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Towards lower carbon footprint patterns of consumption: The case of drinking water in Italy

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ABSTRACT

The effects that individual consumption behaviours have on climate change are explored, focusing on products that satisfy the same need but with different carbon footprints. Two types of drinking water, produced, distributed and consumed in Italy, were compared as a case study: tap water and PET-bottled natural mineral water. The first is the one supplied to the municipality of Siena, while the second is a set of 6 different Italian bottled water brands. The results showed that drinking 1.5 L of tap water instead of PET-bottled water saves 0.34 kg CO_2eq. Thus, a PET-bottled water consumer (2 L per day) who changes to tap water may prevent 163.50 kg CO_2eq of greenhouse gas emissions per year. In monetary terms, this translates into a tradable annual verified emission reduction (VER) between US$ 0.20 and 7.67 per drinker. Analysing a mature bottled water market, such as the Italian one, may provide insights into the growing global bottled-water market and its effects on climate change. The environmental and economic benefits of changing drinking water habits are also discussed.

1. Introduction

Greenhouse gas emissions are considered a criterion for evaluating environmental performance of products and activities (Kollmuss et al., 2008). Among the various indicators, the carbon footprint has become popular for estimating contribution to climate change (Baldo et al., 2008, 2009; Sinden, 2009; Iribarren et al., 2010), being applied to energy production technologies, mobility management and energy efficiency (Capoor and Ambrosi, 2008). Greenhouse gas emissions can also be prevented by changing individual patterns of consumption. Choice of the most virtuous goods and services in terms of environmental impacts should be encouraged. Goods can be considered alternative when they satisfy the same need to the same degree (Arnold, 2008).

Consumption of drinking water is not a matter of consumer preference, since it is necessary for human life. In most of the western world, it is supplied at least as bottled water and domestic tap water; the choice between the two is a matter of consumer preference, at least in those countries, such as Italy, where, on average, the quality of average tap water is found to be not worse than the quality of bottled water (Cidu et al., 2011). Bottled water has become a habit for many people because it is perceived as safer, healthier and of better quality (Ferrier, 2001), but a clearer picture of bottled water consump-

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tion can be achieved when other aspects are also considered such as cultural factors, perceived quality of tap water source, and demographic variables (Doria, 2006). A positive correlation between *per capita* income and bottled water consumption can be found in most countries (IBWA, 2009). This is especially true in developed countries, where consumption trends of bottled water are positive in most cases. In all *top-twenty* bottled-water-drinking countries except in France and Spain, *per capita* consumption increased between 2004 and 2009, as reported in Table 1 (IBWA, 2009).

However, developing countries have played the role of leaders: from 1999 to 2009, countries such as China, Brazil and Indonesia increased their consumption of bottled water, respectively 4.7, 2.8 and 3.2 times (Gleick, 2006; IBWA, 2009). The Italian market is one of the most mature bottled-water markets in the world. In 2009, Italians were the second consumers of bottled water, at 192 L *per capita* (IBWA, 2009). In 1980, before creation of a real national bottled-water market, *per capita* consumption was 47 L (IBWA, 2008). This means that Italians changed their drinking habits from tap water to bottled water in less than 30 years. The Italian national bottled-water market was established in the 1980s, aided by the introduction of polymer bottles instead of glass, which reduced transport costs, and by new ways of promoting bottled water. Therefore, studying a mature bottled-water market, such as the Italian one, could be useful for insights into the future of the growing global bottled-water market and its environmental implications.

The aim of the present study was to evaluate and compare two types of drinking water (tap water and water sold in polyethylene terephthalate (PET) bottles), by applying the carbon footprint methodology to a volume of water of 1.5 L. The comparison provides environmental and economic insights and suggestions for policy makers. In this connection, verified emission reductions (VERs) were calculated from the carbon footprint results in order to provide a solid framework for improving environmental strategies for the drinking-water supply chain. In recent years, voluntary carbon markets have arisen alongside compliance carbon markets created as market-based mechanisms for the attainment of Kyoto GHG emission reduction targets. In voluntary markets, emission reductions resulting from a variety of carbon footprint lowering projects implemented worldwide are purchased by individuals, companies and institutions wishing to take part in climate change mitigation efforts (Capoor and Ambrosi, 2008). Voluntary markets have grown rapidly and since 2007 have begun a new phase of greater stability and transparency (Hamilton et al., 2009). Thus VER can be a valid instrument to make carbon footprint lowering projects economically favourable and attractive for private companies and organizations. The role of consumers in carbon footprint reduction can be developed by voluntary carbon markets as discussed in this paper.

