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Assessing the Environmental Impacts of Household Water Supply: A Case Study Considering Consumption Patterns within a Life-Cycle Perspective

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Abstract: Household water supply can cause different environmental impacts associated with the consumption of energy and materials, the generation of waste, and other inputs and outputs necessary to treat and distribute water. These impacts depend on the population's consumption patterns, due to the potential availability of different water sources. In this work, the environmental impacts of water supply were evaluated from a production-consumption perspective, integrating life cycle assessment (LCA) and a survey for determining the end uses and sources of water at household level. The proposed method was applied in the city of Chillán (Chile), where three main sources exist: tap, bottled, and well water. Two household profiles were evaluated, differentiated by the presence of wells within the household. The results show that bottled water generates impacts up to three orders of magnitude greater than the other sources. Although it is the source with the lowest volumetric contribution (<1%), it accounts for 39–92% of the household impacts. Households with well access present greater per capita consumption of water, mainly associated with outdoor activities, but the environmental impacts are similar between profiles. Overall, this study demonstrates the importance of integrating a consumption perspective into LCA studies, generating better information for decision-making.

Keywords: water supply; life cycle assessment; sustainable production and consumption; consumption patterns; environmental assessment

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1. Introduction

During the last decades, water access and sanitation have become a worldwide priority, being included among the 17 United Nations (UN) Sustainable Development Goals [1]. However, securing an equitable and sustainable access to water is an increasingly difficult target, considering climate change and the growing pressures on water resources, with an estimated 40% of the world population currently living in areas under water stress or scarcity [1].

In order to provide access to water and sanitation services, in urban areas the natural water cycle is altered by human infrastructure and activities. The anthropic water cycle begins with water intake from surface or underground, followed by water potabilization, storage, distribution, consumption, wastewater collection, and finally its treatment and discharge into a natural watercourse. In order to be consumed by the population without public health risks, water must undergo a purification process, which involves a series of unit operations aimed at removing contaminants such as pathogens, organic matter, and minerals that are naturally present in water [2].

In Chile, water supply is provided by 54 companies that are regulated by the Superintendent of Sanitary Services (SISS). According to recent data, water supply reaches 99.8% of the urban population, which represents ~80% of the total population of the country [3]. However, household water requirements are not exclusively covered by the centralized system; rather, there are other supply sources that vary from place to place and may include water extracted from dug or driven wells, bottled water, and locally collected rainwater [4].

In order to supply water in enough quantity and quality, every water production system requires energy, materials, and inputs for the production, storage, and distribution of water, which in turn are associated with the generation of several environmental impacts. Life cycle assessment (LCA) is a tool that allows these impacts to be quantified and compared, based on four fundamental steps: the definition of scope and objectives, inventory analysis, environmental impact assessment, and the interpretation of results [5].

It is important to note that the use and/or consumption stage in the life-cycle of a product can be determinant over its environmental impact; therefore, including this perspective in LCA studies allows for a more complete vision of the environmental impacts of a product or service and the generation of appropriate information to support decision-making [6]. Among the instruments that can be used to assess consumer behavior or preferences, surveys are a simple and reliable methodology that allow researchers to make estimations regarding consumption patterns, assess the variables that intervene in consumption habits, and construct user typologies to support LCA studies [6].

However, most studies published in the literature that focus on residential water consumption have a scope that is limited to the production processes (also knowns as a "*cradle-to-gate*" scope), without assessing how consumption habits and consumer types can affect the resulting environmental impacts [2,7–10]. Consumer preference for one or another supply source may be influenced by personal preferences or factors such as socioeconomic status or education level [11,12]. For example, one of the main drivers of bottled water consumption can be the perceived higher quality of this source of water compared to tap water, as previous studies confirm [13,14]. Even in countries with high quality standards for tap water, bottled water can be perceived as a safer option [14], and other factors such as its portability, convenience, association with certain lifestyles, and allegedly better organoleptic qualities can also influence the population's choices regarding its consumption [13]. This situation can also be extrapolated to other water sources, such as locally extracted well-water, captured rainwater, and reclaimed or desalinated water, sometimes due to concerns that go beyond perception, such as the potential presence of unregulated micropollutants in water [15].

In this scenario, this study proposes a conceptual framework for the integration of water consumption practices into LCA studies based on the development, validation, and application of population surveys in parallel to the environmental assessment of production systems. This method was applied to the analysis of the city of Chillán, in south-central Chile, where three main water supply sources were identified: the centralized purification and distribution system (tap water), bottled water consumption, and household extraction from wells. This study contributes to the area of study through the proposal and application of a simple framework to integrate consumption patterns within LCA, in order to improve the results of the analysis and to generate useful information to improve water production systems and to support the development of public policies on sustainable urban water management.

