

FINANCIAL AND ECONOMIC PERFORMANCE OF RECYCLED CEMENT

Vitor Sousa

CERIS, Department of Civil Engineering, Architecture and Environment, Universidade de Lisboa – IST, Av. Rovisco Pais, 1049-001 Lisbon, Portugal

vitor.sousa@tecnico.ulisboa.pt

Inês Meireles

CERIS, Department of Civil Engineering, University of Aveiro, Campus de Santiago, 3810-193 Aveiro, Portugal

imeireles@ua.pt

José Alexandre Bogas

CERIS, Department of Civil Engineering, Architecture and Environment, Universidade de Lisboa – IST, Av. Rovisco Pais, 1049-001 Lisbon, Portugal

jose.bogas@tecnico.ulisboa.pt

ABSTRACT

Portland cement is used mostly in concrete, being responsible for the largest fraction of both the financial and economic cost of the latter. Recently, a novel magnetic separation-based technology has successfully extracted the hydrate cement paste from concrete waste, unlocking, in practical terms, the possibility of producing recycled cement by thermal reactivation. The present study assesses the financial and economic performance of the recycled cement produced with the technology. The methodology used comprises a hybrid approach combining the analogy with existing production processes, with emphasis in real lime production plants, with corrections based on the actual energy consumption of the RC production process. A financial production cost of 65€/t is estimated and an economic cost of only 13€/t is forecasted by the sale of the excess carbon credits assuming an emission cap identical to Portland cement. These results confirm the substantial cost reduction, both in financial (due to the reduction in energy consumption) and economic (due to the reduction of greenhouse gases emissions) terms.

Keywords

recycled cement; energy consumption; greenhouse gases emissions; financial assessment; economic assessment

1 INTRODUCTION

Most of the GHG emissions from Portland cement (PC) result from the decomposition of carbonates, which make up about 60% of the raw materials used to produce clinker. This portion of PC-related GHG emissions, known as process emissions, account for 5% to 8% of annual global anthropogenic CO₂ emissions (Boden et al. 2017). The combination of process emissions with energy-related emissions—which range from 60% to 90% of process emissions depending on the clinker production technology (IEA 2016)—makes PC production the third-largest anthropogenic source of CO₂ emissions, exceeded only by fossil fuel use and land-use changes (Andrew 2018, 2019).

Consequently, multiple initiatives aimed at reducing GHG emissions from clinker production are being developed, with the main solutions involving (Barcelo et al. 2014; Carriço et al. 2020a): i) increasing the energy efficiency of the clinker production process; ii) using alternative fuels and/or biomass; iii) replacing clinker with other materials; and iv) capturing and storing carbon. Although these solutions can help mitigate the problem, some address only energy-related emissions, while others require the availability of alternative products with compatible properties and in sufficient quantities. Carbon capture is a solution capable of dealing with all CO₂ emissions from clinker production, regardless of its source, but a destination for the CO₂ captured is required. Furthermore, the associated costs are significant and highly variable depending on the technology and studies. For instance, Garcia and Berghout (2019) estimated values between 11€/t and 146€/t of CO₂, Antzaras et al. (2023) values between 50€/t and 80€/t of CO₂ and Moore et al. (2025) values between 144\$/t and 215\$/t CO₂. This increases the PC production cost by 25€/t to 70€/t clinker (CEMBUREAU 2024), which represents a 25% to 100% increase of the PC production cost without accounting for the cost of the final destination of the captured CO₂ (CEMBUREAU 2018).

Noticeably, the option of recycling cement is absent from this list, despite being the solution that would promote a more circular economy by: i) eliminating the process emissions; and ii) contributing to tackle the construction and demolition waste (CDW) challenge. Since producing recycled cement (RC) is technically viable (Carriço et al. 2020a; Meng et al. 2021), this absence is largely explained by the inexistence of uncontaminated hydrated cement paste (HCP) on typical CDW. PC is primarily used in concrete production, with other uses (e.g., mortars, grouts) being substantially less significant in quantitative terms (WBCSD and CSI 2009). Consequently, RC production is hindered by the need to extract the HCP from CDW.

This bottleneck was recently overcome with the development of a magnetic separation-based method that enables extracting HCP from concrete waste (Bogas et al. 2021; Carriço et al. 2021). The present research effort aims at assessing the financial and economic performance of the RC produced with this novel technology, complementing the demonstration of its technical viability (Carriço et al. 2020b; Bogas et al. 2022) and environmental benefits (Sousa and Bogas 2021; Sousa et al. 2023a,b).

2 MATERIAL AND METHODS

2.1 RECYCLED CEMENT PRODUCTION

The RC production technology analysed herein entails three stages, namely: i) release; ii) separation; and iii) reactivation.

The first stage consists in mechanically crushing and milling concrete waste to promote the release of the HCP from the aggregates. The laboratory implementation was set to produce material with less than

1 mm, which was then sieved to split into four fractions: i) 0.5 to 1 mm; ii) 0.25 to 0.5 mm; iii) 0.15 to 0.25 mm; and iv) less than 0.15 mm. These particle size ranges were defined to optimize the efficiency of magnetic separation, providing a good balance between the amount of HCP that can be separated and the fraction that is too small or too large to go through the magnetic separator. However, different settings may be more suitable in an industrial setup and/or for concrete waste with different characteristics.

The two middle fractions, 0.25 to 0.5 mm and 0.15 to 0.25 mm, were found to be the most adequate for obtaining the HCP. This was due to the combination of particle size adequacy for the magnetic separation and the high degree of purity of the HCP particles present in these fractions. The smallest fraction (less than 0.15 mm) can be regarded to be similar to a recycled concrete waste filler (NRF). The largest fraction (0.5 to 1.0 mm) is identical to typical fine recycled concrete aggregates (NRA), despite the potentially lower HCP content. In laboratory conditions, the fraction of particles over 1 mm was between 5% and 10% (8% in average) and is also identical to NRA.

