# The overlooked impact of rising glyphosate use on phosphorus loading in agricultural watersheds

**Marie-Pier Hébert[1,2]*, Vincent Fugère[1,2,3,4], and Andrew Gonzalez[1,3]**

**Glyphosate is the most extensively used pesticide worldwide. In addition to raising ecotoxicological concerns, the use of glyphosate adds phosphorus (P) to agricultural landscapes, influencing the accumulation and cycling of P in soil and nearby surface waters. Yet pesticides have been largely ignored when monitoring anthropogenic sources of P in agricultural watersheds. Estimating the supply of P derived from glyphosate use, both globally and in the US alone, we show that trends have markedly increased over the past two decades. Across the US, mean inputs of glyphosate-derived P increased from 1.6 kg P km$^{-2}$ in 1993 to 9.4 kg P km$^{-2}$ in 2014, with values frequently exceeding 20 kg P km$^{-2}$ in areas planted with glyphosate-resistant crops. Although still a minor source of P relative to fertilizers, P inputs from glyphosate use have now reached levels comparable to those from sources for which P regulations were initiated in the past. We thus argue for greater recognition of glyphosate's influence on P flow in watershed research and management.**

***Front Ecol Environ*** 2019; 17(1): 48–56, doi:10.1002/fee.1985

The industrialization of agriculture and the extensive use of agrochemicals have enhanced global food production, a transformation that has also fundamentally altered Earth-system processes and has led to substantial environmental degradation of soil and water quality (Vitousek *et al.* 1997; Cordell *et al.* 2009). Although fertilizer use dropped notably in many areas of North America in the 1980s and 1990s (eg eastern Canada, northeastern US; MacDonald and Bennett 2009; Hale *et al.* 2013), at the global scale pesticide use has risen dramatically (Gilbert 2013). Glyphosate – a phosphonic acid with the formula $C_3H_8NO_5P$, also known as *N*-(phosphonomethyl) glycine – is now the most widely used pesticide worldwide, ranking first in both sales and treated surface area (Nadin 2007; Environment Canada 2011). Commercialized under the label of Monsanto's Roundup in 1974, glyphosate went off-patent in the 1990s, prompting the development of generic herbicide formulations that are less costly and consequently more accessible, including in developing countries (EPA 2012; Health Canada 2016).

## In a nutshell:

- The herbicide glyphosate dominates the global pesticide market, with applied tonnage increasing steadily worldwide
- Glyphosate's capacity to degrade rapidly is often used to argue against potential toxicological effects, but the phosphorus (P) it contains persists in the environment and can influence soil P accumulation and losses to surrounding freshwater systems, which are prone to nutrient pollution
- Although pesticides are typically regarded as negligible sources of nutrients, we argue that the recent and rapid rise in glyphosate use has magnified its relative importance as a source of anthropogenic P, especially in areas of intensive corn, soybean, and cotton cultivation
- Glyphosate inputs should be considered in P assessments and sustainable management programs in agricultural watersheds; with glyphosate use increasing globally, it is imperative that we broaden the discourse of its environmental impacts

Approximately 8.6 billion kg of glyphosate have been applied globally since 1974 (Benbrook 2016); 1.6 billion kg of which has been applied in the US alone, with two-thirds of this amount having been applied in just the past 10 years. Glyphosate-based herbicides are predominantly used in association with genetically engineered resistant crops (GERCs), especially the genetically modified corn (*Zea mays*), soybean (*Glycine max*), and cotton (*Gossypium* spp) varieties that form the so-called "Roundup Ready" crops first introduced in 1996 (in the US; Figure 1; Duke and Powles 2008). The widespread adoption of GERCs (eg in the US, Brazil, and Argentina) has increased glyphosate application by ~15-fold, with GERC cultivation currently accounting for 56% of global glyphosate use (Givens *et al.* 2009; Benbrook 2016). The combination of the broad-spectrum nature of glyphosate, the introduction of GERCs, and increasingly intensive farming practices (eg no-till cropping, monocultures) has elevated the class of glyphosate-based herbicides to the degree that they have

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[1] *Department of Biology, McGill University, Montréal, Canada *(mphebert4@gmail.com);* [2] *Groupe de Recherche Interuniversitaire en Limnologie et en environnement aquatique (GRIL), Montréal, Canada;* [3] *Québec Center for Biodiversity Science (QCBS), Montréal, Canada;* [4] *current address: Département des sciences biologiques, Université du Québec à Montréal, Montréal, Canada*

become the most applied agricultural chemicals in human history (Gilbert 2013). However, scientists are only beginning to understand all of the potential impacts of pervasive glyphosate use, including potential declines in water quality and the emergence of resistant weed species with unanticipated environmental and economic repercussions (eg due to the renewed necessity of weeding by hand and/or compensatory increases in the use of other pesticides with greater toxicity; Duke and Powles 2008; Hicks *et al.* 2018).

