

Article

Formaldehyde Emissions from Wood Materials and Their Impact on Indoor Air Quality

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Abstract

Wood-based materials are a significant source of formaldehyde (HCHO), which is a volatile organic compound (VOC) classified as a human carcinogen that has adverse effects on the indoor air quality (IAQ) and health. This study evaluates HCHO emissions from six materials commonly used indoors, including medium-density fiberboard, melamine, pine boards, birch boards, oak boards, and alder boards, using the desiccator method in accordance with ASTM D5582. The experimental results were integrated with an indoor air model to estimate exposure under representative residential conditions. The measured emissions ranged from 0.235 ± 0.01 to 1.023 ± 0.10 mg m⁻³, with composite materials exhibiting the highest values. The risk analysis showed that all samples exceeded international reference values, with the greatest concern arising in scenarios of chronic exposure. The modeling indicated that indoor air concentrations depend heavily on material load and ventilation, reaching values as high as 0.155 mg m⁻³ under conditions of low air exchange. Overall, the results show that material composition, installed quantity, and ventilation conditions are key factors influencing indoor HCHO concentrations. This study offers an integrated approach that combines experimental measurement and exposure estimation, contributing to informed material selection and the assessment of their impact on IAQ.

Keywords: formaldehyde; indoor air quality; health risk; wood materials; wood-based panels

1. Introduction

Building materials and furniture are a significant source of volatile organic compounds (VOCs) that directly affect indoor air quality (IAQ) [1–3]. Among these compounds, formaldehyde (HCHO) stands out for its high toxicological significance, as it is classified as a human carcinogen (Group 1) by the International Agency for Research on Cancer (IARC), in addition to being a potent irritant to the mucous membranes and the respiratory system [4–6]. Given that the population spends most of its time indoors, the presence of HCHO in indoor environments represents a significant public health problem, with implications for morbidity associated with environmental exposure [7,8].

In this context, wood-based products are identified as one of the main sources of indoor HCHO emissions [9]. Although solid wood releases relatively low amounts of HCHO due to the thermal degradation of its components, the most significant emissions come from the urea–HCHO resins used in the manufacture of engineered wood panels [10,11].



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Materials such as medium-density fiberboard (MDF), particleboard, and plywood can exhibit varying emissions depending on the type of adhesive, the product formulation, and the surface finishes [12]. This variability highlights the need to systematically characterize their emission behavior to identify materials with a lower potential impact on IAQ.

Various standardized methods have been developed to quantify HCHO emissions from materials, including the desiccator method (ASTM D5582), which is widely used due to its simplicity and applicability in quality control [13,14]. Previous studies have investigated HCHO emissions from solid wood and different categories of engineered wood products [15–17]. However, these investigations have often focused on specific wood species, individual product categories, or the effects of particular environmental and manufacturing parameters [12,18]. Consequently, simultaneous comparisons of solid wood species and engineered wood panels under a consistent experimental framework remain limited. Differences in analytical methods and experimental conditions among studies further complicate direct comparisons of reported emission values [19,20].

Moreover, laboratory-based emission measurements do not directly represent the HCHO concentrations that may develop in indoor environments, where material loading and ventilation strongly influence contaminant accumulation [21,22]. Although these parameters are recognized as important determinants of indoor HCHO concentrations, quantitative assessments considering multiple combinations of material load and ventilation conditions remain limited [23]. Therefore, a gap persists between the experimental characterization of HCHO emissions from wood materials and the estimation of their potential contribution to indoor HCHO levels under representative residential conditions.

The objective of this study is to comparatively evaluate HCHO emissions from six commonly used interior wood materials, including solid wood species and engineered wood panels, under a consistent experimental framework. In addition, their potential contribution to indoor HCHO concentrations is quantitatively estimated under different material-loading and ventilation scenarios representative of residential environments. By integrating the simultaneous comparison of multiple wood materials with scenario-based indoor concentration estimation and toxicological contextualization, this study provides a framework for bridging laboratory emission measurements and the assessment of their potential impact on indoor air quality.

2. Materials and Methods

2.1. Wood

Wood is one of the most widely used materials in the construction industry. It is used to produce plywood and solid wood, which are the most common forms found in carpentry. Each has its own specific characteristics and properties, which is why their specific uses are also noteworthy.

Plywood consists of layers of wood glued together in a crisscross pattern; it is known for being strong and lightweight. Among these, pine plywood stands out and is commonly used to construct the frames of furniture. MDF is made of compressed wood fibers and synthetic resins, which makes it denser and heavier. Melamine is a decorative coating applied to a board with a plywood or MDF core, ideal for furniture fronts. In contrast, solid woods are sourced directly from the tree trunk and are more durable; they tend to be more expensive and are used in structures and the manufacture of high-end furniture, standing out for their unique and natural beauty.

The wood types selected for this study were based on market share and the extent of their use in interior applications: pine boards, MDF, birch boards, melamine, oak, and alder (Table 1).

Table 1. Wood-based panel features.

Wood-Based Panel	Thickness (mm)	Adhesive	Finish	Storage (Weeks)
Alder	25.40	-	Mate/Natural	2
Oak	22.23	-	Mate/Natural	12
Melamine	19.05	Urea-HCHO	Mate	2
Birch	15.88	Urea-HCHO	Mate/Natural	12
MDF	19.05	Urea-HCHO	High-gloss UV varnish and smooth melamine	96
Pine	15.88	Urea-HCHO	Mate/Natural	16

2.2. Sample Obtention and Preparation

The samples were obtained from panels of the material to be studied. For each thickness evaluated, eight test specimens measuring 70 ± 2 mm $\times$ 127 ± 2 mm were prepared. The cuts were made so that the samples adequately represented different areas of the panel. The freshly cut edges were positioned at least 25 mm from the original edges of the panel to avoid any effects associated with the manufacturing process.

The sawdust and any loose debris were then removed. The edges and ends of each specimen were sealed by immersing them in paraffin wax, applying at least two coats to ensure a proper seal. The wax did not cover more than 5 mm of the main faces around the perimeter, so as not to significantly alter the effective emission area.

The sealed samples were conditioned for 7 days (± 4 h) at 24 ± 1.7 °C and $50 \pm 10\%$ relative humidity. During conditioning, the test tubes were placed upright and spaced apart to allow air to circulate freely over all their surfaces. The HCHO concentration in the conditioning environment was maintained below 0.1 ppm (Temtop LKC-1000S, San Jose, CA, USA) within a 30 cm radius around the samples.

2.3. HCHO Emission Rate Determination

HCHO emissions were measured using the desiccator method in accordance with ASTM D5582 [24]. For this purpose, a desiccator with an internal volume of 10.5 L was used, which was rinsed with distilled water and dried before each test. The contact surfaces between the base and the lid were lightly greased with vacuum-grade silicone grease to ensure an airtight seal during the test period.

The eight prepared test tubes were placed on the perforated porcelain plate inside the desiccator, arranged so that air could reach all surfaces. A Petri dish (PYREX, Corning, NY, USA) with a diameter of 100 mm and a height of 20 mm, containing 25 mL of distilled water was placed in the center. The system was sealed tightly and kept for 120 min in a room maintained at 24 ± 0.6 °C. After the exposure time had elapsed, the Petri dish was carefully removed, and chemical analyses were immediately performed. Three independent desiccator experiments were conducted for each material ($n = 3$).