The separation stage is the core of the novel technology used. Taking advantage of the magnetic differences between the HCP and the aggregates, the separation of the HCP is achieved by resorting to high intensity permanent rare earth magnets. To increase the magnetic separation efficiency, the material needs to be cleaned beforehand. Wet (with water) and dry (with air) cleaning methods were found to achieve equivalent results in terms of the amount and purity of the HCP obtained, but the latter requires significantly less energy (Sousa et al. 2023a,b). Since the particles surface needs to be dry for the magnetic separation to work, the wet cleaning method requires an additional drying operation that was found to be the environmental bottleneck of the initial implementation of the method (Sousa and Bogas 2021).

The HCP extracted through the magnetic separation undergoes an additional milling step to reduce the particle sizes to a range comparable to that of PC. The final stage consists in reactivating the HCP by promoting the de-hydration of the PC compounds. In order to achieve this goal, the material undergoes heat treatment at a temperature of 600–700°C, with an average value of 650°C (Bogas et al. 2019, Real et al. 2020). This range of temperatures is consistent with what was used by other authors to produce RC (Wang et al. 2018; Xu et al. 2022). The thermal treatment causes a reduction of about 20–25% in the HCP due to the release of water from the HCP. This reduction depends on the hydration degree of the PC present on the concrete waste and the purity of the HCP obtained in the separation stage. In laboratory conditions, the HCP obtained had a purity between 75% to 90%.

Table 1 resumes the mass fluxes of the process implemented in laboratory to produce RC. The RC amount corresponds to the HCP mass deduced of the hydration water that is released during the reactivation stage, so for each unit of concrete waste mass processed only 5% is converted into RC. This is the major drawback of the process, since a large amount of material needs to be processed for obtaining RC. However, the technology does not only produce RC, but also High Quality Recycled Aggregates (HQRA), Normal Recycled Aggregates (NRA) and Normal Recycled Filler (NRF) that are all useful construction materials.

Table 1 – Mass fraction of each product obtained with the novel RC production technology

Product	Amount [%]
Release and Separation	
HCP - Hydrated Cement Paste	6
HQRA - High Quality Recycled Aggregates	18
NRA - Normal Recycled Aggregates	48

NRF - Normal Recycled Filler	23
Loss	5
Reactivation	
RC - Recycled cement	5

2.2 METHODOLOGY

The assessment developed herein aims to address the financial and some of the economic costs associated with the production of RC. Regarding the financial analysis, all stages of the RC production cost are included. To be consistent with previous research efforts on the environmental performance of RC using this technology, it is also assumed that the release and separation stage take place in existing CDW plants and then the HCP is transported to a plant for reactivation, so an additional transportation stage is accounted for. Concerning the economic analysis, the social costs of GHG emissions were assessed. EC (2021) recommends the use of a social discount rate of 3% and the indexation of the financial discount rate to the interest rate of the financial support provided by the states (3.45% in Portugal in 2004), which were adopted herein.

The raw material for RC is CDW, so the price of recycled aggregates was used as a reference, assuming it includes also the capital costs of the release stage. The separation stage was modelled by analogy with the use of magnetic separation in the mining industry. An analogy with lime production was adopted for estimating the thermal reactivation stage costs.

In all stages, the energy costs were calculated based on the energy consumption and respective prices depending on the energy source. Figure 1 presents the evolution of the monthly electricity and natural gas prices in Portugal over 2025, with average values of 66.21€/MWh and 36.05€/MWh, respectively. Despite the increasing trend throughout 2025, at a daily scale, natural gas prices remain relatively constant, whereas the electricity prices vary significantly depending on variability of the energy sources mix.

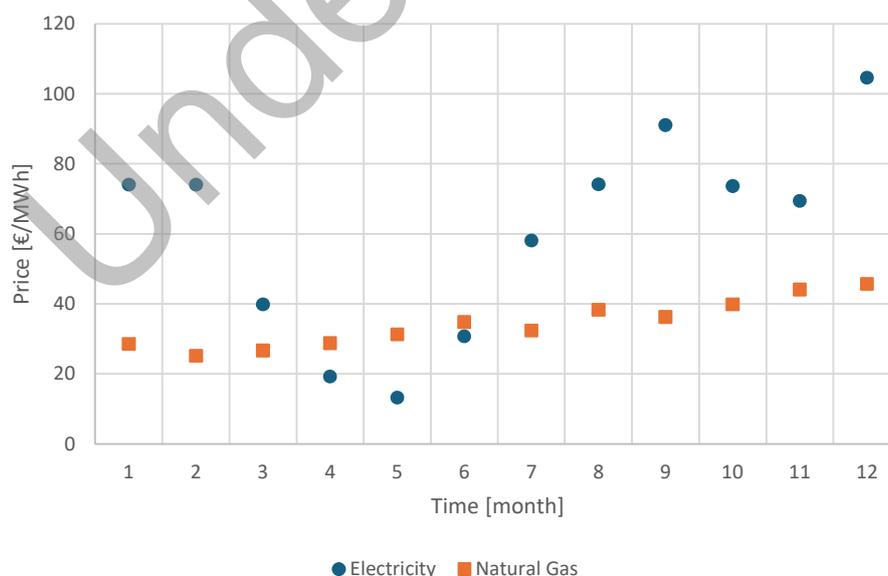


Figure 1 – Electricity and natural gas prices in 2025 (REN 2026)

However, the electricity and natural gas prices to the consumers is not just the energy price and it varies with the type of consumer and power/consumption of the installation. According to DGEG (2026a), the average electricity and natural gas prices for industry in Portugal during the first semester of 2025 (latest available data) were 230.91€/MWh and 71.37€/MWh, respectively, excluding recoverable taxes in the former and all taxes in the latter.

