UNIVERSITÉ DU QUÉBEC À MONTRÉAL

USAGE DE BANDES RIVERAINES COMPOSÉES D'HERBACÉES OU DE SAULES ARBUSTIFS POUR LIMITER LES FLUX AGRO-CHIMIQUES DES GRANDES CULTURES VERS LES COURS D'EAU ET PRODUIRE DE LA BIOMASSE DANS LA PLAINE AGRICOLE DU SAINT-LAURENT

THÈSE

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AVANT-PROPOS

Contribution de chaque co-auteur

Cette thèse a été réalisée dans le cadre du projet CRSNG-Stratégique SABRE, Salix en agriculture pour des bandes riveraines énergétiques, dirigé par Marc Lucotte, et mené en partenariat avec Michel Labrecque, Philippe Juneau et Laurent Lepage. La thèse est rédigée sous forme de trois articles scientifiques en anglais (chapitres 1, 3 et 4), pour fin de publication dans les revues Biomass & Bioenergy, Science of the Total Environmentet Journal of Environmental Quality suite au dépôt final. Pour que les publications correspondent bien à la présente thèse, les articles ont été conservés dans leur intégralité, ce qui occasionne quelques répétitions dans la description des sites d'études et de la méthodologie. Le chapitre descriptif 2 vient en appui technique à la thèse et est ici formaté spécifiquement dans cet objectif. La majeure partie de la planification, de la direction réalisation des travaux de terrain ou de laboratoire ainsi que la rédaction a été réalisée par moi-même, l'auteure principale, et la contribution scientifique de chacun des co-auteurs est détaillée ci-dessous.

Chapitre 1 :

Hénault-Ethier, Louise, Marcelo Pedrosa Gomes, Marc Lucotte, Élise Smedbol, Sophie Maccario, Laurent Lepage, Philippe Juneau and Michel Labrecque. High bioenergetic yields of riparian buffer strips planted with Salix miyabena SX64 along field crops in Québec, Canada. À soumettre dans la revue Biomass & Bioenergy. La planification de l'article et des échantillonnages sur le terrain a été réalisée conjointement avec Marcelo Pedrosa Gomes. Marcelo Pedrosa Gomes, Élise Smedbol et Sophie Maccario ont participé avec moi aux échantillonnages sur le terrain. J'ai effectué les traitements de données et préparé les figures, tableaux et rédigé le texte. Marcelo Pedrosa Gomes a contribué à la révision du texte. Les chercheurs du projet SABRE ont supervisé les travaux et révisé le texte.

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LISTE DES ABRÉVIATIONS, ACRONYMES ET UNITÉS DE MESURES

\$/t	dollars per tone
$\Delta h_{CF\text{-}CR}$	groundwater hydraulic gradient from edge-of-field to edge-of-stream
µm	micrometers
^{o}	degrees
^{0}C	degré Celsius
^{0}C•d	degree-days
1/D	Simpson reciprocal index
3D	three-dimensional
3X	low density Salix buffers treatment or 3 rows/3m width and 33 333 stems/ha (3X)
5X	high density Salix buffers treatment or 5 rows/3m width and 55 556 stems/ha (3X)
Al^{2+}	aluminum divalent cation
alt.	altitude
AMPA	aminomethyl phosphonic acid
aq	aqueous
BB	Boisbriand
BL	black histosol, strongly decomposed
BR	brown histosol, less decomposed
Ca^{2+}	calcium divalent cation
CC	close to the center of the buffer, côté au coeur de la bande
CEAEQ	Centre d'expertise en analyse environnementale du Québec
CEC	cationic electrical conductivity
CF	close to the edge-of-field, côté champ
cm	centimeter
CR	close to the edge-of-stream, côté ruisseau
CS	in the field
CS	clean sand lentils, stratigrahic descriptor
CX	Herbaceous vegetation buffers low density Salix buffers treatment or 3 rows/3m width and 33 3333 stems/ha (3X); hectare (ha)
DEM	digital elevation model
dGPS	differential global positioning system
DOC	dissolved organic carbon
EC ($\mu S \cdot cm^{-1}$)	Electrical conductivity in micro Siemens per centimeter
EPA	US Environmental Protection Agency
Fe^{2+}	Iron divalent cation
GC	Grey clay

GC-ECD	Gas chromatograph – electron capture detector
GPS	Global positioning system
GR	Genetically resistant
H’	Shannon diversity
ha	hectare
HCl	hydrochloric acid
HDPE	high density polyethylene
HNO_3	nitric acid
ID	identity
K^+	Potassium monovalent cation
$K_2S_2O_8$	Potassium persulfate
km	kilometers
kPa	kilopascal, measure of atmospheric pressure
K_{sat}	saturated hydraulic conductivity
L	liters
LDPE	low density polyethylene
M	maize or corn
m^2	square meter
MA	marl
MAPAQ	Ministère de l’Agriculture, des Pêcheries et de l’Alimentation [Ministry of Agriculture, Fisheries and Food]
max	maximum
MDDELCC	Ministère du Développement durable, de l’Environnement et de la Lutte aux changements climatiques [Ministry of Sustainable Development, Environment and Climate Change Mitigation]
Mg^{2+}	Magnesium divalent cation
min	minimum
min, max, average; ^0C	minimum, maximum or daily mean temperature
mm	milimeters
Mn^{2+}	Manganese divalent cation
n. sp.	number of species
Na^{2+}	Sodium divalent cation
NH_4^+	Ammonium ion
NO_2^--NO_3^-	nitrites and nitrates
ns	non-significant
N_{tot}	total dissolved nitrogen, NO_2^-+NO_3^-+NH_4^+
ø	diameter
$\theta_{\perp CF}$ and $\theta_{\perp CR}$	deviation from a perpendicular transect close at the edge-of-field or edge-of-river measured in degrees (^{0})
θ_{CF} and θ_{CR}	angle of incidence close at the edge-of-field or edge-of-river measured in degrees (^{0})

P	pesticide (only in Chapter 2 equations) elsewhere it stands for phosphorus
p	probability
PC1	first principal component
PE	peat lightly decomposed
pH	acidity
PO_4^{3-}	phosphates
PPRLPI	Politique de protection des rives, du littoral et des plaines inondables [Policy for protecting shores, coasts and flood plains]
P_{tot}	total phosphorus
PVC	polyvinyl chloride
QC	province of Québec
r	coefficient of correlation
RBS	riparian buffer strips
RC	reddish clay
RDA	redundancy analysis
S	soy
SD	standard deviation
sig	significant
SL	sandy loam
SR	Saint-Roch-de-l'Achigan
$t\ dw \cdot ha^{-1}$	tones dry weight per hectare
TI	till
TIN	triangulated irregular network
TSS	total suspended solids (particles)
UV	ultraviolet
$watts \cdot m^{-2}$	mean solar radiation measured in watts per square meters
$(\overline{X})$	mean
x, y, z	coordinates : longitude, latitude, altitude
z	altitude

RÉSUMÉ

Le Québec s'est doté d'une politique prônant la conservation de bandes riveraines étroites (≥ 3 m) en bordure de tous les cours d'eau agricoles pour mitiger la pollution diffuse liée aux nutriments et aux pesticides. Cette politique est un compromis entre efficacité de mitigation et impact économique des agriculteurs privés de culture en zone riveraine, ayant une largeur inférieure aux recommandations des scientifiques pour améliorer la qualité de l'eau.

La présente thèse teste donc l'efficacité des bandes riveraines en conformité avec cette politique, en milieu ouvert, sur trois années consécutives. Des bandes riveraines herbacées, typiques des friches le long des cours d'eau, sont comparésà à des plantations de*Salix miyabeana* SX64 Le saule arbustif ayant un potentiel potentiel de phytoremédiation reconnu et sa croissance rapide permet une production de biomasse pouvant pallier aux besoins énergétiques ou économiques des agriculteurs, ce design de bande riveraine constitue une innovation à évaluer. Pour déterminer si l'augmentation de la densité de plantation améliorerait l'efficacité de la bande riveraine étroite ou la productivité de biomasse, les plantations de saules ont été faites sur 3 rangs (33 333 tiges/ha) ou 5 rangs (55 556 tiges/ha). Pour maximiser la portée de nos conclusions, deux types d'environnements communs dans la plaine du Saint-Laurent ont été choisis. Les bandes riveraines de Boisbriand (BB) se situaient au creux de champs valonnés, dans une dépression où l'on retrouve souvent une terre organique avec une nappe phréatique peu profonde. Les bandes riveraines de Saint-Roch-de-l'Achigan (SR) sont dans un champ au relief plat avec une couche d'argile peu profonde. Les trois traitements de bandes riveraines avaient été implantés en triplicata de façon aléatoire en 2009. Les instruments d'échantillonnage de l'eau ont été implantés en 2011 afin de recueillir le ruissellement, l'eau interstitielle en zone non-saturée et phréatique avant et après la bande riveraine, jusqu'au printemps 2014.

La productivité des saules en bande riveraine était plus élevée que celle de plantations commerciales en plein champ (23-34 t bs/ha/an à SR sur un loam sableux compacté et 56-89 t bs/ha/an sur une riche terre organique à BB). Le potentiel de séquestration des nutriments était aussi intéressant : 116-118 Kg-N/ha/an, 23 kg-P/ha/yr et 62-63 Kg-K/ha à SR et 278-447 Kg-N/ha/an, 55-89 kg-P/ha/an et 148-239 Kg-K/ha à BB. Ces potentiels intéressants de production de biomasse et de séquestration de nutriments motivent donc le déploiement de ce type de bande riveraine multifonctionnelle.

Si le ruissellement de surface modélisé suit un parcours très hétérogène sur une échelle locale, reste qu'à l'échelle du champ, la moyenne des trajectoires traverse effectivement la bande riveraine de façon perpendiculaire. De plus, la nappe phréatique s'écoule généralement depuis les champs, vers les cours d'eau, mais ce trajet peut s'inverser en période sèche (lorsqu'il y a connectivité avec le ruisseau àBB). Les trajectoires horizontale et verticale de l'eau influencent l'efficacité perçue de la bande riveraine définie comme la différence de concentration en nutriments ou en pesticides avant ou après la bande riveraine (exprimée en pourcentage).

L'efficacité de la bande riveraine est fortement influencée en fonction d'épisodes saisonniers. Les nutriments sont plus concentrés juste après la fertilisation, et cette période coïncide avec une efficacité accrue dans l'enlèvement des nitrates (77-81% dans le ruissellement à BB, et 92-98 % à 35-70 cm de profondeur à SR) et une ponctuelle supériorité des saules par rapport à la bande enherbée. Le potassium, le phosphore total et l'azote ammoniacal n'étaient retenus que ponctuellement dans le temps et l'espace. Par contre, la rétention des phosphates était nulle tout au long de l'année (depuis la fonte nivale jusqu'après les épandages d'herbicides à base de glyphosate).

La bande riveraine s'est avérée inefficace pour retenir le ruissellement du glyphosate ou de l'AMPA, son sous-produit de dégradation et pourrait même ($p = 0.0513$ à SR) contribuer à l'infiltration du glyphosate vers le sous-sol, où il entrainerait une contamination de la nappe phréatique. La réduction des concentrations de glyphosate (27-54% à SR selon les traitements) dans le sol, suggère que seul le glyphosate adsorbé aux particules de sol est freiné par la bande riveraine.

En conclusion, la bande riveraine de 3 m préconisée au Québec ne suffit pas à atteindre les concentrations requises dans les critères de protection chronique pour la vie aquatique tant pour les nutriments que pour le glyphosate. Enfin, hormis le potentiel intéressant de production de biomasse, les saules n'étaient pas systématiquement plus efficaces que la friche pour mitiger la pollution diffuse.

MOTS CLÉS

Bandes riveraines végétalisées, Nutriments, Glyphosate, Hydrologie, *Salix miyabeana* SX64

INTRODUCTION

I.1 Mise en contexte

Dans plusieurs pays, le lessivage des nutriments, pesticides et particules de terre érodées provenant des exploitations agricoles intensives est considéré comme la plus importante source de pollution diffuse entraînant la dégradation de l'eau (Canada et Europe (Ongley 1997), États-Unis (EPA 2003), Chine (Ongley et al. 2010)). Les fertilisants agricoles seraient l'une des principales cause dans la détérioration de 48% des cours d'eau aux États-Unis (EPA 2003). L'enrichissement excessif en nutriments des cours d'eau représente un défi pour tous les paliers de gouvernements à travers le monde (King et al. 2015). En priorité mentionnons les conséquences néfastes sur la santé liées directement à la consommation d'eau enrichie en nitrates (methemoglobinémie) ainsi qu'aux efflorescences algales toxiques favorisées dans les eaux enrichies en nutriments (Pilotto et al. 1997; Van Dolah 2000; Matson et al. 1997; Townsend et al. 2003; EPA 2003). Les conséquences de l'eutrophisation incluent aussi l'hypoxie et l'anoxie qui s'étendent au-delà des eaux douces intérieures, jusqu'aux régions côtières et marines, menaçant le tourisme, les pêcheries et le fonctionnement des écosystèmes en Amérique du Nord — Baie de Chesapeake (Boesch et al. 2001), Grands Lacs (Rockwell et al. 2005; Hawley et al. 2006), Lac Winnipeg (Schindler et al. 2012) et Golfe du Mexique (Rabalais et al. 2001) — et à l'internationale — Mer Baltique (Conley et al. 2002), Mer de Chine (Chen et al. 2007), Mer Noire et ailleurs (Diaz 2001).

Les grandes cultures constituent une activité économique majeure au Québec, où 365 000 ha de maïs-grain et 318 000 ha de soya ont été ensemencés en 2015 (Institut de la statistique du Québec 2015b). Dans ces champs, la dominance des cultures génétiquement modifiées — 88% du maïs-grain et 59% du soya — continue à prendre de l'ampleur — augmentations de 14% et 7% respectivement depuis 2011, année qui marque le début du présent projet de recherche (Institut de la statistique du Québec 2015a). Ces tendances reflètent la réalité internationale d'une forte adoption des cultures modifiées génétiquement pour résister à des herbicides, à l'exception peut-être encore de l'Europe, où l'on se prépare actuellement à l'arrivée de ces cultures (Tillie et al. 2014; Lemaux 2008; GMO Compass 2014; EPEC 2011). Le glyphosate, aussi connu sous sa première appellation commerciale de Round-Up, est l'un des herbicides à large spectre intimement associé aux cultures transgéniques, dont les ventes progressent rapidement au Québec (MDDEP 2010). En fait, le glyphosate est presque partout au sommet des ventes d'herbicides (Giroux 2015; Giroux and Pelletier 2012; Gorse and Balg 2012; EPA 2011; Health Canada 2011; Environment Canada 2011; Eurostat and European Comission 2007).

Le glyphosate était initialement vu comme une alternative plus sécuritaire par rapport aux herbicides qu'il remplaçait sur le marché (Duke and Powles 2008). Les faibles indices de risque environnemental (IRE) et sur la santé (IRS) du glyphosate en font un choix de prédilection auprès des agriculteurs québécois (http://www.sagepesticides.qc.ca;Québec 2013). Malgré cela, plusieurs études relient le glyphosate à des effets délétères chez les végétaux non-ciblés (Gomes et al. 2014), menacés (Heard et al. 2003) ou vulnérables (Matarczyk et al. 2002) avec leurs populations d'insectes associées (Pleasants and Oberhauser 2013). De plus, les évaluations gouvernementales qualifiant les risques comme minimaux pour les mammifères, les oiseaux et la faune aquatique (EPA 2009) semblent contredites par des études rapportant des effets sur la biodiversité et la productivité des écosystèmes aquatiques (Relyea 2005; Pérez et al. 2007), et ce, même en deçà des critères de protection chroniques pour la vie aquatique (Smedbol et al. 2013). D'un point de vue épidémiologique, le glyphosate est par ailleurs corrélé à une douzaine de maladies humaines des temps modernes (Swanson et al. 2014) dont certains mécanismes métaboliques ont été élucidés (Samsel and Seneff 2013), et sa cancérogénicité a récemment été reconnue (Guyton et al. 2015; IARC 2014). Au Québec, les activités agricoles se concentrent dans la vallée du Saint-Laurent, une source d'eau potable pour 45 % des Québécois (Hébert and Belley 2005). Cependant, dans un maximum de 97.5% (donnée de 2013, intervalle entre 88 et 97.5% de 2011 à 2014) des eaux de surfaces dans les régions productrices de maïs et de soya au

Québec, des concentrations non négligeables de glyphosate ont été mesurées (Giroux and Pelletier 2015) et la contamination des eaux de surface ou sous-terraines est fréquente à travers le monde (Aparicio et al. 2013; GEUS 2013; Horth and Blackmore 2009; Litz et al. 2011; Scribner et al. 2007; Struger et al. 2008). Par conséquent, il est important de mieux documenter la migration du glyphosate vers l'eau, ainsi que les méthodes de la limiter, pour mitiger notre exposition au glyphosate.

I.2 Les bandes riveraines

Parmi les bonnes pratiques agricoles permettant de minimiser les conséquences néfastes des produits agro-chimiques, l'utilisation des bandes riveraines (Figure 1) est une méthode de protection de dernière ligne qui intervient tout juste avant que les intrants agricoles rejoignent les ruisseaux (Moore et al. 2008; Bentrup 2008). Une bande riveraine est essentiellement une zone tampon végétalisée à l'interface des champs et des cours d'eau (Naiman and Decamps 1997) permettant d'atténuer le lessivage des polluants (Dabney et al. 2006). Les mécanismes qui y opèrent incluent : l'atténuation de la dérive éolienne ou du ruissellement; l'augmentation du dépôt des particules de terre érodés; la favorisation de l'infiltration et la dilution des intrants agicoles; l'absorption par le biota; le changement des potentiels d'oxydo-réduction et l'adsorption sur la matière organique ou les particules de sol; la diversification ou l'augmentation des populations microbiennes et de leurs activités enzymatiques dans les sols et la rhizosphère; et l'accélération du métabolisme ou du co-métabolisme des polluants (Locke et al. 2006; Dabney et al. 2006; Osborne and Kovacic 1993, Davis et al, 2007b).

Les bandes riveraines sont largement recommandées en Amérique du Nord (Hickey and Doran 2004) et ailleurs dans le monde (Smethurst et al. 2009). Au Québec la *Politique de protection des rives, du littoral et des plaines inondables* prône la conservation de bandes riveraines étroites (≥ 3 m) en bordure de tous les cours d'eau agricole (MDDEP 2005). Mais la largeur choisie est le fruit d'un compromis socio-économique, entre la mitigation des polluants et l'impact économique pour les producteurs agricoles et faciliter l'application de la politique, plutôt que d'optimiser l'efficacité dans la mitigation des nutriments ou des pesticides (Nolet, 2004). D'où l'intérêt de tester dans des conditions au champ, en milieu ouvert, l'efficacité des bandes riveraines recommandées au Québec.

Dans la littérature, on quantifie l'efficacité des bandes riveraines de différentes façons, soit (a) par égard aux concentrations d'éléments aqueux (nutriment, polluants) ou (b) en référant à un bilan de masse mettant en relation les concentrations et les flux traversant les bandes riveraines (Hill 2000). Les bilans de masse exigent une quantification des flux qui se prête difficilement aux designs expérimentaux visant à ne pas perturber le milieu ou les écoulements naturels. En effet, le positionnement de partitions enpêchant le ruissellement entre les parcelles expérimentales et le creusage de tranchées pour intercepter et quantifier l'ensemble des flux de surface impliquent des modifications importantes du milieu (en plus de décupler les coûts). Par ailleurs, la quantification des flux peut-être biaisée lorsque sont utilisé des équipements de captage de l'eau actif (par exemple des lysimètres sous tension pour capter l'eau souterraine). Parce qu'elles permettent d'augmenter le nombre de parcelles échantillonnée à une fraction du coût, et parce qu'elles impliquent une perturbation minimale du milieu, de nombreuses études de bandes riveraines utilisent plutôt les concentrations d'éléments aqueux pour quantifier l'efficacité d'une bande riveraine. Cette efficacité peut être calculée de deux façons, soit (a) par égard aux concentrations mesurées en absence (contrôle) ou présence d'une bande riveraine ou encore (b) en comparant les concentrations entrantes (contrôle) et sortantes de la bande riveraine (Krutz et al. 2005). Nous avons retenu la méthode (b), car l'utilisation des données avant la bande riveraine permet de limiter l'attribution incorrecte du retrait des polluants à la bande riveraine (contrairement à la méthode (a) où des processus comme la dénitrification et le mélange avec les eaux souterraines seraient impossible à distinguer; Noij et al. 2012). La mesure d'efficacité des bandes riveraines pour retirer un polluant X est ainsi généralement exprimée en pourcentage, en fonction des concentrations retrouvées avant et après la bande riveraine (Eq. 1; Schultz et al. 1995, McKergow et al. 2006, Duchemin et Hogue 2009):

$$\text{Eq. 1: Efficacité (\%)} = (([X_{avant}] - [X_{après}]) \times [X_{avant}]^{-1}) \times 100$$

S'il est nécessaire de tester l'efficacité la bande riveraine étroite préconisée par une politique gouvernementale dans des exploitations agricoles du Québec, c'est parce que l'efficacité de la bande riveraine est souvent jugée proportionnelle à sa largeur (Mayer et al. 2006). Mais comme les herbacées, arbustes ou arbres des bandes riveraines y jouent un rôle clé (Hickey and Doran 2004), le type et la densité des végétaux qui composent la bande riveraine sont importants (Mayer et al. 2006). Dans un contexte où une largeur de bande riveraine unique est préconisée par une politique, la sélection des végétaux et la variation de la densité de plantation deviennent ici des variables clés. Il est donc pertinent de (1) voire si la politique québécoise est efficace et (2) tester une innovation visant la production de biomasse en bande riveraine.

Cette approche innovante jouxtant la rétention des polluants et la production de biomasse s'incrit dans une mouvance vers des bandes riveraines qualifiées de "multifonctionnelles" (Hickey and Doran 2004; Stutter et al. 2012; Fortier et al. 2010a; Adegbidi et al. 2001; Jobin et al. 1997). Outre ces deux fonctions, les bandes riveraines régulent aussi les débits hydriques et sont en général le site d'une plus grande-biodiversité. Il convient donc d'étudier ces bandes riveraines fournissant une panoplie de services écosystémiques sous un angle multidisciplinaire (Stutter et al. 2012) pour mieux comprendre s'il y a des interactions ou des conflits entre les diverses fonctions.

La production de biomasse en bande riveraine semble un des contextes les plus durables dans le milieu agricole (Rockwood et al. 2004; Licht and Isebrands 2005; Fortier et al. 2010b, a). Mais il faut rester prudent sur la possibilité qu'une monoculture de saules exotiques puisse nuire à la biodiversité. Des études antérieures ont montré que les plantations d'autres salicacées (peupliers) peuvent augmenter la diversité floristique à la ferme (Weih et al. 2003), sans causer d'extinction locale ou d'invasion (del Pilar Clavijo et al. 2005), et favorisent même la regénération de la strate arborée naturelle (Lust et al. 2001, D'Amour 2013). Peu de recherches se sont penchées sur le potentiel des plantations d'arbres ou arbustes à croissance rapide à des fins de production de biomasse en bande riveraine pour maintenir la biodiversité tout en minimisant la présence d'espèces exotiques ou invasives (IRSTEA 2014; Cavaillé et al. 2013, Fortier et al. 2011). L'écotone riverain supporte une faune et une flore ne prospérant pas ailleurs dans les champs (Boutin et al. 2003; Jobin et al. 2004). Même les bandes riveraines étroites peuvent avoir un impact positif que la biodiversité dans les fermes (Marshall et al. 2006, Fortier et al. 2011). Une flore diversifiée favorise les pollinisateurs ou agents de lutte biologiques qui améliorent la productivité des terres agricoles (Nicholls and Altieri 2013; Altieri et al. 2005). Paradoxalement, ces bandes riveraines hébergent aussi ce que les agriculteurs considèrent comme des mauvaises herbes (Fortier et al. 2011; Boutin et al. 2003). Par prudence, il convient au minimum de recenser la diversité végétale dans les bandes riveraines laissées en friche et dans les plantations de saules.

I.3 Le projet SABRE

Le projet CRSNG-stratégique SABRE — *Salix* en agriculture pour des bandes riveraines énergétiques — étudie les bénéfices environnementaux et l'acceptabilité socio-économique des bandes riveraines de saules en milieu rural et péri-urbain (Hénault-Ethier et al. 2014). *Salix miyabeana* SX64 a été sélectionné dans le projet SABRE pour sa croissance rapide et sa forte production de biomasse (Labrecque and Teodorescu 2003, 2005). Les chercheurs du projet multidisciplinaire SABRE ont aussi étudié les motivations des agriculteurs dans l'adoption des innovations, comme les bandes riveraines, (Racine 2015) et a pris en considération les nombreux défis de la gouvernance de l'eau dans un milieu où acteurs, enjeux, stratégies et normes sont variées (Dagenais 2015). Le cœur de la présente thèse, testant l'efficacité des bandes riveraines conformes à la politique québécoise pour filtrer la pollution diffuse agro-chimiques en milieu agricole (Chapitres 2, 3 et 4), complète des études en serre et en laboratoire visant à déterminer les capacités de phytoremédiation du glyphosate par les saules (Gomes 2015; Gomes et al. 2015b; Gomes et al. 2015a; Gomes et al. 2014), de même qu'une autre étude portant sur les conséquences du glyphosate sur la productivité et la biodiversité du phytoplankton dans les ruisseaux (Smedbol et al. 2013). Le deuxième élément clé de cette thèse repose sur le test d'une innovation consistant à produire de la biomasse ligneuse par rapport à simplement conserver une friche, une question qui pourraient influencerl'intérêt des agriculteurs pour les bandes riveraines.

I.4 L'organisation de la thèse

L'étude entreprise a été menée *in situ*, sur des bandes riveraines expérimentales matures, implantées deux ans avant le début de la thèse, avec des suivis dans le temps et sur trois ans de différentes variables biologiques et physicochimiques (Figure 2). L'objectif général de la présente thèse est donc de quantifier l'efficacité de deux types de bandes riveraines. Le type de bande riveraine le plus répandu au Québec est une friche colonisée spontanément par une strate herbacée diversifiée (Vézina 2014, communication personnelle). Nos recherches visent à comparer cette situation avec une plantation plus ou moins dense de *Salix miyabeana* SX64, un arbuste reconnu pour son adaptabilité en milieu riverain (Dickmann and Kuzovkina 2008; MAPAQ 2008), sa croissance rapide et sa forte production de biomasse (Labrecque and Teodorescu 2003, 2005) et son potentiel de phytoremédiation (Gasser et al. 2013; Börjesson 1999; Mirck et al. 2005; Kuzovkina and Volk 2009). L'organisation de la présente thèse consiste donc à évaluer le potentiel des saules à séquestrer des nutriments dans leurs tiges qui peuvent ensuite être récoltées comme biomasse énergétique (Chapitre 1), puis évaluer le potentiel de trois traitements de bandes riveraines à intercepter les flux aqueux chargés en nutriments (Chapitre 2) ou en glyphosate (Chapitre 3). Une caractérisation de la structure et de la diversité des herbacées poussant dans les bandes riveraines (présentée en annexe) vient appuyer les Chapitres 1, 3 et 4. Enfin, l'Annexe 4 apporte une description hydrologique des sites d'études pour valider si le ruissellement ou l'eau phréatique peut être intercepté par la bande riveraine.

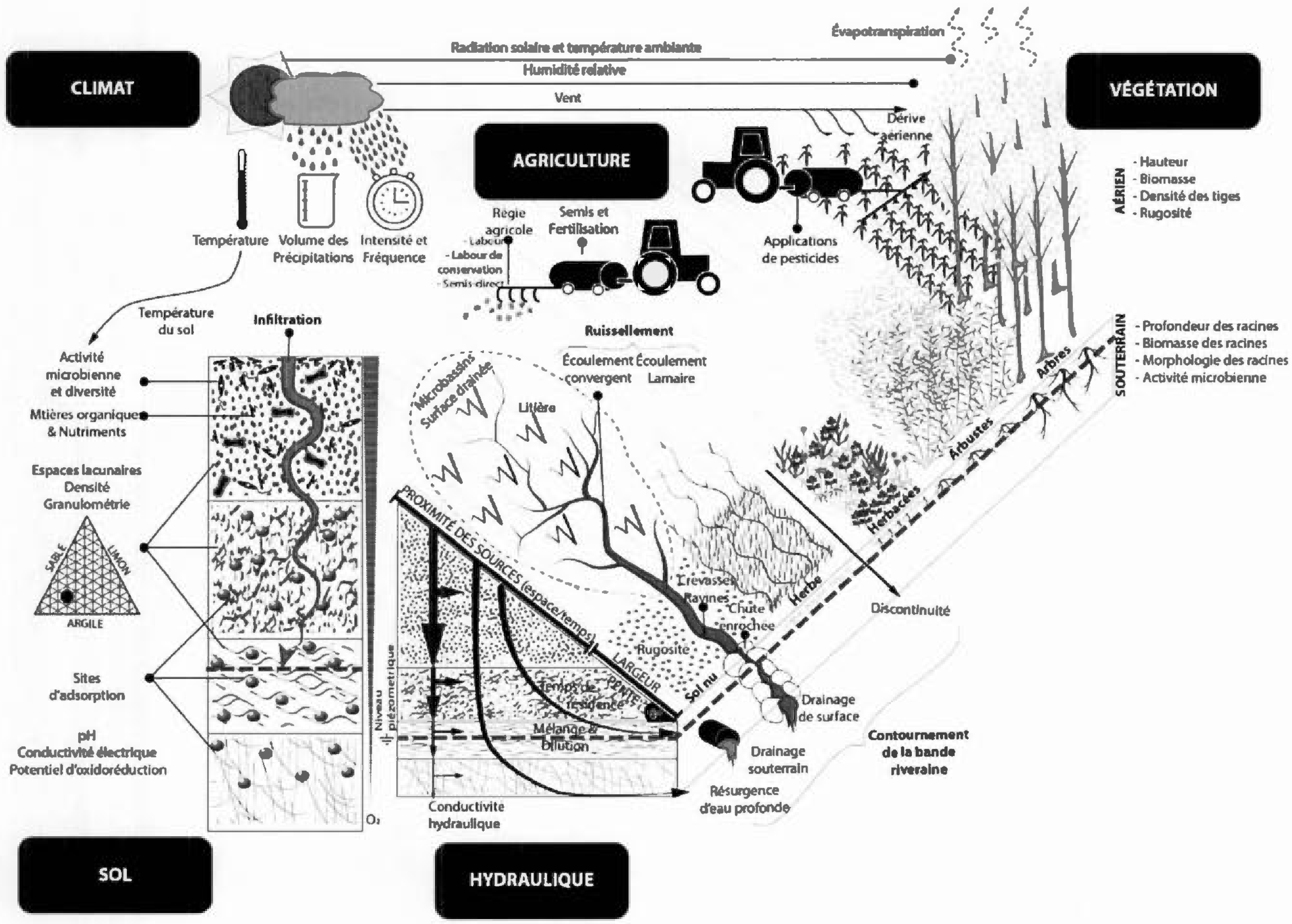

Figure 1: Exemple des diverses fonctions et mécanismes interagissant au sein des bandes riveraines, avec les groupes de variables environnementales qui peuvent jouer un rôle déterminant dans l'efficacité de la bande riveraine. Illustration : Louise Hénault-Ethier.

I.5 Les hypothèses générales pour les chapitres centraux

Chapitre 1 :

Basé sur l'observation que le type de végétaux composant une bande riveraine peut influencer son efficacité à mitiger la pollution diffuse (Mayer et al. 2006), que les saules ont un bon potentiel de phytoremédiation (Gomes 2015; Gasser et al. 2013; Börjesson 1999; Mirck et al. 2005; Kuzovkina and Volk 2009), que la production de biomasse peut-être une fonction soutenable des bandes riveraines multi-fonctionnelles (Rockwood et al. 2004; Licht and Isebrands 2005; Fortier et al. 2010b, a) sans être nécessairement préjudiciable à la biodiversité ou à la conservation végétale (Weih et al. 2003; del Pilar Clavijo et al. 2005; Lust et al. 2001, D'Amour 2013), tester l'hypothèse que les saules arbustifs constituent une option intéressante pour les bandes riveraines étroites. Nos hypothèses spécifiques sont qu'une plantation de saules à haute densité produit plus de biomasse qu'une plantation à faible densité (ce qui peut représenter un intérêt économique pour les agriculteurs).

Chapitre 2 :

Les bandes riveraines végétalisées permettent généralement d'atténuer le lessivage des nutriments (Dabney et al. 2006). Mais son efficacité est proportionnelle à sa largeur, au type et à la densité des végétaux qui la composent (Mayer et al. 2006). Nous émettons donc l'hypothèse que l'efficacité des bandes riveraines peut être augmentée, sans augmenter la largeur de celle-ci, en sélectionnant des espèces végétales plus efficaces (i.e. les saules ont un bon potentiel de phytoremédiation (Gomes 2015; Gasser et al. 2013; Börjesson 1999; Mirck et al. 2005; Kuzovkina and Volk 2009)), et en augmentant la densité de végétation qui y pousse (parce que la séquestration de nutriments est proportionnelle à la biomasse végétale dans les bandes riveraines; Jianqiang et al. 2008). Notre hypothèse spéficique est donc que l'efficacité de la bande riveraine sera proportionnelle à la densité des saules, et que la bande herbacée sera la moins efficace.

Chapitre 3 :

Parce que plusieurs études ont démontré l'efficacité des bandes riveraines pour mitiger la pollution diffuse liée aux herbicides dans différents milieux (Krutz et al. 2005; Schmitt et al. 1999). Aussi, les saules ont un potentiel démontré de phytoremédiation du glyphosate en milieu contrôlé (Gomes et al. 2015). Nous avons émis l'hypothèse que des bandes riveraines de saules pourraient jouer un rôle utile dans la mitigation des effluents de glyphosate dans les champs agricoles. Cette hypothèse s'avère utile parce que l'efficacité des bandes riveraines pour limiter la pollution diffuse liée à l'herbicide le plus vendu sur la planète a peu été étudié, en particulier avec des saules. Notre hypothèse spéficique est encore ici que l'efficacité de la bande riveraine sera proportionnelle à la densité des saules, et que la bande herbacée sera la moins efficace.

I.6 Les objectifs spécifiques

Dans le Chapitre 1 on (1) évalue le potentiel des saules à séquestrer des nutriments et à produire de la biomasse ligneuse dans des BR de 3 m; (2) analyse les interactions entre les variables environnementales qui affectent la croissance et la productivité des saules; et (3) présente une régression linéaire permettant aux agriculteurs de prédire la biomasse des tiges en vue d'optimiser la récolte des bandes riveraines.

Le Chapitre 2 a pour objectif de (1) déterminer si les nutriments retrouvés dans les champs suite aux ajouts d'engrais et d'amendement organiques (NO_2^--NO_3^-, NH_4^+, PO_4^{3-} and K^+) se dissipent lorsqu'ils s'infiltrent à travers les différentes strates de sol sur les marges des champs, en les comparant avec le comportement d'autres cations naturellement présents ou amendés; (2) distinguer l'efficacité de la BR à trois moments clés du calendrier agricole, soit la fonte nivale, après la fertilisation et après les applications d'herbicides à base de glyphosate; et (3) quantifier si l'efficacité de la bande riveraine est proportionnelle à la densité de plantation des saules ou si ces derniers sont plus efficaces que les parcelles en friche enherbées, pour enfin (4) valider si l'eau collectée après ces divers traitements de BR est conforme aux standards de la qualité de l'eau.

Le Chapitre 3 s'intéresse ici au glyphosate et vise à déterminer (1) l'efficacité de la bande riveraine dans la rétention du glyphosate et de l'AMPA issu du ruissellement ou dans l'eau interstitielle qui s'infiltre dans le sol, toujours en fonction de l'hypothèse que les saules à haute densité seront plus efficaces que les saules à faible densité et que la friche herbacée; (2) l'efficacité de la bande riveraine dans la rétention du glyphosate dans un autre substrat, soit adsorbé aux particules de sol; (3) et enfin si l'efficacité de la BR est tributaire des flux en amont de celle-ci, comment varient les concentrations en glyphosate (a) à travers les différentes étapes du calendrier agricole (fonte nivale, après la fertilisation, après les applications d'herbicides à base de glyphosate) et (b) avec la profondeur, lors de l'infiltration dans le sol.

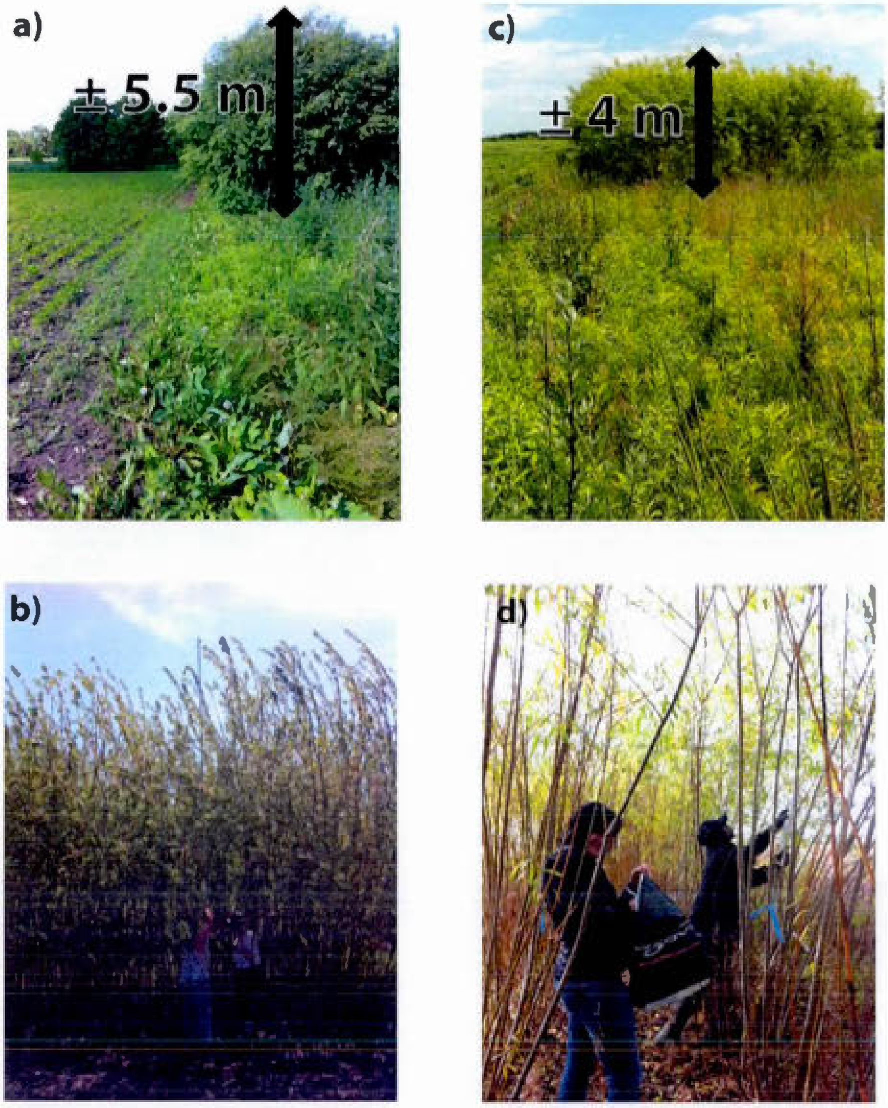

Figure 2: Apparence des bandes riveraines herbacées (avant-plan) et plantées en saules (arrière-plan) en bordure des champs à Boisbriand (a,b) et Saint-Roch-de-l'Achigan (c, d).

I.7 L'approche expérimentale utilisée et sont originalité

Pour tester d'une part (a) l'efficacité des bandes riveraines conformes à la PPRLPI pour mitiger la pollution agro-chimique diffuse, et d'autre part (b) l'innovation que représente les plantations de saules à croissances rapides pouvant intercepter la pollution diffuse tout en produisant de la biomasse, nous avons choisi de travailler dans des conditions ouvertes, au champ, en effectuant un suivi des variables physico-chimiques sur trois ans. Les bandes riveraines ayant été implantées en 2009, deux saisons de croissance avant le début de la thèse, on s'assurait ainsi qu'elles étaient bien établies avant le début des expérimentations. Par ailleurs, les deux sites expérimentaux ont été choisis parce qu'ils représentent deux paysages dinstincts dans la plaine du Saint-Laurent, augmentant ainsi l'intérêt des conclusions dégagées de la présente étude. Les parcelles expérimentales sont situées à Saint-Roch-de-l'Achigan (SR), une région rurale dominée par l'agriculture; et Boisbriand (BB), une région rurale sous l'influence de l'étalement urbain (Figure 3). Comme plusieurs autres champs dans la plaine du Saint-Laurent, la terre minérale de SR, est compactée d'une part à cause de sa granulométrie et du passage de la machinerie agricole (série Achigan) et elle surmonte les argiles de l'ancienne mer de Champlain, (MAPAQ 1990; Lajoie 1965). À BB, une riche terre organique (humisol noir fortement décomposé sous la bande riveraine et série Châteauguay, Dalhousie et Saint-Bernard dans les champs) avec une nappe phréatique affleurant la surface du sol dans les parties les plus basses offre des caractéristiques souvent rencontrées dans des dépressions où sont implantées les bandes riveraines (Lajoie 1960; Collins and Kuehl 2000). Il est envisageable que des résultats similaires aux nôtres puissent être obtenus ailleurs dans la plaine du Saint-Laurent, ou dans des sites ailleurs au monde avec une pédologie, une fertilité, une hydrologie et un climat comparables. Cependant, ce sont les conclusions à l'égard de la politique sur les bandes riveraines et de l'innovation que représentent les bandes riveraines de saules à croissance rapide qui sont les plus généralisables et qui revêtent le plus grand intérêt pour les décideurs et les agriculteurs d'ici et d'ailleurs.

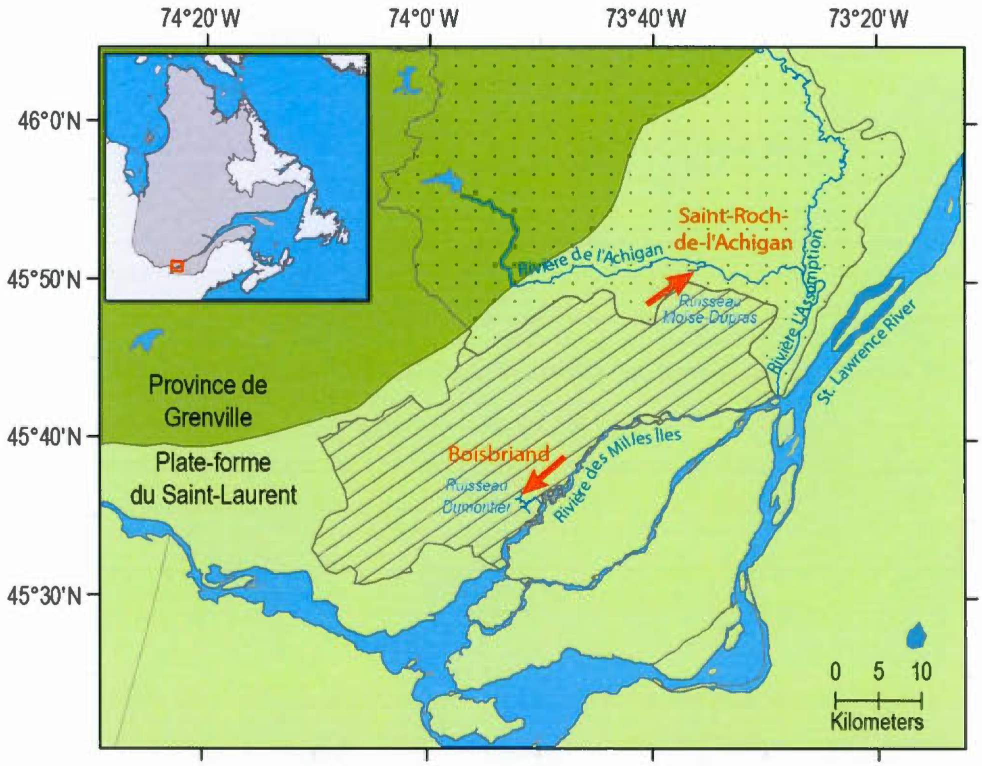

Figure 3: Localisation des deux sites d'études de Boisbriand et de Saint-Roch-de-l'Achigan dans les bassins versants de la Rivière des Milles-Isles et de l'Assomption, respectivement.

Sur chaque site, trois traitements ont été implantés en triplicata et de façon aléatoire. Les trois traitements consécutifs consistent en un une friche d'herbacées spontanées, et deux densités de saules, soit 3 rangs (33 333 tiges/ha) ou 5 rangs (55 556 tiges/ha). Chaque parcelle mesure 3 m de large x 17m de long, et les trois traitements sont disposés bout à bout sur chaque blocs qui sont eux séparés de part et d'autre de la rive ou des chemins de ferme, à une dizaine de mètres de distance environ. Les instruments d'échantillonnage de l'eau ont été implantés en 2011 afin d'échantillonner le ruissellement, l'eau interstitielle en zone non-saturée et phréatique avant et après la bande riveraine. Dans chaque parcelle, il y avait avant et après la bande riveraine des équipements d'échantillonnages pour chaque profondeur (à 0 cm un collecteur de ruissellement de surface, à 35 et 70 cm, des lysimètres sous tension négative).

Dans la présente étude nous avons soigneusement délimité trois périodes critiques pour cibler nos échantillonnages, soit à la fonte nivale, après les semis et la fertilisation et enfin après l'application des herbicides à base de glyphosate. En tout, 18 campagnes d'échantillonnage ont été réalisées entre 2011 et 2014. Ce sont donc plus de 1 100 échantillons qui ont été collectés. Le succès de chaque période d'échantillonnage (basé sur la collecte d'eau dans un équipement au moment ciblé par une campagne) a varié entre 40-53% pour le ruissellement et 56-90% pour l'eau souterraine (moins fortement influencée par les conditions climatiques arides en été). Pour pallier à ceci, tous les échantillons disponibles ont été analysés individuellement en laboratoire, mais les concentrations mesurées ont été regroupées par campagnes pour la suite des analyses statistiques tel que décrit dans les Chapitres 2, 3 et 4.

De plus, contrairement aux études sur les bandes riveraines en milieu contrôlé (parcelles hydrologiquement séparées (Laitinen et al. 2009), pluies artificielles (Tingle et al. 1998; Webster and Shaw 1996; Krutz et al. 2005), ruissellement synthétique (Dosskey et al. 2007)), notre étude a été réalisée dans des exploitations agricoles réelles. On dégage ainsi mieux le potentiel réel des bandes riveraines sujettes à la variabilité des précipitations ou à l'hétérogénéité des trajectoires de ruissellement à travers la bande riveraine (Arora et al. 2010). C'est seulement dans ce contexte que l'on peut réellement tester l'efficacité des bandes riveraines préconisées par la politique québécoise.

Aussi, la persistance, le lessivage et l'infiltration du glyphosate était un phénomène peu étudié dans les conditions agricoles et climatiques propres au Québec, ce qui s'avère pourtant essentiel vu l'importance prépondérante du climat dans le comportement du glyphosate (Helander et al. 2012). L'efficacité des bandes riveraines pour mitiger le glyphosate était aussi quasi-absente dans la littérature internationale (Krutz et al. 2005; Arora et al. 2010; Syversen and Bechmann 2004). Encore au Québec, le gouvernement se prépare à réagir à l'annonce de la cancérogénicité du glyphosate qui arrive simultanément avec de nouvelles données démontrant un taux de contamination alarmant et des concentrations à la hausse dans nos eaux de surface (Giroux 2015). Notre étude arrive donc à point pour renseigner les prochaines actions gouvernementales. Finalement, le Québec s'est doté d'une politique prônant des bandes riveraines étroites, mais il semblait manquer une quantification de leur efficacité à l'échelle des champs, en milieu non-contrôlé sur des terres non drainées et sur plusieurs saisons de croissance.

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CHAPITRE I
HIGH BIOENERGETIC YIELDS OF RIPARIAN BUFFER STRIPS PLANTED WITH *SALIX MIYABENA* SX64 ALONG FIELD CROPS IN QUÉBEC, CANADA

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Abstract

In the province of Québec, Canada, the Policy for protecting shores, coasts and flood plains recommends the presence of 3m wide riparian buffer strips (RBS) along field crops for minimizing agro-chemical leaching to surface waters. Fast-growing woody crops like *Salix miyabeana* SX64 could generate revenues from energy dedicated biomass production within the RBS land area withdrawn from grain production. To determine the potential biomass productivity, experimental RBS were established on two sites — Saint-Roch-de-l'Achigan (SR) and Boisbriand (BB) — with three treatments set up in triplicata composed of mixed herbaceous vegetation (CX), low (3X: 3 rows) and high (5X: 5 rows) density plantations of willows. Growth (stem number, diameter and height) was measured annually from 2011 to 2014; and yield (t dw·ha^{-1}) was measured in 2014. Growth and yield significantly differ between sites (SR < BB). The 23-24 t dw stems·ha^{-1}·year^{-1} yield in compacted sandy loam soil at SR resembles typical field productions but the 56-89 t dw stems·ha^{-1}·year^{-1} in a humisol at BB is a record in the *Salix* literature. At BB, neither intraspecific competition nor competition with understory herbaceous vegetation appears to hamper productivity, in fact enhanced diversity is associated with better productivity. At SR, a greater soil coverage by herbs or introduced species corresponds to reduced willow productivity. In the light of the present research, we conclude that interesting biomass yields may be produced in RBS. This result should be considered along with harvesting, transformation or market opportunities to help farmers assess the viability of this practice on their farm.

1.1. Introduction

1.1.1 Riparian buffer strip policy

Agriculture plays a major role in the economy of rural Quebec, Canada, with a dominance of monoculture and rotational cropping for corn and soybean in the Saint-Lawrence lowlands (Sall et al. 2015). Agriculture- derived diffuse pollution composed of nutrients, pesticides or eroded soil particles is a leading cause of water quality degradation worldwide (EPA 2003; Ongley 1997; Ongley et al. 2010). "Edge-of-field" (Dabney et al. 2006) riparian buffer strip (RBS) (Naiman and Decamps 1997) may mitigate nutrient erosion and leaching by retaining or transforming organically-bound, adsorbed or dissolved nutrients carried by rain and snowmelt (Gagnon and Gangbazo 2007). In Quebec (Canada), the Policy for Protecting Shores, Coasts and Floodplains (PPRLPI, in French) recommends that farmers maintain a 3-m-wide vegetated buffer strip along streams in agricultural zones (MDDEP 2005). However, many farmers feel expropriated from their own lands by this policy which restricts crop culture in riparian zones with the aim of protecting water resources (Dagenais 2015). Considering that Quebec has approximately 2 million ha devoted to agriculture, and that of this, 3m-wide RBS would represent only ≤8780 ha of this land (MENV 1998), it is critical to correctly establish potential alternative uses for this area representing 0.5% of the agricultural landscape.

1.1.2 Riparian buffer strips planted with *Salix*

The most common type of RBS in Québec agricultural landscapes are those spontaneously colonized by a diversified herbaceous strata. But to fulfill ecosystemic services such as phytoremediation of nutrients or pesticides, a larger biomass production may equate greater removal efficiencies (Rockwood et al, 2004), especially if the biomass can be harvested and exported out of the system because for the common alternate RBS in place, decomposing herbs eventually release the sequestered nutrients on site (Hefting et al. 2005). However tall trees may be viewed as undesirable by farmers partly due to the shade trees can cast on nearby crops (Marchand & Masse 2008). Willow buffer strips cut at ground level every three years, would encourages biomass production and multiplication of stem numbers, and it would also limit casting detrimental shade on nearby crops. Hence a shrubby strata may represent a good compromise.

Willow shrubs are well suited for riparian habitats (Dickmann and Kuzovkina 2008; MAPAQ 2008), and are recommended for buffer strips in Quebec (MAPAQ 2008). Willows are renowned for their rapid growth and capacity to intercept nutrients, organic and inorganic chemicals (Mleczek et al. 2009; Hultgren et al. 2010; Kuzovkina and Volk 2009; Gomes et al. 2015a). Among the 450 existing species (Kuzovkina & Quigley 2005), *Salix miyabeana* SX64 is a particularly fast grower and high biomass producer (Labrecque and Teodorescu 2003, 2005), and has documented phytoremediation qualities (Börjesson 1999; Mirck et al. 2005; Kuzovkina and Volk 2009). It's non-point source pollution mitigation capacity has been demonstrated in hydrologically partitioned RBS parcels in Quebec (Gasser et al. 2013, Chapter 2 and 3). Beyond fulfilling desired RBS ecological functions, willows represent a local source of energetic biomass which may be economically beneficial for farmers and landowners (Abrahamson et al. 1998; Börjesson 1999a, b). Multifunctional buffer strips, planted with woody or biomass crops may be amongst the most sustainable options to produce these commodities (Rockwood et al. 2004; Licht and Isebrands 2005; Fortier et al. 2010b, a). Willows are considered a particularly sustainable biomass crop as they rarely rely on insecticides and fungicides (though herbicides are sometimes used for RBS implantation (clearing of pre-existing vegetation) and reclamation (for example removal of willow stumps when productivity declines); Albertsson 2012). On the other hand, woody plantations, often using non-native plants, may be perceived negatively by environmentalists due to their lower diversity than natural stands (Stephens and Wagner 2007; Rosoman 1994), and by farmers due to perceived competition with crops or perceived risk of damaging farming equipment (Marchand and Masse 2008).

1.1.3 Goals

The main objective of this study was test an innovative multifunctional RBS system in the context of the Québec policy which recommends 3 m wide RBS in agricultural areas. We tested how much biomass could be produced by *Salix miyabeana* SX64 buffer strip, and if this yield could be augmented by increasing the plantation density. To help farmers assess field productivity non-destructively, prior to harvest, we built a regression model using willow morphometric growth variables during the 3rd year of growth. To understand the effect of proximity to agricultural fields in biomass production (linked with possible enhancement from nutrients or interference of herbicides in runoff), along with competition between willows or herbaceous plants, a multivariate analysis integrating the influence of environmental variables on willow growth and productivity was conducted.

1.2. Materials and Methods

1.2.1 Study Site

The study was conducted on two sites 33 km apart, Boisbriand (BB: 45°36'40"N; 73°51'40"W) and Saint-Roch-de-l'Achigan (SR: 45°50'48"N; 73°36'17"W), north of Montreal, Canada.

SR, like many fields in the St-Lawrence lowlands, is characterized by a flat topography (less than 3 m elevation difference from the highest point in the field to the RBS, with a 2 m vertical drop from there to the stream) and a deep water table (down to 2.25 m from surface in late summer). The soil series is mapped as Achigan very fine sandy loam (IRDA 2009), which is a gley podzol of alluvial origin with a light texture originating from deposits of the Champlain Sea (Lajoie 1965). Arable soil has an average depth of 30 cm. Quaternary deposits may reach 75-150 cm (Lajoie 1965), sitting atop a 6 m clay bed, a 3 m gravel clay (according to a forage 1 km away) with the bedrock at an average depth of 6 to 9 meters (MDDEP 2006). Drainage is characterized as imperfect due to the texture of the deposits, and lateral ground water movements above the clay are slow due to the flat topography, which may lead to poor aeration, especially in spring. No improvements to surface drainage nor any installation of underground drainage tiles were made to the site.

BB has hilly topography (with ± 15 m elevation difference from the highest point of the field to the stream), the water table depth is even with soil surface at spring melt and 1.3 m deep in summer low water and the buffer strips are established in a rich organic black soil,originating from peat bog decomposition (Guérin 2009).While only four percent of meridional Quebec is constituted of organic soils (Guérin 2009), these soils are common in the depressions near streams where RBS are often implemented (Lajoie 1960; Collins and Kuehl 2000). Pedology is mapped as Châteauguay clayey loam and Dalhousie clay to clayey loam North of the buffer strips. On the South shore, the soil is mapped as Chicot fine sandy loam and Saint Bernard fine sandy loam (IRDA 2008). A drilling-0.5 km away situates the bedrock 13.4 m below the soil surface (MDDEP 2006). No improvements to surface drainage nor any installation of underground drainage tiles were made to the site.

The fields were under rotations of soy (S) and maize (M), with the following cycles between 2010 and 2013: BB: S-S-M-S and SR: S-S-M-M. The crops grown are resistant to glyphosate and spraying was conducted by the farmers once a year (June) in both fields at the recommended rates. See Chapter 2 and 3 for agronomic details.

1.2.2 Pedological Characterization

The soil granulometry (Annexe 1) was characterized at the surface and 35 cm depth according to the wet sifting methodology adapted from the Centre d'expertise en analyse environnementale (2010) which included dissolution of organic matter with 30% H_2O_2 and the use of dispersing and anti-foaming solutions. Two mm, 212 µm and the 63 µm sieves were used and the sand and silt fraction of a surface sample was further differentiated with a sedigraph (Analysette 22 Compact Laser Particle Sizer, FRITSCH, GmbH, Germany). Soil was further characterized for moisture content (oven dry at 105°C), soil organic matter (loss on ignition at 550°C; Carter and Gregorich 2007), soil organic carbon (24h fumigation with HCl), total Carbon and total Nitrogen (elemental analyzer, Carlo Erba NC2500 Milano, Italy), density (on a wet basis), pH (1 part distilled water : 10 parts soil) and carbonates (sequential loss on ignition at 950°C; Heiri et al. 2001; Annexes 2 et 3). In BB, the soil stratigraphy (from top to bottom) includes black histosol (strongly decomposed on von Post Scale), brown histosol (less decomposed), peat (lightly decomposed), till, marl, grey clay and reddish clay. Organic-rich soil is generally present everywhere at 30 cm depth while marl and/or clay is found near 70 cm. In SR, sandy loam, clean sand lentils and clay with traces of iron oxides (FeOX) were observed from top to bottom. Though surface soil appeared homogeneous on both sites, below ground soil strata varied slightly between parcels (a detailed 3D stratigraphic model is provided in Annexe 4).

1.2.3 Climate

Climatic data (precipitation, temperature and degree days) was extracted from the corrected Agro-Meteo online database (Lepage and Bourgeois 2011) based on regional Environment Canada stations (stations Ste-Thérèse West 6.8 km from BB and L'Assomption 13.8 km from SR). Relative ambient humidity and solar radiation were extracted from the Daymet database (Thornton et al. 2014; Thomton et al. 1997). During the 2011-2013 growth period, all climatic variables were constant and did not vary significantly between sites (Figure 1-1). The 30-year average precipitation for the active growing season (April to October) in BB is 628-667 mm and in SR it is 528-627 mm (based on Agriculture and Agro-Alimentaire Canada, Lepage and Bourgeois 2011).

1.2.4 Plantation and vegetation maintenance

The soil of the buffer strips were completely cleared of vegetation prior to plantation using mechanical weeding (no mulch was used). The willow cultivar *Salix miyabeana* SX64 was selected based on its high biomass productivity and its good resistance to diseases and insects (Labrecque and Teodorescu 2005). Willows were planted in spring 2009 and cut back at the end of their first growing season as per the recommended standard practice to encourage the development of multiple-stemmed stools in the following years (Guidi et al. 2013) and facilitate weed control in the first year (Albertsson 2012). The aboveground biomass was then harvested before the beginning of the growth season in spring 2011 (in order to set year zero of shoots for experimental purposes) and again in fall 2013 (normal harvest based on a three year cycle). The control plot and edges of willow plantations were mowed by the farmers once per growing season in August, but the herbaceous vegetation was not harvested.

1.2.5 Experimental design

On each site, an experimental design comprising of three randomized blocks was set up. Each block included three randomized treatments (3 m width x 17 m long), a control zone with ruderal vegetation (CX), and two planted zones with three (3X) or five rows (5X) of willows. On a row, there was 30 cm between plants and 0.75 (3X) or 1.5m (5X) between rows, leading to plantation densities of 33 333 and 55 556 plants·ha^{-1} respectively (see discussion section 1.4.1 concerning plantation density in RBS compared to typical field plantations).

1.2.6 Vegetation Sampling

Willow growth (2011-2013) – Non-destructive willow growth variables were measured at the end of each of the 2011 to 2013 growing seasons. At both BB and SR sites, ten random plants were sampled along three rows (close to the field (CF), in the middle of the buffer width (CC) and close to the stream (CR)) for each treatment (3X, 5X) and each block (3) for a total of 360 samples. The number of stems per plant, stem diameter (3 stems per plant, caliper ±0.1mm, 10cm aboveground) and height (3 tallest stems per plant, from origin to apex) were recorded.

Willow productivity (2013) - The willow biomass was determined at the end of the third growing season. Five willows from each row (CF, CC, CR), for a total of 15 willows per parcel, were cut and weighed wet in the field (±0.1kg). Five subsamples of branches were used to determine the percent humidity (70°C until constant mass) to convert the data into dry mass (dm). Productivity (MT dm/ha) was calculated based on the 33 333 (3X) or 55 5556 (5X) stumps ha^{-1} densities.

Willow nutrient content – The average nutrient content of willow stems was obtained from the literature. Care was taken to select values from similar climatic and cultural environments and where possible, similar clones. The average N (Cavanagh et al. 2011; Labrecque and Teodorescu 2003; Toillon et al. 2013; Adegbidi et al. 2001; Gasser et al. 2013), P (Gasser et al. 2013; Adegbidi et al. 2001) and K (Gasser et al. 2013) content expressed in $g \cdot kg^{-1}$ stems was then converted into $kg \cdot ha^{-1}$ based on the average willow biomass productivity measured in the current study.

Herbaceous vegetation sampling and characterization – Aboveground herbaceous vegetation dry mass was obtained from 65 cm-diameter circular areas taken from the center of each RBS plot. Herbaceous vegetation coverage and height were obtained via the line intercept method (3m stretches on the CF, CC and CR sides; Annexe 5) and recorded as standardized coverage and height classes (Boivin et al. 2000). Plants were identified to the species level using internationally accepted scientific taxa names (Brouillet 2010+). The plants inventoried (Annexe 6) had different dominant communities on both sites (Annexe 7).

1.2.7 Environmental variables

Environmental variables distinguishing willow growth (2011-2013 annual heterogeneity) and willow productivity (2013 only) with **intra-site** environmental variability (spatial heterogeneity within fields and position relative to the stream) were compiled for multivariate statistical analysis (Annexe 8). **(1) Climatic** variables include: annual sum of precipitations (mm), sum of degree-days ($^{0}C \cdot d$), mean temperature (daily min, max, average; ^{0}C), mean solar radiation ($watt \cdot m^{-2}$) and mean ambient humidity (%) (Figure 1- 1). **(2) Global positioning system (GPS)** variables include: cardinal orientation (sun availability) and localization (X, Y, Z coordinates). **(3) Cultural** variables include: total doses of N, P, K, Mg and Ca ($kg \cdot ha^{-1}$) applied to the field during the sampling year, total dose of herbicides containing glyphosate (kg acid equivalents·ha^{-1}) applied to the field during the sampling year, and row crop yield (mt·$year^{-1}$ of sampling year). **(4)** Runoff (from surface collectors) **Water** physico-chemical variables include: pH, PO_4^{3-}, P_{tot}, $NO_2^-+NO_3^-$, NH_4^+, DOC, K^+, Mg^{2+}, Mn^{2+}, Na^+, Zn^{2+}, Ca^{2+}, Fe^{2+}, Al^{3+}, glyphosate as well as its degradation product AMPA (µg·ml^{-1}); (Chapter 2, 3-year means are summarized in Annexe 9). **(5) Soil** Physico-Chemical variables include: moisture (%), organic matter (%), pH, EC (µS·cm^{-1}), carbonates (%) and stratigraphy under the willows (derived from soil cores observations (Annex 2). Soil nutrient contents were not available for every sampling station and time points, and hence were not included in the multivariate analysis. Briefly soil content in P, K, Ca and Mg (kg/ha) were 297, 569, 6395 and 1-10 in BB and 129-239, 90-147, 2057-6263, and 877-1407 in SR (see Table 2- 1 in Chapter 2 for details).

(6) Hydrological variables influencing water availability for growth include: runoff (L) collected on the CF and CR sides of the RBS; drainage basins surface area (m^2) computed from digital elevation models using three geographical assumptions ("bassins" calculated for each runoff collector, "stream" calculated from closest modeled runoff flow collecting area, and "drainage point" calculated where the RBS discharge reaches the stream), slope (actual or absolute value), phreatic table absolute elevation (m), depth from the surface (connectivity or no connectivity model; distance in m), and head measured as the water table height differences between field and stream side (m). Precipitations were grouped with the climatic variables in group 1 above. All methodological details and results for these variables are exposed in Annexe 4.

(7) Herbaceous vegetation variables include: herbaceous biomass (kg dw/ha), sum herbaceous cover (% cover), life cycle (annuals; biennials; perennials; soil cover %) (Marie-Victorin and Rouleau 1964; Flora of North America Editorial Committee 1993+; Agriculture and Agri-Food Canada 2014), weed diversity (sum cover %; proportion %; n sp) (Bouchard et al. 1998; Gouvernement du Québec 1981; Marie-Victorin and Rouleau 1964; Moisan-De Serres et al. 2014; Agriculture and Agri-Food Canada 2014; USDA 2014a), exotic weed diversity (n sp), hydrophytes vs non-hydrophytes (sum cover %; proportion %; n species) (USDA 2014b;

Gauthier et al. 2008, Annex 10), indigenous herbs (soil cover %; proportion; n species) (Brouillet 2010+; Marie-Victorin and Rouleau 1964), shade tolerance (intolerant, intermediate, tolerant; proportion %; n species) (Marie-Victorin and Rouleau 1964; USDA 2014b; Flora of North America Editorial Committee 1993+; USDA 2014a; OMAFRA (Ontario Ministry of Agriculture Food and Rural Affairs) 2013; Mulligan et al. unknown; Klinkenberg 2014), tap vs fibrous root morphology (Caradus 1977), herbaceous plant height (class median, cm), plant diversity (including willow), Shannon diversity H' (including willow) (Shannon 2001), Simpson D^{-1} (Simpson 1949) and bioarea (height x % cover; Elias and Dias 2004, Descoings 1975; Annexe 11). Methodological details for selecting each herbaceous vegetation ecological characteristics (Annexe 12), and how they vary with RBS treatment (CX, 3X, 5X) and side (CF, CC, CR) (Annexes 13 and 14) are presented in Annexes.

1.2.8 Statistical analysis

When data conformed to the normality and homoscedasticity requirements, a factorial ANOVA with buffer strips introduced as random blocks tested factors including treatment (CX, 3X, 5X) and side (CF, CC, CR) when available or required. Biomass was log-transformed prior to analysis (to fit the normal distribution). Post-hoc Tukey tests were conducted when a significant effect was observed and significance was reported in the relevant figures. Growth variables (stems per plant, diameter and height) are analyzed by year, and not repeated measures ANOVA, because random plants were measured from one year to the next. These statistical analyses were conducted using JMP 10 (SAS Institute, Cary, NC).

Many environmental variables were used to interpret willow productivity (Annexe 8). To avoid overparameterization while maximizing the breadth of the multivariate analysis, these variables were used in two distinct and complementary statistical analyses: first, treated as groups of similar nature environmental variables; second, considering highly correlated individual variables independently. (1) For the first analysis, variables of similar nature were grouped in 7 matrices to conduct a principal component analysis (PCA) and extract the coordinates of the first principal axis, which corresponds to maximal variability within each group. The first principal component (PC1) of these 7 groups of environmental variables — Climate, GPS, Culture, Herbs, Hydrology, Soil and Water — were used in a Redundancy Analysis (RDA; Legendre and Legendre 2012) to explore environmental influences on willow growth (stem n, diameter, height; 2011-2013) and productivity (stem number, diameter, height, plant biomass and yield per hectare; 2013). Growth variables of both sites are considered together to appraise site specific influence on growth, while productivity variables are characterized by site to have a better understanding of physical, chemical or biological local heterogeneities (i.e. Annexes 13 and 14). The individual variables most strongly influencing the variability of the PC1 of each group of variables with a similar nature were identified . (2) For the second analysis, a shortlist of variables most strongly correlated with each response ($r \geq 0.50$), was used in a forward election RDA (500 Monte-Carlo permutations, including the 5 most influent variables) to assess their roles on growth and productivity. As a part of the RDA, co-linear variables are automatically excluded. PC1 analyses were conducted with JMP 10 and RDA with CANOCO v4.0 (Lepš and Šmilauer 2003).

1.3. Results

1.3.1 Willow growth

The number of stems per plant (Figure 1- 2) is not statistically different between SR and BB sites. The low and high density plantation treatments only seem to influence the number of stems per plant in the first growth year (2011). In 2013 at BB, plants in the central rows of the RBS (CC) had significantly fewer stems than CF and CR. At SR, CC plants are only distinct from CF plants. Contrary to diameter and height, which increase with time, the number of stems is two- to fourfold lower in 2013 compared to 2011 for BB.

