on the influence of incentives for recycling SHA and the barriers to using HWRCs and retailer take-back schemes. Wil-coxon signed-rank tests were applied (e.g. Ongondo and Williams, 2011a) to rank the incentives and barriers in order of their influence as indicated by survey respondents.

2.3.2. Secondary data analysis

For samples with fewer than 50 data points (10 data points for each appliance within each SHA category) the Shapiro-Wilkinson test was performed to determine if small kitchen appliances and personal care appliances data for weight followed a normal distri-bution. A significance level of 0.05 was applied.

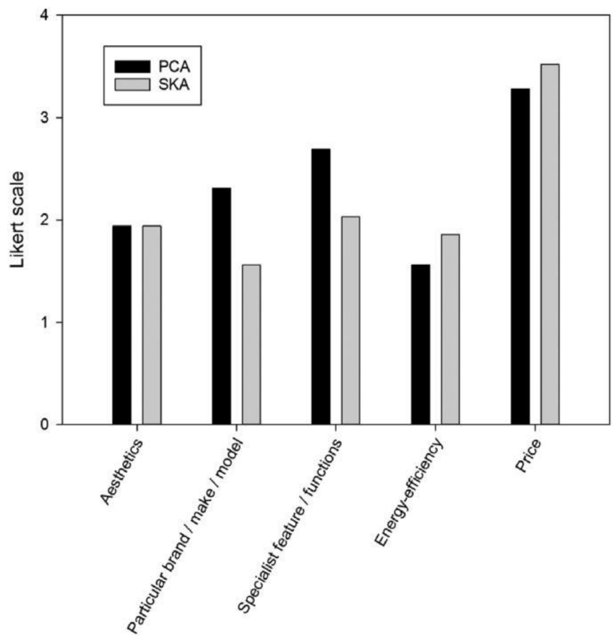
3. Results

A total of 546 surveys were completed in 2011 of which 540 were usable. Some 3030 students were directly contacted using emails, the University of Southampton portal and Facebook groups the research team was affiliated to, representing a response rate of 18%. There were 23,795 students at the UoS in 2015 (The Complete University Guide, 2015) and 2,266,075 students in the UK in 2014/15 (HESA, 2016). Given the time difference between the data collection and the student population estimates, data for both 2011 and 2015 are presented (Table 3). The sample was broadly repre-sentative of the wider UK student population in 2011 and 2015

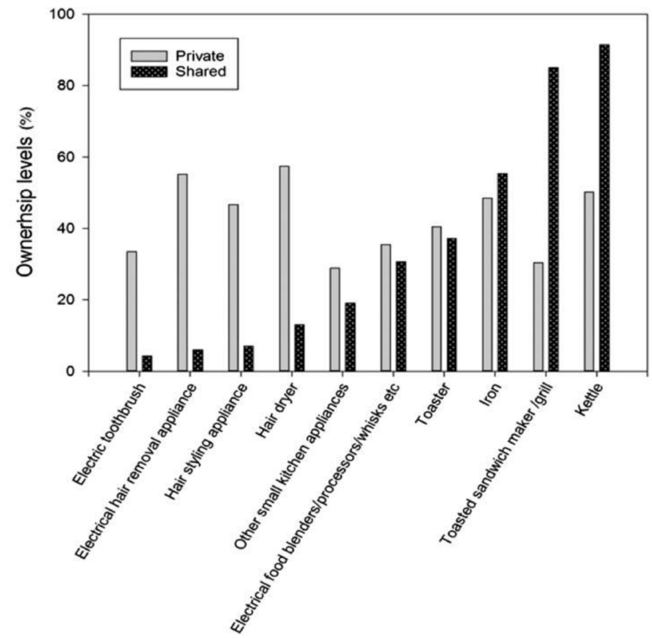
Socio-demographic comparison between the respondent sample and the student population in the UK (Source: HESA, 2011 and 2015).

Socio-demographic categories		% of survey respondents (2011)	% of UK student population (2011)[a]	% of UK student population (2015)[b]
Sex	Females	55	56.4	56.5
	Males	45	43.6	43.5
Age	Young (18–24)	85	69	67
	Mature (25+)	15	31	33
Level of study	Undergraduate	83	76	76
	Post-graduate	17	24	24
Domicile	Home	78	77	81
	Overseas	22	23	19

a https://www.hesa.ac.uk/data-and-analysis/publications/students-2011-12/introduction.
b https://www.hesa.ac.uk/data-and-analysis/publications/students-2015-16/introduction.



PCA
SKA
Likert scale
0
1
2
3
4
Aesthetics
Particular brand / make / model
Specialist feature / functions
Energy-efficiency
Price



Private
Shared
Ownerhsip levels (%)
0
20
40
60
80
100
Electric toothbrush
Electrical hair removal appliance
Hair styling appliance
Hair dryer
Other small kitchen appliances
Electrical food blenders/processors/whisks etc
Toaster
Iron
Toasted sandwich maker /grill
Kettle

Fig. 1. PCA and SKA mean score of importance purchasing factors (scale from 0 = not at all important at to 4 = most important).