2.2.1 Release stage

Recycled aggregates price in Portugal is only available under request, creating an uncertainty if a research inquiry would receive a market representative quote. However, in Spain, publicly disclosed concrete recycled aggregates selling prices range from 6€/t to 7€/t (Femovert 2025; Novaprosa 2025). According to Linares et al. (2024), in 2021, the production costs of recycled aggregates in Spain was 2.27€/t and the selling price 4.17€/t. This corresponds to a gross sales margin of 45.6%. These values were assumed to be representative of the price of recycled aggregates in the Iberian Peninsula.

2.2.2 Separation stage

Magnetic separation is not used in the aggregate, PC, lime, concrete or related industries. However, it is used in the mining industry (Xie et al. 2022) and some authors are also adjusting the patent technology reported herein to separate other CDW (e.g., brick from concrete - Lipowsky et al. 2025). The permanent rare earth magnets used have very low operation cost, since (Ibrahim et al. 2017; Tripathy et al. 2017a,b): i) the energy consumption is just necessary to rotate the rolls; ii) maintenance is very limited, since it only involves the engine rotating the rolls; and iii) requires basically no labour. Based on the data from Drobe et al. (2021), a 500 t/day magnetic separator has an investment cost of 0.48€/t without accounting for the effect of time. This estimate was done assuming a service life of 15 years and 275 days of production per year. Considering social and financial discount rates of 3% and 3.45%, respectively, the capital cost estimates of the magnetic separation is 0.61€/t and 0.63€/t, respectively.

2.2.3 Reactivation stage

An analogy with lime production was adopted as a baseline for estimating the reactivation costs resorting to data available regarding two recent lime plant projects (Table 2). The Mayur Resources project is a new lime plant in Papua New Guinea that entails the investment on all facilities, not restricted to the quarry and the plant, but also a pier for maritime transportation of the lime and crushed aggregates produced and all complementary land infrastructures required for access and energy supply (<https://api.investi.com.au/api/announcements/mrl/7b3a0f90-68d.pdf>). The Microlime project was a new plant developed in an existing quarry in the region of Fátima, Portugal. The investment required the reactivation of the quarry operations, but the core of the financial burden was limited to the lime plant since the region is well served by land transportation (roads and train) and energy (electricity and natural gas) infrastructures (https://www.microlime.pt/wp-content/uploads/2025/01/ficha_de_projeto2030_microlime.pdf).

Table 2 – Lime plants projects data

Component	Units	Mayur Resources	Microlime
Investment			
Cost	M€	86.70	14.19
Quarry and limestone	M€	11.75	
Lime plant and infrastructure	M€	74.92	
Duration	months	18	18

Operation			
Lime Production			
Daily	t/day	1200	400
Annual	kt/year	400	88
Limestone production			
Annual	kt/year	500	
Costs			
Lime	€/t	47.45	
Limestone	€/t	4.24	
Price			
Lime	€/t	95.24	105.00
Limestone	€/t	10.48	7.73

The unit annual capacity costs are 187€/t (74 920 k€/400 kt) and 161€/t (14 190 k€/88 kt) for the Mayur (just the lime plant and infrastructure) and Microlime projects, respectively. These estimates are aligned with the average value of 165€/t for PC plants in Europe reported by Boyer and Ponsard (2013). As such, the capital cost are comparable using the PC or the lime industry as a baseline.

Disregarding the influence of time and considering a linear depreciation for a project horizon of 30 years, the production cost associated with the Mayur Resources project is between 54€/t (only lime plant and infrastructures) and 55€/t (plant, quarry and infrastructures). These values are in the lower range of the lime production costs in the EU, which Stork et al. (2014) estimates to range from 55 €/t to 70 €/t. According to EC (2018), the lowest production costs are associated with Parallel Flow Regenerative Kilns (PFRK) technology (which is the technology adopted in both projects) and the highest with Long Rotary Kilns (LRK) technology.

The main cost components of lime production are: i) raw material; ii) energy; iii) capital; and iv) other.

The raw material costs partly depend on the distance from the geological occurrence of the raw material. Despite the relative abundance of calcium carbonate in the earth's crust, the existence of deposits which contain sufficient amounts with high chemical purity and the right physical and mechanical properties is rare (Schorcht et al. 2013).

The energy is not only the largest component, but also the most susceptible to variations due to the different kiln technologies, and type and price variability of the fuels. The thermal energy cost in the Microlime plant can be estimated between 76.5€/t and 100.4€/t considering that: i) most PFRK lime plants in Europe use natural gas (Schorcht et al. 2013; Stork et al 2014); ii) the average market price of natural gas in Portugal is 86.06€/MWh, as reported above; and iii) the PRGK technology requires between 3.2 and 4.2 GJ/t (Schorcht et al. 2013; Stork et al 2014). This range of energy costs already exceed the EU average production cost of lime (62.5€/t) reported by EC (2018).

For lime production, capital costs are usually low, around 4€/t. Disregarding the effect of time on money, the capital cost of the Mayur project can be estimated at 7.2€/t, with the quarry, and 6.2€/t, without the quarry, making up 13% and 12% of the production costs, respectively. In the Microlime project, the capital costs represent 5.4€/t. Despite the differences, the estimates are of the same order of magnitude. In these conditions, the equivalent annual capital costs are: i) 11.05€/t (financial) and 11.71€/t (social) for the Mayur project with quarry; ii) 9.56€/t (financial) and 10.12€/t (social) for the Mayur project without quarry; and iii) 8.23€/t (financial) and 8.71€/t (social) for the Microlime project. Since the operation costs of the Mayur project is 47.45€/t, the capital costs are 12-13% of the total production costs. This value is

more aligned with the clinker production, which may be explained by the infrastructure investments required in the Mayur project. The difference between the estimates for real projects and the values reported by the industry may be explained by: i) government incentives aimed at attracting investment and promoting regional economic development and employment; and ii) the time lag between the references and the significant inflation in recent years in the sequence of the COVID19 and the war in Ukraine.

