FACETS
a multidisciplinary open access science journal

RESEARCH ARTICLE

The role of management instruments in the diversion of organic municipal solid waste and phosphorus recycling

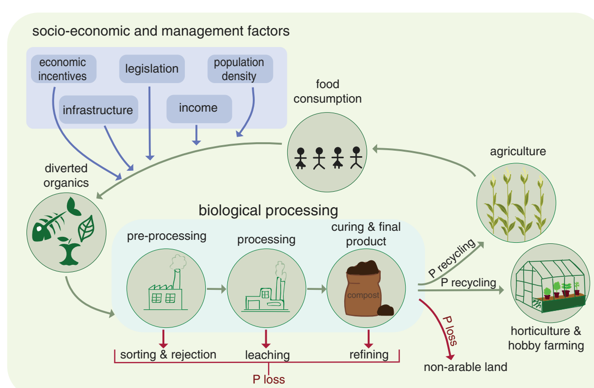
Jillian L. Treadwell[a*], Elena M. Bennett[bc], and O. Grant Clark[ac]

[a]Department of Bioresource Engineering, Faculty of Agricultural and Environmental Sciences, McGill University, 21,111 Lakeshore Road, Ste-Anne de Bellevue, QC H9X 3V9, Canada; [b]Department of Natural Resource Sciences, Faculty of Agricultural and Environmental Sciences, McGill University, 21,111 Lakeshore Road, Ste-Anne de Bellevue, QC H9X 3V9, Canada; [c]McGill School of Environment, McGill University, 3534 University Street, Montreal, QC H3A 2A7, Canada

*jilliantreadwell@gmail.com

Abstract

Organic waste, which contains essential plant nutrients such as phosphorus, constitutes 30%–50% of municipal solid waste in developed countries. Unfortunately, much of this resource is buried in landfills or incinerated. Many jurisdictions have, therefore, adopted the diversion of organic waste and the recycling of nutrients as policy goals. We used data sets from Europe and Ontario, Canada, to explore the impact of socio-economic and management factors on the rates of organic waste diversion and examined the effect of this diversion on phosphorus recycling. Organic diversion rates were highly correlated with income in Europe and with infrastructure, such as source-separated organic waste collection, in Ontario. Significant correlations were also observed between diversion rates and the use of policy instruments such as economic incentives, legislative organic waste bans, and curbside bag limits. We estimated that 39%–63% of the phosphorus in diverted organics is returned to arable land. Ultimately, we found that although socio-economic factors influence the success of organic waste diversion, policies, accessible infrastructure, economic incentives, and legislative requirements can be leveraged to improve the recycling rate of organic waste and the nutrients they contain.



Key words: phosphorus, organic waste, recycling, waste diversion, waste management, nutrient management

OPEN ACCESS

Citation: Treadwell JL, Bennett EM, and Clark OG. 2018. The role of management instruments in the diversion of organic municipal solid waste and phosphorus recycling. FACETS 3: 896–919. doi:10.1139/facets-2018-0005

Handling Editor: Jeffrey Hutchings

Received: February 9, 2018

Accepted: August 5, 2018

Published: September 27, 2018

Published by: Canadian Science Publishing

Introduction

Organic waste constitutes approximately 30%–50% of municipal solid waste streams in developed countries (Ponsá et al. 2008; Fehr and Arantes 2015; Recyc-Québec 2015). Where conventional waste management is practiced, much of this organic waste is buried in landfills, incinerated (with the ash sent to landfill), or stabilized and used as landfill cover (Tsai 2008; Senthilkumar et al. 2014; Metson and Bennett 2015a; Zabaleta and Rodic 2015). The disposal of organic waste in landfills is associated with greenhouse gas emissions and nutrient loading to waterways (Augenstein 1992; Bulkeley and Askins 2009). Landfilling organic waste also contributes to a wasteful one-way flow of nutrients through the anthropogenic food system (Nilsson 1995; Smil 2000). This paper focuses predominantly on the organic fraction of municipal solid waste (OFMSW), which includes biodegradable waste material such as food waste and leaf and yard waste that is produced and disposed of by the public, and often managed centrally. It is sometimes referred to as bio-waste, green waste, biodegradable municipal waste (BMW), organic waste, and (or) putrescible waste. A description of OFMSW, as it pertains to this paper, along with explanations for other relevant terms, can be found in **Box 1**.

Landfilling and incineration of organic waste, combined with agricultural fertilization using non-renewable mineral ores, has substantially altered many nutrient cycles, depleting usable stocks of these nutrients and changing the way they flow through the environment at both global and local scales (Smil 2000; Schlesinger and Bernhardt 2013). Over the last century, global phosphorus (P) cycles have seen intense acceleration caused primarily by mining to produce agricultural fertilizer (Tilman et al. 2002; Cordell and White 2011). This acceleration has multiplied P mobilization by up to four-fold since the Green Revolution resulting in an increase of P delivery to oceans (Bennett et al. 2001; Childers et al. 2011). At the same time, ever-increasing flows of food and other raw materials into cities have led to high levels of nutrient accumulation in urban areas and their landfills (Kennedy et al. 2007; Grimm et al. 2008; Neset and Cordell 2012; Lin et al. 2014).

The acceleration and disruption of the global P cycle results in two principal problems. First, there are long-term issues of P scarcity because of the non-renewable and geographically concentrated nature of phosphate rock reserves (Cooper et al. 2011; Scholz and Wellmer 2013). Phosphorus scarcity raises issues related to food security as P is a non-substitutable plant macronutrient (Cordell et al. 2009; Childers et al. 2011). The second problem is that P runoff from agriculture and urban systems can act as a powerful aquatic pollutant, creating widespread issues related to eutrophication (Carpenter et al. 1998). This can have detrimental consequences for waterways and coastal ecosystems including declines in coral reef health and loss of communities, increased incidence of fish kills, and reductions in species diversity (Smith and Schindler 2009).

Although there is no "silver bullet" solution to the current dependence on synthetic fertilizers and the disruption of biogeochemical cycles, opportunities exist both in agriculture and in waste management; this paper focuses on the waste management aspect. Some organic waste management strategies facilitate the recycling of nutrients back to arable soil and, thus, have the potential to reduce aquatic pollution and simultaneously alleviate nutrient limitations; this can play a key role in a larger, more comprehensive solution addressing issues of altered biogeochemical cycles (Burnley 2001; Taşeli 2007; Cordell and White 2013; Mayer et al. 2016). Increased diversion of the OFMSW from landfills to more sustainable options such as biological processing is a policy goal in many regions, echoed in municipal, provincial, national, and international waste management plans and policies (Conseil fédéral Suisse 1990; European Commission 1999; Québec 2011; City of Toronto 2016). These management plans generally list several reasons for promoting organic waste diversion, including the potential to turn waste into a valuable resource through the recycling of important nutrients (e.g., plans from Metro Vancouver (2010) and the Region of Peel (2009)). Increasing the recycling of nutrients such as P through organic waste diversion can reduce demand for mined P, convert waste

Box 1. Definitions and terms.

Dry recyclable materials: non-organic source-separated materials that can be recycled; for example, glass (e.g., bottles and jars), metal (e.g., aluminum cans), paper, cardboard, and plastics (e.g., rigid containers, bags, etc.).

European Union Landfill Directive (Directive 1999/31/EC): a regulation passed in 1999 that directs waste management and landfill disposal in the European Union.

The Directive outlines procedures that must be followed and includes a list of materials that may not be accepted at landfills. The Directive includes an organic waste to landfill ban that was phased in over three stages as the diversion restrictions increased from 0% to 65%. As many countries were granted varying derogation periods, the annual requirements varied across countries.

Landfill fee: a charge that is set by the operator of a landfill for the disposal of waste at a landfill site (refers to a typical gate fee or tipping fee).

Landfill tax: a levy charged by a public authority for the disposal of waste (in this study, these taxes are only assessed in the European Union case study) (Watkins et al. 2012).

Landfill charge: the sum of landfill fee and landfill tax; the total charge for the disposal of waste in a landfill (Watkins et al. 2012).

Organic fraction of municipal solid waste (OFMSW): includes biodegradable waste material such as food waste and leaf and yard waste that is produced and disposed of by the public, and often managed centrally. It is sometimes referred to as bio-waste, green waste, biodegradable municipal waste (BMW), organic waste, and (or) putrescible waste.

Organic waste ban: a specific type of waste disposal ban that prohibits the disposal of organic waste (including food waste and leaf and yard waste) to landfills and, in some cases, waste-to-energy facilities.

Organic waste diversion: management of organic waste so that it is sent to an alternative processing facility such as a composting plant or an anaerobic digester instead of being buried in landfills or incinerated.

Organic waste diversion rate: the amount of organic waste diverted to biological treatment facilities as a fraction of the municipal organic waste generated in the region.

Reapplication: refers to the application of processed organic waste (e.g., compost or digestate) to arable land.

Recycling: the conversion of waste material into a reusable material.

Source-separated organics (SSO): organic waste that has been separated at the source (e.g., family residence, restaurant, or business) from other waste streams by the generator/user prior to any pre-processing. It does not include human excreta.

Waste disposal ban: legislation or policy that discourages or prohibits the disposal of specific materials or waste streams at landfills and (or) waste-to-energy facilities (sometimes referred to as a disposal ban or a waste ban).

into a resource, and reduce total waste disposal (Nilsson 1995; Eriksson et al. 2005; Childers et al. 2011; Neset and Cordell 2012).

