Identification of criteria to determine the restoration potential of riparian wetlands in highly degraded agricultural environments

-

Final Report

Report produced as part of the project “Identification de critères pour déterminer le potentiel de restauration des milieux humides riverains en milieux agricoles fortement dégradés ” funded by the Fonds National de Conservation des Milieux Humides (FNCMH), Environnement et changements climatiques Canada

William Massey[1], Jean-Philippe Marchand[1], Pascale Biron[1], Thomas Buffin-Bélanger[2], Marie Laroque[3]

1. Department of Geography, Planning and Environment, Concordia University;
2. Département de biologie, chimie et géographie, Université du Québec à Rimouski;
3. Département des sciences de la terre et de l’atmosphère, Université du Québec à Montréal

April 2019

Analysis and writing:
William Massey, M.Sc Environmental studies
Research professional

Jean-Philippe Marchand, M.Sc
Ph.D candidate

Preliminary analysis of climate change scenarios:
Sylvain Gagné, M.Sc.
Research professional

Vegetation assessment/characterization at study sites:
Audréanne Loiselle, M.Sc.
Ph.D candidate at L'institut de Recherche en Biologie Végétale

Researchers:
Thomas Buffin-Bélanger, Ph.D
Professor

Marie Laroque, Ph.D
Professor

Pascale Biron, Ph.D (PI)
Professor

Financial support:

Environment and Climate Change Canada: Fonds National de Conservation des Milieux Humides (FNCMH)
Concordia University
Université du Québec à Rimouski (UQAR)
Université du Québec à Montreal (UQAM)
Institut de Recherche et de Développement en Agroevironnement (IRDA)
Ouranos – Consortium on Regional Climatology and adaptation to Climate Change
Pleine-Terre

For citation:

Massey, W., Marchand, JP., Biron, P. M., Buffin-Bélanger, T., Laroque, M. (2019). Identification de critères pour déterminer le potentiel de restauration des milieux humides riverains en milieux agricoles fortement dégradés. Report presented to the Fonds National de Conservation des Milieux Humides (FNCMH)., Concordia University. 131 pages.

ACKNOWLEDGEMENTS

We would like to thank Environment and Climate Change Canada who partially funded this project on developing criteria to evaluate the restoration potential of riparian wetlands in degraded agricultural landscapes. We would also like to thank all collaborators for their interest, support and feedback on this important subject as well as the various field assistants who worked hard to make the data collection possible.

SUMMARY

Riparian wetlands are closely linked to meander dynamics, with channel migration leading to floodplain development including microtopographies and to meander cut-offs and the formation of oxbow lakes. The abandoned meanders are frequently flooded zones and provide key long-term ecosystem services. Unfortunately, in agricultural watersheds, channelization and backfilling of topographical troughs have resulted in the removal of riparian wetlands as well as of their morphogenetic processes. Many restoration projects fail to meet target objectives as they typically focus on rehabilitating specific wetland functions, often using hard structures to control water levels, for example, without properly addressing the underlying causes of degradation. The current scientific consensus is that the restoration of natural processes, i.e. channel migration, is needed to design wetlands that are dynamic in both time and space.

A key factor in the restoration of riparian wetland ecosystem functions is the nature of the channel/floodplain hydrological connections. It is hypothesized that ancient meanders can be good restoration targets due to their potentially higher hydrological connections associated with their coarser fluvial sediments. However, more information is needed to better understand their potential to be re-connected in degraded agricultural landscapes. The objective of this study was is to define indicators of areas that have the best wetland restoration potential in agricultural watersheds. This goal was attained by 1) determining the state of knowledge regarding ancient meander restoration worldwide to highlight indicators of project success/failure from the literature, and 2) providing examples of the potential for wetland restoration through re-meandering at three straightened headwater streams in the St. Lawrence Lowlands of Quebec. The three stream reaches (approximately 1 km in length) are located in regions of intensive agriculture (mainly corn and soya) on the Des Fèves River (Branche 53, drainage area 2.41 km2; DF), on the Petite rivière Pot-au-Beurre (near Baie Lavallière, drainage area 8.62 km2; PPauB), and on the Ruisseau Martin (Bulstrode River watershed, drainage area 27 km2; RM). All three reaches are located in the clayey silt geological environment of the Champlain Sea partly eroded by the streams and accompanied by coarser sediments in alluvial plain

At each site, physical characteristics of the channel and ancient meander (channel geometry, meander topography, surface geology, water level at gauging stations, groundwater level at wells/piezometers and temperature) and ecological components (riparian vegetation and in-channel fish habitat) were measured through a combination of quantitative (ex: site instrumentation with loggers) and qualitative (ex: expert interpretation and cameras) measurements. The potential for hydrological connections through surface and/or subsurface processes was evaluated using signal processing techniques to determine the lag times between in-channel peak water levels and groundwater levels in response to important precipitation events. Re-meandering potential was evaluated using the RVR Meander model to simulate channel migration.

Results indicated that all three streams have the potential to re-meander, although bank cohesion of clay particles at PPauB and DF likely limits channel migration at these sites. Clear differences between the sites were found in both the potential for restoration and the lateral hydrological connectivity. At RM, factors such as land use change from agriculture to forest following the initial channelization (latest straightening between 1950-1960), the topography of the ancient meanders which allows surface water to pool, and short lag times between the water level response of the piezometers in the ancient meander depression and in the channel, indicate a great potential for lateral hydrological connectivity and passive restoration of riparian wetlands. Indeed, results of vegetation and fish habitat assessments revealed that RM has the highest proportion of its floodplain classified as wetlands and excellent fish habitat quality in channel sections adjacent to the ancient meanders. Conversely, sites PPauB and DF, where persistent human activities such as dredging (latest straightening in 2013 for the two rivers), removal of riparian vegetation and filling of the ancient meander with thick layers of fine material have lead to the disconnection of the coarse alluvial deposits from the channel, with no evidence of lateral subsurface hydrological connectivity at these sites. Without active restoration to improve ancient meander topography and produce a riparian buffer between the ancient meander and adjacent agricultural fields wetland restoration potential appears to be compromised. The active restoration approach is also likely to be needed to promote channel migration in PPauB and DF because bank cohesion of clay particles limits channel migration at these sites, unlike at RM.

This study suggests that, after many decades without intervention in the stream, in conditions similar to that of the RM site, meandering starts naturally and wetlands can begin to redevelop, ensuring a higher quality of habitats and a higher biodiversity. This potential for natural remediation appears to be limited in reaches such as PPauB and DF where human interventions are still ongoing, where infilling has modified significantly the potential for lateral stream-aquifer hydrological connectivity and where bank cohesion may limit channel migration. Restoration of physical processes in these cases would require riparian landowner participation to limit direct interventions in the stream and riparian zones. This study highlights that it is worth working on the restoration potential of straightened agricultural streams as they can provide important ecosystem services when natural hydrogeological processes are able to operate.

RÉSUMÉ

Les milieux humides riverains sont étroitement liées à la dynamique des méandres, la migration des chenaux conduisant au développement des plaines inondables y compris aux microtopographies, aux avulsions et à la formation de bras morts (“oxbow”). Les méandres abandonnés sont fréquemment inondés et fournissent des services écosystémiques essentiels à long terme. Malheureusement, dans les bassins versants agricoles, la linéarisation et le remblayage des creux topographiques ont entraîné l’élimination des milieux humides riverains ainsi que de leurs processus morphogénétiques. De nombreux projets de restauration n'atteignent pas les objectifs visés car ils se concentrent généralement sur la réhabilitation de fonctions spécifiques des milieux humides, utilisant souvent des seuils pour contrôler les niveaux d'eau, par exemple, sans s'attaquer aux causes sous-jacentes de la dégradation. Le consensus scientifique actuel est que la restauration des processus naturels, c'est-à-dire la migration du chenal, est nécessaire pour concevoir des milieux humides dynamiques dans le temps et dans l'espace.

Un facteur clé dans la restauration des fonctions de l'écosystème des milieux humides riverains est la nature des liens hydrologiques entre le chenal et la plaine inondable. L'hypothèse est que d'anciens méandres peuvent être de bonnes cibles de restauration en raison de leurs connexions hydrologiques potentiellement plus élevées associées à leurs sédiments fluviaux plus grossiers. Cependant, davantage d'informations sont nécessaires pour mieux comprendre leur potentiel de reconnexion dans des contextes agricoles dégradés. L'objectif de cette étude était de définir des indicateurs des zones qui offrent le meilleur potentiel de restauration des zones humides dans les bassins versants agricoles. Cet objectif a été atteint 1) en déterminant d'abord l'état des connaissances sur la restauration des anciens méandres afin de mettre en évidence les indicateurs de succès ou d'échec de projet dans la littérature, et 2) en fournissant des exemples du potentiel de restauration des milieux humides par l'étude détaillée de trois cours d'eau redressés dans les Basses Terres du Saint-Laurent au Québec. Les trois cours d'eau (environ 1 km de long) sont situés dans des régions d'agriculture intensive (principalement du maïs et du soja) sur la rivière Des Fèves (Branche 53, bassin versant 2,41 km2; DF), sur la Petite rivière Pot-au-Beurre (près de la baie Lavallière, aire de drainage de 8,62 km2; PPauB), et sur le ruisseau Martin (bassin versant de la rivière Bulstrode, aire de drainage de 27 km2; RM). Les trois tronçons sont situés dans l'environnement géologique limoneux argileux de la mer de Champlain partiellement érodé par les ruisseaux et accompagné de sédiments plus grossiers dans la plaine alluviale.

À chaque site, les caractéristiques physiques du chenal et de l'ancien méandre (géométrie du chenal, topographie du méandre, géologie de surface, niveau d'eau aux stations de jaugeage ainsi que dans les puits / piézomètres, température) et composantes écologiques (végétation riveraine et poissons), ont été mesurés par une combinaison de mesures quantitatives (ex: sondes à pression) et qualitatives (ex: interprétation par des experts et caméras). Le potentiel de connexions hydrologiques par des processus de surface et / ou souterrains a été évalué en utilisant des techniques de traitement du signal afin de déterminer les délais entre les niveaux d'eau maximaux dans le chenal et dans les piézomètres en réponse à des précipitations importantes. Le potentiel de re-méandrisation a été évalué à l'aide du modèle RVR Meander afin de simuler la migration du chenal.

Les résultats indiquent que les trois cours d'eau peuvent potentiellement re-méandre, bien que la cohésion des berges constituées d'argile à PPauB et à DF limite probablement la migration du chenal à ces sites. De nettes différences entre les sites ont été trouvées à la fois dans le potentiel de restauration et dans la connectivité hydrologique latérale. Au RM, des facteurs tels que le changement d'utilisation du sol de l'agriculture vers la forêt suite au redressement initial (dernier redressement entre 1950-1960) et la topographie des anciens méandres qui permettent à l'eau de surface de se regrouper et des temps de latence courts entre la réponse des piézomètres dans l'ancienne dépression de méandre et dans le chenal résultent en un potentiel important de connectivité hydrologique latérale et de restauration passive des milieux humides riverains. En effet, les résultats des évaluations de la végétation et d'habitat du poisson ont révélé que RM avait la plus forte proportion de sa plaine inondable classée comme milieux humides et une excellente qualité d'habitat du poisson dans les sections de chenal adjacentes aux anciens méandres. Inversement, les sites PPauB et DF, où des activités humaines persistantes telles que le dragage (dernier redressement en 2013 pour les deux cours d'eau), l'enlèvement de la végétation riveraine et le remplissage de l’ancien méandre avec d'épaisses couches de matériaux fins, ont conduit à la déconnexion des dépôts alluviaux grossiers du chenal, sans signe de connectivité hydrologique latérale sur ces sites. Sans restauration active pour améliorer la topographie des anciens méandres et la largeur de la bande riveraine entre l'ancien méandre et les champs agricoles adjacents, le potentiel de restauration de ces milieux humides riverains apparaît compromis. L'approche de restauration active sera probablement également nécessaire pour promouvoir la migration du chenal aux sites PPauB et DF car la cohésion des particules d'argile sur la rive limite la migration du chenal à ces sites, contrairement au RM.

Cette étude suggère qu'après des décennies sans intervention dans le cours d'eau, dans des conditions similaires à celles du site du RM, des méandres commencent naturellement à se former et des milieux humides riverains peuvent se redévelopper, garantissant ainsi une meilleure qualité des habitats et une plus grande biodiversité. Ce potentiel de restauration naturelle semble être limité dans des secteurs tels que PPauB et DF où des interventions humaines sont toujours en cours, où le remblayage a considérablement modifié le potentiel de connectivité hydrologique latérale entre le cours d’eau et l’aquifère et où la cohésion des berges peut limiter la migration du chenal. La restauration des processus physiques dans ces cas exigerait la participation des propriétaires riverains afin de limiter les interventions directes dans le cours d'eau et dans les bandes riveraines. Cette étude souligne qu'il est utile de travailler sur le potentiel de restauration des cours d'eau agricoles redressés, car ils peuvent fournir des services écosystémiques importants lorsque les processus hydrogéologiques naturels sont en mesure de fonctionner.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

Acronyms

Field sites:

1) Branche 53 Des Fèves (**DF**)
2) Petite Rivière Pot-au-Beurre (**PPauB**)
3) Ruisseau Martin (**RM**)

DEM: Digital Elevation Model

1 INTRODUCTION

1.1 Background

Riparian wetlands result in part from the physical foundation laid by the dynamics of natural stream processes such as lateral migration and meander cut-offs (Figure 1). Abandoned meanders, known also as oxbows, are rich in biodiversity and constitute riparian wetlands due to the hydraulic connections they maintain with the adjacent watercourse (Phillips, 2013). Riparian wetlands are often priority targets for restoration due to the many ecosystem services they provide that benefit both the environment and society:

- Hydrological: water supply; flood mitigation (Piégay et al., 2005); groundwater recharge; etc.
- Water quality: contribution to nutrient removal, pollutant filtration in agricultural sectors (Hansen et al., 2018); nutrient cycle; capture sediments (Kuenzler, 1990; Gleason, 1996); etc.
- Biodiversity: biological control; formation of lotic habitats and refuges (Phillips 2013); food chain; etc.
- Cultural: recreation (hunting); etc.

To fulfill these functions, it is necessary to maintain a lateral hydraulic connectivity between the riparian wetland and the watercourse, a process that varies depending on the geomorphological context (Larocque et al., 2016). Indeed, it is well known that, in some geological contexts, abandoned meanders may provide important zones of connectivity between the floodplain and the channel (Philips 2013). However, in agricultural basins of the Saint Lawrence Lowlands, the combination of channel straightening (Figure 2) and the addition of drainage networks often result in a disconnection between the channel and its floodplain due to a lowering of the water table (Brunland et al., 2003). Indeed, small water bodies including headwater streams and riparian wetlands are very sensitive to anthropogenic disturbances due to their high level of connectivity within the landscape (Riley et al., 2018). The effect of continued agricultural activity has been great losses in riparian wetland habitats as well as an interference with the processes that create and maintain them.

Figure 1: A) Historical analysis of ancient channel positions of the Mastigouche River; B) restored wetlands of the Calapooia River, Oregon (Henderson, 2018).

Despite their clear ecological importance there is very little knowledge on restoration potential of reconnecting old meandering bends to create riparian wetlands in degraded agricultural streams. For example, many case studies on the restoration / reconstruction of lowland rivers via re-meanderization are generally focused on the morphological response of the bed profile in relation to longitudinal connectivity rather than lateral or vertical hydraulic connections (Eekhout et al., 2015). Furthermore, although there is evidence that restored wetlands can benefit flood attenuation and water quality in agricultural landscapes (DeLaney, 1995) some studies suggest that restoration will largely be ineffective in watersheds where human disturbances persist (Pierce et al., 2012). Unfortunately, many restoration projects do not properly address the underlying sources of degradation and therefore fail at restoring target ecosystem functions (Gleason, 1996) or do not provide long-term benefits when instream structures are used rather than restoration of processes (Beechie et al., 2008). More recently, a three tier approach to restoration is recommended where restoration efforts are applied not only to the channel but also the surrounding riparian zone while also managing activities in the wider catchement (Riley et al., 2018). In order to move towards more effective, long-term restoration practices, a better understanding of the restoration potential of degraded agricultural streams and the role played by hydrological connectivity in ancient meanders is required. The results of this study will provide important information for decision makers to help guide restoration efforts for long-term benefit.

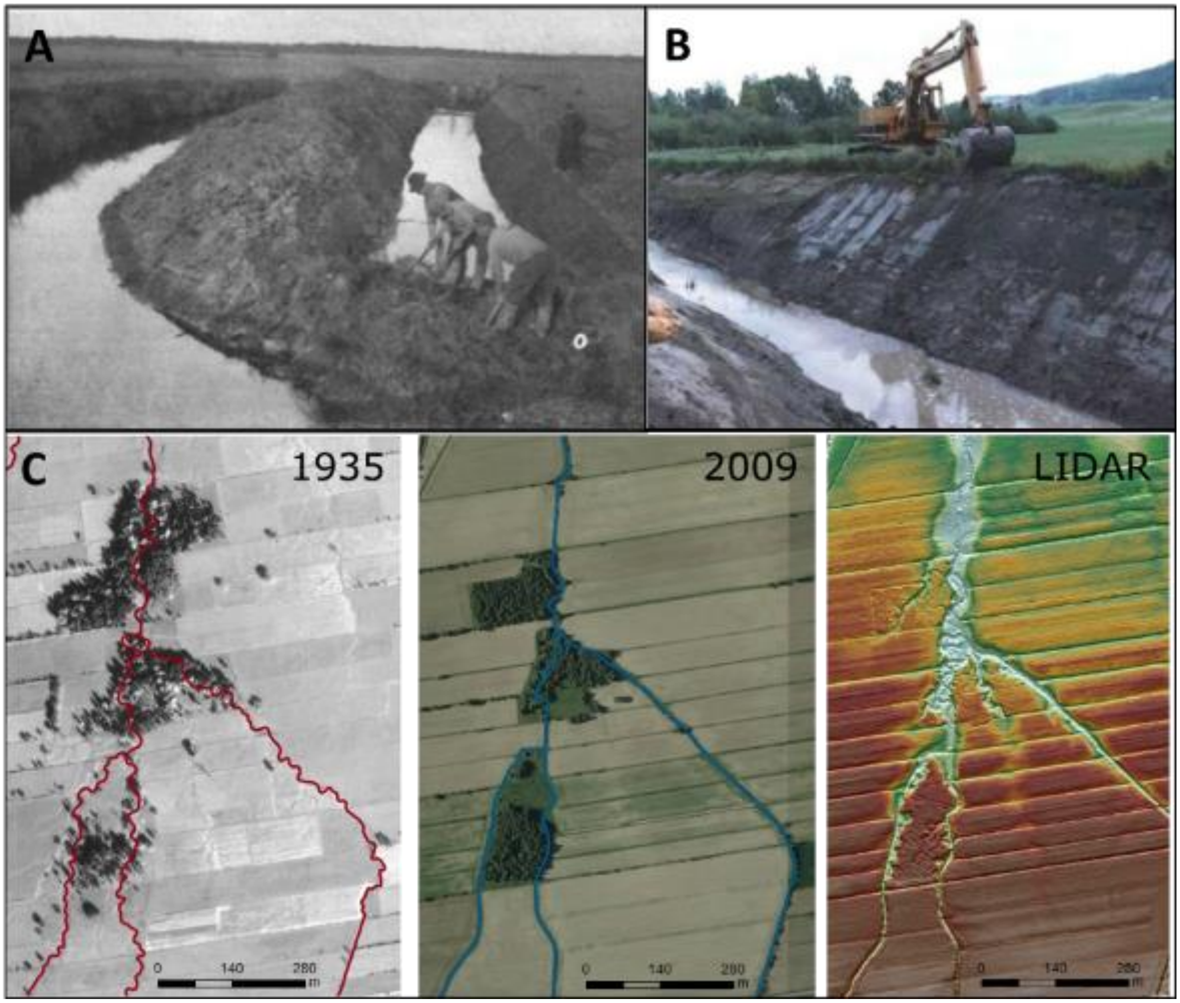


Figure 2: Examples of channelization practices before 1920 A), and after 1930 B), Planform view of the extent of channel straightening commonly found in agricultural areas (here, in the Petite Rivière Pot-au-Beurre watershed) C).

1.2 Main objectives

The ultimate goal of this project is to define indicators of riparian wetlands in agricultural watersheds that have the best potential to be restored and have their ecosystem functions rehabilitated. To accomplish this we established the following five main research objectives:

1. Determine the state of knowledge regarding ancient meander restoration
2. Provide examples of potential wetland restoration for straightened streams
3. Predict potential for degraded streams to re-meander
4. Examine the hydraulic response to wetland restoration scenarios
5. Propose criteria for evaluating the restoration potential of riparian wetlands in the straightened agricultural rivers of Quebec

A key component to this project involves the detailing and clarification of the role that ancient meanders can play in the restoration of riparian wetlands in agricultural watershed (Figure 3).

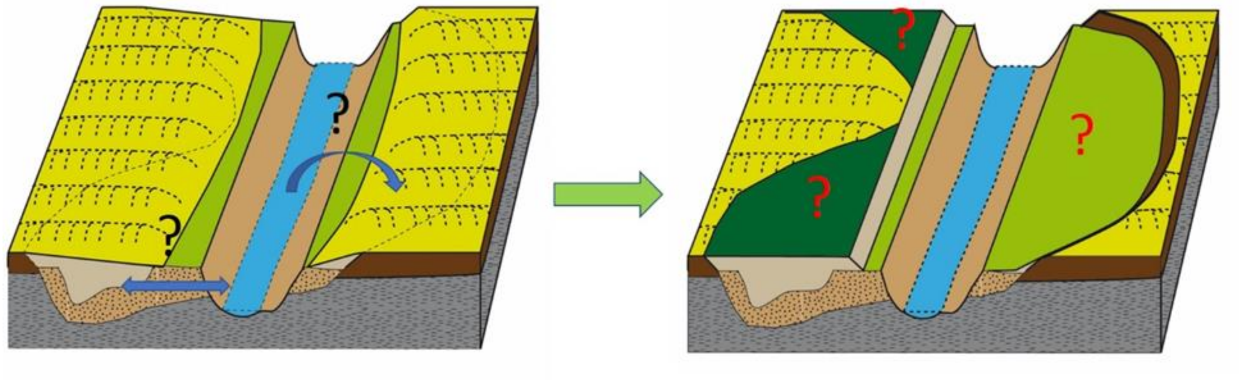


Figure 3: Conceptual model of artificially abandoned meander due to channel straightening illustrating potential hydrological connections and restoration targets.

2 LITERATURE REVIEW

The first research objective was to establish the global state of knowledge about projects involving the reconnection or rehabilitation of ancient meanders and oxbows and more specifically the most recent indicators of success of such projects. To accomplish this we conducted a literature review where the key words (oxbow AND restoration), (abandoned channel AND restoration) and (riverine wetland AND restoration) were used for research in the Geobase-Engineering Village and Web of Science platforms. Out of 308 research results, more than 40 peer-reviewed scientific papers were selected for their relevance about abandoned meander restoration project all around the world (USA, Denmark, France, Germany, Netherlands, China, Romania, Poland, Czech Republic, Austria, Spain). Eight main elements were analyzed through the reading of these articles: the global approach of the study, the types of objectives aimed by the restoration project, the selection criteria for the sites to be restored, the types of rivers targeted and the causes of their degradation, the types of restoration work undertaken, the main results and the main conclusions/suggestions resulting from abandoned meander restoration projects. The results of this literature review will help inform our final selection of criteria for determining the restoration potential of wetlands in degraded agricultural streams.The global approach of the reviewed studies about abandoned meander restoration projects fall into three broad categories: pre- and post-restoration studies (eg, Harrison et al., 2014; Seidel et al. 2017), comparison studies between restored and unrestored environments (Depret et al., 2017; Fischer et al., 2018) and modeling studies (Fisher & Stratford, 2008; Hester et al., 2016). While most post-monitoring studies span 1 to 3 years, some post-restoration follow-ups around 10 years (Kristensen et al., 2014; Stefanik & Mitsch, 2017). Some studies do not deal with ongoing restoration projects but are interested in documenting the hydrological, sedimentary and/or geochemistry of abandoned meanders or riparian wetlands in order to improve the design of a restoration project (Cabezas et al., 2008; Hauer et al., 2014).

Restoration objectives

The most common objectives of restoration projects are to improve water quality (Surridge et al., 2012; Jones et al., 2015; Hester et al., 2016; Vlad & Toma, 2017), the improvement of fish habitat quality (Fischer et al., 2018; Oldenborg & Steinman, 2019) and the mitigation of flood risks (TeLinde et al., 2010; Reckendorfer et al., 2013; Lamouroux et al., 2015). The goal of improving water quality is almost always associated with the reduction/uptake of nitrogen and phosphate inputs from agriculture (Harrison et al., 2012). Other objectives that are often proposed are the general increase in biodiversity (Paganelli & Sconfietti 2013; Riquier et al., 2017), the creation/maintenance/protection of riparian wetlands (Sidle et al. 2000; Gallardo et al., 2012; Stammel et al., 2012) and the improvement of the streams' hydrogeomorphological dynamic (Hein et al., 2004; Seer et al., 2018) and overall physical complexity (Addy and Wilkinson, 2019). Only two projects (Pedersen et al., 2007; Pilařová et al., 2014) address the improvement of the recreational potential of riverine area as a restoration objective. It is interesting to note that several projects taking place on the territory of the European Union present restoration objectives based on transnational guidelines (i.e. EU Water Framework Directives or WFD) which determine both priority issues, but also ecological targets (Fisher & Stratford, 2008; Lüderitz et al., 2011; Obolewski et al., 2016; Seidel et al., 2017).

Selection criteria for abandoned meanders

The selection criteria that were used to determine which abandoned meander should be reconnected or restored are not explicitly mentioned in the vast majority of reviewed studies. Several projects are taking place on severely degraded sections of watercourses following human interventions in the watershed (agricultural drainage, wetland drying) or directly in the channel (linearization, bank stabilization, dams, etc.). In the context of the European Union, the high degree of degradation of certain watersheds designates them as priority intervention basin (under the ecological targets of the WFD) (Fisher & Stratford, 2008; Lüderitz et al., 2011; Obolewski et al., 2016). However, the WFD doesn't specify the precise criteria for selecting the specific restoration sites. The high ecological status of some abandoned meanders, as focal points of riparian biodiversity, justifies in some cases the restoration interventions (Stammel et al., 2012; Paganelli & Sconfietti, 2013; Kristensen et al., 2014). In a specific case where the main objective is the creation of floodwater storage areas, the hydraulic capacity of the riparian environment as well as the low density of human occupation constitute the main selection criteria (Zhang et al., 2014). In Jones et al. (2015) as well as in Fischer et al. (2018), the rivers to be restored are in agricultural areas and the main selection criteria is the low productivity of the land adjacent to the abandoned meander. In Harrisson et al. (2012, 2014), the criteria used for the selection of stream reach to be restored are: a prior knowledge of the geomorphological and biological dynamics of the sites, the presence of a gauging station to perform hydrological monitoring and granted access to the sites by the owners. The latter is also evoked by Pilařová et al. (2014a,b) in the context of an oxbow lake's restoration in an urban center. In this particular case, the fact that the site is municipally owned and that the municipality is in favor of the project has greatly influenced its selection.

Stream types

The vast majority of restoration cases involving the reconnection or the restoration of one or more abandoned meanders have been carried out on high-order streams, often in the lower reaches of their hydrological network. The Strahler order that corresponds to the restored reach is not explicitly mentioned in most read articles, but the streams are often referred as being the main watercourses, used for navigation, harnessed by dams and /or located in estuarine zones. The most cited rivers in the literature related to abandoned meander restoration projects are the Danube (drainage area: 801,463 km²) (Hein et al., 2004; Stammel et al., 2012; Reckendorfer et al., 2013; Pander et al., 2018), the Rhône (98,000 km²) (Lamouroux et al., 2015; Depret et al., 2017; Riquier et al., 2017) and the Ebro River (85,550 km²) (Cabezas et al., 2008; Gallardo et al., 2012). The restored reaches in these large rivers cover several abandoned meanders and are spread over tens of kilometers. The cases of the Skjern River in Denmark (2100 km²) ((Kristensen et al., 2014, Pedersen et al., 2014) and Stör in Germany (Seer et al., 2018) are examples of intermediate stream restoration projects which are also spread over several kilometers. In other cases, the restoration projects are carried out on one or few abandoned meanders abandoned on intermediate-sized rivers (Obolewski & Glińska-Lewczuk, 2011; Fischer et al., 2015). Some projects have also been undertaken on small (second-third order) streams, severely degraded in agricultural settings (Jones et al., 2015), urban settings (Harrison et al. 2012) or both (Hester et al., 2016). Restored river reaches generally have a meandering fluvial style, but some projects have also been also initiated on braided sections (Hein et al., 2004;Te Linde et al., 2010) or wandering reaches (Riquier et al., 2017)

Types of perturbations negatively impacting streams

In the vast majority of restoration projects reviewed, the stream reach shows a high level of degradation. In very few cases, restoration is carried out on natural abandoned meanders with a relatively high ecological value and for which a "rejuvenation" of the riparian area is the goal to be achieved (Paganelli & Sconfietti, 2012; 2013; Hester et al., 2016). The main type of perturbation is undoubtedly stream straightening (for agriculture or navigation purposes) leading to the formation of artificial abandoned meanders (Sidle et al., 2000; Obolewski & Glińska-Lewczuk, 2011; Hauer et al., 2014; Depret et al., 2017; Vlad & Toma, 2017; Fischer et al., 2018; Addy and Wilkinson, 2019). These types of intervention have usually been combined with the installation of bank stabilization or flow control structures (eg. artificial weirs, dykes) (Te Linde et al., 2010), the creation of levees as flood protection (Fisher & Stratford, 2008) and severe drainage measures in the floodplain (Harrison et al., 2012; Kristensen et al., 2014; Lehr et al., 2015). Only one recent study (Addy and Wilkinson, 2019) refers to artificially abandoned meanders that would have been subsequently partially or completely backfilled by local material, as it is often the case.