Fig. 2. SHA ownership levels in percentage of respondents. Items ranked by increasing order of shared use.

for gender and domicile but not fully representative for age and level of study (Table 3).

3.1. Purchasing factors

Price is regarded as the most important purchasing factor for both SKA and PCA (Fig. 1). Specialist features are ranked second for both SKA and PCA, although students have rated specialist fea-tures as more important for SKA than PCA. Almost as importantly rated are aesthetics for PCA and particular brand is ranked third for SKA. A total of 79% of SKA were acquired in new condition com-pared with 92% of PCA.

Chi-square analysis was carried out against the importance of purchasing factors of SKA and sex, age and domicile and then repeated for PCA. Only gender categories yielded significant results for SKA and PCA. Other categories such as age and domicile yielded significant results for PCA and SKA combined. Female students tend to own more PCA than male students. No distinction can be made between genders for SKA. Students over 25 years old tend to value brand, make and aesthetics more than younger students for SHA. Compared with home students, overseas students tend to prefer more specialist features for their SHA.

3.2. Ownership

Grooming and personal care products are more likely to be owned by females (98%) than males (80%) and (unsurprisingly) not shared (Fig. 2). Compared with PCA, SHA used on an everyday basis such as, kettles or toasters are more likely to be shared, espe-cially if respondents lived in a shared accommodation. The larger the household, the more likely that SKA were shared ($X2 = 6.353$, $df = 2$, $P = 0.042$).

3.3. Secondary data distribution

Secondary data on SHA weights were collected online (Tesco, 2016). Students indicated they were mostly attracted by cheaper products, consequently SHA products with the lowest price tags were selected for the data collection. Among online product specifications data relative to product weight, packaging excluded were collected.

Data gathered for PCA were normally distributed but not for SKA (Fig. 3). Given that the sample size was inferior or equal to 50 data points (n = 40 for PCA and n = 50 for SKA), the Shapiro-Wilk normality test was performed for both PCA and SKA. For a degree of significance set at 0.05, the null hypothesis could not be rejected for PCA (p-value = 0.307) but was rejected for SKA (p-value (0.000). Subsequent transformations were made to achieve normality for SKA. PCA and SKA data were merged into a single small household appliances (SHA) category but the normality tests remained negative, thus the data were not normally distributed. SHA data were then transformed with a logarithmic scale and nor-mality achieved. This test was carried out to demonstrate the num-ber of data points from the secondary data collection is sufficient to indicate normality and potential generalisation to the wider popu-lation. The mean value for PCA is 447 ± 172 grammes per product; the mean value for SKA is 1588 ± 721 grammes.

3.4. SHA materials composition

WRAP (2012) data indicate that plastic, ferrous and non-ferrous metals are the most prevalent materials in SHA, accounting for 98% of product composition (Table 4). Strontium, tin, calcium, zinc, bar-ium and nickel represent less than 2% of the total material compo-sition and were not considered in this study. Compared to the amount of plastic and ferrous metals, copper is one of the least pre-sent materials in SHA but is the most valuable material per unit weight (Table 4; WRAP, 2012). Copper is mainly used in limited quantities for wiring motors and cables. In contrast, plastic and fer-rous materials are used for casing and internal elements to sustain the mechanical properties of SHA. Aluminium is used for its light-weight and non-conductive properties to replace ferrous materials when design and economic considerations allow.

Among the University of Southampton students, there is collec-tively more plastic, ferrous, copper and aluminium contained in PCA than SKA (Fig. 4), despite their lower average weight per pro-duct (Table 5). PCA average weight per item ranged from 370 grammes (g.) to 542 g., compared with 1049–2071 g. on average

