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► To cite this version:

Sylvain Corbel, Christian Mougin, Nouredine Bouaïcha. Cyanobacterial toxins: modes of actions, fate in aquatic and soil ecosystems, phytotoxicity and bioaccumulation in agricultural crops - a review. *Chemosphere*, 2014, 96, pp.1-15. 10.1016/j.chemosphere.2013.07.056 . hal-01687895

HAL Id: hal-01687895

<https://hal.science/hal-01687895v1>

Submitted on 31 Jan 2023

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Cyanobacterial toxins: Modes of actions, fate in aquatic and soil ecosystems, phytotoxicity and bioaccumulation in agricultural crops

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Highlights

Phytotoxic effects of cyanotoxins on agricultural plants have been updated.

We report mechanisms of cyanotoxins and target molecules in vegetable organisms.

The effects of cyanotoxins in the terrestrial environment is particularly scarce.

We describe fate of cyanotoxins in aquatic and soil ecosystems.

We examine bioaccumulation of cyanotoxins in vegetable foods.

Keywords

Cyanobacteria, Cyanotoxins, Microcystins, Anatoxins, Cylindrospermopsin, Fate

Abstract

The occurrence of harmful cyanobacterial blooms in surface waters is often accompanied by the production of a variety of cyanotoxins. These toxins are designed to target in humans and animals specific organs on which they act: hepatotoxins (liver), neurotoxins (nervous system), cytotoxic alkaloids, and dermatotoxins (skin), but they often have important side effects too. When introduced into the soil ecosystem by spray irrigation of crops they may affect the same molecular pathways in plants having identical or similar target organs, tissues, cells or biomolecules. There are also several indications that terrestrial plants, including food crop plants, can bioaccumulate cyanotoxins and present, therefore, potential health hazards for human and animals. The number of publications

concerned with phytotoxic effects of cyanotoxins on agricultural plants has increased recently. In this review, we first examine different cyanotoxins and their modes of actions in humans and mammals and occurrence of target biomolecules in vegetable organisms. Then we present environmental concentrations of cyanotoxins in freshwaters and their fate in aquatic and soil ecosystems. Finally, we highlight bioaccumulation of cyanotoxins in plants used for feed and food and its consequences on animals and human health. Overall, our review shows that the information on the effects of cyanotoxins on non-target organisms in the terrestrial environment is particularly scarce, and that there are still serious gaps in the knowledge about the fate in the soil ecosystems and phytotoxicity of these toxins.

Contents

Introduction

Cyanotoxins and their producers

Modes of actions in humans and mammals and occurrence of target biomolecules in vegetable organisms

Neurotoxins

Hepatotoxins

Cytotoxins

Environmental concentrations of cyanotoxins in freshwaters and fate in aquatic and soil ecosystems

Environmental concentrations of cyanotoxins

Fate in aquatic and soil ecosystems

Phytotoxicity effects of cyanotoxins

Neurotoxins

Hepatotoxins

Cytotoxic alkaloids

Bioaccumulation of cyanotoxins in vegetable foods and consequences on animals and human health

Conclusion and future directions

Acknowledgement

References

Introduction

In light of global climate change, and particularly measurable rises in global temperature, as well as increased fluxes of certain nutrients (i.e., nitrates, phosphates) brought either by agricultural run-off or by sewage treatment plants and other anthropogenic sources, it has been suggested that cyanobacteria, including toxin-producing taxa, may be increasing in abundance, and thus represent an emerging human and environmental health concern (For review see in O'Neil et al., 2012). The presence of such toxins has been reported throughout the world and it appears that liver-toxic microcystins are more commonly found in 40–75% cyanobacterial blooms (Sivonen and Jones, 1999). The contamination of surface waters by these cyanotoxins can cause water quality problems for fisheries, aquaculture, farming, and sanitary hazard for human and animals. Humans are exposed to cyanobacteria toxins through many routes, including drinking water, recreational contact, and health food products made from cyanobacteria, and food chain. In recent years, several cyanobacterial toxins were investigated in regard to their ability to enter the food chain via freshwater seafood (Ibelings and Chorus, 2007; Ettoumi et al., 2011), however, their ability to enter the food chain via agricultural crops has not been thoroughly investigated to date. Although no case of poisoning by these products has been reported in the literature, this eventuality must not be ignored. Indeed, a recent epidemiological study showed that the excessive incidence of amyotrophic lateral sclerosis in the population of the islands of Guam in the Pacific was linked to a consumption of the seeds of cycas contaminated by a neurotoxin, β -methylamino-L-alanine (BMAA), produced by a species of cyanobacteria of the genus *Nostoc* living in symbiosis in the roots of this plant (Banack and Cox, 2003; Cox et al., 2003; Murch et al., 2004; Steele and McGeer, 2008). This last cited fact is gaining importance since plants could in a direct or indirect manner contribute to food chain cyanotoxin's transfer, and by the way constitute a potent health risk source. Indeed, numerous studies reported that both submerged and emergent aquatic plants have been shown to absorb microcystins from low external concentrations (Pflugmacher et al., 1998, 2001; Yin et al., 2005; Saqrane et al., 2007). In terrestrial plants, Codd et al. (1999) reported that spray irrigation of commercial lettuce (*Lactuca sativa*) plants with water containing *Microcystis* resulted in colonies and single cells of the cyanobacterium being lodged on the leaves 10 d after the last irrigation. MC-LR was present at 2.5 mg kg⁻¹ dry weight (DW) in the central leaves, 0.833 mg kg⁻¹ (DW) in the distal zone of mature leaves, and 0.094 mg kg⁻¹ (DW) in the basal zone of mature leaves. The last study indicated that toxins were absorbed by the plant as the central leaves would have been protected from irrigation. Similar conclusions were reached for rice (*Oryza sativa*) and rape

(*Brassica napus*) by Chen et al. (2004). Therefore, the accumulation of cyanotoxins in the terrestrial food chain is at present remains more worrying and the proposed quality limits are rare, indeed, many aspects concerning these toxins are particularly scarce, notably those relative to the fate of cyanotoxins in the soil ecosystems and their toxicity and bioaccumulation on agricultural crops. There have been several reviews of the intensification and global expansion of harmful cyanobacterial blooms in terms of abundance, geographic extent, factors that may be promoting this expansion, and prevention and management of cyanobacterial blooms and their toxins, as well as effects on aquatic ecosystem health and transfer on food webs (Wiegand and Pflugmacher, 2005; Ibelings and Chorus, 2007; Paerl and Huisman, 2009; Aráoz et al., 2010; Kinnear, 2010; Merel et al., 2010; Jancula and Maršálek, 2011; O'Neil et al., 2012). However, the purpose of this review is to: (1) Highlight important findings of the last decade of modes of actions of cyanotoxins in humans and mammals and occurrence of target biomolecules in vegetable organisms; (2) Describe the fate of cyanotoxins in aquatic and soil ecosystems and focus in their phytotoxicity; and (3) Emphasize bioaccumulation of these toxins in vegetable foods and its consequences on animals and human health.

