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Mechanical properties, life-cycle assessment, and costs of alternative sustainable binders to stabilise recycled aggregates / Tefa, L., Coppola, B., Palmero, P., Bassani, M.. - In: CLEANER MATERIALS. - ISSN 2772-3976. - 15:(2025), pp. 1-16. [10.1016/j.clema.2025.100302]

Availability:
This version is available at: 11583/2998006 since: 2025-03-03T10:15:04Z

Publisher:
Elsevier

Published
DOI:10.1016/j.clema.2025.100302

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30 October 2026

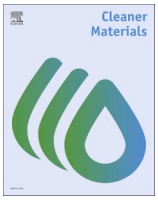
Cleaner Materials 15 (2025) 100302



Contents lists available at ScienceDirect

Cleaner Materials

journal homepage: www.journals.elsevier.com/cleaner-materials



Mechanical properties, life-cycle assessment, and costs of alternative sustainable binders to stabilise recycled aggregates

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ARTICLE INFO

Keywords:
Alternative binders
Recycled aggregates
Road layer stabilisation
Alkali-activated by-products
Life-cycle cost assessment

ABSTRACT

Cement-stabilised subbases provide superior bearing capacity and durability to road pavements compared to unbound aggregate layers. However, stabilisation reduces the environmental benefits derived when recycled aggregates are used. This research compares alternative binders to Portland cement to highlight mechanical, environmental, and economic advantages and disadvantages in a cradle-to-production scenario. Three low-clinker cements with different proportions of pozzolana and three alkali-activated (AA) binders derived from (i) construction and demolition waste fines, (ii) municipal incinerator bottom ash and (iii) waste clay, were compared to Portland limestone cement. The compressive strength of binder pastes was measured after 7 and 28 curing days.

Pozzolanic cements proved viable alternatives to Portland ones, while AA pastes exhibited lower strengths. The crystallinity of alkali-activated silica- and alumina-rich waste precursors was responsible for their limited strength. The life cycle assessment indicated that the replacement of clinker with pozzolana significantly reduces the environmental impact. AA binders with waste precursors can reduce the environmental impact only with a limited quantity of alkaline solution. If the lower strength achieved by AA binders is compensated by adding higher quantities to recycled aggregate, the increase in environmental impact and cost would make them less competitive. The option of using AA binders would be further strengthened with the production of environmentally friendly alkaline solutions and greater local availability of amorphous precursors. At present, cements are cheaper than AA binders due to the current massive production, widespread availability, and competition between producers.

1. Introduction

The subbase of a road pavement contributes to supporting traffic loads and reduces the stresses acting on the subgrade. Ranging in thickness from 20 to 40 cm (Huang, 2003), the subbase is typically composed of compacted unbound granular materials of natural origin. To address sustainability and economic concerns such as the need to conserve non-renewable resources and reduce the amount of inert waste sent to landfill, the use of recycled aggregates in subbase formation is becoming common practice (Zhang et al., 2020; Jiménez, 2013; Abedin Khan et al., 2024). This approach aligns well with the European Commission's green public procurement policy (Shooshtarian et al., 2022; Parliament, 2014; Gálvez-Martos et al., 2018).

However, recycled aggregate typically exhibits lower toughness and durability when compared to natural aggregates (Bassani and Tefa, 2018). It has a greater chance of meeting strength and durability requirements if it is stabilised with cement or other binders (Fanijo et al., 2023; Pham et al., 2023). Cement-stabilised aggregates consist of a mixture of aggregate fractions of different sizes, a small amount of cement (2–6 % of the mass of the aggregate), and water. The latter has a dual role: (i) to ensure sufficient hydration for the binder to react, and (ii) to provide optimum moisture conditions for compaction (Thom,

Abbreviations: AA, Alkali-activated; ANOVA, analysis of variance; AS, Alkaline solution; CDW, Construction and demolition waste; DSCF, Dwass-Steel-Critchlow-Fligner (test); EPD, Environmental Product Declaration; IBA, incinerator bottom ash; IP, Industrial pozzolana; LCA, Life-cycle assessment; LCI, life-cycle inventory; LOI, Loss of ignition; l/s, liquid-to-solid ratio; JCPD, Join Committee Powder Diffraction; PC, Portland cement; RH, relative humidity; WC, Waste clay; XRF, X-ray fluorescence; XRD, X-ray diffraction.

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https://doi.org/10.1016/j.clema.2025.100302
Received 29 July 2024; Received in revised form 17 February 2025; Accepted 17 February 2025
Available online 18 February 2025
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2014). The mix-design of cement-stabilised aggregates relies on achieving the maximum density and compressive strength above a minimum threshold (Xuan et al., 2012). Aggregate properties are the focus of technical standards for cement-stabilised mixes (Committee, 2013; Committee, 2013), while the requirements for cements are limited to compliance with the EN 197–1 standard (Committee, 2011). The Italian technical specification for stabilised mixtures indicates a compressive strength between 2.5 and 4.5 MPa, and a minimum indirect tensile strength of 0.250 MPa after 7 days of curing (Interuniversitario, 2001). Conversely, the Mechanistic-Empirical Pavement Design Guide requires a minimum of 1.7 MPa for compressive strength only (Department of the Army, 1994).

In stabilised granular mixtures, the binder compensates for the heterogeneous properties of recycled aggregates (Chakravarthi and Shankar, 2021). However, the use of Portland cement (PC) raises the impact of the final product to such an extent that the environmental benefits of using recycled aggregates are compromised (Tefa et al., 2022). In this context, fly ash and cement kiln dust have been extensively investigated as alternative binders for stabilised recycled aggregate (Camargo et al., 2013; Arulrajah et al., 2017; Bassani et al., 2016). However, such alternative low-strength binders must be used in larger quantities, which dilutes their economic and environmental advantages. Moreover, some alternative binders are not available locally, and need to be transported across long distances which compromises any of the initial environmental and economic benefits gained (Zhao et al., 2021; Wu et al., 2024). Alkali-activated (AA) binders derived from alumina- and silica-rich precursors, e.g., fly ash and blast furnace slag, have also been used to stabilise recycled aggregates, revealing satisfactory mechanical properties (Mohammadinia et al., 2016; Miranda et al., 2020; Arulrajah et al., 2016; Poltue et al., 2020). The presence of potentially reactive aluminosilicate in the finer particles of construction and demolition waste (CDW) aggregate allowed Bassani *et al.* (Bassani et al., 2019) to successfully stabilise the coarser grains by adding an alkaline solution which achieved stiffness and compressive strengths comparable to those of PC stabilised natural aggregate. Wang *et al.* (Wang et al., 2024) demonstrated that fine powder derived from crushed waste concrete can effectively replace fine silica sand in new geopolymer concrete. The high content of active alkaline compounds in the recycled powder promotes secondary geopolymerisation within the mixture, resulting in improved flexural and compressive strength as compared to unsubstituted concrete. Alternative binders derived from the AA of waste precursors are attracting interest due to their reduced carbon footprint when compared to PC (Robayo-Salazar et al., 2020). Some life-cycle assessment (LCA) studies have shown a reduction in greenhouse gas emissions of between 30 and 80 % in the production of fly ash and slag AA binders compared to PC (van Deventer et al., 2010; McLellan et al., 2011; Danish et al., 2024). For the sake of clarity, the significant variability in these estimates should be noted and they remain the subject of ongoing discussion, primarily due to the impact of the chemical alkaline activator produced. Alkaline solutions (AS) composed of sodium silicate (Na_2SiO_3) and sodium hydroxide (NaOH) are essential for initiating the AA process but derive from energy-intensive processes, which can impact negatively the LCA and cost balance.

The innovation of many binders derived from alternative and recycled sources presents some obstacles when considering their use for stabilising recycled aggregates. The construction sector often faces uncertainties regarding their mechanical properties, environmental impact, and economic viability (Luciano et al., 2020; Cherian et al., 2022). This study addresses a gap in current research by providing a comprehensive evaluation of alternative binders for the stabilisation of recycled aggregate, including the mechanical, environmental, and economic implications of their use. Stabilising road subbases does not necessarily require the best binder, but rather the one that best combines mechanical properties with environmental sustainability and cost-effectiveness (Wang and Baaj, 2021).

Six binders were investigated as alternatives to PC for stabilising recycled aggregates. The basic requirement for the binders was their local availability in the Northwest of Italy, so as not to compromise the environmental benefits achieved with short transport distances. Two categories of binders were under consideration: (i) low-clinker cements, i.e., pozzolanic cements with different percentages of industrial pozzolana – IP – and waste clay – WC, and (ii) AA binders from waste precursors available on the local Italian market, i.e., fine aggregates from undivided CDW with particle sizes less than 0.063 mm, powders from municipal incinerator bottom ash (IBA), and WC derived from the production of expanded clay. A common Portland–limestone cement (CEM-II/B) acted as the reference PC cement. This study specifically addresses two-part alkali activation, which involves the mixing and reaction of a concentrated alkaline aqueous solution with a solid precursor. Fig. 1 provides an outline of the experimental design of this study.

This research introduces several innovations from existing studies. Unlike most research that focuses primarily on the mechanical performance of binders, this study takes a comprehensive approach by simultaneously evaluating their mechanical properties, environmental impact and cost. This multi-dimensional analysis helps in the selection of the most suitable binder for stabilised subbases. Current standards and technical specifications limit selection criteria to the strength characteristics of the mixtures, however a wider understanding of binder performance in terms of environmental and economic impact can help professionals and decision-makers in a more comprehensive evaluation and selection of materials. Moreover, in this study, both cement and alkali-activated binders were selected based on their actual availability in the Northwest of Italy, a region representative of many European areas. It also includes innovative binders with high by-product content (i.e., IP and WP). The context-driven selection process ensures that the study outcomes can be applied to real-world scenarios.

The evaluation of the binders assumed that the higher the compressive strength of the binder paste, the higher the strength and stiffness values of the stabilised recycled aggregate mix containing the same amount of binder. Based on the literature review, the amount of binder required to achieve field performance similar to that of a standard cement-stabilised aggregate mix has been estimated. It is worth noting that binders have been optimised to stabilise recycled aggregates by ensuring the fluidity of fresh pastes to facilitate the mixing stage. This approach differs from other studies where binders were evaluated to achieve their best performance, without necessarily considering their intended application.