The other category includes components such as labour, maintenance and overhead costs. The proportional distribution of the lime production cost in Europe is presented in Figure 2. A survey conducted by EC (2018) obtained a more detailed distribution (Figure 3), revealing that maintenance and labour (salaries) have relatively similar weights on the total cost, marking energy and raw materials as the most relevant cost items.

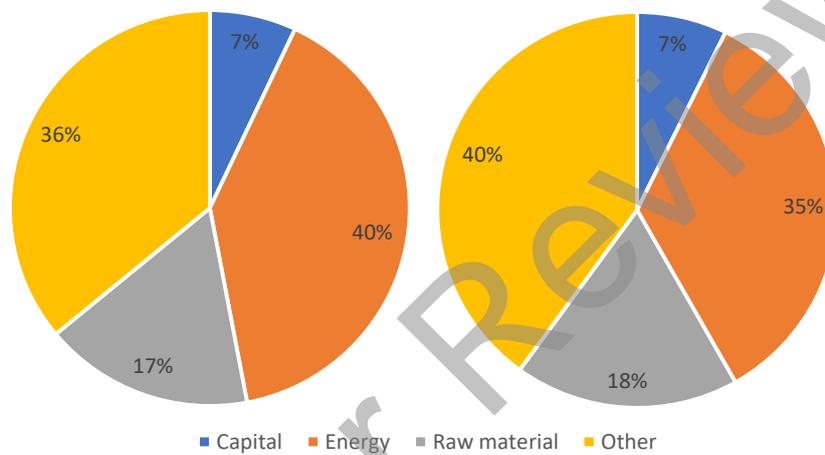


Figure 2 – Average (left) and PFRK (right) total production cost distribution for lime in Europe (Stork et al. 2014; EC 2018)

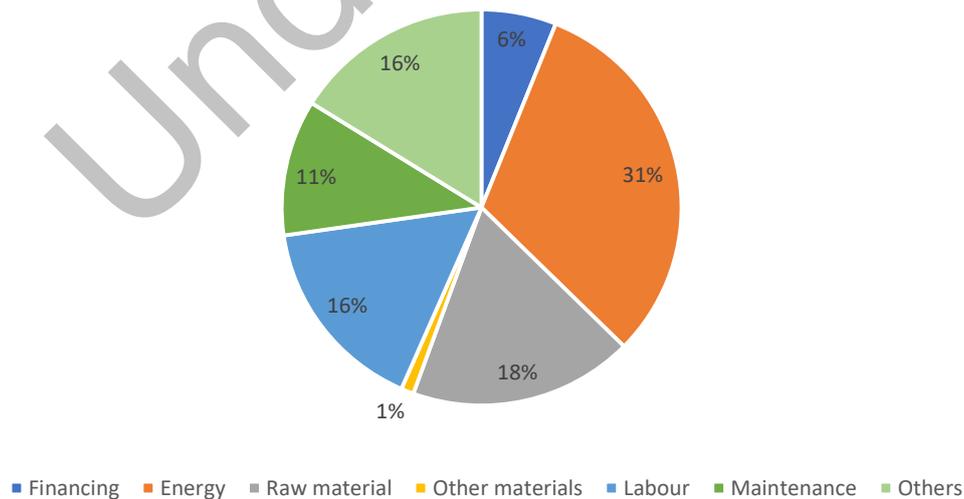


Figure 3 – Detailed total production costs of lime reported by 11 companies in Europe (EC 2018)

2.2.4 Transportation

In 2022, CNR (2022a) estimated the long-haul transportation costs in Portugal to be 1.13€/km for a 40 tonne HGV truck. The study focuses on international transportation and 90% of the fuel (diesel) was purchased in Spain. International transportation companies buy most of the diesel in Spain because it averaged 1.47€/l, without the excise duty discount (CNR 2022b), whereas in Portugal it averaged 1.80€/l (Figure 4). Consequently, national transportation cost in Portugal can be estimated at 0.038€/tkm considering that: i) a 40 tonne HGV truck has an average load capacity of 30 t; ii) the diesel price averaged 1.58€/l in 2025; and iii) the government offers a discount of 0.30€/l up to 35 t of diesel and 0.20€/l onwards for freight road transportation. The Portuguese government offers also a deduction of 120% of the fuel expenses on the deducible profit, but was not accounted for herein.

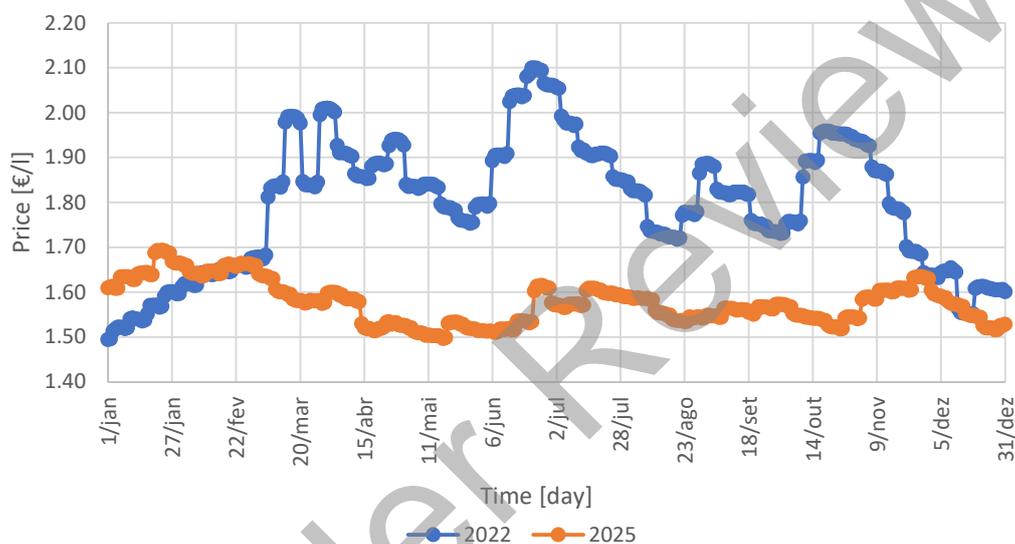


Figure 4 – Daily average diesel price in mainland Portugal (DGEG 2026b)

