



## Emission risks in processing and conversion of lignocellulose-based biocomposites

Nanci Ehman<sup>a</sup>, Sandra Rodríguez Fabià<sup>a</sup>, Julia Catalán<sup>b</sup>, Gary Chinga-Carrasco<sup>a,\*</sup> 

<sup>a</sup> RISE PFI, Høgskoleringen 6B, Trondheim 7034, Norway

<sup>b</sup> Department of Anatomy, Embryology and Genetics, University of Zaragoza, Zaragoza 50013, Spain

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### ABSTRACT

Wood-derived components (e.g. fibers, lignin, nanofibers) are widely studied to develop thermoplastic biocomposites with, for example, improved mechanical properties and reduced global warming potential. Manufacturing of biocomposite products includes compounding and conversion processes (e.g., extrusion, injection molding, and 3D printing). These processes apply mechanical forces and heat to melt thermoplastic polymers and form a given product. However, in some cases, compounding and conversion stages may generate emissions of volatile organic compounds (VOC) and/or ultrafine particles (UFP) and we must consider their effects on human health. Additionally, due to the nano-dimensions cellulose nanofibers are considered UFP. Therefore, its impacts on human health should be evaluated, especially when dried for biocomposite production. This review provides an overview of emissions generated in the production line of lignocellulose-based biocomposites, considering: wood preprocessing, extrusion, 3D printing, and injection moulding. Emissions of VOCs and UFP were considered, including the occupational exposure limits according to the current regulations and the potential health effects associated with such emissions

### 1. Introduction

The bioeconomy arises as a response to environmental and societal challenges, as a measure to mitigate the effects of climate change and reduce the use of fossil resources. In addition, it generates new opportunities for economic development. One of the driving alternatives for bioeconomy is the reduction of plastic materials. Lignocellulose-based materials represent environmentally friendly alternatives by reducing the fossil-based load in biocomposites [1]. In this review, lignocellulose-based biocomposites are materials that combine thermoplastic polymers with wood components, which are expected to contribute to the ongoing transition to a bio- and circular economy. Biocomposites for value-added applications can be based on wood derivatives (e.g., wood chips, sawdust, pulps, lignin, or nanocellulose) [2], which are renewable, highly available, and biodegradable. Wood components can be used to partially replace plastics such as high or low-density polyethylene (HDPE and LDPE, respectively), polyethylene terephthalate (PET), acrylonitrile butadiene styrene, polyvinyl chloride, polylactic acid (PLA), and polypropylene (PP) among others [3].

Lignocellulose-based biocomposites have great potential to replace conventional plastics in automotive, aeronautics, construction, textiles,

electronics, and packaging applications [4]. Leading car manufacturers currently consider the use of biocomposites in internal automobile parts and front or rear bumpers due to the lower weight, better performance, and lower CO<sub>2</sub> emissions [4,5]. Aircraft interior cabin and non-structural components, airframes of unmanned aerial vehicles, drones, and rotorcraft parts are also some options considered in aeronautics [6]. Some studies have evaluated applications like facades, partitions, building panels, and structural applications in the construction sector. Manufacturers are not only looking for improved mechanical properties but also economics and sustainability aspects must be included in the decision-making [7]. Other limiting aspects to consider in the production of lignocellulose-biocomposites are process optimization (leading to a more profitable global economy); and integration into the industrial production chain (acceptance in conventional markets, inclusion in the recycling chain, adaptation to regulations, etc.) [8, 9]. Considering sustainability, during biocomposites production, the use of biobased materials, or cellulose-base materials as fillers could lead to a decrease of greenhouse gas (GHG) emissions [10,11].

Additionally, biocomposites have been a focus in 3D printing over the last few years. The studies include combinations of lignocellulosic fibers, nanocellulose, or lignin with LDPE or HDPE [12–14], PLA

\* Corresponding author.

E-mail address: [gary.chinga.carrasco@rise-pfi.no](mailto:gary.chinga.carrasco@rise-pfi.no) (G. Chinga-Carrasco).

[15–19], polyhydroxyalkanoates [20], acrylonitrile butadiene styrene (ABS) [21,22], PP [23–25], among others. Wood fibers increase the biobased fraction in a given 3D structure and improve the mechanical properties as they act as a reinforcing agent. An increase in mechanical properties is expected to reduce material consumption for a given product, thus reducing cost, improving environmental performance, and encouraging industrial applicability.

Despite the extensive development of lignocellulosic-based biocomposites in recent years, only a few studies cover the emissions generated during extrusion and conversion processes. Scarce review studies on VOC emissions generally focus on pure polymers, either virgin or recycled [26,27]. Hence, there exists a gap in the analysis of emissions of all components in lignocellulose-based biocomposites. Some important research questions are, e.g.: (i) what are the emissions related to VOCs and particulate matter throughout the biocomposites production lines, (ii) what happens when processing raw materials at high temperatures, (iii) how much are the occupational exposure limits (OELs) when these materials are breathed, and (iv) which toxic effects these materials can generate in the body.

Considering the existing status of the bibliography and the research question mentioned above, this review intends to compile and analyze information about these relevant topics. The review focuses on VOCs and particulate matter during the production of lignocellulose-based biocomposites (compounding), and conversion processes (extrusion, injection molding, and 3D printing) (Fig. 1). The biocomposites studied are based on PE, PET, PLA, PP, and polyoxymethylene (POM) matrices. In addition, the OEL values and the toxic effects that exposure to these emissions may induce are also reviewed.

## 2. Methods

The study data collection was based on a PRISMA diagram based on the studies performed by Megist et al. [28] and Page et al. [29]. The data collection follows the guidelines for a systematic review. The steps followed to obtain the information are “formal and systematic”, and include premises such as the main objective, searching for relevant bibliography, selection according to specific criteria, and finally, the review. This literature review involved aspects of VOC and particle material emissions, established OEL values, and the effects of the emissions on health. The scope of the study comprised: thermoplastic polymer emissions (from PE, PET, PP, POM, and PLA, alone or wood fibers biocomposites), established OEL values of VOCs and particulate

emissions, and toxicological outcomes. Data management involves identification, screening, and inclusion stages (Supplementary information, Fig. S1). During the identification stage, Google Scholar, Scopus, and Science Direct databases were utilized for the data search. Keywords were combined using AND as a Boolean element, and English and Spanish language documents were collected. Keywords included type of studied material (biocomposites, PE, PET, PP, POM, PLA), type of emission (VOCs, particulate material, dust, formaldehyde), occupational exposure limits (VOC exposure workplace, occupational limits values VOC/particles), toxicity (in VOCs, nanocellulose, and reactive oxygen species (ROS)). During the screening process, the abstracts were reviewed. The non-relevant documents were not considered (articles assessed for eligibility): articles outside the scope of the review (noise, heat, and vibration emissions), and duplicate articles were removed. We focused our efforts on the processes that were considered industrially relevant. Most of the studies on fiber modification have relatively low technology readiness levels (TRL) and are not industrially implemented yet. Therefore prior chemical or physical modifications of cellulose fiber were not considered in this study. Also, the selection criteria do not include control of emissions in the industry, as it requires extensive evaluation before being implemented in the process lines. Considering the initial selection and the additions after the reviewers’ recommendations, a total of 131 articles were considered. No trend in the publications was found. Articles found about VOCs involve mostly pure components (e.g. pure polymers). Toxicity assessments of the emissions found are mainly performed using in vitro approaches.

## 3. Wood preprocessing, compounding, and conversion

Processing of biocomposite involves several operations: wood preprocessing, compounding, pelleting, and conversion (Fig. 1). “Pre-processing” is the first step in preparing wood-based components. This can involve mainly wood components: (i) sawdust that is conventionally used in thermoplastic biocomposites as fillers, (ii) wood fibers and nanocellulose used as reinforcement, and (iii) lignin, which is mostly a side stream from the pulp industry.