2. Materials and methods

Table 1 shows the typical range in the percentage of different binders added to granular mixtures containing recycled aggregates. To stabilise natural or recycled aggregates, cement is added in a range from 2 to 6 % in mass of the dry aggregate (Bassani et al., 2016; Guthrie et al., 2002; Halsted et al., 2006; Gross and Adaska, 2020). AA waste binders need to be added in greater quantities than cement. For instance, Bassani et al. (2019) highlighted the need to include between 11 and 14 % in the mass of AA-CDW fines to achieve mechanical properties which were similar to those of cement-stabilised mixtures. For AA-WC and AA-IBA, no specific information regarding the percentages of the binder added to the CDW aggregate mixture is available in the literature. Therefore, a range of 10–20 % in mass relative to the dry aggregate was considered appropriate, based on previous experimental studies (Arulrajah et al., 2016; Mohammadinia et al., 2018), in which calcium carbide residue, fly ash, slag, and cement kiln dust precursors were used to stabilise road subbase materials. Moreover, these ranges consider that each source of recycled aggregate may exhibit significant variability in their properties thus affecting the required amount of binder.

By restricting our analysis to the mechanical properties, environmental impact, and cost of binders, this study provides preliminary considerations regarding the use of traditional and alternative binders. It guides practitioners toward a more informed selection of sustainable

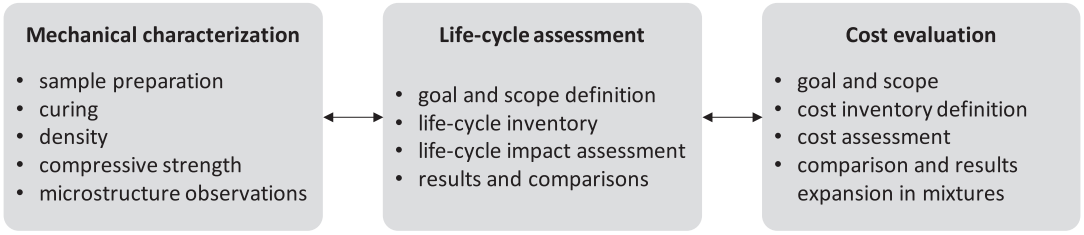


Fig. 1. Flow chart of the experimental study.

Table 1
Typical ranges in percentage terms of binder added to granular mixtures containing recycled aggregates.

Binder	Code	Binder percentage in stabilised CDW aggregates	
		Range (wt. %)	Reference
Portland–limestone cement II/B	C-II/B	2 – 4 (*)	(Guthrie et al., 2002; Halsted et al., 2006; Bassani et al., 2016; Gross and Adaska, 2020)
Pozzolanic cement IV/A	C-IV/A	2 – 4 (*)	
Pozzolanic cement IV/B	C-IV/B	2 – 4 (*)	
Pozzolanic cement IV/H	C-IV/H	3 – 6 (*)	
Alkali-activated CDW fines (liquid/solid ratio = 0.4)	AA-CDW0.4	11 – 14 (**)	(Bassani et al., 2019)
Alkali-activated CDW fines (liquid/solid ratio = 0.5)	AA-CDW0.5	11 – 14 (**)	(Bassani et al., 2019)
Alkali-activated waste clay	AA-WC	10 – 20 (**)	No information. Adapted by (Arulrajah et al., 2016) and (Mohammadinia et al., 2018) (***)
Alkali-activated incinerator bottom ash	AA-IBA	10 – 20 (**)	

Additional notes:
(*) This percentage range indicates the amount of the binder added to the solid phase. The binder includes water that (i) promotes cement hydration and (ii) provides moisture conditions favourable to achieve maximum density after compaction.
(**) This percentage range refers to the addition of AS in the CDW aggregate since the finer fraction (d ≤ 0.063 mm) is already present in the aggregate blends.
(***) No information is available on AA-WC and AA-IBA as binder to stabilise CDW aggregates. The proposed range is retrieved and adapted by (Arulrajah et al., 2016) and (Mohammadinia et al., 2018), where calcium carbide residue, fly ash, slag, and cement kiln dust were employed as precursors.

binders based on the specific context and priorities. In fact, standards and technical specifications usually limit the selection criteria to the strength properties, but environmental and economic impacts can help professionals and decision-makers in a more comprehensive evaluation and selection of materials.

2.1. Reference and low-clinker cements

The experimental reference for the study is a 32.5R CEM-II/B-L Portland-limestone cement conforming to EN 197–1 (Committee, 2011), composed of 65–79 % clinker and 21–35 % limestone. Type-II cement is widely used in Italy, especially in the Northwest region, due to the wide availability of limestone in this territory (Moretti and Caro, 2017). Three alternative low-clinker cements were also investigated: a 32.5 N CEM-IV/A (composed of 65–89 % in mass of clinker, and 11–35 % of IP), a 32.5 N CEM–IV/B (45–64 % clinker, and 36–55 % IP), and a laboratory-produced type IV (denoted as CEM–IV/H) hybrid cement, which combined 50 % of CEM-IV/B with 50 % of waste clay (WC). This type-IV cement is designed to provide low heat development during hydration reactions and for use in high-durability structural works exposed to harsh environmental conditions (i.e., sulphate resistance).

Table 2 shows the chemical composition of cements derived from the X-ray Fluorescence (XRF) analysis. Calcium (CaO) and silicon (SiO_2) oxides were the primary components in CEM-II/B LL in line with the literature (Kristmann, 1977; Khelifi et al., 2017). As expected, moving from CEM-II to CEM-IV types (CEM-IV/A and CEM-IV/B), the quantity of CaO is significantly reduced (from 69.15 % of CEM-II/B-L to 46.65–41.90 % of CEM-IV) and the amount of SiO_2 (27.20 % for CEM-IV/A and 30.00 % for CEM-IV/B) and Al_2O_3 (9.38 % for CEM-IV/A and 9.24 % for CEM-IV/B) is markedly increased. Moreover, both CEM-IV/A and CEM IV/B displayed relatively higher quantities of potassium and iron oxides when compared to CEM-II/B LL, in line with Shagñay *et al.* (Shagñay et al., 2021).

2.2. Precursors of AA binders

Three different waste sources available in the Northwest region of Italy were considered as precursors for AA binder: (i) fine particles from CDW aggregates, (ii) powder bottom ashes derived from the incineration of municipal solid waste, and (iii) WC.

CDW aggregate is obtained through the cleaning, crushing and sieving of waste from the demolition of buildings and civil infrastructures (Oliveira Neto et al., 2017). A 0–8 mm aggregate was collected from the treatment plant and sieved at 0.063 mm to obtain the

Table 2
XRF results for cements and waste precursors of AA binders (LOI = loss of ignition).

Element	Mass (%)						
	Reference binder	Low clinker cements			Waste precursors		
	CEM-II/B	CEM-IV/A	CEM-IV/B	CEM-IV/H	CDW fines	IBA	WC
SiO_2	12.75	27.20	30.00	43.6	52.30	20.83	57.20
Al_2O_3	3.10	9.38	9.24	14.47	16.10	12.63	19.70
CaO	69.15	46.65	41.90	22.8	12.00	37.80	3.79
SO_3	4.99	4.75	4.70	2.67	1.59	5.07	0.63
MgO	1.16	1.63	2.31	3.05	5.53	2.78	3.79
Na_2O	2.59	2.51	2.25	1.82	1.24	2.98	1.38
Fe_2O_3	2.50	4.39	4.89	6.49	6.37	8.23	8.09
Cl	0.33	0.19	0.25	0.167	0.10	2.14	0.09
K_2O	1.53	1.78	2.93	3.46	2.80	1.21	3.98
TiO_2	0.19	0.46	0.43	0.579	0.66	1.16	0.72
LOI	1.16	0.53	0.56	0.30	0.66	0.87	0.05

finer fraction needed for alkali-activation in line with previous studies (Bassani et al., 2019; Tefa et al., 2021). Similarly, fines (d < 0.063 mm) from incinerator bottom ash (IBA) were obtained from the sieving of 0–2 mm sand particles collected at a treatment plant which collects bottom ashes from several municipal solid waste incinerators in Northern Italy. This waste is subjected to a screening process for the removal of iron and metal, before being crushed and sieved to obtain the final sand aggregate. WC was generated in a plant producing lightweight aggregates and expanded clay. Consisting mainly of burnt clay particles (Kaminskas and Savickaite, 2023), WC was used as a supplementary cementitious material in CEM-IV/H (Section 2.1) and as a precursor for AA binders. The WC collected had already been processed into particles smaller than 0.063 mm. XRF characterisation results are detailed in Table 2. The chemical composition of the CDW fines is consistent with previous studies, containing 12.00 %, 16.10 %, 6.37 %, and 5.53 % of CaO, Al_2O_3, Fe_2O_3, and MgO, respectively apart from the SiO_2 content (Bassani et al., 2019; Tefa et al., 2021). The IBA has quite a balanced content of CaO (37.80 %), SiO_2 (20.83 %), and Al_2O_3 (12.63 %), especially in comparison with CDW fines and WC. It also contains relatively large quantities of SO_3 and Fe_2O_3, in line with previous studies (Lam et al., 2011; Chen et al., 2024). Notably, over 75 % of WC consists of SiO_2 (57.20 %) and Al_2O_3 (19.70 %) which is consistent with the composition of the clays used in the production of expanded clay materials and literature (Zhang and Gjørv, 1990; Sharifnia et al., 2016; Arioz et al., 2008). The relatively low value of the loss of ignition (LOI) can reasonably be attributed to the extensive calcination process applied to the clay, ranging between 900 °C and 1250 °C (Arioz et al., 2008).

The phase composition of the different powders was characterised by X-ray diffraction (XRD) analysis in the 2θ range 5-70° (Empyrean Series 3, Malvern Panalytical; operating parameters: 0.007° step size, 22.95 s fixed counting time per step). XRD patterns are collected in Fig. 2, with a focus in the range 10–65° 2θ, provided for clarity. XRD patterns of CEM-II/B and CEM-IV/B, depicted in Fig. 2a, allowed to identify the expected calcium silicate and calcium aluminate phases, which were indexed based on the Join Committee Powder Diffraction (JCPD) database. Specifically, tricalcium silicate (Ca_3SiO_5, JCPD file n. 042-0551), dicalcium silicate (Ca_2SiO_4, JCPD file n. 033-0302), calcium aluminate ($Ca_3Al_2O_6$, JCPD file n. 038-1429) and calcium iron aluminate JCPD fine n. 01-070-1499) phases were identified, along with signals imputable to calcium sulphate hydrate (gypsum, $CaSO_4.2H_20$, JCPD fine n. 01-070-0982). Additionally, a significant amount of calcium carbonate (JCPD file n. 01-085-0849) was detected in CEM-II/B, whereas the quartz phase (JCPD file n. 46-1045) displayed a much higher intensity in the CEM-IV/B, consistent with XRF data.