2.2.5 Social costs

Both PC and lime industries are covered by the European Emission Trading System (EU ETS). However, currently, the GHG emissions are not a cost for many PC and lime producers. The EU ETS allocate free emission allowances for each installation based on GHG emissions benchmarks developed for PC and lime. The benchmark is based on the average GHG emissions from the 10% best performing PC and lime production installations, respectively, in the EU and EEA-EFTA states. As a result, only 1 of the 5 largest PC producers in Europe paid for the carbon emissions (Tamellini and Maury 2025). In Portugal, one of 2 cement producers reported a revenue of over 101 million euros from free emissions allowances and a total revenue of 833 million euros (SECIL 2024). This means that carbon emissions were not a cost, but rather represented more than 12% of the company revenue. However, stricter EU CO₂ taxation policies are expected to change this scenario in the near future, particularly by lowering the allowed emissions cap.

In Portugal, there is also a carbon tax applicable to fuels, including diesel and natural gas, but it is already included in the final retail price of these products.

3 RESULTS AND DISCUSSION

3.1 BASELINE

Since the raw material and production technologies between lime and PC are similar, a cost comparison is done with the latter for validation of the baseline used. According to CEMBUREAU (2021), PC production cost ranges between 46-58€/t for PC with an average clinker to cement factor of 74%. These values are similar to the 2012 estimates of 48€/t and 72€/t (Moya and Boulamanti 2016). The cost estimates exclude the capital costs, only encompassing the fixed and variable operation costs. Moya and Boulamanti (2016) also indicate that the capital costs represent roughly 12% of the total production cost, meaning that the average PC total production cost is 54.5€/t. This value is consistent with Rootzén and Johnsson (2017) total production cost of PC estimate of 58.4€/t, with the capital costs contributing with 5€/t. The cost structure of the PC is presented in Figure 5.

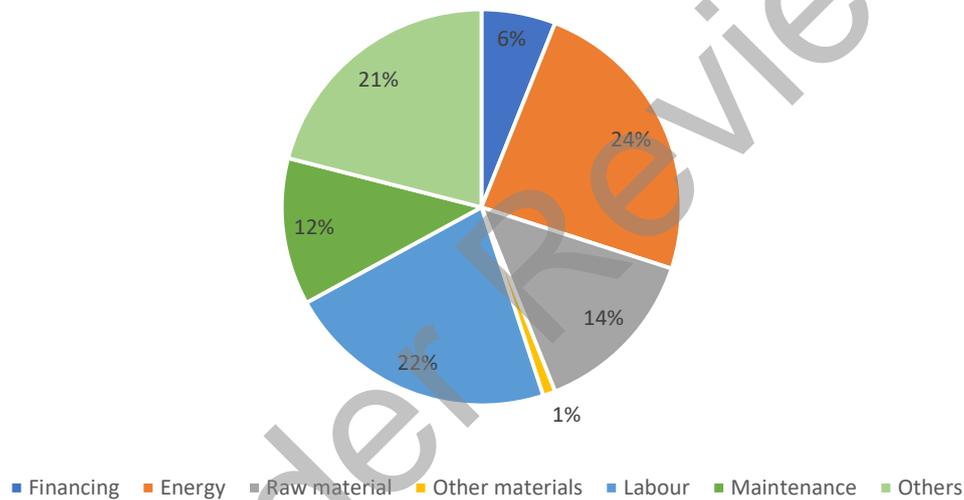


Figure 5 – Detailed total production costs of cement reported by 15 companies in Europe (EC 2018)

The distribution is slightly different from the lime, but that can be explained partly by the fact that it is representative of PC that may have variable clinker content. Also, limestone is between 4 to 15 times cheaper than the additional raw materials required for PC production (Moya and Boulamanti 2016). Disregarding the influence of the cost of the PC additives, the clinker production cost can be estimated to be around 69-85€/t, roughly 20% higher than lime.

Dividing the average lime (60€/t) and clinker (77€/t) total production costs by the respective average manufacture temperature (1050°C, for lime, and 1500°C, for clinker), the values obtained are quite similar ($57 \times 10^{-3} \text{€/t} \cdot \text{°C}$, for lime, and $51 \times 10^{-3} \text{€/t} \cdot \text{°C}$, for clinker). Despite the simplification made with extrapolation from the PC to clinker production cost based on the clinker to cement factor, the comparison indicates that the maximum temperature required in each case explain part of the total production cost differences between lime and clinker.

The comparisons reveals that lime and PC have similar total production costs, despite some differences regarding the cost structure. Considering that the RC is produced with a single raw material, like lime, and

the clinker content associated with PC costs reported is not always disclosed, lime is considered herein to be a more accurate baseline for estimating RC cost.

A relevant aspect of the financial assessment is the size of the of facility. The demand side is not restrictive, since the amount of resources consumed in construction projects largely exceeds the CDW generated. Therefore, the RC production plant size will be restricted by the amount of concrete waste available for processing. Figure 6 presents the evolution of the CDW generated in Portugal over the last decade.