For the colorimetric determination, 4 mL aliquots of the absorbent solution were taken and transferred to screw-cap test tubes. A blank was prepared using 4 mL of distilled water. To each tube, 0.1 mL of 1% chromotropic acid reagent was added, followed by the slow addition of 6.0 mL of concentrated sulfuric acid. The samples were mixed until a homogeneous solution was obtained and then allowed to cool to room temperature.

Absorbance was measured at 580 nm using UV-Vis spectrophotometry (Shimadzu UV-2600; Shimadzu Corporation, Kyoto, Japan). HCHO concentrations were determined using a previously constructed calibration curve. The results are expressed as the concentration of

HCHO in the absorbent solution. Reported values represent the mean $\pm$ standard deviation of three independent experiments.

2.4. Calculation of IAQ Indicators

Based on the average concentration of HCHO in air (C_{air} , mg m^{-3}) obtained using the desiccator method in accordance with ASTM D5582, indicators were calculated to assess the potential impact of the evaluated materials on IAQ. The WHO Hazard Quotient (HQ_{WHO}) was determined using the WHO guideline for indoor HCHO of 0.1 mg m^{-3} (30 min average) as a reference value. This index was calculated using the following [25]:

$$HQ_{\text{WHO}} = \frac{C_{\text{air}}}{0.1} \quad (1)$$

In addition, hazard ratios were calculated for acute and chronic exposure scenarios using the Reference Exposure Levels (RELs) established by the Office of Environmental Health Hazard Assessment (OEHHA) for HCHO. The acute HQ was obtained using a reference value of $55 \text{ } \mu\text{g m}^{-3}$ (0.055 mg m^{-3}); meanwhile, HQ_{chronic} was calculated using $9 \text{ } \mu\text{g m}^{-3}$ (0.009 mg m^{-3}):

$$HQ_{\text{acute}} = \frac{C_{\text{air}}}{0.055} \quad (2)$$

$$HQ_{\text{chronic}} = \frac{C_{\text{air}}}{0.009} \quad (3)$$

Finally, a relative emission index (I_{rel}) was defined in order to compare the performance of the various materials evaluated. This index was calculated by normalizing the concentration of each material relative to the lowest concentration observed in the data:

$$I_{\text{rel},i} = \frac{C_{\text{air},i}}{C_{\text{air},\text{min}}} \quad (4)$$

where $C_{\text{air},\text{min}}$ corresponds to the lowest HCHO concentration in the air among all the records of the analyzed materials.

2.5. Modeling of HCHO in Indoor Air Concentration

In order to estimate the potential contribution of the evaluated materials to IAQ, a well-mixed indoor air model was implemented under steady-state conditions [26].

The concentration of HCHO in indoor air (C_{ss}) was estimated using the following equation:

$$C_{\text{ss}} = \frac{E_A L}{N} \quad (5)$$

where C_{ss} is represented in mg m^{-3} , E_A is the surface emission rate ($\text{mg m}^{-2} \cdot \text{h}^{-1}$), L is the loading factor material (m^{-1}), and N is the ventilation rate (h^{-1}).

E_A was calculated based on the mass of HCHO captured during the testing in accordance with ASTM D5582:

$$E_A = \frac{m}{A_s t} \quad (6)$$

where m is the captured HCHO mass (mg), A_s is the sample area (0.0178 m^2), and t is the test duration (2 h).

L is defined as

$$L = \frac{A}{V} \quad (7)$$

where A is the exposed area of the material in the interior environment (m^2), and V is the volume of the enclosure (m^3).

2.6. Conditions for the Modeling Scenario

To standardize the analysis and allow for direct comparison between materials, all the woods evaluated were hypothetically treated as interior wall panels. This assumption represents a standardized emission scenario and does not necessarily imply that they are primarily used in that application.

For the layout of the space, a representative residential volume of 261 m³ was used, corresponding to an apartment typically featured in residential display models for composite wood products [27].

For the exposed area of the material, two scenarios were considered (10 m² and 20 m²), representing different levels of material load in the indoor environment, ranging from a moderate presence to significant partial coverage of interior surfaces.

Three ventilation rates were evaluated to represent scenarios of low ventilation (0.3 h⁻¹), typical residential ventilation (0.5 h⁻¹) and high ventilation (1.0 h⁻¹) in order to analyze the model's sensitivity. The model assumes steady-state conditions, homogeneous mixing of indoor air, and constant emission of the material, which is widely used in simplified assessments of indoor air pollutants.

3. Results

3.1. Quantification of HCHO Emissions from Wood

The system developed for the quantification of HCHO emissions demonstrated a robust and reproducible analytical response, as evidenced both visually and quantitatively (Figures 1 and 2a). The calibration curve exhibited a linear relationship in the evaluated range of 0.02 to 0.45 µg mL⁻¹, with a coefficient of determination (R²) of 0.9993, confirming the sensitivity and reliability of the method. The slope of the line (0.2744 a.u.·mL·µg⁻¹) indicates an adequate system response, allowing the quantification of HCHO concentrations both below and above the limits typically established in emission standards.

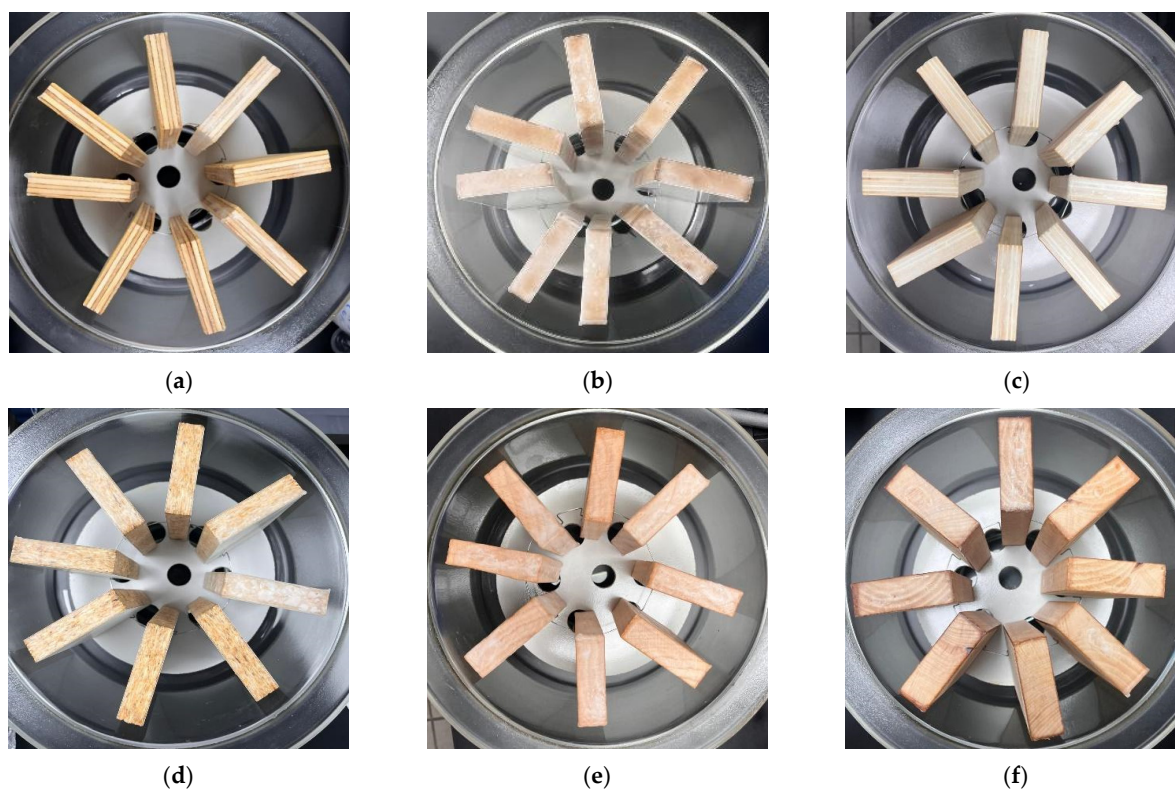


Figure 1. Samples prepared in accordance with ASTM-D5582-00: (a) pine plywood, (b) MDF, (c) birch plywood, (d) melamine, (e) oak, and (f) alder.