Cyanotoxins and their producers

Recent research suggests that eutrophication and climate change are two processes they may promote the proliferation and expansion of harmful cyanobacterial blooms in freshwater, estuarine, and marine ecosystems. These microorganisms are known to biosynthesize a wide range of chemical classes of secondary metabolites such as peptides, macrolides, and glycosides (Patterson et al., 1994; Namikoshi and Rinehart, 1996) possessing a number of bioactivities: antiviral (Patterson et al., 1993, 1994), antifungal (Patterson et al., 1994), cytotoxic (Patterson et al., 1991), enzymatic inhibitor (Honkanen et al., 1995), antineoplastic (Moore, 1996), and allelopathic (Pushparaj et al., 1998). However, some of these cyanobacterial secondary metabolites encompass a diversity of alkaloid and peptide cyanotoxins which have been suggested to both pose threats to human and environmental health worldwide (Hawkins et al., 1985; Carmichael and Falconer, 1993; Kuiper-Goodman et al., 1999; Sivonen and Jones, 1999; Hitzfeld et al., 2000; Ettoumi et al., 2011). Toxic cyanobacteria that have been involved in such incidents belong essentially to the genera *Microcystis*, *Anabaena*, *Aphanizomenon*, *Planktothrix*, *Oscillatoria*, *Cylindrospermopsis* and less often *Gomphosphaeria*, *Coelosphaerium*, *Gloeotrichia*, *Nodularia* and *Nostoc* (Hawkins et al., 1985; Sivonen and Jones, 1999). The cyanotoxins are essentially endotoxins which can be released in the

environment following a cellular lyse (Codd et al., 1989) or following treatment of cyanobacterial blooms with algaecides (Kenefick et al., 1993). They can be classified into four families according to the organs on which they act: neurotoxins (nervous system), hepatotoxins (liver), cytotoxins (several organs: liver, kidneys, adrenal glands, small intestine), and dermatotoxins (irritant toxins). Cyanobacterial neurotoxins are divided in three groups: anatoxins (anatoxin-a, homoanatoxin-a, and anatoxin-a(s)), saxitoxins, and the neurotoxic amino acid L-beta-N-methylamino-L-alanine (BMAA). Anatoxins and the BMAA are specific of cyanobacteria, while, saxitoxins are also synthesized by some marine dinoflagellates and associated with the human disease paralytic shellfish poisoning or PSP (Falconer, 1991; Carmichael, 1994; Kaebernick and Neilan, 2001). By contrast to the other neurotoxins which production depends on the phylogeny of the species, the BMAA can be produced by almost all groups of cyanobacteria from freshwater, brackish, and marine environments (Cox et al., 2005; Banack et al., 2007). Hepatotoxins are divided into two groups: Microcystins (MCs), cyclic heptapeptide hepatotoxins (MW 900-1200), that are regarded as the most frequently occurring and widespread of the cyanotoxins with more than 80 MC variants already reported (Sivonen and Jones, 1999; Cox et al., 2005; del Campo and Ouahid, 2010); and nodularins (MW 800-900) composed of five amino acids with only nine different natural analogs have been characterized (De Silva et al., 1992; Namikoshi et al., 1993; Rinehart et al., 1994; Codd et al., 2005). The hydrophilic alkaloid cytotoxin, cylindrospermopsin (MW 415) has been identified in the freshwater cyanobacteria *Cylindrospermopsis raciborskii* (Ohtani et al., 1992), *Umezakia natans* (Harada et al., 1994), *Aphanizomenon ovalisporum* (Sivonen and Jones, 1999), *Anabaena sp.* (Schembri et al., 2001), and *Raphidiopsis sp.* (Li et al., 2001). Today, only two congeners of cylindrospermopsin have been identified: 7-epicylindrospermopsin and deoxycylindrospermopsin. The freshwater cyanobacterial irritant toxins such as lipopolysaccharides (LPS), or endotoxins as they are commonly called, are major components of the cell wall in most Gram-negative bacteria including cyanobacteria (Jann and Jann, 1984; Mayer and Weckesser, 1984; Kaya, 1996; Stewart et al., 2006).

Modes of actions in humans and mammals and occurrence of target biomolecules in vegetable organisms

Neurotoxins

Anatoxin-a is a potent postsynaptic depolarizing neuromuscular blocking agent that affects both nicotinic and muscarinic acetyl choline receptors (Carmichael et al., 1979; Spivak et al., 1980). It acts as a depolarizing neuromuscular blocking agent mimicking the action of acetylcholine. However, this neurotoxin is not degraded by the acetylcholinesterase, and consequently its action on the muscular cells does not stop and, due to being stimulated, these cells are blocked and thereby resulting to muscle paralysis (Carmichael, 1994; Lilleheil et al., 1997). When the respiratory muscles are affected, the insufficient oxygenation of the brain engenders convulsions and the oppression (Carmichael, 1994; Humpage et al., 1994). The LD50 (lethal dose resulting in 50% deaths) of this neurotoxin is $200 \mu\text{g kg}^{-1}$ (mouse, i.p.) (Carmichael et al., 1979; Skulberg et al., 1992). Homoanatoxin-a is a homologue of anatoxin-a, that was reported to be a potent nicotinic agonist (Wonnacott et al., 1992). It enhances the release of acetylcholine from peripheral cholinergic nerves through opening of endogenous voltage dependent neuronal L-type calcium channels (Aas et al., 1996; Lilleheil et al., 1997). Despite the similarity in their names, anatoxin-a(s) and anatoxin-a are not structurally related and exhibit different physiological properties. Anatoxin-a(s) belongs to the organophosphate class of neurotoxins and it acts as an irreversible inhibition of acetylcholinesterase at the nerve synapse (Mahmood and Carmichael, 1986, 1987). The LD50 (mouse, i.p.) of this toxin is about $20\text{--}40 \mu\text{g kg}^{-1}$ (Mahmood and Carmichael, 1987; Matsunaga et al., 1989; Carmichael et al., 1990). In animals, the mechanisms of action of PSP toxins (saxitoxins) are based on the blockage of sodium conductance in axons (Kao et al., 1967; Henderson et al., 1973). They so inhibit the transmission of the electric activity and prevent the liberation of the acetylcholine (Nishiyama, 1968). Their toxicity is more important than that of anatoxins with a LD50 (mouse, i.p.) in the same conditions for the saxitoxin of $10 \mu\text{g kg}^{-1}$ (Gorham and Carmichael, 1988). Saxitoxins can also bind to calcium (Ca^{+2}) and K^+ channels, interfering with the speed of opening and closing of these channels, which can in turn lead to alteration in the influx of ions to the cell (Wang et al., 2003; Su et al., 2004). In addition, the Na^+ -channel blockage may alter the selective permeability of the membrane and may change the flow of ions, leading to damage to cellular homeostasis (Hille, 1992; Jablonski et al., 2007). Concerning the neurotoxic amino acid (BMAA), it acts in mammals as a glutamate agonist at AMPA, kainite and

NMDA receptors (Spencer et al., 1986, 1987; Andersson et al., 1997; Seawright et al., 1999). Consequently, it increases the intracellular concentration of calcium in neurons and induces neuronal activity by hyperexcitation (Brownson et al., 2002). To our knowledge, no data regarding the toxicity of cyanobacterial neurotoxins in higher plants have been reported. However, interfering of some of them such as saxitoxins with the speed of opening and closing of Na^+ , Ca_2^+ and K^+ channels could modify ions transport in plant cells. For example, a modification of sodium signals can modify osmotic pressure in cells or the assimilation of CO_2 for C4 plants (Brownell and Crossland, 1972). While sodium extrusion in animal cells and microorganisms (including yeast) is directly energized by ATP hydrolysis (Na^+ -ATPases), these Na^+ pumps are absent from higher plants (Horie and Schroeder, 2004).