The XRD pattern of CEM-IV/A is not reported due to its similar composition to that of CEM-IV/B. Fig. 2b depicts the XRD spectrum of the waste clay used in combination with CEM-IV/B to produce CEM-IV/H hybrid cement. The pattern indicates a highly crystalline powder, with quartz as the phase with the predominant intensity. Other aluminosilicate phases, such as albite (JCPD file n. 010-0393) and ringwoodite (JCPD file n. 021-1258) commonly detected in fired clay (Beuntner et al., 2015), were also identified. Ringwoodite (magnesium iron silicate) likely accounts for the higher iron content detected by XRF in CEM-IV/H compared to the other compositions. CDW fines (Fig. 2c) are

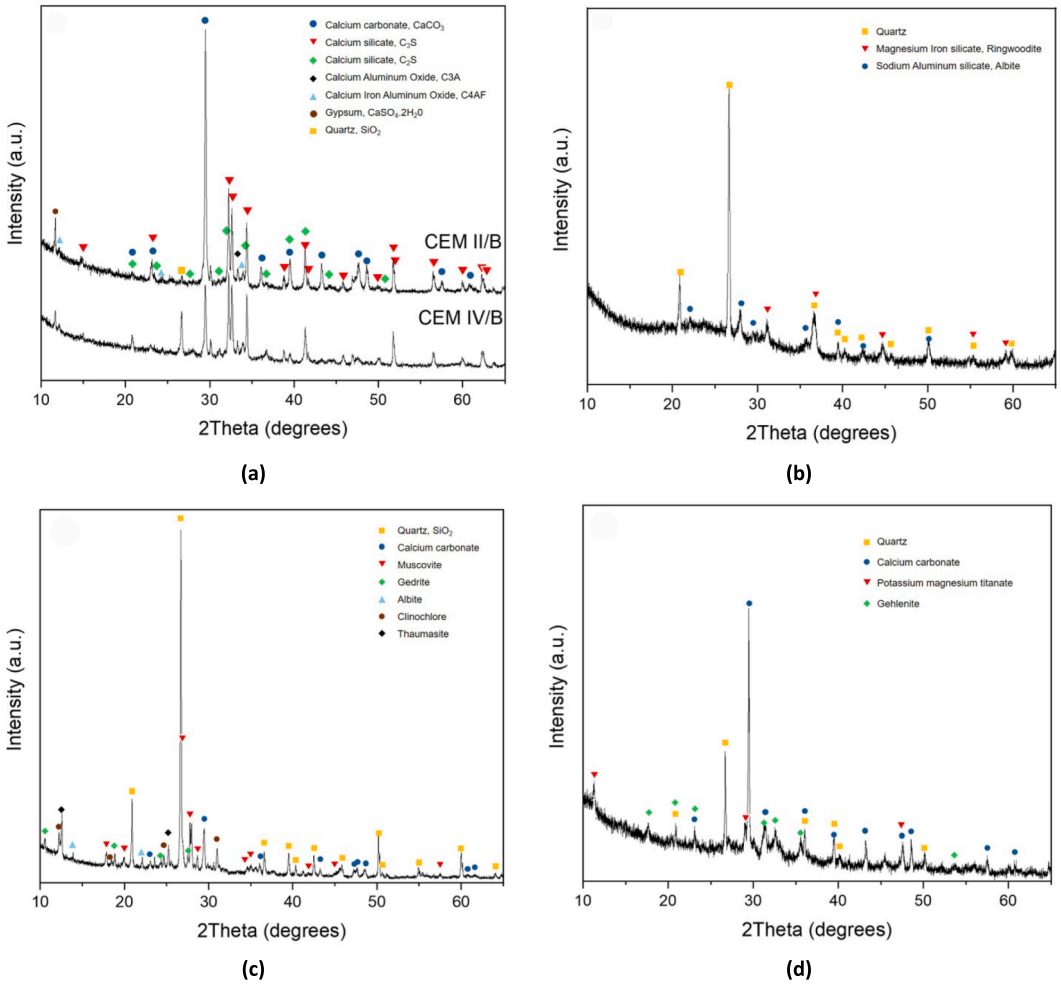


Fig. 2. XRD patterns of (a) CEM-II/B and CEM-IV/B, (b) waste clay, (c) CDW fines, and (d) IBA powders.

composed of calcium carbonate and quartz, with phases from silicate and sulphate groups, including albite, gedrite (JCPD fine n. 013-0506), muscovite (JCPD fine n. 96-900-5472), clinochlore (JCPD fine n. 029-0853) and thaumasite (JCPD fine n. 046-1360) also detected. In the IBA precursor (Fig. 2d), calcite, quartz and a calcium aluminium silicate phase (gehlenite, JCPD file n. 035-0755) were identified, along with the potassium magnesium titanate phase (JCPD file n. 96-100-1130) consistently with XRF results.

2.3. Preparation of specimens and tests

The cements were hydrated with water, while a chemical AS was used to trigger the AA reactions of waste precursors. The AS consists of sodium silicate (Na_2SiO_3) with a SiO_2/Na_2O mass ratio of 1.65 (i.e., $SiO_2 = 22.2$ %, $Na_2O = 13.4$ %, $H_2O = 64.4$ %) and a density at 20 °C of 1.43 g/ml (i.e., 44 °Bé). All the pastes were mixed at room temperature (around 20 °C). The liquid-to-solid ratio (l/s) was selected based on practical considerations to ensure the suitability of each binder for stabilising recycled aggregates. When the aggregate is mixed and compacted with the binder, the letter coats the particles and provides bonded contact points between them. Therefore, each binder was tested at a specific l/s ratio to produce a fresh paste with sufficient fluidity to be easily poured into moulds. This level of fluidity was representative of the consistency required for the binding phase to interact effectively with the recycled aggregates during mixing and compaction, ensuring an optimum bonding and stabilisation effect. This approach differs from previous research where binders were evaluated for optimum performance without necessarily considering their intended use. To achieve the consistency needed in the fresh paste avoiding bleeding during curing, a series of fresh pastes were prepared with incremental adjustments to the l/s ratio, increasing by 0.1 at each step. Ultimately, l/s values of 0.8 and 0.6 were set for IBA and WC, respectively. In the case of CDW fines, l/s ratios of 0.4 and 0.5 were adopted to assess the effects of slightly different workability values for the fresh mixture. It is worth noting that the stabilised recycled aggregates will require a greater amount of liquid phase. Both water (for cements) and AS (for AA binders) have the dual role of (i) chemical reagent for hydration and alkali-activation reactions, respectively, and (ii) lubricant to let the mixture workable at the compaction stage (Bassani et al., 2019; Tefa et al., 2021).

After mixing, fresh pastes were poured into 20-mm side cubic plastic moulds, in 10 replicates, and left to cure in an environmental chamber (Memmert HCP50) at 20 °C and relative humidity (RH) greater than 90 %. After 24 h, the specimens were demoulded and returned to the temperature and humidity-controlled chamber (temperature 20 °C and RH > 90 %) to continue curing for a further 6 and 27 days. The mechanical properties of the hardened binders were evaluated in unconfined compressive configurations with a 50-kN testing machine, by applying a constant strain rate of 0.5 mm/min after 7 and 28 days of curing. In the context of pavement construction, achieving sufficient early strength for immediate use is critical, as roads are opened to construction vehicle traffic within 3 to 7 days. Curing beyond 28 days is rarely feasible in practice due to time constraints. Considering that nearly 80 % of mechanical strength is achieved after 28 days of curing (compared to 100 % after 90 days) (Ibrahim et al., 2020; Najimi et al., 2018), the curing times in this experiment were limited to 7 and 28 days. To highlight any statistically significant differences between the results, the Kruskal–Wallis one-way analysis of variance (ANOVA) coupled with the Dwass-Steel-Critchlow-Fligner (DSCF) test for multiple comparisons was carried out. Table 3 indicates the designation of investigated binders and the l/s ratio adopted, as well as for cements.

2.4. Environmental impact assessment

A comparison of the environmental performance of the binders was conducted by LCA analysis and was carried out using the ILCD Midpoint+ (version 1.0.9) and the *Cumulative Energy Demand* methods, with the impact on climate change and the total consumption of non-renewable resources employed as indicators. The level of equivalent carbon dioxide emissions, which would contribute to the greenhouse effect, was evaluated when determining the impact on climate change (Jolliet et al., 2018; Levasseur et al., 2016). The consumption of non-renewable resources (in MJ) quantifies the use and subsequent depletion of resources that cannot be replaced or regenerated naturally within a human timescale, including fossil, nuclear, and wood-based fuels (Rubio Rodríguez et al., 2011).

The production of 1 m^3 of binder was chosen as the functional unit for the LCA analysis, including all relevant activities required to obtain the end products (extraction of raw minerals, material processing, transportation). The life cycle inventory (LCI) dataset was established with the inclusion of primary data when available from direct measurements at the production plants, interviews, and Environmental Product Declaration (EPD) documentation (Olsson et al., 2024). In the absence of primary data, the LCI data were gathered from relevant literature (Table 4), with a particular focus on the Italian context (Moretti and Caro, 2017). The LCA was conducted using the *openLCA* software (v 1.10.3) supported by the *Ecoinvent* 3.1 database (Wernet et al., 2016). The production of expanded clay had an environmental impact, with only a small proportion of this being attributed to WC, since the production of 1 Mg of expanded clay results in approximately 25 kg of waste material which can be largely reintroduced into the production cycle. Consequently, WC was not categorised as waste, but rather as a by-product. The LCA model for the production of expanded clay was developed by utilising the EPD documentation of a plant operating in the Northwest of Italy. Regarding the production of CDW fines, the material and energy flows considered in a previous study (Zhang et al., 2020) were referenced, while a negative impact (environmental benefit) was included since there was no recourse to landfilling CDW (Khelifi et al., 2017). A similar approach was followed to model the production process of IBA, taking into account the consumption of energy resources for the end-of-waste process and the environmental recovery given that the waste does not go to a landfill but is recycled into a secondary material (Table 5) (Shagñay et al., 2021). The distances travelled by freight trucks transporting the different materials considered in the LCA analysis (raw materials for cement production, waste from the generation zone to the treatment plant, and the recycled material from the treatment plant to the binder factory) are reported in Table 6. All the facilities were in the Northwest of Italy.

Table 4 summarizes the LCI sources, while the input and output flows

Table 3
Designation and liquid/solid ratios (l/s) of investigated binders.

Solid phase	Liquid phase	l/s ratio	Binder designation
CEM-II/B	H_2O	0.4	C-II/B
CEM-IV/A	H_2O	0.4	C-IV/A
CEM-IV/B	H_2O	0.4	C-IV/B
CEM-IV/H	H_2O	0.4	C-IV/H
WC	AS	0.6	AA-WC
CDW fines	AS	0.4	AA-CDW0.4
CDW fines	AS	0.5	AA-CDW0.5
IBA	AS	0.8	AA-IBA

Table 4
Inventory sources per each raw material.

Raw material	LCI source
CEM-II/B, CEM-IV/A, CEM-IV/B	Direct plant measurements
WC	(EPDItaly, 2018)
CDW fines	(Blengini and Garbarino, 2010; Tefa et al., 2022)
IBA	(Technical Committee of the International EPD System, 2020)
AS (liquid Na_2SiO_3)	(Wernet et al., 2016)

Table 5
Input/output flows for different materials.