Alternative processes for managing urban organic waste, such as composting and anaerobic digestion (AD), may help increase the amount of P recycled and reduce pollution (Giusquiani et al. 1995; Childers et al. 2011; Agegnehu et al. 2015; Mayer et al. 2016). Biological processing (e.g., composting and AD) of the OFMSW converts waste material into nutrient-rich, biologically stable end products such as compost and digestate (Levis et al. 2010; Cordell et al. 2011; Senthilkumar et al. 2014). In the case of anaerobic digestion, biological processing also produces valuable biofuel (Khalid et al. 2011). The subsequent application of these end products to land recycles residual organics into useful resources, such as P-rich soil amendment, which can increase soil fertility, enhance agricultural carbon sinks, decrease methane emissions associated with landfilling organic waste, and reduce our reliance on chemical fertilizers (Levis et al. 2010; Dorward 2012; Neset and Cordell 2012). There are various barriers to the application of municipal organic amendments to arable land, such as the proximity of agricultural land, market demand, the presence of contaminants, mineral fertilizer substitutability, and processing and transportation costs. This study, however, focuses primarily on the factors that drive organic waste diversion and the fate of P in organic waste once it is diverted and, therefore, the *potential* for P recycling.

The majority of existing P recovery research has focused on wastewater flows, resulting in a lack of monitoring and data for P recycling associated with solid waste management (Jaffer et al. 2002; Elser and Bennett 2011; Cordell et al. 2012; Linderholm et al. 2012). This data gap means that the effectiveness of organic solid waste processing in real-world contexts is inadequately understood (Cordell et al. 2012; Senthilkumar et al. 2014). Much urban P research to date ignores the OFMSW diversion stream (Antikainen et al. 2005; Garnier et al. 2015), explicitly or implicitly assumes all P contained in diverted OFMSW is returned to arable land (Neset 2005; Neset et al. 2008; Senthilkumar et al. 2014), or neglects the link between diversion and application, looking solely at the land application of compost (Cooper and Carliell-Marquet 2013). Solid waste management provides an important opportunity to increase overall P recycling in a region, given that similar annual tonnages of P output have been found for solid waste as for wastewater in regional systems (Kalmykova et al. 2012), and equal per capita P output was found for sewage sludge and solid waste at a multi-national scale (Ott and Rechberger 2012); however, the flow of P through waste systems must be better understood if organic waste management is to be used as a means to increase P recycling and reuse.

Organic solid waste diversion management and policy: Europe and Canada

Policy and management instruments used to promote OFMSW diversion from landfills and incinerators include landfill taxes (Bulkeley and Askins 2009; Watkins et al. 2012), organic waste bans (Burnley 2001; Metro Vancouver 2015), and the development of sophisticated organic solid waste collection systems and infrastructure (Metcalfe et al. 2013). The success of diversion programs varies across regions, yet little is known about what makes some regions more successful than others. Although several authors have examined the impact of policy and socio-economic factors on the diversion of general waste (Watkins et al. 2012; Mueller 2013) or dry recyclable materials (i.e., glass, paper, cardboard, metal, and plastic; Ferrara and Missios 2005; Sidique et al. 2010; Gellynck et al. 2011), there is limited quantitative research on how such factors influence OFMSW diversion (Williams and Kelly 2003; Refsgaard and Magnussen 2009; Metson et al. 2015). Research examining the reapplication of processed organic waste and nutrients to arable land is even more sparse, especially in North America (Eriksson et al. 2005).

Studies of the diversion of general waste and dry recyclable materials found that a combination of various socio-economic and management factors are likely to play a role in waste diversion (Hornik et al. 1995; Sterner and Bartelings 1999; Nicolli et al. 2012; Watkins et al. 2012; Barr et al. 2013; Miafodzyeva and Brandt 2013; Mueller 2013; Miliute-Plepiene and Plepys 2014). Many studies, including two meta-analyses (Hornik et al. 1995; Miafodzyeva and Brandt 2013), point to infrastructure and convenience as important facilitators with high correlations with overall diversion rates (Hornik et al. 1995; Sterner and Bartelings 1999; Miafodzyeva and Brandt 2013; Miliute-Plepiene and Plepys 2014). Several studies also noted that fee structures (Sterner and Bartelings 1999), monetary rewards (Hornik et al. 1995), and economic disincentives, particularly pay-as-you-throw schemes, may increase diversion and reduce total waste generation, but that results are highly variable across regions (Watkins et al. 2012; Miliute-Plepiene and Plepys 2014). Socio-economic factors such as income, population density, and household size can also be important factors for predicting the recycling behaviour of households (Sterner and Bartelings 1999; Nicolli et al. 2012; Barr et al. 2013; Miafodzyeva and Brandt 2013), whereas some studies found that education, attitudes, and existing social norms may play important roles (Hornik et al. 1995; Sterner and Bartelings 1999; Watkins et al. 2012; Miafodzyeva and Brandt 2013). Finally, legislation and policies such as curbside bag limits and waste disposal bans are expected to increase diversion rates (Nicolli et al. 2012; Mueller 2013). However, it is unclear how these findings from general waste and dry recyclable material diversion translate for OFMSW diversion. More information is required to understand the factors that may impact the feasibility of organic waste management solutions in different locations, and how those solutions might be best implemented (Neset and Cordell 2012; Cordell and White 2013; Metson 2014).

Organic solid waste diversion and policy: European Union

Organic solid waste management strategies have been implemented across the globe, but European countries have historically been at the forefront of centralized organic waste diversion and processing. In 1999, the European Union (EU) passed the EU Landfill Directive (hereinafter referred to as the Directive) (European Commission 1999; European Environment Agency 2013). This legislation required member states to reduce the proportion of organic waste sent to landfill and was phased in over three stages where the diversion restrictions increased at each stage. The first phase of the Directive came into effect in 2006, when most EU countries were required to reduce their biodegradable municipal waste to 75% by weight compared with base year (1995) values. This value decreased to 50% by 2009, and to 35% by 2016. Some countries with a high dependence on landfill disposal (as of 1995) were given additional time (i.e., derogation periods); thus, annual requirements varied across countries. **Table S1** describes the diversion target years. Beyond the Directive, several countries and municipalities within the EU have implemented additional taxes, legislation, and policies to encourage organic waste diversion.

Organic solid waste diversion and policy: Ontario, Canada

Organic solid waste management in Canada lags behind that of Europe. Although some cities have recently implemented legislation and policies on organic waste diversion, there is no national legislation, and the only province-wide legislation banning the disposal of organic waste to landfill is in Nova Scotia (Wagner and Arnold 2008). Other provinces such as Quebec and Ontario have plans to implement province-wide bans on the disposal of organic waste to landfill. Ontario is the most populated province and largest waste producer in Canada. Although there is currently no province-wide legislation on the treatment of organic waste, some jurisdictions have introduced policies, waste management plans, and infrastructure to promote the diversion of organic waste (e.g., Guelph (City of Guelph 2014), Toronto (City of Toronto 2016), and Peel (Region of Peel 2009)).

Understanding the role of existing waste disposal bans, infrastructure, and organic waste diversion policies will provide insight at a time when such policies are gaining prevalence and momentum (Levis et al. 2010). In this study, we explore the role of socio-economic factors, policy, management, and infrastructure on OFMSW diversion rates across Europe and the province of Ontario, Canada, as well as the impact of organic waste diversion and processing on the potential and realized reapplication of P.

Methods

Data collection and synthesis

We collected information on the rates of OFMSW diversion as well as the various factors that we hypothesized might explain the differences in those rates. Using factor analysis and regression models, we explored key factors that influence these diversion rates. Subsequently, we reviewed the existing literature on waste and P flows through centralized organics processing facilities, as well as information on the distribution of end products. This information was used to give a more comprehensive picture of P flows through urban systems.

Waste diversion in Europe

To analyze organic waste diversion rates (i.e., the amount of organic waste diverted to biological treatment facilities as a fraction of the municipal organic waste generated in the region), we used two existing databases containing information on organic waste diversion in Europe between 1995 and 2013 (Eurostat 2014a) and in Ontario, Canada, between 2006 and 2013 (Waste Diversion Ontario 2016). The European data set included annual national values for municipal organic waste (tonnes) sent to biological treatment plants as well as total municipal waste generation (tonnes) for 32 countries in Europe (European Economic Area (EEA) countries, plus Switzerland) from 1995 to 2013. We coded each country based on its Directive derogation period (**Table S1**) and noted when each phase of the Directive was implemented. For simplicity, we will use the term Year 1 to refer to the base year—annual data from before the policy waste announced (1995 for most cases), and we will use the term Year 2 to refer to the second diversion target year (2009 for most non-derogation countries; further details are provided in **Table S1**). National waste composition statistics (Hogg et al. 2002; European Environment Agency 2013) were used to convert Eurostat total waste generation tonnages to organic waste generation tonnages to determine OFMSW diversion rates. Percentage data were used as they allowed for a simplified comparison across regions producing varying amounts of waste. The original data were checked for plausibility and corrected when necessary (**Tables S5**–**S9**).

Waste diversion in Ontario

In Canada, data are available from Waste Diversion Ontario (now the Resource Productivity and Recovery Authority; rpra.ca/datacall/about-the-datacall/) that include annual values for total residential waste generation (tonnes), as well as the proportion of total residential waste sent to biological treatment for over 100 jurisdictions in Ontario (including municipalities, counties, cities, and townships) from 2006 to 2013. Waste from industry and commerce are not included. Source-separated organics and leaf and yard waste are included in the calculations for organic waste diversion. The Waste Diversion Ontario data was used to determine the organic waste diversion rate assuming an average municipal waste composition of 40% organics (Statistics Canada 2005; Region of Waterloo and Golder Associates 2013). For ease of analysis, for Ontario we use the term Year 1 to represent 2006 and Year 2 to represent 2013.

Waste diversion factors

We examined two socio-economic factors and three management factors (**Table 1**). Socio-economic data were collected from 2004 and 2011 for population and income, respectively, in Europe; 2004 represents the median of the time period being examined (1995–2014), whereas 2011 had the most comprehensive income data for all EU countries (Eurostat 2014a). Socio-economic data were

Table 1. Factors used in analysis for Europe and Ontario.