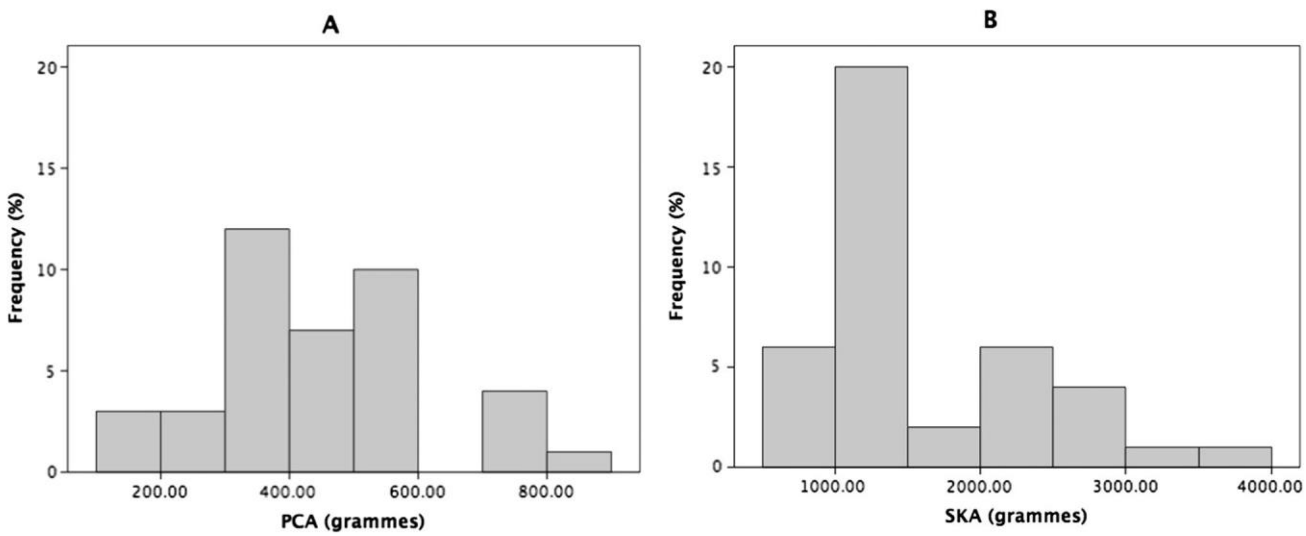


Fig. 3. PCA (A) and SKA (B) sample weight distribution (n = 40).

Table 4
Category 2 WEEE composition. Source data from WRAP (2012); Material prices for July 11th 2016 (London Metal Exchange).

	Ferrous	Plastic	Aluminium	Copper	Sn, Tn, Ca, Ni, Ba
Composition	40%	37%	14%	7%	2%
Price per tonne in USD	214	0	1653	4720	N/A

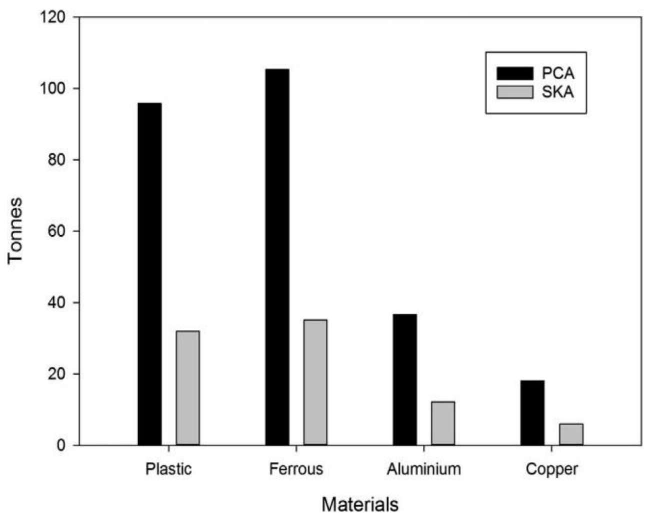


Fig. 4. Local university DUM for PCA and SKA materials content in copper, plastic, ferrous materials and plastic.

for SKA. This can be explained by the higher level of ownership of PCA compared to SKA (Fig. 2). At campus level, this represents 128 tonnes of plastic, 140 tonnes of ferrous materials and 73 tonnes of non-ferrous for PCA and SKA combined. On average there is three times more material content in PCA than SKA (Fig. 4) for plastic, ferrous and non-ferrous materials. The two most abundant mate-rial categories for this SHA DUM are plastic and ferrous metals.

3.5. Distinct Urban Mine potential

End-of-use decisions for PCA and SKA differ if SHA are broken or in working order (Table 6). When SHA are broken, a majority of respondents chose to discard their items as general refuse. Some

Table 5
PCA and SKA items average mass in grammes from online data collection based on products with lowest price tag.

Small household appliances	Average mass (g)
Electric hair removal (PCA)	370 ± 71
Hair dryers (PCA)	543 ± 45
Hair styling (PCA)	435 ± 27
Electric toothbrush (PCA)	440 ± 57
Irons (SKA)	1821 ± 183
Kettles (SKA)	1049 ± 34
Toasters (SKA)	1648 ± 194
Sandwich maker/grill (SKA)	2071 ± 259
Mixers/blenders (SKA)	1583 ± 247

30% of PCA are discarded in the general refuse and 20% of SKA. On the other hand, more SKA tend to be recycled when broken (30%) than PCA (15%). When in working condition, 20% of both SKA and PCA tend to be reused.

The decisions regarding the fate of end-of-use products, when scaled at the UoS level, impact markedly on the amounts of mate-rials bound for different destinations (Fig. 5). If all materials are combined for both PCA and SKA at university DUM level and within 36 months, there are approximately 118 tonnes discarded, 84 tonnes recycled, 70 tonnes stockpiled and 68 tonnes reused of ferrous and non-ferrous materials. PCA are more abundant than SKA: for stockpiling and discarding decisions, there are 153 tonnes of PCA compared with 36 tonnes of SKA.

With regard to DUM potential with stockpile and discard deci-sions, the UoS SHA DUM represents a monetary potential in excess of USD 124,000 for a total of 23,075 students, and USD 11 million at UK level with 2,266,075 students, both equivalent to approxi-mately USD 5 per student (Table 7). More than a third of this potential lies with discarding decisions (Fig. 6). Copper is the least present material (in terms of weight) for both PCA and SKA but its value per tonne is the highest (Table 4). In decreasing order, the highest potential lies first with discarded PCA, stockpiled PCA,

Table 6
End of use fates according to working (W) or broken (B) status expressed in percentage. Data shown are the proportions (%) of SKA and PCA.