While the collar diameter (Figure 1- 2) is somewhat similar in 2011, it becomes increasingly different at both sites as time progresses. Side of the RBS significantly affects plants throughout the whole period. An interaction between treatment and side, and site and side in 2012 leads to SR 5X plants having a lower growth in CF and CC but not in CR.

Stem height (Figure 1- 2) only differs between sites in 2012, though a non-significant trend for BB plants to be taller than SR plants was visible in 2011 ($p = 0.0967$). In 2011, we could not test for the effect of side due to the fact that several plants had only one or two stems.

During willow growth, height and stem diameter increase while stem number decreases, explaining why vectors are diametrically opposed in the RDA (Figure 1- 5a,b). Stem number seems only slightly correlated with environmental variables, both considered as groups (panel a) and individual variables (b). The grouped water variables are most strongly correlated to stem diameter, while height is more closely correlated with the two groups of soil and culture variables (b). Hydrology and GPS appear diametrically opposed to height, while grouped herbs variables are diametrically opposed to diameter, but in a weaker fashion. As for the analysis on the most highly correlated individual variables, it was observed that increasing soil organic matter content increases growth (height and diameter), while increasing annual precipitation or drainage basin reaching the stream across the RBS correlate to reduced growth (Figure 1- 5b).

1.3.2 Willow productivity

Individual plant weights varied from 5.0 to 4.8 kg dw·plant^{-1} at BB, and 2.1 to 1.3 kg dw·plant^{-1} at SR, in the 3X and 5X treatments respectively (Table 1- 1). The total biomass harvested after the three year growth cycle varied from 70 to 268 t dw·ha^{-1} depending on site and treatment (Table 1- 1). Willow biomass (per plant and per hectare; Figure 1- 3) is significantly affected by site and the position of the plants in the RBS. Biomass production is lowest in the CC rows, but the CR row is not statistically distinct from the CC row in SR. When the ANOVAS are constructed with sites separated, treatment significantly affected biomass per hectare in BB ($p = 0.0349^*$).

A regression with willow growth variables shows that stem diameter is the best individual predictor of willow biomass productivity (t dw·ha^{-1};
Figure 1- 4), though the model fit was improved by 15-20% when the three growth variables were used in a multiple regression (Table 1- 2).

To explore potential environmental determinants of willow productivity, we constructed RDAs on each site to maximize within-site discrimination among the different environmental variables matrices, and individual variables. In BB, 2013 productivity was strongly correlated to stem number, contrary to inter-annual fluctuations during the growth period, however stem height was little correlated to all other productivity variables (Figure 1- 5c). Within each site for 2013 and according to the RDA, the grouped culture and climate variables have no influence on productivity (refer to section 1.2.7 for definition of groups), which is an inherent consequence of the absence of variability within sites for these variables. Willow height is diametrically opposed to the groups of soil or GPS variables — and water variables to a lesser extent. Grouped herbs variables (whose most influential PC1 components are the sum of herbaceous vegetation ground cover and the sum of weed ground cover) are positively correlated to the other productivity variables. Among BB individual variables (Figure 1- 5d), the fraction of the RBS covered by herbs or weeds is positively correlated to height, while Al_{aq} is negatively correlated. Introduced species and Shannon diversity are positively correlated with the other productivity variables. In SR (Figure 1- 5e), individual plant biomass, RBS yield and stem diameter are strongly correlated but show little variability. Height and stem number are little correlated with other productivity variables. As in BB, grouped water variables are diametrically opposed to height, but contrary to BB, the soil and GPS groups of variables are negatively correlated to all productivity variables (other than stem number). Among BB individual variables (Figure 1- 5f), shade tolerance of herbaceous vegetation is negatively correlated to diameter, plant biomass and RBS yield, but willow height does not appear to influence this herbaceous plant ecological niche. Height is antagonistically opposed to soil moisture, drainage basin reaching the stream across the RBS, latitude and less strongly to introduced species soil cover.

1.4. Discussion

1.4.1 Productivity and Bioenergetic yields

After a three-year growth cycle, we report annual willow yields equivalent to 56-89 t dw·ha^{-1} in BB and 24-23 t·ha^{-1} in SR for low and high density plantations respectively. This is 2 to 9 times superior to biomass productivity in commercial willow plantations under similar conditions, which yield 10-12 t dw·ha^{-1}·yr^{-1}, with some lesser productive fields reaching only 2-6 t dw·ha^{-1}·yr^{-1} (Keoleian and Volk 2005). Applied research in Sweden, United States of America and United Kingdom reported willow yields of 24-34 t dw·ha^{-1}·yr^{-1} in short rotation plantations with double-rows and 10 000-20 000 plants·ha^{-1} densities (Adegbidi et al. 2003; Labrecque and Teodorescu 2003). However, such a direct RBS vs. field comparison is flawed. When converting the surface area of a 3m wide linear RBS into hectares, we are overestimating the plantation density that could be attained in a real field due to the edge effect. The yields per hectare we report are indeed accurate for a given surface area, but only for 3 m wide linear RBS. The experimental RBS plantation densities were equivalent to 33 333 and 55 556 stems per ha which is three to five times superior to densities conventionally used in commercial plantations in Europe and North America (Adegbidi et al. 2003; Labrecque and Teodorescu 2003). This is because in a field, edge rows cannot simply be juxtaposed. Hence, if 33.3 plants are found in a 10 m long row with one stump every 30 cm, a 90 m^2 field (i.e. 9 m x 10 m) with 1.5 m between the rows would contain 22 222 plants·ha^{-1}, whereas 90 m^2 of 3-m-wide RBS (i.e. 3 stretches of 30 m long) with still 1.5 m between the rows would contain 33 333 plants·ha^{-1}. Similarly, with 0.75 m spacing between the rows, the field calculation would total 44 444 plants·ha^{-1}, whereas the RBS would contain 55 556 plants·ha^{-1}. Gasser et al. (2013) estimated that their willow RBS (0.30 m of space on the row, with 1.83 m space between rows) was equivalent to 18 200 stumps·ha^{-1} short rotation coppicing systems, which seems more akin to field density calculations. However, the reported RBS yields are much lower than in the current study: 3.6 t MS·ha^{-1}·yr^{-1} (after 2 years). A second limitative argument in our field vs. RBS comparison is the fact that on a small scale experimental RBS plantation, meticulous manual harvest when sampling was performed is not comparable to the typical 5-10 % (Graham et al. 1992) or 6-20 % (Vézina et al. 2013; Hébert 2012) harvest losses in commercial plantations. With these in-field equivalence and harvest loss estimates, the corrected SR yields (~14-16 t dw·ha^{-1}) approach typical averages, while BB yields (~33-64 t dw·ha^{-1}) remain quite elevated (Table 1- 1). The remaining difference could then be attributed to the intrinsic conditions within the RBS.

1.4.2 Edge-effect

In a crop field, edges may infer smaller yields (reviewed in Barbour et al. (2007)) and in a natural forest stand the edges too may suffer from a deleterial edge-effect (Saunders et al. 1991). Contrary to fields or forests which have more restricted marginal areas, the RBS linearity could lead to a beneficial rather than a detrimental edge-effect. RBS edge plants produce more biomass possibly due to better sun exposure, as evidenced in BB where central row plants are smaller (Figure 1- 5). In SR, the proportion of shade-tolerant understory herbs is inversely proportional to willow yield (Figure 1- 5). Canopy openness of Salicaceae-planted RBS is known to influence the understory species richness and plant cover, with shade-intolerant plants being excluded as canopy openness decreases (Fortier et al. 2011). Indeed, it has long been known that small plantations are hardly representative of larger scale plantations as they introduce a strong bias with increased stem diameter and biomass productivity of edge rows (Zavitkovski 1981).

Poplar (*Salicaceae*) RBS bordering corn and soy fields may have 20% enhanced annual biomass production (Tufekcioglu et al. 2003) compared to field plantations (Zavitkovski 1981). In poplars, productivity is inversely proportional to the number of rows in a plantation (85.9 to 11.4 t/ha/yr for 1 to 8 row plantations (Zavitkovski 1981). However, if shade was the only driver of productivity reduction with an increasing number of rows, increased biomass production or enhanced growth variables may not only be restricted to edge rows, but could also be visible in the southernmost rows, receiving more sunlight. Indeed, latitude is identified as a predominant factor in SR (northernmost plants produce less). However, both north and south rows (*vs.* central row position) are significantly related to stem diameter, and individual plant biomass yield of both north and south rows are equally enhanced (Annexe 15 et 16), suggesting that water and nutrient limitations, and not solely solar exposure, may also have played a role in this edge-effect. In fact, Gasser et al. (2013) has also observed that interactions with water and nutrients influence willow growth. They studied the effect of swales (depressions parallel to the rows meant to promote ponding and infiltration) in willow RBS. They report that in some treatments (*Salix* without swales) position of the row with respect to the field influenced biomass productivity: there was a reduction of biomass yield from the field to the distal row which could be explained by decreased availability of nutrients as distance from the field source increased due to sequestration by the vegetation. In treatments with five swales, productivity was not affected by row position. However, in other treatments (*Salix* with one swale), the CR yields are superior to the CF yields, probably because the swale design allowed the last rows to better benefit from the nutrients.

Finally, the RBS willows were likely exposed to spray-drift (foliage exposure during application) or leaching (with subsequent root absorption) from nearby applications of herbicides, and particularly glyphosate, applied to the studied fields. Our results on productivity and growth do not suggest obvious side-effects of glyphosate exposure (i.e. no reduced growth on the edge-of-field where glyphosate exposure would originate, and absence of glyphosate effect short-listed among the most significant variables in the cultural group in Annexe 16, and non-significant correlation between glyphosate applications and willow height, or marginal correlation between glyphosate in runoff and willow diameter, but not retained as the most determinant variables in the multivariate analysis). Gomes et al. 2015a,b evidenced that *Salix miyabeana* can effectively absorb glyphosate from the soil (and hence be used as a phytoremediation agent) but that it may also be affected by glyphosate upon shoot or root exposure. Effects on chlorophyll metabolism, photosynthesis and reactive oxygen species visible in greenhouse assays using environmentally relevant doses may only be evidenced through biochemical analyses, and not obvious through the macroscopic measurements we conducted in the field. Furthermore, glyphosate is recommended for eradication of weeds between willow plants, and direct injection in stems or stumps may be required to kill willows in a time-frame that can last up to two years (according to pesticide labels, IPCO 2008). Hence, sub-lethal physiological impacts of glyphosate on the willow RBS may have gone unnoticed in the current study.

1.4.3 Hydrology

During the growth period, the grouped hydrological variables were negatively correlated with stem diameter and height (Figure 1- 5a). The suggestion that a high water table in BB was beneficial for willow growth is elsewhere evidenced with the significant impact of water table depth on growth (Annexe 16), an observation further supported by the positive correlation between biomass production or soil moisture and willow height (Annexe 17). Amongst the most influential variables on the hydrological PC1 axis were "head", the height difference between edge-of-field and edge-of stream, and water table elevation, the latter of which should be viewed here as a geographic variable since BB in the low plain areas presents a lower water table altitude compared to SR in the high plain. Height and diameter were negatively correlated to precipitation (see Figure 1- 1 for annual precipitations. Weekly precipitation for the duration of the study is presented in Annex 26), and micro-basin surface area ("stream") in SR has significantly larger basin (which varied from 3 to 83 m^2 in BB compared to 46 to 1725 m^2 in SR, Table 2-2; Annexe 4), and this may also be a distinguishing factor between sites (and perhaps years).

Within sites, just for the final 2013 productivity variables, no hydrologic variables were selected in the forward-selection RDA as major drivers of BB productivity, suggesting that on a micro-geographic scale hydrology was not a major driver or limitation to willow productivity. Nevertheless, willow height was reduced in the field areas where the water table absolute elevation was highest while the diversity of hydrophilic plants was positively correlated with stem number (Annexe 18). In SR, understanding the global effect of hydrology on willow productivity is more complex. Soil moisture and micro-basins (calculated from runoff flow reaching the stream) did not have much influence on biomass productivity or plant diameter, but were positively correlated with stem number and inversely proportional to stem height. However, an alternate measurement of micro-basin area (calculated from drainage points to streams) were negatively correlated to stem diameter (Annexe 18). This suggests that increased water availability leads to distinguishable growth morphologies, i.e. more available water is correlated to shorter, thinner and more branched plants and vice-versa. Counterintuitively, a shallower water table depth was itself positively correlated with taller plants, suggesting that some underground water availability is important, but that waterlogging in the surface (i.e. soil moisture) could limit growth (Annexe 18). Hydrophytic plants soil cover or proportion is positively linked to SR willow final height, while RBS areas more strongly colonized by upland plants have shorter willow. Plantation of trees and shrubs may help recreate some desired attributes of the riparian ecotone, but historical and ongoing agricultural activities will remain influent on soil and water hydrogeochemistry, which then affect plant biodiversity and spatial patterns (Flinn and Marks 2007; Vellend et al. 2007; Vidon et al. 2010).

1.4.4 Nutrients in runoff water

Nutrient concentrations in surface runoff water were positively correlated with stem diameter and height during willow growth (in Figure 1- 5, panel a, the Water vector is most strongly influenced by PO_4^{3-} and NH_4^+ (see details in Table 1- 2); in panel b, Ca and Mg are the most strongly correlated individual parameters), again suggesting a strong differentiation between sites as nutrient concentrations in runoff waters were also more elevated in BB. Surface runoff was considered representative of site fertility (groundwater and soil nutrients were not included in this analysis, as they were not available for every sampling station and time points).

No major aqueous nutrient concentrations were identified in the RDA amongst the most important individual variables associated to willow growth on both sites nor final productivity at either site (Figure 1- 5). The apparent small correlation between Ca and Mg fertilization and stem numbers may bear no real biological explanation and be an artifact stemming from the sole supplements on these elements in 2011 in BB, the year where the number of stems of recently coppiced plants was highest. However, increased P fertilization in corn and soy fields is associated with an increase in willow height (Annexe 17). Finally, elements like Mn^{2+} (r = 0.37; diameter), NH_4^+ (r = 0.41 with height; r = 0.60 with diameter) and Zn^{2+} (r = 0.05 with height) all showed positive correlations to certain growth variables (Annexe 17). But for final productivity variables at each site, Zn^{2+} concentrations were positively associated with stem number in SR. Several willow clones can hyperaccumulate Zn in their roots or shoots (Utmazian et al. 2007). Zn may induce metabolic changes in plants (i.e. oxidative stress and photosynthesis inhibition (Tsonev and Lidon 2012)) including willows (Landberg and Greger 2002), but despite decreased biomass production, no other visible signs or phytotoxicity or gross morphometric changes have been reported (Wieshammer et al. 2007). Hence, we could not find a direct explanation for the positive correlation between stem number and aqueous Zn^{2+} concentrations in the literature, though morphological features (i.e. leaf number, area and biomass) which affect evapotranspiration have been associated to metal uptake before (Mills et al. 2000). While Zn^{2+} availability in the soil is largely controlled by the soil pH, with soils of pH above 6.5 (BB = 6.6 and SR = 6.6-7.1) potentially leading to Zn deficiency in plants (Muhammad et al. 2012). Also, Al^{3+} concentrations were negatively associated to plant height in BB. Gobran et al. (1993) reported that Al^{3+} decreases willow growth in the field, under naturally occurring soil solution, as well as in the laboratory with rooted cuttings, and this may explain our RBS observations.

Enhanced nutrient concentrations running off from fields have previously been associated with enhanced *Salix* growth on the edge-of-field RBS rows (Gasser et al. 2013). SR field was not only fertilized with mineral fertilizers, but also with organic amendments like sewage sludge and pig slurry (See Chapter 2). The significant biomass productivity increase of willows has previously been evidenced in fertilization experiments using wastewater (Guidi Nissim et al. 2015), sludge (Rockwood et al. 2004; Labrecque et al. 1997) or pig slurry (Cavanagh et al. 2011).

1.4.5 Competition with understory herbaceous plants

During growth, the herbaceous vegetation ecological characteristics (strongly driven by sum of weed cover and Shannon diversity on the PC1 axis of the corresponding environmental matrix), negatively influenced diameter and height of willows. In fact, at BB, four ecological herbaceous plant characteristics appear in the top 5 influential factors in the RDA. Weed soil coverage only weakly influences final willow productivity at BB, even though more herbaceous plant coverage and weeds are strongly associated with taller willow plants. This contradicts the assumption that competing plants reduce willow productivity (Guidi et al. 2013). Shannon diversity or the number of introduced plants all positively correlated to other productivity variables. At SR, herbaceous plants ecological characteristics also occupy 2 of the top 5 most influential variables affecting willow productivity, but in that case, introduced plants coverage is negatively correlated to all productivity variables (except stem number; Figure 1-4). There was also a negative correlation between productivity and shade tolerant plant soil proportion discussed above as a consequence of the edge-effect. Actually, light penetration, rather than tree species, is a more important determinant of understory diversity (Fortier et al. 2011) and the different architecture of other Salicaceae (various poplar hybrid clones) was shown to affect light availability which influenced biomass and biodiversity of understory vegetation (Fortier et al. 2011).

The presence of weeds is generally considered as competition detrimental to willow productivity (Albertsson 2012), as it is in Québec (Vézina et al. 2013; Labrecque et al. 1994). However, herbaceous vegetation is sometimes intentionally grown (i.e. *Lolium multiflorum*) in order to stabilize the soil between rows (Gasser et al. 2013). Such systems of narrow strips of trees and shrubs provide enough light to the ground vegetation that grasses and herbs are able to grow providing excellent soil particle trapping (Schultz et al. 1995). However, we were expecting more shade tolerant plants under the willows due to light limitation but found that shade tolerance was negatively correlated to productivity. In riparian vegetation, understory herbs may account for less than 3% of total evapotranspiration rates (Tabacchi et al. 2000), suggesting that competition with woody plants for water might be minimal. Furthermore, the high herbaceous diversity correlations with high productivity (witnessed at BB) has been explained by others through the temporal stabilization of ecosystem functions emphasizing that a better characterization of homogeneous woody communities (i.e. poplar plantations or coppiced willows) with more diverse natural riparian stands could help to understand the RBS plant communities' ability to retain runoff or favor infiltration (Tabacchi et al. 2000).

The main conclusions from the multivariate analyses presented above are three-fold: (1) Hydrology appears as an important determinant of willow growth, but final productivity and soil moisture, water table depth, drainage basin sizes or the presence of hydrophytic plants might interact differently within each site. Hence, RBS implemented where water is abundant on good draining soils is best (though waterlogging may be detrimental on compacted sites). (2) Nutrient concentration in runoff appears as another determinant of willow growth, which mean that nutrient availability heterogeneity should be considered in the maintenance of RBS. (3) Willow productivity may not always be influenced by weed coverage, and the RBS nutrient capture potential is positively correlated with enhanced biodiversity. This means that farmers may not need to maintain monospecific stands of biomass crops in their RBS as biodiversity doesn't necessarily result in lower productivity or recruitment of weeds in the fields.

1.4.6 Potential of the willow RBS to sequester nutrients

Riparian ecosystems can serve as both a short- and long-term nutrient filters and sinks pending periodical harvest of trees to ensure a net uptake of nutrients (Lowrance et al. 1984). Uptake of nitrogen by non-harvested herbs and deciduous tree leaves may contribute only to short term removal, as remineralization occurs within a few months to a few years (Hefting et al. 2005). Litterfall contributes to annual recycling of nutrients and may lead to some export in the dormant period with approximately 26% of N and 38% of P retained by vegetation and exempt of this annual cycling process (Blackwell et al. 2009). In wetland systems, nutrients may become incorporated into peat (2.5% N and 0.2% P) though accumulation rates are relatively low (Blackwell et al. 2009). Unless *luxury uptake* occurs (plant absorption of nutrients in excess of their essential growth requirements), the nutrient concentration of wood is generally low (Blackwell et al. 2009). Nitrogen concentrations in willow stems varies slightly between different short-rotation field plantations (France : 2.6-6.3 g N·kg^{-1} (Toillon et al. 2013); New-York state : 3.7-9.6 g N·kg^{-1} (Adegbidi et al. 2001); Québec : 3.7-5.0 g N·kg^{-1} (Cavanagh et al. 2011) or 5.3-7.3 g N·kg^{-1} (Labrecque and Teodorescu 2003) or RBS systems (Québec: ≥9 g N·kg^{-1} (Gasser et al. 2013). A conservative N export rate of 5 g N·kg^{-1} was retained for further calculations. P concentrations in *Salix miyabeana* SX64 stems in RBS vary from 1.1-0.9 g-K·kg dw^{-1} (Gasser et al. 2013) while P concentrations in other *Salix* clones planted in fields are lower (0.5-0.7 g·kg dw^{-1}; Adegbidi et al. 2001). The most conservative estimate of 0.5 g·kg dw^{-1} was used in nutrient sequestration calculations. *Salix miyabeana* SX64 K concentrations vary from 2.6-2.7 g-K·kg dw^{-1} (Gasser et al. 2013) but are not affected by *Salix* density in RBS, hence the mean value was retained for K sequestration estimates. We calculated that the harvest of our willow stems after three years of growth may have contributed to the sequestration of 116-118 kg-N·ha^{-1}·yr^{-1} at SR and 278-447 kg-N·ha^{-1}·yr^{-1} at BB. The N sequestration in these stems was considerable compared to those observed in *Salix miyabeana* SX64 in controlled RBS leaching-plots near Quebec city (70 kg N·ha^{-1}, (Gasser et al. 2013) but comparable to the field observations of Labrecque and Teodorescu (2003) with *Salix viminalis* in a clayey soil (150 kg-N·ha^{-1}·yr^{-1}). The calculated RBS sequestration rates for phosphorus is 23 kg-P·ha^{-1}·yr^{-1} in SR and 55-89 kg-P·ha^{-1}·yr^{-1} in BB, which is higher than the previously reported 10-11 kg-P·ha^{-1}·yr^{-1} (Adegbidi et al. 2001). Finally, the potential potassium sequestration represents 62-63 kg-K·ha^{-1} at SR and 148-239 kg-K·ha^{-1} at BB, these estimations being higher than the 20 kg-K·ha^{-1} reported by Gasser et al. (2013).

1.4.7 Limitations

During the course of the study, some factors affecting productivity were witnessed. For instance, during the first year of growth in BB, some plants in the driest parcels died. Furthermore, in April 2012, stems affected by *Janus abbreviatus* (Say) were cut off and removed from the plantation as a prophylactic measure to control the infestation. Low density plantations had more infected stems (84.3 dropped stems ± 43.5 SD between the triplicata) than high density plantations (38.3 ± 34.5). The number of infected stems appeared significantly influenced by both the proximity to the closest treeline of the nearby forest ($p = 0.0178^*$), and by the density of plantation ($p = 0.0297^*$) (General linearized model, interaction was non-significant). Furthermore, giant willow aphid, *Tuberolachnus salignus* (Gmelin) aggregations were noticed on most stems in 2013. In SR, insect infestation was not problematic, but part of a row of willow close to the field was damaged by agricultural machinery. These events led to the replacement of some plants, which could have led to global reductions in our estimated yields, despite the fact that the yields we report are higher than others reported in the literature. The current study was also limited by the need to use non-destructive willow growth variables, except in the last year of growth and harvest, because of the small size of the experimental parcels and parallel ongoing aqueous nutrient and glyphosate removal studies which required minimal disturbances to the system. Nevertheless, we developed a significant regression to help predict willow yields in RBS at the end of three-year growth cycles. A final limitation of this study is the scope of the conclusions that can be drawn from the intersites multivariate analyses. As only two sites were studied, this precludes generalizations outside of our system to the conclusions reached concerning the most influential environmental variables. However, the selected method is interesting to untangle how within site variability of environmental parameters influences the salix growth a productivity outcome. The adundant environmental descriptions was specifically laid here for future use in meta-analyses comparing growth and productivity across different sites.

1.4.8 Perspectives for farmers

The regression developped can be useful to farmers trying to estimate potential yields in order to decide whether they should harvest after three years or extend the harvest cycle by an extra year based on expected yields and market demand and pricing. Average field yields were given by the farmers. Grain yields in BB were 3.65, 10.5 and 3.75 $t \cdot ha^{-1}$ for 2011, 2012 and 2013 (under a soy, maize, soy rotation) respectively. Grain yields in SR were 2.76, 8.6 and 4.4 $t \cdot ha^{-1}$ for 2011, 2012, and 2013 (under a soy, maize, maize rotation), respectively. Some economic aspects were thus considered (see a tabular view in Annexe 19). Though grain price averaged 228 $\$\cdot t^{-1}$ for corn and 482 $\$\cdot t^{-1}$ for soy from 2009-2013, these values appear much higher than historic trends and future predictions (Sall et al. 2015). Hence, considering a conservative current market value of 195 $\$\cdot t^{-1}$ for corn and 410 $\$\cdot t^{-1}$ for soy (www.grainwiz.com, consulted online 2015-06-15), crop losses (opportunity cost) due to the protection of a RBS approached 3 700-5 082 $\$\cdot ha^{-1}$ for three years (or 1 200-1 700 $\$\cdot ha^{-1}\cdot yr^{-1}$) based on SR and BB crop cycling and yields. Potential revenue from willow sales, based on 80-120 $\$\cdot t^{-1}$ market value estimates (personal communications, Francis Allard, Agro-Énergie 2015), thus represent a potential revenue of 12 000-29 000 \$ at BB and 5 000-7 700 \$ at SR only accessible after a three-year period. This means that farmers lose grain revenues each year and would need to wait a full growth cycle before benefiting from potential willow sales revenues. A discount rate could also be applied to this evaluation to account for delayed revenues and to take into consideration the incertainty associated with longer term investments. However, considering harvest costs of 62 $\$\cdot t^{-1}$ for a long and thin RBS or 216 $\$\cdot t^{-1}$ for larger field plantations (Vézina et al. 2013), harvesting costs could range between 10 000-58 000 $\$\cdot ha^{-1}$ at BB and 4 000-15 000 $\$\cdot ha^{-1}$ at SR. Hence, profits from willow RBS harvest and sale range from -29 000 to 1 600 $\$\cdot ha^{-1}$ at BB and -7 700 to 700 $\$\cdot ha^{-1}$ at SR, excluding all costs associated with planting, maintenance or the benefits associated with increased bank stability (i.e. the cost of dredging in wateways can reach 25 000$\$\cdot km^{-1}$, and can be done as frequently as every 6-7 years when erosion retention is deficient; Paradis & Biron 2016, Gravel 2012). Because marginal profits may only be attainable when harvesting costs are minimal and market value is maximal, this stresses the need to correctly assess the perfect harvest time for farmers. But waiting to have sufficient biomass production before harvest must also be outweighed against the potential detrimental effect of taller willows casting shade over edge-of-field crops, larger branches hampering mechanical harvest and leading to greater harvesting losses, or branches falling in the field potentially damaging agricultural machinery. Earlier studies demonstrated that *Populus* or willow RBS may lead to a net economic burden for farmers (Simard 2009). Furthermore, costs may not be the sole challenges to address, considering that harvesting small tonnages on field margins may require farmers to cooperate for the acquisition and operation of the appropriate machinery, or production of sufficient volumes to increase sale value. On the other hand, including externalities (or environmental goods and services) such as the water filtration potential, the RBS may be profitable for society as a whole (Simard 2009).

Several environmental goods and services associated with agroforestry could also be associated with willow RBS: Preservation of soil physical and biochemical structure; preservation of water physical and biochemical quality, equilibration of hydric regimes; preservation of bank stability with associate decrease in dredging work recurrence; control of air quality by reducing pesticide spray drift; climatic control through greenhouse gas sequestration and favorable micro-climates; sustenance of soil, wetlands or aquatic biological diversity; improving pollination or biological insect control; controlling invasive or exotic species while preserving habitats for vulnerable or threatened species; and finally social values such as landscape preservation or agronomic values such as the creation of windbreaks (Simard 2009; Marchand and Masse 2008). Biological diversity, including the control of exotic weeds and preservation of indigenous species, is a topic of economic importance which should be addressed further in RBS settings (Fortier et al. 2011, Annexes 12 and 13 list exotic weeds and indigenous plants encountered in our experimental parcels). The question of incorporating externalities for private farms in a RBS economic evaluations needs to be addressed and eco-conditioned subsidies may not motivate farmers as much as a strict and uniform application of municipal regulations created on the basis of provincial policy (Dagenais 2015).

1.5. Conclusion

The current study reports the highest biomass productivity ever measured for *Salix miyabeana* SX64, which could represent a strong commercial interest for farmers. Indeed, the record breaking 56-89 t dw stems·ha^{-1}·year^{-1} in a humisol at BB may motivate farmers who have organic rich soils in depressions at the edge of their fields to plant this species in their RBS. Farmers, which have compacted sandy loam soil such as SR, may also be interested in the good biomass productivity (23-24 t dw stems·ha^{-1}·year^{-1} yield), which is still close to the highest producing averages for in-field plantations. *Salix* grown in buffer strips seem to be quite productive, even in narrow strips, because they benefit from increased light availability as well as water and nutrient runoff. Cultivating *Salix miyabeana* SX64 is a practice, which could improve both nutrient sequestration potential and productivity of currently uncultured 3 m RBS prescribed under the Quebec policy on riparian areas in agricultural regions (PRLPPI). Stem diameter was the best predictor of *Salix miyabeana* SX64 productivity based on a site specific regression. This model will be useful to estimate biomass yield potential in buffer strips, as we have suggested that field-derived equations would not represent buffer strip growth conditions and yields. The buffer strips' ability to retain nutrient leaching from fields may lead to differential growth of shrubs within the buffer strip, with taller plants facing the fields, only when nutrients are limiting (SR, not BB). Intraspecific competition is displayed on individual plant sizes (kg·plant^{-1}) but when nutrients are not limiting, this differences is not apparent on yields per surface area (t·ha^{-1}). However, though the understory herbaceous vegetation ecological characteristics influenced willow productivity, interspecific competition did not seem to hamper productivity in sites where water and nutrients were not limiting, though introduced species ground cover antagonized productivity in our least fertile site, perhaps due to competition for available soil moisture or nutrients. In the light of the present research, we conclude that interesting biomass yields may be produced in RBS, but we cannot confirm the actual interest of farmers, which depends on harvesting or transformation opportunities and local markets conditions.

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1.7. Tables and Figures

Table 1- 1: Mean *Salix miyabeana* SX64 productivity for the 2011-2013 rotation for low and high density riparian buffer strips and on two sites.

Site	Treatment		n	Biomass per plant ($kg \cdot plant^{-1}$)		RBS Biomass yield ($t\ dw \cdot ha^{-1}$)		RBS Biomass yield ($t\ dw \cdot ha^{-1} \cdot yr^{-1}$)		Estimated yield converted to equally spaced field plantations		Corrected in field yield equivalent accounting 10% harvest losses
	Rows	Plants·ha^{-1}		Mean	SD	Mean	SD	Mean	SD	Density ($plant \cdot ha^{-1}$)	Yield ($t\ dw \cdot ha^{-1} \cdot yr^{-1}$)	Yield ($t\ dw \cdot ha^{-1} \cdot yr^{-1}$)
BB	3X	33 333	45	5.0	2.5	167	85	56	28	22 222	37	33
	5X	55 556	45	4.8	3.0	268	168	89	56	44 444	71	64
SR	3X	33 333	45	2.1	1.7	71	56	24	19	22 222	16	14
	5X	55 556	43	1.3	1.1	70	61	23	20	44 444	18	16

Table 1- 2: Regressions predicting *Salix miyabeana* SX64 biomass (t dw·ha^{-1}) based on non-destructive growth variables.

	Boisbriand				**Saint-Roch-de-l'Achigan**			
Variable	n	r^2	p	Regression Log t dw· ha^{-1} =	n	r^2	p	Regression Log t dw· ha^{-1} =
Stem number (n)	90	0.36	<0.0001*	$1.91+0.07 \cdot n$	88	0.09	0.0056*	$1.53+0.4 \cdot n$
Diameter (d)	90	0.55	<0.0001*	$1.22+2.2 \cdot 10^{-2} \cdot d$	88	0.57	<0.0001*	$0.99+2.5 \cdot 10^{-2} \cdot d$
Height (h)	90	0.25	<0.0001*	$1.26+1.9 \cdot 10^{-2} \cdot h$	88	0.24	<0.0001*	$1.09+1.6 \cdot 10^{-2} \cdot h$
Stepwise (AICc min)	88	0.66	<0.0001*	$1.35+0.09 \cdot n+4.3 \cdot 10^{-3} \cdot d+2.1 \cdot 10^{-3} \cdot h$	88	0.66	<0.0001*	$1.35+0.10 \cdot n+4.3 \cdot 10^{-3} \cdot d+2.2 \cdot 10^{-3} \cdot h$

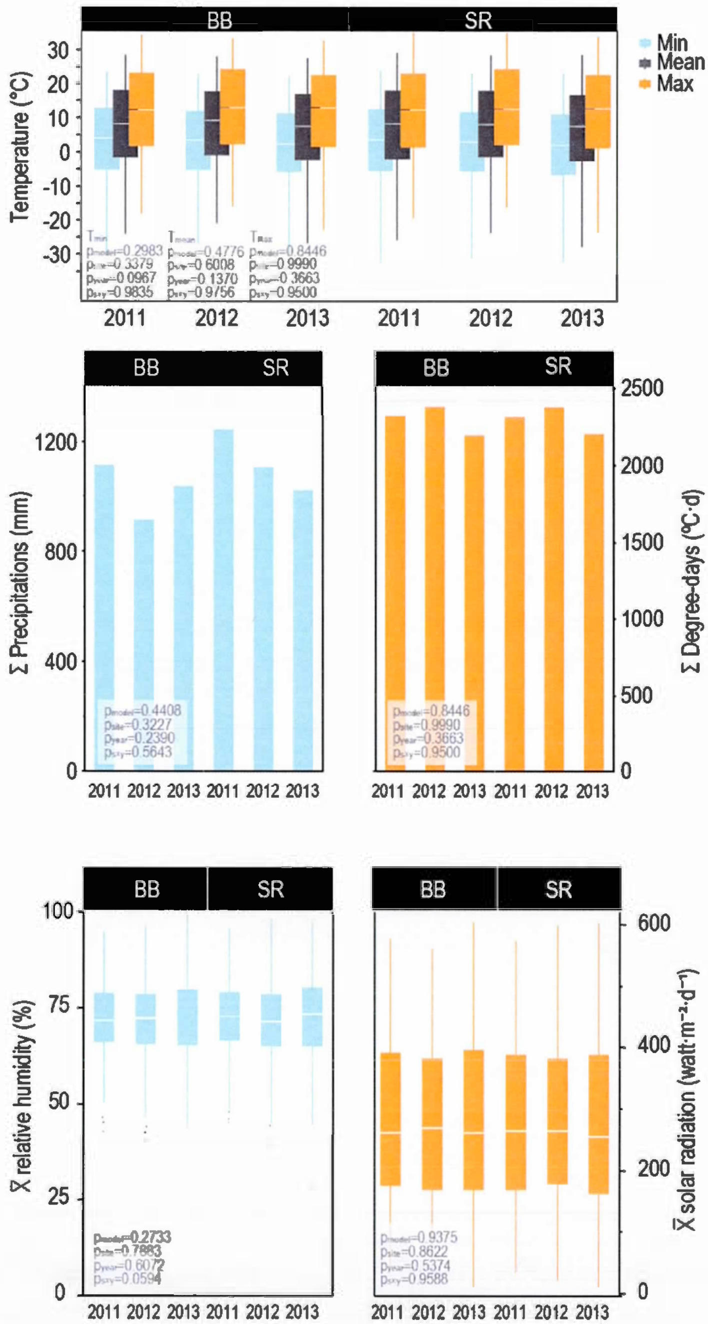

Figure 1- 1: Climatic characteristics of the Boisbriand (BB) and Saint-Roch-de-l'Achigan (SR) sites during the willow growth period.

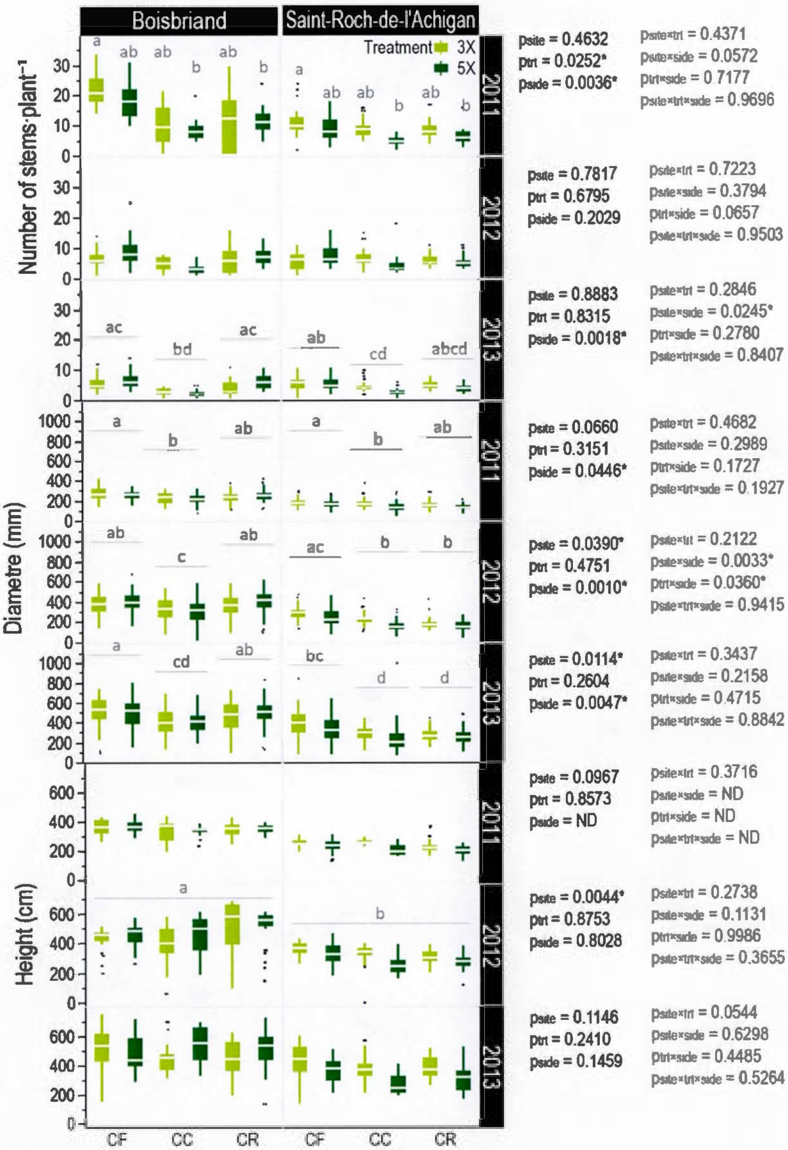

Figure 1- 2: Growth variables (number of stems per plant, diameter and height) of *Salix miyabeana* SX64 in riparian buffer strips.

Data is from 2011 to 2013, on two sites, under two density treatments and with respect to side (CF: Edge-of-field, CC: center of buffer and CR: close to stream). Random block ANOVA treatments are presented on the figure and statistically different groups are represented by different letters.

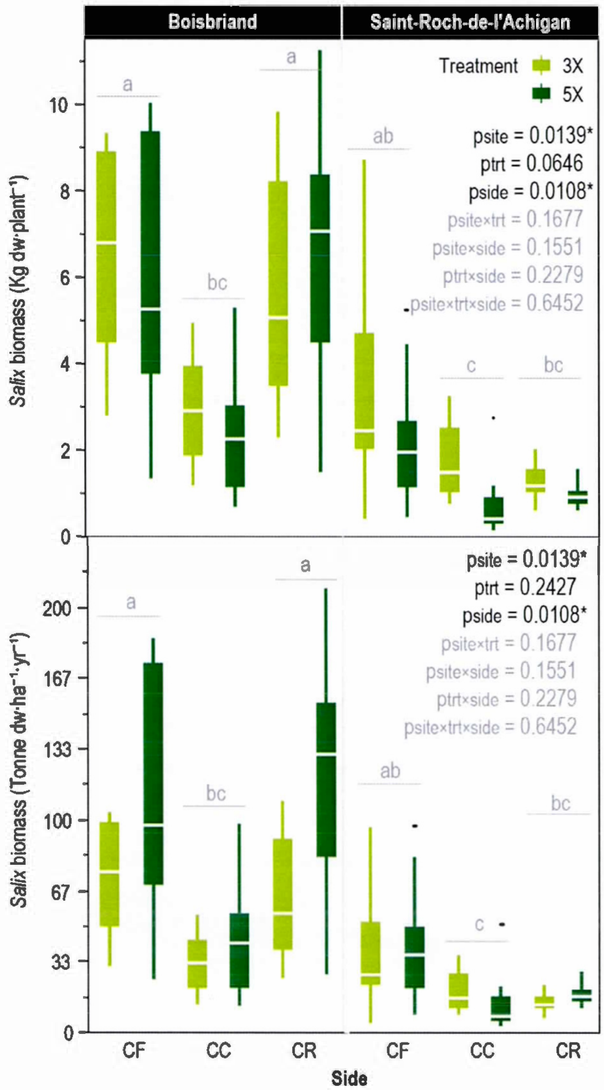

Figure 1- 3: Biomass productivity (per plant and per hectare) of *Salix miyabeana* SX64 in riparian buffer strips.

Data is from 2013, on two sites, under two density treatments and with respect to side (CF: Edge-of-field, CC: center of buffer and CR: close to stream). Random block ANOVA treatments are presented on the figure and statistically different groups are represented by different letters.

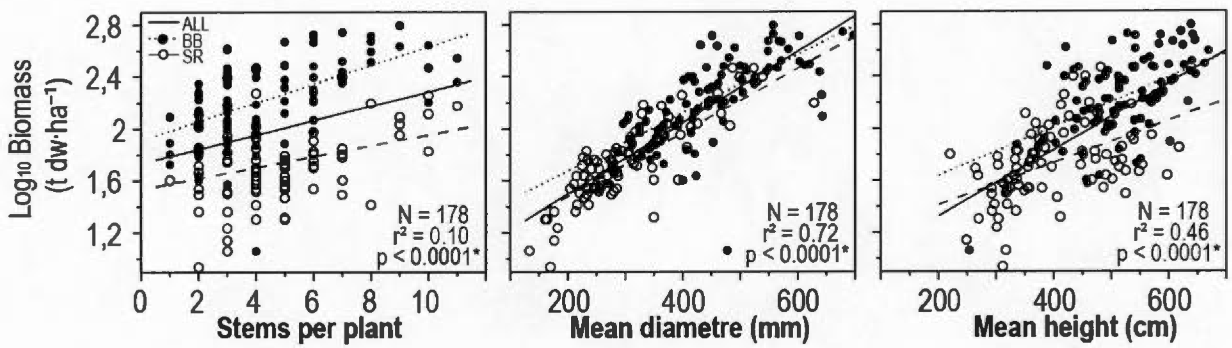

Figure 1- 4: Biomass productivity with respect to 2013 growth variables (number of stems per plant, mean diameter and mean height).

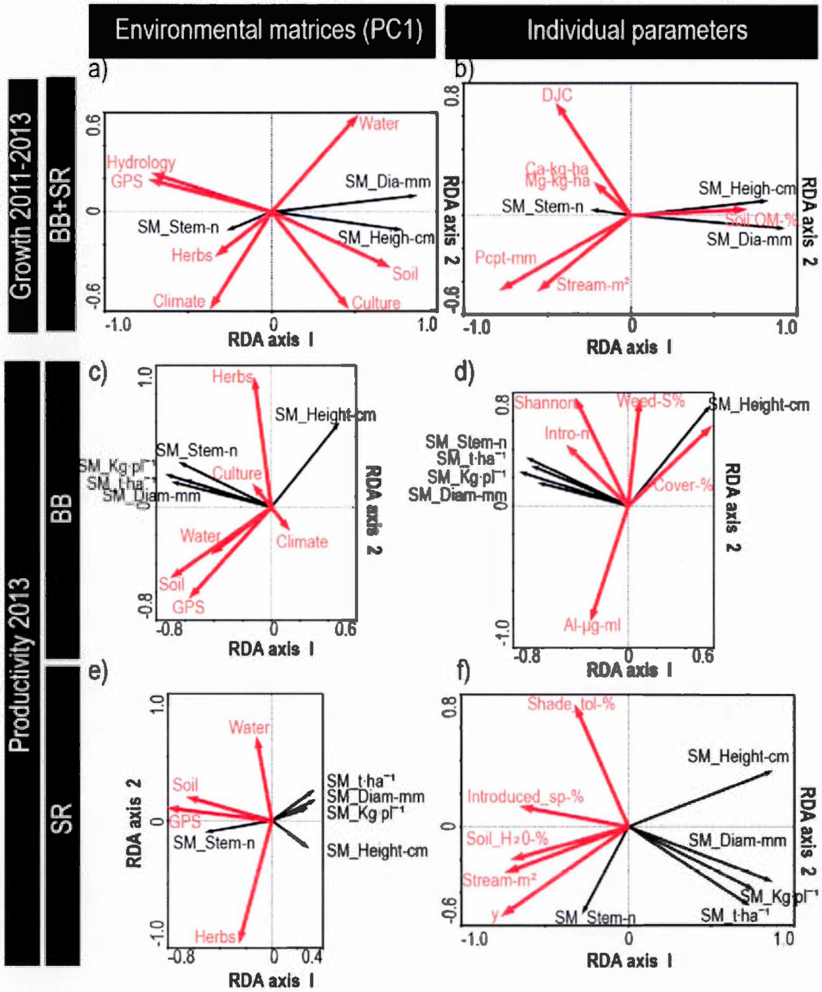

Figure 1- 5: Redundancy Analysis (RDA) on willow growth variables and productivity.

Growth (a-b) includes data from both sites over 2011 to 2013, while productivity (c-f) includes only data from the harvest year (2013) and analysis by site (BB in c,d and SR in e,f) was preferred to distinguish local influences on final productivity variables, as biomass productivity is statistically distinct between sites. Dimension reduction was achieved by extracting the first principal component of environmental matrices (Climate, GPS, Hydrology, Culture, Herbs, Water and Soil) for use in the RDA (a, c, e). Note: In RDA biplots, correlations between Salix variables and environmental variables are equal to the cosine of the angle between two arrows (i.e. vectors). Hence, similar vector orientations reveal positive correlations, opposite vectors depict negative correlations, and perpendicular vectors suggest little correlation. Arrow length represents the strength of these relationships.

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CHAPITRE II

POTENTIAL EFFICIENCY OF GRASSY OR SHRUB WILLOW BUFFER STRIPS AGAINST NUTRIENTS RUNOFF FROM SOY AND CORN FIELDS

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Abstract

The province of Quebec (Canada) policy for protecting shores, coasts and flood plains promotes 3-m wide riparian buffer strips (RBS). Herbaceous RBS and RBS planted with *Salix miyabeana* SX64 at two densities in a randomized block design with triplicates of each treatment were monitored to study nutrient retention (nitrate, ammonium, phosphorus and potassium) in runoff, interstitial and phreatic waters. Two study sites characterized by sandy loam (Saint-Roch-de-l'Achigan; SR) and organic-rich (Boisbriand; BB) soils were sampled 16 and 14 times, respectively, over three consecutive growing seasons (2011-2013). Sampling campaigns followed important agricultural events: (1) snowmelt or ≥ 15 mm natural precipitation events after (2) fertilization and (3) glyphosate-based herbicide applications. The potential efficiency of the buffer strip (expressed as the percent difference in concentration change before and after the RBS). On the edge-of-the-field, waters during post-fertilization had the highest nitrate concentrations. This period also coincided with the highest potential efficiency of the buffer strip to dissipate nitrate (77-81% in runoff at BB, 92-98% at 35-70 cm depth at SR). Ammonium concentrations in surface runoff were significantly lower on the stream side of the RBS compared to the field side at snowmelt at BB, but it increased during its passage across the RBS at SR. Total phosphorus concentrations were significantly lower on the stream side of the RBS compared to the field side post-fertilization at SR, but dissolved phosphate concentrations were never statistically reduced. Potassium concentrations were significantly reduced after the buffer strip at different moments and depths at BB. The potential efficiency of willow RBS to remove nutrients could not be distinguished from the herbaceous RBS based on intercepted aqueous nutrient concentrations. After the 3-m buffer strip, aqueous nutrient concentrations were generally below Quebec's aquatic life protection standard for nitrate (10 $mg \cdot L^{-1}$), but often above those for phosphorus (30 $ug \cdot L^{-1}$) and ammonium (1.5 $mg \cdot L^{-1}$) suggesting that these narrow RBS, if uncoupled to fertilizer input reductions, were insufficient to protect streams from excess nutrients in corn and soy agricultural regions.

2.1 Introduction

Agriculture-derived diffuse pollution composed of nutrients, pesticides or eroded soil particles is a leading cause of water quality degradation worldwide (EPA 2003; Ongley 1997; Ongley et al. 2010) and eutrophication driving hypoxia threatens tourism, fisheries and ecosystems (Diaz 2001). Nutrient excess originating from agriculture is responsible for the impairment of 48% of US rivers by length (EPA 2003). Several jurisdictions encourage the use of vegetated riparian buffer strips (RBS) along shorelines to mitigate non-point source pollution (Hickey and Doran 2004; Smethurst et al. 2009). It has been shown that "edge-of-field" (Dabney et al. 2006) RBS can mitigate nutrient erosion and leaching by retaining or transforming organic bound, adsorbed or dissolved nutrients carried by rain and snowmelt (Gagnon and Gangbazo 2007; Naiman and Decamps 1997). Within the RBS, eroded soil particles can be deposited, dissolved nutrient infiltration can be enhanced and together with higher soil organic matter content, this can further favor sorption, plant absorption, or microbial transformation (Locke et al. 2006; Dabney et al. 2006; Osborne and Kovacic 1993; Staddon et al. 2001). Buffer strip efficiency to minimize nutrients export to nearby streams with runoff or interstitial waters, is a term used indiscriminately of the processes involved (i.e. nutrients adsorption to soil, bacterial degradation and dilution with rain water).

In Quebec (Canada), the Policy for protecting shores, coasts and flood plains (PPRLPI) recommends that farmers maintain a 3-m-wide vegetated RBS along streams in agricultural zones (MDDEP 2005). However, several authors correlate increased nutrient reduction with increasing RBS width (Mayer et al. 2006; Vought et al. 1994), sometimes suggesting that much greater widths are necessary for nutrient reduction (≥30 m, Hickey and Doran 2004; or ≥60 m for long-term efficiency, Wenger 1999). Nevertheless, Norris (1993) and Wenger (1999) suggested that narrow RBS could also improve water quality, though the highly variable efficiency reported mandates further studies in diverse environments (Hickey & Doran 2004).

Soil water chemistry can change greatly over just a few meters in a soil horizon (Hedin et al. 1998), and this can contribute to the sometimes surprising efficiency of narrow RBS (Hickey and Doran 2004). One-meter-wide buffers have thus been shown to slow runoff and trap eroded soil particles, absorb soluble nutrients and favor denitrification and infiltration (Dabney et al. 2006). Indeed, vertical hydraulic gradients can also play a role in the efficiency of RBS, and RBS efficiency may change in surface runoff or groundwaters (Polyakov et al. 2005). Because tile drainage bypasses buffer strips (King et al. 2015), RBS may constitute a more efficient tool in non-drained lands, which are still present in some agricultural landscapes of Canada (Shady 1989; Harker et al. 2004), the USA (McCorvie and Lant 1993; Zucker and Brown 1998) and Europe (Herzon and Helenius 2008).

The limited adoption of RBS in Québec, despite the existing policy, suggests that farmers may need alternate motivations than the preservation of their soils or the common water resource to implement RBS on their farms (Sager 2004, Dagenais 2015). Where farmers use RBS, they often consist of herbaceous vegetation spontaneously colonizing the riparian areas on the outskirts of fields. Accordingly, most knowledge on narrow RBS focuses on the widespread herbaceous buffers (Gasser et al. 2013). However, as both vegetation type and plantation density also influence RBS efficiency (Mayer et al. 2006), there is a place for innovation through careful selection of vegetation and RBS plantation design, in a multifunctional RBS system that could incite more farmers to adopt this diffuse pollution mitigation technology. Willows (*Salix* sp.) are good candidates to increase the efficiency of narrow RBS. They grow naturally in riparian areas (Dickmann and Kuzovkina 2008), are efficient soil- and water-phytoremediation agents (Mirck et al. 2005; Kuzovkina and Volk 2009) and rapidly produce abundant biomass, which could generate revenues for farmers (Labrecque and Teodorescu 2003, 2005). Non-point source pollution mitigation by willow has been demonstrated in Quebec (Gasser et al. 2013) and elsewhere (Börjesson 1999). In Quebec, narrow willow buffer strips have been studied in controlled settings at the plot scale (Gasser et al. 2013), as well as at the watershed scale in uncontrolled agricultural settings where tile drainage was present (Terrado et al. 2014). It is henceforth desirable to test a novel narroe RBS system, where biomass production could increase both farmer incentives and water filtration efficiency, in uncontrolled agricultural settings without tile drainage, according to the guidelines set in the PPRLPI (MDDEP 2005).

The RBS efficiency varies seasonally, and there are critical moments to target when testing the efficiency of an RBS, including peaks in runoff, agro-chemicals concentration, or bioremediation activities (McClain et al. 2003; Vidon et al. 2010). These critical moments are insufficiently characterized (Vidon et al. 2010). For instance, the vegetation "dormant" season (when plant uptake is reduced) is rarely monitored in RBS studies, even though it may be an intense denitrification period (Vought 1994) and coincide with snowmelt which is a peak export period for soil particles and organic litter (Royer et al. 2006)

It is known that vegetated RBS generally allow to attenuate nutrient runoff (Dabney et al. 2006), but that RBS efficiency is proportional to vegetation type and density (Mayer et al. 2006) or standing biomass (Jianqiang et al. 2008). Accordingly, we hypothesized that the RBS efficiency can be increased compared to the common spontaneous herbaceous vegetation, without increasing its width, by selecting vegetation with demonstrated phytoremediation ability (Gomes 2015; Gasser et al. 2013; Börjesson 1999; Mirck et al. 2005; Kuzovkina and Volk 2009) and its augmenting the plantation density. This paper tests the efficiency of vegetated RBS which minimally respect the guidelines of the Quebec policy, in crop fields of the Saint-Lawrence lowlands (Quebec, Canada). We address four specific goals: (1) Determine to which extent nutrients (NO_2^--NO_3^-, NH_4^+, PO_4^{3-} and K^+) concentrations are reduced during infiltration and from the RBS edge-of-field to edge-of-streams; (2) Compare the efficiency of high-density willow plantations to that of low-density willow and ruderal herbaceous vegetation buffers, in mitigating aqueous nutrient exportation; (3) Distinguish RBS efficiency at snowmelt, post-fertilizer and post-glyphosate applications; and (4) assess if any of these RBS treatments suffice to conform to provincial water quality standards for the protection of aquatic life.

2.2 Materials and Methods

2.2.1 Study sites

The two experimental sites border streams in Saint-Roch-de-l'Achigan (SR: 45.84675°, -73.60463°; alt. 46 m) and Boisbriand (BB: 45.61106°, -73.86119°; alt. 44 m; Table 2- 1). The local growing-season precipitations were recorded on site and data was complemented with Environment Canada's regional statistics for precipitation and temperature using the Agrometeo database (Lepage and Bourgeois 2011) from the closest weather stations (Table 2-1). From 2010 to 2013, temperature, precipitations, degree-days of growth, relative humidity and solar radiations were comparable at BB and SR (Table 2- 1; detailed climatic statistics in Chapter 1). Site topography, established during a survey in 2011 using a differential GPS (R8GNN Base and Rover, Sunnyvale, CA, USA) with ~0.01 m vertical accuracy (USGS (United States Geological Survey) 2013), led to the creation of a digital elevation model encompassing the RBS and the proximal field region using ArcGIS (version 2.1.4, Esri, Redlands, CA, USA) with a 0.01 m vertical precision and a 50 cm resolution (details in Annexe 4). The surface runoff network was calculated with ArcHydro Basic Dendritic Terrain Processing (version 2.0, Esri, Redlands, CA, USA). While concentrated flows were visible during intense rain in the field and on aerial photographs, the buffer strip itself was not impaired by channel erosion. The micro-basins draining towards the RBS were 18.6 ± 25.9 m^2 in BB and 676.9 ± 745.2 m^2 in SR (micro-basins were significantly greater in SR). The greater a source-area is, the greater a total quantity of nutrients passing through the RBS at a specific point might be. The surface runoff network revealed that average flows (modeled pathways) cross the buffer strip perpendicularly, although with local heterogeneities. Groundwater depth, measured by piezometers, was shallower at BB during springmelt (Table 2- 1). At BB, connectivity between the water table and the stream was visible from a resurgence zone in the stream, and during dry summer months, the stream appeared to feed groundwater under the BB fields. Field soils were provided by farmers at BB and SR, and analyzed by AgroEnviroLab (LaPocatière, QC, Canada, accredited by CEAEQ and ISO-CEI 17025). Soil pH was near neutral, with organic matter content higher at BB than at SR, while cationic electrical conductivity (CEC) was superior at SR. The soil was richer (P, K, Ca, Mg, Al and Fe) and soil P saturation was also greater at BB (Table 2- 1). BB soil had finer texture in its mineral fraction, but its high organic matter content made it more permeable than that at SR, which had coarser surface texture (Table 2- 1). Soil series found in field include Achigan (SR), Châteauguay (BB), Dalhousie (BB) and Saint-Bernard (BB). A total of nine soil stratigraphic descriptors were characterized visually during manual coring (0-200 cm) across both sites (Table 2- 1). In BB, the soil stratigraphy (from top to bottom) include black histosol (strongly decomposed on von Post Scale), brown histosol (less decomposed), peat (lightly decomposed), till, marl, grey clay and reddish clay. Organic-rich soil is generally present everywhere at 30 cm depth while marl and/or clay is found near 70 cm. In SR, sandy loam, clean sand lentils and clay with traces of iron oxides were observed from top to bottom. Though surface soil appeared homogeneous on both sites, below ground soil strata varied slightly between parcels (a detailed 3D stratigraphic model is provided is Annexe 4). Slight differences between field and RBS soil physico-chemistry are presented in Annex 2.

The cultivation history at BB was conventional corn (2008), glyphosate-resistant corn (2009) and Identity Preserved soy (2010) whereas the crop rotation at SR included both carrot and soy (2008), conventional corn (2009) and conventional soy (2010). During the study period, the soil was under rotations of soy and corn, both glyphosate resistant (Table 2- 2). Glyphosate-based herbicides were applied once a year post-emergence at recommended rates (Table 2- 2). Fertilization was in accordance with agronomic recommendations (Table 2- 2). No organic amendment was supplied at BB, while SR received stabilized sludge from the municipal waste water treatment facility in 2012 and pig slurry in 2013 (Table 2- 2). In BB, soil was worked under minimal till practice with 4"-harrow disc finish, followed by one pass of grubber at sowing and a last conventional tillage occurrence in fall 2010. In SR, conventional tillage at a depth of 8" ended with a last pass in fall 2009. At sowing in 2010, a harrow disc was used for 1 to 2 passes at 4" depth. The field was maintained under no till in 2012 and 2013.

2.2.2 Experimental design and water sampling

At each site, two treatments plus a control were established in triplicates, in a randomized block design (Figure 2- 1). The three treatments (3-m width x 17-m length, i.e. 51 m^2) consisted of natural herbaceous vegetation (treatment CX) and two densities of willows with 3 rows (treatment 3X) and 5 rows (treatment 5X) representing 33 333 and 55 5556 stems·ha^{-1}, respectively. *Salix miyabeana* SX64, a highly productive clone with good insect resistance (Labrecque and Teodorescu 2005), was planted in spring 2009, coppiced in fall 2009, coppiced again prior to next growth season in spring 2011, and harvested after a three-year growth cycle in fall 2013. According to another study with similar stratigraphy nearby the SR site, *S. miyabeana* SX64 roots were most abundant in the top 20 cm of soil (87%), decreased with depth (7% at 20-40 cm, 5% at 40-60 cm) and rarely grew (1%) beyond 60 cm of depth (Jerbi et al. 2015). The control plot and edges of willow plantations were mowed by farmers once every growing season. For details on plantations, vegetation maintenance and biomass production and diversity, refer to Chapter 1.

Surface runoff was collected in high density polyethylene (HDPE) buckets buried over three quarters of their height in the ground and fitted with 30-cm polyvinyl chloride (PVC) gutters sheltered from rain and extending atop soil surface, perpendicular to the buffer strip. These gutters were equipped with 2-mm nylon mesh to keep coarse particles out. At sampling time, total volume of water collected was estimated in-situ (with a ruler). Water was homogeneously mixed throughout physical measurements and sub-sampling for laboratory chemical analysis. Interstitial water was collected in PVC suction (-70 kPa) lysimeters (Soil Moisture Equipment Inc.: 1900-L model) equipped with ceramic cups (pore size: 1.3 ± 10% µm) buried at 35 or 70 cm depth in soil slurry at BB and in clayey soil at SR (crushed silica lanterns sealed with local clay strata) (Osborne and Kovacic 1993). Prior to sampling, residual pressure was checked with a manometer. Total water volume was pumped manually into a low-density polyethylene (LDPE) tube connected to a graduated glass Erlenmeyer with Tygon® tubing, for measurement. Piezometers (UV-resistant PVC, ⌀: 5.7 cm) were installed 2 m below soil surface. Each piezometer's 50-cm long strainer was protected with silica, sealed with clay, repacked with excavated material respecting soil horizons, then capped. Sampling was carried out by inserting a clean LDPE tube connected to a manual peristaltic pump into the piezometer, after purging a volume equivalent to that of the piezometer. Water was collected in a glass Erlenmeyer, transferred into two NalgeneTM 250-mL bottles and stored at 4°C for subsequent centrifugation (10.000 g for 10 min), aliquot preparation and analysis at the laboratory, within 24 hours.

Surface runoff (0 cm), infiltration in the vadose zone (unsaturated soil near 35 or 70 cm) and in the aquifer (saturated soil near 200 cm) were monitored throughout growing seasons spanning 18 sampling campaigns from spring 2011 to spring 2014. Sampling occurred after precipitation events and targeted important agricultural events (spring snowmelt, sowing and fertilization, and application of herbicides). The intent was to sample each equipment twice per period, so 6 times per year, but success rates varied as described below. In total, 36 surface-water collectors, 72 lysimeters and 24 piezometers were designed, installed close to the edge-of-field (CF) and close to the river (CR), and sampled as described in Figure 2- 1. Each runoff collector and lysimeters group (each 1 m apart) was positioned midpoint along the length of each RBS parcel, 50 cm before (CF) or after (CR) de RBS. The piezometers were positioned on the four corners of each block (50 cm away from the RBS) to avoid disturbance of groundwater near the shallower sampling equipments. Microbasins are significantly larger in SR, and may be larger for 5X treatments according to the model, which based area estimations on the closest modeled runoff flowpath. On the other hand, in BB, 3X parcels may receive runoff from significantly greater source areas (Annexe 4). Of the intended 1 122 sampling units, 1 104 samples were collected from SR and BB, between spring 2011 and summer 2013 (SR) or spring 2014 (BB). Sampling and analysis success rates varied from 40-53% for surface-water samples to 56-90% for the deepest soil-water samples, depending on whether there was enough water in sampling apparatus and to conduct all analyses.

Water samples for total suspended solids analyses were collected unfiltered in 250-mL HDPE bottles. Water samples for P_{tot} were collected unfiltered in 30-mL HDPE NalgeneTM bottles (Thermo Fisher Scientific Inc., MA, USA). Water samples were pre-filtered using a WhatmanTM GFF fitted syringe (runoff only), and filtered (PES, Pall Corporation; pore size: 0.2 µm, ⌀: 2.5 cm) directly in the field, or after lab centrifugation for phreatic water, collected in 15-mL Polystyrene centrifuge tubes (Starstedt™) and kept on ice until laboratory analyses (<24 h for NO_2^-) or frozen. Dissolved cations were pre-filtered (runoff) and filtered as described above, but HNO_3 (analytical grade for trace-metal analysis, Sigma-Aldrich, Saint-Louis, MO, USA) was added to the 30-mL HDPE Nalgene™ bottles (pH ≤ 2) prior to freezing. All plastic bottles were previously cleaned with 10% HCl, rinced with distilled then deionized water (MilliQ™) three times. For Dissolved Organic Carbon (DOC), water samples were pre-filtered and filtered as above, and HCl (pH < 2) was used to preserve samples in two 2-mL combusted (450 °C) amber-glass vials fitted with teflon caps.

Basic physicochemical parameters including temperature (T), pH (± 0.2) and electrical conductivity (EC ± 2.0% µS) were measured in situ with a multimeter (model 63, YSI, OH, USA). Dissolved oxygen was measured in the last sampling event at BB (5100 recorder with 510 BOD probe, YSI inc., Yellow Spring, OH, USA). Total suspended solids (TSS) were weighed (± 0.0001 g) on dried (40 °C) pre-weighed 4.2-cm nylon discs (NylaflowTM, PALL Corporation, USA) after 0.2-µm filtration. Dissolved nutrients (NO_2^-, NO_3^-, NH_4^+, PO_4^{3-}) were analyzed on a TrAAcs 800 continuous flow analyzer (Technicon/Bran+Luebbe) following standard methods (APHA 1992; Wetzel and Likens 1995). The sum of NO_2^-+ NO_3^-+ NH_4^+ was expressed as N_{tot}. For P_{tot}, 4-ml unfiltered water samples were digested using 64 µg $K_2S_2O_8$ and autoclaved (121 °C, 45 min) prior to filtration and analysis for PO_4^{3-} as described above.

Only surface runoff samples were analyzed for P_{tot} as overland flow is almost exclusively exported via this pathway in watersheds (Royer et al. 2006). Dissolved cations (Al^{3+}, Fe^{2+}, K^{+}, Mg^{2+}, Mn^{2+}, Ca^{2+}, Na^{+}, Zn^{2+}) were analyzed from filtered water by Atomic Absorption (GBC 906AA, Hampshire, IL, USA) with acetylene-air flame or acetylene-protoxyde flame (Ca^{2+} and Al^{3+} only), according to standard protocols (APHA 1992; Hendershot et al. 2007). DOC was measured in a Total Organic Carbon Analyzer (TOC-5000A, Shimazu, Kyoto, Japan; (Centre d'expertise en analyse environnementale 2011). Soil organic matter was measured by combustion in an elemental analyzer (Carlo Erba NC2500; (Carter and Gregorich 2007).

2.2.3 Statistical analysis

All statistical analyses were conducted using JMP 7 (SAS Institute, Cary, NC). Wherever data did not fit the normality and homoscedasticity criteria, they were log transformed or analyzed non-parametrically. Summer 2012 was hotter and dryer than normal (Environment Canada 2013), leading to difficulties in collecting runoff water samples. Furthermore, uncontrolled field conditions prevented water collection in every sampling equipment and for every campaign. To circumvent this concentrations were pooled by agricultural event (snowmelt, post-fertilization and post-glyphosate) for statistical analysis. We checked that sampling years (2011-2014) were not statistically different with a Wilcoxon test (per site, depth, side of the buffer strip and agricultural event) prior to pooling. To circumvent other data gaps related to some missing data points before or after the RBS, statistical analyses were reorganized from the initially planned repeated time ANOVA on the nutrient concentration before (CF) and after (CR) the buffer strip (pairing proximal CF-CR sampling equipments). This methodology chosen *a priori* to hydrological surveys was based on the hypothesis that runoff and groundwater flowed directly from the field to the stream, crossing the RBS perpendicularly. However, Annexe 4 revealed that runoff incoming and exiting each RBS parcel had heterogeneous incidence angles, whereas when all RBS parcels in the fields were considered together, the incidence angle did approach the assumed 90 degree interception by the RPS. Hence, CF mean concentrations were used to buffer microsite heterogeneities and facilitate interpretation of the RBS effect at a field scale (Annexe 4). The chosen method helps to alleviate the runoff direction heterogeneities, which are present at a local scale but smoothened at a larger scale. This observation had the added benefit of circumventing the aforementioned data gaps. Nutrient concentrations along depth profiles were compared before (CF) and after (CR) the buffer strips with a Wilcoxon multiple pair comparison. While surface and interstitial waters may not always cross the buffer strip horizontally (infiltration fluxes were not quantified herein), the comparison on concentrations measured along whole depth profiles allows to define trends across stratigraphic layers. This means that our study is not necessarily restricted to horizontal flow, as opposed to other RBS studies. Treatments (CX, 3X and 5X) were compared in a post hoc test via a Steel test (using edge-of-field CF as a control) for all statistically distinct profiles and those exhibiting non-significant (ns) trends ($p = 0.1 \geq 0.05$).

RBS potential efficiency (%) equals ($\overline{X}_{CF}$ - $\overline{X}_{CR}$)/ $\overline{X}_{CF}$, positive potential efficiency indicating nutrient concentration reduction. The efficiency was measured on concentrations for each depths, and each depth should be appreciated together with other depths along the profile to understand the potential role of infiltration.