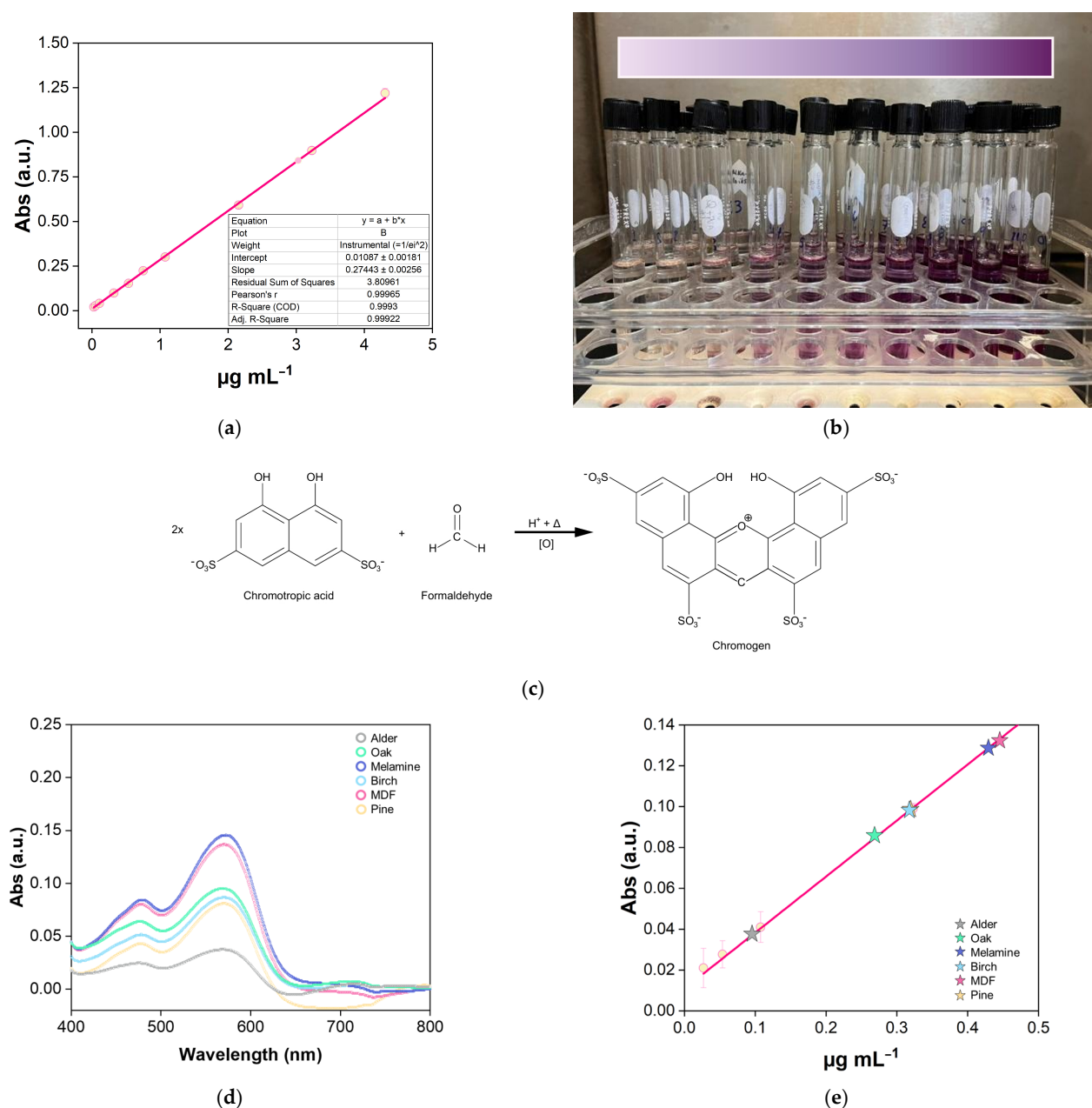


Figure 2. Records of HCHO emissions: (a) calibration curve, (b) evidence of the calibration curve, (c) mechanism of active chromogen formation, (d) absorbance spectrum of the evaluated woods, and (e) ranking of the evaluated woods based on absorbance.

Consistently, both in the standards and in the samples, a color change from colorless to purple was observed, attributable to the formation of the chromogen resulting from the reaction between HCHO and chromotropic acid in an acidic medium (Figure 2b,c) [28]. This behavior confirms the chemical reaction and its usefulness as a method for detecting HCHO.

As for the wood samples, the absorbance spectra (Figure 2d) showed a characteristic peak in the range of 560 to 580 nm, corresponding to the chromogen formed. Among the materials evaluated, MDF exhibited the highest absorbance (0.129 ± 0.005 a.u.) of HCHO, indicating the highest emission among the analyzed samples. Melamine showed the second-highest emission, with an absorbance of 0.126 ± 0.015 a.u., equivalent to 0.41 ± 0.05 µg mL⁻¹.

On the other hand, natural woods (pine, oak, birch, and alder) exhibited significantly lower values that were relatively close to one another, with absorbance values in the range of 0.03 to 0.08 a.u., corresponding to concentrations of 0.08 to 0.22 $\mu\text{g mL}^{-1}$. These results indicate a lower release of HCHO compared to composite materials.

Another way to express HCHO emissions is in terms of mass normalized by the amount of material ($\mu\text{g g}^{-1}$ of wood), which allows for a more direct comparison between matrices. Using this approach (Figure 3a), clear differences were observed among the materials evaluated.