2. Materials and Methods

The study area is the Chillán and Chillán Viejo conurbation (36°36′23.9″ S, 72°6′12.4″ O), located in the Ñuble Region of Chile (Figure 1). The total population of the city is 215,646 inhabitants, with a total of 79,130 households [16].

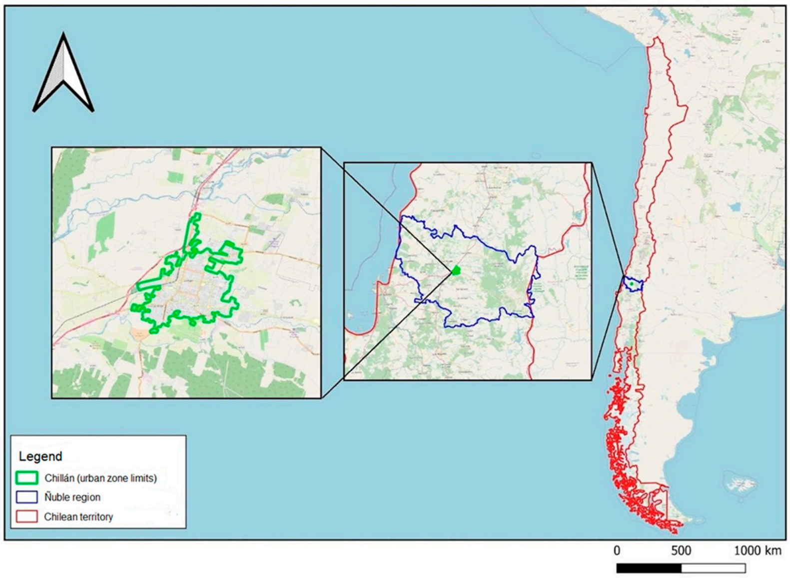

Figure 1. Spatial location of the study area (source: own elaboration based on IDE-Chile data).

The objective of the study was to assess and compare the environmental impacts associated with the drinking water supply in the city of Chillán (Chile), considering different sources and consumption patterns. The proposed methodology has four steps: (1) the construction of life-cycle inventories for the different water supply sources; (2) the quantification and comparation of the environmental impacts of the water sources; (3) the determination of the consumption profiles of households; and (4) the assessment of their environmental impacts. Figure 2 summarizes the scope of the study, and all the methodological steps are described in the following subsections.

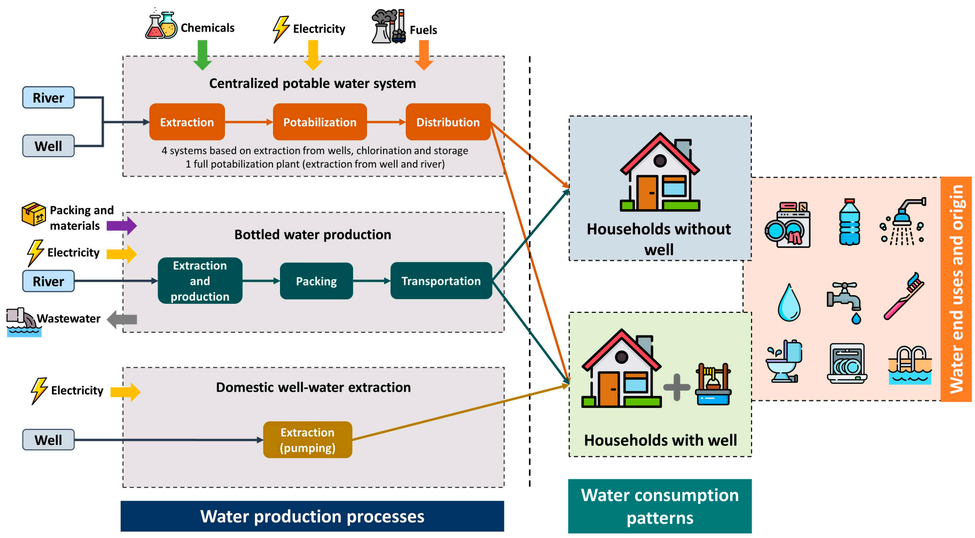

Figure 2. Scope of the study (source: own elaboration).

2.1. Life Cycle Inventories

For the analysis of the different water sources, the functional unit was defined as the "provision of 1 m^3 of water to households" via tap water production systems, bottled water, and decentralized extraction from wells. The scope of the analysis includes the operation of each system, including all steps from water extraction and processing to its distribution to the consumption point. Construction and dismantling of the production systems was not included.

The life cycle inventory for tap water was constructed based on data provided by the local water company (ESSBIO S.A.). Data represents the operation of the systems during the period spanning January 2017 to August 2020.