This paper is structured as follows: a brief presentation of the case studies is reported in Section 2.1, while the carbon footprint definition and calculation procedures are contained in Sections 2.2 and 2.3, respectively. Results are presented in Section 3 and discussed in Section 4, which also provides an evaluation of the economic advantages that a CF downsizing can enable.

Table 1 – Bottled water *per capita* consumption, by leading countries, 1999–2009.

2009 Rank	Countries	Volume (L)		
		1999	2004	2009
1	Mexico	117.4	168.5	234.3
2	Italy	154.8	183.6	191.9
3	United Arab Emirates	84.2	105.6	151.8
4	Belgium–Luxembourg	121.4	148.0	138.9
5	Germany	101.0	124.9	130.6
6	France	117.6	141.6	127.9
7	Lebanon	71.3	101.5	120.4
8	Spain	100.8	136.7	118.9
9	Hungary	29.3	76.1	110.9
10	United States	61.7	87.8	104.5
11	Slovenia	47.3	80.3	102.6
12	Thailand	66.8	76.5	99.9
13	Saudi Arabia	77.0	87.8	99.9
14	Switzerland	91.3	99.6	98.4
15	Croatia	39.1	68.5	96.9
16	Qatar	74.3	78.0	96.6
17	Cyprus	65.7	92.0	92.7
18	Austria	74.8	82.1	89.0
19	Czech Republic	62.1	87.0	88.2
20	Hong Kong	–	58.3	82.9

Source of data: Beverage Marketing Corporation (IBWA, 2009).

2. Materials and methods

2.1. *Case study*

Two types of drinking water were evaluated and compared: tap and PET-bottled water. The tap water (hereafter TW) studied was that supplied to the municipality of Siena (Italy). The water supply network serving 56 smaller municipalities in Siena Province is managed by a single company. It includes different springs, wells and waterworks. In order to isolate and analyse a homogeneous system, only the district of Siena was considered with its 110 km of pipelines, supplying water to users through 220 km of minor pipes. In 2007, the population of Siena (about 55,000 people) consumed about 7.00 GL of water for household uses (AATO 6, 2007).

The PET-bottled water (hereafter BW) studied was natural mineral water, marketed by six Italian companies. The production of the six companies amounted to about 10% (by volume) of the total bottled-water sold in Italy (IBWA, 2008). These companies were selected to provide a representative sample of the Italian market. They differed, among other things, in: (i) plant geographical location, (ii) volume of water bottled per year, (iii) bottling practices, (iv) packing size, (v) market distribution in Italy. Instead of calculating six parameters, we used a weighted mean of six water samples.

As reported in Section 2.3, data for carbon footprint calculations of the two water types was elaborated under a life-cycle perspective (JCR, 2010; SETAC, 2008; ISO, 2006d,e). Inventory data for TW and BW (i.e. quantity of inputs involved in the life-cycle of each drinking water system) is from 2007 unless otherwise specified (see Table A of the supporting material for further details).

2.2. *Carbon footprint and economic assessment*

The carbon footprint (hereafter CF) is a measure of a product's impact on the environment, in terms of greenhouse gases (GHGs) emitted along its supply chain (Finkbeiner, 2009; Wiedmann and Minx, 2008; EPLCA, 2007). Thus it is a subset of the data covered by the more comprehensive life cycle assessment (LCA), which takes into account the consumption of resources and all the impacts associated with a product's full life cycle, from manufacture, through use and maintenance, to final disposal (Baldo et al., 2008; EPA, 2006; EEA, 1997).

CF is measured by converting all GHG emissions to an aggregate "carbon dioxide equivalent" (e.g. kg CO_2eq), which represents global warming potential (GWP). As defined by the Intergovernmental Panel on Climate Change (IPCC, 2007), GWP is an indicator that reflects the relative effect of a greenhouse gas in terms of climate change over a given time period, such as 100 years (GWP_{100}). The GWPs for different emissions (such as CO_2, CH_4, N_2O, HFCs, SF_6 and PFCs, which are the main GHGs) can be added to make a single indicator that expresses overall contribution to climate change. Further information about CF and how it is calculated for goods and services based on LCA techniques can be found in PAS 2050 specific guidelines provided by the British Standards Institution (Carbon Trust, 2008; BSI, 2008).