Sawdust particles are produced through grinding processes, followed by a classification step. The classified wood material goes through a drying stage by air convection, indirectly with heat gas conduction or using electromagnetic IR-radiation [30]. After this stage, the dried sawdust particles enter the compounding step (usually with an extrusion system). This step is similar for, e.g., sawdust particles, wood fibers,

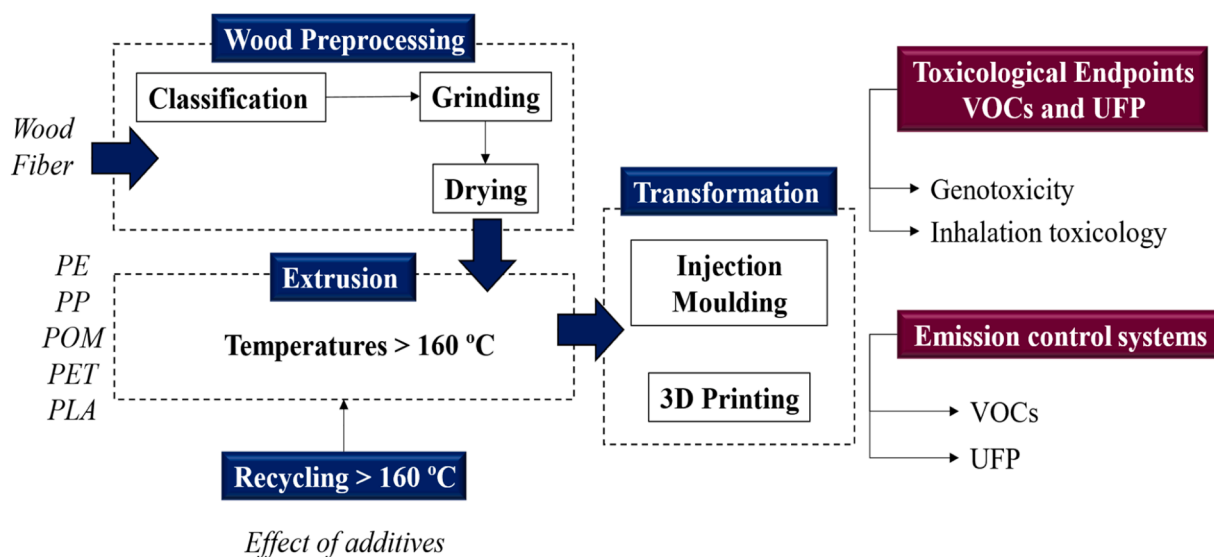


Fig. 1. Scheme of biocomposite production, conversion, and recycling, considering emissions and potential toxicological aspects. Noise, vibration, and heat may also affect health but are not considered in this study.

nanocellulose, and lignin. At the extruder, the wood components are mixed with a given thermoplastic, a compatibilizer agent, and, in some cases, additives. The main function of the extrusion stage is the melting and mixing of the polymers, achieving high homogenization with constant flow. Screw extruders range from single screws to double or multiple screws. The basic operating principle for screw extruders is similar in all cases. The system operating zones are: feeding of pellets, mixing-melting in screws (compression), and finally, the zone where the product is formed into pellets and ready to be converted to end-user products [31].

The conversion of biocomposites into end-user products includes various options. Biocomposites containing sawdust powder are usually used for pipes and profiles. These products are produced through extrusion which is a dynamic process where the melted material is continuously forced to pass through a profiled mold or die. Good designs of transversal profiles, and optimization of pressure, speed, and temperature in the system and the die are foremost to achieving a product with excellent mechanical properties [31,32]. Other processes follow the principle of structure formation by depositing the biocomposite melted in a preformed mold. The processes include injection molding, rotational system, thermoforming, compression casting, or solvent casting [33–41]. Injection molding is advantageous for high-volume products, e.g., automotive parts, electronic components, medical devices, furniture, and single-use devices such as food packaging.

External risk factors during compounding and conversion processes for wood-based biocomposite include volatile organic compounds (VOCs) and wood dust emissions (wood particles) [42]. VOCs and some wood particle emissions are invisible to humans but exist in the immediate environment of the equipment [43].

#### 4. VOCs emitted during biocomposite production

The effect of temperature on VOC emissions is essential since temperature leads not only to the release of substances with toxicological potential into the environment but also because changes can occur in the biocomposite structure and damage to pieces of equipment. The different stages during biocomposite production are carried out at elevated temperatures: drying at temperatures between 60–140°C [44], extrusion and injection molding at temperatures >150°C (both according to the type of polymeric resin) [45]. The release of VOCs from the cellulosic fraction is influenced by wood type (softwood or hardwood), temperature, the wood components present, and humidity [46–48].

#### 4.1. Wood-derived VOC emissions

During softwood preprocessing, terpenes are the first compounds emitted at lower temperatures (60–140°C) [34]. These molecules, composed of isoprene C5 structural units, are widely distributed in nature or can be generated anthropogenically [49]. In nature, many terpenes have high volatility at room temperature because they act as pheromones to protect trees from animals and insects. However, the levels generated anthropogenically are higher and emitted during storage, sawing, and drying [50].

Fig. 2 summarizes the terpene compounds classified according to the isoprene unit numbers [51] and molecular construction. The presence of terpenes causes VOC emissions (~75–80 % of the VOC emissions at  $T > 60^\circ\text{C}$ ) [44,52]. The most emitted terpenes are monoterpenes (pinenes, limonene, and 3-carene) followed by sesquiterpenes [53]. The released terpenes and the amount emitted depends on the species; for example, species from the *Pinus* or *Picea* genus emit high amounts of monoterpenes, while species of the *Larix* genus emit a spectrum of sesquiterpenes and monoterpenes. Terpenes and terpenoids are associated with softwood resins and are removed by 50 % during wood drying [52].

Emissions from softwood also involve aldehydes and carboxylic acids. Hexane-type aldehydes from the fat content of pine are usually frequent, and the emission is even higher for this genus compared to *Picea* species [52]. Pine compounds have similar emitted VOCs at 90–120°C, which mainly include components from the alkene family such as terpenes, some straight-chain alkanes (propanal and hexanal), aldehydes like furfural, and smaller amounts of ketones [54,55]. Similarities in softwood species were also observed in a previous study where the VOCs were emitted at higher temperatures (180–200°C) from Scots pine and Norway spruce mixture, mainly emitting terpene derivatives [56]. Also, at higher temperatures (180–220°C) less volatile compounds such as fatty acids, resin acids, diterpenes, and triterpenes begin to be released into the air [56].

The compounds emitted from hardwoods include carbonyl compounds (aldehydes, organic acids, and ketones), alcohols, alkyl furans, and in some species terpenes [52]. Aldehydes and acetic acids are the ones emitted at the highest levels. Due to hemicellulose content, acetic acid emitted in hardwoods is even higher compared to softwoods [52]. This difference between emitted components was observed in a study of VOC profiles at 40–120°C of 7 wood species that included hardwoods (European alder and beech, Norway maple, sessile oak, acacia, and wych elm) and a softwood (European larch) [57]. Acacia and alder showed the

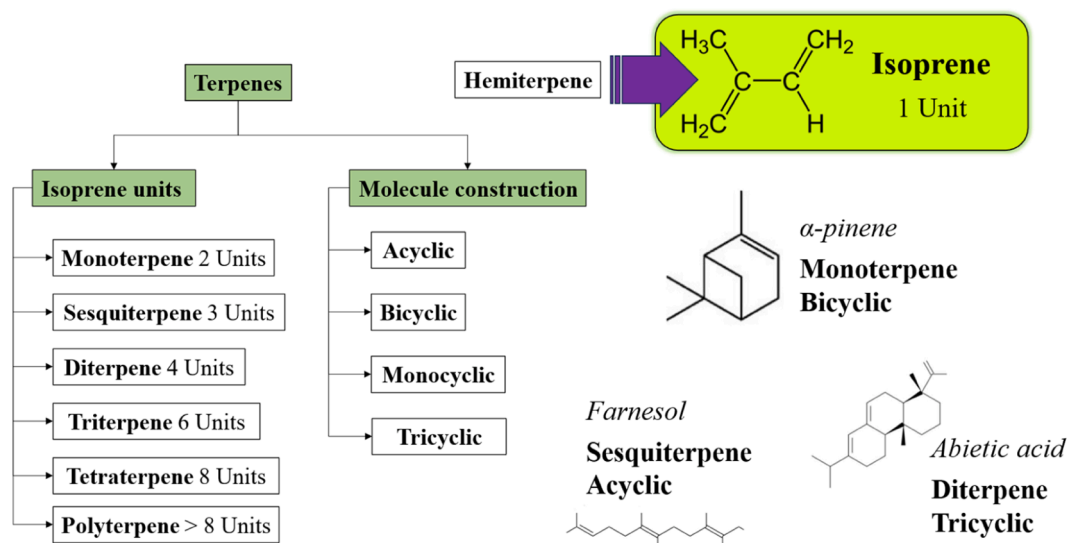


Fig. 2. Terpene compounds classification: Right). The chemical formula of isoprene, the monomeric unit of terpene compounds (top), and examples of terpene molecules (down). Left) Terpenes classification concerning the isoprene units and molecular conformation.

