DOKTORI (PhD) ÉRTEKEZÉS

MACHER GERGELY ZOLTÁN

MOSONMAGYARÓVÁR
2026

SZÉCHENYI ISTVÁN EGYETEM
ALBERT KÁZMÉR MOSONMAGYARÓVÁRI KAR
ALKALMAZOTT FENNTARTHATÓSÁG TANSZÉK

WITTMANN ANTAL NÖVÉNY-, ÁLLAT- ÉS ÉLELMISZER- TUDOMÁNYI MULTIDISZCIPLINÁRIS
DOKTORI ISKOLA

HABERLANDT GOTTLIEB NÖVÉNYTUDOMÁNYI DOKTORI PROGRAM

DOKTORI ISKOLAVEZETŐ:
PROF. DR. VARGA LÁSZLÓ DSC
EGYETEMI TANÁR

PROGRAMVEZETŐ:
PROF. DR. PINKE GYULA DSC
EGYETEMI TANÁR

TÉMAVEZETŐ:
DR. TORMA ANDRÁS, PHD
TANSZÉKVEZETŐ EGYETEMI DOCENS

DR. BEKE DÓRA, PHD
EGYETEMI DOCENS

AZ AZBESZTCEMENT TETŐFEDÉSEK HATÁSAINAK VIZSGÁLATA A CSAPADÉKVÍZ KVALITATÍV PARAMÉTEREIRE ÉS FELHASZNÁLHATÓSÁGÁRA *TRIFOLIUM PRATENSE*, *MEDICAGO SATIVA* ÉS *SOLANUM LYCOPERSICUM* NÖVEKEDÉSÉNEK ÉS FEJLŐDÉSÉNEK VIZSGÁLATÁN KERESZTÜL

KÉSZÍTETTE:
MACHER GERGELY ZOLTÁN

MOSONMAGYARÓVÁR
2026

Az azbesztcement tetőfedések hatásainak vizsgálata a csapadékvíz kvalitatív paramétereire és felhasználhatóságára *Trifolium pratense*, *Medicago sativa* és *Solanum lycopersicum* növekedésének és fejlődésének vizsgálatán keresztül

Írta:

MACHER GERGELY ZOLTÁN

Készült a Széchenyi István Egyetem Wittmann Antal Növény-, Állat- és Élelmiszer-tudományi Multidiszciplináris Doktori Iskola Haberlandt Gottlieb Növénytudományi Doktori Programja keretében

Témavezető: Dr. Torma András, tanszékvezető egyetemi docens

Elfogadásra javaslom (igen / nem)

(aláírás)

Témavezető: Dr. Beke Dóra, egyetemi docens

Elfogadásra javaslom (igen / nem)

(aláírás)

A jelölt a doktori komplex vizsgán megfelelt.

Mosonmagyaróvár, ………………………………

……………………………….

a Komplex Vizsga Bizottság elnöke

Az értekezést bírálóként elfogadásra javaslom (igen/nem)

Első bíráló (Szaszkóné Dr. Decsi Éva Kincső) igen/nem

(aláírás)

Második bíráló (Dr. Czédli Herta) igen/nem

(aláírás)

A jelölt az értekezés nyilvános vitáján ……………%-ot ért el.

Mosonmagyaróvár, ………………………………

A Bírálóbizottság elnöke

Doktori (PhD) oklevél minősítése…………………

Az EDT elnöke

A RÖVID, PUBLIKÁCIÓ ALAPÚ ÉRTEKEZÉS HÁTTERE

A Wittmann Antal Növény-, Állat- és Élelmiszer- tudományi Multidiszciplináris Doktori Iskola (továbbiakban: WAMDI) működési szabályzatának 1. sz. melléklete a hagyományos doktori értekezés mellett lehetőséget biztosít rövid, publikáció alapú értekezés készítésére. A doktori eljárás keretében a doktorjelölt egy összefoglaló jellegű, rövid értekezésben rendszerezi és szintetizálja a doktori kutatómunkája során elért, tudományos folyóiratokban már publikált eredményeit, amelyhez szerves egységként mellékelni köteles magukat a közleményeket is; a rövid értekezés és a csatolt publikációk ily módon együttesen alkotják a bírálatra bocsátott doktori művet. Az értekezés alapjául szolgáló publikációkkal szemben támasztott követelmény, hogy azok elsőszerzős, a SCImago adatbázisban Q1-Q3 minősítéssel rendelkező folyóiratokban megjelent, vagy megjelenésre igazoltan elfogadott, teljes terjedelmű eredeti közlemények (full-length original research paper) legyenek. A jelöltnek a benyújtott közleményekkel összesen legalább 6 pontot kell elérnie az előírt pontozási rendszer alapján, amely szerint egy Q1-es besorolású cikk 4 pontot, egy Q2-es 2 pontot, egy Q3-as pedig 1 pontot ér, azzal a szigorítással, hogy a pontszámok összegétől függetlenül legalább egy publikációnak kötelezően Q1-es vagy Q2-es besorolású folyóiratban kell megjelennie.

A feltételek teljesülése a csatolt publikációk alapján:

Scientific Reports (Q1, IF.: 3.9)	4 pont
Clean Technologies And Environmental Policy (Q1, IF.: 3.9)	4 pont
Discover Sustainability (Q1, IF.: 3.0)	4 pont
International Journal Of Environmental Research And Public Health (Q1, IF.: 3.132)	4 pont
Chemical Engineering Transactions (Q3, IF.: 0.84)	1 pont
Chemical Engineering Transactions (Q3, IF.: 0.84)	1 pont
	18 pont

TARTALOMJEGYZÉK

1. A KITŰZÖTT KUTATÁSI FELADAT RÖVID ÖSSZEFOGLALÁSA ... 1
1.1 Alapprobléma felvázolása ... 1
1.2 A kutatás célkitűzései ... 3
1.3 A kutatás tudományos jelentősége ... 4
2. SZAKIRODALMI ÁTTEKINTÉS ÉS TEORETIKUS HÁTTÉR ... 5
3. ANYAG ÉS MÓDSZERTAN ... 8
3.1 AC-minták FT-IR vizsgálata ... 8
3.1.1 Vizsgálati anyagok és mintacsoportok ... 8
3.1.2 Mintapreparálás ... 9
3.1.3 A krizotil azbeszt vizsgálata FT-IR spektroszkópiával ... 9
3.1.4 Alkalmazott műszer: PerkinElmer Spectrum 400 ... 10
3.2 Csapadékvíz-mintavétel, szimulációs kísérletek és vízanalitikai vizsgálatok ... 10
3.2.1 Kutatási megközelítés és vizsgált anyag ... 10
3.2.2 Mintavételi helyszínek és terepi adatgyűjtés ... 10
3.2.3 Szimulált minták előállítása ... 11
3.2.4 Laboratóriumi vízminőségi vizsgálatok ... 12
3.3 Növényélettani vizsgálatok ... 12
3.3.1 Vizsgált növényfajok és kísérleti koncepció ... 12
3.3.2 A csíráztatási és korai növénynevelési kísérlet körülményei és elrendezése ... 13
3.3.3 Mintaoldatok előkészítése ... 13
3.3.4 Csírázási és korai vegetatív növekedési paraméterek mérése és statisztikai értékelése 14
3.4 Regionális azbesztérintettség és keret az azbesztmentes városi átmenethez ... 14
3.4.1 Az MLP koncepcionális alkalmazása a kutatási modellben ... 14
3.4.2 A regionális különbségek szerepe az azbesztmentes átmenetben ... 15
3.4.3 Az azbesztérintettségi integrált index kialakításának módszertana ... 15
3.4.4 Az index értelmezése az MLP keretrendszerében ... 16
4. EREDMÉNYEK ÉS ÉRTÉKELÉSÜK ... 17
4.1 Az azbesztrostok környezeti mobilitásának vizsgálata a talaj-víz rendszerben ... 17
4.2 Krizotil-azbeszt kimutatása AC-mintákból ... 18
4.2.1 Krizotil-azbeszt kimutatása hullámos azbesztcement lemezekből ... 18
4.2.2 Krizotil-azbeszt kimutatása sík azbesztcement lemezekből ... 19
4.2.3 Krizotil-azbeszt kimutatása azbesztcement csövekből ... 20
4.3 Vízanalitikai vizsgálatok eredményei ... 21
4.3.1 Az azbesztcementtel érintkezett és normál csapadékvíz vízminőségi jellemzői ... 21
4.3.2 Az azbesztcementtel érintkezett és normál csapadékvíz nehézfém-szennyezettsége ... 23
4.3.3 Az azbesztcementtel érintkezett szimulált vízminták vízminőségi jellemzői ... 24
4.3.4 A szimulált azbesztcementtel érintkezett minták nehézfém-szennyezettsége ... 26

4.4 Csíráztatásos és korai vegetációs kísérletek eredményei 28
4.4.1 A *Trifolium pratense* vizsgálatának eredményei 28
4.4.2 A *Medicago sativa* elemzésének eredményei 29
4.4.3 A *Solanum lycopersicum* elemzésének eredményei 30
4.5 Az azbesztmentes városi átmenet rendszerszintű elemzése az MLP keretében 32
4.5.1 Makrokörnyezeti (landscape) tényezők hatása az átmenetre 32
4.5.2 Rezsim-szintű beágyazottság és strukturális korlátok 32
4.5.3 Niche-innovációk és azok felskálázási potenciálja 33
4.5.4 A rendszerszintek közötti kölcsönhatás értékelése 33
4.6 Az azbesztérintettség regionális vizsgálata 33
5. EREDMÉNYEK HASZNOSÍTHATÓSÁGA 36
6. ÚJ TUDOMÁNYOS EREDMÉNYEK 38
7. AZ ÉRTEKEZÉS TÉMAKÖRÉBEN MEGJELENT PUBLIKÁCIÓK 40
7.1 Idegen nyelvű folyóiratban lektorált cikk 40
7.2 Magyarnyelvű folyóiratban lektorált cikk 41
RÖVID ÉRTEKEZÉS IRODALOMJEGYZÉKE 43
ÁBRAJEGYZÉK 46
TÁBLÁZATJEGYZÉK 48
A RÖVID ÉRTEKEZÉS ALAPJÁT KÉPEZŐ KÖZLEMÉNYEK

1. A KITŰZÖTT KUTATÁSI FELADAT RÖVID ÖSSZEFOGLALÁSA

1.1 Alapprobléma felvázolása

Az azbeszttartalmú építőanyagok alkalmazása a 20. század második felében világszerte széles körben elterjedt, elsősorban kedvező mechanikai tulajdonságaik, hő- és korrózióállóságuk, valamint gazdaságos előállíthatóságuk miatt. Az azbesztcement (továbbiakban: AC) termékek, különösen a hullámos tetőfedő lemezek, síkpalák és csővezetékek, az ipari, mezőgazdasági és lakóépületek meghatározó szerkezeti elemeivé váltak, így jelentős mennyiségben kerültek beépítésre a hazai és nemzetközi épített környezetbe egyaránt. Bár az azbeszt humán-egészségkárosító hatásai ma már tudományosan igazoltak, a korábban beépített AC-termékek jelentős mennyisége továbbra is jelen van a települési és mezőgazdasági infrastruktúrában. Az épített környezetben „örökölt" azbesztállomány kezelése ezért nem csupán közegészségügyi, hanem komplex környezetbiztonsági kérdés is, amely hosszú távú monitoring- és kockázatértékelési stratégiákat igényel.

A jelenlegi hazai szabályozási és monitoring gyakorlat elsősorban a levegőminőségi paraméterekre koncentrál, miközben a vízi és talajközegben zajló másodlagos expozíciós útvonalak rendszerszintű vizsgálata még nem kellően kidolgozott. A használatban lévő, illetve elöregedett AC- tetőfedések és egyéb szerkezeti elemek környezeti expozíciója (fizikai erózió, hőingadozás, csapadék, UV-sugárzás és biológiai hatások következtében) a mátrix fokozatos degradációjához vezet. A degradációs folyamat során a cementkötés gyengülése elősegíti az azbesztszálak felszabadulását, valamint a mátrixban jelen lévő fémionok mobilizációját, ami multiplikatív környezeti terhelést eredményezhet. Ennek eredményeként az azbesztszálak és a cementmátrixból kioldódó fémionok a levegőbe, a talajba és a felszíni vagy gyűjtött csapadékvizekbe juthatnak. Az AC-termékek élettartamának végfázisában a kibocsátási folyamatok nem lineáris jellegűek, hanem a mikrorepedések hálózatának kialakulásával és a felületi mállás gyorsulásával exponenciális jellegűvé válhatnak, ami a környezeti terhelés időbeli alulbecslésének veszélyét hordozza. A mátrix-degradáció során felszabaduló komponensek nemcsak fizikai szennyezőanyagként jelennek meg, hanem a vízkémiai egyensúly módosításán keresztül más szennyező anyagok mobilitását is befolyásolhatják.

Miközben a levegőben szálló azbesztszálak humánegészségügyi kockázatai részletesen dokumentáltak, a vízi közegbe jutó szálak és az AC-tel érintkező csapadékvíz minőségi paramétereinek változása lényegesen kevésbé feltárt kutatási terület. A vízben diszpergált rostok fizikai-kémiai viselkedése, ülepedési dinamikája, valamint a vízkémiai paraméterekkel való kölcsönhatása mindmáig számos nyitott kérdést vet fel. Különösen releváns ez a kérdés a mezőgazdasági gyakorlatban, ahol az épületfelületekről összegyűjtött csapadékvíz öntözési célú felhasználása egyre gyakoribb, különösen vízhiányos időszakokban. A decentralizált vízgyűjtési rendszerek terjedése ugyanakkor új típusú, lokális expozíciós forgatókönyveket hoz létre, amelyek kockázatelemzése jelenleg hiányos a hazai szakirodalomban. A potenciálisan kontaminált öntözővíz közvetlen hatással lehet a növények csírázási dinamikájára, morfológiai és élettani paramétereire, valamint oxidatív stresszválaszaira. A szálas szerkezetű ásványi részecskék és a mobilizált nehézfémek együttes jelenléte komplex, akár szinergista toxikus hatásmechanizmust is eredményezhet, amely a növényi sejtszintű folyamatokban is kimutatható változásokat indukálhat. A növények (mint elsődleges ökoszisztéma-receptorok) indikátorszervezetként is szolgálhatnak a környezeti terhelés kimutatásában. Ebből következően a vízminőség-változások és a növényi válaszreakciók integrált vizsgálata komplex környezettoxikológiai megközelítést igényel. A növényi válaszreakciók kvantitatív elemzése ezért nemcsak toxikológiai információval szolgál, hanem a környezeti kockázat integrált bioindikációs értékelésének alapját is képezheti. A növényi gyökérzóna (rizoszféra) kémiai és mikrobiológiai folyamatai módosíthatják a szálas ásványi részecskék és a hozzájuk kötődő fémionok viselkedését, ami a biohasznosulás és a toxikus hatás mértékének differenciáltságát eredményezheti. A környezeti kitettség és a biológiai válaszreakciók közötti összefüggések feltárása hozzájárul olyan prediktív modellek kidolgozásához, amelyek a jövőbeni terhelési forgatókönyvek előrejelzésére alkalmasak.

Kutatásom a környezettudomány, a növényélettan és az analitikai kémia metszéspontjában helyezkedik el. A vizsgálatok középpontjában a krizotil-azbesztet tartalmazó AC-termékek környezeti viselkedésének feltárása, a csapadékvíz-minőség torzulásának jellemzése, valamint a kontaminált víz növényi fejlődésre gyakorolt hatásának elemzése áll. A multidiszciplináris megközelítés lehetővé teszi az anyagszerkezeti, vízkémiai és biológiai folyamatok közötti ok-okozati összefüggések rendszerszintű feltárását. A téma aktualitását az adja, hogy Magyarországon az AC-tetőfedések jelentős része még mindig használatban van, miközben a csapadékvíz-hasznosítás szerepe a fenntartható vízgazdálkodásban folyamatosan növekszik. A klímaváltozás következtében gyakoribbá váló aszályos periódusok tovább növelik a decentralizált vízgyűjtési megoldások jelentőségét, ami a potenciálisan kontaminált csapadékvíz felhasználásának kockázatát is erősíti. E körülmények között a probléma vizsgálata nem csupán tudományos kérdés, hanem a fenntartható vízhasználat, az agrár-ökoszisztémák stabilitása és a környezeti kockázatmegelőzés szempontjából is stratégiai jelentőségű. A két tényező együttes jelenléte olyan potenciális környezeti kockázatot jelent, amelynek tudományos megalapozottságú feltárása elengedhetetlen.

1.2 A kutatás célkitűzései

Kutatási tevékenységem során integrált kutatási célrendszert határoztam meg, amely az AC-termékek anyagszerkezeti vizsgálatától a vízminőségi elemzésen keresztül a növényélettani és toxikológiai értékelésig terjed.

- **C-1:** A kutatás elsődleges célja az AC-termékek krizotil-tartalmának és kimutathatóságának vizsgálata volt különböző terméktípusok esetében. Ennek keretében hullámos karakterisztikájú tetőfedő elemeket, sík AC-lemezeket, valamint AC-csöveket elemeztem Fourier-transzformációs infravörös (továbbiakban: FT-IR) spektroszkópiai módszerrel. Összehasonlítottam a környezeti expozíciónak kitett és attól szeparált minták spektrális illeszkedési paramétereit annak érdekében, hogy feltárjam a mátrix-degradáció és a krizotil kimutathatósága közötti összefüggéseket. Célom annak igazolása volt, hogy a különböző terméktípusok eltérő szerkezeti sajátosságai miként befolyásolják az azbeszt detektálhatóságát és környezeti ellenálló képességét.
- **C-2:** A második kutatási cél az AC-tetőfedéssel érintkezett csapadékvíz minőségi paramétereinek átfogó vizsgálata volt. Ennek során összehasonlító vízanalitikai méréseket végeztem a pH, a fajlagos elektromos vezetőképesség, az összes keménység, a lebegőanyag-tartalom, a teljes szerves széntartalom (TOC), valamint a biokémiai és kémiai oxigénigény (BOI_5, KOI) vonatkozásában. Meghatároztam a nehézfém-koncentrációkat (Zn, Pb, Hg), és kvantifikáltam a normál csapadékmintákhoz viszonyított változási arányokat. A vizsgálatok célja a cementmátrixból történő fémmobilizáció és a vízminőség torzulása közötti oksági kapcsolatok feltárása, valamint a csapadékvíz öntözési célú felhasználhatóságának környezetbiztonsági értékelése volt.
- **C-3:** A harmadik kutatási cél a kontaminált víz növényélettani hatásainak vizsgálata volt. Primer modellnövényként *Trifolium pratense*, szekunder modellként *Solanum lycopersicum* és *Medicago sativa* alkalmazásával elemeztem a csírázási arányt és a csírázási dinamikát különböző dóziskoncentrációk mellett. Kvantitatív módon értékeltem a gyökérhossz és a hajtásmagasság változásait, továbbá lineáris regressziós modellezéssel vizsgáltam a dózis-válasz összefüggéseket a kezdeti vegetációs stádiumban. A növekedési paraméterek alapján meghatároztam az abiotikus stresszhatás mértékét, és értelmeztem annak biológiai relevanciáját.
- **C-4:** A negyedik célkitűzés a környezeti és toxikológiai hatásmechanizmusok integrált értékelése volt. Ennek keretében elemeztem az azbesztszálak és a mobilizált nehézfémek potenciális szinergista hatását, valamint vizsgáltam az oxidatív stressz indukciójára utaló növényi válaszreakciók és morfológiai változások közötti összefüggéseket. A cél a

szennyezett öntözővíz mezőgazdasági és élelmiszerlánc-biztonsági kockázatának kvalitatív értékelése volt.

– **C-5:** Az ötödik célkitűzés a települési azbesztmentesítési és kivezetési stratégiák tudományos megalapozása volt a vízminőségi és biológiai kockázatok integrált értékelése alapján. Ennek keretében vizsgáltam, hogy az AC-tetőfelületekről lefolyó csapadékvíz minőségi torzulása és a növényélettani válaszreakciók miként értelmezhetők települési környezeti kockázati kontextusban. Elemeztem a potenciális expozíciós útvonalakat, különös tekintettel a csapadékvíz-gyűjtés és öntözési felhasználás gyakorlatára, valamint értékeltem az infrastruktúra-örökségből eredő környezeti terhelés szerepét a fenntartható vízgazdálkodás rendszerében. A cél olyan tudományosan megalapozott szempontrendszer kialakítása volt, amely hozzájárulhat a települési azbesztkivezetési programok priorizálásához és a környezetbiztonsági döntéshozatal támogatásához.

1.3 A kutatás tudományos jelentősége

A kutatás újdonságértéke abban rejlik, hogy integrált módon vizsgálja az AC degradációját, és összekapcsolja az anyagszerkezeti változásokat a vízanalitikai paraméterek torzulásával, valamint a növényélettani válaszreakciókkal. A vizsgálatok kvantitatív bizonyítékot szolgáltatnak arra, hogy az AC-tetőfedéssel érintkező csapadékvíz minőségi paraméterei szignifikáns mértékben módosulhatnak, továbbá modellezik a szennyezett víz növényi fejlődésre gyakorolt dózisfüggő hatását. Eredményeim hozzájárulnak a csapadékvíz-hasznosítás környezetbiztonsági újraértékeléséhez, megalapozzák a lakossági és mezőgazdasági kockázatbecslést, valamint tudományos alapot teremtenek a fenntartható települési azbesztkivezetési stratégiák kidolgozásához. A multidiszciplináris megközelítés révén a kutatás nemcsak leíró jellegű eredményeket szolgáltat, hanem oksági összefüggéseket is feltár az anyagszerkezeti degradáció, a vízminőség-változás és a biológiai válaszreakciók között, ezáltal hozzájárulva a környezeti expozíciós modellek továbbfejlesztéséhez és a hazai szabályozási diskurzus tudományos megerősítéséhez.

2. SZAKIRODALMI ÁTTEKINTÉS ÉS TEORETIKUS HÁTTÉR

Az azbeszttel kapcsolatos kockázatoknak kitett munkavállalók védelméről szóló 12/2006. (III. 23.) EüM rendelet megfogalmazása szerint, 6 ásványfaj szálas változatát sorolja a magyarországi szabályozás az azbesztek közé, ezek a szerpentincsoportból a krizotil, míg az amfibolcsoportból az aktinolit, a krokidolit, az antofillit, a grunerit és a tremolit. A WHO kritériumai szerint, szabályszerű azbesztszálnak tekintendő az a részecske, melynek hossza >5 μm, szélessége <3 μm, és oldalaránya (hossz: szélesség) > 3:1 (Cosette, 1984). Az azbesztet és a belőle készült AC-termékeket már az 1970-es évek előtt is széles körben és nagymértékben használták építőanyagként (Leonelli et al., 2006; Bloise et al., 2018). Mára a belélegzett azbesztszálak számos daganatos (tüdőrák, mellhártyarák, hörgőrák, mezotelióma) és nem daganatos betegség (azbesztózis, pleuraplakk) fő okaként is ismertek (McDonald, 1972; Amavais és Hunter, 1977, Mossman et al., 1990). Ennek okán, a legtöbb országban betiltották e termékek újonnani használatát és gyártását, ugyanakkor a tilalom előtt beépített anyagok további használatára, eltávolítására vonatkozóan máig nem áll rendelkezésre szabályozás. A hazai szakirodalomban ugyanakkor továbbra is dominánsan a foglalkozási és levegőminőségi expozíciós útvonalak vizsgálata jelenik meg, miközben az AC-ből származó vízi mobilizáció, valamint annak mezőgazdasági és ökológiai következményei marginális figyelmet kaptak. Magyarországon különösen hiányoznak azok az empirikus, kvantitatív vizsgálatok, amelyek az AC-tetőfelületekről lefolyó csapadékvíz minőségi paramétereit, valamint annak biológiai hatásait integrált módon értékelnék.

Már Spurny et al. (1989), valamint Suzuki et al. (2005) is megfogalmazta, hogy az e termékekre jellemző AC-mátrix sérülékenysége révén, egy sérült és elöregedett 1 m^2-es hullámos karakterisztikával rendelkező AC nagylemezes elem évente megközelítőleg több grammnyi azbesztet veszít el a mátrix szerkezetből. E terméktípus elsődleges alkalmazási területe a magas, illetve hidegtetős épületek héjazata, mely jellemzően mezőgazdasági és kereskedelmi tárolásra, valamint ipari tevékenységre szolgáló épületeknél volt kedvelt fedési mód (Székely, 1981). A hullámos karakterű AC-termékek természetes degradációs tendenciájukon túlmenően elszíneződhetnek, lepattoghatnak, pelyhesedhetnek, valamint a biotikus növekedésre is alkalmas környezetet teremthetnek (Spurny et al., 1989). Ebből kifolyólag az elmúlt évtizedekben a kutatók figyelme a foglalkozási expozíciós és humánegészségügyi kockázatok vizsgálata helyett a környezeti expozíciós forgatókönyvek felé terelődött (Magherini et al., 2023). Baumann et al. (2013) azt írták, hogy noha mára sokat tudunk a levegőben szálló azbeszt és hasonló azbesztiform ásványok belégzése által kiváltott egészségügyi kockázatokról, a vízben lebegő és úszó szálak potenciális egészségügyi és környezeti kockázatai még nem ismeretesek. Mezőgazdasági és vízgazdálkodási szempontból külön figyelmet érdemel a szennyezett talajvíz öntözésre (Turci et al. 2016) történő felhasználása. Emellett az összegyűjtött, AC-tetőfedéssel érintkezett és kontaminált csapadékvíz felhasználása, valamint a magyarországi AC-mederanyaggal rendelkező öntözőcsatornák degradációja is komoly és egyre időszerűbbé váló környezetbiztonsági kérdéssé vált, és még súlyosabbá fog válni a közeljövőben. A települési infrastruktúrában még jelen lévő AC-elemek és az egyre terjedő decentralizált csapadékvíz-hasznosítás együttes jelenléte olyan lokális expozíciós forgatókönyvet teremt, amelynek tudományos feldolgozottsága hazai viszonylatban hiányos. A jelenlegi szabályozási és monitoring gyakorlat nem tér ki differenciált módon a csapadékvíz-gyűjtés során potenciálisan érintett AC-tetőfelületek környezeti kockázatának értékelésére, így a döntéshozatal nem támaszkodik célzott, lokális tudományos bizonyítékokra. Nem áll rendelkezésre olyan magyarországi adatbázis, amely a vízanalitikai paraméterek változását növényélettani válaszreakciókkal összekapcsolva, integrált módon értékelné.

Trivedi et al. (2004) a krizotil-azbeszt vízi környezetben kifejtett hatásmechanizmusainak vizsgálatát kutatták, melynek során a *Lemna gibba* negatív válaszreakcióit vizsgálták a növekedési, fiziológiai és biokémiai paraméterek mentén. Kísérleteik során kimutatták, hogy a krizotil expozíció negatív válaszreakciót váltott ki a levélszám, a gyökérhossz, valamint a biomassza mennyiségében. Mindamellett a klorofill-, karotinoid-, szabad cukor-, fehérje- és keményítőtartalom esetében is negatív hatásokat igazoltak. Ezzel párhuzamosan az oxidatív stresszre irányuló paraméterekben (lipid-peroxid, celluláris hidrogén-peroxid, kataláz, szuperoxid-diszmutáz) idő- és dózisfüggő növekedést mutattak ki. A kísérlet 2007-ben megismétlésre került, melynek során a *Lemna gibba* növényeket itt már négy különböző koncentrációjú (0,5; 1,0; 2,0 és 5,0 mg/l) krizotil-azbeszt oldat hatásának tették ki. A vizsgálatok során igazolták, hogy a krizotil-azbeszt csökkentette a teljes és redukált glutation mennyiségét, mindezzel párhuzamosan növelte az oxidált glutation tartalmat – így a redukált:oxidált glutation arányt – és csökkentette a redukált:oxidált aszkorbát arányt (Trivedi et al., 2007). Trivedi és Ahmad (2011) ezt követően azt vizsgálta, hogy miként hat a mezőgazdasági haszonnövényekre (búza, borsó, mustár) a krizotil-azbeszttel kontaminált talaj. Úgy találták, hogy a krizotil-azbeszt mennyiségének növekedésével a csírázási arány jelentősen csökkent. A krizotil okozta azbeszttoxicitás egyaránt kimutatható volt a hajtás magasságában, a gyökér hosszában, a biomasszában, a klorofillban és a fehérjetartalomban is. A növényi válaszreakciók vizsgálata 2013-ban kiegészült a vöröshagymára gyakorolt hatásmechanizmusok elemzésével. Trivedi és Ahmad (2013) laboratóriumi körülmények között, különböző dózisú (0,5; 1,0; 2,0 és 5,0 g/l) krizotil-azbeszt oldatnak tette ki az *Allium cepa* L. mintanövényeket. A kapott eredmények a korábban ismertetettekhez voltak hasonlatosak, ugyanakkor elsőként vizsgálták a krizotil genotoxicitását is a mintanövények gyökércsúcs merisztémáin. Ugyancsak a növényi növekedési paraméterekre és a tápanyagfelvételre gyakorolt hatások vizsgálatát célozta Saleem et al. (2022) kutatása is, melynek során *Panicum virgatum* és *Phleum pretense* került expozíció alá. Az azbeszttoxicitás mindkét fűféle esetében igazolható, az összes morfológiai paraméter negatív eredménye kimutatható volt. A vizsgált egyedek gyökerében a Cr, Mn, V, As, és Ba felvétele jelentősen fokozódott. Ugyanakkor kutatásuk legfontosabb eredményének mégsem ez a kimutatás tekintendő, hanem az olyan antioxidáns enzimek, mint a szuperoxid-diszmutáz (SOD), kataláz (CAT), glutation-peroxidáz (GPX) és peroxidáz (POD), amelyek aktivitása jelentősen megnövekedett az azbeszterhelés hatására. Mindez arra utal, hogy a növények aktiválták védekező mechanizmusukat a krizotil toxikus hatásának semlegesítésére. A környezeti stresszel szembeni tolerancia emelésekor a növények antioxidánsokat termelnek a túlzott reaktív oxigénfajták (ROS) okozta toxicitás megfékezésére (Dola et al., 2022; Farooq et al., 2022; Ma et al., 2022a; Ma et al., 2022b). A nemzetközi szakirodalomban bemutatott növényélettani és oxidatív stressz-válaszok egyértelműen igazolják, hogy a krizotil-azbeszt vízi közegben is biológiailag aktív, és dózisfüggő hatásmechanizmusokon keresztül befolyásolja az élő rendszerek működését. Mindazonáltal e vizsgálatok többsége izolált laboratóriumi környezetben, szabályozott kísérleti feltételek mellett zajlott, és nem kapcsolódott konkrét települési infrastruktúra-örökségből eredő expozíciós forgatókönyvekhez. Korábbi, rendszerelméleti megközelítésű kutatásomban a települési azbesztmentesítési folyamatokat a multi-level perspective (továbbiakban: MLP) keretrendszerében értelmeztem, rámutatva arra, hogy az „asbestos-free transition” nem pusztán technológiai, hanem intézményi és szabályozási átmenet is.

A nemzetközi szakirodalom egyértelműen igazolja továbbá, hogy az azbeszt toxikológiai és humánegészségügyi kockázatai jól dokumentáltak, valamint hogy vízi közegben is kimutatható dózisfüggő növényélettani és oxidatív stresszválaszokat indukál. Hazai viszonylatban hiányoznak azok az integrált kutatások, amelyek az AC anyagszerkezeti degradációját, a csapadékvíz minőségi paramétereinek torzulását és a biológiai válaszreakciókat egységes, rendszerszintű megközelítésben értelmeznék. Mindezek alapján indokolttá válik egy olyan multidiszciplináris vizsgálati keret alkalmazása, amely az anyagtudományi, vízanalitikai és növényélettani paraméterek integrált elemzésével képes feltárni a hazai környezeti expozíciós forgatókönyvek valós kockázati dimenzióit.

3. ANYAG ÉS MÓDSZERTAN

Jelen disszertáció publikáció-alapú felépítésű, ezért a módszertani keretet a korábban megjelent, lektorált Q1-es közleményeimben részletezett vizsgálati eljárások integrált bemutatására építettem. Kutatásom több, egymásra épülő vizsgálati szintet foglal magában, kezdve az azbesztcement termékek anyagszerkezeti és degradációs jellemzését, a tetőfelületekkel érintkező csapadékvíz fizikai-kémiai paramétereinek elemzését, a kontaminált víz növényélettani hatásainak kísérleti vizsgálatát, valamint a települési szintű fenntarthatósági és kockázati keretrendszer rendszerszintű értékelését. E fejezet célja, hogy egységes szerkezetben, reprodukálható módon és a publikációkon átívelő szakmai koherenciával mutassam be az alkalmazott mintavételi protokollokat, laboratóriumi analitikai eljárásokat, kísérleti beállításokat, valamint a statisztikai adatértékelés módszereit.

3.1 AC-minták FT-IR vizsgálata

3.1.1 Vizsgálati anyagok és mintacsoportok

A vizsgálatok során háromféle azbeszttartalmú anyagmintát alkalmaztam: hullámos karakterisztikájú azbesztcement (AC) tetőlemezeket, sík azbesztcement lemezeket, valamint azbesztcement csöveket. Mindhárom kategórián belül két alcsoportot különítettem el: az egyik csoportba a környezeti hatásoknak (csapadék, szél, hőmérséklet-ingadozás, napsugárzás) kitett minták kerültek ($n = 5$), míg a másik csoportba az ezektől védetten tárolt minták ($n = 5$). Valamennyi mintán ismételt méréseket végeztem a krizotil-spektrumokkal való illeszkedés meghatározása érdekében. Az illeszkedések sikerességét rögzítettem, majd a tíz legjobb spektrális egyezés alapján statisztikai értékelést végeztem. E módszertan lehetővé tette annak vizsgálatát, hogy a környezeti expozíció miként befolyásolja az azbesztcement termékek szerkezeti integritását a védetten tárolt mintákhoz képest. A környezeti hatásoknak kitett minták bontásból és használatból származó, valós körülmények között alkalmazott termékek voltak, amelyeket Győr különböző helyszíneiről gyűjtöttem be.

A szeparált (nem expozíciónak kitett) minták ugyanezen térségből származtak, olyan tárolási helyekről, ahol fel nem használt azbesztcement termékeket raktároztak. A begyűjtött mintákat a vizsgálatok megkezdéséig hermetikusan zárt körülmények között tároltam. A környezeti hatások vizsgálata összehasonlító jellegű volt, mesterséges stresszalkalmazás nélkül. Feltételezésem szerint a használatban lévő minták folyamatosan ki voltak téve a környezeti tényezők (többek között az öregedés, fizikai károsodás, napsugárzás és csapadék) együttes hatásának. Az első mintacsoport hullámos azbesztcement tetőlemezekből állt. Ezek a megközelítőleg 30–40 éves elemek jól látható eróziós és degradációs jeleket mutattak, valamint különféle lerakódások is megfigyelhetők voltak rajtuk. A környezeti hatásoknak kitett minták jelölése AC-S-E-1 és AC-S-E-5 között történt, az átlagértéket AC-S-E-A jelöléssel láttam el. A szeparált minták jelölése AC-S-S-1 és AC-S-S-5 között történt, átlaguk AC-S-S-A jelölést kapott. A második mintacsoport sík azbesztcement lemezekből állt, amelyek hasonlóan mintegy 40 évesek, és a környezeti expozíció hatásai itt is egyértelműen megfigyelhetők voltak. A környezeti hatásnak kitett minták jelölése AC-F-E-1 és AC-F-E-5 között történt (átlag: AC-F-E-A), míg a szeparált minták jelölése AC-F-S-1 és AC-F-S-5 között (átlag: AC-F-S-A). A harmadik mintacsoportot hasonló korú, környezeti hatásoknak kitett, illetve védetten tárolt azbesztcement csövek alkották. A környezeti hatásoknak kitett csövek jelölése AC-P-E-1 és AC-P-E-5 között történt (átlag: AC-P-E-A), míg a szeparált csövek jelölése AC-P-S-1 és AC-P-S-5 között (átlag: AC-P-S-A).

3.1.2 Mintapreparálás

A minták felületét desztillált vízzel és izopropil-alkohollal tisztítottam meg annak érdekében, hogy az esetleges felületi szennyeződések ne befolyásolják a mérési eredményeket. A tisztítás ugyanakkor bizonyos mértékben módosíthatta a minták eredeti felületi állapotát, ezért ezt a környezeti expozíció értékelésének módszertani korlátjaként vettem figyelembe. Ezt követően a mintákat egységes méretre vágtam, majd többkomponensű epoxigyantába ágyaztam. Az epoxigyantát előre meghatározott méretű formába öntöttem, amelyből 30 × 5 mm méretű korongokat készítettem. A korongokat csiszológéppel a kívánt méretre és felületi minőségre alakítottam, majd műszerasztalon rögzítettem vizsgálatra. A mérés során a 0,3 mm átmérőjű gyémántkristályt a minta felületével érintkeztettem, amely lehetővé tette a kiválasztott területek infravörös sugárzással történő pásztázását. Ennek eredményeként abszorpciós térkép készült, amelyen a különböző spektrumokat eltérő színek jelölték. A színkódolt spektrumok a spektrális könyvtárban szereplő anyagokhoz rendelhetők voltak, így biztosítva az összetevők azonosítását.

3.1.3 A krizotil azbeszt vizsgálata FT-IR spektroszkópiával

Az azbeszttípusok erős abszorpciót mutatnak az 1200–900 cm^{-1}, illetve a 600–300 cm^{-1} hullámszám-tartományban. Az azbeszt típusának kvalitatív azonosítása a spektrum 200 cm^{-1}-ig történő elemzésével végezhető el. A krizotil jól elkülöníthető az amfibol típusú azbesztektől, mivel jellegzetes kettős hidroxilcsoport-sávokat mutat 3693 cm^{-1} és 3648 cm^{-1} hullámszámnál. Ezek a sávok a rácsszerkezet elsődleges szilikátrétegeiben elhelyezkedő hidroxilcsoportok közötti kölcsönhatásokból származnak.

3.1.4 Alkalmazott műszer: PerkinElmer Spectrum 400

A mérésekhez a PerkinElmer Spectrum 400 (PerkinElmer, MA, USA) típusú infravörös spektrométert alkalmaztam. A vizsgálatok során az azbesztcement termékek törési felületeiből vett mintákat, valamint a mátrixanyagból származó hulladékmintákat elemeztem összetételük és az esetlegesen jelen lévő adalékanyagok azonosítása céljából. A mintákat a spektrométer tárgyasztalára helyeztem, a 0,3 mm-es gyémántkristályt felfelé irányítva, és körülbelül 100 N nagyságú erővel rögzítettem a megfelelő kontaktus biztosítása érdekében. A mérések érzékenyek voltak a nedvességre és a szén-dioxid jelenlétére, ezért minden egyes mérés előtt rögzítettem a környezeti háttérspektrumot, hogy a műszer képes legyen elkülöníteni a valódi mintaspektrumot a háttérinterferenciától.

3.2 Csapadékvíz-mintavétel, szimulációs kísérletek és vízanalitikai vizsgálatok

3.2.1 Kutatási megközelítés és vizsgált anyag

A tanulmány vegyes módszertani megközelítést alkalmazott, amely terepi mintavételt és laboratóriumi analitikai vizsgálatokat egyaránt magában foglalt. A mixed-methods megközelítés lehetővé tette a valós környezeti expozíciós forgatókönyvek és a kontrollált laboratóriumi körülmények közötti összehasonlíthatóság biztosítását. A vizsgálat középpontjában az azbesztcement (AC) tetőfedő anyag állt, amelynek új alkalmazása Magyarországon már tiltott, ugyanakkor a korábban beépített elemek továbbra is széles körben jelen vannak az épületállományban. A vizsgálat tárgyát képező tetőfedő termék hullámos karakterisztikájú azbesztcement lemez volt, amelyet az 1980-as években épült lakó- és melléképületek esetében széles körben alkalmaztak. Az anyag összetétele elsősorban krizotil-azbesztet tartalmaz (megközelítőleg 8-10 tömegszázalék arányban), míg a fennmaradó részt cement és egyéb kötőanyagok alkotják (Tóth és Weiszburg, 2011). Az anyagösszetétel ismerete elengedhetetlen volt a potenciális kioldódási és mobilizációs mechanizmusok értelmezéséhez.

3.2.2 Mintavételi helyszínek és terepi adatgyűjtés

A csapadékminták gyűjtését Győr területén, összesen 20 mintavételi ponton végeztem. A városban (hasonlóan több magyarországi településhez) még mindig jelentős számban találhatók azbesztcement fedésű épületek, különösen az 1980-as években épült lakóépületek és melléképítmények esetében. Győr ipartörténeti sajátosságai miatt a városban korábban működő, azbeszttartalmú szigetelő- és építőpanelek gyártására specializálódott üzem következtében az azbeszttartalmú anyagok elterjedtsége különösen magas volt. A mintaterület kiválasztása ezért nemcsak praktikus, hanem expozíciós szempontból is indokolt volt, mivel reprezentatív képet ad a hazai városi infrastruktúra-örökség egy részéről.

Az első 10 mintavételi helyszínen a csapadékvizet közvetlenül az azbesztcement tetőfelületről lefolyva, azzal érintkezve gyűjtöttem össze. A további 10 helyszínen a csapadékvizet közvetlenül, felületi érintkezés nélkül, kontaminációmentesen gyűjtöttem be. Ez a kettős mintavételi stratégia lehetővé tette a tetőfelületi hatás kvantitatív elkülönítését a légköri háttérterheléstől. A minták gyűjtése 10 literes üveg edényekben történt. A mintavételi időpontokat az aktuális meteorológiai körülmények és az azt megelőző levegőminőségi mutatók figyelembevételével határoztam meg. A minták gyűjtése során ügyeltem arra, hogy azok ne érintkezzenek horganyzott lemezzel vagy más olyan anyaggal, amely befolyásolhatná a víz kémiai paramétereit. Az anyagkontamináció kizárása kulcsfontosságú volt a nehézfém-analízis megbízhatósága szempontjából. A vizsgált tetőfedő elemek eltérő korúak és különböző mértékben erodált állapotúak voltak. Részletes ásványtani és felületkarakterizálás nem állt rendelkezésre, azonban a minták vizuális dokumentálása megtörtént. Valamennyi vizsgált lemez erősen időjárásnak kitett, repedezett és elszíneződött felületet mutatott. A vizuális degradációs jegyek rögzítése kvalitatív indikátorként szolgált az anyag állapotának értelmezéséhez. A desztillált vizes szimulációhoz használt minták ugyanazon tetőkről származtak. A mintavétel során rögzítettem a csapadék intenzitását és időtartamát is. A terepi körülmények szorosan közelítették a laboratóriumi szimuláció paramétereit: az átlagos csapadékintenzitás megközelítőleg 10 mm/5 perc volt, az események tipikus időtartama pedig 15-20 perc között alakult. Minden csapadékeseményből kizárólag az első 5 liternyi lefolyó vizet gyűjtöttem be, mivel a kezdeti lemosódási (wash-off) fázis várhatóan a legmagasabb szennyezőanyag-koncentrációt tartalmazza. A „first flush” mintavételi stratégia alkalmazása a szennyezőanyag-mobilizáció maximális detektálhatóságát célozta.

3.2.3 Szimulált minták előállítása

A természetes csapadékminták gyűjtésével párhuzamosan mesterséges mintákat is előállítottam desztillált víz felhasználásával. A desztillált vizet egy erodált, hullámos karakterisztikájú azbesztcement lemezen keresztül vezettem át, amely tulajdonságaiban megfelelt a vizsgált tetőfedő anyagoknak. Ezzel a kioldási (leaching) eljárással lehetőség nyílt annak vizsgálatára, hogy a víz kémiai jellemzői miként változnak kizárólag az anyaggal való közvetlen érintkezés következtében, kontrollált körülmények között. A desztillált víz alkalmazása minimalizálta a háttérionok jelenlétét, így az anyag-specifikus hatások izolálhatóvá váltak. A szimuláció során 1 liter desztillált vizet öntöttem át a tetőlemezen 300 másodperc alatt. Ezt az időtartamot azért választottam, hogy modellezzem a vizsgált térségben jellemző, mérsékelt intenzitású csapadékeseményeket. A cél a természetes csapadék intenzitásának laboratóriumi környezetben történő reprodukálása volt. A kontrollált expozíciós idő és térfogat biztosította a kísérlet reprodukálhatóságát és összehasonlíthatóságát.

A mesterséges minták alkalmazását indokolta az is, hogy a természetes csapadékvíz minőségét befolyásolják az aktuális meteorológiai és légszennyezettségi viszonyok. A szimulált minták lehetővé tették annak vizsgálatát, hogy a kontamináció milyen mértékben tulajdonítható közvetlenül az azbesztcement anyaggal való elsődleges érintkezésnek. Ez a módszertani kettősség (terepi és laboratóriumi megközelítés) növelte az eredmények interpretációs megbízhatóságát.

3.2.4 Laboratóriumi vízminőségi vizsgálatok

A begyűjtött csapadékmintákat laboratóriumi analízisnek vetettem alá, amelynek célja a vízminőségi jellemzők átfogó értékelése volt. A vizsgált paraméterek kiválasztása korábbi szakirodalmi ajánlások alapján történt, különös tekintettel az öntözési célú vízhasználat alkalmasságára (Mendez et al., 2011; Nicholson et al., 2010; Qin et al., 2015; Yaziz et al., 1989). A paraméterválasztás célja az volt, hogy a fizikai, kémiai és potenciálisan toxikológiai jellemzők egyaránt lefedésre kerüljenek. A pH, vezetőképesség, lebegőanyag-tartalom, keménység, TOC, KOI és BOI meghatározása a vonatkozó MSZ és EN szabványok szerint történt. A szabványosított módszerek alkalmazása biztosította az eredmények nemzetközi összehasonlíthatóságát és validitását. A nehézfémek (higany, ólom és cink) vizsgálata különös jelentőséggel bírt, mivel ezek a komponensek környezeti és egészségügyi szempontból kritikusak. A fémkoncentrációk meghatározása lehetővé tette a potenciális toxikus terhelés kockázati profiljának kialakítását, valamint az öntözési célú felhasználás környezetbiztonsági értékelését.

3.3 Növényélettani vizsgálatok

3.3.1 Vizsgált növényfajok és kísérleti koncepció

A vizsgálat során három növényfaj csírázási paramétereit és fajtán belüli variabilitását elemeztem: *Trifolium pratense* (Salino, Rozeta, Altaswede), *Medicago sativa* (Emiliana, Gea, Algonquin) és *Solanum lycopersicum* (Manó, Kecskeméti 549, Mobil). A *Trifolium pratense* és *Medicago sativa* vetőmagokat a Pannon-Mag-Agrár Kft.-től, míg a *Solanum lycopersicum* vetőmagokat a Garafarm Trade Kft.-től szereztem be. A vizsgálat célja az volt, hogy kontrollált laboratóriumi körülmények között igazoljam az azbesztcementtel szennyezett öntözővíz növényi fejlődésre gyakorolt kedvezőtlen hatásait, desztillált vízzel kezelt kontrollcsoport alkalmazásával. A csírázási arány és csírázási dinamika szignifikáns csökkenést mutatott valamennyi vizsgált faj és fajta esetében. A kiválasztott növényfajok agronómiai és környezeti relevanciával egyaránt rendelkeznek. A *Solanum lycopersicum* élelmiszerbiztonsági szempontból kiemelt jelentőségű, mivel potenciálisan képes lehet kontaminánsok akkumulációjára szennyezett öntözővíz használata esetén. A *Trifolium pratense* az urbánus zöldfelületek és alternatív vízhasználati rendszerek (pl. szürkevíz-öntözés) kontextusában releváns. A *Medicago sativa* szántóföldi jelentősége miatt alkalmas az azbesztcement alapú öntözőrendszerekből eredő expozíciós kockázatok modellezésére. A három eltérő ökológiai funkciójú növényfaj alkalmazása lehetővé tette az eredmények szélesebb körű értelmezhetőségét agrár- és városi környezetben egyaránt.

3.3.2 A csíráztatási és korai növénynevelési kísérlet körülményei és elrendezése

A dózisfüggő stresszhatás vizsgálata érdekében mindhárom növényfaj három különböző fajtáját teszteltem. A magokat 2,0%-os NaOCl oldattal 2 percen keresztül felületsterilizáltam, majd háromszor steril desztillált vízzel öblítettem. Ezt követően 10-10 magból álló egységeket helyeztem steril Petri-csészékbe, nedvesített vattakorongra. Minden kezelési dózishoz öt ismétlést alkalmaztam ($N = 5 \times 10$). Egy adott növénytípus esetében összesen 350 vizsgálati egységet elemeztem ($N = 7 \times 50$). A csíráztatást 22 ± 1 °C állandó hőmérsékleten végeztem, digitális hőmérséklet-szabályozással. A teljes spektrumú LED-rendszer a kelést követő korai növénynevelési szakasz kontrollált fényviszonyait biztosította programozható fényciklusokkal és szabályozható intenzitással. A növényszintű fényintenzitás maximuma 50 lm volt. A megfelelő levegőáramlást szabályozott szellőztetőrendszer biztosította. A kontrollált környezeti feltételek biztosították, hogy a megfigyelt eltérések kizárólag a kezelési dózisok hatásának legyenek tulajdoníthatók. Az ismétlések száma a biológiai variabilitás csökkentését és a statisztikai megbízhatóság növelését szolgálta.

3.3.3 Mintaoldatok előkészítése

A vizsgálat során Magyarországon elterjedten alkalmazott azbesztcement termékeket használtam, amelyek krizotil-azbeszt tartalma 8,00–10,0%, cementtartalma pedig 90,0–92,0% között mozog. Az extraktum előállításához az azbesztcement mintákat mechanikusan finom porrá őröltem. A port meghatározott koncentrációkban (1,00; 2,00; 5,00; 10,0; 25,0; 50,0 mg/l) kétszeresen desztillált vízben szuszpendáltam, majd 24 órán keresztül szobahőmérsékleten inkubáltam. Az extrakciót követően a szuszpenziókat 0,45 μm pórusméretű szűrőn szűrtem át a nagyobb részecskék eltávolítása érdekében, miközben a finomabb frakció és kolloidális komponensek az oldatban maradtak. Az oldatokat steril üveg edényekben, sötét körülmények között, 4 °C-on tároltam a felhasználásig. A kontrollcsoport kétszeresen desztillált vízzel került kezelésre. A koncentrációsor kialakítása lehetővé tette a dózis–válasz összefüggések kvantitatív értékelését, valamint a potenciális küszöbhatások azonosítását. A 24 órás extrakciós idő biztosította az oldható komponensek és mobilizálható frakciók reprezentatív kioldódását.

3.3.4 Csírázási és korai vegetatív növekedési paraméterek mérése és statisztikai értékelése

A vizsgálat során folyamatosan rögzítettem a csírázási arányt és a csírázási időt. A gyökérhosszt a vattakorong felszínével való érintkezési ponttól a főgyökér csúcsáig mértem. A hajtásmagasságot a 31. napon, korai vegetatív növekedési paraméterként határoztam meg. A mérésekhez Burg-Wächter Precise PS-72150 digitális tolómérőt alkalmaztam (0–150 mm mérési tartomány). Az eredményeket átlag ± standard hiba formájában közöltem, százalékos értékeléssel kiegészítve. A kezelések közötti különbségek statisztikai szignifikanciáját egyutas varianciaanalízissel (ANOVA) vizsgáltam, 0,05-ös szignifikanciaszint mellett. Az ANOVA alkalmazása lehetővé tette a dózisok közötti eltérések objektív kimutatását, míg a standard hiba feltüntetése az adatok megbízhatóságát és szóródását reprezentálta.

3.4 Regionális azbesztérintettség és keret az azbesztmentes városi átmenethez

Az azbesztmentes városi környezetre való átmenet összetett, többdimenziós folyamat, amelyet szabályozási keretek, technológiai innovációk, érintetti részvétel, valamint társadalmi-gazdasági feltételek egyaránt befolyásolnak. Az elemzés elméleti alapját a Geels munkái (Geels, 2002, 2011) nyomán alkalmazott többszintű perspektíva, azaz a Multi-Level Perspective (MLP) képezi, amely olyan átfogó elemzési keretet kínál, amely lehetővé teszi annak vizsgálatát, hogy a kormányzati szintek, az iparági struktúrák és a társadalmi folyamatok közötti kölcsönhatások miként segítik vagy éppen akadályozzák az átmenetet. Az azbeszt jelenlétének csökkentése a városi térben alapvető fontosságú a közegészségügyi kockázatok mérséklése és a fenntartható városi átalakulás elősegítése szempontjából. Az azbeszttartalmú régi épületállomány és infrastruktúra jelentős expozíciós veszélyt jelent a lakosság és a munkavállalók számára, ami összehangolt szakpolitikai beavatkozásokat és stratégiai intézkedéseket tesz szükségessé. Az átmenetelméleti megközelítés lehetővé teszi annak értelmezését, hogy a történeti beágyazottság és az intézményi merevség miként lassíthatja a strukturális változásokat.

3.4.1 Az MLP koncepcionális alkalmazása a kutatási modellben

A kutatási modell a Geels-féle MLP niche-, rezsim- és makrokörnyezeti szintjeire épül, amelyeket az azbesztmentes városi átmenet sajátos feltételrendszerére adaptáltam (Geels, 2002, 2011). Az első szinten a niche-innovációk és szociotechnikai kísérletek állnak, amelyek az azbesztmentes és fenntartható városfejlesztés motorjai. A második szint a rezsimdinamikát foglalja magában, beleértve a szabályozási kereteket, gazdasági struktúrákat és társadalmi normákat, amelyek befolyásolják a fennálló gyakorlatok fennmaradását vagy átalakulását. A harmadik szint a makrokörnyezeti tényezőket reprezentálja, például a globális fenntarthatósági trendeket, klímapolitikát és közegészségügyi prioritásokat. Az MLP alkalmazása lehetővé teszi a szintek közötti kölcsönhatások rendszerszintű elemzését. Az alsó szintű innovációk gyakran ellenállásba ütköznek a mélyen beágyazott rezsimstruktúrák részéről, miközben a makrokörnyezeti nyomások katalizátorként működhetnek az átalakulásban. Az egyszintű elemzési megközelítések kockázata, hogy figyelmen kívül hagyják a strukturális kölcsönhatásokat, ami redukcionista értelmezésekhez vezethet. Az MLP alkalmazásával a kutatás célja, hogy átfogó képet nyújtson az azbesztmentes városi átmenet feltételrendszeréről, feltárva a rendszerszintű akadályokat és előmozdító tényezőket.

3.4.2 A regionális különbségek szerepe az azbesztmentes átmenetben

Az azbesztmentes átmenet nem homogén módon zajlik le a különböző térségekben. A történeti ipari örökség, az épületállomány kora, az infrastruktúra jellege, valamint az intézményi kapacitások jelentős regionális eltéréseket eredményezhetnek. A NUTS (Nomenclature of Territorial Units for Statistics) az Európai Unió statisztikai célú területi osztályozási rendszere; a NUTS-2 ennek második területi szintjét jelöli. A NUTS-2 szintű területi bontás lehetővé teszi a makroszintű (landscape) és mezoszintű (rezsim) sajátosságok térbeli leképezését. A regionális különbségek empirikus megragadása azért indokolt, mert az MLP keretrendszerben a rezsimstruktúrák és pályafüggőségek gyakran területileg differenciáltak. Egyes régiókban erősebb lehet az ipari beágyazottság és az örökölt infrastruktúra jelenléte, míg más térségekben gyorsabb lehet az adaptáció és a niche-innovációk befogadása. A térbeli elemzés ezért nem pusztán leíró jellegű, hanem az átmeneti dinamika strukturális különbségeit is feltárja.

3.4.3 Az azbesztérintettségi integrált index kialakításának módszertana

A regionális különbségek kvantitatív megragadására taxonómiai indexképzési módszert (Oliinyk et al., 2023) alkalmaztam. A cél egy 0 és 1 közötti dimenziómentes mutató meghatározása volt, amely relatív módon fejezi ki az egyes magyarországi NUTS-2 régiók azbesztérintettségi szintjét. Az indikátorrendszer kialakítása során a *Mesothelioma malignum* incidenciaértékeit vettem figyelembe a Nemzeti Rákregiszter adatai alapján. A vizsgált időszak 2005-2020, az elemzési évek: 2005, 2010, 2015 és 2020. Az eltérő mértékegységű adatok standardizálása az alábbi képlet alapján történt:

$$Z_{ij} = \frac{x_{ij} - \bar{x}_{ij}}{\sigma_i}, \quad (1)$$

Ezt követően meghatároztam a referenciaértéket, amely az adott indikátorsorozat maximális standardizált értékének felelt meg.

$$Z_{0i} = \max z'_{ij}\ i \in I, \quad (2)$$

Az egyes régiók referenciaértéktől való eltérését Euklideszi távolsággal mértem:

$$d_{0i} = \sqrt{\sum_{i,j=1}^{n,m} (Z_{ij} - Z_{0i})^2}, \quad (3)$$

Az integrált index számítása az alábbi módon történt:

$$K_i = 1 - \frac{d_{0i}}{d_0},\ d_0 = \overline{d_0} + 2 \cdot \sigma_0, \sigma_0 = \sqrt{\frac{\sum(d_{0i} - \overline{d_0})^2}{n}}, \quad (4)$$

ahol d_0 az Euklideszi távolságok átlagának és kétszeres szórásának összege. Az így kapott K_i érték 0 és 1 közötti dimenziómentes mutató, ahol a magasabb érték nagyobb relatív azbesztérintettséget jelez.

3.4.4 Az index értelmezése az MLP keretrendszerében

Az integrált regionális index nem pusztán statisztikai eszköz, hanem az MLP-keretrendszer empirikus operacionalizálásának egyik eleme. A magasabb indexértékkel rendelkező régiók esetében feltételezhető a rezsimbeágyazottság erősebb jelenléte, a történeti pályafüggőség dominanciája és a strukturális akadályok fennmaradása. Az alacsonyabb indexértékű térségekben nagyobb lehet a niche-innovációk adaptációs potenciálja és a szabályozási beavatkozások hatékonysága. Az index lehetővé teszi a regionális egyenlőtlenségek azonosítását, hozzájárul a célzott szakpolitikai beavatkozások megalapozásához.

4. EREDMÉNYEK ÉS ÉRTÉKELÉSÜK

Disszertációm jelen fejezetében a kutatás empirikus eredményeit mutatom be és értelmezem a korábban ismertetett elméleti keret és módszertani megközelítés tükrében. Az eredmények több vizsgálati szinten kerülnek bemutatásra: az anyagszerkezeti és degradációs elemzések, a vízminőségi paraméterek változásai, a növényélettani reakciók, valamint a regionális azbesztérintettségi index számításának eredményei egymásra épülő logikai struktúrában kerülnek értékelésre. A fejezet célja nem csupán az adatok leíró ismertetése, hanem azok oksági és rendszerszintű értelmezése, különös tekintettel az azbesztmentes városi átmenet feltételrendszerére és a fenntarthatósági implikációkra. Az eredmények értékelése során hangsúlyt helyezek a dózis–hatás összefüggésekre, a területi különbségek jelentőségére, valamint a szakpolitikai és környezetbiztonsági következtetések levonására.

4.1 Az azbesztrostok környezeti mobilitásának vizsgálata a talaj-víz rendszerben

Az áttekintő kutatásomban az azbesztrostok környezeti viselkedését vizsgáltam a víz-talaj kontinuum mentén, különös tekintettel a mobilitási mechanizmusokra és az expozíciós útvonalak komplexitására. Elemzésem célja az volt, hogy rendszerszintű keretben értelmezzem az azbeszt környezeti transzportfolyamatait, és rámutassak azokra a tudományos hiányterületekre, amelyek a hagyományosan légúti expozícióra fókuszáló megközelítésekből adódnak. Eredményeim alapján megállapítottam, hogy az azbeszt környezeti kockázata nem korlátozható a levegőben történő terjedés vizsgálatára. A vízi és talajközegben zajló folyamatok jelentős mértékben befolyásolják a rostok mobilitását, elérhetőségét és potenciális ökotoxikológiai hatásait. Az azbesztrostok fizikai-kémiai tulajdonságai – különösen a rostgeometria, felületi töltés és ásványtani összetétel – meghatározó szerepet játszanak a talaj–víz rendszerben történő mozgásukban.

Elemzésem során rámutattam arra, hogy a víz pH-ja, ionösszetétele és kolloidális frakciója jelentősen befolyásolja az azbesztrostok diszperzióját és aggregációját. A talajkolloidokhoz való kötődés, valamint az oldott szerves anyagokkal való kölcsönhatás módosíthatja a rostok transzportját, ezáltal befolyásolva a felszíni és felszín alatti vizekbe történő bejutás lehetőségét. Megállapítottam, hogy a talaj nem csupán passzív közeg, hanem aktív geokémiai rendszerként működik, amely képes mind mobilizálni, mind immobilizálni az azbesztrostokat. Külön hangsúlyt fektettem az azbesztcement termékek környezeti degradációjának vizsgálatára. Megállapítottam, hogy az elöregedett, málló azbesztcement szerkezetek potenciális diffúz forrásként viselkednek, amelyekből a rostok fokozatosan a talaj–víz rendszerbe juthatnak. Ez a folyamat különösen releváns a csapadékvíz-lefolyás és a felszíni beszivárgás esetében, ahol a fizikai erózió és kémiai oldódási mechanizmusok együttesen járulnak hozzá a környezeti terheléshez.

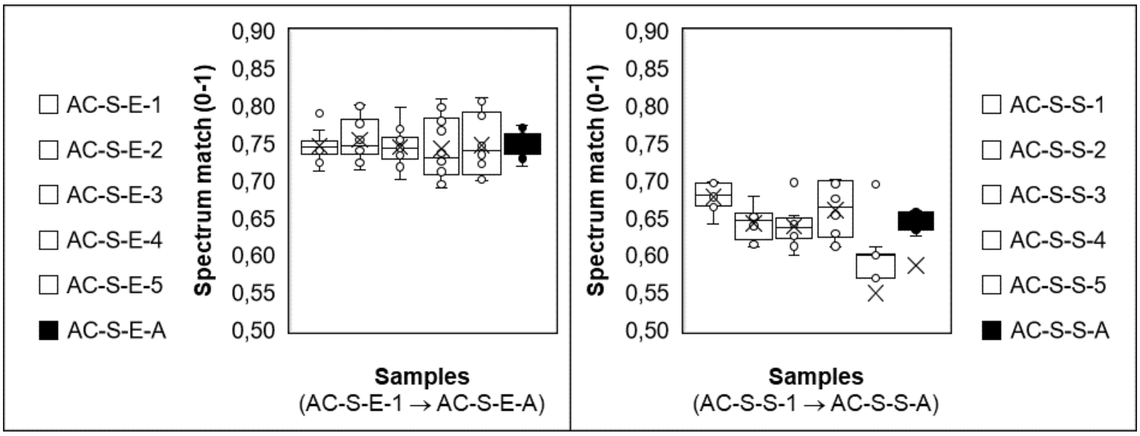
Elemzésem egyik fontos megállapítása, hogy az azbeszt környezeti transzportja nem lineáris folyamat, hanem komplex, többtényezős rendszer eredménye. A talaj–víz interakciók, a hidrológiai viszonyok és az anyag degradációs állapota együttesen határozzák meg az expozíciós pályákat. Rámutattam arra is, hogy a jelenlegi szakirodalomban jelentős hiány mutatkozik a növényi és mikrobiális kölcsönhatások vizsgálatában, ami korlátozza az ökoszisztéma-szintű kockázatbecslések megbízhatóságát. Az azbeszt környezeti viselkedésének értelmezése csak integrált, holisztikus megközelítésben lehetséges. A levegő, a talaj és a víz közötti kölcsönhatások figyelembevétele nélkül az expozíciós modellek hiányosak maradnak. Eredményeim megerősítik, hogy a környezeti mobilitás vizsgálata kulcsfontosságú az azbeszt hosszú távú ökotoxikológiai és közegészségügyi kockázatainak pontosabb meghatározásához, és tudományos alapot teremtenek a komplex környezeti kockázatértékelési modellek továbbfejlesztéséhez.