The CF is a complete methodology that can be used to assess not only the environmental load of a product or process, but also the results of GHG emission reduction projects, for example to calculate carbon offsets. It is therefore also a valid method to make climate change mitigation efforts economically attractive. The *carbon offset* is defined as the act of avoiding or reducing GHG emissions in one place in order to compensate GHG emissions occurring somewhere else (Clean Air – Cool Planet, 2006). One carbon offset unit represents 1 t CO_2eq. GHG emissions cause environmental effects on a global scale, so the geographic source of GHG emissions is irrelevant to their climate-change impact (Gillenwater et al., 2007). Carbon offsets can therefore be traded in global markets.

The so-called "carbon markets" are of two kinds: Kyoto Compliance markets and Voluntary markets. While *emission certificates*, involving countries that signed the Kyoto Protocol, are exchanged in the former, *verified emission reductions* (VERs) are exchanged in the latter. One VER represents 1 t CO_2eq. To assure buyers that VERs correspond to a real emission reduction, VERs must be calculated according to a VER Standard. One of the main problems with VERs is that there are several VER standards with different rules currently in use (Capoor and Ambrosi, 2008). The price of VERs, determined by the market, also depends on which standard is used: the stricter the standard, the higher the price. Since this paper is a preliminary study, aiming to determine the number of VERs that can be created by a project on drinking water systems, no choice was made about the VER standard. The price used in the calculations was therefore US$ 7.34/tCO_2eq, currently the average price of a voluntary carbon certificate, though prices cover a wide range (US$ 1.20–46.90/tCO_2eq) (Hamilton et al., 2009).

The two drinking water systems (BW and TW) were evaluated considering environmental and economic aspects, estimating CF and VERs, respectively. This evaluation may be the first step towards the application of such a tool in the framework of the drinking water industry in Italy.

2.3. *Carbon footprint of drinking water*

GHGs emissions were monitored along the supply chain of the drinking-water systems for BW and TW, including upstream production of all materials, energy carriers, and transport of the products involved, observing the guidelines proposed by the International Standards Organization (ISO, 2006a,b,c). This approach required a life cycle inventory (LCI), which is a core phase of the LCA procedure (Baldo et al., 2008; JCR, 2010; ISO, 2006d,e; SETAC, 2008). The boundaries of the two drinking-water systems were defined from *cradle to gate*, for a *partial* product life cycle: from the manufacture of raw materials (cradle) to the factory gate (i.e. before the use and disposal phases of the product). Since tap water is supplied directly to the final user's home, to enable a better comparison between the two systems, spatial boundaries were expanded to include the transport phase of bottled water to consumers (from the factory gate to the consumer's table). Furthermore end-of-life processes for the waste created by the two systems were not considered. Inputs related to *capital energy*, i.e. energy associated with buildings and machinery involved in the life cycles (Baldo et al., 2008), were also ignored. In general, this omission is not thought to introduce any significant error, as its contribution is usually less than 1% of total-system energy (Bousted and Hancock, 1979). Detailed information about the life cycle inventory and the system boundaries of BW and TW are provided in Table A of the appendix. The functional unit for this study was 1.5 L of drinking water.

The CF of drinking water, in terms of mass of CO_2 equivalent, was calculated with the formula:

$$CF_{\text{drinking-water}} = \left(\sum_{i=1}^{n}(Q_i \times \overline{GWP_i})\right) \quad (1)$$

where Q is the quantity of inputs *i* of the drinking water life cycle and GWP is the average global warming potential (in mass of CO_2eq) of inputs *i* over a given period, such as 100 years (GWP_{100}), as suggested by the IPCC (2007). GWP is calculated as follows:

$$\overline{GWP_i} = \left(\sum_{j=1}^{n}(GhG_j \times GWP_j)\right) \quad (2)$$

where GHG_i are greenhouse gas emissions and GWP_j is the relative global warming potential of greenhouse gas *j*. GHG emissions were obtained from the literature and specific databases usually used for life cycle studies (APME, 2007; CPM LCA Database, 1996–2002; CPM, 2007; FEFCO, 2006; EDIP-Database, 1997; ELCD, 2009; Joshi et al., 2004).