%	Reuse		Recycle		Stockpile		Discard	
Status	W	B	W	B	W	B	W	B
SKA	20	1	9	30	7	9	4	20
PCA	20	1	5	15	10	9	10	30

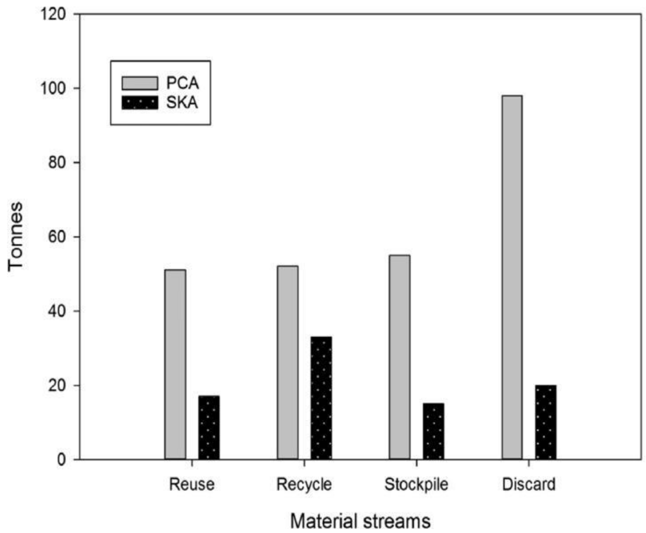


Fig. 5. Local PCA and SKA DUM potential in tonnes for each stream with all decisions
materials combined.

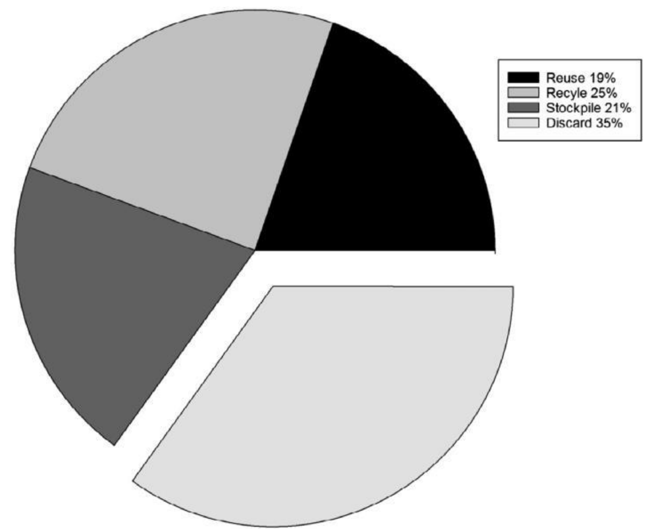


Fig. 6. SHA DUM potential identification from local SHA DUM end of use

extrapolated at UK Higher Education level to prioritise efforts.

followed by discarded SKA and stockpiled SKA. Reuse and recycling decisions are not taken into account in terms of the DUM potential as they are positive outcomes and the DUM potential lies with untapped potential, i.e. stockpiled and discarded SHA.

4. Discussion

Ownership levels were successfully translated into an estimate of the potential of University DUMs potential for SHA, at both local and national levels. PCA have lower metal content per unit than SKA, but high PCA ownership levels result in higher overall mate-rial tonnage (Fig. 4). PCA tend to be individually owned and SKA tend to be shared (Fig. 2). The three most frequently owned PCA items are: hair dryer (58% of respondents), hair removal (56%), and hair styling (50%) appliances. The two most frequently shared appliances are SKA: kettles (91%) and toasters (83%). Almost all female respondents (98%) own at least one PCA compared with 80% of males, but SKA is not gender specific. PCA include hair

Table 7

styling appliances and SKA pertain to the common kitchen envi-ronment. Consequently, SKA tend to be shared but not PCA (Fig. 2). These findings are broadly in line with ownership levels for PCA and SKA for the general population households (Yao and Steemers, 2009). The main difference is that students are more sensitive to price, tending to buy cheaper products, which could imply higher replacement rates due to likely higher failure rates. This behaviour potentially increases the amount of materials dis-carded in the general refuse if students' behaviour as observed (Table 6) prevails.

Depending on the device nature and status, end-of-use deci-sions vary (Table 6). End-of-use decisions according to the device working or broken status is confirmed by Ongondo and Williams (2011a,b) in their mobile phone study. Findings of this study show that broken SKA and PCA are more likely to end as general refuse than similar unbroken appliances. The most frequent end-of-use decision for broken PCA is general refuse (30% of broken appli-ances) and the most frequent end of use decision for broken SKA is recycling (30% of broken appliances). Possibly the ‘‘hygiene”

A and B – Local university PCA and SKA potential based on stockpile and discard decisions.

7A – Local university PCA DUM	USD/tonne	Stockpile in tonnes	Discard in tonnes	Total DUM in USD
Plastic	0	20.5	36.8	0
Ferrous	214	22.6	40.4	13,478
Aluminium	1653	7.8	14.0	36,184
Copper	4720	3.9	6.9	51,033
Total		55	98	100,695

7B – Local university SKA DUM	USD/tonne	Stockpile in tonnes	Discard in tonnes	Total DUM in USD
Plastic	0	5.8	7.5	0
Ferrous	214	6.3	8.3	3128
Aluminium	1653	2.2	2.9	8399
Copper	4720	1.1	1.4	11,845
Total		15	20	23,372

factor and the smaller size of hair appliances, compared with kitchen appliances, entice users to get quickly rid of them, and in the general refuse. PCA items are smaller than SKA (Table 5). The smaller size of PCA entails that social pressure, from non-

Table 8
Choice architecture applied to broken SHA. Adapted from Thaler et al. (2014).