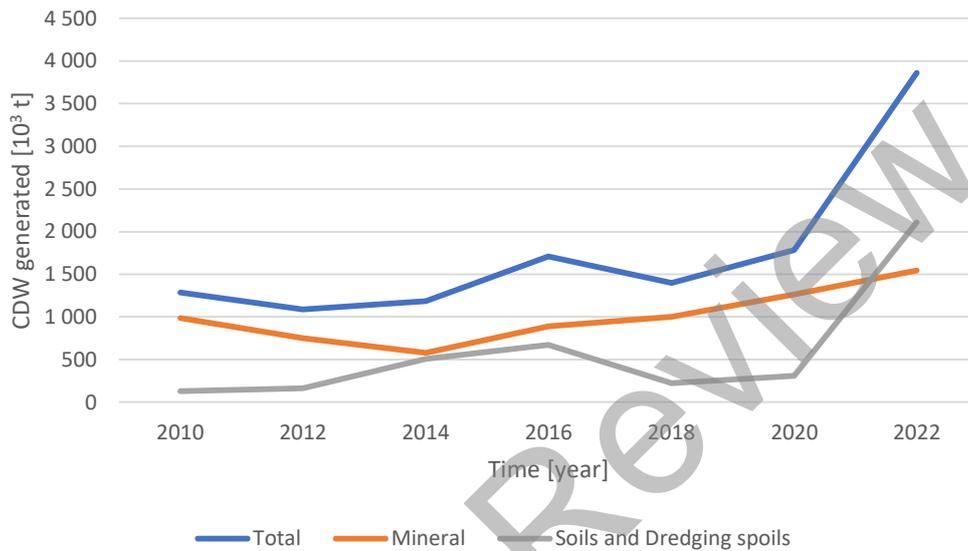


Figure 6 – Evolution of the CDW generated in Portugal (EUROSTAT 2025)

Coelho and Brito (2013) estimated a generation of over 4 Mt/year of CDW, excluding uncontaminated soil and dredging spoils, which exceeds largely the official data. Still, assuming the same proportion in terms of regional (13% generated in metropolitan area of Lisbon and Setúbal) and composition (concrete waste representing 11.6%) distributions, the annual flux of concrete waste ranges from 10 to 27 Kt/year. This dictates a RC plant with capacity between 500 t/year to 1500 t/year, substantially smaller than the reference real lime plant projects used as reference herein, so some scale effects may exist.

3.2 FINANCIAL ANALYSIS

The release and separation stages are expected to take place at the CDW treatment plants because most of the resulting products will be in the form of recycled aggregates and it is possible to make use of the existing equipment for handling and comminution. Therefore, the cost of the release stage is assumed to be the concrete aggregates sale price corrected by the additional energy consumption. The separation stage cost entails the magnetic separation. This approximation disregards the cost of the air compressed system, but the magnetic separation cost reference available in the literature included other operations.

The recycled aggregates cost was estimated considering that the gross sales margin of 45.6% in Spain reported by Linares et al. (2024) in 2021 is still valid and the selling price in 2025 was 6.5€/t based on producer data.

With the exception of the HCP, all other products obtained from the release and separation stages are identical or of better quality than what the existing CDW treatment plants already produce and

commercialize, with the only difference being the granulometry. Consequently, the costs are proportionally distributed by all products based on the amount produced and the stages involved in their production (RC – all stages; HQRA – release and separation; NRA and NRF – release).

Table 3 details the costs for obtaining the HCP, including the transportation to the RC thermal reactivation plant assuming an average distance of 75 km. This implies a journey of 150 km that could be optimized by using the trucks to collect CDW in the return journey. The energy consumption for the release stage was assumed to be only 30% of the total estimate, since part of the costs are already included in the recycled aggregates price. The transportation cost (0.038€/tkm) was corrected by the fact that 1.22 t of HCP needs to be transported to produce 1 t of RC. The results reveal the importance of the location of the RC plant, since transportation between the CDW treatment and the RC thermal reactivation plant makes up 30% of the costs associated with the obtention of the HCP. In fact, the additional processing and the magnetic separation of the concrete waste account for less than 11% of obtaining the HCP.

Table 3 –Costs for obtaining the HCP at the CDW treatment facilities

Resource	Consumption	Units	Cost	Units	Cost	Units
Raw material						
Recycled aggregates	1.22	t/t	3.54	€/t	4.31	€/t
Release						
Electricity	0.48	kWh/t	230.91	€/MWh	0.11	€/t
Fuel	0.56	l/t	1.58	€/l	0.88	€/t
Separation						
Electricity	7.20	kWh/t	230.91	€/MWh	1.66	€/t
Capital	1	-	0.48	€/t	0.48	€/t
Transportation	150	Km	0.046	€/tkm	6.95	€/t
Total					14.40	€/t

Considering that scale effects tend to exist, the capital costs of 10€/t, without state support, and of 6€/t, with state incentives, are estimated for the RC plant where the thermal treatment and complementary operations (e.g., packaging) will take place. These estimates are based on the Mayur project, which has the highest capital costs, and on the difference between the Microlime project and the average capital costs in Europe. Since the facility will be smaller than the reference lime plant projects, a capital cost of 8€/t is assumed. This is a very conservative value, considering that the average capital cost of a PRFK lime plant in Europe is just 4€/t and, usually, includes also the investment in the raw material processing machinery (quarry and comminution equipment) that will not be required in this case.

The labour, maintenance, packaging and other operational costs, with the exception of energy, are assumed to be 22€/t, as indicated for PRFK lime plants in EC (2018).

These values were converted into cost per tonne of calcium carbonate to account for the mass loss in the process (44%).

Table 4 –Cost for producing RC through the reactivation of the HCP

Resource	Consumption	Units	Cost	Units	Cost	Units
Thermal	395.22	kWh/t	71.37	€/MWh	28.21	€/t
Electricity	8.90	kWh/t	230.91	€/MWh	2.06	€/t
Capital	1.22	-	4.48	€/t	5.47	€/t
Other	1.22	-	12.32	€/t	15.03	€/t
Total					50.76	€/t

Overall, the total RC production cost amounts to 65.15€/t, with the distribution of Figure 7. The energy makes up more than 55% of the total, with the thermal energy representing over 85% of it. This is partly explained by the fact that Portugal is in the top 5 countries with most expensive natural gas prices in Europe.