2.3 Results

2.3.1 Water chemistry at the edge-of-the-field

We investigated soil nutrient contents at the exit of fields (before buffer strips) at both BB and SR experimental sites. Higher nutrient concentrations were observed at BB than at SR when comparing depth profiles between 0 and 200 cm (Figure 2- 2). At both sites, surface concentrations of agricultural nutrient inputs (NO_2^-+ NO_3^-, NH_4^+, PO_4^{3-} and K^+) decreased as leachate infiltrated the soil at the edge-of-field (Figure 2- 2). However, after an initial concentration decrease, NH_4^+ appeared to accumulate in the BB water table while at SR, NO_2^- + NO_3^- concentration tended to increase at depths beyond 35 cm. Na^+ and Mn^{2+} displayed a similar trend at both sites, Mn^{2+}_{SR} (subscript refers to the site) concentration reaching a minimum at a depth of 70 cm rather than 35 cm (Figure 2- 2). Concentrations of other nutrients, such as Zn^{2+} and DOC_{SR}, steadily decreased with increasing soil depth (Figure 2- 2). On the contrary, the EC increase coincided with a rise in Mg^{2+}, Fe^{2+}_{BB} and Ca^{2+}_{SR} concentrations along increasing depths. Other variables displayed mixed trends, e.g. pH reaching a minimum near 70 cm or Al^{3+}_{SR} concentrations peaking at 70 cm. Temperature, on the other hand, initially increased before stabilizing with increasing soil depth (Figure 2- 2).

Figure 2- 2 shows proportions of dissolved nitrogen species nitrite (NO_2^-), nitrate (NO_3^-) and ammonium (NH_4^+), relative to the global nitrogen pool (N_{tot}) at snowmelt and post-fertilization stages. NH_4^+ was relatively more abundant at SR than at BB, from the surface down to 70-cm depth (but not in the phreatic zone, at 200 cm). At each site, NO_3^- predominated in the vadose zone (between 35 and 70 cm), and down to 200 cm at SR (Figure 2- 2). Only at the surface, and at 200 cm depth as well in the case of BB, was NH_4^+ most abundant. BB appeared more affected by seasonality than SR, as evidenced by bigger shifts in relative proportions of dissolved nitrogen species from snowmelt to post-fertilization stages (Figure 2- 2). Furthermore, edge-of-field P_{tot} concentration in runoff after fertilization (BB: 9.3 $mg \cdot L^{-1}$ and SR: 3.8 $mg \cdot L^{-1}$) was twice higher than that at snowmelt, but post-fertilization PO_4^{3-} concentration were comparable between sites (BB: 2.4 $mg \cdot L^{-1} \approx$ SR: 2.0 $mg \cdot L^{-1}$). Based on these important discrepancies, each site and sampling period was considered separately in the quantification of RBS potential efficiency.

2.3.2 The behavior of macronutrient across the buffer strip

Macronutrients means per site are first presented as a comparison of $\overline{X}_{CF}$ and $\overline{X}_{CR}$ along depth profiles (Figure 2- 2) and then treatments distinguished via a post-hoc test are found in Figure 2- 5. At both sites, nitrate concentrations were most reduced through RBS after sowing and fertilization, when concentrations were highest (Figure 2- 4). However, significant reductions were found near the surface at BB (0 cm, p = 0.0235*; 35 cm, p = 0.0384*) while significant reductions occurred slightly deeper at SR (35 cm, p = 0.0006*; 70 cm, p = 0.0008*) and reduction trends extended even deeper (200 cm, p = 0.0782). No other significant nitrate loads were recorded, except at 70 cm during snowmelt at SR (p = 0.0227*).

Ammonium concentration was reduced only at BB during snowmelt (p = 0.0463*) (Figure 2- 4). Runoff water ($\overline{X}_{CF}$ = 10 mg NH_4^+-N/L), was almost two orders of magnitudes more concentrated with NH_4^+ than melting snow (~100 µg/L NH_4^+-N; Table 2- 3).

P_{tot} concentration was only significantly reduced at the post-fertilization stage, on soil surface at SR (0 cm, p = 0.0300*), though it was only quantified in runoff water at post-fertilization and snowmelt events (Figure 2- 4). PO_4^{3-} load was never significantly reduced across the RBS. There were, however, non-significant reduction trends at BB, in runoff water during snowmelt (p = 0.0631), and at SR, at 35 cm depth (p = 0.0861).

Decreases in K^+ concentrations across the RBS were common at BB : at snowmelt (35 cm, p = 0.0090*; 70 cm, p = 0.0209*; 200 cm, p = 0.0148*) ; at the post-fertilization stage (35 cm, p = 0.0290*) with a non-significant trend extending to the post-glyphosate stage (35 cm, p =

0.0290*). On the contrary, K^+ concentration reduction was only documented once at SR, at the post-glyphosate stage (200 cm, p = 0.0437*). The absence of concentration gradient across the RBS at SR, at snowmelt, was also observed for other cations such as Ca^{2+}, Mg^{2+}, Mn^{2+}+ and Al^{3+} (Annexe 23).

2.3.3 The effect of vegetation treatments in riparian buffer strips

The choice of vegetation rarely made a significant difference in RBS potential efficiency (Figure 2- 5a and b). Significant differences were observed mainly at the post-fertilization stage, at both sites, while nitrate and nitrite loads were highest (Figure 2- 5a,b). As for nitrate and nitrite retention, the herbaceous buffer strip was significantly more efficient than willow treatments at BB: 35 cm, CX: p = 0.0365*; CX potential efficiency (85.6%) >3X (28.6%) and 5X (31.9%). At SR, on the contrary, the willow treatments were shown to be most efficient: 35 cm, 3X: p = 0.0259*; 5X: p = 0.0421* and 70 cm, 3X: p = 0.0253*; 5X: p = 0.0284*; CX potential efficiency (97.0-92.8%) < 3X (98.3-96.9%) and 5X (97.7-98.0%) for 35-70 cm, respectively (Figure 2- 5a). At snowmelt, herbaceous vegetation at SR leached more nitrate and nitrites: 70 cm, p = 0.0227*; CX potential efficiency (-112.8%) > 3X (6.6%) and 5X (-17.0%). For NH_4^+, no significant difference between vegetation treatments could be distinguished (Figure 2- 5a).

For P_{tot} in post-fertilization runoff water at SR, high-density willow RBS appeared more efficient than other treatments: p =0.0300*; CX potential efficiency (45.7%) and 3X (41.0%) < 5X (77.1%) (Figure 2- 5b). As for PO_4^{3-}, no significant difference between the treatments was observed, either at BB or SR (Figure 2- 5b). For K^+ at BB, the herbaceous vegetation was most efficient at the post-fertilization event: 35 cm, p = 0.0290*; CX potential efficiency (83.8%) >3X (48.5%) and 5X (44.2%), while at snowmelt, the low-density willow treatment was most efficient: 35 cm, $p = 0.0090$*; 3X potential efficiency (47.7%) > CX (-39.6%) and 5X (30.5%) (Figure 2- 5b).

Finally, trends were observed at BB and SR that suggested opposite efficiencies (arrows in Figure 2- 5a,b) of the herbaceous and high-density willow treatments from one depth class to the other, with NO_2^-+ NO_3^-, NH_4^+ and PO_4^{3-} elements, especially at post-fertilization and snowmelt stages (Figure 2- 5a,b).

2.4 Discussion

2.4.1 N-P-K leaching at edge-of-field and decrease via infiltration

In the current experiment, nutrient concentrations before the RBS were averaged to minimize the effect of localized heterogeneity in the agricultural leachate. This method is supported by the discussion of Noij et al. (2012) on the proper use of controls in RBS studies, especially when assessing the effectiveness of a lower riparian zone adjacent to agricultural fields. Several other authors (Lee et al. 2003; Munoz-Carpena et al. 1999; Gasser et al. 2013; Patty et al. 1997; Tingle et al. 1998) have previously compared the average concentrations without a buffer strip (assumed equivalent to CF) to average concentrations after different buffer strip treatments (equivalent to CR) without necessarily pairing by geographic proximity. The statistical rationale for this choice is further explained in Annexe 4.

Decrease of aqueous nutrients (NO_2^-+ NO_3^-, NH_4^+, PO_4^{3-} and K^+) during infiltration was evidenced at the edge-of-field (Figure 2- 2). Nitrate-N predominance in interstitial water at the edge-of-field (Figure 2- 3) was previously reported in other studies (Sabater et al. 2003). It has been explained by higher mobility of NO_3^- through soil layers compared to that of NH_4^+, which adsorbs to clay and hence, remains closer to the surface (Duchemin and Hogue 2009). As for NH_4^+ abundance in the phreatic zone at BB (Figure 2- 3), it could be due to reduced oxygen concentrations in deeper soils, which may have prevented NH_4^+ oxidation to NO_2^- and NO_3^- (Jones and Mulholland 1999), thus favoring longer residence times. Accordingly, decreasing O_2 content at increasing soil depth was confirmed at BB in 2014 during sampling at snowmelt: from soil surface to 70-cm depth, water saturation declined from 55% to 15% (corresponding to a drop from 6.6 to 1.8 $mg \cdot L^{-1}$ of O_2 concentration), and is consistent with the expected lower O_2 content with depth and water saturation. Note, also, that denitrification of the oxidized nitrogen species is favored when the water table is shallow (Burt et al. 1999; Hill 1996; Pabich et al. 2001). The fact that NO_2^- concentrations remained marginal, especially underground at BB (Figure 2- 3), could reflect its status as an intermediate product in both denitrification and nitrification processes, quickly consumed in the subsequent reduction or oxidation steps (Ausland 2014). Also, the lower concentrations of NO_2^- could be due to its smaller redox range of stability. But within microenvironments that possessed a redox potential different from the bulk, its presence could nevertheless be detected (Husson 2013). The NO_2^- concentrations were slightly higher at the surface (Figure 2- 3) likely due to the fact that more NH_4^+ was available from fertilization, for instance from pig slurry (Table 2- 3). Hence, nitrogen speciation along the soil profile could be explained by balance changes between nitrification and denitrification, depending on the redox potential or on the initial proportions of nitrogen species. The pH variation along the depth profile was minimal at both sites (Figure 2- 2), therefore the redox potential may have been more determinant to explain the nitrogen speciation profiles observed (Figure 2- 3; Husson 2013).

Compared to other anions such as NO_2^- and NO_3^-, PO_4^{3-} concentrations decreased more abruptly along the vertical profile, at BB and SR, despite similar concentrations near the surface (Figure 2- 2). Decrease of PO_4^{3-} concentration in interstitial water may have occurred via adsorption or absorption mechanisms, as observed elsewehere (Dorioz et al. 2006). Furthermore, deposition of P associated with soil particles during infiltration is an alternate mechanism to explain decreasing PO_4^{3-} decrease with depth whose importance cannot be quantified here since only the water fraction below 0.2-µm pore size was analyzed (Materials and Methods). Interestingly, BB and SR exhibited similar PO_4^{3-} infiltration behavior despite important site-specific differences. First, their pedology was different: the P-saturation index reached 5.4-7.6% at BB (Table 2- 1), a value just below the 7.8% critical threshold calculated for P enrichment of a soil solution (Beaudin et al. 2008). This was in accordance with generally low levels of P immobilization in rich, organic soils (Vought et al. 1994). On the contrary, SR had shallower clay strata (Figure 2- 2) and a higher CEC (Table 2- 1), which should both have favored greater surface P adsorption (Heathwaite and Dils 2000; Beauchemin et al. 1998). Secondly, tillage management differed between the experimental sites: SR was maintained under no-till, a practice that usually favors infiltration of dissolved P (King et al. 2015). Moreover, cracks were visible at the soil surface, indicating the presence of preferential macropores that generally facilitate P infiltration (King et al. 2015). Thirdly, SR fields received organic fertilizers (pig slurry and sewage sludge, Table 2- 2) that contained PO_4^{3-} at concentrations (Table 2- 3) that could lead to P leaching (Wang et al. 2004). Hence, the similarity of both PO_4^{3-} profiles (Figure 2- 2) may be best explained by biotic rather than abiotic processes, such as the rapid cycling through vegetation or microbial sequestration and decomposition at soil surface (Dorioz et al. 2006).

Compared to N and P, K^+ edge-of-field concentrations were three orders of magnitude lower, in the same range as Ca^{2+} and Mg^{2+} concentrations (Figure 2- 2). However, contrary to those and other cations, except perhaps Zn^{2+} (Figure 2- 2), K^+ concentration decreased along the vertical profile at edge-of-field. Therefore, we propose that K^+ originated mainly from fertilizers (Table 2- 2) while most other cations enriched the interstitial waters as they percolated through the mineral matrix. For instance, the Al^{3+} concentration peak at SR (Figure 2- 2) was consistent with the mineralogy of the Champlain sea clay deposits (Berry et al. 1998), and that of Ca^{2+} at BB, with the mineralogy of lacustrine marl (calcium-carbonate rich mud) strata (Pettijohn 1957).

The edge-of-field nutrient concentrations that we measured were consistent with previous studies, including NO_2^-+ NO_3^- (Schultz et al. 1995; Ginting et al. 2000; Sabater et al. 2003; Young and Briggs 2005); NH_4^+ (Sabater et al. 2003; Young and Briggs 2005); and P_{tot} and PO_4^{3-}, whose levels were similar to those previously published (Ginting et al. 2000; Duchemin and Hogue 2009), or slightly above those measured in runoff water from comparable experimental settings of corn and soy fields (Osborne and Kovacic 1993). Nevertheless, the RBS potential efficiency reported here might not be applicable to drained fields, where groundwater P concentrations, often found to be lower than those measured in tile drainage (King et al. 2015; Heathwaite and Dils 2000), may result in RBS bypass.

2.4.2 Variable potential efficiency of 3-m-wide vegetated riparian buffer strips on nutrients mitigation over time

On the edge-of-field, nutrient concentrations were greatest just after fertilizations and seasonal variability was most pronounced in surface runoff (Figures 3-3 to 3-5, and Annexe 23). RBS potential efficiency was not consistent across sampling periods (Figures 3-3 to 3-5, and Annexe 23). This seasonal fluctuation of RBS efficiency in Québec was also acknowledged by Gasser et al. (2013). They studied a herbaceous control and different *Salix* treatments for 3 consecutive years and calculated efficiency with an unpaired statistical design where runoff (collected from hydrologically partitioned parcels) and interstitial water (collected using lysimeters) were sampled only after the RBS. On the other hand, seasonality was considered unimportant in RBS efficiency to reduce N concentrations in runoff water (where removal rates are expressed as the difference between the input and output nitrate concentration, expressed as a percentage of the input and normalized per unit width of RBS) across 14 scattered

European sites (Sabater et al. 2003). This could possibly be due to less extreme climate fluctuations in Europe than in Quebec.

In the current study, one possibility was that nutrient concentration variations across the RBS changed with sampling time because of seasonal variations in their composition. Accordingly, the PO_4^{3-}/P_{tot} ratio in SR runoff was higher at post-sowing and post-fertilization stages (62%) than at snowmelt (12%) (Figure 2- 4). Ginting et al. (2000) also observed seasonal variations, however with a reverse trend for the PO_4^{3-}/P_{tot} ratio under summer precipitations and snowmelt, with dissolved nutrients that were more concentrated at snowmelt, and more particulate-bound pollutants during rainfall. Perhaps the erosion potential of estival precipitations observed by these authors was lower than that of rapid spring snowmelt in our study. In our experiment, no seasonal variability of this ratio (constant ~21%) was recorded at BB (Figure 2- 4), which could be due to a higher infiltration potential. On the contrary though, N speciation changed very little over time at SR, while BB displayed more obvious differences between snowmelt and post-fertilization events (Figure 2- 3). Finally, contrary to other macro elements which fluctuated widely over time, K^+ concentration fluctuated little as important agronomic periods unfolded at both sites: annual K^+ concentration variation (K^+_{max} - K^+_{min}/K^+_{max}*100) approximated 30% at BB and 50% at SR. Altogether, these observations may suggest that site-specific variations could affect seasonal variations in RBS potential efficiency, hence a closer look on RBS potential efficiency for each sampling season and site follows.

First, sowing and fertilization on barren fields favored nutrient leaching and represented the annual peak in edge-of-field concentrations for NO_2^-+NO_3^-, NH_4^+ and PO_4^{3-} at BB and SR (Figures 3-3 to 3-5, and Annexe 23), similar to observations by Osborne and Kovacic (1993) on PO_4^{3-} and P_{tot}. Decrease in NO_2^-+NO_3^- concentrations during post-fertilization sampling was recorded at both sites (Figures 3-3 and 3-4). However, K^+ concentration reduction was measured only at BB, and P_{tot} concentration reduction only at SR (Figure 2- 4). The higher nutrients at the edge-of-field post-fertilization may stem from both mechanical reworking of the soil and fertilizers inputs; and increased temperatures at late spring could support higher litter mineralization and nutrient leaching from the field. However, this also coincides with potentially greater levels of nutrient absorption by plants and soil denitrification rates, explaining why this time of year was favorable for enhanced RBS efficiency, as suggested by Hefting et al. (2005).

The second sampling period (which followedglyphosate application) occurred once emerged corn and soy plants were actively absorbing nutrients from the soil, coinciding with RBS vegetation's full development, and after most of the labile nutrients had already leached out. However, because precipitations are generally limited during the glyphosate application periods over the three year span of the study (Annexe 26), this challenged the collection of surface runoff (especially at BB, where the soil was permeable). The resulting limited statistical strength could explain the absence of significant RBS potential efficiency in mid-summer as expressed by others (Gasser et al. 2013). Other explanations to the lack of RBS potential efficiency reported during the post-glyphosate sampling period could involve interactions between glyphosate and nutrients. During this period, 3.4 - 3.7 $\mu g \cdot L^{-1}$ of glyphosate was measured in the runoff at BB, and 20 $\mu g \cdot L^{-1}$ concentrations were measured at SR (Chapter 3). Soil concentrations (only measured in SR) averaged 210 $\mu g \cdot kg^{-1}$ dw (Chapter 3).It has previously been reported that glyphosate may interfere with the uptake of various plant nutrients like Ca^{2+} and Mg^{2+} (Duke et al. 1985; Cakmak et al. 2009) and Fe^{2+} and Mn^{2+} (Cakmak et al. 2009), possibly by chelation and subsequent immobilization of nutrients in the soil (Duke et al. 2012; Gordon 2007; Yamada et al. 2009; Zobiole et al. 2012). However, Duke et al. (2012) argued that under normal glyphosate application rates, the glyphosate in soil solution (i.e. 1 $kg \cdot ha^{-1}$ over the top 10 cm would represent a soil concentration of 750 $\mu g \cdot g^{-1}$ and a potential soil solution of 7.44 $\mu g \cdot L^{-1}$) would be much smaller than typical cations in soil solution. Duke's concentration argument does not apply in the current study, as glyphosate concentrations measured in the post-glyphosate period are similar to or greater than the Mg^{2+}, Mn^{2+}, Fe^{2+} and Zn^{2+} concentrations measured in BB or SR (Annexe 23). Hence, interactions between glyphosate and those cations is likely. Another interesting aspect to consider in the potential interaction between glyphosate and nutrients in the soil solution, is that the glyphosate complexes with cations may not all have the same solubility. The least soluble complexes ($Fe^{3+} < Cu^{2+} < Zn^{2+} < Mn^{2+} < Mg^{2+} \sim Ca^{2+}$) (Sundaram and Sundaram 1997) in near-neutral interstitial soil water could have precipitated in the soil (Subramaniam and Hoggard 1988). If the RBS constituted a favorable place for complexation, then it could halt further leaching. This would deserve further insight, and once again, based on the similar glyphosate and nutrient concentrations observed in the current study, this mechanism appears likely. Furthermore, antagonisms between N and glyphosate concentration reduction within the buffer strip — soil types minimizing N leaching may lead to glyphosate leaching (Aronsson et al. 2011) and conditions favoring denitrification may disadvantage glyphosate degradation (Pavel et al. 1999; Vidon and Hill 2004) —will be addressed in Chapter 3. Alternately, the limited potential efficiency measured below soil surface at BB may also have originated from a groundwater flow reversal in the driest summer months, a hypothesis explored in Annexe 4.

Thirdly, nutrient concentrations recorded at snowmelt were the lowest of the year, as Osborne and Kovacic (1993) observed during dormant season. Snowmelt erodes the soil (Agriculture and Agri-Food Canada 2002) and generates extensive runoff water: snow may account for only 30% of annual precipitation, but generates as much as 80% of annual runoff water at snowmelt (Dibike et al. 2012). Consequently, spring melt remains Quebec's peak nutrient leaching season (Terrado et al. 2014; Lapp et al. 1998). Brief rainfall or snowmelt dilute runoff water but large runoff volumes generated add up to important mass transfers (Royer et al. 2006). This is why recording the RBS potential efficiency at snowmelt was critical in the present study. Although nutrient leaching peaked at snowmelt, we observed only limited RBS potential efficiency in nutrient retention from runoff and interstitial waters, at BB: effects were documented mainly for NH_4^+ at soil surface, K^+ at 70-cm depth and non-significant trends for K^+ and PO_4^{3-} at soil surface (Figure 2- 4). Importantly, RBS potential efficiency was generally null at SR (Figure 2- 4). The absence of concentration gradient across the RBS for several cations (Annexe 23) seemed to coincide with a period of very low groundwater hydraulic gradients

($\Delta h_{CF\text{-}CR} \approx 0$ cm in approximately half of the sampling stations, Annexe 4). Overall, lack of potential efficiency at snowmelt was akin to findings by Gasser et al. (2013) that herbaceous vegetation and willow buffers were inefficient in retaining nutrients from snowmelt-induced runoff water near bovine winter enclosures.

2.4.3 Influence of nutrient speciation and type on RBS potential efficiency

While a decrease in PO_4^{3-} concentration across RBS appeared negligible at both sites (Figure 2- 4), NO_2^-+NO_3^- concentration reduction across RBS was generally measurable, except at snowmelt – a potential antagonism reported elsewhere (Vidon and Hill 2004; Vought et al. 1994). Note that at BB, the RBS were implemented in an ancient wetland. Wet riparian areas with low oxygen conditions are favorable to denitrification but hot spots for P release (Vidon and Hill 2004; Vought et al. 1994). Reducing conditions in water favors dissolution of PO_4^{3-} and iron complexes and subsequent plant uptake compared to drier sites where mineral adsorption predominates (Dosskey et al. 2010). Beyond this issue, clear evidence for reduction by RBS of soluble P concentration in groundwater is lacking (Osborne and Kovacic 1993). Moreover, soluble P may be released from RBS when discharge water volumes are large (Dorioz et al. 2006; Osborne and Kovacic 1993), which may have been the case at SR during snowmelt (see 5X effect at soil surface in Figure 2- 5b). Though we only monitored P_{tot} at soil surface, subsurface releases have been documented from RBS planted in fine sandy loam soil (similar to SR) with broad-leaf deciduous trees along corn fields (Peterjohn and Correll 1984). However, contrary to dissolved P for which the RBS had a widely variable efficiency (from -83% to +95%; most commonly 20-30%), P_{tot} is generally well retained by RBS (50-97%), as reviewed by Dorioz et al. (2006). Hence, we are unsurprised to measure no net potential efficiency in PO_4^{3-} reduction, and while the literature suggested that a P_{tot} reduction was common, we only observed it at SR.

Reports exist that corroborate our results of K^+ interception by willow RBS (Figure 2- 5b), and suggest planting willows in swales to increase this potential (Gasser et al. 2013). While we report here no major RBS reductions of the elements Ca^{2+}, Mg^{2+}, Mn^{2+}, Fe^{2+}, Na^+, Al^{3+} and Zn^{2+} (Annex 23), Gasser et al. (2013) demonstrated that some elements (K^+, Ca^{2+}, Na^+, B^+, Cu^{2+}, Zn^{2+}) were affected by the RBS while others (Al^{3+} and Fe^{2+}) were not. Those authors also showed that decreasing nutrient concentration was mediated by water retention across the RBS. Lowrance et al. (1984) demonstrated that forested RBS were a short-term filter for some nutrients (N was more retained than Ca^{2+}> K^+> Mg^{2+}> P), but that the nutrients sequestered in the vegetation was greater than the mass balance between aqueous nutrients inputs and outputs across the RBS. This could explain why our RBS vegetation could sequester nutrients (Chapter 1), while we couldn't detect this based on concentration changes in the waters (current chapter).

2.4.4 Herbaceous vs. woody buffer strips

Herbaceous RBS potential efficiency to retain N-P-K was non negligible compared to woody RBS (Figure 2- 5a,b), a finding corroborated by Mayer et al. (2007) but that contradicts our initial expectations and incentive to the selection of fast-growing willows for this experiment. Although dealing with RBS potential efficiency levels which were lower (> 4% nitrate reduction per RBS meter width) than those presented herein (~10-25%$\cdot m^{-1}$) (Figure 2- 5a,b), Sabater et al. (2003) also failed to discriminate between efficiencies of herbaceous and woody RBS across a wide range of climatic conditions in Europe. The standing biomass in our herbaceous RBS represented only a fraction of that in willow RBS (8% of 3X and 5% of 5X biomass at BB; ≥ 7% of 3X and ≥ 7% of 5X biomass at SR; Annexe 14). Similarly, Hefting et al. (2005), who focused on N retention efficiency in herbaceous versus forested riparian ecosystems using a zonage three point transect system encompassing the stream vincinity, the field and the RBS in between, but measuring N pools in biomass, litter and soil instead of aqueous flows like the current study, observed that herbaceous buffers produced less biomass than woody ones. The correlation between biomass quantity and RBS potential efficiency has not always been substantiated. Thus, Uusi-Kämppä and Yläranta (1996) showed that RBS with similar aboveground biomass production (mowed and harvested annually) could vary in potential efficiency: RBS planted with mixed herbaceous vegetation, shrubs and trees removed more nitrogen than herbaceous RBS solely composed of *Phleum pratense* L. and *Festuca pratensis* L. They monitored orthophosphates and nitrates in runoff using a duplicate design with one control (crops planted in the RBS zone) and two treatments (grass and mixed vegetation), and monitored vegetation yields and runoff prior to RBS implementation and three consecutive years after. Nonetheless, in a controlled laboratory experiment comparing the TN and TP removal efficiency of three types of vegetation, analyzing both water and plants, Jianqiang et al. (2008) stated that RBS nutrient retention capacity was directly proportional to its aboveground biomass production, which contradicts our observations in surface and groundwaters (Figure 2- 5). Interestingly, Gasser et al. (2013) reported changes in buffer strip efficiency from one year to the next due to climatic conditions and vegetation growth stage. Furthermore, using surrogate field runoff, Dosskey et al. (2007) reported that RBS efficiency may change over the first three years after buffer strip implementation, and that this may not necessarily be linked to vegetation type. This finding supports our method of pooling data from similar agricultural sampling periods throughout the entire duration of the experiment, which did start more than three years after RBS vegetation implementation. Most importantly, willow wood harvested after a three-year growth cycle allowed for permanent, annual export of 116-118 kg $N{\cdot}ha^{-1}$, 62-63 kg $K{\cdot}ha^{-1}$ and over 23 kg $P{\cdot}ha^{-1}$ at SR, and 278-447 kg $N{\cdot}ha^{-1}$, 148-239 kg $K{\cdot}ha^{-1}$ and 55-86 kg $P{\cdot}ha^{-1}$ at BB (Chapter 1), while unharvested biomass of herbaceous RBS only temporarily decreased nutrient movements. Indeed, an indispensable condition to long-term P_{tot} storage in vegetated RBS is maintenance (Wenger 1999) and periodical harvest (Dosskey et al. 2010), in order to minimize *in situ* nutrient recycling (Dorioz et al. 2006) and to prevent soil P saturation and subsequent leaching (Vought et al. 1994). Unfortunately, current

RBS policy in Quebec prohibits harvest of over 50% of shrub stems (or of trees over 10-cm diameter), which concerns willow plantations (MDDEP 2005). Based on our results, revision of current policy in favor of better maintenance and periodical harvest of mature woody RBS would allow for more efficient, permanent reduction of agricultural nutrients to prevent leaching out of fields.

The absence of differentiation between herbaceous and woody RBS in nutrient retention potential efficiency may have several possible explanations. Firstly, it may simply stem from the fact that both herbaceous vegetation and woody litter can reduce erosion, sediment and chemical transport to a same extent in runoff water (Uusi-Kämppä and Yläranta 1996; Uusi-Kämppä et al. 2000; Udawatta et al. 2002; McKergow et al. 2006; Dosskey et al. 2007; Dosskey et al. 2010; Sabater et al. 2003). Secondly, herbaceous vegetation that colonized the field and stream edges of experimental willow RBS may have stabilized the surface soil and retained sediments better than would a willow RBS with bare soil and no weeds under their canopy, thus minimizing differences between herbaceous RBS and woody RBS efficiencies in the current experiment (Figure 2- 5). Indeed, total understory vegetation eradication in woody RBS enhances soil erosion and TSS runoff compared to grassed RBS (McKergow et al. 2006). Our observations thus suggest that herbaceous vegetation favored PO_4^{3-} infiltration into the ground (or lower mitigation potential efficiency in deeper waters) while willows better mitigated subsurface flows (Figure 2- 5). Indeed, willows absorbed nitrate from a deeper root zone (BB: 70 cm; SR : 35-70 cm) than herbaceous vegetation, which intercepted nitrate from a shallow root zone (BB: 35 cm, ns trend), supporting Lyons et al. (2000) who suggested that grassy buffers best intercept dissolved nitrogen in runoff (Lyons et al. 2000). A possible explanation is that greater stem density of herbaceous RBS further slows runoff and favors sediment deposition (Dosskey et al. 2010). However, willows removed more P_{tot} (near significant) from runoff than herbaceous RBS, at SR (Figure 2- 5). Hence, greater erosion, sediment and surface P transport prevention by grassy RBS (Lyons et al. 2000) may apply to runoff at BB, but not to runoff at SR. Similarly, discrimination between grassy and woody RBS efficiencies in groundwater nutrient retention was inconsistent across different sites in several studies (Dosskey 2001; Correll 1996; Dosskey et al. 2010; Lyons et al. 2000). Contrary to absence of fully differentiated treatments in the current study, Gasser et al. (2013) noted that P_{tot} and PO_4^{3-} were consistently, highly reduced in soil water (20-40 cm) with *Phalaris* cultivated in a swale (> 10 µg·L^{-1}), compared to different setups of *S. miyabeana* treatments (between 15 and 20 µg·L^{-1} under 5 rows with or without swales). In addition, *S. miyabeana* could remove 90% of N and 85% of P from wastewater (Guidi Nissim et al. 2015), hence the limited potential efficiency of willow in nutrient retention from runoff in the present study may have been due to multiple, unidentified factors and not only to its nutrient absorption capacity.

Finally, plant morphology and physiology may also provide some explanation. The influence of herbaceous vegetation morphometry and ecological characteristics was further explored, together with *Salix* and hydrological variables, to understand their combined impact on nutrient reduction efficiency of the RBS (see Annexe 25). The redundancy analysis (Legendre and Legendre 2012, Lepš and Šmilauer 2003) revealed that several individual herbaceous characteristics were correlated to nutrients concentrations. For instance, hydrophytes abundance was correlated with lower N_{tot} and PO_4^{3-} in runoff (see Annexe 25). Herbaceous vegetation with tap roots correlated with infiltration of dissolved nitrogen (Annexe 25), which is consistent with the enhanced infiltration induced by plants with tap roots (Reubens et al. 2007). In our experiment, the herbaceous RBS were largely covered by plants with tap roots (Annexe 13). The extensive, fibrous root system of willows (Kuzovkina and Volk 2009) is known to reach heterogeneous, P-rich subsurface soil patches (Dunbabin et al. 2004). Willows' greater evapotranspiration potential compared to herbaceous vegetation (Tabacchi et al. 2000) could also enhance their capacity to draw nutrients from underground.

Contrary to the results obtained from herbaceous vegetation variables taken separately, grouping them (using their first principal component), led to no clear evidence of their influence on nutrient concentrations, at any season and depth surveyed (see Annexe 25). This suggests that some of the herbaceous variables surveyed influenced nutrient concentrations differently. This may mask the potential efficiency distinctions between the different treatments, perhaps via compensating mechanisms (i.e. sometimes the herbs plot display more characteristics considered beneficial for agro-chemicals mitigation, sometimes the *Salix* plots do, in a continuum). It is known that RBS over story influences the herb-layer, through altering light availability and soil fertility, and in turn, the low strata influences the woody species through intensive competition and pre-emption of resources such as nutrients (Gilliam 2007).

2.4.5 Observance of water quality criteria

Criteria for water pollution prevention and aquatic life protection against chronic exposure are set, for the Province of Quebec, at 10 $mg \cdot L^{-1}$ NO_3-N, 1.5 $mg \cdot L^{-1}$ NH_4-N and 0.030 $mg \cdot L^{-1}$ (MDDELCC 2013). Levels of NO_3^- in water draining from the fields were lower than the norm throughout the year at SR. At BB, runoff concentrations of NO_3^- were sometimes twice as large as those authorized at the edge-of-field but satisfactory after the RBS (Figure 2- 4). The NH_4^+ threshold was regularly surpassed at SR and water concentrations that satisfied the criteria at the edge-of-field just after application of glyphosate sometimes exceeded the criteria once through the RBS (Figures 2- 4 and 2-5). At BB, snowmelt effluents NH_4^+ concentrations exceeded the criteria, despite a 3-time reduction in concentration across the RBS. Still in BB runoff, NH_4^+ exportations exceeded the norm by one order of magnitude at the post-fertilization stage when the RBS were inefficient, and phreatic waters contained excessive NH_4^+ concentrations after fertilization and glyphosate applications, but not at snowmelt (Figures 2- 4 and 2-5). Within the RBS, NH_4^+ may become oxidized into NO_2^- and NO_3^- where oxygen is abundant, but if the roots favor infiltration to the deeper soil layers where oxygen is less abundant, it could accumulate in the phreatic zone (Jones and Mulholland 1999). Within the RBS, some plants may uptake NH_4^+ (especially near the surface),but preferential NO_3^- uptake is predicted based on this N-source higher availability in the RBS root zone (Figure 2- 1; Aerts & Chapin 1999). Should NH_4^+ become oxidized into NO_2^- and NO_3^-, this would not only favor plant absorption, but could also favor denitrification (carbon exudates by plants favor this process in the RBS) (Vidon & Hill 2004). Hence, while the RBS may influence nitrogen speciation, perhaps leading to oxidation of NH_4^+ and subsequent potential for the NO_3^- criteria of being surpassed, the RBS vegetation plays two important roles, direct N absorption and positive influence on denitrification, which make it an interesting mean to mitigate N pollution.

Despite spatio-temporal variability, edge-of-field's PO_4^{3-} effluents always exceeded the P threshold, at both sites (Figures 2- 4 and 2- 5). After fertilization, at BB, edge-of-field concentrations were over 66 times excessive, and remained 29, 122 and 198 times in excess after the CX, 3X and 5X RBS, respectively. At SR, edge-of-field concentrations were more than 21 times above the criteria, and exceeded the criteria by more than 39, 14 and 7 times compared to the norm after the CX, 3X and 5X treatments, respectively. Therefore, the different treatments potential efficiency were not ordered the same way on both sites. These observations were in line with the extent of P water pollution in agricultural areas of Quebec (Beauchemin et al. 1998). Earlier Quebec studies revealed excessive P_{tot} leaching in more than 50% of fields surveyed, a particularly important issue in clayey soils (Beauchemin et al. 1998), like those at SR. While only inorganic fertilizers were used under reduced tillage at BB, at SR farmers switched to no till early in the study and amended the soil with sludge and pig slurry rich in P_{tot} and PO_4^{3-} (Table 2- 2). Streams did not always satisfy water quality guidelines for NO_3^- (BB: 1.08-1.65 mg/L; SR: 1.9-12 mg/L) (Table 2- 3) (MDDELCC 2013). Quebec water quality guidelines were observed for NH_4^+ in streams (BB: 0.010-0.350 $mg \cdot L^{-1}$; SR: 0.025-0.125 $mg \cdot L^{-1}$). While these guidelines may protect fish from acute toxicity for freshwater species (2.79 mg $NH_3 \cdot L^{-1}$) and seawater species (1.86 mg $NH_3 \cdot L^{-1}$; Randall and Tsui 2002), some authors claim that chronic toxicity for fish may already be initiated at 0.2 mg $NH_3 \cdot L^{-1}$ (Daoust and Ferguson 1984; EPA 2003). Despite dilution of nutrients originating from the land into large water volumes, PO_4^{3-}-loaded streams at BB ($\leq$ 0.202 $mg \cdot L^{-1}$) and SR ($\leq$ 0.286 $mg \cdot L^{-1}$) (Table 2- 3) may have largely exceeded criteria for protection against chronic exposure of aquatic life.

The fact that stream concentrations exceeded regulatory thresholds is uninformative with respect to the potential efficiency of our experimental RBS, which were not continuous throughout the watershed. It nevertheless reflects flawed nutrient management, especially when considering the eutrophic waters at BB (Table 2- 3).

2.5 Conclusion

The current study suggests that the 3 m wide RBS recommended by a Québec policy are insufficient to preserve waters ressources from fertilizers exportations in agricultural settings. The nutrients which are more concentrated in surface runoff than in interstitial waters are attenuated during vertical infiltration through the soil column even though interstitial waters became increasingly charged with cations as they percolated through the mineral matrix. The vegetated RBS occasionally intercepted some nutrients from surface or sub-surface flows. However, nutrient concentration reduction was far from being consistent across sites and across a range of nutrients surveyed, for instance with better interception of nitrate than ammonium; better interception of total phosphorus than dissolved phosphates; and some significant interception of K^{+}, Mn^{2+} and Zn^{2+} but not for other cations surveyed. Seasons also strongly affected RBS potential efficiency, with most potential efficiency observed on most concentrated nutrient waters, when the RBS were actively growing, just after sowing and fertilization. Despite their nutrient sequestration potential exportable via wood harvest for biomass production, fast growing willow did not improve nutrient interception based on monitored water concentrations. The limited potential efficiency of the RBS following glyphosate-based herbicide applications also requires further investigatiion of potential interactions between nutrients and glyphosate. Finally, narrow vegetated RBS alone could not suffice to meet water quality guidelines, urging the need to reinforce the current RBS policy in Québec, perhaps by encouraging wider strips or the use precision buffers where hydrological studies suggest that reinforced efforts are necessary. This issue, in particular, requires immediate action in a context of ever-increasing pig slurry and sewage sludge production and land-application by Quebec's industrial agriculture, which raises alarming environmental problems.

2.6 Tables and Figures

Table 2- 1: Study site characteristics and soil analyses in the fields of Boisbriand and Saint-Roch-de-l'Achigan, based on accredited agronomic laboratory analyses.

	Boisbriand	Saint-Roch-de-l'Achigan
Stream name	Dumontier	Moïse-Dupras
Closest weather station	Ste-Thérèse west (4.7 km)	L'Assomption (13.8 km)
Mean annual temperatures	7.5 ± 0.3 °C	7.0 ± 0.8 °C
Degree days of growth	990 ± 7 °C·d	989 ± 7 °C·d
Annual precipitation	1034 ± 84 mm	1121 ± 92 mm
Coordinates	N 45° 36' 39.8"; W 73° 51' 40.3"	N 45° 50' 48.3"; W 73 ° 36' 16.7"
Elevation	44 m	46 m
Topography	Hilly	Flat
Water table depth from ground surface		
Global average (n=18)	0.63 ± 0.16 m	1.56 ± 0.14 m
Snowmelt (2012)	0.09 ± 0.25 m	0.66 ± 0.18 m
End of summer (2012)	0.59 ± 0.20 m	1.87 ± 0.13 m
Soil classification[1]	Organic-rich black soil, typical humisol	Mineral sandy clay-loam soil sitting atop a clay bed
Soil stratigraphic descriptors (from top to bottom, decomposition according to von Post Scale)[2]	Black histosol (BL; strongly decomposed) Brown histosol (BR; less decomposed) Peat (PE; lightly decomposed) Till (TI) Marl (MA) Grey clay (GC) Reddish clay (RC)	Sandy loam (SL), Clean sand lentils (CS), Grey clay (GC) Reddish clay (RC)
Coarse sand (< 2 mm)	6.1%	43%
Fine sand (< 212 μm)	13%	30%
Silt and clay (<63 μm)	81%	27%
pH (water)[3]	6.6	6.6 - 7.1
pH (buffer)[3]	7.0	7.0 - 7.2
OM (%)[3]	4.5	2.1 - 4.4
CEC (meq/100g)[3]	0.03-2	10.2 - 17.9
P saturation (% P/Al)[3]	12.9	5.4 - 7.6
P-Mehlich (kg/ha)[3]	297	129 - 239
K-Mehlich (kg/ha)[3]	569	90 - 147
Ca-Mehlich (kg/ha)[3]	6395	2057 - 6263
Mg-Mehlich (kg/ha)[3]	1-10	147 - 289
Al-Mehlich (ppm)[3]	nd	877 - 1407
Fe-Mehlich (ppm)[3]	nd	241 - 314

Notes : [1] Soil Classification Working Group (1998); [2] (Soil Classification Working Group 1998) [3]Field soil was sampled on 2012-11-16 at BB and on 2013-05-02 at SR, and analyzed by AgroEnviroLab, La Pocatière, QC, Canada, accredited by CEAEQ and ISO-CEI 17025.

Table 2- 2: Agricultural practices in Boisbriand and Saint-Roch-de-l'Achigan from 2011 to 2013.

Year	Boisbriand				Saint-Roch-de-l'Achigan			
	Culture Date; Seed Variety (Characteristics)	**Fertilization** Date; Dose/Composition (N-P-K)	**Herbicide** Date; Label; Glyphosate Dose	**Yield** (tm/ha)	**Culture** Date; Seed Variety (Characteristics)	**Fertilization** Date; Dose/Composition (N-P-K)	**Herbicide** Date; Label; Glyphosate Dose	**Yield** (tm/ha)
2011	2011-06-10: Soy PS-02-42[1] (RR)	2011-06-10: 145 kg/ha (13-20-20) + 0.21 kg/ha Mg + 0.84 kg/ha Ca	2011-07-08: Factor 540[3] 1.13 kg/ha[4]	3.65	2011-06-04: Soy (RR)[5]	None	2011-07-04: Gardien[10] 0.72 kg/ha	2.76
2012	2012-05-07: Maize 38A85[2] (Bt and RR)	2012-05-07: 336 kg/ha granules (16.6-24.9-12.5) + 359 kg/ha slow release urea (34.4-0-14.3)	2012-06-06: Factor 540[3] 1.13 kg/ha[4]	10.5	2012-05-17: Maize Pride A5840 and A5365[5] (RR)	2012-05-14: 210 kg/ha (33.5-0-16.3) + stabilized sludge[6] (15 mt ww/ha) 2012-05-17: 278 kg/ha (7.9-21.5-10) 2012-06-20: 88 kg/ha (46-0-0)[7]	2012-06-11: Polaris + Ultime[11] 0.89 kg/ha	8.6
2013	2013-05-18: Soy PS-02-42[1] (RR)	2013-05-18: during sowing 23-14-0 ProK from Synagri 157 kg/ha	2013-06-21: Factor 540[3] 1.13 kg/ha[4]	3.75	2013-05-10: Maize N24A[5] (RR)	2013-05-07: pig slurry (45 m^3/ha)[8] 2013-05-10: 143 L/ha (23-14-0)[9]	2013-06-14: Galaxie II[12] 0.81 kg/ha	4.4

Notes: [1]Pride, Chattam, ON, Canada; [2] DuPont Pioneer Hi-Bred Ltd, IA, USA; [3] IPCO Interprovincial Cooperative Ltd, MA, Canada; [4]One liter of 540 $g \cdot L^{-1}$ concentrated glyphosate was diluted in 50-L water and applied over 1 acre (≈0.405 ha); [5]Agro-Lanaudière Ltd (QC, Canada); [6] Based on independent agronomic laboratory analysis, the stabilized sewage sludge contained 8 280 - 9 140 mg $P \cdot kg\ dw^{-1}$ and 12 000 - 18 000 mg $Al \cdot kg\ dw^{-1}$ [7]North-East corner of field received less sludge so amendment was corrected with a higher dose of N in June (117 $kg \cdot ha^{-1}$) as per agronomist recommendation; [8] Based on our analysis, the pig slurry contained ≥13 $mg \cdot L^{-1} PO_4$d. [9]Liquid fertilizer distributed by Agro-100; [10]Gardien, Du Pont Canada, Mississauga, ON, Canada: Classic (Chlorimuron-ethyl 36$g \cdot L^{-1}$) + Polaris (isopropylamine salt of glyphosate 360 g/L), 2.5-L concentrate diluted and spread at 150 $L \cdot ha^{-1}$; [11]2.47-L concentrate diluted and spread at 150 $L \cdot ha^{-1}$; [12]Galaxie II: Ultime (37.5% rimsulfuron and 37.5% nicosulfuron) + Polaris (isopropylamine salt of glyphosate 360 g/L), 2.25-L concentrate diluted and spread at 150 $L \cdot ha^{-1}$.

Table 2- 3: Nutrient concentrations in various compartments surrounding the riparian buffer strip.

Mean ± standard deviation is given wherever more than one sample was analyzed. Sampling periods include snowmelt (SM), post-fertilization (PF) and post-glyphosate (PG).

Sampling period	Site	Sample	PO_4^{3-} (µg/L-P)	$NO_2^-+NO_3^-$ (µg/L-N)	NH_4^+ (µg/L-N)	NO_2^- (µg/L-N)	P_{tot} (µg/L-P)
SM	BB	Snow	7.0	252.2	105.1	1.1	nd
		Surface runoff	21.7	1536.4	380.5	35.6	nd
		Stream	3.9 ± 2.3	1082.7 ± 16.3	25.3 ± 20.0	2.4	nd
	SR	Snow	2.9	24.0	81.3	1.0	nd
		Surface runoff	214.9 ± 16.0	36.9 ± 5.6	168.6 ± 10.4	2.2 ± 0.3	nd
		Stream	6.7	3742.9	24.2	3.4	nd
PF	BB	Rain	14.1	317.7	793.6	4.8	nd
		Stream	27.4	1642.9	340.3	7.9	202.6
	SR	Rain	16.2 ± 9.3	688.5 ± 642.2	1802.8 ± 1179.4	17.6 ± 15.1	nd
		Stream	82.7 ± 61.8	3479.2 ± 1417.9	87.1 ± 39.9	34.9 ± 20.0	286.2
		Liquid fertilizer	3.5E+07	1.1E+08	1.3E+08	0.0	nd
		Pig slurry	13230.4	179.5	1.6E+06	54.8	nd
PG	BB	Stream	2.4	1195.0	32.8	0.0	2.6
	SR	Stream	13.1	1.2E+04	30.1	0.0	29.6
		Granular fertilizer	6.3	2.6E+05	653.4	0.0	0.1

Table 2-3 : (continued)

Sampling period	Site	Sample	K^+ (µg/mL)	Mg^{2+} (µg/mL)	Mn^{2+} (µg/mL)	Na^+ (µg/mL)	Zn^{2+} (µg/mL)	Ca^{2+} (µg/mL)	Fe^{2+} (µg/mL)	Al^{3+} (µg/mL)	DOC
SM	BB	Snow	0.09	0.29	0.00	0.00	0.00	2.65	0.00	0.06	7.53
		Surface runoff	8.88	6.94	0.00	1.78	0.01	44.57	0.15	0.04	38.48
		Stream	3.29 ± 0.27	31.87 ± 2.16	0.03 ± 0.05	13.78 ± 0.45	0.00 ± 0.00	74.69 ± 16.96	0.09 ± 0.13	0.00 ± 0.00	5.04
	SR	Snow	0.03	0.19	0.00	0.00	0.00	45.70	0.00	0.00	1.83
		Surface runoff	3.60 ± 1.06	1.21 ± 0.45	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	3.02 ± 0.65	0.18 ± 0.12	0.09 ± 0.12	12.05 ± 4.18
		Stream	3.39	11.39	0.00	4.81	0.00	0.41	0.06	0.00	6.77
PF	BB	Rain	0.12	0.00	0.00	0.00	0.00	0.76	0.10	0.21	4.66
		Stream	3.35	32.65	0.00	12.80	0.04	101.56	0.12	0.07	0.00
	SR	Rain	0.43 ± 0.47	0.42 ± 0.62	0.00 ± 0.00	0.00 ± 0.00	0.06 ± 0.10	2.34 ± 2.64	0.07 ± 0.06	0.00 ± 0.00	7.53 ± 6.32
		Stream	5.02 ± 0.53	22.31 ± 2.90	0.02 ± 0.04	7.98 ± 0.54	0.00 ± 0.00	64.27 ± 2.08	0.06 ± 0.06	0.12 ± 0.21	13.25 ± 4.12
		Liquid fertilizer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		Pig slurry	nd	nd	nd	nd	nd	nd	nd	nd	6948.00
PG	BB	Stream	3.11	33.69	0.00	13.15	0.00	39.29	0.00	0.00	6.50
	SR	Stream	6.85	21.44	0.00	8.71	0.00	96.04	0.00	0.00	3.61
		Granular fertilizer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

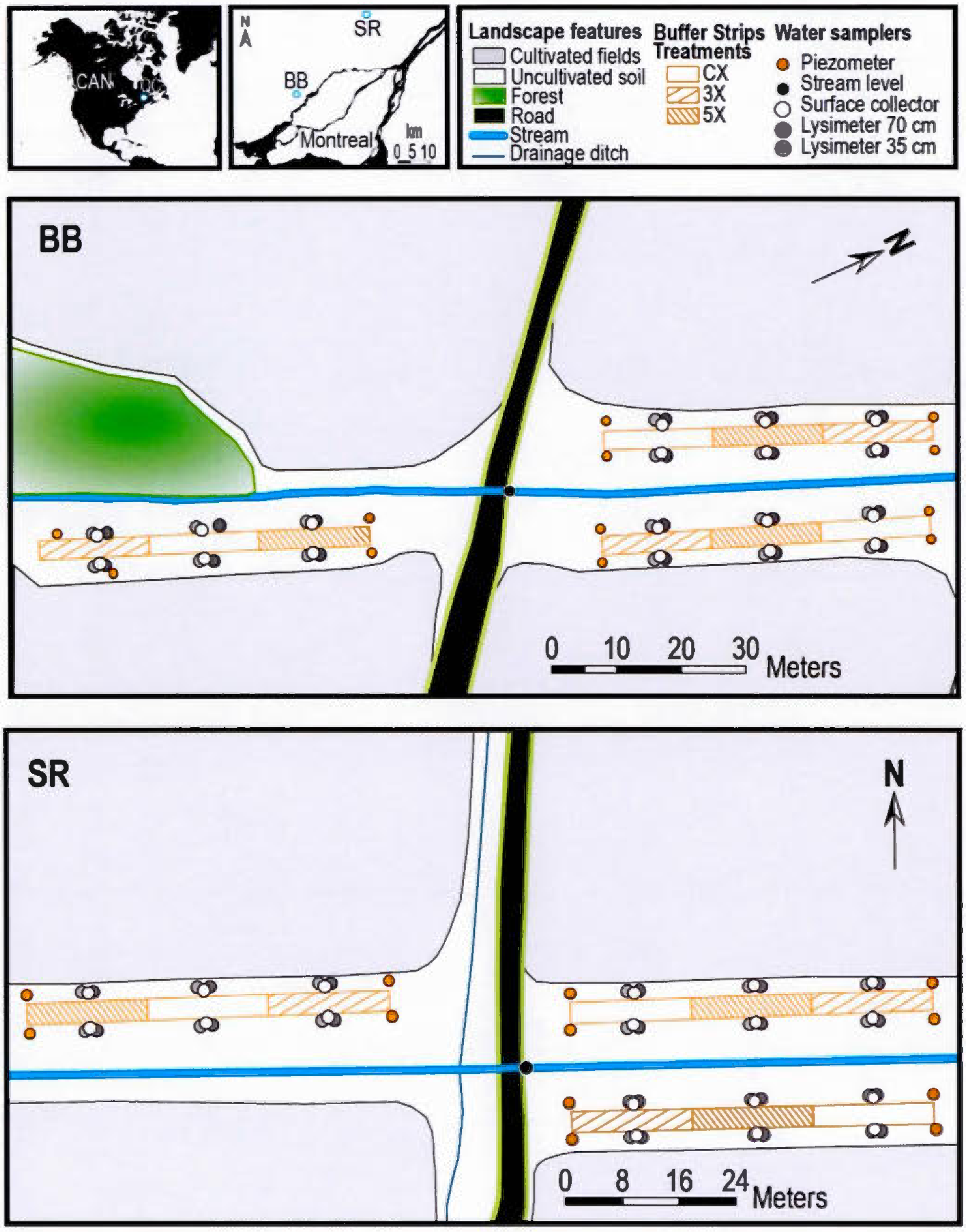

Figure 2- 1: Location of research sites north of Montreal, in Quebec, Canada, with landscape features, treatments and sampling equipments for Boisbriand (BB) and Saint-Roch-de-l'Achigan (SR).

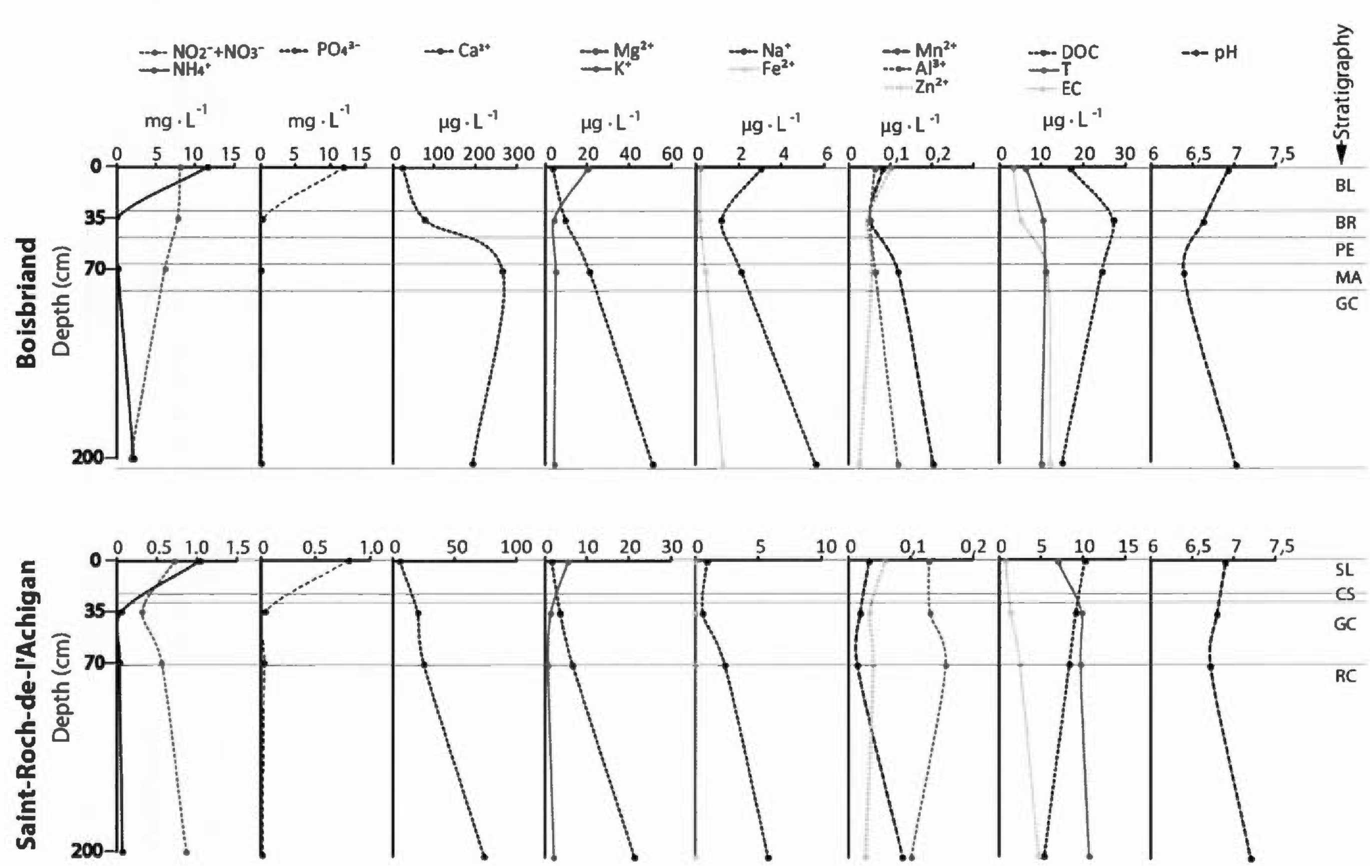

Figure 2- 2: Depth profiles for macronutrients, micronutrients and physicochemical properties of water collected on the edge of the fields (global means on CF side), along the RBS, at snowmelt and post-fertilization stages, in relation to the generalized stratigraphy.

Total number of samples analyzed from 2011 to 2014 is 197 at BB (0, 35, 70 and 200 cm: 33, 59, 63, 42 samples per depth) and 184 at SR (45, 49, 59, 31 samples per depth). Stratigraphic descriptors are detailed in Table 2- 1.

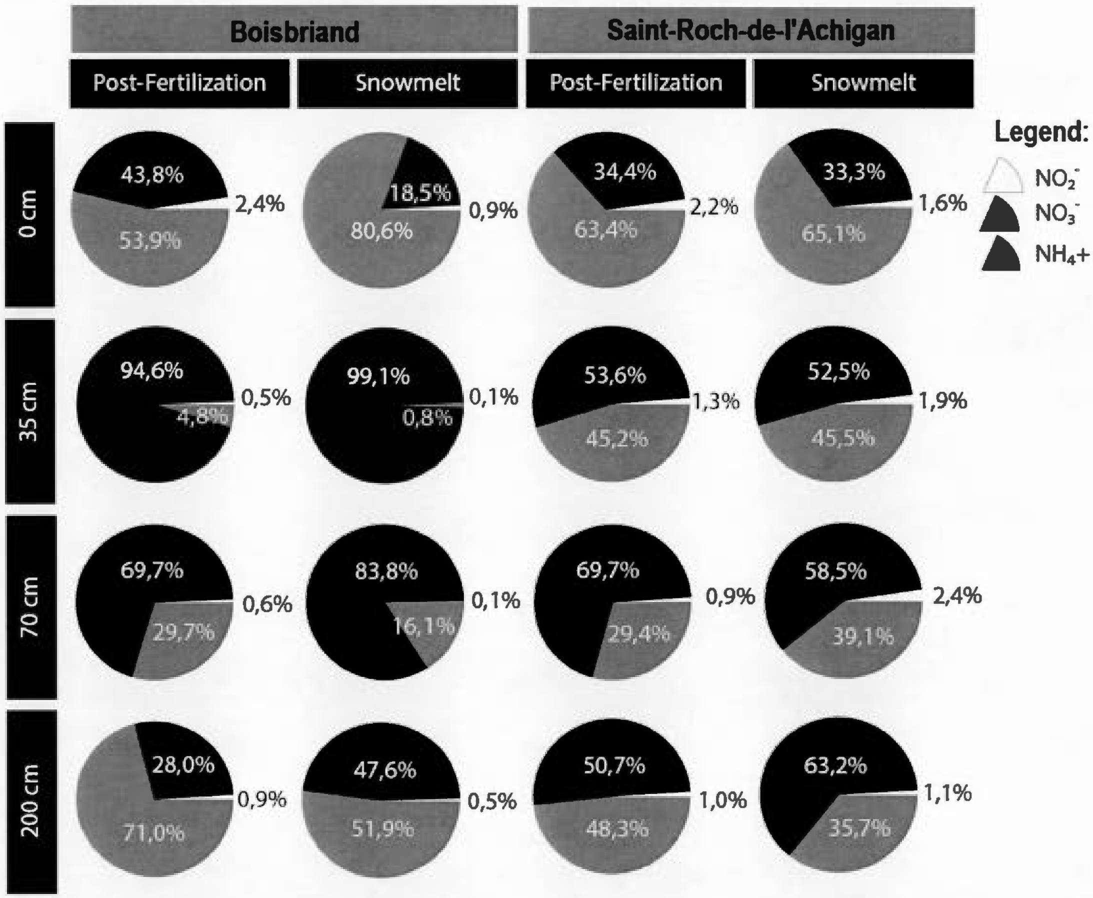

Figure 2- 3: Relative importance of dissolved nitrogen species at post-fertilization and snowmelt stages along vertical gradient, at Boisbriand and Saint-Roch-de-l'Achigan experimental sites.

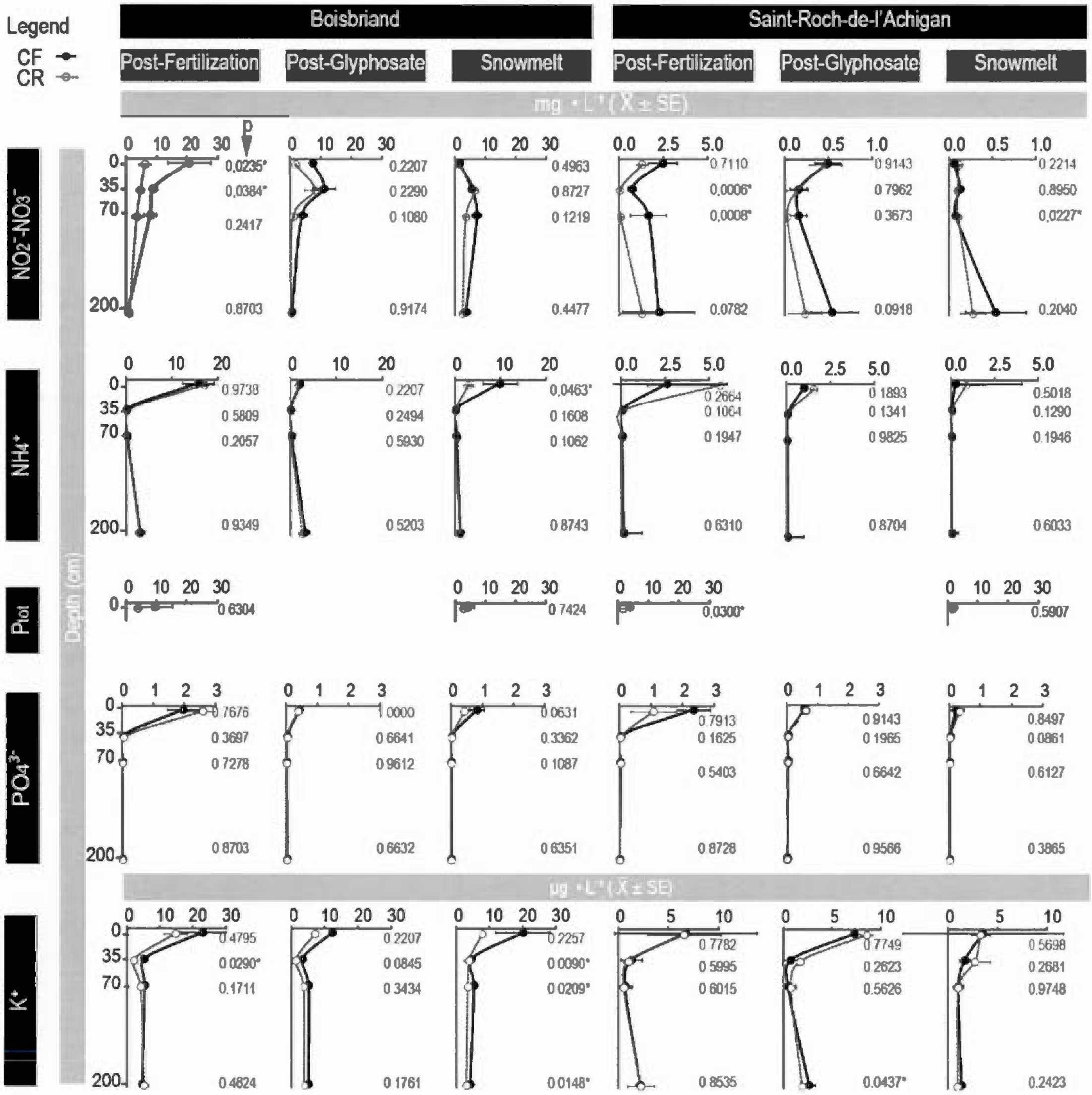

Figure 2- 4: Nutrient (N, P, K) concentrations ($mg \cdot L^{-1}$; mean $\pm$ SE) in Boisbriand and Saint-Roch-de-l'Achigan measured at distinct agricultural events (snowmelt, post-fertilization and post-glyphosate) and presented as depth profiles before (CF) and after (CR) the buffer strip.

The probabilities (look for p with arrow pointing down) are given next to each CF-CR pair of data. Significant figures ($p \leq 0.05$) are marked with an asterisk (*). Ptot was not determined post-glyphosate.

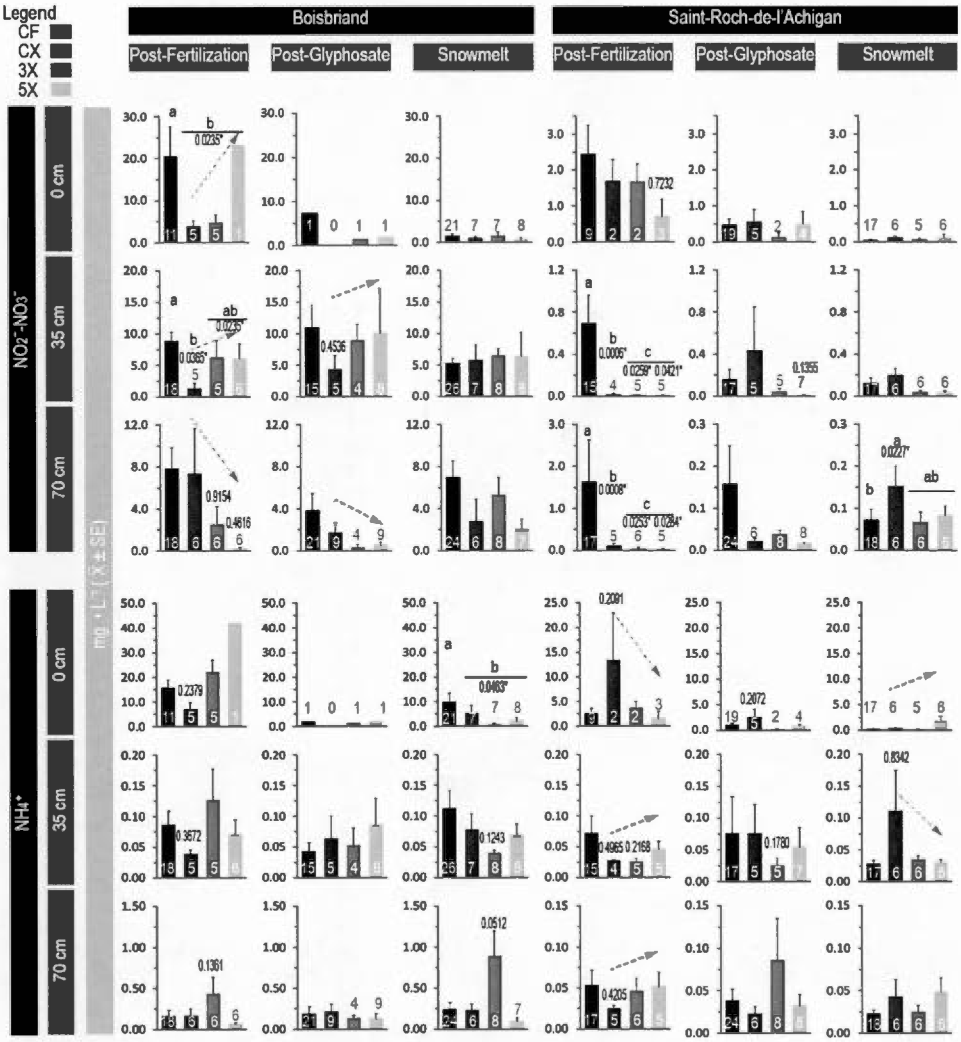

Figure 2- 5a: The effect of RBS treatments (CX, 3X, 5X) on nutrient concentrations (Nitrogen, mg·L^{-1}; mean± SE) in Boisbriand and Saint-Roch-de-l'Achigan, measured at distinct agricultural events (snowmelt, post-fertilization and post-glyphosate) and at different depths.

The edge-of-field (black) and three treatments (CX: herbaceous, 3X: low-density willow and 5X: high-density willow) are presented in the same order from left to right and identified by shades of grey in the legend. Where CF and CR were statistically different ($p < 0.05$) or showed a near significant trend ($p < 0.10$), a *post hoc* Dunnet test was carried out with CF as the control, to verify treatment effects between CX, 3X and 5X treatments. Letters indicate significant differences between treatments, as per the *post hoc* steel test with the edge-of-field (CF) as the control. Above the bars, *p* values (p≤0.05) are reported wherever data is visually confounding, and significant p values are marked with an asterisk (*). Numbers on the bars represent the number of samples. Dashed arrows are presented where potential efficiency reversal trends were observed along the depth profile.