Category	Group	Measurement
Socio-economic	Income	Mean income per capita
	Population	Total population
		Population density
Management	Economic incentives/disincentives	Landfill taxes and fees
		Incineration taxes and fees
		Pay-as-you-throw schemes
	Legislation and policy	Regional waste management regulations
		Curbside bag limits
		Mandatory source separation of waste streams
	Centralized infrastructure	Presence of centralized infrastructure for residential source-separated organics

collected from 2011 for Ontario, as it was the most comprehensive census year available during the study period (2006–2013; Statistics Canada 2011a, 2011b). Management factor data were drawn from published reports for Europe (Sahlin et al. 2007; World Bank 2011; Confederation of European Waste-to-Energy Plants (CEWEP) 2012; Watkins et al. 2012; European Environment Agency 2013) and additional Waste Diversion Ontario data sets, as well as regional websites and municipal government contacts for Ontario (**Tables S10–S11**). Data for management factors in Europe was limited primarily to 2012, whereas we predominantly used 2011 data for Ontario. Missing values were filled with the nearest available value from the same jurisdiction from either the prior or following years. Missing values that remained were estimated from regional means and grey literature values (**Tables S5–S12**).

P recycling in diverted organics

Organic solid waste diversion does not necessitate nutrient and P recycling; however, higher rates of diversion do facilitate better nutrient recycling. Although diversion of the OFMSW facilitates P recycling, it is not guaranteed, as P recycling from organic waste only occurs when that organic waste is reapplied to arable land; therefore, we also examined the fate of P after organic waste diversion. Through a literature analysis, we explored both peer-reviewed and grey literature to find data for P flows through centralized organic waste processing facilities and the distribution of end products. This analysis used a specified search string (see **Supplementary Material 1**), the snowball method using reference lists (Greenhalgh and Peacock 2005), Google™ searches (for grey literature), and personal communication. Based on relevance, we selected 11 (Field et al. 1984; Tiquia et al. 2002; Michel et al. 2004; Massé et al. 2007; Marcato et al. 2008; Andersen et al. 2010; Banks et al. 2011; Schievano et al. 2011; Möller and Müller 2012; Pognani et al. 2012; Zabaleta and Rodic 2015) existing peer-reviewed studies, including one meta-analysis (Zabaleta and Rodic 2015) of nutrients and organic waste management processing, for information on losses of P within the biological treatment process. Additional grey literature studies were found for both Europe (Barth et al. 2008) and Ontario (van der Werf 2013) concerning the distribution of end products.

Data analysis: organic solid waste diversion

We conducted our analyses at the country level for the European data set (29 countries) using data from the second Directive diversion target year (e.g., 2009 for countries with no derogation period).

The same analysis was completed at the jurisdictional level for the Ontario data set (98 jurisdictions) using waste diversion data from 2013.

We performed univariate linear regression analysis between each explanatory variable and the response variables (**Table S2**) to explore correlations between individual factors and organic waste diversion. Data were tested for assumptions of linear regression and, when these assumptions were not met, data were successfully transformed using the most suitable method, as detailed in **Table S2**. Paired *t* tests were performed on the European data to test for a significant difference in organic waste diversion pre- and post-Directive. For the European data, univariate regression and *t* tests were also completed for the change in OFMSW diversion between the second diversion year (generally 2009) and the base year (generally 1995). As historical data prior to 2006 were unavailable for Ontario, the number of years since the implementation of policy or infrastructure was included as an explanatory variable.

We performed a factor analysis to reduce the number of potential input variables for the regression model and assess which factors were significant indicators for OFMSW diversion rates. As this study contained continuous, categorical, and binary data, conventional variable clustering and reduction techniques such as principal component analysis (PCA) were not appropriate; instead, we used factor analysis on mixed data (FAMD; Husson et al. 2006). FAMD is a principal component method capable of analyzing data sets with both continuous and categorical variables (Pagès 2004). It is often described as a mix between principal component analysis (PCA) and multiple correspondence analysis (MCA), and FAMD results are frequently presented as a "correlation circle". The primary purpose for our use of the FAMD model was to assess variables that were closely related and likely to explain similar variance within our regression analyses.

We used multiple regression models to determine the combination of explanatory variables that best explained organic waste diversion rates, optimizing both model fit and complexity. Response variables were transformed using the logit function to satisfy model assumptions of linear regression. We used exploratory building to develop models based on the FAMD results, literature review, variable significance, and univariate regression models (**Table 2** and **Table S2**). Final model selection was based on results of the FAMD analysis, Akaike information criterion (AIC), and log-likelihood ratios. For validation purposes, backward stepwise regression models were also calculated, first using all of the variables listed in **Table 2** and **Table S2** and subsequently using several subsets of the variables (Lebersorger and Beigl 2011).

Results

Although various trends existed in solid waste diversion over time across countries and jurisdictions, in general, organic waste diversion increased with time (**Figs. 1** and **2**). We found a significant difference between the base year (generally 1995) and both the first ($t = -3.83$, $p = 6.42e{-}04$) and second ($t = -5.62$, $p = 5.097e{-}06$) diversion target years for European countries. Although there was a general increase in OFMSW diversion over the study period for both Europe and Ontario, diversion rates varied substantially between and within these two regions; in fact, some countries and jurisdictions experienced a decrease in organic waste diversion over the length of the study period (e.g., Malta; **Fig. 1**).

Along with the OFMSW diversion rate in Years 1 and 2 (i.e., 2006 and 2013 for Ontario, and base year—generally 1995—and Diversion Year 2 for Europe, see **Table S1**), **Figs. 1** and **2** also show per capita waste generation and per capita waste diversion (in kg) for Years 1 and 2. Together, this information gives a more holistic picture of how these systems changed over the study period. Including the per capita waste values provides valuable context; for example, the figures allow us to see not only which countries have increased their diversion of OFMSW over the study period but also those that have managed to reduce their per capita waste generation at the same time—or vice versa.

Table 2. Analysis of explanatory variables for Europe and Ontario; selected and analyzed socio-economic and waste management indicators and individual relationships with OFMSW diversion.

		Relationship with organic waste diversion rate	
Variable name	Variable label[a]	adjusted R^2	Significance
Europe (OFMSW diversion)			
Income (International \$/capita[b])	IN	0.506	<0.001
Population density (No. inhabitants/km^2)	PD	0.011	<1
Landfill charge (€/tonne)	LF	0.435	<0.001
Pay-as-you-throw	PAYT	0.238	<0.01
Legislation	LEG (0)	0.232	<0.05
	LEG (1)		
	LEG (2)		
Mandatory separation	MS	0.097	<0.1
Incineration tax	IT	0.182	<0.05
Incineration ban	IB	−0.028	<1
Europe (Δ OFMSW diversion)			
Income (International \$/capita[b])	IN	0.132	<0.05
Population density (No. inhabitants/km^2)	PD	−0.008	<1
Landfill charge (€/tonne)	LF	0.209	<0.01
Pay-as-you-throw	PAYT	0.09	<0.1
Legislation	LEG (0)	−0.038	—
	LEG (1)	−0.038	<1
	LEG (2)	−0.038	—
Mandatory separation	MS	−0.01	<1
Incineration tax	IT	−0.036	<1
Incineration ban	IB	0.034	<1
Ontario (OFMSW diversion)			
Income (CAD \$/capita)	IN	0.006	<1
Population density (No. inhabitants/km^2)	PD	0.236	<0.001
Population (inhabitants)	POP	0.257	<0.001
Landfill charge (CAD \$/tonne)	LF	0.065	<0.01
Pay-as-you-throw	PAYT	0.096	<0.01
Bag limit	BL	0.0145	<1
Leaf and yard waste	LY	0.211	<0.001
Green bin	GB	0.38	<0.001
Infrastructure	INF (0)	0.461	<0.001

(*continued*)

Table 2. (*concluded*)

Variable name	Variable label[a]	Relationship with organic waste diversion rate: adjusted R^2	Relationship with organic waste diversion rate: Significance
	INF (1)	0.461	<0.001
	INF (2)	0.461	<0.001
	INF (3)	0.461	<0.001
Implementation (No. years since implementation)	IMP	0.358	<0.001

Note: OFMSW, organic fraction of municipal solid waste.

[a]Numbers in parentheses represent the scale level for the categorical variables (e.g., a significant difference between infrastructure levels 1 and 2 would be represented as INF (2)); see Table S2 for details regarding categorical scale values.

[b]International dollars (International $) was based on conversion from local currency using UN Purchasing Power parties (PPP) conversion factors for 2011.