4.2 Krizotil-azbeszt kimutatása AC-mintákból

4.2.1 Krizotil-azbeszt kimutatása hullámos azbesztcement lemezekből

A környezeti hatásoknak kitett hullámos azbesztcement lemezminták esetében az átlagos krizotil-azonosítási arány 0,747193 (1) volt (**1. ábra**). A legmagasabb spektrális egyezés (0,810265 (1)) az AC-S-E-5 minta negyedik mérési pontján jelentkezett, míg a legalacsonyabb detektálási értéket (0,689985 (1)) az AC-S-E-4 minta hatodik mérési pontján rögzítettem. A minták közül a legmagasabb átlagos krizotil-detektálási értéket (0,754616 (1)) az AC-S-E-2 minta mutatta, míg a legalacsonyabbat (0,742706 (1)) az AC-S-E-4 esetében tapasztaltam. A környezeti hatásoktól elzárt (szeparált) minták esetében az átlagos krizotil-azonosítási arány 0,645437 (1) volt. A legmagasabb egyezést (0,701221 (1)) az AC-S-S-4 minta negyedik mérési pontján, míg a legalacsonyabb értéket (0,569845 (1)) az AC-S-S-5 minta nyolcadik mérési pontján mértem. A szeparált minták közül a legmagasabb átlagos érték (0,678396 (1)) az AC-S-S-1 mintánál, a legalacsonyabb (0,605309 (1)) az AC-S-S-5 mintánál jelentkezett.

A teljes mintasorozat összehasonlítási aránya 115,8% volt. Az exponált minták átlagtól való átlagos eltérése 0,00634 (1), míg a szeparált minták esetében 0,00429 (1) értéket mutatott. A hullámos azbesztcement minták esetében jól látható különbség mutatkozott a krizotil spektrális illeszkedésében: a környezeti hatásoknak kitett (E) minták magasabb, megközelítőleg 0,75 (1) körüli átlagértéket mutattak, míg a szeparált (S) minták nagyobb variabilitást és körülbelül 0,65 (1) körüli középértéket jeleztek. Mivel a vizsgált minták azbesztcement-mátrixon alapulnak, a spektrumokban 1400 cm^{-1} környezetében széles csúcs jelent meg. Ez a cementkeverékben jelen lévő kalcium-szilikát-hidrát, kalcium-szulfát és kalcium-karbonát C–O kötéseinek nyúlási rezgéseivel magyarázható. Két jellegzetes spektrális tartomány különíthető el: az O-H nyúlási rezgések a 3700-3500 cm^{-1} tartományban, míg a rácsrezgések a 1200–500 cm^{-1} tartományban jelennek meg.



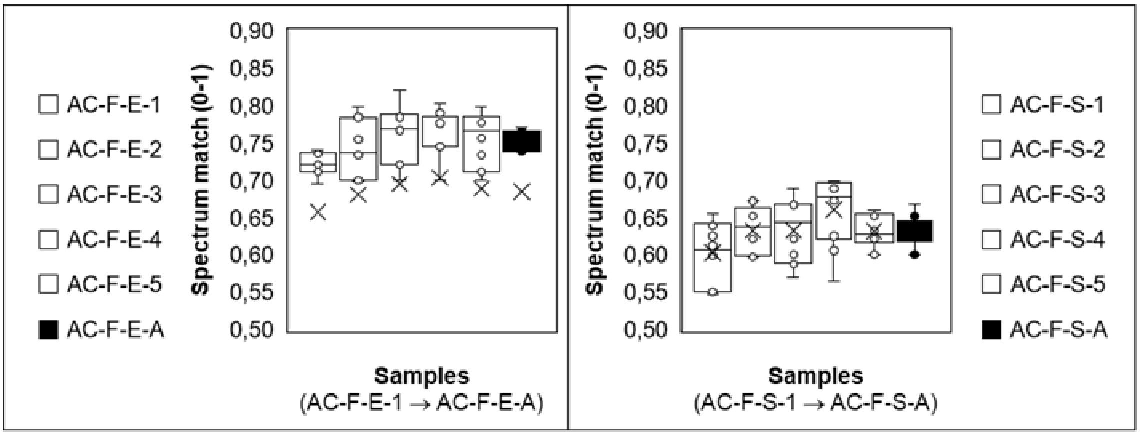
1. ábra: A krizotil detektálás összehasonlító elemzése exponált (E) és szeparált (S) minták esetében (hullámos azbesztcement lemezek)

Forrás: saját számítás, saját szerkesztés

4.2.2 Krizotil-azbeszt kimutatása sík azbesztcement lemezekből

A környezeti hatásoknak kitett sík azbesztcement minták átlagos krizotil-azonosítási aránya 0,753552 (1) volt. A legmagasabb egyezést (0,820001 (1)) az AC-F-E-3 minta tizedik mérési pontján, míg a legalacsonyabb értéket (0,694962 (1)) az AC-F-E-1 minta második mérési pontján mértem. A legmagasabb átlagos detektálási értéket (0,773262 (1)) az AC-F-E-4 minta, a legalacsonyabbat (0,723031 (1)) az AC-F-E-1 minta mutatta (**2. ábra**). A szeparált minták átlagos krizotil-azonosítási aránya 0,632309 (1) volt. A legmagasabb egyezést (0,698857 (1)) az AC-F-S-4 minta ötödik mérési pontján, a legalacsonyabb értéket (0,548113 (1)) az AC-F-S-1 minta negyedik mérési pontján mértem. A legmagasabb átlagos értéket (0,660611 (1)) az AC-F-S-4, míg a legalacsonyabbat (0,602867 (1)) az AC-F-S-1 minta mutatta. A teljes mintasorozat összehasonlítási aránya 119,2% volt. Az exponált minták átlagtól való átlagos eltérése 0,00540 (1), míg a szeparált minták esetében 0,00801 (1) volt.

A környezeti hatásoknak kitett sík lemezminták esetében a krizotil spektrális illeszkedése kifejezettebb volt, a krizotil könnyebben azonosíthatónak és nagyobb egyezési arányúnak bizonyult. A szeparált minták nagyobb variabilitást mutattak, 0,65 (1) alatti középértékkel. A krizotil jellegzetes csúcsai a 3700–3500 cm^{-1} és 1200–500 cm^{-1} tartományban jelentkeztek. A hullámos lemezekhez hasonlóan 1400 cm^{-1} környezetében itt is széles csúcs volt megfigyelhető az azbesztcement-mátrix jelenléte miatt.

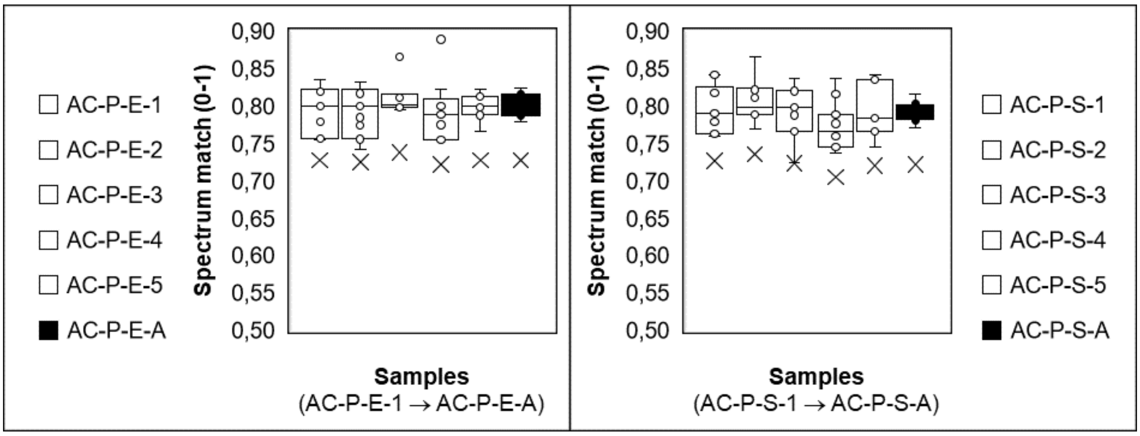


2. ábra: A krizotil detektálás összehasonlító elemzése exponált (E) és szeparált (S) minták esetében (sík azbesztcement lemezek)

Forrás: saját számítás, saját szerkesztés

4.2.3 Krizotil-azbeszt kimutatása azbesztcement csövekből

A környezeti hatásoknak kitett azbesztcement csőminták átlagos krizotil-azonosítási aránya 0,800607 (1) volt. A legmagasabb egyezést (0,888831 (1)) az AC-P-E-4 minta ötödik mérési pontján, míg a legalacsonyabb értéket (0,742121 (1)) az AC-P-E-2 minta nyolcadik mérési pontján mértem. A legmagasabb átlagos értéket (0,811325 (1)) az AC-P-E-3, a legalacsonyabbat (0,793972 (1)) az AC-P-E-4 minta mutatta (**3. ábra**). A szeparált minták esetében az átlagos krizotil-azonosítási arány 0,794369 (1) volt. A legmagasabb egyezést (0,865548 (1)) az AC-P-S-2 minta hetedik mérési pontján, a legalacsonyabb értéket (0,724580 (1)) az AC-P-S-3 minta kilencedik mérési pontján mértem.



3. ábra: A krizotil detektálás összehasonlító elemzése exponált (E) és szeparált (S) minták esetében (azbesztcement csövek)

Forrás: saját számítás, saját szerkesztés

A legmagasabb átlagos értéket (0,808812 (1)) az AC-P-S-2, míg a legalacsonyabbat (0,775610 (1)) az AC-P-S-4 minta mutatta. A teljes mintasorozat összehasonlítási aránya

100,8% volt. Az exponált minták átlagtól való átlagos eltérése 0,00617 (1), míg a szeparált minták esetében 0,00438 (1) volt. A környezeti hatásoknak kitett csőminták esetében a spektrális illeszkedés nagyobb egyöntetűséget mutatott, a krizotil könnyebben detektálható és konzisztensebb volt. Az exponált minták trendje 0,80 (1), míg a szeparált mintáké 0,79 (1) körül alakult. A krizotil jellegzetes csúcsai itt is a 3700–3500 cm^{-1} és 1200–500 cm^{-1} tartományban jelentek meg. A 1400 cm^{-1} körüli széles csúcs az azbesztcement-mátrix jelenlétére utal.

4.3 Vízanalitikai vizsgálatok eredményei

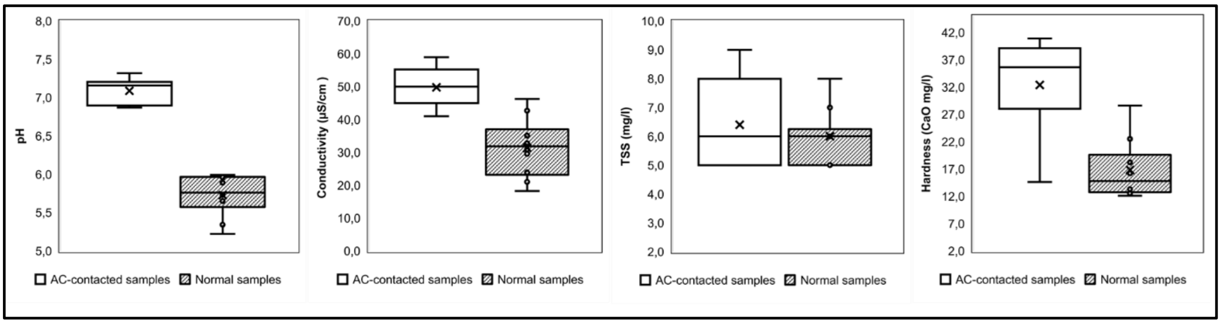
4.3.1 Az azbesztcementtel érintkezett és normál csapadékvíz vízminőségi jellemzői

A vizsgálat jelentős különbségeket tárt fel a normál csapadékvíz és az azbesztcement (AC) tetőfelülettel érintkezett csapadékvíz vízminőségi paraméterei között. Az AC felületekről gyűjtött víz pH-értéke szignifikánsan magasabb volt, átlagosan 7,09 ± 0,162, szemben a normál csapadékvíz 5,73 ± 0,265 értékével, ami 23,8%-os növekedést jelent. Ez arra utal, hogy a tetőfedő anyag lúgosító hatással rendelkezik, amely valószínűsíthetően kalciumtartalmú vegyületek kioldódásával magyarázható. A megváltozott pH-érték befolyásolhatja más szennyezőanyagok oldhatóságát és mobilitását. Hasonló tendencia volt megfigyelhető az elektromos vezetőképesség esetében, amely az összes oldott ion koncentrációját jelzi. Az AC-vel érintkezett minták átlagos vezetőképessége 49,8 ± 6,00 µS/cm volt, míg a normál csapadékvíz esetében 31,4 ± 8,88 µS/cm értéket mértem. Ez arra utal, hogy a tetőfedő anyag további oldott ionokat juttat a vízbe, feltehetően ásványi komponensek és egyéb anyagok kioldódása révén.

Az összes lebegőanyag-tartalom (TSS) vizsgálata csekély növekedést mutatott az AC tetőkről származó mintákban: az átlagos érték 6,40 ± 1,51 mg/L volt, szemben a normál csapadék 6,00 ± 0,94 mg/L értékével. A 6,67%-os emelkedés arra utal, hogy a tetőfedő anyag nem járul hozzá jelentős mértékben a részecsketerheléshez. Ugyanakkor még egy kisebb TSS-növekedés is hatással lehet a vízminőségre, mivel a lebegőanyagok más szennyezőket – például nehézfémeket – is szállíthatnak. A vízkeménység esetében kifejezetten jelentős, 92,6%-os növekedést mértem az AC felületekről származó mintákban a normál csapadékhoz viszonyítva. Az AC minták átlagos keménysége 32,4 ± 9,75 CaO mg/L volt, míg a normál csapadék esetében 16,8 ± 5,35 CaO mg/L értéket mértem (**4. ábra**). A keménység ilyen mértékű emelkedése valószínűleg a cementmátrixból kioldódó kalcium- és magnéziumvegyületeknek tulajdonítható. A megnövekedett vízkeménység vízkőképződéshez vezethet a csőhálózatban, valamint csökkentheti a tisztítószerek hatékonyságát.

A teljes szerves széntartalom (TOC) és az oxigénigény-paraméterek elemzése további információt nyújtott az AC tetőfedés vízminőségre gyakorolt hatásáról. A TOC-értékek 40,9%-

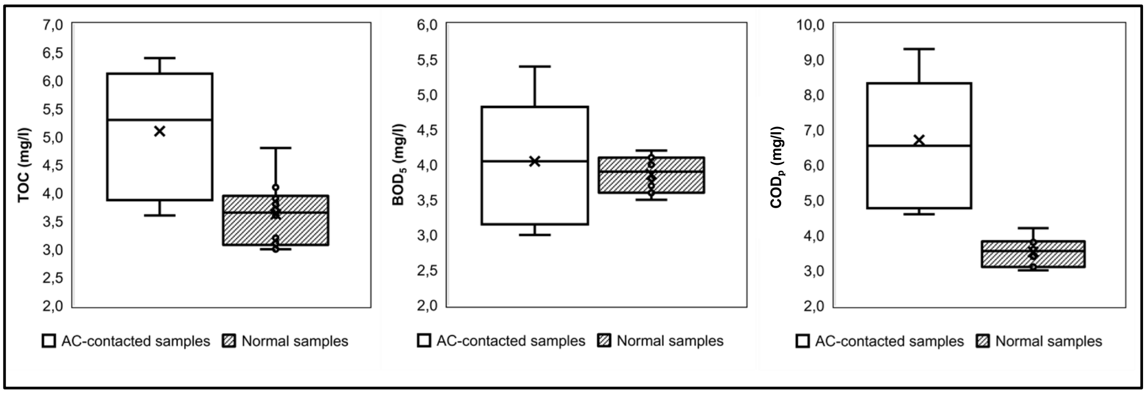
os növekedést mutattak az AC felülettel érintkezett mintákban, amelyek átlagosan 5,10 ± 1,08 mg/L értéket értek el, szemben a normál csapadék 3,62 ± 0,57 mg/L értékével. Ez a jelentős növekedés arra utal, hogy az AC-vel érintkezett vízben több szerves anyag található, amely a tetőfelületen felhalmozódott szerves részecskék bemosódásából vagy szerves komponensek kioldódásából eredhet. A magasabb TOC-szint gyakran fokozott mikrobiális aktivitással társul, ami különösen tárolás vagy elosztás esetén befolyásolhatja a víz minőségét.



4. ábra: A vízminták összehasonlító elemzése pH, vezetőképesség, TSS és keménység paraméterek alapján

Forrás: saját számítás, saját szerkesztés

A biokémiai oxigénigény (BOI_5) értékek enyhe, 4,92%-os növekedést mutattak az AC-vel érintkezett mintákban. Az AC minták átlagos BOI_5 értéke 4,05 ± 0,88 mg/L volt, míg a normál csapadék esetében 3,86 ± 0,25 mg/L értéket mértem. Bár a növekedés mérsékelt, arra utal, hogy az AC-vel érintkezett vízben jelen lévő szerves anyag biológiailag könnyebben hozzáférhető lehet, illetve a mikrobiális terhelés enyhén magasabb lehet, ami befolyásolhatja a víz öntözési vagy ivóvíz-célú felhasználhatóságát. A legkifejezettebb változás a kémiai oxigénigény (KOI) esetében volt megfigyelhető, ahol az AC tetővel érintkezett minták KOI-értékei jelentősen meghaladták a normál csapadék mintákét. Ez a növekedés nagyobb mennyiségű oxidálható szerves és szervetlen anyag jelenlétére utal, amelyek valószínűleg a tetőfedő anyagból vagy a felületen lerakódott szennyezőkből származnak. A magas KOI-érték potenciális szennyezettséget jelez, és megfelelő kezelés hiányában kedvezőtlen hatást gyakorolhat a befogadó vízi ökoszisztémákra (**5. ábra**).



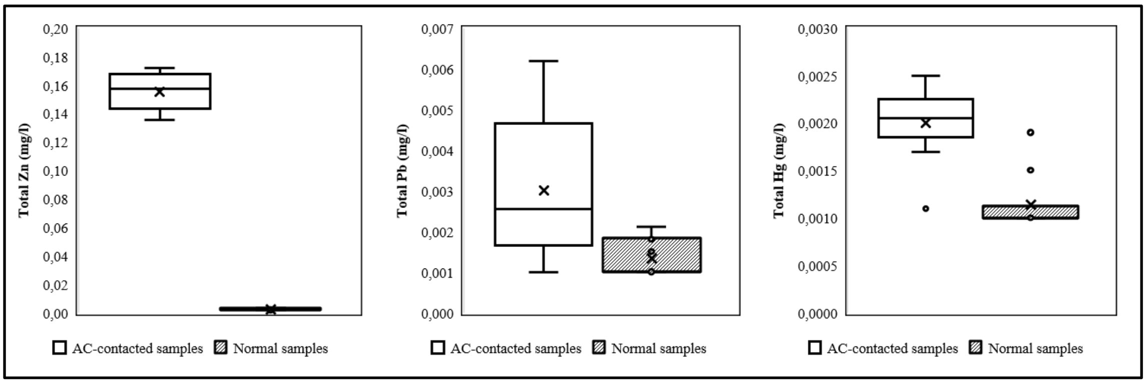
5. ábra: A vízminták összehasonlító elemzése a TOC-, BOI$_5$- (az ábrán: BOD$_5$) és KOI$_p$- (az ábrán: COD$_p$) paraméterek alapján

Forrás: saját számítás, saját szerkesztés

4.3.2 Az azbesztcementtel érintkezett és normál csapadékvíz nehézfém-szennyezettsége

A csapadékvíz fémkoncentrációinak elemzése jelentős különbségeket mutatott a normál minták és az azbesztcement (AC) tetőfelülettel érintkezett minták között. A cink (Zn), az ólom (Pb) és a higany (Hg) koncentrációja számottevően megnövekedett az AC-vel érintkezett mintákban, ami arra utal, hogy a tetőfedő anyag ezen nehézfémek potenciális forrásaként működik. A normál csapadékvízben a teljes cinkkoncentráció igen alacsony volt, átlagosan 0,0024 ± 0,0005 mg/L, amely közel esett a kimutatási határértékhez. Ez arra utal, hogy fémfelületekkel való érintkezés nélkül a csapadékvíz cinktartalma minimális, elsősorban légköri eredetű szennyezésből származik. Ezzel szemben az AC tetőfelülettel érintkezett csapadékvízben a cink koncentrációja drasztikusan, 0,155 ± 0,012 mg/L értékre emelkedett. Ez a jelentős növekedés valószínűsíthetően az azbesztcement anyagában jelen lévő cinkből ered, amely származhat a gyártási folyamatból, illetve a tetőfelületen felhalmozódott légköri szennyeződésekből. A cink koncentrációjának növekedése azt mutatja, hogy az AC-tetőfedés érdemben módosíthatja a gyűjtött csapadékvíz cinktartalmát. A mért 0,155 mg/L koncentráció azonban öntözési célú felhasználás esetén önmagában nem feltétlenül kedvezőtlen, mivel a cink esszenciális mikroelem, és ebben a koncentrációtartományban a növények számára kedvező hatású is lehet.

Az elemzés kimutatta, hogy az AC tetőfelület jelentősen növelte az ólom és a higany koncentrációját is a gyűjtött csapadékvízben (**6. ábra**). Az ólom koncentrációja 127,3%-kal emelkedett: a normál mintákban mért 0,00133 ± 0,0004 mg/L értékről 0,003 ± 0,002 mg/L-re nőtt az AC-vel érintkezett minták esetében. Az ólom kifejezetten toxikus fém, különösen gyermekek esetében, és jelenléte a vízben szigorúan szabályozott. A megemelkedett ólomszint arra utal, hogy az azbesztcement anyag tartalmazhat ólmot – akár gyártási szennyeződésként, akár környezeti lerakódás következtében. Ez különösen indokolttá teszi az ilyen felületekről gyűjtött csapadékvíz rendszeres monitorozását, különösen háztartási felhasználás esetén.



6. ábra: A vízminták összehasonlító elemzése nehézfém-szennyezettség (Zn, Pb, Hg) tekintetében

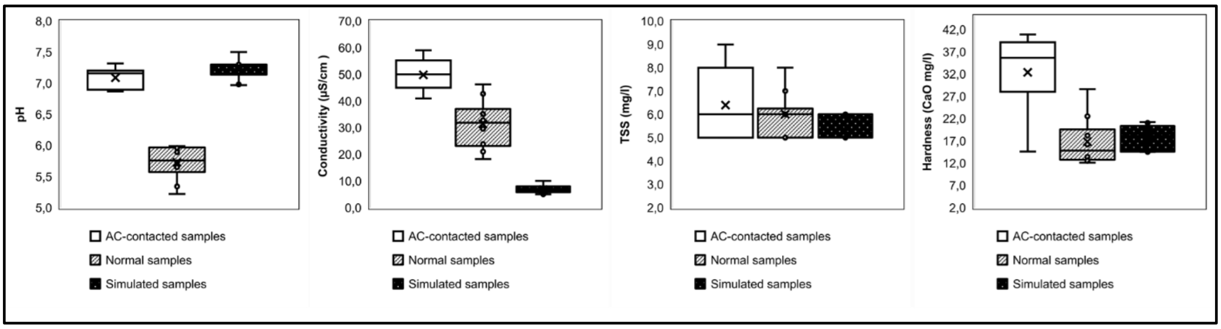
Forrás: saját számítás, saját szerkesztés

Hasonló tendencia volt megfigyelhető a higany esetében is: az AC-vel érintkezett mintákban a koncentráció 75,4%-kal növekedett, a normál csapadékvíz 0,0011 ± 0,0003 mg/L értékéről 0,002 ± 0,0004 mg/L-re. Bár az abszolút koncentrációk alacsonyak, a relatív növekedés jelentős, ami felveti a hosszú távú expozíció és bioakkumuláció lehetőségét.

4.3.3 Az azbesztcementtel érintkezett szimulált vízminták vízminőségi jellemzői

A szimulált vízminták elemzése lehetőséget adott annak vizsgálatára, hogy az azbesztcement (AC) tetőfedés miként befolyásolja a vízminőségi paramétereket semleges, illetve enyhén savas kiindulási körülményekhez viszonyítva. A szimulált minták pH-értéke 7,21 ± 0,154 volt, amely összhangban áll az AC-vel érintkezett természetes csapadékvíz esetében tapasztalt pH-növekedési tendenciával. Ez azt jelzi, hogy az alkalikus kémhatású azbesztcement anyag jelentős semlegesítő hatással rendelkezik, különösen enyhén savas közegben. Ugyanakkor semleges kiindulási állapot esetén a pH-eltolódás minimális volt, ami arra utal, hogy az AC hatása savas víz esetében kifejezettebb. A szimulált minták elektromos vezetőképessége 6,80 ± 1,55 μS/cm értékre növekedett a desztillált víz közel nulla értékéhez képest. Ez az emelkedés arra utal, hogy az AC tetőfedő anyag további oldott ionokat juttat a vízbe, valószínűsíthetően ásványi komponensek és egyéb oldható anyagok kioldódása révén. Az iontartalom növekedése befolyásolhatja a víz ionegyensúlyát és felhasználhatóságát.

Az összes lebegőanyag-tartalom (TSS) a szimulált minták esetében 5,30 ± 0,48 mg/L volt, amely összehasonlítható az AC tetővel érintkezett csapadékvíz minták értékeivel (**7. ábra**). Ez az összhang arra utal, hogy az AC tetőfedés potenciális forrása lehet a részecsketerhelésnek, amely más szennyezőanyagok hordozójaként is működhet, illetve befolyásolhatja a víz átlátszóságát és minőségét.

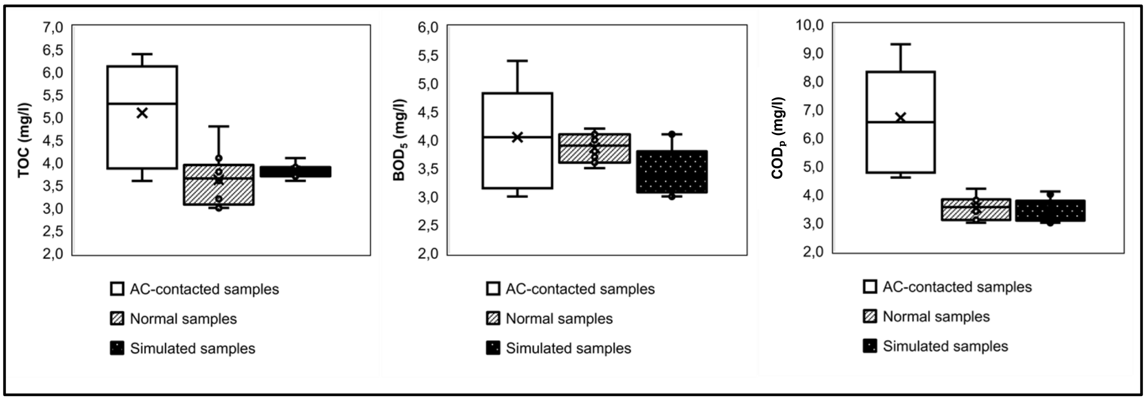


7. ábra: A szimulált minták összehasonlító elemzése pH, vezetőképesség, TSS és keménység paraméterek alapján

Forrás: saját számítás, saját szerkesztés

A szimulált vízminták mért teljes keménysége 17,7 ± 2,67 CaO mg/L volt, amely a normál csapadék tartományán belül helyezkedik el. Ez arra utal, hogy az AC tetőfedő anyag képes növelni a víz keménységét, feltehetően kalciumvegyületek oldódása révén. A megnövekedett keménység befolyásolhatja a víz csőhálózattal, tisztítószerekkel és egyéb folyamatokkal való kölcsönhatását, ami fontos szempont az AC tetők vízminőségre gyakorolt hatásának értékelésekor.

A szimulált és az AC tetővel érintkezett természetes csapadékminták szervesanyag-tartalmának és oxigénigényének összehasonlító elemzése további információt szolgáltatott az anyag hatásmechanizmusáról. A teljes szerves széntartalom (TOC) a szimulált mintákban 3,82 ± 0,14 mg/L volt, amely alacsonyabb, mint az AC-vel érintkezett csapadékvíz minták 5,10 ± 1,08 mg/L átlagértéke (**8. ábra**). Ez a különbség arra utal, hogy az AC tetőfedés további szerves anyaggal járulhat hozzá a víz összetételéhez, feltehetően kioldódás vagy a tetőfelületen felhalmozódott szennyeződések révén. A szimulált minták alacsonyabb TOC-értéke – amelyek nem voltak kitéve a teljes környezeti terhelésnek – alátámasztja azt a feltételezést, hogy a tetőfelületen lerakódott légköri szennyezők jelentős szerepet játszanak a TOC-emelkedésben.



8. ábra: A szimulált minták összehasonlító elemzése a TOC-, BOI$_5$- (az ábrán: BOD$_5$) és KOI$_p$- (az ábrán: COD$_p$) paraméterek alapján

Forrás: saját számítás, saját szerkesztés

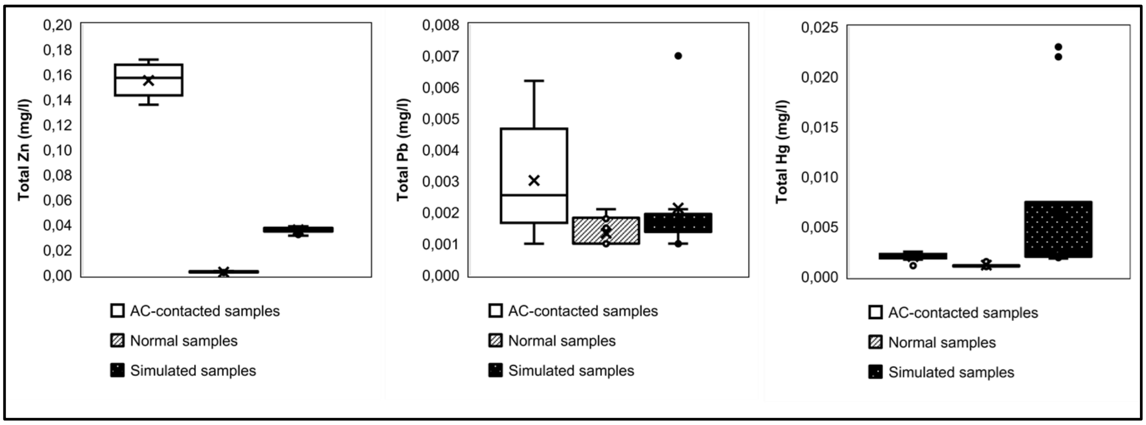
A szimulált csapadékvíz minták átlagos biokémiai oxigénigénye (BOI$_5$) 3,44 ± 0,42 mg/L volt, amely kissé alacsonyabb, mint a normál csapadék és az AC-vel érintkezett minták értékei. Az AC-vel érintkezett minták enyhe BOI$_5$-növekedése arra utal, hogy az ott jelenlévő szerves anyag biológiailag könnyebben hozzáférhető, illetve magasabb mikrobiális terhelés jellemezheti, valószínűsíthetően a tetőfedő anyagból származó szerves komponensek miatt. A szimulált minták BOI$_5$-értéke, amely közel áll a normál csapadékéhoz, azt jelzi, hogy külső szerves terhelés hiányában a biokémiai oxigénigény viszonylag stabil marad, ezáltal kiemelve az AC tetők szerepét a szerves szennyezés növekedésében.

A szimulált minták kémiai oxigénigénye (KOI$_p$) 3,43 ± 0,41 mg/L volt, amely gyakorlatilag megegyezett a normál csapadék értékeivel. Ezzel szemben az AC-vel érintkezett minták KOI$_p$-értéke szignifikánsan magasabb, 6,71 ± 1,78 mg/L volt. Ez a jelentős növekedés nagyobb mennyiségű oxidálható szerves és szervetlen anyag jelenlétére utal, amelyek valószínűleg az azbesztcement anyagból, illetve a tetőfelületen felhalmozódott környezeti szennyeződésekből származnak. A szimulált minták viszonylag alacsony KOI$_p$-értéke arra utal, hogy külső szennyezők hiányában maga az AC anyag önmagában nem okoz jelentős KOI-emelkedést, ugyanakkor környezeti tényezőkkel kombinálva hozzájárulhat az oxidálható komponensek növekedéséhez.

4.3.4 A szimulált azbesztcementtel érintkezett minták nehézfém-szennyezettsége

A szimulált vízminták nehézfém-koncentrációinak elemzése – az azbesztcement (AC) tetőfelülettel érintkezett, illetve a normál csapadékmintákhoz viszonyítva – fontos következtetéseket tett lehetővé az AC anyag vízminőségre gyakorolt hatásáról (**9. ábra**). A szimulált minták átlagos teljes cinkkoncentrációja 0,0361 ± 0,002 mg/L volt, amely jelentősen meghaladja a normál csapadékban jellemző cinkszinteket. Ugyanakkor az AC-vel érintkezett természetes csapadékmintákban a cink koncentrációja ennél is lényegesen magasabb, átlagosan 0,155 ± 0,012 mg/L értéket mutatott. Ez a számottevő növekedés arra utal, hogy az AC tetőfedő anyag jelentős cinkforrásként működik, feltehetően a cementmátrixból történő kioldódás, illetve a gyártás során alkalmazott cinktartalmú adalékanyagok jelenléte miatt.

A szimulált minták emelkedett – bár az AC-vel érintkezett mintákhoz képest alacsonyabb – cinkkoncentrációja azt jelzi, hogy az AC anyag környezeti szennyezők jelenléte nélkül is képes cinket kibocsátani a vízbe. Ez rámutat arra, hogy az azbesztcement tetőfedés önmagában is hozzájárulhat a gyűjtött csapadékvíz cinktartalmának növekedéséhez. A mért koncentráció azonban önmagában nem tekinthető egyértelmű környezeti vagy humánegészségügyi kockázatnak, mivel öntözési felhasználás esetén a cink esszenciális mikroelemként kedvező szerepet is betölthet. A szimulált minták átlagos ólomkoncentrációja 0,0021 ± 0,0017 mg/L volt, amely kissé magasabb a normál csapadék értékeinél, ugyanakkor alacsonyabb az AC-vel érintkezett mintákban mért koncentrációknál. Ez arra utal, hogy az AC anyag bizonyos mértékben hozzájárulhat az ólom jelenlétéhez, de ennek szerepe kevésbé domináns, mint a cink esetében.



9. ábra: A szimulált minták összehasonlító elemzése a teljes Zn-, Pb- és Hg-koncentrációk alapján

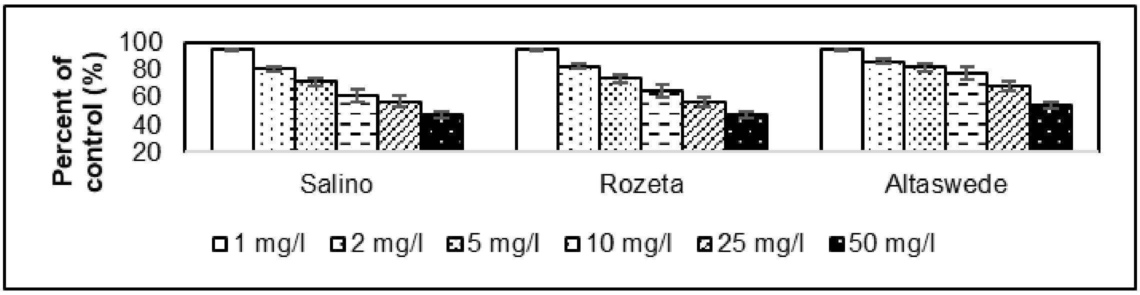
Forrás: saját számítás, saját szerkesztés

A szimulált minták enyhe növekedése azt jelzi, hogy az AC anyagból korlátozott mértékű ólomkioldódás történhet, míg az AC-vel érintkezett természetes csapadékvízben tapasztalt nagyobb koncentrációnövekedés valószínűsíthetően a tetőfelületen felhalmozódott környezeti ólomterhelés következménye. Ez hangsúlyozza annak fontosságát, hogy az AC tetőkről gyűjtött csapadékvíz ólomtartalmának értékelésekor az anyagösszetételt és a környezeti expozíciót egyaránt figyelembe kell venni. A szimulált minták teljes higanykoncentrációja 0,0063 ± 0,009 mg/L volt, amely számottevően meghaladja a normál csapadékban jellemző értékeket. Ez az emelkedett koncentráció arra utal, hogy a higany jelen lehet az AC anyagban, illetve bizonyos körülmények között felszabadulhat abból. A mért értékek nagy variabilitása azonban arra enged következtetni, hogy a higany forrásainak és viselkedésének pontos megértéséhez további vizsgálatok szükségesek.

4.4 Csíráztatásos és korai vegetációs kísérletek eredményei

4.4.1 A *Trifolium pratense* vizsgálatának eredményei

A kontroll csírázási arány 88,0% volt a Salino, 86,0% a Rozeta és 90,0% az Altaswede fajta esetében. A *Trifolium pratense* magok csírázása 25,0 °C-on átlagosan 88,0 ± 2,00%-nak adódott. Már 1 mg/L dóziskoncentráció esetén csökkenés volt megfigyelhető: a csírázási arány 82,0%-ra (Salino), 80,0%-ra (Rozeta) és 84,0%-ra (Altaswede) mérséklődött. Az 50 mg/L dózis hatására a csírázási arány 48,0% (Salino), 46,0% (Rozeta) és 58,0% (Altaswede) értékre csökkent, ami a kontrollcsoporthoz viszonyítva 54,5%, 53,5% és 64,4%-nak felelt meg. A teljes vizsgált csoport átlaga 93,2 ± 0,16% volt 1,00 mg/L esetén, míg 50 mg/L koncentrációnál 57,5 ± 6,04% (kontroll %-ában). A két dózisszint közötti átlagos különbség −35,7 ± 5,90% volt. A determinációs együttható (R^2) Salino esetében 0,9840, Rozeta esetében 0,9974, Altaswede esetében 0,9313 volt, ami erős dózis–válasz összefüggést jelez (**10. ábra**).

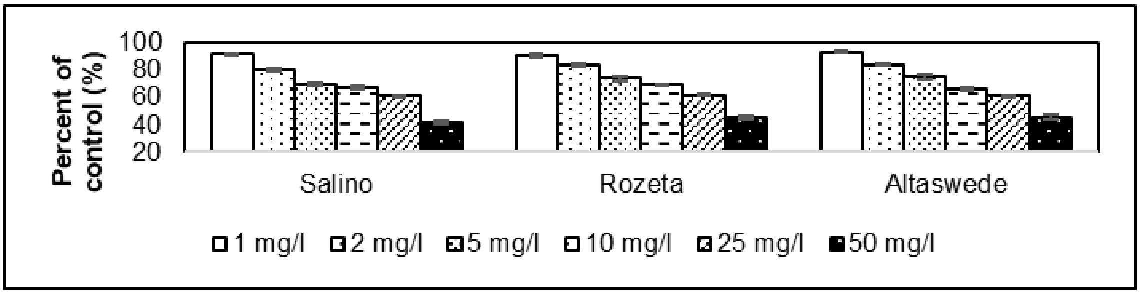


10. ábra: A Trifolium pratense (Salino, Rozeta, Altaswede) fajták gyökérhosszának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására. Mind az 1 mg/L, mind az 50 mg/L koncentráció esetében szignifikáns csökkenés volt kimutatható ($p \leq 0,05$), a nagyobb dózis erőteljesebb gátló hatással.

Forrás: saját számítás, saját szerkesztés

Az 1 mg/L dózis a gyökérhossz 95,2%-ra történő csökkenését eredményezte a kontrollhoz képest Salino esetében, 95,7%-ra Rozeta és 95,5%-ra Altaswede esetében. Az 50 mg/L dózis hatására a gyökérhossz 47,6%-ra (Salino), 47,8%-ra (Rozeta) és 54,5%-ra (Altaswede) csökkent. Ez átlagosan 95,4 ± 0,21%-os (1,00 mg/L), illetve 50,0 ± 3,94%-os (50,0 mg/L) értéket jelentett. Az R^2 értékek 0,9791 (Salino), 0,9941 (Rozeta) és 0,9611 (Altaswede) voltak. Az 1 mg/L dózis a hajtásmagasság 91,3%-ra (Salino), 90,5%-ra (Rozeta) és 93,2%-ra (Altaswede) történő csökkenését eredményezte a kontrollcsoporthoz viszonyítva (**11. ábra**). Az

50 mg/L dózis esetén a hajtásmagasság 41,3% (Salino), 45,2% (Rozeta) és 45,5% (Altaswede) értékre csökkent. Az 1,00 mg/L dózis átlaga 91,7 ± 1,39%, míg 50,0 mg/L esetén 44,0 ± 2,34% volt. Az R^2 értékek rendre 0,9439; 0,9662; illetve 0,9828 voltak.

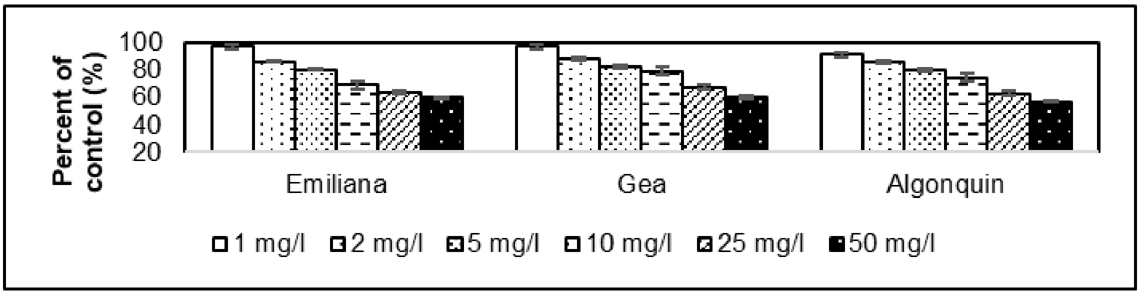


11. ábra: A Trifolium pratense (Salino, Rozeta, Altaswede) fajták hajtásmagasságának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására. A dózis növekedésével a növekedésgátlás mértéke szignifikánsan fokozódott ($p \leq 0,05$). Forrás: saját számítás, saját szerkesztés

4.4.2 A *Medicago sativa* elemzésének eredményei

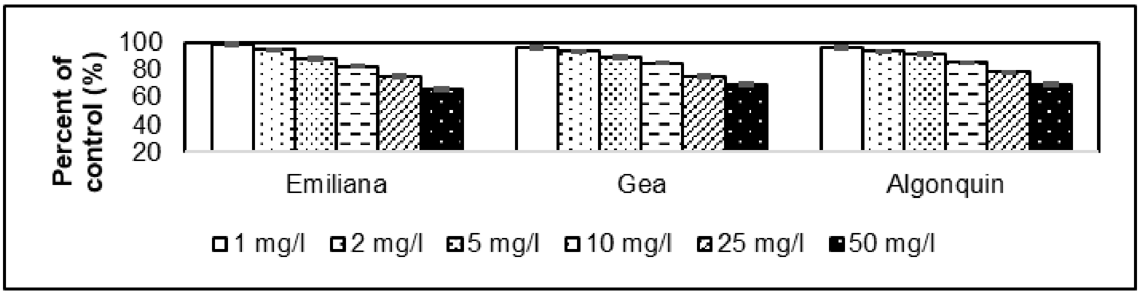
A kontroll csírázási arány 92,0% volt az Emiliana, 94,0% a Gea és 90,0% az Algonquin fajta esetében. A *Medicago sativa* magok csírázása 25,0 °C-on átlagosan 92,0 ± 2,00%-nak adódott. Az 1 mg/L dóziskoncentráció már kimutatható csökkenést eredményezett: a csírázási arány 88,0%-ra mérséklődött Emiliana és Gea esetében, míg 86,0%-ra Algonquin esetében. Az 50 mg/L dózis hatására a csírázási arány 54,0% (Emiliana), 50,0% (Gea) és 52,0% (Algonquin) értékre csökkent. A kontrollcsoport százalékában kifejezve ez 58,7% (Emiliana), 53,2% (Gea) és 57,8% (Algonquin) volt. A teljes csoport átlaga 94,9 ± 1,15% volt 1,00 mg/L esetén, míg 50,0 mg/L dózisnál 56,6 ± 2,95%. A két dózisszint közötti átlagos különbség −38,4 ± 1,81% volt. A determinációs együttható (R^2) Emiliana esetében 0,9827, Gea esetében 0,9842, Algonquin esetében 0,9754 volt, ami erőteljes dózis–válasz összefüggést jelez.

Az 1 mg/L dózis a gyökérhossz 97,2%-ra (Emiliana), 97,1%-ra (Gea) és 91,4%-ra (Algonquin) történő csökkenését eredményezte a kontrollhoz képest (**12. ábra**). Az 50 mg/L dózis hatására a gyökérhossz 59,7%-ra (Emiliana), 60,3%-ra (Gea) és 57,1%-ra (Algonquin) csökkent. Ez átlagosan 95,2 ± 3,30%-os (1,00 mg/L), illetve 59,1 ± 1,68%-os (50,0 mg/L) értéket jelentett. Az R^2 értékek 0,9781 (Emiliana), 0,9788 (Gea) és 0,9844 (Algonquin) voltak.



12. ábra: A Medicago sativa (Emiliana, Gea, Algonquin) fajták gyökérhosszának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására. Mind az 1 mg/L, mind az 50 mg/L koncentráció esetén szignifikáns csökkenés volt kimutatható ($p \leq 0,05$), a magasabb dózis erőteljesebb gátló hatással.
Forrás: saját számítás, saját szerkesztés

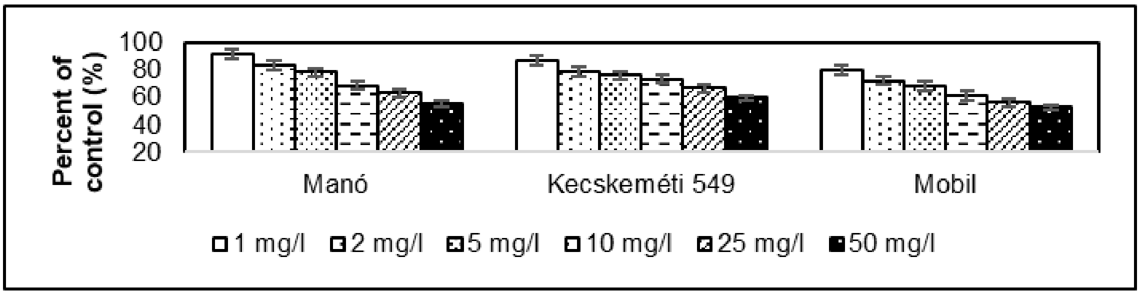
Az 1 mg/L dózis a hajtásmagasság 98,8%-ra (Emiliana), 96,3%-ra (Gea) és 96,4%-ra (Algonquin) történő csökkenését eredményezte a kontrollcsoporthoz viszonyítva (**13. ábra**). Az 50 mg/L dózis esetén a hajtásmagasság 65,9% (Emiliana), 69,1% (Gea) és 69,9% (Algonquin) értékre csökkent. Az 1,00 mg/L dózis átlaga 97,2 ± 1,41%, míg 50,0 mg/L esetén 68,3 ± 2,14% volt. Az R^2 értékek 0,9818 (Emiliana), 0,9604 (Gea) és 0,9429 (Algonquin) voltak.



13. ábra: A Medicago sativa (Emiliana, Gea, Algonquin) fajták hajtásmagasságának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására A dózis növekedésével a növekedésgátlás mértéke fokozódott ($p \leq 0,05$). Forrás: saját számítás, saját szerkesztés

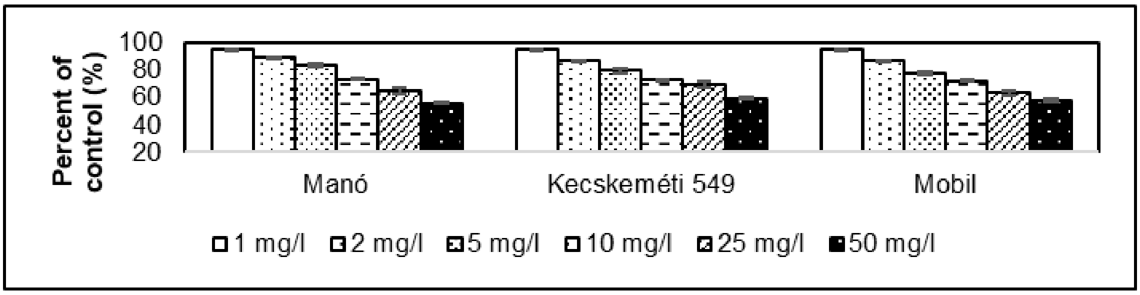
4.4.3 A *Solanum lycopersicum* elemzésének eredményei

A kontroll csírázási arány 92,0% volt a Manó, valamint 90,0% a Kecskeméti 549 és a Mobil fajta esetében. A *Solanum lycopersicum* magok csírázása 25,0 °C-on átlagosan 90,7 ± 1,15%-nak adódott. Az 1 mg/L dóziskoncentráció már mérhető csökkenést eredményezett: a csírázási arány 88,0%-ra mérséklődött a Manó és a Kecskeméti 549, valamint 86,0%-ra a Mobil fajta esetében. Az 50 mg/L dózis hatására a csírázási arány 58,0% (Manó), 60,0% (Kecskeméti 549) és 56,0% (Mobil) értékre csökkent. A kontrollcsoport százalékában kifejezve ez 63,0% (Manó), 66,7% (Kecskeméti 549) és 62,2% (Mobil) volt. A teljes csoport átlaga 96,3 ± 1,26% volt 1,00 mg/L esetén, míg 50,0 mg/L dózisnál 64,0 ± 2,36%. A két dózisszint közötti átlagos különbség −32,4 ± 1,13% volt. A determinációs együttható (R^2) 0,9909 (Manó), 0,9823 (Kecskeméti 549) és 0,9916 (Mobil) volt, ami szoros dózis–válasz kapcsolatot jelez (**14. ábra**).



14. ábra: A Solanum lycopersicum (Manó, Kecskeméti 549, Mobil) fajták gyökérhosszának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására Mind az 1 mg/L, mind az 50 mg/L koncentráció esetén szignifikáns csökkenés volt kimutatható ($p \leq 0,05$), a magasabb dózis erőteljesebb gátló hatással.
Forrás: saját számítás, saját szerkesztés

Az 1 mg/L dózis a gyökérhossz 91,9%-ra (Manó), 87,5%-ra (Kecskeméti 549) és 80,3%-ra (Mobil) történő csökkenését eredményezte a kontrollhoz képest. Az 50 mg/L dózis esetén a gyökérhossz 55,4%-ra (Manó), 59,7%-ra (Kecskeméti 549) és 52,6%-ra (Mobil) csökkent. Ez átlagosan 86,6 ± 5,87%-os (1,00 mg/L), illetve 55,9 ± 3,57%-os (50,0 mg/L) értéket jelentett. Az R^2 értékek 0,9961 (Manó), 0,9710 (Kecskeméti 549) és 0,9908 (Mobil) voltak. Az 1 mg/L dózis a hajtásmagasság 95,6%-ra történő csökkenését eredményezte mindhárom fajta esetében (Manó: 95,6%; Kecskeméti 549: 95,5%; Mobil: 95,6%) a kontrollcsoporthoz viszonyítva (**15. ábra**). Az 50 mg/L dózis hatására a hajtásmagasság 56,0% (Manó), 59,6% (Kecskeméti 549) és 57,8% (Mobil) értékre csökkent. Az 1,00 mg/L dózis átlaga 95,6 ± 0,05%, míg 50,0 mg/L esetén 57,8 ± 1,75% volt. Az R^2 értékek 0,9916 (Manó), 0,9863 (Kecskeméti 549) és 0,9941 (Mobil) voltak.



15. ábra: A Solanum lycopersicum (Manó, Kecskeméti 549, Mobil) fajták hajtásmagasságának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására. A dózis növekedésével a növekedésgátlás mértéke fokozódott ($p \leq 0,05$). Forrás: saját számítás, saját szerkesztés

4.5 Az azbesztmentes városi átmenet rendszerszintű elemzése az MLP keretében

Kutatásom ezen irányú tevékenységének célja annak feltárása volt, hogy a niche-innovációk, a rezsimstruktúrák és a makrokörnyezeti (landscape) tényezők közötti kölcsönhatások miként befolyásolják az azbesztmentes és fenntartható városi fejlődési pályák kialakulását.

4.5.1 Makrokörnyezeti (landscape) tényezők hatása az átmenetre

Elemzésem során elsőként a makrokörnyezeti szint szerepét vizsgáltam az azbesztmentes városi átmenet alakulásában. Eredményeim azt mutatták, hogy a globális fenntarthatósági diskurzus erősödése, a közegészségügyi kockázatokkal kapcsolatos társadalmi tudatosság növekedése, valamint a nemzetközi környezetpolitikai kötelezettségvállalások együttesen jelentős strukturális nyomást gyakorolnak a városi rendszerekre. Ezek a tényezők hosszú távon destabilizálják az azbeszttartalmú infrastruktúrákra épülő megoldásokat, és normatív keretet teremtenek az azbesztmentesítés szükségességének társadalmi elfogadásához. Ugyanakkor megállapítottam, hogy a makroszintű nyomás önmagában nem eredményez automatikus rendszerszintű átalakulást. Amennyiben a rezsimintézmények nem reagálnak adaptív módon a külső impulzusokra, az átmenet lassú és fragmentált marad. A makrokörnyezeti tényezők így elsősorban katalizátor szerepet töltenek be, amely előfeltétele, de nem kizárólagos meghatározója az átmenet sikerének.

4.5.2 Rezsim-szintű beágyazottság és strukturális korlátok

A rezsim-szint elemzése során arra a következtetésre jutottam, hogy az azbeszttartalmú épített környezet fennmaradása mély intézményi és gazdasági beágyazottság következménye. Az építési és bontási ágazatban kialakult gyakorlatok, a meglévő szabályozási struktúrák, valamint a költségoptimalizálásra épülő piaci logika együttesen pályafüggőséget eredményeznek. Ez a strukturális lock-in jelentősen lassítja az alternatív, fenntartható megoldások széles körű elterjedését. Eredményeim azt igazolták, hogy a rezsim akkor válik nyitottabbá az átalakulásra, amikor a szabályozási szigorítás, a pénzügyi ösztönzők és a társadalmi elvárások egyszerre jelentkeznek. Az azbesztmentesítés így nem pusztán technológiai innováció kérdése, hanem intézményi reformot és gazdasági ösztönzőrendszer-átalakítást is igényel. A rezsim rugalmassága döntő tényező az átmenet sebessége és mélysége szempontjából.

4.5.3 Niche-innovációk és azok felskálázási potenciálja

A niche-szint vizsgálata során az azbesztmentes építőanyagok, az innovatív dekontaminációs technológiák, valamint a körforgásos gazdasági modellek megjelenését elemeztem. Eredményeim alapján ezek az innovációk képesek alternatív fejlődési pályákat kínálni a fennálló rendszerrel szemben, és potenciálisan hozzájárulhatnak a rezsim átalakulásához. Ugyanakkor megállapítottam, hogy a niche-megoldások sikeressége nagymértékben függ azok intézményi és piaci integrációjától. A felskálázás feltétele a stabil finanszírozási háttér, a szabályozási környezet támogató jellege, valamint az iparági elfogadottság. Elemzésem rámutatott arra is, hogy a niche-innovációk széles körű elterjedése során fennáll annak veszélye, hogy eredeti fenntarthatósági céljaik részben kompromittálódnak. Ez a „mainstreaming-paradoxon" különösen releváns az azbesztmentesítési folyamat esetében, ahol a költséghatékonysági szempontok gyakran felülírják a hosszú távú környezetbiztonsági prioritásokat.

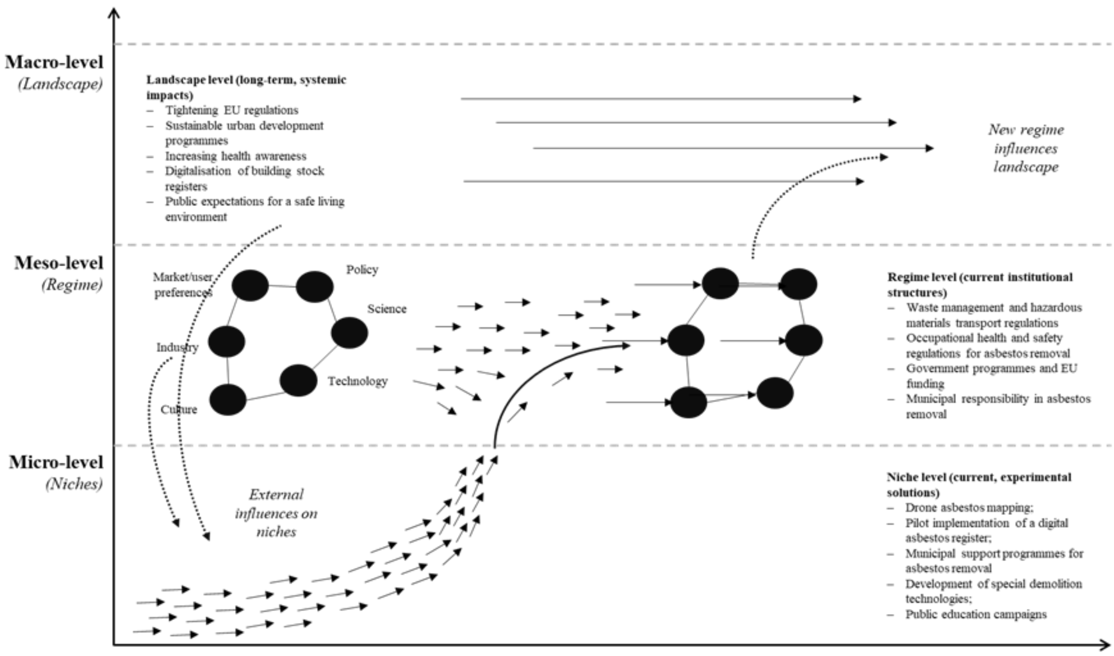
4.5.4 A rendszerszintek közötti kölcsönhatás értékelése

A három szint együttes vizsgálata alapján arra a következtetésre jutottam, hogy az azbesztmentes városi átmenet sikere a rendszerszintek közötti dinamikus kölcsönhatáson múlik. A makrokörnyezeti tényezők normatív és politikai nyomást generálnak, a rezsim biztosítja vagy éppen korlátozza az intézményi kereteket, míg a niche-innovációk konkrét technológiai és szervezeti alternatívákat kínálnak. Az átmenet akkor válik stabil és hosszú távon fenntartható folyamattá, amikor a niche-innovációk intézményesülnek a rezsim struktúráiban, és a makrokörnyezeti elvárások normatív legitimációt biztosítanak az átalakuláshoz. Eredményeim tehát megerősítik, hogy az azbesztmentesítés nem értelmezhető izolált szakpolitikai intézkedésként, hanem komplex szociotechnikai transzformációként, amely a fenntartható városi fejlődés egyik strukturális komponense. Az MLP-szintek közötti azonosított kölcsönhatásokat és az átmenet dinamikáját a **16. ábra** szemlélteti.

4.6 Az azbesztérintettség regionális vizsgálata

Az azbesztérintettség interdiszciplináris értékelésére jelenleg nem áll rendelkezésre egységes módszertan, ezért a jelen tanulmányban bemutatott elemzés is előzetesen kiválasztott indikátorcsoporton alapult. Az adatok rendszerezését követően azok kategorizálását és standardizálását végeztem el. Az egységes szerkezetbe rendezett adatok alapján meghatároztam az Euklideszi távolságértékeket, majd ezek felhasználásával kiszámítottam az alternatív, dimenziómentes azbesztérintettségi indexet (**1. táblázat**). A taxonómiai elemzés során kapott primer és szekunder adatok alapján megállapítható, hogy a magyarországi NUTS-2 régiók helyzete a vizsgált négy évben változatos, ugyanakkor bizonyos értelemben stabil képet mutatott. A 2005 és 2020 közötti időszakban a számított indexérték összességében növekedést mutatott. Az országos átlagos indexérték 2005-ben 0,3098, 2010-ben 0,2377, 2015-ben 0,2612, míg 2020-ban 0,3389 volt. Ez országos szinten alacsony érintettségi szintet jelez, ugyanakkor hangsúlyozandó, hogy a vizsgált időszakban növekedési tendencia figyelhető meg.

A növekedés hátterében a mezotelióma incidenciaértékeinek ingadozó, de tendenciájában emelkedő alakulása, valamint egyes régiókban az azbeszttartalmú hulladék mennyiségének növekedése áll. A 2005 és 2020 közötti teljes időszak átlagos változási üteme +28,5%-os növekedést mutatott. A vizsgált időszak első harmadában (2005–2010) -18,0%-os csökkenés volt tapasztalható, a második harmadban (2010–2015) +24,5%-os növekedés, míg az utolsó harmadban (2015–2020) már +33,2%-os emelkedés következett be.



16. ábra: Az azbesztmentes városi átmenet rendszerszintű dinamikája az MLP keretében

Forrás: saját számítás, saját szerkesztés

Az elemzés eredményei alapján a 2005 és 2020 közötti trendeket vizsgálva jelentős regionális különbségek figyelhetők meg a magyarországi NUTS-2 régiók között. A teljes vizsgált időszakban a legnagyobb növekedés Nyugat-Dunántúlon (HU22) volt kimutatható (+182,9%). Hasonlóan jelentős, 100%-ot meghaladó növekedés volt tapasztalható az Észak-Alföld (HU32) régióban (+111,5%). Ezzel szemben a legnagyobb csökkenés a Dél-Alföld (HU33) esetében jelentkezett (-56,3%). A nyolc régió közül négy esetében (Budapest – HU11, Nyugat-Dunántúl – HU22, Dél-Dunántúl – HU23, Észak-Alföld – HU32) növekedés volt megfigyelhető, míg négy régióban (Pest – HU12, Közép-Dunántúl – HU21, Észak-

Magyarország – HU31, Dél-Alföld – HU33) csökkenés következett be. A vizsgált időszak első harmadában (2005–2010) a legnagyobb növekedést az Észak-Alföld régióban (+48,7%) tapasztaltam, míg a legjelentősebb csökkenés a Dél-Alföld esetében volt kimutatható (-57,9%).

A második harmadban (2010–2015) Nyugat-Dunántúl mutatta a legnagyobb növekedést (+122,2%), míg a legnagyobb visszaesés Pest régióban (-55,9%) volt megfigyelhető. Az utolsó időszakban (2015–2020) az Észak-Alföld (+78,6%) és Észak-Magyarország (-27,0%) jelentette a változási intervallum két szélsőértékét. Az eredmények alapján megállapítható, hogy az azbesztérintettség többdimenziós elemzéssel jellemezhető, és a jelenség a rendelkezésre álló indikátorok mentén kvantifikálható. Ugyanakkor hangsúlyozni szükséges, hogy az indikátorkör bővülésével az indexértékek tovább finomíthatók lennének. A területi eloszlás egyenlőtlenségeit figyelembe véve Magyarország mérsékelten érintett kategóriába sorolható.

1. táblázat: Az Euklideszi távolság és az integrált index számításának eredményei

Code	NUTS-2	Euclidean distance: 2005	Euclidean distance: 2010	Euclidean distance: 2015	Euclidean distance: 2020	K_i: 2005	K_i: 2010	K_i: 2015	K_i: 2020
HU11	Budapest	2.72	3.46	2.77	1.84	0.512	0.395	0.474	0.615
HU12	Pest	3.97	3.74	4.46	3.74	0.286	0.346	0.153	0.218
HU21	Central Transdanubia	2.61	4.06	2.92	2.45	0.530	0.289	0.446	0.487
HU22	Western Transdanubia	4.93	5.20	4.21	3.24	0.114	0.090	0.200	0.322
HU23	Southern Transdanubia	4.44	5.01	4.52	3.68	0.202	0.123	0.140	0.231
HU31	Northern Hungary	4.47	4.90	4.02	3.96	0.197	0.143	0.235	0.172
HU32	Northern Great Plain	4.28	3.75	3.82	2.44	0.231	0.344	0.274	0.489
HU33	Southern Great Plain	3.31	4.74	4.38	3.93	0.406	0.171	0.167	0.177

Forrás: saját számítás, saját szerkesztés

Egyes régiókban széles intervallum és magasabb indexértékek figyelhetők meg (Budapest, Közép-Dunántúl, Észak-Alföld), míg két régió esetében (Dél-Dunántúl, Észak-Magyarország) az indexérték alacsony és szűk variációs tartományban mozgott, jellemzően 0,30 alatti értékekkel. Pest és Nyugat-Dunántúl alacsony, ugyanakkor jelentősen ingadozó indexértékeket mutatott. Az eredmények arra utalnak, hogy az azbesztérintettség területi mintázata differenciált, és a regionális eltérések figyelembevétele elengedhetetlen az azbesztmentes városi átmenet szakpolitikai és stratégiai tervezése során.

5. EREDMÉNYEK HASZNOSÍTHATÓSÁGA

A disszertáció eredményeinek hasznosíthatósága nem szűkíthető le egyetlen alkalmazási területre, mivel a vizsgálatok tárgya önmagában is többdimenziós jelenség. Az azbesztcement nem pusztán építőanyag, hanem olyan komplex kompozit rendszer, amely öregedése során anyagtudományi, kémiai, hidrológiai, biológiai és társadalmi szinten is releváns folyamatokat generál. Ebből következően az eredmények alkalmazhatósága csak rendszerszintű megközelítésben értelmezhető. A kutatás egyik legfontosabb hozzájárulása, hogy az azbesztcement problémáját nem statikus örökségként, hanem dinamikus expozíciós rendszerként írja le. A hatás nem a beépítés pillanatában jelentkezik, hanem a degradáció előrehaladtával válik egyre hangsúlyosabbá. Ez az idődimenzió különösen fontos a környezetpolitikai és kockázatértékelési alkalmazhatóság szempontjából, mivel a jelenlegi infrastruktúra-állomány jelentős része éppen a kritikus öregedési fázisba lépett.