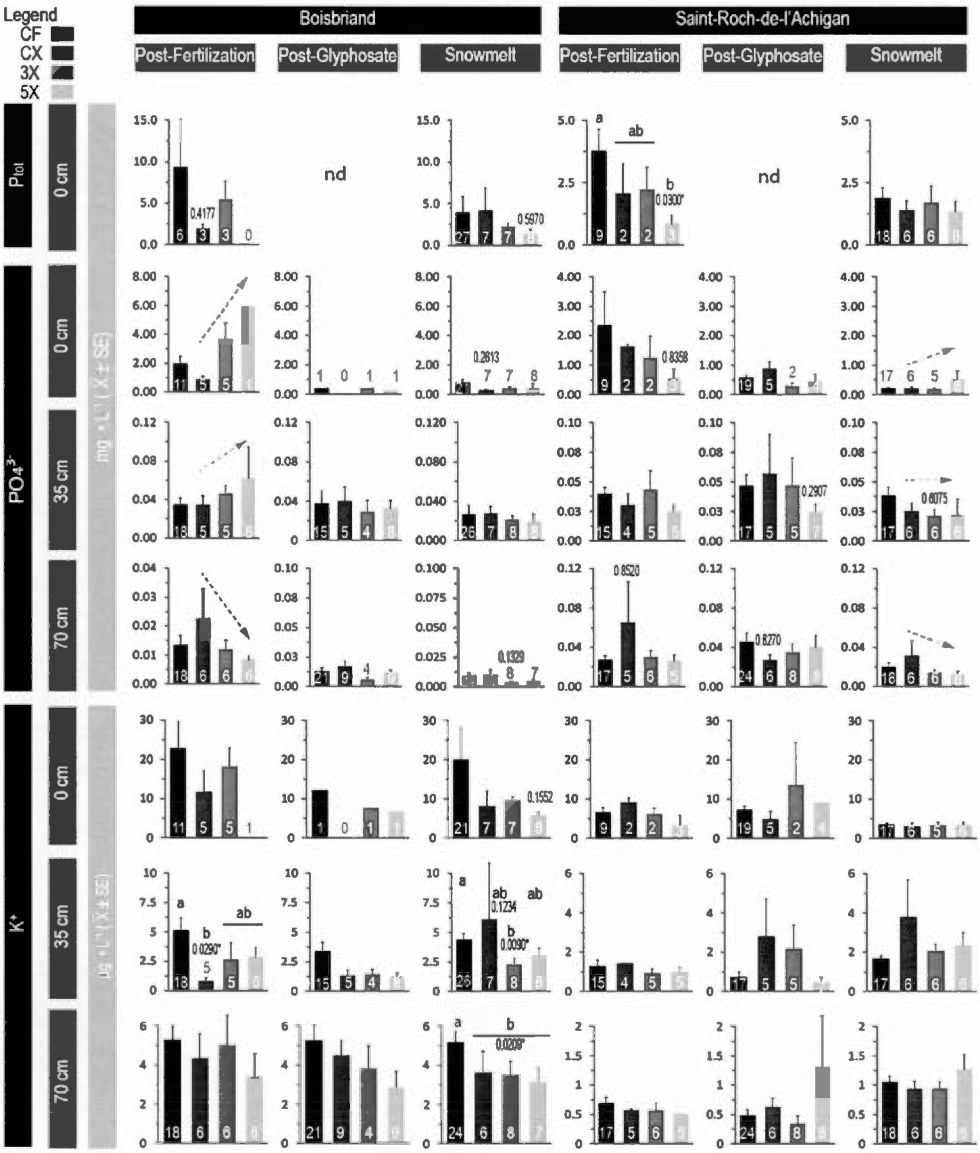

Figure 2-5b: The effect of RBS vegetation treatments (CX, 3X 5X) on nutrient concentrations (Phosphorus & Nitrogen, mean± SE) in Boisbriand and Saint-Roch-de-l'Achigan, pooled by agricultural events.

The edge-of-field (black) and three treatments (CX: herbaceous, 3X: low-density willow and 5X: high-density willow) are presented in the same order from left to right and identified by shades of grey in the legend. Where CF and CR were statistically different ($p < 0.05$) or showed a near significant trend ($p < 0.10$), a *post hoc* Dunnet test was carried out with CF as the control, to verify treatment effects between CX, 3X and 5X treatments. Letters indicate significant differences between treatments, as per the *post hoc* steel test with the edge-of-field (CF) as the control. Above the bars, *p* values ($p \leq 0.05$) are reported wherever data is visually confounding, and significant p values are marked with an asterisk (*). Numbers on the bars represent the number of samples. Dashed arrows are presented where potential efficiency reversal trends were observed along the depth profile.

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CHAPITRE III

HERBACEOUS OR *SALIX MIYABEANA* SX64 NARROW BUFFER STRIPS AS A MEANS TO MINIMIZE GLYPHOSATE LEACHING FROM ROW CROP FIELDS

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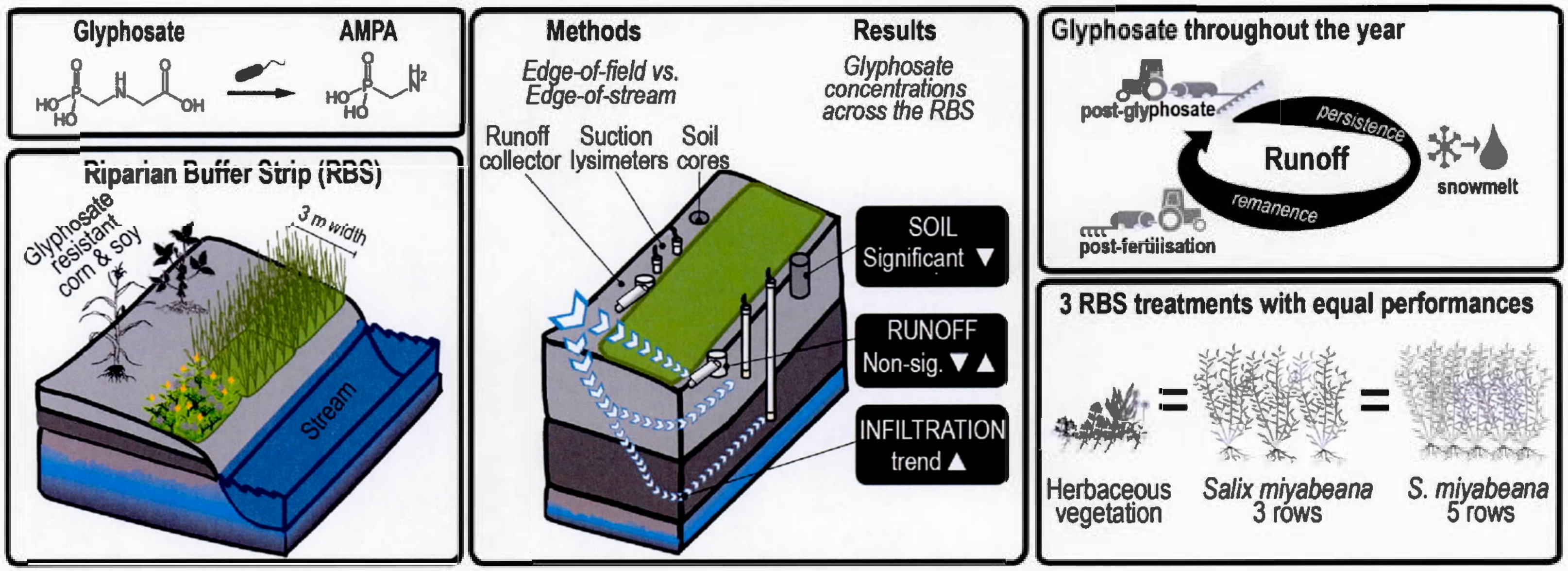

Highlights

- Narrow vegetated riparian buffer strips (RBS) reduce glyphosate loads in surface soils.
- Highly variable potential efficiency of RBS for reducing glyphosate and AMPA leaching.
- Spontaneous herbaceous vegetation RBS is as efficient as *Salix miyabeana* plantations.
- Glyphosate in runoff may not be more concentrated post-glyphosate than post-fertilization.
- RBS may favor glyphosate infiltration to 70cm depth.

Abstract

Glyphosate is the most widely used herbicide, frequently detected in surface waters of agricultural regions, around the world and in Quebec (Canada). Numerous legislations require vegetated riparian buffer strips (RBS) along agricultural streams. Quebec provincial policy requires 3-m-wide RBS. The current research studies the efficiency of narrow herbaceous and low or high density (33 333 and 55 556 stumps/ha) willow RBS, *Salix miyabeana* SX64, to minimize leaching and infiltration of glyphosate and its main degradation product (AMPA) from agricultural fields to streams. Our studies compared triplicate treatments of herbaceous and willow planted RBS located in an organic-rich soil at Boisbriand (BB) and in compacted mineral soil at Saint-Roch-de-l'Achigan (SR). Runoff water was sampled with surface collectors and interstitial water was collected with 35 cm or 70 cm tension lysimeters. Potential efficiency of the RBS is reported as the percent reduction between edge-of-field and edge-of-stream concentrations. Although glyphosate persistence was demonstrated, mean edge-of-field runoff concentrations were lowest at spring melt ($\leq$ 2.4 $\mu g \cdot L^{-1}$). Yet, they nearly doubled after applications of fertilizers and glyphosate at BB (3.4 - 3.7 $\mu g \cdot L^{-1}$) and increased ten folds at SR (20 $\mu g \cdot L^{-1}$). Neither glyphosate nor AMPA in runoff were significantly intercepted by the RBS. After field herbicide spraying, glyphosate measured in SR surface soils (0-20 cm) was on average 210 $\mu g \cdot kg^{-1}$ dw (range from undetected to $\leq$317 $\mu g \cdot kg^{-1}$ dw). Contrary to runoff trends, soil glyphosate was significantly less concentrated on the SR edge-of-stream compared to edge-of-field (27-54% potential efficiency). Loss of correlation between glyphosate and PO_4^{3-} across the RBS may stem from competition for soil adsorption sites. Increased correlations with Ca^{2+}, Al^{3+} and Na^{+} may originate from a complexation hot spot within the RBS. The potential efficiency of herbs, low and high density willow RBS treatments were undifferentiated.

Keywords

Glyphosate, Aminomethyl phosphonic acid (AMPA), vegetated riparian buffer strips, *Salix miyabeana* SX64, corn and soy fields, runoff

Abbreviations

Riparian buffer strips (RBS); Boisbriand (BB); Saint-Roch-de-l'Achigan (SR); genetically resistant (GR); aminomethyl phosphonic acid (AMPA); soy (S); maize (M); dissolved organic carbon (DOC); close to the field-edge (CF); close to the stream-edge (CR); gas chromatograph – electron capture detector (GC-ECD); aqueous (aq); total suspended solids (TSS); pesticide (P)

3.1 Introduction

3.1.1 Glyphosate uses and toxicity

Agriculture uses 60-80% of the world's pesticides (EPA 2011; Health Canada 2011; OECD 2013). Herbicides account for 40% of pesticides (EPA 2011). Glyphosate, first sold as Roundup by Monsanto in 1974 (EPA 2009a), now dominates the world herbicide market (Health Canada 2011; Environment Canada 2011; EPA 2011; Eurostat and European Comission 2007), with 400 formulations used on >400 food crops (EPA 2012). This non-selective herbicide associated with genetically resistant (GR) soy and maize (EPA 2012) is also used as a pre-harvest desiccating agent for non-genetically resistant grains such as wheat (Nader et al. 2013; Jaskulski and Jaskulska 2014). In Quebec (Canada), it ranks 1st for treated surfaces (1.6 million ha/year; Giroux and Pelletier 2012) and the phosphonic acid family ranks 1st in sales (1 388 263 kg active ingredient$\cdot$yr^{-1}; Gorse and Balg 2012). Increasing GR crop production, parallels increasing prevalence in surface water contamination (reaching 97.5% in surveyed soy and maize regions of Quebec; Giroux 2015). However, glyphosate concentration in surface runoff leaching from corn and maize fields in Quebec is unknown. Surface and groundwater contamination is reported worldwide (Aparicio et al. 2013; GEUS 2013; Horth and Blackmore 2009; Litz et al. 2011; Scribner et al. 2007; Struger et al. 2008). The increasing risk for drinking water contamination based on glyphosate's environmental behavior and chemical characteristics should not be overlooked (European Commission 2002; Vereecken 2005). Glyphosate affects weeds and non-target plants (Gomes et al. 2014), threatening indigenous

(Heard et al. 2003) and vulnerable (Matarczyk et al. 2002) plant populations and associated insects (Pleasants and Oberhauser 2013). Inhibition of the 5-enolpyruvyl-shikimate-3-phosphate (EPSPS) synthase enzyme (Boocock and Coggins 1983) induces death by aromatic amino acids starvation in plants (Williams et al. 2000), while also affecting fungi and bacteria (Duke et al. 2012), including gut bacteria essential for human health (Samsel and Seneff 2013). Minimal risk assumptions for mammals, birds and aquatic biota (EPA 2009b) may be misleading. Aquatic communities biodiversity and productivity (amphibians (Relyea 2005); phytoplankton (Pérez et al. 2007)) are impacted by glyphosate even below chronic aquatic toxicity criteria (65 $\mu g \cdot L^{-1}$, Smedbol et al. 2013). Even though Quebec continues to refer to the 65 µg·L-1 criteria, the Canadian council of the ministers of the environment has raised this criteria to 800 $\mu g \cdot L^{-1}$ in 2012 (Giroux 2015). Moreover, it has epidemiologic correlation with a dozen human diseases (Swanson et al. 2014), many of which are supported by metabolic mechanisms (Samsel and Seneff 2013), recently recognized with carcinogenicity (Guyton et al. 2015; IARC 2014).

3.1.2 Glyphosate chemistry and environmental behavior

Glyphosate's behavior has been described extensively in laboratory scale experiments (Bergström et al. 2011; Candela et al. 2007; Dousset et al. 2007; Litz et al. 2011; Zhou et al. 2010), controlled field trials (Aronsson et al. 2011; Candela et al. 2007; Kjær et al. 2011; Laitinen et al. 2009; Landry et al. 2005) or uncontrolled field trials (Kjaer 2005; Laitinen et al. 2009; Simonsen et al. 2008). Several reviews document its environmental leaching and mobility (Borggaard and Gimsing 2008; Giesy et al. 2000; Vereecken 2005), its interactions with phosphate fertilizers (Borggaard 2011), plants and rhizospheric micro-organisms (Duke et al. 2012; Kremer and Means 2009), as well as overall environmental impacts based on cropping systems (Cerdeira and Duke 2010), contamination dispersal over wide geographical areas (Horth and Blackmore 2009) and water treatment removal (Hall and Camm 2007; Jönsson et al. 2013).

Of importance, glyphosate is degraded to non toxic sarcosine by bacteria (Borggaard and Gimsing 2008), but more generally into aminomethyl phosphonic acid (AMPA) which is considered less toxic for aquatic organisms (EPA 2009b) or toxic for the glyphosate resistant crops themselves (Gomes et al. 2015a). Glyphosate's environmental behavior descriptions originate from warmer regions (Central Europe and the USA). Yet, its behavior differs in northern latitudes (Helander et al. 2012). Québec has warm summers but freezing winters, so the low persistence of glyphosate noted elsewhere may not apply to Quebec (Canada) where we could expect rapid degradation in the summer followed by persistance of remaining glyphosate until the following spring. Colder countries such as Northern Europe (Laitinen et al. 2006) may have varying environmental conditions and cultural practices compared to North America (i.e. extensive GR crops culture). Soil composition, microbial activity, climate, timing, tillage and vegetation influence glyphosate leaching (Borggaard and Gimsing 2008). Pesticides (EPA 2003), including glyphosate and AMPA (Borggaard et Gimsing, 2008), may be transported dissolved or particle bound.

3.1.3 Buffer strips potential efficiency in mitigating herbicides

Québec has a policy for the protection of shorelines and littoral and inundating plains, which recommends the use of narrow RBS to protect water resources (MDDEP 2005). RBS potential efficiency to minimize glyphosate export to nearby streams depends on the buffer ability to intercept and attenuate agro-chemicals traveling along the surface or sub-surface pathways (Mayer et al. 2007). Factors controlling RBS efficiency include: edge-of-field concentration, herbicide properties, width, source area ratio, vegetation species, time since establishment and antecedent moisture content (Arora et al. 2010; Krutz et al. 2005; Neary et al. 1993). RBS efficiency may be defined as a global measure of minimizing glyphosate leaching, wherever its adsorption to soil, bacterial degradation and dilution with rain water cannot be discriminated. While some water chemistry studies of RBS efficiency use mass-balance, most (including the current one) rely on input-output concentrations (Hill 2000). RBS efficiency is sometimes defined as the comparison between RBS outputs in presence of a treatment versus that of a control (unplanted, with crops or other), but more generally, it is defined as (inputs-outputs)/inputs (Mayer et al. 2007). This bears the inherent assumption that water flows perpendicularly and horizontally across the RBS (Annexe 4), hence without volumetric quantification of runoff or groundwater, RBS efficiency may be expressed punctually at different depths on a concentration basis.

RBS efficiency is attributed to infiltration, sediment deposition and sorption (based only on organic carbon-water partition coefficient, K_{OC};Arora et al. 2010). Though some dismiss soil organic matter content in controlling glyphosate sorption (Borggaard and Gimsing 2008; Gimsing et al. 2004b) or secondary to cationic binding sites availability, like Fe^{2+} and Al^{3+} (Borggaard and Gimsing 2008; Duke et al. 2012; Sprankle et al. 1975; Yu and Zhou 2005). The RBS may act as a temporary buffer (dilutes contaminants in water, soil or time) or as a definitive sink (irreversible sorption, microbial degradation or plant uptake leading to sequestration, volatilization or decontamination; Krutz et al. 2005). Short soil (Bergström et al. 2011; Duke et al. 2012; Simonsen et al. 2008; Wauchope et al. 2002) and water (Miller et al. 2010; Wauchope et al. 2002) half-life of glyphosate, strongly controlled by microbial degradation (Borggaard and Gimsing 2008; Simonsen et al. 2008) is critical in RBS efficiency, but neglected from some reviews (Arora et al. 2010).

Soil characteristics influence RBS and the efficiency of vegetated ditches, but vegetation's role may be preponderant (Litz et al. 2011; Moore et al. 2008), as plants can uptake organic pesticides (Paterson and Schnoor 1992), including glyphosate (Gomes et al. 2014; Gomes et al. 2015b), and transform or degrade them within their tissues (Dosskey et al. 2010; Juraske et al. 2008; Lin et al. 2008, 2004). On average, RBS decreased herbicide transport by ≥ 27% (Krutz et al. 2005), up to 76% (53-100%) for strongly sorbing pesticides (K_{OC} ≥ 1000 L·kg^{-1}; Arora et al. 2010). However, few studies characterized the effect of vegetation type on RBS herbicide retention (Krutz et al. 2005; Schmitt et al. 1999). Because *Salix miyabeana* SX64 has a demonstrated phytoremediation potential of glyphosate in controlled environments (Gomes et al. 2015), we hypothesize that Salix RBS could play a role in mitigating glyphosate effluents from agricultural fields. The use of fast growing willows in RBS is an innovation worth of being tested, which could improve glyphosate mitigation compared to the commonly used herbaceous RBS. Few studies focused on glyphosate mitigation by the RBS (Syversen and Bechmann 2004). Knowledge on glyphosate mitigation by RBS is essential, especially under uncontrolled field conditions (i.e. without simulated rainfall or physically partitioned runoff parcels), where microtopographic variations influence lateral transport and flow convergence (Arora et al. 2010), and inherent spatio-temporal heterogeneity coupled to multiple biotic and abiotic factors may affect glyphosate leaching and mitigation.

3.1.4 Goals

Due to its distinct climate and agronomic practices, it is critical to quantify glyphosate leaching in runoff samples from fields and narrow RBS efficiency in Québec. Our research will enable decision makers to evaluate whether their RBS policy is likely to improve the water quality of agricultural watersheds. Thus, this study provides supplemental information to constituents charged with conception, implementation and maintenance of narrow RBS, as well as to the provincial and federal government charged with registering pesicides or regulating their uses, to decrease environmental impacts and human exposure.

The present study addresses three questions, which aim to better understand glyphosate's leaching from field and mitigation by the RBS. First, how efficient is the RBS in retaining glyphosate and AMPA from the waters (runoff versus interstitial water), hypothetizing that high density willow RBS are better than low density ones or those composed of spontaneous (naturally recruited, not deliberately introduced) herbaceous vegetation. Second, how efficient is the RBS based on surface soil concentrations of glyphosate. To support this objective, we studied how glyphosate and AMPA leaching at the edge-of-field vary with (a) major agricultural events (snowmelt, post-fertilization and post-glyphosate), and (b), with depth; hypothesizing that concentrations just after glyphosate application and in runoff would be highest.

3.2 Materials and Methods

3.2.1 Study site

The study was conducted in Boisbriand (BB) and Saint-Roch-de-l'Achigan (SR), north of Montreal, Canada. From 2010 to 2013, annual climate was comparable: mean temperatures 7.5 ± 0.3 °C and 7.0 ± 0.8 °C; degree days of growth 990 ± 7 °C·d and 989 ± 7 °C·d and precipitation 1034 ± 84 mm and 1121 ± 92 mm, for BB and SR, respectively. SR (45.84675, -73.60463°; alt. 46 m) has a flat topography, a deep water table and a mineral sandy clay-loam sitting atop a clay bed. BB (45.61106, -73.86119°; alt. 44 m) has a hilly topography, water table is shallow and the buffer strips are established in a typic humisol (Soil Classification Working Group 1998). RBS slopes are >0.5-2 %. This site is thus, representative of organic-rich soils which often accumulate in depressions or low-lying areas around streams (Collins and Kuehl 2000) where RBS are often implemented. Site selection was influenced by the organic-rich soils, which are thought to favor freshwater eutrophication due to their low P binding capacity (Guérin 2009), while flat hydric soils are potentially preferential sites of underground nitrate removal via denitrification (Maître et al. 2005).

From 2011 to 2013, crops rotated between glyphosate resistant soy (S) and maize (M): BB S-M-S and SR S-M-M. Glyphosate was applied once annually with a conventional pneumatic ramp. Spray rate, including water solvent, was approximately 150 $L \cdot ha^{-1}$. Potassium salts of glyphosate (Factor 540, IPCO Interprovincial Cooperative Ltd, Winnipeg, MA, Canada) were applied on 2011-07-08, 2012-06-06 and 2012-06-21 in BB (1.13 kg glyphosate acid equivalents (a.e.)·ha^{-1}). Isopropylamine salts of glyphosate (Polaris) in different formulations (Du Pont Canada, Mississauga, ON) were used in SR on 2011-07-04 (Gardien, 0.72 kg a.e.·ha^{-1}), 2012-06-11 (Polaris + Ultime, 0.89 kg a.e.·ha^{-1}) and 2013-06-14 (Galaxie II, 0.81 kg a.e.·ha^{-1}). Field soil properties are detailed in Chapter 2 (Table 2- 1). Briefly, BB soil had a water extracted pH of 6.6, OM 4.5%, CEC 0.03-2 meq·$100g^{-1}$ and P saturation ($P/Al_{Mehlich-III}$) of 12.9%. SR had a water extracted pH of 6.48-6.83, OM 2.4-3.0%, CEC 15.6-16.5 meq·$100g^{-1}$ and P saturation (P/Al) of 5.46-7.60. Soil series found in field include Achigan (SR), Châteauguay (BB), Dalhousie (BB) and Saint-Bernard (BB). Site characteristics are detailed in Annex 4 and agronomic practices are detailed in Chapter 3.

On each site, a randomized block design of three consecutive treatments (3-m width x 17 m long = $51m^2$) included triplicate treatments of herbaceous vegetation (CX) and two densities of *Salix miyabeana* SX64: 3 (3X) or 5 rows (5X) representing 33 333 or 55 5556 stems/ha. Willows were planted (Spring 2009) and coppiced twice (Fall 2009 and 2010) prior to monitoring. Herbaceous vegetation was mowed once a year but not harvested. Plantation, maintenance, biomass production and diversity are detailed in Chapter 1.

3.2.2 Water sampling

Surface runoff (0 cm) and interstitial water (35 and 70 cm) was sampled 7 times in BB and 9 times in SR during spring melt (2 sampling events on each site), following the first precipitation events ≥ 15 mm, after sowing and fertilization (2 sampling events in BB, 1 in SR) or following the application of glyphosate based herbicides (3 sampling events in BB, and 6 in SR). Thirty-six runoff collectors (polyethylene bucket, PVC gutter, 2 mm mesh), 72 suction lysimeters (PVC tube, ~1.3 µm porous ceramic cup) were designed, installed and sampled as described in Chapter 2. The lysimeters (Soil Moisture Equipment Inc, 1900L, Santa Barbara, CA, USA) were armed with a gauged manual pump (-70 kPa) prior to precipitation events. Piezometers (24 PVC tubes) were used to monitor ground water levels on every sampling event. All plastic sampling bottles were washed with soap and distilled water, soaked in 10% HCl, soaked in distilled water and nanopure water three times and dried prior to sampling. Glass bottles for dissolved organic carbon (DOC) were pre-combusted and rinsed with nanopure water. Pre-filters (Whatman GF/F syringe filter) were rinsed and combusted prior to use. For dissolved nutrient analysis, runoff was pre-filtered to remove coarse debris. Next, runoff (pre-filtered) and interstitial water (directly sampled from the lysimeters) was filtered (0.22 µm 2.5 cm syringe filters, PES, Pall Corporation) directly in field. For glyphosate analysis, unfiltered water was sampled in 250 ml Nalgene bottles preserved at 4°C during field work and frozen at -18°C prior to analysis. Details of the sampling equipment and procedures are available in Chapter 2.

3.2.3 Soil sampling at Saint-Roch-de-l'Achigan

Soil study on RBS potential efficiency was focused to 7 days after glyphosate application. Between glyphosate field spray and sampling, 77 mm of rain fell. Maccario et al. (2015) studied temporal variability of glyphosate concentrations in field (Maccario et al. 2015). Only SR soils were sampled corollarily to a superior water sampling success and to best represent Québec agricultural soils. SR soils had previously been treated with glyphosate prior to the start of the current experiment in 2011 (Traxion, Syngenta in 2009; Gardien, DuPont 2010). Surface cores (0-20 cm) were collected with a manual auger, ≥ 1.5 m away from runoff collectors to avoid disturbance, close to the field-edge (CF) and next to the stream (CR). Manually homogenized soil (debris > 2 mm removed) was frozen (-18 °C) to avoid glyphosate degradation (Puchalski et al. 1999).

3.2.4 Glyphosate and AMPA analyses

Thawed water samples were centrifuged (2000 rpm x 15 min) and filtered (≤0.22 µm; Nylon, Nylaflow™). Filtrate was separated in 50 ml aliquots (centrifugation tubes, Starstedt™). Surface water sample aliquots pH were adjusted to 7-8 using 0.1M NaOH or HCl (analytical reagent grade) prior to solid phase cleaning on 200 mg Chromabond C_{18} packed columns (pre-activated with 3 ml of methanol and 3 ml milli-Q water) at a flow rate ≤ 1 $ml{\cdot}min^{-1}$. Interstitial water samples contain higher concentrations of salts, which can lead to a stronger matrix effect (Basavarajappa & Manjunatha 2015; Ellis et al, 2000). Those were cleaned after pH adjustment (7-7.2) at a flow rate of 1 $ml{\cdot}min^{-1}$ with Chelex cation exchange resin (8g in 20 ml solid phase extraction syringe with PET frit; Na+ form, 16-50 mesh, Bio-Rad®.) and XAD-2 resin to remove hydrophibic compounds (30 ml in a 50 ml solid phase extraction syringe with PET frits; pre-activated 30 minutes with 5 ml methanol, rinced with 3.5 ml milli-Q water).

Extraction of glyphosate and AMPA on 2.3 g AG1-X8 formate form (200-400 mesh, pre-activated with 5 x 3 ml of milli-Q water) was followed by elution (4 x 3 ml HCl 0.6 N) as per Bergström et al. (2011). The analytical method for extraction was modified after Börjesson & Torstensson 2000 and Bergstrom & Borjesson 2010 (modified parameters are given below). Purified samples (12 ml) were placed in a rotary evapotator (to reduce volume to 1 ml), then transferred to a 1.5 ml vial and evaporated under N_2 flux.

Freshly thawed soil samples (5 g, non-dried to avoid glyphosate adsorption changes or lead to glyphosate transformations or atmospheric losses) were extracted with 40 ml water (representing the potential field lixiviation) in a 50 ml Falcon Tube (Sarsteadt™), vortexed (30 sec), agitated (30 minutes; 200 rpm) and sonicated (10 minutes). Centrifugation (10 minutes; 3750 rpm) separated the supernatant (in a clean 50 ml Falcon tube) and pH adjustment (≤ 2; 6N HCl) preceded overnight decantation. Supernatant was pipetted (5 mL), pH was neutralized

(7-7.2; 0.1M NaOH), and the extract was sequentially cleaned on Dowex C-111 (J.T. Baker Chemical Co) and XAD-2 mounted on a peristaltic pump system, respecting pH and flow rates recommended by the manufacturers. Finally, glyphosate and AMPA were was adsorbed on a AG1-X8 column, and eluted with 12 mL of HCl 0.6M at a maximal rate of 1 drop per 4 sec. Samples were first evaporated on a rotary evaporator (≤ 500 µL) and then transferred in a 2 ml glass vial for complete evaporation under N_2 flux.

Both water and soil extracts were then treated identically. Derivatization (90°C; 60 minutes) using 0.5 mL trifluoroethanol (TFE, Fisher Scientific) and 1 mL of trifluoroacetic anhydride (TFAA, Fisher Scientific) preceded complete evaporation (N_2 gas). Samples were resuspended in 800 µL of ethyl acetate and 200 µL of pyridine, instead of 1 mL of ethyl acetate (HPLC grade) as per Börjesson & Torstensson 2000, a small quantity of pyridine (HPLC grade) was used to quench excess sample acidity. Every sample contained an internal quantification standard (1-bromopentadecane, Sigma-Aldrich, St. Louis, USA) to assess injection reproducibility. Samples were injected in a gas chromatograph coupled to an electron capture detector (GC-ECD, Varian GC 3800, EC cell with ^{63}Ni foil model 02-001972-01) equipped with a Restek RXI-5SIL MS (30 m x 0.25 mm ID, 0.25 µm) capillary column, with an injection volume of 2 µL in split mode. Water samples precipitating upon pyridine addition were re-extracted with an additional Cu treatment (to remedy suspected SO_4^{2-} contamination;EPA 1996a, b) prior to evaporation and derivatization. Inside the GC, the temperature was raised from 60°C to 170°C at 6°C·min^{-1} (hold 30 sec), then increased to 250°C at 6°C·min^{-1} (hold 10 min) for injection. This slower temperature ramp (total run 30.17 min) improved peak separation, compared to the method of Bergstrom & Borjesson (2010). The carrier gas, high-purity hydrogen, had a debit of 1.4 mL·min^{-1}. Peak identification and quantification were ascertained with external standards at the beginning and at the end of each sample series. A fresh 1 µg·mL^{-1} working standard of both glyphoaste and AMPA was made daily from stock solutions of (100 µg·mL, in water, stored at 4°C). Glyphosate and AMPA recovery rates 94 ± 6 % and 94 ± 12 % for the surface waters extraction protocol and 97 ± 2 % and 98 ± 7 % for the groundwater extraction protocol, respectively. The detection limits in water were 0.01 and 0.02 $\mu g \cdot L^{-1}$ for glyphosate and AMPA, respectively, as determined through repeated injection of blanks. In environmental samples, the quantification limits were 0.05 $\mu g \cdot L^{-1}$ for glyphosate and 0.1 $\mu g \cdot L^{-1}$ for AMPA. Only glyphosate was quantified from soil water extracts, as AMPA peaks were difficult to resolve due to important matrix effects in the soil.

3.2.5 Water Sampling Effort

Successfully analyzed water samples (n = 129) included 100 runoff samples (BB = 29; SR = 71) and 29 interstitial water samples (BB = 10; SR = 19). More compacted and clayey SR facilitated runoff collection; contrary to the highly permeable BB soils. The dryer and warmer 2012 summer interfered with runoff sampling, challenging RBS studies in the province (Gasser et al. 2013). Some small volume samples were lost due to matrix effects (Ibanez et al. 2005). For example, a duplicate sample preparation (Cu pre-cleaning step) was sometimes required. This limited interannual heterogeneity characterization, hence the pooling of results (2011-2013).

3.2.6 Statistical analyses

In the equation of RBS efficiency, (Eq.1) brackets denote concentrations of the pesticide (P) glyphosate or its degradate AMPA.

$$\text{Eq. 1: Potential efficiency (\%)} = (([P_{CF}] - [P_{CR}]) \times [P_{CF}]^{-1}) \times 100$$

This equation is limited to punctual measurements at different depths, as the water sampling method used prevented the calculation of mass balances, and hence integration of the vertical movement of the water in the equation. The potential efficiency of the RBS was analyzed with an ANOVA on glyphosate and AMPA concentrations in runoff, as well as on glyphosate concentrations in surface soil. Water concentrations (2011-2013) were pooled by agricultural events (snow melt, post-fertilization and post-glyphosate) to alleviate uncontrolled field conditions leading to site and campaign data gaps after checking that data were not statistically different from one year to the next with a Wilcoxon test (per site and agricultural event). Due to localized heterogeneity in surface and groundwater trajectories (see rationale in Annexe 4), we considered edge-of-field samples as a fourth treatment (instead of a side), and due to the impossibility of collecting water from all sampling equipments at all time-points, blocks were dropped from the ANOVA design in the analysis of $[glyphosate]_{aq}$ and $[AMPA]_{aq}$. This was not necessary for soil analyses, which were restricted to the post-glyphosate period of 2013. To understand the influence of various environmental parameters (among which days since glyphosate applications, days since sowing and fertilization, precipitations, pH, TSS, N_{tot}, NH_4^+ PO_4^{3-}, Mg^{2+}, Na^+, Zn^{2+}, Ca^{2+}, Al^{3+}, Mn^{2+} and Fe^{2+} aqueous concentrations, ground cover by herbaceous vegetation, Shannon diversity, anf finally land bare of herbaceous vegetation ground cover will be further discussed in section 4.5 (Discussion). The exhaustive list of environmental parameters used is presented in Annexe 8), multiple regressions were conducted on $[glyphosate]_{aq}$, $[AMPA]_{aq}$ and $[glyphosate]_{soil}$ in which the factors of the original ANOVA (site, side and treatments) were replaced by environmental parameters. To avoid over-parameterization, the number of parameters used in the multiple regression analysis was reduced using two approaches. First, we formed 8 groups of variables with similar nature (time factors; cultural practices; vegetation ecological characteristics; *Salix* growth and productivity; topography; hydrogeology; climate; and water physico-chemistry). Then, we used the first principal component (PC1) of each group in the regression obtained in a Principal Component Analysis. Secondly, we screened for parameters highly correlated with $[glyphosate]_{aq}$, $[AMPA]_{aq}$ and $[glyphosate]_{soil}$ and used these sets of individual variables in the multiple regressions.

To assess the influence of sampling periods on the RBS aqueous input, a Kruskall-Wallis test was conducted on the edge-of-field $[glyphosate]_{aq}$ of both sites. Then, to understand the influence of environmental parameters on leaching of glyphosate and AMPA at the edge-of-fields, we used multiple regressions with the two approaches described above: (1) with the PC1 of 6 pertinent environmental matrices (Cultural practices, Salix, Topography, Time, Water physico-chemistry, and Vegetation ecological characteristics) and (2) with the individual parameters most highly correlated with $[glyphosate]_{aq}$ and $[AMPA]_{aq}$. Finally, to accurately study the relationship between environmental factors and the response variables, $[glyphosate]_{aq}$ or $[AMPA]_{aq}$, pairwise correlations before (CF) and after (CR), the RBS were conducted. All statistical analyses were conducted using JMP 7 (SAS Institute, Cary, NC).

3.3 Results

3.3.1 Buffer strip potential efficiency to retain agrochemicals in runoff

CR $[glyphosate]_{aq}$ was not significantly reduced compared to CF $[glyphosate]_{aq}$ after snowmelt and post-glyphosate (low n precluded statistical analysis post-fertilization; Figure 3- 1). In BB, post-glyphosate $[glyphosate]_{aq}$ sometimes increased, while AMPA decreased, non-significantly after the RBS.

Several environmental parameters may affect RBS potential efficiency, as established via pairwise correlations between $[glyphosate]_{aq}$ (or $[AMPA]_{aq}$) and major nutrients (N-P-K), other elements and environmental characteristics on both sites (BB vs SR) and sides (CF vs CR; Table 3- 1). The aqueous glyphosate runoff concentrations post-glyphosate are strongly correlated with those at snowmelt ($r = 0.75$) and post-fertilization ($r = 0.47$). $[Glyphosate]_{aq}$ and $[AMPA]_{aq}$ are weakly correlated, but the regression is significant at CF in SR ($r = 0.27$; Table 3-1). In CF, $[glyphosate]_{aq}$ is weakly (sig) correlated to PO_4^{3-}, a relationship lost in CR; but the relationship with P_{tot} is weaker (ns), and no correlations exist in BB (Figure 3- 2). Multiple [glyphosate]$_{aq}$ correlations to N are evidenced in SR (NO_2^-+NO_3^-, NH_4^+, N_{tot}). [Glyphosate]$_{aq}$ and K^+ are strongly correlated in SR CF; moderately correlated with Na in SR CR; and correlations with Ca^{2+} and Al^{3+} increased through the RBS. Time since application of glyphosate, fertilization (both at initial sowing and latest, including mid-summer 2nd fertilization) are negatively correlated with [glyphosate]$_{aq}$ in SR, but only time since glyphosate application matters in BB. [AMPA]$_{aq}$ is strongly and positively correlated with all three time measurements in BB (no correlations in SR, Table 3- 1). In BB, increasing [AMPA]$_{aq}$ correlated with greater runoff volumes, but smaller TSS. A very strong correlation between AMPA and pH is found in BB CF (r =0.86, n = 8, Table 3- 1).

Beyond the pairwise correlations described above, groups of similar environmental variables may together play a role in the potential efficiency of the RBS. This is evidenced by multiple regressions on the first principal component axis of each group in Table 3- 2. The most influential parameters along the first axis are abbreviated as PC1. No group of environmental parameters significantly explained [glyphosate]$_{aq}$ at snowmelt (though there is a trend with herbaceous vegetation ecology; PC1: Shannon diversity and herbaceous ground cover); nor post-glyphosate (trend with vegetation; PC1: idem; and water; PC1: P_{tot}, Mg and TSS). [Glyphosate]$_{aq}$ post-fertilization is significantly affected by culture (PC1: no dominant parameter), *Salix* (PC1: stem height and diameter), and topography (PC1: slope and elevation). Irrespective of the sampling period, culture, water and vegetation significantly explain [glyphosate]$_{aq}$. No group of environmental variables significantly explained [AMPA]$_{aq}$ at snowmelt (*Salix* nearly significant), nor post-fertilization. However, herbaceous vegetation ecological characteristics (and perhaps *Salix* nearly significantly) influenced [AMPA]$_{aq}$ post-glyphosate. Altogether, *Salix* and herbaceous vegetation ecological characteristics nearly significantly explain [AMPA]$_{aq}$, while time and water physico-chemistry played secondary roles.

3.3.2 Buffer strip potential efficiency measured in soil samples

After 2013 glyphosate application in SR, $[glyphosate]_{soil}$ at the 0-20 cm depth and on either sides of the RBS ranged from non-detectable to 317 µg·kg dw^{-1}. The mean 28-56% reduction (depending on treatment) in surface soil concentration after the RBS (compared to edge-of-field) is significant (p = 0.0381*), but treatments are undifferentiated (p = 0.3075; Figure 3- 3). Among all the individual environmental variables tested for pairwise correlations, 95 parameters had a 40-60% correlation with $[glyphosate]_{soil}$. Post-glyphosate, $[glyphosate]_{soil}$ are slightly correlated with $[glyphosate]_{aq}$ (post-fertilisation: r = 0.47; post-glyphosate: r = 0.24 and snowmelt: r = 0.03). The 5 highest correlations involve Fe^{2+} (r = 0.69), facultative hydrophytes (r = -0.66), *Salix* (r = -0.63) and litter (r = -0.59) ground covers, and PO_4d (r = -0.47). None of the groups of explanatory parameters PC1 (vegetation, *Salix*, topography, days, water and soil matrix) explain $[glyphosate]_{soil}$, but the herbaceous vegetation ecological characteristics and *Salix* groups of parameters become significant when other groups of parameters are excluded from the model (Table 3- 2).

3.3.3 Edge-of-field glyphosate and AMPA concentrations in runoff

The glyphosate and AMPA edge-of-field concentration in runoff appears to influence the RBS potential efficiency (Annexe 27). Where low concentrations were measured before the buffer strip, a strong negative potential efficiency revealed an increase across the RBS. The RBS potential efficiency seems to level over a wide range of incoming concentrations, and an apparent plateau is inferred from the non-linear regression model, near 51% reduction efficiency for glyphosate and 75% reduction efficiency for AMPA. As the incoming runoff concentrations appeared to affect the RBS potential efficiency, we investigated the effect of depth, sampling period and other environmental parameters on CF concentrations and RBS potential efficiency.

First, along a depth profile encompassing runoff and interstitial waters (0, 35 and 70 cm) $[AMPA]_{aq}$ did not reduce ($p = 0.5017$), while $[glyphosate]_{aq}$ even has a suggestive increasing trend with depth ($p = 0.0513$; Figure 3- 4). The behavior of $[glyphosate]_{aq}$ may differ between BB and SR ($p = 0.0891$). Close-up, SR $[glyphosate]_{aq}$ and $[AMPA]_{aq}$ were highest in CF runoff, and lowest at greater depth. CR's highest means were at the lowest depth sampled (70 cm) suggesting enhanced infiltration.

Secondly, glyphosate ($p = 0.1230$) or AMPA ($p = 0.7056$) concentrations in runoff reaching the edge-of-field during different sampling periods did not significantly differ between both sites (Figure 3- 5). However, the glyphosate concentrations varied with time of sampling ($p = 0.0110^*$), but not AMPA ($p = 0.1444$; Figure 3- 5, Annexe 25). In BB, edge-of-field [glyphosate] at snowmelt ($1.7 \pm 2.3\ \mu g \cdot L^{-1}$), was lower than post-fertilization ($3.4 \pm 2.4\ \mu g \cdot L^{-1}$), but indistinguishable from post-glyphosate ($3.7 \pm 6.0\ \mu g \cdot L^{-1}$), though the latter was punctuated with sporadically higher concentrations (Figure 3- 5a). SR snowmelt ($1.8 \pm 1.9\ \mu g \cdot L^{-1}$) contained less glyphosate than post-fertilization ($16.4 \pm 15.5\ \mu g \cdot L^{-1}$), which was indistinguishable from post-glyphosate ($11.2\ \mu g\text{-}L^{-1} \pm 17.4\ \mu g \cdot l^{-1}$; $p = 0.0110^*$; Figure 3- 5b).

Thirdly, runoff water physico-chemistry likely influenced both CF $[glyphosate]_{aq}$ and $[AMPA]_{aq}$ as suggested by site-specific correlations between both elements, PO_4, $NO_2^- + NO_3^-$, NH_4^+, N_{tot}, K^+, Ca^{2+}, and Al^{3+} (Table 3- 1). Beyond correlations with nutrients, relations between glyphosate and various environmental parameters were observed via multiple regressions ($n = 10$, $r^2 = 1.00$): Sum of degree-days since sampling initiation ($^{o}C \bullet d$; $r = 0.53$, $p = 0.3370$), Sum of precipitations since latest fertilization (mm, $r = -0.50$; $p = 0.0480^*$), Water table depth from surface (m; $r = 0.50$, $p = 0.0826$), Mean T_{min} since sampling initiation (^{o}C; $r = 0.49$, $p = 0.2145$), Mean relative humidity since last glyphosate application (%; $r = 0.45$, $p = 0.0218^*$) and mean $T_{average}$ since latest fertilization (^{o}C; $r = 0.42$, $p = 0.0545$).

3.4 Discussion

3.4.1 Weak potential efficiency of riparian buffer to minimize glyphosate and AMPA export in runoff

In BB and SR, the glyphosate and AMPA concentrations of the runoff is not significantly reduced across the RBS (Figure 3- 1). However in SR, a significant reduction in the soil glyphosate concentration was observed across the RBS (Figure 3- 3). Hence, we need to explain the apparent discrepancy between water and soil. First, the glyphosate and AMPA edge-of-field concentrations affected the RBS potential efficiency (Annexe 27), as previously reported for atrazine, metolachlor and cyanazine (Misra et al. 1996). We hypothesize that localized site heterogeneity in the water movements, coupled to the uncontrolled precipitations and punctual water sampling scheme, may have led to low concentrations in some edge-of-field parcels. However, while water flows, deposited soil particles may be easier to capture because they move less. Thus, it was easier here to confirm RBS potential efficiency by studying water-extractable glyphosate in the soil.

Secondly, the sampling period affected the edge-of-field runoff concentrations (Figure 3- 5). A continuous monitoring of the runoff would have been ideal, in order to analyze global potential reduction efficiency (rather than by period), but limited sampling success prevented this. The average potential reduction efficiency from the surface runoff was greatest in the summer (63-72 %), and while vegetation was dormant at spring melt (14-47 %). Due to runoff dilution by large volumes of snowmelt on saturated soils, spring lows are expected (Daouk et al. 2013). Nevertheless, the potential efficiency reported herein resembles previous reports on 4-8 m RBS with different vegetation (Lin et al. 2011; Syversen and Bechmann 2004), providing some level of confidence on our measurements. Specifically, Lin et al. (2011) observed 60-71% reduction in glyphosate through 4-8m RBS composed of *Festuca arundinacea*, *Festuca* + *Panicum virgatum*, and native *Tripsacum dactyloides* plants, and increasing RBS width improved glyphosate reduction via better particulate trapping efficiency. Furthermore, Syversen and Bechmann (2004) reported 48 and 67 % reduction of glyphosate and AMPA respectively ($\bar{X}$ 4 years) in 5m RBS of fescue, timothy, thistle and common couch (silty clay loam 0.45 ha barley field). Finally, bank filtration, which is basically a reversed RBS pumping river water through a bank to clean drinking water, reduces > 30 % of glyphosate and 46 – 94 % of AMPA (Jönsson et al. 2013). Hence, the non-significance of glyphosate and AMPA reduction from the runoff may likely be attributed to limited statistical power and challenges in collecting runoff in uncontrolled parcels. The inefficiency of the RBS in mitigating the runoffs of glyphosate and AMPA may not necessarily be due to the narrow width of the RBS. Glyphosate is generally considered as a strongly sorbing pesticide, and in a soil like SR (sandy loam with 2-4% OM and near neutral pH, Table 3- 1) adsorption coefficient may be elevated (K_F = 78-93, Vereecken 2005), especially if the soil is not saturated in P and contains high concentrations of Al (Table 2- 1, Vereecken 2005). For herbicides with a strong soil sorption potential, an increase in buffer width may not necessarily lead to an increase in retention efficiency (Krutz et al. 2005), especially if particles are retained within the leading edge of the buffer strip (Dabney et al. 2006). However, the potentially strong sorption does not preclude leaching (see section 4.5.2).

3.4.2 Aqueous and soil glyphosate and AMPA concentrations compared to other studies

Measured glyphosate runoff concentrations are of the same order of magnitude as those measured in surface waters elsewhere (Table 3- 3). The shallow soil interstitial $[glyphosate]_{aq}$ measured, resembles previously published findings (~3.5 $\mu g \cdot L^{-1}$ in drained fields of Denmark; (Kjær et al. 2011); ≤12 $\mu g \cdot L^{-1}$ and in vineyards of Switzerland (Daouk et al. 2013). Glyphosate detection frequency is increasing in Quebec (76.2% to 97.5% from 2005-2013; Giroux and Pelletier 2012; Giroux 2015) and in the US (Scribner et al. 2007) occasionally at levels beyond local water protection criteria (Horth and Blackmore 2009), revealing the need to find sustainable mitigation strategies.

The current study reports higher runoff and interstitial water concentrations of glyphosate than AMPA. Hence, only a fraction of the glyphosate studied may have degraded into AMPA. This may be explained by the stronger soil sorption of AMPA, which limits leaching, despite the fact that AMPA pools in agricultural soils may be greater due to slower degradation of AMPA compared to glyphosate and the fact that aquatic reduction of glyphosate and AMPA may be similar (Giesy et al. 2000). Reports of higher levels of glyphosate than AMPA in surface runoff from small-scale barley plots in Scandinavia and agricultural streams along row crops in Quebec (Horth and Blackmore 2009, Laitinen et al. 2009 and Giroux and Pelletier 2012) corroborate our observations. However, it is contrary to American (Scribner et al. 2007) and European (Horth and Blackmore 2009) reviews. The discrepancy may be explained by slower glyphosate degradation in colder regions (Helander et al. 2012; Stenrød et al. 2005). Glyphosate leaching is strongly controlled by soil characteristics such as pH and Freundlich adsorption coefficient (Bergström et al. 2011). Glyphosate degradation is mainly controlled by its availability for microbial degradation (Borggaard and Gimsing 2008), while AMPA degradation is also influenced by soil organic matter content (Bergström et al. 2011). In our study, the presence of AMPA most likely originated from glyphosate degradation in BB because this site did not receive any sewage sludge and is not likely to receive direct inputs of sewage contaminated waters (nearby houses are connected to municipal sewage system and their position appears hydrologically isolated from the RBS). In addition to microbial degradation, sewage sludge contribution to the AMPA pool (Ghanem et al. 2007) in SR cannot be excluded. Without appropriate source tracking, AMPA has been attributed to detergents and cooling waters in another study (Horth and Blackmore 2009). This led to questioning toxicological relevance (i.e. Deutschland; Schipper et al. 2008), stressing the usefulness of novel source-tracking methods (Kujawinski et al. 2013; Mogusu et al. 2015). Plants (Gomes et al. 2014) like the willows in the RBS have also been shown to breakdown glyphosate into

AMPA, and subsequent root exudates could potentially enrich the soil (Laitinen et al. 2007). It is unknown if microbial versus plant generated AMPA could have been distinguished with the proposed isotopic methods suggested above.

The 2013 post-glyphosate campaign in SR soil reported here, are similar to U.S. data (1-476 $\mu g \cdot kg^{-1}$ dw; Scribner et al. 2007), yet somewhat lower than ranges reported in Argentina (299-2256 $\mu g \cdot kg^{-1}$ extracted with KH_2PO_4; Aparicio et al. 2013) and those reviewed in agricultural soils by Giesy et al. (2000) (800-17 000 $\mu g \cdot kg^{-1}$). Soil glyphosate concentrations correspond to peak field concentrations of the whole study period and are in the same order of magnitude as those measured in the field 7 m before the RBS (Maccario et al. 2015). However, in the post-glyphosate 2013 sampling campaign, the mean buffer's field-edge concentrations (218 ± 26, $\mu g \cdot kg^{-1}$ dw) appeared slightly more elevated than inside the field (117 ± 27, $\bar{X}$ ± SE, $\mu g \cdot kg^{-1}$ dw; Maccario et al. 2015). This suggests potential accumulation of glyphosate before the RBS, perhaps due to the deposition of soil particles with adsorbed glyphosate on the leading edge. Hence, the RBS efficiency may not simply be due to the absence of spraying on the stream-edge (Wenger 1999), with the tall plants limiting aerial spray drift (Wolf and Cessna 2004). While the ANOVA model (with side and treatment) explained 66% of the soil glyphosate concentrations (Figure 3- 3b), multiple regression by replacing site and treatments with environmental characteristics were sometimes more powerful in explaining the RBS potential efficiency (r^2 = 25-99 %; Table 3- 2).

However, though post-glyphosate soil concentrations in the fields were the most elevated of all sampling campaigns (Maccario et al. 2015), runoff concentrations weren't more elevated just after glyphosate application than in the preceding sampling campaign (post-fertilization; Figure 3- 5). This is perhaps due to the strong adsorption of glyphosate on soil (EPA 2009b; Wauchope et al. 2002) which limits leaching (Duke and Powles 2008; Eberbach 1999). Indeed, only severe rainfall just after glyphosate application leaches significant quantities of glyphosate (i.e. ≥ 0.5 % of the applied quantity; Krutz et al. 2005). For this reason, soil and water concentrations may tell different stories. Discontinuous sampling prevented an annual mass balance calculation (glyphosate dosage *vs.* $[glyphosate]_{aq}$ + $[AMPA]_{aq}$ + $[glyphosate]_{soil}$) but typically, < 1 % (Coupe et al. 2011) to 2.4 % (Lin et al. 2011) of applied glyphosate runs off into surface water. As opposed to water samples, the influence of *Salix* and vegetation ecological characteristics on soil glyphosate concentrations were perhaps more important than other environmental groups of parameters, but remained non-statistically significant (Table 3- 2).

3.5 Environmental determinants of glyphosate leaching and RBS potential efficiency

3.5.1 Time and climate

The presence of detectable glyphosate concentrations in spring runoff, 300 days after the last application of the herbidcide, demonstrates its persistence in the environment. Contrary to Horth and Blackmore (2009) study, yet aligned with several other publications (Bergström et al. 2011; Laitinen et al. 2006; Simonsen et al. 2008;Fomsgaard et al. 2003;Laitinen et al. 2009) which noted extended leaching (9 to ≤24 months). Inter-periodic $[glyphosate]_{aq}$ correlations (section 3.2) suggest spatial influences on persistence. Time since glyphosate applications was not the sole significant driver of glyphosate reduction, as days since sowing and fertilization also played a role (Table 3- 1). Such environmental interactions between glyphosate and fertilizers are expected based on the chelating potential of glyphosate (Subramaniam and Hoggard 1988) and complexation in cationic solutions (Chahal et al. 2012). The significant influence of precipitations since the latest fertilization or mean relative humidity since latest glyphosate application, suggest complex interaction between time, climate and agricultural activities (section 3.4). Our observations support that time alone may be less important than timing between application, precipitations or temperature fluctuations as previously suggested (Borggaard and Gimsing 2008).

3.5.2 Hydrology: Topography and Phreatic environmental parameters

Differentiation of influential topographic or cultural parameters is hampered because of their intrinsic dichotomy (differing between BB and SR, but almost or completely homogeneous intra site) (Table 3- 2). A near significant trend ($p = 0.0513$) suggests potential leaching of glyphosate towards groundwater (Figure 3- 4) in BB and SR, supporting earlier controlled (Bergström et al. 2011; Litz et al. 2011) and field experiments (Daouk et al. 2013). However, both sites may not behave identically ($p = 0.0891$) and in SR, our results suggest a potentially increased [glyphosate]$_{aq}$ infiltration in the RBS, an observation supported by similar trends for PO_4^{3-}, Zn^{2+} and Al^{3+} (See Figure 3- 5 in Chapter 2). These observations reinforce the groundwater contamination concerns expressed by Krutz et al. (2005). Glyphosate drainage potential and groundwater contamination potential is theoretically considered low (Cerdeira and Duke 2006; Gustafson 1989; Horth and Blackmore 2009; Scribner et al. 2007) because glyphosate has a strong soil sorption potential (Wauchope et al. 2002; Vereecken 2005). Despite strong sorption potentiel, high water solubility (12.0 $g \cdot L^{-1}$; pH 4.3, 25 °C) (EPA 2009b) may permit glyphosate leaching under conditions of high precipitations, and especially in presence of preferential flow paths, such as macropores (Vereecken 2005, Kjaer 2005). Surface runoff exports the majority (≥ 96 %) of glyphosate from fields, leaving little (4%) to subsurface flows (Daouk et al. 2013). Although once in groundwater, pesticides in general may have a longer longevity (EPA 2003). Glyphosate's half-life is variable in soil (1-197 days; Duke et al. 2012; Wauchope et al. 2002), water (7-91 days; Miller et al. 2010; Wauchope et al. 2002), saltwater (47-315 days; Mercurio et al. 2014) and sediments (14-248 days; EPA 2009b). Common condition in riparian interstitial or groundwater, dark (Mercurio et al. 2014), anaerobic (EPA 2009b), cold (Helander et al. 2012) and salts (Yang et al. 2013), may increase glyphosate persistence. Monsanto reports glyphosate detection in 1.7 % of 28 000 groundwater samples from 8000 sites between 1993-2008 in Europe ($>0.1 \mu g \cdot L^{-1}$ in 0.9 % of the samples; Horth and Blackmore 2009). Pesticide reduction from runoff in vegetated RBS is promising, but the United States department of agriculture USDA considers that there is still little evidence for requiring pesticide removal in shallow groundwater (Bentrup 2008) and our study doesn't reinforce the conclusions of our predecessors with respect to glyphosate reduction by the RBS.

3.5.3 Water and soil physico-chemistry

In SR, edge-of-field correlation between $[\text{glyphosate}]_{aq}$ and PO_4^{3-} (Figure 3- 2; Table 3- 1), and PO_4^{3-} top 5 position within highest correlations explaining SR $[\text{glyphosate}]_{soil}$ (section 3.3), is echoed in the literature. Laitinen et al. (2009) linked $[\text{glyphosate}]_{aq}$ and PO_4^{3-} or P_{tot} in surface runoff ($p < 0.01$) from Finland leaching plots which received glyphosate after fall barley harvest. Elevated glyphosate leaching post-fertilization in SR 2013 (Figure 3- 5) could be linked to remobilization of glyphosate induced by P fertilization. Glyphosate phosphonic acid competes with P for adsorption sites in the soil (Hill 2001; Gimsing et al. 2004b) and P fertilization may induce glyphosate remobilization and subsequent plant reabsorption, leaching or microbial degradation (Borggaard 2011; Simonsen et al. 2008). On the soil series characterizing SR and BB fields, P and glyphosate adsorption sites limitations are unexpected, making the study comparable to the northern European field leaching study of Laitinen et al. (2009). Though reactive Al sites of SR suggest potential P resuspension under high aqueous fluxes (Giroux et al. 2008; Michaud et al. 2002). However, others dismiss P importance in glyphosate leaching (Duke et al. 2012; de Jonge et al. 2000). Nevertheless, the disappearance of the correlation between $[\text{glyphosate}]_{aq}$ and PO_4^{3-} across the RBS suggests differential attenuation processes for both molecules (Figure 3- 2; Table 3- 1). Furthermore, the apparent glyphosate and AMPA potential reduction efficiency plateau, at 51 % and 75 % respectively (Annexe 27), suggests that adsorption site limitations, plays a governing role on RBS potential efficiency. This is aligned with Litz et al. (2011) who attributed low glyphosate potential reduction efficiency due to adsorption site limitations.

The correlations between aqueous glyphosate and most cations (Mg^{2+}, Na^{+}, Zn^{2+}, Ca^{2+}, Al^{3+}) increased during passage through the RBS in SR, and the correlations with Mn^{2+} and Fe^{2+} became increasingly negative (Table 3- 1). SR soil glyphosate concentrations are again correlated with a cation (Fe^{2+}) (section 3.3). This supports earlier observations suggesting that contrary to the general chemistry of the soil solution the presence of certain cations may be critical determinants of glyphosate transport (Daouk et al. 2013). Glyphosate is such a strong complexing agent that some doubt that it could circulate freely without complexing dissolved or mineral cations (Sundaram and Sundaram 1997). Cations mediate glyphosate adsorption to soil particles, like clay (Subramaniam et Hoggard, 1988). Complexes solubility varies depending on the cations in a neutral pH buffer (Sundaram and Sundaram 1997), like RBS interstitial water. Insoluble complexes may precipitate in the soil (Subramaniam and Hoggard 1988). Moreover, if indeed the strengthening correlations between glyphosate and cations in the SR RBS (Table 3- 1) suggest that RBS is a "hot spot" for complexation, precipitation of soluble glyphosate could be a likely removal mechanism. Not only are glyphosate herbicidal properties inactivated by complexation with metals in soil (Fe^{3+}, Fe^{2+}, Al^{3+} but not Ca^{2+}, K^{+} and Na^{+};Hensley et al. 1978) and in solution (Sundaram and Sundaram 1997); glyphosate may interfere with plant uptake of various plant nutrients (Ca, Mg (Duke et al. 1985; Cakmak et al. 2009); Fe, Mn (Cakmak et al. 2009),as chelation immobilizes soil nutrients (Duke et al. 2012; Gordon 2007; Yamada et al. 2009; Zobiole et al. 2012).

N_{tot} and NH_4^{+} are significantly associated with glyphosate on edge-of-field in SR (Table 3- 1). Three concepts may explain strong correlations with NH_4^{+}. First, NH_4^{+} is a determinant in pesticide reduction from artificial wetlands (Stehle et al. 2011). Secondly, isopropylamine (IPA) (from salts of glyphosate applied in SR) may lead to NH_3 release by common soil bacteria (*Pseudomonas* sp.) via IPA dehydrogenase (de Azevedo Wasch 2001). Thirdly, NH_3 fertilizers may solubilize soil humic substances via alkalinization leading to elution of organic-associated glyphosate (de Jonge et al. 2000). The 3 fold correlation strength decreases between NH_3 and glyphosate from CF to CR side, which suggests different processes within the RBS (glyphosate reducing *vs.* NH_4^+ increasing trends; Figure 2- 3 in Chapter 2). This is possibly linked with the glyphosate induction of phenylalanine ammonia lyase (PAL) activity. In plants (i.e. soy (Duke et al. 1980) or corn (Duke and Hoagland 1978)), and microbes (Shende and Patil 2013), PAL catalyzes the conversion of phelylalanine to trans-cinnamate, releasing NH_3 (Duke et al. 1980; Howles et al. 1996). However, the correlation with NO_2^-+NO_3^- strengthens across the RBS (Table 3- 1). Perhaps the result of NH_4^+ (like glyphosate) adsorption to superficial non-saturated soil layers (Jones 1999), while NO_3^- infiltrates the non-saturated zone. In support for this hypothesis, cropping systems and soil types minimizing N leaching, may lead to glyphosate leaching (Aronsson et al. 2011) and conditions which favor denitrification which may weaken glyphosate degradation (Pavel et al. 1999; Vidon and Hill 2004).

TSS is the third most important parameter on the PC1 axis of the water physico-chemistry matrix, influencing glyphosate potential reduction efficiency (Table 3- 2). TSS negative correlations with AMPA is strengthened across the BB RBS (Table 3- 1). Considering that P leaching was mainly particle bound [$(1\text{-}PO_4d)/P_{tot}*100$ = SR: ≥40% in the spring to 90% post-glyphosate; BB: 80% in the spring to 60% post-glyphosate], we may hypothesize that our glyphosate leaching measurements were underestimated (Aronsson et al. 2011). However, others reported low proportions of particle-bound glyphosate transport (Bergström et al. 2011; Daouk et al. 2013; Kjær et al. 2011). Glyphosate reduction within the RBS may be tied to the interception of eroded soil particles (Reichenberger et al. 2007), which may explain the lack of RBS potential efficiency on the dissolved fraction (Figure 3- 1). A characterization of particle-bound glyphosate in runoff may have provided results more similar to those from soil measurements (Figure 3- 3).

The pH was not highlighted among glyphosate influential environmental parameters, even though in other studies it has been deemed to be the best predictor of glyphosate soil sorption

(Borggaard and Gimsing 2008; Gimsing et al. 2004a) and a decisive factor in soil solution transport (Daouk et al. 2013). This is because pH influences ionization of glyphosate and henceforth its ability to bind other ions (Sprankle et al. 1975). Near neutral pH in proximity to fields (~7) and minor changes below the RBS (BB: ≤7.5; SR: ≥6.7; Chapter 1 or Annexe 2), might explain the lack of predictive power for pH. Soil physico-chemistry did not significantly influence soil glyphosate concentrations.

3.5.4 *Salix* and herbaceous vegetation

The absence of significant differences between herbaceous and willow RBS may stem from compensating mechanisms between willow and herbaceous plant effects, as herbaceous plant biomass is inversely proportional to *Salix* density (Chapter 3) and other gradients were observed in herbaceous vegetation ecological characteristics with willow density (Chapter 1). Indeed, the ecological characteristics of the RBS vegetation were determinating factors influencing the RBS potential efficiency on aqueous glyphosate and AMPA, and perhaps even in soil (Table 3- 2).

Ground cover by herbaceous vegetation was the second most influential parameter on the vegetation PC1 axis. This parameter was identified the most important factor affecting pesticide removal from vegetated ditches by Stehle et al. (2011). A regression of glyphosate and AMPA potential reduction efficiency against ground covered by herbaceous vegetation revealed significant effects (Annexe 28). While a higher ground cover increased glyphosate potential reduction efficiency ($r = 0.54$, $p < 0.0001$*), it was linked to a reduced AMPA potential reduction efficiency ($r = -0.72$, $p = 0.0067$*). Glyphosate potential reduction efficiency seemed to reach a plateau around 61%, perhaps due to saturation of soil adsorption sites (Bergström et al. 2011). On the other hand, the negative relation with AMPA potential reduction efficiency may be due to glyphosate degradation into AMPA (Borggaard and Gimsing 2008). Indeed, plants may contribute to glyphosate decontamination (Lin et al. 2011) due to indirect effect on soil microbiota (Borggaard and Gimsing 2008), via absorption (Gomes et al. 2015a; Gomes

2015b; Niti et al. 2013), enhanced infiltration, sedimentation and sorption (Krutz et al. 2005; Patty et al. 1997; Tingle et al. 1998; Webster and Shaw 1996).

Shannon diversity had a prime importance on the PC1 axis of vegetation ecology matrix (Table 3- 2). However, when glyphosate or AMPA potential reduction efficiency were plotted against Shannon diversity, the regression was not significant (Annexe 28). This contradicted the hypothesis that diversified ecosystems enhance functional detoxification capabilities (Altieri 1999), but would still merit further investigations under controlled conditions.

As glyphosate runoff concentrations are above the ≥10mg/L threshold (Table 3- 3), we could see sub-lethal acute toxicity in the species populating the RBS, as it has been shown in ruderal ditch species (Saunders et al. 2013). Within the 3-m-wide RBS, Shannon diversity was significantly lowered on the edge-of-field in SR (interaction between treatment and side parameters) (Annexe 13). This reduced diversity could hypothetically have been due to glyphosate spraying. Indeed, a distance as short a 10 m had unravel differences in Shannon diversity induced by a group of herbicides (including glyphosate) on a field margin (Jobin et al. 1997). Herbicides are expected to shape herbaceous plant communities in fields and contiguous areas (Jobin et al. 1997). While annual plants may be favorably selected under the pressure of herbicides like glyphosate (which kills live plants, not seeds;Jobin et al. 1997), we found no clear evidence for this (Annexe 13). No spray zones as narrow as 2 m may have tangible benefits on plant community diversity (Gove et al. 2007). We have evidence that in SR, bare soil ground cover is significantly reduced beyond the shield of the dense willow RBS (5X), on the edge-of-stream (Annexe 13). This could have been related to glyphosate spray drift protection by the dense trees. And a final clue pointing in the same direction was observed in Annexe 28 (b), where glyphosate is negatively (but not strongly) correlated with the land bare of herbaceous vegetation ground cover. Considering that bare soil patches are enhanced under willows (Annexe 13), that understory vegetation plays a critical role in erosion control (McKergow et al. 2006) and that glyphosate interception may be related to erosion control (section 4.4.2, herein), further analysis on interactions between shrubs, herbaceous ground layers and glyphosate control are recommended. Because glyphosate is often used to establish (clear vegetation) or maintain (eradicate competing weeds) RBS (Schultz et al. 1995; Fortier et al. 2010; Dosskey et al. 2007) or willow plantations (Labrecque et al. 1994; Albertsson 2012), and because residual soil concentrations of glyphosate do not appear to hamper the establishment of willows, this doesn't mean that glyphosate used in or around the RBS will not affect its herbaceous community structure, which in turn can lead to influences on the RBS potential efficiency.

3.6 Conclusion

The 3 m wide RBS did not significantly mitigate glyphosate and AMPA leaching from fields to streams. However, in the mineral soil samples analyzed, the glyphosate concentration reduction by the RBS was significant. Hence, characterization of RBS potential efficiency for RBS policy monitoring purposes should not rely on single substrate analysis as diverging conclusions may be reached by surveying water and soil. In both water and soil sampled, the glyphosate and/or AMPA potential reduction efficiency of low or high density willow treatments could not be differentiated from the spontaneous herbaceous vegetation. Glyphosate and AMPA concentrations measured in runoff from Quebec row crop fields on sandy loam and humisol, are within the same order of magnitude of those in surface soil water or surface water sampled elsewhere in Canada and in the world. Temporal, climatic, topographic, and runoff physico-chemistry parameters influence glyphosate exports, while AMPA leaching is influenced by agricultural practices, runoff physico-chemistry and hydrogeology. In contrast, *Salix* and vegetation ecological characteristics influenced the glyphosate and/or AMPA concentration differences between edge-of-field and edge-of-stream. This suggests that an in depth characterization of the RBS morphometry and diversity should be included in further RBS potential efficiency studies to better distinguish ecological effects beyond those of the intended gross vegetation treatments (i.e. shrubby versus herbaceous). Furthermore, RBS potential efficiency in reducing glyphosate varies strongly with site and time. One cannot extrapolate results to different environments without appropriate testing. Since glyphosate is omnipresent in surface waters of field row crop regions of Quebec (Canada) and elsewhere around the globe, and because it may bear environmental and human toxicological consequences the 3-m-wide RBS promoted by Quebec policy, even with the use of fast growing willows as efficient phytoremediation agents instead of spontaneous herbaceous vegetation, remains insufficient to protect surface waters and groundwater from glyphosate and AMPA contamination. Accordingly, farmers should minimize sole reliance on glyphosate or herbicide sprayings to control weeds wherever possible. As reducing the problem at the source may help to minimize the persistence and potential infiltration problems identified herein.