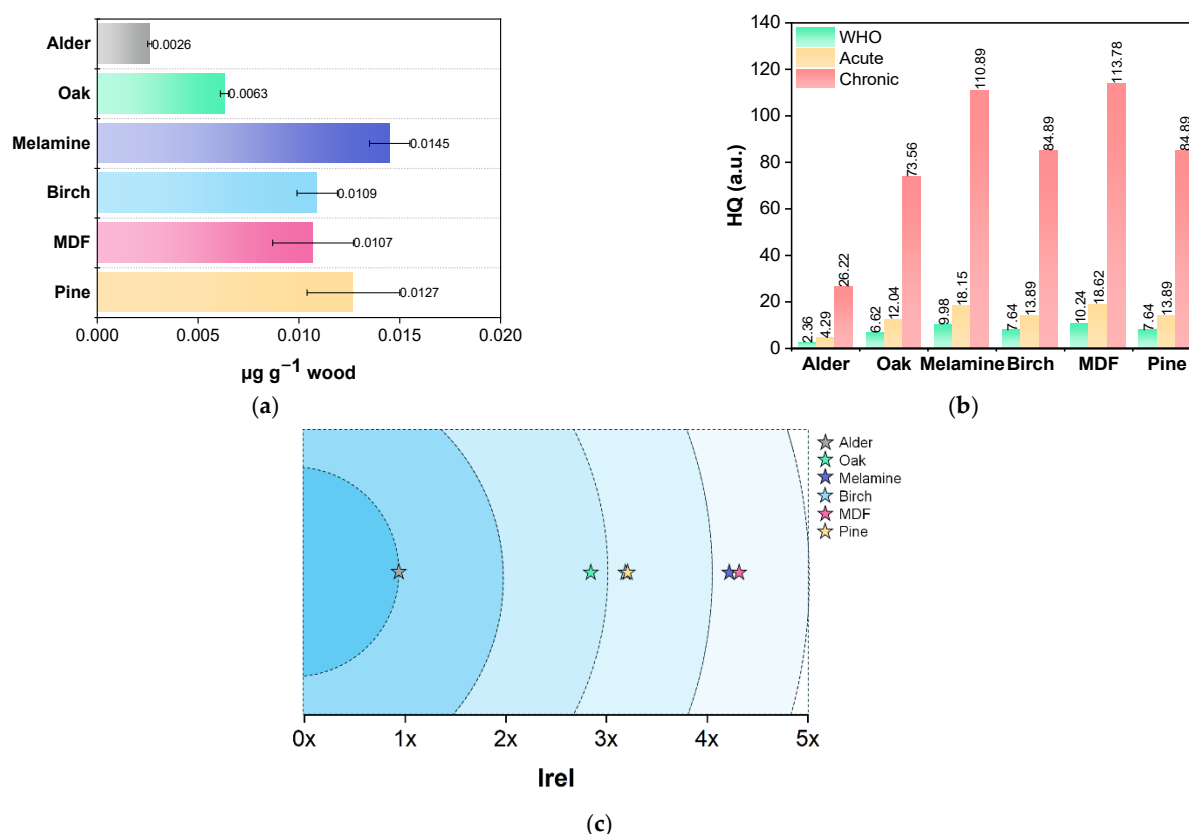


Figure 3. (a) HCHO emissions in $\mu\text{g g}^{-1}$ of wood, (b) hazard ratios, and (c) relative emission index for different types of wood.

Melamine exhibited the highest emission, with a value of $0.0145 \pm 0.001 \mu\text{g g}^{-1}$, confirming its role as the primary source of HCHO among the samples analyzed. In second place, pine showed relatively high emissions ($0.0127 \pm 0.002 \mu\text{g g}^{-1}$), followed by birch ($0.0109 \pm 0.001 \mu\text{g g}^{-1}$) and MDF ($0.0107 \pm 0.0002 \mu\text{g g}^{-1}$), whose values were statistically close within the margin of experimental error.

On the other hand, oak ($0.0063 \pm 0.0002 \mu\text{g g}^{-1}$) and alder ($0.0023 \pm 0.0001 \mu\text{g g}^{-1}$) showed the lowest emissions, demonstrating a lower potential contribution to the release of HCHO in indoor environments.

3.2. Assessment of Risks Associated with HCHO Exposure

To assess the toxicological significance of HCHO emissions, the results were compared with the WHO guideline value (0.1 mg m^{-3}) and expressed as hazard quotients (HQ) for acute and chronic exposure scenarios (Figure 3b).

The results show that all samples exceed the WHO guideline value ($\text{HQ}_{\text{WHO}} > 1$), with values ranging from 2.36 (alder) to 10.24 (MDF). MDF had the highest HQ_{WHO} , followed

by melamine (9.98), pine board and birch board (7.64), and oak (6.62). Alder presented the lowest value (2.36), although it was still above the safety threshold.

In the acute exposure scenario, a similar trend was observed, with HQ_{acute} values ranging from 4.29 (alder) to 18.62 (MDF). Once again, MDF (18.62) and melamine (18.15) had the highest values, followed by pine and birch (13.89), oak (12.04), and alder (4.29).

For the chronic exposure scenario, the $HQ_{chronic}$ values were considerably higher, reaching a maximum of 113.78 for MDF, followed by melamine (110.89), pine and birch (84.89), oak (73.56), and alder (26.22). This substantial increase demonstrates that the risk associated with HCHO intensifies under conditions of prolonged exposure. Overall, the results demonstrate that materials widely used indoors, particularly engineered materials such as MDF and melamine, present the highest levels of potential risk, far exceeding international reference values. Even natural materials such as oak and alder exceed the recommended limits, highlighting the need to consider the contribution of various wood sources in the assessment of IAQ.

3.3. Relative Comparison of Emissions Using I_{ref}

In addition, I_{ref} was evaluated, using alder as the baseline material ($I_{ref} = 1.00$), since it exhibited the lowest HCHO emissions among the samples analyzed (Figure 3c). This index allows for a direct comparison of the magnitude of emissions among different materials relative to an internal reference.

The values obtained showed a clear distinction between the samples, ranging from 1.00 to 4.34. MDF had the highest value ($I_{ref} = 4.34$), followed by melamine ($I_{ref} = 4.23$), indicating that these materials emit more than four times the amount of HCHO compared to the reference material. Pine board and birch board showed identical values ($I_{ref} = 3.24$), while oak had an intermediate value ($I_{ref} = 2.81$).

These results are consistent with the risk indices discussed earlier, reinforcing the finding that engineered materials—particularly MDF and melamine—are the primary sources of HCHO emissions. In contrast, natural woods tend to emit relatively lower levels of HCHO, although these emissions are still significant from an IAQ perspective.

3.4. Relative Comparison of Emissions Using the Reference Index

C_{ss} modeling revealed a strong dependence on both the EA and on ventilation conditions and surface loading (Figures 4 and 5). The estimated C_{ss} values ranged from 0.005 to 0.155 mg m^{-3} depending on the material and the scenario evaluated.

The highest concentrations were observed under low ventilation conditions (0.3 h^{-1}). MDF had the highest value (0.155 mg m^{-3}) for a load of 20 m^2 , followed by melamine (0.151 mg m^{-3}). Pine board and birch board showed identical values (0.115 mg m^{-3}), while oak exhibited intermediate concentrations (0.100 mg m^{-3}). Alder exhibited the lowest values (0.036 mg m^{-3}), consistently maintaining the lowest contribution across all scenarios.

For the typical residential ventilation scenario (0.5 h^{-1}), concentrations decreased proportionally, reaching maximum values of 0.093 mg m^{-3} for MDF and 0.090 mg m^{-3} for melamine (20 m^2). In this same scenario, pine and birch had concentrations of 0.069 mg m^{-3} , while oak and alder recorded values of 0.060 mg m^{-3} and 0.021 mg m^{-3} , respectively.

Under high ventilation conditions (1.0 h^{-1}), concentrations decreased significantly, with maximum values of 0.046 mg m^{-3} (MDF, 20 m^2) and minimum values of 0.005 mg m^{-3} (alder, 10 m^2), thus confirming the dominant role of ventilation in mitigating indoor air pollutants.