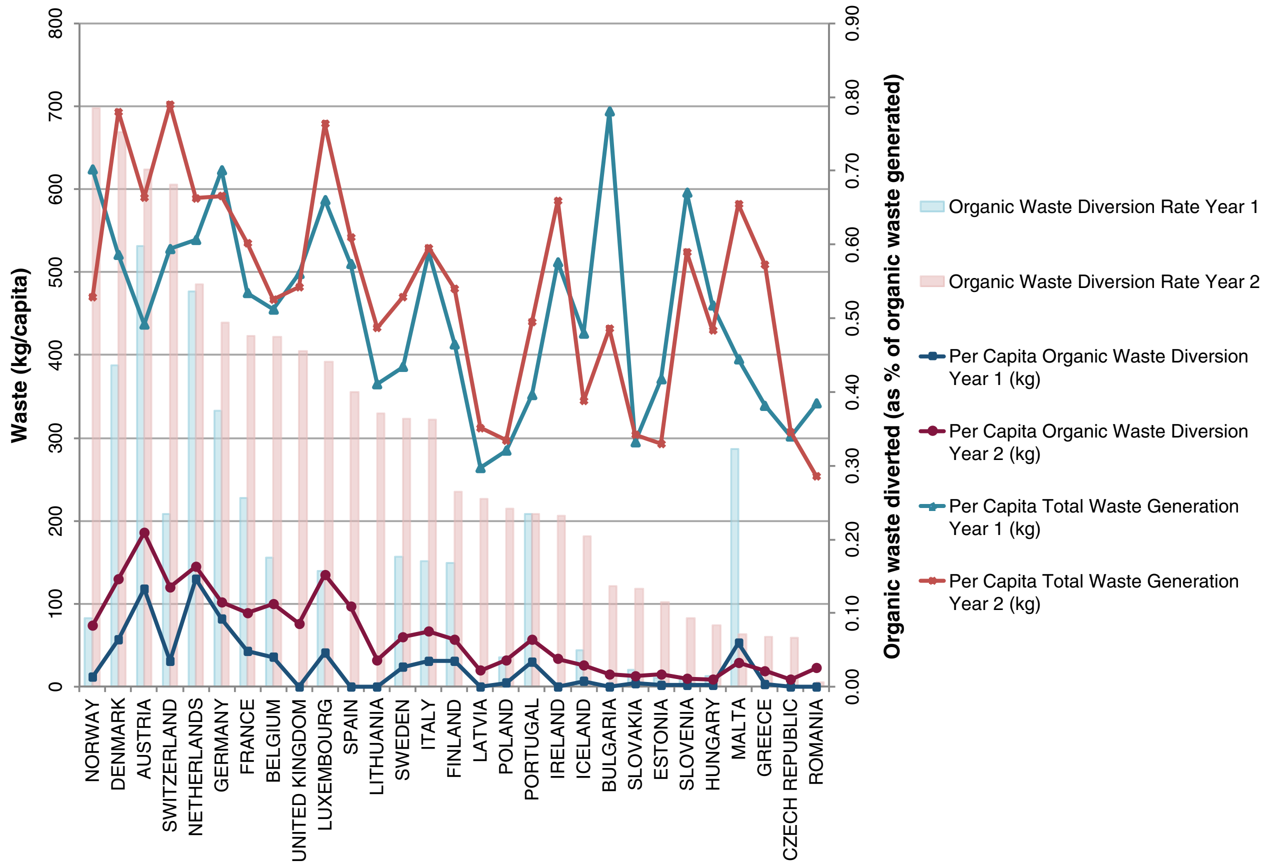


Fig. 1. European organic solid waste diversion by country. Bars show percent organic waste diversion in Year 1 (base year; BY) and Year 2 (second diversion year; DY2), whereas lines indicate per capita organic waste diversion and per capita total waste generation for both Year 1 and Year 2.

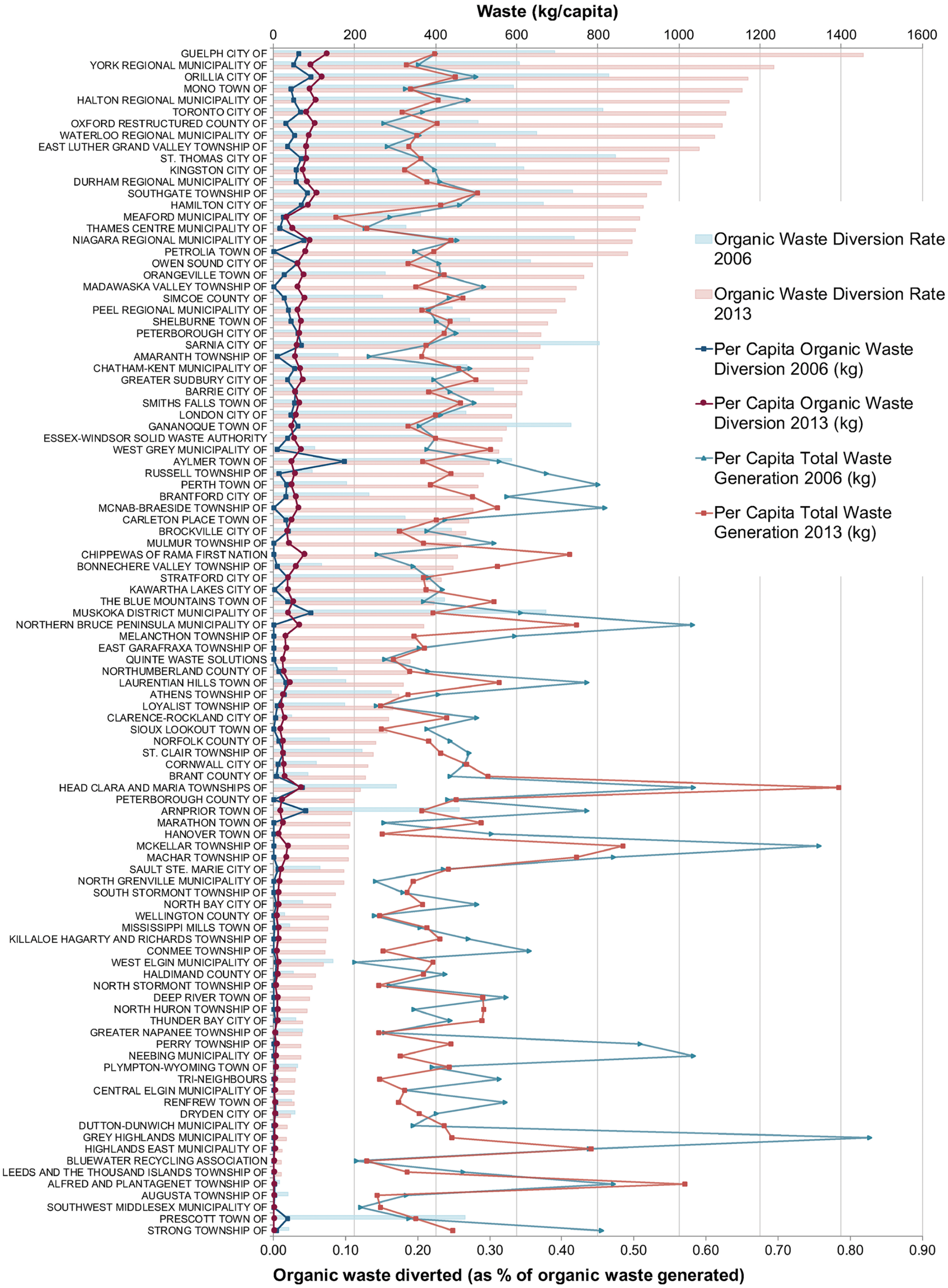


Fig. 2. Ontario organic solid waste diversion by jurisdiction. Bars show percent organic waste diversion in Year 1 (2006) and Year 2 (2013), whereas lines indicate per capita organic waste diversion and per capita total waste generation for both 2006 and 2013.

Regression analyses and key factors for organic waste diversion rates

Regression analyses showed evidence of significant relationships between OFMSW diversion and several explanatory variables for both Europe and Ontario (Table 2 and Table S2). Table 2 shows the univariate regression analysis of explanatory variables for Europe and Ontario. Table 2 includes the variable name and label (individual code used to distinguish categorical variables) for each independent variable. It also includes the coefficient of determination (adjusted R^2) and significance (p value indicator) for each regression. The coefficient of determination indicates how much variance of the dependent variable (diversion of OFMSW) is predicted by its relationship to each of the explanatory factors. Using univariate regression, significant correlations were found between explanatory variables (e.g., income, landfill charge, and infrastructure) and OFMSW diversion (Table S2). In Europe, less significant relationships were observed for the change in OFMSW diversion (between the second diversion year and the base year values) and explanatory variables.

Table 3 presents the results of multiple regression analysis for three models: Europe OFMSW diversion (in Year 2 or Diversion Year 2), Europe change in OFMSW diversion (from Year 1 to Year 2; i.e., before and after the implementation of the Directive), and Ontario OFMSW diversion (in 2013). The table includes the dependent variable (OFMSW diversion) in each of those models as well as the adjusted R^2 and significance for each multiple regression analysis model. The adjusted R^2 represents the percentage of the response variable variation that is explained by the model.

For each final model, Table 3 also lists the explanatory variables that were included in that model based on FAMD, AIC, and log-likelihood ratios. We have included standard coefficient outputs for each explanatory variable. The coefficient estimate represents the mean change in the dependent variable for every one unit of change in the explanatory variable while holding the other variables in the model constant—the larger coefficient estimates indicate that a larger change is observed in the rate of organic waste diversion for every unit increase in the explanatory variable. The t value is the value of the t statistic, which measures how far away the coefficient estimate is (in standard deviations) from 0. In Table 3, we can see that the t values differ from 0, which indicates that there is a relationship between the response variable (OFMSW diversion) and each of the explanatory variables. Finally, the low p values for explanatory variables indicate that they are meaningful additions to the model.

Multiple regression analysis for Europe indicated that income and incineration tax were key factors in the rate of OFMSW diversion (adjusted $R^2 = 0.58$; Table 3), whereas key factors related to change in organic waste diversion were income, population density, and legislation (adjusted $R^2 = 0.23$). In Europe, FAMD showed a strong positive correlation between income, landfill charge, and pay-as-you-throw (PAYT; Fig. 3), indicating that because income is considered in the final model, landfill charge and PAYT are likely to be key factors as well. Figure 3 shows the "correlation circle" from FAMD. The names in the circle represent explanatory variables, whereas the arrows and their positioning indicate (*a*) their relationship to other variables (e.g., if two variables are closely related, they are likely to explain similar variance within a regression model; if they are set at 90 degrees, they likely have little to no relationship, and if two variables are set at 180 degrees, they are likely to have a negative relationship); and (*b*) how much of the variance is accounted for through the first and second dimensions (as represented by the percentages on the y- and x-axes, respectively).