A disszertáció eredményei lehetővé teszik egy strukturált expozíciós-hatás lánc (exposure–effect chain) modell megfogalmazását, amely az anyagszerkezeti degradációtól a társadalmi kockázatig követi nyomon a folyamatokat. Ez a modell nem pusztán elméleti konstrukció, hanem empirikusan alátámasztott, több módszertani szinten validált rendszer. A spektroszkópiai eredmények egyértelműen igazolták, hogy a környezeti expozíció a cementmátrix mikrostrukturális integritásának csökkenéséhez vezet. A karbonátosodás, a mikrorepedések kialakulása és a kötőanyag-fázis átalakulása előfeltétele a komponensek mobilizációjának. Ez a szakasz a kockázat forráspontja.

- **Gyakorlati alkalmazhatóság:** Az FT-IR alapú állapotdiagnosztika alkalmas lehet az öregedett azbesztcement elemek kockázati besorolására. A spektrális eltérések kvantifikálása hozzájárulhat a bontási prioritások meghatározásához és a preventív beavatkozások tervezéséhez.

A vízanalitikai eredmények azt mutatják, hogy az azbesztcementtel érintkező csapadékvíz paraméterei szignifikánsan eltérnek a normál csapadékvízétől. A pH-növekedés, a vezetőképesség emelkedése és a keménység jelentős változása a mátrix aktív ionforrás szerepét igazolja. A Zn, Pb és Hg koncentrációk emelkedése pedig azt mutatja, hogy a mobilizáció toxikológiailag releváns komponensekre is kiterjed.

- **Gyakorlati alkalmazhatóság:** A csapadékvíz-gyűjtési rendszerek esetében indokolt lehet az anyagfüggő minőségi kockázatértékelés bevezetése. Az eredmények alapot adhatnak olyan szabályozási keretek kialakításához, amelyek a tetőfedő anyag típusát a vízhasználat engedélyezési feltételeihez kötik.

A víz a hatáslánc transzportközege. A kémiai paraméterek torzulása nem önmagában jelent kockázatot, hanem azért, mert közvetíti az anyagszerkezeti változások következményeit a biológiai receptorok felé.

- **Gyakorlati alkalmazhatóság:** A vízminőségi paraméterek alapján kialakítható egy előrejelző indikátorrendszer, amely a biológiai hatások bekövetkezésének valószínűségét becsüli. Ez a rendszer alkalmazható mezőgazdasági öntözési döntések előkészítésében.

A növénykísérleti eredmények magas determinációs együtthatóval alátámasztott dózis–válasz kapcsolatot mutattak. A gyökérhossz és hajtásmagasság csökkenése azt jelzi, hogy az expozíció már alacsony koncentrációk esetén is kimutatható biológiai hatással jár.

- **Gyakorlati alkalmazhatóság:** A bioindikáció integrálható a vízminőség-ellenőrzési rendszerekbe, különösen ott, ahol a víz közvetlen mezőgazdasági felhasználásra kerül.

A NUTS-2 szintű indexképzés lehetővé tette az azbesztérintettség területi differenciáltságának kvantifikálását. Ez a makroszintű validáció azt mutatja, hogy az expozíciós lánc nem elszigetelt jelenség, hanem területileg strukturált mintázat.

- **Gyakorlati alkalmazhatóság:** A regionális index alkalmas lehet mentesítési prioritási térképek készítésére és forrásallokációs döntések megalapozására.

A vizsgálatok egyértelműen rámutatnak arra, hogy az azbesztcement örökség nem tekinthető pusztán passzív hulladékgazdálkodási kérdésnek. Az azbesztcement elemek aktív környezeti forrásként viselkednek, amelyek degradációjuk során kémiai és potenciálisan toxikológiai szempontból releváns komponenseket mobilizálnak. Ez a felismerés paradigmaváltást igényel az azbesztkezelési stratégiákban. A fenntartható városi átmenet nem valósítható meg kizárólag új anyagok alkalmazásával vagy jövőorientált szabályozással. Szükséges az anyaginfrastruktúra öregedési folyamatainak integrált kezelése is. A degradáció nemcsak esztétikai vagy statikai kérdés, hanem expozíciós kockázati tényező. Eredményeim indokolják a mentesítési programok priorizálását és felgyorsítását. Indokolt továbbá a csapadékvíz-használat szabályozásának felülvizsgálata olyan esetekben, ahol a gyűjtött víz azbesztcement felületről származik. A bioindikáció beemelése a környezeti kockázatértékelési gyakorlatba új, integrált monitoring-megközelítést kínálhat, amely a kémiai paramétereket biológiai válaszreakciókkal együtt értelmezi. A regionális prioritásalapú beavatkozás pedig lehetővé teszi a források hatékonyabb, kockázatarányos elosztását.

Ez a vertikális integráció (az anyagtudománytól a regionális léptékig) biztosítja a modell tudományos megalapozottságát. A rendszer vitaképes abban az értelemben, hogy további indikátorokkal és hosszabb idősorokkal finomítható, ugyanakkor jelen formájában is koherens, módszertanilag alátámasztott és reprodukálható keretet kínál.

6. ÚJ TUDOMÁNYOS EREDMÉNYEK

1. tézis: Igazoltam, hogy a környezeti expozíciónak kitett azbesztcement termékek mátrix-degradációja szignifikánsan befolyásolja a krizotil FT-IR spektrális kimutathatóságát, és a spektrális illeszkedési arány növekedése indikátora lehet a szerkezeti integritás csökkenésének.

Indokolás: Kimutattam, hogy a környezeti hatásoknak (UV-sugárzás, csapadék, hőingadozás) kitett hullámos és sík azbesztcement elemek esetében a krizotil spektrális illeszkedése szisztematikusan magasabb értékeket mutat, mint a szeparált minták esetében. Ez arra utal, hogy a cementmátrix degradációja növeli a szálas komponensek relatív spektrális láthatóságát. Eredményeim alapján az FT-IR módszer nemcsak kvalitatív azonosításra, hanem állapotdiagnosztikai célra is alkalmazható.

2. tézis: Bizonyítottam, hogy az azbesztcementtel érintkező csapadékvíz vízminőségi paraméterei (pH, vezetőképesség, keménység, TOC, KOI, BOI_5) szignifikáns mértékben eltérnek a normál csapadékvízétől, és ez a változás a cementmátrix ionmobilizációjával magyarázható.

Indokolás: A vizsgálatok igazolták a lúgosító hatást, az ionterhelés növekedését, valamint az oxidálható komponensek emelkedését. Eredményeim alátámasztják, hogy az azbesztcement tetőfelület nem passzív gyűjtőfelület, hanem aktív kémiai forrás, amely módosítja a lefolyó víz összetételét. Ez a megállapítás új megközelítést jelent a csapadékvíz-hasznosítás kockázatértékelésében.

3. tézis: Kimutattam, hogy az azbesztcement mátrixból mobilizálódó nehézfémek (Zn, Pb, Hg) koncentrációja jelentősen meghaladja a normál csapadékvíz értékeit, és a mobilizáció részben a mátrixból, részben a felületi akkumulációból származik.

Indokolás: A szimulált és valós csapadékminták összehasonlítása lehetővé tette a mátrixeredetű és környezeti eredetű komponensek elkülönítését. Eredményeim igazolják, hogy az azbesztcement szerkezeti öregedése potenciális nehézfém-kibocsátási forrássá teszi az építőanyagot. Ez a megállapítás környezettoxikológiai és szabályozási szempontból is releváns.

4. tézis: Bizonyítottam, hogy az azbesztcementtel kontaminált víz dózisfüggő, lineáris regresszióval jól modellezhető növekedésgátló hatást fejt ki a vizsgált modellnövények (*Trifolium pratense*, *Medicago sativa*, *Solanum lycopersicum*) esetében.

Indokolás: A csírázási arány, gyökérhossz és hajtásmagasság szignifikáns csökkenése már alacsony koncentrációk mellett is kimutatható volt, magas determinációs együtthatóval (R^2). Ez igazolja, hogy a vízminőségi torzulás biológiai hatássá transzformálódik. A növényélettani eredmények empirikusan megerősítik az anyagszerkezeti degradáció és az ökológiai válasz közötti oksági kapcsolatot.

5. tézis: Kidolgoztam és empirikusan alátámasztottam egy többszintű, Multi-Level Perspective (MLP) alapú expozíciós–hatás modellt, amely az azbesztcement anyagszerkezeti degradációjától a vízminőségi torzuláson és biológiai válaszreakción keresztül a fenntarthatósági átmenet rendszerszintű értelmezéséig terjed.

Indokolás: A modell mikro-szinten az anyagtudományi és vízanalitikai folyamatokat, mezo-szinten az ökológiai és agrárkörnyezeti hatásokat, makro-szinten pedig a települési és szabályozási átmeneteket integrálja. Igazoltam, hogy az azbesztcement-probléma nem izolált környezeti jelenség, hanem egy szociotechnikai rendszerben beágyazott átmeneti kihívás, amelyben a degradációból eredő környezeti hatások visszahatnak a szabályozási és mentesítési struktúrákra. Ez a rendszerszintű megközelítés új értelmezési keretet kínál az azbesztmentes települési átmenet tudományos megalapozásához.

6. tézis: Igazoltam, hogy a környezeti hatásoknak kitett azbesztcement felületekről lefolyó vízben a pH, a vezetőképesség, a vízkeménység, valamint a Zn, Pb és Hg koncentrációk szignifikáns növekedést mutatnak, és az így torzult víz dózisfüggő, lineárisan modellezhető növekedésgátló hatást fejt ki mezőgazdasági jelentőségű növényfajok esetében.

Indokolás: Kimutattam, hogy az azbesztcementtel érintkező csapadékvíz pH-értéke átlagosan közel egy teljes egységgel magasabb, mint a normál csapadéké, a vízkeménység közel kétszeresére növekedhet, a Zn-koncentráció több nagyságrenddel meghaladja a normál csapadékvíz értékeit, a Pb és Hg koncentrációk szintén szignifikáns emelkedést mutatnak, valamint a 1–50 mg/l koncentrációtartományban alkalmazott extraktum a csírázási arányt, gyökérhosszt és hajtásmagasságot 30–60%-os mértékben csökkentheti. Eredményeim bizonyítják, hogy az azbesztcement degradációja vízközvetített expozíciós útvonalon keresztül mérhető ökológiai hatást generál, így az azbesztkockázat értelmezése nem korlátozható kizárólag a levegőben terjedő szálas expozícióra.

7. AZ ÉRTEKEZÉS TÉMAKÖRÉBEN MEGJELENT PUBLIKÁCIÓK

7.1 Idegen nyelvű folyóiratban lektorált cikk

- Szandi, Á., Balog, Z., Zaka, K. S., & Macher, G. Z. (2026). Persistence of Asbestos-Containing Friction Materials in the Hungarian Waste Stream Twenty Years After the European Union Ban. INTERNATIONAL JOURNAL OF ENVIRONMENTAL RESEARCH AND PUBLIC HEALTH, 23(6), 802. http://doi.org/10.3390/ijerph23060802 (**Q1**)
- Macher, G. Z., Beke, D., & Torma, A. (2026). Index-based FT-IR assessment of chrysotile detectability and environmental durability in asbestos cement materials. SCIENTIFIC REPORTS, 1, 1–32. http://doi.org/10.1038/s41598-026-49524-w (**Q1**)
- Macher, G. Z., Bódizs, D., Sipos, D., & Schmeller, D. (2026). Asbestos Poverty as a New Paradigm for Multidimensional Urban Sustainability. JOURNAL OF URBAN HEALTH-BULLETIN OF THE NEW YORK ACADEMY OF MEDICINE, 1(1), 1–15. http://doi.org/10.1007/s11524-026-01063-5 **(D1)**
- Macher, G. Z., Torma, A., & Beke, D. (2025). The impact of asbestos cement pollution in irrigation water on physiological and germination characteristics of *Trifolium pratense*, *Medicago sativa*, and *Solanum lycopersicum* seeds. SCIENTIFIC REPORTS, 15(1). http://doi.org/10.1038/s41598-025-01011-4 **(Q1)**
- Macher, G. Z., Beke, D., & Torma, A. (2025). Water quality of harvested rainwater from asbestos cement roofs and its suitability for irrigation. CLEAN TECHNOLOGIES AND ENVIRONMENTAL POLICY, Recent Progress in Controlling Pollution, Promoting Sustainable Circularity, and Aiding Resilient Farming(1), 1. http://doi.org/10.1007/s10098-025-03197-7 **(Q1)**
- Macher, G. Z. (2025). The Impact of Rainwater Quality Harvested from Asbestos Cement Roofs on Leaf Temperature in *Solanum lycopersicum* as a Plant Water Stress Indicator. WATER, 17(14). http://doi.org/10.3390/w17142070 **(Q1)**
- Macher, G. Z., & Szigeti, C. (2025). Pathways to asbestos-free and sustainable cities using multi-level perspective approach. DISCOVER SUSTAINABILITY, 6(1). http://doi.org/10.1007/s43621-025-01932-0 **(Q1)**
- Macher, G. Z., Torma, A., & Beke, D. (2025). Examining the Environmental Ramifications of Asbestos Fiber Movement Through the Water–Soil Continuum: A Review. INTERNATIONAL JOURNAL OF ENVIRONMENTAL RESEARCH AND PUBLIC HEALTH, 22(4). http://doi.org/10.3390/ijerph22040505 **(Q1)**
- Macher, G. Z., & Beke, D. (2025a). Mosses in Urban Environments as Passive Biofilters and Organisms Impacted by Asbestos-Contaminated Habitats. INTERNATIONAL

JOURNAL OF ENVIRONMENTAL RESEARCH AND PUBLIC HEALTH, 22(6). http://doi.org/10.3390/ijerph22060838 **(Q1)**

- Macher, G. Z., Károly, F., Teh Boon Sung, C., Beke, D., Torma, A., & Gergely, S. (2024). Spectroscopic Analysis of Chrysotile Asbestos and its Environmental Resistance in Asbestos Cement Waste Products. PERTANIKA JOURNAL OF SCIENCE AND TECHNOLOGY, 32(6), 2441–2458. http://doi.org/10.47836/pjst.32.6.03 **(Q3)**
- Macher, G. Z., Torma, A., & Beke, D. (2024). Analysing the Environmental Durability and Chrysotile Content of Asbestos Cement Products by FT-IR Spectroscopy. CHEMICAL ENGINEERING TRANSACTIONS, 114, 709–714. http://doi.org/10.3303/CET24114119 **(Q3)**
- Gergely, Z. M., Judit, P., & Dóra, S. (2023). The Dissonances of Asbestos Exposure and the Concept of Sustainable Settlements in the Light of the European Union's 2023 Asbestos Neutrality Objective. CHEMICAL ENGINEERING TRANSACTIONS, 107, 145–150. http://doi.org/10.3303/CET23107025 **(Q3)**
- Gergely, Z. M., Dóra, B., & András, T. (2023). Integrated Analysis and Assessment of the Hungarian Regions in Terms of Health Problems and Waste Management Challenges caused by Asbestos. CHEMICAL ENGINEERING TRANSACTIONS, 107, 361–366. http://doi.org/10.3303/CET23107061 **(Q3)**
- Macher, G. Z., & Beke, D. (2024c). The impact of irrigation with harvested rainwater containing asbestos cement matrix on the germination characteristics of *Solanum lycopersicum*. ACTA PHYTOPATHOLOGICA ET ENTOMOLOGICA HUNGARICA, 59(1), 63–73. http://doi.org/10.1556/038.2024.00207
- Macher, G. Z., & Beke, D. (2024b). Effect of asbestos cement contamination in irrigation water on physiological and germination parameters of *Trifolium pratense* and *Solanum lycopersicum* seeds. BIO WEB OF CONFERENCES, 125. http://doi.org/10.1051/bioconf/202412501005

7.2 Magyarnyelvű folyóiratban lektorált cikk

- Macher, G. Z., Torma, A., & Beke, D. (2023a). A települési és mezőgazdasági vízgazdálkodás azbeszt okozta kockázati tényezői a krizotil-azbeszt vonatkozásában. AGROKÉMIA ÉS TALAJTAN, 72(1), 44–63. http://doi.org/10.1556/0088.2023.00147 **(Q4)**

– Macher, G. Z., Beke, D., & Torma, A. (2023). Az azbesztcement-komplexummal kontaminálódott öntözővíz kockázatai a növénytermesztésre, a talaj-víz-növény rendszerre. ACTA AGRONOMICA ÓVÁRIENSIS, 64(Különszám 2.), 11–23.

RÖVID ÉRTEKEZÉS IRODALOMJEGYZÉKE

1. Amavais, R., & Hunter, W. (1977). Public health risks of exposure to asbestos. Pergamon Press.
2. Baumann, F., Ambrosi, J. P., & Carbone, M. (2013). Asbestos is not just asbestos: An unrecognised health hazard. The Lancet Oncology, 14(7), 576–578. https://doi.org/10.1016/S1470-2045(13)70257-2
3. Bloise, A., Catalano, M., & Gualtieri, A. F. (2018). Effect of grinding on chrysotile, amosite and crocidolite and implications for thermal treatment. Anais do III Congresso Brasileiro Ciências da Saúde, 1, 1–9. https://doi.org/10.3390/min8040135
4. Cossette, M. (1984). Defining asbestos for monitoring purposes. In B. Levadie (Ed.), Definitions for asbestos and other health-related silicates (ASTM Special Technical Publication 834, pp. 5–50). American Society for Testing and Materials.
5. Dola, D. B., Mannan, M. A., Sarker, U., Mamun, M. A. A., Islam, T., Ercisli, S., Saleem, M. H., Ali, B., Pop, O. L., & Marc, R. A. (2022). Nano-iron oxide accelerates growth, yield, and quality of Glycine max seed in water deficits. Frontiers in Plant Science, 13, 992535. https://doi.org/10.3389/fpls.2022.992535
6. Farooq, T. H., Rafay, M., Basit, H., Shakoor, A., Shabbir, R., Riaz, M. U., Ali, B., Kumar, U., Qureshi, K. A., & Jaremko, M. (2022). Morphophysiological growth performance and phytoremediation capabilities of selected xerophyte grass species toward Cr and Pb stress. Frontiers in Plant Science, 13, 997120. https://doi.org/10.3389/fpls.2022.997120
7. Geels, F. W. (2002). Technological transitions as evolutionary reconfiguration processes: a multi-level perspective and a case-study. Research Policy, 31(8–9), 1257–1274. https://doi.org/10.1016/S0048-7333(02)00062-8
8. Geels, F. W. (2011). The multi-level perspective on sustainability transitions: Responses to seven criticisms. Environmental Innovation and Societal Transitions, 1(1), 24–40. https://doi.org/10.1016/j.eist.2011.02.002
9. Leonelli, C., Veronesi, P., Boccacini, D. N., Rivasi, M. R., Barbieri, L., Andreola, F., Lancellotti, I., Rabitti, D., & Pellacani, G. C. (2006). Microwave thermal inertisation of asbestos containing waste and its recycling in traditional ceramics. Journal of Hazardous Materials, 135(1–3), 149–155. https://doi.org/10.1016/j.jhazmat.2005.11.035
10. Ma, J., Saleem, M. H., Ali, B., Rasheed, R., Ashraf, M. A., Aziz, H., Ercisli, S., Riaz, S., Elsharkawy, M. M., Hussain, I., Alhag, S. K., Ahmed, A. E., Vodnar, D. C., Mumtaz, S., & Marc, R. A. (2022a). Impact of foliar application of syringic acid on tomato (*Solanum lycopersicum* L.) under heavy metal stress—Insights into nutrient uptake, redox

homeostasis, oxidative stress, and antioxidant defense. Frontiers in Plant Science, 13, 950120. https://doi.org/10.3389/fpls.2022.950120

11. Ma, J., Saleem, M. H., Yasin, G., Mumtaz, S., Qureshi, F. F., Ali, B., Ercisli, S., Alhag, S. K., Ahmed, A. E., Vodnar, D. C., Hussain, I., Marc, R. A., & Chen, F. (2022b). Individual and combinatorial effects of SNP and NaHS on morpho-physio-biochemical attributes and phytoextraction of chromium through Cr-stressed spinach (Spinacia oleracea L.). Frontiers in Plant Science, 13, 973740. https://doi.org/10.3389/fpls.2022.973740
12. Magherini, L., Avataneo, C., Capella, S., Lasagna, M., Bianco, C., Belluso, E., De Luca, D. A., & Sethi, R. (2023). Mobility of crocidolite asbestos in sandy porous media mimicking aquifer systems. Journal of Hazardous Materials, 458, 131998. https://doi.org/10.1016/j.jhazmat.2023.131998
13. McDonald, J. C. (1972). Asbestosis in chrysotile mines and mills. In Biological effects of asbestos (IARC Scientific Publication No. 8, pp. 155–159). International Agency for Research on Cancer.
14. Mendez, C. B., Klenzendorf, J. B., Afshar, B. R., Simmons, M. T., Barrett, M. E., Kinney, K. A., & Kirisits, M. J. (2011). The effect of roofing material on the quality of harvested rainwater. Water Research, 45(5), 2049–2059. https://doi.org/10.1016/j.watres.2010.12.015
15. Mossman, B. T., Bignon, J., Corn, M., Seaton, A., & Gee, J. B. L. (1990). Asbestos: Scientific developments and implications for public policy. Science, 247, 294–301. https://doi.org/10.1126/science.2153315
16. Nicholson, N., Clark, S. E., Long, B. V., Siu, C. Y. S., Spicher, J., & Steele, K. A. (2010). Rainwater Harvesting from Roofs for Non-Potable Reuse. Low Impact Development 2010, 21–30. https://doi.org/10.1061/41099(367)3
17. Oliinyk, O., Mishchuk, H., Vasa, L., & Kozma, K. (2023). Social responsibility: Opportunities for integral assessment and analysis of connections with business innovation. Sustainability, 15(6), 5608. https://doi.org/10.3390/su15065608
18. Qin, H.-P., Tang, Q.-L., Wang, L.-Y., & Fu, G. (2015). The impact of atmospheric wet deposition on roof runoff quality in an urbanized area. Hydrology Research, 46(6), 880–892. https://doi.org/10.2166/nh.2015.209
19. Saleem, K., Asghar, M., Saleem, M., Raza, A., Kocsy, G., Iqbal, N., Ali, B., Albeshir, M., & Bhat, E. (2022). Chrysotile-asbestos-induced damage in *Panicum virgatum* and Phleum pratense species and its alleviation by organic-soil amendment. Sustainability, 14(17)*, 10824. https://doi.org/10.3390/su141710824
20. Spurny, K. R., Marfels, H., Boose, C., Weiss, G., Opiela, H., & Wulbeck, F. J. (1989). Fiber emissions from weathered and corroded asbestos cement products. Part 2. Physical and

chemical properties of the released asbestos fibres. Zentralblatt für Hygiene und Umweltmedizin, 188(3–4), 262–270.

21. Suzuki, Y., Yuen, S. R., & Ashley, R. (2005). Short, thin asbestos fibres contribute to the development of human malignant mesothelioma: Pathological evidence. International Journal of Hygiene and Environmental Health, 208, 201–210. https://doi.org/10.1016/j.ijheh.2005.01.015
22. Székely, L. (1981). Tetőfedés. In Tetőszigetelés, tetőfedés: Segédlet a szakipari munkák tervezéséhez és kivitelezéséhez I. (pp. 200–219). Építésügyi Tájékoztatási Központ.
23. Tóth, E., & Weiszburg, T. (2011). Környezeti ásványtan (T. Weiszburg, Ed.).
24. Trivedi, A. K., Ahmad, É., Musthapa, M. S., & Ansari, F. A. (2004). Environmental contamination of chrysotile asbestos and its toxic effects on growth and physiological and biochemical parameters of *Lemna gibba*. Archives of Environmental Contamination and Toxicology, 47(3)*, 281–289. https://doi.org/10.1007/s00244-004-3161-7
25. Trivedi, A. K., Ahmad, É., Musthapa, M. S., & Ansari, F. A. (2007). Environmental contamination of chrysotile asbestos and its toxic effects on antioxidative system of *Lemna gibba*. Archives of Environmental Contamination and Toxicology, 52(3)*, 355–362. https://doi.org/10.1007/s00244-006-0056-9
26. Trivedi, A. K., & Ahmad, I. (2011). Effects of chrysotile asbestos contaminated soil on crop plants. Soil and Sediment Contamination: An International Journal, 20(7)*, 767–776. https://doi.org/10.1080/15320383.2011.609197
27. Trivedi, A. K., & Ahmad, I. (2013). Genotoxicity of chrysotile asbestos on *Allium cepa* L. meristematic root tip cells. Current Science, 105(6)*, 846–852.
28. Turci, F., Favero-Longo, S. E., Gazzano, S., Tomatis, M., Gentile-Garofalo, L., & Bergamini, M. (2016). Assessment of asbestos exposure during a simulated agricultural activity in the proximity of the former asbestos mine of Balangero, Italy. Journal of Hazardous Materials, 308, 321–327. https://doi.org/10.1016/j.jhazmat.2016.01.056
29. Yaziz, M. I., Gunting, H., Sapari, N., & Ghazali, A. W. (1989). Variations in rainwater quality from roof catchments. Water Research, 23(6), 761–765. https://doi.org/10.1016/0043-1354(89)90211-X

ÁBRAJEGYZÉK

1. ábra: A krizotil detektálás összehasonlító elemzése exponált (E) és szeparált (S) minták esetében (hullámos azbesztcement lemezek) Forrás: saját számítás, saját szerkesztés 19
2. ábra: A krizotil detektálás összehasonlító elemzése exponált (E) és szeparált (S) minták esetében (sík azbesztcement lemezek) Forrás: saját számítás, saját szerkesztés 20
3. ábra: A krizotil detektálás összehasonlító elemzése exponált (E) és szeparált (S) minták esetében (azbesztcement csövek) Forrás: saját számítás, saját szerkesztés 20
4. ábra: A vízminták összehasonlító elemzése pH, vezetőképesség, TSS és keménység paraméterek alapján Forrás: saját számítás, saját szerkesztés 22
5. ábra: A vízminták összehasonlító elemzése a TOC-, BOI_5- (az ábrán: BOD_5) és KOI_p- (az ábrán: COD_p) paraméterek alapján Forrás: saját számítás, saját szerkesztés 23
6. ábra: A vízminták összehasonlító elemzése nehézfém-szennyezettség (Zn, Pb, Hg) tekintetében Forrás: saját számítás, saját szerkesztés 24
7. ábra: A szimulált minták összehasonlító elemzése pH, vezetőképesség, TSS és keménység paraméterek alapján Forrás: saját számítás, saját szerkesztés 25
8. ábra: A szimulált minták összehasonlító elemzése a TOC-, BOI_5- (az ábrán: BOD_5) és KOI_p- (az ábrán: COD_p) paraméterek alapján Forrás: saját számítás, saját szerkesztés 26
9. ábra: A szimulált minták összehasonlító elemzése a teljes Zn-, Pb- és Hg-koncentrációk alapján Forrás: saját számítás, saját szerkesztés 27
10. ábra: A *Trifolium pratense* (Salino, Rozeta, Altaswede) fajták gyökérhosszának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására. Mind az 1 mg/L, mind az 50 mg/L koncentráció esetében szignifikáns csökkenés volt kimutatható ($p \leq 0{,}05$), a nagyobb dózis erőteljesebb gátló hatással. Forrás: saját számítás, saját szerkesztés 28
11. ábra: A *Trifolium pratense* (Salino, Rozeta, Altaswede) fajták hajtásmagasságának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására. A dózis növekedésével a növekedésgátlás mértéke szignifikánsan fokozódott ($p \leq 0{,}05$). Forrás: saját számítás, saját szerkesztés 29
12. ábra: A *Medicago sativa* (Emiliana, Gea, Algonquin) fajták gyökérhosszának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására. Mind az 1 mg/L, mind az 50 mg/L koncentráció esetén szignifikáns csökkenés volt kimutatható ($p \leq 0{,}05$), a magasabb dózis erőteljesebb gátló hatással. Forrás: saját számítás, saját szerkesztés 30
13. ábra: A *Medicago sativa* (Emiliana, Gea, Algonquin) fajták hajtásmagasságának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására A dózis növekedésével a növekedésgátlás mértéke fokozódott ($p \leq 0{,}05$). Forrás: saját számítás, saját szerkesztés ... 30

14. ábra: A *Solanum lycopersicum* (Manó, Kecskeméti 549, Mobil) fajták gyökérhosszának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására Mind az 1 mg/L, mind az 50 mg/L koncentráció esetén szignifikáns csökkenés volt kimutatható ($p \leq 0{,}05$), a magasabb dózis erőteljesebb gátló hatással. Forrás: saját számítás, saját szerkesztés............................. 31
15. ábra: A *Solanum lycopersicum* (Manó, Kecskeméti 549, Mobil) fajták hajtásmagasságának összehasonlító vizsgálata az azbeszttel szennyezett öntözővíz hatására. A dózis növekedésével a növekedésgátlás mértéke fokozódott ($p \leq 0{,}05$). Forrás: saját számítás, saját szerkesztés ... 31
16. ábra: Az azbesztmentes városi átmenet rendszerszintű dinamikája az MLP keretében Forrás: saját számítás, saját szerkesztés 34

TÁBLÁZATJEGYZÉK

1. táblázat: Az Euklideszi távolság és az integrált index számításának eredményei 35

A RÖVID ÉRTEKEZÉS ALAPJÁT KÉPEZŐ KÖZLEMÉNYEK

scientific reports

Check for updates

OPEN

The impact of asbestos cement pollution in irrigation water on physiological and germination characteristics of *Trifolium pratense*, *Medicago sativa*, and *Solanum lycopersicum* seeds

Gergely Zoltán Macher[1]✉, András Torma[1] & Dóra Beke[2]

This paper investigates how plants respond to stress caused by asbestos cement products in irrigation water. It presents a thorough evaluation of the exposure and risk factors for plants, water, and soil when exposed to these materials. The experimental results provide empirical evidence of plant stress responses based on physiological and germination parameters. The research is motivated by concerns about environmental contamination from asbestos cement in irrigation water, which can be toxic to plants and lead to soil pollution, negatively impacting vegetation and soil quality. When exposed to asbestos in water, plants experience toxic stress that can inhibit photosynthesis, nutrient uptake, and germination. Asbestos can also adversely affect cell division and metabolism, risking plant growth, reproduction, and overall health, as well as making them more susceptible to disease and pests under environmental stress. The paper examines the impact on germination and physiological parameters of *Trifolium pratense*, *Medicago sativa*, and *Solanum lycopersicum*, particularly how they were affected by pre-established concentrations of irrigation water mixed with asbestos cement during a controlled germination experiment. The research methodology was developed in the absence of established global practices, standards, and methods, creating an opportunity for further methodological advancement. The findings could serve as a situational analysis for professionals in environmental plant protection and analytical fields.

Keywords Asbestos cement contamination, Asbestos toxicity, Harvested rainwater, Abiotic stress

Asbestos is a collective term for naturally occurring fibrous minerals that were extensively utilized in construction and industry due to their ability to withstand high temperatures and provide insulation. These minerals exhibit durability and cost-effective, but they represent potential health risks when inhaled as airborne particles[1]. Inhalation of asbestos fibres can lead to respiratory illnesses, including asbestosis, lung cancer, and *Mesothelioma malignum*[2]. The presence of asbestos in older structures has been identified as a continuing cause for concern[3]. Asbestos pollution has profound ramifications for environmental wellbeing, especially in soil and water ecosystems.

Asbestos has been shown to significantly affect environmental health, particularly regarding soil and water quality.

The release of asbestos fibres into the environment can contaminate the soil, thereby jeopardizing land-based ecological systems. Soil contamination can contribute to the spread of asbestos through wind erosion or surface runoff, leading to the dispersal of fibres into new areas[4–6]. The enduring nature and resilience of asbestos in the surrounding areas require comprehensive supervision, correction, and precautionary actions to reduce additional environmental and public health consequences[7].

[1]Department of Applied Sustainability, Albert Kázmér Faculty of Agricultural and Food Sciences of Széchenyi, István University in Mosonmagyaróvár, Széchenyi István University, Gyor, Hungary. [2]Department of Plant Sciences, Albert Kázmér Faculty of Agricultural and Food Sciences of Széchenyi, István University in Mosonmagyaróvár, Széchenyi István University, Mosonmagyaróvár, Hungary. ✉email: macher.gergely.zoltan@sze.hu

Asbestos cement products, especially roofing materials, are a major source of asbestos contamination in the environment due to their gradual deterioration. Asbestos cement products, particularly asbestos cement roofing, were widely used due to their durability and resistance to fire and weather. These products consist of a mixture of cement and asbestos fibres, providing structural strength. However, over time, these roofs become increasingly vulnerable to erosion and degradation, leading to the release of asbestos fibres into the environment. Factors such as age, weather conditions, and physical damage can exacerbate this erosion[8]. The deterioration and wearing away of asbestos cement roof materials result in the release of sizable amounts of harmful asbestos fibres into the nearby surroundings. These fibres can endure for extended periods due to their strong resistance to natural degradation processes[9]. Weathering mechanisms like freeze-thaw cycles, acid rain, and ultraviolet radiation expedite the decomposition of these substances, leading to a higher rate of fibre release[10–12]. These fibrous materials can be transported by the wind and distributed in soil or adjacent water sources[13]. Deteriorating roofing can release asbestos fibres, which may contaminate the nearby soil and water, thereby posing serious health risks to human and animal populations[14,15]. Rainwater collected from these roofs has the risk of being contaminated with asbestos fibres, making it unsuitable for drinking or irrigation and possibly causing pollution in local groundwater sources[16–18]. Additionally, the asbestos fibres may settle on plants and vegetation, potentially being incorporated into the food web if these plants are consumed by livestock or wildlife[19].

Chrysotile asbestos, a component of cement roofing, can be released into water as fibres when the cement materials deteriorate. This release is mainly caused by weathering and erosion processes, during which rainwater or surface runoff carries the asbestos fibres away from their original location[17]. In water, chrysotile asbestos is less durable than other forms, but it can still persist and remain suspended for significant periods due to its fibrous nature[9]. Asbestos fibres that are released into the environment not only present risks to human health and soil quality, but also have direct and indirect impacts on plant physiology and growth.

Investigations into the impacts of asbestos on plants, especially those cultivated for gardens and agriculture, have uncovered multiple effects. Exposure to asbestos fibres can inhibit seed germination, likely due to the abrasive nature of the fibres disrupting seed coats and impeding water uptake[17]. Plants that do germinate may display stunted shoot and root growth because of fibrous asbestos inhibiting the development of root systems, affecting nutrient and water absorption[9]. The number of leaves a plant produces may also be reduced, likely due to impaired photosynthesis and overall vitality[20]. Biochemical factors of plants, such as the level of proteins, may be negatively impacted by exposure to asbestos. The abrasive properties of asbestos fibres along with their chemical reactions can disturb regular metabolic activities. When plant tissues are exposed to soil contaminated with asbestos, the protein content frequently diminishes due to the inhibition of crucial nutrient uptake by the fibres, leading to physiological strain[21]. The presence of asbestos fibres in soil also correlates with reduced chlorophyll content, which can impair photosynthesis and lead to reduced plant vigor and yield[20]. Furthermore, root development can be significantly impacted, as fibres physically obstruct root elongation or chemically disrupt normal root function[22]. Additionally, oxidative stress caused by asbestos exposure can lead to the generation of reactive oxygen species (abbreviation: ROS), which can damage cellular components, further reducing biochemical functions like protein synthesis and leaf growth[23]. Exposure to asbestos poses a distinct challenge to plant life, emerging as a novel environmental element in modern times. The gritty and fibrous quality of asbestos has the potential to disturb cellular arrangements, leading to physical harm in root and leaf tissues. This dual impact results in decreased nutrient absorption, compromised protein production, and overall inhibited growth, culminating in a comprehensive reaction to this stressor[21]. Runoff from deteriorating asbestos cement products can mobilize fibres into collected rainwater, making it a potential hazard when used for irrigation or other purposes[17]. Thus, understanding and mitigating the impact of asbestos on plant stress responses is critical for safeguarding agricultural productivity and ecosystem health in a changing climate. Given these environmental and biological impacts, it is crucial to further investigate how asbestos contamination influences plant stress responses and broader ecological dynamics.

Table 1 presents a comprehensive summary of the relevant literature necessary for establishing the research background of this paper. This summary aims to delineate the existing body of knowledge regarding the effects of chrysotile asbestos on vegetation, thereby clarifying the novelty of the current research. By consolidating previous studies, this overview identifies gaps in the literature and emphasizes the significance of the present investigation. Such an approach ensures that the findings of this research are situated within the broader academic discourse, illustrating how it contributes to advancing the understanding of the relationships between environmental contaminants and plant life. Ultimately, this summary underscores the innovative aspects of the paper, positioning it as an important contribution to the field.

The novelty and gap-filling nature of this research can be emphasized by noting that previous literature has typically focused on either the contamination of soil samples taken from polluted sites or analysed aquatic environments. This research integrates these two approaches by examining the various aspects of chrysotile asbestos presence in precipitation and its effects on the soil-plant system. The novelty of this research lies in its comprehensive evaluation of plant stress responses to asbestos cement contamination in irrigation water during the early vegetation phases, an area with limited established global practices and standards. Additionally, there is a notable international gap in that most studies concentrate solely on pure asbestos fibres, overlooking the fact that chrysotile's high surface tension and charge can lead to the formation of complex associations. A key recognition of this paper is that plants do not encounter pure chrysotile but rather chemically and physically altered complexes or composite formations, which influence their bioavailability and toxic effects. Furthermore, in cases of heavy metal contamination, synergistic effects may occur, which are often disregarded. This paper differs from previous work by employing a multi-faceted approach, integrating lab-based germination and growth experiments with advanced analytical techniques to assess asbestos effects on plants. While earlier studies focused on free asbestos fibres in soil and water, this research examines the impact of asbestos cement particles

Author(s) (year)	Title	Findings	Reference
Trivedi A. K., Musthapa A. M. S., Ansari F. A., Rahman Q. (2004)	Environmental contamination of chrysotile asbestos and its toxic effects on growth and physiological and biochemical parameters of *Lemna gibba*	The paper was conducted to assess the toxicity of chrysotile asbestos, a type of asbestos, on Lemna gibba, a water-loving aquatic plant. The research found that asbestos concentrations were present in water, sediment, and aquatic plant samples near an asbestos cement plant. In laboratory experiments, L. gibba plants were exposed to two concentrations of chrysotile asbestos twice weekly for 28 days. The paper evaluated the effects of this exposure on various growth, physiological, and biochemical parameters. The results showed that chrysotile asbestos exposure had inhibitory effects on shoot number, root length, and biomass of the plants. Additionally, it led to changes in chlorophyll, carotenoids, total free sugars, starch, and protein content. On the other hand, electrolyte efflux, lipid peroxidation, cellular hydrogen peroxide, catalase, and superoxide dismutase activity increased in a dose- and time-dependent manner. The findings indicate that chrysotile asbestos causes oxidative stress and phytotoxicity.	19
Trivedi A. K., Musthapa A. M. S., Ansari F. A. (2007)	Environmental contamination of chrysotile asbestos and its toxic effects on antioxidative system of *Lemna gibba*	The paper was conducted to evaluate the toxicity of chrysotile asbestos on duckweed plants. Asbestos was found in plant samples near an asbestos cement factory, and laboratory experiments were carried out using different concentrations of chrysotile asbestos to assess changes in antioxidant systems in the plants. The paper measured the levels of glutathione and ascorbate, which are known biomarkers of chrysotile contamination. The results showed that exposure to chrysotile asbestos led to a decrease in total and reduced glutathione, and an increase in oxidized glutathione and reduced/oxidized glutathione ratios. Additionally, there was an increase in the size of the ascorbate pool and in the levels of reduced and oxidized ascorbate, along with a decrease in the reduced/oxidized ascorbate ratio. These changes in antioxidant levels can be considered as indicators of exposure to unsafe environments, as glutathione and ascorbate play a crucial role in combating oxidative stress caused by environmental factors.	24
Trivedi A. K., Ahmad I. (2011)	Effects of Chrysotile Asbestos Contaminated Soil on Crop Plants	The paper reports on the impact of the growing vegetation near an asbestoscement factory on chrysotile asbestos. Soil samples collected from locations close to the factory exhibited higher concentrations of asbestos fibres compared to those taken from more distant sites. However, other soil characteristics, such as organic carbon, nitrogen, phosphorus, potassium, electrical conductivity, and pH, were similar across all samples. The food crops used in the experiment, including wheat, peas, and mustard, displayed significant germination issues in soil contaminated with chrysotile asbestos fibres. The height of the exposed plants, root length, biomass, as well as chlorophyll and protein content were also reduced. This paper highlights the detrimental effects of chrysotile asbestos on the surrounding vegetation.	21
Trivedi A. K., Ahmad I. (2013)	Impact of chrysotile asbestos contaminated soil on foliar nutrient status of plants	The toxic effects of asbestos on humans are well documented, but little information is available on its effects on plants. In this paper, asbestos contamination was monitored in soil around a cement factory at different distances from the factory. More asbestos contamination was found in the soil near the factory. These soils were analysed for different parameters (organic carbon, EC, pH, macro- and microelements). As the asbestos contamination increased, the levels of magnesium, iron and manganese showed a slight increase, which was accompanied by a decrease in the availability of other nutrients. Wheat, pea and mustard seeds were planted in asbestos-contaminated and control soils. Reduced nutrient content was observed in the leaves of plants grown in asbestos-contaminated soil. The paper reports on the effect of the foliar nutrient status of plants grown near an asbestos cement plant.	25
Trivedi A. K., Ahmad I. (2013)	Genotoxicity of chrysotile asbestos on Allium cepa L. meristematic root tip cells	The genotoxic and mutagenic effects of chrysotile asbestos on animals and humans are well-documented, but little is known about its impact on plants. To address this, researchers conducted a paper using Allium cepa L. bulbs exposed to different concentrations of chrysotile asbestos. They observed changes in cytogenetic parameters over a period of 96 h. The paper found that the progression of the prophase stage in exposed plants was slower compared to control plants. The number of cells in metaphase increased in exposed plants, while it decreased in control plants. Furthermore, the number of cells in anaphase and telophase stages decreased in exposed plants over time. The mitotic index also decreased in a time- and concentration-dependent manner in exposed plants. Additionally, exposed plants showed an increase in interphase nuclei, spindle abnormalities, chromosome sticking, and micronucleus formation. These findings suggest that chrysotile asbestos exhibits genotoxicity in plants.	26
Saleem K., Asghar M. A., Saleem M. H., Raza A., Kocsy G., Iqbal N., Ali B., Albeshr M. F., Bhat E. A. (2022)	Chrysotile-Asbestos-Induced Damage in *Panicum virgatum* and *Phleum pretense* Species and Its Alleviation by Organic-Soil Amendment	In this paper, the hazardous effects of asbestos on the growth and development of switchgrass and timothy grass were investigated, with the aim of finding ways to mitigate its toxicity. Asbestos-contaminated soil samples were collected near a cement factory, and it was found that these soils had increased uptake of heavy metals, such as chromium, manganese, vanadium, arsenic, and barium, which led to reduced plant growth. To counteract this, antioxidant enzymes were activated to maintain redox balance. The soil closest to the cement plant was found to be the most toxic. However, the addition of compost as a biofertilizer significantly reduced the toxic effect of asbestos fibres and reduced metal uptake, improving overall plant growth and development. It was also observed that heavy metal accumulation was higher in the roots of the grass species. In summary, this paper suggests that the investigated grass species can be negatively affected by asbestos contamination, but the addition of compost can help mitigate these effects.	27

Table 1. Summary of the literature background on the effects of Chrysotile asbestos on vegetation.

in irrigation water, simulating real-world contamination. The experimental design uses precise dose-response analyses to evaluate physiological and biochemical stress indicators in plants, while statistical modelling enhances the reliability of findings by quantifying the relationship between asbestos exposure and plant responses. The other novelty of the research lies in the fact that it investigates the combined effect of asbestos-cement matrix, a topic that has been studied only minimally in the literature. Moreover, existing research does not adequately address the involvement of horticulture, further highlighting the significance of the current paper. Thus, this research not only sheds new light on the effects of chrysotile asbestos but also provides a more comprehensive understanding of its impacts on soil-plant systems, particularly focusing on horticulture and the complexities of environmental interactions. Despite the scarcity of such studies, this research provides new insights into the environmental risks posed by asbestos contamination, advances methodological approaches, and offers valuable situational analysis for professionals in environmental plant protection and analytical fields.

Materials and methods

The paper analysed the germination parameters and diversity of three varieties of *Trifolium pretense* (Salino, Rozeta, Altaswede), *Medicago sativa* (Emiliana, Gea, Algonquin) and *Solanum lycopersicum* (Manó, Kecskeméti 549, Mobil). It also evaluated their variability to validate the adverse effects of asbestos cement contaminated irrigation water on plant growth in a controlled experiment using distilled water. The *Trifolium pratense* and *Medicago sativa* seeds were procured from Pannon-Mag-Agrár Kft., whereas the *Solanum lycopersicum* seeds were acquired from Garafarm Trade Kft. The results indicated a significant reduction in both the rate and percentage of germination across all *Trifolium pratense*, *Medicago sativa* and *Solanum lycopersicum* varieties. The selected plant species - *Solanum lycopersicum*, *Trifolium pratense*, and *Medicago sativa* - have relevance beyond their agricultural importance, as they can serve as models to address critical environmental and water management concerns. *Solanum lycopersicum* was chosen due to its significance in horticulture and its potential to accumulate contaminants when irrigated with water from aging asbestos cement channels, which poses risks to food safety. *Trifolium pratense* is particularly pertinent in the context of urban green spaces, where the use of greywater for irrigation is an emerging practice, necessitating further assessment of plant resilience and contaminant uptake. *Medicago sativa*, widely cultivated in arable farming, is crucial for evaluating the risks associated with irrigation infrastructure constructed from asbestos cement materials, which may contribute to pollutant exposure. These plant species, therefore, serve as valuable models for understanding the implications of alternative water sources and irrigation systems on both agricultural and ecological sustainability.

Conditions and arrangement for germination experiment

Three different varieties of *Trifolium pratense*, *Medicago sativa* and *Solanum lycopersicum* were investigated to assess the effects of stress induced by varying doses. The seeds were sterilized on the surface using 2.0% NaOCl for a duration of 2 min, followed by three rinses with sterile distilled water. Subsequently, batches consisting of 10 seeds each were placed in individual sterile Petri dishes containing a moistened cotton disc. Each treatment dose was represented by five petri dishes for analysis ($N = 5 \times 10$). There are a total of 350 test items for each type ($N = 7 \times 50$). The germination experiment was conducted under controlled laboratory conditions to ensure reproducibility. The Petri dishes housing the seeds were maintained at a consistent temperature of 22 ± 1 °C, monitored using a digital temperature controller. The system utilized built-in LED illumination optimized for plant growth, providing a full-spectrum light source. The LED setup, powered via an adjustable USB-A to USB-C connection, enabled programmable light cycles with seven-step dimmable intensity settings. The photon flux density at plant level was maintained at a maximum of 50 lm. The ventilation system was designed to ensure appropriate airflow regulation.

Preparation of sample solutions

During this analysis, asbestos cement products were examined that commonly utilized in Hungary, which contained chrysotile asbestos with an average composition ranging from 8.00 to 10.0%, and a cement content of 90.0–92.0%. It was essential to simultaneously investigate these specific characteristics due to their combined effects on the erosion and deterioration of the product matrix. Solutions at various concentrations (1.00 mg/l, 2.00 mg/l, 5.00 mg/l, 10.0 mg/l, 25.0 mg/l, and 50.0 mg/l) were prepared using doubly distilled water in a laboratory setting for analysis purposes. To prepare the extract, asbestos cement samples were first mechanically ground into fine particles. The resulting powder was then suspended in doubly distilled water at predetermined concentrations and incubated for 24 h at ambient temperature. After the extraction process, the suspensions were filtered through a 0.45 μm filter to remove larger particles while retaining the asbestos fibres and colloidal components. The resulting solutions were stored in sterile glass containers under dark conditions at 4 °C until needed for further analysis. The control group underwent treatment with double-distilled water. Seeds were subjected to their assigned dosage and treatment method during the experiment.

Germination assessment

Throughout the analysis, we consistently monitored the germination rate, germination time, and the root length and shoot height of plants. Root length was measured from the point of contact with the cotton disc's surface to the tip of the primary root, while shoot height measurements were taken on day 31 for each sample. These parameters were assessed utilizing a Burg-Wächter Precise PS-72,150 digital caliper (0 mm to 150 mm range) to ensure precise data on development. Each treatment was replicated multiple times, and our findings are reported as mean values ± standard error. The obtained data was examined in terms of percentages. The statistical significance of the results was evaluated using ANOVA analysis performed in Excel, with a significance level threshold of 0.05 to determine if the findings were statistically significant.

Results

Results of *Trifolium pratense* analysis

The control germination rate is 88.0% for Salino, 86.0% for Rozeta and 90.0% for Altaswede. Germination of *Trifolium pratense* seeds at 25.0 °C under light and dark conditions averaged 88.0 ± 2.00%. The tested samples showed that already at a dose concentration of 1 mg/l the germination rate was reduced to 82.0% in the case of the Salino, 80.0% for Rozeta and 84.0% for Altaswede. This was reduced to 48.0% (Salino), 46.0% (Rozeta) and 58.0% (Altaswede) at the 50 mg/l dose, and to 54.5% (Salino), 53.5% (Rozeta) and 64.4% (Altaswede) as a percentage of the control group. The average value for the whole group is 93.2 ± 0.16% for 1.00 mg/l and 57.5 ± 6.04% for 50 mg/l at the percentage of control group. The average difference in the effect of the two dose concentrations is −35.7 ± 5.90%. The coefficient of determination of Salino is 0.9840, while 0.9974 for Rozeta and 0.9313 for Altaswede. The dose effect of 1 mg/l resulted in a reduction to 95.2% in root length compared to the control group in Salino (Fig. 1). This rate was 95.7% for Rozeta and 95.5% for Altaswede. The values as a percentage of the control group because of the 50 mg/l dose effect was 47.6% for Salino, 47.8% for Rozeta, and 54.5% for Altaswede. This represented an average of 95.4 ± 0.21% (1.00 mg/l) and 50.0 ± 3.94% (50.0 mg/l). The coefficient of determination of Salino is 0.9791, while 0.9941 for Rozeta and 0.9611 for Altaswede.

The dose effect of 1 mg/l resulted in a reduction to 91.3% in height of the shoot compared to the control group in Salino (Fig. 2). This value was 90.5% for Rozeta and 93.2% for Altaswede. The average value as a percentage of the control group because of the 50 mg/l dose effect was 41.3% for Salino, 45.2% for Rozeta, and 45.5% for Altaswede. The average of 1.00 mg/l dose affect is 91.7 ± 1.39% and for 50.0 mg/l is 44.0 ± 2.34%. The coefficient of determination of Salino is 0.9439, while 0.9662 for Rozeta and 0.9828 for Altaswede.

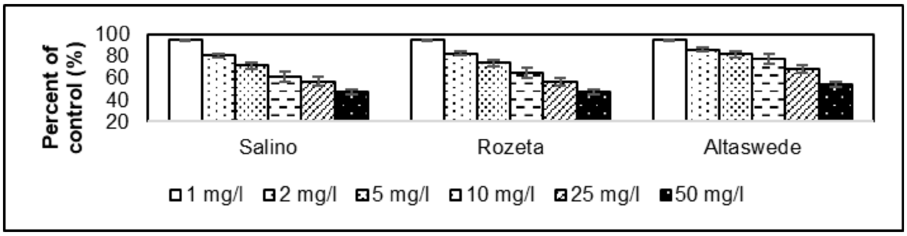


Fig. 1. Comparative examination of root length in *Trifolium pratense* cultivars (Salino, Rozeta, and Altaswede) exposed to asbestos-contaminated irrigation water. Significant reductions in root length were observed at both 1 mg/l and 50 mg/l concentrations ($p \le 0.05$), with greater inhibition at higher doses. The reduction was most prominent in the 50 mg/l dose, where root length was reduced to 47.6% (Salino), 47.8% (Rozeta), and 54.5% (Altaswede) compared to the control group.

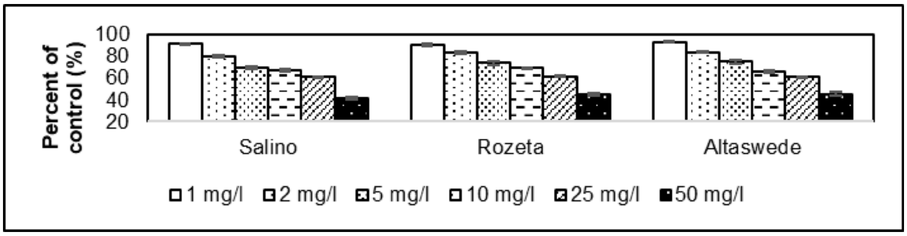


Fig. 2. Comparative examination of shoot height in *Trifolium pratense* cultivars (Salino, Rozeta, and Altaswede) exposed to asbestos-contaminated irrigation water. Significant reductions in shoot height were observed at both 1 mg/l and 50 mg/l concentrations ($p \le 0.05$), with greater inhibition at higher doses. The reduction was most prominent in the 50 mg/l dose, where shoot height was reduced to 41.3% (Salino), 45.2% (Rozeta), and 45.5% (Altaswede) compared to the control group.

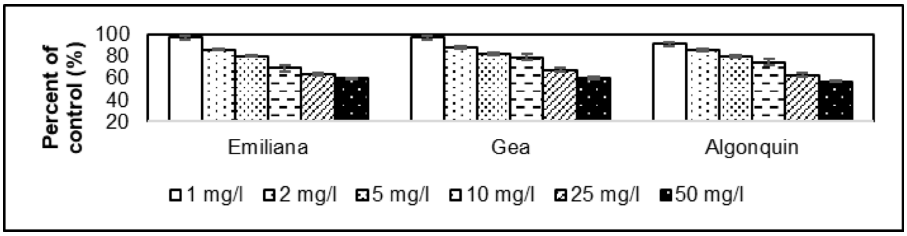


Fig. 3. Comparative examination of root length in *Medicago sativa* cultivars (Emiliana, Gea, and Algonquin) exposed to asbestos-contaminated irrigation water. Significant reductions in root length were observed at both 1 mg/l and 50 mg/l concentrations ($p \le 0.05$), with greater inhibition at higher doses. The reduction was most prominent in the 50 mg/l dose, where root length was reduced to 59.7% (Emiliana), 60.3% (Gea), and 57.1% (Algonquin) compared to the control group.

Results of *Medicago sativa* analysis

The control germination rate is 92.0% for Emiliana, 94.0% for Gea and 90.0% for Algonquin. Germination of *Medicago sativa* seeds at 25.0 °C under light and dark conditions averaged 92.0 ± 2.00%. The tested samples showed that already at a dose concentration of 1 mg/l the germination rate was reduced to 88.0% in the case of the Emiliana and Gea, while it was 86.0% for Algonquin. This was reduced to 54.0% (Emiliana), 50.0% (Gea) and 52.0% (Algonquin) at the 50 mg/l dose, and to 58.7% (Emiliana), 53.2% (Gea) and 57.8% (Algonquin) as a percentage of the control group. The average value for the whole group is 94.9 ± 1.15% for 1.00 mg/l and 56.6 ± 2.95% for 50 mg/l at the percentage of control group. The average difference in the effect of the two dose concentrations is −38.4 ± 1.81%. The coefficient of determination of Emiliana is 0.9827, while 0.9842 for Gea and 0.9754 for Algonquin. The dose effect of 1 mg/l resulted in a reduction to 97.2% in root length compared to the control group in Emiliana (Fig. 3). This rate was 97.1% for Gea and 91.4% for Algonquin. The values as a percentage of the control group because of the 50 mg/l dose effect was 59.7% for Emiliana, 60.3% for Gea, and 57.1% for Algonquin. This represented an average of 95.2 ± 3.30% (1.00 mg/l) and 59.1 ± 1.68% (50.0 mg/l). The coefficient of determination of Emiliana is 0.9781, while 0.9788 for Gea and 0.9844 for Algonquin.

The dose effect of 1 mg/l resulted in a reduction to 98.8% in height of the shoot compared to the control group in Emiliana (Fig. 4). This value was 96.3% for Gea and 96.4% for Algonquin. The average value as a percentage

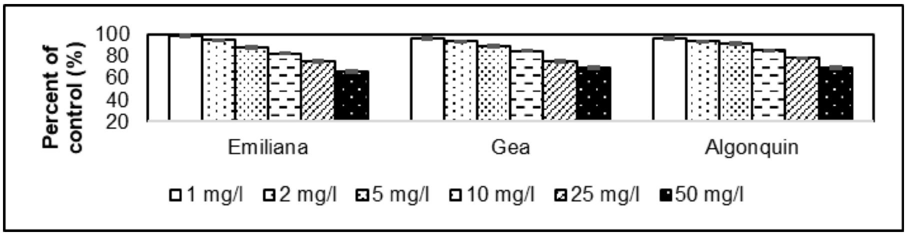


Fig. 4. Comparative examination of shoot height in *Medicago sativa* cultivars (Emiliana, Gea, and Algonquin) exposed to asbestos-contaminated irrigation water. Significant reductions in shoot height were observed at both 1 mg/l and 50 mg/l concentrations ($p \leq 0.05$), with greater inhibition at higher doses. The reduction was most prominent in the 50 mg/l dose, where shoot height was reduced to 65.9% (Emiliana), 69.1% (Gea), and 69.9% (Algonquin) compared to the control group.

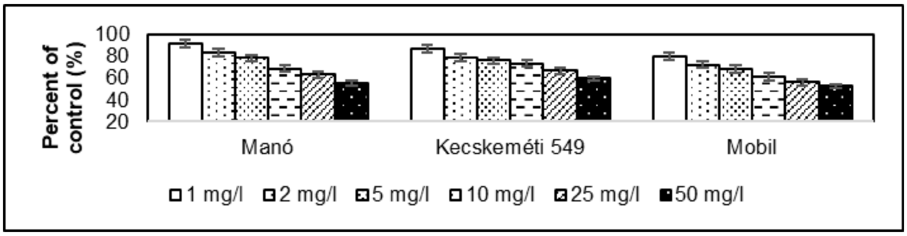


Fig. 5. Comparative examination of root length in *Solanum lycopersicum* cultivars (Manó, Kecskeméti 549, and Mobil) exposed to asbestos-contaminated irrigation water. Significant reductions in root length were observed at both 1 mg/l and 50 mg/l concentrations ($p \leq 0.05$), with greater inhibition at higher doses. The reduction was most prominent in the 50 mg/l dose, where root length was reduced to 55.4% (Manó), 59.7% (Kecskeméti 549), and 52.6% (Mobil) compared to the control group.

of the control group because of the 50 mg/l dose effect was 65.9% for Emiliana, 69.1% for Gea, and 69.9% for Algonquin. The average of 1.00 mg/l dose affect is 97.2 ± 1.41% and for 50.0 mg/l is 68.3 ± 2.14%. The coefficient of determination of Emiliana is 0.9818, while 0.9604 for Gea and 0.9429 for Algonquin.

Results of *Solanum lycopersicum* analysis

The control germination rate is 92.0% for Manó and 90.0% for Kecskeméti 549 and Mobil. Germination of *Solanum lycopersicum* seeds at 25.0 °C under light and dark conditions averaged 90.7 ± 1.15%. The tested samples showed that already at a dose concentration of 1 mg/l the germination rate was reduced to 88.0% in the case of the Manó and Kecskeméti 549, while 86.0% for Mobil. This was reduced to 58.0% (Manó), 60.0% (Kecskeméti 549) and 56.0% (Mobil) at the 50 mg/l dose, and to 63.0% (Manó), 66.7% (Kecskeméti 549) and 62.2% (Mobil) as a percentage of the control group. The average value for the whole group is 96.3 ± 1.26% for 1.00 mg/l and 64.0 ± 2.36% for 50 mg/l at the percentage of control group. The average difference in the effect of the two dose concentrations is −32.4 ± 1.13%. The coefficient of determination of Manó is 0.9909, while 0.9823 for Kecskeméti 549 and 0.9916 for Mobil. The dose effect of 1 mg/l resulted in a reduction to 91.9% in root length compared to the control group in Manó (Fig. 5). This rate was 87.5% for Kecskeméti 549 and 80.3% for Mobil. The values as a percentage of the control group because of the 50 mg/l dose effect was 55.4% for Manó, 59.7% for Kecskeméti 549, and 52.6% for Mobil. This represented an average of 86.6 ± 5.87% (1.00 mg/l) and 55.9 ± 3.57% (50.0 mg/l). The coefficient of determination of Manó is 0.9961, while 0.9710 for Kecskeméti 549 and 0.9908 for Mobil.

The dose effect of 1 mg/l resulted in a reduction to 95.6% in height of the shoot compared to the control group in Manó (Fig. 6). This value was 95,5% for Kecskeméti 549 and 95,6% for Mobil. The average value as a percentage of the control group because of the 50 mg/l dose effect was 56.0% for Manó, 59.6% for Kecskeméti 549 and 57.8% for Mobil. The average of 1.00 mg/l dose affect is 95.6 ± 0.05% and for 50.0 mg/l is 57.8 ± 1.75%. The coefficient of determination of Manó is 0.9916, 0.9863 for Kecskeméti 549 and 0.9941 for Mobil.

Discussion

Previous research[28]has shown that leguminous plants respond to chemical stressors in a dose-dependent manner[29,30]. Our findings confirm this trend, as *Trifolium pratense* exhibited significantly lower germination rates at higher doses.

This sensitivity is critical to understanding the ecological implications of chemical exposure, as legumes play a vital role in soil health and fertility due to their ability to fix atmospheric nitrogen[31,32]. The low germination rates at higher doses indicate that *Trifolium pratense* has a relatively low chemical tolerance. This may negatively impact its population dynamics and the stability of the surrounding ecosystem.

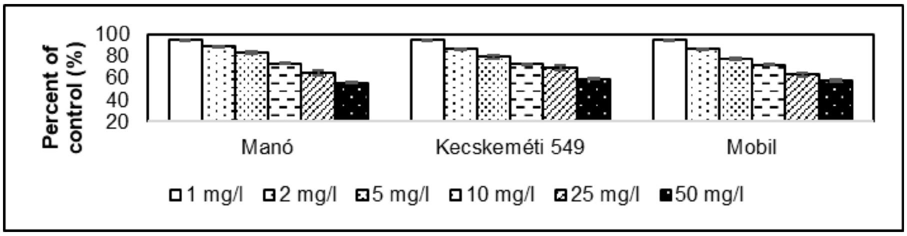


Fig. 6. Comparative examination of shoot height in *Solanum lycopersicum* cultivars (Manó, Kecskeméti 549, and Mobil) exposed to asbestos-contaminated irrigation water. Significant reductions in shoot height were observed at both 1 mg/l and 50 mg/l concentrations ($p \leq 0.05$), with greater inhibition at higher doses. The reduction was most prominent in the 50 mg/l dose, where shoot height was reduced to 56.0% (Manó), 59.6% (Kecskeméti 549), and 57.8% (Mobil) compared to the control group.

Many studies have shown that leguminous plants suffer reduced growth and reproductive success when exposed to pollutants[33,34]. Our findings support this, highlighting the potential risks to their ecological role and agricultural importance. The results are not only striking in the case of *Trifolium pratense*, but also evident in the two other receptors, which similarly responded to the effects of chemical stressors. Throughout the studies, we observed a significant decrease in germination rates for the other two receptors as well, reinforcing the notion that chemical pollution has widespread effects on various plant species. These findings suggest that the examined chemical substances exert influence not only on specific plants but also on surrounding ecosystems. The diversity of reactions triggered by chemical stressors may yield important insights into maintaining ecological balance and underscores the necessity for further research to achieve a deeper understanding of this phenomenon.

It is crucial to consider that the observed effects may be influenced by the concurrent action of multiple stressors present in the environment. For instance, the leaching of other toxic substances, such as heavy metals or construction-related contaminants, from polluted sites could also contribute to the observed responses. Investigating these interactions and isolating the effects of a single chemical stressor would necessitate further research into the potential cumulative impacts of various pollutants.