Like fertilizers, pesticides applied to agricultural fields can be transported to surrounding aquatic ecosystems; indeed, glyphosate is increasingly observed in surface and ground waters around the world (Scribner *et al.* 2007; Environment Canada 2011; Aparicio *et al.* 2013). Despite its known presence in aquatic environments and the associated risk of toxicity for aquatic organisms (Annett *et al.* 2014), glyphosate is reported to degrade rapidly in water, with a half-life that ranges from 1 to 91 days (mean = 10 days), whereas its degradation in soils can take months, and even years in northern areas (Giesy *et al.* 2000; Duke *et al.* 2012; Helander *et al.* 2012). Glyphosate's rapid degradation relative to that of most other herbicides has been a primary argument against rising concerns about its potential toxicological effects (Cerdeira and Duke 2006).

Although the compounds released by glyphosate degradation are claimed to be "naturally occurring substances" (Monsanto 2014), these include phosphorus (P)-containing molecules. Glyphosate acid by mass contains 18.3% P, implying that its application represents an input of anthropogenic P to agroecosystems. In addition to contributing to P load, glyphosate can disrupt watershed P cycling due to its chemical similarities with phosphate ions (the primary bioavailable form of P) by binding to the same soil sites, potentially influencing soil P saturation and losses (de Jonge *et al.* 2001; Gimsing *et al.* 2004; Vereecken 2005). As a result, regardless of the ecotoxicological concerns over glyphosate itself and of the rate at which glyphosate degrades, its application inevitably leads to greater anthropogenic P inputs in the agricultural landscapes where it is used, and potentially to greater P export from soils to water bodies via diverse pathways.

Anthropogenic P inputs are typically regulated in agricultural areas because the P storage capacity of soils is frequently close to saturation due to the legacies of past practices (Sharpley *et al.* 2013). This saturation implies that P derived from fertilizer (hereafter "P-fertilizer") is more likely to flow into lakes, streams, wetlands, and coastal areas, and is therefore one of the primary causes of eutrophication (Smith *et al.* 2006). The use of fertilizers in agriculture has been reduced to mitigate freshwater eutrophication and its often severe ecological consequences (Conley *et al.* 2009), such as harmful algal blooms or the proliferation of macrophyte beds, biodiversity loss, depletion of dissolved oxygen leading to fish kills, and closure of aquatic recreational sites (Smith *et al.* 2006; Carpenter *et al.* 2011). Eutrophication is also expensive to manage and in some cases may even be irreversible (Schindler 2006). In contrast to fertilizers, pesticides have been largely ignored in P regulation policy making, despite the growing use of glyphosate around the world and the concomitant increase of glyphosate contribution to anthropogenic P inputs – an issue that has thus far received little to no attention in watershed management and P assessments.

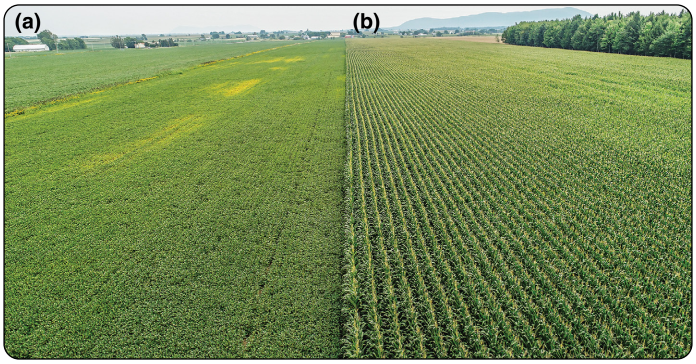

**Figure 1.** Monocultures of genetically engineered resistant (a) soybeans (*Glycine max*) and (b) corn (*Zea mays*), illustrating a common form of intensive land use.

Pollution from synthetic chemicals is a poorly studied dimension of global change (Bernhardt *et al.* 2017). In addition to investigating the toxicological effects of widespread glyphosate use, we believe that assessing its influence on P flow is also an important and timely issue. We illustrate the current state of this problem by quantifying P inputs derived from glyphosate use over time and at multiple spatial scales, and by comparing these values with other anthropogenic P inputs. We then describe the various pathways through which glyphosate can influence soil P accumulation and losses to freshwater systems, and conclude by briefly discussing several mitigation options.

## Trends in anthropogenic P inputs derived from glyphosate use

Although glyphosate use is increasing worldwide, how trends vary across regions or scales and how this P input compares to other anthropogenic P sources remain unclear. Using various open access datasets (eg those from the US Geological Survey, as well as the Food and Agriculture Organization of the UN; WebTable 1), we collated statistics of glyphosate and P-fertilizer usage for comparative purposes. We compiled these data at global and US national scales, as well as for 3055 counties of the conterminous US, and converted these numbers into their corresponding P contents (WebTable 1). For US county-level estimates, we scaled P inputs to agricultural land cover within each county in order to quantify usage per unit area independent of potential agricultural expansion. Synthesizing 91,890 glyphosate and P-fertilizer estimates (WebTable 1), we quantified P inputs derived from glyphosate use over time, compared these trends across scales, and contrasted the importance of glyphosate- to fertilizer-derived P.