Water supply comes from 5 production systems, located at various points in the city:

1. In the central purification system of Chillán (SCh), water is extracted from three points: one surface water intake in the Chillán River and two groundwater intakes. The purification process is based on conventional treatment processes, including flocculation, sedimentation, rapid filtration, and disinfection (chlorination).
2. The northeastern extraction and storage system (NE) comprises two groundwater intake points, with a compact treatment process consisting of filtration and disinfection by chlorination.
3. The northern extraction and storage system (N) comprises five groundwater intake points and a disinfection process by chlorination.
4. The southern extraction and storage system (S) has three groundwater intake points and a disinfection process by chlorination.
5. The southeastern extraction and storage system (SE) includes two groundwater intake points and a disinfection process by chlorination.

Each of these systems has an independent storage system, except for the central purification system of Chillán, which pumps its production to the southern or northern tanks or distributes it directly to meet the immediate demand. As the main pumping stations of the city are located in the same location of the production systems, their electricity consumption is accounted in the corresponding inventory. Furthermore, an additional consumption of ~1.3×10^{-3} kWh per FU was reported by the water company for the remaining pumping processes in the distribution network, which was also accounted in the inventory.

Bibliographic sources were used to construct the bottled water inventory [7,17]. For this process, a surface water intake was assumed, as most of the bottled water companies in Chile process water coming from mineral springs, and do not extract it from underground. The transport distance of the manufactured product (bottled water) was estimated as the distance between a bottling plant located in the O'Higgins Region (34°10′15″ S, 70°44′40″ W) and the point of consumption (Chillán), resulting in 321 km.

For the life cycle inventory of well water, it was assumed that the main consumption is associated with the electricity needed to extract water from shallow wells; therefore, consumption in kWh of a 0.5 HP pump was used as a reference, with estimated maximum and minimum flows determined based on technical data sheets of pumps available on the market.

The life cycle inventories for the background processes were obtained from the Ecoinvent v3.8 database. Electricity consumption for all processes was assumed to come from the national electricity mix. The Chilean energy mix was modified with data updated to 2020 from the National Electric System [18], resulting on a 35.1% contribution from hard coal, 26.6% from hydroelectricity, 17.7% from natural gas, 9.8% from photovoltaic systems, 7.1% from onshore wind turbines, 2.4% from biomass, 0.8% from oil, 0.3% from geothermic plants, and 0.2% from wood co-generation. A summary of the used Ecoinvent processes can be consulted Table S1 of the Supplementary Material.

2.2. *Life Cycle Impact Assessment*

The impact categories used in this study were selected by means of a literature review of similar studies [4,7,19]. The resulting categories were global warming, terrestrial acidification, freshwater eutrophication, freshwater ecotoxicity, human carcinogenic toxicity, land use, and fossil resource scarcity. The ReCiPe 2016 midpoint method in its hierarchical version was used for the assessment, through its implementation in the SimaPro v9.2 software.

2.3. Consumption Profiles

In order to establish the consumption and end use of residential water by the inhabitants of the city, a survey aimed at seven dimensions was developed, validated, and applied. These dimensions include a sociodemographic description; access to the different water sources (tap, well, and bottled); water consumption habits for (a) personal hygiene, (b) cleaning, (c) outdoor uses, and (d) intake; and water saving practices. The instrument was prepared based on a literature review and was subsequently validated by four evaluators (two environmental professionals with postgraduate degrees and two water community managers with a minimum of two years of experience in their positions). The validation process consisted of a review of four criteria using a score sheet with a range of 1 to 4: clarity, relevance, and coherence of each question, and sufficiency of each dimension. The free-marginal multirater Kappa parameter was used as a measure of the level of agreement among the validators [20]. The validation results and comments allowed the instrument to be improved before its application to the population. The average values for the four criteria assessed during the validation, as well as the Kappa values obtained for each category and dimension can be reviewed in Tables S2 and S3 of the supplementary material. Due to the social restrictions resulting from the COVID-19 pandemic, the survey was applied through a digital formulary (between 28 September and 4 November 2021). A total of 123 surveys were received, of which those answered incorrectly were discarded, leaving a total of 104. Considering a total population of 79,130 households, this results in an error margin of 9.41% with a confidence level of 95%.

Based on the survey results and estimated specific consumption values of appliances and activities (Table S4), the water consumption of each household was quantified, as well as the percentage contribution of each water source. The end-uses included in the survey included dish washing, toilet use, tooth brushing, showering, irrigation, and pool use. Due to the difficulty of obtaining reliable data on some water uses by conducting surveys, the following assumptions were made to complement the information collected by the instrument: dish washing time was assumed to be equal to 5 min (with the faucet open); the daily frequency of toilet use was assumed to be six times to urinate and once to defecate; full-loads in washing machines were assumed to be of 9 kg; tooth brushing was assumed to be performed twice a day, with a duration of 3 min if people leave the faucet open and 1 min if the faucet is closed [21].