Precursors	Input/output flows
CEM-II/B-L	• Clinker, limestone, gypsum, organic admixture, iron sulphur (Committee, 2011) • Electricity, heat, fuels • Transportation of materials • Air emissions
CEM-IV/A	• Clinker, limestone, gypsum, organic admixture, iron sulphur, IP, WC (Committee, 2011) • Electricity, heat, fuels • Transportation of materials • Air emissions
CEM-IV/B	• Clinker, limestone, gypsum, organic admixture, iron sulphur, IP, WC (Committee, 2011) • Electricity, heat, fuels • Transportation of materials • Air emissions
WC	• Raw materials for expanded clay production (clay, dolomite, limestone, fly ash, water) • Expanded clay production: electricity (15.2 kW/h per Mg of the final product), heat (3.4 MJ per Mg of the final product) • Transport of materials • Waste clay generated during the production (25 kg each Mg of expanded clay)
CDW fines	• Transport of CDW • Electricity, heat, fuels for end-of-waste treatment (Tefa et al., 2022) • Avoidance of landfilled CDW (Blengini and Garbarino, 2010)
IBA	• Transport of IBA • Electricity (18.6 kW/h per Mg of treated IBA) fuels (22.5 MJ per Mg of treated IBA) for end-of-waste treatment • Avoidance of landfilled IBA (Wernet et al., 2016)

Table 6
Transportation distances.

Material	Route	Travelled distance
Cements	Limestone quarry plant → cement factory	5 km
	Gypsum quarry plant → cement factory	30 km
	Organic admixture factory → cement factory	50 km
	Iron sulphur production plant → cement factory	150 km
	IP production plant → cement factory	200 km
	WC production plant → cement factory	200 km
AA binders	CDW fines → binder/cement factory	20 km
	IBA-treated powders → binder/cement factory	50 km
	AA factory → binder/cement factory	100 km
	WC production plant → binder/cement factory	200 km
CDW (waste)	Demolition site → CDW recycling plant	20 km
IBA (waste)	Incineration site → IBA recycling plant	50 km

included in the analysis for each material are shown in Table 5. The input/output flows corresponding to the proportion of constituents, resource consumption (fuel, heat, electricity), and air emissions of cement products (CEM-II/B, CEM-IV/A, and CEM–IV/B) are not disclosed for reasons of industrial secrecy. The proportions of constituent materials are those specified in EN 197–1 (Committee, 2011); as reported in Section 2.1.

Ultimately, an estimation of the impact on climate change associated with the production of 1 m^3 of stabilised mixture depends on the binder used and the percentage composition of the same in the mixture. It is worth noting that the LCA outcome for stabilised recycled mixtures was derived by scaling the climate change values of the binder, by considering its composition in percentage terms in the stabilised mixtures. The low impact of recycled aggregates on the climate change indicator justifies this approach (Tefa et al., 2022). Therefore, it is reasonable to attribute any change in the climate change indicator (of the stabilised mixtures) exclusively to the binder. Any impact associated with mixing and laying activities was ignored at this stage.

2.5. Costs evaluation

The production costs of the binders were determined by taking into account (i) the cost of extracting the raw material, (ii) the cost of converting the raw material into a finished product, (iii) the cost of transportation (as indicated by distances in Table 6), and (iv) the cost associated with the final treatment of the waste material. Additionally, a system expansion was included to assess the impact of each binder on the production of stabilised CDW aggregate mixtures. Fig. 3 illustrates the information used to estimate the production costs of the cements and outlines the system boundaries. In the case of C-IV/H (Fig. 3b), the transportation costs for the WC derived from expanded clay production were factored into the overall costs, along with the raw material extraction, transportation, and processing operations necessary to obtain C-IV/H cement.

In the case of AA binders (Fig. 4), the production of stabilised mixtures can be streamlined into a single step directly at the mixing plant. This process allows the precursor powder to be mixed with AS and the CDW aggregate at the same time. For the AA-CDW0.4 and AA-CDW0.5 binders, the advantage is further enhanced by the presence of CDW fines in the aggregate blend. In this case, the cost estimate includes (i) the transport of the CDW from the demolition site to the aggregate production plant, (ii) the treatment of waste to produce coarse and fine aggregates, and (iii) the production and transportation of the AS (Fig. 4a). Referring to Fig. 4b, the production and use of AA-WC in stabilised mixtures involve costs associated with the transport of WC and AS production and transport. Finally, for AA-IBA, an intermediate operation is required to treat IBA from municipal solid waste incineration. Consequently, the cost estimate (Fig. 4c) also includes the cost of transporting the material from the incinerator to the treatment plant and the associated treatment operations.

Consistent with the LCA analysis, the production of 1 m^3 of binder was chosen as the reference unit for the cost evaluation (cradle-to-gate approach). In addition, considering the range of binder percentages present in CDW aggregate mixtures (Table 1), the cost of producing 1 m^3 of stabilised mixture (binder and aggregate particles) using each type of binder was estimated. Cost data for North-west Italy were obtained from direct interviews with manufacturers and from 2023 price lists (Piemonte, 2023).

3. Results and discussion

3.1. Strengths

3.1.1. Cements

Fig. 5 shows the average compressive strength ($\sigma_{c,max}$) results obtained from 7- and 28-day cured specimens. After 7 days of curing, both commercial type-IV cements (i.e., C-IV/A and C-IV/B) showed a slightly lower average strength than the reference cement C-II/B (−11 % and −3% for C-IV/A and C-IV/B, respectively). The DSCF test for pairwise comparisons reveals a significant difference between C-II/B and C-IV/A, $t(19) = 4.24$, p-value < 0.001, but no significant difference between C–II/B and C–IV/B, $t(20) = 0.73$, p-value $= 0.48$. The compressive strength values after 7 days of curing are consistent with previous investigations indicating that the strength of pozzolanic cements is generally lower than that of PC at early stages, albeit they achieve similar strengths after longer curing periods (Ghrici et al., 2006).

After 28 days of curing, the average compressive strengths of C-IV/A and C-IV/B were found to be superior (+7% and + 1 % for C-IV/A and C-IV/B respectively) to that of C-II/B, even if the DSCF test for pairwise comparisons indicates no statistical difference between the averages, $t(18) = -1.98$, p-value $= 0.06$, for C-II/B vs. C-IV/A, and $t(19) = 0.22$, p-value $= 0.83$ for C-II/B vs. C-IV/B). This demonstrates that the clinker content in C-IV/A and C–IV/B was sufficient to (i) activate the

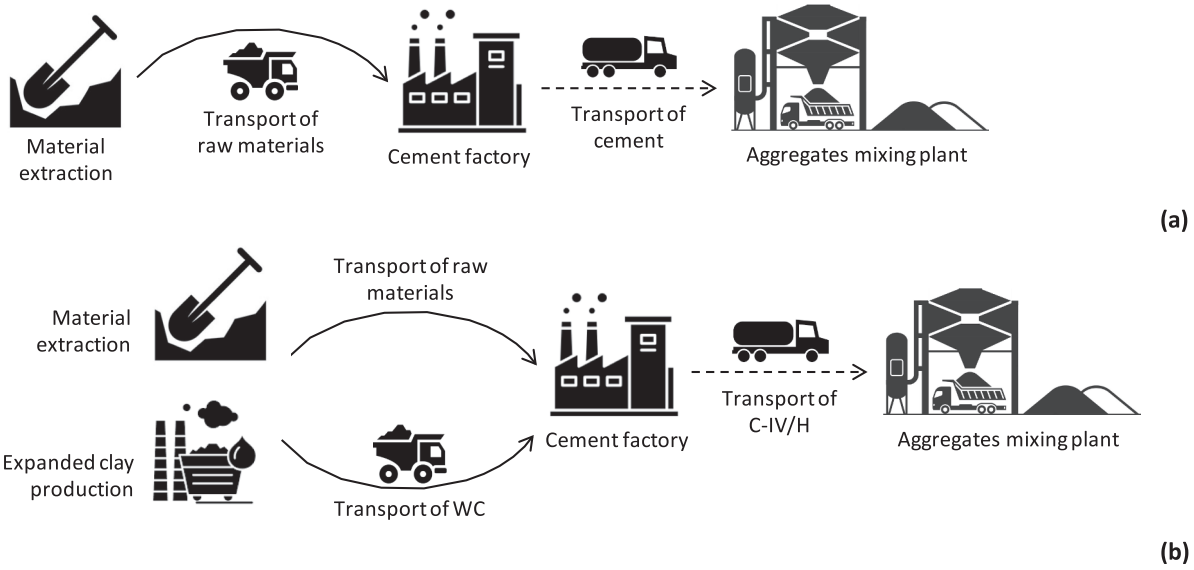


Fig. 3. Diagrams for the cost estimate to produce (a) C-II/B, C-IV/A, C-IV/B, and (b) C–IV/H. A system expansion is also assumed to use those binders for aggregate stabilisation purposes.