From 1994 to 2014, global inputs of P derived from glyphosate (hereafter "P-glyphosate") increased from 10.3 Gg (gigagrams; thousand metric tons) to 151.3 Gg per year (Figure 2a). During

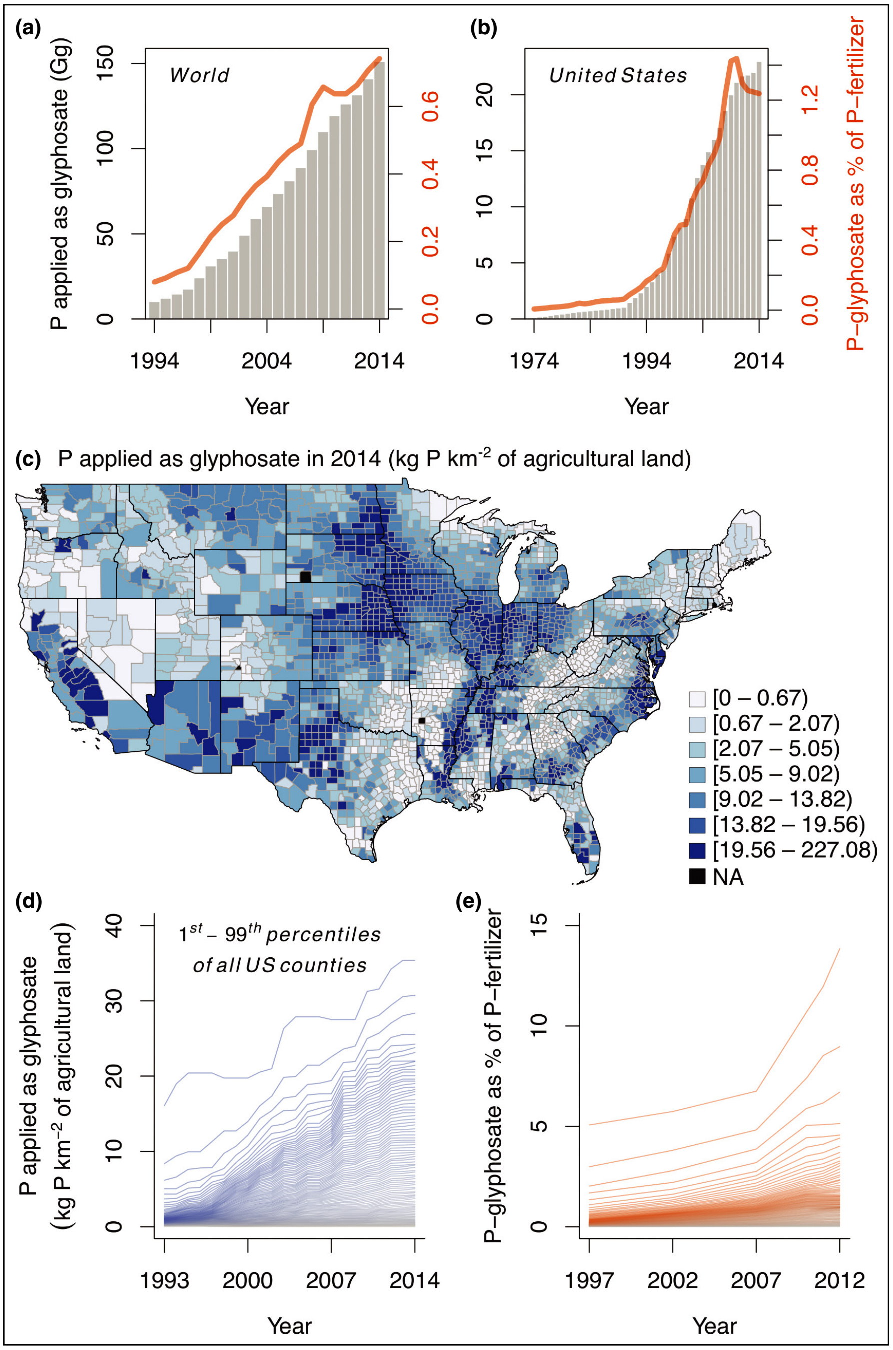

**Figure 2.** Spatiotemporal variation in glyphosate-derived phosphorus (P) inputs. (a and b) Trends in P applied as glyphosate (gray bars) and in the relative proportion of P applied as glyphosate versus P applied as mineral fertilizer (orange lines) at (a) global and (b) US national scales. (c) Mass of P applied as glyphosate per unit area of agricultural land in all counties of the conterminous US in 2014. (d) Trends in glyphosate-derived P inputs and (e) proportions relative to fertilizer for all US counties. Lines correspond to percentiles (ie limits of 100 groups/quantiles that each include >30 counties). The top line is the 99th percentile, below which lie 99% of counties in any given year. The narrower time frame of (e) relative to (d) represents the period over which glyphosate and fertilizer usage statistics overlapped at the county level.