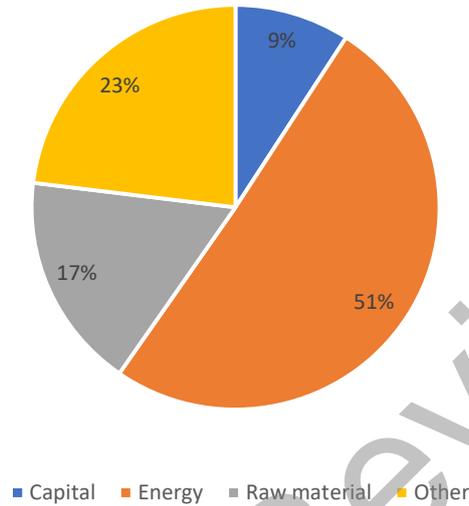


Figure 7 – Distribution of the total RC production costs

3.3 ECONOMIC ANALYSIS

In Portugal, there are two carbon taxes in effect: i) the EU ETS, which has a European Union coverage; and ii) the Portugal carbon tax, which has national coverage. The EU ETS is a cap-and-trade system implemented in the European Union since 2005 that sets decreasing emissions caps in specific sectors (e.g., electricity and heat generation; energy-intensive sectors, such as the PC and lime industries among others; aviation; maritime transport) and creates a market enabling the acquisition of emission allowances. Some industries may receive a share of free emissions allowances that can also be traded (EC 2015), but face a fine (100€/t in 2013) for any excess emissions for which they do not have an allowance (either free or acquired) (EU 2016, 2025a). The Portugal carbon tax was implemented in 2015 as part of a wider package of Green Fiscal Reform and covers all fossils fuels and applies to CO₂ emission mainly from the industry, buildings and transport sectors (EEB 2021). To avoid double taxation, EU ETS operators are exempt from the Portugal carbon tax, as well as certain industrial processes, modes of transport and more vulnerable consumers.

Figure 8 depicts the carbon prices in place in Portugal, being noticeable the substantial increase in recent years. In 2021, the EU ETS passed the 40€/t CO₂eq barrier for the first time. This was due to adjustments done by the European Union, in particular in terms of management of the free allowances (Reid 2024). Since then, the carbon emissions price has averaged at over 60€/t. In 2025, the Portuguese carbon tax increased from the 20-25€/t CO₂eq of the previous years to 67€/t CO₂eq, exceeding the EU ETS for the first time.

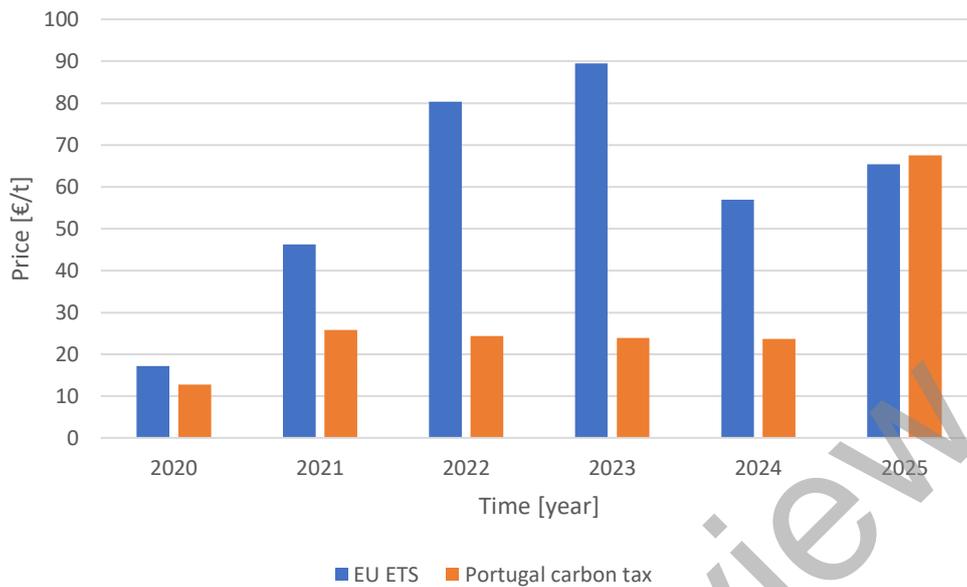


Figure 8 – Average annual GHG emissions price (World Bank - <https://carbonpricingdashboard.worldbank.org/compliance/price>)

RC production only emits 120 kgCO₂eq/t, compared to 615 kgCO₂eq/t reported by SECIL (2024). Assuming that RC production would be attributed the same free allowances that clinker production and adopting a 70€/t average long term price tag for the EU ETS carbon tax, each tonne of RC would allow an extra revenue of 37.5€/t, in comparison to PC. This estimate only accounts for the direct emissions from the thermal treatment to avoid double counting since: i) the electricity prices already include their own EU ETS carbon price; and ii) fuel prices in Portugal already include the Portuguese Carbon Tax. This estimate is comparing RC with an average PC with unknown clinker to cement ratio. According to DAPHabitat (2023), average grey clinker production emits 822 kgCO₂eq/t. In this scenario, the GHG emissions reduction benefits offset a substantial portion of the RC production costs, bringing them down to 13€/t. This estimate can be regarded conservative estimate since: i) free allowances will decrease in the future (EC 2025b); and ii) EU ETS are already over 70€/t and forecasted to increase even further in the future (Reid 2024).

3.4 DISCUSSION

With an estimated value of 65€/t, RC production cost is on the upper end of the range reported for PC. However, it should be noted that the data for PC is not recent and there was a significant increase in the cost components, particularly energy. Before 2022, average natural gas price in Europe was relatively constant at 45€/MWh, but increased to over 80€/MWh in 2022 and has remained at that level since. In the same period, cement price increased roughly 60% (IMPIC 2026) and presently, in Portugal, the price of bagged CEM II/B-L 32.5N from retailers is at 160€/t. With this margin, it is safe to conclude that RC is financially viable.