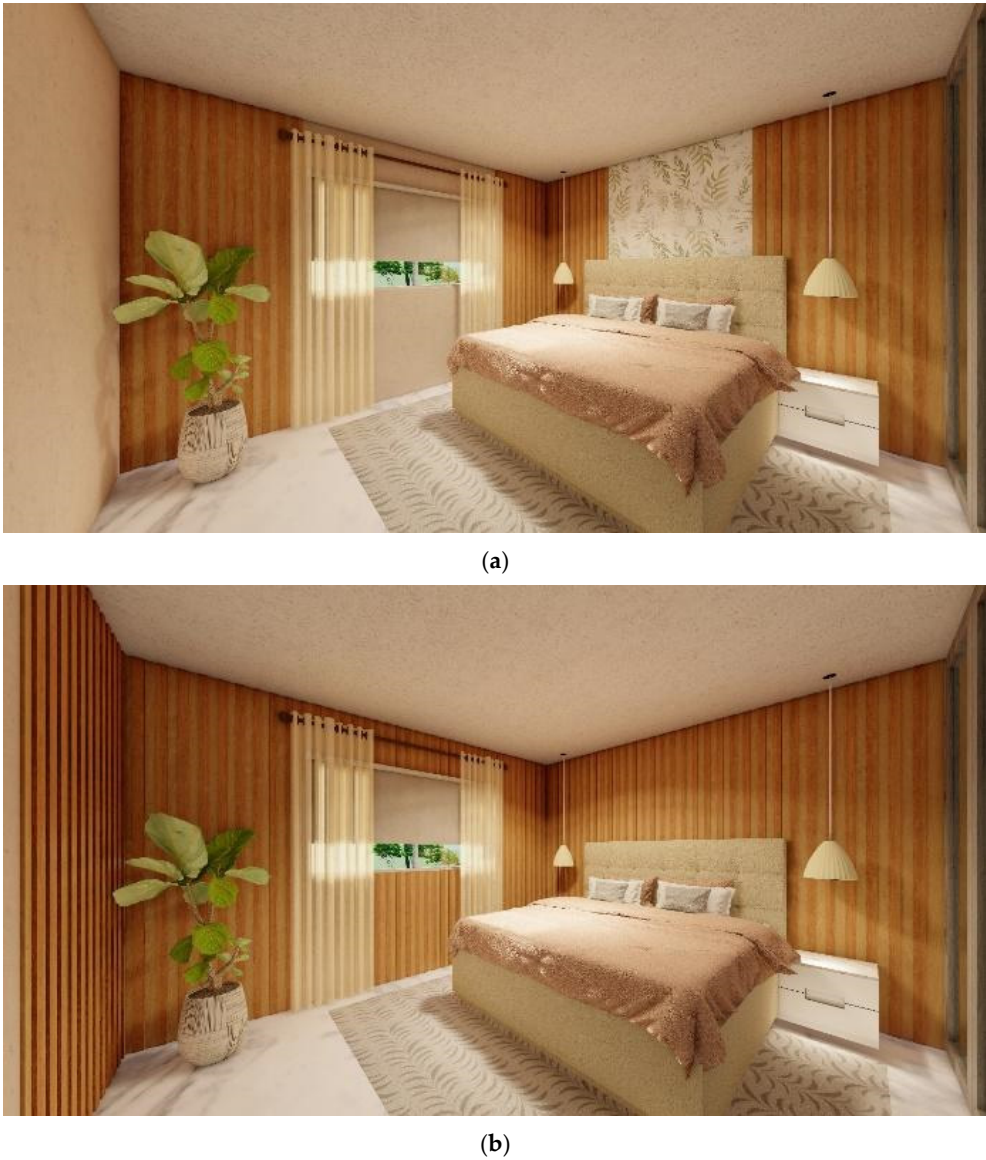


Figure 4. (a) Scenario 1: room with a wooden roof, 10 m²; (b) Scenario 2: room with a wooden roof, 20 m².

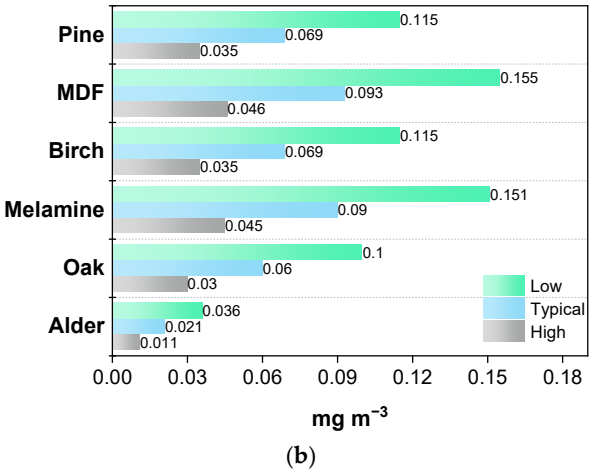
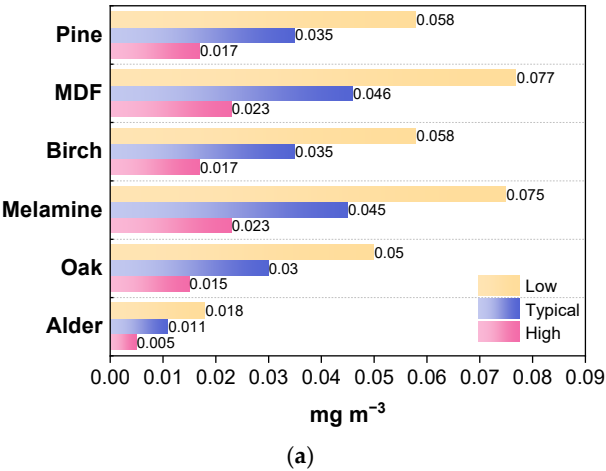


Figure 5. C_{ss} for the different scenarios and modeling conditions: (a) 10 m² and (b) 20 m².

Altogether, the results indicate that engineered materials such as MDF and melamine are the primary potential sources of HCHO in indoor environments, especially under conditions of poor ventilation and high material density. In contrast, woods such as alder and oak make a significantly smaller, though not negligible, contribution in scenarios with limited ventilation.

4. Discussion

4.1. Influence of Urea–HCHO Adhesives

The high emissions observed in MDF and melamine are closely related to the use of resins as binding agents during their manufacture, primarily urea–HCHO (UF), although melamine–urea–HCHO (MUF) and phenol–HCHO (PF) systems are also used due to their low cost and adequate adhesive properties [29,30]. Among these, UF resins are the most widely used in engineered wood products; however, they exhibit lower chemical stability against ambient humidity, which promotes the continuous release of HCHO through both desorption of free monomer and hydrolysis of methylene bonds under varying temperature and humidity conditions [31,32]. In contrast, PF resins exhibit greater stability and, consequently, lower emissions, making them more common in exterior applications [16]. In this context, the industry's preference for UF resins in interior applications represents a significant factor from the perspective of IAQ.

This behavior has been previously documented and explains why composite materials tend to act as the primary sources of HCHO in indoor environments [33–35]. In contrast, solid wood does not contain synthetic adhesives in its structure, so its emissions are limited mainly to the natural release of carbonyl compounds present in the biomass or to minor contributions from surface coatings.