that time, a total of 1484 Gg of P was applied in the form of glyphosate, with the annual tonnage of glyphosate applied increasing at an average rate of 7 Gg per year. A similarly steeply rising trend was observed at the national level in the US, with annual P-glyphosate inputs increasing from 2.9 Gg in 1994 to 23 Gg in 2014 (mean rate of increase = 1 Gg per year, over 20 years), yielding a total application of 278 Gg of P-glyphosate (Figure 2b). The relative proportion of P-glyphosate versus P-fertilizer also increased both globally and nationally within the US over that period (Figure 2, a and b), despite contrasting trends in fertilizer application at these two scales (ie increasing worldwide, but decreasing or stable in the US; WebFigure 1, a and b). In 2014, these relative proportions were 0.74% and 1.24% across the world and the US, respectively (Figure 2, a and b); although modest, these proportions have been and are still increasing. The only observed decrease in relative proportions was in the US for a short period of time (a reduction of ~0.25% between 2010 and 2014), which was due to a rapid rise in the use of P-containing fertilizers (WebFigure 1b); indeed, inputs of P-glyphosate increased by 1.87 Gg (or 8.85%) during that same period (2010–2014; Figure 2b).

These estimates also indicate that the US alone currently accounts for roughly one-sixth of global P-glyphosate inputs, confirming that it ranks among the top glyphosate users worldwide (Benbrook 2016). However, whereas glyphosate use in the US has increased nearly eightfold over the past two decades, global use has increased approximately 15-fold over the same time period (1994–2014; Figure 2, a and b), suggesting that glyphosate use has expanded more rapidly in other countries than in the US. Although we could not find reliable time-series of glyphosate use for other major consumers of this pesticide, point estimates from recent years suggest that P-glyphosate inputs may currently be higher in some developing countries than in the US. For instance, in Brazil, nationwide sales of glyphosate totaled 186.5 Gg (= 34.2 Gg P) in 2012, whereas sales of P-fertilizers totaled 4325 Gg (= 1888 Gg P), resulting in a P-glyphosate-to-P-fertilizer ratio of 1.8% (IBGE 2015). Thus, in 2012, both the absolute amount and the relative contribution of P-glyphosate were higher in Brazil than in the US. Brazil has also been a major producer of GERCs since 2000, along with China and Argentina; for example, 93% and 100% of the Brazilian and Argentinian soybean production, respectively, was glyphosate-resistant in 2014 (CCM 2013; Benbrook 2016). The decreasing costs of generic glyphosate-based herbicides and the recent patent expiration for some GERCs may lead to intensive usage in many developing countries with limited resources to undertake P mitigation and management programs.

The use of glyphosate and fertilizers is highly variable across US counties, reflecting spatial heterogeneity in land use (Figure 2c; WebFigure 1d). In 2014, hotspots of P-glyphosate inputs were key agricultural areas such as the Corn Belt, the Central Valley of California, the Texas Plains, and the Mississippi Alluvial Plain (Figure 2c); all of these regions are characterized by intensive cultivation of GERCs (corn, soybeans, or cotton; USDA NASS 2017), which likely accounts for their extensive glyphosate use. These areas also have high P-glyphosate-to-P-fertilizer ratios (with the exception of the Corn Belt, where fertilizer usage is markedly higher, thereby lowering the ratio; WebFigure 1, d and e). Across the 3055 US counties for which we were able to obtain data, the mean P-glyphosate input on agricultural land increased from 1.6 kg P km$^{-2}$ in 1993 to 9.4 kg P km$^{-2}$ in 2014. Because inputs were scaled to agricultural land area, this increase was most likely due to greater glyphosate use and not agricultural expansion within counties. Maximum P-glyphosate input in 2014 was 227 kg P km$^{-2}$ (Figure 2c); however, there could be considerable uncertainty in estimates for individual years and counties. To generate a more representative estimate of the upper range of P-glyphosate inputs, and to track how this value changed over time, we calculated percentiles of P-glyphosate inputs for each year. Percentiles are limits of 100 groups/quantiles; in this case, each percentile included ~31 counties. We considered that the 99th percentile – the value below which lie 99% of counties in any given year – represented a conservative but robust estimate of maximum inputs, excluding the 30 counties with the highest values (some of which were likely to be outliers). Most percentiles increased from 1993 to 2014 (Figure 2d), confirming the trend of the mean. The 99th percentile of all counties also increased markedly, from 15.3 kg P km$^{-2}$ in 1993 to 35.4 kg P km$^{-2}$ in 2014 (Figure 2d).