most volatile carbonyls; beech and maple emitted higher quantities of aliphatic hydrocarbons; oak showed mainly values of acetic acid; elm showed high amounts of sesquiterpenes and aromatic hydrocarbons; and larch showed mostly terpenes (monoterpenes, oxygenated monoterpenes, sesquiterpenes, and oxygenated sesquiterpenes) [57]. The emissions at higher temperatures are related to hemicellulose, cellulose, and lignin degradation [46]. In carbohydrates, oxidation reactions above 150°C result in acetic and formic acid formation, which lead to the hydrolysis of carbohydrates and are catalyzed by humidity [58]. For example, the acetic release was observed in cork oak wood at 120°C and was attributed to hemicellulose degradation through acetyl group elimination [59]. The same authors also found furfural formed from pentoses and hexoses compounds [59]. Formic and acetic acids were emitted in untreated hardwood at higher values than softwood, but aldehydes were detected in both cases [60]. Extractives, polysaccharides, and lignin components lead to formaldehyde emissions through decomposition pathways generating first hexoses and hydroxymethylfurfural, and finally furfural formation with formaldehyde release [47,61]. Authors have found that within the polysaccharide components, xylose and arabinose lead to increased formaldehyde release, while lignin emits even more formaldehyde than carbohydrates [47]. In addition, it has been found that the removal of extractives reduces the release of formaldehyde [62].

The type of treatment influences the release of VOC components due to the type of chemical bonds present (CH or CO bonds) during the thermal degradation or the secondary reactions in lignin, hemicelluloses, or amorphous cellulose [63]. Considering the influence of chemical components, it should be noted that the wood pulp obtained mechanically will release more VOCs than pulps obtained by chemical processes [64]. Even if we compare a bleached and unbleached chemical pulp, higher amounts of VOCs will be released in the unbleached pulp due to the presence of less thermally stable components [63]. However, a previous study demonstrated that in chemi-thermomechanical pulp (CTMP) the acetic acid release is higher than in thermomechanical pulp (TMP) [48]. The authors attributed the higher release of acetic acid to a greater number of acetic acid groups since saponification of the acetyl groups occurs during CTMP pulping, and also to the presence of formate ions ( $\text{HCOO}^-$ ) that lead to the formation of undissociated acetic acid in contact with water [48,65].

#### 4.2. Established exposure limits of wood-based VOCs in workplaces

Regulations across countries and regions set occupational exposure limits (OELs) in workplaces for some VOCs emitted from wood. For the other VOCs described in this study, no OELs are set either because they have not been described as hazardous or because of the limited available information regarding their potentially hazardous effects. The limits defined are Time-Weighted Average or Threshold Limit Value (TWA or TLVs, average airborne exposure in an 8-h day or 40-h week), Short-Term Exposure Limits (STEL, 15-min time-weighted average exposure, which should not be exceeded during the working day even if the 8-h TWA is below the limit values), the Permissible Exposure Limit (PEL, the basic limit for how much exposure is allowed), and or Ceiling (maximum exposure limit, which cannot be exceeded for any length of time) [66]. Table 1 shows the regulated OELs for VOC emissions that could be generated from wood in workplaces. The levels are regulated nationally and are based on regional regulations. Each country can reduce the limit values concerning regional standards, but the values can never be higher than those established standards.

#### 4.3. VOC emissions from thermoplastic matrices

VOCs can be generated at any point in the life cycle of a thermoplastic: by thermal oxidation in the manufacturing stages (processing or extrusion), irradiation during aseptic storage, prolonged storage, or reaction with the environment or elements it contains [27]. Detecting

**Table 1**

Information about regulatory occupational exposure limits (OELs) in air emissions of wood-related volatile organic compounds.

Component	OELs ( $\text{mg}/\text{m}^3$ )		Refs.
	TWA	STEL	
Terpenes	$\alpha$ -pinene, $\beta$ -pinene, 3-carene	Cal/OSHA and NIOSH $\leq 560$ (turpentine). Scandinavian countries $\leq 140$	Scandinavian countries $\leq 300$ (can vary between countries)
	D-Limonene	Spain $\leq 168$ Scandinavian countries $\leq 140$ – $150$ (can vary between countries) Switzerland/Germany adopted lower values than Scandinavia.	Scandinavian countries $\leq 300$ (can vary between countries). France approx.1000.
	Furfural (2-Furaldehyde)	OSHA $\leq 20$ European and some Asian countries $\leq 8$ East European countries $\leq 10$ New Zealand $\leq 0.80$	Some European countries: 8–25 (can vary between countries)
Aldehydes	Formaldehyde	EU regulation $\leq 0.37$ . Some European countries can accept higher (the UK) or lower (The Netherlands) values. Japan $\leq 0.12$ (JSOH).	EU regulation $\leq 0.74$
	Formic acid	EU regulation, JSOH, OSHA, and NIOSH $\leq 9$ Some countries in Europe $\leq 5$ , and some can reach 9.5	–
Organic acids	Acetic acid	EU, JSOH, OSHA, and NIOSH regulations $\leq 25$ Sweden accepts limits of up to the half.	NIOSH $\leq 37$ and EU regulation $\leq 50$

TWA: Acceptable average exposure over a specified period, usually a nominal eight hours. STEL: Acceptable average exposure over a short period, usually 15 min

VOCs emitted during the extrusion and conversion stages is of relative importance in plastic processing plants. Several studies have evaluated the VOCs emitted from virgin and recycled plastic, considering PE, PET, PP, POM, and PLA. Table 2 summarizes the VOCs emitted from thermoplastic matrices at temperatures above 160°C considering OELs  $< 1000 \text{ mg}/\text{m}^3$ . Acetic acid, formic acid, and formaldehyde are also emitted (see corresponding OELs in Table 1).

Components emitted from virgin PE include aliphatic and aromatic hydrocarbons, alcohols, phenolic compounds, aldehydes, ketones, carboxylic acids, and some esters. Linear hydrocarbons (alkanes and alkenes) are the most frequent molecules in VOCs from PE [26]. Aldehydes (mainly acetaldehyde), organic acids (acetic acid was the most frequent), and several ketones (mainly acetone) were emitted during the extrusion of LDPE pellets to produce films (280–320°C) [71]. For aliphatic hydrocarbons, the list contains mainly alkanes with OEL  $< 1000 \text{ mg}/\text{m}^3$  (Table 2), although there are other components listed (OELs  $> 1000 \text{ mg}/\text{m}^3$ ) like 2-methyl hexane, n-nonane, and n-octane (in HDPE virgin and recycled), 2-methyl pentane and 2,4-dimethyl hexane

**Table 2**

Information of regulatory occupational exposure limits (OELs) in air emissions of virgin polymer-related volatile organic compounds.