Two consumption profiles were constructed based on the information collected in the survey, in which the distinctive criterion was the presence of a well at the respondents' homes. A LCA of each profile was performed, based on the estimated contribution of each water source. The functional unit used for the environmental comparison of the profiles was the "annual per capita water consumption". It is important to mention that the quality of water was not included in the inventory of neither of the water sources, mainly due to the lack of data necessary to quantify the probable presence of potentially toxic micro-pollutants in water.

2.4. Statistical Analysis and Estimation of the Variability of Results

A statistical analysis was carried out for each inventory parameter of the water production systems, including the calculation of the average value, maximum, minimum, mode, and standard deviation. In addition, the normality of the data was evaluated by means of a Shapiro-Wilks test ($\alpha = 0.05$) using the InfoStat v2020 software (Table S5). In this study, the variability of each primary datum of the inventory was represented by means of the standard deviation (for parameters that presented a normal distribution), or assuming a triangular distribution with maximum, minimum, and representative values (mode) (for parameters that presented a non-normal parameters). In the case of data obtained from literature, the variability was estimated by the implementation of the Pedigree matrix through the SimaPro v9.2 software [22]. Variability of the characterization results was estimated by means of a Monte Carlo analysis, using a fixed value of 1000 iterations.

3. Results and Discussion

3.1. Life Cycle Inventories

Table 1 shows the inventory of the five drinking water production systems. In comparative terms, there is a large difference in electricity consumption, which in the case of the northeast extraction system is ~34 times greater than the consumption of the Chillán production system. This difference is mainly related to the scale and efficiency of the systems, as well as the fact that surface intake accounts for 96% of the water extracted in the Chillán production system, unlike the rest of the systems, which have exclusively groundwater intakes and a higher electricity requirement for pumping.

Table 1. Simplified inventory for the operation of the five water production systems.

Parameter	Unit	Production System				
		SCh	NE	N	S	SE
Electricity	kWh/m^3	0.04	1.35	0.79	0.80	0.96
Chlorine gas	kg/m^3	2.4×10^{-3}	8.10×10^{-4}	1.9×10^{-3}	1.2×10^{-3}	1.2×10^{-3}
Calcium hydroxide	kg/m^3	3.1×10^{-4}	-	-	-	-
Oil consumption	kWh/m^3	5.6×10^{-5}	4.9×10^{-3}	1.5×10^{-3}	2.6×10^{-3}	-
Aluminum sulphate	kg/m^3	8.3×10^{-3}	-	-	-	-

SCh: central purification system of Chillán; NE: northeastern extraction system; N: northern extraction system; S: southern extraction plant; SE: southeastern extraction system.

In the case of bottled water, the production system is more complex, as it requires the incorporation of material for bottles, packaging, and the transport requirements for the finished product (Table 2). If we compare the inventory data on energy consumption, bottled water has a consumption ~222 times greater than water extracted from wells (0.09 kWh per m^3) and ~98 times greater than the average Chillán production and distribution system (0.21 kWh per m^3).

Table 2. Inventory of the bottled water production process.

	Parameter	Unit	Mean	References
Inputs	Surface water	m^3	1.26	Garcia-Suarez et al. (2019) [17]
	Polypropylene (PP)	kg	1.40	Garcia-Suarez et al. (2019) [17]
	Corrugated carton boxes	kg	3.01	Garcia-Suarez et al. (2019) [17]
	Kraft paper	kg	0.07	Garcia-Suarez et al. (2019) [17]
	Ozone, liquid	kg	0.35	Garcia-Suarez et al. (2019) [17]
	Packaging film	kg	4.03	Garcia-Suarez et al. (2019) [17]
	Polyethylene Terephthalate (PET)	kg	18	Garcia-Suarez et al. (2019) [17]
	Electricity	kWh	20	Garfí et al. (2016) [7]
Auxiliary processes	Injection molding	kg	1.4	Garcia-Suarez et al. (2019) [17]
	Blow molding	kg	18	Garcia-Suarez et al. (2019) [17]
	Transport, truck	tkm	321	Garcia-Suarez et al. (2019) [17]
Wastes	Wastewater	m^3	0.26	Garcia-Suarez et al. (2019) [17]