Asbestos exposure can disrupt key biological processes in plants. This disruption can lead to abnormal growth patterns, especially in the root and shoot systems[35]. Additionally, asbestos exposure may impair nutrient uptake by damaging root membranes or inducing oxidative stress, which compromises cellular transport proteins involved in nutrient absorption[36–38]. These cellular disturbances contribute to significant reductions in germination rates, root growth, and shoot height observed in the study, as plants exposed to high asbestos concentrations exhibit stunted development and decreased ability to absorb essential nutrients[39,40].

The effects of asbestos on plant growth can also be linked to oxidative stress pathways. When exposed to toxic substances like asbestos, plants often experience an increase in reactive oxygen species, which can damage lipid membranes, proteins, and DNA[41,42]. This oxidative stress disrupts normal cell functioning, leading to impaired metabolic processes and growth inhibition. Specifically, asbestos exposure may trigger an imbalance between antioxidants and ROS, overwhelming the plant's defense systems. As a result, plants may exhibit stunted growth, reduced germination rates, and inhibited root elongation, as observed in our study[43]. These effects align with previous research indicating that oxidative stress is a major factor in plant response to chemical pollutants, as ROS accumulation can interfere with nutrient uptake and cellular respiration[44]. Therefore, oxidative stress pathways may play a central role in the plant's inability to tolerate high concentrations of asbestos, contributing to the observed toxic effects on plant development.

Similarly, the significant reduction in root growth at 50 mg/l aligns with other research[45]showing inhibition of root elongation by comparable phytotoxin concentrations[46–48]. The decrease in shoot height under higher dose conditions is supported by findings[49]documenting growth inhibition under stressful environmental toxin conditions, involving hormonal imbalances and nutritional uptake disruptions[50,51]. These dose-dependent inhibitory effects are consistent with broader plant toxicology results[52]where higher doses impact plant physiological and metabolic processes[53–57].

In light of these complexities, it is essential to consider potential confounding factors, such as variations in nutrient availability, water quality, or climatic conditions, which may further complicate the interpretation of dose-response relationships. These factors could independently influence plant health, irrespective of the chemical stressors under investigation. Therefore, it is recommended that future studies either control for these variables or conduct research in a more controlled experimental setup to more accurately isolate the effects of the pollutants.

These findings suggest important implications for environmental policies and strategies to mitigate pollution. The observed phytotoxic effects emphasize the necessity of regulating contaminant levels in agricultural and natural environments to safeguard plant biodiversity and ecosystem stability[58,59]. Implementing more stringent guidelines on permissible pollutant concentrations in irrigation water and soil can help minimize the adverse impacts of chemical stressors[60]. Additionally, the results highlight the importance of adopting sustainable agricultural practices, such as phytoremediation and buffer zone management, to alleviate the impact of contaminants on vegetation[61,62]. These approaches align with broader environmental policies focused on enhancing soil health, maintaining biodiversity, and ensuring food security[63].

Such findings highlight the importance of considering dose-response relationships in assessing the risks posed by chemical stressors, particularly in environments where these species are cultivated or naturally occur. Further research is warranted to explore the mechanisms behind this sensitivity and to evaluate the long-term implications for plant communities and agricultural practices. The wider environmental ramifications of these findings also highlight the need for more robust mitigation strategies and policy interventions[64,65]. Given the demonstrable adverse impacts of chemical contamination, including asbestos pollution, on plant growth and ecosystem resilience, regulatory frameworks should prioritize monitoring and remediation initiatives[66]. This underscores the imperative of integrating scientific evidence into policy development to mitigate environmental degradation and safeguard biodiversity[5]. The findings highlight the need to address the real-world challenges for farming in areas with asbestos contamination. The negative impacts on plant growth and germination show the risks to crops grown in polluted soils or irrigated with asbestos-tainted water[67]. Farmers and land managers in affected regions should explore alternative water sources, soil remediation methods, and more resilient plant species[68]. This is especially crucial for sustainable agriculture in historically industrial or asbestos-exposed areas[14]. Further research should focus on practical ways to mitigate these impacts while maintaining agricultural productivity and food safety.

Conclusions

The paper's results indicate a relationship between the dosage and its effect on seed germination, root length, and shoot height in *Trifolium pratense*, *Medicago sativa*, and *Solanum lycopersicum*. At 1 mg/l concentration, there was minimal impact on germination and growth similar to the control conditions. However, at the higher dosage of 50 mg/l, all parameters showed significant decreases. This significant decrease demonstrates how vulnerable seed germination and seedling development are to increased chemical exposure. The high determination coefficients for germination, root length, and shoot height underline the strong connection between higher dosages leading to negative effects. Despite consistent patterns observed across different cultivars indicates that inherent genetic factors affect resilience. These findings stress the need to understand specific responses of each species or cultivar for better agricultural management aiming at protecting crop yield as well as environmental health. There is also a necessity for further research into identifying both physiological and molecular mechanisms behind these responses which will help enhance crop resilience strategies.

Data availability

The datasets generated during and analysed during the current paper are not publicly available but are available from the corresponding author on reasonable request.

Received: 12 December 2024; Accepted: 2 May 2025
Published online: 09 May 2025

References

1. Maulida, P. T. et al. Environmental assessment of friable asbestos from soil to air using the releasable asbestos sampler (RAS). *Toxics* **10**, 103390toxics10120748 (2022).
2. Boffetta, P. Classic epidemiology of lung cancer. *IASLC Thorac. Oncol.* **1-8.e3** https://doi.org/10.1016/B978-0-323-52357-8.00001-9 (2018).
3. Mahoney, K. et al. The past, present and future of Asbestos-Related diseases in Australia: what are the data telling Us? *Sustainability* **15** https://doi.org/10.3390/su15118492 (2023).
4. Van Orden, D. R. Asbestos, in Environmental Forensics, *Elsevier*, pp. 19–33. (1964). https://doi.org/10.1016/B978-012507751-4/50024-0
5. Li, X. et al. Asbestos-Environment pollution characteristics and Health-Risk assessment in typical Asbestos-Mining area. *Toxics* **11** https://doi.org/10.3390/toxics11060494 (2023).
6. Lucci, F. et al. Naturally occurring asbestos (NOA) in granitoid rocks, A case study from Sardinia (Italy). *Minerals* **8** https://doi.org/10.3390/min8100442 (2018).
7. Kent, K. et al. The impact of the COVID-19 pandemic on rural food security in high income countries: A systematic literature review. *Int. J. Environ. Res. Public Health.* **19** https://doi.org/10.3390/ijerph19063235 (2022).
8. Krówczyńska, M. et al. Asbestos—Cement roofing identification using remote sensing and convolutional neural networks (CNNs). *Remote Sens.* **12** https://doi.org/10.3390/rs12030408 (2020).
9. Xu, L. Metals contamination along the watershed and estuarine areas of Southern Bohai Sea, China. *Mar. Pollut. Bull.* **74** https://doi.org/10.1016/j.marpolbul.2013.06.010 (2013).
10. Spurny, K. R. On the release of asbestos fibers from weathered and corroded asbestos cement products. *Environ. Res.* **48** https://doi.org/10.1016/S0013-9351(89)80089-1 (1989).
11. Musthaq, M. A. et al. The effect of various environmental conditions on the impact damage behaviour of Natural-Fibre-Reinforced composites (NFRCs)—A critical review. *Polymers* **15** https://doi.org/10.3390/polym15051229 (2023).
12. Wohlleben, W. et al. Everything falls apart: How solids degrade and release nanomaterials, composite fragments, and microplastics. *NanoImpact* **34** (2024). https://doi.org/10.1016/j.impact.2024.100510
13. Xie, S. Y. et al. In vitro inhalation bioaccessibility for particle-bound hydrophobic organic chemicals: method development, effects of particle size and hydrophobicity, and risk assessment. *Environ. Int.* **120** https://doi.org/10.1016/j.envint.2018.08.015 (2018).
14. Peña-Castro, M. et al. A critical review of asbestos concentrations in water and air, according to exposure sources. *Heliyon* **9** https://doi.org/10.1016/j.heliyon.2023.e15730 (2023).
15. Favero-Longo, S. E. et al. Lichens on asbestos–cement roofs: bioweathering and biocovering effects. *J. Hazard. Mater.* **162** https://doi.org/10.1016/j.jhazmat.2008.06.060 (2009).
16. Zdeb, M. et al. The quality of rainwater collected from roofs and the possibility of its economic use. *Resources* **9** https://doi.org/10.3390/resources9020012 (2020).
17. Ervik, T. et al. Mobilization of asbestos fibers by weathering of a corrugated asbestos cement roof. *J. Occup. Environ. Hyg.* **18** https://doi.org/10.1080/15459624.2020.1867730 (2021).
18. Anabtawi, F. et al. Heavy metals in harvested rainwater used for domestic purposes in rural areas: Yatta area, Palestine as a case study. *Int. J. Environ. Res. Public. Health.* **19**, 2683. https://doi.org/10.3390/ijerph19052683 (2022).

19. Trivedi, A. K. et al. Environmental contamination of Chrysotile asbestos and its toxic effects on growth and physiological and biochemical parameters of *Lemna gibba*. *Arch. Environ. Contam. Toxicol.* **47** https://doi.org/10.1007/s00244-004-3161-7 (2004).
20. Bloise, A. et al. Potentially toxic elements (PTEs) associated with asbestos Chrysotile, tremolite and actinolite in the Calabria region (Italy). *Chem. Geol.* **558**, 119896. https://doi.org/10.1016/j.chemgeo.2020.119896 (2020).
21. Trivedi, A. K. & Ahmad, I. Effects of Chrysotile asbestos contaminated soil on crop plants. soil and sediment contamination. *Int. J.* **20** https://doi.org/10.1080/15320383.2011.609197 (2011).
22. Warnasooriya, S. N. & Montgomery, B. L. Spatial-specific regulation of root development by phytochromes in *Arabidopsis thaliana*. *Plant Signal. Behav.* **6** https://doi.org/10.4161/psb.6.12.18267 (2011).
23. Benedetti, S. et al. Reactive oxygen species: a double-edged sword for mesothelioma. *Oncotarget* **6**, 16848–16865. https://doi.org/10.18632/oncotarget.4253 (2015).
24. Trivedi, A. K. et al. Environmental contamination of Chrysotile asbestos and its toxic effects on antioxidative system of *Lemna gibba*. *Arch. Environ. Contam. Toxicol.* **52** https://doi.org/10.1007/s00244-006-0056-9 (2007).
25. Trivedi, A. K. & Ahmad, I. Impact of Chrysotile asbestos contaminated soil on foliar nutrient status of plants. *Indian J. Plant. Physiol.* **18** https://doi.org/10.1007/s40502-013-0036-9 (2013).
26. Trivedi, A. K. & Ahmad, I. Genotoxicity of Chrysotile asbestos on *Allium cepa L.* meristematic root tip cells. *Curr. Sci.* **105**, (2013).
27. Saleem, K. et al. Chrysotile-Asbestos-Induced damage in *Panicum virgatum* and *Phleum pretense* species and its alleviation by Organic-Soil amendment. *Sustainability* **14** https://doi.org/10.3390/su141710824 (2022).
28. Wang, Q. R. et al. Responses of legume and non-legume crop species to heavy metals in soils with multiple metal contamination. *J. Environ. Sci. Health Part. A*. **37** https://doi.org/10.1081/ESE-120003241 (2002).
29. Pandey, A. K. et al. The Anatomical Basis of Heavy Metal Responses in Legumes and Their Impact on Plant–Rhizosphere Interactions. *Plants* **11** (2022). https://doi.org/10.3390/plants11192554
30. Matamoros, M. A. & Becana, M. Molecular responses of legumes to abiotic stress: post-translational modifications of proteins and redox signaling. *J. Exp. Bot.* **72** https://doi.org/10.1093/jxb/erab008 (2021).
31. Basu, S. et al. Diversity and traits of multiple biotic stressors elicit differential defense responses in legumes. *Agriculture* **13** (2093). https://doi.org/10.3390/agriculture13112093 (2023).
32. Gupta, S. & Das, S. Insight into the Molecular Interaction Between Leguminous Plants and Rhizobia Under Abiotic Stress. *In Molecular Plant Abiotic Stress* 301–314. *Wiley*, (2019). https://doi.org/10.1002/9781119463665.ch16
33. Abd-Alla, M. H. et al. Enhancing Rhizobium–Legume symbiosis and reducing nitrogen fertilizer use are potential options for mitigating climate change. *Agriculture* **13** (2092). https://doi.org/10.3390/agriculture13112092 (2023).
34. Gepts, P. et al. Legumes as a Model Plant Family. Genomics for Food and Feed Report of the Cross-Legume Advances through Genomics Conference. *Plant Physiology* **137** (2005). https://doi.org/10.1104/pp.105.060871
35. Łuniewski, S. et al. Plants, microorganisms and their metabolites in supporting asbestos Detoxification—A biological perspective in asbestos treatment. *Materials* **17** https://doi.org/10.3390/ma17071644 (2024).
36. Roy, T. et al. Microplastic/nanoplastic toxicity in plants: an imminent concern. *Environ. Monit. Assess.* **195** https://doi.org/10.1007/s10661-022-10654-z (2023).
37. Burachevskaya, M. et al. Accumulation, translocation, and toxicity of arsenic in barley grown in contaminated soil. *Plant. Soil.* **467** https://doi.org/10.1007/s11104-021-05067-9 (2021).
38. Afzal, S. et al. Nanostructure and plant uptake: assessing the ecological footprint and root-to-leaf dynamics. *Plant. Nano Biol.* **10**, 100122. https://doi.org/10.1016/j.plana.2024.100122 (2024).
39. Shi, G. L. et al. Accumulation and distribution of arsenic and cadmium in winter wheat (*Triticum aestivum L.*) at different developmental stages. *Sci. Total Environ.* **667** https://doi.org/10.1016/j.scitotenv.2019.02.394 (2019).
40. Gonneau, C. et al. Framework for assessment and phytoremediation of asbestos-contaminated sites. *Environ. Sci. Pollut. Res.* **24** https://doi.org/10.1007/s11356-017-0177-x (2017).
41. Pietrofesa, R. et al. Asbestos induces oxidative stress and activation of Nrf2 signaling in murine macrophages: chemopreventive role of the synthetic Lignan secoisolariciresinol diglucoside (LGM2605). *Int. J. Mol. Sci.* **17** https://doi.org/10.3390/ijms17030322 (2016).
42. Jomova, K. et al. Reactive oxygen species, toxicity, oxidative stress, and antioxidants: chronic diseases and aging. *Arch. Toxicol.* **97** https://doi.org/10.1007/s00204-023-03562-9 (2023).
43. Pizzino, G. et al. Oxidative Stress: Harms and Benefits for Human Health. *Oxidative Medicine and Cellular Longevity* (2017). https://doi.org/10.1155/2017/8416763 (2017).
44. Jarin, A. S. et al. Drought stress tolerance in rice: physiological and biochemical insights. *Int. J. Plant. Biology*. **15** https://doi.org/10.3390/ijpb15030051 (2024).
45. Agudo-Jurado, F. J. et al. Status of phytotoxins isolated from necrotrophic fungi causing diseases on grain legumes. *Int. J. Mol. Sci.* **24**, 5116. https://doi.org/10.3390/ijms24065116 (2023).
46. Lin, D. & Xing, B. Phytotoxicity of nanoparticles: Inhibition of seed germination and root growth. *Environ. Pollut.* **150** https://doi.org/10.1016/j.envpol.2007.01.016 (2007).
47. Pan, M. & Chu, L. M. Phytotoxicity of veterinary antibiotics to seed germination and root elongation of crops. *Ecotoxicol. Environ. Saf.* **126** https://doi.org/10.1016/j.ecoenv.2015.12.027 (2016).
48. Wang, X. et al. Validation of germination rate and root elongation as indicator to assess phytotoxicity with *Cucumis sativus*. *Chemosphere* **44** https://doi.org/10.1016/S0045-6535(00)00520-8 (2001).
49. Pérez-Labrada, F. et al. Underlying Mechanisms of Action to Improve Plant Growth and Fruit Quality in Crops under Alkaline Stress. In Abiotic Stress in Crop Plants - Ecophysiological Responses and Molecular Approaches. *IntechOpen* (2024). https://doi.org/10.5772/intechopen.114335
50. Raza, A. et al. Plant hormones and neurotransmitter interactions mediate antioxidant defenses under induced oxidative stress in plants. *Front. Plant Sci.* **13** https://doi.org/10.3389/fpls.2022.961872 (2022).
51. Georgieva, M. & Vassileva, V. Stress management in plants: examining provisional and unique Dose-Dependent responses. *Int. J. Mol. Sci.* **24** https://doi.org/10.3390/ijms24065105 (2023).
52. Hafeez, A. et al. Effect of heavy metals on growth, physiological and biochemical responses of plants. In Plants and Their Interaction to Environmental Pollution (pp. 139–159). *Elsevier*, (2023). https://doi.org/10.1016/B978-0-323-99978-6.00006-6
53. dos Santos, T. B. et al. Physiological responses to drought, salinity, and heat stress in plants: A review. *Stresses* **2** https://doi.org/10.3390/stresses2010009 (2022).
54. Khan, N. et al. Metabolic and physiological changes induced by plant growth regulators and plant growth promoting rhizobacteria and their impact on drought tolerance in *Cicer arietinum L. PLOS ONE*. **14** https://doi.org/10.1371/journal.pone.0213040 (2019).
55. Gudkov, S. V. et al. Effect of ionizing radiation on physiological and molecular processes in plants. *J. Environ. Radioact.* **202** https://doi.org/10.1016/j.jenvrad.2019.02.001 (2019).
56. Hafiz, F. B. et al. Plant nutrition: physiological and metabolic responses, molecular mechanisms and chromatin modifications. *Int. J. Mol. Sci.* **23** https://doi.org/10.3390/ijms23084084 (2022).
57. Zandalinas, S. I. et al. Plant responses to climate change: metabolic changes under combined abiotic stresses. *J. Exp. Bot.* **73** https://doi.org/10.1093/jxb/erac073 (2022).
58. Durante-Yánez, E. V. et al. Phytoremediation of Soils Contaminated with Heavy Metals from Gold Mining Activities Using Clidemia sericea D. Don. *Plants* 11, (2022). https://doi.org/10.3390/plants11050597

59. Sorrentino, M. C. et al. Facing metal stress by multiple strategies: morphophysiological responses of cardoon (*Cynara cardunculus L.*) grown in hydroponics. *Environ. Sci. Pollut. Res.* **28** https://doi.org/10.1007/s11356-021-13242-9 (2021).
60. Alengebawy, A. et al. Heavy metals and pesticides toxicity in agricultural soil and plants: ecological risks and human health implications. *Toxics* **9**, 42. https://doi.org/10.3390/toxics9030042 (2021).
61. Sharma, J. K. et al. Phytoremediation technologies and their mechanism for removal of heavy metal from contaminated soil: an approach for a sustainable environment. *Front. Plant Sci.* **14** https://doi.org/10.3389/fpls.2023.1076876 (2023).
62. Hayyat, M. U. et al. Evaluating the phytoremediation potential of Eichhornia crassipes for the removal of cr and Li from synthetic polluted water. *Int. J. Environ. Res. Public Health*. **20** https://doi.org/10.3390/ijerph20043512 (2023).
63. Cozim-Melges, F. et al. Farming practices to enhance biodiversity across biomes: a systematic review. *Npj Biodivers.* **3** https://doi.org/10.1038/s44185-023-00034-2 (2024).
64. Nowwar, A. I. et al. Impact of irrigation with wastewater on accumulation of heavy metals in Phaseolus vulgaris L. and its remediation. *J. Soil. Sci. Plant. Nutr.* **23** https://doi.org/10.1007/s42729-022-01080-8 (2023).
65. Brunhoferova, H. et al. Removal of 27 micropollutants by selected wetland macrophytes in hydroponic conditions. *Chemosphere* **281**, 130980. https://doi.org/10.1016/j.chemosphere.2021.130980 (2021).
66. Singh, V. et al. Heavy metal contamination in the aquatic ecosystem: toxicity and its remediation using Eco-Friendly approaches. *Toxics* **11** https://doi.org/10.3390/toxics11020147 (2023).
67. Popov, M. et al. Effect of arsenic soil contamination on stress response metabolites, 5-Methylcytosine level and CDC25 expression in spinach. *Toxics* **11**, 10–3390. toxics11070568 (2023).
68. Roy, A. et al. Nanomaterials for remediation of environmental pollutants. *Bioinorg. Chem. Appl.* **2021** https://doi.org/10.1155/2021/1764647 (2021).

Author contributions
All authors reviewed the manuscript.

Funding
Open access funding provided by Széchenyi István University (SZE).

“Supported by the ÚNKP-23-3-I New National Excellence Program of the Ministry for Culture and Innovation from the source of the National Research, Development and Innovation Fund.”

Declarations

Competing interests
The authors declare no competing interests.

Additional information
Correspondence and requests for materials should be addressed to G.Z.M.

Clean Technologies and Environmental Policy (2025) 27:8955–8965
https://doi.org/10.1007/s10098-025-03197-7

ORIGINAL PAPER

Water quality of harvested rainwater from asbestos cement roofs and its suitability for irrigation

Gergely Zoltán Macher[1] · Dóra Beke[2] · András Torma[1]

Received: 1 October 2024 / Accepted: 6 May 2025 / Published online: 30 May 2025

Abstract
Rainwater harvesting is increasingly recognized as a sustainable solution for water management, especially in areas affected by water scarcity. However, the potential pollution risks linked to asbestos cement roofing materials, widely used in residential areas, have not been thoroughly explored. This paper addresses this gap by assessing the effects of asbestos cement roof degradation on the quality of harvested rainwater, focusing on its suitability for irrigation. The research combines controlled laboratory experiments and field-based sampling of rainwater in contact with asbestos cement surfaces. Significant changes in water quality were observed, notably, the analysis highlights a substantial increase in heavy metal contamination, including a 75.4% rise in mercury concentration and a dramatic 127.3% increase in lead levels. The findings underscore the need for extensive investigations into the impact of asbestos cement roofs on water quality in Hungary, considering the varying characteristics and contamination levels of asbestos cement products across countries as well as the limited existing scholarship on the subject in Hungary. This research is novel in its pioneering examination of the effects of asbestos cement roofing on harvested rainwater quality in Hungary, with a particular focus on heavy metal contamination (e.g., lead, mercury, zinc), which poses significant environmental and health risks.

Keywords Asbestos cement · Irrigation water · Harvested rainwater · Water quality · Water contamination

Introduction

Irrigated agriculture, despite occupying only 17% of the total cultivated land, constitutes 40% of global food and agricultural output, highlighting its substantial impact on worldwide food security (Chen et al. 2017). However, the critical role of irrigated agriculture in mitigating food shortages driven by economic and population expansion is eclipsed by its growing water and energy demands, presenting a serious challenge to regional water security and accelerating global climate change (Conway et al. 2015). Rainwater is one of the most commonly used and flexible alternative water sources, which can be employed in agricultural, horticultural, and particularly residential settings.

Rainwater harvesting systems allow communities to access abundant rainwater, reducing pressure on public water supplies and overall water needs (Imteaz et al. 2013). This strategy enhances water security by enabling residential water collection (Abdulla and Al-Shareef 2009). Up to 33% of urban gardens can achieve irrigation water neutrality by collecting rainwater from building rooftops. Additionally, the remaining gardens could fulfill 44% of their water needs through high-efficiency irrigation (Lupia et al. 2017). Rainwater collection is especially recommended to enhance water supply reliability for watering small gardens where traditional irrigation is not viable (Bates et al. 2008). Rainwater harvesting encompasses various methods of gathering, storing, and distributing water to communities from natural and human-made surfaces in rural and urban settings (Pachpute et al. 2009); however, the materials or contamination of these storage mediums are not always a top priority due to infrastructure limitations or cost concerns. Nevertheless,

✉ Gergely Zoltán Macher
macher.gergely.zoltan@sze.hu

1 Department of Applied Sustainability, Albert Kázmér Faculty of Agricultural and Food Sciences of Széchenyi István University in Mosonmagyaróvár, Széchenyi István University, Győr, Hungary

2 Department of Plant Sciences, Albert Kázmér Faculty of Agricultural and Food Sciences of Széchenyi István University in Mosonmagyaróvár, Széchenyi István University, Mosonmagyaróvár, Hungary

the roofing or storage material can impact the quality of collected rainwater (Mendez et al. 2011).

Asbestos cement (abbreviation: AC) roofs, in particular, are a common roofing material in many developing regions, but their impact on rainwater quality remains an underexplored area (Kim et al. 2005).

AC roofing was widely used globally in the mid-to-late twentieth century due to its durability and fire-resistant properties (Martínez et al. 2024). However, the health risks associated with asbestos exposure led to strict regulations and a subsequent decline in its use across many countries (Kamp 2009). Despite this initial durability, these roofs gradually degrade over time, becoming less durable and more susceptible to damage after approximately 30–40 years of use. According to Spurny (1989) and Suzuki et al. (2005), a corrugated sheet made of AC might release several grams of asbestos annually from its matrix structure. These released asbestos fibers and matrix materials from AC have the potential to contaminate rainwater, leading to water pollution, soil contamination, and harmful effects on plant life (Malinconico et al. 2022). The composition and physical state of the roofing material directly influence its interaction with rainwater, which can consequently impact the chemical characteristics of the water and the presence of contaminants (Sanjeeva and Puttaswamaiah 2018). However, comprehensive data on the water quality from different roofing materials remains scarce (Enedoh 2023). As not only asbestos but also other substances such as metal compounds may leach from aged, deteriorated roofing materials, a comprehensive understanding of these interactions is crucial (Chang et al. 2004).

Although AC roofing materials often contain heavy metal additives and potential contaminants, there is a lack of research examining their impacts (Favero-Longo et al. 2009). Asbestos fibers may also be released due to the general deterioration and damage of asbestos-containing materials, as well as in the event of disasters or environmental extremes caused by climate change (Saba et al. 2024). While the health risks associated with inhaling airborne asbestos are well-documented, less is known about the potential hazards posed by asbestos fibers suspended in water, which represents an important environmental exposure pathway (Magherini et al. 2023). Areas with widespread use of AC roofs and rainwater harvesting face heightened risks of water contamination (Ghosh and Ahmed 2022; Gwenzi et al. 2015; Lye 2002). This can harm human health and environmental systems, as prolonged exposure to the polluted water may disrupt soil microbial communities and the overall water–soil–plant system (Gopishankar et al. 2022; Goutam Mukherjee et al. 2022; Rashid et al. 2023). Therefore, it is crucial to phase out and properly dispose of AC construction materials, while also ensuring proper filtration and maintenance of rainwater harvesting systems to address this issue (Peña-Castro et al. 2023). Additionally, it is essential to strengthen water quality regulations and intensify monitoring of pollutants to minimize the risks associated with asbestos-induced water contamination (Kadadou et al. 2024). However, AC roofing remains widespread in areas with less rigorous regulations or where the removal and replacement of existing asbestos materials poses significant financial challenges (Martínez et al. 2024).

This paper aims to address this research gap by evaluating the water quality of rainwater harvested from AC roofs and determining its suitability for irrigation purposes. While previous studies (Y. Kim et al. 2017; Nicholson et al. 2010; Oyilieze Akanwa 2020) have examined rainwater harvesting and its water quality implications, this paper uniquely explores the impacts of a commonly used but understudied roofing material—asbestos cement. These studies have identified issues like heavy metal leaching, bacterial contamination, and other water quality concerns. However, the water quality of rainwater harvested from AC roofs remains an underexplored area. The unique properties of AC roofs and their potential to affect rainwater quality have not been adequately addressed. To fill this research gap, this study provides a comprehensive analysis of the water quality characteristics of rainwater harvested from AC roofs and its suitability for irrigation, which can inform decision-making and water management strategies in regions where AC roofs are prevalent.

Materials and methods

This paper employs a mixed-methods approach, incorporating both field-based data collection and laboratory analysis. The assessment focuses on AC roofing material, which, despite the ban on its new usage, is still widely used in Hungary due to the ongoing presence of previously installed products. The specific roofing product under examination is the corrugated AC, which was extensively employed for residential and ancillary buildings constructed during the 1980s. The composition primarily includes chrysotile asbestos, constituting approximately 8–10% of the total material, with the remaining being cement and other binders (Tóth and Weiszburg 2011).

Sample collection points

Precipitation samples were gathered at 10–10 different locations in Győr. In Győr, as in several other cities in Hungary, a significant number of buildings still feature AC roofs, commonly found in residential and other structures constructed during the 1980s. Győr was historically characterized by a high prevalence of residential buildings featuring asbestos materials, primarily due to a local factory that manufactured asbestos-containing insulation and construction panels.

At the initial 10 sites, rainwater was collected while in direct contact with AC roofing, flowing over it before being gathered. In contrast, at the subsequent 10 points, rainwater was collected directly without any surface contamination or contact. The collection of samples took place using 10-L glass containers. The selection of sampling dates was determined by the weather conditions and the air quality indicators that preceded them. The collected samples were not exposed to any other materials such as galvanized tin or other substances that could alter the water chemistry parameters (Fig. 1).

The examined roofing materials represented varying ages and degrees of erosion. Although detailed mineralogical and surface characterization was not available, visual features of the samples were documented. All examined sheets displayed significantly weathered surfaces, visible cracking, and discoloration. Samples used in the distilled water simulation were collected from the same rooftops. During sample collection, the intensity and duration of precipitation events were also recorded. Field conditions closely resembled the simulated scenario, with an average rainfall intensity of approximately 10 mm per 5 min and typical event durations ranging from 15 to 20 min. From each rainfall event, only the first 5 L were collected to focus on the initial runoff phase, which is expected to contain the highest concentrations of pollutants due to the wash-off effect.

Simulated samples

Simultaneously with gathering the precipitation samples, artificial samples were generated using distilled water. The distilled water was polluted through a leaching process on a weathered AC sheet with corrugated features and properties resembling those of roofing materials. This approach provided insights into how the chemical characteristics of the artificially created sample might be influenced by distorting factors and variable elements when compared to pure distilled water with established parameters.

During the distilled water simulation, 1 L of water was poured across the AC sheet within 300 s. This time frame was selected to simulate a moderate rainfall scenario, corresponding to an average precipitation rate of approximately 10 mm per 5 min in the study area. The aim was to replicate the intensity of natural precipitation within a controlled experimental environment.

The need for artificial samples also arose from the fact that precipitation naturally exhibits a specific water quality influenced by weather conditions and air pollution levels, which could potentially distort data from other sampling locations, possibly in different regions. By creating simulated samples, the study aimed to investigate the possibility of contamination resulting from direct, initial contact with the material. It specifically focused on examining the impact of weathered, degraded corrugated AC roofing material on a medium under normalized conditions, highlighting how such a material might affect the water's chemical composition. This approach allowed for a clearer understanding of how the eroded, degraded nature of AC roofing materials could contribute to contamination in rainwater.

Laboratory analysis

The collected precipitation samples underwent laboratory analyses to evaluate various water quality characteristics.

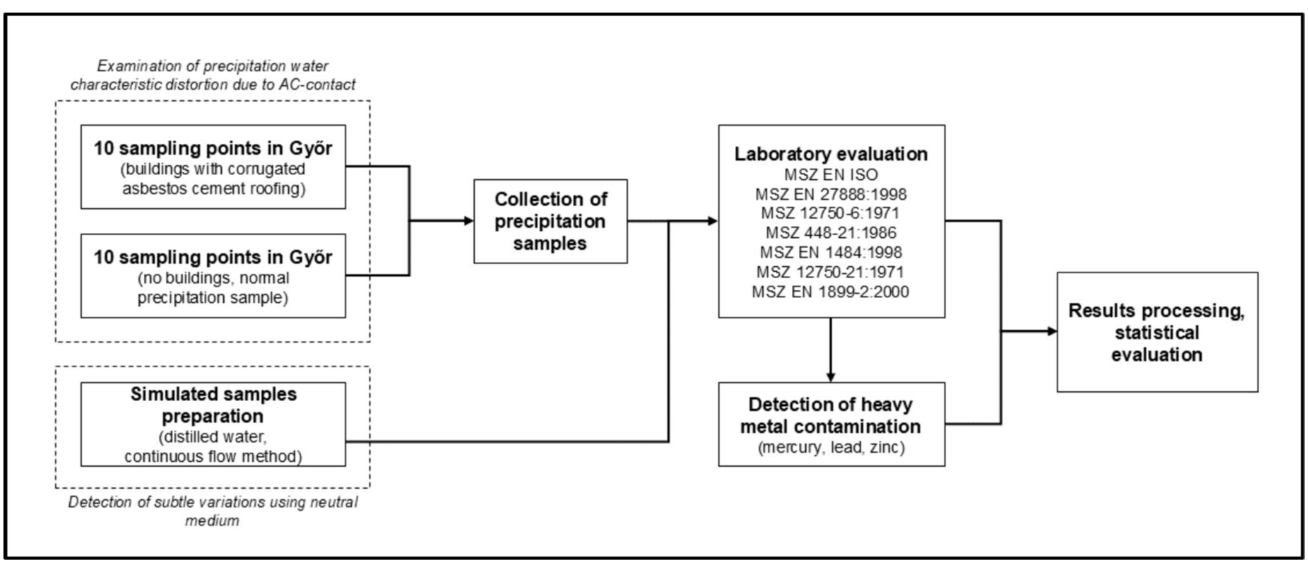


Fig. 1 Schematic diagram of setup process and sampling points. AC = asbestos cement

The selection of quality parameters was based on recommendations from previous research (Mendez et al. 2011; Tengan and Akoto 2022). Rainwater samples were analyzed for a range of water quality parameters, including pH, turbidity, electrical conductivity, oxygen demand, total dissolved solids, and heavy metals (such as mercury, lead, and zinc). These water quality parameters are commonly used to assess the suitability of water for irrigation purposes, as they can provide insights into the chemical, physical, and biological characteristics of the water (Kim et al. 2005; Nicholson et al. 2010; Oyilieze Akanwa 2020). This is particularly applicable when considering water collection and utilization in urban settings, where water quality monitoring also includes the measurement of these parameters. These parameters were chosen to provide a comprehensive understanding of the water quality characteristics and potential suitability for irrigation purposes. The laboratory analysis will follow standard methods for water quality testing, as outlined by relevant national and international guidelines.

The pH was determined according to MSZ EN ISO 10523:2012, providing insight into the water's acidity or alkalinity. Electrical conductivity measured using MSZ EN 27888:1998, served as an indicator of the water's ionic content and overall purity. Total suspended solids were quantified in compliance with MSZ 12750-6:1971, offering information on water clarity. The water's hardness, reflecting the concentration of multivalent cations such as calcium and magnesium, was assessed according to MSZ 448-21:1986. The total organic carbon content was measured based on MSZ EN 1484:1998, providing insights into the presence of organic compounds, while chemical oxygen demand and biological oxygen demand were evaluated following MSZ 12750-21:1971 and MSZ EN 1899-2:2000, respectively, to assess the oxygen-depleting potential and levels of organic pollutants in the water.

The research also examined the presence of heavy metals, such as mercury, lead, and zinc, to identify potential toxic contaminants. The inclusion of these metals is warranted due to their significant environmental and health risks. Mercury is highly toxic and can bioaccumulate in aquatic food chains, posing serious health hazards even at low concentrations. Lead, a prevalent historical contaminant in both industrial and residential environments, remains a significant concern due to its persistent presence in the environment and its harmful effects on neurological development, particularly in children. Zinc, though less toxic, is examined because of its widespread presence in industrial runoff and its potential to cause aquatic toxicity at elevated concentrations. The analysis of these metals offers a comprehensive understanding of the pollution profile and helps assess the risks associated with water contamination by these commonly encountered heavy metals.

Results

Water quality characteristics of asbestos cement contacted and normal rainwater samples

The research found notable differences in water quality between normal rainwater and rainwater that had been in contact with AC roofs. The pH of water from the AC surfaces was significantly higher, averaging 7.09 ± 0.162 compared to 5.73 ± 0.265 for normal rainwater, a 23.8% increase. This suggests the roofing material has an alkaline effect, likely due to the leaching of substances like calcium compounds. The altered pH could impact the solubility and mobility of other contaminants. Similarly, the electrical conductivity, reflecting total ionic content, was higher in the AC samples, averaging 49.8 ± 6.00 µS/cm versus 31.4 ± 8.88 µS/cm in normal rainwater. This indicates the roofing material contributes additional dissolved ions, probably from the release of minerals and other substances. Analysis of total suspended solids indicated a small increase in samples from AC roofs, averaging 6.40 ± 1.51 mg/l compared to 6.00 ± 0.94 mg/l in regular precipitation. This 6.67% change suggests the roofing material does not significantly contribute particulates. However, even a small rise in TSS can impact water quality by transporting other pollutants like heavy metals. The research found a significant increase of 92.6% in water hardness for samples from AC surfaces compared to normal precipitation. The average hardness was 32.4 ± 9.75 CaO mg/L for the AC samples, compared to 16.8 ± 5.35 CaO mg/L in normal rain (Fig. 2). This substantial rise in hardness is likely due to the dissolution of calcium and magnesium compounds from the AC, which contribute to water hardness. Elevated water hardness can lead to scaling in pipes and reduced effectiveness of soaps and detergents.

The analysis of total organic carbon and oxygen demand in the precipitation samples provides further insights into how AC roofing affects water quality. The TOC values show a significant 40.9% increase in samples exposed to AC roofing, averaging 5.10 ± 1.08 mg/l compared to 3.62 ± 0.57 mg/l in normal precipitation. This substantial rise in TOC indicates more organic matter is present in water that has contacted the roofing material, likely due to leaching of organic compounds or accumulation of organic particles on the roof. Elevated TOC levels are often linked to increased microbial activity, which could impact water quality, especially in storage or distribution systems. The biochemical oxygen demand values, which measure the amount of oxygen required by microorganisms to break down organic matter, showed a slight 4.92% increase in samples that contacted AC. The average BOD_5

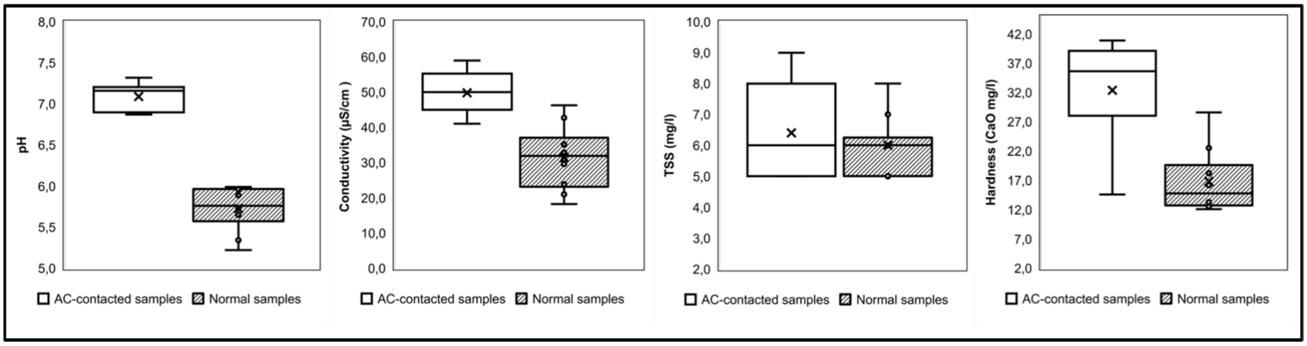


Fig. 2 Comparative analysis of water samples for pH, conductivity, TSS, and hardness

for these samples was 4.05 ± 0.88 mg/L, slightly higher than the 3.86 ± 0.25 mg/L observed in normal precipitation samples. Though modest, this increase suggests the organic matter in AC-exposed water may be more bioavailable or have a marginally higher microbial load, which could affect the water's suitability for irrigation or potable use. The most pronounced change was observed in the chemical oxygen demand, where the COD values for samples in contact with AC roofing were significantly higher than those in normal precipitation samples. This increase indicates a greater presence of oxidizable organic and inorganic matter in the water, likely originating from the roofing material or deposited pollutants. High COD levels suggest potential contamination and can negatively impact aquatic ecosystems if the water is discharged without proper treatment (Fig. 3).

Heavy metal pollution of asbestos cement contacted and normal rainwater samples

The analysis of metal concentrations in rainwater reveals notable differences between normal samples and those that contacted AC roofing. Zinc, lead, and mercury levels increased substantially in samples exposed to AC, indicating the roofing material serves as a source of these heavy metal contaminants. Normal rainwater contained very low total zinc, averaging 0.0024 ± 0.0005 mg/L, near the detection limit. This suggests that without contact with metal surfaces, zinc in rainwater is minimal, primarily from air pollution. However, rainwater that contacted AC roofing had a dramatic zinc increase to 0.155 ± 0.012 mg/L. This sharp rise is attributed to zinc within the AC material, either from manufacturing or atmospheric pollution accumulation on the roof. The substantial zinc elevation highlights AC roofing's potential to contribute significant metal contamination to harvested rainwater, risking environmental and health impacts if the water is used without adequate treatment. The analysis found that AC roofing significantly increased the levels of lead and mercury in harvested rainwater (Fig. 4). The lead concentration rose by 127.3%, from 0.00133 ± 0.0004 mg/l in normal samples to 0.003 ± 0.002 mg/l in AC-contacted samples. Lead is a toxic metal, especially for children, and its presence in water is closely regulated. The increased lead levels suggest the AC

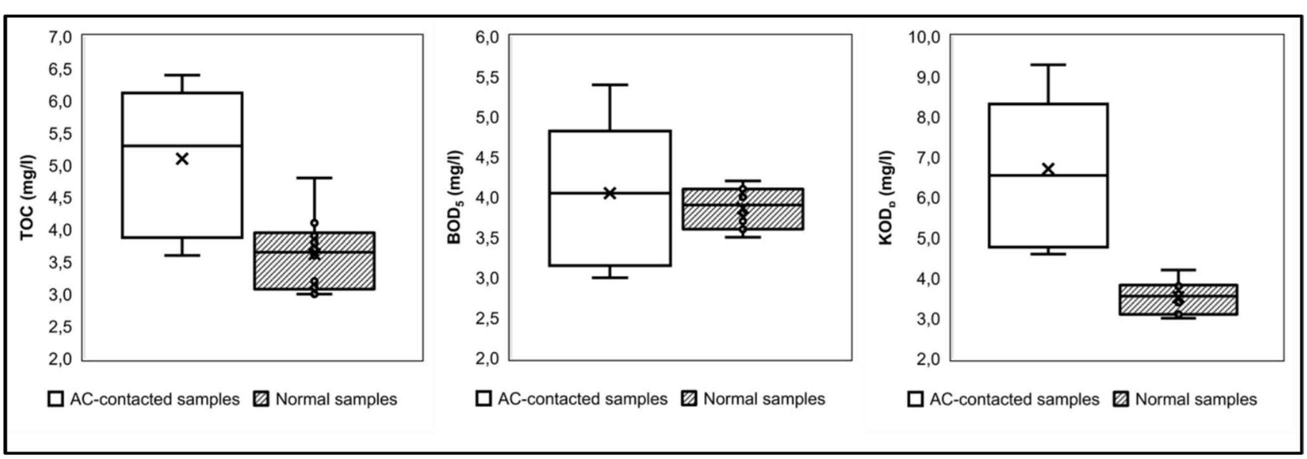


Fig. 3 Comparative analysis of water samples for TOC, BOD_5, and COD_p

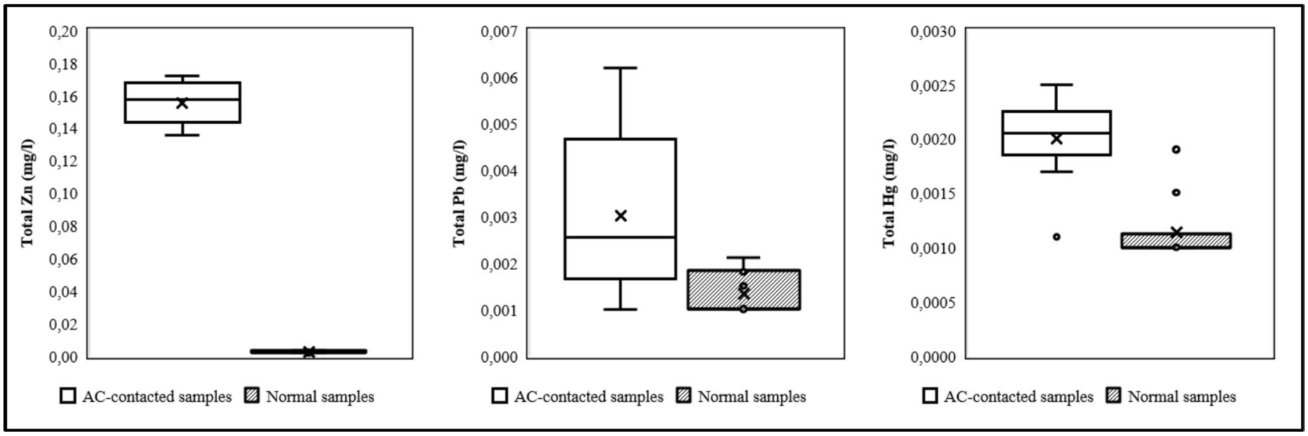


Fig. 4 Comparative analysis of water samples for heavy metal pollution (Zn, Pb, and Hg)

material may contain lead, either as an impurity or from environmental contamination. This highlights the need to monitor and manage lead in rainwater collected from such surfaces, especially where the water is for domestic use. Similarly, mercury concentrations increased by 75.4% in AC-contacted samples, from 0.0011 ± 0.0003 mg/l in normal precipitation to 0.002 ± 0.0004 mg/l.

Water quality characteristics of simulated asbestos cement contacted samples

The simulated water sample analysis provided insights into how AC roofing affects water quality parameters, compared to neutral and slightly acidic precipitation. The pH of the simulated samples was 7.21 ± 0.154, consistent with the trend of increased pH in AC-contacted samples. This suggests the alkaline AC material has a significant neutralizing effect, especially in slightly acidic environments. However, the pH shift was minimal in neutral starting conditions, indicating the AC impact is more pronounced in initially acidic water. Conductivity of the simulated samples increased to 6.80 ± 1.55 μS/cm, compared to near-zero for distilled water. This increase indicates the AC roofing contributes additional dissolved ions, likely from leaching of minerals or other soluble substances. This can affect the water's ionic balance and suitability for use. The average total suspended solids in the simulated samples were 5.30 ± 0.48 mg/l, comparable to precipitation samples contacted with AC roofing. This consistency suggests AC roofing is a source of particulate matter, which could carry contaminants or impact water clarity and quality (Fig. 5).

The measured total hardness of the simulated water samples was 17.7 ± 2.67 CaO mg/l, within the range of normal precipitation. This indicates that the AC roofing material can increase water hardness, likely by dissolving calcium compounds. Elevated hardness can affect water's interactions with plumbing, detergents, and other processes, an important factor in assessing AC roofs' impact on water quality. The analysis of simulated and precipitation samples exposed to AC roofing provides insights into the material's influence on organic content and oxygen demand. The total organic carbon in simulated samples was 3.82 ± 0.14 mg/l, lower than the 5.10 ± 1.08 mg/l average in precipitation samples exposed to AC (Fig. 6). This difference suggests AC roofing

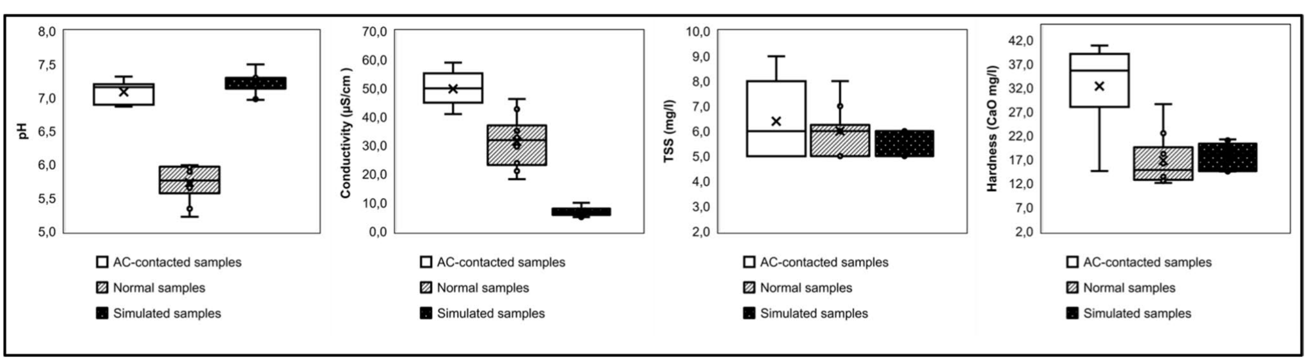


Fig. 5 Comparative analysis of simulated samples for pH, conductivity, TSS, and hardness

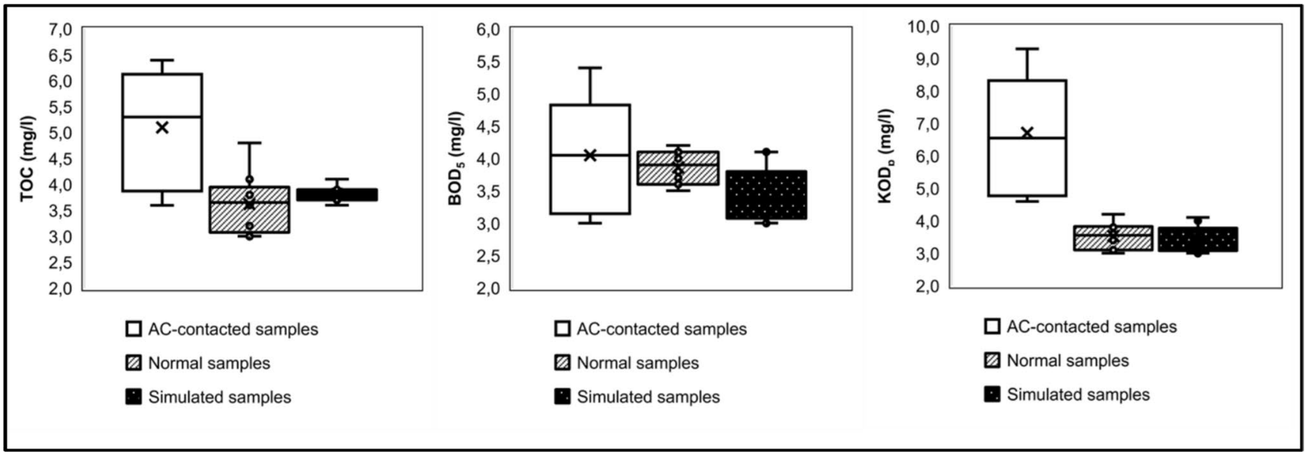


Fig. 6 Comparative analysis of simulated samples for TOC, BOD_5, and COD_p

contributes additional organic matter, potentially through leaching or pollutant accumulation. The lower TOC in simulated samples, not exposed to the full range of environmental conditions, supports the idea that pollutants deposited on AC roofs play a significant role in elevating TOC.

The simulated rainwater samples had an average biochemical oxygen demand of 3.44 ± 0.42 mg/L, slightly lower than normal precipitation and AC-exposed samples. The slight increase in AC-exposed samples suggests the organic matter is more biologically available or there is a higher microbial load, likely due to the organic content from the roofing material. The BOD_5 of the simulated samples, being close to normal precipitation, indicates that without external organic inputs, the biological oxygen demand remains relatively stable, highlighting the contribution of AC roofs to increased organic pollution. The chemical oxygen demand for simulated samples was 3.43 ± 0.41 mg/L, nearly identical to normal precipitation. However, the COD in AC-exposed samples was significantly higher at 6.71 ± 1.78 mg/L. This substantial increase indicates a higher presence of oxidizable organic and inorganic matter, likely originating from the AC material and environmental contaminants on the roof surface. The relatively low COD in the simulated samples suggests that, without external pollutants, the AC material itself may not significantly increase the COD, but when combined with environmental factors, it can contribute to a rise in oxidizable content.

Heavy metal pollution of simulated asbestos cement contacted samples

The analysis of heavy metal concentrations in simulated water samples, compared to those in contact with AC roofing and normal precipitation samples, reveals important

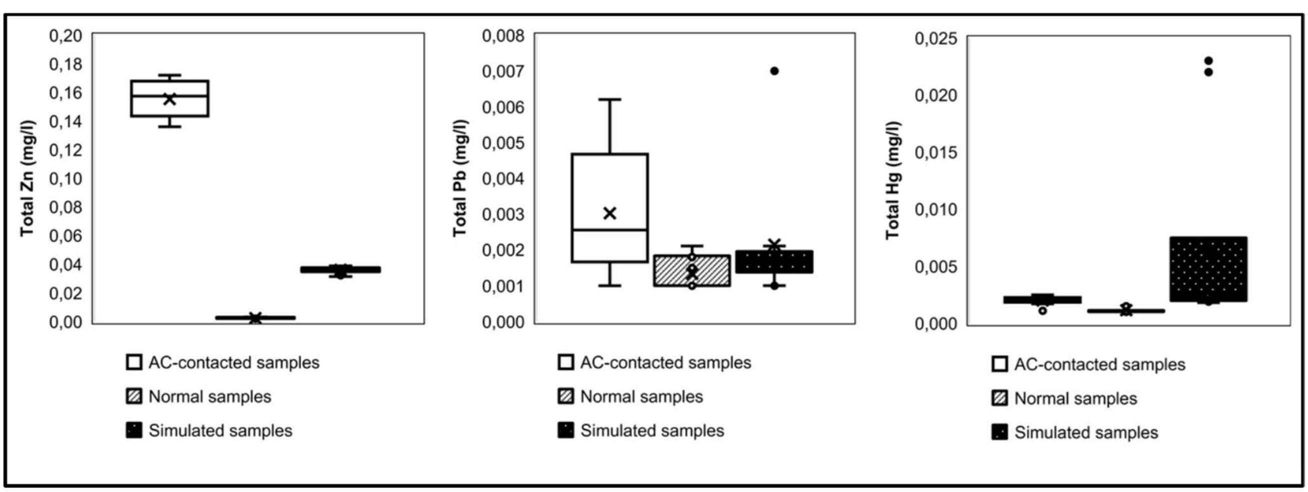


Fig. 7 Comparative analysis of simulated samples for total Zn, Pb, and Hg

insights into the impact of AC materials on water quality (Fig. 7). The simulated samples showed an average total zinc concentration of 0.0361 ± 0.002 mg/l, which is markedly higher than typical zinc levels in precipitation. Furthermore, the zinc concentration in AC-contacted samples was significantly elevated, averaging 0.155 ± 0.012 mg/l. This substantial increase suggests that the AC roofing material is a major source of zinc contamination, likely due to zinc leaching from the AC matrix or zinc-containing additives used in manufacturing. The elevated zinc levels in the simulated samples, though not as high as the AC-contacted samples, indicate that the AC material can release zinc into the water even without environmental pollutants. This highlights the potential for AC roofing to substantially contribute to zinc contamination in harvested rainwater, which could have environmental and human health implications if the water is used without proper treatment. The simulated samples showed an average lead concentration of 0.0021 ± 0.0017 mg/l, slightly higher than in normal precipitation but lower than in AC-contacted samples. This suggests the AC material may contribute to lead, but to a modest extent compared to zinc. The slight increase in simulated samples indicates some lead may leach from the AC material, but the larger rise in AC-contacted samples is likely due to additional environmental lead accumulating on the roof surface. This underscores the importance of considering both material composition and environmental exposure when assessing potential lead contamination in rainwater from AC roofing. The total mercury concentration in simulated samples was 0.0063 ± 0.009 mg/l, notably higher than typical levels in normal precipitation. This elevated mercury suggests it may be present in the AC material or have potential to be released under certain conditions. The high variability indicates further investigation is needed to fully understand mercury sources and behavior related to AC roofing.

Discussion

The quality of harvested rainwater can be influenced by various factors, including the type of roofing material used to collect the water (Mendez et al. 2011). The literature suggests that the quality of rainwater gathered at the start of the rainfall varies from that measured after the initial runoff period (Zdeb et al. 2020). The type of roofing materials is known to affect the chemical composition of runoff due to the leaching of substances and the interaction between precipitation and surface coatings (Bandow et al. 2018; Clark et al. 2008; De Buyck et al. 2021). According to Zdeb et al. (2020), rainwater that exposed to roof surfaces typically experiences deterioration in physical, chemical, and microbiological quality. Given that urban areas are increasingly covered with impervious surfaces, the choice of roofing material plays a crucial role in determining the characteristics of runoff water (Kõiv-Vainik et al. 2022; Öztürk et al. 2024).

The study found that roofing materials significantly affect stormwater quality, with variations in key water chemistry. The pH of the runoff differed based on the roofing material, as seen in the pH increase from contact with AC. These pH changes are crucial, as they can impact the solubility and mobility of pollutants in aquatic environments. Maintaining a neutral pH is vital to prevent harm to aquatic life and the availability of heavy metals (Kumar Dewangan et al. 2007). In parallel, conductivity serves as an indicator of dissolved ion content, and elevated values often correspond to increased pollutant loads (Kumar Dewangan et al. 2007). As expected, roofing materials with metal contamination, such as heavy metals, contributed to the presence of these metals in the runoff, raising concerns about their long-term accumulation in water bodies (Lye 2009). TSS concentrations were elevated in runoff from deteriorating roofing materials, where particulate matter was easily dislodged. This suggests that surface texture and material degradation play a crucial role in the mobilization of suspended particles (Kumar et al. 2021). Higher TSS levels can lead to increased turbidity, reducing light penetration and affecting aquatic ecosystems. Furthermore, suspended solids often act as carriers for other pollutants, such as heavy metals and organic contaminants, amplifying their impact on water quality (Matos et al. 2024). Hardness levels were predominantly affected by mineral-based materials, such as concrete and slate, which contributed to elevated calcium and magnesium concentrations (Müller et al. 2020). Elevated hardness in rainwater may influence its interactions with metal contaminants, potentially reducing the bioavailability of toxic metals through precipitation reactions. TOC levels demonstrated variability among exposed and normal samples, AC contributing higher TOC concentrations. The presence of organic matter in rainwater runoff has implications for microbial activity and nutrient cycling, which may influence downstream water quality (Wardinski et al. 2024). BOD and COD analyses revealed that roofing materials with organic components or coatings contributed to increased oxygen demand, likely due to the leaching of biodegradable substances or organic additives. Higher BOD and COD values indicate greater potential for oxygen depletion in receiving water bodies, which can lead to hypoxic conditions harmful to aquatic life (Mocuba 2010). The results of the analysis indicate that roofing materials significantly influence rainwater quality, particularly through the release of heavy metals such as zinc, lead, and mercury. Accumulated dust and airborne pollutants on roof surfaces may also impact on the metal concentrations detected in the collected rainwater (Ayenimo et al. 2006). Since these materials are not separated from the roofing substrate during runoff, the measured concentrations likely represent a combination of leaching from the roofing material and the wash off of deposited atmospheric contaminants (Hafera 2006). Future research could benefit from a separate examination of roof dust composition to better differentiate between these sources (Nicholson et al. 2010).

Roofing surfaces act as sources of contaminants, affecting runoff composition and posing environmental and health risks (Ibe and Ibe 2016). The weathering of AC and other metal-containing roofing materials contributes to the leaching of these pollutants into rainwater systems (Müller 2016). AC roofs release zinc, lead, and mercury over time, with potential consequences for ecosystems and human health (Gualtieri et al. 2022; Kottek and Yuen 2022). Zinc, often found in galvanized roofing, can disrupt aquatic organisms by affecting enzymatic functions and ecosystem stability (Briffa et al. 2020; Lidman et al. 2020). Lead, a known neurotoxin, can accumulate in the environment, affecting both human populations and wildlife (Assi et al. 2016). Mercury, though less common, is a persistent toxin that bioaccumulates, leading to severe ecological consequences (Pavithra et al. 2023). Stormwater contaminated with these heavy metals poses risks when used for irrigation. Elevated zinc, lead, and mercury concentrations can hinder plant growth, reduce agricultural yields, and lead to bioaccumulation in edible crops, potentially entering the human food chain. Beyond direct toxicity, heavy metals degrade water quality and bioaccumulate in aquatic organisms, affecting entire ecosystems (Saravanan et al. 2024). A shift in contamination levels has been detected even after a single exposure, particularly in a medium regarded as neutral. The persistence of these substances in aquatic ecosystems emphasizes the necessity for continuous monitoring and mitigation measures to address public and environmental health risks. Additionally, the significance of a holistic and integrated approach to water management and policy is reinforced to guarantee the sustainable governance of water resources.

Conclusions

This paper shows that rainwater collected from AC roofs has different chemical properties compared to normal rain. The key finding is that rainwater from AC roofs has higher pH levels, likely due to the roofing materials leaching calcium compounds. This change in pH can impact how other contaminants dissolve and move, which could affect water treatment and the environment. The water samples exposed to AC had higher electrical conductivity, suggesting the AC material contributed more dissolved ions. While total suspended solids only modestly increased, small amounts of particulates can carry other pollutants like heavy metals, potentially degrading water quality. Notably, water hardness nearly doubled in AC-exposed samples compared to normal rainwater. Additionally, total organic carbon concentrations were significantly higher in AC-exposed water, indicating the leaching of organic compounds from the roofing material. This could increase microbial activity and elevate the biochemical oxygen demand, further affecting the quality of stored rainwater. The chemical oxygen demand was also notably higher in AC-exposed samples, implying an increased presence of oxidizable organic and inorganic matter, likely from the roofing material or deposited pollutants.

The paper also identifies a considerable increase in heavy metal contamination, particularly zinc, lead, and mercury, in rainwater exposed to AC roofs. These metals pose significant environmental and health risks, especially if the water is used for irrigation or potable purposes. The increased concentrations of zinc, lead, and mercury in AC-exposed samples highlight the potential for these materials to serve as sources of heavy metal pollution. Zinc was found to be significantly elevated in both AC-exposed and simulated samples, indicating the potential for AC roofing to contribute substantial metal contamination to harvested rainwater. Owing to the considerable specific surface area and surface tension of asbestos fibers, these heavy metals form complexes that markedly enhance each other's effects. The results of simulated samples regarding general water chemistry parameters may not exhibit conspicuous changes; however, concerning heavy metals, they vividly illustrate the straightforward manifestation of pollution.

Overall, the findings underscore the critical role of roofing materials in determining the quality of harvested rainwater. The observed changes in pH, conductivity, hardness, organic content, and heavy metal concentrations emphasize the need for further investigation and monitoring of water quality in areas utilizing AC roofs for rainwater collection. These results suggest that the use of AC roofing materials can lead to significant alterations in water quality, with implications for water treatment, ecosystem health, and public safety. Thus, it is essential to consider these factors in the management of rainwater harvesting systems, particularly in urban areas, to ensure safe and sustainable water use practices. These findings underscore the importance of conditions when assessing the environmental impact of roofing materials on water quality. Further research is warranted to elucidate the mechanisms driving these observed differences and to devise effective strategies for mitigating potential environmental risks associated with AC roofing.

Author contribution All authors reviewed the manuscript.

Funding Open access funding provided by Széchenyi István University (SZE). This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors. The article processing charge (APC) is waived due to the publisher's institutional agreement with my university.

Data availability The datasets generated during and analyzed during the current study are not publicly available but are available from the corresponding author on reasonable request.

Declarations

Conflict of interest The authors declare that they have no competing interests.