Hepatotoxins

Cyanobacterial hepatotoxins type microcystin-LR are generally not able to penetrate vertebrate cell membranes and therefore, require uptake via the bile acid transport system present in hepatocytes and cells lining the small intestine (Runnegar et al., 1991). As a result of this, toxicity of these cyanotoxins is restricted to organs expressing the organic anion transporter on their cell membranes such as the liver (Fischer et al., 2005). However, in vegetable cells one relatively unexplored question regarding these toxins concerns the mechanism of uptake, particularly the variants that would be predicted to be membrane impermeable based on polarity. They may cross cell membranes of plants by other mechanisms, including diffusion or by root absorption. Pflugmacher et al. (2001) have been reported that when the emergent reed plant *P. australis* was exposed to $0.5 \mu\text{g}$ of ^{14}C -labeled microcystin-LR L^{-1} for 3 d, it demonstrated a rapid uptake (since 0.5 h) of the toxin. The main uptake route appeared to be in the stem and rhizome, from which the toxin is transported into the higher parts of the plant to the leaves. Uptake directly through the leaves may also occur by direct contact of small plants or by the lowest leaves of a plant with surface water and with upper leaves by wave and spray contact (Pflugmacher et al., 2001). Once in both vertebrate and vegetable cells, microcystins and nodularins have been shown to be potent and specific inhibitors of protein phosphatases 1 and 2A, and this inhibition accounts for their extreme toxicity (MacKintosh et al., 1990; Kurki-Helasmo and Meriluoto, 1998; Hastie et al., 2005). Those proteins are involved in several physiological and molecular processes in higher plants (Sheen, 1993; Takeda et al., 1994). Indeed, numerous studies reported that microcystins have several perturbatory

effects on plant physiology and metabolism, when sufficient levels of toxin enter the plant cells (MacKintosh et al., 1990; Siegl et al., 1990; Sheen, 1993; Yamasaki, 1993; Smith et al., 1994; Takeda et al., 1994; Abe et al., 1996; Smith, 1996; Kurki-Helasma and Meriluoto, 1998; Weiss et al., 2000; McElhiney et al., 2001; Pflugmacher, 2002; Romanowska-Duda and Tarczynska, 2002; Gehringer et al., 2003; Chen et al., 2004, 2011; Mitrovic et al., 2005; Saqrane et al., 2007, 2008; Stüven and Pflugmacher, 2007; Järvenpää et al., 2007; Jang et al., 2007; Peuthert et al., 2008; Máthé et al., 2009; Huang et al., 2009; El Khalloufi et al., 2011, 2012; Jámbrik et al., 2011; Perron et al., 2012). On the other hand, several studies have also reported that these hepatotoxins induce oxidative stress in mammal cells (Zegura et al., 2003; Botha et al., 2004; Bouaïcha and Maatouk, 2004; Puerto et al., 2010). Therefore, their toxicity on aquatic plants seems to be also more linked to the induction of oxidative stress manifested by elevated reactive oxygen species (ROS) production and malondialdehyde (MDA) content (Lefevre et al., 1950; Pflugmacher, 2004; Hu et al., 2005; Leflaive and Ten-Hage, 2007).

Cytotoxins

The alkaloid cylindrospermopsin (CYN) is known as a general cytotoxin that blocks protein synthesis in mammal cells (Runnegar et al., 1995; Froscio et al., 2001, 2003). Implications of this effect can be also observed in vegetable cells. In fact, Froscio et al. (2008) reported that CYN was shown to inhibit the eukaryotic protein synthesis apparatus with similar potency in plant and mammalian cell extracts, IC₅₀ of 334 nM in wheat germ extract and 110 nM in reticulocyte lysate. Metcalf et al. (2004) also showed that CYN inhibited pollen germination in tobacco plants (*Nicotiana tabacum*), with partial inhibition of protein production in the germinating pollen tubes following exposure to 138 µg mL⁻¹ of toxin.

Environmental concentrations of cyanotoxins in freshwaters and fate in aquatic and soil ecosystems

Environmental concentrations of cyanotoxins

The occurrence of cyanobacterial toxins was reported throughout the world in surface waters, where hepatotoxic microcystins are more commonly found in 50–75% cyanobacterial blooms (Ettoumi et al., 2011). Data on environmental concentrations of cyanotoxins have been compiled and reviewed in numerous papers (Sivonen and Jones, 1999; Falconer and Humpage, 2006; Van

Apeldoorn et al., 2007; Messineo et al., 2009). In this review, we give a summary on environmental concentrations focusing on irrigation waters with the ultimate aim to relate them to phytotoxicological data. Cyanotoxins are intracellular toxins contained within living cells, depending on both the nature of the toxin and the growth stage (Jungmann et al., 1996; Orr and Jones, 1998; Park et al., 1998a,b; Sivonen and Jones, 1999). They are only released into the water, to form dissolved toxin, during cell senescence or cell death and lysis or through water treatment processes such as algaecide application, rather than by continuous excretion (James and Fawell, 1991; Gupta et al., 2001; Babica et al., 2006). The highest total (intracellular plus dissolved) cyanotoxin levels have been found in blooms and scums. For example, total MC concentrations in surface waters vary from trace to several milligrams per liter, being strongly influenced by the occurrence of these forms of cyanobacterial biomass. In surface waters used as irrigation source, total MC concentrations of 4-50 $\mu\text{g L}^{-1}$, up to 6500 $\mu\text{g L}^{-1}$, have been reported in multiple locations, including but not limited to the Morocco (Oudra et al., 2001), Tunisia (El Herry et al., 2008), India (Prakash et al., 2009), Turkey (Gurbuz et al., 2009), and Finland (Spoof et al., 2003), but much higher levels up to 29000 $\mu\text{g L}^{-1}$ in Algeria (Nasri et al., 2008) (Table 1). It should be noted, however, that these very high concentrations of cyano- toxins would be from scums or from very dense cyanobacterial biomass. In the field, water samples with more than 1 $\mu\text{g L}^{-1}$ total MCs, dissolved fraction did not comprise more than 10% of the combined intra and extracellular pool (Lindholm and Meriluoto, 1991; Jones and Orr, 1994; Tsuji et al., 1996; Ueno et al., 1996; Lahti et al., 1997). As well in some laboratory studies, where both intracellular and extracellular cyclic peptide toxins and STXs have been measured, it is generally the case that in healthy log phase cultures, less than 10–20% of the total toxin pool is dissolved in the culture medium (Sivonen et al., 1990; Lehtimäki et al., 1997; Negri et al., 1997; Rapala et al., 1997). On the contrary, CYN may often be found at higher levels in dissolved form than within cells, as it readily leaks from cells under normal growth conditions (Norris et al., 2001; Falconer and Humpage, 2006; Wörmer et al., 2008). For example, Shaw et al. (1999) found that in two instances of *A. ovalisporum* blooms around 80% of the total toxin content of the water was in free solution. Recently, Messineo et al. (2009) reported that in several Italian lakes of different characteristics and human uses, extracellular concentrations of total CYN varied from non-detectable values up to 126 $\mu\text{g L}^{-1}$. However, limited or no information is available about the proportion of dissolved form with respect to the total level for the cyanobacterial neurotoxins.