The CF of TW and BW is given by the sum of three components:

$$CF = CF_{\text{materials}} + CF_{\text{energy}} + CF_{\text{transport}} \quad (3)$$

The meaning and calculation of each component of Eq. (3) depend on the kind of drinking water considered.

For bottled water:

- $CF_{\text{materials}}$ is the CF of production of raw materials for packaging, i.e. plastics (PET, PP, HDPE/LDPE), corrugated

cardboard, glue, wood, for operation of industrial machinery (i.e. lubricating oil), and for water treatment (i.e. additives);
- CF_{energy} is the CF of production (and combustion) of energy carriers (i.e. electric power, gas, fuel oil) required by the plant.
- $CF_{transport}$ is the CF of production (and combustion) of transport fuels (i.e. diesel and gasoline);

For tap water:

- $CF_{materials}$ is the CF of production of raw materials (i.e. steel, cast iron, plastics and fibreglass) involved in maintenance of pipes, waterworks and other structures, and production of chemical compounds used for water treatment (i.e. hydrochloric acid and sodium chlorite). Tap water does not require containers or packaging;
- CF_{energy} is the CF of production (and combustion) of energy carriers (i.e. electric power, gas, fuel oil) required to pump and distribute water and for administrative activities.
- $CF_{transport}$ is the CF of production (and combustion) of transport fuels used by the maintenance fleet. Tap water is not transported but piped from source to users so this component was neglected.

3. Results

A detailed life cycle inventory was performed for all the PET-bottled and public tap water analyzed. Comprehensive information is reported in Table A of the supporting material, where the quantities of the various inputs are provided for the different drinking-water systems. Table 2 includes the carbon footprint of BW and TW calculated by Eq. (1), and the average GWP or the sum of GHG emissions (in kg CO_2eq) calculated for production of the corresponding inputs – see Eq. (2). CF values are shown for single inputs of production, identifying the type of contribution in the life cycle (i.e. materials, energy, transport). Further specific considerations about the use of LCA databases and calculation of average GWPs are provided in the Supporting material.

The average CF of bottled water (CF_{BW}) per functional unit (1.5 L drinking water) is 3.37×10^{-1} kg CO_2eq. The largest contribution comes from $CF_{materials}$ (1.98×10^{-1} kg CO_2eq, 59% of CF_{BW}), which is mainly due to packaging processes. PET bottle production is 46% of the average CF. $CF_{transport}$ is 8.81×10^{-2} kg CO_2eq (26% of CF_{BW}), followed by CF_{energy} 5.10×10^{-2} kg CO_2eq (15% of CF_{BW}).

The CF of tap water is 1.35×10^{-3} kg CO_2eq per 1.5 L drinking water. CF_{energy} is the largest term at 94% where the electricity consumption contributed about 92%. The CF of the steel is the largest contribution to the $CF_{materials}$ of tap water, being about 5% of total CF_{TW}. The contribution of other flows is negligible.

The pie diagram in Fig. 1 shows the differences in CF composition of the two types of drinking water.

Average CF_{BW} is about 250 times greater than CF_{TW} per functional unit. This means that the release of 3.36×10^{-1} kg CO_2eq can be avoided by drinking 1.5 L of TW instead of 1.5 L of BW. This difference can be explained by looking at the CF

Table 2 – Carbon footprint of PET-bottled water and tap water of Siena (Italy), for the terms of Eq. (2). Values are per functional unit: 1.5 L of drinking water.