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3.8 Tables and figures

Table 3-1: Correlation between glyphosate and AMPA and other environmental variables for Boisbriand (BB) and Saint-Roch-de-l'Achigan (SR), on either sides of the buffer strips (close to the field (CF) or to the river (CR)).

Small samples ($n < 10$) are in grey, correlations ($r \geq 0.50$) are in bold and underlined values showed significance in a regression analysis ($p \leq 0.05$).

	Glyphosate				AMPA			
	BB		SR		BB		SR	
	CF	CR	CF	CR	CF	CR	CF	CR
Glyphosate (µg/l)	1	1	1	1	-0.16	-0.11	0.27	-0.05
AMPA (µg/l)	-0.16	-0.11	0.27	-0.05	1	1	1	1
P_{tot} (µg/l-P)	-0.31	-0.35	0.23	-0.07	-0.44	-0.42	-0.05	0.31
PO_4d (µg/l-P)	-0.20	-0.30	0.21	0.06	-0.23	-0.36	-0.05	0.21
$NO_2^-+NO_3^-d$ (µg/l-N)	**0.57**	-0.33	0.30	0.47	-0.26	-0.35	0.23	0.00
NH_4^+d (µg/l-N)	-0.19	-0.31	**0.52**	0.16	-0.24	-0.38	-0.03	0.22
NO_2^-d (µg/l-N)	-0.11	0.06	0.33	0.48	**-0.77**	0.46	0.41	0.22
N_{tot} (ug/l)	-0.04	-0.38	0.48	0.31	-0.26	-0.46	0.13	0.18
NO_2^-/N_{tot} (%)	**0.88**	0.69	0.25	0.02	**-0.88**	-0.18	0.49	0.14
NO_3^-/N_{tot} (%)	**0.96**	0.03	0.11	-0.11	-0.30	0.77	0.48	-0.11
NH_4^+/N_{tot} (%)	**0.96**	0.31	-0.21	-0.01	-0.01	0.20	-0.42	0.42
K^+ (µg/ml)	0.12	-0.28	**0.62**	0.36	-0.17	-0.05	0.06	0.11
Mg^{2+} (µg/ml)	-0.22	-0.21	0.14	0.33	-0.29	0.42	-0.09	-0.09
Mn^{3+} (µg/ml)	**-0.50**	0.00	-0.05	-0.15	**0.85**	**0.76**	-0.31	0.12
Na^+ (µg/ml)	**-0.51**	0.09	0.31	0.47	0.14	**0.75**	-0.06	-0.01
Zn^{2+} (µg/ml)	0.26	-0.27	0.11	0.28	-0.20	-0.33	0.03	-0.05
Ca^{2+} (µg/ml)	-0.03	-0.10	0.30	**0.51**	-0.05	**0.68**	-0.09	-0.10
Fe^{2+} (µg/ml)	-0.08	-0.20	0.00	-0.22	-0.26	**0.70**	0.17	0.09
Al^{3+} (µg/ml)	0.05	0.29	0.22	0.46	-0.31	**0.76**	-0.07	0.13
Days since glyphosate	-0.24	**-0.53**	-0.20	-0.26	0.26	0.33	-0.21	-0.13
Days since Sowing & Fertilization	-0.29	-0.24	-0.43	-0.32	0.38	**0.74**	-0.10	-0.04
Days since last Fertilization (incl 2^{nd})	-0.29	-0.24	-0.46	-0.33	0.38	**0.74**	-0.10	-0.03
Volume (Litres)	-0.10	-0.34	-0.13	-0.27	0.24	**0.77**	-0.15	-0.22
TSS >0,2um (mg/ml)	-0.32	-0.27	0.03	-0.26	-0.24	**-0.62**	0.04	-0.03
DOC (µg/ml)	-0.19	-0.38	0.18	0.41	-0.03	-0.23	-0.22	-0.35
pH	-0.40	0.16	0.22	-0.06	**0.86**	0.02	-0.41	-0.35

Table 3-2: Environmental parameters (belonging to Salix, Topographic, Water, Time, or Culture descriptors) influencing RBS potential efficiency on glyphosate and AMPA aqueous concentrations from 2011 to 2013.

Multiple regression models exploring relationships between glyphosate and AMPA aqueous concentrations across the buffer strip (potential efficiency) at different sampling periods (SM: Snow melt; PF: post-fertilisation; and PG: post-glyphosate) and the principal component of six environmental parameters matrices. Environmental parameter(s) most strongly contributing to the first principal axis of the principal component analysis of the matrices used to explain glyphosate and/or AMPA. 1 All parameters nearly equally contributing to PC1 variability.

Compartment:		**Water** (Runoff)								**Soil** (0-20cm)		
Substance:		**Glyphosate**				**AMPA**						**Glyphosate**
Period:		SM	PF	PG	ALL	SM	PF	PG	ALL			PG
Regression r^2:		0.25	0.99	0.35	0.73	0.56	0.41	0.46	0.49			0.5
N:		21	10	21	22	21	10	21	22			18
Group of parameters *(PC1 1º Parameters)*		**p**				**p**						**p**
Vegetation	*1) Shannon H'* *2) Soil covered by herbs*	**0.0979**	0.1893	**0.0962**	**0.0006***	0.1570	0.5898	**0.0157***	**0.0345***	**Vegetation**	*1) Shannon H'* *2) Soil covered by hydrophytes*	0.1051
Salix	*1) Salix height* *2) Salix diameter*	0.1400	**0.0334***	0.5236	0.2000	**0.0620**	0.5665	**0.0592**	**0.0315***	**Salix**	*1) Salix height* *2) Salix diameter*	0.2162
Topo	*1) x-y-z coordinates*	0.9803	**0.0042***	0.6523	**0.0561**	0.7608	0.8373	0.877	0.7782	**Topo**	*1) x-y-z coordinates*	0.1972
Water	*1) P_{tot}* *2) Mg* *3) TSS*	0.3339	0.1717	**0.0559**	**0.0128***	0.2291	0.7568	0.1402	**0.0771**	**Water**	*1) P_{tot}* *2) Mg* *3) TSS*	0.9498
Time	*1) Days since sowing & fertilization* *2) Days since latest fertilization*	0.3771	0.1366	0.9987	0.5686	0.3032	0.5652	0.2548	**0.0715**	**Time**	*ND[1]*	0.3301
Culture	*ND[1]*	0.6273	**0.0022***	0.5555	**0.0219***	0.5120	0.5622	0.3988	0.2336	**Soil**	*1) N_{tot}* *2) C_{tot}*	0.4072

Table 3-3: Glyphosate and AMPA concentrations measured in surface runoff, interstitial water or surface waters of different regions.

Region	Glyphosate ($\mu g \cdot L^{-1}$)	AMPA ($\mu g \cdot L^{-1}$)	Details	Authors
Boisbriand, Québec (Canada)	≤ 17.4	≤ 7.8	Surface runoff, porous humisol	Current study
Saint-Roch-de-l'Achigan, Québec (Canada)	≤ 67.1	≤ 9.8	Surface runoff, compacted sandy loam	Current study
Québec (Canada)	≤ 16.0 (2008); ≤ 3.3 (2009); ≤ 29 (2010);	≤ 1.1 (2008); ≤ 1.1 (2009); ≤ 3.8 (2010)	Agricultural surface waters; glyphosate detection frequency is increasing (76.2% to 97.5% from 2005-2013	(Giroux and Pelletier 2012; Giroux 2015)
Ontario (Canada)	≤ 40.8	-	Surface waters, None > 65 µg/L aquatic life protection criteria (n= 500 samples); detection frequency (15%)	(Environment Canada 2011; Giroux and Pelletier 2012; Struger et al. 2008)
British Columbia (Canada)	2-9	3-6	Runoff, of all pesticides surveyed, glyphosate and AMPA had the highest concentrations	(Environment Canada 2011)
United States	≤ 9.7	≤ 8.7	National Water-Quality Assessment Program (NAWQA)	(Scribner et al. 2007)
United States	≤ 99	≤ 22	State cooperative studies	(Scribner et al. 2007)
United States	≤ 427	≤ 41	Toxic Substances Hydrology Program (Toxics) Leary Weber Ditch Basin and Sensitive Environments	(Scribner et al. 2007)
Argentina	0.5-4 (April); 1.7-3.6 (August); ≥0.1-7.6 (Sept.)		Surface runoff	(Aparicio et al. 2013)
Switzerland	< 0.3 (2010) 1.75-12 (2011)		Surface soil waters in a viticulture with annual spray limited to the vines immediate footprint	(Daouk et al. 2013)
Denmark	3.5 ug/L	-	Surface soil water (root zone) 1m below drained corn clayey fields (applications post-harvest)	(Kjær et al. 2011)
France	≤ 50	≤ 48,9	Surface water, 27 and 50% of [glyphosateP] and [AMPA] ≥0.1$\mu g \cdot L^{-1}$	(Horth and Blackmore 2009)
Spain	≤ 15.3	ND	Surface water; 11% of [glyphosateP] ≥0.1 $\mu g \cdot L^{-1}$	(Horth and Blackmore 2009)
Sweden	≤ 13	≤ 4.0	Surface water; ≥ 0.9 and ≥ 0.8% of [glyphosateP] and [AMPA] ≥0.1 $\mu g \cdot L^{-1}$	(Horth and Blackmore 2009)
Belgium (Flanders)	< 10	< 10	Surface water; 81% of [glyphosate P] ≥ 0.1 $\mu g \cdot L^{-1}$ in Wallonia	(Horth and Blackmore 2009)
Netherlands	< 1.0	> 8.0	Surface water; 36 and ≥ 16% of sites [glyphosateP] and [AMPA] ≥ 0.1 $\mu g \cdot L^{-1}$	(Horth and Blackmore 2009)

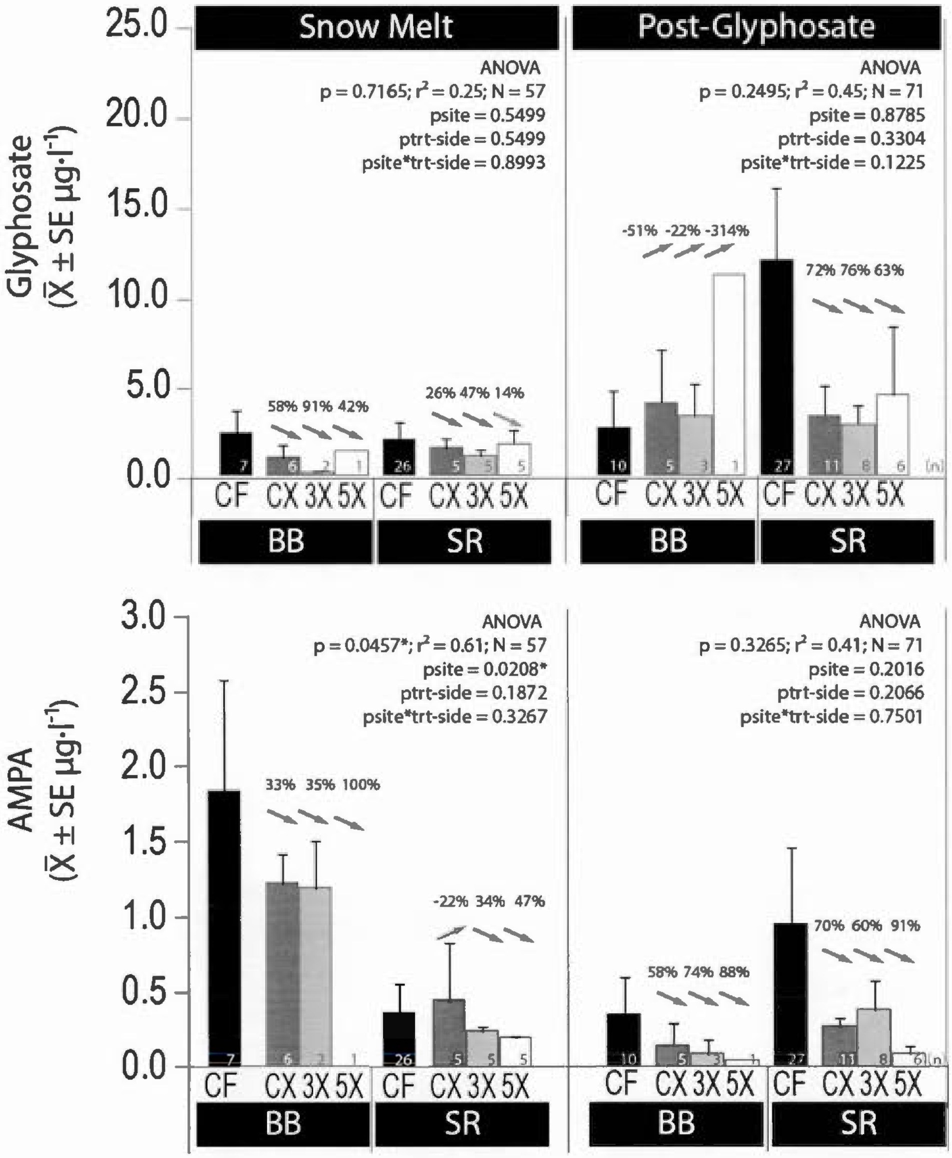

Figure 3- 1: Buffer strip potential efficiency on glyphosate and AMPA concentrations in runoff at Boisbriand and Saint-Roch-de-l'Achigan from 2011 to 2013.

Measurements were taken before the buffer strip (CF) and after the herbaceous buffer (CX), low density (3X) and high density (5X) *Salix Miyabeana* SX64 buffers at snowmelt and after glyphosate based herbicide applications in the fields. The number of samples per bar (n) and the RBS potential efficiency (%) is given on the figure. Note that AMPA concentrations are only a fraction of glyphosate concentrations, and hence are not presented at the same scale for clarity purposes.

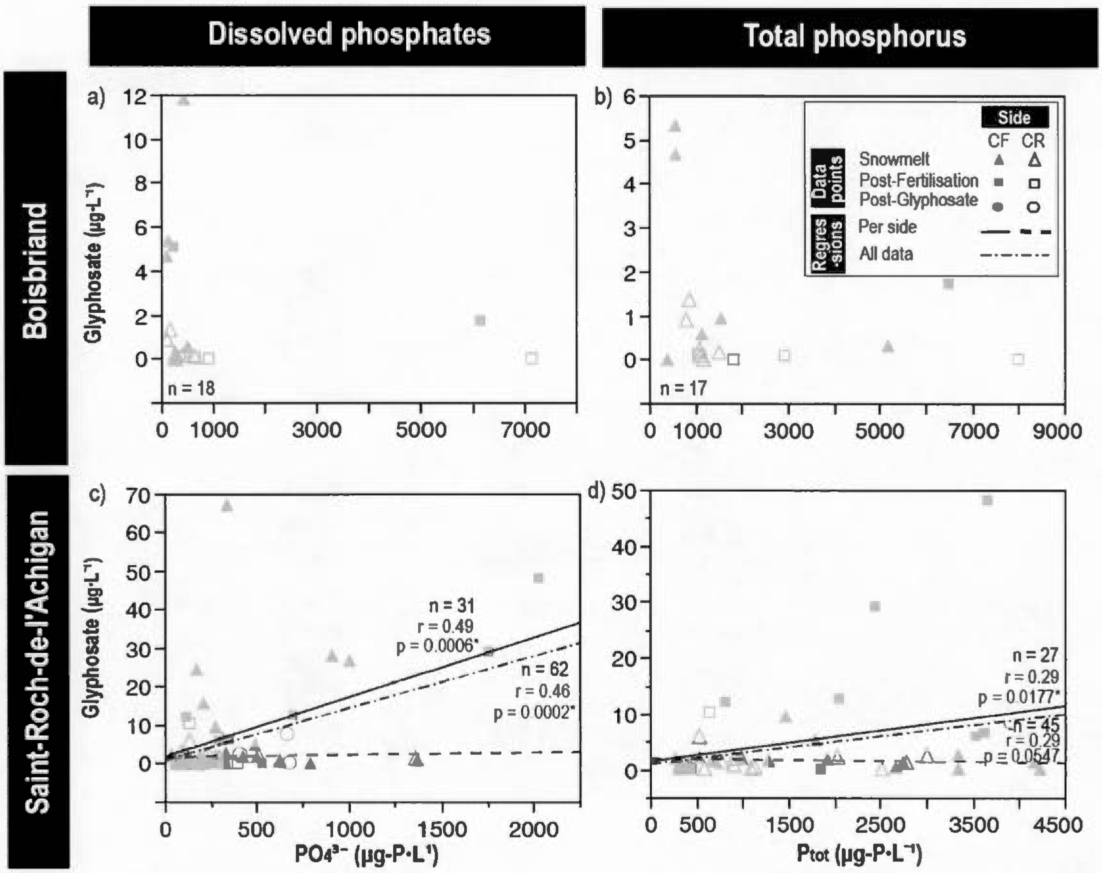

Figure 3- 2: Relationship between glyphosate and dissolved phosphates or total phosphorus concentrations in runoff water (0cm).

No linear relationships between PO_4^{3-} or P_{tot} and glyphosate in Boisbriand. For all data points combined, there is a weak but significant relationship between the concentrations of dissolved phosphates, but not total phosphorous in the aqueous phase of SR. This relationship between glyphosate and PO_4^{3-} is stronger when considering only data points before the buffer strip. Alternately, the weak correlation vanishes in the runoff collected after the buffer strip. This suggests that processes in the buffer strip affect glyphosate and phosphorus concentrations differently. Furthermore, though both glyphosate and phosphorus are removed from the buffer strip, glyphosate might somewhat be less efficiently removed. Keep in mind that glyphosate concentrations are approximately three orders of magnitude lower than concentrations of phosphorus. Glyphosate scales are variable between graphs to ensure optimal visibility of independent correlations.

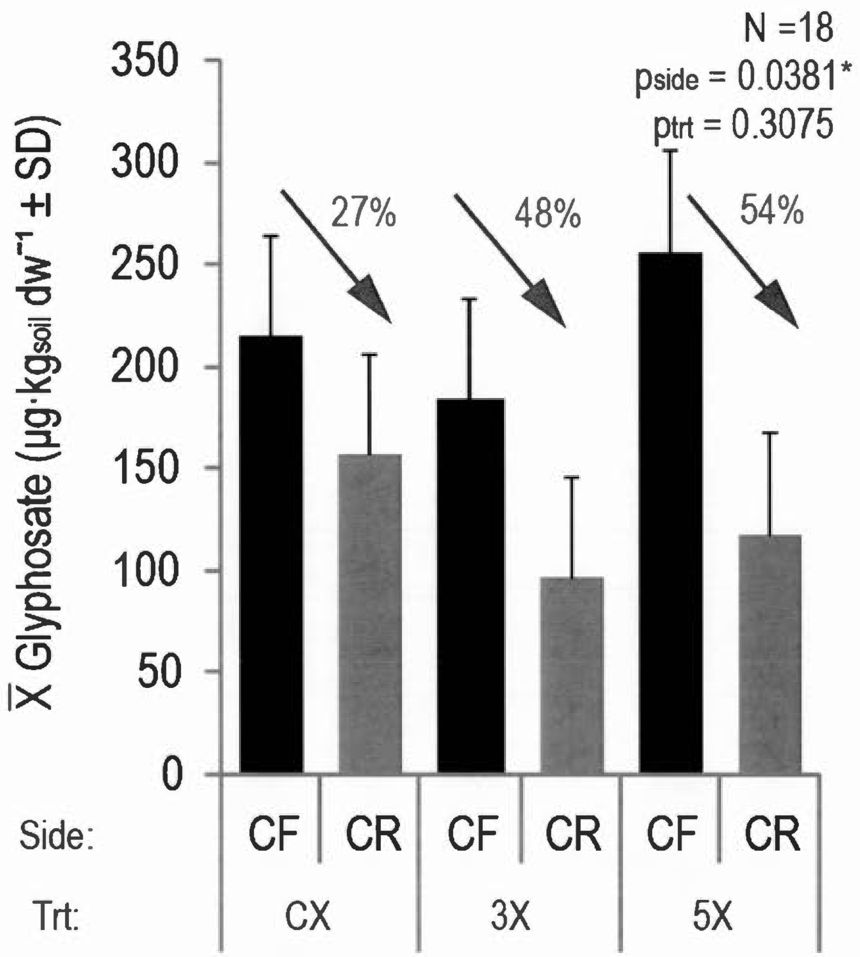

Figure 3- 3: Buffer strip potential efficiency on glyphosate soil concentration at Saint-Roch-de-l'Achigan during the post-glyphosate sampling period.

Samples were obtained during a sampling campaign on 2013-06-27, 7 days after herbicide application in the field followed with 77mm of rain). Glyphosate soil concentrations (based on dry weight), on both sides of the buffer strip (close to the field (CF) or close to the river (CR)) and according to treatment (i.e. willow density). For each bar n = 3 (N=18). The observed reduction (grey arrow) is based on the each mean. Glyphosate is not significantly reduced after the buffer strip and the Willow buffers are not statistically more efficient than the herbaceous buffer. Probabilities reported are from an ANOVA. Blocks were dropped from the model due to insufficient degrees of freedom. Interaction between treatment and side was insignificant ($p = 0.5453$) and the mode explained 66% of the variance in glyphosate concentrations.

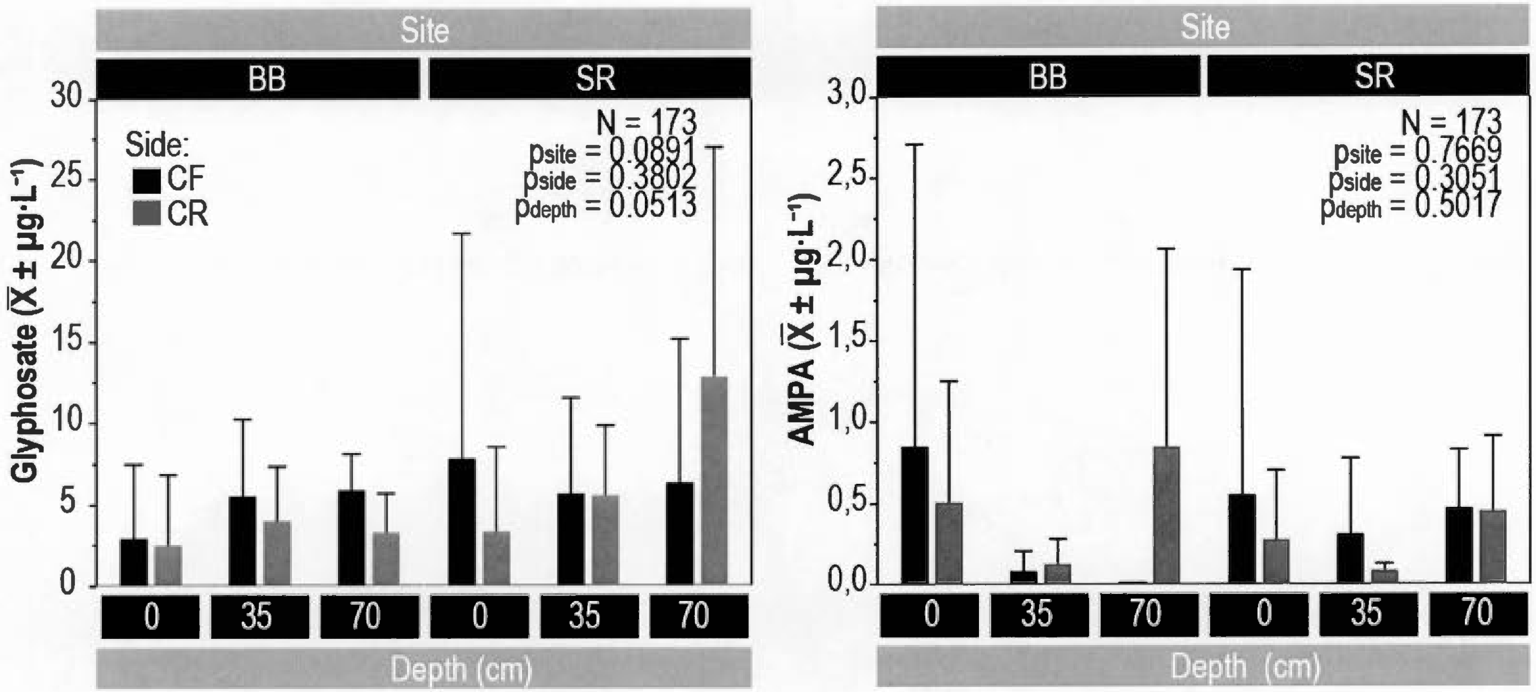

Figure 3- 4: Glyphosate and AMPA aqueous concentrations with respect to depth.

Standard deviations are given in the error bars.

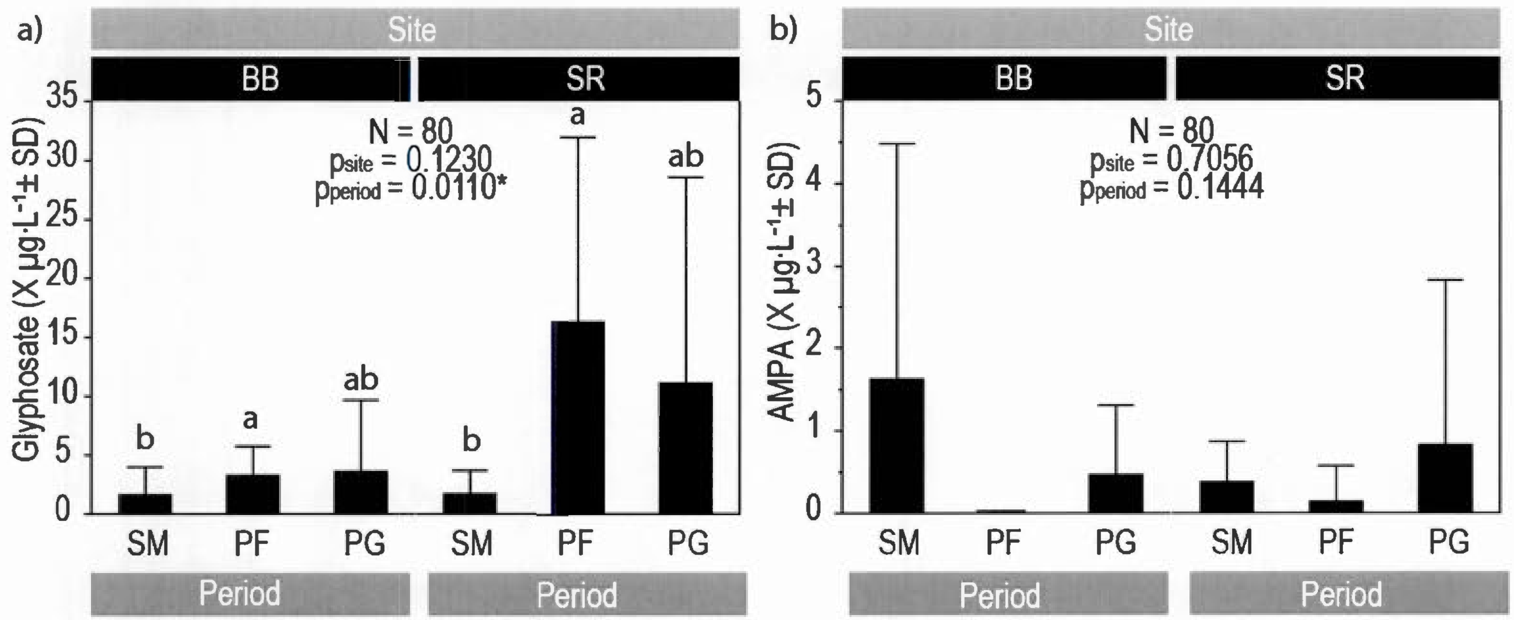

Figure 3- 5: The effect of sampling period on glyphosate in surface runoff before the buffer strip (CF) at Boisbriand and Saint-Roch-de-l'Achigan

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CONCLUSION GÉNÉRALE

II.1 Regard intégrateur sur l'ensemble de la thèse

La présente thèse discute des problèmes de pollution diffuse agricole causés par le lessivage des nutriments, pesticides et particules de terre érodées entraînant la dégradation des cours d'eau. L'objectif général est double. D'une part, nous avons testé la Politique québécoise de protection des rives, du littoral et des plaines inondables (PPRLPI; MDDEP 2005) en quantifiant l'efficacité de bandes riveraines de 3 m établies en condition de champs, sur des terres agricoles qui ont une pédologie et un relief bien distinct, mais similaire à d'autres terres que l'on retrouve dans la plaine agricole du Saint-Laurent. Notre focus était de quantifier l'efficacité de bandes riveraines enherbées ou plantées de saules arbustifs à mitiger les flux de nutriments et d'herbicide à base de glyphosate. Ce faisant, nous avons testé un nouveau type de bande riveraine consistant à utiliser le saule à croissance rapide *Salix miyabeana* SX64 pour produire de la biomasse énergétique Ces bandes qui poussent en friche et dont l'entretien minimal se limite souvent en une tonte annuelle sont répandues. Les bandes enherbées sont populaires parce que les agriculteurs ne s'investissent que peu dans cette parcelle de terre de laquelle ils se sentent expropriés (Dagenais 2015). Pour atteindre nos objectifs centraux, il est rapidement devenu évident que la compréhension de cet écosystème complexe nécessitait de dresser le portrait de diverses composantes en interrelation. Comme nous avons travaillé en milieu agricole plutôt que dans des parcelles expérimentales contrôlées, la caractérisation du milieu physique (Annexe 4) et biotique (Chapitre 1) pouvant influencer l'efficacité des bandes riveraines (Chapitres 3 et 4) était importante. À travers trois chapitres, nous avons donc évalué le potentiel de production de biomasse (Chapitre 1) et étudié la mitigation des nutriments (Chapitre 2) et du glyphosate (Chapitre 3). La biodiversité des bandes riveraines est abordée transversalement dans les Chapitres 1, 3 et 4. Une analyse de la trajectoire de l'eau qui ruisselle ou du comportement de l'eau souterraine aux abord de la bande riveraine est présenté en complément en Annexe 4.

II.2 Une production de biomasse intéressante mais difficile à valoriser

Les saules en bandes riveraines produisent beaucoup de biomasse ligneuse (Chapitre 1). Les rendements équivalents mesurés oscillaient entre 23-34 t base sèche (bs)/ha/an à Saint-Roch-de-l'Achigan sur un loam sableux compacté, ce qui est légèrement supérieur aux rendements typiques des productions commerciales (Keoleian and Volk 2005; Adegbidi et al. 2003; Labrecque and Teodorescu 2003). Par contre, à Boisbriand, sur une riche terre organique où l'eau abonde, des rendements record de 56-89 t bs/ha/an ont été mesurés, de loin supérieurs à tout ce qui n'a jamais été publié dans la littérature. Ces bons rendements s'expliquent possiblement par la position stratégique des bandes riveraines, parce que les plantations en lisière sont plus exposées au soleil, mais aussi parce qu'elles interceptent les nutriments lessivés des champs. En se basant sur des teneurs en nutriments conservatrices dans les tiges de saules, ceux-ci pourraient intercepter 116-118 kg-N/ha/an, 23 kg-P/ha/an et 62-63 kg-K/ha/an à SR sur un site modérément fertile et 278-447 kg-N/ha/an, 55-89 kg-P/ha/an et 148-239 kg-K/ha/an à BB. La récolte des tiges permettrait d'exporter définitivement ces nutriments hors du système contrairement à la bande riveraine herbacée qui n'agirait que comme tampon temporaire parce que les nutriments séquestrés par la végétation sont rapidement reminéralisés en l'absence d'une récolte.

Nous avons montré que les bandes riveraines de saules étaient attrayantes pour leur productivité et leur potentiel à séquestrer des nutriments. Mais qu'en est-il de la viabilité de telles exploitations? Encore faut-il récolter ces plantations marginales, souvent difficiles d'accès et à des coûts importants s'il faut déplacer de la machinerie d'un champ à l'autre pour la coupe. Notre analyse économique sommaire suggère que la vente des tiges pourrait rapporter un revenu intéressant (1200-1700 $/an/ha), malgré leur faible valeur marchande (80-

120 $/t), mais en comptabilisant le coût des récoltes ou la perte de productivité des grains, les profits restent incertains. L'internalisation des coûts liés aux services écosystémiques bénéficiant à l'ensemble de la société, comme la filtration de l'eau ou la stabilisation des berges, pourrait aider à faire pencher la balance économique en faveur des plantations de biomasse énergétique en bande riveraine (Simard 2009). Pour aider les agriculteurs à optimiser leur décision de récolter leur bande riveraine, nous avons proposé une équation permettant d'estimer la biomasse de façon non-destructive pour un cycle de croissance de trois ans. Pour l'instant, la PPRLPI ne permet que la coupe d'assainissement de 50 % des tiges de plus de 10 cm dans les bandes riveraines (MDDEP 2005), ce qui est fonctionnel pour les bandes riveraines arborées mais incompatible avec les bandes riveraines de saules arbustifs dont la récolte des tiges aériennes ne dérange pas le système racinaire (donc possiblement la capacité de séquestration) pouvant continuer à produire des tiges pendant une vingtaine d'années (Guidi et al. 2013). Dans ce contexte, il nous apparait important de recommander une ré-évaluation de cette clause de la PPRLPI par le ministère de l'environnement. Mais même si les saules n'étaient pas récoltés après tout, leur présence pourrait être à la regénération naturelle des bandes riveraines très lente en l'absence de plantations (D'Amour 2013; Lust et al. 2001).

II.3 L'eau fait son chemin

La méthodologie développée dans le second chapitre de la présente thèse nous suggère que le ruissellement de surface traverse effectivement la bande riveraine presque perpendiculairement si l'on considère l'ensemble des trajets modélisés à l'échelle qui englobe la bande riveraine et le champ à partir duquel ce ruissellement prend source (Annexe 4). Par contre, on y note aussi une importante hétérogénéité spatiale. Le moyen le plus réaliste de quantifier l'efficacité de la bande riveraine doit donc englober les hétérogénéités. Tandis que plusieurs fermiers se tournent vers l'agriculture de précision, il n'est plus rare d'avoir des données précises sur la topographie des champs. Une étude du ruissellement de surface telle que présentée au Annexe 4 s'avère maintenant plus envisageable qu'auparavant et permettrait de préciser les zones critiques d'intervention en fonction de la trajectoire du ruissellement de surface et de la superficie des micro-bassins versants qui alimentent les diverses sections des bandes riveraines.

On traite généralement des questions hydrogéologiques dans les bandes riveraines de façon isolée, et plusieurs études sur les polluants ignorent les hétérogénéités réelles de leurs sites, soit en n'y référant tout simplement pas, soit en assumant tout simplement leur homogénéité. Nous avons proposé une méthode simple pour comprendre l'hétérogénéité spatiale et temporelle de l'eau souterraine et utilisé les données acquises pour mieux comprendre l'efficacité de la bande riveraine, à l'aide d'analyses multivariées. L'eau souterraine s'écoule différemment au fil des saisons qui ponctuent le travail des agriculteurs. Lors des crues printanières le sol est par endroits saturé d'eau et l'absence de gradient hydraulique qui en résulte suggère un faible potentiel d'interception par la bande riveraine. Après l'épandage des herbicides, probablement à cause de la faible pluviométrie estivale, des inversions de flux hydrogéologiques peuvent survenir (Boisbriand), et qu'alors l'eau souterraine s'écoulant normalement depuis les champs jusqu'au ruisseau, fait un trajet inverse, influençant notre quantification de l'efficacité des bandes riveraines.

Nous recommandons donc aux décideurs de bien prendre en compte la variabilité spatio-temporelle du ruissellement et de l'eau souterraine pour quantifier l'efficacité des bandes riveraines. Cette recommandation est applicable pour l'amélioration de la PPRLPI, mais aussi pour tous les programmes d'appui financiers aux agriculteurs pour le déploiement de bandes riveraines. Le temps et l'argent investis pour mieux comprendre l'hydrologie propre à chaque site peut faire la différence entre un design de bande riveraine efficace ou un déficient.

II.4 Les nutriments : pour ce qui est bon avec modération, c'est la dose qui fait le poison

Notre étude sur les nutriments dans la bande riveraine suggère que la PPRLPI est insuffisante pour protéger efficacement les eaux de surface et souterraines et que l'approche consistant à planter des saules à croissance rapide n'améliore pas l'efficacité de la bande riveraine par rapport aux bandes enherbées plus communes (Chapitre 2). Les concentrations annuelles maximales en nitrates, phosphate et azote ammoniacal ont été enregistrées après la fertilisation dans les champs. Les réductions des nitrates de 77-81 % dans le ruissellement à BB, et de 92-98% dans l'eau interstitielle (35-70 cm de profondeur) à Saint-Roch-de-l'Achigan juste après la fertilisation sont bénéfiques. Cependant, cette efficacité dans l'enlèvement des nutriments à travers la bande riveraine n'est significative que ponctuellement dans le temps et l'espace pour les nitrates, l'azote ammoniacal, le phosphore total dissous et le potassium. Et pour ce qui est des phosphates dissous, aucune interception significative n'a été notée. Il est vrai que le site de BB n'est pas le milieu idéal pour l'adsorption du phosphore, mais en contrepartie les caractéristiques du site (conditions hypoxiques riches en matières organiques) en font un milieu propice à la dénitrification (selon les paramètres d'importance cités par d'autres chercheurs comme; Vidon and Hill 2004; Vought et al. 1994).

Dans le flux de ruissellement qui sort d'une bande riveraine de 3 m, nous avons noté des concentrations satisfaisantes en nitrates (<10 $mg \cdot L^{-1}$), mais les teneurs en phosphore (>30 $\mu g \cdot L^{-1}$) ou en azote ammoniacal (>1.5 $mg \cdot L^{-1}$) dépassent souvent les critères québécois établis pour la protection chronique de la vie aquatique (MDDELCC 2013). Notre constat nous pousse à recommander au ministère de l'Agriculture de se pencher sur l'optimisation des plans de fertilisation pour réduire en amont des bandes riveraines le risque à la source. Nous recommandons aussi au ministère de l'Environnement de revoir la PPRLPI (MDDEP 2005) pour mieux faire ressortir le rôle que peut jouer la bande riveraine, c'est-à-dire une solution de dernière ligne. Toujours sur la base de nos résultats, nous recommandons aussi au ministère de moduler ses recommandations de design en tenant compte des réalités agronomiques, hydrologiques et végétales locales. Nous avons constaté qu'une bande riveraine de 3 m de largeur n'est pas systématiquement efficace, et d'autres avant nous avaient suggéré que l'efficacité est proportionnelle à la largeur (Mayer et al. 2006). Une bande de largeur unique est peut-être plus facile à faire appliquer, mais nous avons observé que son efficacité varie possiblement en fonction de la taille des bassins versants, des écoulements préférentiels et du potentiel d'infiltration de chaque milieu. De plus, nos observations suggèrent que tous les végétaux ne sont pas égaux, les herbacées freinent peut-être mieux le ruissellement de surface, mais les racines profondes de certaines plantes (souvent ligneuses, mais parfois aussi herbacées) peuvent puiser les nutriments plus en profondeur. Nous suggérons donc que la mixité des strates dans les bandes riveraines devrait être favorisée dans la politique (PPRLPI, MDDEP 2005).

II.5 L'herbicide le plus vendu sur la planète contamine nos eaux et la bande riveraine aide peu

La teneur en glyphosate dans les sols de surface en amont de la bande riveraine (Chapitre 3) est supérieure aux concentrations mesurées au cœur des mêmes champs par d'autres membres du projet SABRE (Maccario et al. 2015). Mais juste en aval de la bande riveraine, la concentration est effectivement réduite de 27-54% (Chapitre 3). Nos observations suggèrent que la bande riveraine freine les particules de terre érodées. Cependant, l'efficacité des bandes riveraines à freiner le glyphosate adsorbé aux particules de terre érodées est différente dans le cas du glyphosate dissous dans le ruissellement. La bande riveraine freine parfois presque tout le glyphosate (91%) et l'AMPA (100%) tandis qu'à d'autres moments ou dans d'autres traitements, elle constitue une source où s'opère une resolubilisation (augmentation jusqu'à 314% pour le glyphosate et 22% pour l'AMPA). Dans l'ensemble donc, l'efficacité de la bande riveraine à contrer le ruissellement est jugée non significative. Contrairement à nos attentes, on n'a pas pu montrer une amélioration dans la phytoremédiation du glyphosate grâce aux bandes riveraines de saules. Ainsi, à l'heure où plusieurs cherchent des solutions faciles et efficaces pour remédier aux problèmes de contamination environnementale, il reste délicat de promouvoir une solution unique, comme la bande riveraine de 3 m de large, pour minimiser la contamination au glyphosate. Une large part de l'acceptabilité des risques dans la révision de l'homologation du glyphosate par l'agence de règlementation de la lutte antiparasitaire repose en fait sur l'utilisation judicieuse de zones tampons pour protéger les plantes terrestres et les écosystèmes aquatiques sensibles (Santé Canada 2015), mais notre étude démontre que certaines zones tampons n'offrent que peu de protection aux écosystèmes en aval.

À cette inefficacité, nous ajoutons l'observation d'une tendance de l'infiltration du glyphosate à travers la bande riveraine (profondeur maximale analysée de 70 cm). Nos résultats contredisent ainsi ceux de nombreux chercheurs qui répètent depuis 25 ans que le glyphosate a théoriquement un faible potentiel de lessivage (Cerdeira and Duke 2006; Gustafson 1989; Horth and Blackmore 2009; Scribner et al. 2007) à cause de sa forte capacité à s'adsorber aux particules de sol (Wauchope et al. 2002). Si nous avons pu détecter du glyphosate dans l'eau interstitielle, c'est parce que le glyphosate est aussi fortement soluble dans l'eau (EPA 2009). Notre détection de glyphosate dans l'eau interstitielle au Québec s'ajoute donc aux nombreuses observations dans les eaux souterraines européennes (1,7% de 28 000 échantillons prélevés entre 1993 et 2008 contenaient du glyphosate, parfois à des concentrations supérieures aux normes européennes; Horth and Blackmore 2009). La potentielle infiltration du glyphosate dans le sol, à travers la bande riveraine, diffère des conclusions de l'Agence de règlementation sur la lutte antiparasitaire qui soutenait que l'infiltration du glyphosate dans les profondeurs du sol et jusque dans la nappe phréatique était peu probable dans son projet préliminaire de ré-homologation de l'usage du glyphosate (Santé Canada 2015). Nous les encourageons donc à moduler leur position, parce que les bandes riveraines que le gouvernement provincial recommande pourraient jouer un rôle dans l'infiltration du glyphosate dans le sol.

Du glyphosate dissous a été détecté dans le ruissellement recueilli à la sortie des champs, juste avant la bande riveraine, près d'un an après sa dernière application, ce qui n'avait jamais été rapporté au Québec. Les chauds étés québécois laissaient à penser que le glyphosate se serait entièrement dégradé avant la fin de la saison de croissance, mais il appert qu'on puisse observer une persistance du glyphosate au-delà de la saison hivernale et sa rémanence au printemps suivant. Nous recommandons donc à l'Agence de règlementation sur la lutte antiparasitaire de prendre en considération la possible persistance du glyphosate dans l'environnement en saison froide, en revenant sur sa position initialement publiée dans son projet de réévaluation du glyphosate (Santé Canada 2015).

Nous avons aussi observé une corrélation entre le glyphosate et les phosphates ruisselant des champs, ce qui va dans le même sens que d'autres études (Borggaard 2011; Simonsen et al. 2008) qui ne font pas l'unanimité dans la communauté scientifique (Duke et al. 2012). Nous suggérons donc une étude plus approfondie de ce processus au Québec, où nous pratiquons une intensive fertilisation des terres, entre autres avec les rejets de l'industrie porcine.

Les concentrations que nous avons mesurées dans les flux aqueux sortant des champs ressemblent aux concentrations mesurées dans des ruisseaux de régions agricoles du Québec (Giroux and Pelletier 2012; Giroux 2015). Elles correspondent à des teneurs qui pourraient avoir un impact délétère sur la faune et la flore terrestre et aquatique (Smedbol et al. 2013; Saunders et al. 2013), ce qui remet en question la suggestion du gouvernement fédéral de reconduire l'homologation du glyphosate (Santé Canada 2015). Nous avons aussi constaté que dans cette même ronde de révision, Santé Canada a aussi négligé d'admettre la cancérogénicité du glyphosate pourtant récemment reconnu par l'Agence internationale de lutte contre le cancer comme potentiellement cancérigène pour les humains (Guyton et al. 2015; IARC 2015). S'il est trop tard pour réagir à la prochaine ré-homologation fédérale du glyphosate, il n'est pas trop tard pour que les provinces ou encore les municipalités imposent des contraintes légales ou règlementaires supplémentaires dans le but de mieux se prémunir contre d'éventuels problèmes liés au glyphosate et qui contamine la quasi-totalité de nos eaux de surface dans la plaine du Saint-Laurent (Giroux 2015). Dans sa révision actuelle du Code de Gestion des Pesticides, Québec pourrait intégrer nos recommandations afin de réduire le risque humain et environnemental de l'utilisation de la substance active la plus utilisée dans la province. Sur la scène internationale, plusieurs législations (fédérales, régionales ou municipales) ont interdit l'utilisation du glyphosate selon une compilation datée de décembre 2015 (Pesticide Action Network UK, 2015). L'Europe doit aussi revoir en 2016 l'homologation du glyphosate. Un vote important du parlement visant la ré-homologation du glyphosate pour une période de 15 ans a été repoussé en mars 2016, et une proposition visant l'acquisition de données supplémentaires et la publication des études confidentielles citées dans le projet de ré-homologation est sur la table (Parlement Européen, 2016).

II.6 La bande riveraine est un écosystème complexe où les végétaux subissent et affectent la qualité de l'eau

Contrairement à notre hypothèse de départ, l'efficacité des bandes riveraines à intercepter les nutriments et le glyphosate n'est pas directement proportionnelle à la densité de plantation des saules (sauf pour les nitrates en post-fertilisation; Chapitres 3 et 4). Ceci est contraire à nos attentes. En effet, le potentiel de séquestration des nutriments dans les plantations de saules calculé au Chapitre 1 suggérait que les saules absorbaient des nutriments hors de l'eau. À long terme, cette récolte des tiges exporterait les nutriments hors du système. Mais aussi parce que l'on s'attendait à ce que l'efficacité d'enlèvement des nutriments soit proportionnelle à la biomasse aérienne des végétaux. C'est d'ailleurs le fort potentiel de phytoremédiation du saule qui avait motivé l'utilisation de ce genre dans le projet SABRE. L'absence de différence entre les divers traitements dans notre étude pourrait simplement être liée à l'hétérogénéité spatio-temporelle du ruissellement couplée à notre difficulté à intercepter ponctuellement les flux aqueux particulièrement lorsque les précipitations sont limitées, ou encore à la strate herbacée qui colonisait partiellement les parcelles de saules.

II.7 Perspective multidisciplinaire, forces et limites du projet

Le regard jeté par les disciplines choisies (géochimie, hydrologie, biologie, écologie) répond bien à l'objectif initial visant d'évaluer l'efficacité de la bande riveraine pour limiter la pollution agrochimique des cours d'eau en milieu agricole. C'est précisément à cette interface entre les disciplines que se situe l'utilité des sciences de l'environnement qui nous ont ici permis de faire des observations multidisciplinaires et d'engendrer des recommandations permettant à la société de prendre les meilleures décisions pour assurer la durabilité de l'exploitation agricole. Le jumelage de nos expériences en milieu agricole à des expériences en milieu contrôlé par d'autres membres de l'équipe (Gomes et al. 2015a,b) constitue une force du projet CRSNG stratégique SABRE et ont permis d'établir que les saules pourraient être des agents de phytoremédiation du glyphosate dans la bande riveraine. Tout comme l'est le jumelage des études physico-chimiques avec celle des aspects socio-politiques (Dagenais 2015, Racine 2015).

L'impossibilité de réaliser un bilan de masse avec des données en continue est possiblement la plus grande limite du présent projet. En contre-partie, notre design expérimental a maximisé la représentativité du milieu agricole. Nous avons minimisé les dérangements pédologiques et hydrologiques liés à l'échantillonnage (c.-à-d. l'absence de partitions étanches entre les parcelles expérimentales et l'absence de vastes tranchées pour recueillir l'ensemble du ruissellement). De plus, nous avons respecté l'hétérogénéité spatio-temporelle naturelle du système en misant sur les activités agricoles normales et les précipitations naturelles (vs. l'utilisation de pluies artificielles ou de ruissellement synthétique avec des teneurs uniformes de fertilisants ou pesticides).

II.8 Innovations et contribution à l'avancement des sciences

La présente recherche vient combler un manque d'études sur les bandes riveraines de quelques mètres de large, typiquement rencontrées au Québec et ailleurs au Canada et constitue un apport intéressant à la science parce que nous avons travaillé à l'échelle des champs, en milieu non-contrôlé sur des terres non drainées et sur plusieurs saisons de croissance. La conclusion générale voulant que la bande riveraine ne soit pas suffisamment efficace pour mitiger les flux agro-chimiques vient documenter l'utilité de la PPRLPI et permettra peut-être d'améliorer le cadre politico-légal qui régit les bandes riveraines.

En particulier, nos recherches complémentent d'autres travaux effectués au Québec qui démontrent que la séquestration des nutriments dans la biomasse ligneuse du saule produite en bande riveraine a un excellent potentiel de phytoremédiation et nous pensons qu'il serait souhaitable de revoir la PPRLPI de façon à y inclure une ouverture pour la récolte des parties aériennes des espèces qui peuvent rejeter et continuer à croître sur plusieurs cycles de récolte. Bien que le Québec se soit doté d'une politique prônant la protection des bandes riveraines il y a plus de deux décennies, la faible largeur qu'elle recommande semble inefficace et les agriculteurs ne seraient pas nécessairement enthousiastes à l'idée de devoir augmenter significativement la largeur des bandes riveraines, notamment à cause de la perte de superficie cultivable qui entraîne des pertes de revenus. Les bandes riveraines de saule à croissance rapide que nous avons testées sont en ce sens une innovation qui a un potentiel intéressant.

La présente recherche apporte aussi des éléments pertinents au sujet du glyphosate (persistance, co-élution avec d'autres nutriments dissous, potentiel d'infiltration potentiellement accru sous les bandes riveraines) qui n'avaient jamais auparavant été étudiés en sol québécois. Ces éléments pourront éclaircir les décideurs chargés de la ré-homologation du glyphosate au Canada, ou de l'encadrement législatif de son utilisation au Québec et ou encore pour améliorer l'encadrement règlementaire municipal de cette substance potentiellement dangereuse, mais toujours disponible en vente libre.

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ANNEXES

ANNEXE 1

GRANULOMETRY OF THE BOISBRIAND AND SAINT-ROCH-DE-L'ACHIGAN SOILS.

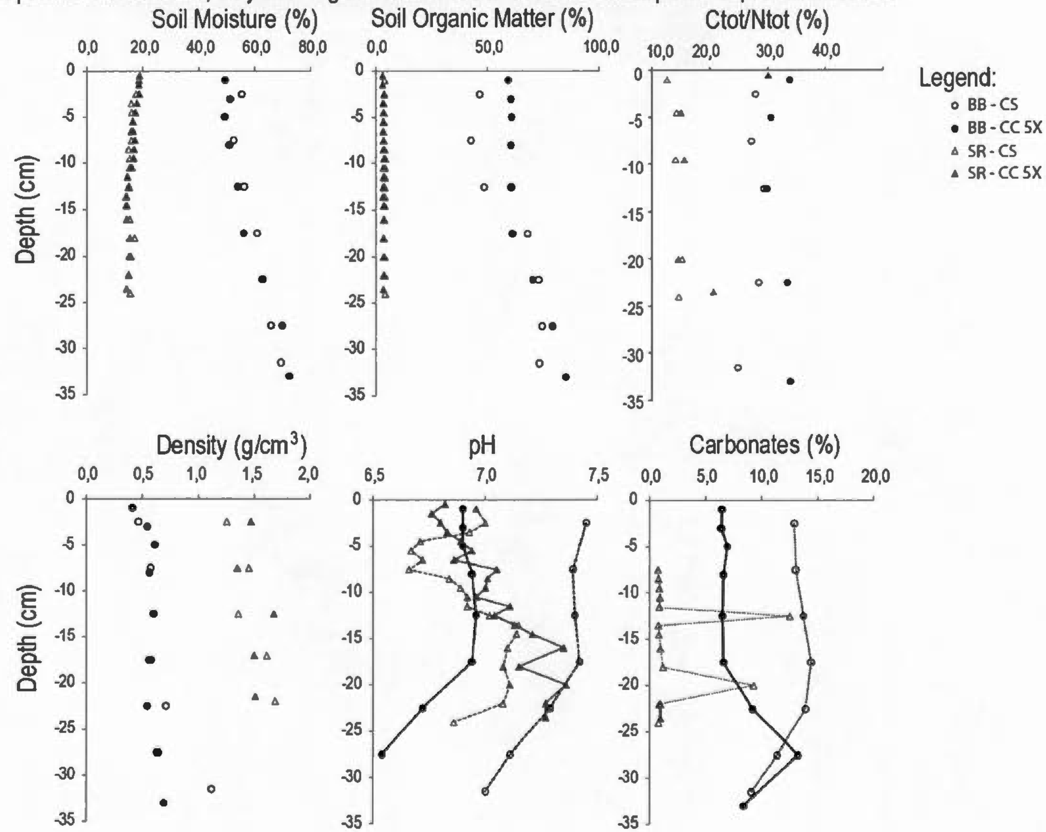
Sedigraph results of surface samples indicate the < 63 µm fraction is composed of silt in a proportion of 72.2 % at Boisbriand and 76.7 % at Saint-Roch-de-l'Achigan.

Site	Depth (cm)	Coarse sand < 2 mm	Fine sand < 212 µm	Silt and Clay <63 µm
Boisbriand	0-10	6,1	13,3	80,5
	30-40	6,4	13,9	79,7
Saint-Roch	0-10	43,2	30,1	26,7
	30-40	37,2	33,3	29,4

ANNEXE 2

SOIL PHYSICO-CHEMICAL CHARACTERISTICS PER DEPTH ON BOTH SITES FROM 2011 SAMPLES.

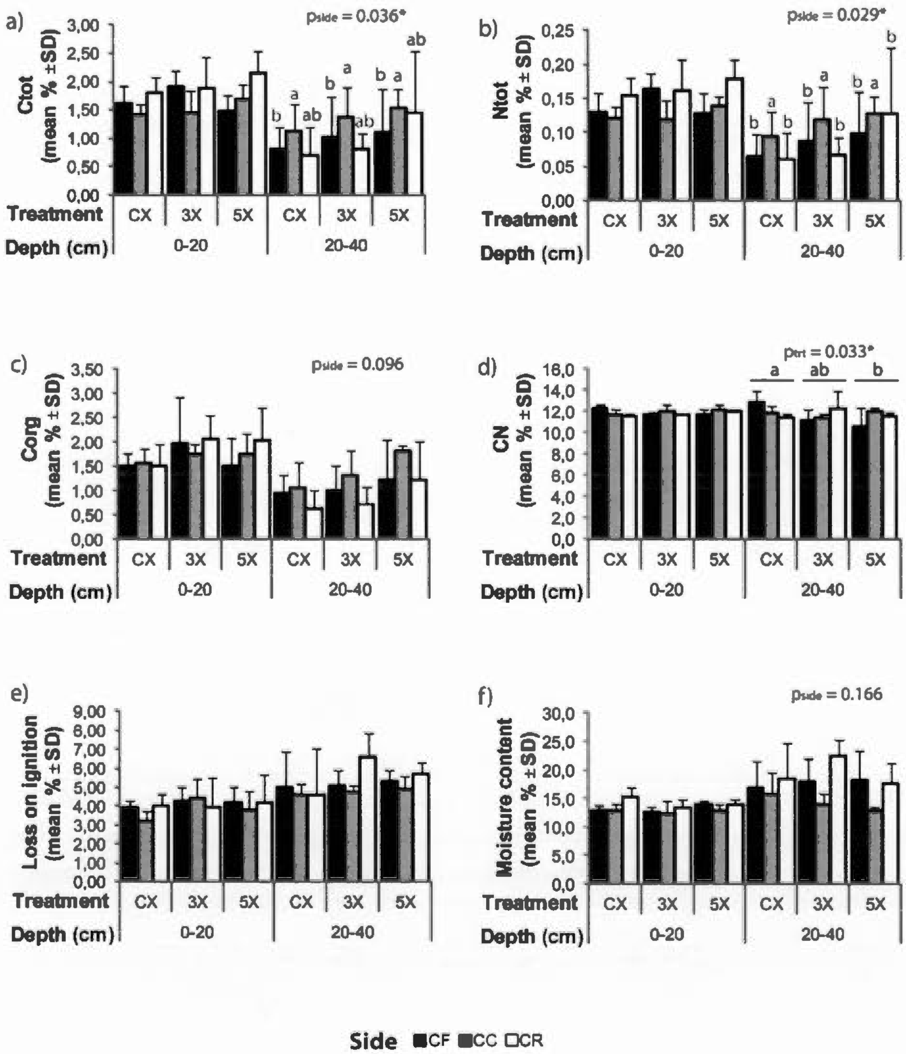
Soil cores were obtained in 2011 below the cropped area (CS) adjacent to the RBS and close to the center of the high density Salix plantations (CC 5X). SR soil physico-chemistry varies little from the field to the RBS, except perhaps for pH. BB soil physico-chemistry varies slightly from the field to the RBS for soil organic matter, organic carbon, pH and carbonates. Soil Organic Matter was based on weight loss after ignition whereas Soil Organic Carbon was obtained with an elemental analyzer, as both profiles were similar, only Soil Organic Matter which has more data points is presented here.



ANNEXE 3:

SOIL PROPERTIES IN SAINT-ROCH-DE-L'ACHIGAN AT DIFFERENT DEPTH FOR 2013 POST-GLYPHOSATE CAMPAIGN 9000

a) Ctot, b) Ntot, c) Corg, d) CN, e) Loss on ignition and f) Moisture content. Data presented is the mean (n = 3 for each bar) and error bar represents the standard deviation. Treatments CX, 3X and 5X correspond to *Salix* plantation densities of 0, 33 333 and 55 556 stems/ha. Sides correspond to position on the transect perpendicular to the buffer strip starting on the side of the field (CF), through the center of the buffer strip (CC) and ending on the edge of the brook (CR). Propabilities reported are based on a fully randomized ANOVA showing the effect of treatment (trt) or side when they are statistically significant (*) or close to significance. Letters (a and b) denote statistically different sides or treatments based on a post hoc Tukey HSD analysis.



ANNEXE 4

HYDROLOGY HETEROGENEITY IN AGRICULTURAL RIPARIAN BUFFER STRIPS

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Abstract

Riparian buffer strips (RBS) are widely recommended to protect surface water and groundwater in agricultural watersheds. However, RBS which prove effective at a field scale may not always lead to the expected results at a watershed level, because the scale of study influences the measurement of effectiveness. Leaching plot studies using hydrological partitions between parcels (referred to as controlled conditions) suggest that narrow RBS may be efficient at retaining nutrients and pesticides, but this has not been demonstrated in an uncontrolled field study in northern latitudes where spring melt brings heavy and rapid runoff discharge, while summer months are hot and dry. The objective of this research was to characterize the spatio-temporal heterogeneity of hydrology near fields with conventional agro-fertiliser inputs, only irrigated by natural precipitations and without terrain alteration to capture runoff (i.e. no partitions between parcels, no transverse ditch to collect all runoff). Within this agricultural riparian buffer strips experimental settings, we aimed at ensuring that RBS vegetation could intercept runoff and groundwater leaching from agricultural fields. Two sites, one with a mineral soil (Saint-Roch-de-l'Achigan) and one organic rich (Boisbriand), were equipped with water sampling equipment (passive runoff collectors, suction cup lysimeters and piezometers), and monitored between summer 2011 and spring 2014. The sites have 3-m-wide RBS vegetated with herbs or willows bordering corn/soy fields. Furthermore, drainage lines from surface runoff may on average intersect the RBS perpendicularly, but they are subject to wide local heterogeneity. Groundwater may be closer to the potentially active root zones at snowmelt, but it can also flow very little from the fields to the streams at this period. Furthermore, in some cases, groundwater flow does not occur perpendicularly to the RBS. Under low water regimes with stream/phreatic connectivity, water can even flow in the opposite direction, from the stream to the field, a situation that could affect agro-chemical filtering potential efficiency of the RBS. The variability of groundwater flow direction can be influenced by stratigraphy, localized soil physico-chemistry and historical modifications of agricultural stream beds. Hence, assessing the spatial heterogeneity of surface water flows, and temporal heterogeneity of groundwater flow is essential for proper characterization of RBS potential

efficiency in intercepting agro-chemical pollution, and assumption that water flows across vegetated RBS from field to the stream should always be verified.

Keywords

Riparian buffer strips, surface water, groundwater, agricultural watersheds, runoff, spatio-temporal heterogeneity, multi-scalar perspective

Abbreviations

- Riparian buffer strips (RBS);
- Herbaceous vegetation treatment (CX);
- *Salix miyabeana* SX64 at 33 333 stumps/ha (3X);
- *Salix miyabeana* SX64 55 556 stems/ha (5X);
- Saint-Roch-de-l'Achigan (SR);
- Boisbriand (BB);
- kilometers (km);
- hectares (ha);
- centimeter (cm);
- millimeters (mm),
- micrometers (µm);
- saturated hydraulic conductivity (K_{sat});
- Ministère de l'Agriculture, des pêcheries et de l'alimentation [Ministry of Agriculture, fisheries and food] MAPAQ;
- degré Celsius (°C);
- Degree-days of growth (°C·d);
- edge-of-field (CF);
- middle of the buffer strip (CC);
- close to the river (CR);
- in the field (CS);
- high density polyethylene (HDPE);
- polyvinyl chloride (PVC);
- Global positioning system (GPS);
- differential Global positioning system (dGPS);
- Digital elevation model (DEM);
- Altitude (z);
- Angle of incidence (θ_{CF} and θ_{CR}) measured in degrees (°);
- deviation from a perpendicular transect ($\theta_{\perp CF}$ and $\theta_{\perp CR}$);
- Three-dimensional (3D);
- Identity (ID);
- Triangulated irregular network (TIN);
- Mean ($\overline{X}$);

A4.1 Introduction

A4.1.1 The role of RBS and intrinsic hydrological processes

Riparian buffer strips (RBS) are one of several best management practices for the protection of surface waters (Moore et al. 2008; Bentrup 2008) and are recommended in agricultural regions around the world to mitigate non-point source pollution (Hickey and Doran 2004; Smethurst et al. 2009). Multidisciplinary perspectives are essential to the design, study and evaluation of RBS efficiency (Stutter et al. 2012). Therefore, addressing hydrologic flowpaths, together with chemical and biological processes, is essential to better understand riparian zone functioning (Hill 2000). Hence, to characterize 3-m-wide RBS potential efficiency to mitigate leaching of nutrients (Chapter 2) and glyphosate-based herbicide (Chapter 3), it was deemed important to integrate essential aspects of hydrology (current chapter). Measuring buffer strip potential efficiency primarily depends on the targeted buffer strip function and measurements can be done in various ways.

A4.1.2 The influence of scale on agricultural RBS hydrology

Buffer strips may be studied from a multi-scalar perspective (Wiens 1989), ranging from laboratory studies focusing on processes in controlled settings, i.e. Ausland (2014); Gomes et al. (2015), to watershed or catchment studies (Smethurst et al. 2009; Ratté-Fortin 2014; Uriarte et al. 2011; Terrado et al. 2014; Dosskey 2001) encompassing or levelling local heterogeneities (Wiens 1989; Baker et al. 2001) to make real-life assessments or predictions of global efficiency (Norris 1993; Verstraeten et al. 2006; Smethurst et al. 2009; Baker et al. 2001) (Annexe 20, a-c). The current study focused on the intermediate scale sometimes referred to as field scale (Lee et al. 2003), plot scale (Gasser et al. 2013) or along transects (Munoz-Carpena et al. 1999; Osborne and Kovacic 1993). This scale allows to study the transverse passage of water and diffuse pollutants through the RBS (Lee et al. 2004) and is predominantly studied in the literature (Stutter et al. 2012; Dosskey 2001). Most studies relying on runoff plots and confined field experiments demonstrate some efficiency in filtering a variety of contaminants in the runoff (Norris 1993). However, agricultural catchment studies surprisingly found limited effect of RBS to control surface water pollution, despite general successes at the plot scale (Norris 1993; Verstraeten et al. 2006; Stutter et al. 2012).

A4.1.3 Determinants of field runoff and RBS capture

Determinant factors affecting RBS efficiency from a hydrology perspective include precipitation, flow convergence, infiltration rate, water storage capacity, topography and vegetation cover (Polyakov et al. 2005). The RBS vegetation may enhance infiltration (Dosskey et al. 2010), deposition of sediment (Polyakov et al. 2005) and soil-bound agro-chemicals (Krutz et al. 2005) and grasses in particular, disperse convergent overland flows (Lowrance et al. 1997; Dosskey et al. 2010), while shrubby vegetation promotes more evapotranspiration (Allen et al. 1998; Dosskey et al. 2010). All of these minimize leaching to surface waters (Krutz et al. 2005). The vegetation type, density and spacing can influence soil porosity (Dosskey et al. 2010) and sediment interception by the RBS (Polyakov et al. 2005).

Runoff (i.e., volume, sheet vs. convergent channel flow, heterogeneous agro-chemical concentrations) and groundwater (water table height and flow) may be difficult to measure in uncontrolled settings (Krutz et al. 2005), and thus require greater spatio-temporal characterization to assess RBS efficiency. Slope within the RBS is also considered a good indicator of trapping efficiency. Runoff flow convergence induces more concentrated surface flows that can overwhelm the RBS capacity (Polyakov et al. 2005; Michaud et al. 2005). These ephemeral gullies are inherent to topography and may become more permanent "classic" gullies under no-till practices (Dabney et al. 2006). In uniform width RBS, some zones with larger source-area (Herron and Hairsine 1998; Dosskey et al. 2002; Polyakov et al. 2005) due to converging flow paths, end up insufficiently protected (Dosskey et al. 2002; Polyakov et al. 2005), arguing in favor of precision RBS with a width optimized for actual terrain characteristics

(Polyakov et al. 2005). Though width variability has extensively been studied, the inherent small topographic variability of the source area in a natural field with a fixed RBS width has not yet been studied.

A4.2.1 Local influences on ground water movements within an RBS

In humid climates where aquifers are connected to rivers, groundwater generally flows laterally towards streams (Winter et al. 1998). Though some substrates allow faster water movement, groundwater movement is generally slower than aboveground runoff (Winter et al. 1998; Dosskey et al. 2010). As different soil layers with different hydraulic properties can dictate how the water migrates horizontally and vertically, this in turn influences pollutant residence time, interaction with the root zone, interaction with organic rich or microbiologically active horizons and subsurface leaching, which all affect the efficiency of the RBS in mitigating underground diffuse pollution (Polyakov et al. 2005). For instance, a high water table alone is not sufficient to predict denitrification in RBS (Vidon and Hill 2004), but pairing with elevated dissolved organic carbon measurements gives a better prediction potential for denitrification in micro-anaerobic hot spots (Burt et al. 1999; Hill 1996; Pabich et al. 2001). Sometimes, groundwater seeps to the surface, leading to a rapid flow across the RBS that does not allow efficient treatment of the water (Bentrup 2008). Alternately, for deeply incised streams, the groundwater may be too deep for the RBS vegetation to significantly intercept it (Bentrup 2008). To correctly assess the potential efficiency of an RBS, historic land disturbances, restricting soil layers, preferential groundwater flow paths and other features that control diffusion and infiltration of dissolved or particulate aqueous pollutants should be considered (Polyakov et al. 2005).

Subsurface drainage may contribute to a direct export of nutrients (i.e. P; King et al. 2015), effectively bypassing vegetated buffer strips (Osbome and Kovacic 1993). However, tile drainage is not present everywhere, accounting for 5-100% of fields depending on the area (McCorvie and Lant 1993; Zucker and Brown 1998; King et al. 2015; Harker et al. 2004; Herzon and Helenius 2008). The exact proportion of row crop lands with tile drainage is poorly quantified, hence it is critical to complement RBS potential efficiency studies conducted in drained areas (Terrado et al. 2014) with some studies conducted in non-drained areas. The current study was conducted in agricultural settings without drainage tiles (Chapter 2 and 3), and this justified the need to characterize hydrological heterogeneity.