3.2. Environmental Impact Assessment: Production and Distribution Systems

The production systems that make up the city's network present differences in their unit operations and processing capacities. The Chillán production system provides the greatest amount of water to the network, equivalent to 80% of the total, followed by the northern extraction system (9%), southern and southeastern systems (5% each), and northeastern system (1%). Correspondingly, an analysis of the contribution of each production system to the environmental impacts of the tap water network shows that the Chillán production system has the highest contribution, with 8.7–50.6% depending on the category, while the lowest contribution is the northeastern extraction system, with 3.6–6.1%. Electricity consumption represented the highest contributing process in all impact categories. In fact, in the case of the southeastern production system the contribution is close to 99% in all categories. Due to the relevance of electricity consumption and the differences observed between systems based on surface intake and sub-surface intake, the comparison of the production systems (Figure 3) reveals a notable difference between the Chillán production system and the other systems, which are mainly based on extraction from wells. In terms of climate change potential, the Chillán system presents an impact 23 times smaller than the northeastern extraction system. It is important to mention that the high variability observed in the characterization results is mainly associated with the variability of the inventory parameters reported by the water company (Table S5).

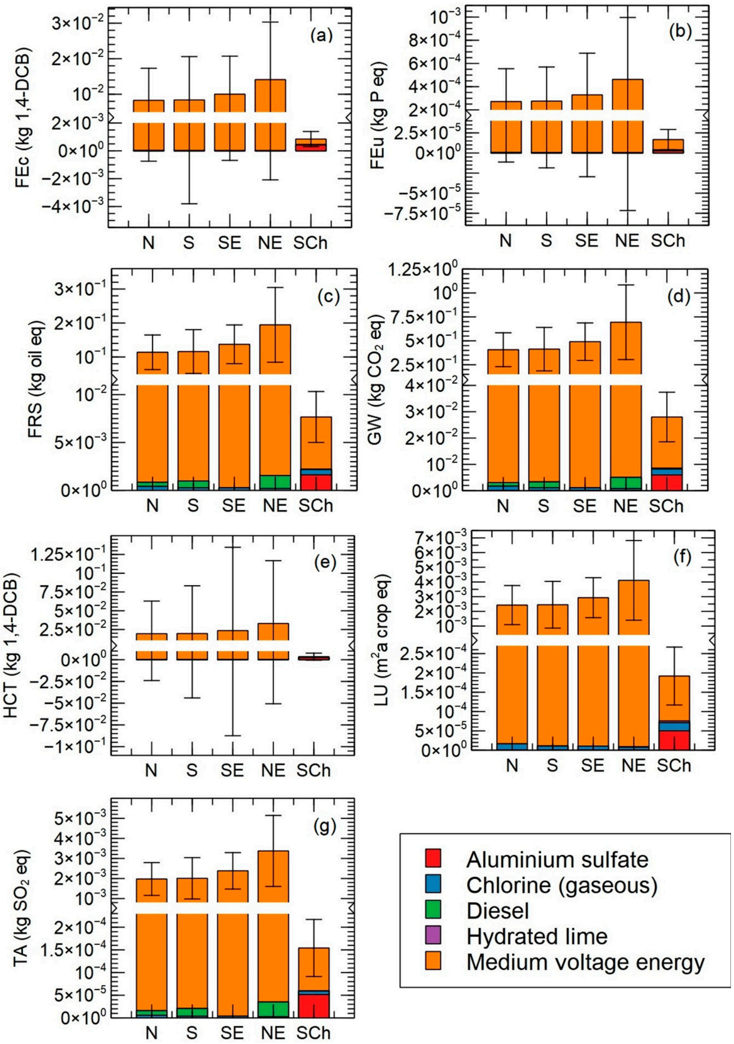

Figure 3. Results of the environmental impact assessment of the water production systems for each impact category, showing the contribution of each process to the environmental load. (**a**) Freshwater ecotoxicity, (**b**) freshwater eutrophication, (**c**) fossil resource scarcity, (**d**) global warming, (**e**) human carcinogenic toxicity, (**f**) land use, and (**g**) terrestrial acidification. SCh: central purification system of Chillán; NE: northeastern extraction system; N: northern extraction system; S: southern extraction plant; SE: southeastern extraction system. Error bars represent the coefficient of variation obtained through Monte Carlo analysis.

Similar to our results, Ortiz-Rodríguez et al. (2014) [19] reported that in the case of four water treatment plants in Colombia, 86% of the impact for the global warming category comes from electricity (which is attributed mainly to the pumping stage), with results ranging from 0.013 to 0.383 kgCO_2 eq per m^3 of tap water produced. The differences between the systems of that study were associated to the type of treatment, the quantity of chemical products used, and the energy mix.

3.3. Environmental Impact Assessment: Bottled Water

Figure 4 presents the contribution of the different processes to the environmental burden of bottled water production. The consumption of polyethylene terephthalate (PET) for the bottles, and the transportation of the bottled water to the retailers are the processes with the highest contributions, with ranges of 8.7–53.6% and 12.3–39.2% depending on the category analyzed, respectively.