4.2. Differences Between Composite Materials and Solid Woods

The results show that MDF and melamine have the highest EA rates and, consequently, the highest C_{ss} values, followed by particleboard. This behavior is consistent with the higher resin content in these materials, as well as their larger effective surface area and porosity, which facilitate the release of HCHO into the environment [36].

Comparison with previously reported HCHO emission data provides further context for the differences observed among the evaluated materials. As summarized in Table 2, HCHO emissions from wood materials vary considerably across previous studies. Direct quantitative comparison of these values is limited by differences in analytical methods, experimental conditions, sample preparation, and units of expression. Nevertheless, previous reports generally show higher HCHO emissions from engineered wood products than from solid wood, which is consistent with the emission behavior observed in the present study.

The wide range of reported HCHO emissions also highlights the influence of material composition and processing conditions. MDF, for example, has shown substantial variability in reported emission values [15,39,44], whereas lower emission levels have generally been reported for solid woods such as oak and pine [15,18,41]. Similarly, the variability observed for melamine-faced materials [29] indicates that surface finishing does not necessarily result in uniformly low HCHO emissions and that the underlying panel, adhesive system, and surface treatment may influence emission behavior.

In addition, composite panels undergo intensive industrial processes, including drying, pressing, and thermohydrolysis, which alter the chemical structure of the wood and promote the formation and release of carbonyl compounds [15,46]. Factors such as temperature and pressing time, as well as panel density, significantly influence emissions. Higher density means a higher adhesive content per unit volume, which increases the release of HCHO [47].

Table 2. HCHO emissions reported in previous studies for wood-based panels evaluated in this study.

Wood-Based Panel	HCHO Emission	References
Alder	26.20 mg kg ⁻¹ ^a	[37]
	78.80 mg kg ⁻¹ ^b	
	3.45 mg kg ⁻¹	[38]
	14.50 mg kg ⁻¹	
Oak	0.009 mg L ⁻¹	[39]
Melamine	0.06 mg L ⁻¹	[40]
	0.71 mg L ⁻¹	
	12.37 mg kg ⁻¹ ^c	[41]
	14.43 mg kg ⁻¹ ^c	
Birch	0.023 mg m ⁻³	[15]
	0.048 mg m ⁻³ ^d	
	0.31 mg L ⁻¹ ^c	[29]
	0.09 mg L ⁻¹ ^e	
MDF	0.630 mg m ⁻³	[42]
	0.380 mg L ⁻¹	[43]
	3.48 mg L ⁻¹	[44]
Pine	2.8 µg L ⁻¹	[45]
	0.003 mg L ⁻¹ ^f	[39]
	0.0036 mg L ⁻¹ ^f	[15]

Notes: ^a Veneer drying temperature at 20 °C; ^b Veneer drying temperature at 180 °C; ^c UF adhesive; ^d Paint coating; ^e PF adhesive; ^f Solid wood.

Furthermore, the type of wood used in the manufacture of the panel also influences emission behavior. Differences in acetic acid content—which can react with HCHO—can alter the final emissions [15]. However, this effect may be offset by physical properties such as the material’s density. In this study, pine and birch boards exhibited similar emissions, suggesting that factors associated with the manufacturing process may outweigh the intrinsic characteristics of the wood species.

The variability reported for alder and birch further demonstrates the importance of processing and experimental conditions. HCHO emissions from alder veneers varied with drying temperature [37], whereas differences reported for birch were associated with surface coating and adhesive type [15,29]. These findings indicate that material classification alone cannot fully explain HCHO emission behavior and that manufacturing conditions, adhesive chemistry, and surface treatments must also be considered.

In contrast, solid wood has significantly lower emissions due to the absence of synthetic adhesives in its structure, which significantly limits the release of HCHO [48]. Previous studies have reported particularly low emissions for species such as alder, which is consistent with the results obtained in our experiments. This difference highlights the critical role of industrial processes and chemical additives used during manufacturing in the generation of indoor pollutants [46,49]. In this regard, the results are consistent with previous reports identifying engineered wood products as one of the main sources of HCHO in indoor environments [23,50].

4.3. Potential Impact on IAQ in Homes

Modeling steady-state concentrations made it possible to translate the experimental results into indoor exposure scenarios. Under baseline ventilation conditions (0.5 h⁻¹), composite materials such as MDF and melamine reached concentrations close to the WHO guideline value (0.1 mg m⁻³) when exposed surfaces of 20 m² were considered. This result

is particularly relevant, as this area is representative of common configurations in homes, such as the combination of closets, cabinets, and interior finishes.

Under low-ventilation conditions (0.3 h^{-1}), the estimated concentrations of MDF and melamine exceeded the guideline value, suggesting a potential risk of exposure in homes with insufficient ventilation. In contrast, increased ventilation (1.0 h^{-1}) significantly reduced concentrations, highlighting the critical role of air exchange rate as a control mechanism.

Other material-related factors, such as the type of surface finish, can also affect emissions. It has been suggested that coatings such as impregnated decorative papers can act as diffusion barriers, reducing the release of HCHO [51]. However, there are reports with conflicting results, indicating that certain finishes may increase emissions [44]. In the present study, high-gloss finished MDF exhibited higher emissions than those reported for uncoated MDF, suggesting that the interaction between the finish and the substrate may influence the release dynamics of the contaminant [18,52]. These findings suggest that IAQ depends not only on the type of material but also on the amount installed and on the ventilation conditions in the space. In particular, the combination of composite materials with high surface loads and low ventilation rates can lead to HCHO levels close to or exceeding the recommended health limits.

Nevertheless, the ASTM D5582 desiccator method is a static, closed-system test and does not directly reproduce indoor conditions, particularly in terms of air exchange, airflow, temperature, and relative humidity. In contrast, climate chamber methods provide greater control over these parameters and are better suited to assessing emissions under representative indoor conditions. In this context, the desiccator method provides a standardized basis for comparing the emission behavior of different wood materials under equivalent conditions, which is consistent with the comparative scope of this study. Therefore, the resulting emission data should be interpreted as comparative inputs for scenario-based modeling rather than as direct measurements of emissions in occupied indoor environments.

5. Conclusions

The results show that wood-based materials used indoors can contribute significantly to HCHO levels in the air, with direct implications for the health and sustainability of living spaces. It was also observed that composite materials, particularly MDF boards and melamine, exhibit the highest emissions, while solid wood contributes less, though not insignificantly. The combination of experimental results and modeling demonstrated that indoor air concentrations depend not only on the type of material but also on the amount installed and on ventilation conditions. In scenarios with limited ventilation and high material loads, levels exceeding recommended values are reached, increasing the risk associated with prolonged exposure. From a sustainability perspective, these findings underscore the importance of selecting low-emission materials, optimizing interior design to reduce pollutant loads, and ensuring adequate ventilation. They also highlight the need to strengthen policies and standards related to VOC emissions in building materials. Overall, the study contributes to the development of strategies aimed at improving IAQ and promoting healthier and more sustainable environments.