A4.2.2 Goals

This study was performed within a larger project designed to identify environmental parameters which may influence nutrient (Chapter 2) and pesticide fluxes (Chapter 3) and RBS interception. A short literature review (above) introduced the necessary background to understand the scale in which the current study falls, as well as the determinants of runoff and groundwater movements. The rest of this chapter aims at supporting statistical analysis and interpretation within other chapters of the current thesis by providing specific data on hydrology. This study focuses of characterizing the hydrology of two fields and associated RBS within the St-Lawrence plain. The two sites have the same cool and humid temperate climate, but different soils and relief (one flat, the other hilly). It was designed to ensure that experimental parcels on each site were similar and that water flows across the RBS in a way that vegetation can intercept runoff and groundwater.

While the hydrological characterization was originally intended to be conducted anteriorly to the layout of the sampling equipments, it was eventually judged that such an analysis would delay the initiation of water sampling and would thus restrict the intended length of the sampling study (3 years). Hence, the hydrological characterization was conducted in parallel to the active water sampling, and efforts were made to have this characterization as detailed as possible to acknowledge for confounding variables in the aqueous concentrations data interpretations (hence the multivariate analysis design used).

The specific objectives are to assess the spatio-temporal heterogeneity of hydrology. Topographic, stratigraphic and pedologic site characterization was performed on each experimental RBS parcel to quantify homogeneity of slopes, strata and physical properties of the soil within the experimental fields. The surface runoff spatial heterogeneity was characterized. The effect of time on groundwater flow and fluctuations was also assessed, with additional historic and stratigraphic considerations. Finally, the RBS runoff interception potential was estimated and different methods to quantify the contributing source-area were performed to assess their role on edge-of-field runoff collection potential.

A4.2 Methods

A4.2.1 Experimental sites

The experimental design on each site is a triplicate of random blocks with three treatments each: Herbaceous vegetation (CX), *Salix miyabeana* SX64 at 33 333 (3X) and 55 556 stems/ha (5X). The two experimental sites (Figure A4- 1) where corn and soy were cropped in rotation from 2011-2013 border two first order streams. The Moïse-Dupras stream in Saint-Roch-de-l'Achigan (SR: N45°50'48.3"; W73° 36'16.7"; alt. 46 m) flows towards the L'Achigan River 1.3 km downstream from the site. The Dumontier stream in Boisbriand (BB: N 45°36'39.8", W 73°51'40.3"; alt. 44 m) reaches the Des-Milles-Isles river 4.8 km downstream. SR has a relatively flat topography, a deeply incised artificially dug drainage ditch, while BB has a gentle hilly topography. The regional watershed of the streams in the vicinity of the RBS (not necessarily draining through the RBS) is ~10.1 ha in BB and ~8.3 ha in SR. The Dumontier stream has been rectilineated since the 1930's, and flows through an ancient wetland, as evidenced by aerial photographies of the site (Figure A4- 2).

Climatic parameters from regional Environment Canada stations — Sainte-Thérèse Ouest 6.8 km from BB and L'Assomption 13.8 km from SR — were extracted from Agro-Meteo online database (Lepage and Bourgeois 2011). From 2010 to 2013, mean temperatures 7.5 °C and 7.0 °C; degree days of growth 990 °C·d and 989 °C·d and precipitation 1034 mm and 1121, for BB and SR respectively were statistically comparable (see Chapter 1).

A4.2.2 Surface and groundwater sampling

A total of 36 surface water collectors, 72 lysimeters and 24 piezometers were designed, installed at the edge-of-field (CF, acting as a reference) and close to the river (CR) (Figure A4-) and sampled as described in Chapter 2. Surface runoff was collected in high density polyethylene (HDPE) buckets buried over three quarters of their height in the ground and fitted with polyvinyl chloride (PVC) gutters sheltered from rain and extending at the soil surface, perpendicular to the buffer strip, over a length of 60 cm, and equipped with 2 mm nylon mesh to restrict coarse particles. At the time of sampling, the total volume of water collected was estimated in situ, with a ruler. A statistical analysis was conducted to check if the runoff volume collection changed between sites and/or was influenced by the side of the RBS (CF vs. CR). Soil water was collected in polyvinyl chloride suction lysimeters (Soil Moisture Equipment Inc, 1900L, Santa Barbara, CA, USA) equipped with ceramic cups buried at 35 or 70 cm depth. The groundwater level was recorded at every water sampling campaign from 2011-2014 (± 0.5 cm). The piezometers were installed on the outer margin of each RBS field block, which comprised three treatments.

A4.2.3 Topography and slopes

The precise topography of the buffer strip and neighboring fields was obtained in July 2011 with a Trimble differential GPS with a base fixed near the center of the study area (R8GNN base and rover, Sunnyvale, California, USA). The vertical precision of the instrument is approximately 1 cm (USGS (United States Geological Survey) 2013). To obtain a precise topography in order to model surface runoff close to the buffer strips, the sites were surveyed every 0.5 m to localize the exact positions of the water collecting devices, soil cores, and buffer strip margins, and to account for important hydrological features (i.e. rock chutes, which are engineered passages to prevent erosion in preferential runoff pathways, or other obvious water flow passages). A coarser sampling interval (~15 m grid) was used over the proximal regions of the adjacent fields in order to establish the area drained by each experimental buffer strip. Finally, a regional DEM was used to confirm regional water flow over the width of the whole field as GPS data was lacking at the extremities of both sites. For better results, a regional 1:50 000 DEM was transformed into vector data (isoline), so it could be integrated to the interpolation to get more accurate results. The regional DEM for BB was obtained from the database of the local watershed comitee OBAMIL (Louis Tremblay, Comité de Bassin Versant de la Rivière des Milles-Isles, personal communication) and for SR it was obtained from the regional municipality geomatic services office (Adam Pelletier, MRC Montcalm, personal communication).

Three scales of DEMS were created (Annexe 20,d-f) to visualize the terrain, understand where surface runoff could flow and what surface area from the field contributed to the surface runoff collectors and could be intercepted through the RBS using ArcGIS (version 2.1.4, Esri, Redlands, California, USA. The finest RBS scale (1:250) had a precision z = 1 cm, xy = 10 cm and resolution of 10 cm. The intermediate scale called proximal (1:1000) had a precision z = 1 m, xy = 10 m and a resolution of 1.5 m. Finally, the Field scale (1:30 000) had a precision z = 1 m, xy = 10 m and a resolution of 1.5 m). Slopes within the RBS were calculated using z values of proximate water sampling equipment pairs (on the CF and CR sides of the RBS). Slopes further infield were estimated by extending a 5 m transect from the CF sampling equipment within the field at a 90° angle relative to the stream and extracting the corresponding point value from the DEM. The homogeneity of slopes across sites and treatments was tested statistically.

A4.2.4 Surface runoff and basins

To simulate the directions of surface runoff, the r.watershed tool of GRASS GIS (version 6.0, Champaign, Illinois, USA) was used for basin visualization and definition of stream flow channels. The minimum basin size was set to allow approximation of streamflow channels in as much details as possible without overcrowding the visualization.

In a second phase, the smoothened DEM was used as the workflow input in ArcHydro Basic Dendritic Terrain Processing (version 2.0, esri, Redlands, California, USA). Modeling based on the DEM generated from field sampling positioned the stream on its historic position (which was inherent to the natural topography). Because this was in slight disagreement with the sampling data codification (i.e. points on the stream margin vs in the RBS or field were tagged manually), the stream and drainage ditch correct position were set in the model. The input DEM used in surface hydrological modeling was reconditioned according to the AGREE method (Hellweger and Maidment 1997) used in the ArcHydro extension (Version 2.0 beta). To avoid distortion in drainage paths caused by minor depressions, which are interpolation artefacts, the function "fill sinks" was used to smooth the surface. Then the surface water flow directions were calculated using the three resolutions of DEM created above (RBS, proximal and field), by adjusting the number of working cells to the scale of each modeling (ranging from 500 to 1000 for the field or field scales for both sites). Using the most representative scale (i.e. proximal scale, see results), drainage line angles of incidence with the RBS and microbasin surface were calculated.

The drainage line angles of incidence (°) with the RBS were estimated with a protractor on both edges of the RBS (θ_{CF} and θ_{CR}) as illustrated in (Figure A4- 3). The corresponding deviation from a perpendicular transect ($\theta_{\perp CF}$ and $\theta_{\perp CR}$) was then calculated to account for drainage lines which change direction as they cross the RBS. We validated statistically that the incidence angles were similar across sites, sides and treatments.

The drainage area of microbasins (m^2) draining towards surface water collectors of each experimental buffer strip was computed in ArcGIS. Using the output of the hydrological modeling, four methods were used to estimate microbasins drainage area: basins (catchment — surface drained by smaller arms of the drainage lines), nearest stream, affiliated basins (BB only, (including several smaller basins — adjunct catchment— including larger ramifications of the runoff)) or drainage points (SR only, manually located points positioned on the drainage lines — i.e. rock chutes created by farmers to favor drainage from fields to stream — from which the software can compute drainage surface). The homogeneity of the drainage microbasins across sites, side and treatments was validated statistically.

A4.2.5 Groundwater level and flow

In order to assess if lysimeters were in the saturated soil region, the water table measured in each piezometers of the RBS region were interpolated using the "Topo-to-raster" function of ArcGIS. In BB, a model in which the water table was connected to the stream was tested in addition to the RBS interpolations due to the presence of a visible resurgence zone in the eastern section of the RBS. Water table depth near each water sampling equipment were then tabulated ("extract-value-to-point"). Depth relative to surface was used directly in other analysis including submersion of the water sampling equipment. A lysimeter was considered submerged when the water table level was at least 10cm above the ceramic porous cup (accounting for z measurements precision). Groundwater flow direction, head differences were calculated (z_{CF}-z_{CR}). Groundwater flow was considered directional, rather than stagnant, only if the difference in elevation from the CF to CR sides was greater than ≥ 20 cm.

A4.2.6 Stratigraphy

The buffer strips were established in a typic humisol in BB (derived from an ancient wetland) and in a mineral sandy clay-loam sitting atop a clay bed in SR (Soil Classification Working Group 1998). The granulometry (Table A4- 1) was characterized at the surface and at 35 cm depth according to the wet sifting methodology adapted from Centre d'expertise en analyse environnementale (CEAEQ) (2010) which included dissolution of organic matter with 30% H_2O_2 and the use of dispersing and anti-foaming solutions. The 2 mm, 212 μm and the 63 μm sieves were used and the sand and silt fraction of a surface sample was further differentiated with a sedigraph (Analysette 22 Compact Laser Particle Sizer, FRITSCH, GmbH, Germany). BB neighboring field drainage ranges from good to imperfect while SR is imperfectly drained (Gagné et al. 2013). Because BB soil in the vicinity of the RBS was much different from the rest of the field, in situ Guelph permeameter (Soil Moisture, Model 2800K1, Santa Barbara, CA, USA) measurements of the saturated hydraulic conductivity (K_{sat}) was conducted for the surface soils (0-10 cm). The K_{sat} of all soil series mapped within the limits of the BB and SR fields were taken from the literature (Gagné et al. 2013; MAPAQ 1990).

The stratigraphy was characterized at every 10 cm depth during the installation of the water collecting devices in May 2011 and then again, during water sampling campaigns from 2011-2013. Soil cores were collected near the stream (CR), in the middle of the buffer strip (CC), next to the buffer on the side of the field (CF) and finally in the field itself (CS), at a minimum distance of 1,5 m from the water sampling equipment to minimize disturbance. Gross granulometric observations, compaction and color (Munsell Soil Color chart) were used to classify the samples collected. A 3D representation of the sites was built using GMS (v10.0, Aquaveo™, Provo, Utah, USA). Each borehole was assigned a soil ID and a horizon ID (at the contact between the layers). Cross-sections were automatically generated and filled, and manually reviewed. The GPS data was used to generate a TIN with linear interpolations. Each cross-section was snapped onto the topographic model (snapping is program function aiming to make the cross-section coincide with the surface elevation of the DEM) prior to being transformed in a solid. Transects centered on the CF and CR axis, as well centered mid-distance on each RBS where the water sampling equipment is located, were manually positioned.

A4.2.7 Statistical analysis

The runoff volume and incidence angles with the RBS were analyzed with an ANOVA on side (CF vs CR), treatments (CX, 3X vs 5X) and interactions. When data did not conform to the normality assumption, non-parametric estimations on ranks were conducted. The slopes, absolute slopes, "basins", and "closest streams" microbasin surface area were tested with an ANOVA on site (BB vs SR), treatments (CX, 3X vs 5X) and interactions. The "affiliated basins" in BB and rock chute "drainage points" in SR were log transformed to fit normality and analyzed for treatment effect with an ANOVA. All statistical analysis was conducted using JMP 7 (SAS Institute, Cary, NC).

A4.3 Results and Discussion

A4.3.1 Topography

Slopes within the RBS at the two sites are > 0.5 - 2 %. In the vicinity of the RBS (up to 5 m into the field), slopes in BB vary from 0 to 5%, while they vary from 0 to ≥ 15 % in SR. Approximately 50 % of the terrain is nearly leveled (> 0.5 - 2 %) at both sites (Figure A4- 4). The wider range of estimated slopes in SR is caused by localized minor mounts and small depressions, the terrain being much more levelled at the field scale than in BB. Topographic minima and maxima range from 48.4 - 54.2 m in SR with a sharp > 2 m drop from the buffer riverside CR edge to the actual stream level. In BB, the minima and maxima range from 35.7 - 41.9 m but the major elevation differential is in the fields and the drop from the buffer edge close to the river (designated as CR) is less than 0.5 m. Overall, neither site ($p = 0.9400$) nor the RBS vegetation treatments ($p = 0.0723$) had statistically different slopes. However, absolute slopes (which can affect residence time, but are independent of slope direction) were significantly smaller in BB ($p = 0.0008^*$) and there was an interaction with the RBS vegetation treatment ($p = 0.0032^*$). Within the RBS, we did not observe slopes > 6 % which may fail to retain sediments (Polyakov et al. 2005) because they lead to higher overland flow velocity while minimizing infiltration and particle deposition (Knies 2009). Because slopes were relatively uniform across sites, sides and treatments (despite some variability within absolute slopes), we are confident that this parameter.

A4.3.2 Stratigraphy

A total of nine soil types were observed across both sites (see Figure A4- 5 for 3D stratigraphic rendering). In BB, black histosol, brown histosol, peat, marl, rocks (till) and clay were observed from surface to the bottom. In SR, sandy loam, clean sand lentils and clay with traces of iron oxides (FeOX) were observed from top to bottom. Though surface soil appeared homogeneous on both sites, below ground soil strata varied slightly between parcels. While black, brown histosol and peat are mapped differently, they represent arbitrary stages on a continuum of organic soil pedogenesis, with the black histosol being the most humified form. Hence, apparent changes between stratigraphic layers 3D representation represent a transition of peat oxidation stage rather than abrupt physico-chemical changes. On the other hand, rocks (likely washed till) found near the F-F', E-E' and to a lesser extent east of the C-C' transects may have a more important impacts on groundwater movements, which may be explained by the historic position of the stream (Figure A4- 2). Organic-rich soil generally surrounds the 30 cm lysimeters while marl and/or clay surrounds the 70 cm lysimeters.

A4.3.3 Surface runoff

Drainage lines at the RBS, proximal and field scales (Annexe 20) appear coherent with each other (Annexe 21). In particular, the proximal and field scales in SR are almost exactly superimposed within the model limits. The RBS scale was modeled with 10X greater precision (± 0.1 m) than the proximal scale, and there are likely several water flowpaths across the RBS, and perhaps not only nearly unique concentrated flow paths as suggested by the other scales. The larger the scale, the more likely realistic values will be obtained due to minor spatial heterogeneities levelling, but the more likely micro-site specific process variability will be lost (Krutz et al. 2005). The narrow limits of the RBS scale lead to several potential hydrologic flow path artifacts (i.e. water appearing to drain from the RBS to the field contrary to other modeled scales; small and unconnected drainage lines intercepted by the RBS model limits). Furthermore, the finer drainage lines output from the narrow RBS scale are likely to change with time. Thus, the narrow RBS scale may not be very instructive for the purpose of modeling runoff over a few years necessary for the quantification of nutrient (Chapter 2) and glyphosate (Chapter 3) retention by the RBS. While the runoff flowpaths obtained based on the regional DEMs suggested heavy flow convergence within the RBS, this was not observed during rainy day field visits. Concentrated flows may be observed in the majority, but not all, of agricultural RBS (Dosskey et al. 2002). The field model, which had a lower vertical precision of 1 m and resolution of 1.5 m, could not e entirely circumvented to calculate the extent of adjunct microbasins (those basins which extend beyond the region encompassed in the proximal model) in BB. Hence, the proximalscale (which still relied on the dGPS data with 0.01 m vertical and 1 m horizontal precision with a 50 cm resolution) was judged best for the characterization of surface runoff flow path across the RBS, and most of the calculations of microbasins surface area described below. At the proximal scale, the ephemeral cropland gullies visible may be somewhat more permanent, though not necessarily to the extent of becoming severely eroded classic gullies (Dabney et al. 2006).

The spatial variability in preferential surface runoff can be appraised when considering all the field runoff intercepting the RBS (Figure A4- 6). There was no significant difference between side and treatment on the overall incidence angle (θ; ANOVA), and though there is local variability in each parcel relative to the perpendicular transects across the buffer strip ($\theta_{\perp}$), there was no significant difference, which could be linked with side or treatment (testing for ranks on paired data). This means that globally, the incoming runoff crosses the buffer strip in a perpendicular fashion (~ 90°), but on a local scale incoming (CF) and exiting (CR) preferential surface flows may enter and exit the buffer test parcels at variable angles. This is critical for the statistical analysis of RBS potential efficiency in mitigating nutrients (Chapter 2) and glyphosate (Chapter 3).

Microbasins were on average smaller in BB than SR ($p < 0.0001$* for both "basins" and "closest streams" models Table A4- 2and Annexe 18). Though the "stream" model did not reveal surface area differences between treatments ($p = 0.3897$), the "closest stream" model revealed that drained area was statistically larger for the RBS composed of 5 rows of willows (5X), than the RBS with 3 rows of willows (3X) (the herbaceous treatment (CX) being statistically undistinguishable from each; $p = 0.0073$* with a significant interaction with site: 0.0102*). While calculations based on the drainage points superimposed on the rockchute (common erosion protection structure found on the edge of field in SR) in SR yielded similar results (CX = 5X ≥ 3X; $p = 0.0009$*), affiliated basins in BB were statistically larger in 3X and smaller in 5X (CX not distinguishable; $p = 0.0408$*). While the surface collectors were not installed specifically where the hydrological model suggests the passage of concentrated runoff (because the model was built only after the installation of the sampling equipment) the surface runoff collectors were nevertheless efficient in catching the water that flowed through. A single method for calculating microbasins may fail to be ubiquitously applicable. For instance, affiliated basins (which enable to calculate the extended region draining into the RBS using the field model's less precise data in a few regions where the proximal model appeared too narrow to fully capture the whole surface area of the micro-basin) were only pertinent in BB, and drainage points on rock at the nearest chutes (engineered erosion control systems in place) could only be positioned in SR. Finally, some automatically generated hydrological microbasins may be relatively small (1-72 % smaller) compared to what can be expected if the closest modeled runoff path effectively intercepted the collector. However, there was not always a runoff stream located in the realistic vicinity of the runoff collector, and hence, three SR parcels could not be attributed a surface area under the "closest stream" model. This is an inherent limitation to the subsequent use of these modeled data to interpret the RBS potential efficiency to filter aqueous fluxes of nutrients and glyphosate.

Because of site specific differences in topography, stratigraphy, microbasin sizes and K_{sat}, runoff volumes were analyzed independently for both locations. The 2011 runoff volumes (recorded on eight occasions;Figure A4- 7) were unaffected by side ($p = 0.7204$) and treatment ($p = 0.3320$) in BB, but the RBS significantly reduced runoff volumes in SR (side: $p = 0.0110$*) even though it was again irrespective of treatment ($p = 0.7005$). Except for a very weak but significant regression between runoff volume collected on the edge-of-field and microbasins size based on the "closest stream" model in BB ($r^2 = 0.10$, $p < 0.0001$*, $n = 162$; Figure A4- 8), no other significant relationships between runoff volumes and slopes of microbasins size models were found (data not shown). Runoff volume could not be linearly related to any source area measurement models, except in BB where the "nearest stream" microbasin model was significantly related to collected runoff volume. Several others found a direct relation between runoff volume and source area (Herron and Hairsine 1998; Dosskey et al. 2002; Polyakov et al. 2005). Two reasons may explain the absence of a link between runoff volume and source area. First, experiments under uncontrolled field conditions where no restrictions on surface waters are imposed by partitions between parcels or via interception of all runoff and infiltrated water with transverse ditches will lead to more variable water capture sampling. This may eventually affect how we interpret RBS potential efficiency between uncontrolled versus controlled conditions. For instance, the effective area of an RBS (through which water actually flows) may be only a fraction of the gross RBS area (the whole vegetated surface adjacent to the stream), especially if concentrated runoff occurs (Dosskey et al. 2002). Secondly, though we tried various models to calculate source area (the size of the micro-basin draining towards the RBS or water samplers), based on the most precise and pertinent scales, these estimations remain strongly dependent on the accuracy of the dGPS sampling and constructed topographic models. Hence, it may not be excluded that the lack of predictive power for runoff volume collection based on source surface area could be due to model assumptions. Assuming that the whole (gross) RBS area (54 m^2) contributed to runoff interception, our source area : RBS area ratio oscillated from 0 - 17.8 based on the "closest stream" model. However, if we only consider the 60 cm gutter to efficiently intercept runoff which flows across the 3-m width RBS (effective area of 1.8 m), then our source:RBS area ratios (effective area) oscillate between 0 and 958. The majority of previous studies varying the source area: RBS ratio in a controlled manner (range 5 : 1 – 45 : 1) found that ratio did not influence RBS potential efficiency significantly, because of variability in the infiltration rates across studies (Krutz et al. 2005). Hence, studying this ratio within an apparently uniform field where the source area varies naturally due to inherent topography was hoped to control for the across-site variability of earlier studies and allow for better discernment of the source area effect, but this was not the case. In our study, we intended to target the source area, or micro-basins, draining towards the RBS or water samplers, to take into accounts the fact that some of the water from the larger field watersheds (~10.1 ha in BB and ~8.3 ha in SR) was draining towards ditches rather than the RBS.

A4.3.4 Water table

As expected, groundwater levels are higher in the spring than in the summer (Figure A4- 9). Our observations suggest connectivity of the groundwater and surface water at BB, visible by a resurgence zone in the eastern region of the study area. The groundwater generally moves from the fields to the stream (except in driest periods in BB), though once again, not necessarily at an angle perpendicular to the buffer strip. During snowmelt in SR, water in the saturated soils did not appear to flow in half of the sampling zones, as evidenced by the lack of a gradient from the CF to CR sides. Furthermore, the variability of groundwater movement may be influenced by stratigraphy and localized soil physico-chemistry (Figure A4- 5) and historical modifications of agricultural stream beds (Figure A4- 2). Hence, spatio-temporal heterogeneity of the groundwater flow needs to be taken into consideration in the interpretation of RBS potential efficiency to remove nutrients (Chapter 2) and glyphosate (Chapter 3).

A4.3.5 Implications of the spatio-temporal heterogeneity of aqueous fluxes in the evaluation of RBS potential efficiency to mitigate nutrients and glyphosate

The common assumption that most runoff reaches a buffer enters the buffer and flows through it (except for a portion that infiltrates) perpendicularly appears erroneous, based on our observations and those of Dabney and Vieira (2013) before us. We have demonstrated that over the proximal field scale, modeled runoff incidence does enter and exit the RBS at a near perpendicular angle, however, within each parcel, the runoff incidence angle deviates widely from the expected perpendicular flow. This appears critical to truly appreciate the potential efficiency of the RBS in the two subsequent chapters on nutrients (Chapter 2) and glyphosate (Chapter 3). The observations made herein suggest a specific potential efficiency calculation to avoid the confounding effect of local heterogeneities. The movement of surface water in the field influences the ability to collect runoff in the surface sampling equipment, and this is especially critical on a local scale where sampling equipment before the buffer strip may receive more or less water than the equipment on the other side of the buffer, due to local topography/hydrology and not to specific buffer strip treatments. However, because the mean incidence angle is perpendicular to the buffer strips at the regional field scale, pooling data before the buffer strip should minimize the confounding effects of local heterogeneities. Our initial plan to pair proximal stations before and after the buffer strip did not appear pertinent after analyzing the modeled trajectory of the surface runoff. Hence, scaling up to analyze mean pollutant loads before and after the buffer strip should minimize concentration variability which would have otherwise been exacerbated in a paired statistical design. Paired designs would have been fit for hydrologically isolated experimental plots, but this was not the case here. Pollutant removal effectiveness of the RBS is often measured in a way which appears more akin to an unpaired design, that is by measuring pollutant loads in presence versus absence of an RBS in plots that are running parallel (Lee et al. 2003; Munoz-Carpena et al. 1999; Noij et al. 2012; Uusi-Kämppä and Yläranta 1996; Duchemin and Hogue 2009). For others (McKergow et al. 2006; Sabater et al. 2003; Hook 2003), RBS effectiveness is calculated as the difference between the input and output volumes or concentrations, and expressed as a percentage of the initial input value.

This formula is sometimes normalized per meter width of RBS to allow inter-site comparisons (Sabater et al. 2003). Where surrogate runoff is applied (Dosskey et al. 2007; Schmitt et al. 1999) RBS input concentrations can be estimated from the tank mix. When partitioned runoff plots are used (Dosskey et al. 2007; Patty et al. 1997; Duchemin and Hogue 2009; Schmitt et al. 1999), sheets of metal physically separate the parcels and all the runoff from the source area (minus infiltrated water) is assumed to be collected after the RBS. However, in uncontrolled settings with natural rainfall, the influx and outflow concentrations may not be homogeneous, hence we suggest that averaging edge-of-field or influx concentrations over the whole field region may compensate for localized heterogeneity leading to high or low concentrations in the influx which does not necessarily migrate from the field to the stream perpendicularly to the RBS. This is somewhat similar to the approach of McKergow et al. (2006) who reported aggregate concentrations and loads rather than individual plot values to minimize the spatial variability among multiple RBS plots, but who contrary to us still used a paired statistical design. Hence, the methodology used in the Chapter 2 and 4 uses the average incoming and outcoming concentrations at the field scale, rather than a statistical design solely based on geographic proximity pairing.

Sampling of discrete points in the field may have missed small ridges, berms and furrows which could have altered the modeling of surface runoff, but we are confident that our observations provide sufficient basis for reasonable interpretation of runoff path, as Dosskey et al. (2002) pleaded before us. In reality, tillage and other agricultural mechanical activities are performed parallel to the RBS and inevitably form berms at the leading edges of the RBS (Dabney and Vieira 2013). These berms can then act as linear roughness elements, which can interact with topography and soil properties and eventually alter runoff movements. To minimize imprecision of our field source calculation models under the influence of berms growth over time (due to the annual passage of farm machinery (Vieira and Dabney 2011), we limited our RBS runoff capture and models of the source area analysis to the 2011 year where the topographic measurements were conducted. RBS renovations can avoid the persistent effect of these berms (Dabney et al. 2006; Bentrup 2008).

A4.4 Conclusion

The objective of this study was to ensure that experimental parcels used in parallel agro-chemical removal potential efficiency studies were similar and that the water flowed across the RBS in a way where vegetation could intercept runoff and groundwater. This is important for the interpretation of nutrient (Chapter 2) and glyphosate (Chapter 3) removal potential efficiency. The assumption that water flows from the fields to the streams in a nearly perpendicular fashion is not verified everywhere. However, when averaging all runoff stream at the proximal field scale, the runoff streams globally appear to cross the RBS at a perpendicular angle. Hence, pooling the results of localized water samples quantifying agro-chemical concentrations may help to buffer the localized heterogeneity of surface runoff. This strategy was adopted in Chapters 3 and 4. Based on the closest stream assumption, the micro-basins draining towards the RBS were 18.6 ± 25.9 m^2 in BB and 676.9 ± 745.2 m^2 in SR (micro-basins were significantly greater in SR). Modeled surface runoff flow paths suggest concentrated water passage locations (though no permanent gullies were visible on these areas in the field), and the greater a source-area is, the greater a total quantity of nutrients passing through the RBS at a specific point might be.

Below surface runoff, phreatic waters may also deviate from the implicit assumption that water should flow from fields to stream. Phreatic waters may indeed flow from fields to stream in a nearly perpendicular fashion most of the time. However, soil saturation in the spring may lead to little horizontal water movements (i.e. SR), heterogeneous soil stratigraphy may lead to flows that are not necessarily perpendicular to the RBS (i.e. BB), and connectivity with regional aquifers may lead to water flowing from the streams to the fields in the driest summer months (i.e. BB). Bank storage (i.e. underground flow from stream to field) may also occur due to temporary flood peaks or intense evapotranspiration by the streamside vegetation (Winter et al. 1998). Hence, when the water table below the fields and RBS are low due to low precipitation or intense evapotranspiration, an underground source emerging from a confined aquifer or an intense precipitation pulse, may lead to flow reversal. This was taken into account in the RBS potential efficiency presented in Chapters 3 and 4. Furthermore, historical straightening of streams may alter the normal hydrogeology, such that groundwater may flow in its natural course if the substrate is more conductive there, despite what is apparent from the above ground superficial modifications of the stream bed (see Figure A4- 2). As preferential groundwater flow channels may be the cause of the observed subaqueous spring (Winter et al. 1998), this is a plausible explanation for the groundwater flow reversal post-glyphosate application in BB.

A4.5 Acknowledgements

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A4.6 Tables and Figures

Table A4- 1: Granulometry and saturated hydraulic conductivity (K_{sat}) at Boisbriand and Saint-Roch-de-l'Achigan.

Granulometry was obtained by sifting across indicated diameter mesh, and a proportion of silt of 72.2% and 76.7% was observed, in Boisbriand and Saint-Roch-de-l'Achigan respectively, in the smallest fraction using a sedigraph. K_{sat} was measured in Boisbriand using a Guelph Permeameter, on the field-edge (CF) of the riparian buffer strip. Other K_{sat} values by soil series were obtained from the litterature (1) Gagné et al. (2013), (2) MAPAQ (1990).

			Boisbriand			**Saint-Roch-de-l'Achigan**		
Parameter	Depth (cm)	Coarse sand < 2 mm	Fine sand < 212 µm	Silt and Clay <63 µm		Coarse sand < 2 mm	Fine sand < 212 µm	Silt and Clay <63 µm
Granulometry	0	6,1	13,3	80,5		43,2	30,1	26,7
	35	6,4	13,9	79,7		37,2	33,3	29,4
Soil series		CF	Dalhousie[1]	Châteauguay[1]	Saint-Bernard[1]	Achigan[1]	Achigan[2]	
K_{sat} (cm/h)	0-10	0.03 to 4.02	N/D	N/D	N/D	N/D	N/D	
	0-30	N/D	0.53	4.00	8.00	0.61	1.30	
	30-40	N/D	0.12	2.33	4.28	1.50	1.31	
	>40	N/D	0.47	2.00	N/D	N/D	N/D	

Table A4- 2: Source area dimensions based on four different models and source area: RBS ratio considering the effective area or total RBS area.

Station	Site	Block	Side	Trt	Source area (m²) *based on 4 different model estimations* basin	closest stream	affiliated basins (BB only)	drainage point near rock chute (SR only)	RBS effective area (m²)	Source area: effective RBS ratio basin	closest stream	affiliated basins (BB only)	drainage point near rock chute (SR only)	RBS total area (m²)	Source area: total RBS ratio basin	closest stream	affiliated basins (BB only)	drainage point near rock chute (SR only)
73	BB	NE	CR	CX	11	8	60		1.8	5.9	4.5	33.6		54	0.1	0.1	0.6	
74	BB	NE	CR	5X	11	3	75		1.8	5.9	1.7	41.7		54	0.1	0.0	0.8	
75	BB	NE	CR	3X	30	3	287		1.8	16.7	1.7	159.6		54	0.3	0.0	3.0	
76	BB	NE	CF	3X	14	4	97		1.8	7.6	2.3	53.6		54	0.1	0.0	1.0	
77	BB	NE	CF	5X	20	83	144		1.8	10.9	46.0	80.0		54	0.2	0.9	1.5	
78	BB	NE	CF	CX	5	3	114		1.8	2.7	1.4	63.3		54	0.0	0.0	1.2	
79	BB	SE	CR	3X	8	4	82		1.8	4.7	2.4	45.7		54	0.1	0.0	0.8	
80	BB	SE	CR	5X	36	78	108		1.8	19.7	43.2	59.9		54	0.4	0.8	1.1	
81	BB	SE	CR	CX	15	21	57		1.8	8.3	11.8	31.7		54	0.2	0.2	0.6	
82	BB	SE	CF	CX	4	3	115		1.8	2.5	1.8	63.9		54	0.0	0.0	1.2	
83	BB	SE	CF	5X	28	22	176		1.8	15.7	12.4	98.0		54	0.3	0.2	1.8	
84	BB	SE	CF	3X	12	12	72		1.8	6.8	6.4	40.0		54	0.1	0.1	0.7	
85	BB	SW	CR	5X	8	5	76		1.8	4.5	2.6	42.5		54	0.1	0.0	0.8	
86	BB	SW	CR	CX	17	6	158		1.8	9.4	3.4	88.0		54	0.2	0.1	1.6	
87	BB	SW	CR	3X	19	56	285		1.8	10.3	31.3	158.1		54	0.2	0.6	2.9	
88	BB	SW	CF	3X	11	9	250		1.8	6.1	4.9	139.1		54	0.1	0.1	2.6	
89	BB	SW	CF	CX	19	9	130		1.8	10.7	5.2	72.1		54	0.2	0.1	1.3	
90	BB	SW	CF	5X	9	5	36		1.8	4.9	2.6	19.7		54	0.1	0.0	0.4	
Mean					**15,4**	**18,6**	**129,0**			**8,5**	**10,3**	**71,7**			**0,2**	**0,2**	**1,3**	
SD					**8,8**	**25,9**	**76,3**			**4,8**	**14,3**	**42,4**			**0,1**	**0,3**	**0,8**	
91	SR	NE	CR	CX	45				1.8	25.2	0.0			54	0.5	0.0		
92	SR	NE	CR	5X	86	1468		1485	1.8	47.6	815.5		825.1	54	0.9	15.1		15.3
93	SR	NE	CR	3X	204	121		204	1.8	113.5	67.3		113.5	54	2.1	1.2		2.1
94	SR	NE	CF	3X	204	186		204	1.8	113.5	103.4		113.5	54	2.1	1.9		2.1
95	SR	NE	CF	5X	86	1408		1485	1.8	47.6	782.5		825.1	54	0.9	14.5		15.3
96	SR	NE	CF	CX	57	54		198	1.8	31.9	30.2		110.1	54	0.6	0.6		2.0
97	SR	NW	CR	3X	81	63		81	1.8	45.1	34.9		45.1	54	0.8	0.6		0.8
98	SR	NW	CR	CX	263			<u>2374</u>	1.8	146.3	0.0		1318.7	54	2.7	0.0		24.4
99	SR	NW	CR	5X	22	1725		1734	1.8	12.3	958.5		963.5	54	0.2	17.8		17.8
100	SR	NW	CF	5X	14	1610		1734	1.8	7.5	894.7		963.5	54	0.1	16.6		17.8
101	SR	NW	CF	CX	127	46		127	1.8	70.5	25.4		70.5	54	1.3	0.5		1.3
102	SR	NW	CF	3X	81	49		81	1.8	45.1	27.5		45.1	54	0.8	0.5		0.8
103	SR	SE	CR	3X	55	145		154	1.8	30.4	80.5		85.6	54	0.6	1.5		1.6
104	SR	SE	CR	5X	151			140	1.8	83.8	0.0		78.0	54	1.6	0.0		1.4
105	SR	SE	CR	CX	46	1570		1579	1.8	25.7	872.5		877.5	54	0.5	16.2		16.2
106	SR	SE	CF	CX	46	1543		1579	1.8	25.7	857.1		877.5	54	0.5	15.9		16.2
107	SR	SE	CF	5X	59	56		140	1.8	32.6	31.0		78.0	54	0.6	0.6		1.4
108	SR	SE	CF	3X	55	110		154	1.8	30.4	61.1		85.6	54	0.6	1.1		1.6
Mean					**93,4**	**676,9**		**791,4**		**51,9**	**313,5**		**439,8**		**1,0**	**5,8**		**8,1**
SD					**69,4**	**745,2**		**815,4**		**38,6**	**402,5**		**453,0**		**0,7**	**7,5**		**8,4**

Note: Underlined data may be an overestimate.

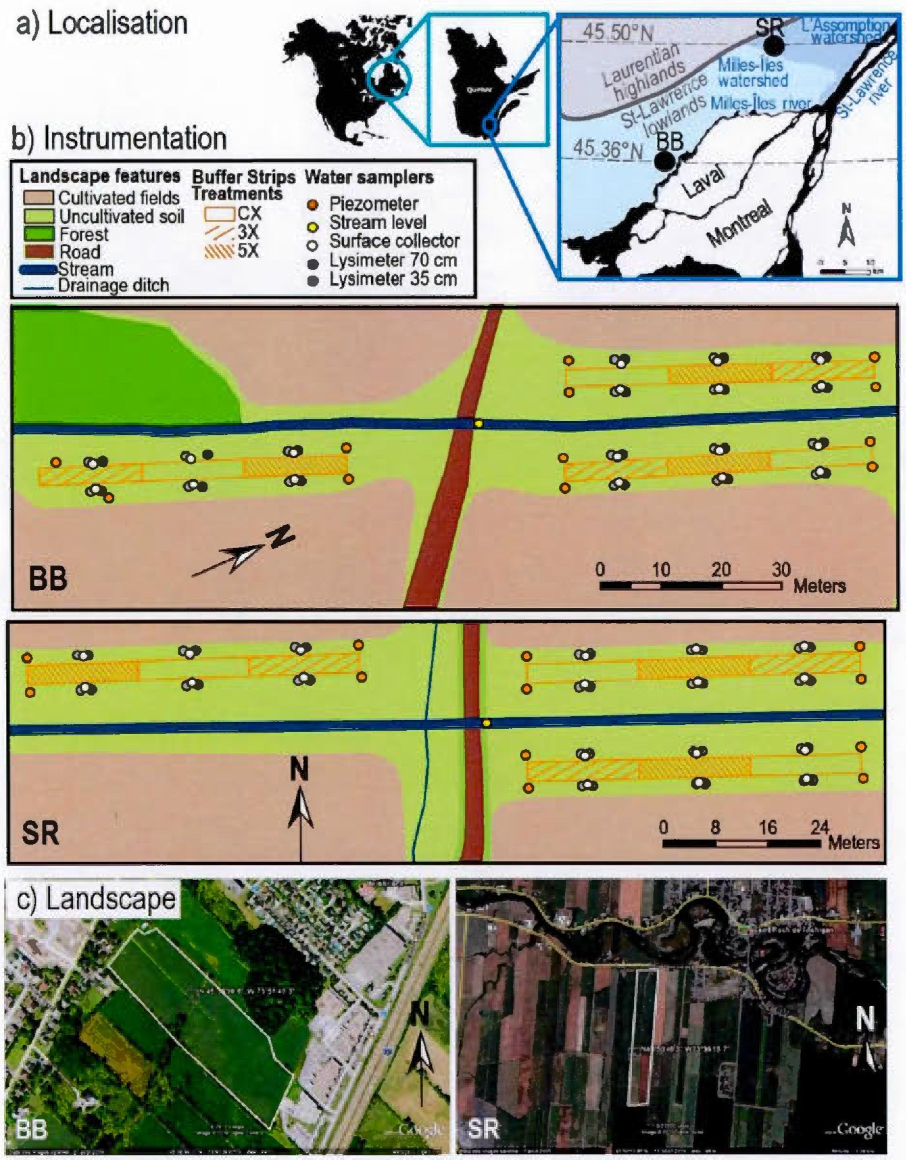

Figure A4- 1: (a) Location maps, (b) water sampling equipment of the Boisbriand (BB; left) and Saint-Roch-de-l'Achigan (SR; right) sites in Quebec, Canada. (c) Satellite images showing the landscape. The stream flows south-west in BB and east in SR.

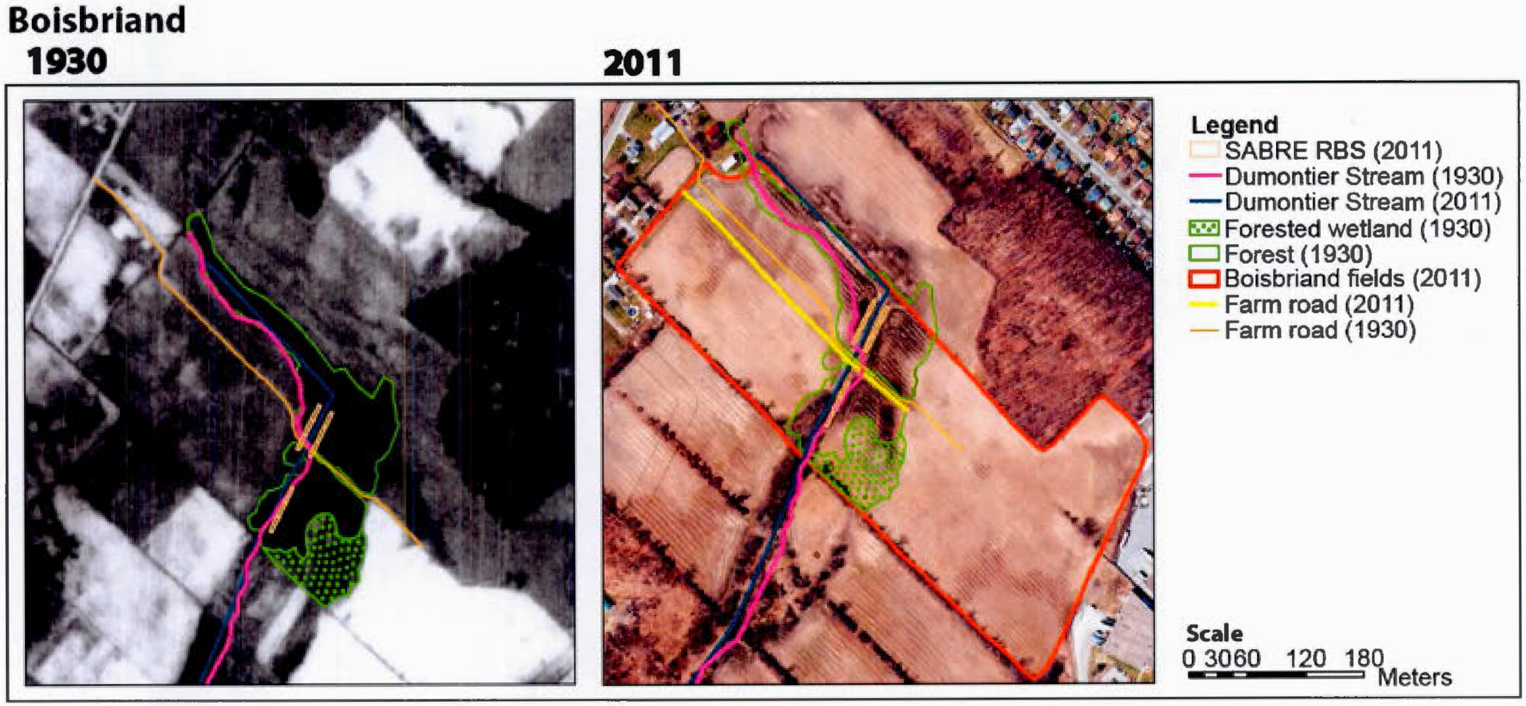

Figure A4- 2: Important landmarks in Boisbriand during the experimental time versus their historical positioning in 1930.

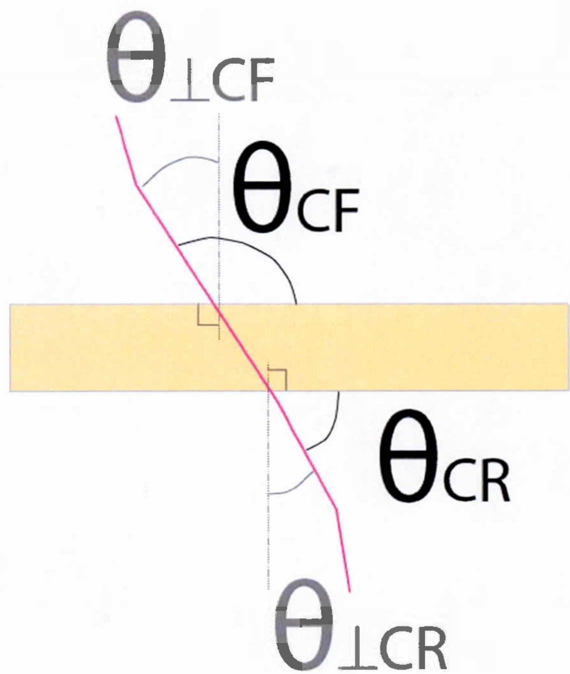

Figure A4- 3: Methodology for calculating the incidence angle of incoming runoff on the edge of field (CF) and the edge of stream (CR).

Mean angle relative to buffer strip for all drainage lines is marked as θ_{side}. Mean deviation from perpendicular for local drainage lines ($\theta\perp_{side}$).The CF or CR RBS edges were used to calculate the angle of incidence (θ) or the angle relative to a perpendicular flow line crossing the buffer strip $\theta_{\perp}$

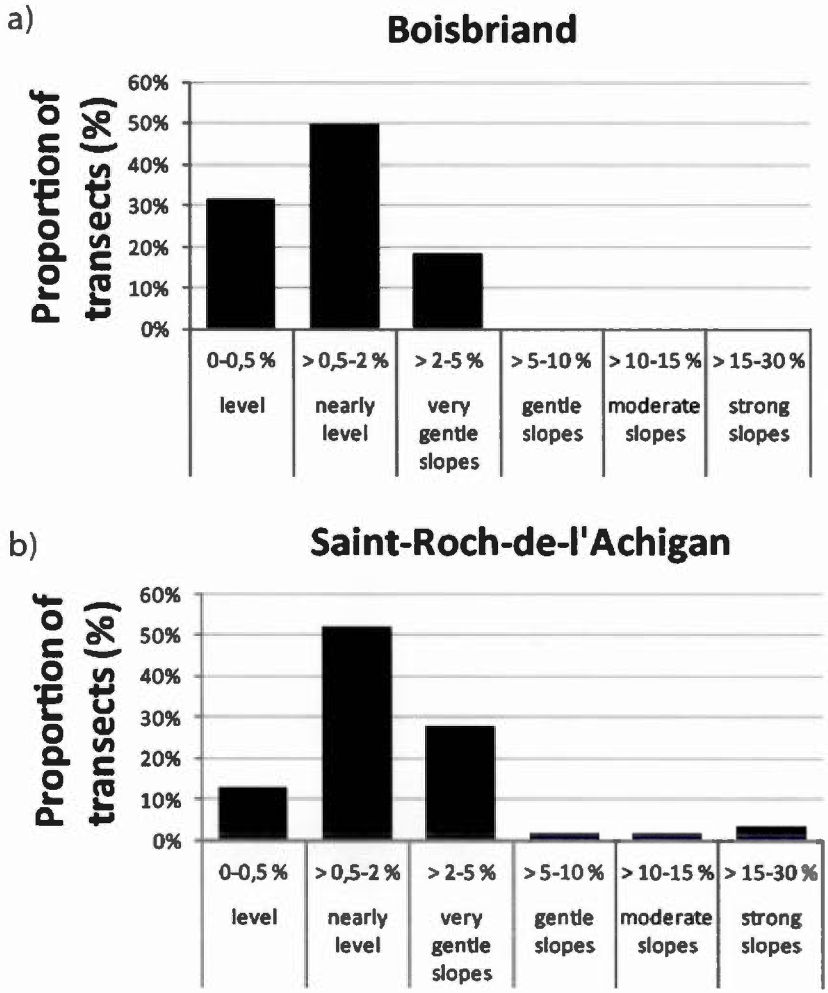

Figure A4- 4: Distribution of slopes for (a) Boisbriand and (b) Saint-Roch-de-l'Achigan.

For each site, 54 transects (4 m) were measured for slope both across the buffer strip and just before the buffer strip. Slopes greater than 5% at Saint-Roch-de-l'Achigan suggest preferential runoff flow paths (scale 1:1000).

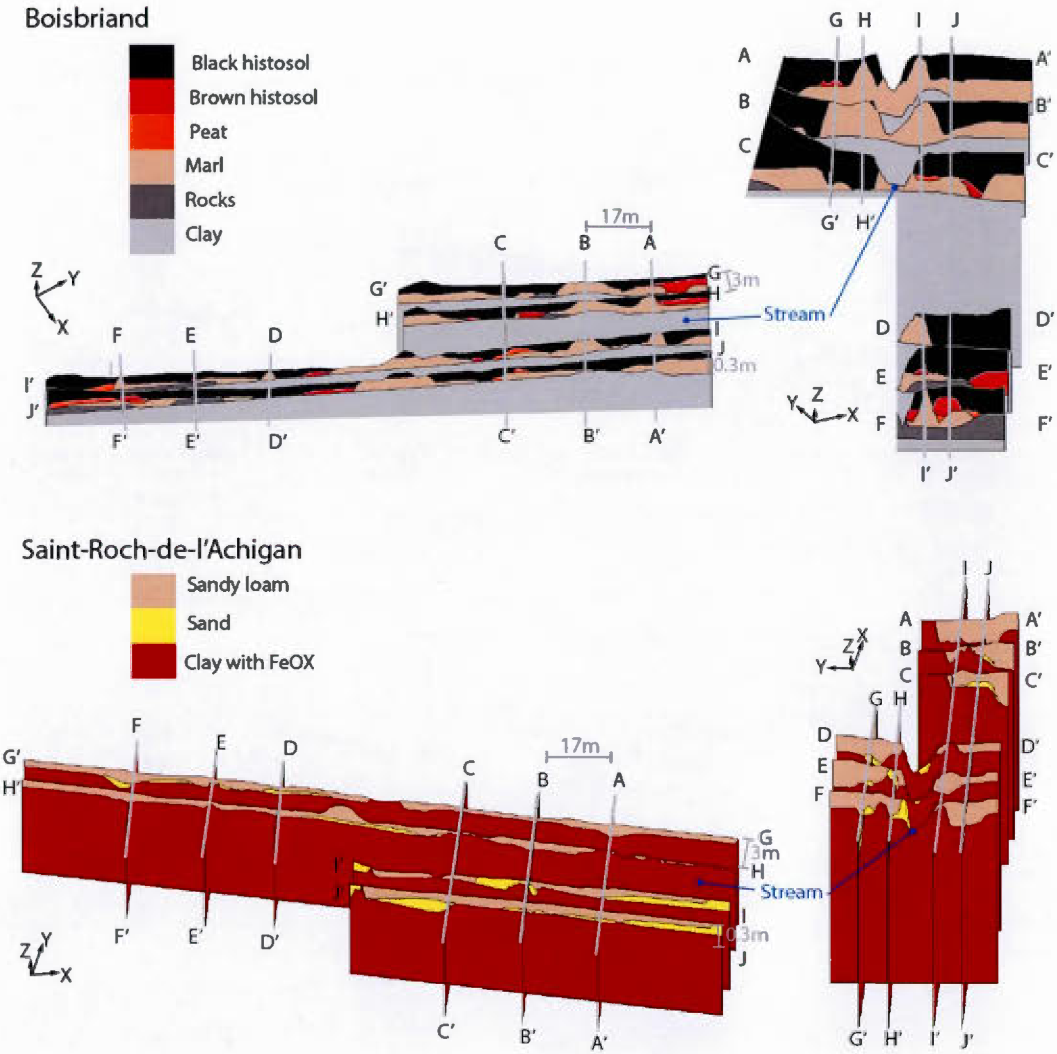

Figure A4- 5: Three dimensional stratigraphic models of Boisbriand (top) and Saint-Roch-de-l'Achigan (bottom).

Transects G-G' and J-J' are located on the edge-of-field; transects H-H' and I-I' are located on the edge-of-stream, and both sets of transects are separated by 3m. Transects A-A', B-B' and C-C'; as well as D-D', E-E' and F-F' are separated by 17m and are located at mid-point of each riparian buffer treatment parcels. The 0, 35 and 70cm water sampling equipment is situated near the intersection of perpendicular transects. Note that depth (Z axis) is magnified by a factor of 10X to facilitate discernment of stratigraphic layers.

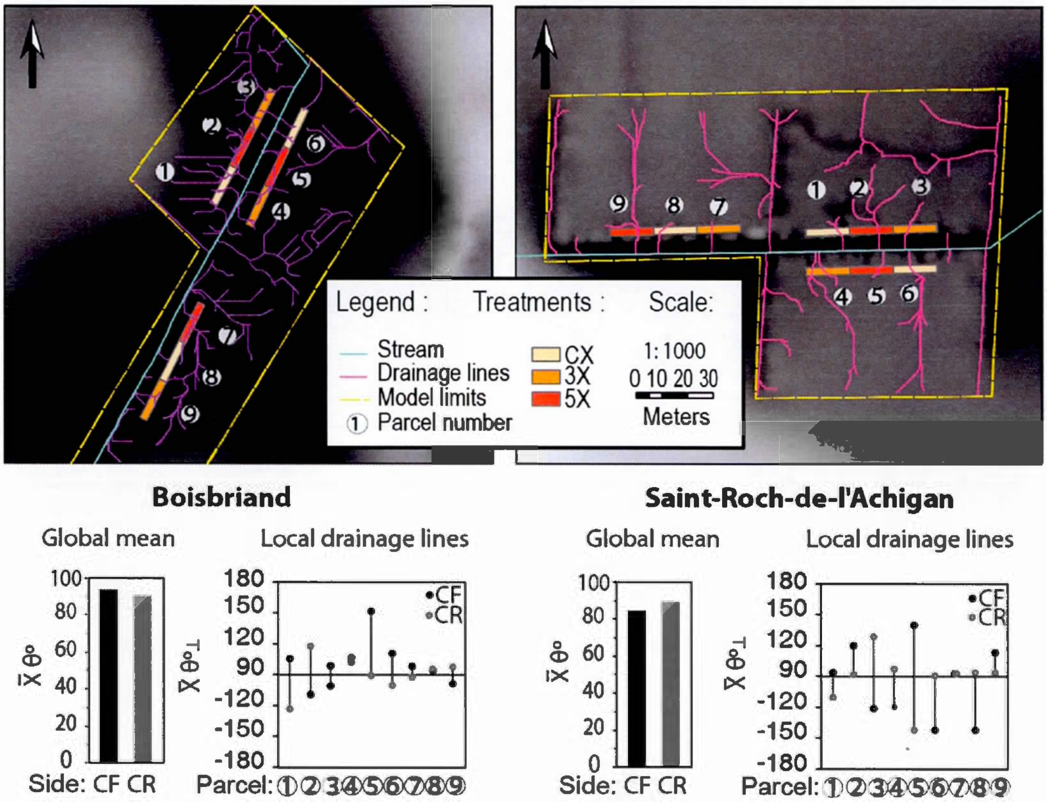

Figure A4- 6: Surface runoff incidence angle in Boisbriand and Saint-Roch-de-l'Achigan based on the proximal field scale drainage lines.

No significant difference in runoff incidence angle for side (CF vs CR), treatment (CX, 3X, 5X), geographic quadrant (NE, SE, SW) or site (BB, SR) for mean θ but θ⊥ varies for each parcel. There was no significant difference (ANOVA) between side and treatment on the overall runoff incidence angle (θ) (histograms), and though there is local variability in each parcel relative to the perpendicular transects across the buffer strip ($\theta_{\perp}$)(needle diagram), there was no significant difference which could be linked with side or treatment (testing for ranks on paired data). This means that globally, the incoming runoff crosses the buffer strip in a perpendicular fashion (~ 90°), but on a local scale incoming (CF) and exiting (CR) preferential surface flows may enter and exit the buffer test parcels at variable angles.

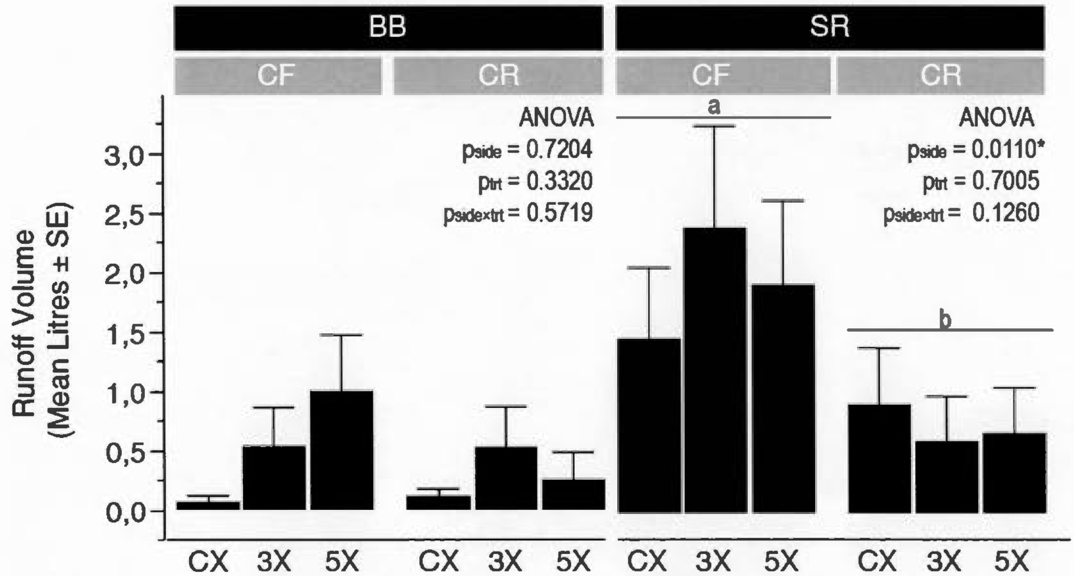

Figure A4- 7: Average runoff volume collected in 2011 on two sites (BB vs SR), two sides (CF vs CR) and three treatments (CX, 3X, 5X).

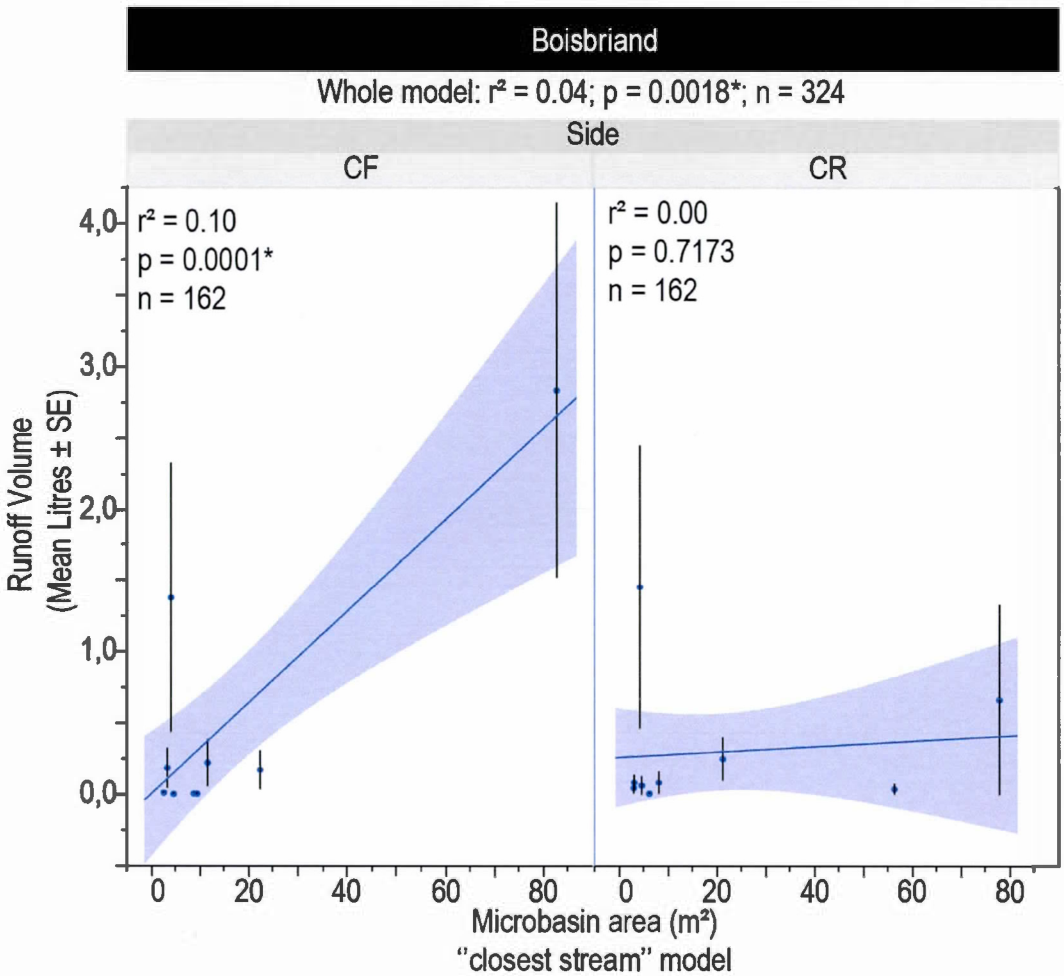

Figure A4- 8: Average runoff collected in 2011 in Boisbriand, before (CF) or after (CR) the buffer strip, in relation to the size of the source microbasin area calculated from the "closest stream" model.

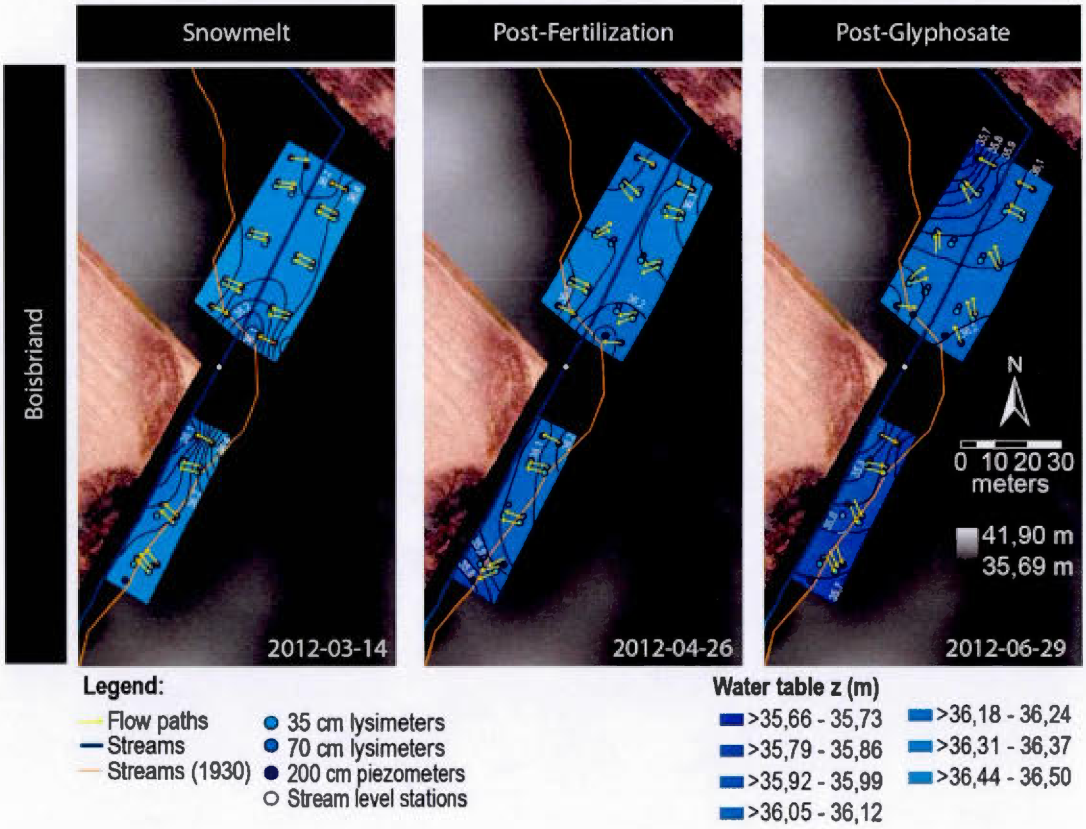

Figure A4- 9: (a) Boisbriand: Water table altitude (blue scale) during characteristic agricultural sampling periods within the contextual field surface elevation (black and white scale).

The water table is highest at snowmelt on both sites, and lowest post-glyphosate. Amplitude of the phreatic water table vertical movement is approximately 85 cm in BB and 75cm in SR from the spring to summer. In BB, spring water table flows towards the stream, and resurgence zones were observed east of the stream water level station. In dryer months, there is a reversal of groundwater flows and the stream appears to feed the phreatic water table with water flowing towards the north for the eastern parcels and flowing towards the east in the south-western parcels. In these moments, water seems to deviate from the current stream position, perhaps under the geological influence of the stream bed prior to linearization (1930). In SR, the groundwater appears disconnected from the stream, and no flow reversal occur in the dryer months. Furthermore, note that the ground appears totally saturated with water in the spring and no flow direction could be discerned in half of the stations based on water table altitude isobars (water assumed to flow perpendicularly to them).

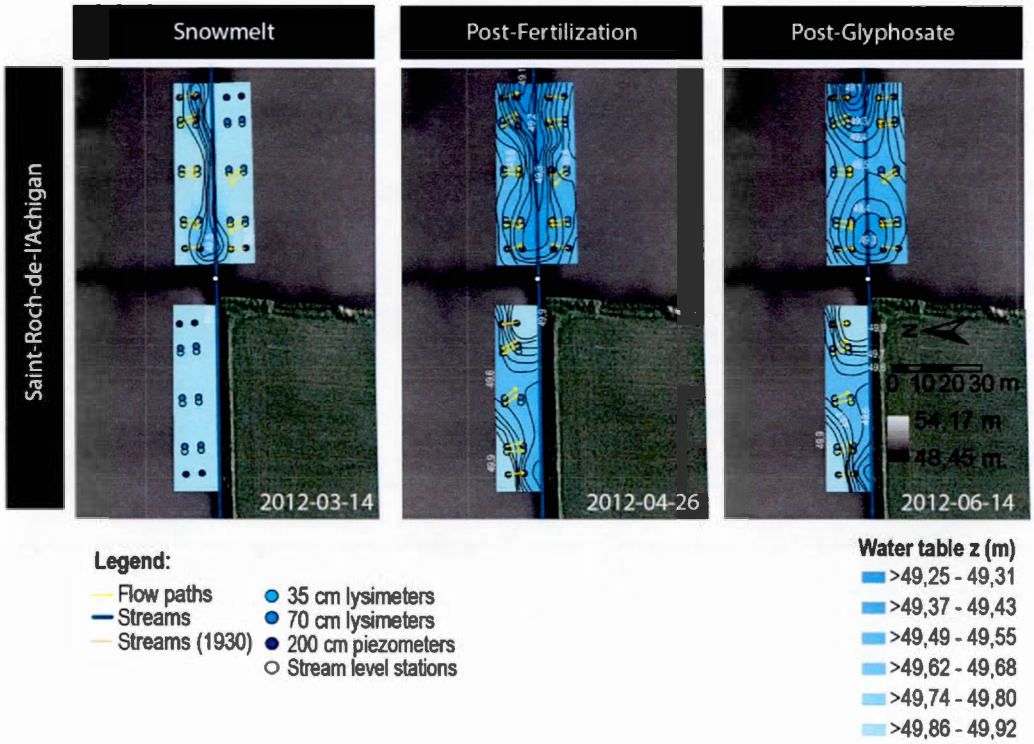

Figure A4- 9: (b) Saint-Roch-de-l'Achigan: Water table altitude (blue scale) during characteristic agricultural sampling periods within the contextual field surface elevation (black and white scale).