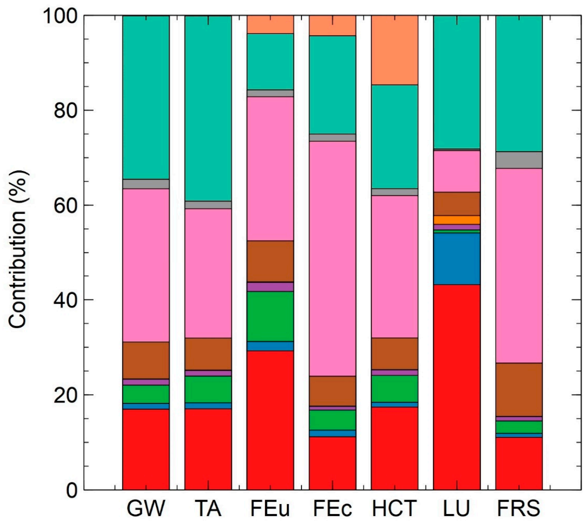

Figure 4. Results of the environmental impact assessment for bottled water. The figure shows the contribution of each process to the environmental load. ■ Blow molding, ■ corrugated board boxes, ■ medium voltage energy, ■ injection molding, ■ Kraft paper, ■ packaging film, ■ polyethylene terephthalate, ■ polypropylene, ■ transport, ■ wastewater from water production. GW: Global warming, TA: terrestrial acidification, FEu: freshwater eutrophication, FEc: freshwater ecotoxicity, HCT: human carcinogenic toxicity, LU: land use, FRS: fossil resource scarcity.

Similarly, the study published by Thomassen et al. (2021) [4] reported that PET and the transport of bottled water to the retailer contributed around 27% and 25% to the global warming category, respectively. In the study conducted by Garcia-Suarez et al. (2019) [17], bottle production was also the process with the greatest contribution to the environmental burdens of bottled water, with values of 89% for freshwater eutrophication, 80% for fossil resource depletion, 77% for terrestrial acidification potential, and 73% for global warming.

3.4. Comparison of the Environmental Impacts of Water Sources

A comparison of the environmental impacts of the three studied water supply sources reveals large differences. For example, in the global warming category, the potential impact of bottled water is around 3300 times greater than that of well water. This difference is smaller between well water and drinking water, with the latter having a potential impact 2.2 times greater. The behavior of the other categories follows a similar trend, as can be seen in Table 3. In the case of drinking water and well water, electricity is almost the only significant factor from an environmental perspective. For bottled water, a series of additional bottling, packaging and transport operations are required, resulting in comparatively major impacts.

In the study conducted by Thomassen et al. (2021) [4], tap water had the lowest environmental load in the climate change category, with 0.17 $kgCO_2$ eq, followed by well water with 0.9 $kgCO_2$ eq, and bottled water with 259 $kgCO_2$ eq. Our results are similar to those reported by Fantin et al. (2014) [23], who based on an extensive review of several LCA studies, reported average values of 0.9 kg CO_2 eq per m^3 for tap water, and 162.4 kg CO_2 eq per m^3 for bottled water. The observed differences between the studies are most likely related to the scope of the study and the processes included in the assessment, as not all LCA studies include aspects such as the materials used for infrastructure, its decommissioning after the end of lifetime, or the management of post-consumption waste.

Table 3. Comparison of environmental impacts of the three water supply sources.

Impact Category	Unit	Well Water		Tap Water		Bottled Water	
		Average	SD	Average	SD	Average	SD
GW	kg CO_2 eq	0.05	0.01	0.11	0.03	165.99	5.81
TA	kg SO_2 eq	2.4×10^{-4}	1.0×10^{-4}	5.6×10^{-4}	1.3×10^{-4}	0.56	0.02
FEu	kg P eq	3.3×10^{-5}	0.00	7.3×10^{-5}	3.19×10^{-5}	0.04	0.00
FEc	kg 1,4-DCB	1.0×10^{-3}	1.2×10^{-3}	2.5×10^{-3}	2.1×10^{-3}	5.88	0.29
HCT	kg 1,4-DCB	2.3×10^{-3}	7.8×10^{-3}	7.1×10^{-3}	0.02	4.99	0.65
LU	m^2a crop eq	2.9×10^{-4}	1.04×10^{-4}	6.9×10^{-3}	2.1×10^{-4}	8.18	0.41
FRS	kg oil eq	0.01	3.9×10^{-3}	0.03	8.1×10^{-3}	67.75	2.46

SD: standard deviation, estimated from the coefficient of variation; GW: global warming; TA: terrestrial acidification; FEu: freshwater eutrophication; FEc: freshwater ecotoxicity; HCT: human carcinogenic toxicity; LU: land use; FRS: fossil resource scarcity.