Fate in aquatic and soil ecosystems

Once they enter in aquatic and soil ecosystems, cyanotoxins can be removed according to various processes such as photochemical degradation by UV, adsorption in particles in suspension or onto sediments, and biodegradation (Tsuji et al., 1994; Rapala et al., 1994; Lahti et al., 1996; Chiswell et al., 1999; Welker and Steinberg, 1999; Kaebernick and Neilan, 2001; Mazur-Marzec et al., 2006; Wörmer et al., 2008; Burns et al., 2009; Klitzke et al., 2010, 2011; Thirumavalavan et al., 2012). However, the four groups of cyanotoxins: hepatotoxins, neurotoxins, cytotoxins, and dermatotoxins, exhibit quite different chemical stabilities in these ecosystems. Hepatotoxin cyclic peptide cyanotoxins, microcystins and nodularins, are extremely stable compounds and may persist in aquatic systems for weeks after being released from the cells (Jones and Orr, 1994; Chen et al., 2008; Edwards et al., 2008). According to other studies, these toxins in natural conditions could persist for several months or years (Harada et al., 1996; Sivonen and Jones, 1999). However, numerous studies reported that photochemical degradation by sunlight UV and exposure to degrading bacteria may speed up their removal from the water (Bourne et al., 1996; Heresztyn and Nicholson, 1997; Sivonen and Jones, 1999; Park et al., 2001; Song et al., 2009; Ho et al., 2012). The photodegradation of MCs in full sunlight can take as little as two weeks or longer than six weeks, depending on the presence of water-soluble cell pigments (Tsuji et al., 1994; Welker and Steinberg, 2000). More recently, Thirumavalavan et al. (2012) showed in a laboratory experiment that the presence of humic acid and turbidity affected the photo-degradation process. Additionally, in sea water the rate of nodularin photolysis can be accelerated by the presence of some cell components and humic substance (Welker and Steinberg, 1999). Conversely, during the benthic phase, the photodegradation of these cyanotoxins is expected to be almost negligible due to low radiation penetration (Wörmer et al., 2010). In fact, Welker and Steinberg (2000) found that the half-life of MCs in the deep lakes is longer than the season of cyanobacteria growth, what suggests that the photolysis is significant only for shallow lakes. The alkaloid cytotoxin, CYN, is relatively stable in the dark; however, in sunlight and in the presence of cell pigments degradation occurs quite rapidly with more than 90% within 2–3 d (Chiswell et al., 1999). The neurotoxin, anatoxin-a, is also relatively stable in the dark, but it undergoes rapid photochemical degradation in sunlight particularly in alkaline conditions, even in the absence of cell pigments (Stevens and Krieger, 1991; Smith and Sutton, 1993). However, no data are available for other cyanobacterial neurotoxins and LPS dermatotoxins.

Cyanotoxins can also be retained on suspended particles or onto sediments in aquatic systems.

Wörmer et al. (2011) showed the great importance of sedimentation processes in the fate of MCs in freshwaters with an amount of toxin associated to settling particles to be in the range of $\text{mg d}^{-1} \text{m}^{-2}$. But other studies reported that no more than 20% of toxins can be adsorbed on sediments (Rapala et al., 1993; Lahti et al., 1996). Furthermore, it was suggested that the removal of cyanotoxins in this process was the result of both adsorption and biodegradation (Lahti et al., 1996). Therefore, biodegradation would appear to be the main fate for most cyanotoxins in aquatic systems and the relative performance of this process would be very site specific and dependent upon local sediment characteristics and microbial activity. It was recently reported that the data generated in laboratory and field studies strongly indicate that, in shallow lakes, low persistence and natural eliminations of MCs are due to biodegradation; suggesting that sediments play a crucial role in biodegradation by continuously supplying toxin-degrading bacteria to the water column (Chen et al., 2008, 2010; Mazur-Marzec et al., 2009). However, in deep sediments, biodegradation might be limited due to anoxic conditions (Holst et al., 2003; Grützmaier et al., 2002, 2010) and sediments only bring nutrients for bacteria responsible of cyanotoxins biodegradation. Degradative heterotrophic bacteria of hepatotoxic cyanotoxins (MCs and NOD), and cytotoxins (CYN) have been found in various media, such as water columns (Jones and Orr, 1994; Cousins et al., 1996; Christoffersen et al., 2002; Hyenstrand et al., 2003; Lemes et al., 2008; Mazur-Marzec et al., 2009; Chen et al., 2010), sediments (Rapala et al., 1994; Holst et al., 2003), sewage effluents (Lam et al., 1995) or soils (Miller et al., 2001; Grützmaier et al., 2002), with specific enzymatic pathways well characterized (Bourne et al., 1996; Okano et al., 2009; Zhang et al., 2010). Several previous studies have been indicated that MCs can be degraded by aquatic bacteria identified as pertaining especially to the genus *Sphingomonas* (Bourne et al., 1996; Harada et al., 2004; Ishii et al., 2004; Maruyama et al., 2006; Manage et al., 2009). Therefore, a microcystin-degrading gene cluster, *mlrA*, *B*, *C* and *D* was identified in these microorganisms, sequenced and the degradation process was proposed (Bourne et al., 2001; Saito et al., 2003; Imanishi et al., 2005). In the last two decades, several other species of bacteria capable of degrading peptidic cyanotoxins were identified, *Sphingomonas sp.* strain ACM-3962 (Jones et al., 1994), *Paucibacter toxinivorans* (Rapala et al., 2005), *Sphingosinicella microcystinivorans* (Maruyama et al., 2006), *Burkholderia sp.* (Lemes et al., 2008). The most toxic congener, Microcystin-LR, was also found susceptible to breakdown by *Sphingomonas*, which initiated ring-opening and the production of a linear compound 200 times less toxic (Bourne et al., 1996). Recently, Ho et al. (2012) identified another bacterium strain (TT25) whose genome is similar to *Sphingopyxis sp.* that it is able to degrade MCs. The ability of

these all species to degrade other congeners of MCs and NODs was investigated and revealed that peptides with the Adda–Arginine bond were successfully degraded while MC-LF, with Adda-Phenylalanine bond and 6(z)-Adda-MC-LR and 6(z)-Adda-MC-RR were not significantly degraded (Imanishi et al., 2005). Another Japanese *Sphingomonas* isolate, 7CY, was shown to degrade a wider range of MCs, including MC-LR, -RR, -LY, -LW, and -LF but it was unable to degrade NOD-Har a NOD analog where arginine is replaced by homo-arginine (Ishii et al., 2004). Biodegradation has also been shown to be an important process for the removal of the alkaloid cytotoxin, CYN, from contaminated water (Chiswell et al., 1999; Senogles et al., 2002). By contrast, a laboratory study investigating biodegradation of CYN with bacterial communities from two water bodies in Spain, one having frequent exposure to CYN, the other rarely, has been shown that biodegradation of this toxin by an active microbial community does not take place during a 40-d (Wörmer et al., 2008). A recent study demonstrated that CYN was degraded by indigenous microbial flora in waters with a history of *Cylindrospermopsis* blooms (Smith et al., 2008). Despite isolation of many bacteria from CYN enriched cultures, only a single isolate (*Delftia* sp.) capable of degrading CYN has been obtained (Smith, 2005). However, for cyanobacterial neurotoxins there are few reports on their persistence and biodegradation compared to cyanobacterial heptotoxins, although the increasing occurrence of these toxins in surface waters. A recent study indicated that saxitoxins (STXs) are predisposed to bacterial degradation during passage through bioactive treatment plant (Kayal et al., 2008). However, this study showed that structural modification during the biological treatment resulted to decrease of the predominant C-toxins variants and an increase in GTX2 and GTX3 which are more toxic than the C-toxins. Early work by Kiviranta et al. (1991) reported the isolation of a *Pseudomonas* sp. capable of rapid degradation of anatoxin-a, with a rate of 6-30 mg mL⁻¹ per 3 d. A later study reported by Rapala et al. (1994) has been shown the removal of anatoxin-a by microbial populations isolated from water and sediments of a eutrophic, oligotrophic, and humic lake. In conclusion, the period of photodegradation of cyanotoxins is relatively long in comparison to the degradation caused by the microbial activity. Recently, Hu et al. (2012) found that the *Bacillus* sp. strain EMB is able to completely remove 2.99 mg L⁻¹ of MC-RR and 2.15 mg L⁻¹ of MC-LR within 24 h. However, the biodegradation speed of cyanotoxins in aquatic ecosystems can be influenced by the initial concentration and nature of toxins (Edwards et al., 2008; Ho et al., 2012) and by additional factors such as the water temperature (Park et al., 2001; Ho et al., 2007a,b; Smith et al., 2008; Hoefel et al., 2009) and the bacterial community composition within the water body; not only the types of organisms present, but also their abundance (Hoefel et al., 2009; Ho et al., 2012). Hoefel et