Multiple regression analysis for OFMSW diversion in Ontario indicated that infrastructure, PAYT, and population density were driving factors (adjusted $R^2 = 0.52$; Table 3). PAYT and bag limits, as well as infrastructure and landfill fees, were closely correlated in Ontario (Fig. 3); therefore, landfill

Table 3. Final regression models for Europe ($n = 29$) and Ontario ($n = 98$) for organic waste diversion.

Dependent variable	Adjusted R^2	Model significance	Explanatory variable[a]	Coefficient values: Coefficient estimate	t	p
Europe						
OFMSW Diversion (DY2)	0.5841	<0.001	Constant	−3.38	−7.85	<0.001
			Income	1.08E−4	5.25	<0.001
			Incineration tax	0.78	2.11	<0.05
Change in OFMSW Diversion	0.2282	<0.001	Constant	−3.19	−5.84	<0.001
			Income	7.64E−5	−5.84	<0.01
			Incineration ban	1.40	1.95	<0.1
			Legislation (2)	−1.09	−1.73	<0.1
Ontario						
OFMSW Diversion (2013)	0.5214	<0.001	Constant	−3.51	−12.1	<0.001
			Population density	5.46E−4	2.81	<0.01
			Pay-as-you-throw	0.50	2.06	<0.05
			Infrastructure (1)	1.11	3.68	<0.01
			Infrastructure (2)	1.00	2.60	<0.05
			Infrastructure (3)	2.72	7.65	<0.001

Note: DY2, second diversion year; OFMSW, organic fraction of municipal solid waste.
[a]Numbers in parentheses represent the scale level for the categorical variables (e.g., a significant difference between infrastructure levels 1 and 2 would be represented as Infrastructure (2)); see Table S2 and for details regarding categorical scale values.

fees and bag limits should also be considered key factors. FAMD results from Europe, and especially Ontario, also showed clustering among the factor levels for the categorical variables of legislation and infrastructure, respectively (Fig. S1).

The fate of P through centralized organic waste processing

The literature analysis on P flows through centralized waste systems suggested significant P losses during both processing and the distribution of the final product. An average of 21% of P entering biological treatment facilities is lost as a result of rejected waste, leaching, and refining of finished product (Table S3; Fig. 4). This loss can vary anywhere from 1% to 100% depending on feedstock, treatment process, and extenuating circumstances (e.g., plant malfunction). A literature review completed in 2015 (Zabaleta and Rodic 2015) found that AD typically sees P losses of <10%, whereas losses from composting may range from 1% to 38%. Significant amounts of P can be lost through leachate and runoff in open systems such as windrow composting (Tiquia et al. 2002), whereas the predominant losses of P in more closed systems such as AD occur during sorting and rejection in the pre-treatment and refining stages (Pognani et al. 2012). Of the total compost produced, an average of 65% (59%–70%) is applied to land used for food production; the remainder (30%–41%) is predominantly used in landscaping or remediation (Table S3). This also varied drastically by location, where the amount of finished product sent to arable land ranged from 0% to 100% (Table S3). Therefore, from the total P available in diverted organics, approximately 50% (40%–63%) is reapplied to arable land, whereas the remaining 50% is lost during processing (11%–33%) or used on non-arable land (20%–36%).

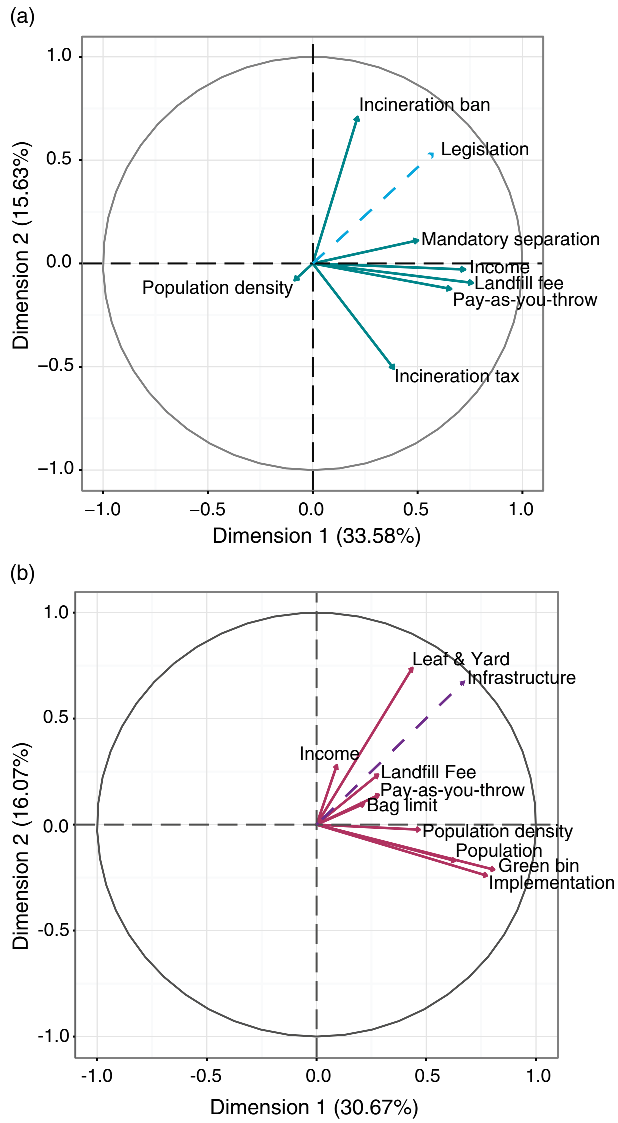


Fig. 3. (a) Factor analysis of mixed data (FAMD) results for Europe, illustrating the relationships between explanatory variables for continuous and binary variables (solid teal arrows) as well as categorical data (dashed blue arrow; overlaid using interclass correlation). (b) FAMD results for Ontario jurisdictions, showing explanatory variables for continuous and binary variables (solid maroon arrows) and categorical data (dashed purple arrow; overlaid using interclass correlation). Variables that are closely related are likely to explain similar variance within a regression model. Percentages on the *y*- and *x*-axes represent how much of the model variance is accounted for by the first and second dimensions, respectively, of the FAMD model (33.58% and 15.63% for Europe; 30.67% and 16.07% for Ontario).

Data availability and limitations

This study was limited by data availability. For example, results from Ontario and existing literature (Sterner and Bartelings 1999; Bernstad 2014), indicate that infrastructure may play a significant role in OFMSW diversion rates; however, we were unable to analyze this in Europe because such data were

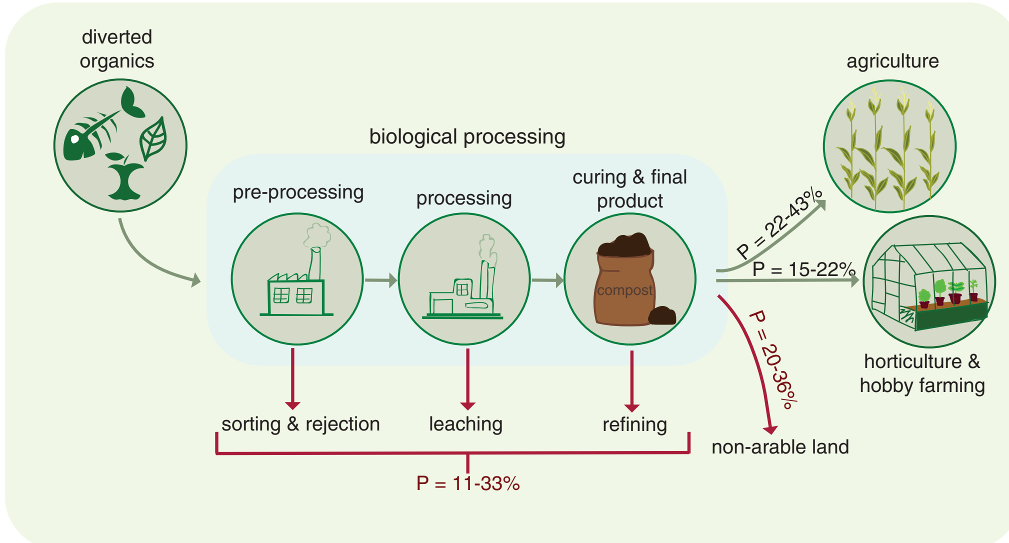


Fig. 4. Typical phosphorus (P) flow through organic solid waste processing streams. Losses of P in centralized biological processing are estimated at 11%–33% (see Tables S2 and S3). An additional 30%–41% of the final compost product (containing 20%–36% of the original P) is sent to non-arable land. Therefore, 40%–63% of the P in diverted organics is recycled to arable land.

unavailable at the national level. Further research of the role of infrastructure in Europe may provide additional insight into diversion rates. Scale might also affect which factors appear to most strongly influence organic waste diversion and recycling; however, multi-scale data were generally not available. Additionally, the European data set used for this study was not well-suited to making strong inferences outside Europe or the national scale on which it was studied. When looking at national and municipal level data, behaviours that exist at a household level and that might strongly affect diversion and reapplication rates can be missed. There are existing studies related to household behaviour for recycling and some that include organics (Sterner and Bartelings 1999; Martin et al. 2006; Bulkeley and Askins 2009), but future research integrating national and municipal data with household behavioural findings could prove extremely valuable. Despite these limitations, we feel that this study still has significant value in acting as a guide for future studies of factors influencing organic waste diversion, for acknowledging the factors that played significant roles in the areas that were studied, and for bringing to attention to issues associated with data availability. Furthermore, as the Ontario data set was much larger, it is possible to make stronger generalizations from the results of that analysis, which we consider valuable, especially in the North American context.