References

Abdulla FA, Al-Shareef AW (2009) Roof rainwater harvesting systems for household water supply in Jordan. Desalination 243(1–3):195–207. https://doi.org/10.1016/j.desal.2008.05.013

Assi MA, Hezmee MNM, Haron AW, Sabri MY, Rajion MA (2016) The detrimental effects of lead on human and animal health. Veterinary World 9(6):660–671. https://doi.org/10.14202/vetworld.2016.660-671

Ayenimo JG, Adekunle AS, Makinde WO, Ogunlusi GO (2006) Heavy metal fractionation in roof run off in Ile-Ife, Nigeria. Int J Environ Sci Technol 3(3):221–227. https://doi.org/10.1007/BF03325929

Bandow N, Gartiser S, Ilvonen O, Schoknecht U (2018) Evaluation of the impact of construction products on the environment by leaching of possibly hazardous substances. Environ Sci Eur 30(1):14. https://doi.org/10.1186/s12302-018-0144-2

Bates BC, Kundzewicz ZW, Wu S, Palutikof J (2008) Climate change and water. Intergovernmental Panel on Climate Change Secretariat

Briffa J, Sinagra E, Blundell R (2020) Heavy metal pollution in the environment and their toxicological effects on humans. Heliyon 6(9):e04691. https://doi.org/10.1016/j.heliyon.2020.e04691

Chang M, McBroom MW, Scott Beasley R (2004) Roofing as a source of nonpoint water pollution. J Environ Manage 73(4):307–315. https://doi.org/10.1016/j.jenvman.2004.06.014

Chen Y, Marek G, Marek T, Brauer D, Srinivasan R (2017) Assessing the efficacy of the SWAT auto-irrigation function to simulate irrigation, evapotranspiration, and crop response to management strategies of the Texas high plains. Water 9(7):509. https://doi.org/10.3390/w9070509

Clark SE, Steele KA, Spicher J, Siu CY, Lalor MM, Pitt R, Kirby JT (2008) Roofing materials' contributions to storm-water runoff pollution. J Irrig Drain Eng 134(5):638–645. https://doi.org/10.1061/(ASCE)0733-9437(2008)134:5(638)

Conway D, van Garderen EA, Deryng D, Dorling S, Krueger T, Landman W, Lankford B, Lebek K, Osborn T, Ringler C, Thurlow J, Zhu T, Dalin C (2015) Climate and southern Africa's water–energy–food nexus. Nat Clim Chang 5(9):837–846. https://doi.org/10.1038/nclimate2735

De Buyck P-J, Matviichuk O, Dumoulin A, Rousseau DPL, Van Hulle SWH (2021) Roof runoff contamination: establishing material-pollutant relationships and material benchmarking based on laboratory leaching tests. Chemosphere 283:131112. https://doi.org/10.1016/j.chemosphere.2021.131112

Enedoh MC (2023) Determination of the leaching tendencies of various roofing sheets modelled as rain water harvesters in Imo state Nigeria. J Chem Soc Niger. https://doi.org/10.46602/jcsn.v48i4.909s

Martínez DE, Saba M, Gil LK (2024) Assessment of asbestos-cement roof distribution and prioritized intervention approaches through hyperspectral imaging. Heliyon 10(3):e25612. https://doi.org/10.1016/j.heliyon.2024.e25612

Favero-Longo SE, Castelli D, Fubini B, Piervittori R (2009) Lichens on asbestos–cement roofs: Bioweathering and biocovering effects. J Hazard Mater 162(2–3):1300–1308. https://doi.org/10.1016/j.jhazmat.2008.06.060

Ghosh S, Ahmed T (2022) Assessment of household rainwater harvesting systems in the southwestern coastal region of Bangladesh: existing practices and household perception. Water 14(21):3462. https://doi.org/10.3390/w14213462

Gopishankar T, Baraiya Divyeksha H, Vasantha VL, Praveen N (2022) Bioremediation and Detoxification of Asbestos from Soil. In: Advances in Bioremediation and Phytoremediation for Sustainable Soil Management. Springer International Publishing, pp 211–228. https://doi.org/10.1007/978-3-030-89984-4_14

Goutam Mukherjee A, Ramesh Wanjari U, Eladl MA, El-Sherbiny M, Elsherbini DMA, Sukumar A, Kannampuzha S, Ravichandran M, Renu K, Vellingiri B, Kandasamy S, Valsala Gopalakrishnan A (2022) Mixed contaminants: occurrence, interactions, toxicity, detection, and remediation. Molecules 27(8):2577. https://doi.org/10.3390/molecules27082577

Gualtieri AF, Lassinantti Gualtieri M, Scognamiglio V, Di Giuseppe D (2022) Human Health Hazards Associated with Asbestos in Building Materials. In: Ecological and Health Effects of Building Materials. Springer International Publishing, pp 297–325. https://doi.org/10.1007/978-3-030-76073-1_16

Gwenzi W, Dunjana N, Pisa C, Tauro T, Nyamadzawo G (2015) Water quality and public health risks associated with roof rainwater harvesting systems for potable supply: review and perspectives. Sustain Water Qual Ecol 6:107–118. https://doi.org/10.1016/j.swaqe.2015.01.006

Hafera J (2006) Metals contamination in stormwater from common roofing materials. World Environ Water Resour Congress 2006:1–8. https://doi.org/10.1061/40856(200)459

Ibe FC, Ibe BO (2016) Roof runoff water as source of pollution: a case study of some selected roofs in Orlu metropolis, Imo state, Nigeria. Int Lett Nat Sci 50:53–61. https://doi.org/10.18052/www.scipress.com/ILNS.50.53

Imteaz MA, Ahsan A, Rahman A, Mekanik F (2013) Modelling stormwater treatment systems using MUSIC: accuracy. Resour Conserv Recycl 71:15–21. https://doi.org/10.1016/j.resconrec.2012.11.007

Kadadou D, Tizani L, Alsafar H, Hasan SW (2024) Analytical methods for determining environmental contaminants of concern in water and wastewater. MethodsX 12:102582. https://doi.org/10.1016/j.mex.2024.102582

Kamp DW (2009) Asbestos-induced lung diseases: an update. Transl Res 153(4):143–152. https://doi.org/10.1016/j.trsl.2009.01.004

Kim R-H, Lee S, Lee J-H, Kim Y-M, Suh J-Y (2005) Developing technologies for rainwater utilization in urbanized area. Environ Technol 26(4):401–410. https://doi.org/10.1080/09593332608618545

Kim Y, Dao AD, Kim M, Nguyen V-A, Han M (2017) Design and management of rainwater harvesting systems to control water quality for potable purposes in Cu Khe. Vietnam Water Sci Technol:

Water Supply 17(2):452–460. https://doi.org/10.2166/ws.2016.154

Kõiv-Vainik M, Kill K, Espenberg M, Uuemaa E, Teemusk A, Maddison M, Palta MM, Török L, Mander Ü, Scholz M, Kasak K (2022) Urban stormwater retention capacity of nature-based solutions at different climatic conditions. Nat-Based Solut 2:100038. https://doi.org/10.1016/j.nbsj.2022.100038

Kottek M, Yuen ML (2022) Public health risks from asbestos cement roofing. Am J Ind Med 65(3):157–161. https://doi.org/10.1002/ajim.23321

Kumar Dewangan S, Shrivastava S, Tigga V, Lakra M (2007) Review paper on the role of pH in water quality implications for aquatic life, human health, and environmental sustainability. Int Adv Res J Sci, Eng Technol ISO. https://doi.org/10.17148/IARJSET.2023.10633

Kumar R, Sharma P, Verma A, Jha PK, Singh P, Gupta PK, Chandra R, Prasad PVV (2021) Effect of physical characteristics and hydrodynamic conditions on transport and deposition of microplastics in riverine ecosystem. Water 13(19):2710. https://doi.org/10.3390/w13192710

Lidman J, Jonsson M, Berglund ÅMM (2020) The effect of lead (Pb) and zinc (Zn) contamination on aquatic insect community composition and metamorphosis. Sci Total Environ 734:139406. https://doi.org/10.1016/j.scitotenv.2020.139406

Lupia F, Baiocchi V, Lelo K, Pulighe G (2017) Exploring rooftop rainwater harvesting potential for food production in Urban areas. Agriculture 7(6):46. https://doi.org/10.3390/agriculture7060046

Lye DJ (2002) Health risks associated with consumption of untreated water from household roof catchment systems[1]. JAWRA J Am Water Resour Assoc 38(5):1301–1306. https://doi.org/10.1111/j.1752-1688.2002.tb04349.x

Lye DJ (2009) Rooftop runoff as a source of contamination: a review. Sci Total Environ 407(21):5429–5434. https://doi.org/10.1016/j.scitotenv.2009.07.011

Magherini L, Avataneo C, Capella S, Lasagna M, Bianco C, Belluso E, De Luca DA, Sethi R (2023) Mobility of crocidolite asbestos in sandy porous media mimicking aquifer systems. J Hazard Mater 458:131998. https://doi.org/10.1016/j.jhazmat.2023.131998

Malinconico S, Paglietti F, Serranti S, Bonifazi G, Lonigro I (2022) Asbestos in soil and water: a review of analytical techniques and methods. J Hazard Mater 436:129083. https://doi.org/10.1016/j.jhazmat.2022.129083

Matos T, Martins MS, Henriques R, Goncalves LM (2024) A review of methods and instruments to monitor turbidity and suspended sediment concentration. J Water Process Eng 64:105624. https://doi.org/10.1016/j.jwpe.2024.105624

Mendez CB, Klenzendorf JB, Afshar BR, Simmons MT, Barrett ME, Kinney KA, Kirisits MJ (2011) The effect of roofing material on the quality of harvested rainwater. Water Res 45(5):2049–2059. https://doi.org/10.1016/j.watres.2010.12.015

Mocuba J (2010) Dissolved oxygen and biochemical oxygen demand in the waters close to the quelimane sewage discharge. Acknowledgements. https://doi.org/10.13140/2.1.1504.9288

Müller A (2016) Potential Sources of Stormwater Pollutants: Leaching of Metals and Organic Compounds from Roofing Materials. https://doi.org/10.13140/RG.2.1.1239.1281

Müller A, Österlund H, Marsalek J, Viklander M (2020) The pollution conveyed by urban runoff: a review of sources. Sci Total Environ 709:136125. https://doi.org/10.1016/j.scitotenv.2019.136125

Nicholson N, Clark SE, Long BV, Siu CYS, Spicher J, Steele KA (2010) Rainwater harvesting from roofs for non-potable reuse. Low Impact Development 2010:21–30. https://doi.org/10.1061/41099(367)3

Oyilieze Akanwa A (2020) Rural harvested rainwater: effect of roof types and its design on water quality and health: a case for CBP approach in Anambra state. Rev Environ Earth Sci 7(1):1–14. https://doi.org/10.18488/journal.80.2020.71.1.14

Öztürk Ş, Yılmaz K, Dinçer AE, Kalpakcı V (2024) Effect of urbanization on surface runoff and performance of green roofs and permeable pavement for mitigating urban floods. Nat Hazards 120(13):12375–12399. https://doi.org/10.1007/s11069-024-06688-w

Pachpute JS, Tumbo SD, Sally H, Mul ML (2009) Sustainability of rainwater harvesting systems in rural catchment of sub-saharan Africa. Water Resour Manage 23(13):2815–2839. https://doi.org/10.1007/s11269-009-9411-8

Pavithra KG, SundarRajan P, Kumar PS, Rangasamy G (2023) Mercury sources, contaminations, mercury cycle, detection and treatment techniques: a review. Chemosphere 312:137314. https://doi.org/10.1016/j.chemosphere.2022.137314

Peña-Castro M, Montero-Acosta M, Saba M (2023) A critical review of asbestos concentrations in water and air, according to exposure sources. Heliyon 9(5):e15730. https://doi.org/10.1016/J.HELIYON.2023.E15730

Rashid A, Schutte BJ, Ulery A, Deyholos MK, Sanogo S, Lehnhoff EA, Beck L (2023) Heavy metal contamination in agricultural soil: environmental pollutants affecting crop health. Agronomy 13(6):1521. https://doi.org/10.3390/agronomy13061521

Saba M, Coronado-Hernández OE, Gil LKT (2024) Energy efficiency in subtropical homes: replacing asbestos-cement roofs with sustainable alternatives. Buildings 14(12):4082. https://doi.org/10.3390/buildings14124082

Sanjeeva A, Puttaswamaiah SG (2018) Influence of atmospheric deposition and roof materials on harvested rainwater quality. J Environ Eng. https://doi.org/10.1061/(ASCE)EE.1943-7870.0001460

Saravanan P, Saravanan V, Rajeshkannan R, Arnica G, Rajasimman M, Baskar G, Pugazhendhi A (2024) Comprehensive review on toxic heavy metals in the aquatic system: sources, identification, treatment strategies, and health risk assessment. Environ Res 258:119440. https://doi.org/10.1016/j.envres.2024.119440

Spurny KR (1989) On the release of asbestos fibers from weathered and corroded asbestos cement products. Environ Res 48(1):100–116. https://doi.org/10.1016/S0013-9351(89)80089-1

Suzuki Y, Yuen SR, Ashley R (2005) Short, thin asbestos fibers contribute to the development of human malignant mesothelioma: pathological evidence. Int J Hyg Environ Health 208(3):201–210. https://doi.org/10.1016/j.ijheh.2005.01.015

Tengan BM, Akoto O (2022) Comprehensive evaluation of the possible impact of roofing materials on the quality of harvested rainwater for human consumption. Sci Total Environ 819:152966. https://doi.org/10.1016/j.scitotenv.2022.152966

Tóth E, Weiszburg T (2011) Környezeti ásványtan

Wardinski K, Wall H, Scott D (2024) Soil-derived dissolved organic matter and nutrient sources from urban stormwater control measures. Sci Total Environ 957:177288. https://doi.org/10.1016/j.scitotenv.2024.177288

Zdeb M, Zamorska J, Papciak D, Słyś D (2020) The quality of rainwater collected from roofs and the possibility of its economic use. Resources 9(2):12. https://doi.org/10.3390/resources9020012

Discover Sustainability

REVIEW

Open Access

Pathways to asbestos-free and sustainable cities using multi-level perspective approach

Gergely Zoltán Macher[1*] and Cecília Szigeti[2]

*Correspondence:
Gergely Zoltán Macher
macher.gergely.zoltan@sze.hu
[1]Department of Applied Sustainability, Széchenyi István University, Győr, Hungary
[2]Institute of Sustainable Studies, Budapest Metropolitan University, Budapest, Hungary

Abstract
Urban policymakers, researchers, and municipal planners increasingly face the challenge of managing complex sustainability transitions, particularly in contexts involving persistent environmental hazards such as asbestos contamination. This systematic review applies the Multi-Level Perspective (MLP), which examines interactions between niche innovations, socio-technical regimes, and broader landscapes, to the underexplored area of asbestos-free urban transitions. The concept of an "asbestos-free city" is introduced in this paper as a novel analytical lens to describe urban transitions aiming to eliminate asbestos-related risks through systemic, sustainable interventions. The review was conducted through a structured qualitative analysis of peer-reviewed academic literature, guided by predefined thematic criteria and relevance to urban asbestos-related transitions. The review highlights the factors that enable or hinder the adoption of asbestos-free and strong sustainable solutions, as well as the role of various actors, such as policymakers, industry, and civil society, in driving these transitions. Despite the growing body of work on sustainability transitions, the integration of MLP into asbestos-related urban transformation remains limited. This paper fills that gap by offering a structured synthesis and proposing a roadmap for future research and practice. Our findings provide actionable insights for actors across policy, civil society, and industry seeking to accelerate transitions toward asbestos-free and sustainable cities.

Keywords Sustainability transitions, Multi-level perspective, Asbestos-free cities, Strong sustainability, Urban development

1 Introduction

With increasing global concerns about public health risks and environmental sustainability, the transition towards asbestos-free urban environments has gained significant attention [1]. Asbestos, historically used in construction and industrial applications due to its durability and fire resistance, has been identified as a severe health hazard, leading to diseases such as asbestosis, lung cancer, and mesothelioma [2–5]. In the 20th century, approximately 125 million tons of asbestos were used worldwide, with its use peaking in the 1970s [6, 7]. The extensive legacy of asbestos use, particularly in building construction, water infrastructure, and a wide array of manufactured goods, has led to a ubiquitous environmental and public health challenge due to its carcinogenic properties and potential for long-term exposure [8, 9]. Notwithstanding the extensive awareness of asbestos-related risks and the implementation of usage bans in more than 50 countries, the global consumption of asbestos persists at an estimated 2 million tons per year [10–12].

Consequently, numerous nations have enacted regulations to eliminate asbestos-containing products and encourage the adoption of sustainable alternatives in urban settings [13, 14]. Nevertheless, realizing a comprehensive transition necessitates a thorough comprehension of the multilayered factors shaping policy execution, industry adaptation, and public acceptance [15, 16]. Traditionally, the construction and urban planning industries have emphasized cost-effectiveness and material durability, often to the detriment of environmental and public health considerations [17–20]. However, as sustainability goals and public health awareness have advanced, cities are increasingly implementing policies to enable asbestos-free development while integrating broader sustainability principles [21]. The transition to asbestos-free cities is not merely a regulatory challenge, but also a strategic approach to enhancing long-term urban resilience and sustainability [1, 22]. This paper introduces the concept of an "asbestos-free city" as a novel analytical lens to describe and analyse integrated urban transitions aimed at eliminating asbestos-related risks through sustainable, multi-actor interventions. Eliminating asbestos-containing materials is a vital step in reducing public health risks and ensuring safer urban environments [23]. Furthermore, this aligns with global sustainability initiatives by promoting environmentally friendly construction materials and circular economy principles [1, 24]. According to the Ellen MacArthur Foundation [25], the circular economy model focuses on optimizing resource management by minimizing waste and pollution, maximizing the utilization of products and materials, and fostering the regeneration of natural ecosystems. These tenets are critical in the context of sustainably eliminating asbestos and transitioning to alternative materials. Multifaceted factors, encompassing government policies, industrial characteristics, and societal engagement, play a pivotal role in this transitional process [26]. Regulatory frameworks establish the legal and compliance requirements for the removal of asbestos and its replacement with sustainable alternatives [27]. Concurrently, industrial determinants, such as technological progress, market incentives, and competitive dynamics within the construction industry, shape the effective adoption of asbestos-free practices [13, 28–30]. Additionally, public consciousness and societal backing are crucial in stimulating the demand for safe and sustainable urban development [31]. Despite the increasing acknowledgment of the necessity for asbestos-free urban environments, the efficacy of transition strategies continues to be an active area of research [13, 26]. While regulatory frameworks offer a foundation, their impact fluctuates based on industry-specific challenges and regional socioeconomic conditions [32]. The cost implications associated with asbestos removal and substitution can be substantial, and without sufficient financial and technological support, certain industries may face difficulties in compliance [30, 33, 34]. To address these financial challenges, innovative business models can play a crucial role in facilitating the transition. Green financing initiatives, including government-backed subsidies and ESG (Environmental, Social, and Governance) investments, can help mitigate the financial burden on construction firms [35]. Additionally, public-private partnerships (abbreviation: PPPs) may encourage cooperation between government entities and private companies to fund large-scale asbestos removal projects [36]. Furthermore, circular economy business models—such as recycling and repurposing materials from asbestos-containing buildings—can create economic value while supporting sustainability efforts [37].

Sustainable business models encompass mechanisms of value creation, delivery, and retention that are not only economically viable but also generate positive environmental and social impacts [38]. According to [39], such models challenge conventional logics of value creation and offer alternative pathways that promote long-term ecological and social sustainability. In the practical application of sustainability, a critical distinction must be made between initiatives that genuinely foster sustainable outcomes (namely, circular economy principles, resource efficiency, and the mitigation of social inequalities) and those that are superficial attempts at appearing eco-friendly, often driven by marketing objectives [40]. In addition to definitional clarity, attention should also be paid to the degree to which different sustainable business models are embedded in the prevailing structures of consumer society [41, 42]. As [38] emphasize, models based on digitalization, sharing, and resource efficiency tend to exhibit stronger rebound effects, cause a return to previous, often environmentally harmful, consumption patterns. In contrast, sufficiency- and stewardship-oriented models appeared less susceptible to such effects and more clearly reflected the long-term goals of sustainability [43]. This understanding supports the notion that the term 'sustainable business model' is an overarching concept that necessitates differentiating between genuinely transformative models and those more deeply embedded within existing consumerist frameworks [44].

Additionally, stakeholders, including policymakers, construction firms, and urban planners, must comprehend the long-term value of sustainable transitions beyond immediate fiscal concerns [45–47].

The integration of "sustainability-as-a-service" business models can facilitate the transition to asbestos-free and sustainable construction. In this approach, companies offer asbestos remediation and sustainable construction services on a subscription or pay-per-use basis [48]. This model enables property owners to transition gradually while spreading the associated costs over time. Additionally, market-based incentives, such as tax benefits for companies that utilize asbestos-free and sustainable materials, could accelerate the shift toward safer and more environmentally-friendly urban environments [49].

A comprehensive review of the factors influencing asbestos-free and sustainable urban transitions is essential to connect policies with practical implementation [50]. The paper adopts the Multi-Level Perspective (abbreviation: MLP) framework [51–53], which conceptualizes sustainability transitions as interactions across three levels: niche innovations, socio-technical regimes, and the broader socio-technical landscape. This approach allows for a systemic assessment of how diverse factors shape the pace and direction of asbestos-free urban transitions [54]. The external context, including industry, economics, and regulations, directly impacts how cities manage this transition [16]. Without considering these contextual factors, policy recommendations may lack the adaptability and practicality needed for effective implementation [55]. By aligning business strategies with policy and market forces at different levels of the MLP model, the transition to asbestos-free and sustainable construction can be facilitated [56]. For instance, at the niche level, startups developing environmentally-friendly construction materials may benefit from government incentives. At the regime level, established firms can adopt sustainable supply chain practices and circular economy principles. Concurrently, at the broader landscape level, global sustainability commitments can catalyse large-scale industrial transformations [57]. Leveraging these market-driven approaches is crucial, as policy recommendations alone may lack the adaptability and practical feasibility required for effective implementation.

As municipalities develop more sustainable practices, comprehending the relationships among regulations, industry operations, and community involvement is essential [29]. Examining the influence of multifaceted elements on the transition to asbestos-free environments can bolster urban resilience and advance worldwide sustainability goals [58]. This paper systematically reviews the existing literature and empirical evidence to identify the principal enablers and barriers, and to provide insights into successfully implementing a sustainable, asbestos-free urban model. The research's novelty lies in its systemic approach to the multilevel drivers of sustainable urban development, linking regulatory, industrial, and social aspects. The research gap stems from the fact that existing studies often focus on a single factor, whereas this paper adopts a holistic approach, enabling a deeper understanding of complex interrelations and interactions.

While several studies have addressed aspects of asbestos removal and sustainable urban development [48, 59, 60], a comprehensive synthesis integrating the Multi-Level Perspective (MLP) framework into asbestos-free urban transitions remains absent. Existing reviews often focus narrowly on policy, technological innovation, or social acceptance independently, overlooking the systemic interactions across niche, regime, and landscape levels that the MLP framework captures. This gap limits our understanding of the multi-dimensional challenges and opportunities in asbestos-free urban transformations. To address this gap, the paper is guided by the following research questions:

- Q1 What are the main enabling factors and barriers across niche, regime, and landscape levels influencing the adoption of asbestos-free and sustainable urban solutions?
- Q2 What roles do different actors (policymakers, industry, and civil society) play within these multi-level interactions to accelerate or hinder the transition?
- Q3 How can the Multi-Level Perspective framework elucidate the complex dynamics shaping asbestos-free urban transitions?

By answering these questions, the paper aims to provide a structured synthesis that not only fills the existing literature gap but also offers actionable insights for advancing asbestos-free sustainable cities.

2 Theoretical background

2.1 Is an asbestos-contaminated community capable of achieving sustainability?

The discourse surrounding sustainability has expanded significantly over recent decades, encompassing not only environmental concerns but also considerations of social justice, economic viability, and intergenerational equity [61–63]. While the concept of weak sustainability posits that natural capital can be substituted with human-made capital, proponents of strong sustainability argue that certain environmental assets, particularly those crucial for health and ecosystem stability, are irreplaceable [64, 65]. The case of asbestos-contaminated communities illustrates the profound challenges associated with achieving sustainability in the presence of persistent environmental hazards [66]. Asbestos, a recognized carcinogen, does not degrade over time like many other pollutants, and its health consequences, including asbestosis, lung cancer, and mesothelioma, may not manifest until decades after exposure [3]. This latency period creates a disconnect between immediate mitigation efforts and long-term health outcomes, thereby complicating the implementation of sustainability principles in affected areas [50]. From a strong sustainability standpoint, an asbestos-contaminated environment represents a fundamental contradiction [67]. Robust sustainability necessitates safeguarding ecological integrity, maintaining essential life-supporting systems, and preventing irreversible environmental damage [61]. However, asbestos contamination directly undermines these objectives, as exposure to airborne fibres poses an ongoing threat to human health and natural ecosystems [68]. Unlike pollutants that can be effectively neutralized or biologically degraded, asbestos remains persistent once released into the environment [3]. Even when abatement measures, such as encapsulation or controlled removal, are implemented, the residual risks remain significant, particularly in aging infrastructure and inadequately regulated disposal sites [32]. The mere presence of asbestos-containing materials in built environments, even if undisturbed, conflicts with the precautionary principle—a cornerstone of sustainable development—due to the potential for future exposure incidents [69].

Intergenerational equity, a core tenet of sustainability [70], is severely compromised in asbestos-contaminated communities. The prolonged latency period of asbestos-related illnesses means that current exposure disproportionately burdens future generations, contradicting the fundamental objective of sustainability: ensuring a safe and liveable environment for succeeding populations [69]. Additionally, the economic and social expenditures associated with asbestos remediation and related healthcare services divert financial and institutional resources from other sustainability initiatives [71]. In many instances, the remediation process itself generates further environmental and social challenges, as the safe removal, transportation, and disposal of asbestos materials necessitates extensive regulatory oversight and technological infrastructure [72]. Furthermore, the socioeconomic disparities that often characterize asbestos-exposed populations exacerbate the issue, as lower-income communities may lack the political influence or financial capacity to advocate for comprehensive remediation efforts [50, 68]. The systemic nature of asbestos-related risks underscores the challenges of reconciling sustainability with scenarios where environmental degradation is effectively irreversible without large-scale intervention. While certain sustainability aspects, such as economic resilience or community engagement [73], may be promoted in asbestos-affected areas, the fundamental principles of strong sustainability remain difficult to achieve as long as asbestos contamination persists [74]. This raises broader ethical and policy questions about the extent to which sustainability can be realized in compromised environments [75], and whether the presence of hazardous, non-degradable pollutants constitutes an absolute barrier to sustainability. Consequently, in any effort to integrate sustainability strategies into asbestos-contaminated communities [76], the primary focus must be on the complete elimination of exposure risks as a prerequisite for broader environmental and social progress.

2.2 Challenges of asbestos-free transition

Conceptualizing social transitions is a complex endeavour that frequently encompasses both qualitative and quantitative dimensions [77, 78]. The shift towards an asbestos-free environment encounters various challenges, including the multi-level and regulatory aspects of transforming industrial and construction systems [79], the interplay between stability and the emergence of new solutions within existing economic and social structures [80], and the integrated management of social, economic, and technological transformations [81].

The pervasive utilization of asbestos in industrial and construction settings has engendered long-term reliance [22]. Transitioning to an asbestos-free environment necessitates substantial financial and infrastructural investments, encompassing the replacement or modification of existing buildings and equipment, as well as the implementation of appropriate safety measures [49]. The success of asbestos elimination is intrinsically linked to cooperation between the public and private sectors [70], as mitigating health and environmental risks requires a long-term strategic approach. Furthermore, as this transition remains in its nascent stages in numerous countries, a comprehensive assessment is indispensable, considering the safety, cost-effectiveness, and environmental impact of alternative materials [82]. Collaboration between nations and stakeholders plays a pivotal role in realizing this transition, as long-term partnerships provide opportunities to foster innovation, facilitate knowledge sharing, and optimize the allocation of financial resources [83, 84].

The evaluation of methods for facilitating asbestos-free transitions can be divided into qualitative and quantitative approaches [85, 86]. Qualitative methods enable in-depth theoretical and conceptual investigation of transition processes [87, 88]. Complementing these, qualitative models or analytical frameworks help organize and interpret the complex findings arising from qualitative data. Quantitative approaches, by contrast, provide measurable and precise results but may be limited in their capacity to incorporate factors that cannot be readily translated into mathematical expressions [38, 89]. As such, developing a comprehensive qualitative framework alongside a validated quantitative model is essential for establishing effective strategies for asbestos elimination [90].

2.3 Application of the MLP in asbestos-free and sustainable urban transitions

The MLP constitutes a salient analytical framework for investigating the dynamics underpinning socio-technical transitions [78, 82, 91]. Its utility is noteworthy when considering the promotion of asbestos-free and sustainable transitions in urban environments. This approach delineates three interacting levels - the landscape, the regime, and the niche - each of which plays a pivotal role in shaping the trajectories of such transitions [92, 93].

The landscape level encompasses overarching contextual factors such as macroeconomic conditions, regulatory frameworks, and societal values [70, 94]. In the case of asbestos-free transitions, this landscape is characterized by global concerns regarding health and the environment, the implementation of more stringent regulations on hazardous materials, and growing public consciousness about the long-term risks associated with asbestos exposure. These broad, systemic trends exert gradual yet significant pressure on existing urban systems, compelling them to adapt accordingly over time [95].

The regime level is characterized by the prevailing infrastructure, policies, and institutional norms that shape urban planning, construction practices, and environmental policies [96]. The continued presence of asbestos in older buildings, coupled with regulatory and economic constraints, generates significant lock-in effects that impede rapid transitions [49]. Existing regulatory frameworks may either facilitate or hinder the phasing out of asbestos, depending on factors like enforcement mechanisms, available financial incentives, and industry resistance [97, 98]. Nonetheless, external pressures such as legal mandates, advocacy campaigns, and public demand for healthier urban environments can challenge the stability of this regime, opening up avenues for systemic change [99].

From a business perspective, financial mechanisms such as green bonds, impact investing, and sustainability-linked loans can play a pivotal role in mitigating the economic challenges associated with asbestos remediation [100]. Collaborative efforts between governments and private sector stakeholders through public-private partnerships can help fund large-scale asbestos abatement projects. Additionally, circular economy frameworks can facilitate the creation of value from recycled construction materials [32]. Furthermore, performance-based contracting models have the potential to incentivize innovation by linking financial rewards to successful asbestos removal and sustainable urban development outcomes [101]. At the niche level, innovative solutions and alternative approaches arise, frequently driven by pioneering stakeholders such as sustainable construction companies, research organizations, and policy innovators [78]. These niche-level actors explore and experiment with asbestos-free building materials, novel decontamination methods, and circular economy strategies for waste management [102]. Entrepreneurial business models, such as leasing arrangements for sustainable construction materials and performance-based contracts for asbestos remediation, can offer scalable solutions that reconcile economic incentives with sustainability objectives [103]. By nurturing entrepreneurship in the asbestos-free building industry and supporting startup accelerators focused on green construction innovations, actors at the niche level can catalyse systemic transformation [104]. As these innovations develop and gather momentum, they can shape and influence the broader regime, thereby accelerating the transition towards more sustainable urban systems [105]. Extant research has applied the MLP framework to examine urban sustainability transitions, highlighting the significance of niche innovations in disrupting established socio-technical regimes [106]. For instance, studies on sustainable urban development have shown that grassroots initiatives focused on green architecture, circular construction methods, and policy-driven incentives play a vital role in transforming urban environments [107–110]. Similarly, the adoption of asbestos-free construction techniques in pioneering cities offers valuable insights into overcoming institutional and economic barriers to change. The application of the MLP to the transition towards asbestos-free urban environments underscores the necessity for a comprehensive and strategic approach [78]. Successful transitions require not only regulatory interventions and technological advancements, but also active public engagement and cross-sectoral collaboration. By harnessing the synergies across the landscape, regime, and niche levels, cities can systematically phase out asbestos while cultivating healthier and more sustainable built environments [111].

Asbestos phase-out processes are deeply embedded in social and institutional contexts, which fundamentally shape the direction, pace, and depth of sustainability transitions. These transitions are not merely technological or regulatory in nature; they also involve complex processes of social learning, trust-building, and justice-oriented governance. Identifying both barriers and enabling factors across different levels—individual, organizational, and systemic—provides valuable insights into critical leverage points where targeted interventions can enhance the effectiveness and equity of the transition. Table 1 presents these multilevel challenges and enablers in a structured format, highlighting how individual mindsets, institutional dynamics, and systemic conditions interact to either hinder or foster transformative change.

The examples provided are particularly relevant in high-risk contexts, where asbestos exposure is compounded by social vulnerability and institutional underrepresentation, factors that collectively obstruct the implementation of just and sustainable remediation pathways.

Table 1 Multilevel challenges and enabling factors in transformative sustainability transitions

Level	Type	Description	Examples / Notes
Individual	Challenge	Limited understanding of systemic sustainability concepts (e.g. strong sustainability, social metabolism)	Sustainability literacy gaps among citizens or professionals in high-risk environments
	Challenge	Cognitive and emotional resistance to change; psychological distance from long-term risks	"Not in my backyard" (NIMBY) attitudes towards remediation or infrastructure change
	Challenge	Low sense of agency in high-uncertainty or post-contamination contexts	Demotivation in asbestos-affected populations due to long-term neglect
	Enabler	Sustainability-oriented value systems and intergenerational thinking	Environmental ethics education, storytelling, local heritage revitalization
	Enabler	Empowerment through citizen science and participatory monitoring	Community mapping of contaminated zones; involvement in risk assessment processes
	Enabler	Psychological ownership of solutions; emotional connection to place	Community gardening on remediated land; local circular economy pilots
Organizational	Challenge	Institutional lock-ins, rigid hierarchies resistant to horizontal collaboration	Traditional remediation agencies focused on compliance rather than co-creation
	Challenge	Misalignment between sustainability goals and operational mandates	Health and safety policies disconnected from environmental or innovation strategies
	Challenge	Lack of diversity in decision-making processes	Marginalized or vulnerable groups excluded from asbestos phase-out planning
	Enabler	Mission-driven innovation aligned with strong sustainability principles	Organizations integrating public health, environmental justice, and innovation
	Enabler	Use of transdisciplinary and inclusive methods in program design and implementation	Co-design workshops, shared governance structures
	Enabler	Institutional learning and reflexivity mechanisms	After-action reviews, embedded monitoring and learning systems
Systemic	Challenge	Fragmented governance systems and incoherent policy mixes	Environmental, public health, and innovation policies operating in silos
	Challenge	Path dependency and lack of disruptive innovation incentives	Continued preference for cost-minimization over long-term social and ecological gains
	Challenge	Unequal access to information, funding, or remediation infrastructure	Under-resourced municipalities, urban-rural divide in environmental health interventions
	Enabler	Adaptive regulatory environments that reward sustainability co-benefits	Green public procurement, incentives for nature-based remediation
	Enabler	Integration of science-policy-practice interfaces	Dedicated knowledge brokers, sustainability transition hubs
	Enabler	Long-term investment in justice-oriented sustainability infrastructure	Strategic remediation programs addressing both contamination and socio-economic disparities

2.4 The MLP and strong sustainability

The integration of the MLP with strong sustainability offers a robust analytical framework [112] to comprehend and enable asbestos-free transitions in urban settings. Strong sustainability underscores the imperative of maintaining ecological integrity and ensuring intergenerational equity, principles that align with the systemic and multi-scalar nature of the MLP [113, 114]. By examining the interplay between landscape, regime, and niche levels, it becomes feasible to identify both obstacles and prospects for progressing towards asbestos-free, sustainable cities [115]. At the landscape level, sustainability influences economic trends, policies, and public awareness of asbestos risks [116]. Stricter regulations, health guidelines, and demand for sustainable materials put pressure on existing systems. The shift toward circular economy further supports eliminating hazardous materials like asbestos and promoting safer alternatives [117]. The established policies, industry standards, and economic structures governing construction, waste management, and urban planning make up the regime level [118]. Despite increased awareness of asbestos dangers, many urban areas still rely on infrastructure containing asbestos-based materials. Institutional inertia, economic constraints, and path dependency hinder systemic change [119]. However, regulations, technology, and incentives can disrupt current practices and enable a transition to sustainable, asbestos-free urban systems. At the niche level, emerging innovations drive change. Researchers, policymakers, and industry pioneers develop alternative materials, remediation techniques, and policy frameworks aligned with sustainability principles [52, 120]. Pilot projects and small-scale initiatives serve as incubators, providing proof-of-concept models that can be scaled up and integrated into mainstream urban development [107]. Collaboration among stakeholders at this level is crucial for exchanging knowledge, reducing barriers, and accelerating the adoption of sustainable practices [121, 122]. The MLP provides a valuable framework for policymakers and planners to devise comprehensive strategies addressing the structural and behavioural transformations required for sustainable and asbestos-free urban transitions [51]. Integrating regulations, incentives, and participatory governance approaches is critical to overcoming resistance and ensuring an equitable transition [115].

3 Research hypotheses and design

3.1 Research hypothesis development

The transition to asbestos-free urban environments is a multifaceted process influenced by various factors, including regulatory policies, technological innovations, stakeholder involvement, and socioeconomic conditions [78]. The research addresses multiple factors, among which the analysis of regulatory frameworks receives particular focus, as it plays a pivotal role in shaping asbestos-free and sustainable urban transformations. This analytical perspective is closely linked to other key aspects, such as the management of health risks, the inclusion of societal actors, and the pace and implementation of interventions. The centrality of regulatory frameworks lies not only in establishing legal boundaries, but also in their direct impact on societal acceptance, the mitigation of health-related risks, and the timing and effectiveness of policy implementation. The MLP offers a comprehensive framework for examining how interactions among different levels of governance, industry, and societal influences either enable or impede this transition [81, 123]. Mitigating the presence of asbestos in urban areas is essential for safeguarding public wellbeing and facilitating sustainable city transitions [124]. Asbestos-containing older buildings and infrastructure pose considerable threats to inhabitants and workers, necessitating coordinated policies and strategic actions to confront this critical concern [125, 126]. Drawing on transition theory, we propose the following hypothesis:

Hypothesis (H1) The resistance of legacy industries and path dependencies in construction and infrastructure sectors hinder the effectiveness of asbestos-free policies, necessitating targeted interventions.

We hypothesize that the success of transitioning to asbestos-free practices is shaped by the degree of regulatory oversight and institutional backing. Stringent national bans, local ordinances, and international accords significantly influence the pace and breadth of asbestos removal initiatives. In jurisdictions with robust regulations, municipalities and industries are more inclined to implement proactive asbestos-free measures, whereas weak enforcement allows the persistent presence of asbestos to remain a major obstacle. Consequently, we propose the following hypothesis:

Hypothesis (H2) The stringency of regulatory frameworks moderates the relationship between asbestos-free policies and urban sustainability outcomes.

The role of industry and technological innovation is vital in enabling the transition. Novel asbestos-free materials, safer demolition methods, and improved waste management solutions help mitigate exposure risks. Highly innovative industries, with the availability of alternatives and adherence to best practices, can expedite asbestos-free transitions. Conversely, industries resistant to change may experience slower transitions due to economic and structural impediments. Based on these observations, we propose the following hypothesis:

Hypothesis (H3) Technological innovations and industrial advancements influence the connection between policies promoting asbestos-free materials and transitions towards sustainable urban development.

Adopting a MLP is crucial for understanding how the interplay of policy, technological advancements, and societal engagement facilitates asbestos-free transitions. The paper examines transformation processes through the lens of the MLP, with particular attention to the interactions among the niche (innovation), regime (systemic), and landscape (macro-environmental) levels. A detailed analysis of these levels is presented in Chap. 4. While this paper is grounded in transition theory, it incorporates empirical insights from the reviewed literature to illustrate how theoretical constructs manifest in real-world contexts. Future work may benefit from a more structured quantitative synthesis to further validate the observed patterns across studies.

3.2 Pathways of the strong sustainability transitions

Analyses of strong sustainability transition processes often examine the interactions among various levels, facilitating an understanding of how social and technological changes shape these transition dynamics [127]. The MLP framework utilizes pathway descriptions to outline transition pathways, offering a comprehensive and nuanced analysis of sociotechnical transformations [128]. These descriptions provide a holistic understanding of different levels, ranging from niche innovations to the transformation of established systems, and capture their intricate interactions [78]. Pathway characterizations provide a comprehensive understanding that takes into account the intricate interrelationships between social dynamics and technological advancements across societal, economic, and environmental domains [129]. The characterization of resilient sustainability transition pathways involves classifying the developmental process of a system, functioning as a practice-oriented and policy-driven research methodology for exploring socio-technical complexity [123, 130]. Within this approach, the MLP framework emphasizes the interactions between different levels—innovative niches, the dominant regime, and macro-environmental pressures—thereby enabling a comprehensive interpretation of the evolutionary dynamics of social phenomena and technological transformations [131]. Pathway descriptions allow for the identification of pivotal junctures where experimental, radical innovations interact with stable yet adaptable systemic structures, while macro-environmental challenges—such as ecological crises and shifts in societal values—also play a decisive role in system transformation [132, 133]. This holistic viewpoint highlights that the success of resilient sustainability transitions is not merely the result of technological or economic factors but is fundamentally contingent on profound social, cultural, and institutional changes [134]. These changes integrate the protection of natural capital and adherence to ecological limits into the functioning of society [135]. Thus, pathway descriptions also serve as strategic guidelines for policymakers, assisting them in identifying critical intervention points that can facilitate the transition towards a sustainable future [136].

3.3 The conceptualisation and application of MLP

This research model adopts a hierarchical perspective to examine urban transitions. At the first level, the focus is on niche innovations and socio-technical experiments that drive the development of asbestos-free and sustainable cities. The second level encompasses the regime dynamics, including regulatory frameworks, economic structures, and social norms that influence the persistence or transformation of existing urban practices. The third level represents the broader landscape factors, such as global sustainability trends, climate policies, and public health imperatives, which shape the overall conditions for transition. The analysis employs a MLP framework to investigate the interactions between these levels and their role in enabling or impeding sustainable urban transitions. The research model applied in this paper is illustrated in Fig. 1. The MLP is a methodology that facilitates the assessment of how interactions across diverse levels shape transition pathways.

It is particularly advantageous for analysing intricate socio-technical systems, wherein lower-level niche innovations interact with deeply entrenched regime structures, subject to the influence of external landscape pressures. Examining transitions using single-level analytical tools risks overlooking the interdependencies between these levels, leading to incomplete or misleading interpretations of transition dynamics. This could culminate in a reductionist approach that fails to account for the systemic barriers and enablers of sustainable change. By employing MLP, this paper systematically evaluates the role of various factors in asbestos-free urban transitions, ensuring a comprehensive understanding of the conditions required for sustainability-oriented urban transformation.

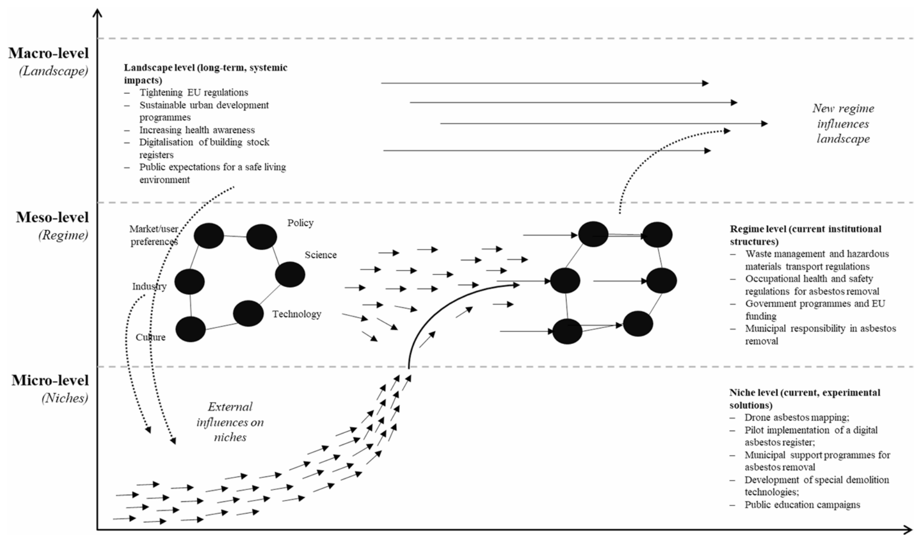


Fig. 1 MLP on the transition to strong sustainability for asbestos-free settlements

4 Conceptualisation and operationalization of MLP elements

4.1 Niches

Niche innovations catalyse interdependent transformations involving emerging technologies, practices, stakeholder configurations, values, networks, and policies [86]. Niches represent new technological, regulatory, organizational, or conceptual developments [137]. In the context of transitioning cities to be asbestos-free, niche innovations encompass novel asbestos-free construction materials, alternative demolition and waste management approaches, and policy-driven urban renewal initiatives [138, 139]. According to the multi-level perspective framework, the transition to asbestos-free cities requires niche-level innovations. These include new technologies, policy changes, and stakeholder-led advocacy to eliminate asbestos in urban areas [49]. Achieving this transition requires the adoption of new material alternatives, the implementation of stricter regulatory frameworks, and the transformation of construction and demolition practices [62]. Research indicates that successful transitions are contingent upon the presence of robust and well-supported niche innovations that can challenge the prevailing socio-technical regime [78, 140–142]. Niches are initiatives where alternative practices and rules emerge within networks of actors [143–145]. The niche concept is analogous to alternative urban renewal strategies, but it extends beyond just these networks. Instead, it originates from evolutionary economics, which examines technological progress and how alternative networks facilitate the adoption of new technologies [146]. Examples of niches in urban sustainability include green building certifications, circular economy projects, and sustainable urban planning frameworks [147]. The promotion of non-toxic materials and sustainable construction policies can be considered niche-level innovations [147–149]. However, alternative approaches do not always lead to major changes [150–152]. Therefore, it is crucial to clearly define the niche being studied and its distinctiveness from the existing system [153].

The MLP framework originally conceptualized niches as protective environments fostering innovation [51, 78, 131]. However, in the context of asbestos-free urban transitions, scholars commonly use the niche concept to describe emerging regulatory policies, alternative building materials, and industry best practices [154–156]. The degree to which niche innovations disrupt the dominant socio-technical system depends on their maturity and capacity for upscaling [157, 158]. The internal niche processes of formation, development, and actor-network configurations are crucial for enabling sustainable transitions [159]. From a business model perspective, niche innovations necessitate sustainable financing mechanisms, viable revenue streams, and market demand to achieve successful scaling. The widespread adoption of asbestos-free alternatives relies not only on regulatory pressures but also on the ability of firms to integrate cost-effective production, distribution, and value propositions that appeal to industry stakeholders. For instance, firms developing asbestos-free materials must ensure competitive pricing, secure supply chain partnerships, and effectively communicate the long-term economic benefits of their products. Furthermore, innovative business models, such as product-service systems, can facilitate niche expansion by enabling companies to offer integrated demolition and asbestos-free reconstruction services, rather than merely selling materials [160, 161].

The scaling-up of asbestos-free niche innovations, however, presents challenges [32]. As niche solutions become more widespread, they risk dilution or partial compromise of their original sustainability attributes. For instance, while regulatory frameworks encourage asbestos-free materials, economic and logistical constraints may result in partial compliance rather than complete adoption. The "paradox of mainstreaming" in sustainability transitions, describing how the upscaling of niche innovations often leads to trade-offs between extensive diffusion and adherence to core sustainability principles [123, 162–165]. The diversity among niche actors further complicates the dynamics of transitioning to asbestos-free environments [82]. These niche actors often hold a range of motivations, backgrounds, and strategic visions, which can lead to fragmentation within efforts to transition away from asbestos [166]. Differing normative perspectives on public health and environmental safety shape the trajectories of these transitions, with actors' views on asbestos regulation influencing whether they pursue gradual reforms or advocate for more radical transformations, regardless of their alignment with the existing regime or niche positions [32, 50, 166, 167]. Furthermore, niche actors must navigate the interactions with the established regime, balancing their engagement with the mainstream construction and demolition sectors while simultaneously advocating for systemic change [168, 169].

Business model innovation is vital for addressing these challenges. Leveraging new funding sources like green bonds, public-private partnerships, and circular economy models can help niche players create more stable financial conditions for asbestos-free transitions [170]. Startups focused on sustainable construction can benefit from impact investment funds and carbon credit schemes that incentivize non-toxic building practices. Additionally, business models incorporating lifecycle cost assessments can demonstrate the long-term financial benefits of asbestos-free materials, reducing adoption barriers for construction firms and real estate developers [171]. The sustainable construction movement leverages market mechanisms to amplify its influence while challenging traditional industry practices [172, 173]. Similarly, in transitioning to asbestos-free environments, regulatory authorities, industry leaders, and grassroots groups must work together to advance niche innovations while mitigating the risk of co-option. The dynamic interplay between niche actors and the broader socio-technical system underscores the complexity of shifting towards asbestos-free urban settings, emphasizing the need for strategic interventions that reinforce niche resilience and enable systemic transformation [167].

4.2 Regimes

Within the broader field of transition research, studies on asbestos-free and sustainable urban transitions frequently focus on the dynamics and management of transitions rather than explicitly defining the underlying regime structures [106, 123]. In the context of asbestos-related transitions, the dominant urban regime primarily encompasses the entrenched construction and infrastructure sectors, which are governed by established regulations, industry norms, economic structures, and socio-political arrangements that have historically facilitated the continued use of asbestos-based materials [49, 174–176]. This regime spans multiple dimensions, including regulatory policies, economic interests, technological standards, and professional practices, all of which shape the persistence or transformation of asbestos-related urban practices [32].

The asbestos-related regime is characterized by institutionalized regulations, long-standing industry practices, material supply chains, and socioeconomic interdependencies that sustain conventional construction methods [32, 49]. The built environment sector is defined by deeply embedded regulatory frameworks governing the demolition, renovation, and disposal of asbestos-containing materials, often leading to gradual changes rather than transformative shifts towards sustainable alternatives [177–179]. Modern construction and urban management approaches are highly standardized, with extensive rules regulating materials, labour safety protocols, and waste management systems [102]. The regime encompasses key governmental agencies, regulatory bodies, industry actors, and professional associations that shape the prevailing discourse on urban safety, environmental sustainability, and public health regarding asbestos removal and mitigation [62, 180]. A comprehensive understanding of the urban regime should extend beyond policy and industry dynamics to incorporate social and economic dimensions, including public perceptions, risk awareness, and the economic interests of stakeholders within the urban development sector [181–183].

Despite increasing recognition of the hazards posed by asbestos, the prevailing system resists change due to the entrenched interests of industry stakeholders, institutional inertia, and regulatory complexities [50]. In many urban areas, legacy infrastructure containing asbestos presents significant challenges, as comprehensive removal and replacement necessitate substantial financial resources, technological innovations, and coordinated policy efforts [184, 185]. Moreover, research indicates that the dominant regulatory discourse tends to be technocratic, prioritizing incremental risk mitigation and compliance with existing safety standards, rather than advocating for fundamental shifts towards sustainable urban materials and construction practices [186, 187]. Similarly, scholars have argued that the construction and demolition sectors are largely controlled by major industry players and regulatory institutions, reinforcing established practices and hindering the pace of transformation [168]. Existing research on asbestos-free transitions predominantly concentrates on the regulatory aspects of socio-technical regimes, possibly due to their tangible and enforceable nature, while the normative and cognitive dimensions are less frequently considered [82, 188]. However, the cultural and cognitive dimensions play a pivotal role in shaping resistance to change, as public and industry perceptions regarding asbestos safety measures and alternative materials significantly influence the adoption of new practices [167, 189].

The asbestos-related regime faces significant barriers to transformation, including 'lock-in' and 'path dependency' issues that underlie the structural and institutional resistance to change within urban construction and infrastructure systems [49]. For instance, regulatory lock-ins in many cities manifest as bureaucratic obstacles and legal ambiguities concerning liability for asbestos removal, while economic path dependencies stem from the high costs of replacing asbestos-laden infrastructure [190]. Additionally, research indicates that socio-technical regimes persist despite evident contradictions due to the well-established routines, financial incentives, and risk-averse nature of institutional decision-making processes [191]. Nevertheless, scholars have observed that all socio-technical regimes exhibit inherent contradictions, which create prospects for gradual shifts and system reconfigurations [141, 192].

From a business model perspective, the persistence of asbestos-based systems is heavily influenced by financial feasibility and market incentives. The economic self-interests of construction firms, material suppliers, and waste management companies shape the pace of transition toward asbestos-free alternatives [167]. Business models leveraging cost-effective yet sustainable materials, circular economy approaches, and financial incentives for compliance with asbestos-free regulations could play a pivotal role in accelerating this shift. However, the transition also entails economic risks, such as elevated costs of asbestos abatement and alternative materials, which may discourage industry actors from proactively embracing change [48].

Recent academic works underscore the need for heightened focus on lateral interactions between established regimes and emerging niches, highlighting the significance of multi-regime and multi-niche dynamics in urban sustainability transitions [78]. Some scholarly investigations examine the intersections between asbestos-related transitions and broader sustainability initiatives, encompassing green building programs, circular economy models, and public health policies [193]. The integration of innovative business models that incorporate sustainability certifications, environmentally-friendly financing mechanisms, and extended producer responsibility frameworks can aid in bridging the divide between niche innovations and entrenched industry structures, thereby facilitating a more economically feasible transition toward asbestos-free urban environments [32, 48, 194]. Concurrently, the shift towards asbestos-free cities as an exemplar of multi-regime interactions, where the realignment of construction, waste management, and environmental health regimes is necessary to achieve meaningful systemic transformation [193]. Defining distinct system boundaries remains a persistent obstacle in studies of asbestos-related transitions. The delineation of 'the regime' to be investigated and potentially managed is not inherently predetermined, but rather a matter of framing and deliberation - a challenge that continues to shape contemporary research on asbestos-free and sustainable urban transformations [167]. Given the interdependencies among regulatory frameworks, industry structures, and societal behaviours, future inquiries must adopt a comprehensive perspective that considers both the systemic impediments and facilitating conditions for asbestos-free urban transitions [195].

4.3 Landscape

Research on sustainability transitions often overlooks the landscape level, including transitions to asbestos-free cities. While some scholars say they don't examine landscape dynamics in depth to stay focused, most studies simply leave out the landscape level without explanation [196, 197]. This suggests the landscape category is sometimes seen as a 'catch-all' for factors that don't fit neatly into the niche or regime levels. However, external trends and outside factors play a key role in shaping asbestos-free urban transitions, showing the need to better incorporate the landscape level in these analyses [198, 199]. Key landscape pressures that facilitate asbestos-free urban transitions encompass global health regulations, technological advancements in safer construction materials and demolition practices, economic factors including global trade policies and fluctuations in the cost of alternative materials, as well as socio-political changes such as growing public awareness and advocacy against hazardous substances. Landscape-level pressures can not only destabilize existing regimes but also generate opportunities for niche innovations. Likewise, increased international investment in sustainable urban development projects and more stringent environmental regulations can accelerate asbestos-free transitions by incentivizing alternative construction techniques and supporting innovative industries in this domain [102, 150]. Additionally, disruptive occurrences such as natural disasters and urban renewal initiatives can serve as pivotal moments where momentum builds for asbestos removal. From a business model perspective, these disruptions present opportunities for new market participants to introduce asbestos-free construction offerings, granting a competitive edge to firms that emphasize sustainability and regulatory adherence [200].

The landscape protects new practices from dominant industry approaches. Policies can shield emerging niches; for asbestos-free transitions, government support and legal frameworks can protect innovative, asbestos-free construction from conventional, asbestos-reliant methods [32, 201]. Evolving consumer preferences and corporate commitments to sustainability compel construction firms to adopt more environmentally-friendly business models, thereby shaping investment patterns and strategic choices in the industry [101]. Asbestos bans and urban renewal can create parallel opportunities for green building, circular construction, and environmental health monitoring. The leveraging of financial instruments like green bonds, impact investment funds, and collaborative public-private initiatives can play a crucial role in facilitating the scaling of asbestos-free urban innovations. Regulatory interventions at the landscape level alter legal frameworks and reshape urban practices and socioeconomic interactions [105]. The literature on the MLP suggests that landscape pressures can both destabilize and reinforce the status quo. The landscape analysis should consider stabilizing forces, which are relevant in phasing out asbestos in cities where factors like weak enforcement or industry inertia can delay or block progress [202]. In this context, financial incentives, tax incentives, and sustainability-focused financing instruments can help mitigate economic barriers to transitional efforts, thereby supporting the long-term sustainability of businesses engaged in asbestos-free construction.

5 Discussion

5.1 Understanding of MLP and sustainable asbestos-free transition pathways

The MLP framework conceptualizes sustainability transitions as the product of dynamic interplays among niche, regime, and landscape levels [106]. In the context of asbestos-free transitions in urban settings, these multilevel interactions determine the breadth, pace, and magnitude of change. Nevertheless, a key limitation is the inadequate examination of how these distinct levels interrelate and mutually shape one another [69, 91, 113]. This limitation pertains to the MLP framework itself, which tends to emphasize the individual levels rather than thoroughly analyzing the dynamic and reciprocal interactions between them [158, 203]. Existing research on sustainability transitions often focuses on how dominant sociotechnical regimes resist alternative solutions, or how niche innovations challenge incumbent systems [123]. In the context of transitioning to asbestos-free settlements, a key challenge for niche initiatives is their limited influence over broader regulatory and market-driven dynamics [167]. The success of such transitions hinges on whether niche-regime interactions catalyse systemic shifts, thereby enabling asbestos-free alternatives to be integrated into mainstream construction and urban planning practices. The timing and sequencing of the transition are critical factors, as the phased implementation of policies prohibiting asbestos permits a gradual adaptation by industries, regulatory bodies, and communities, thereby mitigating economic disruptions and ensuring long-term sustainability (Table 2).

The dynamics of transition in asbestos-free transitions depend on whether niche actors work within the existing regime or actively challenge and reshape it. Niche development is essential but not always enough to trigger a full regime shift [78]. These transitions involve interactions across legislation, financial incentives, technology, and public awareness, including lobbying, negotiation, and knowledge-sharing [204]. Crucial to these transitions is the exchange of expertise and institutional support, which can facilitate the adoption of safer alternatives. However, entrenched practices, economic dependencies, and lack of incentives often obstruct these processes [205]. Successful asbestos-free transitions depend on aligning networks of actors, social dynamics, and strategic alliances. Building connections between niche and regime actors can create opportunities for policy, funding, and institutional support [87, 206]. Engagement with regime actors can help niches gain legitimacy and sustainability, particularly where regulatory frameworks dictate the transition. Balancing challenging dominant norms and aligning with existing structures ensures a smooth transition, though the outcome depends on context-specific factors [207].

Business model innovation is vital for scaling up asbestos-free alternatives. Sustainable construction firms can make niche solutions economically feasible and competitive. Financial incentives and circular economy models can further drive the adoption of these alternatives by aligning market dynamics with sustainability objectives [208]. Despite its widespread application in transition studies, the MLP framework has faced various critiques, particularly regarding its capacity to account for agency, power dynamics, and structural rigidity within transition processes. A key criticism is that MLP often assumes a relatively linear and structured transition trajectory, potentially underestimating the complexity and unpredictability inherent in real-world change dynamics. Furthermore, the framework has been faulted for inadequately addressing power struggles, vested interests, and institutional resistance, all of which are highly salient

Table 2 Phases of asbestos-free transition

Transition phase	Key processes	Major risks and barriers	Key stakeholders and their roles	Intervention strategies and instruments
Preparation (0–5 years)	Mapping asbestos presence and collecting data	Incomplete databases and lack of asbestos location knowledge	Government (legislation, funding programs)	Analysis of international and EU directives
	Legislative preparation and international benchmarking	Uncertainty in legislation and financing	Local authorities (local regulations, pilot projects)	Development of GIS-based asbestos databases
	Pilot programs and local test projects		Research institutions (technological solutions)	R&D funding and support for pilot initiatives
Early transition (5–10 years)	Introduction of regulations and financial incentives	Market resistance to new regulations	Construction companies (transition to asbestos-free materials)	Tax incentives and subsidies to facilitate the transition
	Support for the transition of construction industry players	High initial investment costs	Waste management entities (new recycling technologies)	Awareness campaigns for industry stakeholders
	Expansion of waste management capacities		Financial institutions (green financing mechanisms)	Cost-benefit analyses to highlight long-term advantages
Accelerated transition (10–20 years)	Mainstreaming recycling and circular economy solutions	Challenges in infrastructure adaptation (machinery, waste processing)	Companies (development of sustainable business models)	Green public procurement and mandatory sustainability standards
	Workforce training programs and industrial capacity expansion	Shortage of skilled labour for new technologies	Educational institutions (vocational training for new technologies)	EU and national funding for industrial investments
			Technology providers (innovation in processing and materials)	Implementation of extended producer responsibility (EPR) schemes
Stabilization (20 + years)	Complete removal of asbestos from urban environments	Persistence of illegal asbestos use in certain segments	Civil society organizations (awareness, monitoring)	Strict enforcement against illegal asbestos trade
	Green building standards becoming the norm	Social justice issues (affordable housing concerns)	Local communities (participatory governance, oversight)	Social housing programs to ensure a just transition
	Full establishment of waste processing infrastructure		International organizations (WHO, UNEP – global standardization)	Adoption of international sustainability frameworks

in asbestos-free transitions where industry stakeholders may actively resist regulatory changes. These limitations suggest that while MLP offers a useful conceptual tool for understanding asbestos-free transitions, its explanatory power would benefit from being complemented by other perspectives that more effectively capture the socio-political dimensions of change.

5.2 Assessment of strong sustainable asbestos-free transition

Aligning networks of diverse stakeholders, navigating social complexities, and forging strategic alliances are crucial for successful asbestos-free transitions. Bridging the gap between niche innovators and established regime actors can unlock opportunities for policy interventions, funding streams, and institutional backing [70, 201]. Sustainability transitions research grapples with the tension between the inherent uncertainty of

Table 3 Business model strategies for the transition to asbestos-free urban environments

MLP level	Business model components	Market and financial mechanisms	Regulatory and institutional frameworks	Social and environmental impacts
Niche	Gradual replacement of asbestos-containing materials with existing alternatives, development of renovation and demolition services, utilization of recycling opportunities, circular economy-based strategic planning.	Municipal support for demolition and renovation projects, R&D incentives for recycling technology development, financing for circular economy initiatives.	Pilot programs and experimental regulations at the local level, cooperation with research institutions, involvement of professional organizations, development of a municipal "value inventory" for circular economy integration.	Community engagement, awareness-raising on asbestos risks, implementation of circular economy-based community services, waste-to-resource conversion.
Regime	Transition of existing construction and waste management companies to asbestos-free materials, integration of circular economy principles into corporate operations, development of green employment strategies.	National and EU funding for renovation and demolition projects, tax incentives for circular economy practices, financial instruments supporting green procurement.	Strengthened building and waste management regulations, mandatory asbestos removal in public buildings, regulatory frameworks supporting circular business models.	Reduction of asbestos exposure in urban environments, improvement of public health indicators, enhancement of living conditions through sustainable construction practices.
Landscape	Global trends in construction sustainability and health protection, long-term goals for green building and circular economy adaptation, promotion of shared economy models.	International financial mechanisms (e.g., EU cohesion funds, sustainability investments), ESG requirements for construction companies, economic incentives for circular production.	Tightening of EU and global regulations, recommendations from WHO and other international organizations, sustainable urban development strategies.	Long-term public health benefits, sustainable urban development, reduction of construction waste and material recirculation, knowledge management, and awareness-building on circular economy potential.

transition processes and the normative objective of sustainability [123]. This challenge is especially salient in asbestos-free transitions, where the impact of change must be evaluated not only in terms of regulatory adherence, but also with respect to public health, environmental safety, and long-term socioeconomic implications [209, 210]. Successful asbestos-free transitions address the key challenges of health risks, high costs, limited disposal options, and industry resistance. Transition initiatives should aim to reduce asbestos-related illnesses, develop safe replacement materials, and establish regulatory frameworks that prevent future asbestos contamination [32]. Table 3 provides a summary of the key business models, regulatory mechanisms, and market incentives that drive the sustainable transition to asbestos-free urban environments across the three levels of the transition framework—niche, regime, and landscape.

These models must generate sustainable value by enhancing public well-being and environmental protection, while ensuring the economic viability of asbestos-free alternatives through scalable and cost-effective solutions [48]. Research on sustainability transitions often focuses on the change process, with fewer studies providing detailed, quantitative assessments of actual impacts. This is particularly true for asbestos-free transitions, where key indicators like reduced asbestos-related diseases, improved air quality, or construction sector changes are often underdeveloped or inconsistently applied [49, 50]. Despite challenges in measuring systemic influence, specific indicators can offer insights into transition progress, such as adoption rates of asbestos-free materials, industry practice changes, and shifts in public perception and policy. Importantly, the normative aspects of asbestos-free transitions raise equity questions about who benefits, who bears costs, and whether solutions are equitable and sustainable long-term [26, 30]. These considerations highlight the need for a comprehensive approach that goes beyond technical substitution to include social, economic, and political dimensions [211].

A key concern is the potential risk of prioritizing transition processes without rigorously evaluating their outcomes. Simply substituting asbestos with alternative materials does not necessarily ensure a sustainable transition, particularly if those alternatives present their own environmental or health-related drawbacks [185]. Accordingly, an assessment of the transition should not only measure progress but also undertake a critical examination of whether new practices align with broader sustainability objectives. This is crucial to ensure that asbestos-free transitions contribute to the development of safer, healthier, and more resilient urban environments [82].

Moreover, a critical examination of the validity of transition pathways is necessary, given that the MLP tends to emphasize the structural aspects of change while overlooking the role of conflicts, political contestation, and the agency of specific actors. In the case of transitioning away from asbestos, various stakeholders, ranging from regulatory bodies to construction industry actors, may have competing interests, leading to a fragmented and uneven adoption of alternative materials. The limitations of MLP in accounting for these diverging interests suggest that additional analytical approaches, such as institutional theory or political economy perspectives, could provide a more comprehensive understanding of the challenges associated with transitions and their broader societal implications.