Group	Compound	Polymer	OELs (mg/m <sup>3</sup> )		Refs.
			TWA	STEL	
Alkanes	n-hexane	High-density polyethylene Polypropylene	OSHA ≤1800 (PEL). NIOSH ≤180 (REL) EU regulation ≤72. China ≤100	Some European countries ≤144. Sweden and China ≤180	[26, 69,77]
Aromatics	ethylbenzene	High-density polyethylene	EU regulation ≤442. NIOSH and OSHA ≤435 (PEL/REL). Some European countries ≤88. Japan ≤217. Norway ≤20	EU regulations ≤884. Some European countries adopted lower limits	
	xylene	High-density polyethylene Low-density polyethylene	OSHA, NIOSH ≤435 (PEL/REL). EU regulation ≤221. Some European countries have established lower values	NIOSH ≤655. EU regulation ≤442. Some European countries adopted lower levels	
	naphthalene	Polypropylene	EU regulation, NIOSH, OSHA ≤50 Some European, Asian, and Australian countries accept lower values	European and Asia countries <75–80 (vary between countries). Some European countries <10	
Alcohols	phenanthrene	Polypropylene	OSHA and ACGIH ≤0.2 (PEL as benzene soluble). NIOSH ≤0.1 (REL, cyclohexane extractable)	–	
	isooctanol	Polypropylene	EU regulation ≤5.4. Japan ≤4.3	–	
Aldehydes	1-decanol	Polypropylene	EU regulations ≤100. Germany and Switzerland ≤66	EU regulations ≤200	[69, 76]
	2-methyl-2-propanol	Polypropylene	NIOSH, OSHA ≤300 (PEL/REL). JSOH ≤150. Some European countries adopted lower values Norway ≤75	NIOSH, OSHA ≤450	[69, 78]
	Acrolein	Polypropylene	NIOSH, OSHA ≤0.5 (PEL/REL). EU regulation ≤0.05. Some Asian countries ≤0.23	STEL ≤0.80 (NIOSH, OSHA), ≤0.23 (EU regulations)	
Carboxylic acids	Acetaldehyde	Polypropylene	OSHA ≤360 (PEL). European countries except France ≤100. Some European, Asian, and Australian countries ≤50	–	[69, 71]
	Propanoic acid	Polypropylene	EU regulation ≤31. NIOSH ≤30 (REL)	EU regulation ≤62. NIOSH ≤45	[26, 69]
Ketones	Methyl isobutyl ketone	High-density polyethylene	OSHA ≤410 (PEL) NIOSH ≤205 (REL) EU regulations ≤83	OSHA ≤416. NIOSH ≤300 EU regulation <208. Some European countries adopted lower values	[69, 78]
	4-hydroxy-4-methyl-2-pentanone	Polypropylene	NIOSH and OSHA ≤240 (PEL/REL). Some European, Asian, and Australian countries adopted lower values	European countries <480 (vary between countries)	

PEL: Permissible exposure level. REL: Recommended exposure limit.

(in PP virgin), and heptane (in recycled PET) [26]. The limits varied in all components. Even in the same country, as is the case of the USA, there are variations between NIOSH and OSHA [67,68]. In recycled PE, halogenated substances, sulfuric acids, pyridines, imides, pyrazines, amides, and amines are also identified [26] and the components are related to the incorporated additives or contamination during the recycling process.

Several configurations are used to reduce the generation of VOCs in post-consumer PE. VOC reduction in recycled HDPE could be performed by heating and is common in the purification of materials for food packaging, where it is required to achieve a standard according to standard norms referring to total and specific migration [72]. Although the number of VOCs of the components present in virgin PE is reduced when the recycling cycles increase, new components such as flavors and fragrances appear [26].

Both PE and PP samples yielded similar oxidation [73]. Usually, alkanes and alkenes emitted from PP are in lower concentrations and quantities than PE emissions. But more aldehydes and carboxylic acids are released in PP, some examples include acetaldehyde, acrolein, heptanal, octanal, nonanal, and decanal. Carboxylic acids emitted from PP were formic, acetic (also in HDPE and LDPE), propanoic, tetra, and hexadecanoic acids.

Table 2 also indicates the OEL values of some ketones emitted by PP and PE. In PP, mainly acetone (not indicated in the table, OELs > 1000 mg/m<sup>3</sup>) and some pentanones and hexanones are emitted; while in PE butanones, hexanones, decanones, and dodecanones are emitted [26]. Some alcohols emitted from PE, PP, and PET with the lowest OEL values included 2-butoxyethanol, 1-dodecanol in recycled HDPE; 3-methyl-1-butanol and isooctanol in recycled LDPE; and 1-decanol, 1-dodecanol in virgin PP. Other alcohols emitted from recycled plastics were p-cresol (HDPE), butylated hydroxytoluene (from HDPE, and PET), and furfuryl alcohol (LDPE) [26]. Similarly, it is possible to find new components in

PET recycling.

Components in virgin PET are aldehydes, aromatic and aliphatic hydrocarbons. However, terpenes, such as  $\alpha$ -limonene, have been found in post-consumer PET [74]. Fabris et al. [75] also found the presence of terpenes (limonene and myrcene in post-consumer options) in PET. Just as PE, PP, and PET are widely used and recycled in the packaging industry, PLA is one of the most used materials for 3D printing. Väisänen et al. [76] evaluated VOC emissions in the filament production and 3D printing of PLA and birch, aspen, and spruce wood biocomposites. The VOCs emitted at the highest concentration during 3D printing were lactide (lactone cyclic ester, PLA dimmer), 2-methyl-2-propanol, hexamethylcyclotrisiloxane, ethyl acetate, and tetrachloroethylene. During the production of filaments, lactide, ethanol, and 2-methyl-2-propanol were mainly produced. Carbonyl compounds were also evaluated during the two stages, and it was found that during filament production, acetone and formaldehyde emissions were mainly generated.

The biocomposite filaments showed the release of VOCs such as  $\alpha$ -limonene, 3-carene,  $\alpha$ -pinene, isoprene, and p-cymene [76]. When compared with ABS, PLA emits up to 7 times fewer VOCs when 3D structures are printed at 220°C [79]. However, when compared to PET at 210°C, it emits more VOCs, and if colored PLA filaments are used, the emissions are even higher [80].

In POMs, the thermal degradation leads to formaldehyde and CO<sub>2</sub> release. Formaldehyde emissions can be reduced by adding amides or amines that react with aldehydes or reducing the unstable formic acid ester form [81]. These additives are low molecular modifiers and must be optimized to prevent a decrease in mechanical properties [82].

#### 4.4. VOC emissions studies in the industry and 3D printing

The concentration values of VOCs emitted in PE plants are lower than those generated in PP recycling plants. However, in both cases,

they are below the OEL values [42]. The total VOCs (aliphatic hydrocarbons, aromatic compounds, and formaldehyde) emitted in the PP and PE extrusion areas in two recycling plants in an 8-h work day were <3 mg/m<sup>3</sup> [42]. The measured levels in a study of emissions in the extrusion zone of 3 factories in the plastic industry also remained below the established limits: factory 1 (PE and paraffin wax), factory 2 (PE, PS, ethylene/octene copolymer), and factory 3 (PP, HDPE, polyethylene wax and ABS) [83]. The authors detected mainly alkanes (n-octane, concentration <8 mg/m<sup>3</sup>; n-decane, <0.20 mg/m<sup>3</sup>) and naphthalene <0.60 mg/m<sup>3</sup> in factory 1. Ethyl acetate (<3 mg/m<sup>3</sup>), n-butane (<0.60 mg/m<sup>3</sup>), isopropyl-alcohol (<0.30 mg/m<sup>3</sup>) isobutane, and propane (<0.50 mg/m<sup>3</sup>) were mainly detected in the factory 2. Finally, in factory 3 aromatic compounds were detected at concentrations <5 mg/m<sup>3</sup> [83].