al. (2009) have demonstrated a direct relationship between the abundance of degrading organisms and the rate of degradation of MC-LR. Furthermore, although MCs are degraded by most of bacteria species, it seems that a lasting day's delay or weeks are necessary before the degradation is introduced. This result was in agreement with the conclusion of Hyenstrand et al. (2003) indicating that bacteria species have to adapt themselves at first to the cyanobacteria metabolites before the degradation of MCs becomes effective. Indeed, the results of this last study indicate a weaker degradation of the MC-LR in May compared with September where the occurrence of cyanobacteria is higher. Similarly, Smith et al. (2008) found that CYN was degraded by indigenous microbial flora in waters with a history of *Cylindrospermopsis* blooms.

The physicochemical fate and the environmental concentrations of cyanotoxins in soil have been the subject of a range of recent studies. Several classes of these toxins have been detected in field soils, and the sorption behavior and degradation and transfer to vegetables have been studied to a large extent (Morris et al., 2000; Miller et al., 2001; Chen et al., 2006b; Bibo et al., 2008; Sathishkumar et al., 2011). The use of water from sources containing cyanobacterial blooms and toxins for spray irrigation of terrestrial plants, including food crop plants presents both a harmful effect on growth and development of plants and on soil ecosystems and potential health hazards through several exposure routes, including uptake into the food chain and accumulation of toxins on the external surfaces of edible plant material. Questions, therefore, arise about the persistence of total cyanotoxins (dissolved and within the cyanobacterial cells) when reach the soil ecosystem to produce phytotoxic effects. Once reach the soil ecosystem, cyanotoxins persist in the environment, depending on the efficiency of degradation (i.e., photolysis, hydrolysis and bacterial degradation). Microcystins can persist in agriculture soils for relatively long times, with a half-life ranging between 6 and 17.8 d (Chen et al., 2006b). Jones et al. (1995) reported that scums of *M. aeruginosa* that dry on the shores of lakes may contain high concentrations of MCs for several months. Recently, Metcalf et al. (2012) found that MCs were detected in herbarium specimens of cyanobacteria which had been collected from aquatic and terrestrial environments in 11 countries throughout the world, dried, and stored at ambient temperatures in the dark for up to 170 years. Microcystins were also detected by HPLC and ELISA assays in desert crust samples from Qatar at concentrations between 1.5 and 53.7 ng g⁻¹ dry weight (Metcalf et al., 2012). Thus, the persistence of these toxins within dried cyanobacterial cells for long period suggests that they will be released back into the soil when re-immersed by irrigation water, particularly when cyanobacterial blooms are used in some countries as an organic fertilizer (Chen et al., 2006a,b). However, as mentioned

above for aquatic ecosystems, adsorption on sediments and especially exposure to degrading bacteria may also speed up their removal from the soil.

The information on the adsorption of cyanotoxins in agriculture soil ecosystems is particularly scarce. However, adsorption of cyanobacterial hepatotoxins was measured in several batch studies to determine the applicability of bank filtration as an efficient removal strategy of these toxins from drinking water. For example, in batch experiments Miller et al. (2001) studied the adsorption of cyanobacterial hepatotoxins, MC-LR and NOD, in five soils with different physicochemical properties collected from regions around South Australia. They found that the soils with the high clay and/or organic carbon contents had the higher toxins adsorption coefficients. In similar experiments, Miller and Fallowfield (2001) found that the soils with the highest organic carbon content (2.9%) and the highest clay content (16.1%) were the most effective at removing these toxins in batch experiments. However, the sandy soil (98.5% sand) was incapable of the removal of cyanotoxins. This finding was supported by Morris et al. (2000) who reported that the clay content and its quality may be more important for the adsorption than other soil characteristics. However, Eynard et al. (2000) suggested that soil was unable to protect groundwater from cyanotoxins that originated from surface waters. Thus, it seems that cyanotoxins sorption in soils is low and could potentially result in their high bioavailability to soil organisms and plants. In several studies, it seems that the major dissipation process for cyanotoxins in soil ecosystems is mainly via microbial degradation (Miller and Fallowfield, 2001; Chen et al., 2006b). In fact, numerous soil bacteria as *Arthrobacter* sp., *Brevibacterium* sp. and *Rhodococcus* sp. are able to breakdown MCs (Manage et al., 2009). Bourne et al. (2001) observed the same thing with *Sphingomonas* sp. that possesses a gene cluster involved in the degradation of MC-LR. Furthermore, Falconer et al. (1983) and Lambert et al. (1996) conclude that sand filtration alone is unable to remove dissolved cyanotoxins. However, slow sand filters can be expected to remove 99% of dissolved cyanotoxins (Keijola et al., 1988; Grützmacher et al., 2002). This can be explained by the formation of a biofilm on top of the filter that it allows for some biodegradation of cyanotoxins in slow sand filtration. No data are available for other cyanobacterial toxins such as neurotoxins and dermatotoxins, but some degradation may be expected, again depending on the chemical conditions of soil. In conclusion, the scarce results on the fate of cyanotoxins in soil ecosystems are very variable, which do not allow affirming with certainty the necessary time for a complete disappearance of these toxins. This variability ensues partially from used methods (e.g. studies led in laboratory with non environmental concentrations of toxins and in free-soil systems). Therefore, the fate of cyanotoxins in soil ecosystems will require more studies before we are capable to formulate an opinion on their

persistence and uptake into the food chain.

Phytotoxicity effects of cyanotoxins

The information on the effects of cyanotoxins on non-target organisms in the terrestrial environment is particularly scarce. However, despite the impressive amount of information on their toxicity on mammals compiled during the last two decades, there are still serious gaps in the knowledge about the phytotoxicity of these toxins. The phytotoxic effects of cyanotoxins on higher plants were firstly focused on aquatic photoautotrophic organisms (algae and macrophytes) that are naturally exposed to cyanotoxins (Harper, 1992; Papke et al., 1997; Weiss et al., 2000; Yu et al., 2000; Ikawa et al., 2001; Pietsch et al., 2001; Mitrovic et al., 2004; Ha and Pflugmacher, 2013). Since few years, scientists were also interested by the effect of these toxins on terrestrial plants because, irrigation waters from sources containing cyanobacterial blooms and toxins are generally used without treatment for spraying agricultural crops and plants that might, therefore, induce a food chain contamination with a considerable health risk and potential economic losses.