Factors	Definition	Broken or unwanted SHA application

cohabitants at least, cannot be activated as it is more difficult to discern a non-environmental behaviour, as identified by Darby and Obara (2005). The difference between working and broken sta-tus has limited impact on the amount of PCA or SKA being stock-piled; 7–10% of both broken and working SKA and PCA are stockpiled (Table 6). Compared to the findings of Ongondo and Williams (2011a), stockpiling decisions are reported as almost the least preferred alternatives. It seems that students do not now stockpile working or broken PCA or SKA. Potential explana-tions for this change include the limited space available in student accommodation and that alternatives (recycling, discarding, reus-ing, selling) are now more readily available than previously.

The potential of a DUM is evaluated on stockpiling and general refuse decisions. Compared with reuse and recycle streams, they are negative outcomes from a circular economy perspective. At local level, the studied SHA DUM represents 189 tonnes (Fig. 5) and USD 124,000 (Table 7). PCA represent three times as much as SKA in terms of both quantity and value. Ongondo et al. (2015) estimated that 20 tonnes of IT equipment were currently stockpiled on the same campus and indicated that a further 87 ton-

Defaults	Presented of most preferred alternative with regards to desired outcome
Feedback	Information provided to user on action performed
Errors	Expected errors in decision making process

Small collection container located in student accommodation hallway and next to entrance. Vibrant colours to make it stand out and remind regularly its presence to dwellers. Efforts to make it visually appealing with attractive design

Message informing the equivalent in resources to produce a certain amount of metals or plastic. Transparent container to see contributions made

General waste thrown in container. Pictograms reminding Do’s and Don’ts. Access point wide enough to accept SHA but small enough to obstruct large refuse bags nes would also likely be available within 36 months of the time of the study, although the study does not specify how much e-waste was discarded. IT equipment is composed of critical materials such as rare earth metals but they are difficult to extract (Binnemans et al., 2013). SHA materials are less valuable with mostly ferrous and non-ferrous materials such as copper, but the recycling pro-cess is simpler for small household appliances compared with IT equipment. There are fewer components and the components are complex. Consequently, exploiting a DUM of small household appliances has lower technical requirements than exploiting an IT-equipment DUM. This SHA DUM indicates that the ‘‘low hanging fruits” are PCA, as they are the most frequently discarded (Table 6). Despite their lower average weight compared with SKA, higher ownership levels of PCA yield higher absolute values for their metal content, both in tonnage and monetary value. At UK level for the Higher Education sector, PCA DUMs represent a potential of USD 9.4 million and USD 2.35 million for SKA. For a total in excess of USD 11 million at UK level for SHA. Overall, it appears that 35% of an SHA DUM flow through the discard stream (Fig. 6).

This monetary potential could serve as the maximum necessary investment to capitalise on small household appliances DUMs at UK level within Higher Education. Estimates presented (Table 7) represent the market value these materials constitute after, hypo-thetically, being processed and fully recovered. Therefore, further discussions with public and private organisations should be under-taken to estimate communication, collection and processing costs. Timlett and Williams (2008) advocate low cost and simple solu-tions for most effective collection results. Tompson et al. (2015) propose that social media, especially among young adults, are cost effective methods to communicate consistently and regularly. These simple, yet efficient methods could be used to inform users of improved choice architecture for PCA and SKA end-of-use deci-sions. Table 8 illustrates how a DUM could be exploited based on core rules to design choice architecture (Thaler et al., 2014).

Choice architecture applied to WEEE should focus on conve-nience, sorting and intrinsic motivators (Table 8).

Ongondo and Williams (2011a,b), Saphores et al. (2012) and Wang et al. (2011) recognised that recycling convenience is an important aspect in WEEE recycling behaviour and inconvenient collection methods are barriers to recycling. Convenient locations near to the daily journeys made by consumers should be preferred.

Attractive design and colours would also raise awareness and increase acceptance (Fraige et al., 2012).

Conveniently located containers would incorporate different entry points according to the type of WEEE collected and their working status. Zeng et al. (2017) advocate gradually shifting from a ‘‘macroscopic” to a ‘‘microscopic” perspective by taking into account the substances used to produce materials. Whilst sorting WEEE is still considered macroscopic at product level, choice archi-tecture to segment products could be further refined to integrate subtler elements with regard to materials and substances. Whereas there are generally positive attitudes towards source separation for solid waste (Song et al., 2016), errors in usage should be expected. Participants might mistakenly assign a specific type of WEEE to an incorrect container entry point. To prevent these problems, trans-parent containers with pictograms reminding the nature of WEEE could be displayed.