Discussion

We showed that although existing socio-economic factors are likely to play a role in OFMSW diversion and reapplication, management options such as economic incentives and infrastructure can be leveraged to increase organic waste diversion. We also found that many regions have low P reapplication associated with organic waste management, with significant room for improvement. Information from this research can be used in conjunction with pre-existing qualitative urban- (Bulkeley and Askins 2009; Metson and Bennett 2015b) and household-level studies (Sterner and Bartelings 1999; Martin et al. 2006; Refsgaard and Magnussen 2009; Bernstad 2014; Parizeau et al. 2015) to assess organic waste diversion potential in specific regions or case studies.

Although OFMSW diversion generally increases over time, our study showed that there is broad variation between regions, and that most countries and jurisdictions recycle only a fraction of their potential P from solid waste. For example, Germany—a region with a relatively successful organic solid waste diversion program and high reapplication rates—diverted 50% of their organic solid waste in 2013 (Eurostat 2014a), and has an expected reapplication rate of 69% (Barth et al. 2008). This means that approximately 34% of the P in organic waste in Germany is recycled back to arable land. There are considerable opportunities for diverting and recycling significantly more P from organic solid waste.

Our work also leads to a better understanding of why OFMSW diversion programs in some regions may be more successful than others. Countries such as Austria or cities such as Guelph have high diversion rates (**Figs. 1** and **2**) and can act as models for other locations. Although Austria has a high income, they also use PAYT systems to discourage waste disposal, have high taxes on landfill and incineration disposal, and have strong legislation on the disposal of organic waste to landfill (**Table S5**). In line with key factors in Ontario, Guelph has sophisticated infrastructure for the collection of source-separated organics and imposes a PAYT economic incentive program (**Table S10**). It is also important to investigate countries such as Lithuania which, despite having the second lowest income of European countries in this study (Eurostat 2014b; UNData 2014, **Table S5**), had a relatively high diversion rate of 37% in 2013 (**Fig. 1**, **Table S5**). Our research indicates that the use of landfill taxes and PAYT economic incentives are likely to have contributed to this admirable diversion rate. Alternatively, some regions such as Iceland have moderate to high incomes, but relatively low diversion rates (20%) (**Table S5**), corroborating existing literature that suggests that economic prosperity alone does not guarantee greater waste diversion, and that diversion must also be a policy and management priority (Mazzanti et al. 2009). Research regarding the influence of the EU Landfill Directive was inconclusive. The *t* tests indicated a significant difference between means before and after the implementation of the Directive; however, analysis of the change in OFSMW diversion was inconclusive and lacked significant relationships. It is possible that countries with high diversion prior to the Directive continued to do well or, as evidenced by some of the individual country trends, that there is a saturation level or plateau in diversion rates.

Conclusions

Urban P recycling can play an important role in ensuring a lasting supply of this crucial nutrient and non-renewable resource. However, unless urban P flows are returned to arable land, many of their potential benefits are lost; thus, a clear understanding of the use and reapplication of organic waste end products is critical. Although most existing literature on P recovery focuses on flows associated with wastewater treatment (Morse et al. 1998; Jaffer et al. 2002; Antikainen et al. 2005; Elser and Bennett 2011; Cordell et al. 2012; Linderholm et al. 2012), solid waste management provides an important opportunity to recycle more P than wastewater alone, given that similar levels of P output have been found for solid waste as for wastewater (Kalmykova et al. 2012; Ott and Rechberger 2012). Here, we have demonstrated that economic incentives, local legislation, and accessible infrastructure can be effective ways to increase OFMSW diversion and P recycling to move beyond waste and toward valuable resource recovery.

Acknowledgements

We acknowledge McGill University, the Natural Sciences and Engineering Research Council (NSERC) of Canada, GHD Incorporated, Ice River Springs, and the Environmental Research and Education Foundation for their support.

Author contributions

JLT, EMB, and OGC conceived and designed the study. JLT performed the experiments/collected the data. JLT, EMB, and OGC analyzed and interpreted the data. JLT, EMB, and OGC contributed resources. JLT, EMB, and OGC drafted or revised the manuscript.

Competing interests

The authors have declared that no competing interests exist.

Data availability statement

All relevant data are within the paper and in the Supplementary Material.

Supplementary Material

The following Supplementary Material is available with the article through the journal website at doi:10.1139/facets-2018-0005.

Supplementary Material 1

References

Agegnehu G, Bird MI, Nelson PN, and Bass AM. 2015. The ameliorating effects of biochar and compost on soil quality and plant growth on a Ferralsol. Soil Research, 53: 1–12. DOI: 10.1071/SR14118

Andersen JK, Boldrin A, Christensen TH, and Scheutz C. 2010. Mass balances and life-cycle inventory for a garden waste windrow composting plant (Aarhus, Denmark). Waste Management & Research, 28: 1010–1020. PMID: 20980476 DOI: 10.1177/0734242X09360216

Antikainen R, Lemola R, Nousiainen JI, Sokka L, Esala M, Huhtanen P, et al. 2005. Stocks and flows of nitrogen and phosphorus in the Finnish food production and consumption system. Agriculture, Ecosystems & Environment, 107: 287–305. DOI: 10.1016/j.agee.2004.10.025

Augenstein D. 1992. The greenhouse effect and US landfill methane. Global Environmental Change, 2(4): 311–328. DOI: 10.1016/0959-3780(92)90048-c

Banks CJ, Chesshire M, Heaven S, and Arnold R. 2011. Anaerobic digestion of source-segregated domestic food waste: performance assessment by mass and energy balance. Bioresource Technology, 102: 612–620. PMID: 20797849 DOI: 10.1016/j.biortech.2010.08.005

Barr S, Guilbert S, Metcalfe A, Riley M, Robinson GM, and Tudor TL. 2013. Beyond recycling: an integrated approach for understanding municipal waste management. Applied Geography, 39: 67–77. DOI: 10.1016/j.apgeog.2012.11.006

Barth J, Amlinger F, Favoino E, Siebert S, Kehres B, Gottschall R, et al. 2008. Compost production and use in the EU. Report for the European Commission DG/JRC. 182 pp. Available from organics-recycling.org.uk/dmdocuments/compostproduction_and_usein_EU.pdf.

Bennett EM, Carpenter SR, and Caraco NF. 2001. Human impact on erodable phosphorus and eutrophication: a global perspective: increasing accumulation of phosphorus in soil threatens rivers, lakes, and coastal oceans with eutrophication. Bioscience, 51: 227–234. DOI: 10.1641/0006-3568(2001)051[0227:HIOEPA]2.0.CO;2

Bernstad A. 2014. Household food waste separation behavior and the importance of convenience. Waste Management, 34: 1317–1323. PMID: 24780762 DOI: 10.1016/j.wasman.2014.03.013

Bulkeley H, and Askins K. 2009. Waste interfaces: biodegradable waste, municipal policy and everyday practice. The Geographical Journal, 175: 251–260. DOI: 10.1111/j.1475-4959.2008.00310.x

Burnley S. 2001. The impact of the European landfill directive on waste management in the United Kingdom. Resources, Conservation and Recycling, 32: 349–358. DOI: 10.1016/S0921-3449(01)00074-X

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, and Smith VH. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications, 8: 559–568. DOI: 10.1890/1051-0761(1998)008[0559:NPOSWW]2.0.CO;2

Childers DL, Corman J, Edwards M, and Elser JJ. 2011. Sustainability challenges of phosphorus and food: solutions from closing the human phosphorus cycle. Bioscience, 61: 117–124. DOI: 10.1525/bio.2011.61.2.6

City of Guelph. 2014. Solid waste management master plan [online]: Available from guelph.ca/plans-and-strategies/solid-waste-management-master-plan/.

City of Toronto. 2016. Long term waste strategy. City of Toronto [online]: Available from toronto.ca/services-payments/recycling-organics-garbage/long-term-waste-strategy/.

Confederation of European Waste-to-Energy Plants (CEWEP). 2012. Landfill taxes and bans [online]: Available from cewep.eu/landfill-taxes-and-bans/.

Conseil fédéral Suisse. 1990. Ordonnance sur les traitements des déchêts. Bern, Switzerland. Available from admin.ch/opc/fr/classified-compilation/19900325/index.html.

Cooper J, and Carliell-Marquet C. 2013. A substance flow analysis of phosphorus in the UK food production and consumption system. Resources, Conservation and Recycling, 74: 82–100. DOI: 10.1016/j.resconrec.2013.03.001

Cooper J, Lombardi R, Boardman D, and Carliell-Marquet C. 2011. The future distribution and production of global phosphate rock reserves. Resources, Conservation and Recycling, 57: 78–86. DOI: 10.1016/j.resconrec.2011.09.009

Cordell D, and White S. 2011. Peak phosphorus: clarifying the key issues of a vigorous debate about long-term phosphorus security. Sustainability, 3: 2027–2049. DOI: 10.3390/su3102027

Cordell D, and White S. 2013. Sustainable phosphorus measures: strategies and technologies for achieving phosphorus security. Agronomy, 3: 86–116. DOI: 10.3390/agronomy3010086

Cordell D, Drangert J-O, and White S. 2009. The story of phosphorus: global food security and food for thought. Global Environmental Change, 19: 292–305. DOI: 10.1016/j.gloenvcha.2008.10.009

Cordell D, Rosemarin A, Schröder JJ, and Smit AL. 2011. Towards global phosphorus security: a systems framework for phosphorus recovery and reuse options. Chemosphere, 84: 747–758. PMID: 21414650 DOI: 10.1016/j.chemosphere.2011.02.032

Cordell D, Neset T-SS, and Prior T. 2012. The phosphorus mass balance: identifying ‘hotspots’ in the food system as a roadmap to phosphorus security. Current Opinion in Biotechnology, 23: 839–845. PMID: 22503084 DOI: 10.1016/j.copbio.2012.03.010

Dorward LJ. 2012. Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)? A comment. Food Policy, 37: 463–466. DOI: 10.1016/j.foodpol.2012.04.006

Elser J, and Bennett E. 2011. Phosphorus cycle: a broken biogeochemical cycle. Nature, 478: 29–31. PMID: 21979027 DOI: 10.1038/478029a

Eriksson O, Reich MC, Frostell B, Björklund A, Assefa G, Sundqvist J-O, Granath J, Baky A, and Thyselius L. 2005. Municipal solid waste management from a systems perspective. Journal of Cleaner Production, 13: 241–252. DOI: 10.1016/j.jclepro.2004.02.018

European Commission. 1999. EU Landfill Directive (99/31/EC) [online]: Available from ec.europa.eu/environment/waste/landfill_index.htm.