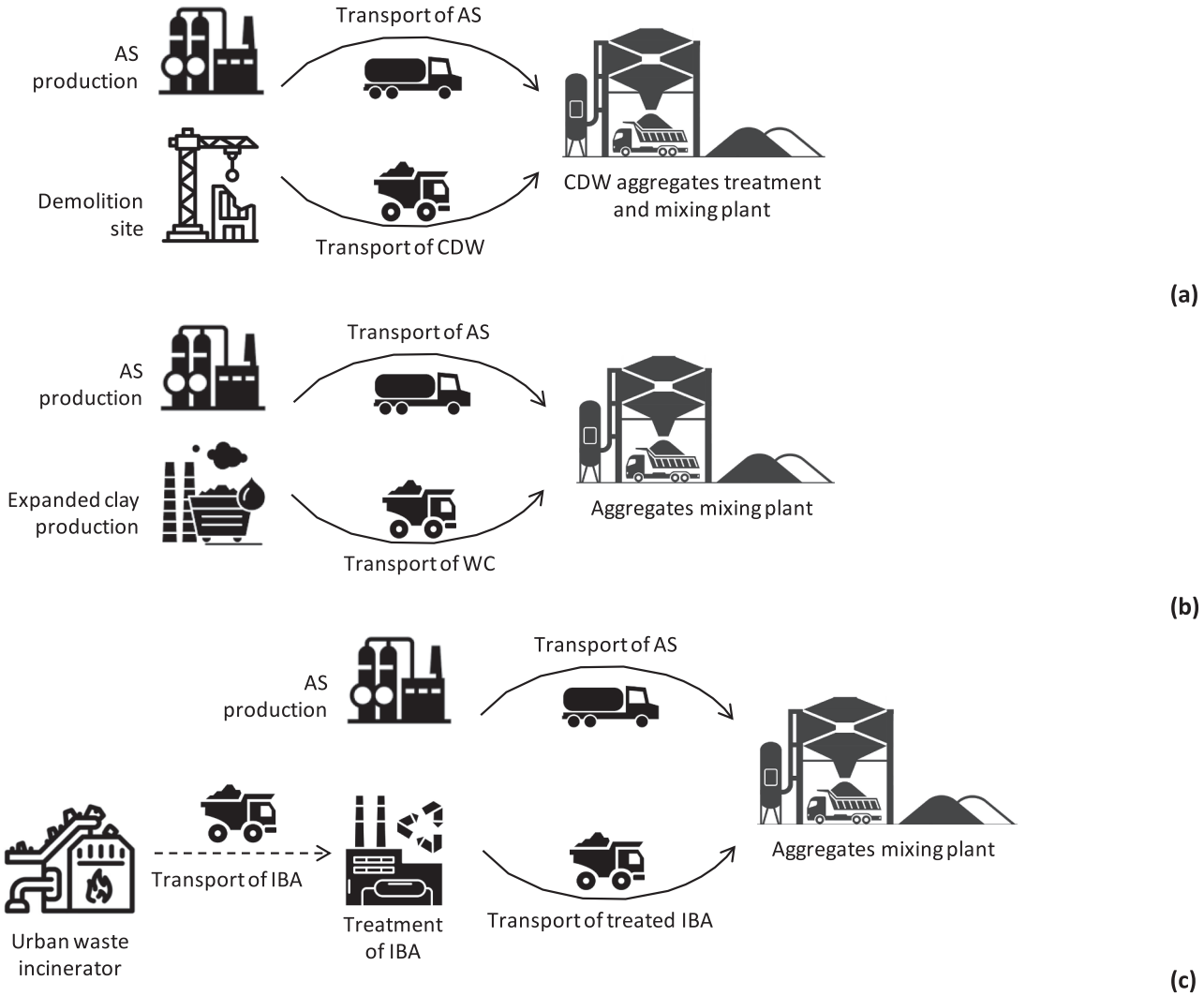
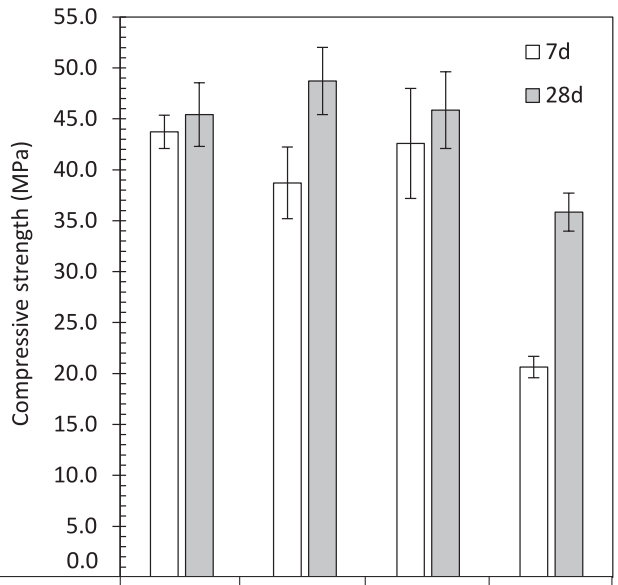


Fig. 4. Diagrams for the cost estimate to produce (a) AA-CDW0.4 and AA-CDW0.5, (b) AA-WC, and (c) AA-IBA. A system expansion is also assumed to use those AA binders for aggregate stabilisation purposes.



Binder	C-II/B	C-IV/A	C-IV/B	C-IV/H
$\sigma_{c,max}$ (MPa) 7 days	43.7	38.7	42.6	20.6
$\sigma_{c,max}$ (MPa) 28 days	45.4	48.7	45.9	35.8

Fig. 5. Average compressive strength results for cementitious binders (error bars indicate one standard deviation).

pozzolanic properties of the supplementary cementitious materials (IP) included and (ii) contribute to the progression of the setting and hardening reactions (Thomas, 2013).

After 7 days, C-IV/H specimens reached an average compressive strength of 20.6 MPa, significantly lower than the reference cement (C-II/B). In this case, the excess WC used in the formation of C-IV/H had an adverse effect on the development of early-stage strength, resulting in a halving of the average $\sigma_{c,max}$ value compared to C-IV/B (42.6 MPa). The clinker content in the C-IV/H binders can be regarded as insufficient to adequately trigger the pozzolanic reactions of the added WC. This finding is in line with the outcomes of Marangu *et al.* (Marangu et al., 2014), who noted a considerable reduction in compressive strength when 50 %wt of waste pozzolana was added to ordinary PC. They attributed this reduction to the limited formation of calcium-hydrated species due to an excessive concentration of inactive minerals and the presence of a substantial volume of pores (Darweesh and Nagieb, 2007; Tironi et al., 2014). However, the average compressive strength after 28 days of curing was 74 % higher (average $\sigma_{c,max}$ = 35.8 MPa) than that obtained after 7 days, thus showing that the addition of WC slowed down the kinetic reactions (Amer and El-Hoseny, 2017). The DSCF test for pairwise comparisons reveals a significant difference between C-II/B and C-IV/H, $t(25) = 44.4$, p-value < 0.001, and between C-IV/B and C-IV/H, $t(25) = 16.0$, p-value < 0.001, thus demonstrating the influence of the cement type on the strength development.

FESEM observations carried out on 28-day cured specimens and reported in Fig. 6a highlight the poorly compacted microstructure of hardened C-IV/H specimens due to the excessive content of IP and WC. However, Fig. 6b illustrates the presence of finely faceted grains, likely attributable to C-S-H gel, together with large hexagonal grains associated with the portlandite phase. These microstructural findings suggest correct clinker hydration, leading to the formation of the initial C-S-H phase. The presence of unconsumed portlandite particles indicates that the pozzolanic reaction has yet to occur. It is noteworthy that this low-clinker cement does not meet the limits set by the EN-197 standard (Committee, 2011) due to its elevated WC content, rendering it unsuitable for structural applications in building construction.

3.1.2. Alkali-activated binders

Binders derived from the AA of waste treatments exhibited significantly lower compressive strengths compared to the cementitious pastes (Fig. 7). AA-CDW0.4 and AA-CDW0.5 pastes cured for 7 days exhibited average $\sigma_{c,max}$ values of 3.0 and 1.7 MPa, respectively (DSCF test for pairwise comparisons, $t(22) = 18.8$, p–value < 0.001. However, after 28 days of curing, the AA-CDW0.5 material reached twice the compressive strength (8.6 MPa) of the same with l/s = 0.4 (4.3 MPa). Increasing the amount of the alkaline solution resulted in a more effective alkaline activation of the CDW fines, and consequently a more significant strength development. The adoption of an even higher l/s ratio for CDW fines, such as 0.6, was ruled out to avoid the bleeding phenomena, which was a factor in previous works (Bassani et al., 2019).

In the case of AA-IBA, after 7 days of curing at room temperature, the strength was similar to that observed for AA-CDW0.4 specimens, $t(24) = 0.13$, p-value > 0.90, thus indicating no statistical differences between the average values, with an average $\sigma_{c,max}$ of 3.0 MPa, in line with the literature (Chen et al., 2016). After 28 days of curing, the strength increased to 6.9 MPa, reaching intermediate values between AA-CDW0.4 and AA-CDW0.5. Moreover, it is important to highlight that the IBA powders required a high l/s ratio (0.8) to achieve the desired flowability. In addition to the increased environmental impact and cost (Sections 3.2 and 3.3), these high quantities of AS can result in several drawbacks, such as the previously mentioned bleeding phenomenon (Gharzouni et al., 2016), the formation of a porous, hardened microstructure, and interference with the polymerisation of Al and Si species, thus resulting in weaker alkali-activation reactions and strength development (Ruiz-Santaquiteria et al., 2012).

The very low mechanical strength of AA-WC can be explained by

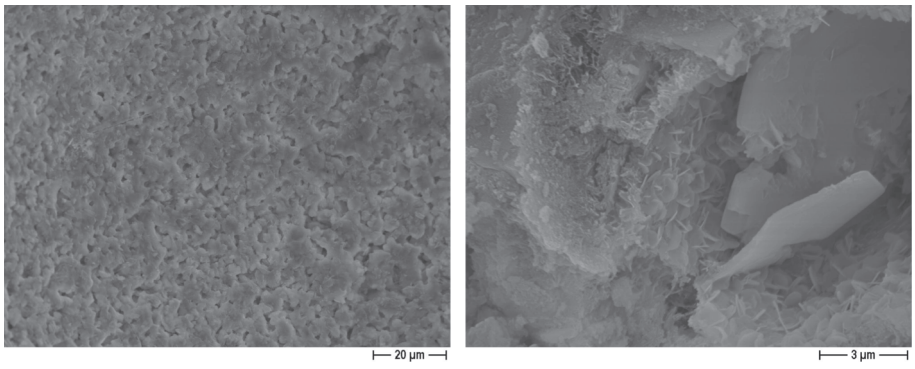
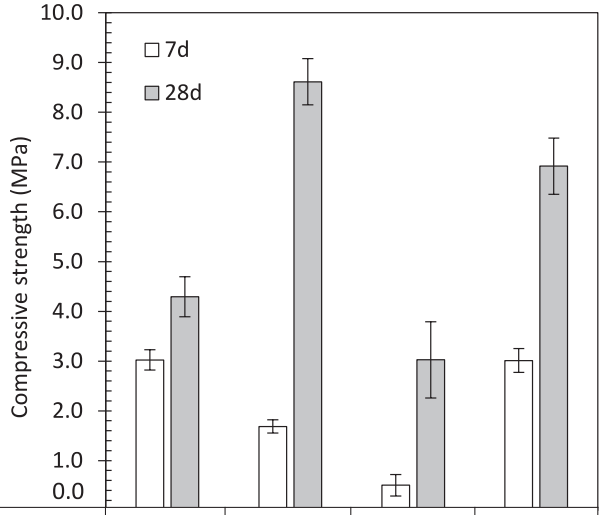


Fig. 6. FESEM micrographs of cured specimens of C-IV/H binder at (a) 1000x and (b) 8000x magnifications.