Neurotoxins

The cyanobacterial neurotoxins have not received more research attention than have cyanobacterial hepatotoxins. This is a consequence of the many livestock deaths caused by cyanobacterial species producing hepatotoxic microcystins and their more widespread occurrence rather than species producing neurotoxins (Ettoumi et al., 2011). In addition, the recent inclusion of microcystin-LR as a toxic chemical in the World Health Organisation (WHO) drinking water guidelines has further accelerated investigation of the toxic effects of microcystins on mammals and vegetables rather than cyanobacterial neurotoxins. Therefore, there are only few studies reported in the literature on the effects of cyanobacterial neurotoxins on crops and plants. Mitrovic et al. (2004) were exposed the free-floating aquatic plant *L. minor* and the filamentous macroalga *Chladophora fracta* to anatoxin-a at 0.1-25 $\mu\text{g L}^{-1}$ under laboratory conditions for 4-7 d. They found in both organisms significantly increase of peroxidase activity after 4 d exposure at 25 $\mu\text{g L}^{-1}$ but not at lower concentrations. After 7 d exposure to this neurotoxin significant increase of GST activity and reduction of photosynthetic oxygen production were observed at 5 and 20 $\mu\text{g L}^{-1}$ but not at lower concentrations in *L. minor*. In addition, Ha and Pflugmacher (2013) reported that this

alkaloid neurotoxin at an environmentally relevant concentration ($15 \mu\text{g L}^{-1}$), induced phytotoxic effects on the submerged aquatic macrophyte *Ceratophyllum demersum*, mediated by oxidative stress. Recently, Esterhuizen-Londt et al. (2011) investigated in vitro study the effect of BMAA at different environmentally concentrations (0.5, 1, 5, 50 and $100 \mu\text{g L}^{-1}$) for 24 h on the oxidative stress responses of the macrophyte *C. demersum*. The most pronounced effects found were activity-inhibiting effects on all the oxidative stress response enzymes at all exposure concentrations. However, enzymes not related to oxidative stress response were not affected by the BMAA in these experiments. For other neurotoxins, the literature search did not yield any results.

Hepatotoxins

The effects of cyanobacterial toxins on photoautotrophic organisms have been most intensively studied for MCs, in line with their abundance and their mode of action. First experiments were focused on the ability of these hepatotoxins to act as general allelopathic compounds against planktonic microalgae, macroalgae and macrophytes in aquatic ecosystems. The allelopathic effects of Aphanizomenon and other cyanobacteria bloom formers on chlorophyte species are early documented in several studies (Lefevre et al., 1950; Tassigny and Lefevre, 1971; Boyd, 1973). Subsequently, Ikawa et al. (2001) and Papke et al. (1997) observed that cyanobacterial metabolites can induce the growth inhibition of the green alga *Chlorella pyrenoidosa* and the photosynthesis of other cyanobacteria species, respectively. Similarly, Sukenik et al. (2002) found that *Microcystis* sp., a MCs producer, severely inhibited the growth of the freshwater dinoflagellate *Peridinium gatunense* in mixed laboratory cultures which was attributed to the excretion of allelopathic substances rather than to successful competition for nutrients. Hu et al. (2005) found that the growth of *Synechococcus elongatus* was reduced by 53.6% after 6 d of exposure to $100 \mu\text{g L}^{-1}$ of MC-RR suggesting that oxidative stress manifested by elevated ROS levels and MDA contents might be responsible for the toxicity of MC-RR to this species. Moreover, Singh et al. (2001) demonstrated that MCs are strongly algicidal and point to the possibility that they may have an important role in establishment and maintenance of toxic blooms of *M. aeruginosa* in freshwater ecosystems. Valdor and Aboal (2007) demonstrated the inhibitory effect of both cyanobacterial extracts and pure MCs on the growth of microalgae. Bártová et al. (2010) examined effects of semipurified *Microcystis* extract containing MCs (0.2-20 nM) on age-induced cell differentiation of the filamentous cyanobacterium *Trichormus variabilis* and they found that heterocyst and akinete formation was significantly decreased after exposure to extract containing 2 or 20 nM of MCs

within 10 d of exposure. Recently, Perron et al. (2012) evaluated the effect of four microcystins standards (variants MC-LF, -LR, -RR, -YR) at different concentrations ($0.01-10 \mu\text{g mL}^{-1}$) and 0.01 , 0.1 , and $1 \mu\text{g mL}^{-1}$ equivalent microcystins extracted from *Microcystis aeruginosa* (CPCC299), which is known to produce mainly MC-LR, on the fluorescence of four green algae (*Scenedesmus obliquus* CPCC5, *Chlamydomonas reinhardtii* CC125, *Pseudokirchneriella subcapitata* CPCC37 and *Chlorella vulgaris* CPCC111) and how they can affect the flow of energy through photosystem II. Their results showed that MCs affect the photosynthetic efficiency and the flow of energy through photosystem II from $0.01 \mu\text{g mL}^{-1}$ within only 15 min and that MC-LF was the most potent variant, followed by MC-YR, -LR and -RR.

It was also noticed that in eutrophic freshwaters dominated by cyanobacteria, a decrease in species diversity and in the growth of macrophytes often occurs (Harper, 1992; Weiss et al., 2000; Yu et al., 2000; Pietsch et al., 2001). Casanova et al. (1999) found that the abundance and the variety of macrophytes are reduced in the presence of cyanobacterial blooms. In 1986, Kirpenko showed for the first time the inhibition growth of water plants Elodea and Lemna by MCs isolated from a natural bloom. This allelopathic action was recently confirmed by Weiss et al. (2000) further to the coculture of the plant *Lemna minor* with the cells of *M. aeruginosa*. Moreover, Pflugmacher (2002) revealed that MC-LR induces allelopathic effects on the aquatic macrophytes such as *C. demersum* and *Myriophyllum spicatum*, resulting in growth inhibition, reduction in photosynthetic oxygen production, and changes in pigment pattern. Jang et al. (2007) found by examining cyanobacterial toxin production in response to direct exposure to an axenically cultured aquatic plant (*Lemna japonica* Landolt) using two toxic monoclonal strains of *M. aeruginosa* Küzing (NIES strains 103 and 107) that reciprocal allelopathic responses have been observed between these two species *Microcystis* and *Lemna*. In several other studies, it occurred that MCs have the potential to exert toxic effects on growth and physiological processes, which all might be related to the inhibition of protein phosphatase activity or oxidative stress in aquatic moss (Wiegand et al., 2002) and in higher aquatic plants such as *Lemna gibba* (Saqrane et al., 2007), *Lemna* genus (Mitrovic et al., 2005), *L. japonica* (Jang et al., 2007), *Spirodela oligorrhiza* (Romanowska-Duda and Tarczynska, 2002), *Phragmites australis* (Yamasaki, 1993; Máthé et al., 2009; Jámbrik et al., 2011), and *C. demersum* (Pflugmacher, 2004).

There are also several indications that terrestrial plants, including food crop plants, can be altered by MCs present in irrigation waters, resulting principally to their serine/threonine phosphatases inhibition and reactive oxygen species (ROS) production. Sheen (1993) found that the marine

phycotoxin okadaic acid, a potent inhibitor of serine/threonine protein phosphatases like MCs, efficiently blocks chlorophyll accumulation induced by light in etiolated maize leaves. It seems also that this phycotoxin blocks root hair growth and alter cortical cell shape of *Arabidopsis thaliana* L. at 3 nM (Smith et al., 1994). Takeda et al. (1994) found that okadaic acid and MC-LR, inhibitors of protein phosphatases type 1 and 2A block the sugar-inducible gene expression in petioles of sweet potato *Ipomoea batatas*. Similarly, Siegl et al. (1990) reported that in vivo these toxins prevented the light-induced activation of sucrose-phosphate synthase (SPS) that is generally activated by dephosphorylating by protein phosphatase 2A, and decreased sucrose biosynthesis and CO₂ fixation in spinach leaves. Yin et al. (2005) reported that MC-LR at 5 mg L⁻¹ is able to cause oxidative damage resulting in lipid peroxidation and decrease of glutathione GSH content and increases of superoxide dismutase (SOD) and catalase (CAT) activities on *A. thaliana* cells. Later, Stüven and Pflugmacher (2007) provide further evidence that cyanobacterial toxins as well as cyanobacterial crude extract containing MC-LR induce oxidative stress response in *Lepidium sativum* seedlings, manifested by lipid peroxidation, elevation of alpha- and beta-tocopherol concentrations and elevated activities of antioxidative enzymes like the glutathione peroxidase, glutathione S-transferase and glutathione reductase. El Khalloufi et al. (2012) showed that 30 d exposure of *Lycopersicon esculentum* to a cyanobacterial crude extract containing 2.22–22.24 µg MCs mL⁻¹ caused enhancement on peroxidase activity and phenolic content indicated that the extract caused an oxidative stress. The exposure of rice plants (*Oriza sativa*) to toxic *M. aeruginosa* cyanobacterial extracts containing 50 µg MC-LR L⁻¹ resulted in a significant increase in the GST activity in leaves of this plant (Prieto et al., 2011). Therefore, by acting as protein phosphatase inhibitors and inducers of ROS production, MCs could be involved in several physiological and molecular processes in higher terrestrial plants.