Types of restoration work

Restoration works undertaken in abandoned meanders vary according to the state of the meander (eg. stage of evolution), the level of degradation of the watercourse and with the pursued objectives. In many cases, the restoration of abandoned meanders involves physically reconnecting the abandoned meander to the channel (Hein et al., 2004; Stammel et al., 2012; Riquier et al., 2017; Seidel et al., 2017; Addy and Wilkinson, 2019; Oldenborg & Steinman, 2019). This implies that the upstream and downstream ends of the abandoned meander are mechanically excavated to allow the flow of river water. Rigid stabilization structures are sometimes used to maintain the stability of the openings (Kristensen et al., 2014; Seidel et al., 2017; Seer et al., 2018). Vegetation planting in restored meander banks is also used to stabilize the work and counter the invasion of exotic species (Gallardo et al., 2012; Jones et al., 2015; Seer et al., 2018). In some cases, backfills or artificial dams are used to block the old stream route and force water into the abandoned meander (Pedersen et al., 2014; Lehr et al., 2015). When the elevation gradient between the main stream and the abandoned meander is too large to allow mass transfer (at predefined flow rates), the excavation can extend to the abandoned meander bed (Jones et al., 2015; Lehr et al., 2015; Obolewski et al., 2018). Excavation of the abandoned meander bed, as well as the removal of macrophytes, are also used for projects which goal is to "rejuvenate" a riparian wetland (Gallardo et al., 2012; Paganelli & Sconfietti, 2013), meaning bringing back the oxbow lake to an earlier stage of evolution. We are referring here to the stage of evolution of an abandoned meander that transits from a semi-lotic environment (oxbow lake) to a completely terrestrialized environment, gradually being filled with overbank sediment and integrated with the alluvial plain. When the restoration project includes the creation of fish habitat and aims at increasing the hydrogeomorphological dynamics of the watercourse, physical elements are introduced into the channel, such as coarse sediments and woody debris (Stammel et al., 2012; Hester et al. al., 2016; Pander et al., 2018; Seer et al., 2018). Some projects create hydraulic units (riffle, pool) inside the restored meanders (Harrison et al., 2012). In some cases, the abandoned meander remains completely or partially disconnected (Lüderitz et al., 2011; Obolewski et al., 2016). In such case, some projects use pumping systems to redirect part of the stream flow to an abandoned meander or a restored riparian wetland (Sidle et al., 2000; Fisher & Stratford, 2008). In one case, a network of PVC pipes were installed within the alluvial plain to hydrologically connect the stream to the abandoned meander even beneath bank full stage (Obolewski & Glińska-Lewczuk, 2011). Finally, in the design plan of a restoration project of an urban riparian wetland, which includes an oxbow lake, recreational elements were incorporated into the landscape, including a playground, a bicycle path and interpretation panels (Pilařová et al., 2014a,b).

Main restoration measured impacts

The literature review has identified some common positive impacts among the different abandoned meanders restoration projects. Some impacts are unequivocal while others are more ambiguous. Clear positive impacts include increased aquatic and riparian biodiversity and improved hydrogeomorphological dynamics. Several studies show that the reconnection of one or more meanders has led to a local increase in ichthyic diversity (Gallardo et al., 2012; Jones et al., 2015; Lamouroux et al., 2015), avian diversity (Kristensen et al. 2012; Pander et al., 2018), macrobenthic diversity (Obolewski & Glińska-Lewczuk, 2011; Obolewski et al., 2016; Seidel et al., 2017) and plant diversity (Pedersen et al., 2014; Seer et al., 2018). The increase in fish diversity is related to the observed increase in the morphological quality of reconnected meanders (Lüderitz et al., 2011; Hauer et al., 2014). Changes in sediment chemistry, following the excavation of the fine sediment layer, were identified as one of the main causes of improvement for benthic diversity (Obolewski et al., 2018). An increase in the lateral hydrological connectivity (i.e. hydrological transfer between the channel and adjacent meanders) and an increase in the morphological and hydraulic heterogeneity of the alluvial plain are the main factors responsible for the general increase in biodiversity (Obolewski et al., 2018). Another clear positive impact of abandoned meander restoration through reconnection is the increase in flooding frequency and duration (hydro-period) of the riparian wetland i.e. oxbow (Fisher & Stratford, 2008; Kristensen et al., 2014). Restoration projects that sought to rejuvenate abandoned meanders in terms of oxbow's evolutionary stage were also successful (Paganelli & Sconfietti, 2013). Finally, in the context of a restoration project on the Rhône River, the increase in riparian ecological quality has led to an increase in the degree of attachment of local residents to their river (Lamouroux et al., 2015).

Impacts with more ambiguous results include nutrient uptake in restored meanders, changes in physical complexity, and impacts on underground hydrological connectivity. Several studies show reductions in nitrate concentration (Harrison et al., 2012), phosphate (Oldenborg & Steinman, 2019), or both (Harrison et al., 2014; Pedersen et al., 2014; Jones et al., 2015) in streams whose abandoned meanders have been restored. However, a recent study (Addy and Wilkinson, 2019) concluded that the newly created backwaters may help increase a reaches capacity to retain nutrients and fine sediments. Addy and Wilkinson (2019) also noted that in the 3 years following ancient meander reconnection, although changes such as channel widening, increased deposition and decreased bed slope did occur, no clear differences in physical complexity or pool volume was observed (compared to the control reach), indicating that positive effects may take more time or that the degraded conditions of the watershed are counteracting the restoration efforts. The better sorption capacity of the deep sediments of abandoned meanders, revealed by the excavation of the upper fine sediments, can explain the decline in phosphorus concentration (Oldenborg & Steinman, 2019). Harrisson et al. (2014) suggest that the positive impacts of restoration on nutrient uptake in abandoned meanders varies annually depending on the level of hydrological connectivity, the hydraulic gradient between the main channel and the restored meander and the water retention time within the meander and its riparian area. However, other studies show conflicting results, for example an increase in the nitrogen and phosphorus load in the abandoned meander in relation to the increase of the hydro period (Fisher & Stratford, 2008), while Oldenborg & Steinman (2019) suggest that this increase applies to the duration of the restoration works. The modeling study by Hester et al. (2016) suggests that restoration work will have minimal impact on nitrogen uptake. A study about the impact of the reconnection of an abandoned meander on the level of groundwater exchange (Lehr et al., 2015) suggests that the excavation of the restored meander bed did not have a significant influence. Rather, it suggests that the natural heterogeneity of the hydraulic conductivity of meander bed sediments determines the level of exchange between surface water and groundwater.

Lessons learned from past restoration projects

Two main lessons can be learned from the various abandoned meander restoration projects analyzed. The first is that the main key to success of that type of restoration project is to target the increase of the level of hydrological connectivity between the main stream and its abandoned meander/floodplain (Lüderitz et al., 2011). More importantly, restoration works should aim to increase the diversity of hydrological connection types (Hein et al., 2004; Obolewski et al., 2016; Pander et al., 2018; Seer et al., 2018). This diversity can be increased by promoting heterogeneity in the morphology of the excavated meander bends, by introducing natural physical elements that can influence flow conditions and water retention time in the meander, such as coarse sediments and woody debris and/or by promoting hydrological exchanges at several levels of discharge. The second lesson is to avoid the trap of aiming at restoring pre-disturbance conditions of the watercourse (Hauer et al., 2014; Brown et al., 2018). Restoration planning should prioritize works that can evolve in space and time. For these reasons, restoration project should plan for the implementation of buffer zone adjacent to the restored meander to allow morphological, but also biological mobility of the restored riparian environment. Projects should also avoid the installation of rigid stabilization structures that limit this mobility (Kristensen et al., 2014).

It is important to make distinctions while comparing the response time of restored meanders belonging to stream reach with different hydrogeomorphological dynamics. Highly dynamic fluvial environments, such as streams with significant bedload transport (braided, wandering) or large rivers are likely to respond more quickly to restoration/reconnection than smaller, less dynamic streams (Pander et al., 2018). This situation highlights the importance of implementing long-term monitoring of stream response to meander restoration project, especially for small streams (Gallardo et al., 2012). In more practical terms, some actions may be taken when carrying out the restoration work to help achieve objectives and avoid collateral environmental damages. That includes environmental management of excavated sediments to avoid major inputs of pollutants and nutrients to the watercourse (Obolewski et al., 2018) and measures to counter the colonization of restored meanders by invasive species (Gallardo et al., 2012; Seidel et al., 2017). Planting native species at the restored site can be beneficial for controlling invasive alien species and for initiating riparian forest canopy development (Seer et al., 2018). However, one study in particular shows that after fourteen years, restored planted and non-planted sites exhibit a similar diversity of plant species (Stefanik & Mitsch, 2017).

Finally, several studies indicate that the participation of local stakeholders, especially riparian owners such as farmers, is beneficial for the good conduct of restoration project (Pilařová et al., 2014; Jones et al., 2015; Flávio et al., 2017). The partial or complete integration of local stakeholders to the decision-making process has led to the development of public interest toward restoration projects, particularly in terms of water quality issues and recreational uses of riparian environments. It also appears that the implementation of programs governed by national or transnational bodies (eg. Water Framework Directive (EU) or Clean Water Act (USA)) and whose achievement of objectives becomes the responsibility of regional entities (watershed management organization for example) is a predominant factor in setting up restoration projects (Flávio et al., 2017). However, it is important to recognize that such programs are accompanied by rules and criteria that may limit restoration initiatives on certain types of watercourse. This is particularly the case for small headwater streams, which, for reasons of watershed size (in the case of the WFD) or level of hydrological connectivity (intermittent or ephemeral watercourse) (in the case of the Clean Water Act in the USA) are excluded from funding programs.

3 STUDY REGION AND FIELD SITE CHARACTERISTICS

Three sites in the St-Lawrence Lowlands of Quebec were chosen to study (Figure 4): Branch 53 of the Des Fèves River - DF (site 1) near Ste-Martine, Petite Rivière Pot-au-Beurre River – PPauB (site 2) in the Lavallière Bay near Saint-Robert, and Ruisseau Martin - RM (site 3) near Saint-Samuel. The sites were chosen from three separate drainage basins to represent diversity in headwater streams. This is a region of relatively flat topography (ex: water surface slopes of the three sites range from 0.2% to 0.3%) and low elevations ranging from around 15 m above sea level at PPauB to 85 m above sea level at RM . Due to the physiographic characteristics of the Saint Lawrence Lowlands a large proportion of the area is designated for agricultural practices and many small first and second order headwater streams have been straightened with their ancient meanders filled to improve drainage and crop yields. As such the drainage areas of our study sites are also quite small ranging from 2.41 km^2 (DF) to 27 km^2 (RM). The study sites all showed evidence of a meandering channel pattern prior to their straightening during the middle of the 20th century and were chosen due to previous work that had been conducted by other studies and the availability of LiDAR data which allowed us to produce high resolution Digital Elevation Models (DEMs) of the area.

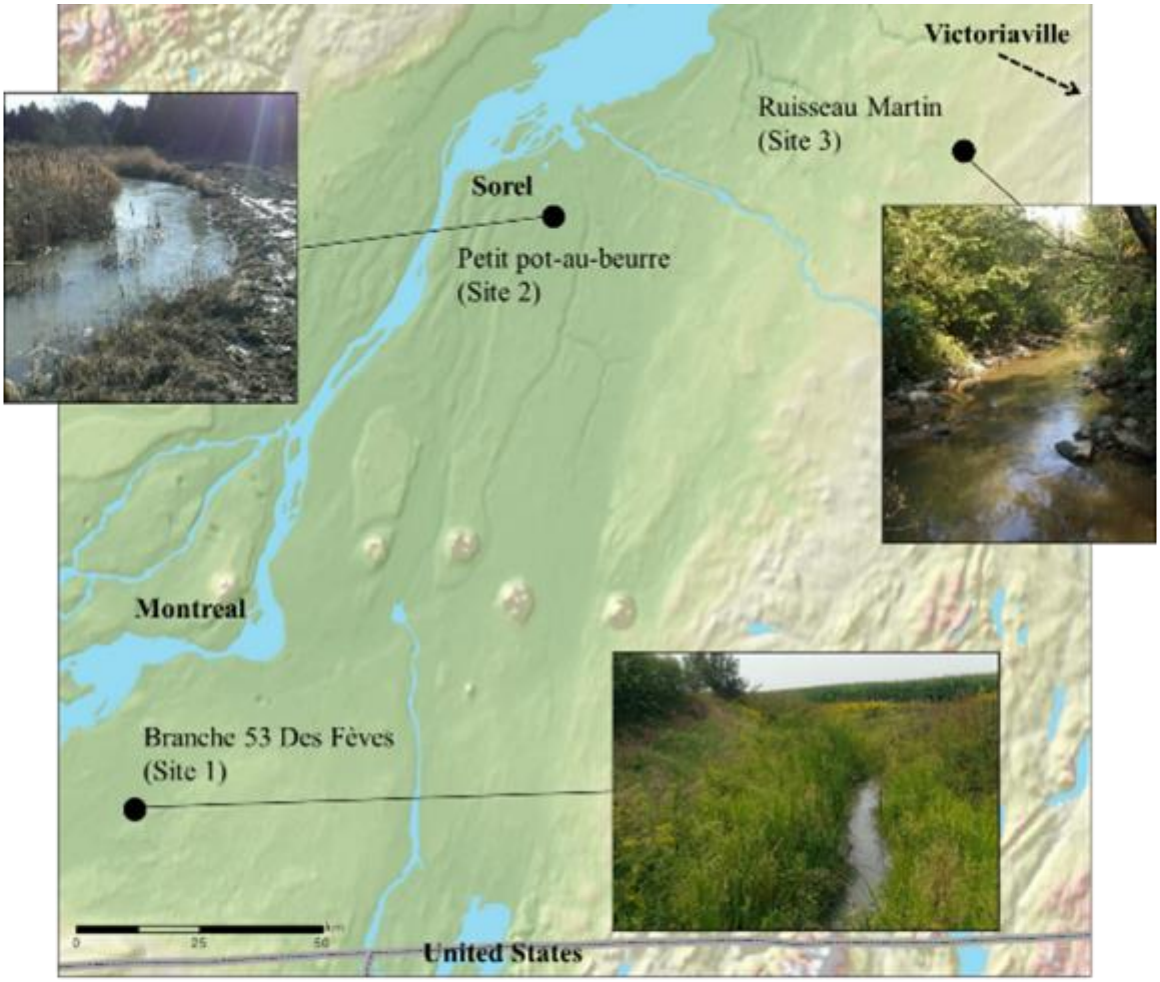


Figure 4 : Location of study sites in the Saint Lawrence Lowlands of Québec. Site1- Branche 53 Des Fèves; site 2 – Petite Rivière Pot-au-Beurre; site 3 - Ruisseau Martin

Site 1 - Branch 53 of the Des Fèves River:

This site is located within the Châteauguay River drainage basin which drains an area of approximately 2 540 km^2 south of Montreal. The basin is shared with the United-States although 55% of it is found within Québec borders, most of which is agricultural land use. Surface geology consists mainly of marine clay sediments from the ancient Champlain sea up to 18m in thickness (Tremblay, 2018). Alluvial sediments are sparse, although irregular mounds of very permeable till ranging in depths of 10 – 20m are spread throughout the study region.

Analysis of historical photography reveals that this headwater tributary of the Des Fèves River has already been partially straightened prior to 1930 (Table 1) although some sinuous sections are still visible in the downstream section (Figure 5). Indeed, the dominant landuse is agriculture with a very restricted riparian zone. As described by Paradis and Biron (2016), an extensive subsurface drainage is present in the surrounding fields of corn and soy. Such a network is typically maintained by frequent dredging of the channel bed, the most recent of which occurred in 2013. Despite the great degree of anthropogenic disturbance in the region, past dynamics of channel lateral migration are clearly visible from the LiDAR (Figure 5). LiDAR and field surveys of the site also revealed the presence of coarse glacial till deposits associated with a drumlin at the upstream section of our study reach which likely explains the two observed resurgence points where fresh groundwater enters the stream.

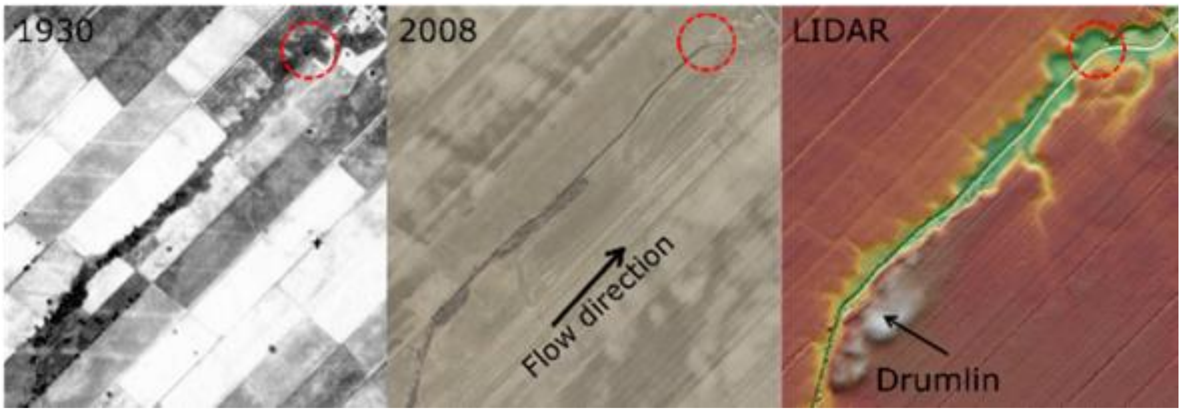


Figure 5 : Historical air photography and LiDAR showing clear indications of ancient meander bends and a drumlin.

The DF site is quite narrow with wetted width and depth measured during summer baseflow conditions ranging from 0.9 - 5.1m and 0.1 - 0.65m, respectively, along the 1km study reach. Values of water surface elevation extracted from the LiDAR reveal a mild slope of 0.23% typical for Lowland headwater streams. Water temperature measured during the study period ranged from 0.6 - 22.8°C with quite a bit of longitudinal variation as well as a moderating effect related to cooler groundwater flowing from the resurgence points.

A geomorphic assessment conducted at DF during the late summer of 2017 allowed for the identification of three distinct sub-reaches across the study site (Figure 6).

The first reach (Figure 6A) in the upstream section of the study reach is lined by corn fields along the left bank and a small forested area along its right bank. This is likely due to the presence of the drumlin and associated coarse sediments creating a hill and discouraging encroachment of agriculture right up to the stream. As expected, bed and bank sediments in this reach are fairly coarse with cobbles, gravels and sands being abundant, particularly at the resurgence zone. While there is some diversity in depth and flow downstream of the first resurgence point, upstream conditions are stagnant with a lack of flow resulting in the accumulation of a thick layer of silt.

In the second reach (Figure 6B) we see a greater diversity in morphology with the presence of a few partially developed riffle/pool units. As both banks of this reach are lined with a narrow buffer of forest the observed morphological diversity is likely due to the presence of woody debris. Still in proximity to the drumlin sediments are diverse ranging from gravels and sand along the bed to a higher proportion of clay and silt along the banks. This reach is also wider with a shallower slope to the banks allowing for some sinuosity to develop since the last dredging in 2013.

The third reach (Figure 6C) in the downstream section of the study reach is completely surrounded by fields of corn. This is a reach of very low morphological diversity dominated by clay and silt with a straightened channel pattern and little variation in width or depth, typical of small channelized streams in agricultural areas.

A closer look at cross sections extracted from the LiDAR (Figure 7) clearly shows the presence of several ancient meander bends in the downstream section of the study reach. Soil samples extracted using a manual auger showed a dominance of filling material and marine clay in these meander bends indicating that hydraulic connectivity through subsurface processes is likely limited to the resurgence points.

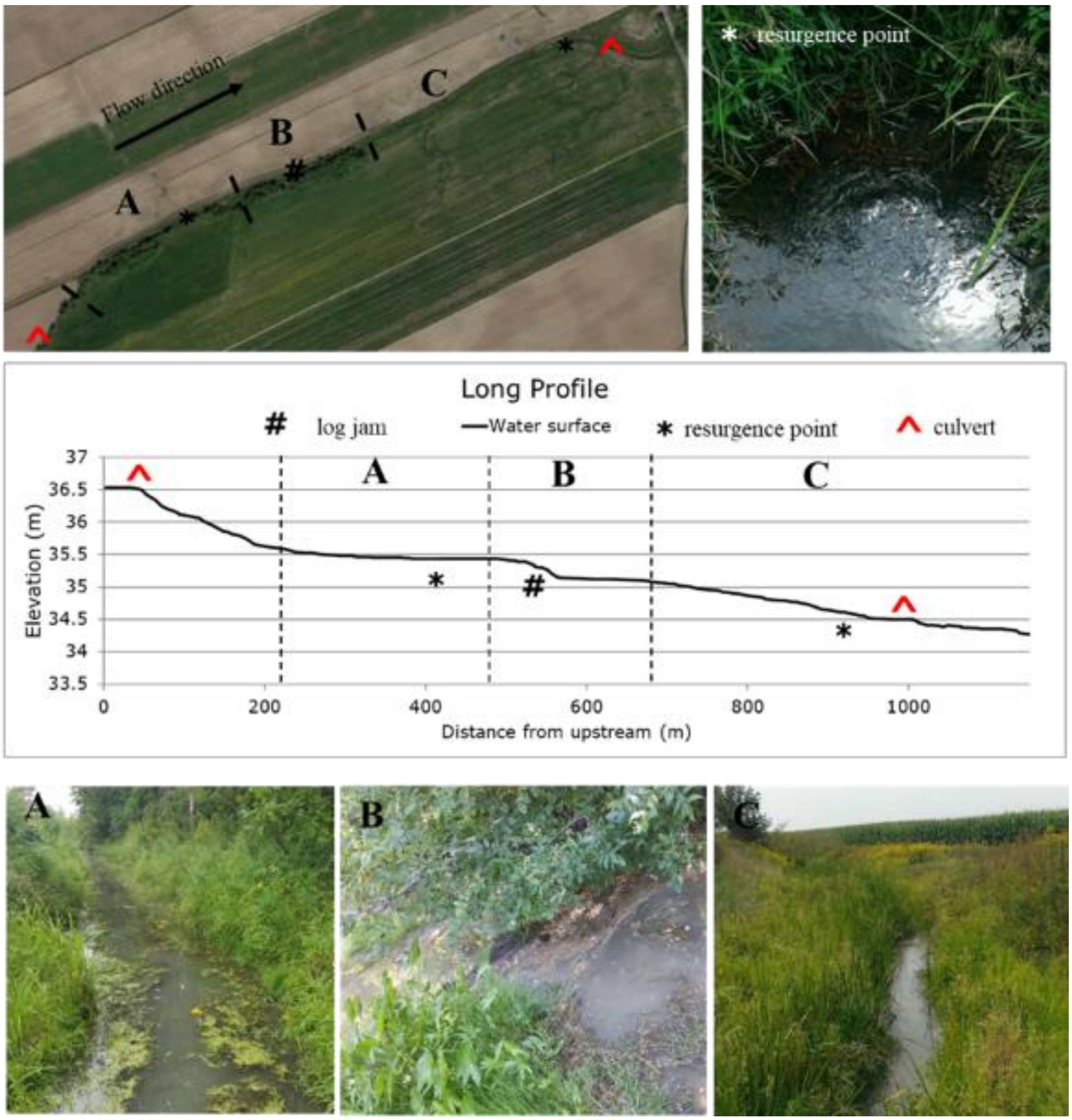


Figure 6 : Study site at Branch 53 of the Des Fèves River divided into three distinct reaches following a geomorphic assessment conducted in late summer. Reach A) upstream, reach B) middle, reach C) downstream.

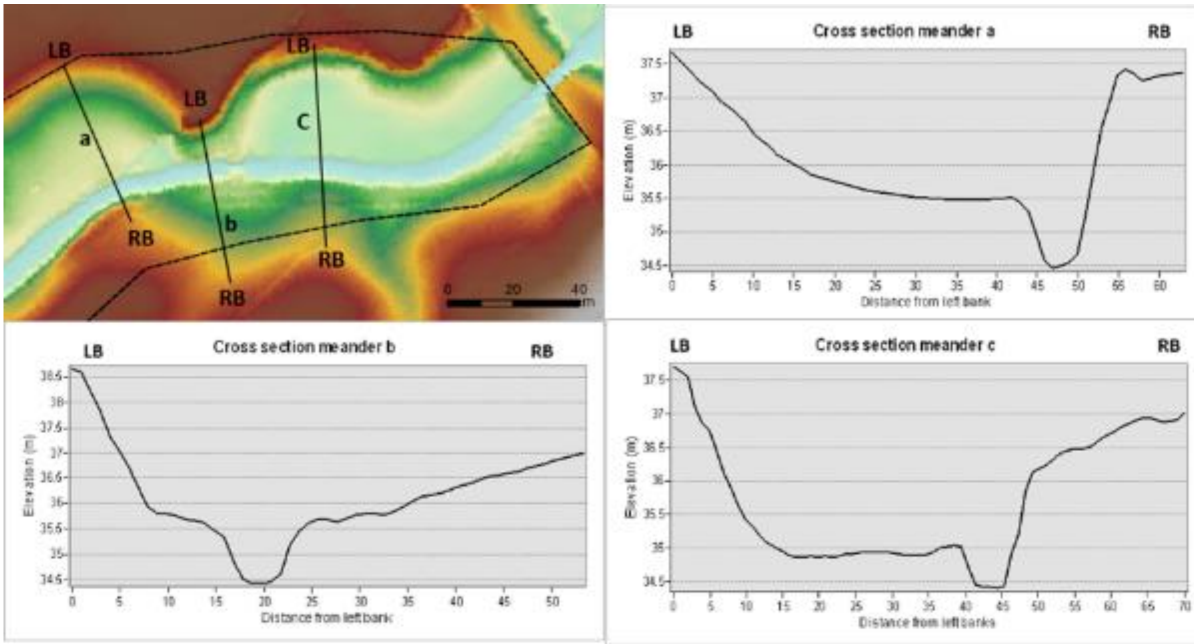


Figure 7 : Three cross sections extracted from the LiDAR clearly showing ancient meanders in the downstream reach of Branch 53 of the Des Fèves River.

Site 2 - Petite Rivière Pot-au-Beurre (PPauB)

This site is located within the Lavallière Bay watershed (192.5km^2) situated 80km to the north-east of Montreal. It is a highly degraded watershed due to the dominance of agricultural land use in the area and is classified as one of the most polluted in Quebec. The poor water quality, high sediment input from agricultural runoff and the drying up of riparian wetlands is well documented for this region (Fourcier et al., 2007).

A recent report (Carrier et al., 2013), produced to detail the groundwater resources of Montérégie Est, Quebec, describes the surface geology of the region around our study site as dominated by silty-clayey sediments from the former Champlain sea with a strong presence of silty-sandy fluvial sediments. The layer of fine sediments can be quite dramatic with a thickness of greater than 60m being reported in some locations. Indeed, results of the types and proportions of quaternary surface sediments observed showed that the main units consisted of fine marine sediments (45%), alluvial sediments (28%) and littoral sediments (16%). It was also indicated in the report that there were no outcrops of sediments of fluvioglacial or glaciolacustrine origin in our study area.

The PPauB occupies a small sub-basin of the Lavallière Bay draining an area of about 58.5km^2, with a drainage area of 8.62 km^2. Analysis of historical photos shows that this river network experienced significant channel straightening after 1964 (Figure 8) (Table 1). LiDAR data provide evidence of past meander dynamics in the floodplain, a state which is confirmed by the highly sinuous river pattern observed for some sections in the 1930 and 1964 photos. Comparison of 1935 and 2009 photos shows a loss in forest cover and a reduction in riparian zone area in addition to the channelization.

Figure 8 : Historical air photography and LiDAR showing a clear meandering pattern in both 1935 and 1964 while the stream has a straightened pattern in 2009. Gradual deforestation is also observed from 1935 to 2009 and meander scars are visible in the LiDAR. The dashed circle shows the site where groundwater monitoring devices were installed.

The study reach of the PPauB highlights the typical low relief and mild slopes of this region with a water surface slope of 0.2% and an elevation range from about 15 – 17m above sea level. Land use is as expected dominated by corn and soy agriculture with a few patches of forested areas that maintain some sinuosity. Stream wetted width as measured during the summer low flow conditions was very narrow with a minimum width of 0.4m recorded and an average wetted width of 1.5m. Depths were also quite shallow with as little as 0.05m of water in some sections. One exception was a relatively large pool which had formed downstream of a culvert (Figure 9C) that had a wetted width of 7m and a depth of 0.5m. Discharge values ranged from completely dry during the month of September to 1.27m^3/s during spring floods in April. Water temperatures were also quite variable ranging from a maximum of 27.5°C recorded in late summer to a minimum of 0.02°C recorded in December.

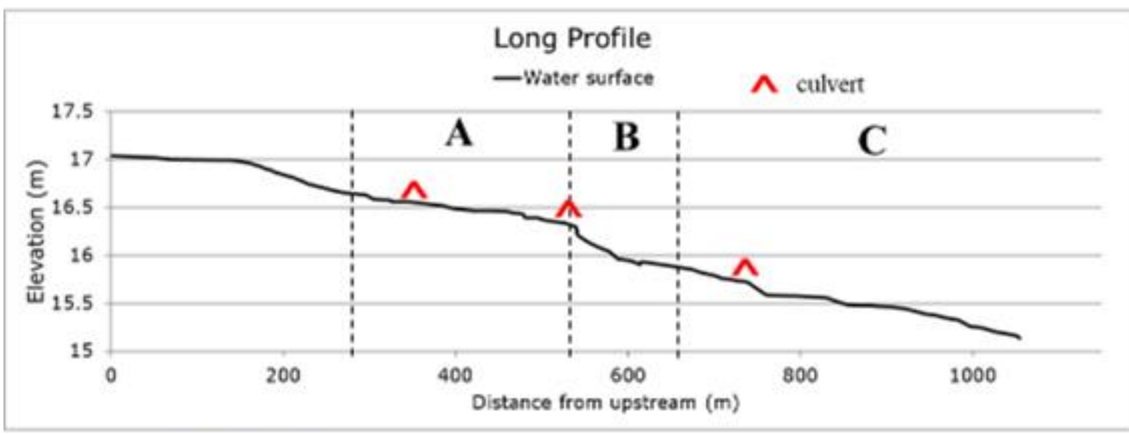
A geomorphic assessment conducted at PPauB during the late summer of 2017 allowed for the identification of three distinct sub-reaches across the study site (Figure 9).