Because P-fertilizer inputs at the county scale were stable or decreased from 1997 to 2012 (ie the range for which we had overlapping P-glyphosate and P-fertilizer data at the county level; WebFigure 1c), the importance of P-glyphosate inputs relative to P-fertilizer inputs has increased over time (Figure 2e). Between 1997 and 2012, the median proportion increased from 0.1% to 1%, whereas the maximum proportion (99th percentile) increased from 5% to 14% (Figure 2e). These data confirm that glyphosate is becoming an increasingly important source of P, but that these inputs remain a much smaller source of anthropogenic P as compared to P-fertilizers (as quantified here and in other large-scale assessments; MacDonald *et al*. 2011). However, even in counties with relatively low P-glyphosate-to-P-fertilizer ratios, P-glyphosate inputs could still impact P losses from soils if those soils are already saturated (eg due to past P legacies). This could apply to many counties in the Corn Belt, in which ratios were low but absolute inputs of P-glyphosate were among the highest (Figure 2c; WebFigure 1e). Moreover, and perhaps more importantly than the comparison with P-fertilizers, the magnitude of P-glyphosate inputs is now similar to that of previous issues of concern that have been addressed in watershed management, such as P derived from detergents. For instance, prior to legislation banning P in detergents (eg in Canada and the US), the supply of P via the use of detergents was 8.54 kg P km$^{-2}$ (in 1971) throughout the sub-basin of the St Lawrence River (76 watersheds), whereas in 2011, the estimated use had dropped to 1.08 kg P km$^{-2}$ (Goyette *et al.* 2016). In comparison, nearly one-half of the US counties reported P-glyphosate inputs exceeding 8.5 kg P km$^{-2}$ in 2014. In more urbanized watersheds, however, reported P inputs derived from detergents were often higher (eg an average of 31 kg P km$^{-2}$ for 18 watersheds around Lake Michigan between 1974 and 1992; Han *et al.* 2011). Agricultural areas associated with intensive farming practices currently have similar values (99th percentile of all counties in 2014 = 35.4 kg P km$^{-2}$; Figure 2d). Thus, the range of P-glyphosate inputs now overlaps with that of other P sources, such as detergents, that prompted regulation.

## Glyphosate and phosphorus: accumulation, transport, and bioavailability to organisms

The influence that glyphosate may exert on the export of P from soils to waterways depends on many factors. Some laboratory and field trials have tested the environmental behavior, mobility, and fate of glyphosate, for instance, to document how it interacts with soil particles, how likely it is to be degraded or assimilated by organisms, and what drives accumulation in soil versus runoff (ie surface and sub-surface water flow) and leaching (ie flow into and out of the ground) to freshwater systems (Vereecken 2005; Borggaard and Gimsing 2008). However, the results of much of this research have been published in soil science and environmental chemistry journals, such that watershed scientists may not be familiar with glyphosate's influence on P flow. Accordingly, the potential effects of glyphosate on P accumulation in soil and losses to water bodies have not been effectively integrated into management guidelines for agricultural regions. In the following sections, we briefly review the primary mechanisms through which glyphosate can influence the cycling and fate of P in agricultural landscapes, and the implications this may have for freshwater organisms and ecosystems.

### Fate after application

Glyphosate is most often directly sprayed onto fields using industrial machinery, leading to a small fraction being lost to the atmosphere and transported elsewhere. The remainder is taken up by plants or penetrates into the soil, the former releasing some of it into the soil via root transport (Figure 3; Helander *et al.* 2012). Once in the soil, glyphosate can then (1) be carried away via surface water runoff; (2) be assimilated by the roots of non-target plants; (3) temporarily adsorb to soil particles until detachment, to then reach ground or surface waters; and (4) adsorb to soil particles and progressively degrade in the soil after longer-term storage (Figure 3; Borggaard 2011;