Life cycle input processes	$\overline{GWP}$[a] kg CO_2eq/unit	GHG data source	Contribution type	BW kg CO_2eq	TW kg CO_2eq
PET	4.68E+00	APME (2007)	M	1.56E−01	–
PP	1.96E+00	APME (2007)	M	2.94E−03	3.59E−09
HDPE/LDPE	2.08E+00	APME (2007)	M	3.56E−02	4.22E−09
Steel	1.14E+01	CPM (2007)	M	–	6.65E−05
Fibreglass	2.04E+00	Joshi et al. (2004)	M	–	2.10E−08
PVC	3.20E+00	APME (2007)	M	–	1.14E−08
Cast iron	2.51E+00	EDIP (1997)	M	–	8.06E−06
Hydrochloric acid (HCl)	1.50E+00	APME (2007)	M	–	8.06E−07
Corrugated cardboard	4.97E−01	FEFCO (2006)	M	2.57E−03	–
Glue	4.14E−01	EDIP (1997)	M	8.72E−06	–
Wood (pallet waste)	2.02E+00	CPM (2007)	M	7.13E−04	–
Lubricating oil	2.93E−01	EDIP (1997)	M	6.61E−07	–
Additives (O3, O2, CO2, N)	1.47E−01	ELCD (2009)	M	6.85E−04	–
Electric power	7.11E−01[b]	ELCD (2009)	E	4.36E−02	1.24E−03
Gas	3.16E+00	ELCD (2009)	E	4.27E−03	1.26E−07
Fuel oil	3.53E+00	ELCD (2009)	E	3.06E−03	3.12E−05
PET preforms to plants (by truck)	1.02E−06[c]	ELCD (2009), IPCC (2006)	T	2.04E−04	–
BW to stores (by truck: 82%)	6.06E−05[c]	ELCD (2009), IPCC (2006)	T	2.32E−02	–
BW to stores (by rail: 18%)	2.07E−05[c]	ELCD (2009), IPCC (2006)	T	1.74E−03	–
BW to consumers (by diesel car: 50%)	5.60E−03[c]	ELCD (2009), IPCC (2006)	T	2.80E−02	–
BW to consumers (by gasoline car: 50%)	6.99E−03[c]	ELCD (2009), IPCC (2006)	T	3.50E−02	–
		Total carbon footprint		3.37E−01	1.35E−03

BW, bottled water sample; TW, public tap water of Siena Municipality; M, materials; E, energy; T, transport.
[a] Sum of GHG emissions (in kg CO_2eq) of input processes. Inputs are in kg if not otherwise expressed. See Appendix A (Supporting Material) for inventory data.
[b] kg CO_2eq per kWh.
[c] kg CO_2eq per bottle/km.

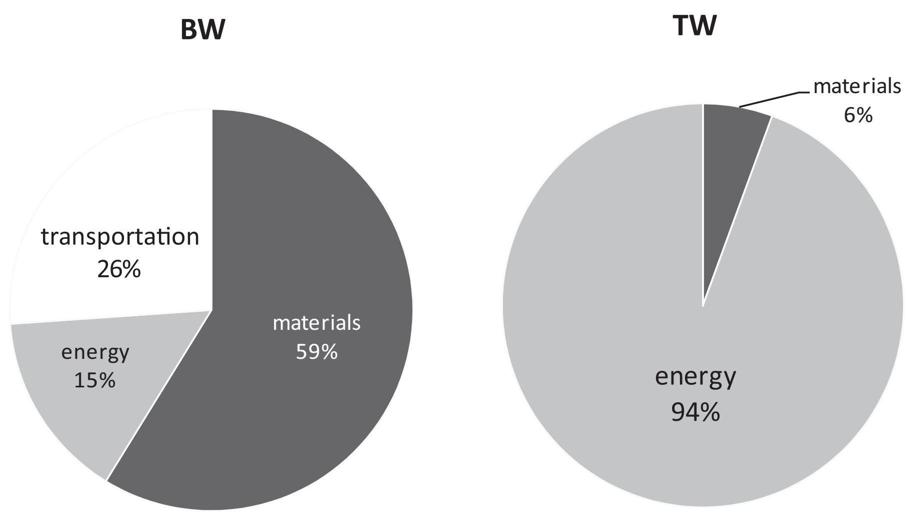

Fig. 1 – The composition of carbon footprint for the two types of drinking water: BW-bottled water (left) and TW-tap water (right).

components. TW does not have a $CF_{transport}$, because it is piped to users' homes. The $CF_{material}$ and CF_{energy} components of TW make up for this, since $CF_{material}$ is related to maintenance of pipes and other infrastructure and CF_{energy} is related to water pumping and distribution. The absence of $CF_{transport}$ in TW avoids emission of 8.81×10^{-2} kg CO_2eq per functional unit. A minor difference is observed between the two CF_{energy} components: in this case drinking TW instead of BW means avoiding emission of 4.97×10^{-2} kg CO_2eq per functional unit. The main difference is between the two material components. The former is about 2600 times larger than the latter, since TW does not require any packaging. It only requires materials for pipe maintenance and chemicals for water treatment. Emission of about 1.98×10^{-1} kg CO_2eq can be avoided from materials alone by drinking TW instead of BW.