Intrinsic motivators triggered by feedback should be preferred over extrinsic motivators based on monetary rewards. According to our calculations, the maximum monetary reward would be approximately USD 5 per student for SHA. This is the value that could be retrieved from secondary materials. In reality, it would be lower as logistic and recycling costs would need to be taken into account. A low monetary reward usually fails to trigger an intended behaviour (Kamenica, 2012). In addition, Benabou and Tirole (2003) argue that incentives are weak behaviour reinforcers over the short term and have a negative impact over the long term.

Feedback could be produced for each item deposited in the con-tainer. According to the product weight and type, the amount of energy and materials saved to produce its equivalent for primary materials could be displayed on a screen. This type of information is a valuable intrinsic motivator (Baxter and Gram-Hanssen, 2016). Feedback designed to foster intrinsic motivators is advocated by Schultz et al. (1995) and technology can produce individualised messages to engage with sustainable behaviour change (Tompson et al., 2015). Li et al. (2015) support the view that long term improvements to the WEEE current situation should be sus-tained by local consumer participation. Keramitsoglou and Tsagarakis (2013) argue that empowerment leads to positive beha-viour change.

As a result, efforts to improve collection points based on choice architecture should not exceed USD 124,000 for the University of

Southampton and at UK Higher education level should not exceed USD 11 million (Table 7).

5. Conclusions and recommandations

This study has confirmed the validity of the DUM concept to evaluate secondary resources potential. At UK higher education level, a value of USD 2.35 million lies with SKA and USD 9.4 million with PCA, bringing the value of SHA DUMs to a value in excess of USD 11 million. Within a defined urban area, large amounts of cop-per and ferrous metals can be sourced and directed towards pro-cessing centres.

DUMs are composed of stocks that need to be tapped into and negative flows such as discarding that need to be diverted. To reduce these stocks and redirect these flows, end-of-use decisions need to be altered durably. Consumers' end-of-use decisions takes into account their product's type and status. For long term changes, collection systems need to acknowledge the factors influencing end of use decisions.

Behavioural economics, and more precisely choice architecture, acknowledge these variable factors by offering a set of techniques that can be freely adapted to shape behaviours towards desired outcomes. Choice architecture applied to a DUM is a step towards unlocking additional resources. These concepts evaluate available resources potential and propose solutions taking into account con-sumers' decision-making process. In addition to advocating the use of behavioural economics to improve WEEE end of use decisions, this study is the first attempt to quantify the value of an existing DUMs.

Key recommendations from this study are:

Collection containers should be designed to guide consumers to sort between broken or unbroken WEEE.
Feedback should be based on benefits provided for each action performed.

Intrinsic motivators fostering positive emotions should be pre-ferred over extrinsic motivators.
Enhanced collection system investments should not exceed anticipated materials retrieval value.

Some limitations were identified. SKA evaluation at UK level is not as precise as for PCA. SHA replacement rate was not measured in this study and only estimated according to students' average stay on campus during their studies. Data accuracy could be improved due to time lag between SHA ownership levels and materials valuation. DUM valuation could be increased by identify-ing the value of items to be reused. Future studies could focus on the cost of communicating, collecting and processing these streams to refine the necessary investment to improve choice architecture in specific DUMs.

References

Afroz, R., Masud, M.M., Akhtar, R., Duasa, J.B., 2013. Survey and analysis of public knowledge, awareness and willingness to pay in Kuala Lumpur, Malaysia – a case study on household WEEE management. J. Clean. Prod. 52, 185–193.

Ajzen, I., 1985. From intentions to actions: a theory of planned behavior. In: Kuhl, P. D.J., Beckmann, D.J. (Eds.), Action Control, SSSP Springer Series in Social Psychology. Springer, Berlin Heidelberg, pp. 11–39.

Ariely, D., Bracha, A., Meier, S., 2009. Doing good or doing well? Image motivation and monetary incentives in behaving prosocially. Am. Econ. Rev. 99 (1), 544– 555.

Balleine, B.W., Dezfouli, A., Ito, M., Doya, K., 2015. Hierarchical control of goal-directed action in the cortical–basal ganglia network. Curr. Opinions Behav. Sci., Neuroecon. 5, 1–7.

Baxter, J., Gram-Hanssen, I., 2016. Environmental message framing: enhancing consumer recycling of mobile phones. Resour., Conserv. Recy. 109, 96–101.

Benabou, R., Tirole, J., 2003. Intrinsic and extrinsic motivation. Rev. Econ. Stud. 70, 489–520.

Binnemans, K., Jones, P.T., Blanpain, B., Van Gerven, T., Yang, Y., Walton, A., Buchert, M., 2013. Recycling of rare earths: a critical review. J. Clean. Prod. 51, 1–22.
Bouton, M.E., 2014. Why behavior change is difficult to sustain. Prev. Med., Behav. Change, Health, Health Disparities 68, 29–36.
Brown, Z.S., Johnstone, N., 2014. Better the devil you throw: experience and support for pay-as-you-throw waste charges. Environ. Sci. Pol. 38, 132–142.
Caniato, M., Vaccari, M., Visvanathan, C., Zurbrügg, C., 2014. Using social network and stakeholder analysis to help evaluate infectious waste management: a step towards a holistic assessment. Waste Manage. 34, 938–951.
Chan, K., 1998. Mass communication and pro-environmental behaviour: waste recycling in Hong Kong. J. Environ. Manage. 52, 317–325.
Cossu, R., Williams, I.D., 2015. Urban mining: concepts, terminology, challenges. Waste Manage., Urban Min. 45, 1–3.
Darby, L., Obara, L., 2005. Household recycling behaviour and attitudes towards the disposal of small electrical and electronic equipment. Resour. Conserv. Recycl. 44, 17–35.