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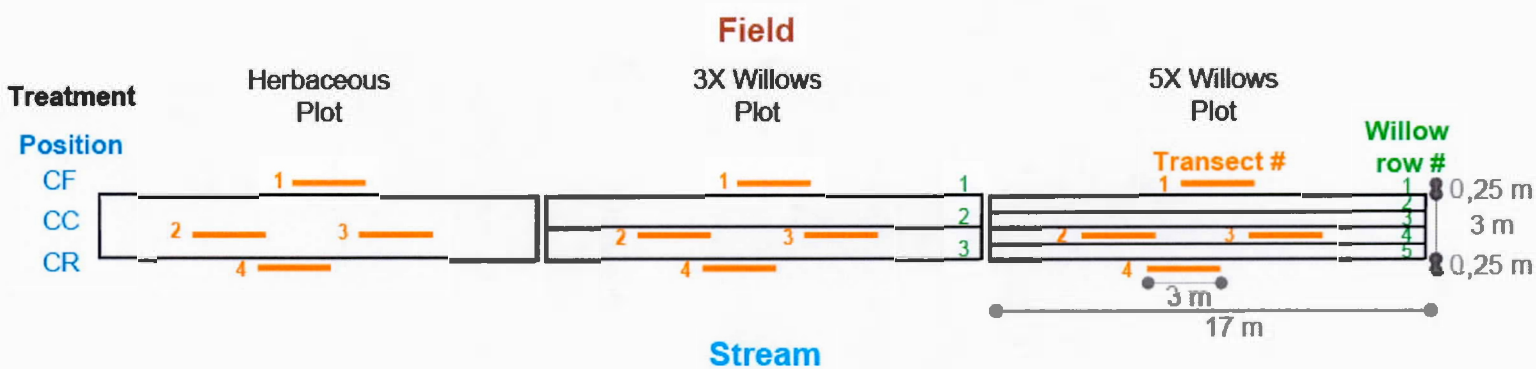
ANNEXE 5:

POSITIONING OF THE 3-M-LINES FOR THE LINE-INTERCEPT METHOD (ORANGE) IN RELATION TO THE FIELD AND STREAM, AS WELL AS WILLOW ROW NUMBER IN THE 33 333 (3X) AND 55 556 (5X) STEM/HA TREATMENTS.

The herbaceous plot and willow understory vegetation were sampled prior to annual mowing, on the 5th and 6th of August 2013, a time of the year corresponding to maximal deployment of biomass (Kelly et al. 2007). Vegetation sampling was conducted 5 years after RBS establishment, while the willow shoots were 3 years of age. **Biomass:** Samples were taken only in the center of the buffer strip, at random locations. Three round subplot of 65 cm diameter were delimited along the center of each planted or unplanted parcels (CC axis). Vascular plants were harvested by clipping aerial parts down to the ground level. Samples were weighed wet in the field, then all 18 samples were dried and re-weighed. **Diversity**: Four lines were sampled in each parcel according to the *line intercept* method. Three meters long lines were stretched between two pickets in along the longitudinal axis of the RBS: close to the field (CF), in the middle of the buffer width (CC) or close to the stream (CR) at fixed places relative to the willow rows. However, transects location on the longitudinal axis was random, avoiding edges of the parcel and locations disturbed by water or soil sampling. Both external transects (CF and CR) are 25 cm away from the proximal willow rows. In the herbaceous plots, lines were aligned with those of the planted plots. On each line, the herbaceous and shrubby vegetation intersecting the string was characterized. The relative importance of each plant species was recorded by **cover classes**: 1 = 0-1%, 2 = 1-5%, 3 = 5-10%, 4 = 10-25%, 5 = 25-50%, 6 = 50-75% and 7 = 75-100% and **height classes**: 1 = 0-5 cm, 2 = 5-30 cm, 3 = 30-150 cm (Boivin et al. 2000).



ANNEXE 6

HERBACEOUS STRATA PLANT SPECIES OBSERVED IN VEGETATED BUFFER STRIPS AT BOISBRIAND AND SAINT-ROCH-DE-L'ACHIGAN AND THEIR BIOLOGICAL OR ECOLOGICAL CHARACTERISTICS

Details on families, habit, habitat, status, life cycle, attraction to pollinators, flowering, weed status, propagation strategies and root morphology are given, along with specific references when various sources were consulted. Notes and references are explained at the end of the continued table. All specimens were identified to the species level, except for a few immature, senescent or damaged specimens generically referred to the genus level or *Poacea* sp. (Brouillet 2010+). A total of 78 species were identified in the two sites, with 48 species observed in BB and 47 in SR. The majority of the plants growing in the buffer strips were herbaceous (68) and only a few trees (6), shrubs (3) and vines (1) were observed. In Boisbriand, young maples appear to be more favorably recruited in the willow planted buffer strips, and this is consistent with the observations of Lust et al. (2001). The vast majority of the herbaceous strata can be considered as weeds (65 species), but this number drops to 47 species when excluding indigenous plants as per conservationists distinctions. A total of 64 species encountered in humid or damp habitats (Habitat H), and 57 commonly found on river banks and stream sides (Habitat B in Annexe 6) were observed on both sites. The only obligate hydrophyte, *Acer saccharinum*, was observed in BB. Six species of nitrogen fixing plants covered 9% of the surveyed on the least fertile soil RBS (SR). This observation suggests two hypotheses: (1) the poorer soil fertility of Saint-Roch-de-l'Achigan could make nitrogen fixation an important survival characteristic and (2) the importance of these plants may bear consequences in the nitrogen budget of the buffer strips in Saint-Roch-de-l'Achigan.

Annex 6 (Continued)

Site [a]	Family	Species [b]	Habit [c] (1)	Habitat [d]	Wetland status	Riparian buffer [e]	Status [f] (1)	Life cycle [g]	Seed propagation [i]	Vegetative propagation rate[j]	Rhizomes	Stolons	Root morphology [l]	Shade tolerance [J]
BB SR	Sapindaceae	*Acer negundo*	T	H, B, N (8)	FAC (43)	Y (8)	I	P	S slo (8)	slow			F (8)	Tol (8)
BB	Sapindaceae	*Acer rubrum*	T	H, N, T (2, 8)	FACW (44)	Y (6, 37)	N	P	S mod (8)	none			T (13)	Med (8, 37)
BB	Sapindaceae	*Acer saccharinum*	T	H, B, N (3, 8)	OBL (44)	Y (8, 37)	N	P	S slo (8)	none			T (13)	Med (8, 37)
SR	Asteraceae	*Achillea millefolium*	H	H, B, P, F, R, W, G, S, C,(2, 4, 8)	FACU (43)		I	P	S slo (5, 8)	slow	R (9, 4, 8)	S (4)	F (8, 13)	Med (8)
BB SR	Poaceae	*Elymus repens*	H	H, F, G (2, 42)	FACU (43)		I	P	S mod (8)	rapid	R (2, 9, 8)		F (13)	Int (8)
BB	Poaceae	*Agrostis stolonifera*	H	H, B, P, F, R, G (2, 3, 4, 42)	FACW (44)		I	P	S slo (8, 42)	rapid	R (42)	S (4, 8, 42)	F (13)	Int (8)
BB	Amaranthaceae	*Amaranthus powellii*	H	H,F,R,G, (39, 40, 42)	UPL* (40, 41)		I	A	S rap (5, 16, 38)	none			T (5, 42)	Int (16)
BB SR	Asteraceae	*Ambrosia artemisiifolia*	H	H, B, F,R,G (2, 3, 39)	FACU (3, 43)		I	A	S rap (5, 34)	none			T* (13)	Med (14)
BB	Ranunculaceae	*Anemone canadensis*	H	H, B, R, W (2, 3, 42)	FACW (3, 43)	Y (37)	N	P	S slo (14, 35)	rap	R (14, 42)		F (13)	Med (14, 37)
BB	Asteraceae	*Arctium minus*	H	H, B, P, R, G (2, 3, 39, 42)	FACU (3, 43)		I	B	S rap (5, 16)	none			T (5, 9, 13)	Med (17)
BB SR	Asteraceae	*Artemisia vulgaris*	H	H, B, F, R, G, (3, 38, 39, 42)	UPL (3, 43)		I	P	S rap (33, 38)	slow	R (2, 9)		T (13)	Int (14)
BB SR	Apocynaceae	*Asclepias syriaca*	H	H, B, P, F, R, W, G, S, C,(2, 3, 8)	UPL (3, 43)		N	P	S rap (5, 8)	rap	R (9, 8)		T (13)	Int (14)
BB	Brassicaceae	*Brassica rapa*	H	F, R, G, (3, 38, 39)	UPL (3, 43)		I	A	S mod (8)	none			T (13, 42)	Med (8)
SR	Brassicaceae	*Brassica nigra*	H	B, P, F, R, G, S,	UPL (3)		I	A	S rap (36)	none			T (13, 42)	Int (14)
SR	Poaceae	*Bromus inermis*	H	H, P, F, R, W, G, S, C, (2, 3, 4, 8, 39)	UPL (3, 43)	Y (8)	I	P	S slo (8)	rap	R (8, 9)		F (13)	Int (8)
BB	Caryophyllaceae	*Cerastium fontanum*	H	H, B, P, F, R, W, G, S (2, 3, 42)	FACU (3, 43)		I	B	S (5)	yes			T* (13, 42)	Int (14)
BB	Amaranthaceae	*Chenopodium album*	H	H, B, F, R, G (2, 3, 39)	FACU (3, 43)		I	A	S rap (5, 16)	none			T* (5)	Int (16)
BB	Amaranthaceae	*Oxybasis glauca*	H	H, B, P, F, R, G, S, C (2, 3, 38)	FACW (3)		N	A	S (4, 38)	none			T* (13, 38)	Med (14)
SR	Asteraceae	*Cichorium intybus*	H	H, B, P, F, R, G (2, 3, 39)	FACU (3, 43)		I	P	S rap (5, 8)	none			T (2, 5, 42)	Int (8)
SR	Asteraceae	*Cirsium arvense*	H	H, B, P, F, R, G (2, 3, 39)	FACU (3, 43)		I	P	S mod (5, 16, 33)	rap	R (2, 16)		T (9, 30)	Int (16)
BB	Asteraceae	*Cirsium vulgare*	H	H, B, P, F, R, W, G (2, 3, 38, 39)	FACU (3, 43)		I	B	S rap (5, 16, 39)	none			T (9, 14)	Int (16)
BB	Convolvulaceae	*Calystegia sepium subsp. americana*	H	H, B, F (2, 3, 42)	FAC (3, 43)		N	P	S rap (32)	rap	R (9, 42)		F (13)	Int (14)
BB	Cornaceae	*Cornus alternifolia*	A	H, B (3)	FACU (3, 43)	Y (6)	N	P	S slo (8)	none			T (13)	Tol (8)
SR	Poaceae	*Dactylis glomerata*	H	H, B, P, F, R, W, G, C, (2, 3, 4, 8)	FACU (3, 43)		I	P	S slo (8)	none	R (4)		F (2, 42)	Tol (8)
SR	Equisetaceae	*Equisetum arvense*	H	H, B, R, W, G, C (2, 3, 39)	FAC (3, 43)		N	P	S slo (8)	mod	R (2, 8, 9)		F (13)	Med (8)
BB	Asteraceae	*Erigeron canadensis*	H	H, B, P, F, R, G (2, 3, 39)	FACU (3, 43)		I	A	S rap (5, 18, 38)	none			T* (5, 18)	Med (19)
SR	Asteraceae	*Erigeron philadelphicus*	H	H, B, P, F, R, W, (2, 3, 8, 39)	FACW (44)		N	P	S slo (8, 38)	mod		S (8, 38)	F* (13)	Med (8)
SR	Asteraceae	*Erigeron sp.*	H				N	A/B/P	ND				F (13)	ND
BB	Brassicaceae	*Erysimum cheiranthoides*	H	H, B, P, F, R, G, S, C (3, 38)	FACU (3, 43)		I	A	S slo (38, 39)	none			T (13, 42)	Int (19)
BB	Polygonaceae	*Fallopia convolvulus*	H	H, B, F (3, 38)	FACU (3, 43)		I	A	S (5, 38)	none			F (5)	Int (41)

Annex 6 (Continued)

Site [a]	Family	Species [b]	Habit [c] (1)	Habitat [d]	Wetland status	Riparian buffer [e]	Status [f] (1)	Life cycle [g]	Seed propagation [i]	Vegetative progapation rate[i]	Rhizomes	Stolons	Root morphology [l]	Shade tolerance [j]
SR	Poaceae	*Festuca rubra*	H	H, B, S, (2, 3, 8)	FACU (3, 43)	Y (8)	N	P	S slo (8)	rap	R (2, 4, 8, 9)	R (4, 42)	F (14)	Tol (8)
BB SR	Rosaceae	*Fragaria virginiana*	H	H, B, P, F (2, 3)	FACU (3, 43)		N	P	S slo (15)	rap	R (8)	S (2, 8)	F (13, 42)	Med (19)
BB	Lamiaceae	*Galeopsis Tetrahit*	H	H, B, P, W, G (2, 3, 39, 42)	FACU (3, 43)		I	A	S mod (16)	None			T/F (13, 16, 42)	Med (19)
SR	Asteraceae	*Galinsoga parviflora*	H	H, G, (3, 42)	UPL (3, 43)		I	A	S rap (20)	mod			F (13, 42)	Int (19, 20)
BB	Apiaceae	*Heracleum maximum*	H	H, B, R, (2, 3)	FACW (3, 43)		N	B/P	S rap (8)	mod			T (13)	Int (8)
BB	Balsaminaceae	*Impatiens capensis*	H	H, B, (2, 3)	FACW (44)		N	A	S slo (5, 8)	none			F* (13)	Tol (2, 8)
BB SR	Asteraceae	*Lactuca serriola*	H	H, B, F, R, G (2, 3, 39)	FACU (3, 43)		I	A	S rap (5, 21, 38)	none			T (13)	Int (21)
SR	Poaceae	*Lolium sp.*	H	F (2)			I	A/P	ND				F (13)	ND
SR	Fabaceae	*Medicago lupulina*	H	H, B, P, F, R, G (2, 3, 39)	FACU (3, 43)		I	A	S rap (5, 8, 22)	mod			F (13)	Int (8, 22)
SR	Fabaceae	*Melilotus albus*	H	H, B, R, G, S, C, N, T (2, 3, 8)	FACU (3)		I	A	S rap (8)	none			T (8, 42)	Int (8)
BB SR	Poaceae	*Muhlenbergia frondosa*	H	H, B (2, 3)	FACW (3, 43)		N	P	S slo (8)	mod	R (2, 8, 9)		F (13)	Tol (8)
BB	Oxalidaceae	*Oxalis stricta*	H	H, B, F, G, (2, 3)	FACU (3, 43)		I	P	S mod (5)	rap	R (9, 23)	S (2)	F (13)	Int (19)
BB	Poaceae	*Panicum dichotomiflorum subsp. dichotomiflorum*	H	H, B, F, G, (3, 42)	FACW (44)		I	A	S mod (5, 8)	none			F (5, 42)	Int (8)
SR	Poaceae	*Panicum sp.*	H				N	A	S ND (5, 8)	ND			F (13)	ND
BB	Vitaceae	*Parthenocissus quinquefolia*	V	H, B, F, W, S, C, (2, 3, 8)	FACU (3, 43)	Y (6, 37)	I	P	S mod (8)	rap	R (13)	S (8)	F (13)	Med (2,8)
BB SR	Apiaceae	*Pastinaca sativa*	H	H, B, F, R, G, (2, 39, 42)	FACU (45)		I	B	S rap (5, 24)	none			T (2, 24, 42)	Med (24)
BB	Polygonaceae	*Persicaria maculosa*	H	H, B, F, R, G, (2, 3, 42)	FACW (44)		I	A	S rap (5)	none			T (5, 13, 42)	Med (19)
BB	Poaceae	*Phalaris arundinacea*	H	H, B, R, N, T (2, 8, 42)	FACW (44)		N	P	S slo (8)	rap	R (8, 9)		F (13)	Int (8)
BB SR	Poaceae	*Phleum pratense*	H	H, B, F, R, C, T (2, 3, 8)	FACU (3, 43)	Y (8)	I	P	S slo (8)	none			F (8, 42)	Med (8)
BB SR	Plantaginaceae	*Plantago major*	H	H, B, F, R, G, (2, 3, 39)	FACU (3, 43)		I	P	s mod (5)	none	R (2)		F (5, 42)	Med (8)
BB SR	Poacea	*Poacea sp.*	H				N	ND	S ND (8)	ND			F (13)	Tol (46)
BB	Polygonaceae	*Persicaria pensylvanica*	H	H, B (2, 3)	FACW (44)		N	A	S rap (8)	none			T (13)	Int (8)
SR	Salicaceae	*Populus deltoides*	T	H, B, S, T (2, 3, 8)	FACW (44)	Y (37)	N	P	S rap (8)	slow			F* (13)	Int (8, 37)
SR	Salicaceae	*Populus sp*	T				N	P	S ND (8)	ND			F* (13)	ND
SR	Salicaceae	*Populus tremuloides*	T	H, B, N, T, (2, 3, 8, 42)	FACU (3, 43)	Y (8, 37)	N	P	S mod (8)	mod			F* (13)	Int (8, 37)
BB SR	Ranunculaceae	*Ranunculus repens*	H	H, B, P, G, S (2, 3, 39)	FAC (3, 43)		I	P	s rap (5, 8)	slow	R (8)	S (9)	F (13, 42)	Tol (8)
SR	Anacardiaceae	*Toxicodendron radicans*	H	H, B, R, (2, 3, 42)	FAC (3, 43)		N	P	s rap (5, 25, 39)	slow	R (5, 39)	S (9)	F (13)	Tol (25)
BB	brassicaceae	*Rorippa palustris*	H	H, B (2)	FACW (44)		N	B					T (13, 42)	Med (19)
SR	Rosaceae	*Rubus occidentalis*	A	F, S (2)	UPL* (2)	Y (37)	N	P	S slo (8)	rap			F (13)	Med (6, 8, 37)
BB	Asteraceae	*Senecio vulgaris*	H	H, B, F, R, G (2, 3, 42)	FACU (3, 43)		I	A	s rap (5, 8)	none			T* (13, 42)	Med (6, 8)

Annex 6 (Continued)

Site [a]	Family	Species [b]	Habit [c] (1)	Habitat [d]	Wetland status	Riparian buffer [e]	Status [f] (1)	Life cycle [g]	Seed propagation [i]	Vegetative propagation rate [j]	Rhizomes	Stolons	Root morphology [l]	Shade tolerance [J]
BB	Poaceae	*Setaria pumila*	H	H, B, F, G (2, 3)	FAC (3, 43)		I	A	s mod (5, 39)	none			F (5, 39, 42)	Int (8)
BB	Caryophyllaceae	*Silene latifolia*	H	F, R, G (42)	UPL* (41)		I	B/P	s rap (5, 26)	mod			T (13)	Int (26)
BB SR	Asteraceae	*Solidago canadensis*	H	H, B, P, F,R, W, G, S, T (2, 3, 8)	FACU (3, 43)	Y (37)	N	P	S slo (8)	rap	R (8)		F (13, 42)	Int (8, 27, 37)
BB SR	Asteraceae	*Euthamia graminifolia*	H	H, B, P, F, R, W, S, R, T (2, 3, 8)	FAC (3, 43)		N	P	S slo (8)	rap	R (8, 9)		F (13)	Med (8)
BB SR	Asteraceae	*Sonchus arvensis*	H	H, B, F, R, G (2, 3)	FACU (3, 43)		I	P	s rap (5, 28)	rap	R (2, 28)		T (9, 14, 28)	Med (19, 28)
SR	Caryophyllaceae	*Stellaria graminea*	H	H, B, F, (40)	UPL (3, 43)		I	P			R (2)		F (13)	Med (41)
BB SR	Asteraceae	*Taraxacum officinale*	H	H, B, P, F, R, G (2, 3)	FACU (3, 43)		I	P	s rap (5, 8, 33)	none			T (5, 2, 9)	Med (8)
SR	Asteraceae	*Tragopogon pratensis*	H	P, F, R, G, (2, 38, 42)	FACU (41, 45)		I	B/P	s rap (5, 29, 39)	none			T (5, 2, 9)	Int (29)
SR	Fabaceae	*Trifolium campestre*	H	H, F, G, S (2, 40, 42)	FACU (41, 45)		I	A/B					T* (13, 42)	Int
SR	Fabaceae	*Trifolium hybridum*	H	H, B, G, S, C, T (2, 3, 8)	FACU (3, 43)		I	P	S slo (8)	none			T (8, 42)	Int (8)
SR	Fabaceae	*Trifolium pratense*	H	H, B, P, G, S (2, 3, 8)	FACU (3, 43)		I	P	S slo (8)	none			T (8, 42)	Int (8)
SR	Fabaceae	*Trifolium repens*	H	H, B, G, S, T (2, 3, 8)	FACU (3, 43)		I	P	S slo (8)	mod		S (2, 8)	F (13, 39)	Int (8)
SR	Fabaceae	*Trifolium sp*	H				I							Int
BB	Asteraceae	*Tussilago farfara*	H	H, B, G, (2, 3, 42)	FACU (3, 43)		I	P	s rap (5, 33)	mod	R (2, 9)		T (5, 13)	Int (41)
SR	Scrophulariaceae	*Verbascum thapsus*	H	P, R, G, S, (2, 3, 39)	UPL (3, 43)		I	B	s rap (5, 31)	none			T (5, 42)	Int (41)
SR	Fabaceae	*Vicia cracca*	H	H, P, F, R, W, G (2, 39)	FAC (41, 45)		I	P	S slo (8)	mod	R (8, 9)		F (13)	Med (8)

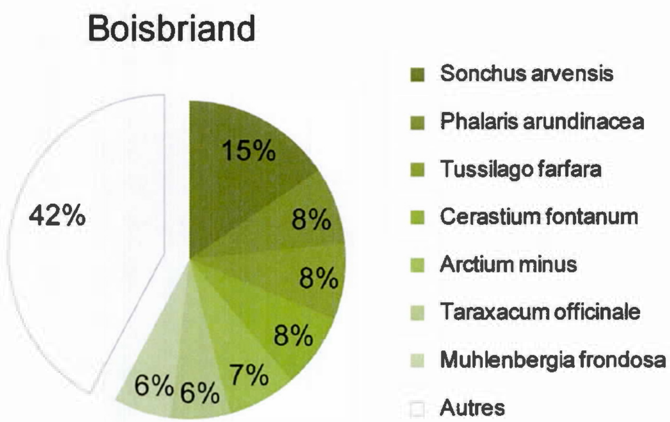
Annex 6 (Continued)

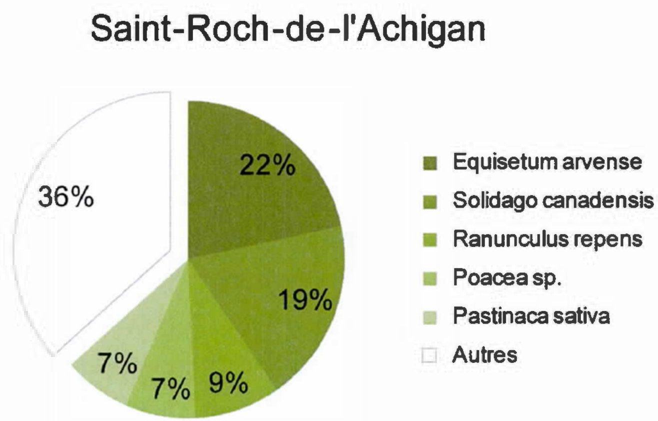
Notes : (a) Study sites: BB: Boisbriand; SR: Saint-Roch-de-l'Achigan; (b) VASCAN accepted nomenclature; (c) T: Tree, A: Arbustive, H: Herbaceous, V: Vine; (d) Wetland indicator status references described in the text (an asterisk denoted the author's best judgement for unclassified species); H: humid or damp habitat; R: river bank or stream side; P: Pasture or grassland; F: Field; R:Roadside and ditches ; W: Woodland forest or forest edges ; G: Waste ground or disturbed area ; S: dry, sandy or well-drained soils; C: Clayey soil; O: Partially or totally shaded areas ; T: species used for technical characteristics in rehabilitation, reclamation, erosion control or phytoremediation; (e) Y: Yes, species recommended in the litterature for use in riparian buffers; (f) N: Indigenous; I: Introduced; (g) P: perennial, A: annual, B: Biannual; (h) S: propagates by seeds; (i) Seed propagation: Slo: Slow, Mod: Moderate, and Rap: Rapid; and (i) T: Tap, F: Fibrous based on the following definitions: tap roots — large, thick, vertically penetrating root including radical of certain seedlings — and fibrous roots — branched with no dominant root (Caradus 1977), and based on a literature review (see references in table) complemented with similar soil, texture and habitat observations of preserved specimens (Herbier Marie-Victorin, Centre de biodiversité de l'Université de Montréal, Canada); (j) Tol: Tolerant, Mid: Intermediate Tolerant, Int: Intolerant

References: (1) (Brouillet 2010+); (2) (Marie-Victorin and Rouleau 1964); (3) USDA 2014 (Wetland indicator status); (4) (Flora of North America Editorial Committee 1993+); (5) (Agriculture and Agri-Food Canada 2014); (6) (Moisan-De Serres et al. 2014); (7) Payette 2011; (8) USDA 2014 (Plant Guide); (9) (Bouchard et al. 1998); (10) USDA 2014 (Noxious weed composite US); (11) USDA 2014 (US States); (12) USDA 2014 (Weed North-eastern States); (13) Specimens from Herbier Marie-Victorin; (14) (Anonymous 2014); (15) Smreciu et al. 2013; (16) (British Columbia Ministry of Agriculture Food and Fisheries 2002); (17) Lee 2013; (18) (Weaver and Downs 2003) Weaver 2001; (19) Plants for a Future; (20) Warwick & Sweet 1983; (21) Weaver & Down 2003; (22) (Turkington and Cavers 1979); (23) (Doust et al. 1985); (24) (Cain et al. 2010); (25) (Mulligan and Junkins 1977); (26) (McNeill 1977); (27) (Werner et al. 1980); (28) (Lemna and Messersmith 1990); (29) (Clements et al. 1999); (30) (Nadeau and Born 1989); (31) (Gross and Werner 1978); (32) (Pfirter et al. 1997); (33) (Bostock and Benton 1979); (34) (Friedman and Barrett 2008); (35) (Molano-Flores and Hendrix 1999); (36) (Commonwealth Agricultural Bureaux International (CABI) 2014); (37) (FIHOQ 2008); (38) (OMAFRA (Ontario Ministry of Agriculture Food and Rural Affairs) 2013); (39) (Mulligan et al. unknown); (40) (New England Wild Flower Society 2014); (41) (eFlore 2014); (42) (Klinkenberg 2014); (43) (Lichvar et al. 2014); (44) (Gauthier et al. 2008); (45) (Blaney 2012); (46) (D'Amour 2013).

ANNEXE 7

DOMINANT SPECIES IDENTIFIED (MORE THAN 5% COVERAGE) IN BOISBRIAND AND SAINT-ROCH-DE-L'ACHIGAN ARE MUTUALLY EXCLUSIVE ACROSS BOTH SITES.





ANNEXE 8:

LIST OF ENVIRONMENTAL VARIABLES QUANTIFIED TO INTERPRET AGROCHEMICAL LEACHING FROM FIELDS, RIPARIAN BUFFER STRIP SEQUESTRATION POTENTIAL, *SALIX MIYABEANA* SX64 PRODUCTIVITY, HYDROLOGICAL CYCLING AND ECOLOGICAL NICHE OF HERBACEOUS COMMUNITIES, WATER AND SOIL PHYSICO-CHEMISTRY.

This set of variable is referred to in Chapter 1, 3 and 4. Each individual variable was chosen (rationale) as a proxy of one or few environmental processes. Types of data include numerical and binary. Time and space variability are assigned based on data availability and model assumptions (i.e. Soil physico-chemistry may change with time and vary between sites but was only used to characterize one time-point in one site). Nd: Not-Determined.

Individual Environmental Variables in groups of similar nature	Rationale	Type	Restrictions	Time variability	Inter-sites spatial variability	Intra-site spatial variability
Time						
Sampling periods (Snowmelt, Post-Fertilization, Post-glyphosate)	Target high runoff or chemical concentrations	N		Y	Y	N
Year	Interannual variability	N		Y	N	N
Number of days since last glyphosate application	Proxy for reduction	N		Y	Y	N
Number of days since last spring fertilization (only the first annual fertilization at sowing)	Proxy for reduction	N		Y	Y	N
Number of days since last fertilization (including mid-summer secondary fertilization)	Proxy for reduction	N		Y	Y	N
Cultures						
Field culture (maize or soy; at year X)	Proxy for crop balance (inputs vs needs)	B		Y	N	N
Antecedent culture (maize or soy; year X-1)	Proxy for residual soil nutrient concentrations	B		Y	Y	N
Total dose N (kg/ha; year X)	Proxy for nutrient input	N		Y	Y	N
Total dose P (kg/ha; year X)	Proxy for nutrient input	N		Y	Y	N
Total dose K (kg/ha; year X)	Proxy for nutrient input	N		Y	Y	N
Total dose Mg (kg/ha; year X)	Proxy for nutrient input	N		Y	Y	N
Total dose Ca (kg/ha; year X)	Proxy for nutrient input	N		Y	Y	N
Total dose glyphosate (kg acid equivalents/ha; year X)	Proxy for glyphosate input	N		Y	Y	N
Total dose Potassium salts of glyphosate (kg a.e./ha; year X)	Proxy for glyphosate/nutrient interactions	N		Y	Y	N
Total dose Isopropylamine salts of glyphosate (kg a.e./ha; year X)	Proxy for glyphosate/nutrient interactions	N		Y	Y	N
Chlorimuron (presence/absence)	Proxy for pesticide interaction	B		Y	Y	N
Rimsulfuron + Nicosulfuron (presence/absence)	Proxy for pesticide interaction	B		Y	Y	N
Yield (mt/year)	Proxy for crop balance (inputs vs needs)	N		Y	Y	N

Annexe A- 8 (Continued)

Matrices and Individual Environmental Variables	Rationale	Type	Restrictions	Time variability	Inter-sites spatial variability	Intra-site spatial variability
Vegetation ecological characteristics						
• Salix Final Biomass (2013 Kg dw/ha)	Proxy for nutrient uptake/shade	N		N	Y	Y
• Herbaceous biomass (Kg dw/ha)	Proxy for nutrient uptake	N		N	Y	Y
• Total biomass (Salix + Herbaceous vegetation; Kg dw/ha)	Proxy for nutrient uptake	N		N	Y	Y
• Litter soil cover (g dw/m²)	Proxy for surface rugosity	N		N	Y	Y
• Soil herbaceous cover (% cover)	Proxy for surface rugosity	N		N	Y	Y
• Herbaceous bare soil cover (% cover)	Proxy for erosion potential	N		N	Y	Y
• Life cycle (Annuals; Biennials; Perennials; soil cover %)	Proxy for ecological niche selection factors	N		N	Y	Y
• Weed diversity (Soil cover %; Proportion %; n sp)	Proxy for ecological niche selection factors	N		N	Y	Y
• Exotic weed diversity (n sp)	Proxy for ecological niche selection factors	N		N	Y	Y
• Roots (Fibrous, Tap, ND; Soil cover %; Proportion %; n species)	Proxy for nutrient infiltration potential	N		N	Y	Y
• Hydrophytes (UPL, FAC, FACU, FACW, OBL, ND; Hydrophytes vs non-hydrophytes; Soil cover %; Proportion %; n species)	Proxy for water flow	N		N	Y	Y
• Indigenous herbs (Soil cover %; Proportion; n species)	Proxy for ecological niche selection factors	N		N	Y	Y
• Shade tolerance (Intolerant, Intermediate, Tolerant; Soil cover %; Proportion %; n species)	Proxy for ecological niche selection factors	N		N	Y	Y
• Herbaceous plant height (class median, cm)	Proxy for nutrient uptake	N		N	Y	Y
• Salix (soil cover %)	Proxy for shade	N		N	Y	Y
• Plant diversity (including Salix)	Proxy for ecological niche selection factors	N		N	Y	Y
• Plant diversity attracting pollinators (including or excluding Salix)	Proxy for ecological niche selection factors	N		N	Y	Y
• Shannon diversity H' (including Salix)	Proxy for ecological niche selection factors	N		N	Y	Y
• Simpson 1/D	Proxy for ecological niche selection factors	N		N	Y	Y
• Bioarea (height x % cover)	Proxy for nutrient uptake	N		N	Y	Y

Annexe A- 8 (Continued)

Matrices and Individual Environmental Variables	Rationale	Type	Restrictions	Time variability	Inter-sites spatial variability	Intra-site spatial variability
Salix						
• Salix Mean Diameter	Proxy for nutrient uptake	N		Y	Y	Y
• Salix Mean Height	Proxy for nutrient uptake	N		Y	Y	Y
• Salix Mean number of stems	Proxy for nutrient uptake/rugosity	N		Y	Y	Y
• Salix Biomass (Kg ww/plant)	Proxy for nutrient uptake	N		Y	Y	Y
• Salix Biomass (Kg dw/plant)	Proxy for nutrient uptake	N		Y	Y	Y
Topography						
• Localization (X, Y, Z GPS cordinates)	Proxy for climate/sun	N		N	Y	Y
• Slope (actual or absolute value)	Proxy for erosion/runoff potential	N		N	Y	Y
• Drainage area of sampling equipments (m²) Four models tested "Bassins"; "Nearest stream"; "Affiliated bassins" OR "drainage points"	Proxy for runoff and agrochemicals leaching potential.	N		N	Y	Y
• Stratigraphy (Qualitative or quantitative)	Proxy for infiltration/hydraulic conductivity	B		N	Y	Y
• Stratigraphic disturbances (linearization of stream)	Proxy for anomaly	B		N	Y	Y
Hydrogeology						
• Phreatic table depth (connectivity or no conectivity model; distance in m)	Proxy for dilution/underground transport/denitrification potential	N		Y	Y	Y
• Water sampling equipment submerged (qualitative or quantitative in m)	Proxy for dilution/underground transport/denitrification potential	N-B		Y	Y	Y
• Water table head height difference from fields to stream (quantitative in m or qualitative)	Proxy for water flow/residence time	N-B		Y	Y	Y
Climate						
• Value since (1) sampling initiation; (2) since sowing & fertilization; (3) since last fertilization (including 2nd annual); (4) since glyphosate application						
○ Σ Precipitations (mm)	Proxy for leaching and dilution	N		Y	Y	N
○ Σ Degrés-jours (proxy de la croissance végétale; °C·d)	Proxy for vegetation growth	N		Y	Y	N
○ Mean Temperature (daily min, max, average; °C)	Proxy for evapotranspiration/vegetation growth	N		Y	Y	N
○ Mean Ambiant humidity (%)	Proxy for evapotranspiration/vegetation growth	N		Y	Y	N

Annexe A- 8 (Continued)

Matrices and Individual Environmental Variables	Rationale	Type	Restrictions	Time variability	Inter-sites spatial variability	Intra-site spatial variability
Water physico-chemistry						
• Runoff volume (L)	Proxy for agrochemical leaching/dilution	N		Y	Y	Y
• Glyphosate (µg/l)	Proxy for agrochemical loading	N		Y	Y	Y
• AMPA (µg/l)	Proxy for agrochemical loading	N		Y	Y	Y
• pH	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• Salinity (µS)	Proxy for agrochemical loading	N		Y	Y	Y
• Electrical conductivity (µS)	Proxy for agrochemical loading	N		Y	Y	Y
• Temperature (°C)	Proxy for biotic activity	N		Y	Y	Y
• Turbidity (PPT)	Proxy for erosion	N		Y	Y	Y
• O_2 (%)	Proxy for biotic activity/denitrification potential	N	1	N	nd	Y
• PO_4^{3-} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• P_{tot} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• NO_2^-+NO_3^- (µg/mL)	Proxy for agrochemical loading/denitrification potential	N		Y	Y	Y
• NO_2^- (µg/mL)	Proxy for agrochemical loading	N		Y	Y	Y
• NH_4^+ (µg/mL)	Proxy for agrochemical loading	N		Y	Y	Y
• DIN : NO_2^-+NO_3^-+NH_4^+ (µg/mL)	Proxy for agrochemical loading	N		Y	Y	Y
• NO_3^- (%)	Proxy for N cycling processes	N		Y	Y	Y
• NO_2^- (%)	Proxy for N cycling processes	N		Y	Y	Y
• NH_4^- (%)	Proxy for N cycling processes	N		Y	Y	Y
• DOC (µg/mL)	Proxy for organic matter cycling/denitrification potential	N		Y	Y	Y
• K^+ (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• Mg^{2+} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• Mn^{2+} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• Na^{2+} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• Zn^{2+} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• Ca^{2+} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• Fe^{2+} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• Al^{3+} (µg/mL)	Proxy for agrochemical loading/glyphosate adsorption	N		Y	Y	Y
• TSS ≥ 0.2µm (mg/mL)	Proxy for erosion	N	2	Y	Y	Y
• TSS ≥ 7µm (mg/mL)	Proxy for erosion	N	3	N	N	Y
• TVS (mg/mL)	Proxy for erosion/organic matter cycling	N	3	N	N	Y

Annexe A- 8 (Continued)

Matrices and Individual Environmental Variables	Rationale	Type	Restrictions	Time variability	Inter-sites spatial variability	Intra-site spatial variability
Soil Physico-Chemistry						
• Soil Moisture (%)	Proxy for biotic activity/water availability	N	4	N	nd	Y
• Soil Organic Matter (%)	Proxy for organic matter cycling/glyphosate adsorption	N	4	N	nd	Y
• N_{tot} (%)	Proxy for organic matter cycling	N	4	N	nd	Y
• C_{tot} (%)	Proxy for organic matter cycling	N	4	N	nd	Y
• C_{org} (%)	Proxy for organic matter cycling/glyphosate adsorption	N	4	N	nd	Y
• C/N	Proxy for organic matter cycling	N	4	N	nd	Y
• pH	Proxy for organic matter cycling	N	5	N	nd	Y
• EC	Proxy for organic matter cycling	N	5	N	nd	Y
• Glyphosate (µg/Kg dw)	Proxy for glyphosate adsorption	N	4	N	nd	Y
• AMPA (µg/Kg dw)	Proxy for glyphosate adsorption	N	4	N	nd	Y

Notes : (1) Only for BB 2014 Snowmelt; (2) Same samples as those analyzed for dissolved glyphosate; (3) Only SR 2013 Post-glyphosate (Campaign 8000) and BB 2014 Snowmelt (campaign 10 000); (4) Data from SR 2013 Post-glyphosate; (5) Data from field instrumentation in 2011

ANNEXE 9:
LANDSCAPE, CLIMATIC, FIELD SOIL AND AQUEOUX RBS FLUX CHARACTERISTICS IN BOISBRIAND (BB) AND SAINT-ROCH-DE-L'ACHIGAN (SR).

Average ± standard deviation (where available) environmental parameters on both research sites. Raw data was used in the RDA. Both sites have a similar climate, but BB has a richer soil, leading to nutrient enriched runoff and interstitial water in plant root zones. Furthermore, the water table of BB is shallower, suggesting enhanced accessibility for the RBS vegetation (see Chapters 1 to 4 for detailed methods to obtain these data).

	BB	SR
Landscape		
Elevation	44 m	46 m
Topography	Hilly	Flat
Water table depth (m)	0.36 ± 0.26	1.16 ± 0.14
Climatic		
Mean temperature (°C)	7.5 ± 0.3	7.0 ± 0.8
Degree days of growth (°C·d)	990 ± 7	989 ± 7
Precipitation (mm)	1034 ± 84	1121 ± 92
Relative ambient humidity	72.6 ± 0.8	72.5 ± 0.9
Solar radiation	284.0 ± 2.0	283.0 ± 3.4
Field soil		
Soil classification[1]	Organic-rich black soil, typic humisol	mineral sandy clay-loam soil sitting atop a clay bed
pH_{water}	6.6	6.48-6.83
pH (buffer)[2]	7.0	7.0-7.2
Organic Matter (%)	4.5%	2.4-3.0
CEC (meq/100g)	0.03-2	15.6-16.5
Granulometry (<2000 - <212 - <63 μm %)	6.1 – 13 – 81	43 – 30 – 27
P-Mehlich (kg/ha)[2]	297	129-239
K-Mehlich (kg/ha)[2]	569	90-147
Ca-Mehlich (kg/ha)[2]	6395	2057-6263
Mg-Mehlich (kg/ha)[2]	1-10	147-289
Runoff (0 cm)		
pH	7.0 ± 0.5	7.1 ± 0.6
Salinity	199 ± 231	62 ± 72
Electrical conductivity	316 ± 323	70 ± 62
Dissolved Organic Carbon	19.8 ± 17.7	10.9 ± 6.1
NO_2+NO_3	5315 ± 11902	604 ± 1163
NH_4	9352 ± 13395	1469 ± 3129
PO_4	1142 ± 1580	700 ± 1323
K	15.1 ± 24.02	5.4 ± 3.99
Interstitial water from		
Shallow root zone (35 cm)		
pH	6.7 ± 0.5	6.8 ± 0.4
Salinity	371 ± 270	101 ± 74
Electrical conductivity	534 ± 374	132 ± 49
Dissolved Organic Carbon	28.4 ± 16.7	9.2 ± 6.3
NO_2+NO_3	7117 ± 8805	201 ± 530
NH_4	79 ± 98	54 ± 118
PO_4	33 ± 40	37 ± 33
K	3.4 ± 4.39	1.7 ± 1.85
Deep root zone (70 cm)		
pH	6.4 ± 0.8	6.8 ± 0.4
Salinity	814 ± 567	128 ± 107
Electrical conductivity	1145 ± 695	215 ± 135
Dissolved Organic Carbon	25.3 ± 15.3	8.2 ± 6.4
NO_2+NO_3	4407 ± 6696	316 ± 1668
NH_4	242 ± 406	41 ± 61
PO_4	11 ± 12	32 ± 34
K	4.5 ± 2.84	0.8 ± 0.69

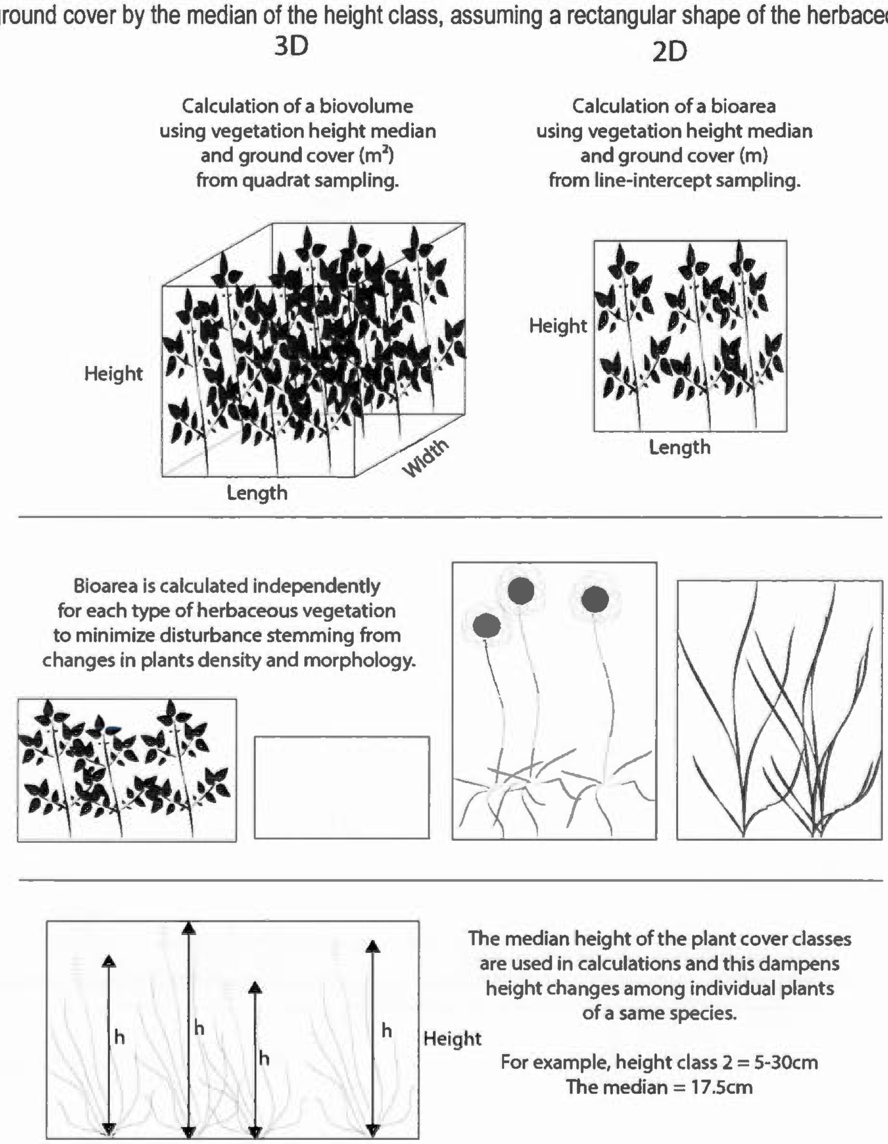
ANNEXE 10: NUMBER OF SPECIES AND RELATIVE PROPORTION OF HYDROPHYTES (OBLIGATE, FACULTATIVE WETLAND AND FACULTATIVE HYDROPHYTES) VERSUS NON-HYDROPHYTES (FACULTATIVE UPLAND, OBLIGATE UPLAND OR NON-CLASSIFIED) AT BOISBRIAND AND SAINT-ROCH-DE-L'ACHIGAN.

Hydrophytes (obligate or facultative) are more abundant at BB (38.3%) while non-hydrophytes (upland) dominate SR (78.0%), and this is likely influenced by topography and hydrogeology. In BB, hydrophytes are slightly more abundant on the edge-of-stream (CR) and non-hydrophytes least abundant in the center of the RBS (CC). The abundance of hydrophytes on the stream edge appears as a fundamental ecological niche (Tiner 1991; Gauthier et al. 2008). At both sites, there are generally more hydrophytic plants (coverage and diversity) in the herbaceous plots (CX), and in terms of coverage, there is sometimes a gradient between the three treatment instead of a dichotomic distinction (as in species diversity) between unplanted and willow parcels. This is consistent with a potentially lowered soil moisture below the willows (Annexe 3) which may be due to its strong evapotranspiration potential (Tabacchi et al. 2000). This hydrophytic plant heterogeneity may also indicate hydrology heterogeneity within sites (Tiner, R. W. (1991). "The Concept of a Hydrophyte for Wetland Identification." BioScience 41(4): 236-247).

	BB	**SR**	**Both**
Obligate Wetland	1	0	1
Facultative Wetland	12	3	14
Facultative Hydrophytes	5	6	8
Facultative Upland	24	24	37
Obligate Upland	5	8	11
Non-classified	2	7	7
Hydrophytes (n)	**18**	**9**	**23**
Hydrophytes (%)	**38,3%**	**22,0%**	**32,4%**
Non-hydrophytes (n)	**29**	**32**	**48**
Non-hydrophytes (%)	**61,7%**	**78,0%**	**67,6%**

ANNEXE 11:
THE CALCULATION OF BIOVOLUME (3D) VS BIOAREA (2D) FOR THE HERBACEOUS VEGETATION STRATA.

Biovolume is the product of size and cover. It represents the amount of space occupied by vegetation, and conveys information differing from biomass (Descoings 1975). The square approximation can be used for the herbaceous strata, and the minor distorsions that this implies are usually resolved when averaging several over- and underestimates. As plants were sampled along a line, and not a three-dimensional quadrat, relative plants **bioarea**, an adaptation of the plant biovolume measurement (Elias and Dias 2004) was calculated by multiplying absolute ground cover by the median of the height class, assuming a rectangular shape of the herbaceous strata.



ANNEXE 12:
HERBACEOUS VEGETATION ECOLOGICAL CHARACTERISTIC USED TO CHARACTERIZE THE DIVERSITY OF THE RBS AND TO EXPLORE RELATIONS WITH *SALIX*, HYDROLOGY, NUTRIENTS AND GLYPHOSATE IN THE CHAPTERS 1 TO 3.

List of the morphometric and ecological characteristics of the herbaceous vegetation selected as proxy for various RBS functions (i.e. biomass productivity, nutrient and glyphosate reduction efficiency, indigenous biodiversity and presence of exotic weeds). A rationale for the selection of each variable is then presented, with the corresponding references.

Variable	Proxy for	Rationale
Salix Final Biomass (2013 Kg dw/ha)	nutrient uptake shade	Above-ground biomass production is directly proportional to RBS nutrient retention (Jianqiang et al. 2008).Restricted light availability under a riparian *Salicaceae* canopy has previously been identified as a major driver for understory diversity (Fortier et al. 2011). Biomass was selected as a proxy for shade (Fortier et al. 2011). Vegetation diversity is known to play a positive role in the stability of biomass production (Proulx et al. 2010).
Herbaceous biomass (Kg dw/ha)	nutrient uptake	Competition with herbaceous vegetation may be detrimental to willow biomass production (Labrecque et al. 1994; Albertsson 2012). Above-ground biomass production is directly proportional to RBS nutrient retention (Jianqiang et al. 2008). Herbaceous buffers produce less biomass than woody ones Hefting et al. (2005). RBS with similar aboveground biomass production (mowed and harvested annually) are not all equally efficient (Uusi-Kämppä and Yläranta 1996)
Total biomass (Salix + Herbaceous vegetation; Kg dw/ha)	nutrient uptake	Above-ground biomass production is directly proportional to RBS nutrient retention (Jianqiang et al. 2008).
Litter soil cover (g dw/m^2)	surface rugosity temporal buffering of nutrients	Litter affects soil roughness which impacts hydrological processes such as runoff and erosion (Tabacchi et al. 2000). Litter can be used as a proxy for leaf biomass production in RBS, and litter can act as a temporary storage for nutrients (N) preventing sharp releases (Hefting et al. 2005).
Soil herbaceous cover (% cover)	surface rugosity	Herbaceous vegetation can reduce erosion, sediment and chemical transport in runoff (Uusi-Kämppä and Yläranta 1996; Uusi-Kämppä et al. 2000; Udawatta et al. 2002; McKergow et al. 2006; Dosskey et al. 2007; Dosskey et al. 2010; Sabater et al. 2003). Greater stem density of herbaceous RBS further slows

Annexe 12 (Continued)

Variable	Proxy for	Rationale
		runoff and favor deposition (Dosskey et al. 2010).
Herbaceous bare soil cover (% cover)	erosion potential	Bare soil is most at risk of erosion and vegetated cover decreases this risk (Correll 1996). Understory vegetation control in woody RBS would normally enhance soil erosion and TSS runoff compared to grassed RBS (McKergow et al. 2006).
Life cycle (Annuals; Biennials; Perennials; soil cover %)	ecological niche selection factors	Herbicides can affect plant community structure by selecting for annual herbaceous plants which are protected from herbicides like glyphosate that affects live plants and not plants protected by a seed coat (Jobin et al. 1997).
Weed diversity (Soil cover %; Proportion %; n sp)	ecological niche selection factors	Riparian zones may be a recruitment habitat and dispersal corridor for invasive weeds (Boutin et al. 2003). Weeds represent major economic burdens and conservation threats worldwide (Pimentel et al. 2005). However, weeds also play an important role in maintaining farmland biodiversity (Storkey 2006), and tolerable levels (below economic threshold) of certain weeds is recommended to aid in the survival of pollinators (Nicholls and Altieri 2013).
Exotic weed diversity (n sp)	ecological niche selection factors	Conservationists may disagree with agronomists with respect to weed definitions, hence exotic weeds were distinguished from weed diversity. Exotic species invasion may be greater in riparian ecotones compared to upland habitats, especially in fragmented landscapes with a modified hydrology (Hood and Naiman 2000; Planty-Tabacchi et al. 1996). Weeds represent major economic burdens and conservation threats worldwide (Pimentel et al. 2005). Free-growing herbaceous RBS (early successional stage) may be more sensitive to exotic invaders than RBS where woody plants are planted (D'Antonio and Chambers 2006). The presence of willows has been shown to decrease invasive species (IRSTEA 2014; Cavaillé et al. 2013).
Roots (Fibrous, Tap, ND; Soil cover %; Proportion %; n species)	nutrient infiltration potential	Belowground, tap roots favor infiltration (Reubens et al. 2007) while fibrous roots may better uptake immobile and mobile nutrients because they colonize more surface (Dunbabin et al. 2004). Willows have fibrous root systems (Kuzovkina and Volk 2009), but the root morphologies of other plants colonizing planted or unplanted buffer may vary.
Hydrophytes (UPL, FAC, FACU, FACW,	hydrology	The hydrophitic status of plants is an indicator of local hydrological processes, for instance, hydrophytes may establish themselves where the water table is high or where runoff is available (Tiner 1991). Obligate

Annexe 12 (Continued)

Variable	Proxy for	Rationale
OBL, ND; Hydrophytes vs non-hydrophytes; Soil cover %; Proportion %; n species)		and facultative hydrophytes may be used to infer the hydrology of a site, such as the high water line, for regulatory purposes (Gauthier et al. 2008). Riparian plants can lower water table depth and reduce contact between groundwater and plant roots which can in turn affect water storage potential and infiltration or denitrification potential (Dosskey et al. 2010).
Indigenous herbs (Soil cover %; Proportion; n species)	ecological niche selection factors	Salicaceae canopy closure may favor indigenous plants and disfavor exotic plants compared to free-growing herbaceous RBS closure (Fortier et al. 2011). Shade may increase the sensitivity of plants to herbicides (Lin et al. 2004).
Shade tolerance (Intolerant, Intermediate, Tolerant; Soil cover %; Proportion %; n species)	ecological niche selection factors	Restricted light availability under a riparian *Salicaceae* canopy has previously been identified as a major driver for understory diversity (Fortier et al. 2011).
Herbaceous plant height (class median, cm)	influence on biodiversity	Height of plants, as a measurement of plant architecture, may influence biodiversity indicators in vegetation communities (Schwab et al. 2002).
Salix (soil cover %)	shade	Restricted light availability under a riparian *Salicaceae* canopy has previously been identified as a major driver for understory diversity (Fortier et al. 2011).
Plant richness (including Salix)	ecological niche selection factors	Diversified ecosystems potentially enhance functional detoxification capabilities (Altieri 1999). Glyphosate may exert a sub-lethal acute toxicity in the ruderal ditch species populating RBS potentially affecting plant diversity (Saunders et al. 2013)
Shannon diversity H' (including Salix)	ecological niche selection factors	Vegetation diversity is known to play a positive role in the stability of biomass production and in the temporal stability of ecosystemic functions (Proulx et al. 2010). Diversity is a desirable landscaping attribute (Nagendra 2002).
Simpson 1/D	ecological niche selection factors	This index represents evenness, and tells us if plants are equitably distributed or not (Nagendra 2002). Vegetation diversity is known to play a positive role in the stability of biomass production and in the temporal stability of ecosystemic functions (Proulx et al. 2010).

Variable	Proxy for	Rationale
Bioarea (height x % cover)	nutrient uptake influence on biodiversity	Plant biovolume, influenced by soil stability and fertility (Velazquez and Gomez-Sal 2009) or woody canopy (Menaut and Cesar 1979), may also affect sediment trapping efficiency (Corenblit et al. 2013). Bioarea is an adaptation from plant biovolume (Elias and Dias 2004), as plants were sampled along a line, and not a three-dimensional quadrat. Plantations architecture may also influence biodiversity (Schwab et al. 2002).

ANNEXE 13:
ECOLOGICAL CHARACTERISTICS OF THE HERBACEOUS LAYER IN THE RIPARIAN BUFFER STRIPS.

Herbaceous vegetation diversity indexes, origin and weediness, hydrophitic status, life cycle, root morphology and shade tolerance on both sites, according to treatment and side of the RBS. Methodological notes for calculation of diversity indexes[1] and statistical analyses[2] are presented at the end of the table. Statistics reported aim at explicating how vegetation varies between experimental sites (BB vs SR), vegetation treatments (0, 33 333 or 55 556 stems/ha of willows, abbreviated CX, 3X and 5X, respectively) and side of the RBS (close to the edge-of-field, center or closer to the edge-of-stream, abbreviated CF, CC and CR, respectively). Significant probabilities ($p < 0.05$) and trends ($p \leq 0.10$) are reported and non-significant probabilities are identified as NS. Small letters (a-d) identify statistically distinct groups. Weeds are discussed at the end of the table[3].

				Indexes				Ground cover (%)		Ground cover (%)								Richness (n species)								
				Shannon's		Simpson's reciprocal		Bare soil		Indigenous		Exotic		All weeds		Exotic weeds		Indigenous		Exotic		All weeds			Exotic weeds	
Site	Trt	Side	n	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	SD	Mean	SE
BB	CX	CF	3	2.1	0.1 a	5.2	0.5 a	0.0	0.0 d	27.3	4.2 ab	72.7	4.2 ab	100.0	0.0 a	72.7	4.2 ab	3.7	0.3 a	4.0	0.6 a	7.7	0.7	1.15 a	6.0	0.6 a
		CC	6	2.0	0.1 ab	4.6	0.5 a	0.0	0.0 d	31.3	14.5 ab	68.7	14.5 ab	98.5	1.5 ab	67.2	13.9 ab	3.7	0.4 a	4.8	0.7 a	8.3	0.4	1.03 a	5.5	0.7 a
		CR	3	1.4	0.1 bc	3.0	0.2 a	0.0	0.0 d	48.3	12.8 a	51.7	12.8 b	100.0	0.0 a	51.7	12.8 bc	2.7	0.3 a	4.3	0.9 a	7.0	1.0	1.73 ab	4.3	0.3 ab
	3X	CF	3	1.0	0.3 cd	2.6	0.7 b	62.5	7.2 bc	7.5	5.4 b	92.5	5.4 a	91.5	4.8 bc	87.0	4.3 a	2.3	0.7 b	2.3	0.9 c	4.0	1.5	2.65 cd	3.3	1.2 bc
		CC	6	0.8	0.1 d	2.6	0.2 b	75.0	3.6 ab	9.2	6.0 b	90.8	6.0 a	85.1	9.2 bc	77.6	8.9 a	2.5	0.3 b	1.3	0.4 c	3.3	0.4	1.03 d	3.0	0.4 cd
		CR	3	1.1	0.4 cd	4.5	1.4 b	35.8	7.9 c	29.8	20.9 ab	70.2	20.9 ab	87.8	7.2 bc	68.2	20.5 ab	3.3	0.3 b	2.7	0.9 c	5.0	0.6	1.00 cd	3.7	0.3 bcd
	5X	CF	3	1.1	0.1 cd	3.1	0.1 b	70.8	4.2 ab	21.6	5.6 ab	78.4	5.6 ab	93.3	3.4 b	71.7	5.7 ab	2.0	0.6 c	3.3	0.7 b	4.7	0.3	0.58 cd	3.3	0.7 bcd
		CC	6	0.7	0.2 d	3.4	0.6 b	87.5	0.0 a	39.1	9.7 a	60.9	9.7 b	55.8	13.1 c	23.9	8.9 c	1.5	0.5 c	2.7	0.4 b	2.7	0.7	1.75 b	1.5	0.5 d
		CR	3	1.3	0.2 c	3.4	0.2 b	55.8	10.2 bc	26.3	12.0 ab	73.7	12.0 ab	90.6	4.7 bc	70.5	10.3 ab	3.0	0.6 c	3.7	0.9 b	5.7	1.8	3.06 bc	3.7	1.2 bc
p_{trt}				<0,0001*		<0,0001*		<0,0001*		0,0004*		0,0006*		<0,0001*		<0,0001*		<0,0001*		<0,0001*		<0,0001*			<0,0001*	
p_{side}				0.0307*		0,9318		0,0001*		0,0311*		0,0331*		0.0059*		0,0004*		0,0521		0,1801		0,0443*			0.0153*	
p_{trt} X p_{side}				<0,0001*		<0,0001*		0,0071*		0.0425*		0,0458*		0.1627		<0,0001*		<0,0001*		0.1018		0.0007*			0.0072*	
SR	CX	CF	3	1.5	0.4 abc	3.5	0.7 ab	0.0	0.0 e	76.4	9.6 a	23.6	9.6 b	86.0	12.7 b	23.5	9.6 b	1.7	0.7	7.0	2.1 a	8.0	2.1	3.61 a	5.7	1.5
		CC	6	2.3	0.3 a	5.6	0.8 a	0.1	0.0 e	47.3	5.6 b	52.7	5.6 a	90.1	5.2 b	52.7	5.6 a	2.3	0.6	7.3	0.7 a	8.5	1.0	2.43 a	6.5	0.9
		CR	3	2.2	0.5 ab	5.1	0.9 ab	0.0	0.0 e	54.9	13.3 ab	45.1	13.3 ab	74.3	12.9 b	45.1	13.4 ab	2.7	1.7	8.0	2.1 a	9.7	3.2	5.51 a	7.3	2.6
	3X	CF	3	1.2	0.0 cd	3.1	0.4 b	24.2	3.3 bcd	54.5	9.4 a	45.5	9.4 b	99.3	0.7 a	45.2	9.0 b	2.3	0.9	5.3	0.9 ab	7.0	0.6	1.00 ab	5.0	0.6
		CC	6	1.4	0.1 abc	4.5	0.4 ab	35.0	3.8 abc	47.7	7.3 b	52.3	7.3 a	93.5	4.7 a	52.0	7.4 a	2.2	0.2	5.7	0.3 ab	7.2	0.5	1.17 ab	5.0	0.4
		CR	3	1.9	0.3 ab	4.7	1.1 ab	9.3	2.1 cd	62.0	18.9 ab	38.0	18.9 ab	98.3	0.9 a	38.0	18.9 ab	2.0	0.6	8.3	0.9 ab	9.7	1.2	2.08 ab	6.7	1.2
	5X	CF	3	1.0	0.1 cd	3.5	0.6 ab	47.5	7.5 ab	61.6	18.3 a	38.4	18.3 b	95.9	4.1 a	38.4	18.3 b	1.0	1.0	5.7	0.9 b	6.0	2.0	3.46 b	4.3	1.9
		CC	6	0.9	0.1 d	4.0	0.7 ab	59.2	1.7 a	45.6	11.8 b	54.4	11.8 a	87.7	8.1 a	54.4	11.8 a	1.7	0.3	5.2	0.5 b	6.3	0.5	1.21 b	4.8	0.5
		CR	3	1.3	0.4 bcd	3.1	0.8 ab	21.0	10.4 d	51.7	17.1 ab	48.3	17.1 ab	99.3	0.7 a	48.3	17.1 ab	2.3	0.3	5.3	2.0 b	7.3	1.8	3.06 b	6.0	1.5
p_{trt}				<0,0001*		0,0210*		<0,0001*		0.3524		0.3483		0.0123*		0.3136		0.3790		0.0064*		0.0108*			0.0684	
p_{side}				0,0007*		0,0033*		<0,0001*		0.0144*		0.0141*		0.0636		0.0166*		0.5187		0.3311		0.0921			0.0767	
p_{trt} X p_{side}				0,0355*		0.2942		0,0028*		0.5214		0.5256		0.5564		0.4786		0.3215		0.0504		0.6223			0.5760	

Annexe 13: Continued

Hydrophytes

Total coverage (sum %)

Site	Trt	Side	FAC Mean	FAC SE	FACU Mean	FACU SE	FACW Mean	FACW SE	OBL Mean	OBL SE	UPL Mean	UPL SE
BB	CX	CF	13.5	6.0^{a}	108.5	8.5^{a}	58.5	5.3^{a}	0.0	0.0	2.5	1.3^{ab}
		CC	14.3	4.8^{a}	81.3	7.6^{ab}	48.6	11.2^{ab}	0.0	0.0	18.3	4.2^{a}
		CR	7.8	3.2^{a}	75.0	9.5^{abc}	62.5	7.2^{a}	0.0	0.0	1.0	0.5^{ab}
	3X	CF	2.5	1.3^{b}	54.7	4.3^{abc}	3.5	1.1^{d}	0.0	0.0	0.0	0.0^{b}
		CC	1.3	0.6^{b}	46.3	12.8^{bcd}	9.2	4.6^{d}	0.0	0.0	0.0	0.0^{b}
		CR	6.8	2.7^{b}	66.0	17.7^{abc}	16.0	5.6^{cd}	0.0	0.0	0.0	0.0^{b}
	5X	CF	1.0	0.5^{b}	42.8	3.8^{cd}	9.3	2.1^{cd}	0.0	0.0	0.0	0.0^{b}
		CC	1.3	0.6^{b}	10.9	3.1^{d}	10.3	4.4^{cd}	0.5	0.3	0.5	0.3^{b}
		CR	1.0	0.5^{b}	69.5	15.4^{abc}	19.7	2.2^{bc}	0.0	0.0	0.0	0.0^{b}
p_{trt}			0.0013*		<0.0001*		<0.0001*		NS		<0.0001*	
p_{side}			NS		0.0029*		0.0042*		NS		0.0185*	
$p_{trt} \times p_{side}$			NS		0.0169*		NS		NS		NS	
SR	CX	CF	81.2	27.1	80.8	9.8^{ab}	0.0	0.0^{a}	0.0	0.0	1.0	0.5^{a}
		CC	50.9	9.3	94.8	5.6^{a}	6.3	3.1^{a}	0.0	0.0	15.4	4.1^{a}
		CR	40.3	11.5	74.3	6.5^{ab}	0.0	0.0^{a}	0.0	0.0	12.8	3.2^{a}
	3X	CF	39.3	6.6	25.5	2.9^{de}	0.0	0.0^{ab}	0.0	0.0	0.0	0.0^{b}
		CC	31.2	5.1	42.3	5.0^{cd}	0.0	0.0^{ab}	0.0	0.0	0.0	0.0^{b}
		CR	46.2	6.5	58.5	8.0^{bc}	2.0	0.5^{ab}	0.0	0.0	1.0	0.5^{b}
	5X	CF	35.2	11.3	41.3	5.5^{cd}	0.0	0.0^{b}	0.0	0.0	0.0	0.0^{b}
		CC	25.5	8.8	10.8	1.0^{e}	0.0	0.0^{b}	0.0	0.0	0.0	0.0^{b}
		CR	54.5	13.2	31.7	3.0^{d}	0.0	0.0^{b}	0.0	0.0	1.0	0.5^{b}
p_{trt}			NS		<0.0001*		0.0244*		0.0618^{NS}		<0.0001*	
p_{side}			NS		0.0159*		NS		NS		NS	
$p_{trt} \times p_{side}$			NS		<0.0001*		0.0425*		NS		NS	

Proportional coverage (%)

Site	Trt	Side	FAC Mean	FAC SE	FACU Mean	FACU SE	FACW Mean	FACW SE	OBL Mean	OBL SE	UPL Mean	UPL SE
BB	CX	CF	0.1	0.0^{a}	0.6	0.0^{bcd}	0.3	0.0^{ab}	0.0	0.0	0.0	0.0^{ab}
		CC	0.1	0.0^{a}	0.5	0.0^{cd}	0.3	0.1^{ab}	0.0	0.0	0.1	0.0^{a}
		CR	0.1	0.0^{a}	0.5	0.1^{cd}	0.4	0.0^{a}	0.0	0.0	0.0	0.0^{ab}
	3X	CF	0.0	0.0^{ab}	0.8	0.0^{a}	0.0	0.0^{bcd}	0.0	0.0	0.0	0.0^{b}
		CC	0.0	0.0^{ab}	0.7	0.0^{ab}	0.1	0.0^{cd}	0.0	0.0	0.0	0.0^{b}
		CR	0.1	0.0^{ab}	0.6	0.1^{bcd}	0.2	0.1^{abc}	0.0	0.0	0.0	0.0^{b}
	5X	CF	0.0	0.0^{b}	0.7	0.0^{ab}	0.1	0.0^{bcd}	0.0	0.0	0.0	0.0^{b}
		CC	0.0	0.0^{b}	0.3	0.1^{d}	0.2	0.1^{bcd}	0.0	0.0	0.0	0.0^{b}
		CR	0.0	0.0^{b}	0.7	0.0^{abc}	0.2	0.0^{abc}	0.0	0.0	0.0	0.0^{b}
p_{trt}			0.0111*		<0.0001*		<0.0001*		0.1500		<0.0001*	
p_{side}			NS		<0.0001*		0.0007*		0.7522		0.0105*	
$p_{trt} \times p_{side}$			NS		0.0002*		NS		NS		NS	
SR	CX	CF	0.3	0.1^{b}	0.5	0.1	0.0	0.0^{a}	0.0	0.0	0.0	0.0
		CC	0.2	0.0^{b}	0.5	0.0	0.0	0.0^{a}	0.0	0.0	0.1	0.0
		CR	0.2	0.1^{b}	0.4	0.0	0.0	0.0^{a}	0.0	0.0	0.1	0.0
	3X	CF	0.5	0.0^{a}	0.3	0.0	0.0	0.0^{ab}	0.0	0.0	0.0	0.0
		CC	0.4	0.0^{a}	0.5	0.0	0.0	0.0^{ab}	0.0	0.0	0.0	0.0
		CR	0.4	0.1^{a}	0.5	0.1	0.0	0.0^{ab}	0.0	0.0	0.0	0.0
	5X	CF	0.4	0.0^{ab}	0.6	0.1	0.0	0.0^{b}	0.0	0.0	0.0	0.0
		CC	0.4	0.1^{ab}	0.3	0.0	0.0	0.0^{b}	0.0	0.0	0.0	0.0
		CR	0.4	0.1^{ab}	0.3	0.0	0.0	0.0^{b}	0.0	0.0	0.0	0.0
p_{trt}			0.0139*		0.1174		0.0244*		0.0618^{NS}		<0.0001*	
p_{side}			NS		0.0558^{NS}		NS		NS		NS	
$p_{trt} \times p_{side}$			NS		<0.0001*		0.0425*		NS		NS	

Richness (n species)