Cytotoxic alkaloids

Cylindrospermopsin, a protein synthesis inhibitory cyanobacterial cytotoxin also led to a clear growth inhibition and anatomy modification through the alteration of microtubules organization of the common reed *P. australis* at concentrations 0.5–40 µg mL⁻¹ (Beyer et al., 2009). Previous study demonstrated that CYN inhibited the growth of *Sinapsis alba* mustard seedlings at 18.2 µg mL⁻¹ (Vasas et al., 2002). Short term exposure of rice plants (*Oriza s.*) to toxic *A. ovalisporum* cyanobacterial extracts containing 0.13 µg CYN L⁻¹ can lead to an increase of oxidative stress (increase in GST and GPx activities). Moreover, longer exposure periods can lead to tissue necrosis

(loss of tissue fresh weight) concomitant with the oxidative stress. In addition, the plant exposure to a mixture of *A. ovalisporum* and *M. aeruginosa* cell extracts containing $0.13 \mu\text{g CYN L}^{-1}$ and $50 \mu\text{g MC-LR L}^{-1}$, respectively, resulted in a significant increase in the GST and GPx activities, suggesting a synergistic effect of both extracts (Prieto et al., 2011).

Bioaccumulation of cyanotoxins in vegetable foods and consequences on animals and human health

In aquatic ecosystems, several studies have been reported the bioaccumulation of cyanotoxins in common aquatic vertebrates and invertebrates, including zooplankton, mollusks and crustaceans, and fish, which pose a potential risk to both animal and human health if such aquatic animals are consumed (Ibelings and Chorus, 2007; Ettoumi et al., 2011). However, their ability to enter the food chain via agricultural crops has not been thoroughly investigated to date. Questions, therefore, arise about the health significance of spray irrigation of crops with water from sources containing cyanobacterial blooms and toxins. Nevertheless, several studies have been shown the accumulation potential of cyanotoxins in aquatic vegetable organisms, suggesting that terrestrial plants, including food crop plants, can also take up these toxins. Mitrovic et al. (2005) reported that the filamentous alga *C. fracta* accumulates MC-LR at a rate of $8 \text{ ng g}^{-1} \text{ d}^{-1}$. In addition, few amounts of MCs were detected in *C. vulgaris* and *Scenedesmus quadricauda* cells only during the first 3 d of exposure, but not during the remaining period of the experiment, suggesting a possible biotransformation of MCs in these algae (Mohamed, 2008). The emergent reed plant *P. australis* showed an apparent distribution of MC-LR in the different parts of the plant, after exposure to this toxin at $0.5 \mu\text{g L}^{-1}$ with highest uptake was detected in the stem and then the rhizome (Pflugmacher et al., 2001). In addition, *Lemna minor* has also been shown to accumulate MC-LR up to a concentration of $0.2887 \pm 0.009 \text{ ng mg}^{-1}$ wet wt plant material, after 5 d of exposure to this toxin at $20 \mu\text{g L}^{-1}$ with an accumulation rate equivalent to $58 \text{ ng g}^{-1} \text{ d}^{-1}$ (Mitrovic et al., 2005). However, Saqrane et al. (2007) reported that *L. gibba* could take up and biotransform microcystins. The chronic exposure of plant led to dose-dependent MCs accumulation which reached $2.24 \mu\text{g g}^{-1}$ dry weight after being exposed to $0.3 \mu\text{g mL}^{-1}$ of MCs (Saqrane et al., 2007). Recently, it has been shown that collected water chestnut (*Trapa natans*) from Lake Tai accumulated MCs at highest level up to $7.02 \text{ ng g}^{-1} \text{ dw}$ (Xiao et al., 2009).

Terrestrial plants could be exposed to cyanobacterial toxins via the use of eutrophic water that may contain cyanobacterial blooms and toxins from irrigation and, therefore, they can take up

cyanotoxins. Peuthert et al. (2007) have been reported that MC-LR could be absorbed by roots and be translocated from roots to shoots in seedlings of eleven agricultural plants. A second study by Crush et al. (2008) that used different species too, revealed a high level of MCs accumulation in lettuce (*L. sativa*) exceeding the tolerable daily intake of $0.04 \mu \text{ kg}^{-1}$ of body weight d^{-1} recommended by the World Health Organization (Sivonen and Jones, 1999). However, the most of these studies have been performed in hydroponic conditions where the roots have been in direct contact with the toxin solutions and can, therefore, overestimate the bioaccumulation rate. In our knowledge the only study reported in the literature that was realized in soil showed that MC concentrations in roots did not exceed the tolerably limit, however, the concentration of MCs in aerial parts of the plant are not determined (Järvenpää et al., 2007). Both the roots and shoots of rice were reported to accumulate MC-LR in a laboratorial study (Chen et al., 2004). In addition, a recent study by Chen et al. (2012) reported for the first time the accumulation of MC-LR in rice grains harvested from Lake Taihu in China. However, the concentration of MC-LR detected in rice grains was very low and thus may not pose a threat to human health currently. In addition to the possibility of internal accumulation of MCs, irrigation may lead to accumulation of toxins on the external surfaces of edible plant materials when the contaminated water dries on the plant surface between irrigation periods or when the water becomes trapped in the centers of, for example, salad plants. In fact, Codd et al. (1999) have been reported that colonies and single cells of *M. aeruginosa* and microcystins were retained by salad lettuce after growth with spray irrigation water containing the microcystin-producing cyanobacteria. Recently, Kittler et al. (2012) reported that treatment of *Brassica oleracea* var. sabellica, *Brassica juncea*, and *S. alba* under varying experimental conditions showed significant CYN uptake, with CYN levels ranging from 10% to 21% in the leaves compared to the CYN concentration applied to the roots ($18\text{--}35 \mu \text{g L}^{-1}$). These results suggest that crop plants irrigated with CYN-containing water may represent a significant source of this toxin within the food chain. However, further research is needed into the uptake and fate of microcystins and other cyanobacterial toxins by food plants and the persistence of these toxins in the edible plant materials.

Conclusion and future directions

This review has established that cyanobacterial cells and toxins can be associated with crop plants after spray irrigation with water containing these agents. Therefore, the use of water from sources containing cyanobacterial blooms and toxins for spray irrigation of crop plants may not only inhibit

growth of plants, but also can induce a food chain contamination with a considerable health risk and potential economic losses. Several studies have been shown that cyanotoxins could be absorbed by roots, transported to shoots, and then be translocated to grains and/or fruits. Nevertheless, the concentration of MC-LR detected, for example, in rice grains was very low and thus may not pose a threat to human health currently. Cyanotoxins could be partially metabolized during the long distance transportation from roots to grains or fruits, which may result in the lower level of cyanobacterial hepatotoxins type microcystins detected in rice grains. In addition, MCs could bind to serine/threonine phosphatases during transport and thus could also affect their accumulation in grains and fruits. Therefore, further investigations are needed into the uptake and fate of microcystins and other cyanobacterial toxins by food plants during the total period of vegetative and fruit development.