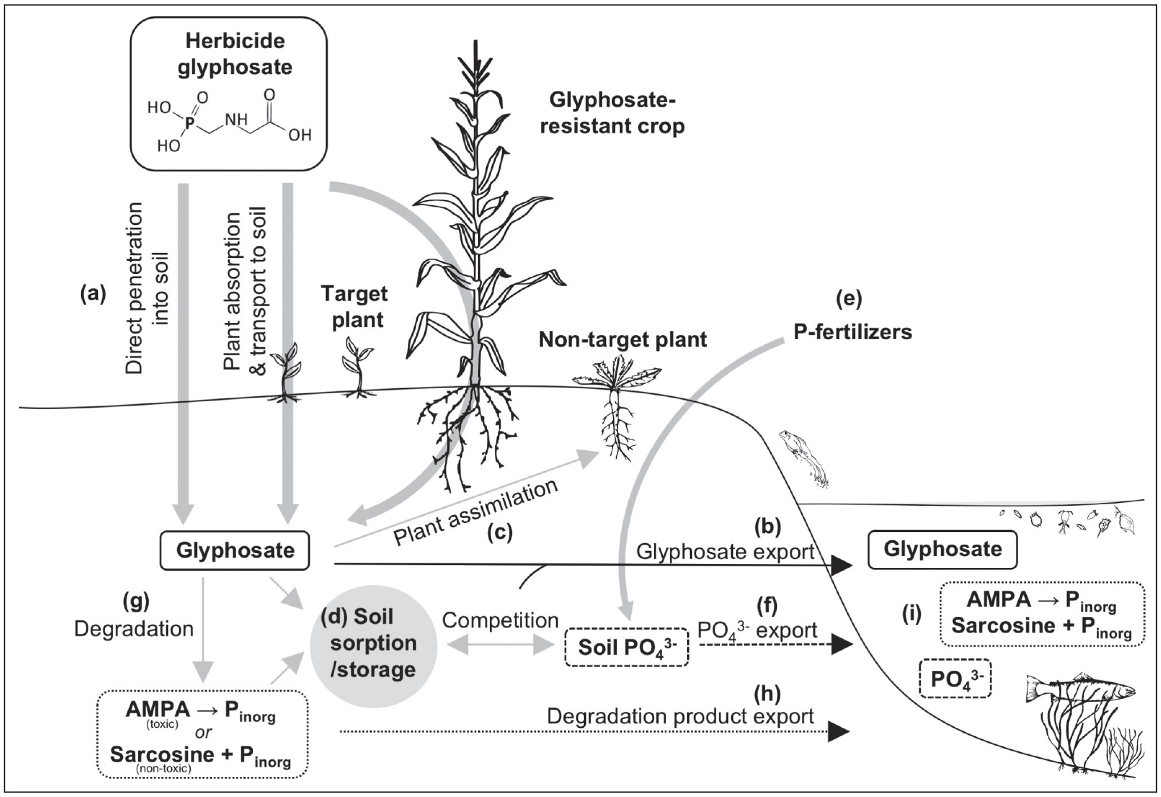

**Figure 3.** Transport and fate of glyphosate in agricultural landscapes. (a) Once sprayed (and ignoring atmospheric loss), glyphosate can either penetrate the soil surface directly or be absorbed by plants via their foliage and translocated via phloem down to the roots, where it is exuded into the soil. (b) In the soil, a fraction of glyphosate can be transported by runoff or can leach into surface waters, either directly following application or after a period of soil storage; (c) another fraction can also be assimilated by nearby non-target plant roots. (d) Most glyphosate, however, will adsorb to soil particles, with soil retention capacity depending on soil mineral content, pH, and phosphate ($PO_4^{3-}$) content, (e) either natural or originating from other anthropogenic inputs, such as fertilizers. (b and f) By adsorbing to soil, glyphosate may compete with $PO_4^{3-}$ for sorption sites, potentially influencing the mobility of glyphosate and/or $PO_4^{3-}$; the extent of these processes may vary with soil composition and structure. (g) Glyphosate is also subject to degradation in soil, with aminomethyl phosphonic acid (AMPA) and sarcosine being the main degradation intermediates, and with both degradation pathways releasing inorganic P; (h) these compounds can also enter into surface waters. (i) Finally, glyphosate degradation can also occur in water itself, releasing degradation products and inorganic P. (Schematic inspired by Helander *et al*. [2012]).

Coupe *et al.* 2011). Several environmental and climatic conditions, as well as the timing and frequency of pesticide and fertilizer applications, can influence the relative importance of each of these pathways and therefore determine the fate of glyphosate post-application.

## From soil to water

Regardless of whether glyphosate is transported immediately to waterways via runoff or is exported after a longer leaching process in the ground, the presence of glyphosate in surface waters is influenced by the timing, doses, and frequency of pesticide application, climate (eg rainfall intensity and frequency), hydrology (eg flow routes), and soil composition and structure (Kjaer *et al.* 2005; Candela *et al.* 2010). These variables influence the flow of glyphosate and the amount that reaches surface waters; reported concentrations in surface runoff can vary considerably, ranging from ~0.1 $\mu g\ L^{-1}$ to as high as 5200 $\mu g\ L^{-1}$ (Edwards *et al.* 1980; Coupe *et al.* 2011; Hénault-Ethier *et al.* 2017).

Soil properties such as mineral content, phosphate concentration, and pH can also influence glyphosate leachability (Figure 3; Borggaard and Gimsing 2008). These characteristics affect glyphosate sorption (binding) or desorption to soil particles and thus the capacity of a given soil to store glyphosate; greater sorption mitigates leaching or assimilation by non-target plants. Specifically, minerals such as iron, aluminum, and calcium provide sorption sites to glyphosate and favor soil accumulation. In contrast, phosphate-rich soils and fertilization events can reduce glyphosate sorption, as glyphosate and phosphate bind to similar sorption sites (Vereecken 2005). Soil pH can also modulate glyphosate sorption, with glyphosate mobility being greater when soil pH is higher (Gimsing *et al.* 2004). Importantly, even if glyphosate remains and degrades in the soil, glyphosate or its degradation products can accumulate and contribute to soil P saturation.

Climate is known to influence the flow of agrochemicals within watersheds. Under a changing climate, precipitation and the occurrence of storms or floods are becoming

### Panel 1. Intensifying agriculture: the case of Montérégie in southern Québec

Glyphosate is by far the most extensively used pesticide in the Montérégie region of Québec, Canada (as well as in the province overall), with total area sprayed having increased eightfold since the introduction of GERCs, reaching 2 million ha in 2014 (Giroux 2015). Monocultures of corn (*Zea mays*) and soybean (*Glycine max*) cover 34% of the area, with GERCs now accounting for 88% and 59% of their production, respectively (compared to only 32% and 17%, respectively, in 2003; ISQ 2017). Water bodies in the region provide important ecosystem services, such as drinking water and recreational opportunities, but they are often found near sites of GERC cultivation (Figure 4). In Canada, the concentration of glyphosate considered safe for aquatic life was revised upward, from 65 μg $L^{-1}$ in 1999 to 800 μg $L^{-1}$ in 2012 (CCME 2012). Traces of glyphosate have now been observed in nearly all water bodies in the Montérégie region monitored by the provincial government (Giroux 2015). Although maintenance of riparian buffer strips along waterways is a commonly employed preventive strategy in Québec (Figure 4), the results of a recent study indicate that the 3-m buffers required by legislation are ineffective at intercepting glyphosate (Hénault-Ethier *et al.* 2017).