This paper is subject to several limitations that should be acknowledged. First, although the analysis is grounded in transition theory and supported by selected empirical examples, the synthesis remains largely conceptual. The inclusion of more systematic empirical data, such as comparative case studies or quantitative meta-analyses, would enhance the robustness of the findings and allow for stronger generalizability across contexts. Second, while the review identifies key drivers and barriers of asbestos-free urban transitions, it does not provide a comprehensive geographic analysis. Differences in regulatory stringency, implementation capacity, and transition progress across countries or regions are not systematically addressed. Future research should map these spatial variations to better understand the enabling or hindering conditions in diverse national and local settings. Third, although business models are frequently referenced as potential enablers of sustainable transitions, empirical data on their practical implementation and effectiveness are limited. This paper primarily discusses their theoretical significance; however, evaluating their real-world performance remains an important direction for further investigation. These limitations open up promising avenues for future research, particularly the need for geographically differentiated studies, quantitative synthesis of policy impacts, and empirical assessments of organizational innovation in transition contexts.

6 Conclusions

This research leveraged the MLP approach to examine the transition towards asbestos-free built environments, underscoring the complex interplay between policy, industry dynamics, and societal influences. The findings suggest that while the availability of novel asbestos-free materials, more stringent regulations, and enhanced waste management practices have enabled progress, deeply entrenched industry norms and institutional path dependencies persist as significant obstacles.

The research findings elucidate the systemic obstacles that obstruct asbestos-free transitions. Resistance from established industries, economic interdependencies, and regulatory sluggishness frequently hinder the enactment of asbestos-free policies. Nonetheless, external forces, such as evolving global sustainability priorities and technological innovations, generate avenues for change, motivating local governments and industries to embrace safer substitutes. Regulatory frameworks are fundamental in determining the success of initiatives aimed at eliminating asbestos. Regions with robust enforcement mechanisms are more likely to realize effective policy implementation, while lax oversight allows asbestos-containing materials to remain prevalent in urban areas. This highlights the necessity for coordinated, multi-level governance approaches that harmonize national policies with local enforcement capacities. Technological advancements and industrial adaptability are also essential for expediting the transition to asbestos-free alternatives. The accessibility of cost-effective substitutes, progress in demolition safety techniques, and enhanced waste management solutions all contribute to the practicability of phasing out asbestos usage. However, realizing widespread adoption necessitates both regulatory backing and industry commitment to invest in sustainable practices. The MLP framework provides valuable insights into the interactions between various levels of governance and industry in shaping sustainability transitions.

Existing research using the MLP framework suggests limitations in addressing the socio-political complexities of transitioning to asbestos-free construction. To gain a more nuanced understanding, future studies should incorporate alternative analytical approaches. Examining power dynamics, institutional resistance, and stakeholder strategies will be crucial in assessing the feasibility and long-term impacts of transition policies. By integrating complementary theoretical perspectives, researchers can provide a more comprehensive view of how sustainability transitions unfold in highly contested and regulated domains, such as asbestos-free construction.

Future research should focus on quantitative evaluations of the outcomes of asbestos-free policies, assessments of the long-term implications of alternative materials, and investigations into the role of public engagement in driving change. A comprehensive approach that integrates policy interventions, technological advancements, and cross-sector collaboration will be crucial for ensuring a successful and sustainable transition to asbestos-free urban environments.

Author contributions

Gergely Zoltán Macher and Cecília Szigeti jointly conceived and wrote the manuscript. Both authors contributed equally to the drafting, revision, and final approval of the text.

Funding

No funding was received for this study.

Data availability

No datasets were generated or analysed during the current study.

Declarations

Ethical approval
This study does not contain any studies with human participants or animals performed by any of the authors. Therefore, ethical approval was not applicable.

Consent to publish
Not applicable. This study does not contain any individual person's data in any form.

Consent to participate
Not applicable. This study does not involve human participants.

Clinical trial number
Not applicable.

Competing interests
The authors declare no competing interests.

Received: 17 March 2025 / Accepted: 8 September 2025
Published online: 29 September 2025

References

1. Moerman L, van der Laan S. Toxic Products and Hazardous Waste: The Global Asbestos Issue, 2015, pp. 103–20. https://doi.org/10.1108/S2043-052320150000008005
2. Pathak A, Giri M, Pokhrel B, Wagle MN. Identification of asbestos content in bulk materials imported in Nepal. Sci World. 2020;13:5–7. https://doi.org/10.3126/sw.v13i13.30481.
3. Gualtieri AF. Naturally occurring asbestos: A global health concern? State of the Art and open issues. Environ Eng Geosci. 2020;26:3–8. https://doi.org/10.2113/EEG-2271.
4. K KW. S.H. C. Malignant pleural mesothelioma and the role of Non– operative therapies. Principles and practice of cardiothoracic surgery. InTech; 2013. https://doi.org/10.5772/55722.
5. Ascoli V, Comba P, Pasetto R. Urban mesothelioma: Is there an emerging risk of asbestos in place? Int J Cancer. 2004;111:975–6. https://doi.org/10.1002/ijc.20346.
6. Ross M, Nolan RP. History of asbestos discovery and use and asbestos-related disease in context with the occurrence of asbestos within ophiolite complexes. Ophiolite concept and the evolution of geological thought. Geological Society of America; 2003. https://doi.org/10.1130/0-8137-2373-6.447.
7. Frank AL. Global use of asbestos - legitimate and illegitimate issues. J Occup Med Toxicol. 2020;15:16. https://doi.org/10.1186/s12995-020-00267-y.
8. Butkevics J, Atstaja D. Diversion of asbestos-containing waste from landfilling: opportunities and challenges. Sustainability. 2025;17:4529. https://doi.org/10.3390/su17104529.
9. van Zandwijk N, Rasko JEJ, George AM, Frank AL, Reid G. The silent malignant mesothelioma epidemic: a call to action. Lancet Oncol. 2022;23:1245–8. https://doi.org/10.1016/S1470-2045(22)00269-8.
10. Frank L, Joshi A. The global spread of asbestos. Ann Glob Health. 2014;80:257. https://doi.org/10.1016/j.aogh.2014.09.016.
11. Damiran N, Frank A. Asbestos consumption in mongolia: 1996–2014. Int J Environ Res Public Health. 2018;15:136. https://doi.org/10.3390/ijerph15010136.
12. Furuya S, Chimed-Ochir O, Takahashi K, David A, Takala J. Global asbestos disaster. Int J Environ Res Public Health. 2018;15:1000. https://doi.org/10.3390/ijerph15051000.
13. Park E, Takahashi K, Jiang Y, Movahed M, Kameda T. Elimination of asbestos use and asbestos-related diseases: an unfinished story. Cancer Sci. 2012;103:1751–5. https://doi.org/10.1111/j.1349-7006.2012.02366.x.
14. Kumar A, Prasad MNV, Maiti SK. Tripti. Asbestos. Environmental Materials and Waste, Elsevier; 2016, pp. 285–305. https://doi.org/10.1016/B978-0-12-803837-6.00013-5
15. Geiser K. Making materials matter. New solutions: a journal of environmental and occupational. Health Policy. 2002;12:157–75. https://doi.org/10.2190/NW04-YTLL-C8PV-UFMH.
16. Dixon T, Lannon S, Eames M. Reflections on disruptive energy innovation in urban retrofitting: methodology, practice and policy. Energy Res Soc Sci. 2018;37:255–9. https://doi.org/10.1016/j.erss.2017.10.009.
17. Song Y, Zhang H. Research on sustainability of building materials. IOP Conf Ser Mater Sci Eng. 2018;452:022169. https://doi.org/10.1088/1757-899X/452/2/022169.
18. Janjua SY, Sarker PK, Biswas WK. A review of residential buildings' sustainability performance using a life cycle assessment approach. J Sustain Res 2019. https://doi.org/10.20900/jsr20190006
19. Hauke B, Kuhnhenne M, Lawson M, Veljkovic M, Siebers R. What does 'sustainable construction' mean? An overview. Sustainable Steel Buildings, Wiley; 2016, pp. 1–12. https://doi.org/10.1002/9781118740828.ch1
20. Zajemska M, Wojtyto D, Michalik J, Berski S. Review of current trends in sustainable construction. Energies (Basel). 2025;18:2559. https://doi.org/10.3390/en18102559.
21. Lohunova O, Wyjadłowski M. Modification of vibratory driving technology for sustainable construction works. MATEC Web Conf. 2018;251:03063. https://doi.org/10.1051/matecconf/201825103063.
22. Zimmer AT, Ha H. People, planet and profit: unintended consequences of legacy building materials. J Environ Manag. 2017;204:472–85. https://doi.org/10.1016/j.jenvman.2017.09.026.
23. Brandt T, Ketter W, Kolbe LM, Neumann D, Watson RT. Smart cities and digitized urban management. Bus Inform Syst Eng. 2018;60:193–5. https://doi.org/10.1007/s12599-018-0537-1.
24. Safitri Zen I, Ahamad R, Gopal Rampal K, Omar W. Use of asbestos building materials in Malaysia: legislative measures, the management, and recommendations for a ban on use. Int J Occup Environ Health. 2013;19:169–78. https://doi.org/10.1179/2049396713Y.0000000028.

25. Ellen MacArthur Foundation. Towards the Circular economy: economic and business rationale for an accelerated transition. Cowes: 2013.
26. Vincenten J, George F, Martuzzi M, Schröder-Bäck P, Paunovic E. Barriers and facilitators to the elimination of asbestos related diseases—Stakeholders' perspectives. Int J Environ Res Public Health. 2017;14:1269. https://doi.org/10.3390/ijerph14101269.
27. Li J, Dong Q, Yu K, Liu L. Asbestos and asbestos waste management in the Asian-Pacific region: trends, challenges and solutions. J Clean Prod. 2014;81:218–26. https://doi.org/10.1016/j.jclepro.2014.06.022.
28. Silvestri S. Managing asbestos in Italy: twenty years after the ban. NEW SOLUTIONS: a journal of environmental and occupational. Health Policy. 2013;22:489–96. https://doi.org/10.2190/NS.22.4.g.
29. Wu HY-J, Lin R-T, Wang J-D, Cheng Y. Transnational dynamics amid poor regulations: Taiwan's asbestos ban actions and experiences. Int J Environ Res Public Health. 2017;14:1240. https://doi.org/10.3390/ijerph14101240.
30. Allen L, Baez J, Stern M, Takahashi K, George F. Trends and the economic effect of asbestos bans and decline in asbestos consumption and production worldwide. Int J Environ Res Public Health. 2018;15:531. https://doi.org/10.3390/ijerph15030531.
31. Irwig HG, Oliver LC, Page T, Wegman DH, Ellenbecker MJ. Asbestos in place: a building management perspective. Ann N Y Acad Sci. 1991;643:589–96. https://doi.org/10.1111/j.1749-6632.1991.tb24510.x.
32. Nabango H, Majale C. Towards a circular economy: a review on asbestos waste management regulations in Kenya. East Afr J Environ Nat Resour. 2022;5:34–47. https://doi.org/10.37284/eajenr.5.1.548.
33. Hinze J, Hinze MM. Problems associated with removal of asbestos. J Constr Eng Manag. 1986;112:211–9. https://doi.org/10.1061/(ASCE)0733-9364(1986)112:2(211).
34. Chrostowski PC, Foster SA, Anderson EL. Human health risks associated with asbestos abatement. Risk Anal. 1991;11:465–81. https://doi.org/10.1111/j.1539-6924.1991.tb00632.x.
35. Durrani N, Raziq A, Mahmood T, Khan MR. Barriers to adaptation of environmental sustainability in smes: a qualitative study. PLoS ONE. 2024;19:e0298580. https://doi.org/10.1371/journal.pone.0298580.
36. Obmiński A. Comparison of the effects of use, protection, improper renovation and removal of asbestos products on the example of typical old office buildings in Poland. Sci Rep. 2023;13:13577. https://doi.org/10.1038/s41598-023-37257-z.
37. Campitelli A, Kannengießer J, Schebek L. Approach to assess the performance of waste management systems towards a circular economy: waste management system development stage concept (WMS-DSC). MethodsX. 2022;9:101634. https://doi.org/10.1016/j.mex.2022.101634.
38. Csutora M, Harangozo G, Szigeti C. Sustainable business models—crisis and rebound based on Hungarian research experience. Resources. 2022;11:107. https://doi.org/10.3390/resources11120107.
39. Bocken NMP, Short SW, Rana P, Evans S. A literature and practice review to develop sustainable business model archetypes. J Clean Prod. 2014;65:42–56. https://doi.org/10.1016/j.jclepro.2013.11.039.
40. Høgevold NM, Svensson G, Padin C. A sustainable business model in services: an assessment and validation. Int J Qual Service Sci. 2015;7:17–33. https://doi.org/10.1108/IJQSS-09-2013-0037.
41. Tunn VSC, Bocken NMP, van den Hende EA, Schoormans JPL. Business models for sustainable consumption in the circular economy: an expert study. J Clean Prod. 2019;212:324–33. https://doi.org/10.1016/j.jclepro.2018.11.290.
42. Reinecke PC, Küberling-Jost JA, Wrona T, Zapf AK. Towards a dynamic value network perspective of sustainable business models: the example of RECUP. J Bus Econ. 2023;93:635–65. https://doi.org/10.1007/s11573-023-01155-7.
43. Sesini G, Castiglioni C, Lozza E. New trends and patterns in sustainable consumption: a systematic review and research agenda. Sustainability. 2020;12:5935. https://doi.org/10.3390/su12155935.
44. Nosratabadi S, Mosavi A, Shamshirband S, Kazimieras Zavadskas E, Rakotonirainy A, Chau KW. Sustainable business models: a review. Sustainability 2019;11:1663. https://doi.org/10.3390/su11061663.
45. Ramaswami A. Chapter 20 Engineering sustainable urban infrastructure, 2006, pp. 411–34. https://doi.org/10.1016/S1871-2711(06)80027-9
46. Obinna Iwuanyanwu I, Gil-Ozoudeh AC, Okwandu. Chidiebere somadina ike. Retrofitting existing buildings for sustainability: challenges and innovations. Eng Sci Technol J. 2024;5:2616–31. https://doi.org/10.51594/estj.v5i8.1515.
47. Hariram NP, Mekha KB, Suganthan V, Sudhakar K. Sustainalism: an integrated socio-economic-environmental model to address sustainable development and sustainability. Sustainability. 2023;15:10682. https://doi.org/10.3390/su151310682.
48. Zoraja B, Živančev M, Ubavin D, Nakomčić-Smaragdakis B. Circular economy as possible solution for asbestos burden. IOP Conf Ser Mater Sci Eng. 2021;1163:012033. https://doi.org/10.1088/1757-899X/1163/1/012033.
49. Singh R, Frank AL. Does the presence of asbestos-containing materials in buildings post-construction and before demolition have an impact on the exposure to occupants in non-occupational settings? Cureus. 2023. https://doi.org/10.7759/cureus.37305
50. Emmett EA. Asbestos in high-risk communities: public health implications. Int J Environ Res Public Health. 2021;18:1579. https://doi.org/10.3390/ijerph18041579.
51. Geels FW. The multi-level perspective on sustainability transitions: responses to seven criticisms. Environ Innov Soc Transit. 2011;1:24–40. https://doi.org/10.1016/j.eist.2011.02.002.
52. Geels F. The Multi-Level Perspective on Sustainability Transitions: Background, overview, and current research topics 2024. https://doi.org/10.33774/coe-2024-c15gb
53. Geels FW. Technological transitions as evolutionary reconfiguration processes: a multi-level perspective and a case-study. Res Policy. 2002;31:1257–74. https://doi.org/10.1016/S0048-7333(02)00062-8.
54. Lim TK, Rajabifard A, Khoo V, Sabri S, Chen Y. The smart city in Singapore: How environmental and geospatial innovation lead to urban livability and environmental sustainability. Smart Cities for Technological and Social Innovation, Elsevier; 2021, pp. 29–49. https://doi.org/10.1016/B978-0-12-818886-6.00003-4
55. Iwaniec DM, Cook EM, Barbosa O, Grimm NB. The framing of urban sustainability transformations. Sustainability. 2019;11:573. https://doi.org/10.3390/su11030573.
56. Pardalis G, Mahapatra K, Mainali B. A triple-layered one-stop-shop business model canvas for sustainable house renovations. IOP Conf Ser Earth Environ Sci. 2020;588:022060. https://doi.org/10.1088/1755-1315/588/2/022060.
57. Cimprich A, Young SB, Schrijvers D, Ku AY, Hagelüken C, Christmann P, et al. The role of industrial actors in the circular economy for critical Raw materials: a framework with case studies across a range of industries. Mineral Econ. 2023;36:301–19. https://doi.org/10.1007/s13563-022-00304-8.

58. Gallagher TP. Proactive environmental management is the key to full compliance. Environ Qual Manag. 1991;1:37–44. https://doi.org/10.1002/tqem.3310010107.
59. Bolan S, Kempton L, McCarthy T, Wijesekara H, Piyathilake U, Jasemizad T, et al. Sustainable management of hazardous asbestos-containing materials: containment, stabilization and inertization. Sci Total Environ. 2023;881:163456. https://doi.org/10.1016/j.scitotenv.2023.163456.
60. Järvholm B, Burdorf A. Asbestos and disease—A public health success story? Scand J Work Environ Health. 2024;50:53–60. https://doi.org/10.5271/sjweh.4146.
61. Smith ME, Lobo J, Peeples MA, York AM, Stanley BW, Crawford KA et al. The persistence of ancient settlements and urban sustainability. Proc Natl Acad Sci 2021. https://doi.org/10.1073/pnas.2018155118
62. Stevulova N, Estokova A, Holub M, Singovszka E, Csach K. Characterization of demolition construction waste containing asbestos, and the release of fibrous dust particles. Appl Sci. 2020;10:4048. https://doi.org/10.3390/app10114048.
63. Buchanan J. Environmental trust? Sustainability and renewables policy and practice in the school years. Curriculum Perspect. 2021;41:163–73. https://doi.org/10.1007/s41297-020-00130-4.
64. Shi L, Han L, Yang F, Gao L. The evolution of sustainable development theory: types, goals, and research prospects. Sustainability. 2019;11:7158. https://doi.org/10.3390/su11247158.
65. Da Silva RM. A influência do Pensamento econômico Na ideia de sustentabilidade e Suas implicações Para a Percepção e Conservação do Mundo Natural. Desenvolvimento E Meio Ambiente. 2018;46. https://doi.org/10.5380/dma.v46i0.55806.
66. Baden D, Frei R. Product returns: an opportunity to shift towards an access-based economy? Sustainability. 2021;14:410. https://doi.org/10.3390/su14010410.
67. Li X, Chen Y, Li X, Wang M, Xie W, Ding D, et al. Asbestos-Environment pollution characteristics and Health-Risk assessment in typical Asbestos-Mining area. Toxics. 2023;11:494. https://doi.org/10.3390/toxics11060494.
68. Jadhav A, Gawde N. Current asbestos exposure and future need for palliative care in India. Indian J Palliat Care. 2019;25:587. https://doi.org/10.4103/IJPC.IJPC_51_19.
69. Epting S. Affordable housing in regions with naturally occurring asbestos. Environ Justice. 2020;13:15–20. https://doi.org/10.1089/env.2019.0027.
70. Dragomir C-C, Foris T. The collaborative approach to sustainability:a model of commissioning system intervention in supporting multi-stakeholder partnerships from National to global levels. Sustainability. 2022;14:1536. https://doi.org/10.3390/su14031536.
71. Calabrese M, Suparaku S, Santovito S, Hysa X. Preventing and developmental factors of sustainability in healthcare organisations from the perspective of decision makers: an exploratory factor analysis. BMC Health Serv Res. 2023;23:797. https://doi.org/10.1186/s12913-023-09689-w.
72. Aryal A, Morley C. Mitigation of contamination and health risk: asbestos management and regulatory practices. Sustainability. 2024;16:9740. https://doi.org/10.3390/su16229740.
73. Csutora M, Zsoka A, Harangozo G. The grounded survey – an integrative mixed method for scrutinizing household energy behavior. Ecol Econ. 2021;182:106907. https://doi.org/10.1016/j.ecolecon.2020.106907.
74. Sokienah Y. Aspects of sustainability in the design elements of traditional Jordanian houses. Civil Eng Archit. 2020;8:1194–201. https://doi.org/10.13189/cea.2020.080604.
75. Cairns J. Ethics in environmental politics and sustainable use of the planet. Ethics Sci Environ Polit. 2001;1:38–45. https://doi.org/10.3354/esep001038.
76. Lee H, Sam K, Coulon F, De Gisi S, Notarnicola M, Labianca C. Recent developments and prospects of sustainable remediation treatments for major contaminants in soil: a review. Sci Total Environ. 2024;912:168769. https://doi.org/10.1016/j.scitotenv.2023.168769.
77. Lode ML, te Boveldt G, Macharis C, Coosemans T. Application of multi-actor multi-criteria analysis for transition management in energy communities. Sustainability. 2021;13:1783. https://doi.org/10.3390/su13041783.
78. Oates L. Sustainability transitions in the global south: a multi-level perspective on urban service delivery. Reg Stud Reg Sci. 2021;8:426–33. https://doi.org/10.1080/21681376.2021.1995478.
79. Marra A, Colantonio E. The institutional and socio-technical determinants of renewable energy production in the EU: implications for policy. J Ind Bus Econ. 2022;49:267–99. https://doi.org/10.1007/s40812-022-00212-6.
80. Harangozo G, Csutora M, Kocsis T. How big is big enough? Toward a sustainable future by examining alternatives to the conventional economic growth paradigm. Sustain Dev. 2018;26:172–81. https://doi.org/10.1002/sd.1728.
81. De Herde V, Baret PV, Maréchal K. Coexistence of cooperative models as structural answer to Lock-Ins in diversification pathways: the case of the Walloon dairy sector. Front Sustain Food Syst. 2020. https://doi.org/10.3389/fsufs.2020.584542.
82. Fayad MF, Elmahdy M, Sabri S, Kasem AA. Managing of City transition toward sustainable eco urbanism (Proposed paradigm). IOP Conf Ser Mater Sci Eng. 2020;974:012033. https://doi.org/10.1088/1757-899X/974/1/012033.
83. United Nations Conference on Trade and Development. Plastic pollution: the pressing case for natural and environmentally friendly substitutes to plastics. United Nations. 2023. https://doi.org/10.18356/9789210027229
84. Kobayashi T, Nakajima L. Sustainable development goals for advanced materials provided by industrial wastes and biomass sources. Curr Opin Green Sustain Chem. 2021;28:100439. https://doi.org/10.1016/j.cogsc.2020.100439.
85. Rosário A, Dias J. Sustainability and the digital transition: a literature review. Sustainability. 2022;14:4072. https://doi.org/10.3390/su14074072.
86. Peris J, Bosch M. The paradox of planning for transformation: the case of the integrated sustainable urban development strategy in València (Spain). Urban Transform. 2020;2:7. https://doi.org/10.1186/s42854-020-00011-z.
87. Frantzeskaki N. Bringing transition management to cities: building skills for transformative urban governance. Sustainability. 2022;14:650. https://doi.org/10.3390/su14020650.
88. Jedi ZAJ, Al-Jawari SM. Analysing formal transformations in sustainable urban structures in Kufa city, iraq. Environ Socio-Econ Stud. 2023;11:52–63. https://doi.org/10.2478/environ-2023-0011.
89. Aspers P, Corte U. What is qualitative in qualitative research. Qual Sociol. 2019;42:139–60. https://doi.org/10.1007/s11133-019-9413-7.
90. Kaur N, Vedel I, El Sherif R, Pluye P. Practical mixed methods strategies used to integrate qualitative and quantitative methods in community-based primary health care research. Fam Pract. 2019;36:666–71. https://doi.org/10.1093/fampra/cmz010.

91. El Bilali H. The multi-level perspective in research on sustainability transitions in agriculture and food systems. Syst Rev Agric. 2019;9:74. https://doi.org/10.3390/agriculture9040074.
92. Keller M, Sahakian M, Hirt LF. Connecting the multi-level-perspective and social practice approach for sustainable transitions. Environ Innov Soc Transit. 2022;44:14–28. https://doi.org/10.1016/j.eist.2022.05.004.
93. Herrfahrdt-Pähle E, Schlüter M, Olsson P, Folke C, Gelcich S, Pahl-Wostl C. Sustainability transformations: socio-political shocks as opportunities for governance transitions. Glob Environ Change. 2020;63:102097. https://doi.org/10.1016/j.gloenvcha.2020.102097.
94. Rabnawaz Ahmed R, Zhang X. Multi-layer value stream assessment of the reverse logistics network for inert construction waste management. Resour Conserv Recycl. 2021;170:105574. https://doi.org/10.1016/j.resconrec.2021.105574.
95. Gorissen L, Spira F, Meynaerts E, Valkering P, Frantzeskaki N. Moving towards systemic change? Investigating acceleration dynamics of urban sustainability transitions in the Belgian City of Genk. J Clean Prod. 2018;173:171–85. https://doi.org/10.1016/j.jclepro.2016.12.052.
96. Cao W, Dong L, Cheng Y, Wu L, Guo Q, Liu Y. Constructing multi-level urban clusters based on population distributions and interactions. Comput Environ Urban Syst. 2023;99:101897. https://doi.org/10.1016/j.compenvurbsys.2022.101897.
97. Gu X, An X, Liu A, Environmental regulation, corporate economic performance and Spatial technology spillover: evidence from China's heavily polluting listed corporations. Int J Environ Res Public Health. 2022;19:1131. https://doi.org/10.3390/ijerph19031131.
98. Markard J, Geels FW, Raven R. Challenges in the acceleration of sustainability transitions. Environ Res Lett. 2020;15:081001. https://doi.org/10.1088/1748-9326/ab9468.
99. Phulkerd S, Rachmi CN, Sameeha MJ, Borazon EQ, Thow A-M, Trevena H, et al. Identifying opportunities for strategic policy design to address the double burden of malnutrition through healthier retail food: protocol for South East Asia obesogenic food environment (SEAOFE) study. Int J Environ Res Public Health. 2022;19:528. https://doi.org/10.3390/ijerph19010528.
100. Ning Y, Cherian J, Sial MS, Álvarez-Otero S, Comite U, Zia-Ud-Din M. Green bond as a new determinant of sustainable green financing, energy efficiency investment, and economic growth: a global perspective. Environ Sci Pollut Res. 2022;30:61324–39. https://doi.org/10.1007/s11356-021-18454-7.
101. Marek M, Krejza Z. Sustainable building: circular economy as a key factor for cost reduction. E3S Web Conf. 2024;550:01009. https://doi.org/10.1051/e3sconf/202455001009.
102. Roque AJ, Paleologos EK, O'Kelly BC, Tang AM, Reddy KR, Vitone C, et al. Sustainable environmental geotechnics practices for a green economy. Environ Geotech. 2022;9:68–84. https://doi.org/10.1680/jenge.21.00091.
103. Küpfer C, Bastien-Masse M, Devènes J, Fivet C. Environmental and economic analysis of new construction techniques reusing existing concrete elements: two case studies. IOP Conf Ser Earth Environ Sci. 2022;1078:012013. https://doi.org/10.1088/1755-1315/1078/1/012013.
104. Bui S. Enacting transitions—the combined effect of multiple niches in whole system reconfiguration. Sustainability. 2021;13:6135. https://doi.org/10.3390/su13116135.
105. Cihan Kayacetin N, Versele A. A circular and Bio-based renovation strategy for low-income neighbourhoods. IOP Conf Ser Earth Environ Sci. 2022;1078:012080. https://doi.org/10.1088/1755-1315/1078/1/012080.
106. Larbi M, Kellett J, Palazzo E, Mehdipour A. Urban sustainability transitions in two frontrunner cities: insights from the multi-level perspective. Plann Pract Res. 2021;36:494–513. https://doi.org/10.1080/02697459.2021.1919430.
107. Kim HW, Aaron McCarty D, Lee J. Enhancing sustainable urban regeneration through smart technologies: an assessment of local urban regeneration strategic plans in Korea. Sustainability. 2020;12:6868. https://doi.org/10.3390/su12176868.
108. Larbi M, Kellett J, Palazzo E. Urban sustainability transitions in the global south: a case study of Curitiba and Accra. Urban Forum. 2022;33:223–44. https://doi.org/10.1007/s12132-021-09438-4.
109. Schmeller D, Pirisi G. Green capital East of the leitha? The chances and disadvantages of major cities in the Pannonian basin to win the European green capital award. Hung Geographical Bull. 2023;72:287–309. https://doi.org/10.15201/hungeobull.72.3.5.
110. Schmeller D, Sümeghy D. Is the rival City always greener? An analysis of the indicators for European green capital award shortlisted and applicant cities, 2010–2024. Reg Stat. 2024;14:197–228. https://doi.org/10.15196/RS140201.
111. Hallett S, Hoagland L, Toner E. Urban agriculture: environmental, economic, and social perspectives. hortic rev (Am Soc hortic Sci). Hoboken, NJ, USA: John Wiley & Sons, Inc.; 2016. pp. 65–120. https://doi.org/10.1002/9781119281269.ch2.
112. Ragheb G, El-Wahab MA, Ragheb RA. Sustainable indicators framework for strategic urban development: a case study of Abu Teeg City in Assiut, Egypt. Int J Sustainable Dev Plann. 2022;17:91–107. https://doi.org/10.18280/ijsdp.170109.
113. Trisciuoglio M, Barosio M, Ricchiardi A, Tulumen Z, Crapolicchio M, Gugliotta R. Transitional morphologies and urban forms: generation and regeneration processes—an Agenda. Sustainability. 2021;13:6233. https://doi.org/10.3390/su13116233.
114. Dinda S, Chatterjee N, Das, Ghosh S. Modelling the future vulnerability of urban green space for priority-based management and green prosperity strategy planning in kolkata, india: a PSR-based analysis using AHP-FCE and ANN-Markov model. Geocarto Int. 2022;37:6551–78. https://doi.org/10.1080/10106049.2021.1952315.
115. Xie F, Liu G, Zhuang T. A comprehensive review of urban regeneration governance for developing appropriate governance arrangements. Land (Basel). 2021;10:545. https://doi.org/10.3390/land10050545.
116. Zeng X, Yu Y, Yang S, Lv Y, Sarker MNI. Urban resilience for urban sustainability: concepts, dimensions, and perspectives. Sustainability. 2022;14:2481. https://doi.org/10.3390/su14052481.
117. Mady K, Anwar I, Abdelkareem RS. Nexus between regulatory pressure, eco-friendly product demand and sustainable competitive advantage of manufacturing small and medium-sized enterprises: the mediating role of eco-innovation. Environ Dev Sustain. 2024. https://doi.org/10.1007/s10668-024-05096-1.
118. Yu ATW, Wong I, Wu Z, Poon C-S. Strategies for effective waste reduction and management of building construction projects in highly urbanized cities—a case study of Hong Kong. Buildings. 2021;11:214. https://doi.org/10.3390/buildings11050214.
119. Stål HI. Inertia and change related to sustainability – an institutional approach. J Clean Prod. 2015;99:354–65. https://doi.org/10.1016/j.jclepro.2015.02.035.

120. Firoozi AA, Firoozi AA, Oyejobi DO, Avudaiappan S, Flores ES. Emerging trends in sustainable building materials: technological innovations, enhanced performance, and future directions. Results Eng. 2024;24:103521. https://doi.org/10.1016/j.rineng.2024.103521.
121. Ribeiro LA, Kunz M, da Silva TL, Benincá L. Initiatives for smarter and more sustainable cities in two Brazilian cities. Proceedings of the 13th International Conference on Theory and Practice of Electronic Governance, New York, NY, USA: ACM; 2020, pp. 678–81. https://doi.org/10.1145/3428502.3428602
122. Andreoni V, Ruiz Vargas V. Tracking the interlinkages across sdgs: the case of hill centered education network in Bogota. Colombia Sustain. 2020;12:7924. https://doi.org/10.3390/su12197924.
123. Geels FW, Kern F, Clark WC. System transitions research and sustainable development: challenges, progress, and prospects. Proc Natl Acad Sci. 2023. https://doi.org/10.1073/pnas.2206230120
124. Capolongo S, Rebecchi A, Dettori M, Appolloni L, Azara A, Buffoli M, et al. Healthy design and urban planning strategies, actions, and policy to achieve salutogenic cities. Int J Environ Res Public Health. 2018;15:2698. https://doi.org/10.3390/ijerph15122698.
125. He W, Li T, A research review on the transformation of old districts. IOP Conf Ser Earth Environ Sci. 2021;632:052104. https://doi.org/10.1088/1755-1315/632/5/052104.
126. Mora L, Deakin M, Zhang X, Batty M, de Jong M, Santi P, et al. Assembling sustainable smart City transitions: an interdisciplinary theoretical perspective. J Urban Technol. 2021;28:1–27. https://doi.org/10.1080/10630732.2020.1834831.
127. Schot J, Kanger L. Deep transitions: emergence, acceleration, stabilization and directionality. Res Policy. 2018;47:1045–59. https://doi.org/10.1016/j.respol.2018.03.009.
128. Geels FW, Schot J. Typology of sociotechnical transition pathways. Res Policy. 2007;36:399–417. https://doi.org/10.1016/j.respol.2007.01.003.
129. Andersson J, Hellsmark H, Sandén B. The outcomes of directionality: towards a morphology of sociotechnical systems. Environ Innov Soc Transit. 2021;40:108–31. https://doi.org/10.1016/j.eist.2021.06.008.
130. Gaziulusoy I, Veselova E, Hodson E, Berglund E, Erdogan Öztekin E, Houtbeckers E, et al. Design for sustainability transformations: a deep leverage points research agenda for the (Post-)Pandemic context. Strategic Des Res J. 2021;14:19–31. https://doi.org/10.4013/sdrj.2021.141.02.
131. Geels FW. Micro-foundations of the multi-level perspective on socio-technical transitions: developing a multi-dimensional model of agency through crossovers between social constructivism, evolutionary economics and neo-institutional theory. Technol Forecast Soc Change. 2020;152:119894. https://doi.org/10.1016/j.techfore.2019.119894.
132. Ghosh B, Kivimaa P, Ramirez M, Schot J, Torrens J. Transformative outcomes: assessing and reorienting experimentation with transformative innovation policy. Sci Public Policy. 2021;48:739–56. https://doi.org/10.1093/scipol/scab045.
133. Lanhoso F, Coelho DA. Emergence fostered by systemic analysis—Seeding innovation for sustainable development. Sustain Dev. 2021;29:768–79. https://doi.org/10.1002/sd.2173.
134. Markard J, Raven R, Truffer B. Sustainability transitions: an emerging field of research and its prospects. Res Policy. 2012;41:955–67. https://doi.org/10.1016/j.respol.2012.02.013.
135. Seddon N. Harnessing the potential of nature-based solutions for mitigating and adapting to climate change. Science (1979) 2022;376:1410–6. https://doi.org/10.1126/science.abn9668
136. D'Amato D. Sustainability narratives as transformative solution pathways: zooming in on the circular economy. Circular Econ Sustain. 2021;1:231–42. https://doi.org/10.1007/s43615-021-00008-1.
137. van der Jagt APN, Raven R, Dorst H, Runhaar H. Nature-based innovation systems. Environ Innov Soc Transit. 2020;35:202–16. https://doi.org/10.1016/j.eist.2019.09.005.
138. Katsou E, Nika C-E, Buehler D, Marić B, Megyesi B, Mino E, et al. Transformation tools enabling the implementation of nature-based solutions for creating a resourceful circular City. Blue-Green Syst. 2020;2:188–213. https://doi.org/10.2166/bgs.2020.929.
139. Kropp C, Aicher S. Material transitions in architecture and construction: social implications, appropriation, and outcomes for sustainability. TATuP - Z Für Technikfolgenabschätzung Theorie Und Praxis. 2024;33:8–56. https://doi.org/10.14512/tatup.7157.
140. Jin X, Shen GQP, Wang Q-C, Ekanayake EMAC, Fan S. Promoting construction industrialisation with policy interventions: a holistic review of published policy literature. Int J Environ Res Public Health. 2021;18:12619. https://doi.org/10.3390/ijerph182312619.
141. Jin X, Shen GQP, Ekanayake EMAC. Improving construction industrialization practices from a socio-technical system perspective: a Hong Kong case. Int J Environ Res Public Health. 2021;18:9017. https://doi.org/10.3390/ijerph18179017.
142. Timchuk OG, Nikityuk LG, Bakhtairova EA. Integration of innovations in the construction industry. IOP Conf Ser Earth Environ Sci. 2021;751:012181. https://doi.org/10.1088/1755-1315/751/1/012181.
143. Miörner J, Binz C. Towards a multi-scalar perspective on transition trajectories. Environ Innov Soc Transit. 2021;40:172–88. https://doi.org/10.1016/j.eist.2021.06.004.
144. Wittmayer JM, Avelino F, van Steenbergen F, Loorbach D. Actor roles in transition: insights from sociological perspectives. Environ Innov Soc Transit. 2017;24:45–56. https://doi.org/10.1016/j.eist.2016.10.003.
145. Avelino F, Wittmayer JM. Shifting power relations in sustainability transitions: a multi-actor perspective. J Environ Planning Policy Manag. 2016;18:628–49. https://doi.org/10.1080/1523908X.2015.1112259.
146. Wittmayer JM, Backhaus J, Avelino F, Pel B, Strasser T, Kunze I, et al. Narratives of change: how social innovation initiatives construct societal transformation. Futures. 2019;112:102433. https://doi.org/10.1016/j.futures.2019.06.005.
147. Popescu AI. Long-term city innovation trajectories and quality of urban life. Sustainability. 2020;12:10587. https://doi.org/10.3390/su122410587.
148. Štrbac S, Kašanin-Grubin M, Pezo L, Stojić N, Lončar B, Ćurčić L, et al. Green infrastructure designed through nature-based solutions for sustainable urban development. Int J Environ Res Public Health. 2023;20:1102. https://doi.org/10.3390/ijerph20021102.
149. Yılmaz DG. Nature-based solutions for climate-resilient cities: a proposal of a model for successful implementation. J Des Resil Archit Plann. 2023;4:189–203. https://doi.org/10.47818/DRArch.2023.v4i2092.
150. Calheiros CSC, Stefanakis AI. Green roofs towards circular and resilient cities. Circular Econ Sustain. 2021;1:395–411. https://doi.org/10.1007/s43615-021-00033-0.

151. Chodnekar H, Yadav P, Chaturvedi H. Review and assessment of factors associated with green building rating systems. IOP Conf Ser Earth Environ Sci. 2021;795:012033. https://doi.org/10.1088/1755-1315/795/1/012033.
152. Castellar JAC, Popartan LA, Pueyo-Ros J, Atanasova N, Langergraber G, Säumel I, et al. Nature-based solutions in the urban context: terminology, classification and scoring for urban challenges and ecosystem services. Sci Total Environ. 2021;779:146237. https://doi.org/10.1016/j.scitotenv.2021.146237.
153. Pelli P. Service innovation and sustainable construction: analyses of wood vis-à-vis other construction projects. Clean Eng Technol. 2021;2:100061. https://doi.org/10.1016/j.clet.2021.100061.
154. Huijben JCCM, Verbong GPJ, Podoynitsyna KS. Mainstreaming solar: stretching the regulatory regime through business model innovation. Environ Innov Soc Transit. 2016;20:1–15. https://doi.org/10.1016/j.eist.2015.12.002.
155. Bohnsack R. Local niches and firm responses in sustainability transitions: the case of low-emission vehicles in China. Technovation. 2018;70–71:20–32. https://doi.org/10.1016/j.technovation.2018.02.002.
156. Pesch U, Vernay A-L, van Bueren E, Pandis Iverot S. Niche entrepreneurs in urban systems integration: on the role of individuals in niche formation. Environ Plann A: Econ Space. 2017;49:1922–42. https://doi.org/10.1177/0308518X17705383.
157. Goyal N, Howlett M. Technology and instrument constituencies as agents of innovation: sustainability transitions and the governance of urban transport. Energies (Basel). 2018;11:1198. https://doi.org/10.3390/en11051198.
158. Kivimaa P, Kern F. Creative destruction or Mere niche support? Innovation policy mixes for sustainability transitions. Res Policy. 2016;45:205–17. https://doi.org/10.1016/j.respol.2015.09.008.
159. Hamburg I. Learning for sustainable development through innovation in SMEs. Adv Soc Sci Res J. 2020;7:371–81. https://doi.org/10.14738/assrj.78.8867.
160. Hu Y, Zeng Y. Achieving sustainable operations: challenges, countermeasures, and the case of unilever. SHS Web Conf. 2024;181:01036. https://doi.org/10.1051/shsconf/202418101036.
161. Sharma D, Yadav V, Singh D. Advancements and implications of product service systems in the automobile industry: a comprehensive review. RAiSE-2023. Basel Switzerland: MDPI; 2023. p. 98. https://doi.org/10.3390/engproc2023059098.
162. Luo BN, Tang Y, Chen EW, Li S, Luo D. Corporate sustainability paradox management: a systematic review and future agenda. Front Psychol 2020. https://doi.org/10.3389/fpsyg.2020.579272
163. Herrero M, Thornton PK, Mason-D'Croz D, Palmer J, Bodirsky BL, Pradhan P, et al. Articulating the effect of food systems innovation on the sustainable development goals. Lancet Planet Health. 2021;5:e50–62. https://doi.org/10.1016/S2542-5196(20)30277-1.
164. Veland S, Gram-Hanssen I, Maggs D, Lynch AH. Can the sustainable development goals Harness the means and the manner of transformation? Sustain Sci. 2022;17:637–51. https://doi.org/10.1007/s11625-021-01032-8.
165. Haack P, Rasche A. The legitimacy of sustainability standards: a paradox perspective. Organ Theory. 2021. https://doi.org/10.1177/26317877211049493.
166. Wydra S, Hüsing B, Köhler J, Schwarz A, Schirrmeister E, Voglhuber-Slavinsky A. Transition to the bioeconomy – analysis and scenarios for selected niches. J Clean Prod. 2021;294:126092. https://doi.org/10.1016/j.jclepro.2021.126092.
167. Peton H, Blanc A. Temporal dynamics of deinstitutionalization – the case of asbestos in France. Management. 2021. https://doi.org/10.37725/mgmt.v24.4570
168. Gyadu-Asiedu W, Ampadu-Asiamah A, Fokuo-Kusi A. A framework for systemic sustainable construction industry development (SSCID). Discover Sustain. 2021;2:25. https://doi.org/10.1007/s43621-021-00033-y.
169. Andersson R, Buser M. From waste to resource management? Construction and demolition waste management through the lens of institutional work. Constr Manag Econ. 2022;40:477–96. https://doi.org/10.1080/01446193.2022.2081989.
170. Börner T, Zinn M. Key challenges in the advancement and industrialization of biobased and biodegradable plastics: a value chain overarching perspective. Front Bioeng Biotechnol. 2024. https://doi.org/10.3389/fbioe.2024.1406278.
171. Sandanayake MS. Environmental impacts of construction in Building Industry—a review of knowledge advances, gaps and future directions. Knowledge. 2022;2:139–56. https://doi.org/10.3390/knowledge2010008.
172. Fei W, Opoku A, Agyekum K, Oppon JA, Ahmed V, Chen C, et al. The critical role of the construction industry in achieving the sustainable development goals (SDGs): delivering projects for the common good. Sustainability. 2021;13:9112. https://doi.org/10.3390/su13169112.
173. Adabre MA, Chan APC, Darko A, Hosseini MR. Facilitating a transition to a circular economy in construction projects: intermediate theoretical models based on the theory of planned behaviour. Building Res Inform. 2023;51:85–104. https://doi.org/10.1080/09613218.2022.2067111.
174. Roberts P, Carleton WC, Amano N, Findley DM, Hamilton R, Maezumi SY, et al. Using urban pasts to speak to urban presents in the anthropocene. Nat Cities. 2024;1:30–41. https://doi.org/10.1038/s44284-023-00014-4.
175. Ang YQ, Berzolla ZM, Letellier-Duchesne S, Reinhart CF. Carbon reduction technology pathways for existing buildings in eight cities. Nat Commun. 2023;14:1689. https://doi.org/10.1038/s41467-023-37131-6.
176. Bibri SE. The core academic and scientific disciplines underlying data-driven smart sustainable urbanism: an interdisciplinary and transdisciplinary framework. Comput Urban Sci. 2021;1:1. https://doi.org/10.1007/s43762-021-00001-2.
177. Wickramasuriya P, De Silva V. An investigation of the difficulty of implementing design for disassembly principles in the construction industry of Sri Lanka. FARU J. 2023;10:18–28. https://doi.org/10.4038/faruj.v10i1.184.
178. Muthukumaraswamy kamalakannan S, Murali SAA, Malik MARA, Saravanan NA, Vimal Kumar B, Vijayaragavan R, et al. Retrofitting of an existing structure design and techniques. IOP Conf Ser Mater Sci Eng. 2021;1197:012059. https://doi.org/10.1088/1757-899X/1197/1/012059.
179. Bertino G, Kisser J, Zeilinger J, Langergraber G, Fischer T, Österreicher D. Fundamentals of building deconstruction as a circular economy strategy for the reuse of construction materials. Appl Sci. 2021;11:939. https://doi.org/10.3390/app11030939.
180. Hong MH, Choi HM, Yoon J, Kim SC. Detoxification of asbestos and recovery of valuable metals from detoxified asbestos. Bull Korean Chem Soc. 2021;42:907–14. https://doi.org/10.1002/bkcs.12288.
181. Peponi A, Morgado P. Transition to smart and regenerative urban places (SRUP): contributions to a new conceptual framework. Land (Basel). 2020;10:2. https://doi.org/10.3390/land10010002.
182. Hölscher K, Frantzeskaki N. Perspectives on urban transformation research: transformations in, of, and by cities. Urban Transform. 2021;3:2. https://doi.org/10.1186/s42854-021-00019-z.
183. Yaskova NY, Sarchenko VI, Khirevich SA. Main principles of comprehensive approach to formation of comfortable urban environment. IOP Conf Ser Mater Sci Eng. 2021;1079:032031. https://doi.org/10.1088/1757-899X/1079/3/032031.

184. Franzblau A, Demond AH, Sayler SK, D'Arcy H, Neitzel RL. Asbestos-containing materials in abandoned residential dwellings in Detroit. Sci Total Environ. 2020;714:136580. https://doi.org/10.1016/j.scitotenv.2020.136580.
185. Wallis SL, Emmett EA, Hardy R, Casper BB, Blanchon DJ, Testa JR et al. Challenging global waste management – bioremediation to detoxify asbestos. Front Environ Sci 2020. https://doi.org/10.3389/fenvs.2020.00020
186. Sandanayake M, Bouras Y, Haigh R, Vrcelj Z. Current sustainable trends of using waste materials in concrete—a decade review. Sustainability 2020;12:9622. https://doi.org/10.3390/su12229622.
187. Teicher HM. Making embodied carbon mainstream: a framework for cities to leverage waste, equity, and preservation policy to reduce embodied emissions in buildings. J Environ Stud Sci. 2023;13:404–18. https://doi.org/10.1007/s13412-023-00836-7.
188. Henry MS, Bazilian MD, Markuson C. Just transitions: histories and futures in a post-COVID world. Energy Res Soc Sci. 2020;68:101668. https://doi.org/10.1016/j.erss.2020.101668.
189. Deng W, Alias SN, Md Rami AA, Ismail IA. Antecedents of resistance to organizational change: a systematic literature review. Int J Acad Res Bus Social Sci 2023. https://doi.org/10.6007/IJARBSS/v13-i2/16382
190. Sharafat A, Khan MS, Latif K, Tanoli WA, Park W, Seo J. BIM-GIS-based integrated framework for underground utility management system for earthwork operations. Appl Sci. 2021;11:5721. https://doi.org/10.3390/app11125721.
191. Chadee A, Ray I, Chadee X. Systemic issues influencing technical certainty in social housing programmes in a small Island developing state. Buildings. 2021;11:65. https://doi.org/10.3390/buildings11020065.
192. Duygan M, Kachi A, Oliveira TD, Rinscheid A. Introducing the Endowment-Practice-Institutions (EPI) framework for studying agency in the institutional contestation of socio-technical regimes. J Clean Prod. 2021;296:126396. https://doi.org/10.1016/j.jclepro.2021.126396.
193. Sorensen A, Brenner A-K, Cities, urban property systems, and sustainability transitions: contested processes of institutional change and the regulation of urban property development. Sustainability. 2021;13:8429. https://doi.org/10.3390/su13158429.
194. Dirisu JO, Fayomi OSI, Oyedepo SO, Udoye NE. Asbestos-free aluminium dross brake pad: a mini review. IOP Conf Ser Mater Sci Eng. 2021;1107:012034. https://doi.org/10.1088/1757-899X/1107/1/012034.
195. Neitzel RL, Sayler SK, Demond AH, d'Arcy H, Garabrant DH, Franzblau A. Measurement of asbestos emissions associated with demolition of abandoned residential dwellings. Sci Total Environ. 2020;722:137891. https://doi.org/10.1016/j.scitotenv.2020.137891.
196. Long H, Zhang Y, Ma L, Tu S. Land use transitions: progress, challenges and prospects. Land (Basel). 2021;10:903. https://doi.org/10.3390/land10090903.
197. Shao J, Li F. Multi-Function tradeoffs of land system in urbanized areas—a case study of Xi'an. China Land (Basel). 2021;10:640. https://doi.org/10.3390/land10060640.
198. Colding J, Marcus L, Barthel S. Promoting partnership between urban design and urban ecology through social-ecological resilience building. urban transition - perspectives on urban systems and environments, IntechOpen; 2023. https://doi.org/10.5772/intechopen.96642
199. Gao S, Liu S. Exploration and analysis of the aesthetic cognitive schema of contemporary Western urban landscapes. Int J Environ Res Public Health. 2021;18:5152. https://doi.org/10.3390/ijerph18105152.
200. Frangioudakis Khatib G, Collins J, Otness P, Goode J, Tomley S, Franklin P, et al. Australia's ongoing challenge of legacy asbestos in the built environment: a review of contemporary asbestos exposure risks. Sustainability. 2023;15:12071. https://doi.org/10.3390/su151512071.
201. Arachi D, Furuya S, David A, Mangwiro A, Chimed-Ochir O, Lee K, et al. Development of the National asbestos profile to eliminate asbestos-related diseases in 195 countries. Int J Environ Res Public Health. 2021;18:1804. https://doi.org/10.3390/ijerph18041804.
202. Aboufazeli S, Jahani A, Farahpour M. A method for aesthetic quality modelling of the form of plants and water in the urban parks landscapes: an artificial neural network approach. MethodsX. 2021;8:101489. https://doi.org/10.1016/j.mex.2021.101489.
203. Billi M, Zurbriggen C, Urquiza A, Allendes A. Transition in action: toward a social theory of the governance of transitions. Front Sociol. 2024. https://doi.org/10.3389/fsoc.2024.1206050.
204. Kanger L, Sovacool BK, Noorkõiv M. Six policy intervention points for sustainability transitions: a conceptual framework and a systematic literature review. Res Policy. 2020;49:104072. https://doi.org/10.1016/j.respol.2020.104072.
205. McDonald PL, Foley TJ, Verheij R, Braithwaite J, Rubin J, Harwood K et al. Data to knowledge to improvement: creating the learning health system. BMJ 2024;384:e076175. https://doi.org/10.1136/bmj-2023-076175
206. Ehnert F, Egermann M, Betsch A. The role of niche and regime intermediaries in Building partnerships for urban transitions towards sustainability. J Environ Planning Policy Manag. 2022;24:137–59. https://doi.org/10.1080/1523908X.2021.1981266.
207. Simoens MC, Leipold S. Trading radical for incremental change: the politics of a circular economy transition in the German packaging sector. J Environ Planning Policy Manag. 2021;23:822–36. https://doi.org/10.1080/1523908X.2021.1931063.
208. Iluyomade TD, Okwandu AC. Innovative materials in sustainable construction: a review. Int J Sci Res Archive. 2024;12:2435–47. https://doi.org/10.30574/ijsra.2024.12.1.1048.
209. Engler J-O, Abson DJ, von Wehrden H. Navigating cognition biases in the search of sustainability. Ambio. 2019;48:605–18. https://doi.org/10.1007/s13280-018-1100-5.
210. Parida V, Wincent J. Why and how to compete through sustainability: a review and outline of trends influencing firm and network-level transformation. Int Entrep Manag J. 2019;15:1–19. https://doi.org/10.1007/s11365-019-00558-9.
211. Goforth T, Nock D. Inequality in energy transitions: subnational air pollution disparities resulting from national decarbonization strategies 2021. https://doi.org/10.21203/rs.3.rs-945021/v1

Publisher's note

Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.

International Journal of
Environmental Research and Public Health

MDPI

Review

Examining the Environmental Ramifications of Asbestos Fiber Movement Through the Water–Soil Continuum: A Review

Gergely Zoltán Macher [1,2,*], András Torma [1] and Dóra Beke [3]

[1] Department of Applied Sustainability, Albert Kázmér Mosonmagyaróvár Faculty of Agricultural and Food Sciences, Széchenyi István University, 9026 Győr, Hungary; torma@sze.hu

[2] Wittmann Antal Crop-, Animal- and Food Sciences Multidisciplinary Doctoral School, Albert Kázmér Mosonmagyaróvár Faculty of Agricultural and Food Sciences, Széchenyi István University, 9200 Mosonmagyaróvár, Hungary

[3] Department of Plant Sciences, Albert Kázmér Mosonmagyaróvár Faculty of Agricultural and Food Sciences, Széchenyi István University, 9200 Mosonmagyaróvár, Hungary; beke.dora@sze.hu

* Correspondence: macher.gergely.zoltan@sze.hu

Abstract: The environmental pollution potential of asbestos products is a worldwide health issue, but their dissemination through the water–soil continuum is often an overlooked aspect. Similarly, the behavior of asbestos fibers released from the products is still not fully understood, although our knowledge is based on studies concerning their mineralogical characteristics, health effects, and waste disposal. It has been claimed and contradicted that asbestos harm is only found in air and humans. Asbestos fibers are found not only in industrial settings but also through the industrial use of asbestos cement products, which has contributed to asbestos emissions and its movement in water and soil. Asbestos fibers are diverse in their physicochemical properties, and this diversity has a significant influence on their behavior in the environment. Recent research has confirmed that asbestos can be transported by water and spread to other parts of the environment. However, the mechanisms underlying this, such as the settling of fibers, their attachment to soil particles, or their movement in groundwater, as well as the environmental and health implications, require further investigation. This paper examines the process and impact of asbestos contamination in the interconnected water, soil, and plant environmental sectors, providing a systematic review of the latest literature.

Keywords: asbestos; chrysotile; asbestos mobility; asbestos toxicity; soil pollution; water pollution

Academic Editor: William A. Toscano

Received: 29 January 2025
Revised: 24 March 2025
Accepted: 25 March 2025
Published: 26 March 2025

Citation: Macher, G.Z.; Torma, A.; Beke, D. Examining the Environmental Ramifications of Asbestos Fiber Movement Through the Water–Soil Continuum: A Review. *Int. J. Environ. Res. Public Health* **2025**, *22*, 505. https://doi.org/10.3390/ijerph22040505

1. Introduction

Asbestos is a naturally occurring mineral fiber with a complex and often contentious history [1,2]. While it was widely used for its heat-resistant, durable nature and insulating properties throughout the 20th century in various industrial and commercial applications [3,4], its environmental mobility remains a critical concern. Asbestos can be transported through soil and water, affecting a wide range of ecosystems and human health [5]. The widespread presence of asbestos in various environmental media, including construction materials and other industrial products, has led to significant occupational and environmental exposures [6], contributing to far-reaching health consequences. Asbestos is a collective term referring to specific naturally occurring fibrous silicate minerals [7,8]. Specifically, the term encompasses two types of silicates with a fibrous structure [9,10]: serpentines and amphiboles. Six varieties of minerals are considered fibrous asbestos.

These are chrysotile, a member of the serpentine group [11], and actinolite, crocidolite, anthophyllite, grunerite, and tremolite, all members of the amphibole group [12,13]. All these minerals have different physical and chemical properties, which vary with their hazard and the manner of formation. The World Health Organization defines a regular asbestos fiber as a particle with a length greater than 5 μm, a diameter less than 3 μm [14], and an aspect ratio exceeding 3:1 [7,15]. Given their size, these minuscule fibers can easily be inhaled and can permeate through regions of the respiratory system (such as the alveoli of the lungs) [16]. When inhaled, the fibers can become lodged in lung tissue, resulting in a number of serious health problems over time [17]. Long-term exposure to asbestos has been shown to lead to asbestosis, lung cancer, and mesothelioma, a rare but aggressive cancer that develops in the lining of the lungs, chest, or abdomen [18]. Asbestos exposure is linked to serious health risks, particularly through airborne pathways, leading to strict regulations and bans in many countries. However, asbestos also remains a threat in older buildings and consumer products, where it can be transported through soil and water, posing additional environmental and public health risks. This exposure route, often overlooked, must be addressed in environmental regulations and health assessments [19,20]. Despite asbestos having been a widely utilized material, its legacy persists, posing an ongoing significant public health concern that necessitates sustained efforts to manage and remediate its presence in the environment. The widespread use of asbestos has had a significant impact on public health, with a well-established link between asbestos exposure and the development of various respiratory diseases, including asbestosis, lung cancer, and malignant mesothelioma [21,22]. Asbestos exposure can occur through both occupational and environmental sources, with construction workers, shipbuilding personnel, and individuals living in proximity to asbestos-containing materials (ACM) being at particularly high risk [23–25]. The health risks associated with asbestos exposure have led to the implementation of various regulatory measures aimed at mitigating its impact [26].

In the United States, the Occupational Safety and Health Administration (OSHA) has established permissible exposure limits for asbestos in the workplace [16], while the Environmental Protection Agency (EPA) regulates the removal and disposal of asbestos-containing materials. However, there are significant regulatory gaps when it comes to the presence of asbestos in soil and water, and current environmental regulations fail to address the risks associated with non-airborne asbestos exposure. These gaps highlight the need for more comprehensive environmental policies and research on asbestos contamination in these media [10,27,28].

Chrysotile, commonly referred to as white asbestos, is the most extensively utilized form of asbestos and is considered carcinogenic [29,30]. This type of asbestos exhibits distinct characteristics compared to other asbestos varieties, with a structural composition comprising octahedral magnesium and tetrahedral silicon layers, and an outermost magnesium hydroxide layer [31–33]. In the pH range found in soil, the outer magnesium layer dissolves rapidly, followed by the dissolution of the silicon layers, which becomes the rate-limiting step in this process [31,34,35]. Chrysotile demonstrates high solubility in moderately acidic pH conditions [31,36,37] but incongruent dissolution patterns in more acidic and neutral environments [34,35]. This indicates that the rate of white asbestos dissolution is largely influenced by the acidity of the surrounding medium [35,38]. Additionally, certain asbestos minerals, including chrysotile, exhibit a net positive surface charge in water around neutral pH, attributable to their outer brucite-like layer [39]. This surface charge behavior impacts the interaction of chrysotile fibers with environmental and biological systems, influencing their mobility and bioavailability [40–42]. However, crucial questions remain regarding the fate of chrysotile in soil, particularly concerning its long-term stability and potential for leaching into groundwater [33,43].

The global usage of asbestos experienced a significant rise between the 1940s and 1980s until it reached its peak [44,45]. This surge in usage was driven by the material's favorable properties and widespread industrial applications [46,47]. However, this trend shifted when reports emerged of adverse health consequences [48] related to environmental exposure to asbestos in various European countries, including the Netherlands, Poland, and Italy [49–51]. These reports highlighted the serious health risks posed by asbestos, particularly in occupational settings where exposure levels were high. The detrimental effects of asbestos and asbestos-containing products were not fully recognized until the latter half of the 20th century, despite earlier warnings from researchers [52–54]. This delayed recognition of the health hazards associated with asbestos led to widespread occupational diseases [20], including asbestosis, lung cancer, and mesothelioma. In response to mounting evidence of its carcinogenicity, the International Agency for Research on Cancer (IARC) classified all forms of asbestos as Group 1 carcinogens in 1977 [55,56]. Despite these findings, the 1980s marked the highest point of asbestos-containing product utilization in numerous nations, such as Hungary, where asbestos was extensively used in construction and manufacturing [57]. The primary risks associated with asbestos are attributed to the inhalation of its fibers [58].

Lifetime cancer risk estimates are primarily based on the concentrations of asbestos contamination and exposure [5,59,60]. Consequently, air has been the primary medium of investigation for asbestos, with the unit of concentration being the number of asbestos fibers per volume of air [8]. Monitoring airborne asbestos levels remains a critical aspect of occupational health and safety regulations aimed at minimizing exposure and protecting public health [8].

2. Asbestos Cement Products and Their Environmental Risks

Over 90% of the global asbestos usage has been directed towards the production of asbestos cement sheets and pipes, which continues in certain regions today [61,62]. Asbestos cement is a composite material consisting of asbestos fibers embedded within a cement matrix, comprising cement binder and water reaction products [63]. Asbestos fibers have been extensively utilized as reinforcement in cement-asbestos composites, enhancing their tensile strength and thermal resistance [64–66]. Asbestos cement products represent high-binder content applications, commonly in the form of flat sheets, corrugated sheets, and pipes, where asbestos accounts for up to 8–10% of the composition, combined with 90–92% cement binder [7]. As noted, the same physicochemical properties that made asbestos useful in various engineering applications are also responsible for its environmental persistence and remediation challenges [67]. For instance, asbestos is resistant to the chemical, biological, and thermal treatments commonly employed for organic pollutants [68]. For these asbestos-containing products, a well-documented alteration process is fragmentation, cracking, and spalling, which can occur due to exogenous or anthropogenic factors [69,70]. This can lead to the release of asbestos fibers into the environment, posing significant health risks [71]. Given the hazards associated with asbestos exposure, the demolition and reclamation of asbestos-containing materials from construction and demolition waste [72] has become a major concern [71]. Several studies have explored the potential for recycling and repurposing asbestos-containing waste in a safe and environmentally responsible manner [71,73,74].

According to Tóth and Weiszburg [7], higher binder content significantly reduces the likelihood of asbestos fibers being released into the air, although the rate of spalling increases when the product is damaged. This protective effect of higher binder content is crucial in mitigating airborne asbestos exposure, particularly in construction materials where wear and tear are common [75]. However, elevated asbestos concentrations in the air surrounding dilapidated sheet-roofed buildings have been found to be detrimental to human health [76]. The degradation of these materials over time leads to the release of asbestos fibers, posing significant health risks to nearby residents and workers [77–79]. Asbestos cement materials, commonly used in various building applications, can release hazardous asbestos fibers into surrounding environments [80]. Climatic factors, including severe weather events and increased precipitation associated with climate change, exacerbate this issue by accelerating the deterioration of asbestos-containing materials [81]. These environmental conditions, including precipitation, water movement, evaporation, erosion, ablation, and solar radiation, as well as human activities like vehicle transport, land displacement, earthmoving, and agriculture, can facilitate the mobilization and dissemination of asbestos fibers. This contamination can impact the quality of water, air, and soil, and specifically, these processes play a crucial role in the movement of asbestos fibers through the soil profile and water systems. For instance, erosion caused by rainfall or wind can dislodge asbestos fibers from the surface of contaminated materials, such as asbestos-cement products, and transport them across the land [82]. Precipitation acts as a natural transport mechanism, washing the fibers into water bodies or deeper soil layers, especially during heavy rainfalls. Water transport further enhances the dispersal, with rivers and streams acting as conduits, carrying asbestos particles to new locations [8]. Evaporation can lead to the concentration of fibers in certain areas, and solar radiation can alter the physical properties of asbestos fibers, making them more prone to dispersal [83]. Human activities, such as vehicle transportation, can stir up dust containing asbestos fibers, facilitating their dissemination in the environment. Similarly, land displacement and earthmoving associated with construction, mining, or road building can release asbestos from buried materials, causing it to become airborne or mobilized in the soil [84]. Agricultural practices, including irrigation, plowing, and fertilizer application, can disturb contaminated soil, enabling asbestos fibers to spread more extensively. Consequently, these factors contribute to the movement of asbestos fibers into the air, water, and soil, where they can contaminate water supplies, infiltrate food chains, and pose long-term health risks to humans and wildlife [85,86]. Zhang et al. [87] identify one of the key challenges with the proposed solution, namely that the removal and disposal of asbestos cement sheets is a long-term undertaking, given the widespread use of these materials in buildings and the high costs associated with their proper removal and disposal [88]. Asbestos-containing products are typically landfilled, where they are buried in soil to prevent erosion and scattering [89]. However, wear and damage during transport or storage can lead to the formation of small asbestos fibers, which have a greater potential for water transport than bulk asbestos and can also infiltrate groundwater [8].