The first reach (Figure 9A) in the upstream section, despite previously being channelized and surrounded by agricultural fields, does show some development of sinuosity as well as variability in depth and flow. Bank slope in this section is relatively mild. Several runs were identified, and pools were typically observed upstream and downstream of most culverts. Clay is the dominant sediment (likely filling material from the adjacent agricultural fields) with a thick layer of siltation in zones of sluggish flow.

The second reach (Figure 9B) consists of a very short section with a forested riparian zone that does not appear to have been channelized post 1964. Some woody debris are present, and the sinuosity is much higher compared to upstream and downstream reaches, explaining the diversity in depth and flow observed in this reach. Both pools and runs were identified as well as an equal proportion of sand and clay sediments. Erosion was observed along both banks.

The third reach (Figure 9C) in the downstream section is lined by agricultural fields on both banks. With the exception of a large pool that has formed downstream of a culvert, the channel pattern is narrow and uniform with very low and slow flow, particularly during late summer. Sediments are dominated by clay and filling material with a layer of siltation in the pool.

A cross section extracted from the LiDAR (Figure 10) indicates the presence of an ancient meander. Taking into account the vertical exaggeration of the cross-section profile, this stream appears to be disconnected from its floodplain. Soil samples extracted in the abandoned meander reveal the presence of alluvial deposits (fine gravel and sand) beneath a thick layer of filling material and a clayey/silty horizon.



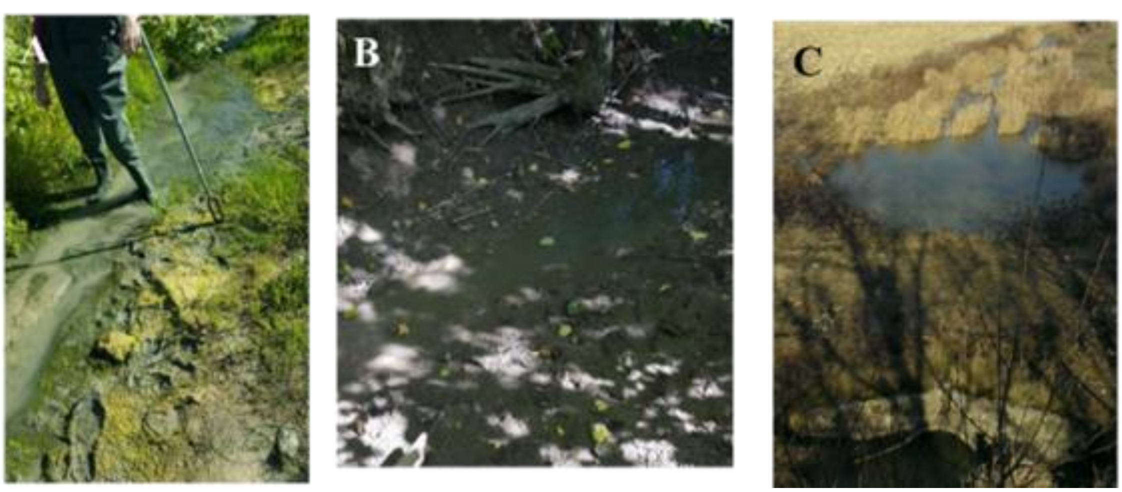


Figure 9 : Study site at PPauB divided into three distinct reaches following a geomorphic assessment conducted in late summer. Reach at low flow A) upstream, reach B) middle, reach C) downstream.

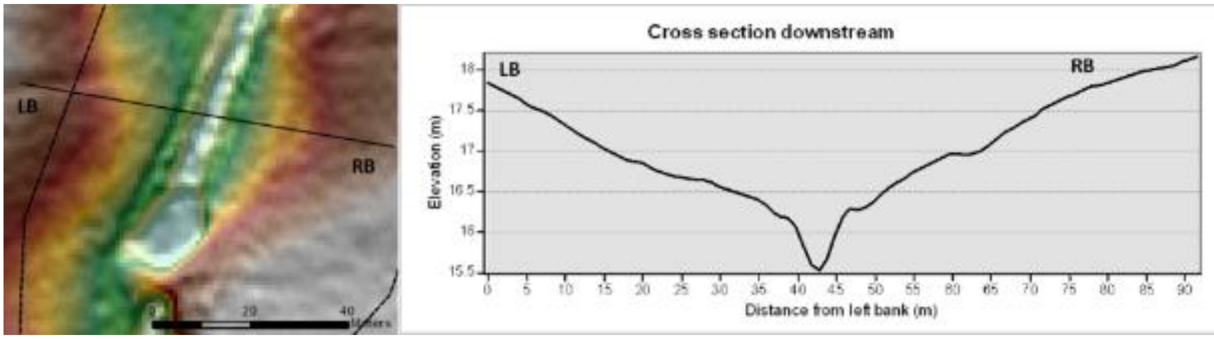


Figure 10 : Cross-section extracted from the LiDAR highlighting disconnection from the ancient meander in the downstream reach of PPauB.

Site - Ruisseau Martin (RM)

As described by Olsen et al. (2012) RM drains an area of approximately 40km^2 into the Bulstrode River, a tributary of the Nicolet river. The study reach drains a smaller area of 27km^2. The watershed is again dominated by agricultural land use. Analysis of historical air photography (Figure 11) reveals that many sections of the stream were straightened between 1950 and 1960 (Table 1) resulting in a clear decrease in sinuosity. The ancient channel paths, floodplain and abandoned meanders are also quite visible in the LiDAR. Interestingly, despite heavy agriculture, reforestation is observed along some reaches of RM (Figure 12). A recent report (Larocque et al., 2015), produced to further develop the groundwater resources knowledge base in the Nicolet and lower zone of the Saint-Francois regions describes the quaternary surface geology of the area surrounding our study site as stratigraphically complex, consisting of a succession of tills, separated by layers of glaciolacustrine sediments which are effectively impermeable due to their large size and texture. Surface deposits are reported to be dominated by sands of littoral and eolian origin, and clay, above the impermeable basal unit of tills. Rock outcrops are also common in our study area and there are clear connections between the regional aquifer and the river.

The study site (Figure 12) is fairly different from the previous two site locations boasting a wider range of wetted width and depth (1.5 – 6.2m and 0.1 – 0.6m respectively), a higher slope of water surface (0.3%) and an elevation range of 81.5 – 84.5m above sea level. As the drainage area is the largest of the three sites it is not surprising that discharges recorded were also larger, ranging from 0.11m^3/s during summer baseflow conditions to 1.73m^3/s during high spring flows. Maximum temperature values are also less (range of (0.5 – 22˚C), likely due shading from the forest cover and potential hydrologic connectivity with groundwater. The higher morphological diversity and larger riparian buffer observed at this site indicates that it may be a good case for illustrating the potential of passive restoration.

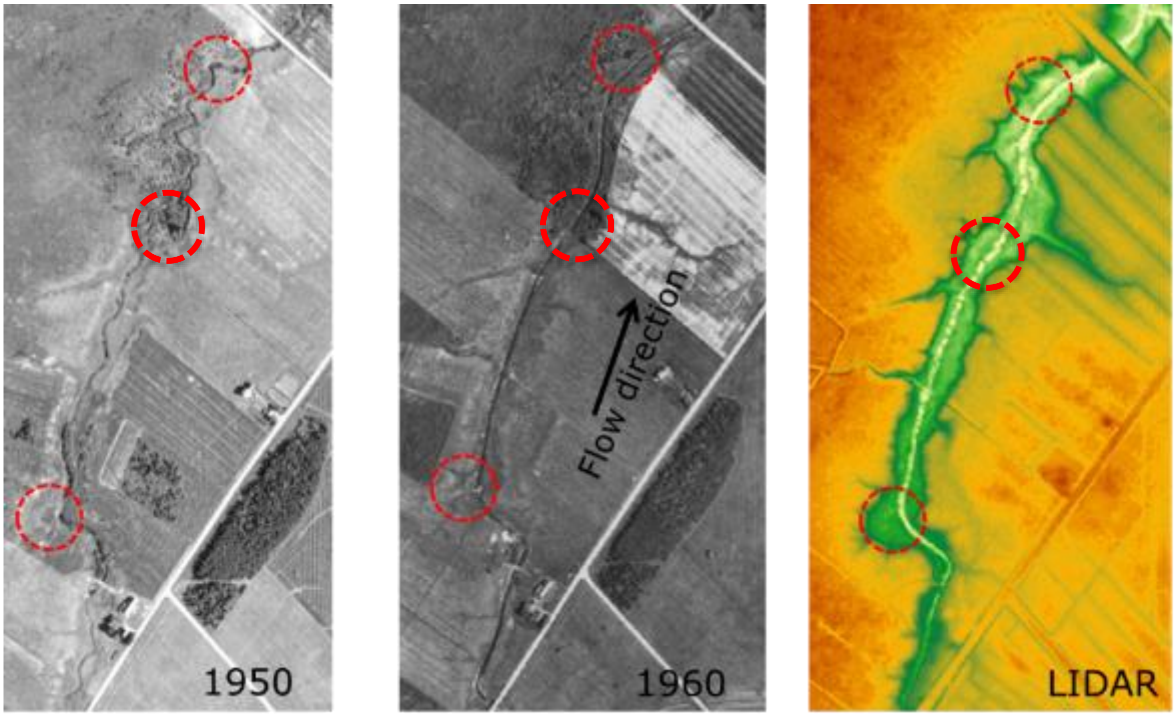


Figure 11 : Historical air photography showing a clear meandering pattern in 1950 and a straightened pattern in 1960. Traces of abandoned meanders (red circles) are also visible in the LiDAR. Red circles also show sites where groundwater monitoring devices were installed.

A geomorphic assessment conducted at RM during the late summer of 2017 allowed for the identification of three distinct sub-reaches across the study site (Figure 12).

The first reach (Figure 12A) in the upstream section is characterised by a high gradient straight planform geometry. The presence of rapids, riffles and pools creates a large degree of habitat and flow diversity despite the low sinuosity. The dominant sediment here is a mix of boulders and cobbles.

In the second reach (Figure 12B) planform geometry is again straightened but the gradient is lower. Several log jam accumulations are observed (Figure 12A) which explain the presence of riffles and deep pools as well as the large diversity in depth and flow. Boulders and cobbles are still common although there is now a strong presence of gravel and sand.

In the third reach (Figure 12C) we see much less diversity in depth and flow conditions. In this case, although there are still several log jams (Figure 13B), they appear to have more of a backwater effect with sluggish flow. A vegetated mid channel bar is observed downstream of a particularly large accumulation of woody debris and overall sinuosity is low. Sediments are smaller with less cobbles and gravels while more sand and silt sections are observed.

Cross-sections extracted from the LiDAR (Figure 14) indicate the presence of two distinct ancient meanders (several others exist along this site but two are showcased here). Soil samples extracted in the abandoned meanders reveal the presence of alluvial deposits (coarse gravel and sand) beneath a thick layer of sand. The topography and nature of sediment deposits indicate that hydraulic connectivity is likely maintained though both surface and subsurface processes at these meanders. The presence of surface water year-round on the downstream meander supports this hypothesis.

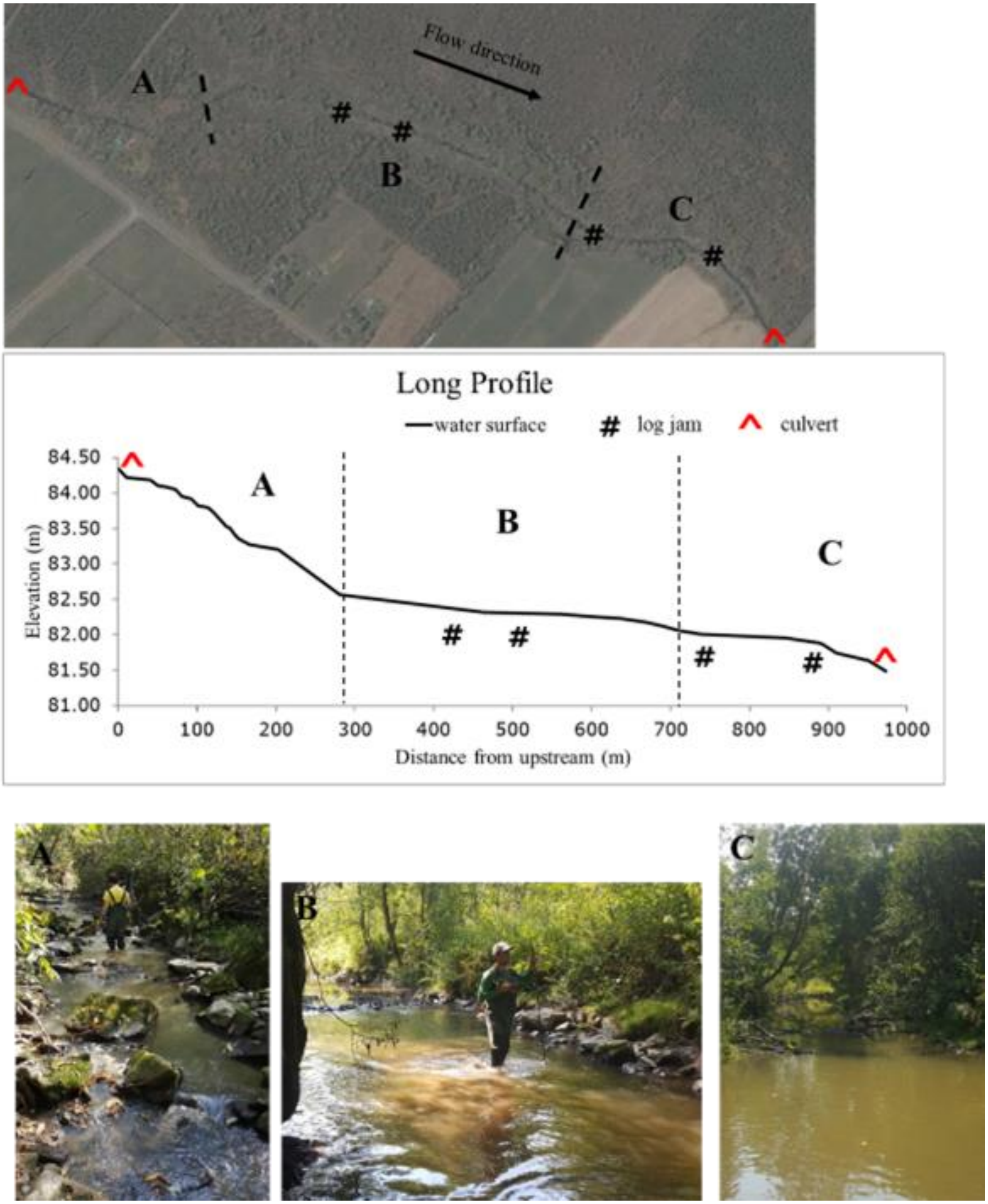


Figure 12 : Study site at RM divided into three distinct reaches following a geomorphic assessment conducted in late summer. Reach A) upstream, reach B) middle, reach C) downstream.

Figure 13 : Examples of woody debris accumulations (beaver dam in some case) at RM A) midstream reach B and B) downstream reach C.

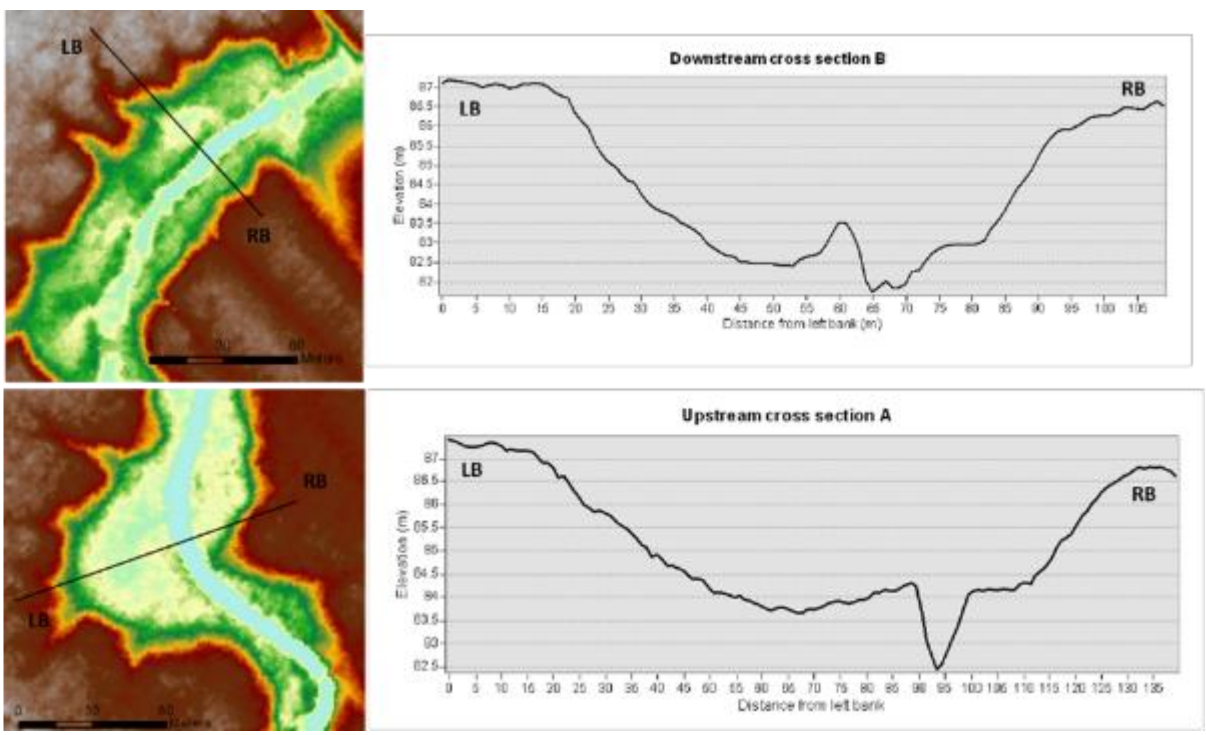


Figure 14 Cross-sections extracted from the LiDAR showing a clear ancient meander in the upstream and downstream reaches of RM.

Table 1: Summary of human interventions at the study sites and general characteristics.

Site	Approx. date of straightening	Last known maintenance	Historical (pre-straightening) channel length (m)	Current site channel length (m)	Drainage area (km^2)	Bankfull discharge (m^3/s)	Valley slope	Mean bankfull width
DF	After 1930	2013	1508	1405	2.41	0.9	0.002	4.14
PPauB	After 1964	2013	1125	994	8.62	1.99	0.002	3.11
RM	Between 1950-1960	None	1262	1020	27	5.8	0.002	7.75

4 METHODOLOGY

In this section, we evaluate current and past conditions of the channel/floodplain environment to determine suitability for potential wetland restoration for the three different straightened streams in agricultural areas (DF, PPauB, RM). For each site physical characteristics of the channel and ancient meander (channel geometry, surface geology, hydrology, hydrogeology and temperature) and ecological components (riparian vegetation and in-channel fish habitat) are measured through a combination of quantitative (ex: site instrumentation with loggers) and qualitative (ex: expert interpretation and cameras) measurements. The methodology aimed at determining suitability for wetland restoration potential qualitatively, taking into account the presence of existing wetlands and the potential for hydrological connections via surface and subsurface processes and ongoing anthropogenic perturbations (ex: drainage networks and culverts). Here the focus is on concepts of process-based restoration for long-term rehabilitation of wetland hydrogeological and ecosystem functions (Beechie et al., 2010 Wohl et al., 2015).

4.1 Data collection

Figures 15-17 provide a planform view of the three study sites and positioning of all instruments installed to refer to during the methodology and results described throughout section 4. Table 2 provides a summary of data collected over the two year period.

Table 2: Summary of data collected at the three study sites.

Data category	Data source	Primary use
Hydrology	• Field measurements of discharge (low, moderate and peak flows) • Pressure transducers in channel (water level upstream and downstream of study site)	• Identify significant flooding events • Compare water level fluctuations with groundwater
Barometric pressure	• Barologger installed at each site	• Compensation
Topography/ Geomorphology	• LiDAR (high resolution Digital Elevation Model) • Historical air photographs (1930, 1950, 1965) • Total station and differential GPS (ancient meander) • Geomorphic assessment 1km reach	• Identify ancient meanders and historical channel positions • Evaluate the HGM characteristics of each site

Hydrogeology	• Pressure transducers installed in piezometers (ground water level) • Boreholes (stratigraphy) of ancient meanders and floodplain	• Stratigraphic composition • Compare water level fluctuations with channel
Water temperature	• Longitudinal: Hoboware temperature loggers • Lateral: pressure transducers in piezometers record groundwater temperature	• Identify subsurface connections (groundwater inputs)
Photographic	• Reconyx hyperfire camera (every 30 min) facing ancient meander	• Snow cover and flood extents
Vegetation	• Vegetation units in floodplain • Dominant species and soil characteristics	• Determine presence and extent of wetlands
Fish habitat	• Qualitative habitat evaluation index (QHEI)	• Consider in channel habitat

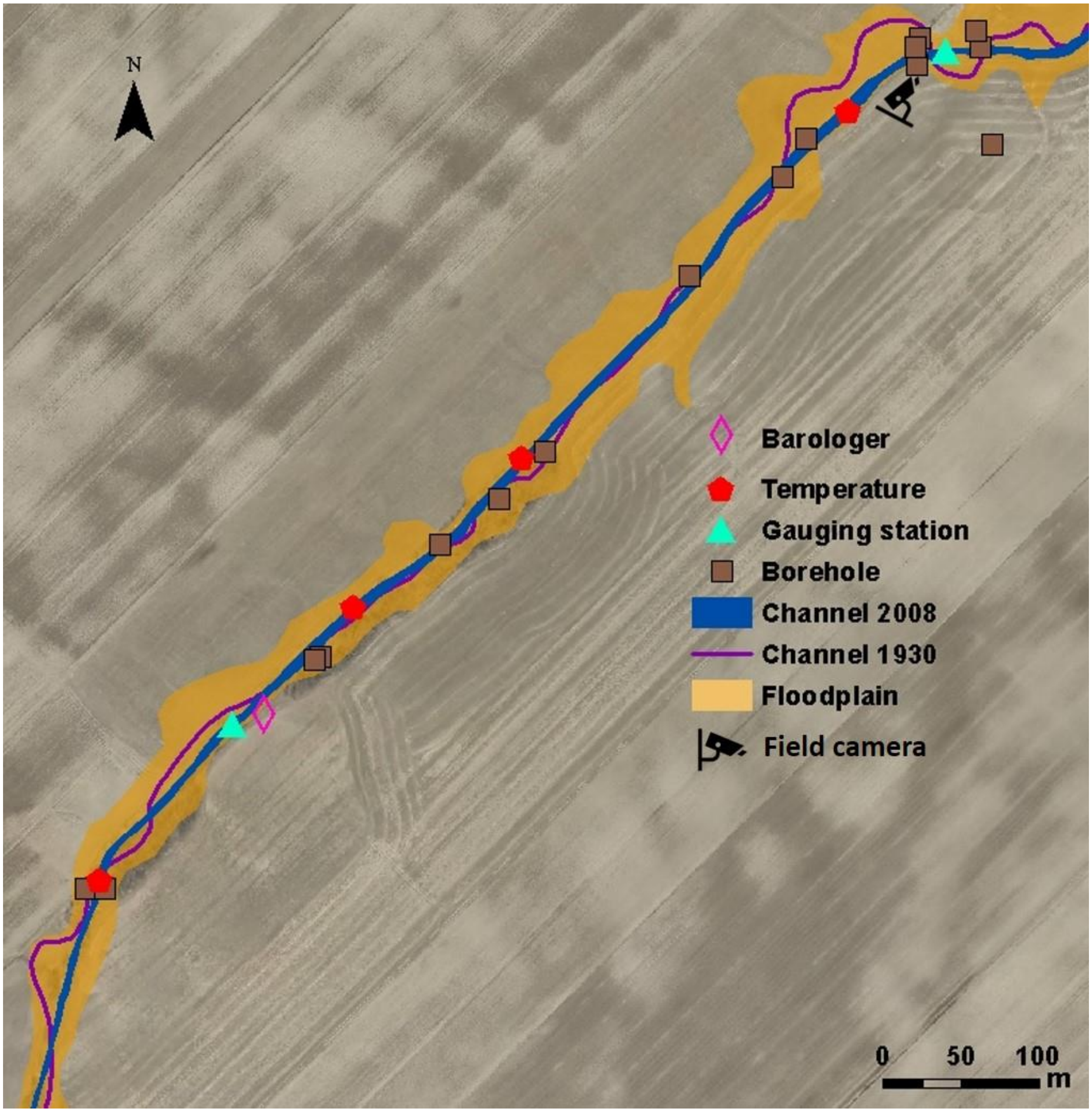


Figure 15: Summary of data collection and instrumentation for DF.

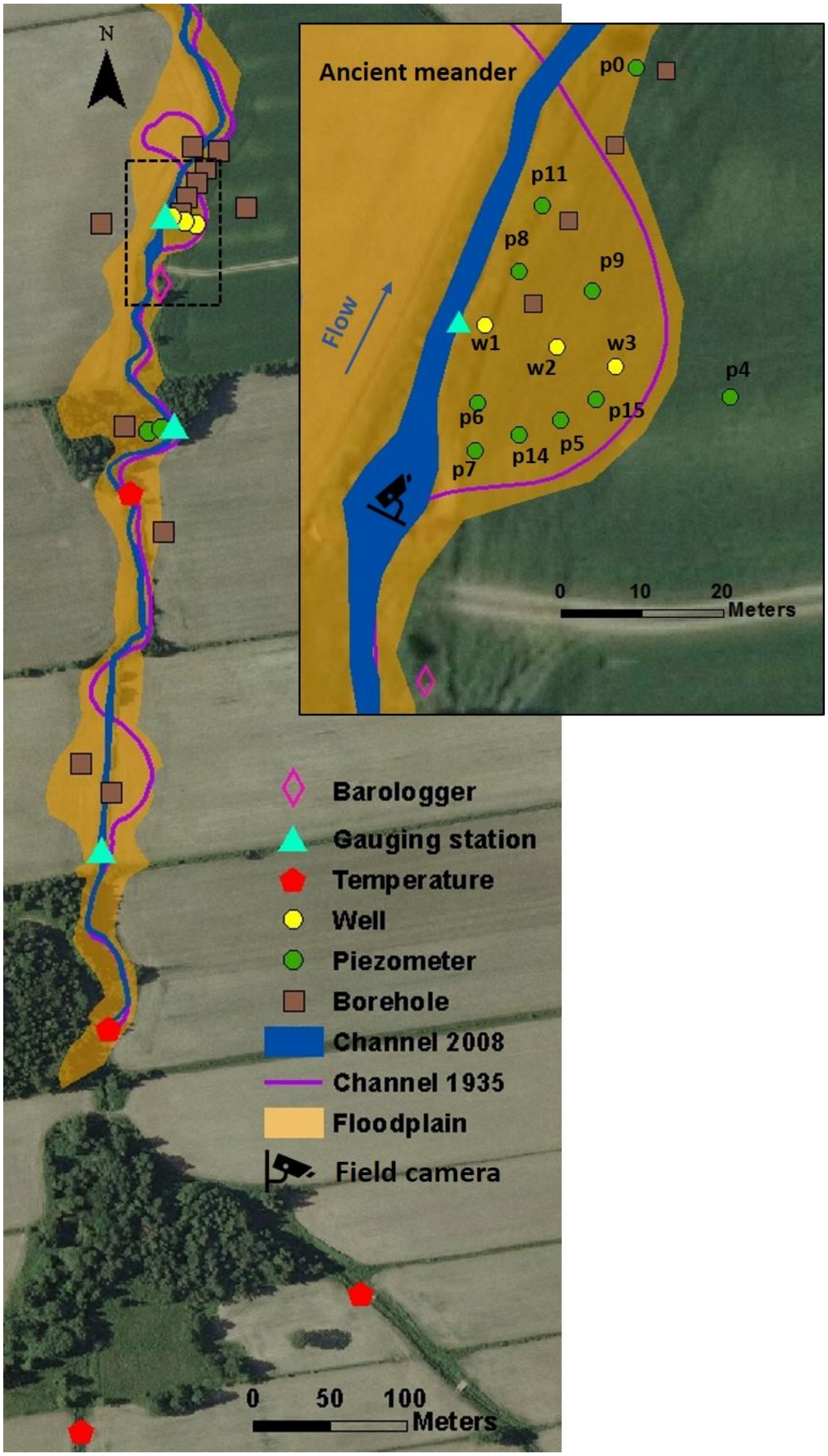


Figure 16: Summary of data collection and instrumentation for PPauB.