Binder	AA-CDW0.4	AA-CDW0.5	AA-WC	AA-IBA
$\sigma_{c,max}$ (MPa) 7 days	3.0	1.7	0.5	3.0
$\sigma_{c,max}$ (MPa) 28 days	4.3	8.6	3.0	6.9

Fig. 7. Average compressive strength results for AA binders (error bars indicate one standard deviation).

considering the crystalline nature of the powder (Fig. 2). The significant difference between the mechanical data at 7 and 28 days indicates again slow reaction and hardening kinetics, contrary to the behaviour of traditional geopolymer and alkali-activated materials. While conventional alkali-activated materials derived from highly reactive silico-aluminate powders, such as metakaolin, fly ash and blast furnace slag are characterised by short setting and hardening times (Sun et al., 2023; Ranjbar et al., 2020), the mineral and highly crystalline nature of CDW fines reduce their activity in the presence of the alkaline solution (Palmero et al., 2017). Previous works by the Authors have shown that the reactions can be accelerated by high-temperature curing (i.e., 60 – 80 °C) (Coppola et al., 2020), but this treatment is not compatible with the road applications for which these binders are intended.

Fig. 8 provides a summary of the mechanical performance results for the binders analysed in this study and relates them with the potential quantities of binder required in stabilised recycled aggregates (Table 1). The areas depicted in Fig. 8 are derived from a combination of ranges of compressive strengths (after 28 days of curing) and binder percentages in the stabilised mixtures. Cementitious binders (C-II/B, C-IV/A, C-IV/B, and C-IV/H) exhibit the highest strengths; therefore, they are added in small quantities (2–4 % by weight) relative to the aggregate in stabilised mixes, in line with the literature. Conversely, alkali-activated binders derived from waste materials exhibit significantly reduced mechanical properties and necessitate a higher proportion in the mixture, following the data presented in Table 1. In this latter case, the range in the percentage of the binder added is wider than for cements due to the more uncertain and less guaranteed performance of alkali-activated binders when employed at ordinary temperatures in the field (5 – 30 °C).

3.2. Environmental impact assessment

Fig. 9 illustrates the LCA outcomes associated with the production of 1 m^3 of each binder (dry cement powder and water, or precursors powder and AS). In assessing the climate change impact indicator (Fig. 9a), it is noteworthy that C-II/B and C-IV/A exhibit very similar environmental footprints. Despite variations in the use of supplementary cementitious materials (such as 21–35 % limestone in C-II/B and 11–35 % IP in C-IV/A), their equivalent CO_2 emissions align with literature, underscoring the complex interplay of the constituent materials in determining the overall impact (Fayomi et al., 2019; Sanjuán et al., 2020). Similar results are found when examining the consumption of non-renewable resources (Fig. 9b). Specifically, C-IV/A has a marginally higher impact (+6% with respect to the reference C-II/B), attributable to the longer transport distances required for some materials (Nguyen et al., 2018). In line with the literature, the reduced amount of clinker in C-IV/B compared to C-IV/A is reflected in a 25 % and 22 % decrease in the climate change and non-renewable natural resource consumption indicators, respectively (Salas et al., 2016). A further reduction in clinker content, as seen in C-IV/H, results in a significant decrease in the level of environmental impact observed. In this case, the equivalent CO_2 emissions and non-renewable resource consumption values are nearly halved compared to C-IV/B. Climate change percentage contribution graphs of Fig. 10 demonstrate the decisive role of clinker in the environmental impact of cements. For C-II/B, almost 96 % of the climate change is attributed to clinker, while for C-IV/B, it accounts for over 90

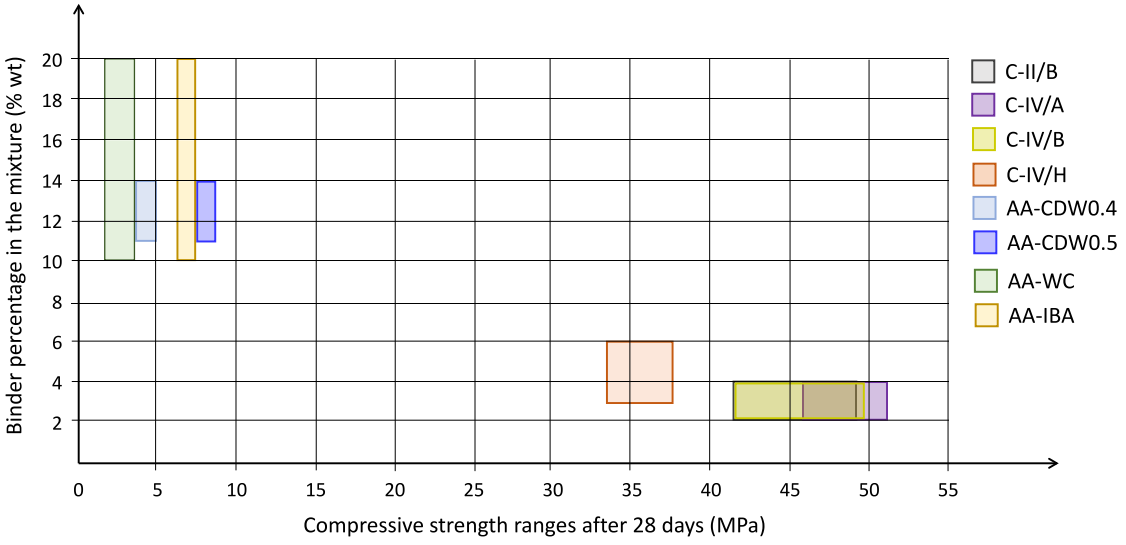


Fig. 8. Comparison of the mechanical strengths of different binders considering the range in the percentage of the same (binders) added to granular mixtures containing recycled aggregates.

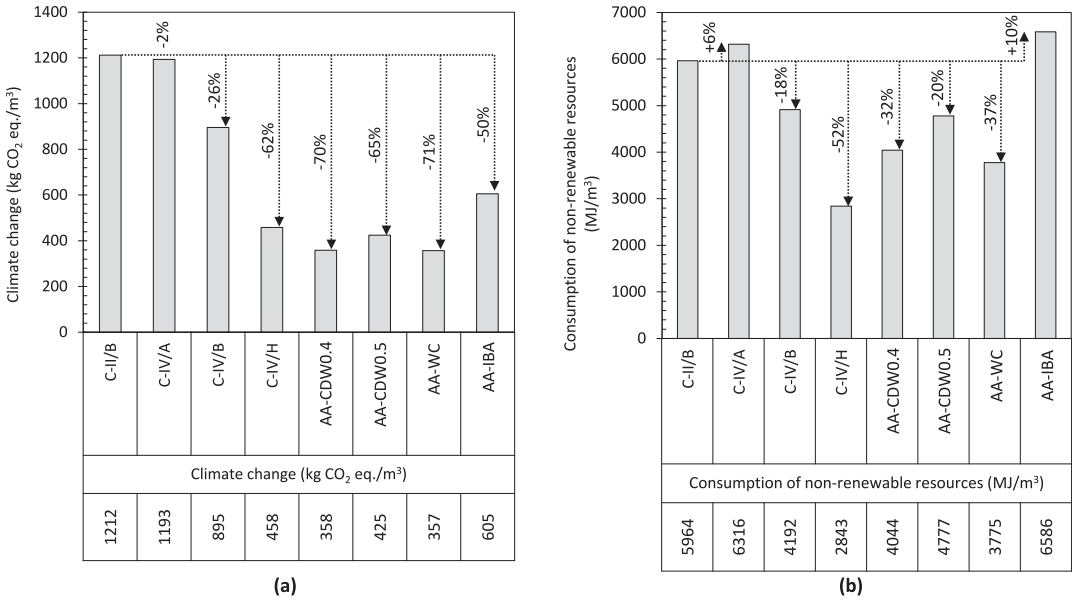


Fig. 9. LCA analysis indicators: (a) climate change and (b) total consumption of non-renewable resources. Percentages indicate the variation of the indicator with respect to the reference binder (C-II/B).

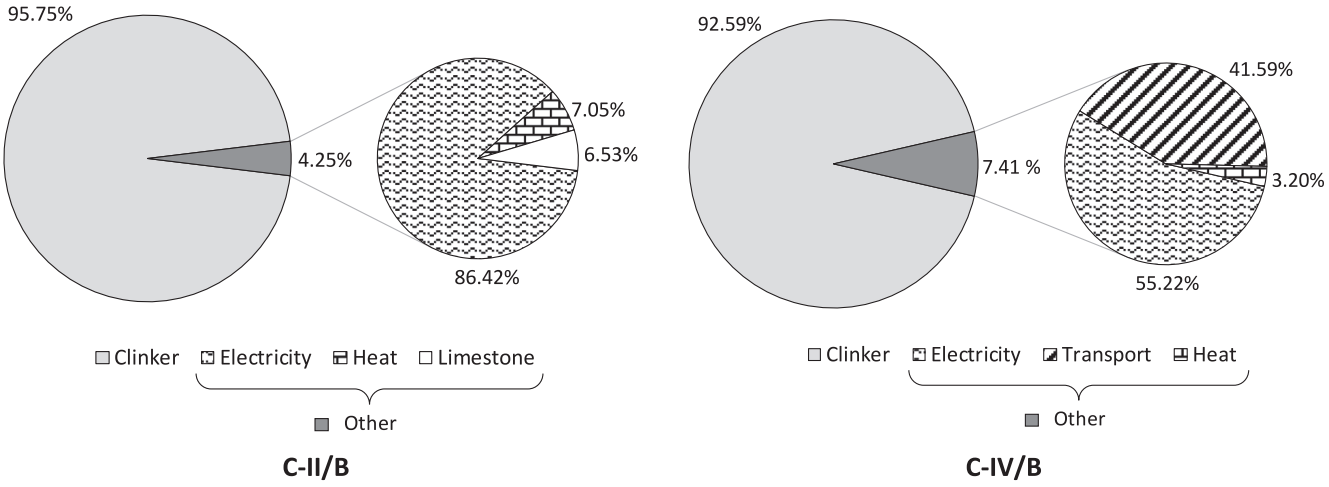


Fig. 10. Percentage contribution to climate change associated with the production of 1 m^3 of C-II/B and C–IV/B binder. Factors with a contribution lower than 0.1 % have been omitted from the charts.

%. Additionally, electricity and the transportation of raw materials contribute 2.9 % and 5.4 % respectively to the overall environmental impact.

The contribution analysis shown in Fig. 11 reveals that over 90 % of the climate change associated with the production of the C-IV/H binder is attributable to CEM-IV/B, with a negligible contribution (0.1 %) from WC, despite both being present in equal mass within the mixture.

A closer look at the binders derived from the AA of waste precursors reveals their lower climate change impact values, which are primarily attributed to the absence of clinker, which is known for its highly energy-consuming and impactful manufacturing process (Huntzinger and Eatmon, 2009). However, the use of AS prevents the environmental impact from being as low as expected for a waste-based material. AA-WC and AA-CDW0.4 binders showed the lowest equivalent CO_2 emissions (357 and 358 kg of eq. CO_2 per m^3, respectively), while the consumption of non-renewable resources indicator for both is almost 30 % higher than C-IV/H. Increasing the l/s from 0.4 to 0.5 in AA CDW fines leads to a 19 and 18 % increase in the climate change and total consumption of non-renewable resources indicators, respectively. The negative effect of the AS on the environmental impact of AA binders is even more evident with AA-IBA. In this case, the high AS content (l/s = 0.8) contributes to an increase in the climate change impact (605 kg of CO_2 eq. per m^3) and results in the binder having the highest consumption of non-renewable resources.