European Environment Agency. 2013. Managing municipal solid waste—a review of achievements in 32 European counties. Publications Office of the European Union, Luxembourg. 36 pp. Available from eea.europa.eu/publications/managing-municipal-solid-waste.

Eurostat. 2014a. Municipal waste statistics [online]: Available from ec.europa.eu/eurostat/statistics-explained/index.php/Municipal_waste_statistics.

Eurostat. 2014b. Mean and median income by household type [online]: Available from appsso.eurostat.ec.europa.eu/nui/show.do?dataset=ilc_di04&lang=en.

Fehr M, and Arantes CA. 2015. Making a case for recycling biodegradable municipal waste. Environment Systems and Decisions, 35: 483–489. DOI: 10.1007/s10669-015-9568-z

Ferrara I, and Missios P. 2005. Recycling and waste diversion effectiveness: evidence from Canada. Environmental and Resource Economics, 30: 221–238. DOI: 10.1007/s10640-004-1518-z

Field JA, Caldwell JS, Jeyanayagam S, Reneau RB, Kroontje W, and Collins ER. 1984. Fertilizer recovery from anaerobic digesters. Transactions of the American Society of Agricultural Engineers, 27(6): 1871–1876.

Garnier J, Lassaletta L, Billen G, Romero E, Grizzetti B, Némery J, et al. 2015. Phosphorus budget in the water-agro-food system at nested scales in two contrasted regions of the world (ASEAN-8 and EU-27). Global Biogeochemical Cycles, 29: 1348–1368. DOI: 10.1002/2015GB005147

Gellynck X, Jacobsen R, and Verhelst P. 2011. Identifying the key factors in increasing recycling and reducing residual household waste: a case study of the Flemish region of Belgium. Journal of Environmental Management, 92: 2683–2690. PMID: 21704444 DOI: 10.1016/j.jenvman.2011.06.006

Giusquiani PL, Pagliai M, Gigliotti G, Businelli D, and Benetti A. 1995. Urban waste compost: effects on physical, chemical, and biochemical soil properties. Journal of Environmental Quality Abstract, 24: 175–182. DOI: 10.2134/jeq1995.00472425002400010024x

Greenhalgh T, and Peacock R. 2005. Effectiveness and efficiency of search methods in systematic reviews of complex evidence: audit of primary sources. BMJ, 331: 1064–1065. PMID: 16230312 DOI: 10.1136/bmj.38636.593461.68

Grimm NB, Faeth SH, Golubiewski NE, Redman CL, Wu J, Bai X, et al. 2008. Global change and the ecology of cities. Science, 319: 756–760. PMID: 18258902 DOI: 10.1126/science.1150195

Hogg D, Favoino E, Nielsen N, Thompson J, Wood K, Penschke A, et al. 2002. Economic analysis of options for managing biodegradable municipal waste: final report to the European Commission. Eunomia Research And Consulting, ZREU, LDK, HDRA Consultants and Scuola Agraria del Parco di Monza (for ECOTEC Research & Consulting), Bristol, England.

Hornik J, Cherian J, Madansky M, and Narayana C. 1995. Determinants of recycling behavior: a synthesis of research results. The Journal of Socio-Economics, 24: 105–127. DOI: 10.1016/1053-5357(95)90032-2

Husson F, Lê S, and Mazet J. 2006. Factor analysis and data mining with R [online]: Available from ftp.auckland.ac.nz/software/CRAN/doc/packages/FactoMineR.pdf.

Jaffer Y, Clark TA, Pearce P, and Parsons SA. 2002. Potential phosphorus recovery by struvite formation. Water Research, 36: 1834–1842. PMID: 12044083 DOI: 10.1016/S0043-1354(01)00391-8

Kalmykova Y, Harder R, Borgestedt H, and Svanang I. 2012. Pathways and management of phosphorus in urban areas. Journal of Industrial Ecology, 16: 928–939. DOI: 10.1111/j.1530-9290.2012.00541.x

Kennedy CA, Cuddihy J, and Engel-yan J. 2007. The changing metabolism of cities. Journal of Industrial Ecology, 11: 43–59. DOI: 10.1162/jie.2007.1107

Khalid A, Arshad M, Anjum M, Mahmood T, and Dawson L. 2011. The anaerobic digestion of solid organic waste. Waste Management, 31: 1737–1744. PMID: 21530224 DOI: 10.1016/j.wasman.2011.03.021

Lebersorger S, and Beigl P. 2011. Municipal solid waste generation in municipalities: quantifying impacts of household structure, commercial waste and domestic fuel. Waste Management, 31: 1907–1915. PMID: 21689921 DOI: 10.1016/j.wasman.2011.05.016

Levis JW, Barlaz MA, Themelis NJ, and Ulloa P. 2010. Assessment of the state of food waste treatment in the United States and Canada. Waste Management, 30: 1486–1494. PMID: 20171867 DOI: 10.1016/j.wasman.2010.01.031

Lin T, Gibson V, Cui S, Yu CP, Chen S, Ye Z, et al. 2014. Managing urban nutrient biogeochemistry for sustainable urbanization. Environmental Pollution, 192: 244–250. PMID: 24746891 DOI: 10.1016/j.envpol.2014.03.038

Linderholm K, Mattsson JE, and Tillman AM. 2012. Phosphorus flows to and from Swedish agriculture and food chain. Ambio, 41: 883–893. PMID: 22627872 DOI: 10.1007/s13280-012-0294-1

Marcato CE, Pinelli E, Pouech P, Winterton P, and Guiresse M. 2008. Particle size and metal distributions in anaerobically digested pig slurry. Bioresource Technology, 99: 2340–2348. PMID: 17600701 DOI: 10.1016/j.biortech.2007.05.013

Martin M, Williams ID, and Clark M. 2006. Social, cultural and structural influences on household waste recycling: a case study. Resources, Conservation and Recycling, 48: 357–395. DOI: 10.1016/j.resconrec.2005.09.005

Massé DI, Croteau F, and Masse L. 2007. The fate of crop nutrients during digestion of swine manure in psychrophilic anaerobic sequencing batch reactors. Bioresource Technology, 98: 2819–2823. PMID: 17400445 DOI: 10.1016/j.biortech.2006.07.040

Mayer BK, Baker LA, Boyer TH, Drechsel P, Gifford M, Hanjra MA, et al. 2016. Total value of phosphorus recovery. Environmental Science & Technology, 50: 6606–6620. PMID: 27214029 DOI: 10.1021/acs.est.6b01239

Mazzanti M, Montini A, and Nicolli F. 2009. The dynamics of landfill diversion: economic drivers, policy factors and spatial issues: evidence from Italy using provincial panel data. Resources, Conservation and Recycling, 54: 53–61. DOI: 10.1016/j.resconrec.2009.06.007

Metcalfe A, Riley M, Barr S, Tudor T, Robinson G, and Guilbert S. 2013. Food waste bins: bridging infrastructures and practices. The Sociological Review, 60: 135–155. DOI: 10.1111/1467-954X.12042

Metro Vancouver. 2010. Integrated solid waste and resource management: a solid waste management plan for the Greater Vancouver Regional District and member municipalities [online]: Available from metrovancouver.org/services/solid-waste/SolidWastePublications/ISWRMP.pdf.

Metro Vancouver. 2015. Food scraps recycling [online]: Available from metrovancouver.org/foodscraps.

Metson GS. 2014. Urban phosphorus sustainability: how human diet, urban agriculture and socioecological context influence phosphorus cycling and management. McGill University, Montreal, Québec.

Metson GS, and Bennett EM. 2015a. Phosphorus cycling in Montreal's food and urban agriculture systems. PLoS ONE, 10: e0120726–18. DOI: 10.1371/journal.pone.0120726

Metson GS, and Bennett EM. 2015b. Facilitators & barriers to organic waste and phosphorus re-use in Montreal. Elementa: Science of the Anthropocene, 3: 70. DOI: 10.12952/journal.elementa.000070

Metson GS, Iwaniec DM, Baker LA, Bennett EM, Childers DL, Cordell D, et al. 2015. Urban phosphorus sustainability: systemically incorporating social, ecological, and technological factors into phosphorus flow analysis. Environmental Science & Policy, 47: 1–11. DOI: 10.1016/j.envsci.2014.10.005

Miafodzyeva S, and Brandt N. 2013. Recycling behaviour among householders: synthesizing determinants via a meta-analysis. Waste and Biomass Valorization, 4: 221–235. DOI: 10.1007/s12649-012-9144-4

Michel FC Jr, Pecchia JA, Rigot J, and Keener HM. 2004. Mass and nutrient losses during the composting of dairy manure amended with sawdust or straw. Compost Science & Utilization, 12: 323–334. DOI: 10.1080/1065657X.2004.10702201

Miliute-Plepiene J, and Plepys A. 2014. Does food sorting prevents and improves sorting of household waste? A case in Sweden. Journal of Cleaner Production, 101: 182–192. DOI: 10.1016/j.jclepro.2015.04.013

Möller K, and Müller T. 2012. Effects of anaerobic digestion on digestate nutrient availability and crop growth: a review. Engineering in Life Sciences, 12: 242–257. DOI: 10.1002/elsc.201100085

Morse GK, Brett SW, Guy JA, and Lester JN. 1998. Review: phosphorus removal and recovery technologies. Science of the Total Environment, 212: 69–81. DOI: 10.1016/S0048-9697(97)00332-X

Mueller W. 2013. The effectiveness of recycling policy options: waste diversion or just diversions? Waste Management, 33: 508–518. PMID: 23312779 DOI: 10.1016/j.wasman.2012.12.007

Neset T-SS. 2005. Environmental imprint of human food consumption Linköping, Sweden 1870–2000. Linköping University, Linköping, Sweden.

