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Unintended consequences of secondary legislation: A case study of the UK landfill tax (qualifying fines) order 2015

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ABSTRACT

Increasing attention is being paid to the use of policy instruments in promoting progressive waste management and supporting the transition to a circular economy. To be effective in this context, instruments must be balanced, providing the correct amount of sanction and incentive to ensure environmental protection, enhance resource recovery, and promote innovation and investment in beneficial technologies. Focusing on the UK landfill tax, and adopting a stakeholder-oriented approach, this paper presents a case study illustrating how the ineffective implementation of secondary legislation can have unintended consequences on the aims of primary legislation. Specifically, it examines the Landfill Tax (Qualifying Fines) Order 2015 (QFO), which introduced a Loss On Ignition (LOI) test regime to classify fines for tax purposes. Results from a stakeholder survey (n = 44) revealed that the introduction of the QFO has dis-incentivised material recovery and discouraged investment in separation technologies, thereby creating a perverse incentive to landfill waste. Major weaknesses identified include the poorly defined LOI test regime, the timing of and responsibility for conducting LOI testing, the lack of compliance checks, and the marked discontinuity in tax rates at the somewhat arbitrary 10% LOI threshold. Furthermore, the system was widely viewed to be open to abuse by unscrupulous traders. A set of recommendations are made to address these shortcomings, where it is proposed that the LOI threshold should be replaced by multiple tax bands or a sliding scale and ideally combined with a direct incentive for investment such as an enhanced capital allowance for resource efficient technologies.

1. Introduction

Transitioning from a linear to a Circular Economy (CE) could overcome consequences of unsustainable consumption such as environmental degradation, resource depletion, and climate change (Moreno et al., 2016). A CE mimics a natural biological system by recirculating resources through successive generations, where resource efficiency is promoted through the optimisation of production systems, resource utility is maintained by extracting the maximum value when in use, and any remaining value at end-of-life is recovered through progressive waste management strategies (Smol et al., 2015).

There is now growing attention on the role of policy in delivering the CE. Soderman et al. (2016) notes that the European Union (EU) is increasingly recognising the role of policy in supporting the transition from end-of-pipe waste management to efficient resource management, whilst Jimenez-Rivero and Garcia-Navarro (2017) highlight the need for government to strengthen and enforce instruments that adhere to CE principles. One CE-aligned approach is the use of Extended Producer Responsibility (EPR), which places responsibility for end-of-life management with the producer (Lindhqvist, 2000). Currently the use of EPR (in the EU and elsewhere, e.g. Mrkajić et al., 2018; Wang et al., 2018) is restricted at a practical level to packaging waste, waste electrical and electronic equipment, end-of-life vehicles, and hazardous household wastes (Lifset et al., 2013). For an ideal CE approach, this would extend to up-stream issues such as eco-design, along with full internalisation of waste management costs to shift responsibility from taxpayers and local authorities to companies and consumers (Lifset et al., 2013). While this may be realised in the future, during the transition end-of-pipe waste management remains a key concern. Indeed, EU policy initiatives, the most recent being the 'Circular Economy Package (CEP)' (2015-ongoing; European Commission EC, 2016), place an increased emphasis on both CE models and the efficient use of wastes (Gregson et al., 2015; Smol et al., 2015).

With respect to waste management, two key EU directives are the Waste Framework Directive (WFD) (2008/98/EC), which introduces the waste hierarchy and sets recycling targets, and the Landfill Directive (LD) (1999/31/EC), which sets targets requiring a reduction in the landfilling of biodegradable and other polluting solid wastes

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(European Commission EC, 2008, 1999). Both the WFD and the LD have been amended by the CEP, which reiterates the waste hierarchy, strengthens recycling targets, and extends landfill diversion targets to include all municipal wastes (European Commission EC, 2015a,b).

Although all member states are obliged to transpose EU directives into national policy, economic and social differences between countries are reflected in the disparity of waste management systems employed (Mihai and Apostol, 2012; Pires et al., 2011). Concerning landfill diversion, several countries have achieved very low landfilling rates, where this has been attributed to effective national policy and the use of fiscal measures such as Landfill Taxes (LFTs) (European Environment Agency EEA, 2000; Mazzanti et al., 2009).

While LFTs have been successful in diverting waste from landfill, to what extent they promote material recovery is less clear. The financial competitiveness of secondary materials can be enhanced through taxation on competing virgin materials or on waste disposal, where Solderholm (2011) argues that the latter can be more effective due to low administration costs and increased policy acceptance. However, Martin and Scott (2003) found that while the United Kingdom (UK) LFT had increased landfill diversion, it had been less successful in promoting the top waste hierarchy priorities. Likewise, in an EU-wide study, Mazzanti and Zoboli (2008) concluded that while LFTs can lead to the management of waste being promoted up the waste hierarchy (to recovery or recycling), they do not create a backwards incentive to reduce waste generation. To address such issues, researchers have called for a re-framing of the waste hierarchy in terms of resource use and productivity, arguing that this would help policy makers ensure that they not only disincentivise disposal, but also adequately incentivise preferred environmental options (Gharfalkar et al., 2015; Van Ewijk and Stegemann, 2014).