However, there are gaps remaining concerning information on the future of cyanotoxins in soil in terms of speciation, persistence, mode of degradation and impact on biological life in soils. The results of many existing tests and particularly laboratory studies on phytotoxicity of cyanotoxins are done in soil-free systems and using non realistic environmental concentration of toxins. Therefore, they are difficult to compare to field studies because both abiotic (e.g., soil conditions) as well as biotic (composition of the degrading biological community) factors can influence the outcome of such studies. In order to assess the relevance of phytotoxicity of cyanotoxins and their bioaccumulation in crop plants in the terrestrial environment, further research seems thus appropriate.

Acknowledgement

The research was supported with a Grant to S. Corbel from Région Ile-de-France, DIM-ASTREA program No ast110055.

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Table 1

Overview of some published cyanobacterial toxin concentrations from various countries. Concentrations are presented in $\mu\text{g g}^{-1}$ dry weight (DW) or else in $\mu\text{g L}^{-1}$ as indicated.

Country	Location	Use	Type	Concentrations ($\mu\text{g L}^{-1}$ or $\mu\text{g g}^{-1}$ dw)	Reference
Algeria	L. Oubeira	□	microcystin-LR	3–29,163	Nasri et al. (2008)
Argentina	R. San Roque	§, £	microcystin-LR	920	Conti et al. (2005)
Australia	–	–	microcystin-LR	48.6	Giannuzzi et al. (2011)
	R. of drinking water	§	saxitoxin	30	Orr et al. (2004)
	L. Julius	§, £, □	cylindrospermospin	2×10^6	Saker and Griffiths (2001)
	R. Carria	§, £, □	cylindrospermospin	18.9	McGregor and Ibbro (2000)
	L. Cobaki Village	§, £, □	cylindrospermospin	101.4	Everson et al. (2011)
	Narrung Chammel	§	nodularin-R	1.6	Heresztyn and Nicholson (1997)
	L. Coolmunda	£	microcystin-LR	12	Stewart et al. (2006)
	L. Wivenhoe	£	cylindrospermospin	1–2	
Brazil	L. Bolonha	§	microcystin	1.25	Vieira et al. (2005)
	Sao Paulo	§, £	microcystin	0.5–100	Nobre (1997)
	Parana	§	microcystin	0.2–6.6	Hirooka et al. (1999)
	D. Itaipu	£	microcystin	6.4–10	
	R. Tapacura	§	saxitoxin	52	Molica et al. (2005)
	R. Armando Ribeiro Goncalves	§	microcystin-LR saxitoxin	8.8 3.14	Costa et al., 2006
China	R. Haimen	§	microcystin-LR	1,556	Ueno et al. (1996)
	L. Taihu	§, £, □	microcystin-LR	34.2	Liu et al. (2011)
Denmark	L. Knud so	–	homoanatoxin-a	2,300*	Henriksen (1996)
	–	–	homoanatoxin-a	800*	
	–	–	homoanatoxin-a	60*	
	L. Ravn so	–	homoanatoxin-a	2,300*	
	L. Salten Langso	–	homoanatoxin-a	20*	
	L. Agerso	–	saxitoxin	37*	Kaas and Henriksen (2000)
	L. Bastrup so	–	saxitoxin	6.4*	
Finland	L. Hvidesø	–	saxitoxin	85.1–182.5**	
	L. Visoi ggaard so	–	saxitoxin	224.1*	
	Prästräsket	§, £, □	microcystin-LR	42	Spoof et al. (2003)
France	Södra Slemmen	§, £, □	nodularin-R	0.2	
	Högskär	§, £, □	nodularin-R	0.5	
Germany	La loue	£	anatoxin-a	8,000*	Gugger et al. (2005)
	L. Champs-sur-marne	£	saxitoxin	4.8–6.7	Iedreux et al. (2010)
Greece	20 water bodies	£	anatoxin-a	0.39–6.7	Bumke-Vogt et al. (1999)
	55 water bodies	–	microcystin-LR	10	Fastner et al. (1999)
	–	£	microcystin-LR	36	Ueno et al. (1996)
	Berlin water bodies	£	microcystin-LR	0.14–119	Fromme et al. (2000)
Ireland	33 water bodies	–	microcystin-LR	50–1,600*	Cook et al. (2004)
	L. Caragh	§, £	anatoxin-a	112–444	James et al. (1997)
Italy	–	§	homoanatoxin-a	1.4	Furey et al. (2003b)
	L. Lough Sillan	§	homoanatoxin-a	24	Furey et al. (2003a,b)
	R. Innincarra	§	homoanatoxin-a	34	Furey et al. (2003b)
	L. Lough Key	§	homoanatoxin-a	12	
	L. Corbally	§, £	anatoxin-a	60–100*	James et al. (1997)
	R. Monteleone	§	microcystin	226	Messineo et al. (2009)
	L. Albano	£	cylindrospermospin	126	
	L. Inbanuma	§, £	microcystin	52	Ueno et al. (1996)
Japan	L. Suwa	–	microcystin	3.61	Park et al. (1998a,b)
	L. Baringo	□	anatoxin-a	0.05–0.21	Ballot et al., 2004
Kenya	L. Nakuru	–	anatoxin-a	5–223*	
	L. Baringo	□	anatoxin-a	0.05–0.21*	Ballot et al. (2003)
	L. Bogoria	–	anatoxin-a	10–18*	Krienitz et al. (2003)
	–	–	anatoxin-a	0.3–9*	Ballot et al. (2004)
	L. Simbi	–	microcystin-LR	19.7–39	Ballot et al. (2005)
	L. Sonachi	–	microcystin-LR	1.6–12*	
	L. Noriwashu	§, □	microcystin-LR	0.041	Krienitz et al. (2013)
Morocco	R. Lalla takerkoust	§, £, □	microcystin	73*	Oudra et al. (2001)
	–	□	microcystin	95.4	El Ghazali et al. (2011)
Netherlands	L. 't Joppe	§, £	microcystin-LR	2.5	Kardinaal et al. (2007)
	L. Volkerak	§, £	microcystin-LR	7	
	L. Kinselmeer	§, £	microcystin-LR	18	
New Zealand	L. Waitawa	–	microcystin-LR	28,000	Wood et al. (2006b)
	L. Horowhenua	–	microcystin-LR	16,291	
	L. Ngaroto	–	microcystin-LR	1,535	
	L. Taupo	–	microcystin-LR	708	
	Neuma Pond	–	microcystin-LR	22,58	
	L. Rotoiti	□	microcystin-LR	10–760	Wood et al. (2006a)
	L. Rotoehu	□	microcystin-LR	23	
Portugal	–	§, £	microcystin	13.7	Ueno et al. (1996)
Poland	R. Sulejow	§, £	microcystin-LR	1.17	Gagala et al. (2010)
	L. Brinskie	£	microcystin-LR	1.87	
S. Africa	D. Nhanganzwane	–	microcystin	23,718	Oberholster et al. (2009b)
	D. Makhohlolo	–	microcystin	0.317	
	L. Krugersdrift	–	microcystin	43.7	Oberholster et al. (2009a)