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Abbreviations

VOC	Volatile organic compound
HCHO	Formaldehyde
IARC	International agency for research on cancer
MDF	medium-density fiberboard
IAQ	Indoor air quality
RELS	Reference exposure levels
OEHHA	Office of Environmental Health Hazard Assessment
UF	Urea–formaldehyde
MUF	Melamine–urea–formaldehyde
PF	Phenol–formaldehyde

References

1. Ruiz-Jimenez, J.; Heiskanen, I.; Tanskanen, V.; Hartonen, K.; Riekkola, M.-L. Analysis of indoor air emissions: From building materials to biogenic and anthropogenic activities. *J. Chromatogr. Open* **2022**, *2*, 100041. [[CrossRef](#)]
2. Xu, L.; Hu, Y.; Liang, W. Composition and correlation of volatile organic compounds and odor emissions from typical indoor building materials based on headspace analysis. *Build. Environ.* **2022**, *221*, 109321. [[CrossRef](#)]
3. Horvat, T.; Pehnek, G.; Jakovljević, I. Volatile Organic Compounds in Indoor Air: Sampling, Determination, Sources, Health Risk, and Regulatory Insights. *Toxics* **2025**, *13*, 344. [[CrossRef](#)] [[PubMed](#)]
4. Wu, X.; Apte, M.G.; Maddalena, R.; Bennett, D.H. Volatile Organic Compounds in Small- and Medium-Sized Commercial Build-ings in California. *Environ. Sci. Technol.* **2011**, *45*, 9075–9083. [[CrossRef](#)] [[PubMed](#)]
5. Bhat, A.A.; Afzal, M.; Goyal, A.; Gupta, G.; Thapa, R.; Almalki, W.H.; Kazmi, I.; Alzarea, S.I.; Shahwan, M.; Paudel, K.R.; et al. The impact of formaldehyde exposure on lung inflammatory disorders: Insights into asthma, bronchitis, and pulmonary fibrosis. *Chem.-Biol. Interact.* **2024**, *394*, 111002. [[CrossRef](#)] [[PubMed](#)]
6. Protano, C.; Buomprisco, G.; Cammalleri, V.; Pocino, R.N.; Marotta, D.; Simonazzi, S.; Cardoni, F.; Petyx, M.; Iavicoli, S.; Vitali, M. The Carcinogenic Effects of Formaldehyde Occupational Exposure: A Systematic Review. *Cancers* **2021**, *14*, 165. [[CrossRef](#)] [[PubMed](#)]
7. Barrese, E.; Valentini, M.; Scarpelli, M.; Samele, P.; Malacaria, L.; D’Amico, F.; Lo Feudo, T. Assessment of Formaldehyde’s Impact on Indoor Environments and Human Health via the Integration of Satellite Tropospheric Total Columns and Outdoor Ground Sensors. *Sustainability* **2024**, *16*, 9669. [[CrossRef](#)]
8. Chatterjee, A.; Pál, L.; Lovas, S.; McKee, M.; Diószegi, J.; Kovács, N.; Szűcs, S. Assessment of health risks from exposure to indoor volatile organic compounds in European educational buildings. *Sci. Rep.* **2026**, *16*, 6554. [[CrossRef](#)] [[PubMed](#)]
9. Kelly, T.J.; Smith, D.L.; Satola, J. Emission Rates of Formaldehyde from Materials and Consumer Products Found in California Homes. *Environ. Sci. Technol.* **1999**, *33*, 81–88. [[CrossRef](#)]
10. Adamová, T.; Hradecký, J.; Pánek, M. Volatile Organic Compounds (VOCs) from Wood and Wood-Based Panels: Methods for Evaluation, Potential Health Risks, and Mitigation. *Polymers* **2020**, *12*, 2289. [[CrossRef](#)] [[PubMed](#)]
11. Roffael, E. Volatile organic compounds and formaldehyde in nature, wood and wood based panels. *Holz Roh Werkst.* **2006**, *64*, 144–149. [[CrossRef](#)]
12. Salem, M.Z.M.; Böhm, M.; Srba, J.; Beránková, J. Evaluation of formaldehyde emission from different types of wood-based panels and flooring materials using different standard test methods. *Build. Environ.* **2012**, *49*, 86–96. [[CrossRef](#)]
13. Perera, H.; Lebanov, L.; Rodriguez, E.S.; Taoum, A.; Paull, B.; Sivret, E. Analytical approaches for sampling and assessing volatile organic compounds emitted from engineered wood products. *Build. Environ.* **2025**, *271*, 112578. [[CrossRef](#)]
14. Kim, S.; Choi, Y.-K.; Park, K.-W.; Kim, J.T. Test methods and reduction of organic pollutant compound emissions from wood-based building and furniture materials. *Bioresour. Technol.* **2010**, *101*, 6562–6568. [[CrossRef](#)] [[PubMed](#)]