Furthermore, the public's improper disposal of these hazardous asbestos cement products exacerbates the issue, as such actions can result in the unchecked erosion and release of asbestos fibers into the soil [23,25,67]. Asbestos emission from building materials, particularly in older structures, is a significant concern due to the potential health risks posed by asbestos fibers [8,20]. Several studies have found that the weathering and deterioration of asbestos-containing materials, such as cement sheets and roofing felts, can lead to the release of asbestos fibers into the air [23]. The risk is particularly heightened in urban areas with a prevalence of older buildings and infrastructure, where the fibers can become airborne and increase the likelihood of exposure among the general population [90]. The natural release of asbestos from shale can also indicate the amount of asbestos released due to the aging and weathering of the shale surface [87]. The release of asbestos fibers from naturally occurring sources adds another layer of complexity to managing environmental contamination. In a study conducted by Cely-García [91] in Colombia, activity-based sampling was used to evaluate the release of asbestos fibers from various contaminated soil samples, revealing a relatively low asbestos risk. This finding suggests that, under certain conditions, the risk of asbestos fiber release from soil may be manageable [92]. However, contrasting findings from Bornemann and Hildebrandt [81] reported that older sheet roofs or those with damaged surfaces can emit asbestos fibers into the air at average annual concentrations of several grams [93]. This substantial release poses significant health risks, particularly in urban areas with older infrastructure [94]. Additionally [76], previously documented that the surface of asbestos cement sheets undergoes weathering-induced corrosion at a rate of approximately 0.01–0.024 mm per year [76,87]. This gradual degradation process continuously releases asbestos fibers into the environment, contributing to long-term contamination and health risks [20]. The ongoing weathering and deterioration of asbestos-containing materials, combined with improper disposal practices, highlight the critical need for effective asbestos management and remediation strategies to protect public health and the environment [95,96].

Asbestos fibers are known to be highly durable and can remain suspended in the air for long periods, posing a persistent threat to respiratory health [97]. The latency period for these diseases can be several decades, meaning that the health effects of asbestos exposure may not become apparent until many years after the initial exposure [98]. This underscores the importance of proactive measures to minimize asbestos release and exposure. Moreover, the challenge of managing asbestos contamination is compounded by the variability in the types of asbestos fibers and their differing toxicological properties [8,26]. Chrysotile, amosite, and crocidolite are among the most common types, each with unique characteristics and health risks [38]. Effective asbestos management requires a comprehensive understanding of these differences and the implementation of tailored strategies to address the specific risks associated with each type [99,100]. The emission of asbestos fibers from both building materials and natural sources represents a multifaceted environmental health challenge [96]. Urban areas with aging infrastructure are particularly vulnerable, and the continuous weathering of asbestos-containing materials further exacerbates the risk [20]. Studies have shown varying levels of risk associated with asbestos release from different sources, highlighting the need for ongoing research and adaptive management strategies [101]. To protect public health, it is imperative to develop and enforce stringent regulations on asbestos use, ensure proper disposal practices, and invest in the remediation of contaminated sites [26]. Public awareness campaigns and education on the dangers of asbestos exposure are also crucial components of a comprehensive asbestos management plan [26].

3. Soil Contamination by Asbestos Complexes and Its Risks

Asbestos, a naturally occurring mineral, can become a significant environmental pollutant when released into the soil [102]. The terrestrial environment can be contaminated with asbestos through natural [103–105] or human-induced processes [33]. Naturally occurring asbestos in rocks is transferred to the soil through geological phenomena [106], resulting in the soil inheriting the mineralogical and geochemical properties of the underlying bedrock [69,107]. Natural asbestos deposits are discontinuous, highly tectonized, and fragile and can break down under minimal mechanical stress [108], transforming into incoherent material that can be easily incorporated into the soil. Asbestos fibers can be easily incorporated into soil through natural weathering processes or human activities, such as the degradation of asbestos-containing construction materials [82,102,104].

Asbestos may enter soils and sediments through natural processes but primarily through release from anthropogenic sources [109,110]. Nonetheless, naturally occurring asbestos fibers in rocks and soils can also pose an environmental risk by becoming airborne. Previously, soil was considered a layer that effectively insulated asbestos in environmental waste [106,111]. Recent studies suggest that organic acids commonly found in soils can enhance the leaching of small asbestos fibers [76]. Although the transport rate of these fibers is not rapid, they can become a source of respirable dust in the air due to various mechanical factors [112,113]. Dry conditions, strong winds, and mechanical disturbances—such as excavation, construction activities, or heavy machinery movement—can further contribute to fiber resuspension, increasing the likelihood of airborne asbestos exposure [114]. Additionally, seasonal variations in soil moisture levels may influence fiber stability, with prolonged droughts exacerbating dust generation and fiber dispersal [115]. Without proper mitigation measures, these processes can lead to long-term contamination risks, affecting both agricultural workers and nearby populations.

Once present in the soil, asbestos can alter its physical, chemical, and biological properties in multiple ways, thereby influencing soil quality and ecosystem health [116]. Physically, asbestos fibers can modify soil texture and porosity, potentially affecting water retention and infiltration rates. These changes may lead to altered soil aeration, influencing root growth and microbial activity [89]. Chemically, asbestos minerals can interact with soil components, affecting pH levels and the availability of essential nutrients. Some studies suggest that asbestos-containing minerals can release magnesium, iron, and silica into the soil, which may alter nutrient cycling and plant uptake dynamics [117–119]. Biologically, asbestos contamination can impact microbial communities by disrupting their composition and metabolic functions. Certain fungi and bacteria capable of interacting with silicate minerals may play a role in asbestos weathering, but the overall ecological consequences remain poorly understood [120–122]. The mechanisms of asbestos entry into the soil profile include atmospheric deposition, industrial waste disposal, and weathering of asbestos-containing rocks. Once in the soil, fibers can adsorb onto mineral surfaces or organic matter, influencing their mobility. While asbestos is relatively immobile under neutral to alkaline conditions, acidic environments, and organic acids can facilitate fiber desorption and leaching. Additionally, surface runoff and erosion can transport asbestos particles into aquatic systems, where they may pose further environmental and health risks [100].

The mobility of asbestos fibers in soil is influenced by various factors, including fiber characteristics, soil properties, and environmental conditions [82,102,123]. Asbestos concentrations in soil can lead to partial displacement, including re-entrainment into the air, as well as changes in the electrical charge of asbestos fibers that can affect the organic soil layer and facilitate fiber migration [124]. In natural settings, asbestos can occur in large deposits, with chrysotile being the most commonly found form, as its fibers are present in serpentine rock formations [106]. The primary anthropogenic sources are asbestos cement sheets, tiles, and panels used in construction, which predominantly contain chrysotile [67]. Environmental concentrations can vary widely due to human activities, land use, and natural factors, as the aging of asbestos cement products is the main cause of fiber release in urban environments, depending on the source type, degradation level, and treatment/manipulation [8,89,106].

The limited research on the mobility of asbestos fibers in porous media restrains our understanding of the environmental factors influencing the transport of fibers in soil [76]. The previously established relationship between fiber concentration and measured soil concentration, first demonstrated by Addison et al. [125] and expanded upon by the RIVM National Institute for Public Health and the Environment, can be leveraged to quantify asbestos levels in soil [60,112]. However, the risks of asbestos in soil remain uncertain, necessitating further research to evaluate the relationship between fiber release and health impacts [126,127]. The mobility of asbestos fibers in soil can be estimated based on studies of the transport of mineral colloids in water [128].

The distinctive physical and surface chemical characteristics of asbestos fibers, including their high aspect ratio and lengths of up to 100 μm [129], significantly influence their mobility, retention, and persistence in soil environments [67,112,130]. Unlike spherical colloids, asbestos fibers exhibit complex transport dynamics due to their elongated structure, which affects their interaction with soil particles, water flow pathways, and biological components. According to [89], the transport and removal of colloids in soil are influenced by both physical and chemical factors. Among the physical factors, particle size [131], shape [132], and soil pore size distribution [133] determine the extent to which asbestos fibers migrate or become immobilized. Larger fibers, particularly those exceeding the typical pore sizes in fine-textured soils such as clay and silt, are more likely to be trapped by mechanical filtration or straining. Conversely, smaller fibers may be transported more easily through macropores, preferential flow pathways, and fractures in structured soils. The fibrous nature of asbestos also allows for entanglement with organic matter and root systems, which can further restrict mobility.

Chemical factors significantly modulate asbestos fiber stability and movement within the soil matrix. The pH of the soil [134] plays a crucial role in determining the surface charge of asbestos minerals, which in turn affects their adhesion to soil particles. Under acidic conditions, the positive surface charge of asbestos fibers may enhance their interaction with negatively charged clay minerals and organic matter, leading to increased retention. In contrast, alkaline conditions can promote dispersion, enhancing mobility in groundwater and surface runoff. The ionic strength [135] of the soil solution further influences fiber behavior. High ionic strength environments, often associated with saline or contaminated soils, can lead to particle aggregation and flocculation, reducing the likelihood of fiber transport. Conversely, lower ionic strength may promote fiber dispersion, increasing the potential for groundwater contamination. The presence of phosphates and other solutes [136] can also impact asbestos mobility by altering electrostatic interactions, either stabilizing fibers in suspension or enhancing their adsorption onto mineral surfaces [113]. In addition to these factors, soil organic matter content plays a dual role in asbestos transport. Organic compounds, including humic and fulvic acids [137], can bind to asbestos surfaces, enhancing their retention in the soil profile. However, under certain conditions, organic ligands may also facilitate fiber mobilization by altering surface charge characteristics and promoting desorption [138]. Furthermore, microbial activity within the soil can influence asbestos weathering processes, potentially leading to fiber fragmentation or dissolution, which may modify their environmental impact [89]. Colloids are transported through the soil by infiltrating water and deposited on soil particles through sedimentation, absorption/adsorption, and diffusion, as predicted by colloid filtration theory [89,139]. Previous research on asbestos in soils has primarily focused on former asbestos mining areas [113], but there is also a need to assess other impacted regions and determine their level of exposure.

Moreover, agricultural activities can cause asbestos particles in the soil to decompose and become airborne [113,140,141], rendering the agricultural sector particularly vulnerable despite the paucity of studies in this domain [89,125,142]. The disturbance of soil through plowing, irrigation, and other farming practices can enhance the risk of asbestos fiber release, posing health risks to farmers and surrounding communities [8]. Additionally, irrigation can facilitate fiber transport through water movement, while plowing and soil tillage may increase fiber resuspension and bioavailability [143]. Fertilizer application, especially when combined with organic amendments, could further influence fiber interactions in the soil matrix, potentially altering their persistence and uptake by crops [144].

Additionally, remediation of asbestos-contaminated soils presents unique challenges. Traditional soil remediation techniques may not be effective for asbestos fibers due to their size, durability, and tendency to become airborne [67]. Innovative approaches such as phytoremediation, where certain plants are used to stabilize and contain asbestos fibers, and in situ stabilization, where chemical agents are added to soil to bind fibers and reduce mobility, are being explored [145]. These methods aim to mitigate the release of asbestos fibers and reduce the associated health risks [120].

The presence of asbestos in soil is a complex issue with significant implications for environmental and public health [106]. The mobility of asbestos fibers in soil, influenced by various physical and chemical factors, underscores the need for comprehensive research and targeted remediation strategies [89]. Understanding the interaction of asbestos fibers with soil and developing effective management practices are crucial steps in mitigating the risks associated with asbestos contamination [26]. Further studies are essential to evaluate the long-term impact of asbestos in soil and to develop innovative solutions to protect human health and the environment [100].

4. Water Contamination by Asbestos Complexes and Its Risks

In stark contrast, the potential dangers of waterborne exposure to ingested asbestos have received considerably less scrutiny despite studies indicating an elevated risk of stomach cancer from such exposure [146]. This discrepancy in research focus is notable, given the substantial health implications. Asbestos-containing materials primarily pose a hazard when their fibers are released into the air, creating an inhalation risk. However, the pathogenicity of ingested asbestos fibers remains a critical and ongoing area of investigation. Studies have explored various aspects of this issue, including early examinations by [147–149] and more recent work by [150]. Despite these efforts, the body of literature remains less extensive compared to airborne asbestos risks [57]. Furthermore, very few studies have delved into the potential risks arising from airborne asbestos and asbestos-forming minerals that are released during water mobilization or evaporation processes [151]. This gap in research is significant, as it suggests that there could be overlooked pathways for asbestos exposure [152]. Previous research had suggested that environmental exposure to asbestos fibers through water or soil was negligible because the fibers were assumed to adhere to soil particles and be filtered out during infiltration [153]. However, this assumption has been challenged by more recent studies reporting the presence of asbestos fibers in deep underground aquifers [105,154,155]. These findings indicate the emergence of an alternative transport pathway for asbestos exposure through shallow groundwater systems [5,89,156].

The transport of asbestos fibers from soil to surface water is highly dependent on a range of site-specific factors, including precipitation patterns, soil and rock degradability, slope morphology, vegetation cover, and the degree of anthropogenic influence [157]. This complex interplay of factors means that understanding the environmental conditions that enhance the mobility of asbestos fibers between contaminated soil and groundwater is of utmost importance [89]. This knowledge is crucial for developing effective strategies to mitigate the health risks associated with both airborne and waterborne asbestos exposure, ensuring better public health outcomes in areas affected by asbestos contamination [26]. Asbestos fibers can infiltrate groundwater sources through the natural weathering of asbestos-containing rocks and improper disposal of asbestos-containing waste [158]. Previous research [157] has concentrated primarily on surface water contamination by asbestos, while groundwater studies remain in the experimental phase. In general, asbestos pollution in water is attributable to both anthropogenic and natural factors [8]. Studies have shown that chrysotile asbestos can be present in groundwater near mining sites, contributing to potential health risks. The persistence of asbestos fibers in groundwater is largely due to their resistance to chemical and biological degradation [159]. Groundwater contamination by asbestos is a concern near industrial sites where asbestos was historically used or disposed of. Asbestos fibers in groundwater can be transported over long distances, posing risks to distant populations [8,120].

Of particular note is the fact that the impact of asbestos fibers on aquatic ecosystems is largely unknown, and very few studies have been conducted to elucidate the various mechanisms of action [160]. In countries where industrial asbestos use is prohibited or strictly regulated, anthropogenic sources include the disposal of asbestos-contaminated water from mines and quarries [8]. Another anthropogenic source of emissions is the erosion of asbestos cement pipes used in drinking water and wastewater networks, which can also release asbestos fibers [161]. Additionally, water flowing through improperly disposed asbestos-contaminated waste can become a source of pollution [89]. Although extensive research has been conducted on the health hazards [162] associated with the inhalation of airborne asbestos and analogous asbestos-like minerals [151], there remains a paucity of information regarding the potential health risks posed by asbestos fibers suspended in water, which represents one of the most concerning mobilization pathways in the environment [25,163]. Caramuscio et al. [164] investigated the presence of asbestos contamination in groundwater and surface water near an asbestos mine. Their findings revealed that the asbestos content of stream water near the mine ranged from 1.00 to 3.60 mg/L, while groundwater levels were between 1.00 and 4.10 mg/L. Previous studies [155,165,166], as reported [167], have demonstrated that asbestos pollutants from mining activities can lead to metal contamination and increased asbestos levels in surface waters and sediments [168–170], particularly in areas surrounding active and inactive asbestos mines [155,171,172]. The asbestos fibers detected were predominantly short, measuring less than 5 μm in length, but the potential for asbestos dispersion from stream water to air was not examined [83,173,174]. Despite the recognized health risks, European Union regulations on water quality still do not specify limit values or guideline values for asbestos in drinking water [175,176], nor for natural surface and groundwater. This regulatory gap presents significant challenges in assessing and mitigating exposure risks, particularly in regions where aging asbestos cement pipelines, industrial effluents, or natural geological sources contribute to fiber release [8,177].

The absence of harmonized standards for asbestos monitoring in water bodies impedes the development of risk-based management strategies and hinders cross-border comparability of exposure data. Moreover, the lack of legally binding thresholds limits enforcement mechanisms, potentially delaying preventive interventions even in areas with documented contamination. Given the growing body of evidence on the potential carcinogenicity of ingested asbestos fibers, the absence of EU-wide regulations may result in insufficient protection of public health, particularly for vulnerable populations relying on untreated or minimally treated water sources [178]. Without targeted epidemiological studies and toxicological assessments, the long-term implications of chronic low-dose asbestos ingestion remain uncertain, reinforcing the need for regulatory advancements and precautionary policy measures at both national and supranational levels [179,180].

The World Health Organization (WHO) has yet to establish a safe concentration level for asbestos in water, a notable omission given the organization's global influence on health standards [181]. In contrast, the United States Environmental Protection Agency (US-EPA) has set a maximum contaminant level for asbestos in drinking water at 7×10^6 asbestos fibers per liter, primarily addressing fibers longer than 10 μm [182,183]. This standard reflects a more proactive approach to managing the risks associated with asbestos in water. Several studies have focused on monitoring and determining the background levels of asbestos in drinking water, with some of the highest values measured exceeding 10^7 asbestos fibers per liter in the USA [184–186] and 10^8 asbestos fibers per liter in Canada [187]. These findings underscore the variability and potential for high concentrations of asbestos in water sources. Further complicating the issue [186] demonstrated a correlation between airborne and waterborne asbestos in buildings where asbestos [chrysotile and amphibole] was detected in tap water [8]. They reported that 2.4×10^7 asbestos fibers per liter of water could result in air concentrations as high as 120 asbestos fibers per liter. This correlation suggests that asbestos fibers in water can become airborne, thereby increasing the risk of inhalation and the associated health impacts [151]. Consequently, potential emergency situations should not be overlooked, as the concentration of asbestos in surface waters and drinking water can dramatically increase during natural disasters [188]. Natural disasters can disrupt and mobilize asbestos fibers, leading to sudden spikes in contamination levels that pose immediate health risks [189]. From an agricultural and water management perspective, special attention should be given to the use of contaminated groundwater for irrigation [113] and indoor water usage [190]. Using contaminated water for irrigation can introduce asbestos fibers into the food chain, while indoor water use can increase the risk of asbestos inhalation as fibers become aerosolized during household activities [89]. Despite these concerns, there is a lack of research on the extent to which asbestos fibers bioaccumulate in plants and how they are transferred through trophic levels. The mechanisms of fiber uptake by plant roots, potential translocation within plant tissues, and the implications for human and animal consumption remain largely unexplored. Additionally, the physicochemical properties of asbestos may influence its interaction with soil microbiota and plant exudates, potentially altering bioavailability and accumulation patterns [42,121]. These knowledge gaps highlight the urgent need for targeted studies on asbestos behavior in agricultural systems to better inform risk assessments and regulatory frameworks. These scenarios elevate human exposure levels and underscore the need for comprehensive monitoring and regulation to mitigate the risks associated with asbestos in water. Addressing these issues requires coordinated efforts at the policy, research, and community levels to ensure that water quality standards are robust and protective of public health [191,192].

5. Conclusions

In our research, we have meticulously elucidated that the issues stemming from asbestos and various asbestos-containing products have long constituted a critical environmental domain, one that fundamentally transcends the traditional boundaries of waste management concerns. This study underscores the importance of re-evaluating our understanding of asbestos-related environmental hazards, particularly in light of recent developments in international scholarly discourse. The prevailing assumption that asbestos fibers remain immobile in soil and that water, as a transmission medium, has only a minimal role in the mobilization of these fibers has been robustly challenged by contemporary research findings. Table 1 presents a comprehensive critical analysis of the impacts on soil and water, as well as the identified gaps in the current body of research. These findings suggest a significantly higher potential for asbestos fiber dispersion, thereby elevating the risks of human exposure and associated health hazards. Despite the ongoing experimental nature of research in this field, a significant gap persists in the availability of standardized analytical methods and procedures. These methods are crucial for providing concrete guidelines for assessing the contamination levels in various water and soil types. The absence of such standardized protocols hampers the ability to accurately quantify and address the extent of asbestos contamination in different environmental media.

Table 1. Impacts on the water–soil system and associated research gaps.

Aspect	Effects on Water	Effects on Soil	Gaps and Research Needs
Physical characteristics	Asbestos fibers can remain suspended in water and settle slowly [8].	Persist in soil for extended periods without natural degradation [82].	Lack of precise data on sedimentation and transport dynamics.
Chemical stability	Type-dependent, but generally does not degrade in water [83].	Stable structure, slow weathering [193].	Limited data on long-term transformations and interactions.
Contaminant adsorption	Can adsorb heavy metals and organic pollutants [194].	May bind toxic substances, making them bioavailable to plants [195].	Unclear which contaminants can accumulate.
Biological effects	Not a food source but can enter the food chain [86].	Potentially toxic to microorganisms and plants [196].	Insufficient research on ecosystem-level impacts.
Drinking water contamination	Asbestos fibers can be present in drinking water [8].	Can leach from soil into groundwater [8].	Unclear relationship between exposure and health risks.
Toxicity	Potentially carcinogenic if inhaled [197].	Long-term presence may pose risks to living organisms [198].	Further studies needed on effects on soil-dwelling organisms.
Degradation and removal	Does not degrade naturally, only removable through physical filtration [199].	Remains in soil unless mechanically disturbed [119].	Development of effective removal techniques required.
Entry into surface waters	Introduced via rainfall and industrial discharge [67].	Transported by wind erosion and precipitation [200].	Transport mechanisms are not fully understood.
Ecological impact	May clog filter-feeding aquatic organisms [199].	Can alter soil microbial activity and reduce biodiversity [113].	Limited experimental data on ecological damage.
Sedimentation rate	Slow sedimentation, but turbulence can enhance dispersion [151].	Movement varies across different soil types [200].	Poorly understood mobility in various soil compositions.
Water treatment challenges	Difficult to remove with standard filtration methods [201].	Soil remediation is complex and costly [119].	Need for innovative methods for effective removal.
Groundwater contamination	Can infiltrate deeper layers via precipitation [151].	Mobility depends on soil pH and organic matter content [67].	Limited data on movement and concentrations in groundwater.
Temperature effects	Temperature variations may influence sedimentation [201].	Freeze-thaw cycles could alter distribution [202].	Insufficient knowledge of the impact of extreme climate conditions.
Impact on plants	Can be absorbed by plants through water uptake [198].	Toxic effects on root systems and plant growth [198].	Limited research on cellular effects in plants.
Interactions with sediments	May accumulate in sediments [67].	Functions as a temporary reservoir, releasing fibers back into the environment over time [20].	Further research needed to understand storage and mobilization in sediments.
Hydrodynamic influence	Flow conditions affect distribution [199].	Behavior varies in soils with different porosity levels [67].	Limited modeling data on flow velocity impacts.
Human exposure	Inhalation or ingestion possible through drinking and bathing [8].	Can be inhaled or ingested as soil dust [113].	Need for epidemiological studies to assess exposure risks.
Long-term environmental impact	May persist in aquatic environments for decades [199].	Can remain unchanged in soil for centuries [67].	Lack of long-term data on degradation or transformation potential.

Previous investigations have typically reported very low concentrations of asbestos, predominantly consisting of fibrous chrysotile asbestos.

However, these findings should not be misconstrued as indicative of a negligible risk. Rather, they highlight the necessity for more nuanced and comprehensive analytical approaches. Furthermore, the methodology for assessing water contamination due to asbestos remains rudimentary and parallels the approaches used for air quality assessments. This underscores the urgent need for developing more sophisticated procedures that can accurately quantify asbestos contamination in water bodies on a volumetric basis. The current reliance on air-based assessment models for water contamination is inadequate and does not capture the complexity of asbestos dispersion in aqueous environments.

Given these insights, it becomes evident that future research and policy frameworks must adopt a holistic approach to addressing the asbestos problem. This involves a thorough and integrated examination of air, soil, and water as interconnected environmental spheres and transmission media. Such a comprehensive approach will enable a more accurate assessment of the environmental and health risks posed by asbestos, thereby facilitating the development of more effective mitigation and management strategies. In conclusion, our research advocates for a paradigm shift in the way asbestos-related environmental hazards are understood and addressed, emphasizing the need for integrated, multidisciplinary investigations and interventions.

Author Contributions: G.Z.M., writing—original draft preparation writing—review and editing; A.T., writing—review and editing, supervision; D.B., writing—review and editing, supervision. All authors have read and agreed to the published version of the manuscript.

Funding: This research received no external funding. The APC was funded by Széchenyi István University, grant number: 063PTP2025.

Acknowledgments: The authors would like to thank Professor Heriberto Cabezas y Fernández for his preview and consultation of the manuscript.

Conflicts of Interest: The authors declare no conflicts of interest.

References

1. Brostrøm, A.; Harboe, H.; Fonseca, A.S.; Frederiksen, M.; Kines, P.; Bührmann, W.; Bønløkke, J.H.; Jensen, K.A. Asbestos fiber levels from remediation work. *J. Hazard. Mater. Adv.* **2025**, *17*, 100552. [CrossRef]
2. Abel, E.L.; DiGiovanni, J. Environmental Carcinogenesis. In *The Molecular Basis of Cancer*; Elsevier: Amsterdam, The Netherlands, 2008; pp. 91–113. [CrossRef]
3. Siengchin, S. A review on lightweight materials for defence applications: Present and future developments. *Def. Technol.* **2023**, *24*, 1–17. [CrossRef]
4. Yakubu, A.U.; Zhao, J.; Jiang, Q.; Ye, X.; Liu, J.; Yu, Q.; Xiong, S. A comprehensive review of primary cooling techniques and thermal management strategies for polymer electrolyte membrane fuel cells PEMFC. *Heliyon* **2024**, *10*, e38556. [CrossRef] [PubMed]
5. Pira, E.; Donato, F.; Maida, L.; Discalzi, G. Exposure to asbestos: Past, present and future. *J. Thorac. Dis.* **2018**, *10*, S237–S245. [CrossRef] [PubMed]
6. Frank, A.L.; van Zandwijk, N. Asbestos history and use. *Lung Cancer* **2024**, *193*, 107828. [CrossRef]
7. Tóth, E.; Weiszburg, T. *Környezeti Ásványtan*; Typotex Kiadó: Budapest, Hungary, 2011.
8. Peña-Castro, M.; Montero-Acosta, M.; Saba, M. A critical review of asbestos concentrations in water and air, according to exposure sources. *Heliyon* **2023**, *9*, e15730. [CrossRef]
9. Ross, M.; Langer, A.M.; Nord, G.L.; Nolan, R.P.; Lee, R.J.; Van Orden, D.; Addison, J. The mineral nature of asbestos. *Regul. Toxicol. Pharmacol.* **2008**, *52*, S26–S30. [CrossRef]
10. Strohmeier, B.R.; Huntington, J.C.; Bunker, K.L.; Sanchez, M.S.; Allison, K.; Lee, R.J. What is asbestos and why is it important? Challenges of defining and characterizing asbestos. *Int. Geol. Rev.* **2010**, *52*, 801–872. [CrossRef]
11. Millet, P. Electrochemical membrane reactors. In *Current Trends and Future Developments on (Bio)Membranes*; Elsevier: Amsterdam, The Netherlands, 2023; pp. 285–313. [CrossRef]
12. Seaman, D.M.; Meyer, C.A.; Kanne, J.P. Occupational and Environmental Lung Disease. *Clin. Chest Med.* **2015**, *36*, 249–268. [CrossRef]

13. Mannan, S. Chapter 18—Toxic Release. In *Lees' Loss Prevention in the Process Industries, Sam Mannan*; Elsevier: Amsterdam, The Netherlands; Butterworth-Heinemann: Oxford, UK, 2012; pp. 1679–1760. [CrossRef]
14. Cossette, M. Defining Asbestos for Monitoring Purposes. In *Definitions for Asbestos and other Health-Related Silicates (ASTM Special Technical Publication 834)*; Levadie, B., Ed.; American Society for Testing and Materials: Philadelphia, PA, USA, 1984; pp. 5–50.
15. Fitzgerald, S.M. Resolving asbestos and ultrafine particulate definitions with carcinogenicity. *Lung Cancer* **2024**, *189*, 107478. [CrossRef]
16. INSERM (Expertise Collective). *Effets sur la Santé des Principaux Types D'exposition à L'amiante*; INSERM: Paris, France, 1997.
17. Peng, W.; Mi, J.; Jiang, Y. Asbestos exposure and laryngeal cancer mortality. *Laryngoscope* **2016**, *126*, 1169–1174. [CrossRef] [PubMed]
18. Markowitz, S. Asbestos-Related Lung Cancer and Malignant Mesothelioma of the Pleura: Selected Current Issues. *Semin. Respir. Crit. Care Med.* **2015**, *36*, 334–346. [CrossRef] [PubMed]
19. LaDou, J.; Castleman, B.; Frank, A.; Gochfeld, M.; Greenberg, M.; Huff, J.; Joshi, T.K.; Landrigan, P.J.; Lemen, R.; Myers, J.; et al. The Case for a Global Ban on Asbestos. *Environ. Health Perspect.* **2010**, *118*, 897–901. [CrossRef] [PubMed]
20. Curado, A.; Nunes, L.J.R.; Carvalho, A.; Abrantes, J.; Lima, E.; Tomé, M. The Use of Asbestos and Its Consequences: An Assessment of Environmental Impacts and Public Health Risks. *Fibers* **2024**, *12*, 102. [CrossRef]
21. Visonà, S.D.; Villani, S.; Manzoni, F.; Chen, Y.; Ardissino, G.; Russo, F.; Moretti, M.; Javan, G.T.; Osculati, A. Impact of Asbestos on Public Health: A Retrospective Study on a Series of Subjects with Occupational and Non-Occupational Exposure to Asbestos during the Activity of Fibronit Plant (Broni, Italy). *J. Public Health Res.* **2018**, *7*, 122–127. [CrossRef]
22. Wagner, G.R.; Lemen, R. Asbestos. In *International Encyclopedia of Public Health*; Elsevier: Amsterdam, The Netherlands, 2008; pp. 238–245. [CrossRef]
23. Kamp, D.W. Asbestos-induced lung diseases: An update. *Transl. Res.* **2009**, *153*, 143–152. [CrossRef]
24. Wagner, G.R.; Lemen, R.A. Asbestos. In *International Encyclopedia of Public Health*; Elsevier: Amsterdam, The Netherlands, 2017; pp. 176–182. [CrossRef]
25. Noonan, C.W. Environmental asbestos exposure and risk of mesothelioma. *Ann. Transl. Med.* **2017**, *5*, 234. [CrossRef]
26. Aryal, A.; Morley, C. Mitigation of Contamination and Health Risk: Asbestos Management and Regulatory Practices. *Sustainability* **2024**, *16*, 9740. [CrossRef]
27. Boffetta, P. Epidemiology of environmental and occupational cancer. *Oncogene* **2004**, *23*, 6392–6403. [CrossRef]
28. Bloise, A.; Catalano, M.; Barrese, E.; Gualtieri, A.F.; Gandolfi, N.B.; Capella, S.; Belluso, E. TG/DSC study of the thermal behaviour of hazardous mineral fibres. *J. Therm. Anal. Calorim.* **2016**, *123*, 2225–2239. [CrossRef]
29. Donaldson, K.; Poland, C.A.; Murphy, F.A.; MacFarlane, M.; Chernova, T.; Schinwald, A. Pulmonary toxicity of carbon nanotubes and asbestos—Similarities and differences. *Adv. Drug Deliv. Rev.* **2013**, *65*, 2078–2086. [CrossRef] [PubMed]
30. Stayner, L.; Welch, L.S.; Lemen, R. The Worldwide Pandemic of Asbestos-Related Diseases. *Annu. Rev. Public Health* **2013**, *34*, 205–216. [CrossRef] [PubMed]
31. Bales, R.C.; Morgan, J.J. Dissolution kinetics of chrysotile at pH=7 to 10. *Geochim. Cosmochim. Acta* **1985**, *49*, 2281–2288.
32. Iwaszko, J.; Zawada, A.; Przerada, I.; Lubas, M. Structural and microstructural aspects of asbestos-cement waste vitrification. *Spectrochim. Acta Part A Mol. Biomol. Spectrosc.* **2018**, *195*, 95–102. [CrossRef]
33. Walter, M.; Geroldinger, G.; Gille, L.; Kraemer, S.M.; Schenkeveld, W.D. Soil-pH and cement influence the weathering kinetics of chrysotile asbestos in soils and its hydroxyl radical yield. *J. Hazard. Mater.* **2022**, *431*, 128068. [CrossRef]
34. Gronow, J.R. The dissolution of asbestos fibres in water. *Clay Miner.* **1987**, *22*, 21–35. [CrossRef]
35. Walter, M.; Schenkeveld, W.D.C.; Reissner, M.; Gille, L.; Kraemer, S.M. The Effect of pH and Biogenic Ligands on the Weathering of Chrysotile Asbestos: The Pivotal Role of Tetrahedral Fe in Dissolution Kinetics and Radical Formation. *Chem. Eur. J.* **2019**, *25*, 3286–3300. [CrossRef]
36. Choi, I.; Smith, R.W. Kinetic study of dissolution of asbestos fibers in water. *J. Colloid Interface Sci.* **1972**, *40*, 253–262. [CrossRef]
37. Rozalen, M.; Ramos, M.E.; Fiore, S.; Gervilla, F.; Huertas, F.J. Effect of oxalate and pH on chrysotile dissolution at 25 C: An experimental study. *Am. Miner.* **2014**, *99*, 589–600. [CrossRef]
38. Bernstein, D.; Dunnigan, J.; Hesterberg, T.; Brown, R.; Velasco, J.A.L.; Barrera, R.; Hoskins, J.; Gibbs, A. Health risk of chrysotile revisited. *Crit. Rev. Toxicol.* **2013**, *43*, 154–183. [CrossRef]
39. Pollastri, S.; Gualtieri, A.F.; Gualtieri, M.L.; Hanuskova, M.; Cavallo, A.; Gaudino, G. The zeta potential of mineral fibres. *J. Hazard. Mater.* **2014**, *276*, 469–479. [CrossRef] [PubMed]
40. Engin, A.B.; Nikitovic, D.; Neagu, M.; Henrich-Noack, P.; Docea, A.O.; Shtilman, M.I.; Golokhvast, K.; Tsatsakis, A.M. Mechanistic understanding of nanoparticles' interactions with extracellular matrix: The cell and immune system. *Part. Fibre Toxicol.* **2017**, *14*, 22. [CrossRef] [PubMed]
41. Holmes, E.P.; Wilson, J.; Schreier, H.; Lavkulich, L.M. Processes affecting surface and chemical properties of chrysotile: Implications for reclamation of asbestos in the natural environment. *Can. J. Soil Sci.* **2012**, *92*, 229–242. [CrossRef]

42. Martino, E.; Cerminara, S.T.; Prandi, L.; Fubini, B.; Perotto, S. Physical and biochemical interactions of soil fungi with asbestos fibers. *Environ. Toxicol. Chem.* **2004**, *23*, 938–944. [CrossRef]
43. Baur, X.; Soskolne, C.L.; Lemen, R.A.; Schneider, J.; Woitowitz, H.-J.; Budnik, L.T. How conflicted authors undermine the World Health Organization (WHO) campaign to stop all use of asbestos: Spotlight on studies showing that chrysotile is carcinogenic and facilitates other non-cancer asbestos-related diseases. *Int. J. Occup. Environ. Health* **2015**, *21*, 176–179. [CrossRef]
44. Kim, Y.-C.; Hong, W.-H.; Zhang, Y.-L. Development of a model to calculate asbestos fiber from damaged asbestos slates depending on the degree of damage. *J. Clean. Prod.* **2015**, *86*, 88–97. [CrossRef]
45. Virta, R.L. *Worldwide Asbestos Supply and Consumption Trends from 1900 Through 2003*; U.S. Geological Survey: Reston, VA, USA, 2006.
46. du Plessis, A.; Razavi, N.; Benedetti, M.; Murchio, S.; Leary, M.; Watson, M.; Bhate, D.; Berto, F. Properties and applications of additively manufactured metallic cellular materials: A review. *Prog. Mater. Sci.* **2022**, *125*, 100918. [CrossRef]
47. Nazir, A.; Gokcekaya, O.; Billah, K.M.M.; Ertugrul, O.; Jiang, J.; Sun, J.; Hussain, S. Multi-material additive manufacturing: A systematic review of design, properties, applications, challenges, and 3D printing of materials and cellular metamaterials. *Mater. Des.* **2023**, *226*, 111661. [CrossRef]
48. Driece, H.A.L.; Siesling, S.; Swuste, P.H.J.J.; Burdorf, A. Assessment of cancer risks due to environmental exposure to asbestos. *J. Expo. Sci. Environ. Epidemiol.* **2010**, *20*, 478–485. [CrossRef]
49. Szeszenia-Dabrowska, N.; Wilczyńska, U.; Szymczak, W.; Laskowicz, K. Environmental exposure to asbestos in asbestos cement workers: A case of additional exposure from indiscriminate use of industrial wastes. *Int. J. Occup. Med. Environ. Health* **1998**, *11*, 171–177.
50. Ferrante, D.; Mirabelli, D.; Tunesi, S.; Terracini, B.; Magnani, C. Pleural mesothelioma and occupational and non-occupational asbestos exposure: A case-control study with quantitative risk assessment. *Occup. Environ. Med.* **2016**, *73*, 147–153. [CrossRef] [PubMed]
51. Musti, M.; Pollice, A.; Cavone, D.; Dragonieri, S.; Bilancia, M. The relationship between malignant mesothelioma and an asbestos cement plant environmental risk: A spatial case–control study in the city of Bari (Italy). *Int. Arch. Occup. Environ. Health* **2009**, *82*, 489–497. [CrossRef] [PubMed]
52. Doll, R. Mortality from lung cancer in asbestos workers. *Occup. Environ. Med.* **1955**, *12*, 81–86.
53. Selikoff, I.J.; Churg, J.; Hammond, E.C. The occurrence of asbestosis among insulation workers in the United States. *Ann. N. Y. Acad. Sci.* **1965**, *132*, 139–155. [CrossRef]
54. Wagner, J.C.; Sleggs, C.A.; Marchand, P. Diffuse Pleural Mesothelioma and Asbestos Exposure in the North Western Cape Province. *Occup. Environ. Med.* **1960**, *17*, 260–271. [CrossRef]
55. IARC (International Agency for Research on Cancer). *IARC Monographs on the Evaluation of Carcinogenic Risks to Humans. Asbestos (Chrysotile, Amosite, Crocidolite, Tremolite, Actinolite, and Anthophyllite)*; IARC: Lyon, France, 2012; Volume 100C.
56. INRS (Institut National de Recherche et de Sécurité). Dates Clefs dans la Connaissance et la préVention du Risque Amiante. Available online: https://www.inrs.fr/risques/amiante/historique-problematique-amiante.html (accessed on 25 March 2024).
57. Thives, L.P.; Ghisi, E.; Júnior, J.J.T.; Vieira, A.S. Is asbestos still a problem in the world? A current review. *J. Environ. Manag.* **2022**, *319*, 115716. [CrossRef]
58. Rosell-Murphy, M.-I.; Abós-Herràndiz, R.; Olivella, J.T.; Alberti-Casas, C.; Allas, I.G.; Artés, X.M.; Günther, I.K.; Malet, I.G.; Martínez, R.O.; Canela-Soler, J. Risk factors associated with asbestos-related diseases: A community-based case–control study. *BMC Public Health* **2013**, *13*, 723. [CrossRef]
59. Henderson, D.; Leigh, J. The History of Asbestos Utilization and Recognition of Asbestos-Induced Diseases. In *Asbestos*; CRC Press: Boca Raton, FL, USA, 2011; pp. 1–22. [CrossRef]
60. Swartjes, F.A.; Tromp, P.C.; Wezenbeek, J.M. *Assessing Risks of Soil Contamination with Asbestos*; RIVM Report 711701034; TNO Environment, Energy & Process Innovation (MEP); Grontmij: Bilthoven, The Netherlands, 2003; pp. 1–69. (In Dutch)
61. Ramazzini, C. Asbestos Is Still With Us: Repeat Call for a Universal Ban. *Arch. Environ. Occup. Health* **2010**, *65*, 121–126. [CrossRef]
62. Rice, J. The Global Reorganization and Revitalization of the Asbestos Industry, 1970–2007. *Int. J. Health Serv.* **2011**, *41*, 239–254. [CrossRef]
63. Yvon, Y.; Sharrock, P. Characterization of Thermochemical Inactivation of Asbestos Containing Wastes and Recycling the Mineral Residues in Cement Products. *Waste Biomass-Valorization* **2011**, *2*, 169–181. [CrossRef]
64. Wang, J.; Schlagenhauf, L.; Setyan, A. Transformation of the released asbestos, carbon fibers and carbon nanotubes from composite materials and the changes of their potential health impacts. *J. Nanobiotechnol.* **2017**, *15*, 15. [CrossRef]
65. Hosseinpourpia, R.; Varshoee, A.; Soltani, M.; Hosseini, P.; Tabari, H.Z. Production of waste bio-fiber cement-based composites reinforced with nano-SiO2 particles as a substitute for asbestos cement composites. *Constr. Build. Mater.* **2012**, *31*, 105–111. [CrossRef]
66. Vimalanathan, M. Study on Fiber Reinforced Concrete Using Asbestos. *Int. J. Res. Appl. Sci. Eng. Technol.* **2022**, *10*, 4531–4534. [CrossRef]

67. Malinconico, S.; Paglietti, F.; Serranti, S.; Bonifazi, G.; Lonigro, I. Asbestos in soil and water: A review of analytical techniques and methods. *J. Hazard. Mater.* **2022**, *436*, 129083. [CrossRef] [PubMed]
68. Spasiano, D.; Pirozzi, F. Treatments of asbestos containing wastes. *J. Environ. Manag.* **2017**, *204*, 82–91. [CrossRef]
69. Burragato, F.; Gaglianone, G.; Gerbasi, G.; Mazziotti Tagliani, S.; Papacchini, L.; Rossini, F.; Sperduto, B. Fibrous mineral detection in natural soil and risk mitigation (1st paper). *Period. Mineral.* **2010**, *79*, 21–35.
70. Bint, L.; Hunt, S.; Dangerfield, D.; Mechaelis, M. *New Zeland Guidelines for Assessing and Managing Asbestos in Soil*; Printlink: Wellington, New Zealand, 2017.
71. Gualtieri, A.F. Recycling asbestos-containing material (ACM) from construction and demolition waste (CDW). In *Handbook of Recycled Concrete and Demolition Waste*; Elsevier: Amsterdam, The Netherlands, 2013; pp. 500–525. [CrossRef]
72. Bódizs, D. The Hungarian situation of construction and demolition waste recovery/processing plants. In *Az Építési és Bontási Hulladékot Hasznosító/Feldolgozó Művek Magyarországi Helyzetképe*; Széchenyi István University: Győr, Hungarian, 2022.
73. Leonelli, C.; Veronesi, P.; Boccaccini, D.N.; Rivasi, M.R.; Barbieri, L.; Andreola, F.; Lancellotti, I.; Rabitti, D.; Pellacani, G.C. Microwave thermal inertisation of asbestos containing waste and its recycling in traditional ceramics. *J. Hazard. Mater.* **2006**, *135*, 149–155. [CrossRef]
74. Plescia, P.; Gizzi, D.; Benedetti, S.; Camilucci, L.; Fanizza, C.; De Simone, P.; Paglietti, F. Mechanochemical treatment to recycling asbestos-containing waste. *Waste Manag.* **2003**, *23*, 209–218. [CrossRef]
75. Gualtieri, A.F.; Gualtieri, M.L.; Scognamiglio, V.; Di Giuseppe, D. Human Health Hazards Associated with Asbestos in Building Materials. In *Ecological and Health Effects of Building Materials*; Springer International Publishing: Cham, Switzerland, 2022; pp. 297–325. [CrossRef]
76. Spurny, K. On the release of asbestos fibers from weathered and corroded asbestos cement products. *Environ. Res.* **1989**, *48*, 100–116. [CrossRef]
77. Rodilla, J.M.R.; Cerrada, B.C.; Pujadas, C.S.; Delclos, G.L.; Benavides, F.G. Fiber burden and asbestos-related diseases: An umbrella review. *Gac. Sanit.* **2022**, *36*, 173–183. [CrossRef]
78. Favero-Longo, S.E.; Matteucci, E.; Siniscalco, C. Plant Colonization Limits Dispersion in the Air of Asbestos Fibers in an Abandoned Asbestos Mine. *Northeast. Nat.* **2009**, *16*, 163–177. [CrossRef]
79. Obmiński, A.; Janeczek, J. The effectiveness of asbestos stabilizers during abrasion of asbestos-cement sheets. *Constr. Build. Mater.* **2020**, *249*, 118767. [CrossRef]
80. Bolan, S.; Kempton, L.; McCarthy, T.; Wijesekara, H.; Piyathilake, U.; Jasemizad, T.; Padhye, L.P.; Zhang, T.; Rinklebe, J.; Wang, H.; et al. Sustainable management of hazardous asbestos-containing materials: Containment, stabilization and inertization. *Sci. Total Environ.* **2023**, *881*, 163456. [CrossRef] [PubMed]
81. Bornemann, P.; Hildebrandt, U. On the problem of environmental pollution by weathering products of asbestos cement. *Staub. Reinhaltung Luft* **1986**, *11*, 487–489.
82. Li, X.; Chen, Y.; Li, X.; Wang, M.; Xie, W.; Ding, D.; Kong, L.; Jiang, D.; Long, T.; Deng, S. Asbestos-Environment Pollution Characteristics and Health-Risk Assessment in Typical Asbestos-Mining Area. *Toxics* **2023**, *11*, 494. [CrossRef]
83. Avataneo, C.; Capella, S.; Luiso, M.; Marangoni, G.; Lasagna, M.; De Luca, D.A.; Bergamini, M.; Belluso, E.; Turci, F. Waterborne asbestos: Good practices for surface waters analyses. *Front. Chem.* **2023**, *11*, 1104569. [CrossRef]
84. Labagnara, D.; Patrucco, M.; Sorlini, A. Occupational Safety and Health in Tunnelling in Rocks Formations Potentially Containing Asbestos: Good Practices for Risk Assessment and Management. *Am. J. Appl. Sci.* **2016**, *13*, 646–656. [CrossRef]
85. Sachanbiński, M. Results of research on asbestos pollution of the natural environment of Lower Silesia. In *Risk Assessment of Buildings Contaminated with Asbestos Dust and Risk Control Methods*; Seminar; Warsaw-ITB: Warsaw, Poland, 2009.
86. Kurunthachalam, S.K. Asbestos and Its Toxicological Concern. *Hydrol. Curr. Res.* **2013**, *4*, 1–5. [CrossRef]
87. Zhang, Y.-L.; Kim, Y.-C.; Hong, W.-H. Visualizing distribution of naturally discharged asbestos fibers in Korea through analysis of thickness changes in asbestos cement slates. *J. Clean. Prod.* **2016**, *112*, 607–619. [CrossRef]
88. Bassani, C.; Cavalli, R.M.; Cavalcante, F.; Cuomo, V.; Palombo, A.; Pascucci, S.; Pignatti, S. Deterioration status of asbestos-cement roofing sheets assessed by analyzing hyperspectral data. *Remote Sens. Environ.* **2007**, *109*, 361–378. [CrossRef]
89. Mohanty, S.K.; Salamatipour, A.; Willenbring, J.K. Mobility of asbestos fibers below ground is enhanced by dissolved organic matter from soil amendments. *J. Hazard. Mater. Lett.* **2021**, *2*, 100015. [CrossRef]
90. Capacci, L.; Biondini, F.; Frangopol, D.M. Resilience of aging structures and infrastructure systems with emphasis on seismic resilience of bridges and road networks: Review. *Resilient Cities Struct.* **2022**, *1*, 23–41. [CrossRef]
91. Cely-García, M.F.; Lysaniuk, B.; Pasetto, R.; Ramos-Bonilla, J.P. The challenges of applying an Activity-Based Sampling methodology to estimate the cancer risk associated with asbestos contaminated landfilled zones. *Environ. Res.* **2020**, *181*, 108893. [CrossRef]
92. Gray, C.; Carey, R.N.; Reid, A. Current and future risks of asbestos exposure in the Australian community. *Int. J. Occup. Environ. Health* **2016**, *22*, 292–299. [CrossRef]

93. Obmiński, A. The concentration of asbestos fibres during buildings exploitation analysis of concentration variability. *Arch. Civ. Mech. Eng.* **2021**, *22*, 6. [CrossRef]
94. Chen, F.; Zhang, W.; Mfarrej, M.F.B.; Saleem, M.H.; Khan, K.A.; Ma, J.; Raposo, A.; Han, H. Breathing in danger: Understanding the multifaceted impact of air pollution on health impacts. *Ecotoxicol. Environ. Saf.* **2024**, *280*, 116532. [CrossRef] [PubMed]
95. Kottek, M.; Yuen, M.L. Public health risks from asbestos cement roofing. *Am. J. Ind. Med.* **2022**, *65*, 157–161. [CrossRef]
96. Singh, R.; Frank, A.L. Does the Presence of Asbestos-Containing Materials in Buildings Post-construction and Before Demolition Have an Impact on the Exposure to Occupants in Non-occupational Settings? *Cureus* **2023**, *15*, 1–11. [CrossRef]
97. Miserocchi, G.; Sancini, G.; Mantegazza, F.; Chiappino, G. Translocation pathways for inhaled asbestos fibers. *Environ. Health* **2008**, *7*, 4. [CrossRef]
98. Huh, D.-A.; Chae, W.-R.; Choi, Y.-H.; Kang, M.-S.; Lee, Y.-J.; Moon, K.-W. Disease Latency according to Asbestos Exposure Characteristics among Malignant Mesothelioma and Asbestos-Related Lung Cancer Cases in South Korea. *Int. J. Environ. Res. Public Health* **2022**, *19*, 15934. [CrossRef]
99. Zhang, Y.-L.; Byeon, H.-S.; Hong, W.-H.; Cha, G.-W.; Lee, Y.-H.; Kim, Y.-C. Risk assessment of asbestos containing materials in a deteriorated dwelling area using four different methods. *J. Hazard. Mater.* **2021**, *410*, 124645. [CrossRef]
100. Durczak, K.; Pyzalski, M.; Brylewski, T.; Juszczyk, M.; Leśniak, A.; Libura, M.; Ustinovičius, L.; Vaišnoras, M. Modern Methods of Asbestos Waste Management as Innovative Solutions for Recycling and Sustainable Cement Production. *Sustainability* **2024**, *16*, 8798. [CrossRef]
101. Tetik, Y.Ö.; Zümrüt, I.B.; Çamurcu, A.G.; Kale, Ö.A.; Baradan, S. Measurement and removal of asbestos in residential dwellings to be demolished—Urban transformation experience in Izmir, Turkey. *Environ. Sci. Pollut. Res.* **2024**, *31*, 9857–9866. [CrossRef] [PubMed]
102. Gualtieri, A.F. Naturally Occurring Asbestos: A Global Health Concern? State of the Art and Open Issues. *Environ. Eng. Geosci.* **2020**, *26*, 3–8. [CrossRef]
103. Bloise, A.; Ricchiuti, C.; Punturo, R.; Pereira, D. Potentially toxic elements (PTEs) associated with asbestos chrysotile, tremolite and actinolite in the Calabria region (Italy). *Chem. Geol.* **2020**, *558*, 119896. [CrossRef]
104. Baumann, F.; Buck, B.J.; Metcalf, R.V.; McLaurin, B.T.; Merkler, D.J.; Carbone, M. The Presence of Asbestos in the Natural Environment is Likely Related to Mesothelioma in Young Individuals and Women from Southern Nevada. *J. Thorac. Oncol.* **2015**, *10*, 731–737. [CrossRef] [PubMed]
105. Buck, B.J.; Goossens, D.; Metcalf, R.V.; McLaurin, B.; Ren, M.; Freudenberger, F. Naturally Occurring Asbestos: Potential for Human Exposure, Southern Nevada, USA. *Soil Sci. Soc. Am. J.* **2013**, *77*, 2192–2204. [CrossRef]
106. Obmiński, A. Asbestos cement products and their impact on soil contamination in relation to various sources of anthropogenic and natural asbestos pollution. *Sci. Total Environ.* **2022**, *848*, 157275. [CrossRef]
107. Ricchiuti, C.; Pereira, D.; Punturo, R.; Giorno, E.; Miriello, D.; Bloise, A. Hazardous Elements in Asbestos Tremolite from the Basilicata Region, Southern Italy: A First Step. *Fibers* **2021**, *9*, 47. [CrossRef]
108. Evans, B.W. The Serpentinite Multisystem Revisited: Chrysotile Is Metastable. *Int. Geol. Rev.* **2004**, *46*, 479–506. [CrossRef]
109. Bowes, D.R.; Langer, A.M.; Rohl, A.N.; Zussman, J. Nature and Range of Mineral Dusts in the Environment. *Philos. Trans. R. Soc. Lond. Ser. A Math. Phys. Sci.* **1977**, *286*, 593–610.
110. Lee, R.; Strohmeier, B.; Bunker, K.; Van Orden, D. Naturally occurring asbestos—A recurring public policy challenge. *J. Hazard. Mater.* **2008**, *153*, 1–21. [CrossRef]
111. U.S. Environmental Protection Agency. Reuse and Revitalization in Ambler, Pennsylvania. Available online: https://www.epa.gov/ambler/reuse-and-revitalization-ambler-pennsylvania (accessed on 28 June 2022).
112. Swartjes, F.A.; Tromp, P.C. A Tiered Approach for the Assessment of the Human Health Risks of Asbestos in Soils. *Soil Sediment Contam. Int. J.* **2008**, *17*, 137–149. [CrossRef]
113. Turci, F.; Favero-Longo, S.E.; Gazzano, C.; Tomatis, M.; Gentile-Garofalo, L.; Bergamini, M. Assessment of asbestos exposure during a simulated agricultural activity in the proximity of the former asbestos mine of Balangero, Italy. *J. Hazard. Mater.* **2016**, *308*, 321–327. [CrossRef] [PubMed]
114. Zuo, J.; Rameezdeen, R.; Hagger, M.; Zhou, Z.; Ding, Z. Dust pollution control on construction sites: Awareness and self-responsibility of managers. *J. Clean. Prod.* **2017**, *166*, 312–320. [CrossRef]
115. González-Zamora, Á.; Benito-Verdugo, P.; Martínez-Fernández, J. On the Variability in the Temporal Stability Pattern of Soil Moisture Under Mediterranean Conditions. *Span. J. Soil Sci.* **2024**, *14*, 12839. [CrossRef]
116. Steffan, J.J.; Brevik, E.C.; Burgess, L.C.; Cerdà, A. The effect of soil on human health: An overview. *Eur. J. Soil Sci.* **2018**, *69*, 159–171. [CrossRef]
117. David, S.R.; Fritsch, S.; Forster, A.; Ihiawakrim, D.; Geoffroy, V.A. Flocking asbestos waste, an iron and magnesium source for Pseudomonas. *Sci. Total Environ.* **2020**, *709*, 135936. [CrossRef]
118. Holmes, E.P.; Lavkulich, L. The effects of naturally occurring acids on the surface properties of chrysotile asbestos. *J. Environ. Sci. Health Part A* **2014**, *49*, 1445–1452. [CrossRef]

119. Berry, T.-A.; Wallis, S.; Doyle, E.; de Lange, P.; Steinhorn, G.; Vigliaturo, R.; Belluso, E.; Blanchon, D. A Preliminary Investigation into the Degradation of Asbestos Fibres in Soils, Rocks and Building Materials Associated with Naturally Occurring Biofilms. *Minerals* **2024**, *14*, 106. [CrossRef]
120. Łuniewski, S.; Rogowska, W.; Łozowicka, B.; Iwaniuk, P. Plants, Microorganisms and Their Metabolites in Supporting Asbestos Detoxification—A Biological Perspective in Asbestos Treatment. *Materials* **2024**, *17*, 1644. [CrossRef]
121. Wong, L.C.; Rodenburg, U.; Leite, R.R.; Korthals, G.W.; Pover, J.; Koerten, H.; Kuramae, E.E.; Bodelier, P.L. Exploring microbial diversity and interactions for asbestos modifying properties. *Sci. Total Environ.* **2024**, *951*, 175577. [CrossRef]
122. David, S.R.; Geoffroy, V.A. A Review of Asbestos Bioweathering by Siderophore-Producing *Pseudomonas*: A Potential Strategy of Bioremediation. *Microorganisms* **2020**, *8*, 1870. [CrossRef]
123. Ricchiuti, C.; Bloise, A.; Punturo, R. Occurrence of asbestos in soils: State of the art. *Episodes* **2020**, *43*, 881–891. [CrossRef]
124. American Chemical Society. The Fate of Asbestos in Soil: Remediation Prospects and Paradigms. In Proceedings of the National Conference in Philadelphia, Philadelphia, PA, USA, 21–25 August 2016. Session TOXI 25.
125. Addison, J.; Davies, L.S.T.; Robertson, A.; Willey, R.J. *Release of Dispersed Asbestos Fibres from Soils*; Technical Report; Institute of Occupational Medicine: Edinburgh, UK, 1988.
126. Baker, K.; Sole, S.; Hay, S.; Mitcheson, B.; Thomas, L. *Discussion Paper on Guidelines for Airborne Concentrations of Asbestos Fibres in Ambient Air: Implications for Quantitative Risk Assessment; Society for Brownfield Risk Assessors*; SoBRA: Edinburgh, Scotland, 2018.
127. Hellawell, E.E.; Hughes, S.J. Asbestos contamination on brownfield development sites in the UK. *Environ. Res.* **2021**, *198*, 110480. [CrossRef] [PubMed]
128. Ryan, J.N.; Elimelech, M. Colloid mobilization and transport in groundwater. *Colloids Surfaces A Physicochem. Eng. Asp.* **1996**, *107*, 1–56. [CrossRef]
129. Skinner, H.C.W.; Ross, M.; Frondel, C. *Asbestos and Other Fibrous Materials: Mineralogy, Crystal Chemistry, and Health Effects*; Oxford University Press: Oxford, UK, 1988.
130. Agency for Toxic Substances and Disease Registry (ATSDR). *Toxicological Profile for Asbestos*; U.S. Department of Health and Human Services, Public Health Service: Atlanta, GA, USA, 2002.
131. Pelley, A.J.; Tufenkji, N. Effect of particle size and natural organic matter on the migration of nano- and microscale latex particles in saturated porous media. *J. Colloid Interface Sci.* **2008**, *321*, 74–83. [CrossRef]
132. Seymour, M.B.; Chen, G.; Su, C.; Li, Y. Transport and retention of colloids in porous media: Does shape really matter? *Environ. Sci. Technol.* **2013**, *47*, 8391–8398.
133. Bradford, S.A.; Yates, S.R.; Bettahar, M.; Simunek, J. Physical factors affecting the transport and fate of colloids in saturated porous media. *Water Resour. Res.* **2002**, *38*, 63-1–63-12. [CrossRef]
134. Bergendahl, J.; Grasso, D. Prediction of colloid detachment in a model porous media: Thermodynamics. *AIChE J.* **1999**, *45*, 475–484. [CrossRef]
135. Tufenkji, N.; Elimelech, M. Breakdown of Colloid Filtration Theory: Role of the Secondary Energy Minimum and Surface Charge Heterogeneities. *Langmuir* **2005**, *21*, 841–852. [CrossRef]
136. Hofmann, A.; Liang, L. Mobilization of colloidal ferrihydrite particles in porous media—An inner-sphere complexation approach. *Geochim. Cosmochim. Acta* **2007**, *71*, 5847–5861. [CrossRef]
137. Pignatello, J.J.; Kwon, S.; Lu, Y. Effect of Natural Organic Substances on the Surface and Adsorptive Properties of Environmental Black Carbon (Char): Attenuation of Surface Activity by Humic and Fulvic Acids. *Environ. Sci. Technol.* **2006**, *40*, 7757–7763. [CrossRef] [PubMed]
138. Inam, M.A.; Khan, R.; Akram, M.; Khan, S.; Park, D.R.; Yeom, I.T. Interaction of Arsenic Species with Organic Ligands: Competitive Removal from Water by Coagulation-Flocculation-Sedimentation (C/F/S). *Molecules* **2019**, *24*, 1619. [CrossRef]
139. Elimelech, M. Effect of Particle Size on the Kinetics of Particle Deposition under Attractive Double Layer Interactions. *J. Colloid Interface Sci.* **1994**, *164*, 190–199. [CrossRef]
140. Petriglieri, J.R.; Laporte-Magoni, C.; Salvioli-Mariani, E.; Ferrando, S.; Tomatis, M.; Fubini, B.; Turci, F. Morphological and chemical properties of fibrous antigorite from lateritic deposit of New Caledonia in view of hazard assessment. *Sci. Total Environ.* **2021**, *777*, 146185. [CrossRef]
141. Baumann, F.; Maurizot, P.; Mangeas, M.; Ambrosi, J.-P.; Douwes, J.; Robineau, B. Pleural Mesothelioma in New Caledonia: Associations with Environmental Risk Factors. *Environ. Health Perspect.* **2011**, *119*, 695–700. [CrossRef]
142. Ma, H.; Bolster, C.; Johnson, W.P.; Li, K.; Pazmino, E.; Camacho, K.M.; Anselmo, A.C.; Mitragotri, S. Coupled influences of particle shape, surface property and flow hydrodynamics on rod-shaped colloid transport in porous media. *J. Colloid Interface Sci.* **2020**, *577*, 471–480. [CrossRef]
143. Quilliam, R.S.; Pow, C.J.; Mwesiga, J.J.; Shilla, D.A.; Woodford, L. Microplastics in agriculture—A potential novel mechanism for the delivery of human pathogens onto crops. *Front. Plant Sci.* **2023**, *14*, 1152419. [CrossRef]

144. Tripathi, S.; Srivastava, P.; Devi, R.S.; Bhadouria, R. Influence of synthetic fertilizers and pesticides on soil health and soil microbiology. In *Agrochemicals Detection, Treatment and Remediation*; Elsevier: Amsterdam, The Netherlands, 2020; pp. 25–54. [CrossRef]
145. Sánchez-Castro, I.; Molina, L.; Prieto-Fernández, M.; Segura, A. Past, present and future trends in the remediation of heavy-metal contaminated soil—Remediation techniques applied in real soil-contamination events. *Heliyon* **2023**, *9*, e16692. [CrossRef]
146. Fortunato, L.; Rushton, L. Stomach cancer and occupational exposure to asbestos: A meta-analysis of occupational cohort studies. *Br. J. Cancer* **2015**, *112*, 1805–1815. [CrossRef] [PubMed]
147. World Health Organization (WHO). *Elimination of Asbestos-Related Diseases (No. WHO/SDE/OEH/06.03)*; World Health Organization: Geneva, Switzerland, 2006.
148. Gamble, J. Risk of gastrointestinal cancers from inhalation and ingestion of asbestos. *Regul. Toxicol. Pharmacol.* **2008**, *52*, S124–S153. [CrossRef]
149. Totaro, M.; Giorgi, S.; Filippetti, E.; Gallo, A.; Frendo, L.; Privitera, G.; Baggiani, A. Asbestos in drinking water and the dangers to human health: A narrative synthesis. *Ig Sanita Pubbl.* **2019**, *75*, 303–312.
150. Cheng, T.J.; More, S.L.; Maddaloni, M.A.; Fung, E.S. Evaluation of potential gastrointestinal carcinogenicity associated with the ingestion of asbestos. *Rev. Environ. Health* **2021**, *36*, 15–26. [CrossRef] [PubMed]
151. Avataneo, C.; Petriglieri, J.R.; Capella, S.; Tomatis, M.; Luiso, M.; Marangoni, G.; Lazzari, E.; Tinazzi, S.; Lasagna, M.; De Luca, D.A.; et al. Chrysotile asbestos migration in air from contaminated water: An experimental simulation. *J. Hazard. Mater.* **2022**, *424*, 127528. [CrossRef]
152. Lin, R.-T.; Soeberg, M.J.; Chien, L.-C.; Fisher, S.; Takala, J.; Lemen, R.; Driscoll, T.; Takahashi, K. Bibliometric analysis of gaps in research on asbestos-related diseases: Declining emphasis on public health over 26 years. *BMJ Open* **2018**, *8*, e022806. [CrossRef]
153. Fuller, W.H. *Movement of Selected Metals, Asbestos, and Cyanide in Soil: Applications to Waste Disposal Problems*; Municipal Environmental Research Laboratory, Office of Research and Development, US Environmental Protection Agency: Washington, DC, USA, 1977.
154. Buzio, S.; Pesando, G.; Zuppi, G.M. Hydrogeological study on the presence of asbestos fibres in water of northern Italy. *Water Res.* **2000**, *34*, 1817–1822. [CrossRef]
155. Koumantakis, E.; Kalliopi, A.; Dimitrios, K.; Gidarakos, E. Asbestos pollution in an inactive mine: Determination of asbestos fibers in the deposit tailings and water. *J. Hazard. Mater.* **2009**, *167*, 1080–1088. [CrossRef] [PubMed]
156. Klebe, S.; Leigh, J.; Henderson, D.W.; Nurminen, M. Asbestos, Smoking and Lung Cancer: An Update. *Int. J. Environ. Res. Public Health* **2019**, *17*, 258. [CrossRef]
157. Avataneo, C.; Belluso, E.; Bergamini, M.; Capella, S.; De Luca, D.A.; Lasagna, M.; Turci, F. Waterborne Naturally Occurring Asbestos: A case study from Piedmont (NW Italy). In Proceedings of the 22nd EGU General Assembly, online, 4–8 May 2020; p. 19615.
158. IARC Working Group on the Evaluation of Carcinogenic Risks to Humans. Asbestos (chrysotile, amosite, crocidolite, tremolite, actinolite and anthophyllite). In *Arsenic, Metals, Fibres and Dusts*; International Agency for Research on Cancer: Lyon, France, 2012.
159. Li, X.; Ding, D.; Xie, W.; Zhang, Y.; Kong, L.; Li, M.; Li, M.; Deng, S. Risk assessment and source analysis of heavy metals in soil around an asbestos mine in an arid plateau region, China. *Sci. Rep.* **2024**, *14*, 7552. [CrossRef]
160. Gwinn, M.R.; DeVoney, D.; Jarabek, A.M.; Sonawane, B.; Wheeler, J.; Weissman, D.N.; Masten, S.; Thompson, C. Meeting Report: Mode(s) of Action of Asbestos and Related Mineral Fibers. *Environ. Health Perspect.* **2011**, *119*, 1806–1810. [CrossRef] [PubMed]
161. Wang, D.; Cullimore, R.; Hu, Y.; Chowdhury, R. Biodeterioration of asbestos cement (AC) pipe in drinking water distribution systems. *Int. Biodeterior. Biodegrad.* **2011**, *65*, 810–817. [CrossRef]
162. Baumann, F.; Ambrosi, J.-P.; Carbone, M. Asbestos is not just asbestos: An unrecognised health hazard. *Lancet Oncol.* **2013**, *14*, 576–578. [CrossRef]
163. Jacques, O.; Pienitz, R. Assessment of asbestos fiber contamination in lake sediment cores of the Thetford Mines region, southern Quebec (Canada). *Environ. Adv.* **2022**, *8*, 100232. [CrossRef]
164. Caramuscio, P.; Annoni, P.; Maran, S.; Zuppi, G.M.; Pesando, G. *Simulation of Nickel Transport in the Shallow Aquifer of Stura Valley (Northern Italy)*; Brookfield: Rotterdam, The Netherlands, 1992.
165. Monaro, S.; Landsberger, S.; Lecomte, R.; Paradis, P.; Desaulniers, G.; P'An, A. Asbestos pollution levels in river water measured by proton-induced X-ray emission (PIXE) techniques. *Environ. Pollut. Ser. B Chem. Phys.* **1983**, *5*, 83–90. [CrossRef]
166. Meck, M.; Love, D.; Mapani, B. Zimbabwean mine dumps and their impacts on river water quality—A reconnaissance study. *Phys. Chem. Earth Parts A B C* **2006**, *31*, 797–803. [CrossRef]
167. Kusiorowski, R.; Lipowska, B.; Kujawa, M.; Gerle, A. Problem of asbestos-containing wastes in Poland. *Clean. Waste Syst.* **2023**, *4*, 100085. [CrossRef]
168. McGuire, M.J.; Bowers, A.E.; Bowers, D.A. Asbestos analysis case history: Surface water supplies in Southern California. *J. AWWA* **1982**, *74*, 470–478. [CrossRef]