These CF results are used to quantify the VERs that can be earned by changing from BW to TW. When the difference is expressed in monetary terms, drinking TW instead of BW means saving US$ 0.0004–0.0158 per unit as tradable VERs. For a person drinking 2 L water per day (WHO, 2006), this is US$.0.20–7.67 per year.

4. Discussion

The CF of tap water and PET-bottled water were remarkably different, indicating that individual choice of drinking water may have big consequences in terms of GHG emissions. The life cycle of BW has much greater global warming potential than that of TW, though both quench thirst to the same extent.

These results warrant a comment about the substitutability of PET-bottled water and tap water. In microeconomics and common sense, two goods are substitutes if they satisfy similar needs or desires (Arnold, 2008). Generally, a rise in the price of one increases demand for the other. Hence, consumer choice between substitutes is based on price: the substitute with the lower price is preferred (Baumol and Blinder, 2008). Thus if BW and TW are substitutes, two conditions should be met: both should satisfy the same need and the demand for one should rise when the price of the other increases. At first glance, these conditions seem to be met, but closer scrutiny reveals that although water is drunk to satisfy thirst, a physiological need, consumption of BW does not seem to depend on the price of TW or *vice versa*. Consumption of BW increases with income (IBWA, 2009). Bottled water prices are generally 500–1000 times higher than those of TW (Ferrier, 2001), and BW is considered to reflect an opulent life style and is generally perceived as healthier and safer than TW due to advertising (Kunze, 2008). Hence, consumer choice between TW and BW seems to be more complex than raw price comparison and BW may satisfy more complex needs than thirst (Doria, 2006). Defining BW and TW as substitutes cannot be taken for granted, for the above reasons. From a consumer point of view they may be defined as *weak substitutes*, while from a physiological point of view they satisfy the same need. This is particularly true in those countries where the average quality of TW is not inferior to the average quality of BW (Doria et al., 2009).

In this paper we have shown that the consumer's choice has consequences for climate change, and growing public awareness about climate change may help consumers to perceive TW and BW as substitutes. Another aspect should not be overlooked: whether or not one drinks TW, it is in any case provided to everyone for drinking and other domestic uses. The production system of TW would therefore exist even if everyone drank BW. Moreover, the amount of TW drunk is negligible compared to other uses. Italian mean domestic consumption of water is about 197 L *per capita* per day (Martire and Tiberi, 2007) compared to about 2 L of water per day drunk by adults (WHO, 2006), about 1%. This means that domestic TW consumption is added to the CF of BW drinkers. In fact, 99% of TW consumed for other uses is usually of drinking-water quality (as in the case of Siena Municipality). In this paper the CF of a BW-drinker consuming 2 L per day is 1.64×10^{2} kg CO_2eq per year, plus the CF for domestic use of TW (195 L per day), namely 6.41×10^{1} kg CO_2eq per year. Hence drinkers of BW have a CF of 2.28×10^2 kg CO_2eq per year for water in general. Instead, the CF of a TW-drinker is composed of a fraction used for drinking (2 L per day) and a fraction for other domestic uses (195 L per day), namely 6.48×10^1 kg CO_2eq per year. The difference between a TW-drinker and a BW-drinker in terms of GHG emissions related to their overall "drinking" water consumptions is about 1.64×10^2kg CO_2eq per year. To limit climate change, the substitutability of BW and TW perceived by consumers needs to be closer to physiological needs than to current consumer preferences. This could be achieved by informing consumers about the different amounts of GHGs associated with TW and BW and about the role they can play in preventing climate change. This role becomes important at population level. For instance, if the whole population of Siena (55,000 persons) drank BW, the annual CF of the population for all water use would be about 1.26×10^7 kg CO_2eq per year (i.e. CF_{BW} + 99%CF_{TW}). If they all drank TW, it would be about 3.56×10^6 kg CO_2eq per year. The people of Siena could therefore be responsible for emitting or not emitting 8.99×10^6 kg CO_2eq per year, depending on the number of BW-drinkers in the population.