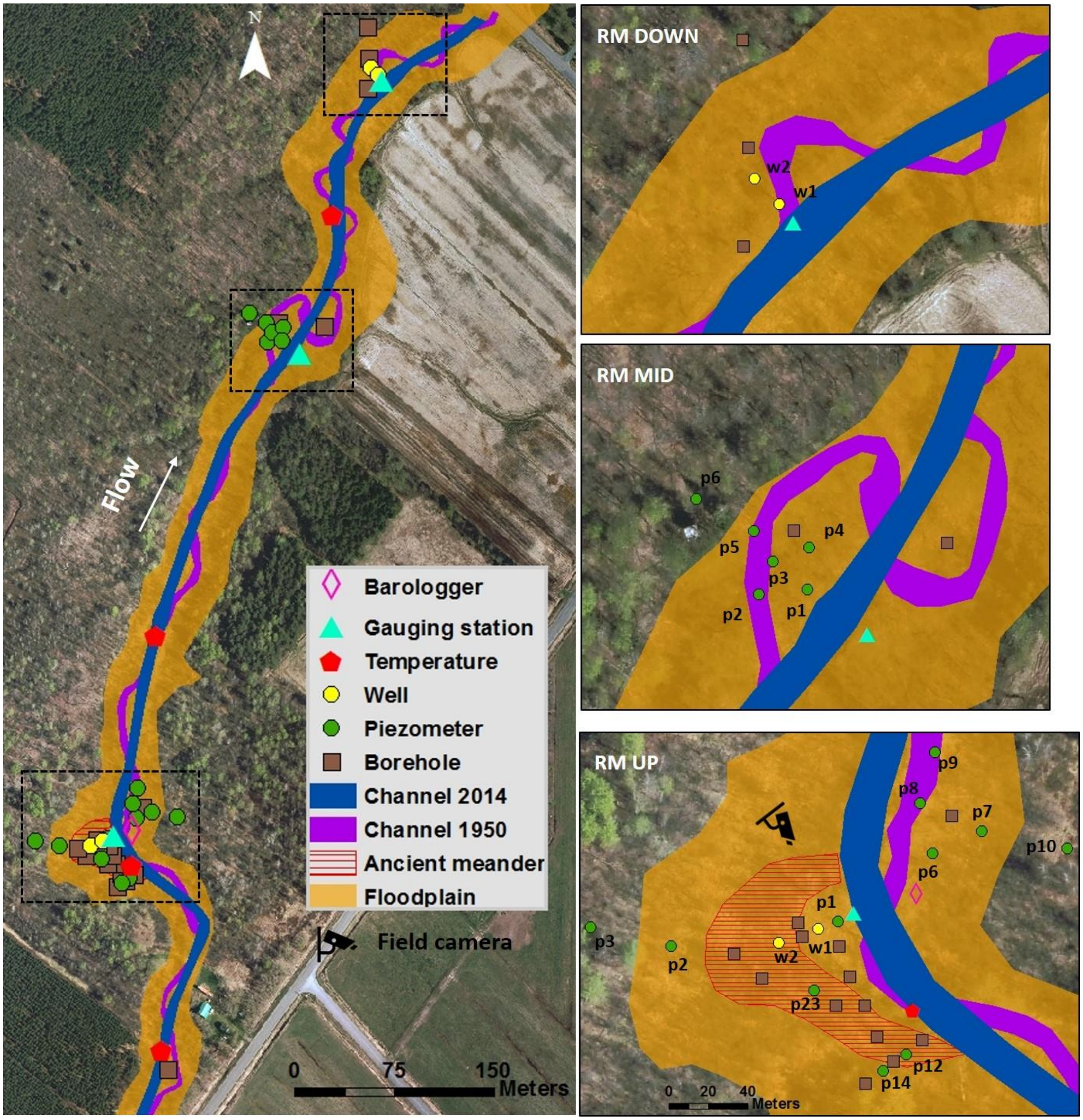


Figure 17: Summary of data collection and instrumentation for RM.

4.1.1 Geomorphic characteristics and topography of ancient meanders

A geomorphic assessment was performed over a 1km section of the channel for each site during early visits of the first year of data collection (2017). This assessment involved measures of channel geometry (wetted width and depth), grain size (bed and banks) while also noting signs of bank erosion, deposition (bars) and woody debris dynamics (accumulations and log jams).

Wetted width and depth were measured using measuring tape and a wading rod while grain size was evaluated visually. Measures were not taken at regular intervals but rather when clear changes in either channel planform geometry or bank and floodplain slope were observed. Other characteristics were evaluated upon occurrence, including approximate area coverage of log jams which was measured when possible.

This data was used to identify distinct reaches (as seen in section 4) but also aided in explaining differences in wetland restoration potential between sites as well as their potential to re-meander (section 6). For instance, areas of significant bank erosion originating from fluvial processes, presence of mid channel bars and woody debris typically indicate lateral channel migration.

A high density of topographic points was obtained using a Total Station (Leica TC805L) and georeferenced with a dGPS (differential GPS, model Spectra precision 80) (Figure 18). Measures were taken of the channel and floodplain in proximity to ancient meanders which had been selected for further study. This was done to spatially position field sensors (discussed in later sections) as well as to supplement the elevation data from the LiDAR.

Figure 18: Total station setup at DF A), and dGPS during a survey of RM B).

4.1.2 Survey of surface geology

Initially, soil samples (boreholes) were taken sparsely along the floodplain and channel bank when obvious changes in topography were observed. This was done in order to produce a map of surface geology for each of the three sites as well as to confirm the presence of ancient meanders through the identification of coarser alluvial deposits (representing bedload). In the second year (2018), samples were taken more intensively within the ancient meanders to better characterize their stratigraphic profiles and to identify the extent of alluvial bedload deposits.

The sampling was conducted using either an auger (Figure 19A) (83 mm ($3^{1/4}$in) diameter, 0.50m length) and up to 3 extensions of 1m each for a total potential borehole of 3.35m or a manual gas-powered drill (Pionjar 120) (Figure 19B) which produced samples of 51 mm (2 in) diameter and 1m length up to a maximum of 3m. The nature of sampling with the auger (twisting motion) made determining exact depths of samples less accurate than sampling with the Pionjar which removed long (1m) intact cores (Figure 19C, E). We accounted for this by frequently verifying depths with measuring tape, minimizing potential error. The Pionjar also had the advantage of allowing us to take samples of stratigraphic units that are below the water table, which was not be possible with the auger as it cannot pull up non-cohesive material (Figure 19D).

Figure 19: A) Borehole dug at DF using auger; B) borehole dug at PPauB using Pionjar; C) 1m core sample extracted by the Pionjar; D) soil sample extracted with the auger; E) soil sample obtained from Pionjar.

The general sampling procedure once the borehole locations were selected (combination of studying LiDAR, orthoimages and on-site investigation) was to dig at regular intervals depending on the instrument (auger ~ 0.35m, Pionjar ~1m). Samples were assessed visually and classified into distinct units (ex: regional deposits, bankfill till, alluvial (bar/bedload), floodplain soil) based on dominant grain size, color, humidity and presence of organic material. A sample of each stratigraphic unit was retained in a Ziploc bag and a depth measurement was recorded, paying particular attention to contact between units. Due to the disturbed nature of ancient meanders in agriculture settings many of the contact zones between layers were transitional (Figure 20A) rather than abrupt (Figure 20B, C). Sampling of each borehole continued until the basal unit was identified (Figure 20D) or an obstruction (boulder, log) was encountered.

Figure 20: A) example of a transition area between stratigraphic units from a sample extracted at PPauB; clear contact depth between units for a sample extracted at RM B) and PPauB C); D) basal unit of marine clay at DF.

The identification of stratigraphic units by visual assessment is subject to error of interpretation due to multiple assessors in the field as well as presence of transition sections in the samples. Samples collected in Ziploc bags allow for us to confirm field observations in the laboratory if necessary.

Post-processing of core sample data was done using the Stater 4 software (Golden Software Inc.) which displayed the depth and unit data for each borehole (Figure 21A) allowing cross-sections between boreholes to be established. These cross-sections were used to inform detailed conceptual models of ancient meander stratigraphy for each site (Figure 21B) which were further drawn from Adobe Illustrator software (Adobe Inc.).

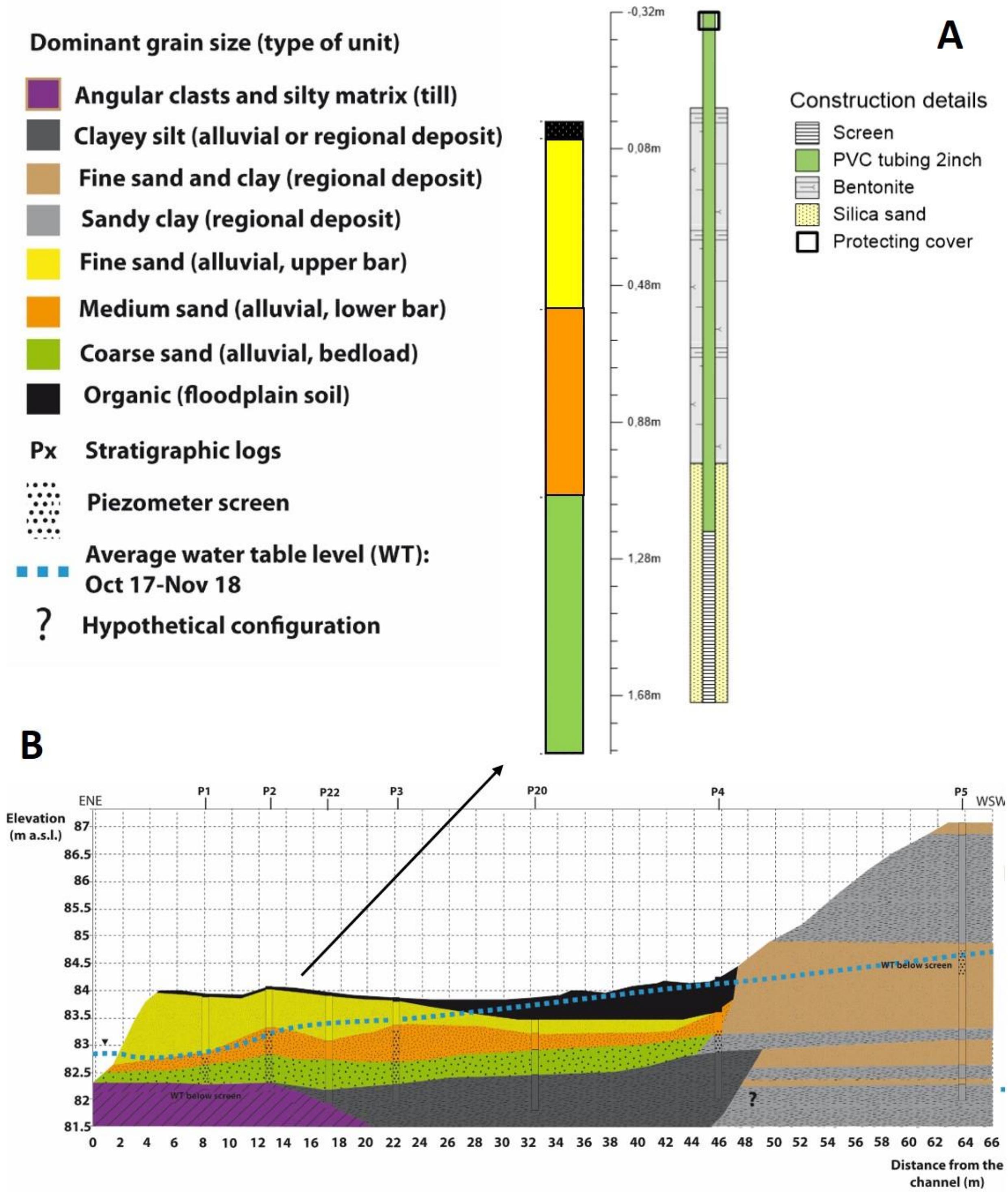


Figure 21: A) Example of borehole view layout for a stratigraphic log obtained at RM – P2; B) example of cross section view of an ancient meander at RM upstream.

4.1.3 Hydrology

Channel hydrology provides important information for understanding both surface and subsurface connections between the channel and floodplain, connections which are important for developing and maintaining wetlands (Larocque et al., 2016). We evaluated the hydrology of our sites in three main ways: 1) consulting past studies where variables such as drainage area and bankfull discharge had already been documented, allowing us to calculate unit stream power, 2) continuous measurements of flow stage through on-site instrumentation coupled with regular discharge measurements at a variety of flow stages to establish rating curves for each site (relationship between flow stage and discharge), and 3) installation of cameras to visually document the channels response to flooding events and validate flooding into ancient meanders (though surface processes, i.e. overbank flow).

Past studies:

The study sites were chosen in part because past studies were available. For DF we consulted Paradis (2015), for PPauB Roux (2012) and Michaud et al. (2014), and for RM Olsen and Buffin-Bélanger (2012). These documents also provided data which helped us verify/confirm our own calculated values (ex: bankfull width, bankfull discharge evaluated from our rating curves, slope of study reach).

Flow stage:

Continuous measures of flow stage were required at each site in order to establish rating curves with discharge measurements as well as to evaluate potential hydraulic connections between the channel, ancient meander/floodplain and regional deposits. This was accomplished by installing at least two pressure transducers for each site (RM had three installed), one in the channel cross-section adjacent to the ancient meander of interest and the other at the other end of the study area at least 500m distance streamwise. The exact positioning of logger installation was chosen based on field evaluation of cross-section stability (no signs of significant erosion, sediment accumulation and not directly upstream from a log jam) in order to ensure consistent channel dimensions throughout the study period. A total station was used to measure the channel cross-section dimensions as well as the adjacent floodplain to eventually interpret recorded flow stage data.

To protect the pressure transducers and to allow them to be easily readout a traditional L-pipe installation was performed (Figure 22A). The L-pipe consists of two ABS tubes with a diameter of 64 mm (2.5 in) cut to length appropriate for the dimensions of the particular bank it is to be installed in (Figure 22B). One end of the tube is placed into a cut excavated from the bank while the other end extends into the channel slightly above the bed. The submerged end of the tubing is protected by a screen and large rocks to both stabilize its perpendicular position and inhibit the deposition of fines (Figure 22C). The pressure transducer is attached via non-stretch cord and is positioned to rest on the bottom of the pipe connection to ensure consistent replacement after readouts.

Figure 22: ABS pipes allow for easier access to loggers during readout at DF A), example of L-piping to be installed at RM B), finished installation of L-pipe at RM C).

Pressure transducers of various models were used from devices available at UQAM, UQAR and Concordia University (Onset Hoboware (0-30ft) Titanium, Solinst Levelogger Gold,Junior and Edge model 3001). Beyond their respective readout devices and software user interfaces there is little difference in their output (accuracies ranging from +/- 0.3cm for the M5's and 1cm for the M20's). The pressure transducers were set to record at 30-min intervals (see Table 3 for specifics on data recorded). For each site a barologger (accuracy +/- 0.05kPa, 0.5cm) was installed with no more than a 10m difference in elevation from the pressure transducers in order to provide barometric compensation. A total of 7 pressure transducers were installed across the study site channels (2 per site with an extra at RM middle site which was added in year 2).

Discharge:

Repeated measures of discharge were taken at each field site over a range of flow stages (Table 3) in order to establish a stage/discharge relationship with our flow stage data. Discharge was measured using the typical cross-section methodology which involves summing the partial discharge calculated over panels established in the field (Figure 23). For DF and PPauB width, depth and average velocity were measured at 10cm intervals from bank to bank while the larger RM was done at 20cm intervals. Velocity was measured at 0.4 times the depth using a Swoffer (model 2100) propeller current meter (accuracy +/- 2cm/s).

Table 3: Summary of discharges measured at each site

Site	Date	Time	Discharge (m^3/s)	Channel stage (m)
DF	01/10/2017	12:00	0.013	34.37
	27/10/2017	13:15	0.018	34.37
	16/11/2017	11:00	0.025	34.39
	17/04/2018	8:30	0.245	34.63
	01/05/2018	16:00	0.240	34.83
	20/05/2018	9:00	0.033	34.38
PPauB	28/10/2017	10:00	0.002	14.79
	01/11/2017	10:30	0.137	15.24

	06/12/2017	11:00	0.389	15.6
	17/04/2018	15:00	1.359	15.82
	14/05/2018	10:30	0.043	14.99
	17/05/2018	11:00	0.025	14.93
	23/05/2018	9:40	0.014	14.91
RM	20/09/2017	18:00	0.011	82.72
	10/05/2017	19:30	0.011	82.75
	28/10/2017	13:00	0.571	83.09
	01/11/2017	13:30	0.528	83.06
	06/12/2017	16:17	0.923	83.18
	17/04/2018	12:00	1.729	83.34
	12/05/2018	15:30	0.136	82.85
	23/05/2018	15:30	0.098	82.81

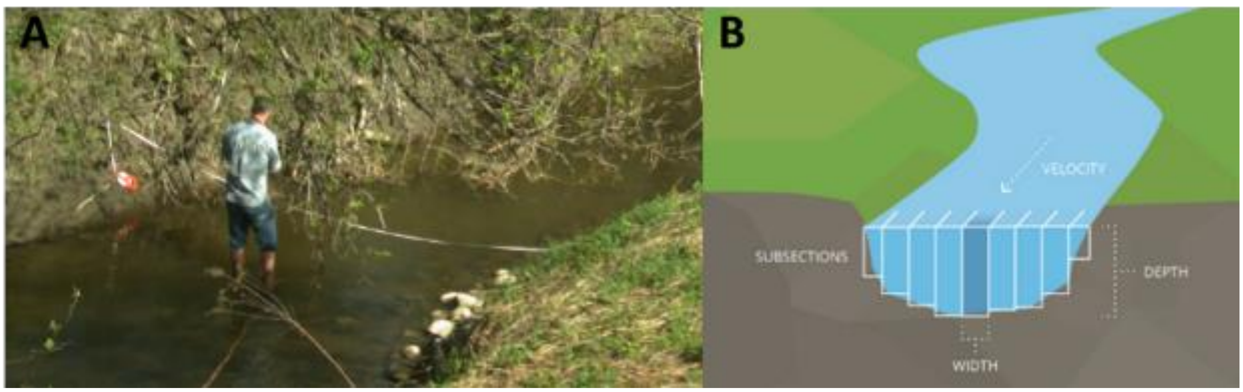


Figure 23: Example of discharge measurement taken at RM A), and illustration of panels B) (photo credit: Fondriest Environmental, Inc.).

Cameras:

Three cameras (Reconyx hyperfire H500) were installed, one for each site (Figure 24) facing the ancient meander and recording an image every 30 min with a field of view of 42° (Figure 25). The cameras proved very useful for visually documenting flooding events (spring floods) and validating potential surface connections with the ancient meander (Figure 26A). Cameras were also useful for identifying times of snow cover where logger data would not be useable (Figure 26B). Interestingly, even recording once every 30 min, our cameras caught a lot of wildlife (Figure 27A, B).

Figure 24: Camera installation at DF A), PPauB B), and RM C).

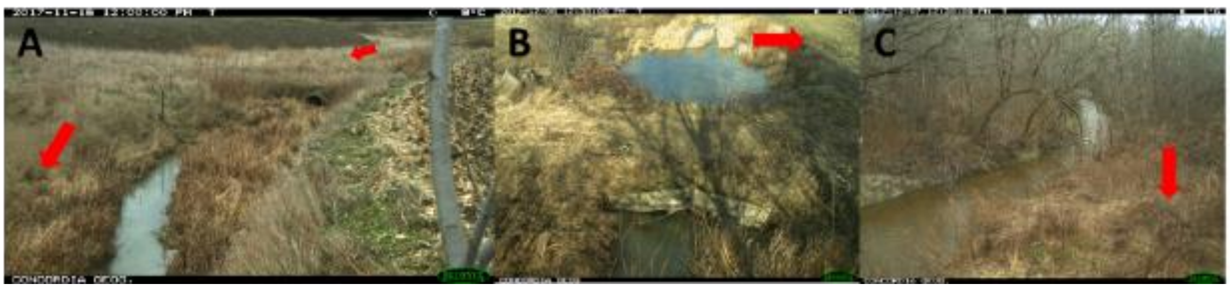


Figure 25: Examples of images captured from Reconyx cameras (red arrows denote ancient meanders) for DF A), PPauB B), and RM C).

Figure 26: A) Example of overbank flow into ancient meander during early spring flood at DF, B) photos such as this of PPauB in winter allow us to evaluate when snow cover may interfere with pressure transducer readings.

Figure 27: Examples of wildlife captured on camera. A) geese at DF, B) deer at RM.

4.1.4 Hydrogeology

In order to assess potential hydraulic connections (one of the main criteria for evaluating an areas' wetland restoration potential) between the channel, ancient meander and superficial deposits, quantitative information regarding the shallow groundwater hydrodynamics was required. Groundwater levels were continuously measured with pressure transducers (same models as described above) installed in a series of wells and piezometers (Figure 28). As we wished to compare channel water level and groundwater level responses to flooding events all loggers were set to record at 30-min intervals, measuring simultaneously.

Experimental design:

In year 1 (2017), wells were installed within the ancient meander and perpendicular to the channel level logger with the goal of providing a cross-section profile of the local water level. Three wells were installed at the PPauB meander while 2 wells were installed for each of the RM meanders for a total of 7 wells. No wells were installed at DF which, as assessed through field observations and soil samples, had a thick layer of clay sediments and therefore was unlikely to have measurable subsurface connections with the channel. At this site, hydrological connections between groundwater and channel for DF were evaluated differently (see section 4.1.5 Temperature and resurgence points). It should also be noted that despite the presence of fine gravel and sand in the alluvial plain, there is no surface aquifer at the PPauB site. Therefore, any hydrological connectivity between the river and sediments would be localised to the irregular pockets, accentuating their importance as potential restoration targets.

Year 2 (2018) brought a densification of water level measurements through the installation of many piezometers within the ancient meander and regional deposits, and both parallel and perpendicular to the channel in order to provide a more 3D perspective on groundwater levels (Figure 28). For each site effort was made through evaluation of LiDAR, historical photos and boreholes, to identify and install at least one piezometer within the coarse alluvial sediments of the ancient channel bed (essentially the pre-straightened position of the channel). A total of 17 piezometers were installed across the sites at RM and 14 were installed at PPauB for a total of 31. It should also be noted that a second site was established at PPauB in a small forested patch that was not straightened consisting of 2 piezometers and 1 channel L-piping. A third site was also added for RM between both upstream and downstream study sites in order to better characterise the ancient meanders present at the location (6 piezometers and 1 L-piping).

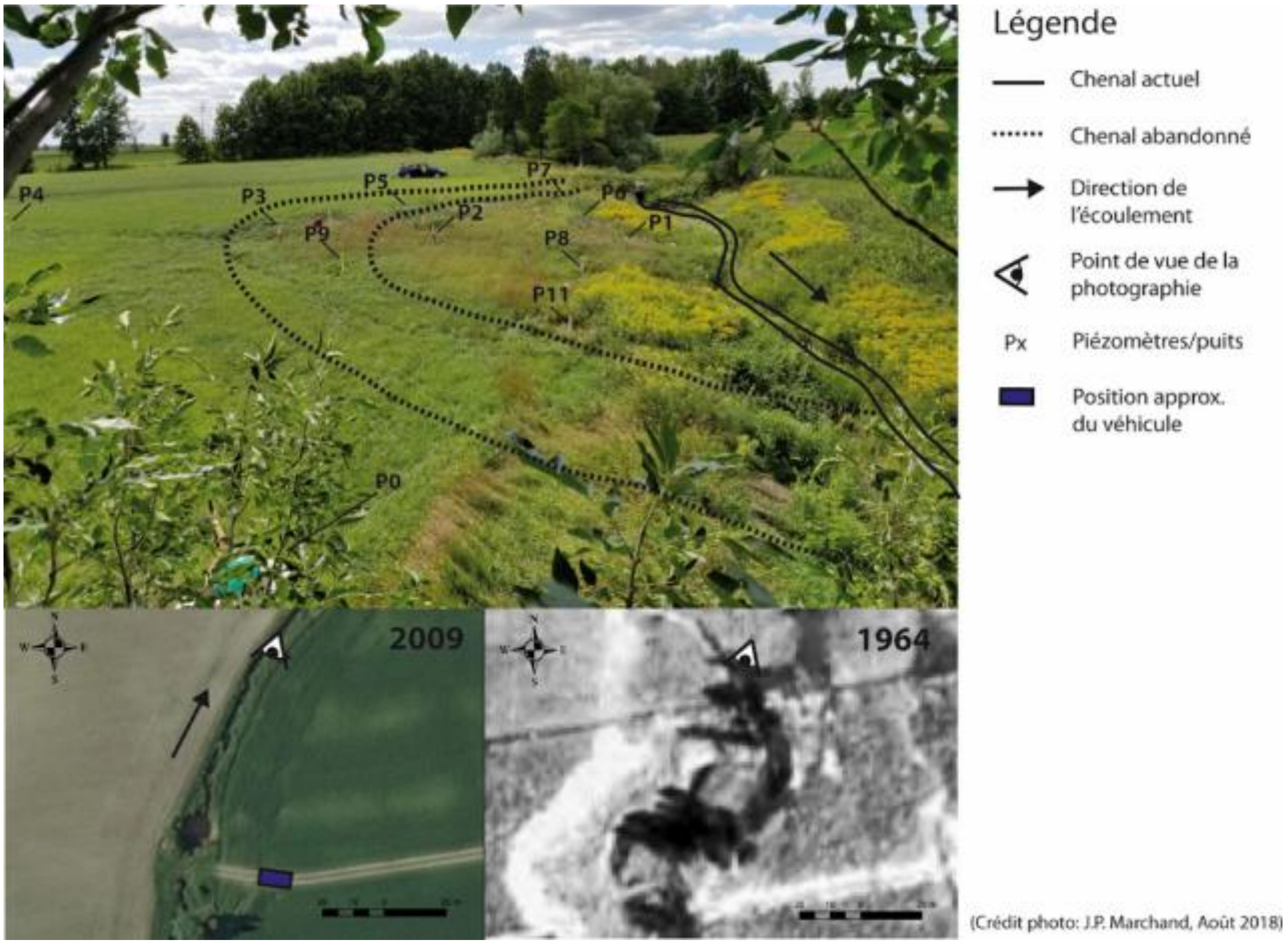


Figure 28: Example of setup at PPauB

Differences between wells and piezometers:

Several differences exist in the construction designs between wells and piezometers (Figure 29) affecting what the groundwater level being measured represents. For wells, the perforations start at the base of the pipe and typically extend to near ground surface. Therefore, the water level measured inside the pipe is actually the average water pressure along the entire length of the pipe and is the result of the partial contribution of the various stratigraphic units present, which may or may not have different values of conductivity. Piezometers on the other hand are only perforated at a narrow band at the bottom of the pipe so that water level in the pipe measures the pressure of a particular stratigraphic unit of interest, which in our case is the coarse alluvial unit of the ancient channel beds. A bentonite seal (impermeable) is created around the outside of the pipe just above the perforated section to restrict percolation from the ground surface as well as the other stratigraphic units.

We installed wells in year one (essentially recording water table) as we did not yet have sufficient details on stratigraphy to install piezometers. In year two, once our stratigraphic survey was completed, we added piezometers to better capture the response of the ancient channel bed. We do not expect much difference in measured water level between the installed wells or piezometers as the layer we are interested in is by far the coarsest material in the stratigraphic profile with much higher values of hydraulic conductivity (theoretically).

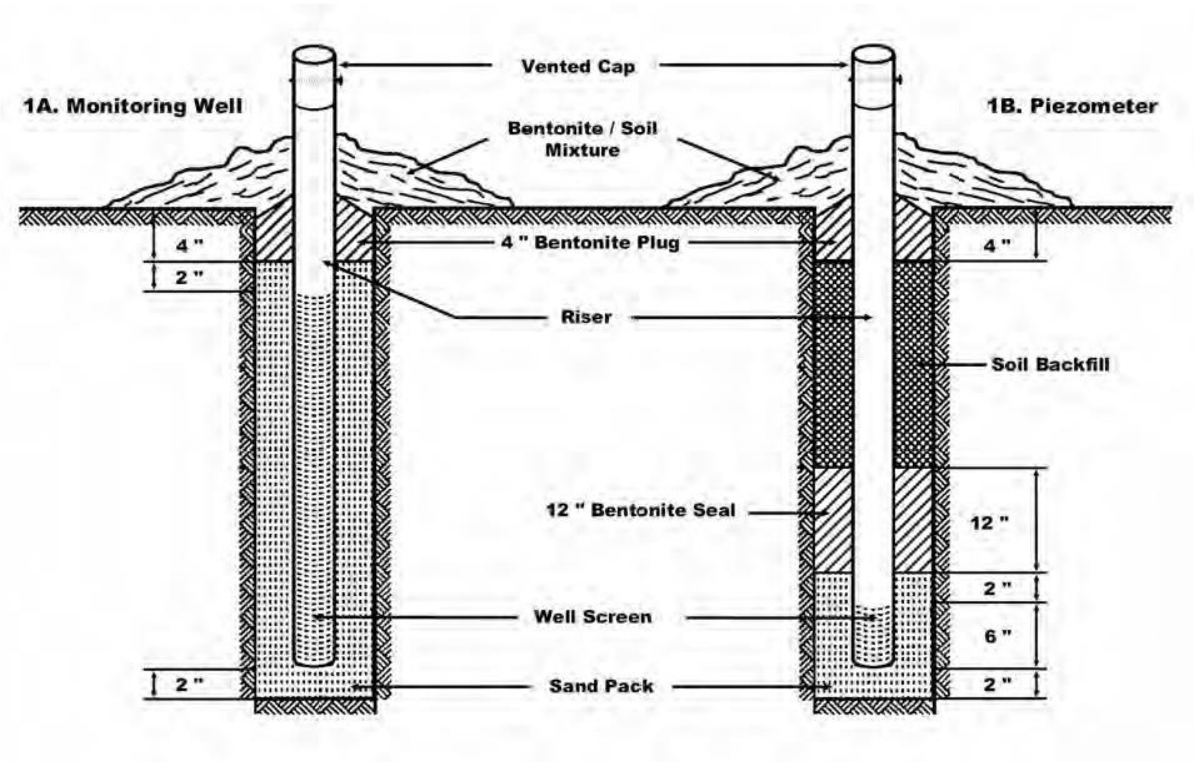


Figure 29: Schematic diagram of installed monitoring well A) and piezometer B). From Washington State Department of Transportation, www.wsdot.wa.gov/sites/default/files/2017/07/24/Env-Wet-InstallMonWellsPiezometers.pdf. Note: the values of depth and width shown here are to provide an example, actual installation values are site specific depending on the depth of the basal unit (see below).

Construction of wells:

Figure 29A) shows the general concept of a well installation. Our installation process in the field was as follows:

1. Dig a borehole (well and piezometer installations were always combined with collection of stratigraphic information, see 5.2.2 survey of surface geology) with manual auger. Borehole width was consistently 83 mm ($3^{1/4}$in) in diameter with a maximum depth of 0.30m below the water table. Due to limitations of the auger which could not bring up very saturated non-cohesive material we could not go further below the water table. However, wells were installed under late summer drought conditions and it was therefore not expected for the water table to lower more than installation levels over the course of the study period.
2. PVC piping with a 51 mm (2 in) diameter was then inserted into the borehole. All wells were a total length of 3m consisting of a 1.5m perforated section, capped 5cm off the bottom of the borehole, and a 1.5m length non-perforated section extending above ground. Perforated sections of PVC (and most of the other components) were obtained from PlastechPlus, a company specializing in well/piezometer materials (Figure 30A). These perforated sections were designed for environments with heavy clay content although we did provide extra protection via socks (Figure 30B) to further prevent infiltration of fine material without obstructing flow.