3.5. Consumption Profiles: End-Uses of Water in Households

Figure 5 shows the percentage contributions of the different end-uses in the two household profiles. Around 25% of the survey respondents declared to possess a well within their households. It can be seen that shower and toilet use together account for most of the total consumption, with 59.7% and 66.5% of total water use in homes with and without well access, respectively, which is similar to results in previous studies [21,24]. The greatest difference between the two profiles is associated with the contribution of water use for outdoor activities, namely pools and irrigation. These activities are associated with a significant water consumption and represent an additional monetary cost for users, which suggests that access to well water (whose operating costs are lower) favors an increased water consumption associated with outdoor uses.

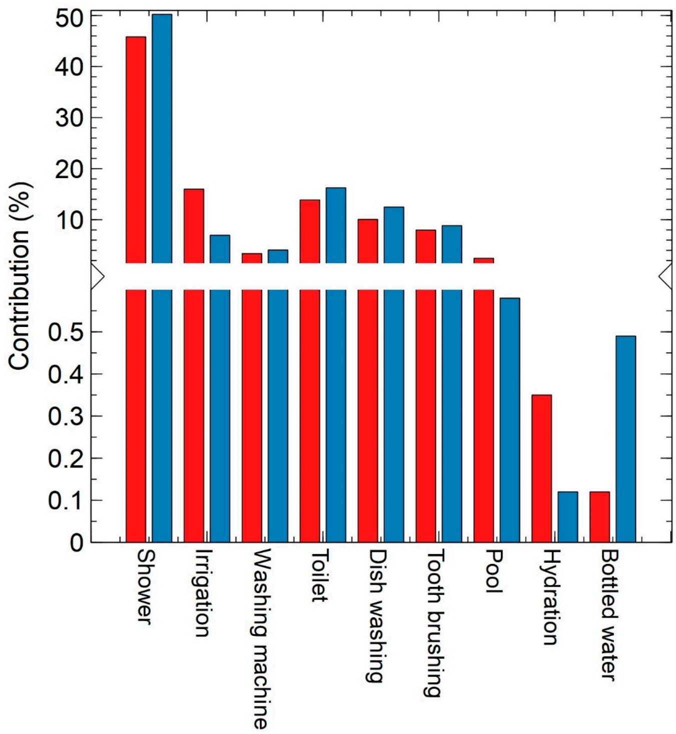

Figure 5. Comparison of the percentage contribution of each water end-use in households with well (■) and without well (■).

A comparison of the results found in this study with those obtained by Castillo-Ávalos et al. (2013) [21] reveals differences mainly in the use of water for tooth brushing, with these authors finding a contribution of 14.9% compared to the 6.0–6.9% of this study. According to Castillo-Ávalos et al. (2013) [21], turning off the faucet during brushing generates a yearly difference of 13.1 m^3 per capita. In our study, an estimation of water consumption for this use resulted in an average consumption value of 4.9 m^3, below the range reported by Castillo-Ávalos et al. (2013) [21] (6.6–19.7 m^3). Another important difference observed with respect to the literature is related to the consumption associated with washing machine use. In a study conducted in USA by the Water Research Foundation (2016) [24], the water consumption of laundry washing reached a value of 3504 gallons per capita-year (13.3 m^3), accounting for 17% of the total consumption. In our study this end use represents only 3.4–4.1% of total consumption, which may be due to the type of household appliances (including washing machine size and water efficiency), cultural practices (such as washing frequency and load), or the estimation methods applied in both studies.

3.6. Comparison of Consumption Profiles and Their Environmental Impacts

Table 4 shows that in both studied profiles, volumetric consumption of tap water accounts for most of the total consumption, with 99.5% in households without wells and 68.7% in households with wells. This may be associated with the accessibility, low monetary value, and high quality of tap water in Chile, which allows its use in any type of domestic application. The differences between the two profiles are associated with well water consumption, which can reach 31.5%. In terms of economic costs, the overall spending of money in water supply is mainly determined by the higher costs of bottled water compared to the alternatives (Table 4).

Table 4. Contribution of each water source to total water consumption in households with and without wells.

Water Source	Homes without Well			Homes with Well		
	m^3 inhab^{-1} year^{-1}	Percentage Contribution	US$ inhab^{-1} year^{-1} *	m^3 inhab^{-1} year	Percentage Contribution	US$ inhab^{-1} year^{-1} *
Tap water	55.5	99.5%	291.8	44.1	68.4%	148.6
Well water	-	-		20.3	31.5%	
Bottled water	0.3	0.5%		0.1	0.1%	
Total	55.8	100%		64.5	100%	

* Estimated cost: tap water: 1.99 US$ per m^3; well water: 0.02 US$ per m^3; bottled water: 604.6 US$ per m^3.