Emission levels were also low in a study carried out in a 3D printing chamber where pure PLA was used, finding total VOC (aliphatic hydrocarbons, aromatic compounds, acetic acid, ketones) values between 0.30–0.40 mg/m<sup>3</sup> [78]. Similar values were found during the 3D printing of pure PLA filaments and combined with cellulose fibers and wood dust [76]. In pure PLA, lactide was the most abundant compound, but acetic acid, aldehydes, and alcohol compounds were also detected. Terpenes were also detected in the biocomposites. The author found that the total VOC levels were similar to pure PLA polymer. The emissions during 3D printing were lower than 0.056 mg/m<sup>3</sup>. The authors concluded that the total concentration of VOCs from PLA-based biocomposites was not higher than pure PLA, although emissions of terpenes and furfural in non-significant air concentrations were found in biocomposites [76].

### 5. Particulate matter released during biocomposite production

Dust is dispersed atmospherically and causes not only air pollution but also health effects when the concentration of particles is greater than those allowed. Understanding and controlling the aspects related to the emission of UFPs makes it possible to control the impacts on health and the environment. The deposition of particles in the respiratory tract is based on their physicochemical properties, charge, size, density, and shape [84]. Deposition follows mechanisms concerning diffusion, gravitational sedimentation, inertial impaction, interception, and electrostatic interaction [84–86]. In the size range greater than 1 μm, particles are deposited due to impaction and sedimentation. Deposition in

the upper bronchial and extrathoracic airways is determined by impaction. Sedimentation regulates deposition in the lower bronchial airways. In the size range of 0.1 to 1 μm, particles are deposited simultaneously by gravitational and diffusional transport. At smaller sizes, diffusion predominates [86,87]. The classification of particles is based on their aerodynamic equivalent diameter (AED) and is defined as the diameter of a sphere of unit density that would have the same settling velocity as the particle studied (it is used to compare particles of different sizes, shapes, and densities) [88]. Fig. 3 summarizes the AED, predicted deposition, and mechanisms of the particles in the respiratory tract.

The AED reveals if they are inhalable (breathed into the nose or mouth), thoracic (fraction of inhalable particles that pass through the larynx), or respirable (the inhalable particles that reach the lower part of the respiratory tract, and the alveoli of the lungs). Inhalable particles lower than 100 μm are deposited in the nasopharyngeal region (upper airway passages-nose, nasal cavity, and throat) largely by impaction [85].

The World Health Organization (WHO) defines particulate matter (PM) as sizes lower than 10 μm: PM<sub>2.5</sub> (0.10–2.50 μm), PM<sub>10</sub> (2.50–10 μm) [89], and finally, the smaller ones are UFP particles (<0.10 μm or 100 nm) [85]. During breathing at rest of UFP, 5 % of the particles larger than 100 nm of the total present in the environment can enter through the nose, but up to 80 % if the particles are closer to 1 nm [90].

Wood dust corresponds to an accumulation of any wood particulate generated during the pre-processing of wood and is the most important emission at this stage. Wood dust may be caused by intrinsic [91,92] or external factors [93]. Both hardwood and softwood dust have OEL values that should not be exceeded (Table 3). For Western red cedar (WRC), a highly allergenic species of softwood, the limits can be lower. The epidemiological evidence of the carcinogenicity for softwood dust is more limited than for hardwood dust [92]. But, in general, the harder the wood, the more tightly packed the cells are; therefore, hardwoods experience greater fragmentation, which generates more dust [94].

Within the limits of UFP, we can also find nanocellulose. Nanocellulose is a wood derivative that comprises particles and nanofibers with at least one of their dimensions in the nanoscale. Nanocellulose can be highly crystalline (cellulose nanocrystals, CNC) or rather fibrillated and more amorphous (microfibrillated cellulose: MFC, bacterial cellulose: BC, and cellulose nanofibrils: CNF). Due to the promising

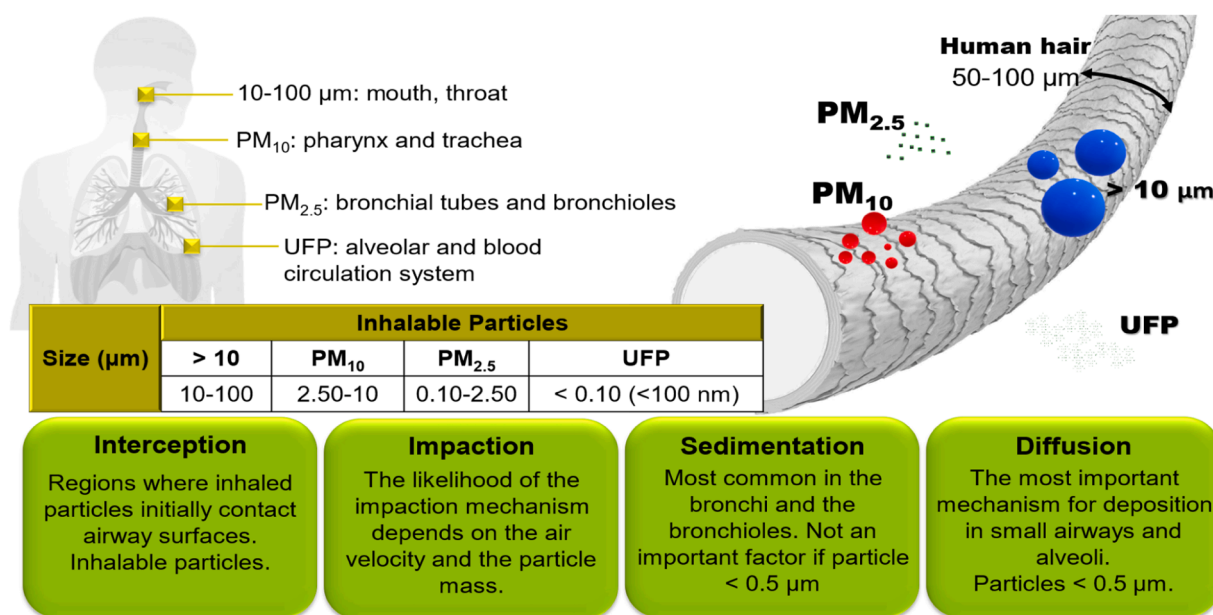


Fig. 3. Summary of aerodynamic equivalent diameters (AED), mechanisms (interception, impaction, sedimentation, and diffusion), and predicted place deposition of inhalable particles in the respiratory tract.

**Table 3**  
Information about time-weighted average (TWA) values established for wood respirable dust.

Wood	TWA (mg/m <sup>3</sup> )	Refs.
Softwood	OSHA ≤5 (PEL) NIOSH ≤1 (REL) Some countries in Europe adopted ≤1–3 (vary between countries) UK ≤5 Western red cedar wood not included	[69,95,96]
Hardwood	EU regulation ≤2 (when hardwood dust is mixed with other wood dust, the limit value concerns all the mixture) NIOSH ≤1 UK ≤3 (when hardwood dust is mixed with other wood dust, the limit value concerns all the mixture) Western red cedar wood not included	[67,95]
Western Red Cedar	OSHA ≤2.5 CAL/OSHA ≤0.5 (TLV, PEL-TWA)	[68,69,96]

PEL: Permissible Exposure Levels. REL: Recommended exposure limit. TLV: Threshold limit value. TWA: Average exposure over a specified period, usually a nominal eight hours

mechanical properties of nanocellulose [97], these nanomaterials have been proposed to reinforce thermoplastic polymers, including PE [98–100], PLA [101], and PP [102]. Assessment of the toxicological profile of nanocelluloses is thus important regarding, e.g., exposure during production processes. Table 4 summarizes the nanocellulose average sizes [103–111], where one or both dimensions belong to the UFP group.

Regardless of whether nanocellulose forms a nanofibril or a nanoparticle, its dimensions are in the order of nanometers which allows it to reach the lung alveoli if inhaled [112]. Currently, nanocellulose is not included specifically in regulation like another nanomaterials (gold, silver, platinum, copper nanoparticles, or carbon nanotubes) [113]. Therefore nanocellulose is included within UFPs, without additional specifications. For example, OSHA established a limit of 15 mg/m<sup>3</sup> for total dust and 5 mg/m<sup>3</sup> for respirable dust. Whilst, NIOSH recommended 5 mg/m<sup>3</sup> for the respirable fraction. However, these regulations do not mention anything specific about nanocellulose [114].