169. McMillan, L.M.; Stout, R.G.; Willey, B.F. Asbestos in raw and treated water: An electron microscopy study. *Environ. Sci. Technol.* **1977**, *11*, 390–394. [CrossRef]
170. Wei, B.; Ye, B.; Yu, J.; Jia, X.; Zhang, B.; Zhang, X.; Lu, R.; Dong, T.; Yang, L. Concentrations of asbestos fibers and metals in drinking water caused by natural crocidolite asbestos in the soil from a rural area. *Environ. Monit. Assess.* **2012**, *185*, 3013–3022. [CrossRef]
171. Anastasiadou, K.; Gidarakos, E. Toxicity evaluation for the broad area of the asbestos mine of northern Greece. *J. Hazard. Mater.* **2007**, *139*, 9–18. [CrossRef] [PubMed]
172. Kashansky, S.V.; Slyshkina, T.V. Asbestos in water sources of the Bazhenovskoye chrysotile asbestos deposit. *Int. J. Occup. Med. Environ. Health* **2002**, *15*, 65–68.
173. Schreier, H.; Omueti, J.A.; Lavkulich, L.M. Weathering processes of asbestos-rich serpentinitic sediments. *Soil Sci. Soc. Am. J.* **1987**, *51*, 993–999.
174. Avataneo, C.; Belluso, E.; Capella, S.; Cocca, D.; Lasagna, M.; Pigozzi, G.; De Luca, D.A. Groundwater asbestos pollution from Naturally Occurring Asbestos (NOA): A preliminary study on the Lanzo Valleys and Balangero Plain area, NW Italy. *Ital. J. Eng. Geol. Environ.* **2021**, *1*, 5–19.
175. Malinconico, S.; Cappa, F.; Zamengo, L. International and Italian regulations concerning asbestos limits in liquids. In Proceedings of the International Conference on Asbestos Monitoring and Analytical Methods (AMAM), Venezia, Italy, 5–7 December 2005.
176. European Parliament and Council. Directive (EU) 2020/2184 of the European Parliament and of the Council of 16 December 2020 on the Quality of Water Intended for Human Consumption (Recast) (Text with EEA Relevance). Official Journal of the European Union, L 435, 23 December 2020, pp. 1–62. Available online: https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32020L2184 (accessed on 21 March 2024).
177. Lisco, S.; Lapietra, I.; Laviano, R.; Mastronuzzi, G.; Fracchiolla, T.; Moretti, M. Sedimentological features of asbestos cement fragments in coastal environments (Taranto, southern Italy). *Mar. Pollut. Bull.* **2023**, *187*, 114469. [CrossRef]
178. Vincenten, J.; George, F.; Martuzzi, M.; Schröder-Bäck, P.; Paunovic, E. Barriers and Facilitators to the Elimination of Asbestos Related Diseases—Stakeholders' Perspectives. *Int. J. Environ. Res. Public Health* **2017**, *14*, 1269. [CrossRef] [PubMed]
179. Pirani, G. Amianto nelle acque: Inquadramento normativo e proposte operative. In *Rischio Amianto in Italia: Da Minerale Pregiato a Minaccia per la Salute e per L'ambiente (Geologia dell'Ambiente, Addendum to No.4/2017)*; Baldi, D., Ed.; SIGEA: Castelfidardo, Italy, 2017; pp. 53–62.
180. Go, J.; Farhat, N.; Leingartner, K.; Insel, E.I.; Momoli, F.; Carrier, R.; Krewski, D. Review of epidemiological and toxicological studies on health effects from ingestion of asbestos in drinking water. *Crit. Rev. Toxicol.* **2024**, *54*, 856–894. [CrossRef]
181. WHO. International Programme on Chemical Safety—Asbestos. Available online: https://www.who.int/health-topics/chemical-safety#tab=tab_1/ (accessed on 11 November 2020).
182. U.S. Environmental Protection Agency (EPA). *Safe Drinking Water Act*; EPA 816-F-04-030; U.S. EPA: Washington, DC, USA, 2004. Available online: https://www.epa.gov/sdwa (accessed on 20 March 2024).
183. *EPA/600/R-94/134*; Method 100.2 Determination of Asbestos Structures over 10 µm in Length in Drinking Water. U.S. EPA-Environmental Protection Agency: Washington, DC, USA, 1994.
184. Millette, J.R.; Clark, P.J.; Pansing, M.F.; Twyman, J.D. Concentration and size of asbestos in water supplies. *Environ. Health Perspect.* **1980**, *34*, 13–25. [CrossRef]
185. Millette, J.R.; Clark, P.J.; Stober, J.; Rosenthal, M. Asbestos in water supplies of the United States. *Environ. Health Perspect.* **1983**, *53*, 45–48. [CrossRef]
186. Webber, J.S.; Syrotynski, S.; King, M.V. Asbestos-contaminated drinking water: Its impact on household air. *Environ. Res.* **1988**, *46*, 153–167. [CrossRef]
187. Chatfield, E.J.; Dillon, M.J. *A National Survey for Asbestos Fibres in Canadian Drinking Water Supplies (79EHD-34)*; Department of National Health and Welfare: Ottawa, ON, Canada, 1979.
188. Capuano, F.; Fava, A.; Bacci, T.; Sala, O.; Paoli, F.; Biancolini, V.; Motta, E. La ricerca di amianto nelle acque potabile. In *Arpa Agenzia Regionale Prevenzione e Ambiente dell'Emilia-Romagna, Ecoscienza n. 3 /2014*; Acque potabili un sistema di controllo integrato all'altezza delle sfide; Arpae Emilia-Romagna (Agenzia Prevenzione Ambiente Energia dell'Emilia-Romagna): Bologna, Italy, 2014.
189. Hernández, M.; Pereira, D.; Bloise, A. Asbestos: Communicating the Health Issues Derived from Fibrous Minerals to Society. *Appl. Sci.* **2024**, *14*, 8980. [CrossRef]
190. Roccaro, P.; Vagliasindi, F.G. Indoor release of asbestiform fibers from naturally contaminated water and related health risk. *Chemosphere* **2018**, *202*, 76–84. [CrossRef]
191. Al-Hashimi, O.; Hashim, K.; Loffill, E.; Čebašek, T.M.; Nakouti, I.; Faisal, A.A.H.; Al-Ansari, N. A Comprehensive Review for Groundwater Contamination and Remediation: Occurrence, Migration and Adsorption Modelling. *Molecules* **2021**, *26*, 5913. [CrossRef]

192. Mahjoub, O.; Mauffret, A.; Michel, C.; Chmingui, W. Use of groundwater and reclaimed water for agricultural irrigation: Farmers' practices and attitudes and related environmental and health risks. *Chemosphere* **2022**, *295*, 133945. [CrossRef] [PubMed]
193. Berry, T.-A.; Belluso, E.; Vigliaturo, R.; Gieré, R.; Emmett, E.A.; Testa, J.R.; Steinhorn, G.; Wallis, S.L. Asbestos and Other Hazardous Fibrous Minerals: Potential Exposure Pathways and Associated Health Risks. *Int. J. Environ. Res. Public Health* **2022**, *19*, 4031. [CrossRef]
194. Ranaweera, K.H.; Grainger, M.N.C.; French, A.D.; Mucalo, M.R. Construction and demolition waste repurposed for heavy metal ion removal from wastewater: A review of current approaches. *Int. J. Environ. Sci. Technol.* **2023**, *20*, 9393–9422. [CrossRef]
195. Della Ventura, G.; Rabiee, A.; Marcelli, A.; Macis, S.; D'arco, A.; Iezzi, G.; Radica, F.; Lucci, F. A new approach to deposit homogeneous samples of asbestos fibres for toxicological tests in vitro. *Front. Chem.* **2023**, *11*, 1116463. [CrossRef]
196. Trivedi, A.K.; Ahmad, I. Effects of Chrysotile Asbestos Contaminated Soil on Crop Plants. *Soil Sediment Contam. Int. J.* **2011**, *20*, 767–776. [CrossRef]
197. Lee, E.-S.; Kim, Y.-K. Asbestos Exposure Level and the Carcinogenic Risk Due to Corrugated Asbestos-Cement Slate Roofs in Korea. *Int. J. Environ. Res. Public Health* **2021**, *18*, 6925. [CrossRef]
198. Saleem, K.; Asghar, M.A.; Saleem, M.H.; Raza, A.; Kocsy, G.; Iqbal, N.; Ali, B.; Albeshr, M.F.; Bhat, E.A. Chrysotile-Asbestos-Induced Damage in *Panicum virgatum* and *Phleum pretense* Species and Its Alleviation by Organic-Soil Amendment. *Sustainability* **2022**, *14*, 10824. [CrossRef]
199. Schreier, H. (Ed.) Chapter 3 Asbestos in the Aquatic Environment. In *Asbestos in the Natural Environment*; Elsevier: Amsterdam, The Netherlands, 1989; Volume 37, pp. 45–67. [CrossRef]
200. Maulida, P.T.; Kim, J.W.; Jung, M.C. Environmental Assessment of Friable Asbestos from Soil to Air Using the Releasable Asbestos Sampler (RAS). *Toxics* **2022**, *10*, 748. [CrossRef] [PubMed]
201. Iwaszko, J. Making asbestos-cement products safe using heat treatment. *Case Stud. Constr. Mater.* **2019**, *10*, e00221. [CrossRef]
202. Kusiorowski, R.; Zaremba, T.; Piotrowski, J.; Adamek, J. Thermal decomposition of different types of asbestos. *J. Therm. Anal. Calorim.* **2012**, *109*, 693–704. [CrossRef]

CHEMICAL ENGINEERING TRANSACTIONS
VOL. 114, 2024

A publication of AIDIC
The Italian Association of Chemical Engineering
Online at www.cetjournal.it

Guest Editors: Petar S. Varbanov, Min Zeng, Yee Van Fan, Xuechao Wang

ISBN 979-12-81206-12-0; **ISSN** 2283-9216
DOI: 10.3303/CET24114119

Analysing the Environmental Durability and Chrysotile Content of Asbestos Cement Products by FT-IR Spectroscopy

Gergely Z. Macher[a,*], András Torma[a], Dóra Beke[b]

[a]Department of Applied Sustainability, Albert Kázmér Faculty of Agricultural and Food Sciences of Széchenyi István University in Mosonmagyaróvár, Széchenyi István University, Győr.
[b]Department of Plant Sciences, Albert Kázmér Faculty of Agricultural and Food Sciences of Széchenyi István University in Mosonmagyaróvár, Széchenyi István University, Mosonmagyaróvár.
macher.gergely.zoltan@sze.hu

The aim of this paper is to examine the environmental resistance and chrysotile content of different asbestos cement products and to prove the relationship between these two factors by analytical results. The paper includes Fourier-transform infrared (FT-IR) spectroscopic analysis of asbestos cement pipes and asbestos cement products with corrugated and flat characteristics. The methodology of the study is based on FT-IR spectroscopy and general statistical approaches, and the results obtained are compared using correlation analysis. The background to the topic is that asbestos cement products are still widely used today despite their harmful effects on health, contrary to European Union asbestos-free targets. A damaged and eroded asbestos cement product loses several grams of asbestos and cement per year from its matrix structure, which is exacerbated by exposure to various environmental influences. The match rate of the chrysotile spectrum for analysed samples has been over 50 % in each case. In the number of measurements, the chrysotile detection rate was 7.64 % higher for degraded and eroded samples. In addition, in the samples exposed to environmental factors, the percentage variance was approximately 10 % or higher, with the exception of asbestos cement pipes. The results provide a basis for situational awareness options. Analytic practitioners, material science researchers, and analysts can use them.

1. Introduction

Asbestos minerals are naturally occurring fibrous silicate minerals that have been used in a variety of insulating, refractory (Mahini, 2005), and construction materials because of their durable fibres and ability to withstand high temperatures. Asbestos is a collective term used to describe a group of naturally found fibrous minerals, either serpentine (chrysotile) or amphibole (actinolite, amosite, anthophyllite, crocidolite and tremolite) in nature (Mahini, 2005).

Chrysotile comprises approximately 95 % of naturally occurring asbestos and is the most widely used industrially (Zholobenko et al., 2021). It also constitutes over 90 % of asbestos in water samples. All asbestos types, including chrysotile and amphiboles (crocidolite, amosite, tremolite), are recognised as carcinogenic to human lungs, although chrysotile may exhibit lower carcinogenic potency (Boffetta, 2018). In Europe, asbestos fibres are defined as those longer than 5 µm, wider than 3 µm, and with an aspect ratio greater than 3:1 (Dichicco et al., 2017).

A defining trait of the relevant type is that when chrysotile fibres are disrupted, such as through grinding, crushing, or exposure to water, the structure of the chrysotile fibre disintegrates, and individual fibres are created (Bernstein and Hoskins, 2006). Chrysotile asbestos consists of a fibrous $Mg_3(Si_2O_5)(OH)_4$ arranged in a 1:1 ratio, with interconnected silica tetrahedra attached to a brucite-like trioctahedral sheet. This structural arrangement results in curvature, giving rise to the fibrous nature of chrysotile (Mendelovici et al., 2001).

The worldwide industrial use of asbestos has expanded quickly over the past 150 y due to its valued properties, such as conductivity, fire resistance, and reinforcement in construction and manufacturing (Furuya et al., 2018).

Paper Received: 19 May 2024; Revised: 10 October 2024; Accepted: 1 December 2024
Please cite this article as: Macher G.Z., Torma A., Beke D., 2024, Analysing the Environmental Durability and Chrysotile Content of Asbestos Cement Products by FT-IR Spectroscopy, Chemical Engineering Transactions, 114, 709-714 DOI:10.3303/CET24114119

Asbestos is currently prohibited in over 50 countries globally, but it continues to be extracted and utilised in numerous nations. The primary asbestos producers encompass the Russian Federation, China, Kazakhstan, Brazil, Canada, Zimbabwe, and Colombia (Frassy et al., 2014). Presently, it is estimated that illnesses related to asbestos exposure (Castiblanco et al., 2020) cause approximately 255,000 deaths annually (Furuya et al., 2018). Despite the considerable knowledge of the risks linked to asbestos, this hazardous material continues to pose a significant health risk on a global scale, leading to numerous annual fatalities and making a substantial contribution to cancer-related deaths. However, there is ongoing and increasing worry regarding the potential public health hazard presented by still-in-use asbestos-containing materials (abbreviation: ACM) and their possible influence on future cases of asbestos-related illnesses.

Asbestos-containing materials include flat and corrugated sheets for roofing and compressed panels for cladding and facades in residential structures (Krówczyńska et al., 2014). Asbestos cement (abbreviation: AC) slates are made by mixing 10-20 % chrysotile with 80-90 % cement (Jeong et al., 2013). While asbestos cement sheets are used for roofing and cladding, they are inherently brittle and prone to cracking from minor impacts, repetitive stress, or faulty fastenings (Bassani et al., 2007).

Exposure to asbestos remains a significant concern during the natural weathering, repair, or demolition of buildings with asbestos shingles. The deterioration of these materials releases asbestos fibres into the air and water runoff, leading to indirect soil contamination (Hikuwai et al., 2023). A study by Suzuki et al. (2005) indicates that a corrugated asbestos cement sheet can release several grams of asbestos annually from its matrix structure.

Standard techniques for identifying asbestos currently include using polarised light microscopy for analysing bulk samples and phase-contrast microscopy for examining airborne fibres (Zholobenko et al., 2021). In addition, optical microscopy, scanning electron microscopy, transmission electron microscopy, and scanning transmission electron microscopy are frequently utilised for identifying asbestos fibres in environmental and human samples (Janik and Wrona, 2005). Infrared spectroscopy is also a valuable technique for swiftly identifying various kinds of clay minerals (Farro et al., 2023), as well as detecting asbestos contamination and assessing its qualitative parameters (Wu et al., 2021), the Fourier-transform infrared (abbreviation: FT-IR) method has also been utilised for the characterisation of asbestos fibres by analysing their distinct silicate vibrational bands in the range of 900-1,200 cm^{-1}.

Analysing asbestos content using FT-IR spectroscopy is not a new approach. Previous studies have employed this technique to determine the presence (Giacobbe et al., 2010) and identify different types of asbestos (Zholobenko et al., 2021). However, the role of environmental factors in the detectability of asbestos content beyond its release has received limited attention. Our research aims to address this gap, providing a valuable contribution to the field.

2. Materials and methods

The paper seeks to explore the detectability of chrysotile and the environmental durability of asbestos cement products using FT-IR spectroscopy. This section outlines the principal research methods and parameters that were required for this paper.

2.1 Experimental materials and samples

Three categories of asbestos-containing materials samples were utilised: asbestos cement slates with corrugated characteristics, AC-flat slates, and AC pipes. Two subgroups were formed within each category: one was exposed to environmental elements (rain, wind, temperature, sunshine) with five samples, and one was shielded from these factors with another five samples. Measurements were repeatedly conducted on each sample to determine conformity with chrysotile spectra. The success of these matches was recorded, and a statistical analysis was performed based on the top 10 matches. This methodology facilitated the evaluation of how environmental exposure affects the integrity of these products compared to their preserved counterparts. The environmentally exposed samples were actual products obtained from demolition and usage, sourced from various sites in Győr. The separated samples were obtained from the same area, specifically from locations where unused asbestos cement products had been stored. The collected samples were hermetically stored until the commencement of the experiments. The investigation of environmental factors is theoretical, as the experiment does not employ a stress or pressure chamber. The research is predicated on the assumption that the samples in use have been continuously exposed to environmental influences, including ageing, damage, sunlight, and rain, among others.

The initial batch of samples consists of undulating AC-corrugated sheets. These samples, approximately 30-40 y old, displayed visible signs of erosion and degradation due to environmental exposure alongside various deposits. The samples' labelling exposed to environmental influences ranged from AC-S-E-1 to AC-S-E-5, with

the mean value being labelled as AC-S-E-A. The labelling of separated samples ranged from AC-S-S-1 to AC-S-S-5, with the mean value being labelled as AC-S-S-A.
The second set comprises flat AC sheets with similar ageing (approximately 40 y), which exhibited comparable effects from environmental influences. The samples' labelling exposed to environmental influences ranged from AC-F-E-1 to AC-F-E-5, with the mean value being labelled as AC-F-E-A. The labelling of separated samples ranged from AC-F-S-1 to AC-F-S-5, with the mean value being labelled as AC-F-S-A.
Lastly, the third group includes AC pipes with characteristics consistent with those mentioned earlier. The samples' labelling exposed to environmental influences ranged from AC-P-E-1 to AC-P-E-5, with the mean value being labelled as AC-P-E-A. The labelling of separated samples ranged from AC-P-S-1 to AC-P-S-5, with the mean value being labelled as AC-P-S-A.
Before conducting the analysis, all three sets of samples were meticulously cleaned with distilled water and isopropyl alcohol to remove any possible impurities that could impact the findings. Following this step, they were then cut into consistent sizes before being encased in a multi-component epoxy resin.

2.2 Sample preparation

The specimens underwent a cleaning process using distilled water, which may have affected the investigation of environmental influences. Epoxy resin was poured into a mould according to specified dimensions to create discs measuring 30 x 5 mm. These discs were then ground with a grinder to achieve the desired size and subsequently placed on an instrument slide for examination. A 0.3-mm diamond crystal facing downwards allowed for the scanning of specific areas in the sample with infrared rays, resulting in the generation of an absorption map showing different spectra marked by varying colours. These coloured spectra corresponded to materials in the spectrum library, facilitating component identification.

2.3 Analysis of chrysotile asbestos by FT-IR spectroscopy

All types of asbestos display strong absorption in the 1,200-900 cm^{-1} and 600-300 cm^{-1} ranges. Qualitative identification of the asbestos type can be achieved by analysing the spectrum up to 200 cm^{-1}. Chrysotile exhibits distinct differences from amphiboles, as it demonstrates significant double hydroxyl groups at 3,693 cm^{-1} and 3,648 cm^{-1}, which are created between layers of hydroxyl groups situated within the primary silicate layers of the lattice.

2.4 Used instrument: PerkinElmer Spectrum 400

The PerkinElmer Spectrum 400 instrument from PerkinElmer, MA, U.S. was utilized for conducting measurements. The samples taken from fractured sections of asbestos cement products and collected scrap samples from the matrix material were analysed for their composition, particularly to identify any additives present. Following this step, these materials were placed on the object table of the spectroscope with a 0.3-mm diamond crystal facing upwards and secured in place with a force approximately equivalent to 100 N for analysis. These tests were sensitive to moisture and carbon dioxide. Thus, prior to each measurement, it was necessary to record the spectrum of the environmental background first in order for the instrument to differentiate between the true sample spectrum and background interference.

3. Results

3.1 Detection of chrysotile asbestos from asbestos cement corrugated sheets

The asbestos cement corrugated sheet samples have shown an average chrysotile asbestos detection rate of 0.747193 (1) under environmental exposure (Figure 1). The most significant match (0.810265 (1)) was recorded at the fourth measuring point for sample AC-S-E-5, while the lowest detection value (0.689985 (1)) was identified at the sixth measuring point for sample AC-S-E-4. Among the samples, the highest chrysotile detection average value (0.754616 (1)) was observed in the case of AC-S-E-2, whereas the lowest (0.742706 (1)) was found in AC-S-E-4. The asbestos cement corrugated sheet samples have shown an average chrysotile asbestos detection rate of 0.645437 (1) according to separated samples. The most significant match (0.701221 (1)) was recorded at the fourth measuring point for sample AC-S-S-4, while the lowest detection value (0.569845 (1)) was identified at the eighth measuring point for sample AC-S-S-5. Among the samples, the highest chrysotile detection average value (0.678396 (1)) was observed in the case of AC-S-S-1, whereas the lowest (0.605309 (1)) was found in AC-S-S-5. The overall rate of comparison within the entire sample series is 115.8 %. The average deviation from the mean is 0.00634 (1) for the exposed samples. In contrast, for the separated samples, this value is 0.00429 (1). Asbestos cement corrugated sheet samples exhibit a noticeable difference in spectral matching for chrysotile exposed to the environment (E), displaying higher values with an average of approximately 0.75 (1). Conversely, samples separated from environmental influence (S) demonstrate greater variability in values, with a central tendency around 0.65 (1). The analysed samples are based on an asbestos-cement matrix. Therefore, a wide peak was detected at 1,400 cm^{-1} in the spectra. This can be attributed to the presence of calcium silicate hydrate, calcium sulphate, and calcium carbonate found in the cement mixture, specifically related to the stretching vibrations of C-O bonds. Similar to the findings in Foresti et al. (2003), there are two specific regions of the spectrum that are noteworthy: the O-H stretching vibrations contribute to the 3,700-3,500 cm^{-1} region, while various lattice vibrations account for the bands observed in the 1,200-500 cm^{-1} region.

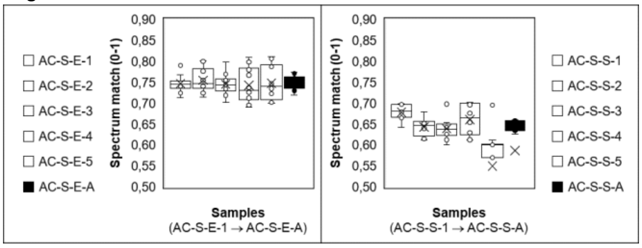


Figure 1: Comparative analysis of chrysotile detection for exposed (E) and separated (S) samples (asbestos cement corrugated sheets)

Asbestos cement corrugated sheet samples exhibit a noticeable difference in spectral matching for chrysotile exposed to the environment (E), displaying higher values with an average of approximately 0.75 (1). Conversely, samples separated from environmental influence (S) demonstrate greater variability in values, with a central tendency around 0.65 (1). The analysed samples are based on an asbestos-cement matrix. Therefore, a wide peak was detected at 1,400 cm^{-1} in the spectra. This can be attributed to the presence of calcium silicate hydrate, calcium sulphate, and calcium carbonate found in the cement mixture, specifically related to the stretching vibrations of C-O bonds.

3.2 Detection of chrysotile asbestos from asbestos cement flat sheets

The asbestos cement flat sheet samples have shown an average chrysotile asbestos detection rate of 0.753552 (1) under environmental exposure. The most significant match (0.820001 (1)) was recorded at the tenth measuring point for sample AC-F-E-3, while the lowest detection value (0.694962 (1)) was identified at the second measuring point for sample AC-F-E-1. Among the samples, the highest chrysotile detection average value (0.773262 (1)) was observed in the case of AC-F-E-4, and the lowest (0.723031 (1)) was found in AC-F-E-1. Figure 2 illustrates a contrast in the chrysotile detection results for asbestos cement flat sheets according to the exposed or separated character.

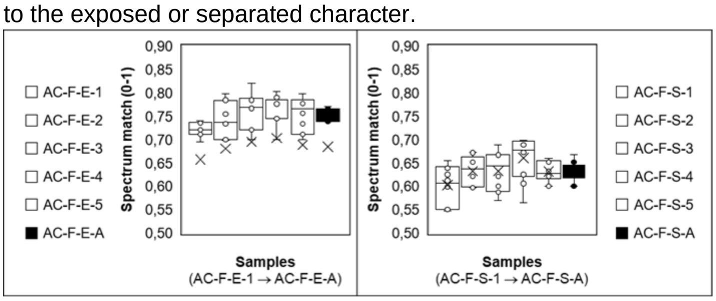


Figure 2: Comparative analysis of chrysotile detection for exposed (E) and separated (S) samples (asbestos cement flat sheets)

The asbestos cement flat sheet samples have shown an average chrysotile asbestos detection rate of 0.632309 (1) according to separated samples. The most significant match (0.698857 (1)) was recorded at the fifth measuring point for sample AC-F-S-4, while the lowest detection value (0.548113 (1)) was identified at the fourth measuring point for sample AC-F-S-1. Among the samples, the highest chrysotile detection average value (0.660611 (1)) was observed in the case of AC-F-S-4, whereas the lowest (0.602867 (1)) was found in AC-F-S-1. The overall rate of comparison within the entire sample series is 119.2 %. The average deviation from the mean is 0.00540 (1) for the exposed samples. In contrast, for the separated samples, this value is 0.00801 (1). The environmentally exposed (E) asbestos cement flat sheet samples exhibit a more noticeable difference in spectral matching for chrysotile. Chrysotile was more readily identifiable and had a higher agreement rate in the samples that were exposed to environmental effects. Conversely, samples separated from environmental influence (S) demonstrate greater variability in values, with a central tendency of less than 0.65 (1). Characteristic peaks of the chrysotile were identified at 3,700-3,500 cm^{-1} and 1,200-500 cm^{-1} ranges. The analysed samples are based on an asbestos-cement matrix; therefore, a wide peak was detected at 1,400 cm^{-1} in the spectra, similar to the asbestos cement corrugated sheets.

3.3 Detection of chrysotile asbestos from asbestos cement pipes

The asbestos cement pipe samples have shown an average chrysotile asbestos detection rate of 0.800607 (1) under environmental exposure. The most significant match (0.888831 (1)) was recorded at the fifth measuring point for sample AC-P-E-4, while the lowest detection value (0.742121 (1)) was identified at the eighth measuring point for sample AC-P-E-2. Among the samples, the highest chrysotile detection average value (0.811325 (1)) was observed in the case of AC-P-E-3, and the lowest (0.793972 (1)) was found in AC-P-E-4. Figure 3 illustrates a contrast in the chrysotile detection results for asbestos cement pipes according to the exposed or separated character. The asbestos cement pipe samples have shown an average chrysotile asbestos detection rate of 0.794369 (1) according to separated samples. The most significant match (0.865548 (1)) was recorded at the seventh measuring point for sample AC-P-S-2, while the lowest detection value (0.724580 (1)) was identified at the ninth measuring point for sample AC-P-S-3. Among the samples, the highest chrysotile detection average value (0.808812 (1)) was observed in the case of AC-P-S-2, whereas the lowest (0.775610 (1)) was found in AC-P-S-4. The overall rate of comparison within the entire sample series is 100.8 %. The average deviation from the mean is 0.00617 (1) for the exposed samples. In contrast, for the separated samples, this value is 0.00438 (1).

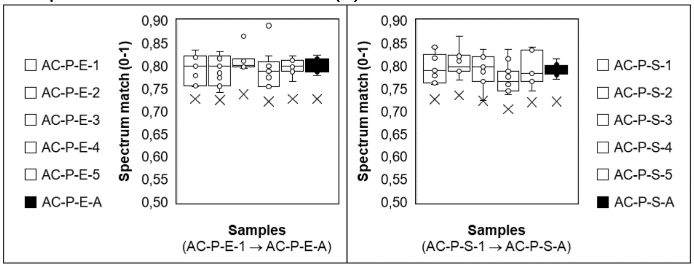


Figure 3: Comparative analysis of chrysotile detection for exposed (E) and separated (S) samples (asbestos cement pipes)

The environmentally exposed (E) asbestos cement pipe samples exhibit a more noticeable uniformity in spectral matching for chrysotile. Chrysotile was more easily detected and showed higher consistency in samples exposed to environmental conditions. The trend revolved around 0.80 (1) for the exposed samples (E) and about 0.79 (1) for the separated samples (S). The chrysotile displayed distinctive peaks at wavelengths of 3,700-3,500 cm^{-1} and 1,200-500 cm^{-1}. As the analysed samples are composed of an asbestos cement mixture, a broad peak was observed at 1,400 cm^{-1} in the spectra.

4. Conclusions

The research aims to assess the environmental impact and detectability of chrysotile asbestos in various asbestos cement products, such as flat and corrugated sheets and pipes, particularly focusing on samples shielded from environmental factors. It demonstrates that FT-IR spectroscopy is effective for identifying chrysotile asbestos by pinpointing characteristic spectral features. This method could become a standard for detecting chrysotile, given its accuracy and practicality. However, the presence of both chrysotile and the cement matrix complicates qualitative analysis. Notable variations were found between exposed (E) and separated (S) samples in corrugated and flat sheets, though less so in pipes.

The main findings of the research include: 1) FT-IR spectroscopy can be used for chrysotile detection and should be standardised in the future; 2) Chrysotile was more detectable in samples exposed to environmental conditions, with matching coefficients consistently around or above 0.75 (1); 3) Environmental impacts like erosion, degradation, product ageing, and corrosion contribute to the easier release and detectability of chrysotile asbestos.

The paper suggests that environmental conditions and ageing play a significant role in the release and detectability of chrysotile asbestos from asbestos cement products. Understanding these factors is essential for evaluating the environmental impact and health hazards linked to asbestos cement products and should be considered in forthcoming assessments. Continuing and expanding this analysis in the future is crucial, and its findings can benefit environmental and analytical researchers.

Nomenclature

FT-IR spectroscopy – Fourier-transform infrared spectroscopy
ACM – asbestos containing materials
AC – asbestos cement

References

Bassani C., Cavalli R.M., Cavalcante F., Cuomo V., Palombo A., Pascucci S., Pignatti S., 2007, Deterioration status of asbestos-cement roofing sheets assessed by analyzing hyperspectral data. Remote Sensing of Environment, 109(3), 361–378, DOI: 10.1016/j.rse.2007.01.014.

Bernstein D.M., Hoskins J.A., 2006, The health effects of chrysotile: Current perspective based upon recent data. Regulatory Toxicology and Pharmacology, 45(3), 252–264, DOI: 10.1016/j.yrtph.2006.04.008.

Boffetta P., 2018, Classic Epidemiology of Lung Cancer. In IASLC Thoracic Oncology (pp. 1-8.e3). Elsevier, DOI: 10.1016/B978-0-323-52357-8.00001-9.

Castiblanco Y., Perilla A., Velasquez P., Santis A., 2020, Design of Occupational Health and Safety Management System Based on ISO 45001, for a Company That Manufactures and Commercializes Low Voltage Electrical Boards. Chemical Engineering Transactions, 82, 91-96.

Dichicco M.C., De Bonis A., Mongelli G., Rizzo G., Sinisi R., 2017, μ-Raman spectroscopy and X-ray diffraction of asbestos' minerals for geo-environmental monitoring: The case of the southern Apennines natural sources. Applied Clay Science, 141, 292–299, DOI: 10.1016/j.clay.2017.02.024.

Farro N.W., Reyes W., Mendoza J.L., Veleva L., Quintana P., Azamar J.A., Aguilar D., 2023, Characterization by XRD and FTIR of Zeolite A and Zeolite X Obtained from Fly Ash. Chemical Engineering Transactions, 99, 679–684.

Frassy F., Candiani G., Rusmini M., Maianti P., Marchesi A., Nodari F., Via G., Albonico C., Gianinetto M., 2014, Mapping Asbestos-Cement Roofing with Hyperspectral Remote Sensing over a Large Mountain Region of the Italian Western Alps. Sensors, 14(9), 15900–15913, DOI: 10.3390/s140915900.

Foresti E., Gazzano M., Gualtieri A.F., Lesci I.G., Lunelli B., Pecchini G., Renna E., Roveri N., 2003, Determination of low levels of free fibres of chrysotile in contaminated soils by X-ray diffraction and FTIR spectroscopy. Analytical and Bioanalytical Chemistry, 376(5), 653–658, DOI: 10.1007/s00216-003-1965-3.

Furuya S., Chimed-Ochir O., Takahashi K., David A., Takala J., 2018, Global Asbestos Disaster. International Journal of Environmental Research and Public Health, 15(5), 1000, DOI: 10.3390/ijerph15051000.

Giacobbe C., Gualtieri A., Francesco Quartieri S., Rinaudo C., Allegrina M., Andreozzi G., Battista, 2010, Spectroscopic study of the product of thermal transformation of chrysotile-asbestos containing materials (ACM). European Journal of Mineralogy, 22(4), 535–546, DOI: 10.1127/0935-1221/2010/0022-2038.

Hikuwai M., Patorniti N., Vieira A., Frangioudakis Khatib G., Stewart R., 2023, Artificial Intelligence for the Detection of Asbestos Cement Roofing: An Investigation of Multi-Spectral Satellite Imagery and High-Resolution Aerial Imagery. Sustainability, 15(5), 4276, DOI: 10.3390/su15054276.

Janik H., Wrona M., 2005, ASBESTOS. In: Encyclopedia of Analytical Science, 141–148. Elsevier, Amsterdam, Netherlands, ISBN: 978-0-12-369397-6, DOI: 10.1016/B0-12-369397-7/00023-6.

Jeong J.-W., Cho S., Park G.-T., Lee S.-J., 2013, Health Risk Assessment and Evaluation of Asbestos Release from Asbestos-cement Slate Roofing Buildings in Busan. Journal of Environmental Science International, 22(12), 1579–1587, DOI: 10.5322/JESI.2013.22.12.1579.

Krówczyńska M., Wilk E., Zagajewski B., 2014, The Electronic Spatial Information System – tools for the monitoring of asbestos in Poland. Miscellanea Geographica, 18(2), 59–64, DOI: 10.2478/mgrsd-2014-0019.

Mahini X., 2005, Asbestos. In: Encyclopedia of Toxicology, 179–182). ElsevierAmsterdam, Netherlands, DOI: 10.1016/B0-12-369400-0/00089-2.

Mendelovici E., Frost R.L., Kloprogge J.T., 2001, Modification of Chrysotile Surface by Organosilanes: An IR–Photoacoustic Spectroscopy Study. Journal of Colloid and Interface Science, 238(2), 273–278, DOI: 10.1006/jcis.2001.7524.

Suzuki Y., Yuen S.R., Ashley R., 2005, Short, thin asbestos fibers contribute to the development of human malignant mesothelioma: pathological evidence. International Journal of Hygiene and Environmental Health, 208(3), 201–210, DOI: 10.1016/j.ijheh.2005.01.015.

Wu S., He M., Yang M., Zhang B., Wang F., Li Q., 2021, Near-Infrared Spectroscopy Study of Serpentine Minerals and Assignment of the OH Group. Crystals, 11(9), 1130, DOI: 10.3390/cryst11091130.

Zholobenko V., Rutten F., Zholobenko A., Holmes A., 2021, In situ spectroscopic identification of the six types of asbestos. Journal of Hazardous Materials, 403, 123951, DOI: 10.1016/j.jhazmat.2020.123951.

CHEMICAL ENGINEERING TRANSACTIONS

VOL. 107, 2023

A publication of AIDIC
The Italian Association of Chemical Engineering
Online at www.cetjournal.it

Guest Editors: Petar S. Varbanov, Bohong Wang, Petro Kapustenko

ISBN 979-12-81206-07-6; **ISSN** 2283-9216

DOI: 10.3303/CET23107061

Integrated Analysis and Assessment of the Hungarian Regions in Terms of Health Problems and Waste Management Challenges caused by Asbestos

Gergely Zoltán Macher[a,*], Dóra Beke[b], András Torma[a]

[a]Department of Applied Sustainability, Albert Kázmér Faculty of Agricultural and Food Sciences, Széchenyi István University, Győr, Hungary
[b]Department of Plant Sciences, Albert Kázmér Faculty of Agricultural and Food Sciences, Széchenyi István University, Mosonmagyaróvár, Hungary
macher.gergely.zoltan@sze.hu

Nowadays, more information is available regarding the health problems and waste management challenges caused by asbestos, and the research directions that focus on alternative agents, transport processes and mobilization routes of asbestos have gained ground. Only in recent years has the published research on the subject begun to deal more intensively with the development of evaluation methods that would ensure the consistent numerical qualification of individual territorial units. This paper examines the exposure of certain regions of Hungary along the lines of asbestos-related waste management and health aspects. After that, the individual relationships were identified and explored based on the specific patterns of the formed groups. The methodology of this paper is the calculation of an internationally applied integral index, the advantage of which is that it enables the comparison of numerical values with different dimensions. The focus of research is not the development of a new scientific methodology, but rather an exploration of the situation and regional comparability of asbestos exposure that is much more multidisciplinary, complex and multidimensional than the previous viewpoints. The value of the calculated integral index was 0.310±0.155 in 2005, while 0.339±0.170 in 2020. The rate of change in the value of the asbestos involvement and exposure integral index shows an increase of +28.5 % between 2005 and 2020. Based on the results, there are significant differences between the individual regions of Hungary along the values of the calculated multidimensional integral indices.

1. Introduction

How does the asbestos prevalence of a settlement, a region or a country, the population's exposure to asbestos over time, affect the risk ratio of malignant tumors and the opportunities for the waste hierarchy and the circular economy to gain ground? The approach to the problem cannot be defined only along the lines of waste management, environmental monitoring and epidemiological data, a complex, interdisciplinary approach to these aspects is important. At the same time, asbestos is an area of environmental protection for which there is neither environmental monitoring, nor permanent background pollution testing, nor public risk assessment (Gray et al., 2016). The problem is a given, the risks are known, but at the same time, asbestos contamination can still be considered a not analyzed environmental protection topic. Asbestos is a hydrated silicate of magnesium, iron, calcium and sodium, divided into two fiber groups, serpentines (chrysotile) and amphiboles (amosite, crocidolite, tremolite, anthophyllite and actinolite (Castro et al., 2003). The characteristic properties of asbestos are high fire resistance and low temperature and microbiological resistance to contact with bacteria or fungi (Kusiorowski et al., 2023). Important advantages of asbestos fibers include high tensile strength, abrasion resistance (Iwaszko et al., 2018), and resistance to acids and alkalis (Malinconico et al., 2022). Good thermal insulation, sound absorption (Park et al., 2012), and high electrical they are characterized by resistance (Santana et al., 2023). Most asbestos minerals also have high mechanical strength (Virta, 2005). Only three of the asbestos minerals are widely used commercially, especially in the construction industry. These include chrysotile (white asbestos), crocidolite (blue asbestos) and amosite (brown asbestos). Chrysotile accounts for

Paper Received: 15 April 2023; Revised: 29 June 2023; Accepted: 26 October 2023
Please cite this article as: Macher G.Z., Beke D., Torma A., 2023, Integrated Analysis and Assessment of the Hungarian Regions in Terms of Health Problems and Waste Management Challenges caused by Asbestos, Chemical Engineering Transactions, 107, 361-366 DOI:10.3303/CET23107061

90-95% of asbestos mined in the world (L. Frank and Joshi, 2014). Asbestos and asbestos cement products now have a historical legacy (Bolan et al., 2023) in the built environment worldwide (Obmiński, 2021). Their use has been typical since the era of industrialization (Janela and Pereira, 2016). More widely, these products have been used in the construction industry throughout history, mainly as insulators and as structural stabilizers for various materials such as various cement or plastic products (Virta, 2005), hence today more than 3,000 different commercial products contain or may contain asbestos in varying concentrations. (Harris and Kahwa, 2003). More than 50.0 % of asbestos sold worldwide was used in Europe between 1920 and 2000 (Paglietti et al., 2016). According to Ramazzini (2010), more than 90.0 % of the asbestos used was used in the form of various asbestos cement products, sheets, roofing materials and pipes. The first asbestos-related health cases were reported in the 1950s, mainly among miners and factory workers dealing with asbestos (Jung et al., 2021). A study conducted in the 1960s identified asbestos as a carcinogen and stated that the health diseases caused by it cause malignant mesothelioma and lung cancer with a latency period of approximately 20-50 years (Mossman et al., 1996). As a result of these health effects, the use of asbestos has gradually declined since the 1970s, and some countries have banned or restricted its use by law since the 1990s (Bahk et al., 2013). The greatest danger and risk are fibers penetrating the alveoli of the lower respiratory tract. These are so-called respirable fibers, fibers with a diameter of less than 3 μm and a length of more than 5 μm (WHO, 1986).

In Hungary, the new use of asbestos has been prohibited since 2005, but the regulation does not cover the prohibition, restriction or regulation of the further use of previously installed products. Pursuant to the European directive (148/2009), the maximum permissible level of asbestos in the air at workplaces is 100 f/l (fibers/liter) (WHO, 1986). According to Li et al. (2014), all asbestos in use in the world today is chrysotile, also known as white asbestos. As the main health risk, scattered asbestos insulation came into focus, for which an exemption and removal program was launched several times. At the same time, the asbestos cement products used in buildings, on the other hand, were not considered to be directly harmful to health (Bornemann and Hildebrandt, 1986), since the asbestos fibers are bound by a strong layer of cement, but at the same time, as a result of weather conditions and erosion and degradation over time, asbestos fibers are released into the surrounding atmosphere, which can reach up to 3.00-5.00 g/m^2 can be (Brown, 1987). At the same time, asbestos fibers can also be released directly as a result of much occupational safety (Gyenes and Wood, 2016) and household accidents (pipe breaks). The verification of the presence of asbestos should be conducted, where discovered its presence, there should be a clearance, or, in the best of conditions, continuous monitoring (Cecchini et al., 2017). Consequently, the environmental and health problems associated with asbestos cement in these buildings are potentially strongly associated with asbestos cement roofing (Zhang et al., 2022). Malignant mesothelioma has been diagnosed in several cases in residents of properties using asbestos cement roofing (Jung et al., 2006). A survey on the use of asbestos-cement roofing in areas with poor living conditions is therefore essential to manage the associated risks (Zhang et al., 2022). At the same time, such a series of surveys is extremely expensive, given their widespread use.

Several methods can be used to determine asbestos exposure and involvement, which also have different advantages and disadvantages. The first is conducting on-site field visits and surveys in order to count the number of properties and buildings with asbestos-cement roofing (Kim et al., 2010), thus establishing the involvement of a given area (Zhang et al., 2022). At the same time, this process is extremely time- and human-resource-intensive. The second option is modeling (Kim et al., 2016); based on the type (Zhang et al., 2016) and area data on roofing found in the building register (Zhang et al., 2022). At the same time, the data in the register is not public, and in many cases, the data in the register does not reflect reality. Monitoring can also be done using drones, which are based on the first method (Lee et al., 2016), also requiring high resources. In recent years, situational studies have also been published that established the involvement of a given area or settlement through satellite images, with particular regard to the state of erosion of individual asbestos cement sheets. Management and risk assessment tools can also be applied, which infer the involvement of certain territorial units through the continuously collected secondary data. Such data may include waste generation, waste management, health and discharge data. At the same time, this process is greatly influenced by the quality and quantity parameters of individual input data, as well as their actual existence. For this reason, in this study, the impact assessment of Hungary's NUTS-2 regions was chosen as a basis. The purpose of this research is to use an alternative indexing method that relies on currently available indicators directly related to asbestos. The results may provide insight into an alternative asbestos involvement assessment approach based on the applied health and waste generation values.

2. Materials and methods

The aim of the research is to calculate an alternative index value to determine the asbestos involvement of Hungarian regions (NUTS-2). The used taxonomic index formation method is a procedure suitable for the uniform handling of multidimensional data, which results in a dimensionless value between 0.00 and 1.00.

During the calculation, indicators were used that are directly related to the topic of asbestos and asbestos cement. However, it must be emphasized that the amount of public, directly accessible data in this regard is minimal, and extremely insignificant. The range of indicators used consisted of the following: mesothelioma (*Mesothelioma malignum*) incidence values collected and treated by the Hungarian Cancer Registry (2023); the data of the Waste Management System Module of the National Environmental Protection Information System (2023) regarding regional waste generation along the waste types 170605 - asbestos-containing construction material and 170601 - asbestos-containing insulation material. During the analysis, waste disposal data were not used, due to the regional differences in distribution and waste management. The examined period is between 2005 and 2020, and the examined years are 2005, 2010, 2015 and 2020. The elements of the taxonomic assessment used to determine the integral index of asbestos involvement were the same as Oliinyk et al. (2023) with the method used.

Eq(1) Standardization of initial data with different units and dimensions in order to reduce them to a single metric scale, using the following formula: Eq(1), where: Z_{ij} is the standardized value of the i-th indicator of the j-th NUTS-2 region ($i = \overline{1,n};\ j = \overline{1,m}$); x_{ij} the arithmetic mean value of the i-th indicator of the j-th NUTS-2 region; σ_i is the standard deviation value of the i-th indicator.

$$Z_{ij} = \frac{x_{ij} - \bar{x}_{ij}}{\sigma_i}, \tag{1}$$

Eq(2) Creation of reference point Z_{0i} ($Z_{01}, Z_{02}, \ldots Z_m$), which means comparing the values of the NUTS-2 regions to the maximum value of the given value series.

$$Z_{0i} = \max z'_{ij}\ i \in I, \tag{2}$$

Where: I is the set of indicators.

Eq(3) Determination of the Euclidean distance, which shows the distance of the indicators relative to a given reference point, where: d_{0i} is the Euclidean distance of the indicator value from the reference point.

$$d_{0i} = \sqrt{\sum_{i,j=1}^{n,m} (Z_{ij} - Z_{0i})^2}, \tag{3}$$

Eq(4) Calculation of the taxonomic index of the asbestos involvement integral index, which reflects the alternative measure of asbestos involvement of the Hungarian NUTS-2 regions (K_i) with formula (Eq(4)), where: K_i is the picture of the given level of asbestos involvement in each Hungarian NUTS-2 region; $\overline{d_0}$ is the arithmetic mean of the corresponding Euclidean distance; σ_0 is the standard deviation of the corresponding Euclidean distance.

$$K_i = 1 - \frac{d_{0i}}{d_0},\ d_0 = \overline{d_0} + 2 \cdot \sigma_0, \sigma_0 = \sqrt{\frac{\sum (d_{0i} - \overline{d_0})^2}{n}}, \tag{4}$$

3. Results

There is no uniform methodology available for the interdisciplinary assessment of asbestos involvement, so the analysis summarized in this paper was also based on a pre-selected group of indicators. After sorting the data, it was categorized and standardized. Through the data formed into a unified system, the Euclidean distance values were also determined which was followed by the calculation of the alternative, dimensionless asbestos involvement index values (Table 1). According to the primary and secondary data obtained by the taxonomic analysis, the Hungarian NUTS-2 regions were characterized by a varied but at the same time stable situation during the four relevant years of the examined period. During the analysis, the value of the calculated index between 2005 and 2020 was a significant increase. The average value of the calculated index value was 0.3098 in 2005, 0.2377 in 2010, 0.2612 in 2015, and 0.3389 in 2020, which represents a low level at the national level, but at the same time, it should be emphasized that a growth rate was experienced in the examined period. The reason for this is the fluctuating but ever-increasing tendentious increase in the incidence of mesothelioma, as well as the increased amount of asbestos-containing waste generated in certain regions. The average rate of change between 2005 and 2020 was a +28.5 % increase, which in the first third of the examined period (between 2005 and 2010) was a -18.0 % decrease, in the second third (between 2010 and 2015) +24.5 % increase, while in the last third (between 2015 and 2020) there was already a +33.2 % increase.

Table 1: Results of Euclidean distance and integral index calculation

Code	NUTS-2	Euclidean distance: 2005	Euclidean distance: 2010	Euclidean distance: 2015	Euclidean distance: 2020	K_i: 2005	K_i: 2010	K_i: 2015	K_i: 2020
HU11	Budapest	2.72	3.46	2.77	1.84	0.512	0.395	0.474	0.615
HU12	Pest	3.97	3.74	4.46	3.74	0.286	0.346	0.153	0.218
HU21	Central Transdanubia	2.61	4.06	2.92	2.45	0.530	0.289	0.446	0.487
HU22	Western Transdanubia	4.93	5.20	4.21	3.24	0.114	0.090	0.200	0.322
HU23	Southern Transdanubia	4.44	5.01	4.52	3.68	0.202	0.123	0.140	0.231
HU31	Northern Hungary	4.47	4.90	4.02	3.96	0.197	0.143	0.235	0.172
HU32	Northern Great Plain	4.28	3.75	3.82	2.44	0.231	0.344	0.274	0.489
HU33	Southern Great Plain	3.31	4.74	4.38	3.93	0.406	0.171	0.167	0.177

Data from: own calculated, own edited

The results of the analysis showed that, based on the trend between 2005 and 2020, there are significant regional differences between the individual Hungarian NUTS-2. During the entire study period, Western Transdanubia (HU22) showed the largest increase (+182.9 %). Similarly, the Northern Great Plain (HU32) region showed an increase of over 100.0 % (+111.5 %). In contrast, the biggest decline (-56.3 %) was experienced in the case of the Southern Great Plain (HU33). An increase in four of the eight regions (Budapest-HU11, Western Transdanubia-HU22, Southern Transdanubia-HU23, Northern Great Plain-HU32), in four (Pest-HU12, Central Transdanubia-HU21, Northern Hungary-HU31, Southern Great Plain-HU33) and a reduction was experienced. In the first third of the examined period (2005-2010), the largest increase could be detected in the case of the Northern Great Plain (+48.7 %), while the largest decrease was observed in the examination of the Southern Great Plain (-57.9 %). In the second third (2010-2015), Western Transdanubia (+122.2 %) showed the largest increase. The largest decline was experienced in Pest (-55.9 %). Against all this, in the last third (2015-2020), Northern Great Plain (+78.6 %) and Northern Hungary -27.0 % were the two endpoints of the change interval.

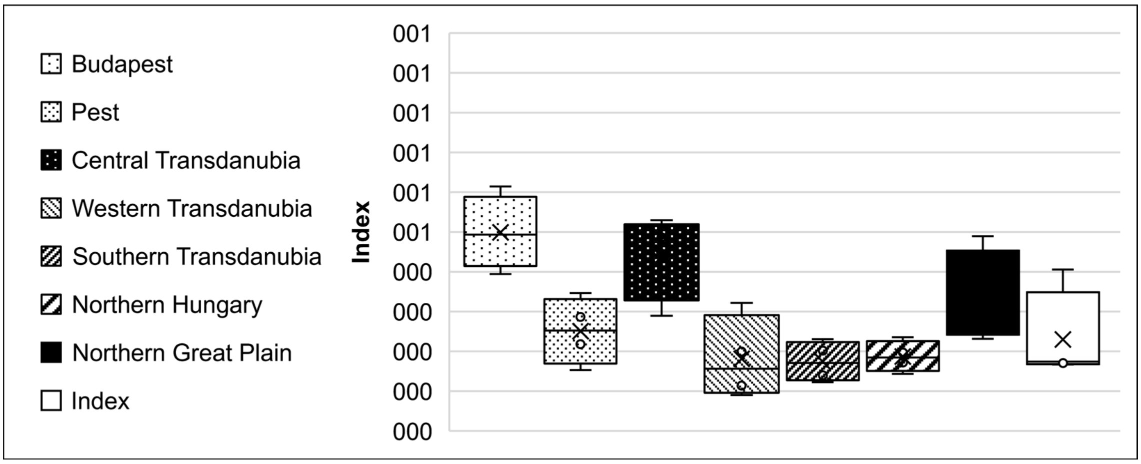


Figure 1: Comparison of calculated asbestos involvement index values in NUTS-2 regions. Data from: own calculated, own edited

Based on the obtained results, it can be concluded that the involvement of asbestos can also be characterized by multidimensional analysis, and the phenomenon can be quantified along the existing indicators. At the same time, it is important to emphasize that if the range of available indicators expands, the values can be continuously refined. If the above results and the inequalities of their territorial distribution were taken into account, Hungary could be classified as moderately affected. The diagram in Figure 1 compares the index values characteristic of each region through the quartiles for the period between 2005 and 2020. It can be concluded that a wide interval was experienced in some regions, with significantly higher index values (Budapest, Central Transdanubia, Northern Great Plain). On the other hand, this is not fulfilled in the case of two regions, the index value showed a low variation spectrum, typically with low values below 0.30 (Southern Transdanubia, Northern Hungary). Pest and Western Transdanubia had a low, but at the same time significantly variable index value.

4. Conclusions

This paper examines the situational picture of asbestos involvement along different indicators in order to highlight regional disparities in mesothelioma incidence and asbestos-containing waste generation. Asbestos involvement can be approached in many ways, but at the same time, there is no uniform, large-scale risk assessment or situational assessment procedure available. The simplest way of assessment is to perform an analysis based on indicators that are directly related to the topic of asbestos. The problem is that the scope of such data and indicators is extremely narrow, and in many cases incomplete. This data can also be health or waste management data. *Mesothelioma malignum*, as a malignant neoplastic disease, is the primary cause of persistent, long-term involvement with asbestos, and its incubation period can be 10-40 years. The other two investigated indicators are the amount of waste generated by the different asbestos cement products and the dusting, dispersed asbestos insulation. During the analysis, a taxonomic method was used whose decisive part element is the number and quality of the indicators used. Based on the results, it can be confirmed that asbestos involvement is indeed a detectable phenomenon following an increasing trend in the NUTS-2 regions of Hungary during the period under review. The results show that there are significant differences between the individual regions, while in some regions the level of asbestos involvement is high, while in others it is typically low. During the entire study period, Western Transdanubia (HU22) showed the largest increase (+182.9 %) in the value of the complex index, while in contrast, the biggest decline (-56.3 %) was experienced in the case of the Southern Great Plain (HU33). The average value of the calculated integral index was 0.310 ± 0.155 at the national level in 2005. In contrast, this value was 0.238 ± 0.119 in 2010, 0.261 ± 0.131 in 2015 and 0.339 ± 0.170 in 2020. The rate of change in the value of the integral index shows an increase of +28.5 % between 2005 and 2020, based on the average of regional values. The research should be continued, the exploration of the asbestos involvement situation is a gap-filling, current topic, which unfortunately depends to a large extent on the quality, quantity and availability of the collected indicators and data. The current results are applicable to environmental scientists and can serve as a starting point for future research.

References

Bahk J., Choi Y., Lim S., Paek D., 2013, Why some, but not all, countries have banned asbestos, International Journal of Occupational and Environmental Health, 19(2), 127–135.

Bolan S., Kempton L., McCarthy T., Wijesekara H., Piyathilake U., Jasemizad T., Padhye L. P., Zhang T., Rinklebe J., Wang H., Kirkham M.B., Siddique K.H.M., Bolan N., 2023, Sustainable management of hazardous asbestos-containing materials: Containment, stabilization and inertization, Science of The Total Environment, 881, 163456.

Bornemann P., Hildebrandt U., 1986, Problem Concerning Environmental Dust Evidence by Weathered-off Asbestos-cement, Dust, Keeping the air clean, 46(11), 487–489 (in German).

Brown S.K., 1987, Asbestos Exposure During Renovation and Demolition of Asbestos-Cement Clad Buildings, American Industrial Hygiene Association Journal, 48(5), 478–486.

Castro H., Giannasi F., Novello C., 2003, The struggle to ban asbestos in the Americas: a public health issue, Science & Public Health , 8(4), 903–911 (in Portuguese).

Cecchini M., Colantoni A., Monarca D., Cossio F., Riccioni S., 2017, Survey on the Status of Enforcement of European Directives on Health and Safety at Work in some Farms of Central Italy, Chemical Engineering Transactions, 58, 103-108.

Gray C., Carey R.N., Reid A., 2016, Current and future risks of asbestos exposure in the Australian community, International Journal of Occupational and Environmental Health, 22(4), 292–299.

Gyenes Z., Wood M.H., 2016, Lessons Learned from Major Accidents Relating to Ageing of Chemical Plants, Chemical Engineering Transactions, 48, 733-738.

Harris L., Kahwa I., 2003, Asbestos: old foe in 21st century developing countries, The Science of The Total Environment, 307(1–3), 1–9.

Hungarian Cancer Registry, Eds.: Dr. István Kenessey, 2023, Hungarian Cancer Registry and Center for Biostatistcs, Budapest (in Hungarian).

Iwaszko J., Zawada A., Przerada I., Lubas, M., 2018, Structural and microstructural aspects of asbestos-cement waste vitrification, Spectrochimica Acta Part A: Molecular and Biomolecular Spectroscopy, 195, 95–102.

Janela J.M.E.M., Pereira P.J.S., 2016, History of asbestos in the world and in Portugal Cultura, Space & Memory (CEM) No. 7, Environmental Citizenship and Participation in the Open University, 193-206 (in Portuguese).

Jung H.-S., Jang J., Park E.-K., Cho Y., Lee J.-C., Kim, H., 2021, Changes in concentrations and characteristics of asbestos fibers dispersed from corrugated asbestos cement sheets due to stabilizer treatment, Journal of Environmental Management, 285, 112110.

Jung S.H., Kim H.R., Koh S.B., Yong S.J., Choi B.S., Ahn Y.S., Park T.I., Chung M.J., Kim Y.M., Song J.S., Chung Y.K., Myung J.P., 2006, Epidemiologic Characteristics Revealed with a Malignant Mesothelioma Surveillance System in Korea, Korean Journal of Occupational and Environmental Medicine, 18(1), 46.

Kim Y.-C., Hong W.-H., Zhang Y.-L., Son B.-H., Seo Y.-K., Choi, J.-H., 2016, Estimating the Additional Greenhouse Gas Emissions in Korea: Focused on Demolition of Asbestos Containing Materials in Building, International Journal of Environmental Research and Public Health, 13(9), 902.

Kim Y.C., Son B.H., Hong W.H., 2010, A Research on the actual conditions of roofing asbestos use and residents consciousness in rural areas, Journal of the Architectural Institute of Korea Planning & Design, 26(11), 363–370.

Kusiorowski R., Lipowska B., Kujawa M., Gerle, A., 2023, Problem of asbestos-containing wastes in Poland, Cleaner Waste Systems, 4, 100085.

Frank L., Joshi T.K., 2014, The Global Spread of Asbestos, Annals of Global Health, 80(4), 257-262.

Lee S., van Riessen A., Chon C.-M., Kang N.-H., Jou H.-T., Kim, Y.-J., 2016, Impact of activator type on the immobilisation of lead in fly ash-based geopolymer, Journal of Hazardous Materials, 305, 59–66.

Li J., Dong Q., Yu K., Liu L., 2014, Asbestos and asbestos waste management in the Asian-Pacific region: trends, challenges and solutions, Journal of Cleaner Production, 81, 218–226.

Malinconico S., Paglietti F., Serranti S., Bonifazi G., Lonigro, I., 2022, Asbestos in soil and water: A review of analytical techniques and methods, Journal of Hazardous Materials, 436, 129083.

Mossman B.T., Kamp D.W., Weitzman S.A., 1996, Mechanisms of Carcinogenesis and Clinical Features of Asbestos-Associated Cancers, Cancer Investigation, 14(5), 466–480.

National Environmental Protection Information System, 2023, Waste Management System Module, Ministry of Energy, Budapest (in Hungarian).

Obmiński A., 2021, Asbestos waste recycling using the microwave technique – Benefits and risks, Environmental Nanotechnology, Monitoring & Management, 16, 100577.

Oliinyk O., Mishchuk H., Vasa L., Kozma K., 2023, Social Responsibility: Opportunities for Integral Assessment and Analysis of Connections with Business Innovation, Sustainability, 15(6), 5608.

Paglietti F., Malinconico S., della Staffa B.C., Bellagamba S., De Simone P., 2016, Classification and management of asbestos-containing waste: European legislation and the Italian experience, Waste Management, 50, 130–150.

Park E.-K., Takahashi K., Jiang Y., Movahed M., Kameda T., 2012, Elimination of asbestos use and asbestos-related diseases: An unfinished story, Cancer Science, 103(10), 1751–1755.

Ramazzini C., 2010, Asbestos Is Still With Us: Repeat Call for a Universal Ban, Archives of Environmental & Occupational Health, 65(3), 121–126.

Santana H.A., Júnior N.S.A., Carneiro G.O., Ribeiro D.V., Cilla M.S., Dias C.M.R., 2023, Asbestos-cement wastes as supplementary precursors of NaOH-activated binders, Construction and Building Materials, 364, 129921.

Virta R.L., 2005, Mineral commodity profiles- Asbestos: U.S. Geological Survey Circular, 1255–KK, 56 p.

WHO., 1986, Asbestos and other natural mineral fibres, World Health Organization.

Zhang Y.-L., Hong W.-H., Kim Y.-C., 2022, Methods for assessing asbestos-containing roofing slate distribution in an area with poor dwelling conditions, Journal of Cleaner Production, 358, 132032.

Zhang Y.-L., Kim Y.-C., Hong W.-H., 2016, Visualizing distribution of naturally discharged asbestos fibers in Korea through analysis of thickness changes in asbestos cement slates, Journal of Cleaner Production, 112, 607–619.