Another factor that requires consideration is the evolution of policy instruments in response to technological advancements in waste processing, with particular attention paid to the interaction between the negative externalities of pollution and the positive externalities of technological innovation (Leme et al., 2014; Luz et al., 2015). Jaffe et al. (2003) argue that policies targeting pollution reduction should also support technological change. Thus, there is a case for combining environmental taxes with direct incentives if the signal from a single instrument is insufficient to promote innovation and adoption of beneficial technologies (Jaffe et al., 2005). Likewise, Bennear and Stavins (2007) argue that in such “second-best” settings, which are common in the area of environmental and resource management, the use of multiple instruments is both the norm and justifiable. However, they also caution that this requires a high level of policy coordination, which may extend to an instrument designed to address one issue being modified in light of another to achieve an overall positive outcome (Bennear and Stavins, 2007).

While the design of appropriate policy instruments is clearly important, it is equally important to ensure the desired impact is achieved through effective implementation (Soderman et al., 2016). In this context, Bailey and Rupp (2005) contend that implementation cannot be fully understood or improved without due consideration of stakeholder perspectives, arguing that industry is uniquely placed to make a valuable contribution towards understanding the strengths and weaknesses of environmental policy instruments. Indeed, numerous stakeholder-related factors have been identified that could undermine implementation, including a lack of competent staff, ineffective administrative capabilities, incoherent or uncomprehensive written documentation, poor inter-organisational communication and support, a lack of cooperation, and competing priorities (Bailey and Rupp, 2005; Khan and Khandaker, 2016; Mosannenzadeh et al., 2017; McTigue et al., 2018). In relation to the latter point, Bailey and Rupp (2005) found that eco-taxes may be counter-productive if a reduction in profitability leads to the de-prioritisation of environmental issues. This again highlights the need to find a balance between competing priorities (or multiple market failures) in waste management policy in order to encourage development of optimal systems. Otherwise, under-regulation may lead to the careless handling of wastes, while over-regulation, regulation that is unclear, or an absence of compensatory incentives, may hinder the re-use of waste materials by creating excessive bureaucracy and stifling innovation (Gharfalkar et al., 2015; Jaffe et al., 2005).

This paper presents a case study illustrating how the ineffective implementation of secondary legislation can have unintended consequences on achieving the aims of primary legislation. Focusing on the UK LFT, it employs a stakeholder survey to examine how the introduction of the Landfill Tax (Qualifying Fines) Order 2015 (QFO) (House of Commons HoC, 2015), a statutory instrument used to classify waste, has impacted on stakeholders. Expanding on a preliminary analysis (Fletcher et al., 2017) it examines how the QFO may disincentivise material recovery and thereby limit landfill diversion, where consideration is given to potential modifications that would ensure sufficient environmental protection while enhancing the economic viability of waste processing. The paper is structured as follows. Section 2 outlines the development of the UK LFT and QFO. Section 3 details the methods used to conduct the analysis. Section 4 discusses stakeholder views on the design and implementation of the QFO, highlighting barriers to material recovery and landfill diversion, and suggesting potential policy developments. Finally, Section 5 reviews the wider implications and conclusions of the study.

2. The UK landfill tax

The UK LFT facilitates the implementation of the LD (Calaf-Forn et al., 2014; Morris et al., 2000), and was introduced in the 1996 Finance Act (HM Stationary Office HMSO, 1996) and modified in the Landfill Tax (Amendment) Regulation 2009 (HM Stationary Office HMSO, 2009). A regulatory incentive administrated by Her Majesty’s Revenue and Customs (HMRC), the LFT applies differential tax rates to wastes disposed of to landfill in order to reflect the environmental burden of this disposal option (Calaf-Forn et al., 2014; Grigg and Read, 2001; Morris et al., 2000). It defines inert (or inactive) waste, which qualifies for a lower tax rate, as non-hazardous (as described by the WFD), with a low Greenhouse Gas (GHG) emission potential (not biodegradable) and low polluting potential (contaminants unlikely to become mobile or leach). Any waste that does not conform to these criteria is classed as active and is liable for the standard tax rate (HMRC, 2016a). In accordance with Section 42(2) of the Finance Act 1996(a), a definitive list of materials that were deemed to meet the definition of inert waste (for the purposes of setting the LFT rate, and based on well characterised properties) was published. Originally delivered through the Landfill Tax (Qualifying Materials) Order 1996 (QMO) and updated in 2011, the materials listed include; naturally occurring materials (rocks, sand and soils), low activity processed materials (glass, ceramics or concrete), processed or prepared minerals (silica, mica or clay), furnace slags, ash, low activity inorganic compounds, calcium sulphate, and calcium hydroxide (including brine) (House of Commons HoC, 2011, 1996).

When first introduced, the LFT rates were £2/tonne for inert waste and £7/tonne for active waste, thus with gate fees of around £5–£15 (ENDS, 1994) total disposal costs remained relatively low. As such, the LFT provided little financial incentive for diversion and had minimal effect on the amount of waste being disposed to landfill (Martin and Scott, 2003). To address this legislative failure, the LFT escalator was introduced (HM Treasury, 1999; Martin and Scott, 2003), where the price of landfilling active waste increased by a fixed amount each year from 2000 to 2014. Since 2015, both the active and inert tax rates have been index linked (HMRC, 2016b), standing at £84.40/tonne for active waste and £2.65/tonne for inert waste in 2016/17 (HMRC, 2016a). Although gate fees have also increased (partly reflecting improved landfill management practices) they have been relatively stable since 2008, with a mean of £22/tonne in 2016 (The Waste and Resources