Directive 2002/96/EC of the European Parliament and of the Council of 27 January 2003 on waste electrical and electronic equipment (WEEE) – Joint declaration of the European Parliament, the Council and the Commission relating to Article 9, Official Journal L 037, 13/02/2003, pp. 0024–0039.
Directive 2011/65/EU of the European Parliament and of the Council of 8 June 2011 on the restriction of the use of certain hazardous substances in electrical and electronic equipment. Official Journal L174, 01/07/2011. P.0088-0110.
Eurostat, 2016. Waste Electrical and Electronic Equipment (WEEE) – Eurostat [WWW Document]. <http://ec.europa.eu/eurostat/web/waste/key-waste-streams/weee> (accessed 9.3.16).
Fraige, F.Y., Al-khatib, L.A., Alnawafleh, H.M., Dweirj, M.K., Langston, P.A., 2012. Waste electric and electronic equipment in Jordan: willingness and generation rates. J. Environ. Planning Manage. 55, 161–175.
Fishbein, M., Ajzen, I., 1975. Belief, Attitude, Intention, and Behavior: An Introduction to Theory and Research. Addison-Wesley Publication Company.
Guillard, V., Pinson, C., 2012. Toward a better understanding and measurement of consumer hoarding. Res. Appl. Market. 27, 57–78.
HESA, 2016, Higher Education Statistics Agency. Hesa.ac.uk. Available at: <https:// www.hesa.ac.uk/stats> (accessed 1 Jun. 2016).
Jones, N., Evangelinos, K., Halvadakis, C.P., Iosifides, T., Sophoulis, C.M., 2010. Social factors influencing perceptions and willingness to pay for a market-based policy aiming on solid waste management. Resour. Conserv. Recycl. 54, 533–540.
Kamenica, E., 2012. Behavioral economics and psychology of incentives. Ann. Rev. Econ., Rev. Adv. 4, 1–26.
Keramitsoglou, K.M., Tsagarakis, K.P., 2013. Public participation in designing a recycling scheme towards maximum public acceptance. Resour. Conserv. Recycl. 70, 55–67.

Krook, J., Carlsson, A., Eklund, M., Frändegård, P., Svensson, N., 2011. Urban mining: hibernating copper stocks in local power grids. J. Clean. Prod. 19, 1052–1056.
de Leeuw, A., Valois, P., Ajzen, I., Schmidt, P., 2015. Using the theory of planned behavior to identify key beliefs underlying pro-environmental behavior in high-school students: Implications for educational interventions. J. Environ. Psychol. 42, 128–138.
Li, J., Zeng, X., Chen, M., Ogunseitan, O.A., Stevels, A., 2015. ''Control-Alt-Delete'': rebooting solutions for the E-waste problem. Environ. Sci. Technol. 49, 7095–7108.
Lme.com, 2016. London Metal Exchange: Home. Available at: <https://www.lme.com/> (accessed 11 Jul. 2016).
Martinho, G., Pires, A., Saraiva, L., Ribeiro, R., 2012. Composition of plastics from waste electrical and electronic equipment (WEEE) by direct sampling. Waste Manage. 32, 1213–1217.
Milovantseva, N., Saphores, J.-D., 2013. Time bomb or hidden treasure? Characteristics of junk TVs and of the US households who store them. Waste Manage., Spec. Thematic Issue: Urban Min. Urban Min. 33, 519–529.
Mueller, S., Wäger, P.A., Widmer, R., Williams, I.D., 2015. A geological reconnaissance of electrical and electronic waste as a source for rare earth metals. Waste Manage. 45, 226–234.
Ongondo, F.O., Williams, I.D., 2011a. Greening academia: use and disposal of mobile phones among university students. Waste Manage. 31, 1617–1634.
Ongondo, F.O., Williams, I.D., 2011b. Mobile phone collection, reuse and recycling in the UK. Waste Manage. 31, 1307–1315.
Ongondo, F.O., Williams, I.D., Whitlock, G., 2015. Distinct urban mines: exploiting secondary resources in unique anthropogenic spaces. Waste Manage., Urban Min. 45, 4–9.

Pakpour, A.H., Zeidi, I.M., Emamjomeh, M.M., Asefzadeh, S., Pearson, H., 2014. Household waste behaviours among a community sample in Iran: an application of the theory of planned behaviour. Waste Manage., Waste Manage. Asia 34, 980–986.

Read, A.D., 1999. ''A weekly doorstep recycling collection, I had no idea we could!'': overcoming the local barriers to participation. Resour. Conserv. Recycl. 26, 217– 249.