3. Filling around the well (space between the PVC and borehole) consisted of coarse sand (Figure 30C) for the majority of pipe length with a bentonite cap at the top 50cm (Figure 30D). The bentonite cap included a concave mound around the ground/borehole interface to prevent infiltration and discourage pooling.
4. The top of the well was then capped with air holes drilled on the sides of the cap to allow air pressure to equalize but prevent entry of rain water. Pressure transducers were then installed hanging from the cap using non-stretched string (error tested in the lab to add a max of 1cm) to the base of the well (Figure 30E).

Construction of piezometers:

Figure 29B shows the general concept of a piezometer installation. Our installation process in the field was as follows:

1. Boreholes were dug through a combination of Pionjar to go well below the water table if necessary and auger to expand the borehole where needed. Maximum depth of the piezometer installation was determined by the depth of the basal unit which was in turn determined through visual assessment of the stratigraphy. The base of the piezometer is always at the contact layer of the basal unit or just below it, although care is taken to ensure that the narrow, perforated section is always in contact with the stratigraphic unit of interest.
2. The diameter of PVC piping used for installation was variable (25 - 64 mm (1 – 2.5 in)), and the perforated section was consistently 40cm in length. Total length of the piezometers is variable depending on the depth of the basal unit and materials available.
3. Filling around the piezometer involved coarse sand extending from the base to just above the perforated section. The space above the perforated section is then capped with bentonite over a length of 30-50cm. The rest of the borehole length is again filled with coarse sand or compact material excavated when digging the borehole. Again a bentonite capping is performed at the top 50cm of the borehole similar to the process use in the well installations.
4. Instrumentation followed the same process as the well installations.

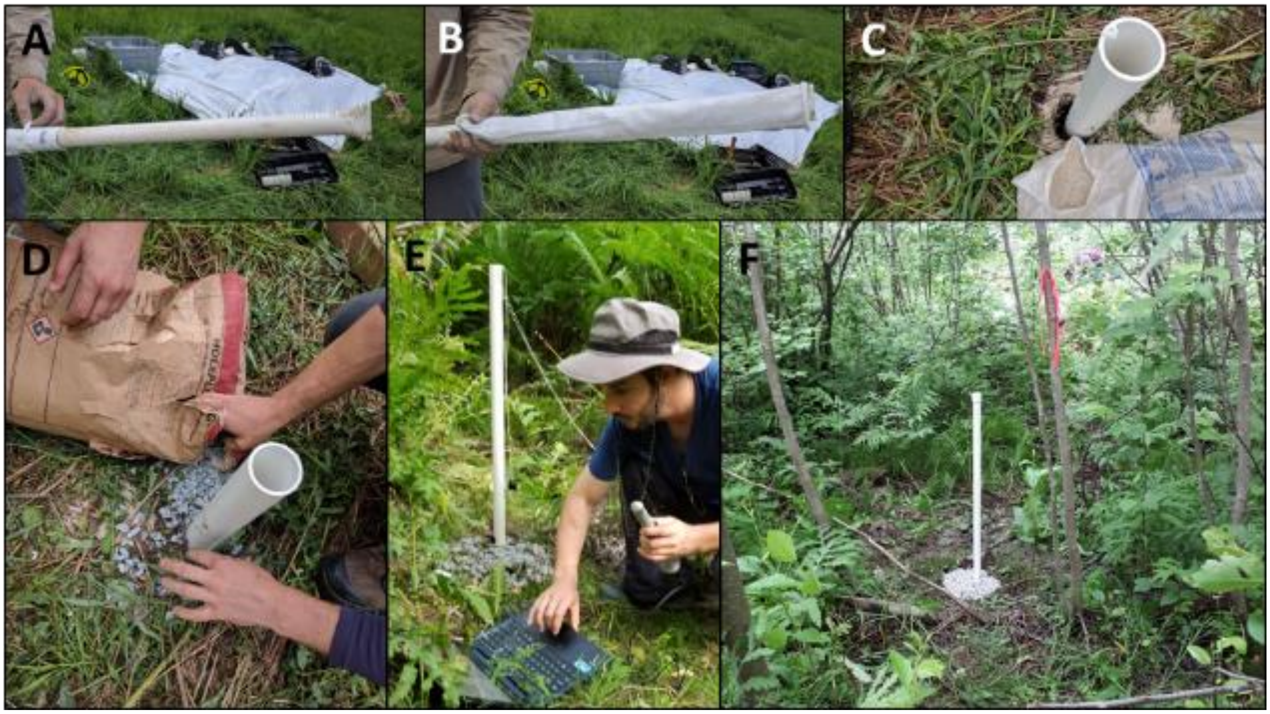


Figure 30: Example of piezometer installation. Perforated section A), sock used to prevent infiltration of fines B), addition of coarse sand C), bentonite D), logger setup E) and finished installation, P4 at RM mid site F).

Measurements of hydraulic conductivity:

A series of slug tests were performed a month after each piezometer was installed. The general procedure was to add a known volume of water (slug) to the piezometer which corresponded to a particular height of water added (volume depending on the dimensions of the PVC pipe). Water level within the pipe was then recorded at 1-sec intervals over a 6-hr period, when the level returned to pre-slug values. Approximation of the hydraulic conductivity (K) based on the slug test results was done using the Hvorslev method. Unfortunately, approximated values were difficult to interpret due to limitations of our slug tests which were performed at the end of the summer when water tables were low and the perforated sections were not completely saturated. As a result, much of the water level response involved the “filling up” of the space around the PVC, i.e. into the borehole, rather than capturing the full response of the alluvial unit of interest.

4.1.5 Temperature and resurgence points

In addition to the piezometer installations at ancient meanders, we wanted to identify alternative groundwater/channel connections throughout the study sites. We therefore measured channel temperature longitudinally with 4 – 5 temperature sensors (Hoboware 64MB) installed in the channel bed in order to see if we could pick up a temperature signal of cooler groundwater entering into the warmer channel. The layout of sensors also allowed us to look at the longitudinal and seasonal variability in channel water temperature. The potential effect of shading was accounted for by installing two sensors at each site outside of the channel measuring air temperature, one in full sun and the other in full to partial shade.

Channel loggers were installed using re-bar and zip-ties or cement blocks and cord. Loggers remained in the channel recording temperature every 30 minutes from September to early December in year one and from mid May to mid November in year two.

As we had decided not to install piezometers at DF due to the high clay content of the floodplain material, temperature information and determining alternate potential sources of groundwater/surface water connection was deemed important at this site. Upon initial inspection of the site two groundwater resurgence points were identified (Figure 31A). Water samples collected just above the stream bed 5m upstream, 5m downstream, and directly on top of the resurgence points were tested for radon in the lab. Results of radon tests confirmed that there is indeed groundwater exfiltration into the channel at these points. Visual inspection of the LiDAR generated DEM for the DF area revealed the possible presence of a drumlin (Figure 31B) which may explain the groundwater resurgence points in a region otherwise dominated by clay. Consultation of the most recent report of quaternary geology for the Châteauguay River drainage basin (Tremblay, 2008) revealed that there is a strong presence of drumlins (mix of coarse and fine glacial till) near our study area. This was confirmed in the field where many deposits of cobble sized till were observed (Figure 31C).

Temperature sensors at DF were strategically placed to determine the influence of the groundwater resurgence points on overall channel temperature (mean daily maximum, minimum and average) as well as potential seasonal effects (cooling in hot summer months and warming in cooler fall months).

Note that all pressure transducers in L-pipe, well and piezometer installations also record water temperature at 30-min intervals. In the case of the wells and piezometers, this is our best approximation of groundwater temperatures.

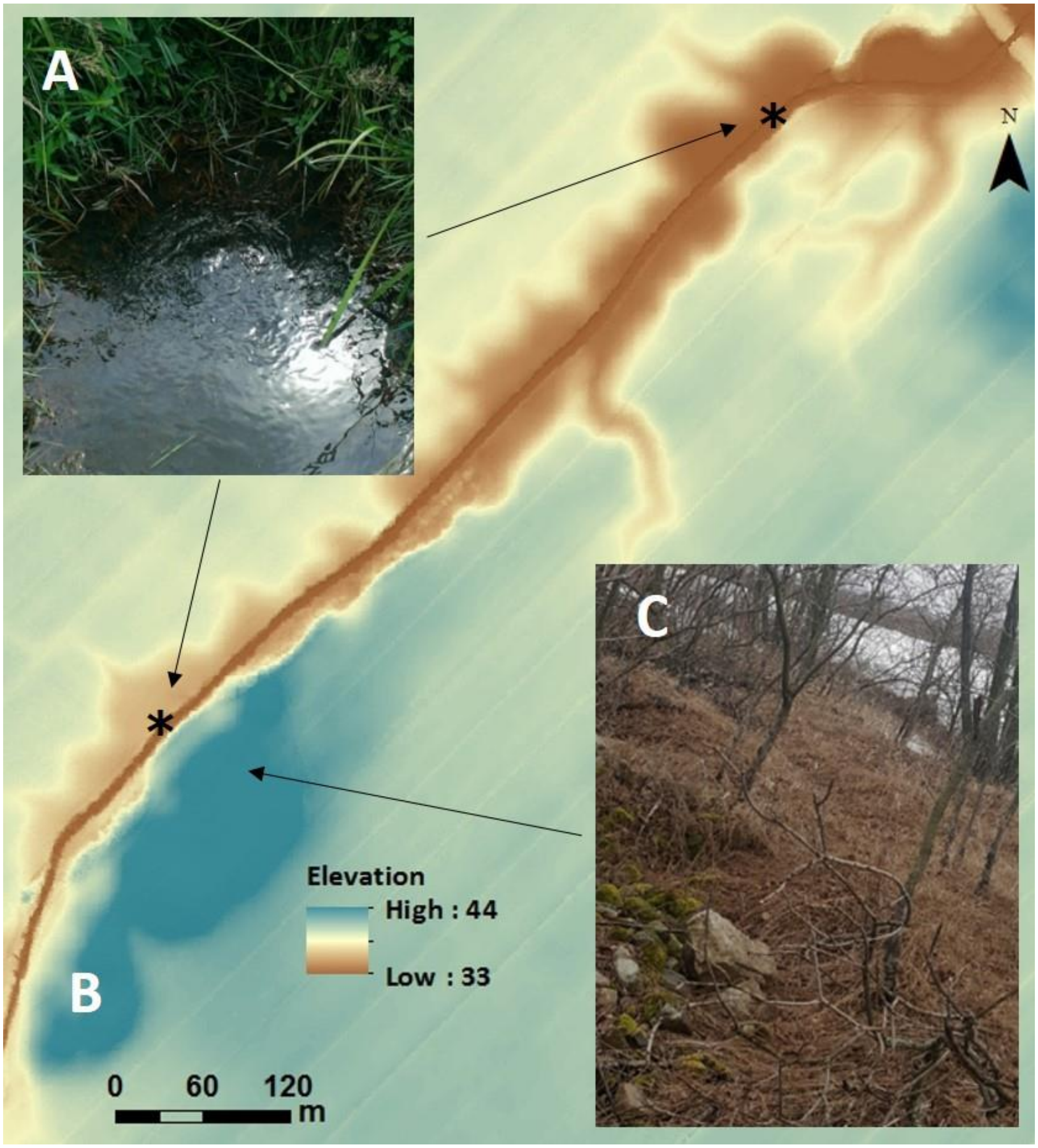


Figure 31: Example of groundwater resurgence point at DF A), drumlin clearly visible in the LiDAR B), cobble sized glacial till on floodplain C).

4.1.6 Vegetation survey

A vegetation survey of the three study sites was conducted during the mid summer period in 2018. This survey was conducted in order to identify existing wetlands as well as to identify signs in terms of species composition and soil characteristics that represent favorable conditions for potential wetlands to develop. For the full report that was produced and more detailed methodology please refer to Appendix 1.

In general, the methodology consisted of delimiting homogeneous vegetation units across the 1km study reaches for the three sites. Vegetation was assessed from the banks of the channel to the top of the adjacent floodplain talus. Vegetation units are defined as the portion of land dominated by one or several of the same species and differing in composition from neighbouring units. Distinction of vegetation units are therefore generally based on the dominant species in each strata present (herbaceous, shrubs, trees). A pedon was also dug within each unit to determine the nature of the soil (sandy, organic, silty etc.), its characteristics (humidity, texture, color) and the presence of speckles (mouchetures) which are a strong indicator of past/present wetlands (Figure 32B). The combination of vegetation inventory and evaluation of soil permitted the classification of units into one of 6 main categories: Meadow, Swamp, Shrub, Forest, Shrub Swamp and Woodland Swamp.

Figure 32: Photo of a homogeneous unit at PPauB A) and a pedon at the same site B) (photo credit: Audréanne Loiselle, 2018).

4.2 Data analysis

4.2.1 Post treatment of logger data and determination of significant rain events

Post treatment of logger data was straightforward involving barometric compensation with a barologger that was installed nearby to the respective pressure transducer and within 10m difference in elevation. Unusable data was then identified (winter periods where channel was frozen over and time where water level was below the sensor) as well as missing data due to instrument readout or slug tests when relevant. Information regarding each pressure transducer can be found in Table 4. Since the wells were installed during year 1 they have the longest period of data recorded (1.1 years) compared to the piezometers which were installed in the summer and/or fall of year 2 (0.4 to 0.2 years).

Table 4: Summary table of sampling info for gauging stations and wells/piezometers

Site	Instrument	Sampling dates		Sampling period (yr)	Time stream frozen over (yr)	Time water level below sensor (yr)	Useable data (%)	Sensor elevation (m)	Sensor depth (m)
		From	To						
DF	Gauging station downstream	2/10/2017	10/11/2018	1.11	0.10	0.00	91.2	34.24	N/A

	Gauging station upstream	2/10/2017	10/11/2018	1.11	0.10	0.00	91.4	35.23	N/A
PPauB	Gauging station downstream	3/10/2017	12/11/2018	1.11	0.23	0.41	42.0	14.8	N/A
	Gauging station upstream	3/10/2017	12/11/2018	1.11	0.23	0.20	61.3	16.04	N/A
	Well 1	3/10/2017	12/11/2018	1.11	0.23	0.30	52.1	14.57	1.63
	Well 2	3/10/2017	12/11/2018	1.11	0.23	0.19	62.2	14.2	2.45
	Well 3	3/10/2017	12/11/2018	1.11	0.23	0.29	53.2	14.55	2.43
	Piezometer 4	12/6/2018	12/11/2018	0.42	0.00	0.30	28.5	14.99	2.77
	Piezometer 5	12/6/2018	12/11/2018	0.42	0.00	0.23	45.5	14.42	2.51
	Piezometer 6	12/6/2018	12/11/2018	0.42	0.00	0.21	49.6	14.47	1.84
	Piezometer 7	12/6/2018	12/11/2018	0.42	0.00	0.01	97.8	14.05	2.56
	Piezometer 8	12/6/2018	12/11/2018	0.42	0.00	0.24	42.3	14.55	1.59
	Piezometer 9	12/6/2018	12/11/2018	0.17	0.00	0.13	23.5	14.69	2.06
	Piezometer 11	12/6/2018	12/11/2018	0.42	0.00	0.32	24.6	14.88	1.36
	Piezometer 0	12/6/2018	12/11/2018	0.42	0.00	0.17	60.6	13.53	3.34
	Piezometer 14	16/8/2018	12/11/2018	0.24	0.00	0.17	29.5	14.32	2.37
	Piezometer 15	16/8/2018	12/11/2018	0.24	0.00	0.22	9.4	14.76	2.25
	Gauging station forested	18/7/2018	12/11/2018	0.32	0.00	0.20	37.5	15.86	N/A
	Piezometer 1 forested	18/7/2018	12/11/2018	0.32	0.00	0.19	40.7	14.81	1.61
	Piezometer 2 forested	18/7/2018	12/11/2018	0.32	0.00	0.21	33.2	14.85	2
RM	Gauging station downstream	5/10/2017	12/11/2018	1.10	0.12	0.00	89.3	80.99	N/A
	Gauging station midstream	29/06/2018	12/11/2018	0.37	0.00	0.00	100.0	81.2	N/A
	Gauging station upstream	05/10/2017	12/11/2018	1.10	0.12	0.00	89.6	82.63	N/A
	Well 1 downstream	5/10/2017	12/11/2018	1.10	0.12	0.00	89.3	81.53	1.62
	Well 2 downstream	5/10/2017	12/11/2018	1.10	0.12	0.00	89.3	81.45	1.19
	Piezometer 1 midstream	29/06/2018	12/11/2018	0.37	0.00	0.00	99.1	81.5	1.91
	Piezometer 2 midstream	29/06/2018	12/11/2018	0.37	0.00	0.02	94.5	81.76	1.19
	Piezometer 3 midstream	29/06/2018	12/11/2018	0.37	0.00	0.06	84.3	81.82	1.01
	Piezometer 4 midstream	29/06/2018	12/11/2018	0.37	0.00	0.00	99.1	81.51	1.15
	Piezometer 5 midstream	29/06/2018	12/11/2018	0.37	0.00	0.00	99.1	81.69	1.83
	Piezometer 6 midstream	29/06/2018	12/11/2018	0.37	0.00	0.21	43.1	83.3	3.39
	Well 1 upstream	05/10/2017	12/11/2018	1.10	0.12	0.00	89.6	82.66	1.46

	Well 2 upstrean	05/10/2017	12/11/2018	1.10	0.12	0.00	89.6	82.66	1.26
	Piezometer 1 upstream	22/06/2018	12/11/2018	0.39	0.00	0.00	99.1	82.37	1.59
	Piezometer 2 upstream	22/06/2018	12/11/2018	0.39	0.00	0.00	99.1	82.95	1.31
	Piezometer 3 upstream	22/06/2018	12/11/2018	0.39	0.00	0.01	98.4	84.25	2.9
	Piezometer 6 upstream	14/08/2018	12/11/2018	0.25	0.00	0.20	19.4	82.8	1.35
	Piezometer 7 upstream	14/08/2018	12/11/2018	0.25	0.00	0.15	38.0	83.12	0.9
	Piezometer 8 upstream	14/08/2018	12/11/2018	0.25	0.00	0.22	10.8	82.91	1.3
	Piezometer 9 upstream	14/08/2018	12/11/2018	0.25	0.00	0.19	21.4	82.7	1.35
	Piezometer 10 upstream	14/08/2018	12/11/2018	0.25	0.00	0.25	0.0	84.36	2.4
	Piezometer 12 upstream	28/09/2018	12/11/2018	0.12	0.00	0.04	65.0	83	1.11
	Piezometer 14 upstream	28/09/2018	12/11/2018	0.12	0.00	0.03	73.6	82.48	1.8
	Piezometer 23 upstream	28/09/2018	12/11/2018	0.12	0.00	0.00	100.0	82.01	1.95

Significant rain events were then determined through interpretation of the hydraulic response in the channel. Due to the effect of ice and snow cover on ground, rain events during the winter period Dec 10, 2017 to April 1, 2018 were excluded. The following criteria had to be met for an event to be considered significant:

Table 5: Criteria used for determining significant rain events

Site	Change in river stage (cm)	Minimum duration of event (h)	Minimum rising limb (h)
DF	>=15	>=12	>=2
PPauB	>=20	>=24	>=2
RM	>=10	>=36	>=4

These minimum values of change in river stage, minimum duration of event and rising limb duration (Table 5) were chosen based on evaluation of water level time series plots for the respective sites. Differences in cross-section geometry at the gauging station and drainage area between the sites were taken into account when deciding on final threshold values. Significant rain events were evaluated over the same time period for each site October 2, 2017 to November 11, 2018.

4.2.2 Cross-correlation analysis

Hydrological connectivity consists of water-mediated fluxes within and between the different components of the fluvial system (Amoros and Roux, 1988). It consists of both surface and subsurface processes, including exchanges between upstream-downstream (longitudinal), stream-floodplain (lateral) and surface water-groundwater (vertical). Several recent studies highlight that surface water-groundwater exchanges in particular act as a keystone for many stream's ecological functions such as denitrification (Vidon et al, 2010) and runoff management (Larocque et al., 2016). Indeed hydrological connectivity is an important consideration for riparian wetland restoration as water delivered to these floodplain ecosystems may facilitate the flux of organisms and nutrients while also increasing habitat and resource access (Reid et al., 2015). For example, a recent study on the MacIntyre River in Australia found that plant abundance in wetlands strongly depended on the availability of waterlogged soil and required hydrological connections and the resultant delivery of water for germination cues (Reid et al., 2015).

Defining the level of hydrological connectivity within a specific stream however, remains a difficult task as it requires the understanding and integration of a diverse and complex distribution of processes operating at different spatial and temporal scales. Furthermore, a lot of information is missing to fully understand and characterize the processes of subsurface flux exchanges between streams and riparian wetlands (Larocque et al., 2016). For this project, we focused on abandoned meanders which are known to be important zones of lateral hydrological connectivity that potentially facilitate exchanges between the coarse material in the floodplain and the channel, and may thus be priority restoration areas (Philips, 2013). We hypothesize that even artificially abandoned meanders in agricultural areas can be wetland restoration hotspots if processes of hydrological connectivity are maintained or can be rehabilitated.

The hydrological connectivity of ancient meanders at PPauB and RM was evaluated via cross-correlation analyses (signal processing using the PAST software, Hammer et al., 2001) between the channel stage and groundwater level time series data collected. Previous studies have used cross-correlation to characterize the intensity of the relationship and the time lag between stage and groundwater peaks (Larocque et al., 1998; Vidon 2012; Cloutier et al., 2014; Buffin-Bélanger et al., 2016; Larocque et al., 2016). We hypothesize that the more intense the relationship between channel stage and groundwater levels and the shorter the lag times between peaks, the greater the potential for hydrological connections. It should be noted here that we insert the word potential as these types of analyses are based on hydraulic pressure gradients, which only infer energy exchanges.

Although cross-correlation analysis may be applied to the entire time-series (Larocque et al., 2016) we found that event-based applications for individual flood events (Buffin-Bélanger et al., 2016) were more applicable for our purposes due to the small spatial and temporal scale of the project. We therefore determined the lag time between the channel stage and groundwater level peaks for the significant rain events identified at PPauB and RM by extracting the time at which the maximum correlation coefficient (r_{xy}) was obtained. This process was repeated for all piezometers. In year 1 our experimental design consisted of well installations perpendicular to channel flow while year 2 piezometer densification permitted a more complex perspective of the spatial distribution of channel/groundwater peak lag times within an ancient meander. Results of cross-correlation were then graphed against other variables (average depth of water table, depth of the alluvial unit, and rain event metrics: duration, time to peak, amplitude) in an attempt to explain the variability within (the range of lag times observed at a particular piezometer) and between (differences in lag times for a particular rain event) piezometers.

4.3 Results

4.3.1 Significant rain events and water level time series

Based on our selection criteria the greatest number of significant rain events was identified at RM (23 events) compared to PPauB (12 events) and DF (9 events) (Figure 33 – A). Comparing the median values of event characteristics reveals that overall responses at DF are small (Figure 33 – B), the total duration of events at PPauB is longer (Figure 33 – C) and there is a much higher time to peak at PPauB (Figure 33 – D). Many natural reasons exist which could explain the differences between the relative responses at each site (ex: drainage area, hydrological connections, development of riparian vegetation, channel geometry, valley slope, topography etc.) but anthropogenic perturbation is likely a major factor here (ex: narrow culverts and dense network of drainage ditches et PPauB in particular). Furthermore, complex rain events (back to back events where the channel was responding to one when another event would begin) were common in the fall months for both PPauB and DF, reducing the total number of significant rain events we were able to identify at these sites.

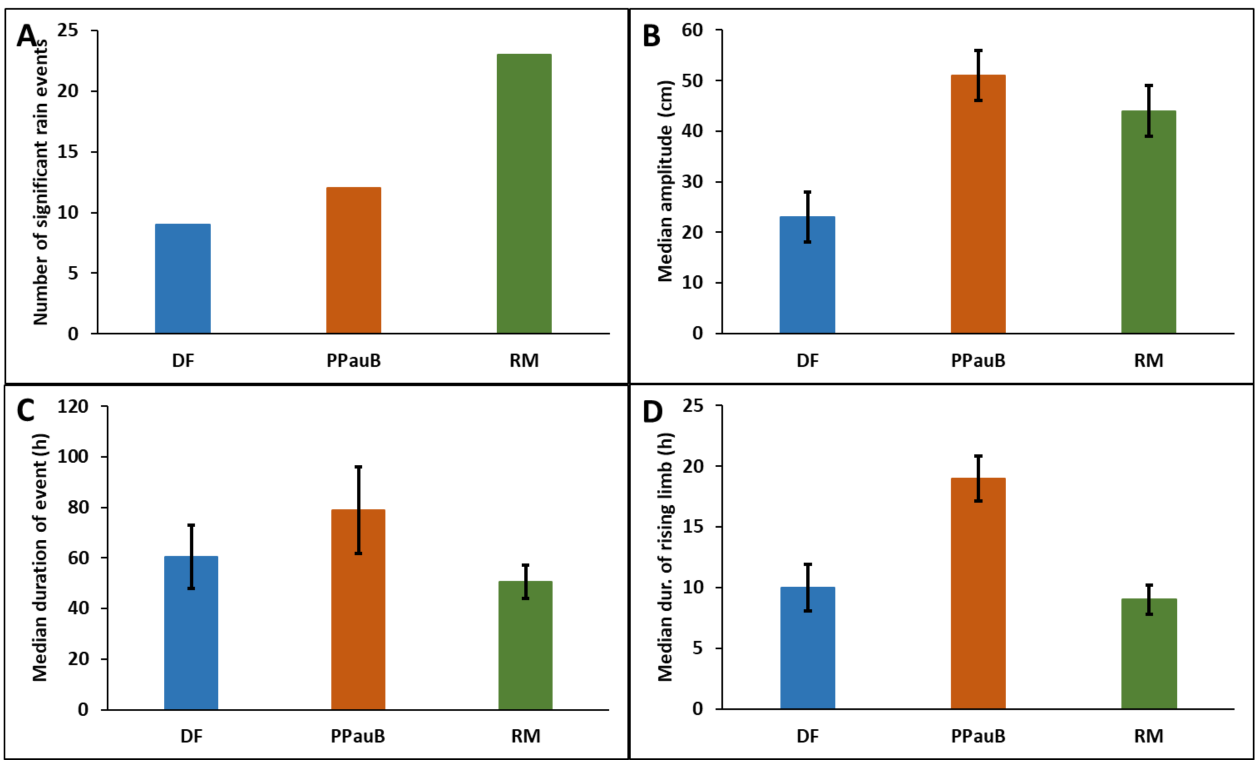


Figure 33: Comparison of channel response to significant rain events between the three sites. A) total number of events from October 2017 to November 2018; B) Median amplitude (initial stage to peak); C) Median duration of event (time to return to baseflow conditions or another event started); D) Median duration of the rising limb (time to peak water level in channel).

The following figures (34 to 41) showcase the channel stage and groundwater levels collected from our series of wells and piezometers at each site. Data recorded is limited for PPauB (Figure 35) due to a very low water table, which fell below the sensor depths for most of the summer and early fall months. Data is also limited for our site at RM upstream right bank (Figure 41) where the water table is below the sensor for most of the time series, only responding during peak fall floods (regional piezometer 10 did not respond at all). In the case of RM upstream right bank the presence of till made it impossible to install the piezometers any lower.

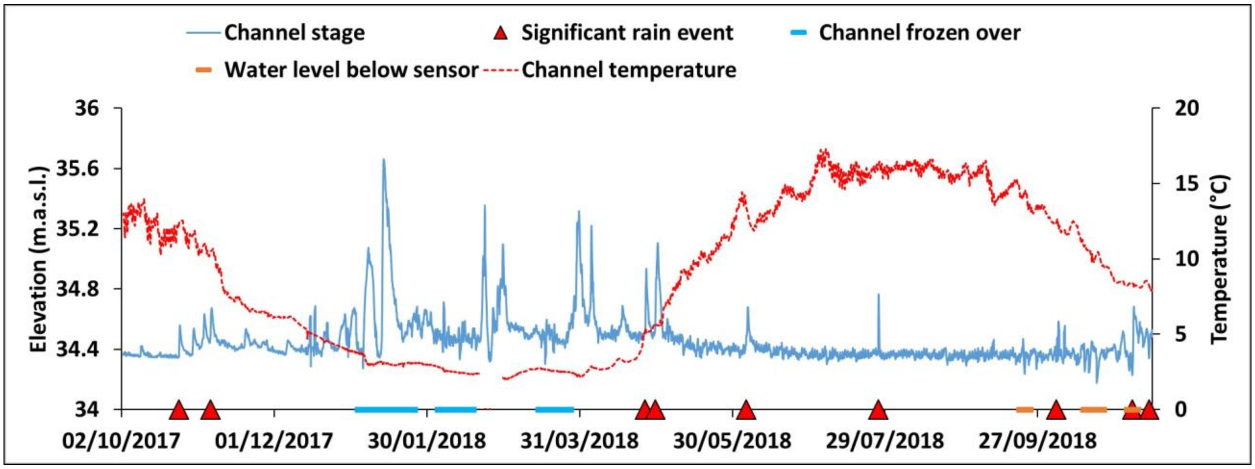


Figure 34: Channel stage (blue line) and temperature (red dashed line) as a time series over the study period taken at the downstream gauging station at DF. Significant rain events are at the start of each event.