The analysis presented in Fig. 11 highlights also the crucial role AS has in determining the values for both the climate change and consumption of non-renewable resource indicators for AA binders. Indeed, AS production is highly energy intensive and this significantly affects the LCA balance (Colangelo et al., 2020). For all AA binders, transportation contributes relatively little to the climate change indicator, accounting for 3.7–3.9 %. The LCA-based studies of Ouellet-Plamondon & Habert (Ouellet-Plamondon et al., 2015) and Passuello *et al.* (Passuello et al., 2017) confirm that AS have a significant impact on the final environmental assessment of AA materials. Therefore, a reduction in their use is a requisite for sustainable outcomes.

Fig. 12 illustrates the impact on the climate change indicator when different binders are used (in varying quantities) to stabilise different mixtures of recycled aggregate. The range in the percentage values of

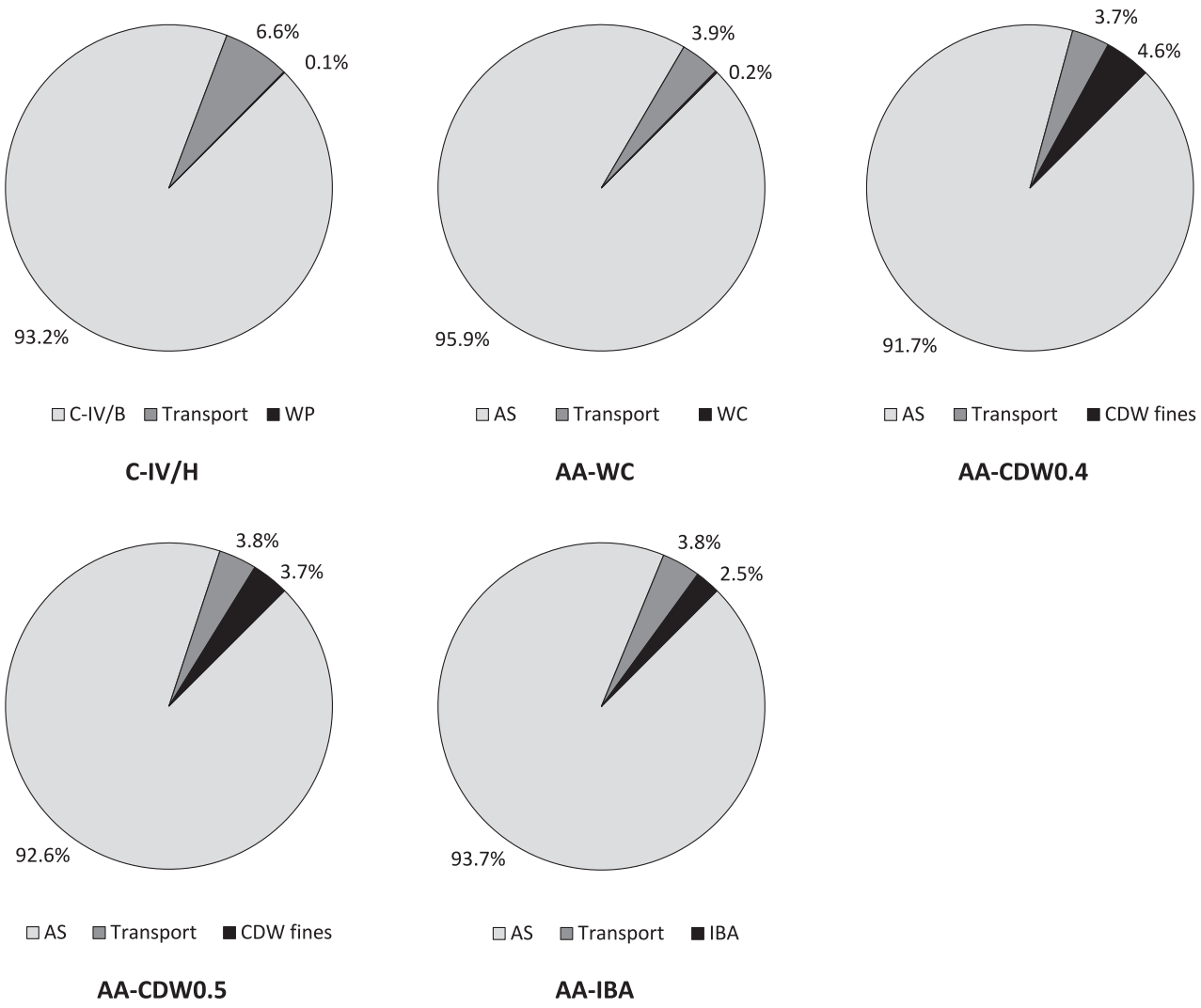


Fig. 11. Percentage contribution to climate change associated with the production of 1 m^3 of C-IV/H, AA-WC, AA-CDW0.4, AA-CDW0.5, AA-IBA binder. Any factors with a contribution lower than 0.1 % have been omitted from the charts.

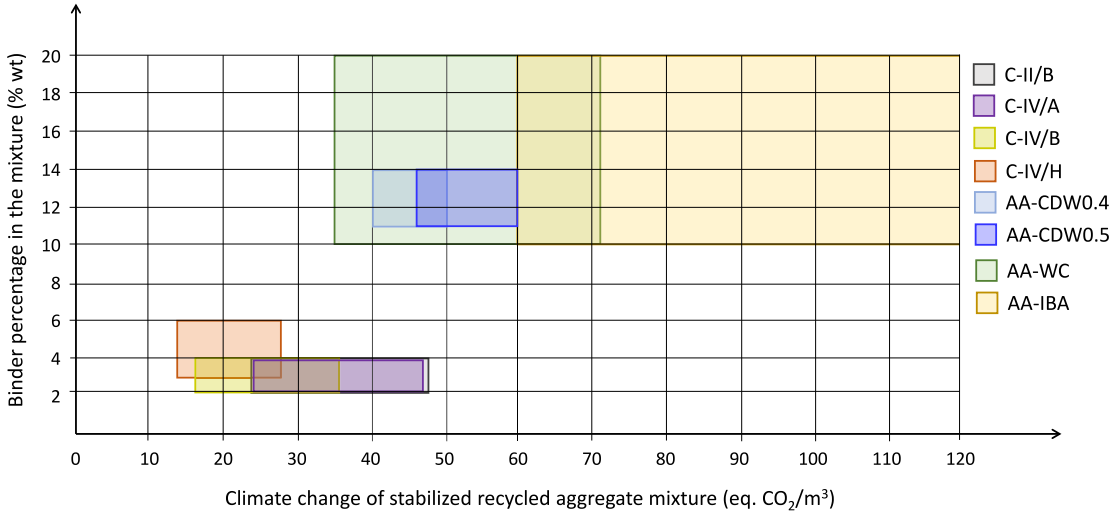


Fig. 12. Climate change of the stabilised CDW mixture depending on the binder type, considering the differences in the percentage of binders added.

binder added comes from literature and is outlined in Table 1. The image identifies the areas of potential climate change impact for recycled aggregate mixtures stabilised with different types of binder.

The climate change impacts of recycled aggregate mixtures stabilised with 2–4 % of C-II/B and C-IV/A are comparable, with values ranging from 24 to 49 and 24 to 48 kg CO_2 eq./m^3 of the stabilised mixture, respectively. A slight reduction in the climate change value is observed when C-IV/B is used instead of C-IV/A for stabilisation purposes. This reduction is due to the lower inherent climate change impact of the C–IV/B cementitious binder. The mixture of recycled aggregate stabilised with 3–6 % of C-IV/H shows the lowest climate change impact even though the addition percentages are higher than for other cementitious binders (2–4 %). In this case, the impact associated with the production of 1 m^3 of stabilised mixtures (aggregate and binder) ranges between 17.9 and 35.8 kg CO_2 eq. Conversely, mixtures containing alkali-activated binders have a greater overall environmental impact than mixtures stabilised with cements even though the binders used have lower climate change values. Mixtures comprising 11–14 % of AA-CDW0.4 or AA-CDW0.5 achieve climate change values of 39–50 and 47–59 kg CO_2 eq./m^3 respectively. Furthermore, the use of AA-WC and AA-IBA may result in even higher impact values. This is because alkali-activated binders need to be added in significantly higher percentages to compensate for their lower mechanical strength. Consequently, although these binders are preferable to cement in strictly environmental terms, the fact that they must be used in greater quantities means they do not ultimately have a lower climate change impact.

3.3. Production costs

The estimated costs associated with the production of each binder are outlined in Table 7. The price of cementitious pastes (C-II/B, C-IV/A, and C-IV/B) are consistent with those in literature ranging between 102 and 112 €/m^3 (Ali et al., 2011; Ndahirwa et al., 2022). The higher costs of C-IV/A (+9%) and C-IV/B (+8%) compared to C-II/B are attributed to the long transport distance (Table 6) of the IP from the expanded clay production plant to the cement factory. In contrast, the limestone present in C-II/B comes from a quarry located very close to the cement factory (5 km), resulting in lower transportation costs (Supino et al., 2016). When C-IV/B and WC are combined in equal parts to form the C-IV/H binder, the cost is reduced to 77.69 €/m^3 (30 % less than the cost of C-IV/B). The WC is a by-product of expanded clay production and incurs transportation costs of 44.63 €/m^3 (Table 6). However, using WC as a precursor to producing AA-WC binder significantly increases the costs (147.57 €/m^3) because of the addition of a considerable quantity of the expensive AS (Abdollahnejad et al., 2015). The production of AS via the hydrothermal process, involving high temperatures and pressures for treating quartz with sodium hydroxide, is highly energy intensive and, consequently, more expensive (Fawer et al., 1999). The high cost of using AS (180.00 €/m^3) has an important impact on the final cost of all AA binders investigated in this study, which is consistent with previous works (Lima et al., 2022; Guo et al., 2023). The estimated costs of producing AA-CDW0.4 and AA-CDW0.5 pastes are 104.03 €/m^3 and 124.48 €/m^3, respectively, and largely attributed to the cost of AS. In fact, the CDW aggregate is the least expensive one (6.20 €/m^3), its main costs being due to waste transportation and treatment. AA-IBA emerges as the most expensive binder, because of (i) the massive quantity of AS (l/s = 0.8) required, and (ii) the relatively high cost of the IBA precursors. The production cost of 53.52 €/m^3 associated with the IBA precursor includes (i) the waste transportation from the incinerator to the treatment plant (50 km according to Table 6), (ii) the operational costs for waste treatment (Wiles, 1996; Bruno et al., 2021), (iii) and the transportation of the IBA precursor to the binder/cement production factory (50 km according to Table 6).