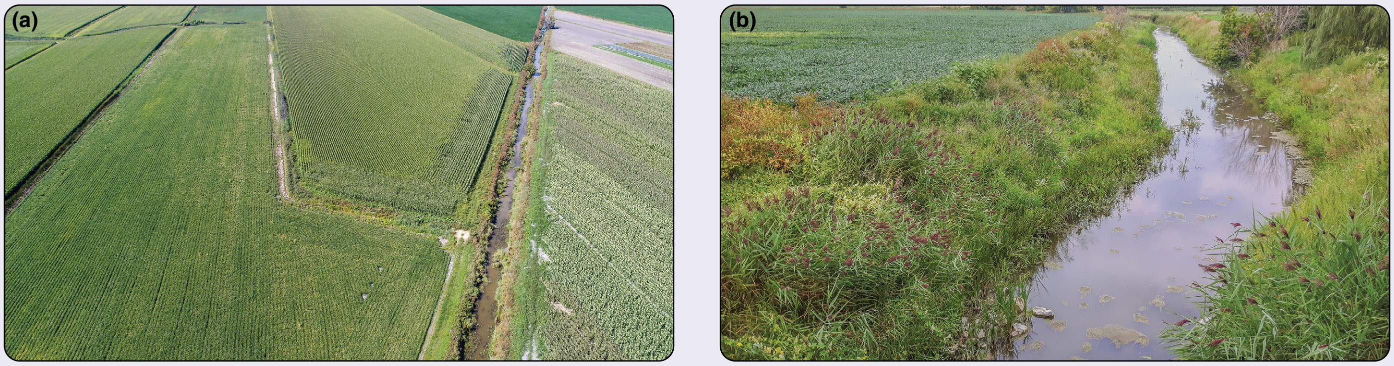

**Figure 4.** The Montérégie region of Québec, Canada, in which fields ([a] and [b]) of genetically engineered resistant crops (GERCs) are located near the Rivière des Hurons.

increasingly frequent in many regions, leading to greater P load in agricultural water bodies (Carpenter *et al.* 2018). For instance, Coupe *et al.* (2011) reported systematically higher glyphosate concentrations in surface waters following storm events (ranging from 34 to 430 μg $L^{-1}$), with the highest levels occurring when glyphosate application preceded a storm. Soil losses of agrochemicals like glyphosate could therefore potentially be exacerbated by climate change in certain areas.

## Glyphosate-derived P in freshwater systems

That glyphosate rapidly degrades in water is often used as an argument against potential health and environmental hazards associated with rising glyphosate concentrations in tested water bodies. However, rapid degradation in water implies two additional concerns: (1) an equally rapid release of degradation products, including P compounds, in aquatic ecosystems sensitive to excess P; and (2) to some extent, a possible underestimation of the actual amount of glyphosate entering freshwater systems. Glyphosate degradation can occur via two biological mechanisms involving either aminomethyl phosphonic acid (AMPA) or sarcosine as catabolic intermediates, with both pathways releasing inorganic P (as a byproduct of sarcosine, or as a metabolite when AMPA is in turn degraded; Figure 3; Hove-Jensen *et al.* 2014). Furthermore, rapid biodegradation in water is due in large part to the ability of some heterotrophic bacteria to use glyphosate as a resource for growth, which can enhance microbial proliferation (Kamat *et al.* 2011; Hove-Jensen *et al.* 2014). Many algal species can also use glyphosate as a source of P (Wang *et al.* 2016), with some taxa being naturally tolerant to the herbicide (eg several strains of cyanobacteria; Forlani *et al.* 2008; Harris and Smith 2016). Moreover, irrespective of whether a given species is capable of using glyphosate as a P source, bioavailable P compounds released via biodegradation can be assimilated by other organisms, stimulating their proliferation, as has been observed in cyanobacterial communities in Lake Erie (Saxton *et al.* 2011) and in experimental communities of periphyton and phytoplankton (Austin *et al.* 1991; Pérez *et al.* 2007). Glyphosate can therefore influence organismal growth and community composition in aquatic ecosystems both via its toxicological effects (favoring resistant species) and via P enrichment and bottom-up effects (Pérez *et al.* 2007; Saxton *et al.* 2011; Wang *et al.* 2016). However, such ecological impacts remain relatively poorly understood, and warrant further investigation.