Neset T-SS, and Cordell D. 2012. Global phosphorus scarcity: identifying synergies for a sustainable future. Journal of the Science of Food and Agriculture, 92: 2–6. PMID: 21969145 DOI: 10.1002/jsfa.4650

Neset T-SS, Bader HP, Scheidegger R, and Lohm U. 2008. The flow of phosphorus in food production and consumption—Linköping, Sweden, 1870-2000. Science of the Total Environment, 396: 111–120. DOI: 10.1016/j.scitotenv.2008.02.010

Nicolli F, Mazzanti M, and Iafolla V. 2012. Waste dynamics, country heterogeneity and European environmental policy effectiveness. Journal of Environmental Policy & Planning, 14: 371–393. DOI: 10.1080/1523908X.2012.719694

Nilsson J. 1995. A phosphorus budget for a Swedish municipality. Journal of Environmental Management, 45: 243–253. DOI: 10.1006/jema.1995.0072

Ott C, and Rechberger H. 2012. The European phosphorus balance. Resources, Conservation and Recycling, 60: 159–172. DOI: 10.1016/j.resconrec.2011.12.007

Pagès J. 2004. Analyse factorielle de données mixtes. Revue de Statistique Appliquée, 54: 93–111.

Parizeau K, von Massow M, and Martin R. 2015. Household-level dynamics of food waste production and related beliefs, attitudes, and behaviours in Guelph, Ontario. Waste Management, 35: 207–217. PMID: 25445261 DOI: 10.1016/j.wasman.2014.09.019

Pognani M, Barrena R, Font X, and Sánchez A. 2012. A complete mass balance of a complex combined anaerobic/aerobic municipal source-separated waste treatment plant. Waste Management, 32: 799–805. PMID: 22261421 DOI: 10.1016/j.wasman.2011.12.018

Ponsá S, Gea T, Alerm L, Cerezo J, and Sánchez A. 2008. Comparison of aerobic and anaerobic stability indices through a MSW biological treatment process. Waste Management, 28: 2735–2742. DOI: 10.1016/j.wasman.2007.12.002

Québec. 2011. Québec residual materials management policy [online]: Available from legisquebec.gouv.qc.ca/en/ShowDoc/cr/Q-2,%20r.%2035.1.

Recyc-Québec. 2015. Caractérisation des matières résiduelles du secteur résidentiel. Éco Entreprises Québec and RÉCYC-QUÉBEC, Montréal, Québec [online]: Available from recyc-quebec.gouv.qc.ca/sites/default/files/documents/carac-2012-2013-rapport-synthese.pdf.

Refsgaard K, and Magnussen K. 2009. Household behaviour and attitudes with respect to recycling food waste—experiences from focus groups. Journal of Environmental Management, 90: 760–771. PMID: 18394776 DOI: 10.1016/j.jenvman.2008.01.018

Region of Peel. 2009. Integrated waste management discussion paper [online]: Available from peelregion.ca/planning/officialplan/pdfs/Discussion_Paper_-_Peel_Integrated_Waste_Management_Feb.2009.pdf.

Region of Waterloo, and Golder Associates. 2013. Region of Waterloo waste management master plan: final master plan report [online] Available from: regionofwaterloo.ca/en/living-here/resources/Documents/Waste/DOCS_ADMIN-1505628-v1-WMMP_Final_WMMP_Report_7Nov13_pdf.pdf.

Sahlin J, Ekvall T, Bisaillon M, and Sundberg J. 2007. Introduction of a waste incineration tax: effects on the Swedish waste flows. Resources, Conservation and Recycling, 51: 827–846. DOI: 10.1016/j.resconrec.2007.01.002

Schievano A, D'Imporzano G, Salati S, and Adani F. 2011. On-field study of anaerobic digestion full-scale plants (Part I): an on-field methodology to determine mass, carbon and nutrients balance. Bioresource Technology, 102: 7737–7744. PMID: 21715157 DOI: 10.1016/j.biortech.2011.06.006

Schlesinger W, and Bernhardt E. 2013. Biogeochemistry: an analysis of global change, 3rd edition. Academic Press, Cambridge, Massachusetts.

Scholz RW, and Wellmer F-W. 2013. Approaching a dynamic view on the availability of mineral resources: what we may learn from the case of phosphorus? Global Environmental Change, 23: 11–27. DOI: 10.1016/j.gloenvcha.2012.10.013

Senthilkumar K, Mollier A, Delmas M, Pellerin S, and Nesme T. 2014. Phosphorus recovery and recycling from waste: an appraisal based on a French case study. Resources, Conservation and Recycling, 87: 97–108. DOI: 10.1016/j.resconrec.2014.03.005

Sidique SF, Joshi SV, and Lupi F. 2010. Factors influencing the rate of recycling: an analysis of Minnesota counties. Resources, Conservation and Recycling, 54: 242–249. DOI: 10.1016/j.resconrec.2009.08.006

Smil V. 2000. Phosphorus in the environment: natural flows and human interferences. Annual Review of Energy and the Environment, 25: 53–88. DOI: 10.1146/annurev.energy.25.1.53

Smith VH, and Schindler DW. 2009. Eutrophication science: where do we go from here? Trends in Ecology & Evolution, 24: 201–207. PMID: 19246117 DOI: 10.1016/j.tree.2008.11.009

Statistics Canada. 2005. Human activity and the environment: solid waste in Canada. Statistics Canada: Environment Accounts and Statistics Division, Ottawa, Ontario.

Statistics Canada. 2011a. Census profile [online]: Available from www12.statcan.gc.ca/census-recensement/2011/dp-pd/prof/index.cfm?Lang=E.

Statistics Canada. 2011b. National Household Survey profile [online]: Available from www12.statcan.gc.ca/nhs-enm/2011/dp-pd/prof/index.cfm?Lang=E.

Sterner T, and Bartelings H. 1999. Household waste management in a Swedish municipality: determinants of waste disposal, recycling and composting. Environmental and Resource Economics, 13: 473–491. DOI: 10.1023/A:1008214417099

Taşeli BK. 2007. The impact of the European Landfill Directive on waste management strategy and current legislation in Turkey's Specially Protected Areas. Resources, Conservation and Recycling, 52: 119–135. DOI: 10.1016/j.resconrec.2007.03.003

Tilman D, Cassman KG, Matson PA, Naylor R, and Polasky S. 2002. Agricultural sustainability and intensive production practices. Nature, 418: 671–677. PMID: 12167873 DOI: 10.1038/nature01014

Tiquia SM, Richard TL, and Honeyman MS. 2002. Carbon, nutrient, and mass loss during composting. Nutrient Cycling in Agroecosystems, 62: 15–24. DOI: 10.1023/A:1015137922816

Tsai W-T. 2008. Management considerations and environmental benefit analysis for turning food garbage into agricultural resources. Bioresource Technology, 99: 5309–5316. PMID: 18178429 DOI: 10.1016/j.biortech.2007.11.025

UNData. 2014. PPP conversion [online]: Available from data.un.org.

van der Werf P. 2013. Ontario organic waste management report 2013–2033. 2cg Inc., London, Ontario [online]: Available from 2cg.ca/articles.php.

Wagner T, and Arnold P. 2008. A new model for solid waste management: an analysis of the Nova Scotia MSW strategy. Journal of Cleaner Production, 16: 410–421. DOI: 10.1016/j.jclepro.2006.08.016

Waste Diversion Ontario. 2016. Municipal data: residential GAP diversion rates [online]: Available from rpra.ca/datacall/about-the-datacall/.

Watkins E, Hogg D, Mitsios A, Mudgal S, Neubauer A, Reisinger H, et al. 2012. Use of economic instruments & waste management performances. Report to the European Commission DG ENV, Bio Intelligence Service S.A.S, Paris, France. 180 pp. Available from ec.europa.eu/environment/waste/pdf/final_report_10042012.pdf.

Williams ID, and Kelly J. 2003. Green waste collection and the public's recycling behaviour in the Borough of Wyre, England. Resources, Conservation and Recycling, 38: 139–159. DOI: 10.1016/S0921-3449(02)00106-4

World Bank. 2011. Solid waste management in Bulgaria, Croatia, Poland and Romania: a cross-country analysis of sector challenges towards EU harmonization. The World Bank, Sustainable Development Department, Europe and Central Asia Region (ECSSD), Washington, D.C. [online]: Available from documents.worldbank.org/curated/en/240901468233089624/Solid-waste-management-in-Bulgaria-Croatia-Poland-and-Romania-a-cross-country-analysis-of-sector-challenges-towards-EU-harmonization.

Zabaleta I, and Rodic L. 2015. Recovery of essential nutrients from municipal solid waste—Impact of waste management infrastructure and governance aspects. Waste Management, 44: 178–187. PMID: 26248488 DOI: 10.1016/j.wasman.2015.07.033