Table 7
Cost of binders (based on 2023 prices).

Binder	Cost (€/m^3)
C-II/B	102.23
C-IV/A	112.55
C-IV/B	110.74
C-IV/H	77.69
AA-CDW0.4	104.03
AA-CDW0.5	124.48
AA-WC	147.57
AA-IBA	224.51

Fig. 13 presents the estimated costs of the different mixtures of recycled aggregate stabilised with each binder investigated. It is worth highlighting that the binder addition percentage (included in Table 1) plays a crucial role in estimating the cost of stabilised mixtures, but this can only be determined through a dedicated mix-design study. Consequently, for this research, these costs were sourced from literature (see Table 1).

Considering (i) the cost of 0–40 mm recycled aggregate at 2.82 €/Mg and (ii) the typical range of 2–4 % (in mass of the dry aggregate) for the additional percentage of ordinary cement (Table 1), the stabilised mixtures would be priced between 9.50 and 13.80 €/m^3. Although C-IV/H binder is less expensive than cements such as C-II/B, C-IV/A, and C-IV/B, its use for recycled aggregate stabilisation would not significantly lower the cost of the stabilised mixture. This is due to its inferior mechanical properties (as discussed in Section 3.1.1), which may require larger quantities of the binder to be added, somewhat negating the economic advantage of using it. If it were possible to use recycled granular materials in a similar percentage to traditional cement, it would be economically advantageous. For instance, a comparison between a mixture of recycled aggregate with 3 % of C-II/B and the same mixture with 3 % of C-IV/H shows a saving of 1.29 €/m^3. This saving can only be achieved through a dedicated mix-design and mechanical performance assessment of the mixture containing recycled granular materials and C-IV/H binder.

The elevated costs associated with AA binders, along with the increased quantities required for stabilisation, result in a doubling or even tripling of the price of stabilised recycled aggregate mixtures, thus rendering these products economically unfeasible. When considering the use of CDW fines as a stabiliser through alkali–activation, 11–14 % in mass (relative to the dry aggregate mass) of AA-CDW0.4 and AA-CDW0.5 binder in the 0–40 mm fraction is necessary to achieve mechanical strength values similar to those attained from traditional cement stabilisation. With the addition of this binder material, the costs of the stabilised mixtures increase to 19.00–22.49 €/m^3 and 22.20–26.57 €/m^3 for the AA-CDW0.4 and AA-CDW0.5 binders, respectively. Based on an analysis of costs, cement-based binders are more economically competitive than hypothetical mixtures containing AA-WC and AA-IBA binders due to the significantly high costs of these binders. This finding is consistent with previous research (McLellan et al., 2011; Ramagiri et al., 2021; Weil et al., 2009). The widespread presence of cement factories throughout the territory ensures that raw materials for cement production do not need to undergo lengthy transportation. Moreover, the multitude of competitors fosters a highly competitive market environment, thus ensuring that pricing remains competitive. Lastly, it is noteworthy that significant funding and extensive research efforts are directed towards optimising production plants to minimize the use of resources, environmental footprints, and, most importantly, costs (Uwasu et al., 2014). In contrast, the limited diffusion of AS production plants results in longer transportation costs for the raw materials needed, thereby increasing the final price (Ponomar et al., 2023). The limited number of plants for producing sodium silicate restricts the competitiveness and availability of the activating liquids for manufacturing AA binders on the market. The stabilisation of recycled aggregates with AA binders becomes economically viable when (i) the precursors for the AA process are produced locally, thereby eliminating long transportation distances, (ii) a significantly lower quantity of activating liquid (e.g., the AS) is required, and

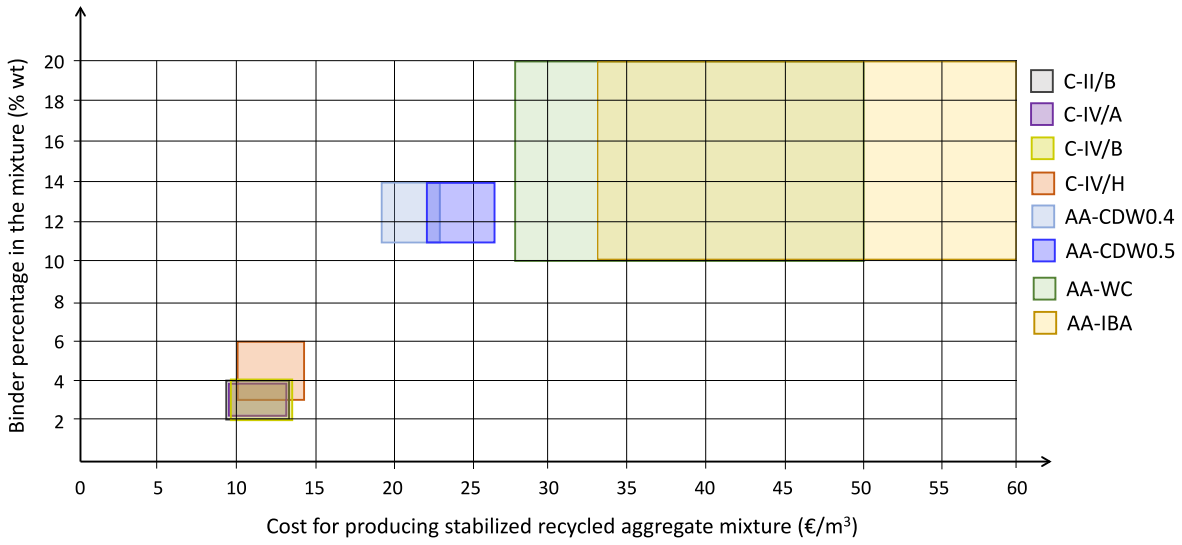


Fig. 13. Estimated costs associated with the production of stabilised aggregates with different types of binders (considering different percentages of addition in the stabilised mixtures).

(iii) the AS is produced using less energy-intensive methods, resulting in a lower initial cost (Abdulkareem et al., 2021).

4. Conclusion

The stabilisation of recycled aggregates is achieved by the addition of a limited amount of binder which increases their stiffness and strength. Portland cement remains the reference binder, but alternative and more sustainable ones should be investigated in response to the growing emphasis on green public procurement policies advocated by the European Union. At present, contractors and professionals are uncertain about their mechanical and environmental performance, and associated costs.

This study fills these gaps by evaluating three low-clinker pozzolanic cements and three alkali-activated fines precursors derived from waste treatment. Unlike most research that focuses solely on mechanical performance, this study uses a comprehensive approach by evaluating mechanical, environmental and economic aspects simultaneously. The selection of binders was based on their local availability in Northwest Italy, which is representative of other industrialised European regions. Their proportions in the mixtures were optimised for their specific use to stabilise recycled aggregates. This contextual and application-oriented approach ensures that the results of the study are practical and applicable in full-scale scenarios.

The key findings from the experimental investigation, LCA analysis, and production cost estimation are as follows:

- pastes made of pozzolanic cement (C-IV/A and C-IV/B) showed comparable compressive strengths to the paste of the reference binder (C-II/B);
- cements CEM-IV/A and CEM-II/B had higher environmental impact indicators (+25 % on average) than CEM-IV/B due to their higher clinker content. This aspect is, in turn, reflected in higher production costs (+10 % on average);
- the hybrid pozzolanic cement (CEM-IV/H) had a lower environmental impact and cost compared to the reference binder (− 62 % of climate change, − 52 % of total consumption of non-renewable resources, − 32 % of costs), but its strength after 7 days was almost halved with respect to the reference binder. However, the compressive strength increased significantly after 28 days of curing, reaching values just below those of the other cements;
- the alkaline activation of CDW fines (AA-CDW) was effective as confirmed by the satisfactory strengths achieved and the relatively low amount of alkaline solution required; this is also reflected in the relatively low environmental impact (−70 % of climate change, –32 % of total consumption of non-renewable resources), with the same costs;
- the alkali-activation of waste clay (AA-WC) had a similar environmental impact to the pastes derived from the alkali-activation of CDW fines but failed to develop mechanical properties, suggesting the inability of the waste clay precursors to develop alkaline reactions;
- the alkali-activation of incinerator bottom ash (AA-IBA) yielded compressive strengths similar to those of AA-CDW fines but required a high amount of alkaline solution thus increasing the environmental impact and cost.

The alkali activation of silica- and alumina-rich waste materials resulted in significantly lower strengths than those achieved using conventional cement. The relatively high crystallinity of mineral phases in the precursors used is responsible for the low mechanical performances of the final mixtures. It should be noted that for stabilised construction materials, excessively strong material is not a prerequisite. However, it is worth noting that when stabilising CDW recycled aggregate, the lower the strength of the binder, the greater the binder quantity to be used in the mix. This leads to an increase in their environmental impact and cost and renders the stabilised mixtures with alkali-activated binders less competitive than cement-stabilised ones. The option of using AA materials will be further strengthened when the challenges associated with the relevant environmental impact arising from the production of AS and the limited local availability of precursor materials are addressed.

Current standards and technical specifications focus primarily on the mechanical properties of stabilised granular mixtures. A broader understanding that includes environmental and economic impacts is essential to enable professionals and decision-makers to make more informed and sustainable choices for their specific context. This study provides a methodological framework that can be applied to different building materials and contexts to support decision-making in constructing sustainable pavements. The scope of this study is limited to an evaluation of the binding phase of stabilised recycled aggregates used in subbase pavement layers and assumes that the binder determines the mechanical properties of the mixture. However, the strength of the

mixture is a result of both the internal cohesion of the binder and its adhesion to the aggregate surface. The amount of the binder in a stabilised granular mixture also determines its final mechanical response. For a full evaluation of alternative binders for stabilised materials, further research should determine the optimum binder content to meet the structural requirements of road pavements.

CRediT authorship contribution statement

Luca Tefa: Writing – review & editing, Writing – original draft, Methodology, Investigation, Data curation, Conceptualization. **Bartolomeo Coppola:** Writing – review & editing, Methodology, Investigation, Data curation, Conceptualization. **Paola Palmero:** Writing – review & editing, Validation, Supervision, Methodology, Formal analysis, Conceptualization. **Marco Bassani:** Writing – review & editing, Validation, Supervision, Resources, Methodology, Funding acquisition, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

The presented investigation is part of the INTREC (*INnovative Technologies for RECycled aggregates from construction and demolition waste in road constructions*) project funded by the European Regional Development call (Regione Piemonte*, Italy, POR*–FESR 2014–2020*, Asse I, Azione I.1b.1.2, Call* PRISM-E). The collaboration with CAVIT S.p.A. in this project was greatly acknowledged.

Data availability

Data will be made available on request.

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