## Glyphosate–phosphate competition: implications for P losses to freshwater

Another mechanism through which glyphosate could increase soil P losses to surface waters is through adsorption to the same sites as phosphate ions. Because of their chemical resemblance, glyphosate and phosphate can compete for the same (limited) sorption sites, with phosphate-rich soils and fertilization (ie increasing P saturation) potentially reducing glyphosate sorption and enhancing its mobility (Gimsing *et al.* 2004; Vereecken 2005; Figure 3). Moreover, intense glyphosate use could also increase the occupancy of sorption sites, potentially favoring phosphate mobility; that is, even if P-glyphosate itself never reaches a water body, glyphosate–phosphate competition for sorption sites could potentially cause greater phosphate losses from soils. Although more difficult to quantify (de Jonge *et al.* 2001; Candela *et al.* 2010), this mechanism could potentially represent one of the most important pathways through which glyphosate conveys P to aquatic systems. In addition, regardless of the pathway and whether P compounds reaching surface waters are in the form of glyphosate, its degradation products, or phosphate, glyphosate is likely increasing soil P accumulation and/or losses. Glyphosate use can therefore alter water quality via its potential effects on both toxicity and P load, an issue that adds to the large existing environmental footprint of agriculture on soil P saturation and eutrophication. Soils that have long been fertilized with P in excess of what was assimilated and removed by crops are much more prone to leach recently applied fertilizers (del Campillo *et al.* 1999), which could also influence the export of glyphosate. Therefore, P legacies due to past practices should be considered not only for the regulation of fertilizer usage but also for P-containing pesticides, especially glyphosate.

## Watershed management

Increasing P loading to freshwater systems is a major concern for watershed management. For example, despite decades of management efforts, P levels have been rising across the US over the past decade (Stoddard *et al.* 2016). We have shown here that P inputs derived from glyphosate application have been increasing steadily, and discussed how this could influence P stocks and flow within watersheds. But what solutions or mitigation strategies could be employed to address this emerging problem? An obvious recommendation would be to include P-glyphosate in monitoring and/or nutrient management programs in areas where glyphosate is used extensively (eg when measuring P levels and P-sorption sites in soils to inform farming practices). Such local characteristics could also help regulate glyphosate use, as they also influence soil accumulation and losses of glyphosate. Rather than having a single recommended dose and application frequency, maximum glyphosate application could be adjusted based on the local soil conditions of any given farm. However, regulating glyphosate application as a function of soil P content may imply further reducing the use of P-fertilizers (ie crediting glyphosate-related P in nutrient management). In addition, due to their additive effects on P inputs in agricultural landscapes, ensuring that glyphosate and P-fertilizers are applied at different times could also reduce P losses to surface waters. Another alternative would involve better regulation of conservation tillage practices; although reducing tillage aims to reduce total P losses by mitigating erosion, it would also lead to unintended increases in dissolved P loss and increased usage of pesticides (Sims and Vadas 2005; Cerdeira and Duke 2006). Given the novelty of this issue, mitigation strategies should be developed, with input from farmers and agricultural extension agents, in order to assess their potential economic and environmental repercussions. For instance, regulating glyphosate use could influence crop yields and/or lead to compensatory increases in the use of other pesticides with lower P content but greater toxicity. Such trade-offs must be evaluated carefully from the perspectives of diverse stakeholders.

## Conclusions

Glyphosate has become the most widely used agricultural chemical in history due to its ability to enhance food production and a relative lack of concern about its persistence and toxicity as compared to other types of herbicides. The prevalent view that glyphosate has a comparatively low risk of adverse health or toxicological impacts has led to permissive regulation of its use, and in some cases maximum exposure thresholds tolerated in aquatic ecosystems have actually been revised upward (eg in Canada; Panel 1). The increasing use of glyphosate has resulted in unforeseen issues, such as the emergence of glyphosate-resistant weeds and, as discussed above, an increasing amount of anthropogenic P in agricultural landscapes, especially where GERCs are grown extensively. Human influence on P flow is central to the problem of environmental degradation and freshwater eutrophication, and efficient exploitation of this limited resource is imperative (Elser and Bennett 2011). With P-glyphosate inputs reaching levels comparable to those of previous sources for which P inputs were eventually regulated by legislation (eg detergents), glyphosate use should no longer be disregarded in P management, and is an issue that must be brought to the attention of policy makers, agricultural extension agents, and conservation practitioners. Collecting data on P-glyphosate inputs at finer spatial scales while considering local soil and/or hydrological characteristics would also help to identify watersheds of greatest concern and better quantify the role of glyphosate in exporting P from land to water across agroecosystems. Given the rise in global glyphosate use and the large-scale adoption of GERCs in multiple countries, we believe that it is now critical to broaden the discourse on the environmental impacts of glyphosate.

## Acknowledgements

We thank Y Prairie, E Bennett, and H Swanson for insightful discussions and comments on earlier versions of the manuscript.

M-PH is supported by a Natural Sciences and Engineering Research Council of Canada (NSERC) scholarship, an NSERC-CREATE Écolac fellowship in affiliation with GRIL, and McGill doctoral awards; VF is supported by an NSERC postdoctoral fellowship; and AG is supported by the Canada Research Chair program, an NSERC discovery grant, a Liber Ero Chair in Conservation Biology, and a Killam fellowship.

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