### 5.1. Wood dust during wood pre-processing

The dust generation by induced factors comprises processes like grinding or size classification of wood particles, as well as during transportation within the same mill. The higher fraction sizes of wood dust consist of particles between 10–100 µm [115]. Particles > 10 µm are found in high percentages in primary wood processing mills or during wood panel production [116,117]. Despite these particles being inhalable, the sizes are relatively large for the respiratory tract. Hence, they tend to be deposited in the external or thoracic area of the

**Table 4**  
Average sizes of different nanocellulose classes.

	Production	Mean Length (nm)	Mean Width (nm)	Shape	Refs.
CNC	Chemical hydrolysis and sonication	100–300 (Wood) 100–500 (Cotton) 70–200 (Ramie) 100–300 (Sisal)	3–5 (Wood) 5–10 (Cotton) 5–15 (Ramie) 3–5 (Sisal)	Rod-like (L/D~10–100)	[105]
CNF	Mechanical fibrillation, disc grinder, and homogenization	500–3000	~20–60	Ribbon-shaped fibrils (L/D~150)	[106,111]
CNF-TEMPO	Oxidation and mechanical fibrillation	3000	3–4	Ribbon-shaped fibrils (L/D~1000)	[104]
CNF-Enz	Pretreatment with enzymes followed by mechanical fibrillation	200–800	4–12	Ribbon-shaped, aspect ratio (L/D~10–200)	[108]
BC	Bacterial cellulose	100–1000	< 100	Ribbon-shaped, aspect ratio (L/D~50)	[107,109,110]

L/D: Ratio between length and diameter of the nanocellulose.

respiratory system (including the pharynx). However, the coarse PM<sub>10</sub> and fine particles PM<sub>2.5</sub> reach the area of the lower respiratory tract: particles up to 10 µm penetrate the trachea region, particles smaller than 3 µm reach the bronchial tubes and those smaller than 2 µm reach the bronchioles [118]. Finally, according to Leikauf et al. [119], UFPs reach the alveolar region, enter the blood circulation system, and exacerbate respiratory diseases (bronchial asthma, lung fibrosis, or cancer) [119].

### 5.2. Particle emissions from biocomposite processing

Studies on particle emissions during extrusion and injection molding are relatively limited and some notable efforts have been made to quantify these emissions [43,120]. On the other hand, wood-based 3D printing filaments have gained popularity in recent years. The emission of particles was evaluated in extruders available for home use. Byrley et al. [43] applied a scanning mobility particle sizer (limit of detection: 15.1 nm) and ultrafine particle counter (minimum detection: 20.0 nm) to evaluate the particles generated by a commercially available 3D printer filament extruder at a temperature of 180°C. The authors found that the highest particle emissions occur at the beginning of extrusion, in the first 20 min.

Particle number emission rates were higher for ABS particles (3.5 × 10<sup>11</sup> particles/min) than for PLA particles (5.6 × 10<sup>10</sup> particles/min). The particle sizes varied between 20–300 nm, but during the peak of the emission values, the emitted particles were below 100 nm. The analysis also demonstrated the persistence of the particles in the environment, i. e., UFP were detectable 70 min after the extrusion of the ABS filament finished and 80 min in the case of PLA, with the environment permanence being greater for PLA-emitted particles [43].

A comprehensive analysis of filament production and 3D printing of biocomposite materials with 15 % and 30 % wood sawdust (mixture of 40 % birch, 40 % spruce, and 20 % aspen) and PLA was performed by Väisänen et al. [76]. In both the extrusion process and in 3D printing, UFP predominated (up to a thousand times more), followed by PM<sub>10</sub>, and finally, in smaller quantities, PM<sub>2.5</sub> particles, with the concentration of particles being higher in the 3D printing stage. It was also shown that the majority of UFP emitted during 3D printing comes from filaments with a relatively high proportion of sawdust [76].

## 6. Toxicological effects

### 6.1. VOCs emitted from wood-based components

VOC exposure does not always result in immediate hazards but can be associated with causes of chronic health disease and even cancer induction. Most times they cause acute symptoms: headaches, dizziness, irritations, and nausea. The prolonged exposure damages internal organs such as the kidney and the liver [121].

Formaldehyde is one of the most reactive aldehydes and has been classified as a carcinogen (humans and animals) by the International

Agency for Research on Cancer (IARC) [122]. The effects of formaldehyde vary from person to person. In some cases, discomfort during short-term exposure to formaldehyde occurs at very low levels, while in others it is detected at a higher level [123]. However, there exists a correlation between doses and damage: lower doses (0.05–2 ppm) cause neurophysiologic effect, respiration air threshold, and ocular irritation, while high doses can produce lung and lower respiratory tract effects (5–30 ppm), pneumonia and pulmonary edema (50–100 ppm) and finally at doses higher than 100 ppm, death [123]. Inhalation of formaldehyde causes genotoxicity by forming reactive oxygen species (ROS) and DNA damage and asthma [124].

Long-term exposure to low concentrations has been associated with occupational asthma. It has also been reported to be a common cause of rhinitis, allergies, or mechanical blockages due to turbinate hypertrophy. The irritating effect has been associated with morphological changes in the respiratory epithelium [125]. A review study on the carcinogenic effects of formaldehyde after occupational exposure evidenced a limited correlation between formaldehyde occupational exposure and the occurrence of cancer. However, the authors mentioned that it is required to perform more studies to reduce the bias concerning co-exposure to other carcinogenic compounds in the occupational environment [126].

Dosis and exposure period are important in estimating the effects of all VOCs. For example, Ergnsgård et al. [127] studied the dose-effect relation between acetic acid exposure (0.5 and 10 ppm) and acute effects in humans during controlled chamber exposure. The subjective ratings of nasal irritation and smell by volunteers increased significantly with exposure level. Above 10 ppm, an increased tendency to blink was observed, suggesting an irritant effect. However, no effects on lung function, nasal airway resistance, or plasma inflammatory markers were observed during or after exposure [127]. Similarly, Møhalve et al. [128] studied eye irritation and olfactory perception in 12 healthy non-smoker people (reporting no allergies or previous illnesses) after exposure (2 min) to four terpenes ( $\alpha$ -pinene,  $\alpha$ -terpineol, 3-carene, and limonene), and n-butanol. Only a few participants reported eye irritation for  $\alpha$ -pinene and  $\alpha$ -terpineol. The onsets of irritation were 1250 mg/m<sup>3</sup> for 3-carene, and between 1700 and 3400 g/m<sup>3</sup> for limonene (in both cases well above the OELs). Whilst in n-butanol was 750 mg/m<sup>3</sup> (also above

the established OEL). The olfactory perceptions were in all cases lower than the OELs [128].

Inhalation of VOCs is the predominant route of exposure (are absorbed by the lungs and diffused up to the alveoli and capillary blood, depending on the dose and metabolic route). Although VOCs can be absorbed by the eyes, the absorption depends on the thickness of the stratum corneum and external surface area, exposure dose, and chemical properties of the VOC [129].

Through in vitro studies, it has been elucidated that the possible mechanism of how VOCs generate inflammation in humans, is through oxidative stress that stimulates the generation of inflammatory mediators in pulmonary epithelial cells (Fig. 4) [121].

Exposure to VOCs leads to the overproduction of ROS [130] like hydrogen peroxide, superoxide anion, hydroxyl radical, peroxide, etc.

The inflammation is caused by transcription and activator factors and could damage DNA, proteins, and lipids. At the same time, the activation of adaptative immune cells could lead to autoimmune diseases [131]. However, no direct effect has been found between exposure to VOCs and autoimmune diseases. Usually, autoimmune diseases are associated with genetic and environmental factors but some previous studies revealed a possible influence of VOCs in asthma and lungs [132], bones (bone mineral density, osteoporosis) [133,134], skin (eczema, psoriasis) [134,135], and blood (hemolytic anemia, diabetes) [134, 136].