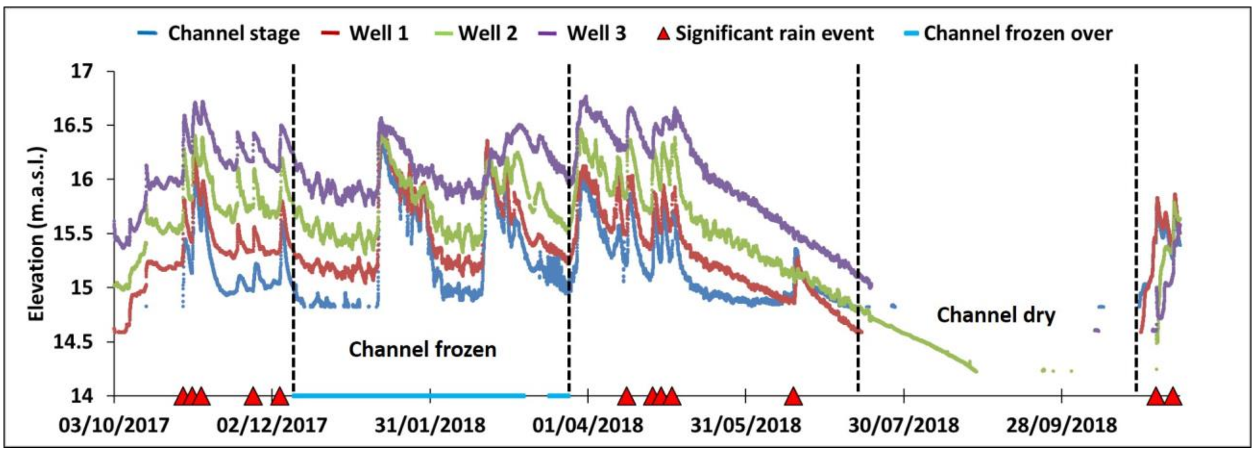


Figure 35: Channel, groundwater levels and significant rain events as a time series over the study period taken at the downstream gauging station at PPauB.

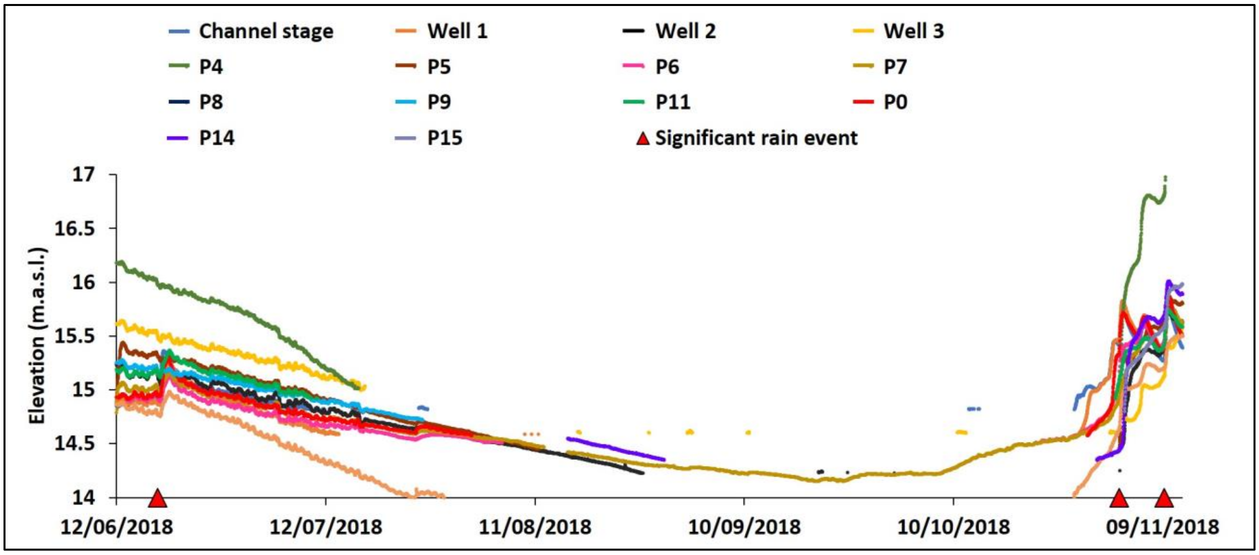


Figure 36: Channel, groundwater levels and significant rain events as a time series for the summer/fall period (including piezometers added) taken at the downstream gauging station at PPauB.

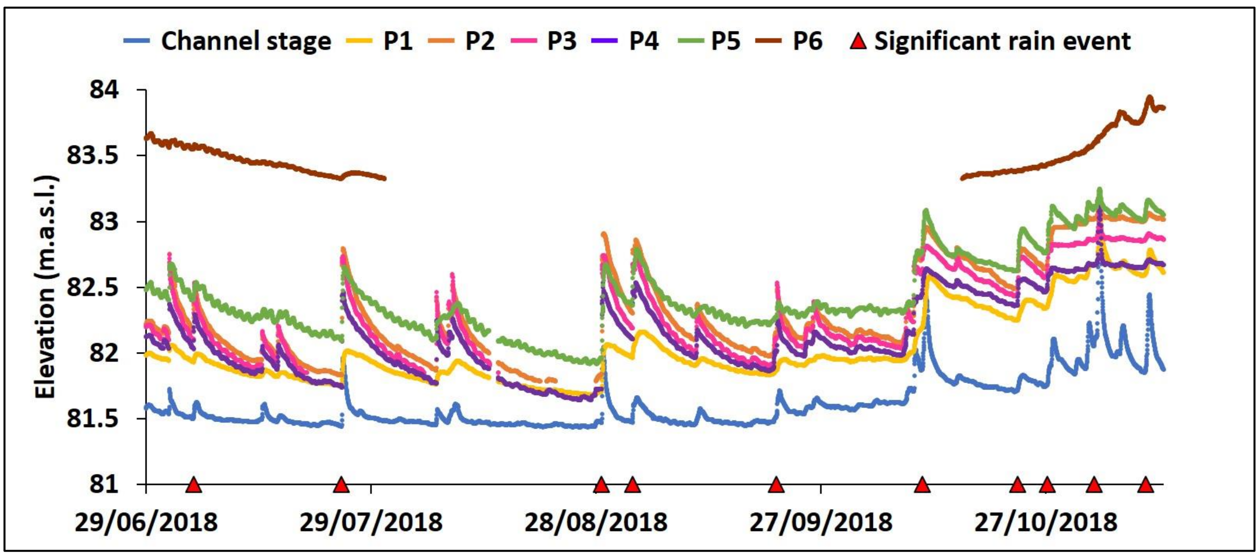


Figure 37: Channel, groundwater levels and significant rain events as a time series for the summer/fall period (including piezometers added) taken at the midstream gauging station at RM.

Note: P6 has no data from July to October as the regional water table was below the sensor position.

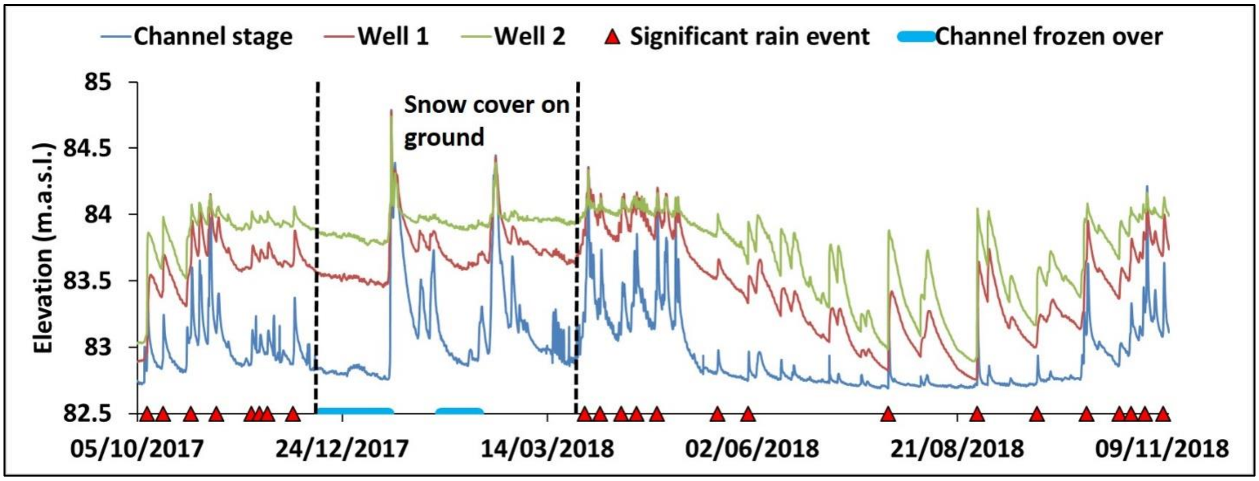


Figure 38: Channel, groundwater levels and significant rain events as a time series over the study period taken at the upstream gauging station at RM left bank.

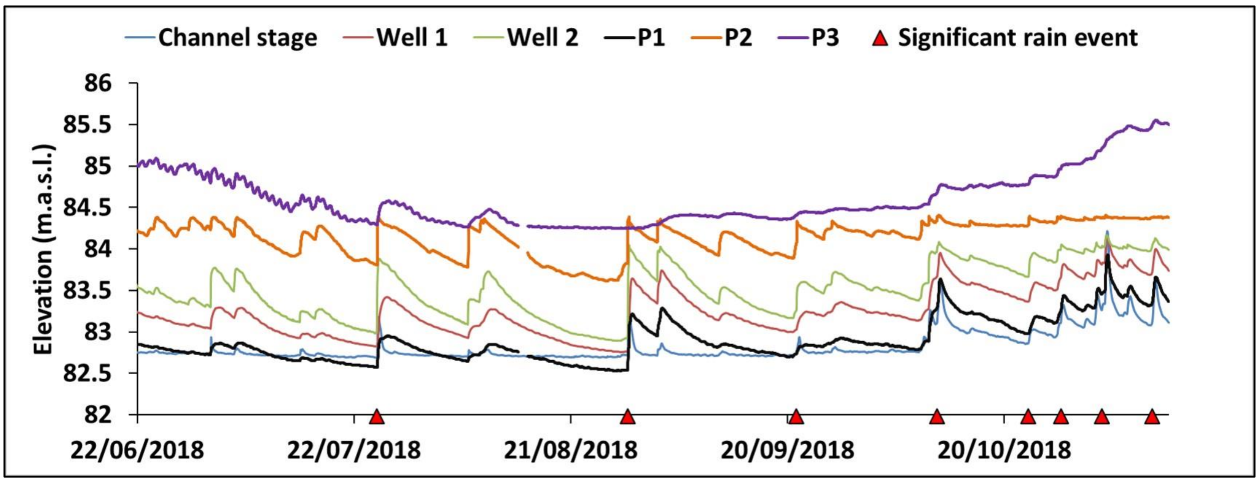


Figure 39: Channel, groundwater levels and significant rain events as a time series for the summer/fall period (including piezometers added) taken at the upstream gauging station at RM left bank.

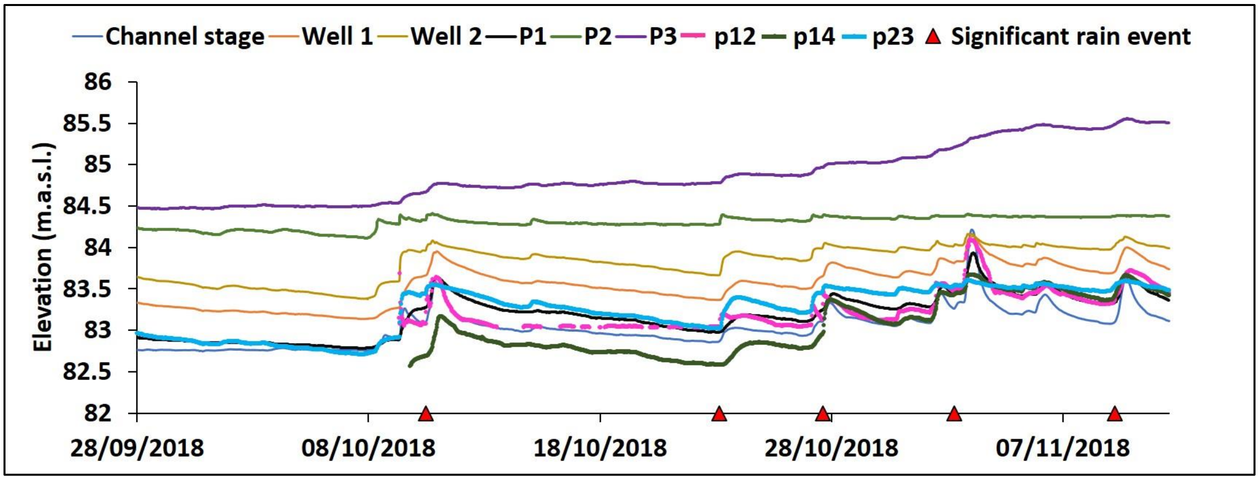


Figure 40: Channel, groundwater levels and significant rain events as a time series for the fall period (including piezometers added) taken at the upstream gauging station at RM left bank.

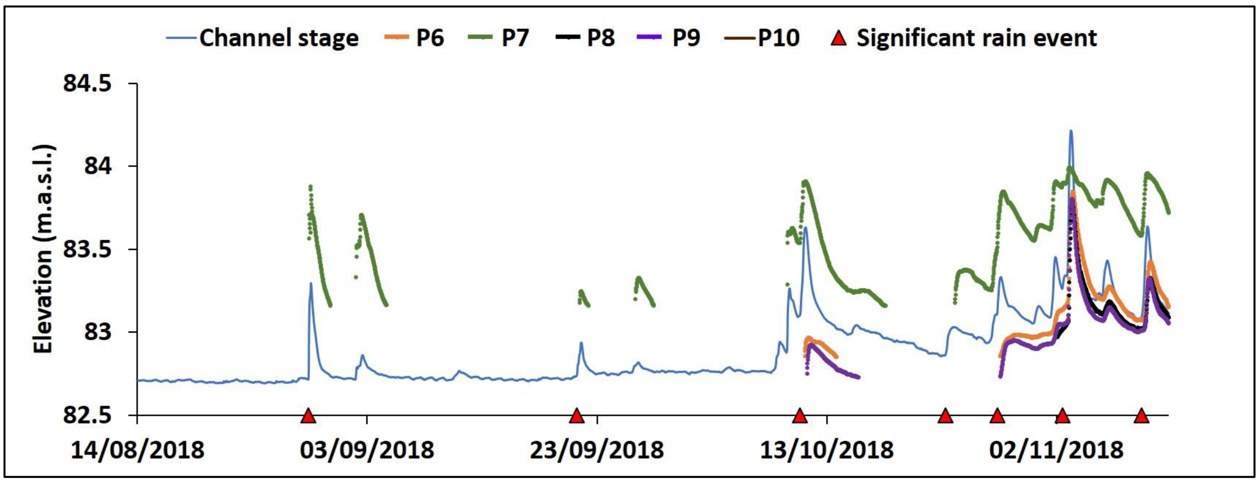


Figure 41: Channel, groundwater levels and significant rain events as a time series for the fall period (including piezometers added) taken at the upstream gauging station at RM right bank.

4.3.2 Cross-correlations

Table 6 provides a summary of lag times obtained for our site at RM up and Figure 42 provides an example of cross-correlation results for two contrasting piezometers installed at RM mid during summer/fall period.

Table 6: Summary of cross-correlation (CC) and lag times for RM up for different wells and piezometers compared to the channel. Negative values of lag indicated a rise in level in the respective well/piezometer occurring concurrently or before that of the channel.

Event	Well 1		Well 2		P1		P2		P3	
	CC	Lag (h)	CC	Lag (h)	CC	Lag (h)	CC	Lag (h)	CC	Lag (h)
1	0.76	21.5	0.76	8.5						
2	0.88	4	0.89	-1						
3	0.86	2.5	0.92	-1.5						
4	0.99	-1.5	0.93	-4.5						
5	0.99	2	0.99	-3.5						

6	0.92	5.5	0.98	-1						
7	0.98	1.5	0.96	-2.5						
8	0.98	1.5	0.99	-2.5						
9	0.97	1	0.92	0						
10	0.95	-4.5	0.93	-4						
11	0.98	-5	0.92	-3.5						
12	0.79	0	0.8	-1.5						
13	0.96	-1	0.99	-2.5						
14	0.87	3	0.93	-2.5						
15	0.77	3	0.81	-0.5						
16	0.83	24.5	0.6	6	0.89	27.5	0.54	0	0.76	28
17	0.8	11	0.73	6	0.8	9.5	0.68	30.5	N.C.	N.C.
18	0.57	23	0.6	9	N.C.	N.C.	0.61	-0.5	N.C.	N.C.
19	0.94	1	0.96	-2.5	0.95	2	0.97	-4.5	0.78	7
20	0.96	5.5	0.98	0	0.96	8.5	0.84	-3.5	0.81	9.5
21	0.91	2.5	0.97	-3	0.91	3.5	0.91	-4	0.67	26
22	0.95	0.5	0.99	-1.5	0.97	1	0.85	-3	0.9	25
23	0.96	1.5	0.97	-1	0.97	2.5	0.36	0	0.9	1.5
Median annual	0.94	2	0.93	-1.5						
Median fall / summer	0.925	4	0.965	-0.5	0.95	3.5	0.76	-1.75	0.795	17.25

Note: only correlation coefficients where p<0.001 are reported; N.C. = No correlation; No data for P1-3 for events 1-15 as they were not yet installed.

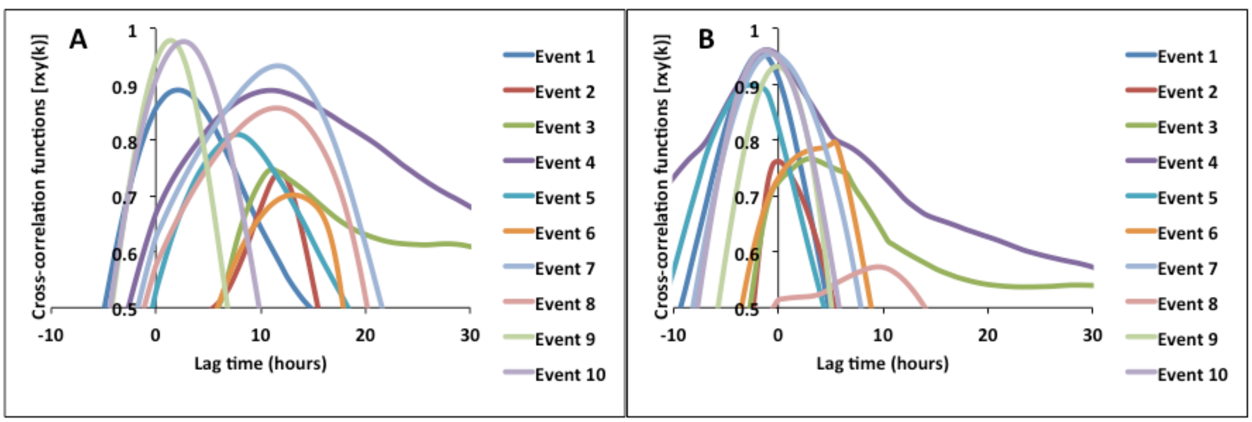


Figure 42: Comparison of cross correlation results for 10 significant rain events identified during the summer/fall period of 2018 for RM Mid P1 (between the channel and ancient meander deposits) A) and P2 (Inside the ancient meander deposits) B)

Results show variability in both the intensity of the relationship between water stage and groundwater levels and the lag time between peaks on a per well/piezometer basis as well as between them. Figure 42 illustrates this variability, and for both piezometers 1 and 2 (A and B) at RM we see correlations ranging from 0.58 to 0.98 and lag times ranging from -2 to 14 hours for the rain events evaluated. Furthermore, piezometer 1 (A) shows all positive lag times indicating that the groundwater peak is consistently occurring after the channel stage peak, up to 14 hours for several events. On the other hand, negative lag times -2 to 0 hours are the most common response to the precipitation events evaluated at piezometer 2 (B), indicating that the groundwater peak here is occurring before that of the channel stage.

In terms of Intensity, cross correlation results at RM tended to have higher cross-correlation coefficients (> 0.75) compared to PPauB. Higher coefficients may be the result of several factors related to characteristics of the rain event (greater amplitude and duration of in channel response), distance from the channel (well/piezometers closer to the channel would respond more intensely), the presence of coarse alluvial deposits and the topography of the ancient meander, which may influence patterns of surface drainage and water table depth. Lower coefficients on the other hand may represent a lack of hydrological connectivity in that groundwater levels at the piezometers or wells are simply not responding to the event or the response at a particular location is too complex. Lower coefficients are also common for piezometers installed in the regional deposits (ex: P5 for RM up and P6 for RM mid sites) where groundwater is either not responding to rain events or the response is too long (for example in some cases the groundwater was still responding to an event when a new event would begin).

Plots of lag time against characteristics of the rain events did not show any clear trends with the exception of time since last rain event where a mild positive correlation was observed (Figure 43 A). Related to time since last rain event we hypothesized that lag time variability would also depend on the degree of saturation prior to the event hence the depth of the water table. Indeed, this trend was observed for several wells and piezometers at RM. This is illustrated in Figure 43 B) which shows that the closer the water table is to the ground surface at the start of the precipitation event the lower the lag time response of that well.

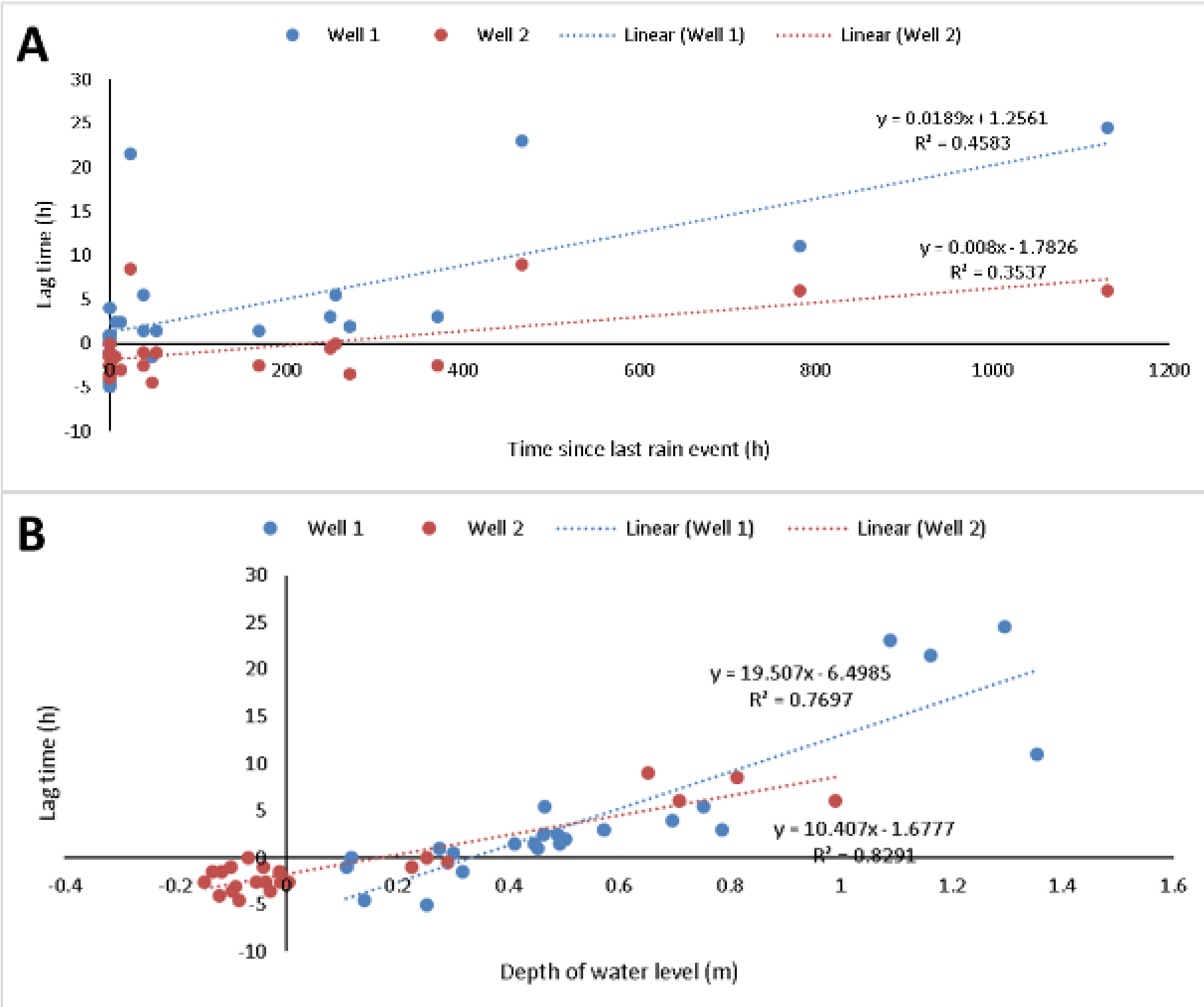


Figure 43: Lag time response of wells installed at RM UP site to all 23 precipitation events plotted against time since last rain event A), and initial depth of water at the respective well B).

The following Figures 44 - 48 illustrate the differences in lag times between water level recordings at the various wells and piezometers within their respective sites. For RM UP (Figure 44) zero or negative median lag times are observed at W2 and P2 which are in the coarse alluvial deposits of the ancient channel bed or in the topographic depression where surface water pools during a rain event. These low lag time values indicate that they respond to precipitation events either before or simultaneously to the channel. P1 and W1 also have low lag times as they are in the fine to medium floodplain deposits and closest to the channel, whereas P3 which is the furthest and in regional deposits has the longest median lag time. A strong correlation is seen between the average depth of the water table and median lag time (Figure 44B) which suggests that there are likely strong hydraulic connections here and a good potential for flux. However, since we are looking at pressure gradients, it is not possible to confirm exchange based on these results. A strong correlation is also seen between median lag time and the depth of the medium to coarse alluvial deposits (Figure 44C) suggesting that the depth of the alluvial unit has an effect on potential hydrological connectivity.

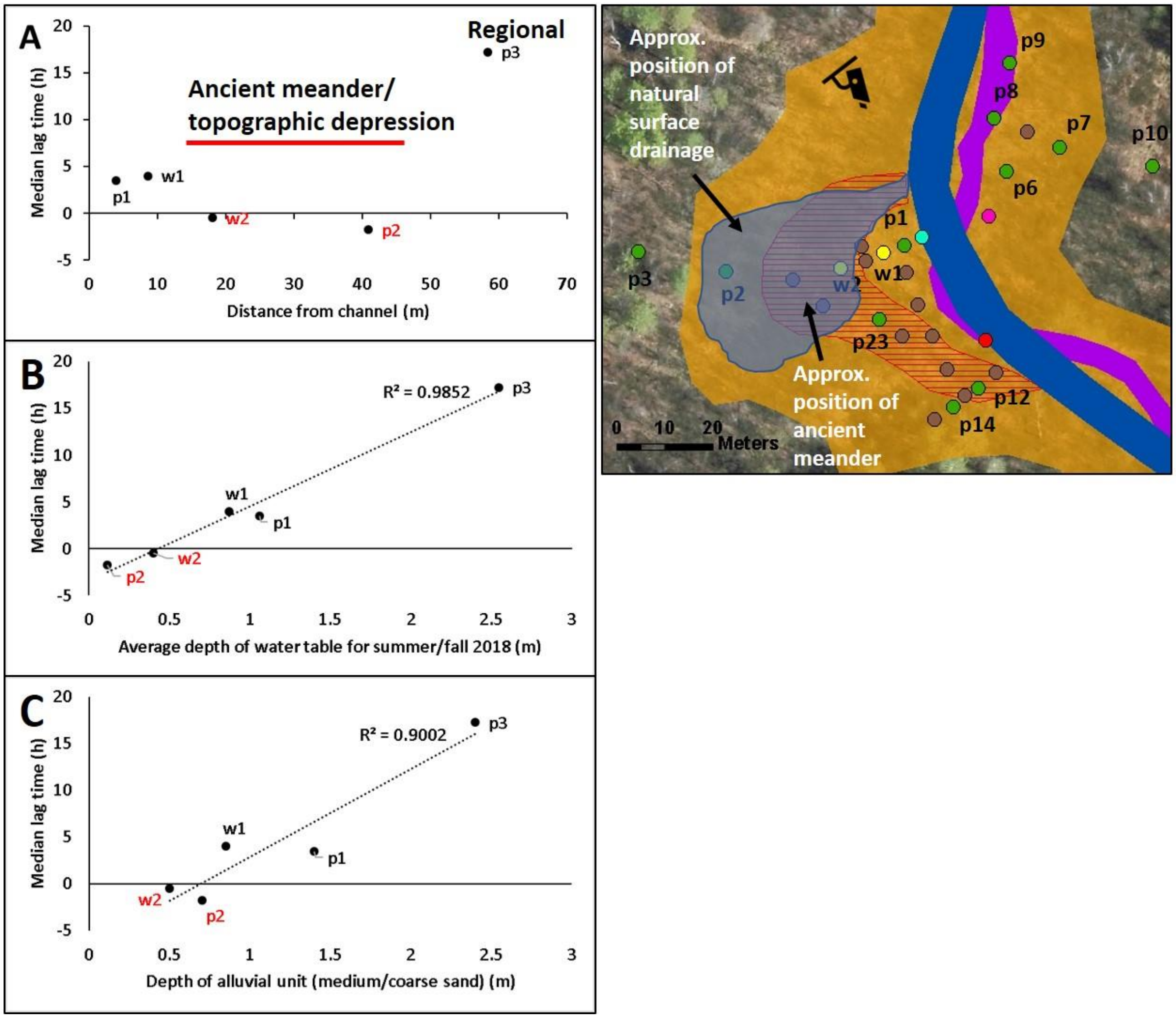


Figure 44: Examining differences in lag times between wells/piezometers for RM UP. A) Median lag time vs. Distance from channel, B) Median lag time vs. Average depth of the water table, C) Median lag time vs. depth of alluvial unit.

Results of cross-correlations for RM MID (Figure 45) show almost identical results to RM UP as P2, P3, P4 and P5 all demonstrated low median lag times for the significant precipitation events considered. In this case, P2, P3 and P5 are located within the coarse alluvial deposits of the ancient channel position and are also in the topographic depression (including P4) where surface water pools and is connected to the main channel.