In order to capture the full economic costs of GHG emissions the social cost of carbon (SCC) should be used instead. SCC has a more ambitious scope of trying to capture the future economic damages of GHG emission presently. It is an economic metric that attempts to adequately reflect the true cost of pollution and climate changes by estimating the present value of the future damages from one additional unit of

carbon emissions in a particular year (Newbold et al. 2011; Gössling and Humpe 2024). This wider scope is a challenge that is hard to tackle given the diversity of areas of impact, interactions and the time frame involved. For instance, Chen-Xu et al. (2024) assessed the health burden and costs of the carbon footprint of the health sector in the European Union using the disability-adjusted life-years (DALYs) as a metric. In Portugal, the health sector emissions amount to 6 298 kt of CO₂eq and is estimated to be responsible for 9 447 DALYs in the baseline scenario. Attributing a value of 70 000€/DALY, with a range of 50 000€/DALY to 110 000€/DALY (de Bruyn et al., 2018), these emissions represent a cost of 661 290 000€. An extrapolation based on the emissions reported by the cement industry in Portugal (3 238 kt, excluding white cement) leads to a SCC of 340 054 785€, which corresponds to roughly 105€/tCO₂eq. EPA (2023) estimate carbon emission cost ranging from 110USD/t to 370USD/t in 2020 depending on the interest rate and modelling approach used. In 2030, emissions costs increase to 140USD to 420USD because of the increased stress in the natural systems. Figure 9 presents the estimated evolution of the social costs of carbon dioxide (in 2021 Canadian dollars – CAD). Other GHG, such as methane and nitrous oxide, have much higher social costs, starting at 1 607CAD/t and 45 053CAD/t in 2020 for 2.5% near-term Ramsey discount rate.

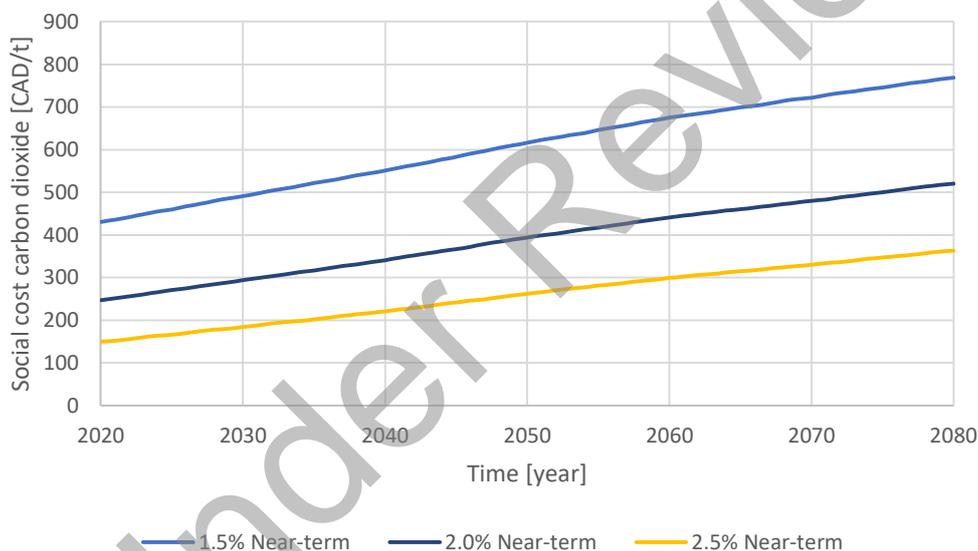


Figure 9 – Evolution of the social cost of carbon dioxide in Canada depending on the discount rate (<https://www.canada.ca/en/environment-climate-change/services/climate-change/science-research-data/social-cost-ghg.html#toc15>)

The variability of the SCC estimates between different studies is even higher, as depicted in the recent and extensive review by Moore et al. (2024). These authors propose a synthetic SCC distribution for 2020 with median of 185USD/t, an average of 283USD/t and an interquartile range of 97USD/t to 369USD/t.

Linares et al. (2024) also estimate a production costs of natural limestone aggregates of 3.40€/t. This implies that natural aggregates production is roughly 50% more expensive than recycled aggregates production (2.27€/t), with part of the difference associated with the energy consumption in the processes. As such, the HQRA obtained in the process are another financially viable product obtained in the RC production process.

4 CONCLUSIONS

The present research explores the financial and economic performance associated with producing RC using a novel magnetic separation-based technology. The results reveal that, compared to PC and lime, the solution is financially viable. With a production cost of 65€/t, the margin to the current sale price of PC in Portugal is of 146%. Including the potential benefits from the substantially lower carbon emissions from RC production, the viability of RC is even more overwhelming. Assuming the same emissions cap of PC, the revenue from selling the emission credits at current carbon prices would almost offset the RC production costs.

The EU ETS clearly underestimates the total cost of GHG emissions, with SCC estimates ranging from at least 130% to over 400% higher. As a result, when the environmental and social benefits associated with GHG emissions reduction of RC are accounted for, its economic benefits are undeniable in comparison to PC. In the absence of carbon emission credits and assuming an average SCC of 200€/t, the price of PC would increase 123€/t, while the increase in RC would be only 16€/t.

RC will not replace PC, in part since the technical characteristics are slightly poorer, but mainly because the concrete waste generation is not enough to meet the cement demand. However, RC has an important role to play in reducing the clinker content of cements in a context where traditional sources (e.g., fly ash and ground granulated blast furnace slag) are decreasing with closure of coal power plants and increase of steel recycling. Furthermore, by using concrete waste as the source for raw material, it contributes towards facing the increasingly stringent waste management targets.

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