Figure 6 shows the contribution of each water source to the total environmental load of the assessed impact categories. In both profiles the greatest contribution comes from bottled water, which accounts for 38.8–90.3% despite its consumption being less than 1% of the total volume. Tap water accounts for between 9.7 and 61.1% of the total impacts in the profile without a well, and between 43.2 and 91.9% in the profile with a well. In the latter, well water contributed between 6.8 and 46.8% depending in the impact category. These results are similar to those reported by Thomassen et al. (2021) [4], who determined that even though bottled water accounts for only 0.4% of the population's water use in Flanders, it is responsible for 80% and 66% of the contributions to the global warming and resource footprint categories, respectively.

The high contribution of bottled water to the environmental impacts is associated with the factors discussed in Section 3.3. Well water is the source with the smallest contribution due to its operational impacts being only associated with energy consumption. The high contribution of bottled water to the environmental impacts also causes that although the per capita water consumption is 7.2% greater in households with wells, the environmental loads in the two profiles are similar, with differences lower than 2%. It is important to mention that, as this study did not include information regarding the quality of water and the possible presence of potentially toxic compounds in the consumed water, the per capita results of categories such as HCT is most likely underestimated, and the contribution of the different processes could be different depending on the quality of the different water sources. Future studies should account the presence of the most important pollutants in water sources, in order to improve the quality and reliability of the obtained results and to quantify the potential health and environmental trade-offs associated with different consumption patterns [25].

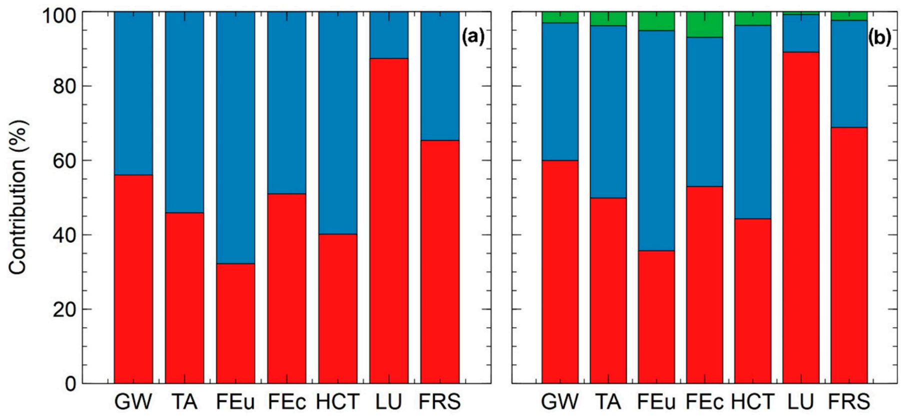

Figure 6. Results of the environmental impact assessment for the consumption profiles. (**a**) households with and (**b**) without wells. Contribution of each source to the environmental load: ■ bottled water, ■ tap water, ■ well water. GW: global warming, TA: terrestrial acidification, FEu: freshwater eutrophication, FEc: freshwater ecotoxicity, HCT: human carcinogenic toxicity, LU: land use, FRS: fossil resource scarcity.

4. Conclusions

From a production standpoint, electricity consumption for potabilization and distribution of tap water results in the greatest contribution to its potential environmental impacts, accounting for 26.7 to 99.0% depending on the production system and impact category. One of the most important factors for electricity consumption during tap water production is the type of source from which the water is taken (surface or groundwater), due to the higher pumping costs of groundwater intake systems. With respect to bottled water, PET bottle production and transportation from plant to the retailer represent the greatest contributions, accounting for up to 53.6% and 39.2% depending on the impact category, respectively. A comparison of the impacts of the production systems showed a large difference in potential impacts, as the results of bottled water are up to 3300 greater than those associated with well water.

From a consumption standpoint, in terms of volume the most important end-uses of water in the assessed households are shower and toilet use, without differences associated with the presence of a well at the respondents' homes. The study of the profiles showed that the presence of wells results in an increase in water consumption for outdoor uses (irrigation and pools), without replacing other water sources or resulting in a decrease in the evaluated per capita environmental impacts. Even though bottled water accounts for <1% of the per capita volumetric consumption, it makes the greatest contribution to the studied impact categories.

Overall, the results demonstrate the importance of including the consumption stage in LCA studies of water supply, as this allows the identification of possible trade-offs that are not identified using process-oriented LCA, improving the scope of the tool as a support for decision-making.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/su15031946/s1, Table S1: List of Ecoinvent processes used in the inventory; Table S2: Average survey validation results by dimension and corresponding free-marginal Kappa values; Table S3: Average survey validation results by category and corresponding free-marginal Kappa values; Table S4: Specific water consumption per appliance or activity; Table S5: Statistical parameters of the drinking water production system inventory. Annex S1: Survey used in the study.

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