## 6.2. Toxicological effects of nanocellulose as UFP

UFPs can interact with the cells of the lung tissue, and even in some cases enter the bloodstream. Elimination of these UFPs will depend on the ability of alveolar macrophages to phagocytize the particles, and this will depend on the size, shape, and composition of the particles, as well as the conditions of the lung [90].

In vitro studies of nanocellulose toxicity evaluated the (geno)toxic effects after exposure to the nanomaterials. Ventura et al. [137] exposed MG-63 (human osteosarcoma tumor) and V79 (Chinese hamster lung fibroblast) cells to 1.5–12.5  $\mu\text{g}/\text{cm}^2$  of nanocellulose (CNF-TEMPO, CNF-Enz, and CNC). The authors concluded that CNF-TEMPO and CNF-Enz were genotoxic in both mammalian cells, whilst CNCs only in

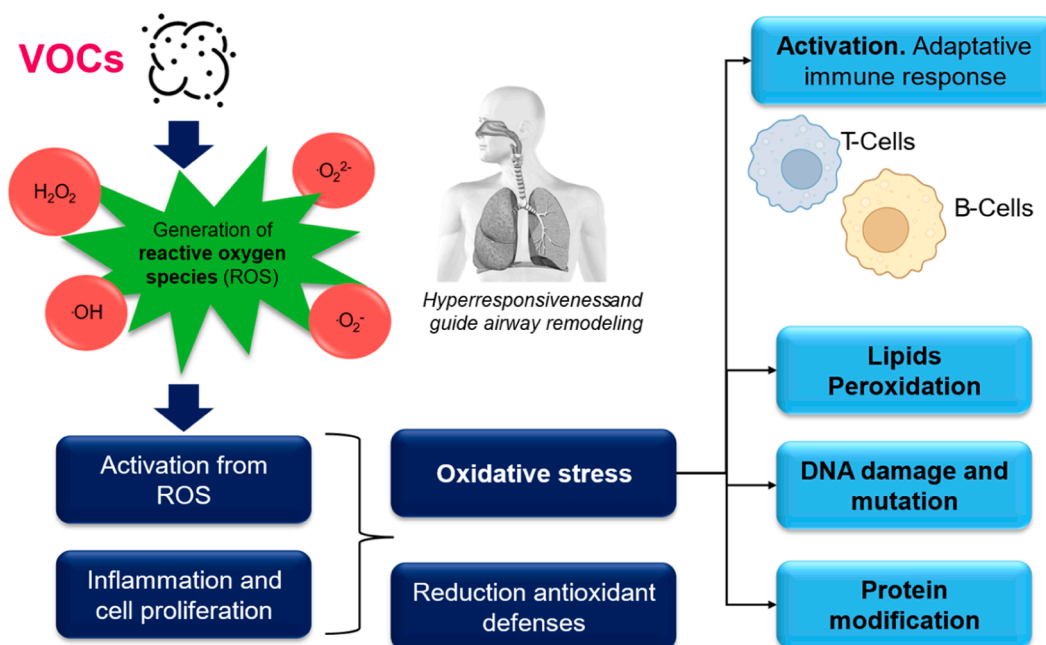


Fig. 4. Consequences in the body of the oxidative stress resulting from the generation of reactive oxygen species (ROS) after inhaling volatile organic compounds (VOCs). Based on [121]. Frontiers, Creative Commons Attribution License (CC BY).

MG-63 cells [137]. Similarly, another in vitro study evaluated the effect of 3.9–500  $\mu\text{g}/\text{mL}$  of unmodified (from bleached sulfite softwood) and modified CNFs (carboxymethylated, modified with hydroxypropyl trimethylammonium, and phosphorylated) on BEAS-2B cells (normal human bronchial epithelium cells) [138]. The authors found that all CNFs showed low cytotoxicity and there were no induced genotoxic effects in the studied cells [138]. However, in a later study using the same cell type, the same authors evaluated the combined effect of size and surface modification of CNFs [139]. Similar cytotoxic effects were detected with all the tested materials, whereas significant genotoxic effects were observed when cells were treated with up to 1000  $\mu\text{g}/\text{mL}$  of fine hydroxypropyl trimethylammonium-substituted CNF (length: 1020 nm, diameter: 4.8 nm), and coarse carboxymethylated CNF (length: 2692 nm, diameter: 17.5 nm) [139]. The authors also found that all modified CNF, unmodified ones (except medium size), and the original pulp were responsible for inducing a dose-dependent generation of intracellular ROS [139]. Hence, secondary genotoxicity, mediated by inflammatory response, could be induced through the generation of ROS triggered by the inflammatory cells [140]. The oxidative stress is induced by the formation of intracellular ROS. As regards inflammation, the innate and adaptive response is triggered because immune system cells can communicate with each other once they have been activated by cytokines (Fig. 5) [139]. Catalán et al. [141] evaluated the inflammatory response of human monocyte-derived macrophages (hMDMs) after exposure to CNC and microcrystalline cellulose (MCC). The authors found a release of pro-inflammatory cytokines IL-1 $\beta$  and TNF- $\alpha$  at high doses of MCC (300  $\text{mg}/\text{mL}$ ), but no cytokine generation was observed when hMDMs were exposed to CNC [141].

Pulmonary and systemic toxicity in vivo was also assessed by the same authors using modified and unmodified CNF [142]. Female mice were administered via repeat pharyngeal aspiration with up to 56  $\mu\text{g}/\text{mouse}$ /administration of each material and the effects were evaluated up to 90 days after the administration. The results showed that exposure to modified CNFs affects the immune response and damages the DNA. The authors concluded that CNF could translocate from the lungs to the liver, which together with the well-known high persistence of CNF raises concerns about long-term effects [142]. Previously, Catalán et al. [143] exposed via single aspiration female mice to 10–200  $\mu\text{g}/\text{mouse}$  of TEMPO-CNF (width 10–25 nm, length up to 1000 nm) for 24 h. The mice developed an immune response to CNF exposure (increased neutrophils, macrophages, eosinophils, and lymphocytes were observed), and DNA damage in the lung cells was detected [143]. Dose-dependent oxidative stress, tissue damage, and inflammatory responses in the lungs were also reported when female mice were exposed, through pharyngeal aspiration, to CNC (50–200  $\mu\text{g}/\text{mouse}$ ) for 24 h

[144].

Besides the above-mentioned effects of size and surface modification, the effect of nanocellulose shape has also been evaluated. A study performed by Li et al. [145] has demonstrated that the size and shape of nanocellulose fibers are determinant factors of the cytotoxic and inflammatory response of macrophages and hepatocytes. The authors applied CNF (length: 6500 nm) and CNC (length: 150 nm) to different mouse macrophages (KUP5, RAW 264.7, and J774A.) and hepatocyte (Hepa 1 – 6) cell lines. All CNCs were cytotoxic when tested up to 200  $\mu\text{g}/\text{mL}$ , while CNF did not induce cytotoxicity for the same concentration range. Above 200  $\mu\text{g}/\text{mL}$ , CNFs showed cytotoxicity in KUP5 cells. An inflammatory response was also observed after CNC phagocytosis by KUP5 cells, inducing the generation of mitochondrial ROS. Besides, the generated lysosomal damage led to the cathepsin B release inducing caspase-1 activation [145].

As happens with other nanomaterials commonly used in bio-composites (i.e., carbon nanotubes, silica nanoparticles), the main health concern associated with nanocellulose is the long bio-persistence in human tissues (i.e., in the lung after being inhaled). This may result in chronic inflammatory effects, leading to fibrosis and cancer development [146]. According to the fiber paradigm, high aspect ratio materials (fiber-shaped) show increased toxicity compared to those with low aspect ratios. In addition, straight and stiff materials, such as some types of carbon nanotubes, induce more severe toxic effects as a consequence of the lack of capacity of macrophages to clear them from the tissue (frustrated phagocytosis) [147]. Furthermore, inhaled nanocellulose has been suggested to induce in the long term a T-helper (Th) 2 type of inflammatory response, which was earlier associated with exposure to asbestos and high aspect ratio nanomaterials, but not with granular nanomaterials. However, nanocellulose did not induce granuloma formation as it has been observed for high aspect ratio materials, which suggests it operates by a different mechanism [142].