Site	Trt	Side	FAC Mean	FAC SE	FACU Mean	FACU SE	FACW Mean	FACW SE	OBL Mean	OBL SE	UPL Mean	UPL SE
BB	CX	CF	0.7	0.2^{a}	4.0	0.3^{a}	2.0	0.3^{a}	0.0	0.0	0.3	0.2^{ab}
		CC	0.7	0.2^{a}	4.3	0.1^{a}	2.2	0.2^{a}	0.0	0.0	1.0	0.0^{a}
		CR	1.0	0.3^{a}	4.0	0.3^{a}	1.0	0.0^{a}	0.0	0.0	0.3	0.2^{ab}
	3X	CF	0.3	0.2^{b}	3.0	0.5^{b}	0.7	0.2^{c}	0.0	0.0	0.0	0.0^{b}
		CC	0.2	0.1^{b}	2.7	0.1^{b}	0.3	0.2^{c}	0.0	0.0	0.0	0.0^{b}
		CR	0.7	0.2^{b}	3.3	0.2^{b}	1.0	0.3^{c}	0.0	0.0	0.0	0.0^{b}
	5X	CF	0.3	0.2^{b}	3.0	0.0^{b}	1.0	0.0^{b}	0.0	0.0	0.0	0.0^{b}
		CC	0.2	0.1^{b}	1.8	0.3^{b}	1.0	0.3^{b}	0.2	0.1	0.2	0.1^{b}
		CR	0.3	0.2^{b}	3.0	0.5^{b}	2.3	0.3^{b}	0.0	0.0	0.0	0.0^{b}
p_{trt}			0.0007*		<0.0001*		<0.0001*		NS		<0.0001*	
p_{side}			NS		NS		NS		NS		0.0468*	
$p_{trt} \times p_{side}$			NS		NS		0.0002*		NS		NS	
SR	CX	CF	2.3	0.4	4.0	0.5	0.0	0.0^{a}	0.0	0.0	0.3	0.2^{a}
		CC	2.0	0.1	4.3	0.5	0.2	0.1^{a}	0.0	0.0	0.5	0.1^{a}
		CR	2.0	0.6	5.0	1.2	0.0	0.0^{a}	0.0	0.0	1.3	0.4^{a}
	3X	CF	1.3	0.2	4.3	0.4	0.0	0.0^{ab}	0.0	0.0	0.0	0.0^{b}
		CC	1.8	0.2	4.5	0.0	0.0	0.0^{ab}	0.0	0.0	0.0	0.0^{b}
		CR	2.0	0.0	4.7	0.7	0.7	0.2^{ab}	0.0	0.0	0.3	0.2^{b}
	5X	CF	1.7	0.2	4.0	0.8	0.0	0.0^{b}	0.0	0.0	0.0	0.0^{b}
		CC	1.8	0.3	3.2	0.2	0.0	0.0^{b}	0.0	0.0	0.0	0.0^{b}
		CR	1.7	0.2	3.7	0.4	0.0	0.0^{b}	0.0	0.0	0.3	0.2^{b}
p_{trt}			NS		0.0588^{NS}		0.0283*		0.0618^{NS}		<0.0001*	
p_{side}			NS		NS		NS		NS		0.0762^{NS}	
$p_{trt} \times p_{side}$			NS		NS		0.0344*		NS		NS	

Annexe A- 13 (Continued)

			Life cycle – Total coverage (sum %)								Roots – Total coverage (sum %)				Roots – Proportional coverage (%)			
			Annuals		Biennials		Perennials		A+B		Tap		Fibrous		Tap		Fibrous	
Site	Trt	Side	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
BB	CX	CF	79.5	14.9 a	26.7	13.3	88.5	8.5 abc	106.2	11.2 a	147.8	5.7 a	46.8	4.7 ab	6.3	0.5 a	1.3	0.2 b
		CC	31.3	8.5 a	25.1	12.5	114.3	10.8 ab	56.3	6.0 a	95.2	13.7 ab	74.9	15.5 ab	5.3	1.5 ab	2.8	0.5 b
		CR	6.8	2.7 a	33.5	2.0	114.3	4.8 a	40.3	4.3 a	68.2	13.1 abc	79.7	10.9 a	3.3	1.8 bc	3.0	0.5 a
	3X	CF	6.0	2.3 b	6.8	2.7	50.3	2.0 cd	12.8	3.2 b	56.2	4.7 abc	9.5	2.2 c	3.0	1.5 b	1.7	0.3 b
		CC	6.8	3.0 b	4.8	0.9	50.0	11.0 de	11.5	3.2 b	45.0	12.6 cd	19.4	3.8 c	2.5	0.4 c	1.3	0.1 b
		CR	15.5	6.3 b	1.0	0.5	74.8	16.1 bcd	16.5	6.0 b	50.5	19.4 cd	43.8	16.1 bc	2.0	0.9 c	3.3	0.6 a
	5X	CF	12.8	3.2 ab	12.5	6.3	31.3	6.9 de	25.3	9.4 b	42.8	4.8 bc	16.3	2.2 c	3.0	0.0 c	2.3	0.2 b
		CC	6.8	2.7 ab	4.7	2.0	19.3	2.0 e	11.5	4.6 b	11.9	3.2 d	18.3	3.6 c	2.2	1.1 c	1.8	0.2 b
		CR	14.8	4.5 ab	42.7	13.1	35.2	2.6 de	57.5	17.6 b	68.0	16.2 abc	28.2	1.8 bc	3.0	1.7 c	3.7	0.2 a
p_{trt}			0.0075*		0.0799		<0.0001*		<0.0001*		<0.0001*		<0.0001*		<0.0001*		0.1966	
p_{side}			0.0609		0.1704		0.0366*		0.2690		0.0004*		0.0020*		0.0049*		<0.0001*	
$p_{trt} \times p_{side}$			0.0153*		0.0065*		0.3086		0.1350		0.0006*		0.7799		0.1056		0.0686	
SR	CX	CF	3.7	1.1	14.5	6.5 ab	159.5	18.4 a	18.2	7.5 abc	35.5	7.3 a	112.3	30.9 bc	3.4	0.9 a	4.0	1.0 b
		CC	11.6	3.0	19.6	2.2 a	162.8	10.0 a	31.2	4.6 a	50.0	4.7 a	159.8	2.4 a	3.2	0.7 a	6.3	0.5 ab
		CR	8.0	3.3	18.3	5.4 ab	131.3	9.4 a	26.3	3.6 ab	39.7	1.5 a	168.0	6.4 a	3.3	2.0 a	7.3	1.1 a
abc	3X	CF	2.3	0.6	1.0	0.5 b	64.7	10.1 cd	3.3	0.1 c	25.7	1.2 b	48.3	7.4 bc	3.0	0.0 ab	4.7	0.2 b
		CC	10.8	4.2	3.0	0.8 ab	65.3	6.5 cd	13.8	3.5 abc	21.7	2.2 b	60.9	11.5 bc	2.8	0.7 ab	5.0	0.1 ab
		CR	2.2	0.4	5.0	1.3 ab	112.5	3.9 ab	7.2	1.1 bc	15.0	4.0 b	107.7	6.2 ab	2.7	1.0 ab	7.7	0.3 a
	5X	CF	7.0	3.4	1.0	0.5 b	70.7	18.1 bcd	8.0	3.2 c	12.5	3.7 b	66.2	19.3 bc	2.3	2.0 b	4.3	0.3 b
		CC	2.3	0.3	3.0	0.8 ab	36.3	7.5 d	5.3	0.5 c	14.3	3.3 b	36.0	7.8 c	2.5	1.1 b	4.3	0.2 ab
		CR	4.5	1.6	13.5	6.0 ab	97.3	13.0 abc	18.0	6.0 abc	19.0	6.2 b	96.3	13.7 ab	2.0	0.9 b	5.7	0.9 a
p_{trt}			0.3176		0.0003*		<0.0001*		<0.0001*		<0.0001*		<0.0001*		0.0106*		0.0911	
p_{side}			0.2012		0.0113*		0.0003*		0.0034*		0.5525		<0.0001*		0.2572		0.0117*	
$p_{trt} \times p_{side}$			0.7307		0.7155		0.0002*		0.2507		0.0851		0.0177*		0.9277		0.2796	

			Shade tolerance – Total coverage (sum %)						Shade tolerance – Proportional coverage (%)					
			Intolerant		Medium		Tolerant		Intolerant		Medium		Tolerant	
Site	Trt	Side	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
BB	CX	CF	61.0	12.8 abc	99.3	4.6 a	34.3	9.5 ab	2.3	0.2 a	4.3	0.2 a	1.0	0.3 b
		CC	97.3	2.4 ab	56.2	5.0 b	20.1	8.0 ab	4.2	0.3 a	3.7	0.2 abc	0.7	0.2 b
		CR	112.7	7.7 a	42.0	6.8 bc	0.0	0.0 ab	3.0	0.3 a	4.0	0.3 ab	0.0	0.0 b
	3X	CF	5.5	2.0 d	55.2	4.4 b	5.0	1.3 b	1.3	0.4 b	2.7	0.4 cd	0.7	0.2 b
		CC	30.2	13.4 cd	27.0	4.0 cd	7.3	3.3 b	1.3	0.3 b	2.0	0.0 d	0.5	0.1 b
		CR	63.0	14.3 abc	25.8	3.7 cd	7.5	2.2 b	2.3	0.2 b	2.7	0.3 cd	1.0	0.3 b
	5X	CF	22.0	7.6 cd	27.8	6.6 cd	9.3	2.1 a	2.0	0.3 b	2.3	0.3 d	1.0	0.0 a
		CC	5.2	0.9 d	11.2	2.8 d	14.4	3.0 a	0.7	0.1 b	2.2	0.3 d	1.3	0.2 a
		CR	46.8	19.7 bcd	39.8	4.7 bc	9.5	1.0 a	2.0	0.5 b	3.0	0.3 bcd	1.7	0.3 a
p_{trt}			<0.0001*		<0.0001*		0.0416*		<0.0001*		<0.0001*		0.0021*	
p_{side}			<0.0001*		<0.0001*		0.0546		0.1349		0.0376*		0.8512	
$p_{trt} \times p_{side}$			0.0407*		<0.0001*		0.0060*		0.0004*		0.4431		0.0071*	
SR	CX	CF	53.2	12.2 a	111.8	35.3 ab	34.5	7.8 ab	2.0	0.3 a	5.0	0.9	1.7	0.2 ab
		CC	78.0	7.6 a	66.3	6.7 b	55.8	5.6 ab	3.2	0.2 a	4.3	0.4	1.3	0.2 abc
		CR	68.2	15.6 a	67.7	17.0 a	71.8	3.9 a	3.7	0.8 a	5.7	0.9	1.3	0.2 abc
abc	3X	CF	18.8	3.2 b	54.7	8.0 ab	0.2	0.1 f	2.3	0.2 b	4.3	0.4	0.3	0.2 d
		CC	29.9	5.5 b	44.4	5.1 b	7.8	1.2 cd	2.7	0.2 b	3.8	0.3	1.2	0.2 bc
		CR	33.5	3.6 b	60.0	1.5 a	29.2	5.7 bc	3.0	0.6 b	5.0	0.5	2.3	0.2 a
	5X	CF	38.0	6.1 b	37.8	11.9 ab	2.7	1.2 ef	2.3	0.4 b	3.3	0.6	0.7	0.2 cd
		CC	12.4	1.9 b	29.3	7.7 b	8.7	3.2 de	2.5	0.3 b	3.2	0.1	1.2	0.1 bc
		CR	16.3	3.4 b	87.7	6.9 a	10.3	2.6 cd	2.0	0.3 b	4.0	0.8	1.3	0.2 abc
p_{trt}			<0.0001*		0.2098		<0.0001*		0.2271		0.0550		0.0063*	
p_{side}			0.7506		0.0251*		<0.0001*		0.0977		0.2846		<0.0001*	
$p_{trt} \times p_{side}$			0.0038*				0.0050*		0.5078		0.9765		<0.0001*	

Annexe 13: (Continued)

Notes:

[1]**Calculation of diversity indexes**: Shannon's — H', Eqn. 1, (Shannon 2001) — and Simpson's reciprocal diversity index — D^{-1}, Eqn. 2, (Simpson 1949) — were calculated as follow:

$$\text{Eqn. 1: } H' = -\sum_i^N p_i \times \log p_i$$

$$\text{Eqn. 2: } D^{-1} = \left[\sum_i^N \frac{n(n-1)}{N(N-1)}\right]^{-1}$$

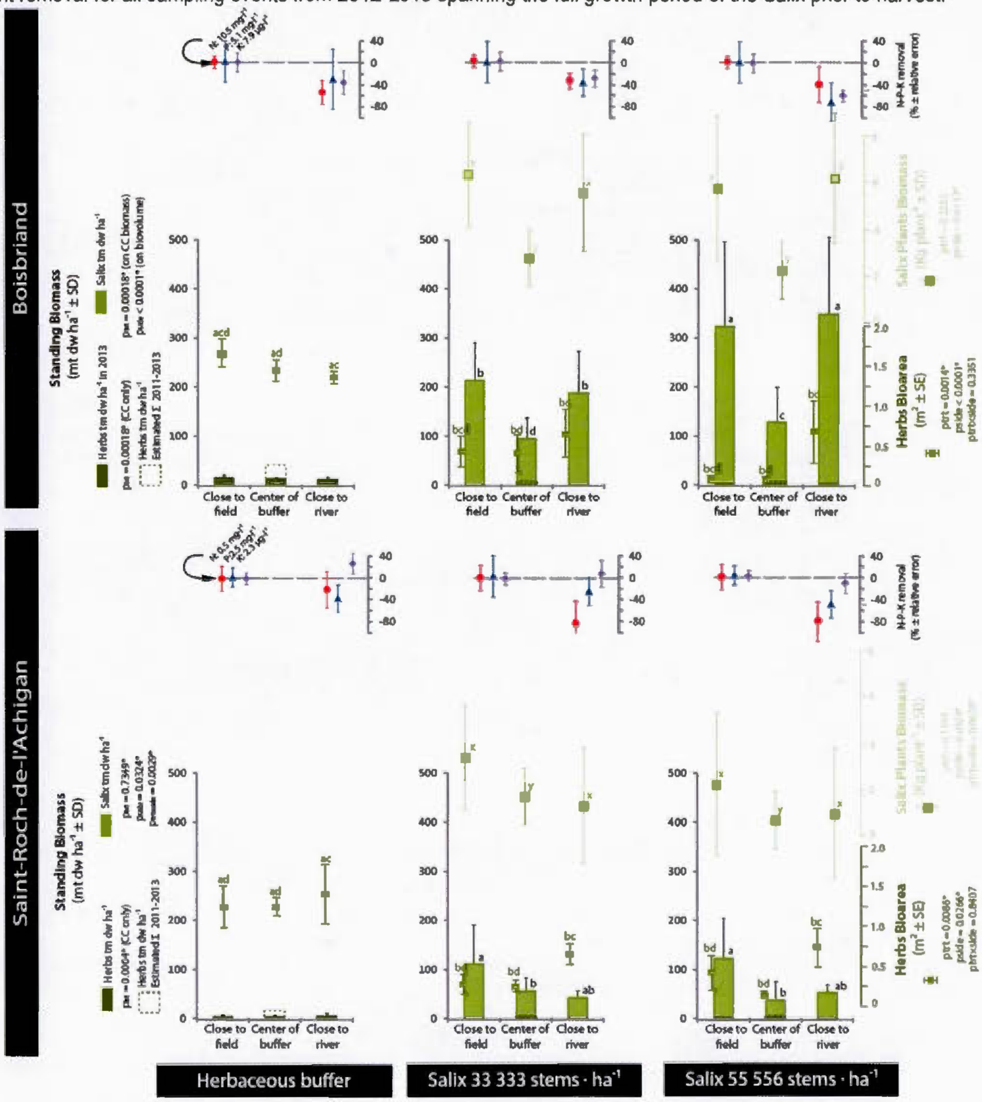
Where n described the cover of a given species and N is the total plant cover of each plot, and p_i refers to n/N. The median of each cover class was used in calculations.

[2]**Statistical analysis**: When data conformed to the normality and homoscedasticity requirements, a full factorial ANOVA was used, with factors tested including treatment (CX, 3X, 5X) and side (CF, CC, CR). Blocks could not be tested as measurements within each parcels (treatment x side) were not replicated. Post-hoc Tukey tests were conducted when a significant effect was observed and significance was reported in the relevant figures. Non-parametric ANOVAs were conducted on the ranked variables where necessary. These statistical analyses were conducted using JMP 10 (SAS Institute, Cary, NC).

[3]**Weeds and exotics weeds:** *Ground cover* - Weeds are significantly more abundant in the unplanted parcel (BB: $p < 0.0001^*$; SR: $p = 0.0123^*$) and significantly more important on the herbaceous plot margins than in the field-edge and center of the high density willow plantations in BB (no effect of side in SR). In BB, the least abundant exotic weeds ground cover is found within the high density willow parcels ($p < 0.0001^*$). In SR, exotic weed species took refuge directly in the middle of the RBS ($p = 0.0166^*$) and treatment did not influence this result. *Richness* – Exotic plants and exotic weeds are enriched (n species) in the herbaceous parcel compared to the low density willow plantation in BB ($p < 0.0001^*$), though exotic weed species richness is most reduced in the center of high density willow plantations compared to the edge-of-field and center of the herbs plots ($p = 0.0153^*$; interaction $p = 0.0072^*$) . Weeds are significantly enriched in the herbaceous strata ($p = 0.0108^*$), though side ($p = 0.0443^*$) and interactions ($p = 0.0007^*$) between both factors complexify the portrait. In SR, exotic plants and weed species are richer on the edge-of-field than on the stream-edge ($p = 0.0064^*$ and $p = 0.0108^*$, respectively) but there are no effect of side, nor any interactions.

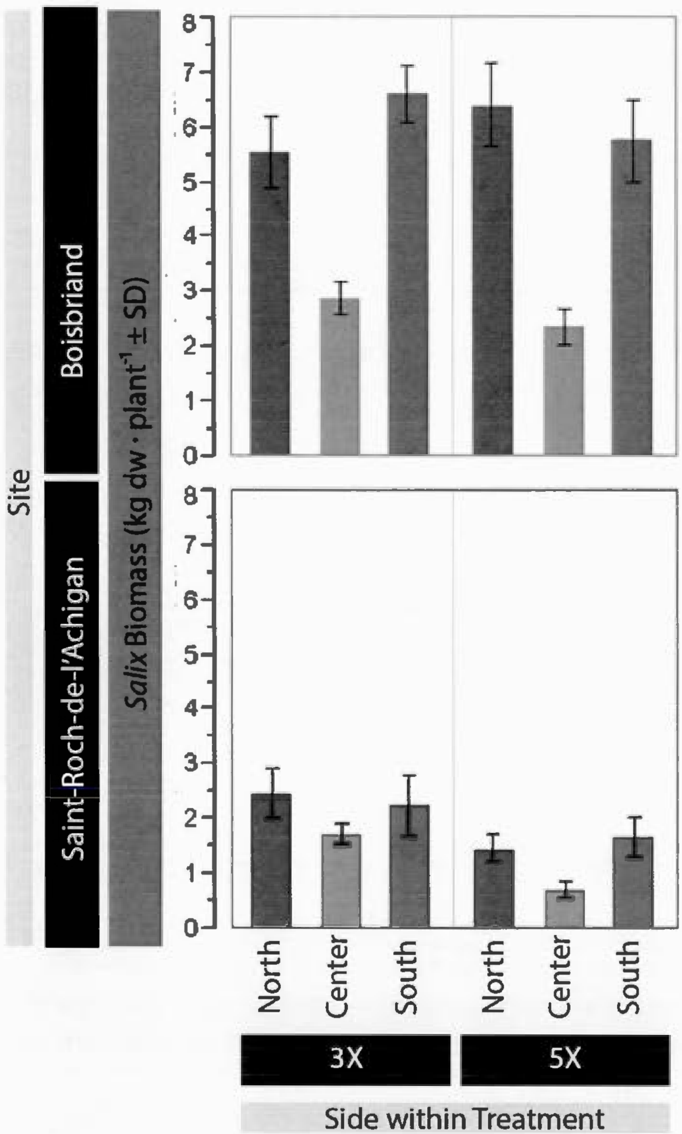
ANNEXE 14:
COMPARISON BETWEEN STANDING BIOMASS AND MEAN NUTRIENT POTENTIAL REDUCTION EFFICIENCY FROM 2011 TO 2013.

Standing biomass is composed of the herbaceous layer (prior to annual mowing) and the *Salix* biomass (prior to the first harvest of a three year growth cycle). Under *Salix*, the herbaceous biomass is negligible (appearing only as a minute line at the bottom of the bars). Individual *Salix* biomass is also presented for comparison. The herbaceous plants biomass in the CX RBS plots is ~230-260 times more elevated than the herbaceous biomass under willows in BB (ANOVA, $p = 0.0001^*$), whereas it is only ~ 2-4 times more elevated in SR (ANOVA, $p = 0.0008^*$). The herbaceous RBS standing biomass was a minor fraction of that in willow RBS (BB: 3X = 8% and 5X = 5%; SR: 3X-5X ≥7%). The herbaceous plants bioarea is significantly different across treatments (BB: $p = 0.0014^*$; SR: $p = 0.0086^*$) and bioarea is not equally deployed on the edge-of-field, center of RBS and closer to the river (BB: $p < 0.0001^*$; SR: 0.0266^*). Herbaceous plant bioarea was significantly correlated to *Salix* height (Chapter 1).The buffer strip potential efficiency (Chapter 2) was added to facilitate comparison. It is represented by global nutrient removal for all sampling events from 2012-2013 spanning the full growth period of the *Salix* prior to harvest.



ANNEXE 15:
CARDINAL ORIENTATION OF WILLOW ROWS WITHIN LOW AND HIGH DENSITY BUFFER STRIPS TO STUDY THE EFFECT OF FUN EXPOSURE ON INDIVIDUAL PLANT BIOMASS YIELD.

The southern-most exposed plants should have been taller if sun exposure (vs shade) was a driving factor of biomass production.



ANNEXE 16:
PRINCIPAL COMPONENTS OF VARIOUS ENVIRONMENTAL MATRICES USED IN THE RDA ON WILLOW GROWTH AND PRODUCTIVITY.

Matrix	Growth (BB+SR; 2011-2013)			Productivity BB (2013)			Productivity SR (2013)		
	% of variance explained by PC1 axis	Sig	Main vectors	% of variance explained by PC1 axis	Sig	Main vectors	% of variance explained by PC1 axis	Sig	Main vectors
Climate	70.8	<0.0001*	1- T mean 2- Sum DJC	100	<0.0001*	All equal	nd	nd	nd
GPS	60	<0.0001*	1=2=3- x-y-z	41.6	<0.0001*	1- y 2- x	35.1	0.0006*	1- z 2- North (= South)
Culture	53.9	<0.0001*	1- Dose N 2- Dose K	100	<0.0001*	All equal	nd	nd	nd
Water	32	<0.0001*	1- PO_4d 2- NH_4	37.3	0.0111*	1- K 2- NH_4	34.6	<0.0001*	1- Mg 2- Ca
Soil	69.1	<0.0001*	1- Soil moisture % 2- Soil OM %	57	<0.0001*	1- Soil OM % 2- Carbonates%	43.4	0.0434*	1- Carbonates % 2- Soil OM %
Hydrology	51.7	<0.0001*	1- nc-m 2- head ncr-m	35.9	<0.0001*	1- d_ncr-m 2- nc-m (= ncr-m)	48.4	<0.0001*	1- d_ncr-m 2- Bassin-m^2
Herbs	39	<0.0001*	1- Weed (Sum cover) 2- Shannon H'	48.6	<0.0001*	1- Herbs (Sum cover) 2- Weed (Sum cover)	34.5	<0.0001*	1- Shannon H' 2- Herbs diversity (n)

ANNEXE 17:
SELECTED INDIVIDUAL ENVIRONMENTAL VARIABLES USED AS INPUTS IN THE WILLOW GROWTH RDA (BOTH SITES TOGETHER; 2011-2013)

The short list of variables chosen amongst those most highly correlated (r > 0.50) with willow growth variables. As the total number of samples for the RDA could be augmented (without over-parameterization; n samples for RDA = 107), this selection of correlations (n = 14) was complemented with 26 significant variables from a stepwise (AICc minimum) multiple regressions with each Salix growth variable were also included (4 variables were common to both selection processes). Variables in bold were forwardly selected in the RDA.

Individual variable	Short title	r	p	Stat	Salix (SM) variable
Distance of phreatic water from soil surface (m)	d_nc-m	-0.27	0.0002	S	stem-n
Annual field GLYP applications (kg-ha)	GLYP-µg/l	0.64		C	height-cm
Water table height difference from field to stream side (m)	head_ncr-m	-0.52		C	height-cm
Soil carbonates (%)	Carbonat-%	0.35	0.0051	S	height-cm
Soil organic matter (%)	**OM-%**		**<0,0001**	S	diam-mm
"		0.60		C	diam-mm
Soil moisture (%)	soil_moist-%	0.58		C	height-cm
Runoff AMPA concentration (µg-ml)	AMPA-µg/l	0.32	0.0002	S	height-cm
Drainage bassin - calculated for each runoff sampling equipment (m^2)	Stream-m2	-0.15	0.0058	S	stem-n
Annual & Biennial plants - total for each transect (Sum %)	AnBi-%	0.19	<0,0001	S	height-cm
Perennial plants - total for each transect (Sum %)	Pere-%	-0.18	<0,0001	S	height-cm
Runoff DOC concentration (µg-ml)	DOC-µg/l	0.40	<0,0001	S	height-cm
Runoff GLYP concentration (µg-ml)	GLYP-µg/l	0.07	0.0270	S	diam-mm
Bare soil cover - proportion (%)	Bare-%	0.27	0.0016	S	height-cm
Runoff Mn concentration (Mg-ml)	Mn-µg-ml	0.37	0.0233	S	diam-mm
Total plants diversity - including Salix (n species)	Tot_Div-n	0.11	0.0167	S	height-cm
Runoff NH_4d concentration (mg-ml)	NH_4-µg-ml	0.41	0.0002	S	height-cm
"		0.60		C	diam-mm
Slope	Slope-deg	0.12	0.0033	S	height-cm
Runoff pH	pH	-0.03	0.0081	S	diam-mm
Drainage bassin - calculated from runoff reaching the stream (m^2)	**Stream-m^2**	-0.47	**<0,0001**	S	diam-mm
"		-0.52		C	height-cm
Total plants soil cover - including Salix (Sum %)	Tot_Cover-%	-0.22	0.0001	S	stem-n
Herbaceous plants bioarea (height x % cover)	Bioarea	-0.01	<0,0001	S	height-cm
Shade tolerance herbaceous plants soil cover - proportion (%)	Stol-%	-0.15	0.0246	S	diam-mm
Runoff Zn concentration (µg-ml)	Zn-µg-ml	0.05	0.0199	S	height-cm
Phreatic water altitude (m)	nc-m	-0.61		C	height-cm
Northward orientation	North	0.09	<0,0001	S	diam-mm
Sum Degree days of growth per year (DJC)	**DJC**	-0.44	**<0,0001**	S	diam-mm
Herbaceous plants soil cover - total for each transect (Sum %)	Cover-%	-0.11	<0,0001	S	height-cm
Sum Precipitations pear year (mm)	**Pcpt-mm**	**-0.67**		C	height-cm
Sum Solar radiations (watt/m^2)	SRAD	0.44	0.0020	S	stem-n
Southward orientation	South	0.10	<0,0001	S	diam-mm
Annual field Ca applications (kg-ha)	**Ca-kg-ha**	**-0.08**	**<0,0001**	S	height-cm
"		**0.60**		C	stem-n
Annual field Mg applications (kg-ha)	**Mg-kg-ha**	**0.60**		C	stem-n
Annual field P applications (kg-ha)	P-kg-ha	0.50		C	height-cm
Longitude NAD83 MTM8 (m)	x	-0.60		C	height-cm
Latitude NAD83 MTM8 (m)	y	-0.60		C	height-cm
Elevation NAD83 MTM8 (m)	x	-0.60		C	height-cm

ANNEXE 18:
SELECTION OF INDIVIDUAL ENVIRONMENTAL VARIABLES USED AS INPUTS IN THE *SALIX* PRODUCTIVITY RDA (PER SITE; 2013 ONLY).

The short list (BB = 16; SR = 15) of variables chosen amongst those most highly correlated (r > 0.50) with Salix 2013 productivity variables. No further variables were selected to avoid over-parameterization (n samples for RDA = 18 per site). Variables in bold were forwardly selected in the RDA.

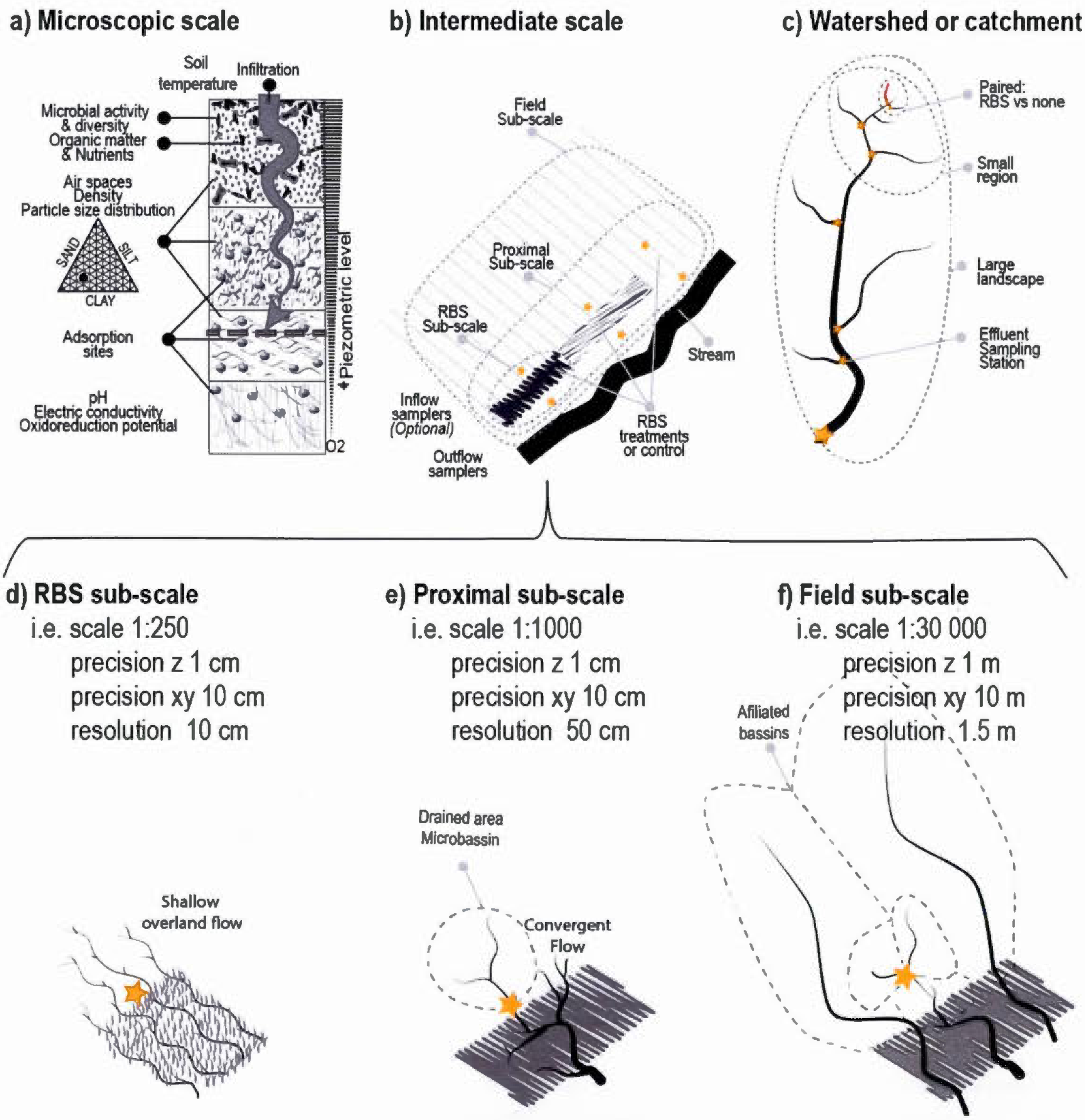
Site	Individual Variable	Short title	r	Salix variable
BB	Soil carbonates (%)	Carbonat-%	0.50	height-cm
	Soil organic matter (%)	OM-%	-0.58	height-cm
	Runoff Al concentration (µg-ml)	**Al-µg-ml**	**-0.75**	height-cm
	Herbaceous plants soil cover - proportion per transect (%)	CoverAll-%	0.56	stem-n
	Weed diversity (n species)	Weed-n	0.51	stem-n
	Hydrophytes diversity (n species)	Hydro-n	0.51	stem-n
	Introduced species diversity (n species)	**Intro-n**	**0.58**	stem-n
	Runoff pH	pH	0.58	height-cm
	Shannon diversity (H')	**Shannon**	**0.57**	stem-n
	Herbaceous plants diversity (n species)	Div-n	0.56	stem-n
	Weed soil cover (Sum %)	**Weed-S%**	**0.54**	height-cm
	Herbaceous plants bioarea (height x % cover)	Bioarea	0.52	height-cm
	Phreatic water altitude (m)	nc-m	-0.58	height-cm
	Herbaceous plants soil cover - total for each transect (Sum %)	**Cover-%**	**0.69**	height-cm
	Longitude NAD83 MTM8 (m)	x	-0.61	height-cm
	Latitude NAD83 MTM8 (m)	y	-0.65	height-cm
SR	Distance of phreatic water from soil surface (m)	d_nc-m	0.67	height-cm
	Soil moisture (%)	**soil_H_20-%**	**-0.64**	height-cm
	Runoff AMPA concentration (µg-ml)	AMPA-µg/l	0.60	kg-plant^{-1}
	Perennial plants - total for each transect (Sum %)	Pere-%	0.54	height-cm
	Hydrophytes soil cover - proportion (%)	Hydro-%	0.67	height-cm
	Hydrophytes soil cover - total for each transect (Sum %)	Hydro-S%	0.68	height-cm
	Introduced species soil cover - proportion (%)	**Introduced-%**	**-0.57**	diam-mm
	Upland plants soil cover - proportion (%)	NoHyd-%	-0.67	height-cm
	Runoff pH	pH	0.51	kg-plant^{-1}
	Drainage bassin - calculated from runoff flow reaching the stream (m^2)	**Stream-m^2**	**-0.56**	diam-mm
	Shade tolerance herbaceous plants soil cover - proportion (%)	**Shade_tol-%**	**-0.51**	kg-plant^{-1}
	Runoff Zn concentration (µg-ml)	Zn-µg-ml	0.53	stem-n
	Drainage bassin - calculated from drainage points to stream (m^2)	Aff_Pt_m2	-0.54	diam-mm
	Simpson reciprocal (1/D)	Simpson	-0.55	t-ha^{-1}
	Latitude MTM8 (m)	**y**	**0.67**	stem-n

ANNEXE 19: CROP LOSS, POTENTIAL REVENUE AND MAXIMAL MAINTENANCE COST OF *SALIX* RBS TO ACHIEVE PROFITABILITY BY THE FARMER.

All externalities related to environmental services provided by the RBS were intentionally left out of this gross estimate. The two extreme scenarios are presented (min-max) were prepared by accounting for the best and worst possible yield, revenues and cost of harvest, hence,at maximal cost of harvest the net profit is the lowest. Note: (1) Cost of harvest includes site preparation, transportation, pressing and handling estimated from Vézina et al. (2013), the minimal cost being associated to an RBS and maximal costs associated to larger plantations.

Site		BB			SR		
year		2011	2012	2013	2011	2012	2013
t/ha		3.65	10.5	3.75	2.76	8.6	4.4
Crop		S	M	S	S	M	M
Crop value ($/t)		$410	$195	$410	$410	$195	$195
Net loss ($/ha)		$1,497	$2,048	$1,538	$1,132	$1,677	$858
			2011-2013			2011-2013	
Net loss ($/ha/3 yrs)			$5,082			$3,667	
Net loss ($/ha/yr)			$1,694			$1,222	
Willow value ($/t)	*min*		$80			$80	
	max		$120			$120	
Willow yield (t/ha)	*min*		167			70	
	max		268			71	
Willow yield (t/ha) including 10% net harvest loss	*min*		150			63	
	max		241			64	
Potential revenue ($/ha)	*min*		$12,024			$5,040	
	max		$28,944			$7,668	
Maximal cost to allocate towards plantation, maintenance and harvest to avoid losses	*min*		$6,943			**$1,373**	
	max		$23,863			**$4,001**	
Cost of harvest[1] ($/t)	*min*		$62			$62	
	max		$216			$216	
Cost of harvest[1] ($/ha)	*min*		$10,354			$4,340	
	max		$57,888			$15,336	
Net profit	*min*		$1,670			$700	
	max		$(28,944)			$(7,668)	

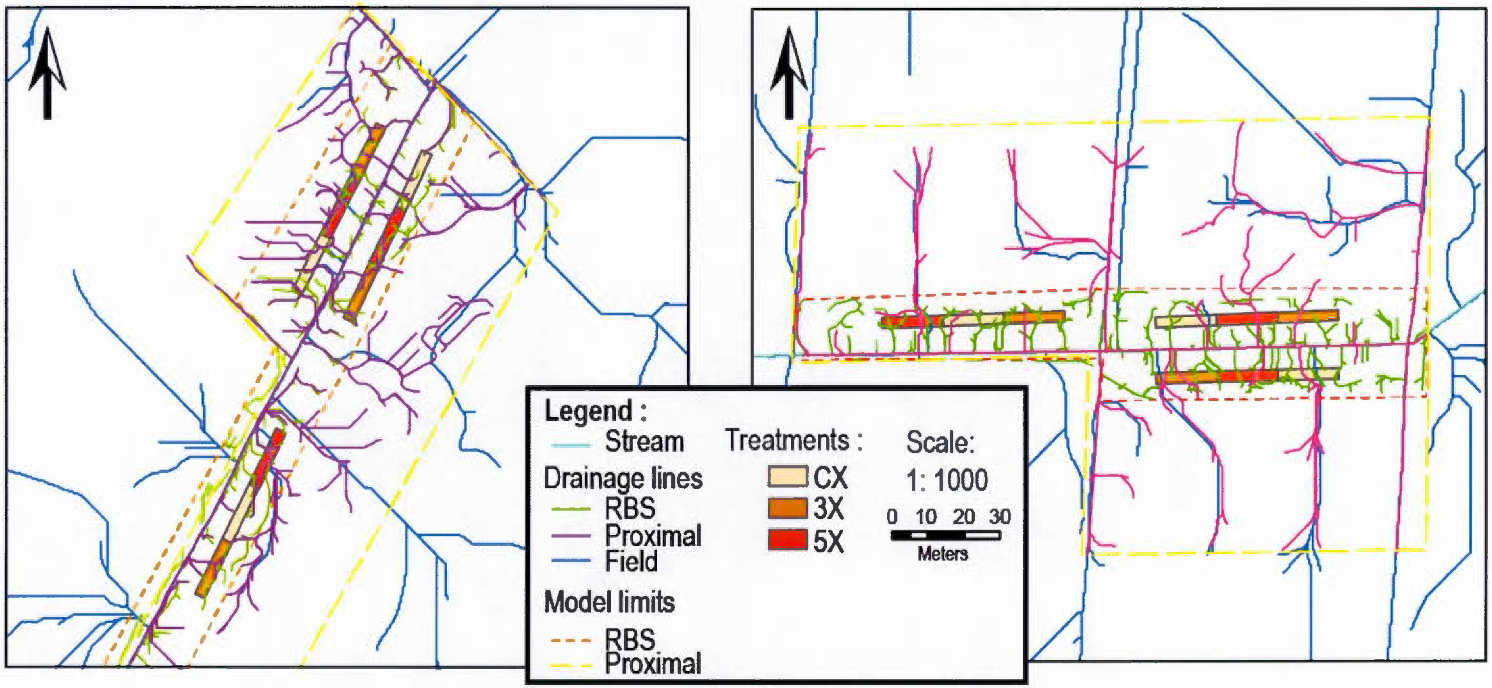
ANNEXE 20:
RBS SCALES AND SUB-SCALES



ANNEXE 21:

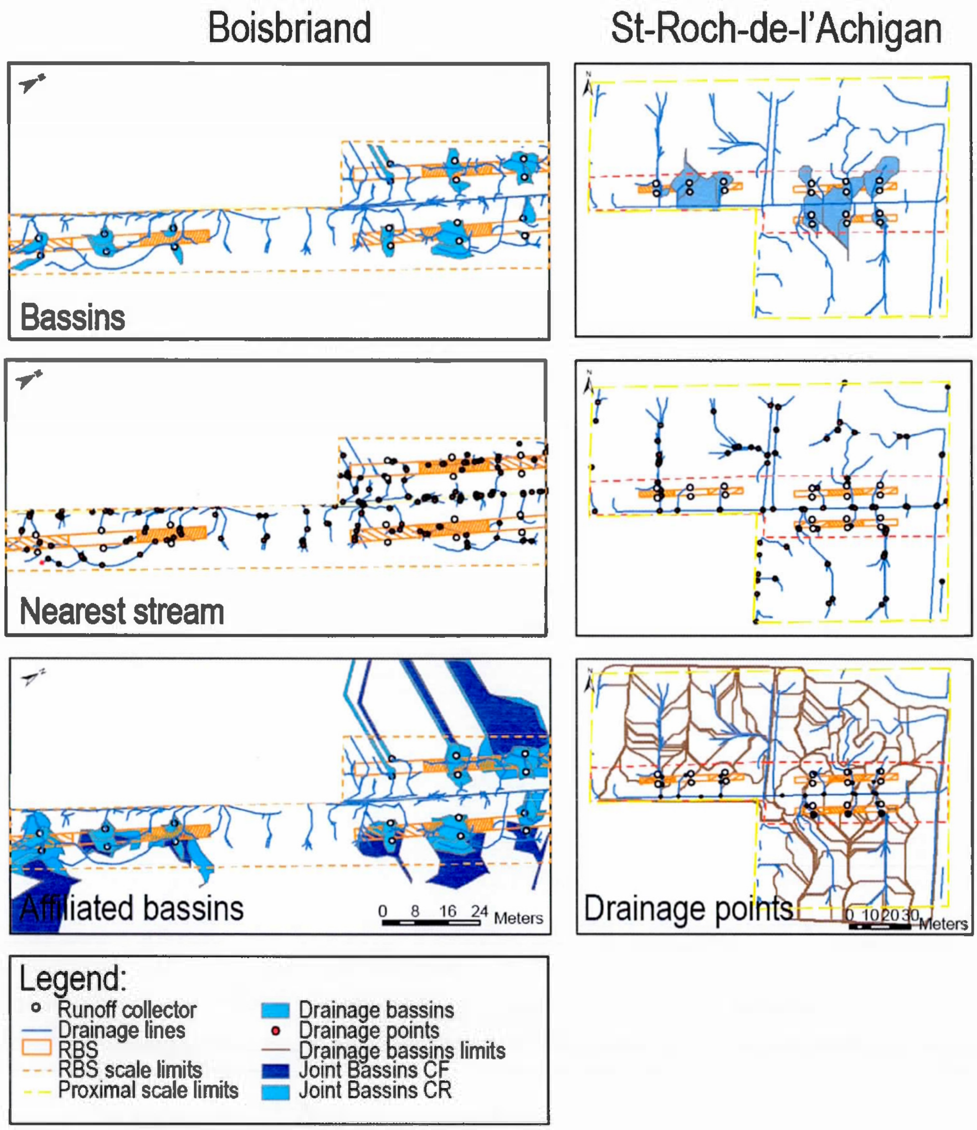
DRAINAGE LINES IN BOISBRIAND (LEFT) AND SAINT-ROCH-DE-L'ACHIGAN (RIGHT) AT THREE DIFFERENT SCALES: RBS (GREEN), PROXIMAL (PURPLE) AND REGIONAL OR FIELD SCALE (BLUE).

Drainage lines nearly overlap at the three scales, though smaller unconnected lines are visible at the RBS scale and only major drainage lines are visible at the field scale. For further analysis, only the proximal scale is used.



ANNEXE 22:
DRAINAGE BASIN SURFACE AREA MODELS SCHEMATIC REPRESENTATIONS.

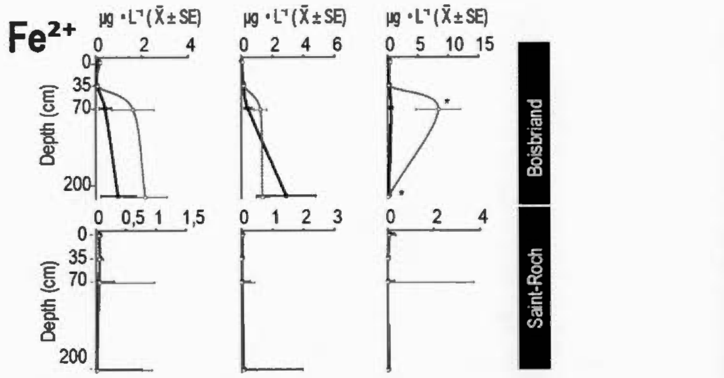
(1) Basins (Catchment) in light blue; (2) Nearest stream (drainage points is black dot placed on closest drainage line); (3) Affiliated basins (BB Only, CF in dark blue and CR in light blue) and (4) Drainage points to nearest rock chute (SR Only, small black dots). Figures are presented side by side to avoid overcrowding of information.

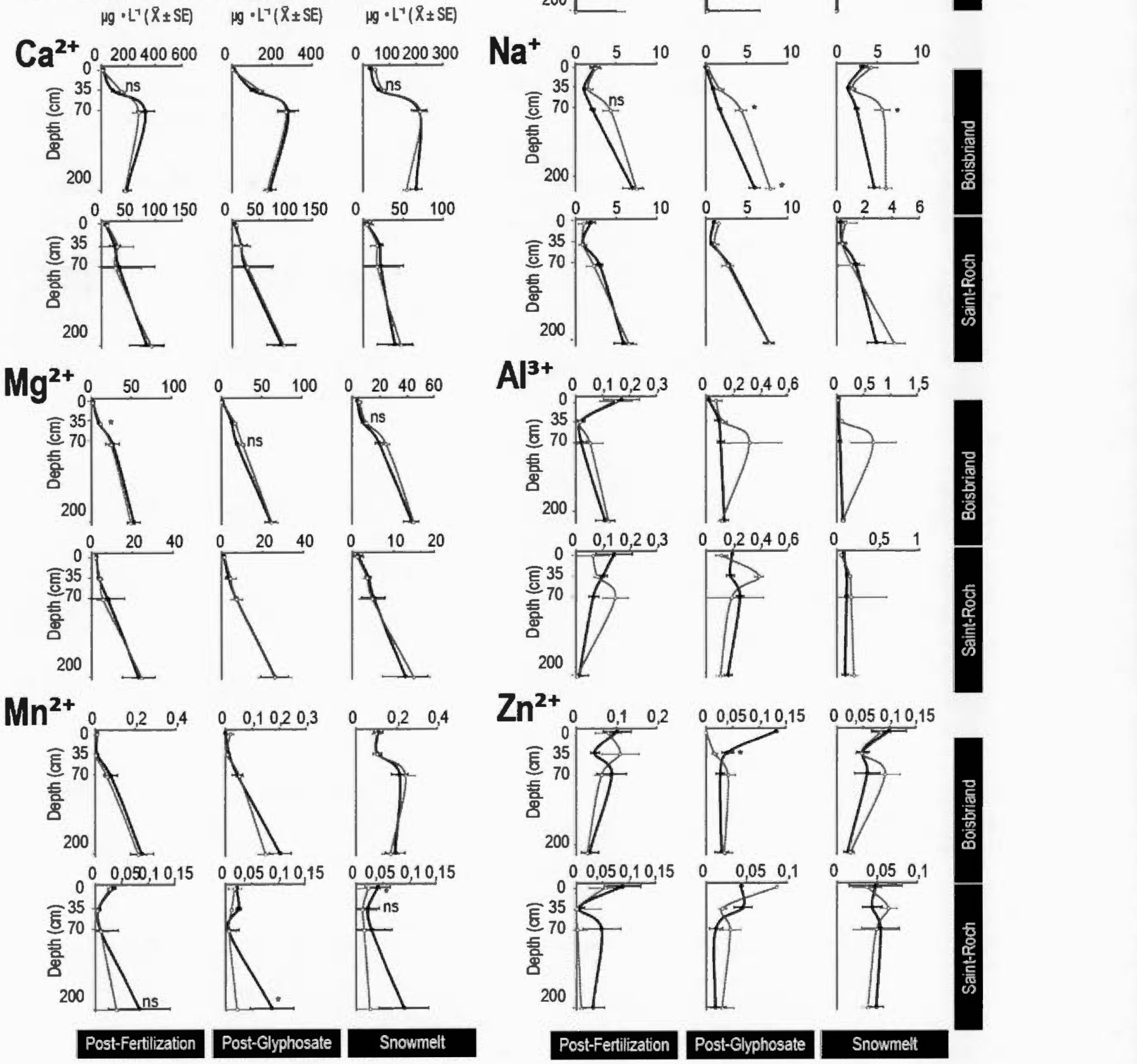


ANNEXE 23:

OTHER NUTRIENT CONCENTRATIONS IN BOISBRIAND AND SAINT-ROCH-DE-L'ACHIGAN POOLED BY AGRICULTURAL EVENT PRESENTED AS DEPTH PROFILES BEFORE (CF) AND AFTER (CR) THE BUFFER STRIP.

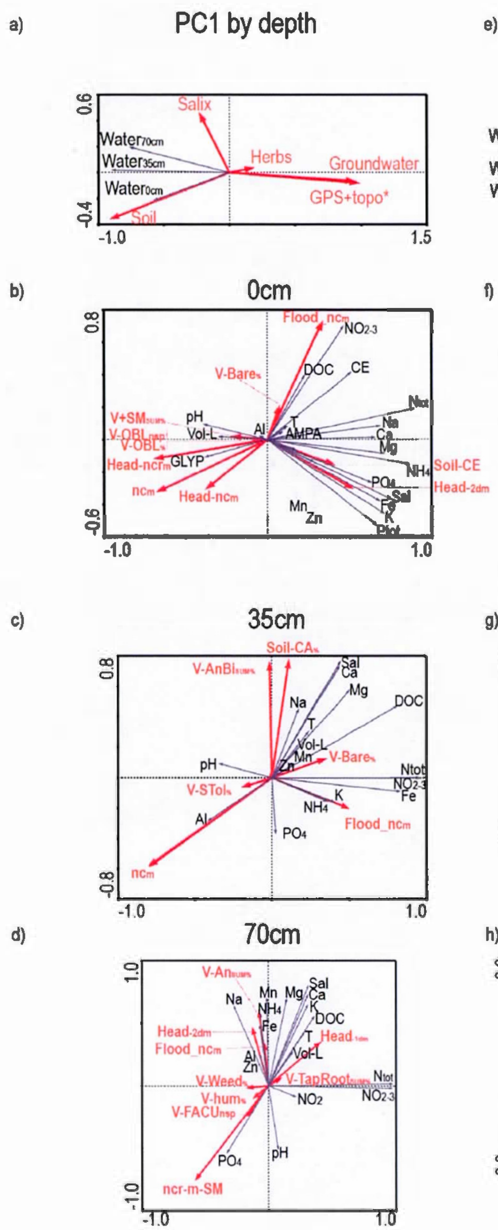
CF is represented as black dots with a solid line while CR has opened circles with a dotted line. The difference between treatments was not presented here as statistical significance was only exhibited in elements with minute concentrations.





ANNEXE 24:
THE INFLUENCE OF HERBACEOUS VEGETATION ON NUTRIENT CONCENTRATIONS MEASURED IN WATER IN THE RBS AT DIFFERENT DEPTHS AND SAMPLING PERIODS.

The influence of environmental groups of similar nature parameters (a, e) or individual variables (b-d; f-h) on aqueous nutrient concentrations measured at different depth (a-d) or different sampling periods (e-h). Legend and statistical method on next page.



Annexe 24: (continued)

Legend: Contrary to individual variables, grouped variables observed at different water sampling depths or periods, showed marginal influence on the aqueous nutrient concentrations in the RBS (a,e). Obligate hydrophytes (V-OBL) considered as richness (nsp) or land cover percentage (%) are negatively correlated with N_{tot} and PO_4^{3-} in runoff (0cm) (b). The total land covered by annuals and biennials (V-AnBiSUM%) or annuals alone (V-AnSUM%) is related to several cations concentrations and perhaps diametrically opposed to PO_4^{3-} concentrations in interstitial waters below the RBS (c,d). Herbaceous vegetation with tap roots (V-TapRoot$_{SUM\%}$) may play a marginal role on N_{tot} or NO_2^-+NO_3^- concentrations measured at 70 cm. Vegetation with tap roots were again selected when individual parameters were analyzed by sampling time. It grouped close to the total volume of water collected at Snowmelt, along with the bare ground cover (V-Bare$_{\%}$) (f) and probably influenced NO_2^-+NO_3^- concentrations measured post-fertilization and was diametrically opposed to non-hydrophilic herbaceous vegetation ground cover (V-Non-Hydro$_{SUM\%}$) (g). On the other hand, the influence of tap roots on nitrogen infiltration (d) could have been regarded as marginal (due to length of vector) if tap roots (and bare ground cover) had not appeared once again as a potentially determinant factor of water collected at various depth during snowmelt (f), and potentially playing a role on NO_2^-+NO_3^- concentrations measured post-fertilization.

Abbreviations: Richness (nsp), land cover proportion (%), total land cover (SUM%), obligate hydrophytes (V-OBL), facultative hydrophytes (V-FACU), annuals and biennials (V-AnBiSUM%), annuals (V-AnSUM%), Non-hydrophytic herbaceous vegetation (V-Non-Hydro), Herbaceous vegetation with tap roots (V-TapRoot$_{SUM\%}$), bare ground cover (V-Bare$_{\%}$), non-hydrophilic herbaceous vegetation ground cover (V-Non-Hydro$_{SUM\%}$), Weedy herbaceous vegetation (V-weed), herbaceous vegetation biomass productivity (V-mass$_{Kg/ha}$), Herbaceous vegetation which tolerate shade (V-ShadeTol$_{SUM\%}$), shade intolerant plants (V-ShadeInt$_{SUM\%}$), Herbaceous vegetation and *Salix miyabeana* SX64 (V+SM), *Salix* productivity (SM$_{t/ha}$), *Salix* individual plant biomass (SM$_{kg/plant}$), *Salix* stem diameter (SM-dia$_{mm}$), Hydraulic gradient between edge-of-field and edge-of-stream under the connectivity model (Head-ncr), Depth of water table in meters under the non-connectivity model (nc$_m$), Water sampling equipment likely in contact with phreatic water (Flood), Surface area of the microbasin (Bassin), longitude and latitude coordinates (xy), Proportion of carbonates in the soil (Soil-CA%), Electrical conductivity of soil (Soil-CE).

Source data and statistical procesure: The herbaceous vegetation and *Salix* variables are taken from Chapter 1, and the Groundwater and GPS+Topo data were taken from Annexe 4. To understand the influence of various environmental parameters on nutrients potential reduction efficiency within the RBS, two approaches were used. First, a forward selection RDA (500 Monte-Carlo permutation, maximum 10 parameters) was used to assess the influence of individual parameters on nutrient behavior within the RBS using a Redundancy Analysis (RDA) (Legendre and Legendre 2012). Water sampling depth (0, 35, 70 cm) and period (snow melt, post-fertilization and post-glyphosate) were analyzed in different RDAs. Secondly, the individual variables were grouped into categories of similar nature (*Salix*, Soil, Herbs, Groundwater and GPS+Topo;), to understand their joint effect. The first principal component (PC1) of these 5 groups of environmental factors were used to interpret nutrient behavior in the RBS using a RDA. PC1 analyses were conducted with JMP 10 and RDA with CANOCO v4.0 (Lepš and Šmilauer 2003).

Annexe 24: (continued)

Summary of findings and explanations: The redundancy analysis (Legendre and Legendre 2012, Lepš and Šmilauer 2003) revealed that several individual herbaceous characteristics were correlated to nutrients concentrations.

A- **Hydrophytes abundance was correlated with lower N_{tot} and PO_4^{3-} in runoff**. The inverse correlation between obligate hydrophytes and N_{tot} and PO_4^{3-} in runoff (0cm) (b) may simply be due to the fact that BB has richer leachate and more accessible moisture, as obligate hydrophytes were only observed on this site (Annexe 10). Alternately, *Phalaris arundinacea* which covers approximately 8% of the surveyed transects in BB (Annexe 7) is known to have a high tolerance to anaerobiosis (USDA 2014), and could tolerate conditions favorable to denitrification (Vidon and Hill 2004). Hence, plant diversity may bear witness to plausible denitrification in BB as a likely explanation for the observed nitrate reduction observed herein.

B- **Herbaceous vegetation with tap roots correlated with infiltration of dissolved nitrogen**, which is consistent with the enhanced infiltration induced by plants with tap roots (Reubens et al. 2007). Herbaceous vegetation with tap roots correlated with infiltration of dissolved nitrogen (Annexe 24), which is consistent with the enhanced infiltration induced by plants with tap roots (Reubens et al. 2007). Indeed, considering that willows have a fibrous system, there were significantly more plants with tap roots (proportional coverage) in the herbaceous parcels (BB: $p < 0.0001$*; SR: 0.0106*), and these changes were gradual from one treatment to another, with some influence of side in BB ($p = 0.0049$*; Annexe 13).

Contrary to the results obtained from herbaceous vegetation variables taken separately, grouping them (using their first principal component), led to no clear evidence of their influence on nutrient concentrations, at any season and depth surveyed. This suggests that some of the herbaceous variables surveyed influenced nutrient concentrations differently. Together with the idea that the gradients in ecological characteristics of the herbaceous vegetation were observed from the free-growing herbaceous parcels to the highest-density *Salix* plantations (Chapter 1), this may mask the potential efficiency distinctions between the different treatments, perhaps via compensating mechanisms (i.e. sometimes the herbs plot display more characteristics considered beneficial for agro-chemicals mitigation, sometimes the *Salix* plots do, in a continuum). It is known that RBS over story influences the herb-layer, through altering light availability and soil fertility, and in turn, the low strata influences the woody species through intensive competition and pre-emption of resources such as nutrients (Gilliam 2007).

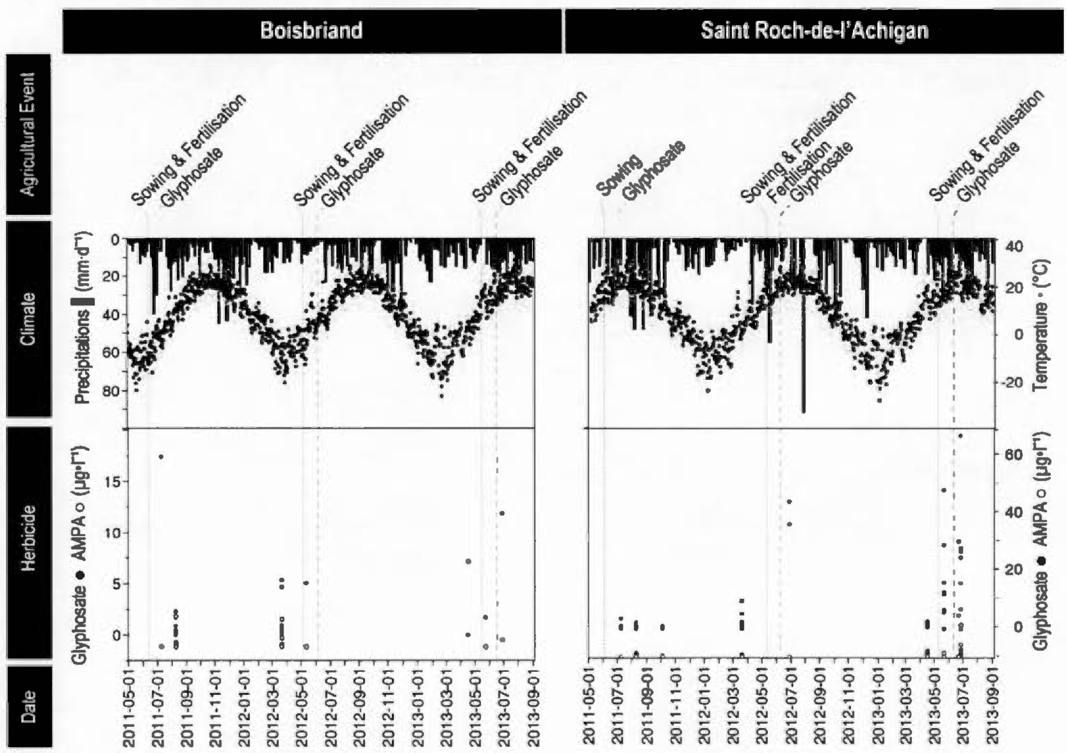
ANNEXE 25: ENVIRONMENTAL PARAMETERS INFLUENCING AQUEOUS NUTRIENTS CONCENTRATIONS AROUND THE RBS FROM 2011 TO 2013.

Multiple correlations exploring relationships between nutrient aqueous concentrations near the buffer strip at different sampling depth (0, 35, 70 cm) and periods (SM: snowmelt; PF: post-fertilization; and PG: post-glyphosate). The principal component (PC1) of five groups of environmental parameters is given. Numbers given are the pairwise correlations between each groups of variables and values in bold were significant ($p < 0.05$).

	Analysis:	**By depth**				**By Sampling period**			
	Group:	0 cm	35 cm	70 cm	ALL	PF	PG	SM	ALL
Group of parameters ↓	*PC1 1° Parameters* ↓								
Vegetation	*1) Shannon H'* *2) Weed diversity*	-0.1814	-0.1594	-0.1904	-0.2503	-0.1762	-0.1923	-0.0414	-0.1605
Salix	*1) Salix height* *2) Salix diameter*	0.8279	0.2861	**0.3737**	0.1631	0.2917	0.2533	0.3240	0.2948
Hydrogeology	*1) Water table head* *2) Water table z elevation*	**-0.5422**	**-0.8535**	**-0.7348**	**-0.7966**	**-0.8260**	**-0.7130**	**-0.7930**	**-0.8285**
Soil	*1) Soil moisture (%)* *2) Organic matter content (%)*	**0.5941**	**0.9049**	**0.8806**	**0.8840**	**0.9153**	**0.8132**	**0.8917**	**0.9367**
GPS + Topo	*1) x-y-z coordinates*	**-0.5308**	**-0.8620**	**-0.7416**	**-0.8130**	**-0.8440**	**-0.7690**	**-0.8421**	**-0.8668**

ANNEXE 26:
TEMPERATURE AND PRECIPITATION FLUCTUATIONS FROM 2011 TO 2013.

The events which characterize the snowmelt, post-glyphosate and post-fertilization campaigns are market, along with the glyphosate and AMPA concentrations recorded on the edge-of-field at each sampling event.

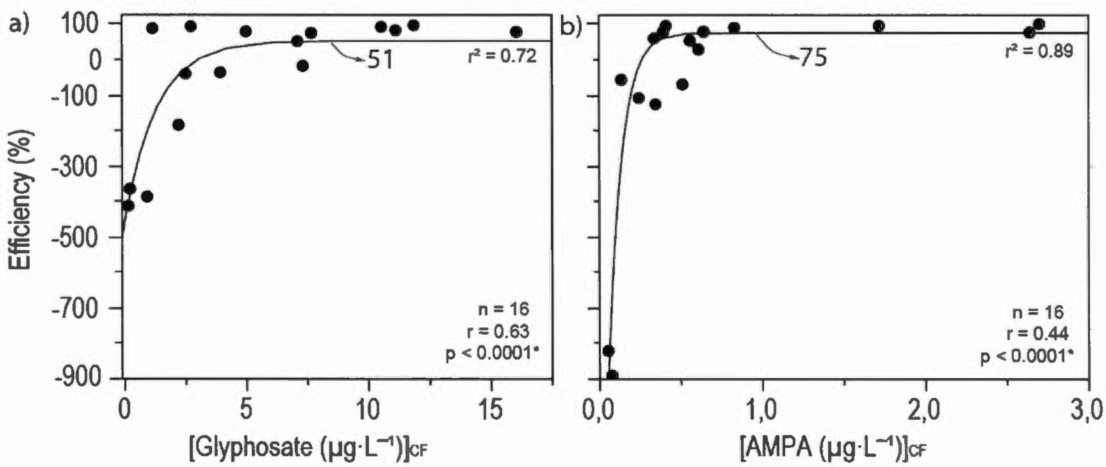


ANNEXE 27:
GLYPHOSATE AND AMPA POTENTIAL REDUCTION EFFICIENCY AS A FUNCTION OF EDGE-OF-FIELD CONCENTRATIONS.

To assess if the potential efficiency of the RBS was dependent on [glyphosate]$_{aq}$ and [AMPA]$_{aq}$ at the edge-of field, we analyzed the spatial organization of both factors using a three parameter non-linear regression, where θ represent parameters statistically optimized to fit the model, and [P] represent the pesticide glyphosate or its residue AMPA (Eq. 2).

$$\text{Eq.2: } Efficiency\ (\%) = \theta_1 \times (1 - \theta_2 \times Exp\ (-\theta_3 \times [P]_{CF}))$$

The value of the plateau is indicated with an arrow and the r^2 corresponds to the fit of the non-linear regression, while r corresponds to the correlation between both variables.

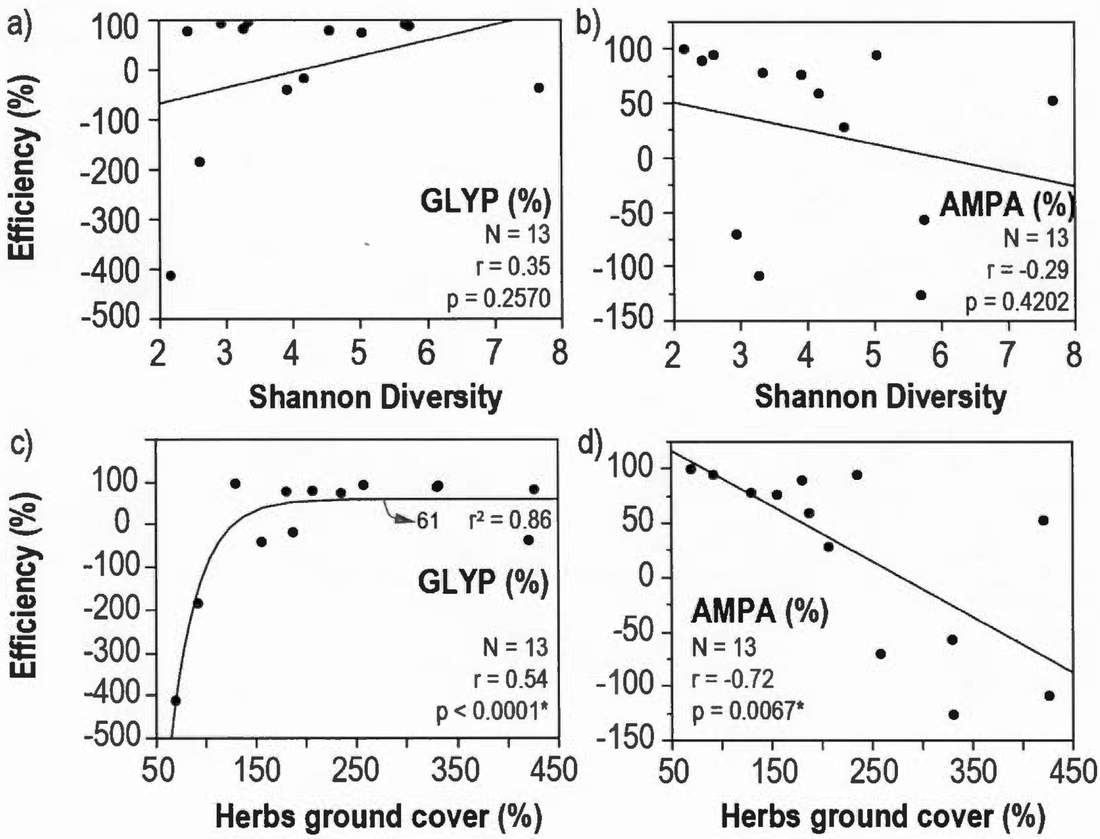


ANNEXE 28:
GLYPHOSATE AND AMPA POTENTIAL REDUCTION EFFICIENCY AS A FUNCTION OF SHANNON DIVERSITY AND HERBACEOUS VEGETATION GROUND COVER.

Shannon diversity is only weakly positively correlated with glyphosate potential reduction efficiency (a) and weakly negatively correlated with AMPA potential reduction efficiency (b). Glyphosate and AMPA were not put on the same scale to allow better visualization of the independent correlations. The herbaceous vegetation variables were taken from Chapter 1. The spatial organization of one relationship incorrectly displayed by a linear regression was analyzed fitted to a three parameter non-linear regression (Eq. 2).

Eq.2: $Predicted\ Efficiency\ (\%) = \theta_1 \times (1 - \theta_2 \times Exp\ (-\theta_3 \times [P]_{CF}))$

Reported r values represent the coefficient of correlation, p values represent the probability that the regression is significant and the r^2 is given where Eq.2 was used to show how closely the model fitted the data set.



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