15. Böhm, M.; Salem, M.Z.M.; Srba, J. Formaldehyde emission monitoring from a variety of solid wood, plywood, blockboard and flooring products manufactured for building and furnishing materials. *J. Hazard. Mater.* **2012**, *221*–222, 68–79. [[CrossRef](#)] [[PubMed](#)]
16. Salem, M.; Böhm, M.; Barčík, Š.; Beránková, J. Formaldehyde Emission from Wood-Based Panels Bonded with Different Formal-dehyde-Based Resins. *Drv. Ind.* **2011**, *62*, 177–183. [[CrossRef](#)]
17. Song, W.; Cao, Y.; Wang, D.; Hou, G.; Shen, Z.; Zhang, S. An Investigation on Formaldehyde Emission Characteristics of Wood Building Materials in Chinese Standard Tests: Product Emission Levels, Measurement Uncertainties, and Data Correlations between Various Tests. *PLoS ONE* **2015**, *10*, e0144374. [[CrossRef](#)] [[PubMed](#)]
18. Que, Z.-L.; Wang, F.-B.; Ma, L.-F.; Furuno, T. Effect of loading ratio, conditioning time and air exchange rate on the formaldehyde emission from wood-based board using large chamber and desiccator method. *Compos. Part B Eng.* **2013**, *47*, 278–282. [[CrossRef](#)]
19. Salthammer, T. The reliability of models for converting formaldehyde emissions from wood-based materials to different environmental conditions. *Build. Environ.* **2024**, *247*, 111041. [[CrossRef](#)]
20. Salthammer, T. Formaldehyde sources, formaldehyde concentrations and air exchange rates in European housings. *Build. Environ.* **2019**, *150*, 219–232. [[CrossRef](#)]
21. Liang, W.; Lv, M.; Yang, X. The combined effects of temperature and humidity on initial emittable formaldehyde concentration of a medium-density fiberboard. *Build. Environ.* **2016**, *98*, 80–88. [[CrossRef](#)]
22. Wang, H.; Xiong, J.; Wei, W. Measurement methods and impact factors for the key parameters of VOC/SVOC emissions from materials in indoor and vehicular environments: A review. *Environ. Int.* **2022**, *168*, 107451. [[CrossRef](#)] [[PubMed](#)]
23. Salthammer, T.; Mentese, S.; Marutzky, R. Formaldehyde in the Indoor Environment. *Chem. Rev.* **2010**, *110*, 2536–2572. [[CrossRef](#)] [[PubMed](#)]
24. ASTM D5582-22; Standard Test Method for Determining Formaldehyde Levels from Wood Products Using a Desiccator. ASTM: West Conshohocken, PA, USA, 2023. [[CrossRef](#)]
25. Adamkiewicz, G.; World Health Organization (Eds.) *WHO Guidelines for Indoor Air Quality: Selected Pollutants*; World Health Organization, Regional Office for Europe: Copenhagen, Denmark, 2010.
26. Rostami, A.; Pithawalla, Y.; Liu, J.; Oldham, M.; Wagner, K.; Frost-Pineda, K.; Sarkar, M. A Well-Mixed Computational Model for Estimating Room Air Levels of Selected Constituents from E-Vapor Product Use. *Int. J. Environ. Res. Public Health* **2016**, *13*, 828. [[CrossRef](#)] [[PubMed](#)]
27. US. Environmental Protection Agency. *Formaldehyde Indoor Air Model—Pressed Wood Products, Version 2.0 (FIAM-pwp v2.0): Model User Manual and Documentation*; US. Environmental Protection Agency: Washington, DC, USA, 2012.
28. Fagnani, E. Chromotropic acid–formaldehyde reaction in strongly acidic media. The role of dissolved oxygen and replacement of concentrated sulphuric acid. *Talanta* **2003**, *60*, 171–176. [[CrossRef](#)] [[PubMed](#)]
29. Conner, A.H. Wood: Adhesives. In *Encyclopedia of Materials: Science and Technology*; Elsevier: Amsterdam, The Netherlands, 2001; pp. 9583–9599.
30. Bekhta, P.; Sedliačik, J.; Bekhta, N. Effect of Veneer-Drying Temperature on Selected Properties and Formaldehyde Emission of Birch Plywood. *Polymers* **2020**, *12*, 593. [[CrossRef](#)] [[PubMed](#)]
31. Papadopoulou, E.; Pappa, C.P.; Karidi, K.; Triantafyllidis, K.S. Furfural-based phenolic resins for plywood panels: From basic concepts to upscale production. *Sustain. Chem. Pharm.* **2025**, *48*, 102205. [[CrossRef](#)]
32. Nuryawan, A.; Rahmawaty; Tambun, K.D.S.; Risnasari, I.; Masruchin, N. Hydrolysis of particleboard bonded with urea-formaldehyde resin for recycling. *Heliyon* **2020**, *6*, e03936. [[CrossRef](#)] [[PubMed](#)]
33. Guo, M.; Pei, X.; Mo, F.; Liu, J.; Shen, X. Formaldehyde concentration and its influencing factors in residential homes after deco-ration at Hangzhou, China. *J. Environ. Sci.* **2013**, *25*, 908–915. [[CrossRef](#)] [[PubMed](#)]
34. Sherzad, M.; Jung, C. Evaluating the emission of VOCs and HCHO from furniture based on the surface finish methods and reten-tion periods. *Front. Built Environ.* **2022**, *8*, 1062255. [[CrossRef](#)]
35. Salthammer, T. Data on formaldehyde sources, formaldehyde concentrations and air exchange rates in European housings. *Data Brief* **2019**, *22*, 400–435. [[CrossRef](#)] [[PubMed](#)]
36. Pibiri, E.; Omelan, A.; Uhde, E.; Salthammer, T. Effect of surface covering on the release of formaldehyde, acetaldehyde, formic acid and acetic acid from particleboard. *Build. Environ.* **2020**, *178*, 106947. [[CrossRef](#)]
37. Aydin, I.; Colakoglu, G. Formaldehyde Emission, Surface Roughness, and Some Properties of Plywood as Function of Veneer Drying Temperature. *Dry. Technol.* **2005**, *23*, 1107–1117. [[CrossRef](#)]
38. Demir, A.; Aydin, I.; Ozturk, H. Formaldehyde Release from Plywood Manufactured with Two Types of Urea Formaldehyde Resins after Fire Retardant Treatment of Veneers. *Drv. Ind.* **2018**, *69*, 193–199. [[CrossRef](#)]
39. Meyer, B.; Boehme, C. Formaldehyde emission from solid wood. *For. Prod. J.* **1997**, *47*, 45–48. [[CrossRef](#)]
40. Wang, S.-Y.; Yang, T.-H.; Lin, L.-T.; Lin, C.-J.; Tsai, M.-J. Properties of low-formaldehyde-emission particleboard made from recy-cled wood-waste chips sprayed with PMDI/PF resin. *Build. Environ.* **2007**, *42*, 2472–2479. [[CrossRef](#)]

41. Popovic, M.; Todorovic, T.; Djiporovic-Momcilovic, M.; Gavrilovic-Grmusa, I.; Popovic, J. The influence of decorative surface of commercial particleboards on the formaldehyde release measured by flask method. *Glas. Suma. Fakul.* **2018**, 137–142. [[CrossRef](#)]
42. Que, Z.-L.; Wang, F.-B.; Li, J.-Z.; Furuno, T. Assessment on emission of volatile organic compounds and formaldehyde from building materials. *Compos. Part B Eng.* **2013**, 49, 36–42. [[CrossRef](#)]
43. Cinar, H. Effects of temperature and thickness of wood based boards on formaldehyde emission. *Wood Res.* **2018**, 63, 895–908.
44. Kim, S.; Kim, J.-A.; Kim, H.-J.; Do Kim, S. Determination of formaldehyde and TVOC emission factor from wood-based compo-sites by small chamber method. *Polym. Test.* **2006**, 25, 605–614. [[CrossRef](#)]
45. Myers, G. Formaldehyde emission from particleboard and plywood paneling: Measurement, mechanism, and product standards. *For. Prod. J.* **1983**, 33, 27–37.
46. Xu, J.; Zhang, Y.; Shen, Y.; Li, C.; Wang, Y.; Ma, Z.; Sun, W. New Perspective on Wood Thermal Modification: Relevance between the Evolution of Chemical Structure and Physical-Mechanical Properties, and Online Analysis of Release of VOCs. *Polymers* **2019**, 11, 1145. [[CrossRef](#)] [[PubMed](#)]
47. Nemli, G.; Öztürk, İ. Influences of some factors on the formaldehyde content of particleboard. *Build. Environ.* **2006**, 41, 770–774. [[CrossRef](#)]
48. Gonçalves, F.D.; Carvalho, L.H.; Rodrigues, J.A.; Ramos, R.M. Wood-Based Panels and Volatile Organic Compounds (VOCs): An Overview on Production, Emission Sources and Analysis. *Molecules* **2025**, 30, 3195. [[CrossRef](#)] [[PubMed](#)]
49. Çolakoğlu, G.; Aydın, İ.; Çolak, S. The effects of waiting time of alder (*Alnus glutinosa* subsp. *barbata*) veneers before drying on shear and bending strength of plywood. *Holz Roh Werkst.* **2002**, 60, 127–129. [[CrossRef](#)]
50. Matthews, T.G.; Fung, K.W.; Tromberg, B.J.; Hawthorne, A.R. Surface emission monitoring of pressed-wood products containing urea-formaldehyde resins. *Environ. Int.* **1986**, 12, 301–309. [[CrossRef](#)]
51. Medved, S.; Forštnarič, J. Formaldehyde barrier efficiency of melamine impregnated paper and melamine edge-banding tape surfaced particleboard and MDF. *Acta Silvae Ligni* **2018**, 116, 43–50. [[CrossRef](#)]
52. Brown, S.K. Chamber Assessment of Formaldehyde and VOC Emissions from Wood-Based Panels. *Indoor Air* **1999**, 9, 209–215. [[CrossRef](#)] [[PubMed](#)]

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