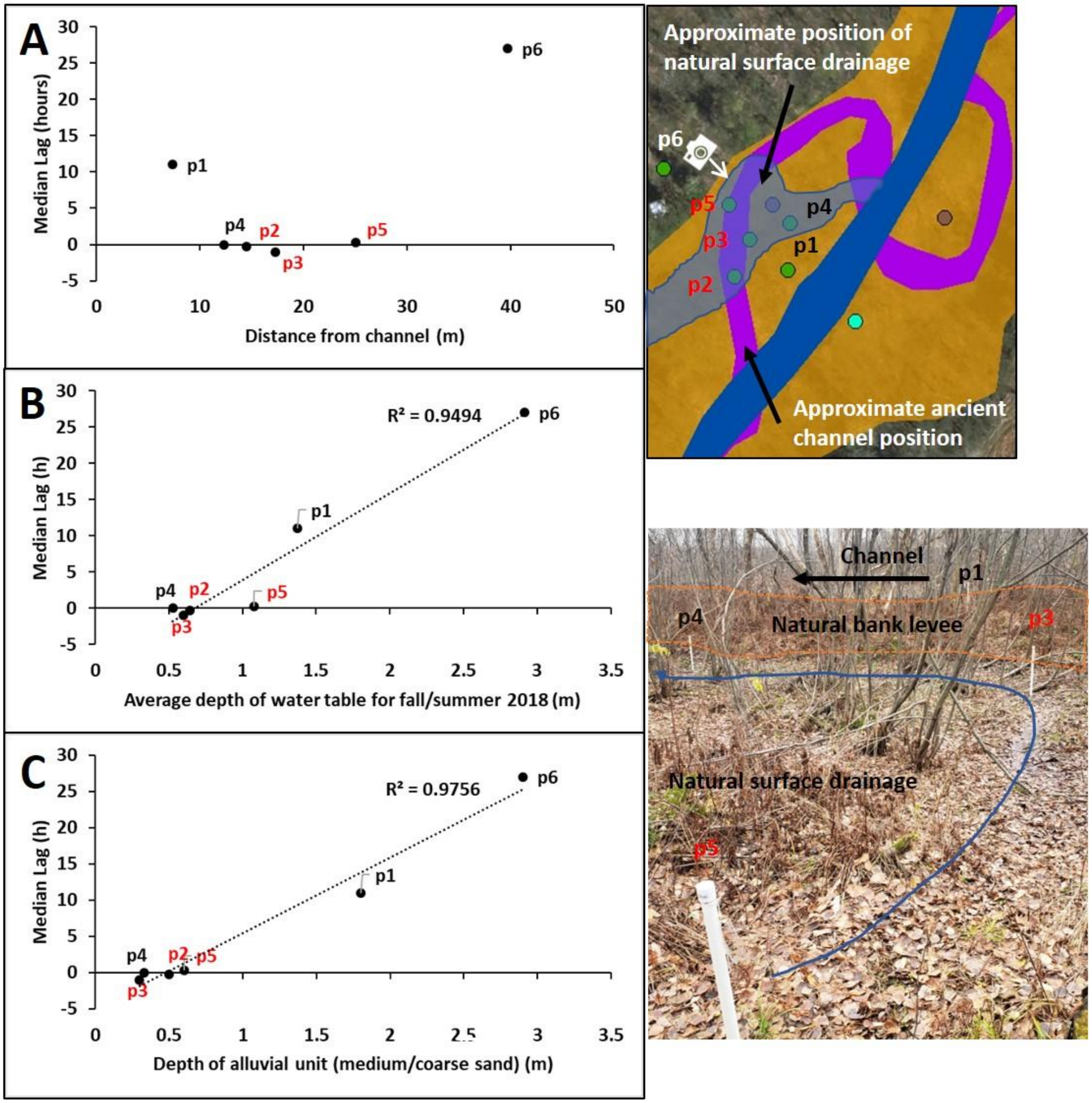


Figure 45: Examining differences in lag times between wells/piezometers for RM MID against A) distance from channel, B) average depth of the water table, C) depth of alluvial unit. Top right shows planform view of site and piezometer positions and bottom right photo shows local topography.

In contrast to RM cross correlation, results for PPauB (Figure 46) did not yield any clear relationship between median lag time and average water table depth or depth of the alluvial unit. Indeed, many piezometers did not respond to smaller, summer rain events (Figure 47). This may be related to the markedly deeper coarse sediment left behind by the ancient channel at PPauB (1.99m, Figure 48), compared to RM (1.17m, Figure 47). This difference is even greater if only considering wells/piezometers closest to the ancient meander bed (2.17m for PPauB and 0.6m for RM). Furthermore, they are covered by remblais from the surface and potentially disconnected from the river due to the presence of clay plugs along the bank. This explains in part why we don't see any evidence for hydraulic connectivity with the ancient meander for PPauB but we do for

RM. It should also be noted that while there is a good density of piezometers at the PPauB site, they had only been recording for a short time and therefore have less data points to compare.

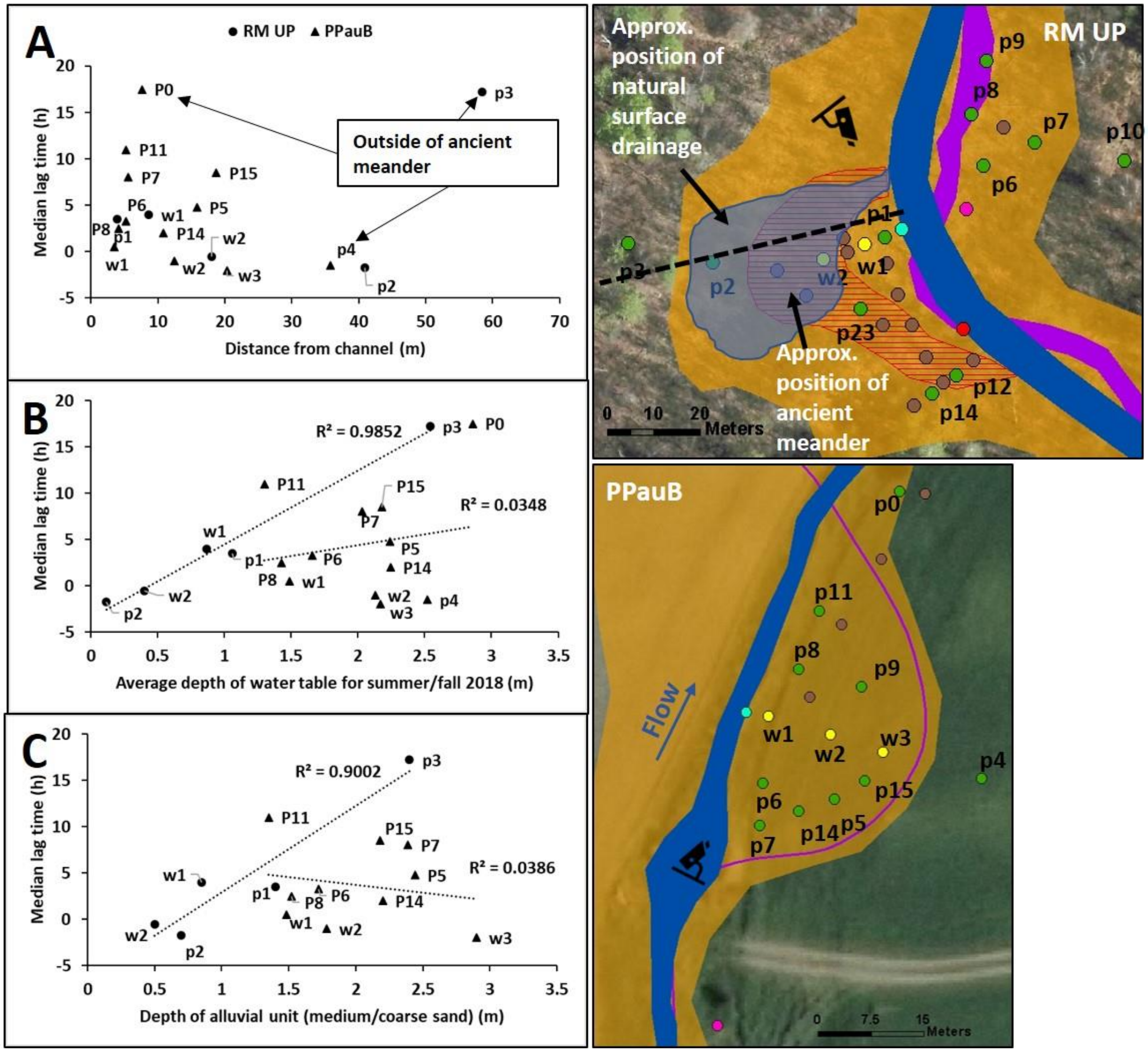


Figure 46: Results from PPauB contrasted with those for RM showing no trend between lag time and A) distance, B) average water table depth or C) depth of alluvial unit. . Note: median lag time for PPauB is based on only 1-3 values as many instruments didn't yield a significant lag time and there are few significant events.

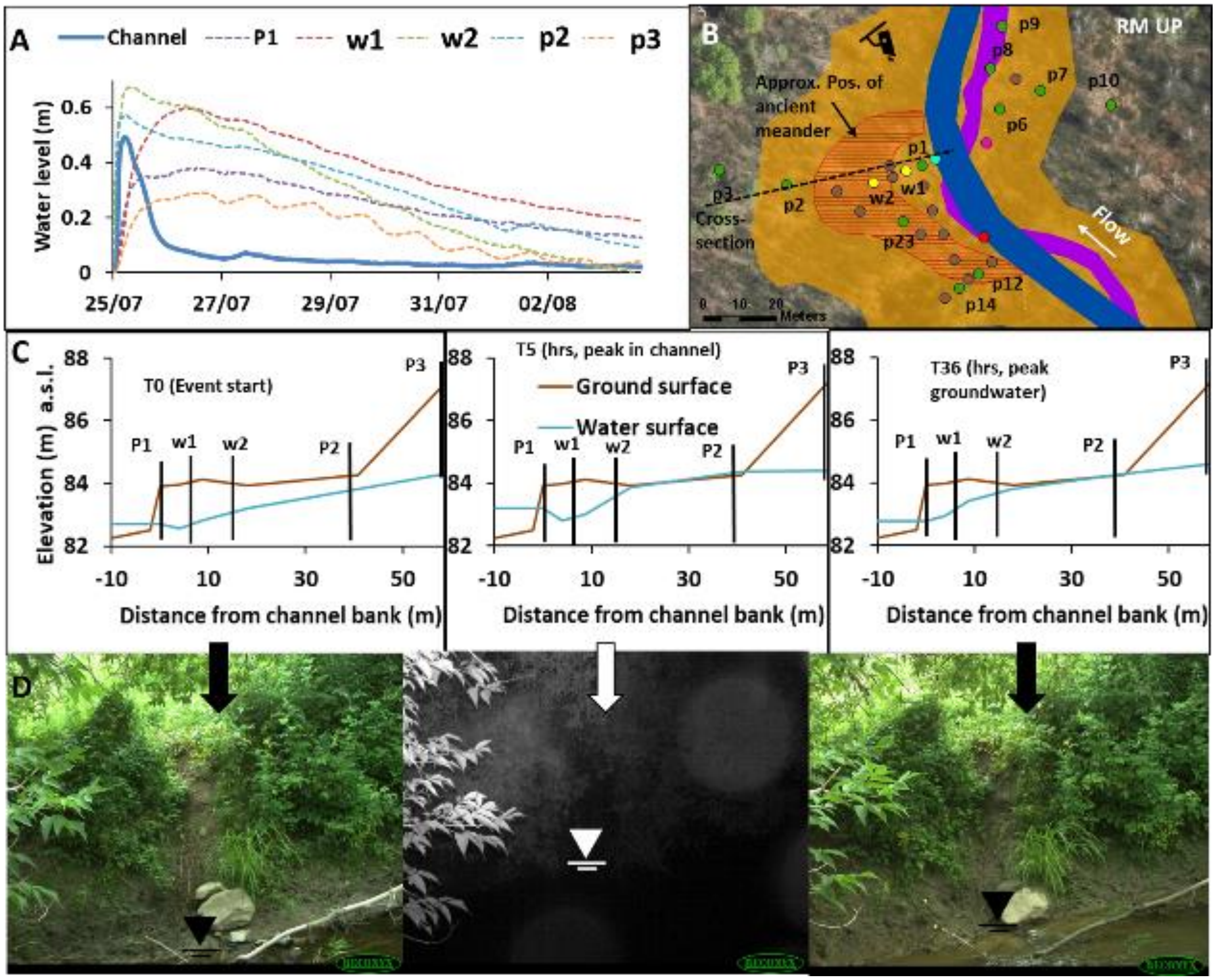


Figure 47: Hydro(geo)logic response of RM UP site to a moderate summer rain event (Table 6. event 16). A) water level response to the event (relative to initial elevation), B) Planform view of meander and cross-section placement, C) cross-sections of water level and ground surface at 3 times during an event (start, channel peak and groundwater peak), D) photos captured from field camera for the same time steps.

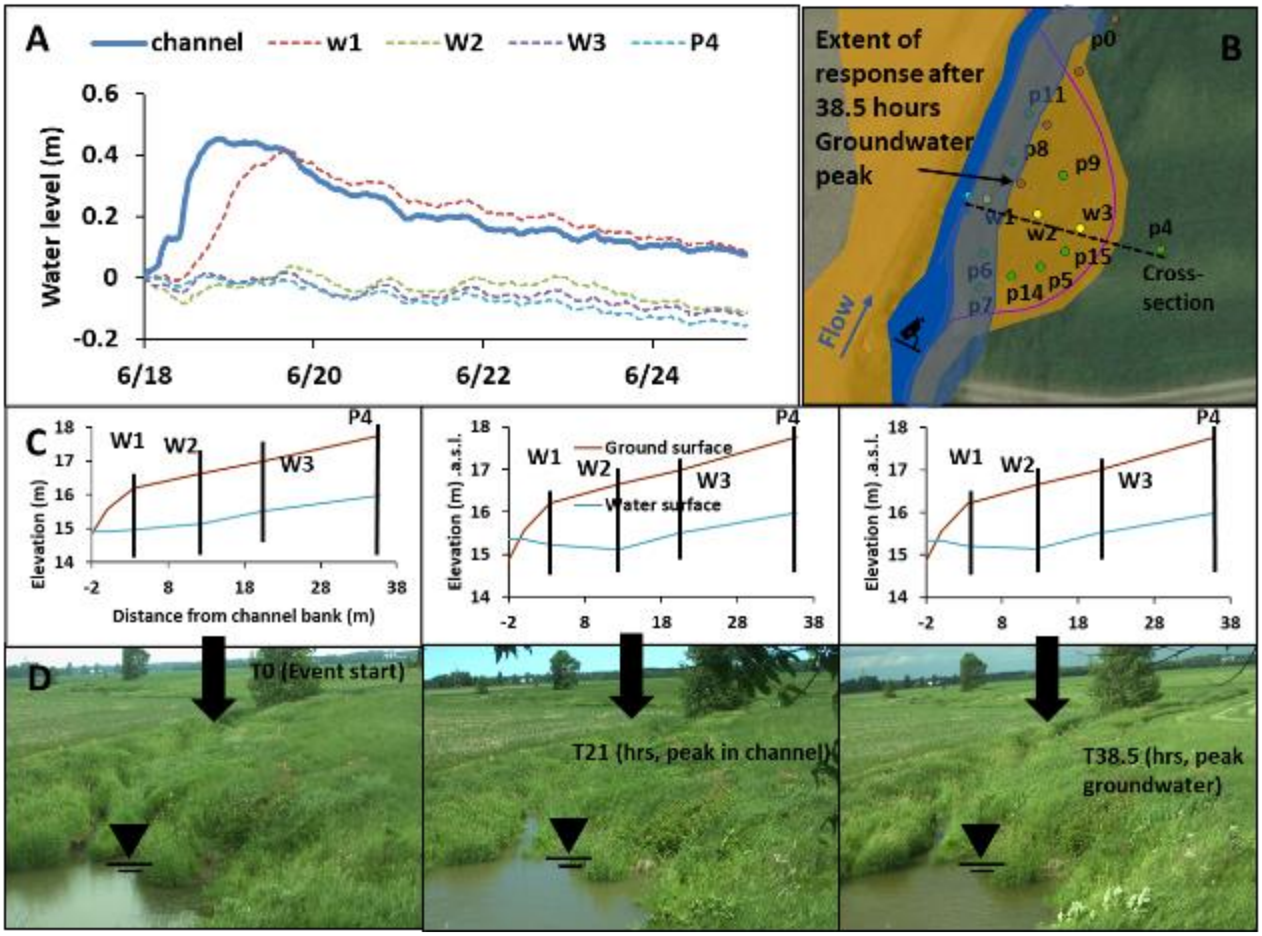


Figure 48: Hydro(geo)logic response of PPauB to a moderate summer rain event. A) water level response to the event (relative to initial elevation), B) Planform view of meander and cross-section placement, C) cross-sections of water level and ground surface at 3 times during event (start, channel peak and groundwater peak), D) photos captured from field camera for the same time steps.

4.3.3 Stratigraphy

The following figures present the results of the stratigraphic surveys at RM (Figure 49) and PPauB (Figure 50), interpolated to show parallel and perpendicular cross-section views of the ancient meanders. The profiles indicate that these artificially abandoned meanders present a stratigraphic composition that is coarser than that of the surrounding floodplain and regional deposits for both sites. Overall grain size of floodplain sediments at RM are coarser than those at PPauB which may partly explain the greater potential hydraulic connectivity observed. Furthermore, these deposits are spatially heterogeneous particularly in the case of PPauB where clay plugs and filling material may reduce connectivity with the channel. Although an impermeable layer is present at both sites, till/clay in the case of RM and clayey/silt in the case of PPauB as well as an organic layer indicative of past wetland presence, the topography of the meander at PPauB, in contrast to that of RM, inhibits the surface pooling of water that would be important for wetland rehabilitation.

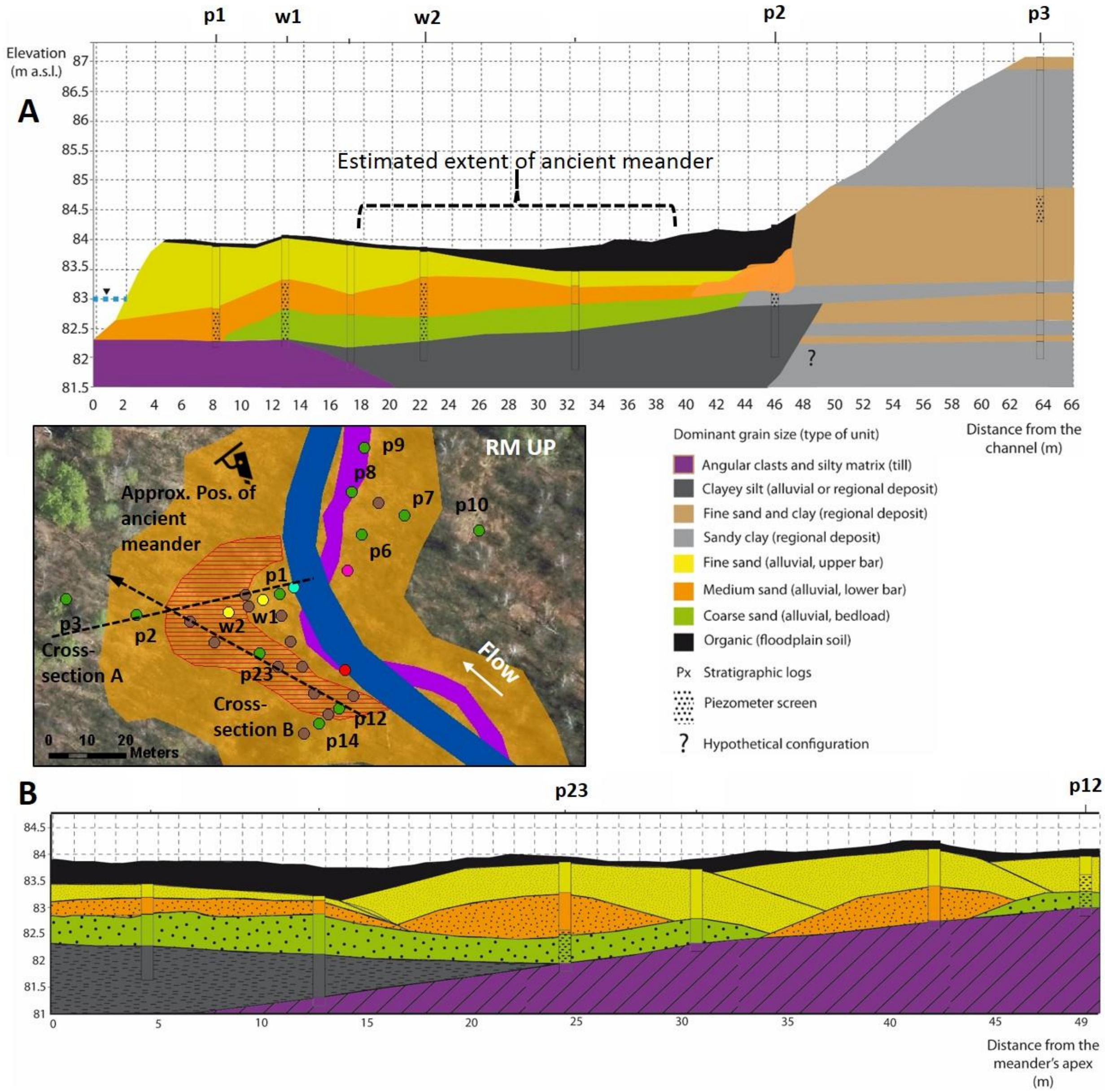


Figure 49: Stratigraphic results for RM UP. A) cross-section perpendicular to main channel, B) planform view.

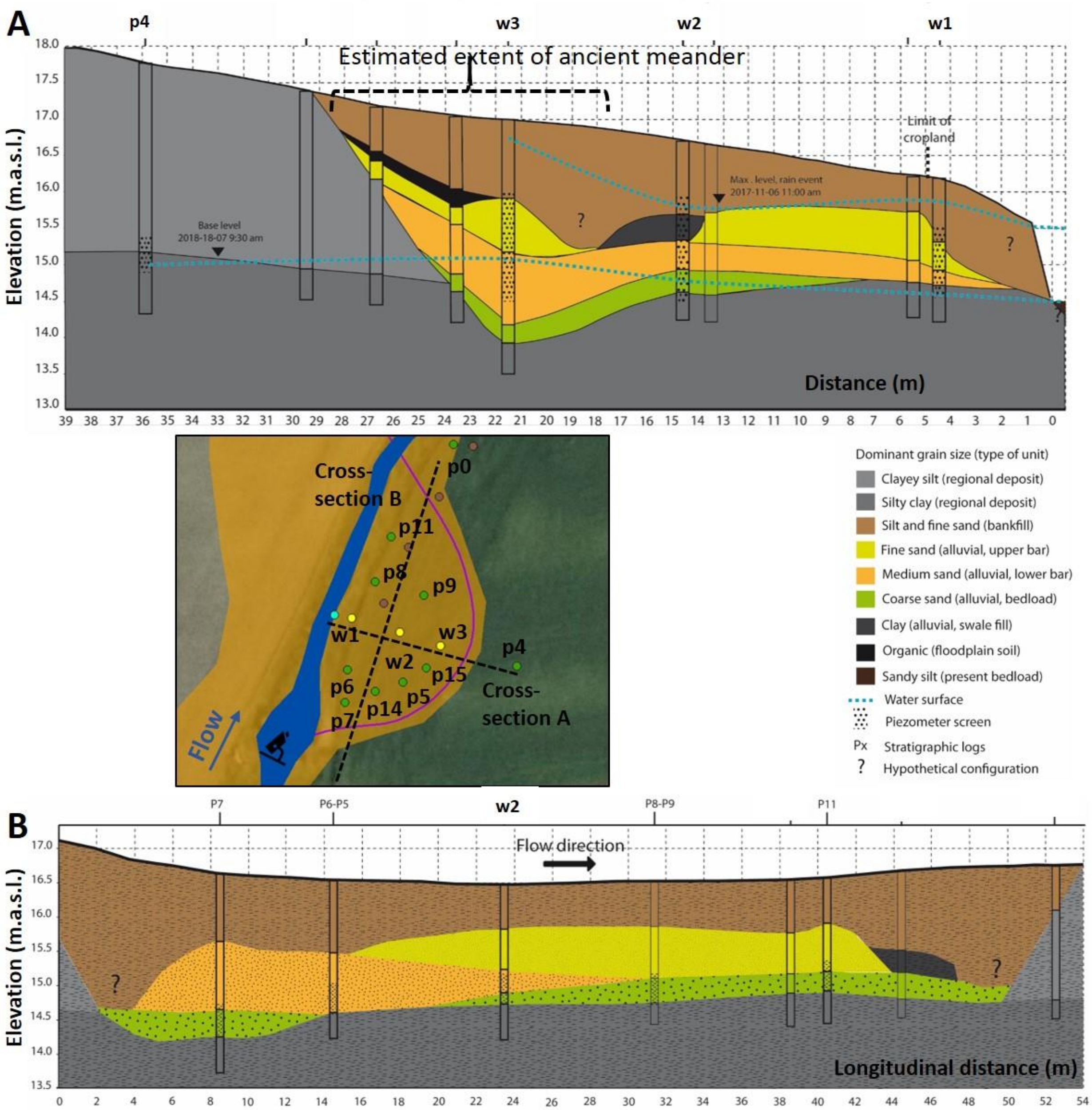


Figure 50: Stratigraphic results for PPauB. A) cross-section perpendicular to main channel, B) cross-section parallel to channel.

4.3.4 Temperature

Temperature gradients are assessed laterally between the channel and groundwater via the pressure transducers installed in the various wells and piezometers. Figure 51 provides a typical example of expected water temperature decreasing with increasing distance from the channel during the summer at P1 and P6. However, the gradient is possibly interrupted by the ancient meander, where recorded temperatures at P2, P3, and P4, which are either within the topographic depression of the ancient meander where surface water pools and drains, are more similar in magnitude and variability to that of the channel. P5 represents an intermediate case as it is on the edge of the ancient meander and the regional deposits and therefore presents warmer temperatures despite its distance from the channel, similar to other piezometers inside the topographic depression, but at the same time temperature is less variable, similar to temperature recordings from piezometers outside the ancient meander.

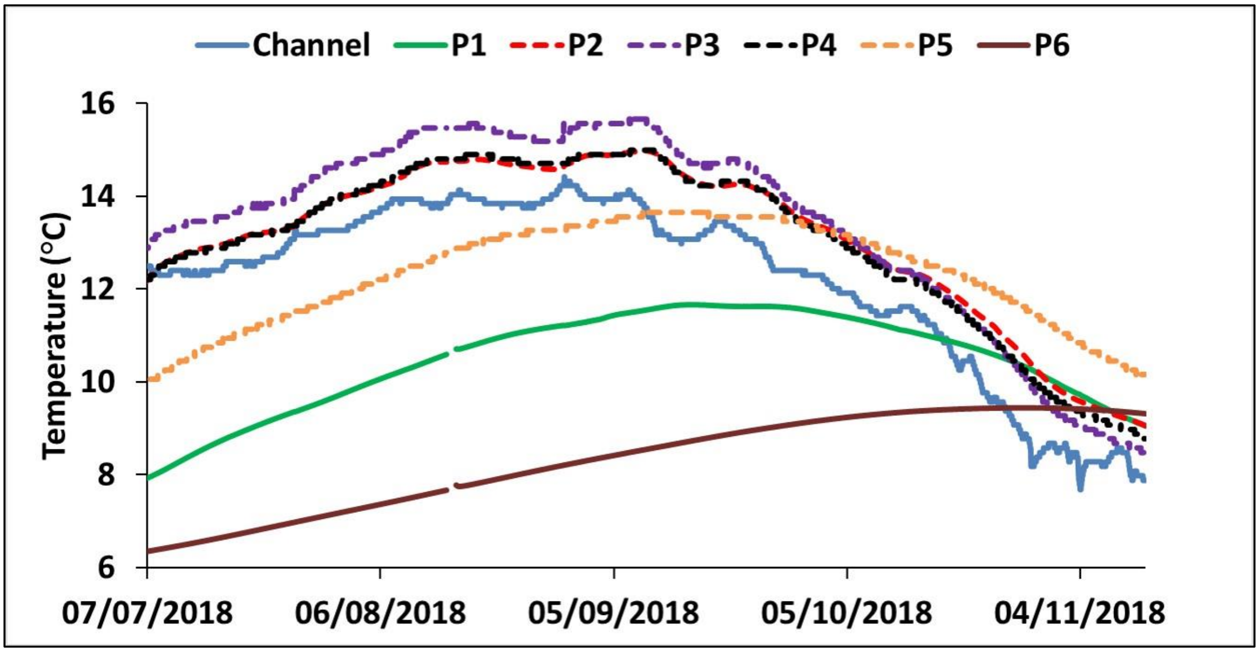


Figure 51: Temperature data of channel and groundwater collected at RM mid for the summer/fall period of 2018. Dashed lines indicate piezometers installed in the topographic depression associated with the ancient meander.

Temperature data collected longitudinally from our Hobo sensors placed in channel was summarized as daily average, minimum and maximum (Figure 52 – 54 show daily average for the three sites). Due to the small number of sensors installed, we used visual interpretation to provide potential explanations for differences between them. Little difference was noted longitudinally between sensors for PPauB and RM. In the case of PPauB this is not surprizing as the channel consists of intermittent stagnant pools for much of the summer. For RM, the consistent shading from the developed riparian zone throughout the site coupled with the strong potential for hydrological connections related to the ancient meanders may explain the observed spatial uniformity of temperature. Unfortunately, in year 2 two sensors were lost from DF and RM, which limited inter-annual comparisons. It is, however, very likely that the resurgence points at DF are having a moderating effect on temperature. We see in Figure 52 that logger 1 which is upstream of the most upstream resurgence point is much more variable and follows more closely air temperature trends. Loggers 2, 3 and 4 on the other hand maintain stable temperatures throughout the summer likely due to the cooler groundwater input from the resurgence points. These differences at DF become lesser once fall discharges dominate the temperature profile.