In most of the in vitro studies, unmodified nanocellulose was evaluated. Some found factors involved doses, cell type studied, size and shape, surface chemistry, and exposure time [141,143,148–151]. Quin et al. [152] highlighted the limited number of studies on the toxic effect of modified CNFs. The authors mentioned that the expected effects can depend on the type and quantity of functional groups inserted on the surface of the nanocellulose [152]. A recent study also mentioned the effect of the degradation of nanocellulose in the body, that is, if they have a slow degradation they could lead to reactions to foreign bodies by the body [153]. Along the same line, Brand et al. [154] mentioned the need for studies comparing both modified and unmodified nanocellulose to better understand the effect of surface modifications on toxicity. Surface properties could affect the composition, conformation, and

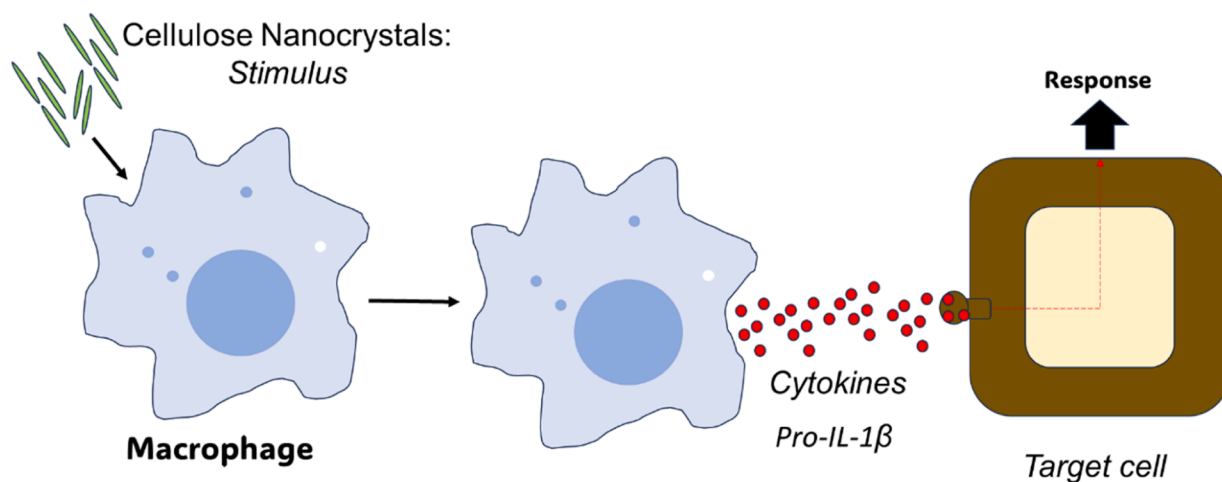


Fig. 5. Mechanism to cytokines generation. Upon recognizing damage, the immune cell generates intracellular signals that activate specific genes to produce cytokines, a key molecule in the immune response.

concentration of adsorbed proteins, and this could lead to cells responding to these changes [153]. Some studies have worked on strategies like the addition of biocompatible functional groups like carboxylates, or amines to reduce the nanocellulose toxicity [139,155]. Even, similar to other nanoparticles, one possibility would be the nanocellulose PEGylation to reduce host immune responses [156].

Based on the existing evidence, exposure to nanocellulose via inhalation may cause health concerns. Therefore, although non-nanocellulose specific, the above-mentioned limits for respirable dust could be applied to protect workers and researchers when handling these materials. In addition, the general principles for preventing risks associated with hazardous chemicals and materials (i.e., in the EU, they are set out in articles 6(1) and (2) of Occupational Health and Safety Framework Directive 89/391/EEC and in article 5 of the Chemical Agents Directive 98/24/EC) can also be applied to the risk management of nanocellulose. In applying the principles, the well-established hierarchy of controls should be followed to ensure the safe handling of nanocellulose [157,158].

## 7. An overview of existing literature

During the production of biocomposites, aspects related to VOC emissions and particulate matter must be considered. This review provided an overview of the compounds emitted at different temperatures and the exposure limits that should not be exceeded in the workspace. Also, toxicity aspects due to prolonged exposure are reviewed. Studies on VOCs emitted in the different stages of obtaining biocomposites are scarce and mostly focused on emissions from the thermoplastic polymers used as a matrix. During the review, we found that the studies that evaluated VOCs produced in pure polymers and biocomposites have shown the great influence of processing temperatures and the components involved in the VOC emissions. The type of components emitted depends on the species of wood and intermediate products, such as wood pulp fibers. Additionally, the temperatures of the pre-processing stage mainly release terpenes, aldehydes, and organic acids. The regulations specify OEL values for these components, mainly aldehydes and organic acids. Also, we observed that, in some countries, OEL values are defined for specific terpenes. It was also found that at temperatures of compounding and conversion stages, the emitted VOCs are more associated with the thermoplastic polymers rather than with the lignocellulosic components. Alkanes, alkenes, acetaldehyde, and acetic acid were the frequent compounds in the evaluated emissions. On the other hand, it can be assumed that pulping or the removal of extractives will reduce the release of aldehydes, terpenes, and organic acids from lignocellulosic material, which would lead to lower emissions in the drying zone and later in compounding and extrusion stages. Comparative studies between VOC emissions from pure polymers and biocomposites are quite scarce, highlighting the need for such studies to reach conclusions regarding VOC emissions. It is also noted that the studies on particulate matter are limited. For particulate matter, the highest emissions were observed during the wood pre-processing stage, although studies also showed emissions during 3D printing. Regulations set specific OELs for hardwoods and softwoods and focus on WRC because it is highly allergenic.

In contrast, the trend was different in toxicity studies, with more existing literature on the toxic effects of exposure to VOCs or particulate matter. However, the studies are mostly in vitro and focus on specific components rather than a comprehensive analysis of all emitted compounds. According to the literature, in vitro studies have shown that factors influencing immune response include the type of nanocellulose, surface modification, size and shape, dose, type of cell studied, and exposure time. The limited number of investigations evaluating the influence of surface modification on toxicity was also observed, demonstrating the extensive development that is still required in this area.

Considering the reviewed literature, it is important to highlight the relevance of controlling emissions in the biocomposite production lines.

This will allow us to establish exhibitions considering health aspects. Human biomonitoring studies exploring the potential health effects associated with VOCs and UFP emissions are urgently needed. However, these studies may still present challenges due to other factors that may be involved, such as variability in the exposure environment.

## 8. Conclusions

This literature review examined studies on VOCs and UFP emissions during biocomposite production. Emissions were observed at the evaluated stages, and it was found that the raw material and temperature are key parameters. Alkanes, alkenes, aldehydes, ketones, organic acids, and terpenes were identified as emitted components. However, existing regulations set OEL values for these compounds, which must be considered in industrial production lines. Prolonged exposure to the emitted compounds may lead to toxicological effects in the body. Hence, proper control of e.g. the biocomposite formulation and temperature during compounding and conversion to reduce emissions, and technologies (e.g., advanced sensors and air purification) to capture potential emissions, should be considered in research and industrial settings. Future research could include a comprehensive assessment of emissions in process lines and, human biomonitoring studies to better understand the potential consequences of prolonged exposure on human health.

## CRedit authorship contribution statement

**Nanci Ehman:** Writing – review & editing, Writing – original draft, Visualization, Investigation, Formal analysis, Data curation, Conceptualization. **Sandra Rodríguez Fabià:** Writing – review & editing, Methodology, Conceptualization. **Julia Catalán:** Writing – review & editing, Supervision, Methodology. **Gary Chinga-Carrasco:** Writing – review & editing, Visualization, Supervision, Resources, Project administration, Methodology, Funding acquisition, Conceptualization.

## Declaration of competing interest

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

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## Supplementary materials

Supplementary material associated with this article can be found, in the online version, at [doi:10.1016/j.jcomc.2025.100595](https://doi.org/10.1016/j.jcomc.2025.100595).

## Data availability

No data was used for the research described in the article.

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