Richetin, J., Sengupta, A., Perugini, M., Adjali, I., Hurling, R., Greetham, D., Spence, M., 2010. A micro-level simulation for the prediction of intention and behavior. Cogn. Syst. Res. 11, 181–193.
Saphores, J.-D.M., Ogunseitan, O.A., Shapiro, A.A., 2012. Willingness to engage in a pro-environmental behavior: an analysis of e-waste recycling based on a national survey of U.S. households. Resour. Conserv. Recycl. 60, 49–63.
Schultz, P.W., Oskamp, S., Mainieri, T., 1995. Who recycles and when? A review of personal and situational factors. J. Environ. Psychol. 15, 105–121.
Schwartz, S.H., 1968. Awareness of consequences and the influence of moral norms on interpersonal behavior. Sociometry 31, 355–369.

Shaw, P.J., Maynard, S.J., 2008. The potential of financial incentives to enhance householders' kerbside recycling behaviour. Waste Manage. 28, 1732–1741.
Song, Q., Wang, Z., Li, J., 2016. Exploring residents' attitudes and willingness to pay for solid waste management in Macau. Environ. Sci. Pollut. Res. 23, 16456– 16462.

Strömberg, H., Rexfelt, O., Karlsson, I.C.M., Sochor, J., 2016. Trying on change – trialability as a change moderator for sustainable travel behaviour. Travel Behav. Soc. 4, 60–68.

Sunstein, C.R., Thaler, R.H., 2012. Nudge: Improving Decisions About Health. Penguin, Wealth and Happiness.
Sun, Z.H.I., Xiao, Y., Sietsma, J., Agterhuis, H., Visser, G., Yang, Y., 2015. Characterisation of metals in the electronic waste of complex mixtures of end-of-life ICT products for development of cleaner recovery technology. Waste Manage. 35, 227–235.

Sun, Z., Xiao, Y., Agterhuis, H., Sietsma, J., Yang, Y., 2016. Recycling of metals from u ban mines – a strategic evaluation. J. Clean. Prod. 112 (Part 4), 2977–2987.
Tesco.com, 2016. Buy Home Electrical from Our Range - Tesco. Available at: <http:// www.tesco.com/direct/home-electrical/?icid=TopNav_flyoutlink_ HomeElectrical> (accessed 2 Nov. 2016).
Thaler, R.H., Sunstein, C.R., Balz, J.P., 2014. Choice architecture. In: The Behavioral Foundations of Public Policy. Eldar Shafir, pp. 428–439.
Thecompleteuniversityguide.co.uk., 2016. Home. Available at: <http://www. thecompleteuniversityguide.co.uk/> (accessed 31 Oct. 2016).
Timlett, R.E., Williams, I.D., 2008. Public participation and recycling performance in England: a comparison of tools for behaviour change. Resour. Conserv. Recycl. 52, 622–634.

Tompson, S., Lieberman, M.D., Falk, E.B., 2015. Grounding the neuroscience of behavior change in the sociocultural context. Curr. Opin. Behav. Sci., Neuroecon. 5, 58–63.

Wallsten, B., Magnusson, D., Andersson, S., Krook, J., 2015. The economic conditions for urban infrastructure mining: using GIS to prospect hibernating copper stocks. Resour. Conserv. Recycl. 103, 85–97.

Wang, Z., Zhang, B., Yin, J., Zhang, X., 2011. Willingness and behavior towards e-waste recycling for residents in Beijing city, China. J. Clean. Prod. 19, 977–984.

Werner, C.M., 2003. Changing homeowners' use of toxic household products: a transactional approach. J. Environ. Psychol. 23, 33–45.

Williams, I.D., 2015. A change of emphasis: waste to resource management, pp. 207–252. In: Harrison, R.M., Hestor, R.E. (Eds.), Issues in Environmental Science and Technology No. 40. Still Only One Earth: Progress in the 40 Years Since the First UN Conference on the Environment. RSC Publishing, Cambridge. Print ISBN: 978-1-78262-076-1. PDF eISBN: 978-1-78262-217-8. EPUB eISBN: 978-1-78262-644-2.

Wrap.org.uk, 2012. Available at: <http://www.wrap.org.uk/sites/files/wrap/ Electrical%20product%20material%20composition%20overview.pdf> (accessed 1 Jun. 2016).

Waste and Resources Action Programme (WRAP), 2016. Increase Your Small WEEE Collection Rates | WRAP UK. Available at: <http://www.wrap.org.uk/content/ increase-your-small-weee-collection-rates> (accessed 11 Jun. 2016).

Yao, R., Steemers, K., 2009. Overview of an innovative EU–China collaboration in education and research in sustainable built environment. Renew, Energy, Spec. Issue: Build. Urban Sustain. 34, 2080–2087.

Zen, I.S., Siwar, C., 2015. An analysis of household acceptance of curbside recycling scheme in Kuala Lumpur, Malaysia. Habitat Int. 47, 248–255.

Zeng, X., Yang, C., Chiang, J.F., Li, J., 2017. Innovating e-waste management: from macroscopic to microscopic scales. Sci. Total Environ. 575, 1–5.

Zhang, K., Kleit, A.N., 2016. Mining rate optimization considering the stockpiling: a theoretical economics and real option model. Resour. Pol. 47, 87–94.