In order to better appreciate the scale of this amount, we can say that it is the amount emitted by 5000 economy cars, each travelling 15,000 km; it is the amount that can be saved by generating 11 GWh from wind instead of coal; it is also the amount avoided by replacing 9000 incandescent light bulbs with compact fluorescent bulbs for one year of uninterrupted use (source: Europe's Energy Portal, http://www.energy.eu/). Our calculations can be expanded to the top 10 BW consumption leading countries. Theoretically, if a complete switch from BW to TW could be possible, a reduction of GHG emitted would be around 34×10^6 t CO_2eq per year. This amount is equivalent to, for example, 7% of the total Italian annual CF or half the annual CF of Finland (source: United Nation Development Programme, International Human Development Indicators, http://hdr.undp.org/en/).

Nevertheless, policy makers usually overlook the role of substitute goods when searching for solutions to reduce their country's contribution to climate change. Traditional ways of reducing CF concern energy production, household efficiency and mobility management. However, our results suggest that non-traditional sectors such as consumer habits are also potential sources of reduction.

Besides these remarkable environmental benefits, reducing CF can also have economic benefits. Even if an economic mechanism incentivizing consumers to change their habits to lower their CF does not exist already, we suggest that something should be implemented. Here we try to propose the application of the VERs mechanism to consumer's choices. For example, in the case of the CF for water of the population of Siena, the economic benefit could be between US$ 10,000 (lowest VER price in 2008) and US$ 422,000 (highest VER price in 2008) when considering the difference in GHG emissions (i.e. 8.99×10^6 kg CO_2eq per year). This earned money can be used, for example, by the population to contribute to lower the price of TW. Voluntary carbon markets therefore suggest the implementation of CF reduction projects related to drinking water. On the other hand, a bottled-water company could use voluntary carbon markets to compensate use of materials with high CF or inefficient transport. Since industrial technologies are already concerned with improving process efficiency (reducing costs, use of resources and related emissions), the carbon footprint for transport should be the first parameter to reduce in a supply chain such as the production and distribution of bottled water. Thus, if Italian BW production decreased, for instance, its CF for transport by 10–20% by improved mechanical engineering or logistic management of goods distribution, the VER earned could decrease by 2–5%, respectively. In terms of money, this means a reduction of US$ 200–10,000 (10% scenario) or US$ 500–20,000 (20% scenario) per year from the above VERs, respectively, in the municipality of Siena.

Despite the feasibility of this preliminary evaluation of two substitute goods (TW and BW), in terms of carbon footprint calculations the consistency of data processed might be improved (e.g. using specific life cycle inventory data with more consistent spatial and temporal characteristics), system boundaries expanded (e.g. including important life cycle phases of the product, such as end-of-life), and an appropriate economic mechanism identified to encourage consumers change their habits in favour of lower CF substitutes. Such refinements would make this tool reliable and applicable to real contexts of production of substitute goods on local, national and international scales.

5. Conclusions

When CFs of PET-bottled and tap water were calculated and compared to quantify the contribution of individual patterns of consumption to climate change, the results showed that choosing substitute goods with lower CFs may avoid a remarkable amount of GHG emissions. An Italian population of 55,000 persons, drinking 2 L of water a day per person, can prevent emission of about 9000 t CO_2eq per year by choosing tap water instead of PET-bottled water. Our analysis suggests that economic incentives schemes, such as voluntary carbon markets, combined with CF assessments, are a powerful tool available to companies, organizations and consumers to reduce environmental load and to favour private and public economic competition and improvement. As demonstrated for Italy, drinking tap water instead of PET-bottled mineral water is associated with environmental and economic benefits that are far from negligible. However, greater margins of life-cycle improvement can be expected in the BW than TW supply chain, limiting the difference in CF between these substitute goods.

Policy makers should seriously consider the role of substitute goods in reducing GHG emissions. Traditional ways of decreasing the CF have concerned energy production, household efficiency and mobility management. Good results could also be achieved by modifying consumer habits, as our results suggest in the case of drinking water. Voluntary carbon markets also offer an opportunity to make reduction of the CF pay in economic terms.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.envsci.2011.01.004.

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