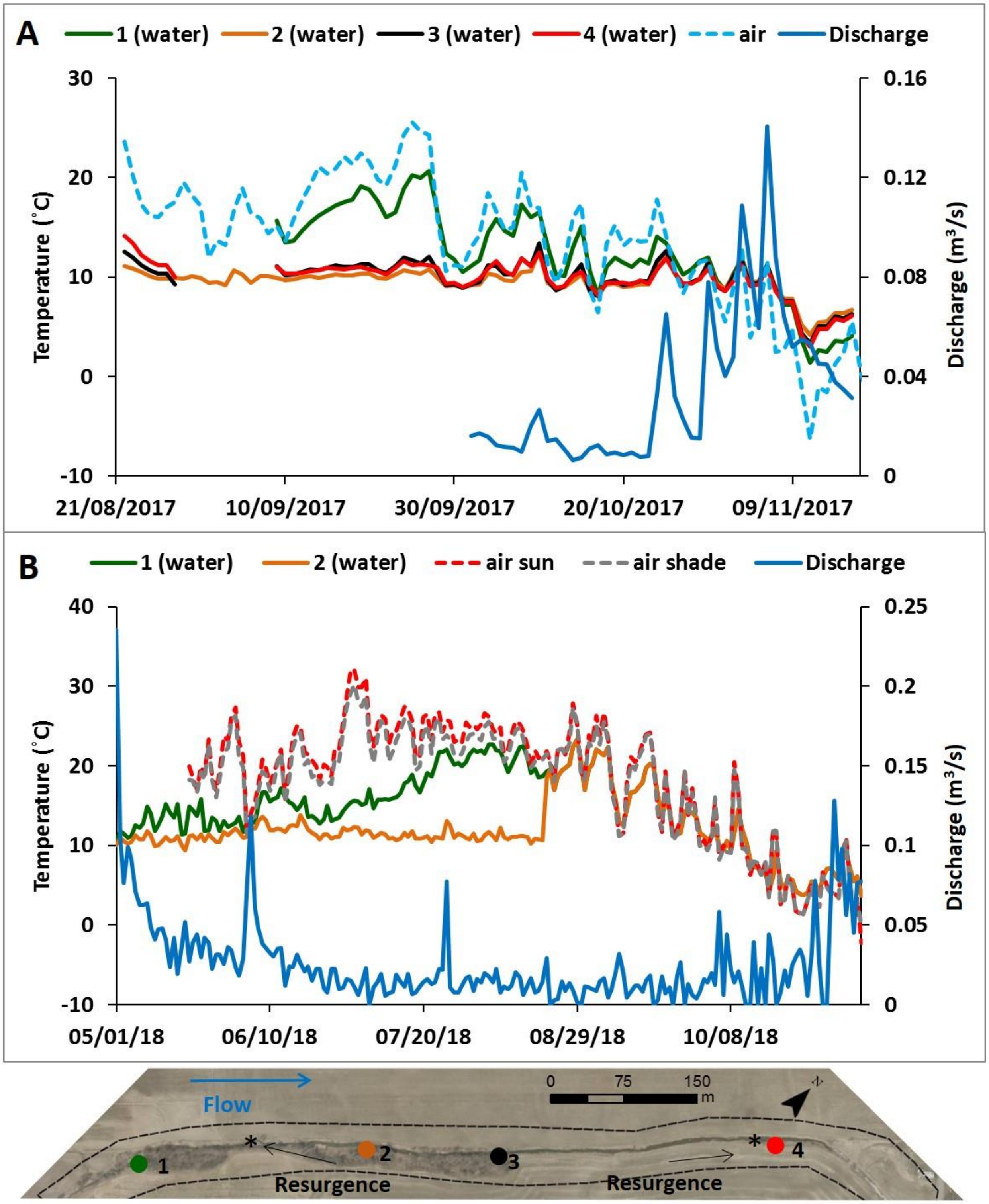


Figure 52: Average daily water temperature recorded at DF in summer/fall of 2017 A) and 2018 B)

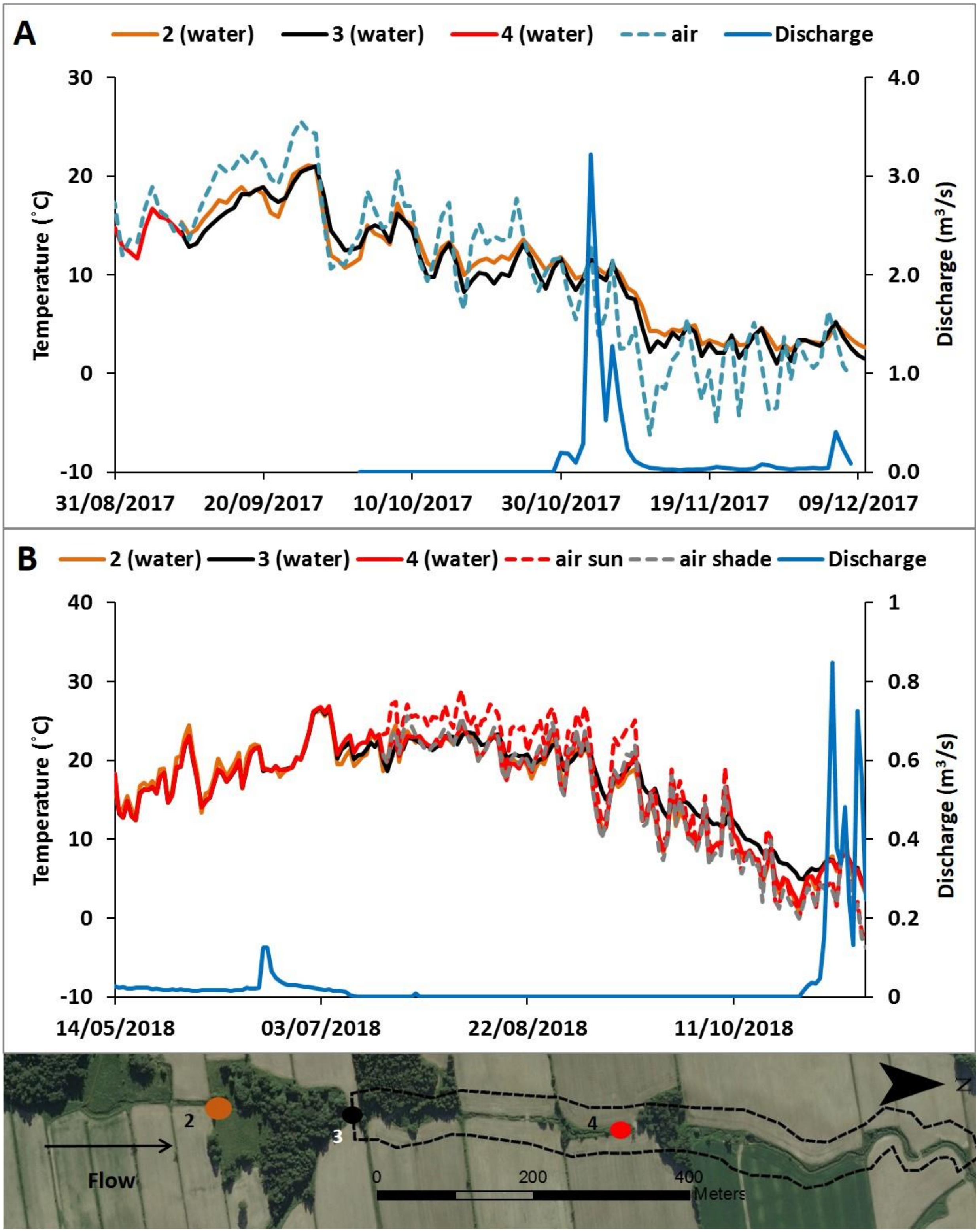


Figure 53: Average daily water temperature recorded at PPauB in summer/fall of 2017 A) and 2018 B)

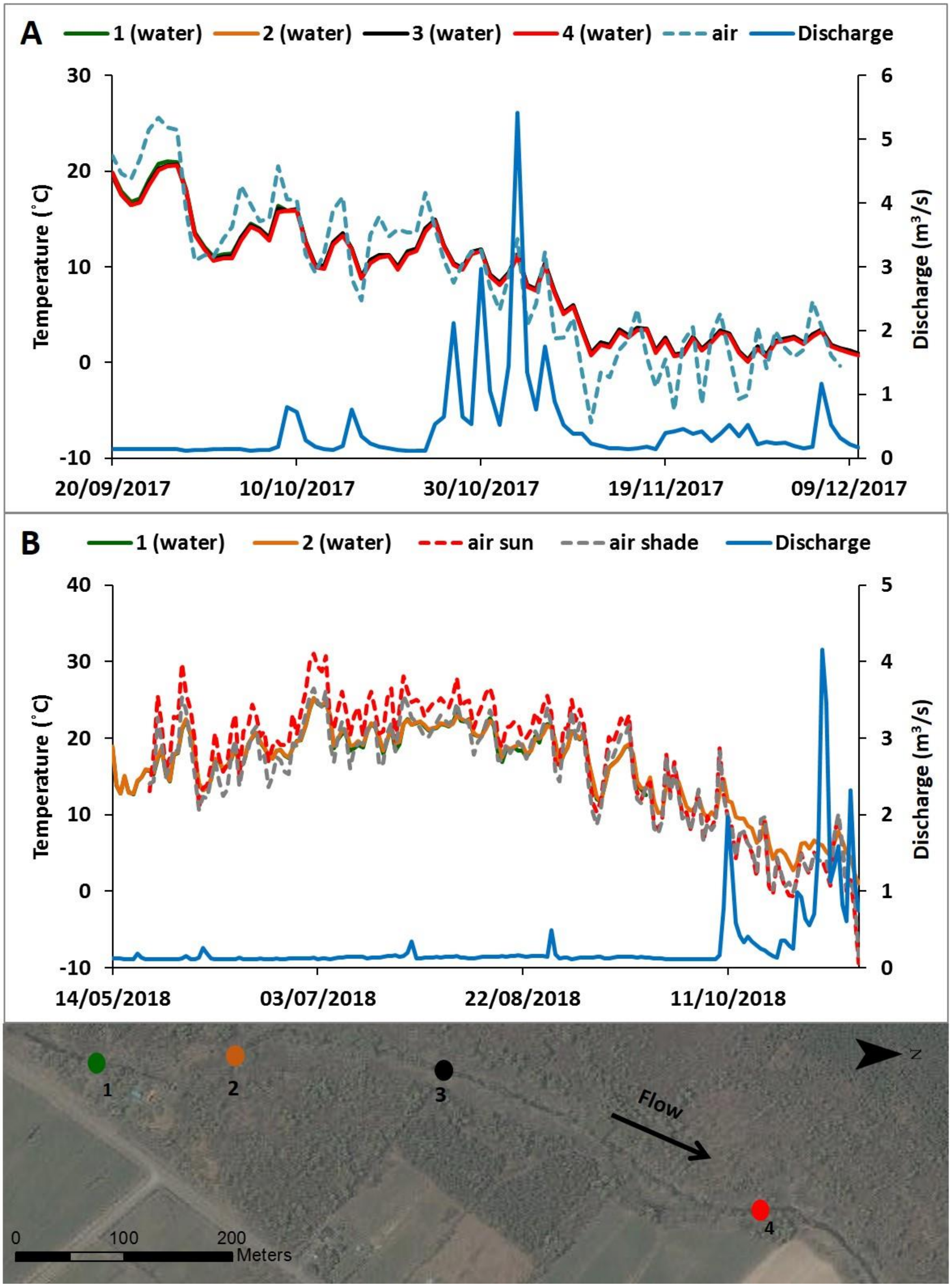


Figure 54: Average daily water temperature recorded at RM in summer/fall of 2017 A) and 2018 B)

Results for temperature data reveal several differences in spatial and temporal variability both longitudinally and laterally with respect to the channel. For instance, at DF we see clear differences in the longitudinal temperature gradient indicating a potential moderating effect of the resurgence points where fresh groundwater inputs provide cooling during hot summer months and a slight warming in December. This is clear when we compare temperature values from the logger which is upstream from both resurgence points (Figure 55) and which is characterized by more extreme daily median temperatures compared to loggers downstream from the resurgence points.

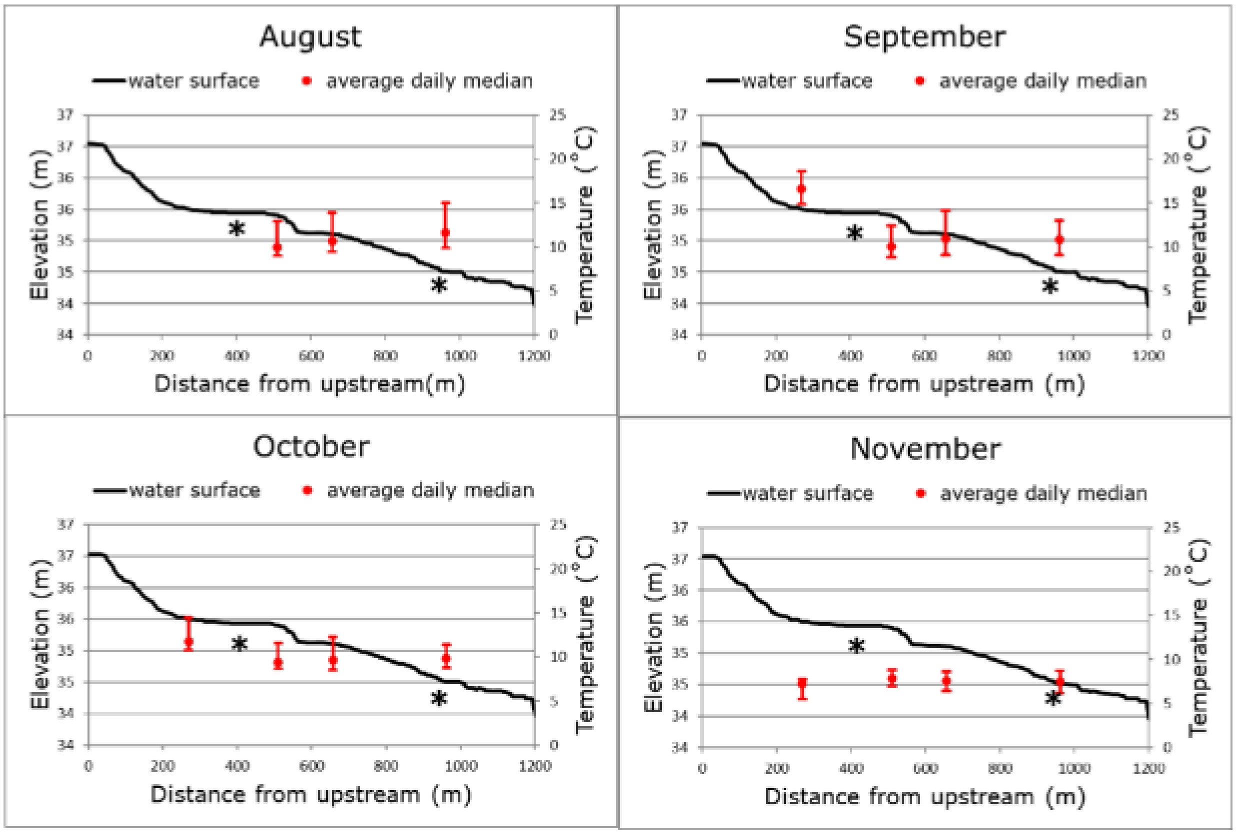


Figure 55 : Spatial and temporal variability of recorded water temperature at Branch 53 Des Fèves in 2017

While RM exhibited the least spatial variability in temperature, likely due to the consistent shading from forest cover throughout the study reach and the evidence of potential hydraulic connections with groundwater throughout the site (i.e. both RM UP and RM MID), seasonality still appears to have an effect on spatial temperature differences at all sites with greater differences in hot summer months (Figure 56).

Figure 57 focuses on the fall 2018 results regarding the potential flux exchanges within the meander at RM UP. Here we see the channel and groundwater response to fall precipitation events which have a cooling effect on all loggers with the exception of P1, which is located between the ancient meander and the channel. Interestingly, P1 has a peak increase in temperature rather than a decrease. Our hypothesis is that there may be flux from groundwater in the ancient meander in the direction of P1. Although groundwater is cooling throughout the season and in response to the precipitation events, it is still warmer in comparison to the channel temperature.

Lateral and longitudinal temperature at PPauB did not show any clear responses to precipitation events or obvious differences between their position in the channel beyond overall seasonal increases in the summer and decreases in the winter. Any longitudinal variability observed is likely related to the depth of the pool in which the logger was placed, since much of the system was dry, consisting of small disconnected pools for several months in the summer.

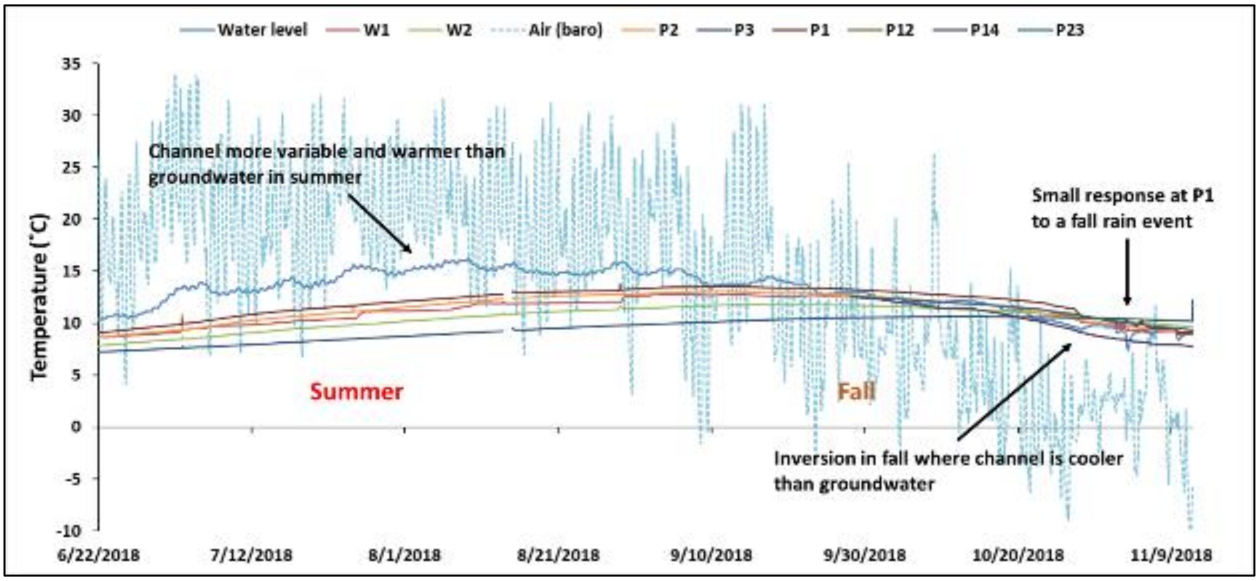


Figure 56: Summer/Fall time series of temperature for RM UP for channel and well/piezometers

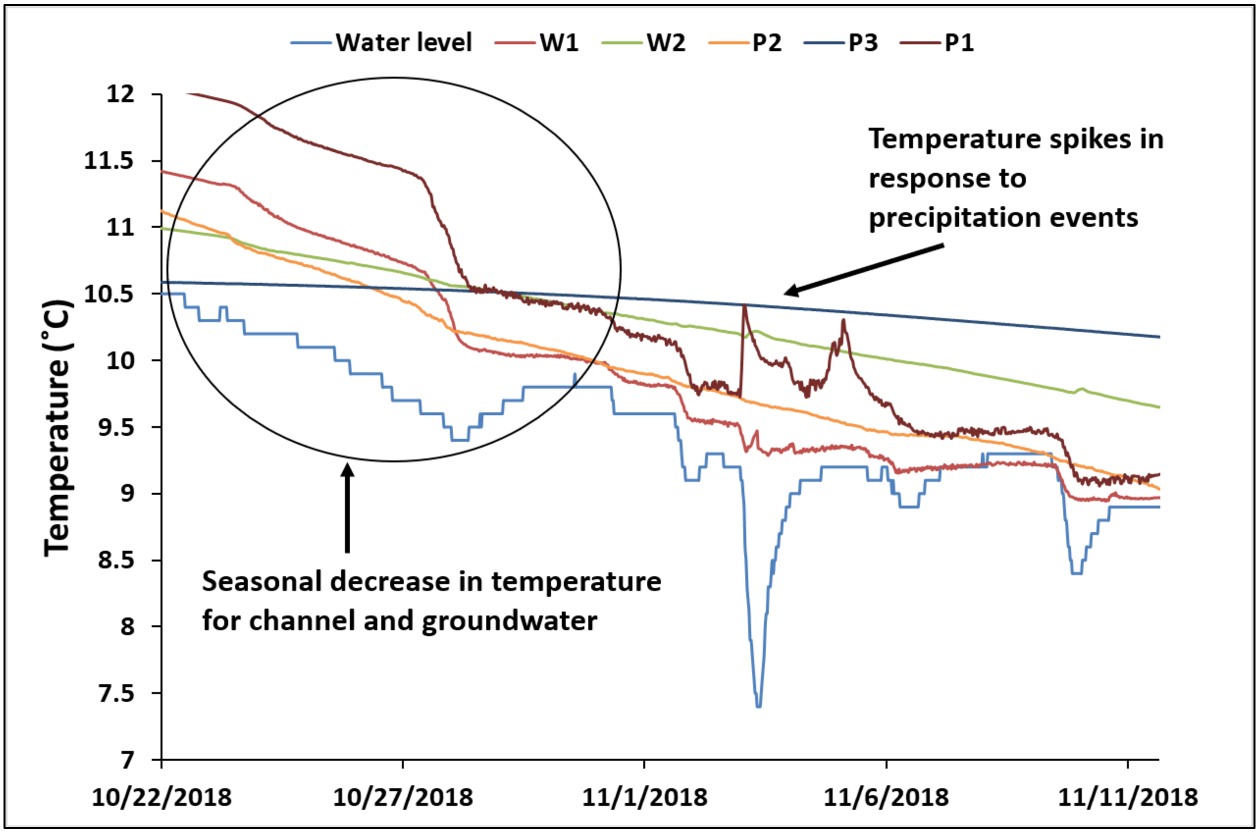


Figure 57: Zoom in of Temperature response to a series of fall floods (3 precipitation events) for RM UP.

4.3.5 Vegetation survey

Figures 58 - 60 present the results of the vegetation assessment showing the extent of wetland habitat units mapped on the LiDAR DEM, which allows to see the ancient floodplain in red (lower elevation).

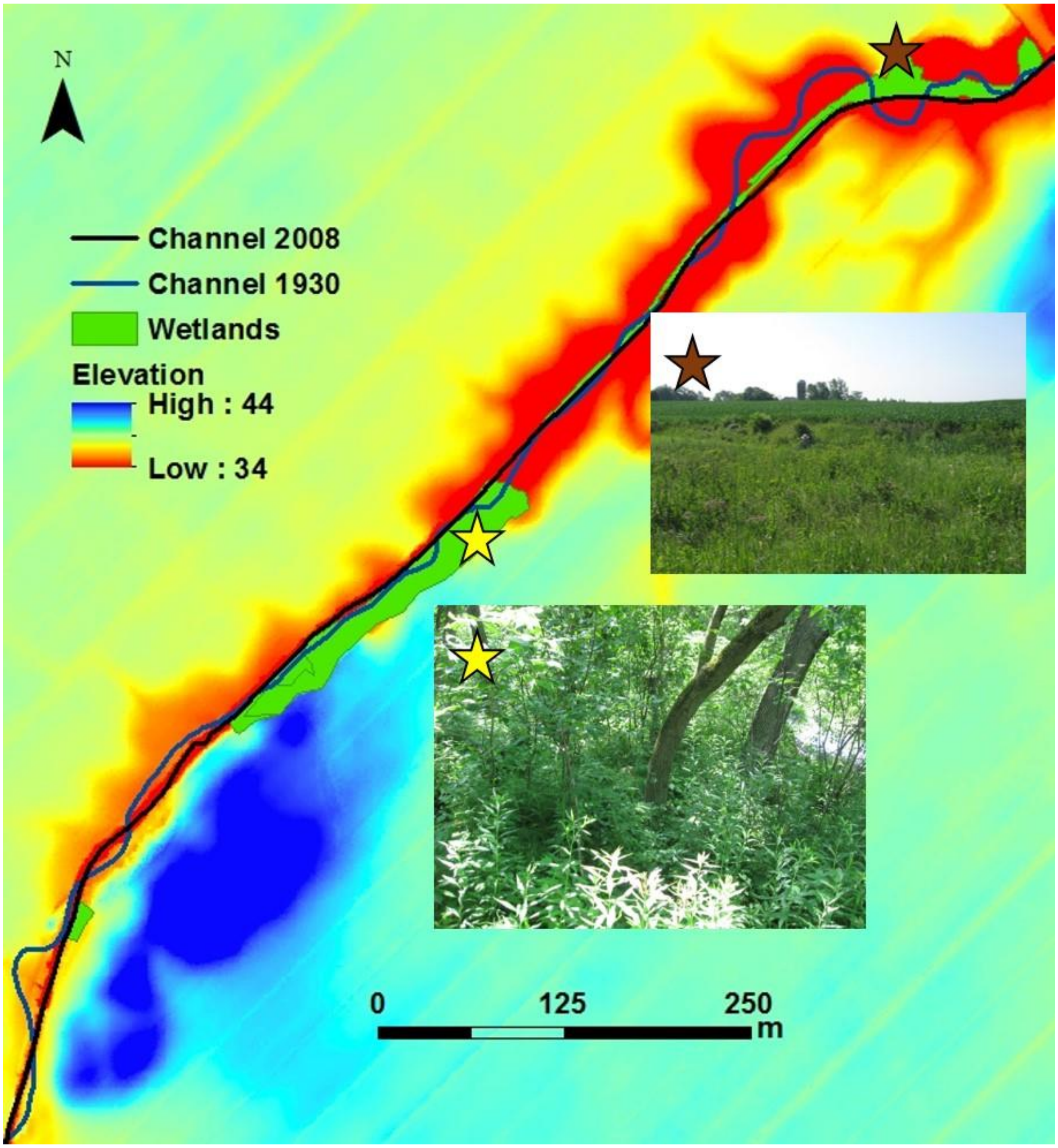


Figure 58: Vegetation units classified as wetland type for DF. Brown star photo of unit classified as a swamp, yellow star photo of unit classified as wooded swampland.

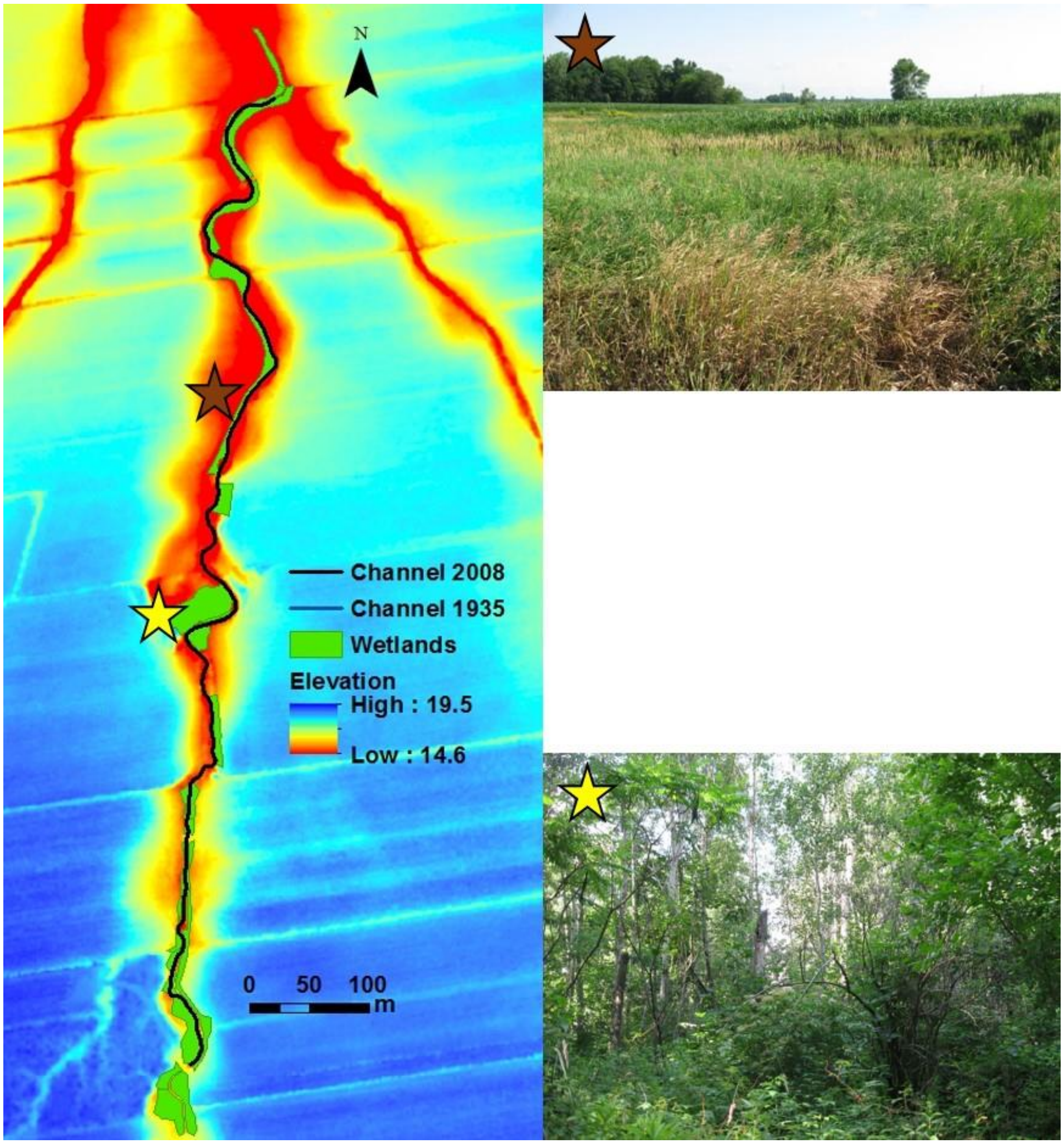


Figure 59: Vegetation units classified as wetland type for PPauB. Brown star photo of unit classified as a swamp, yellow star photo of unit classified as wooded swampland.

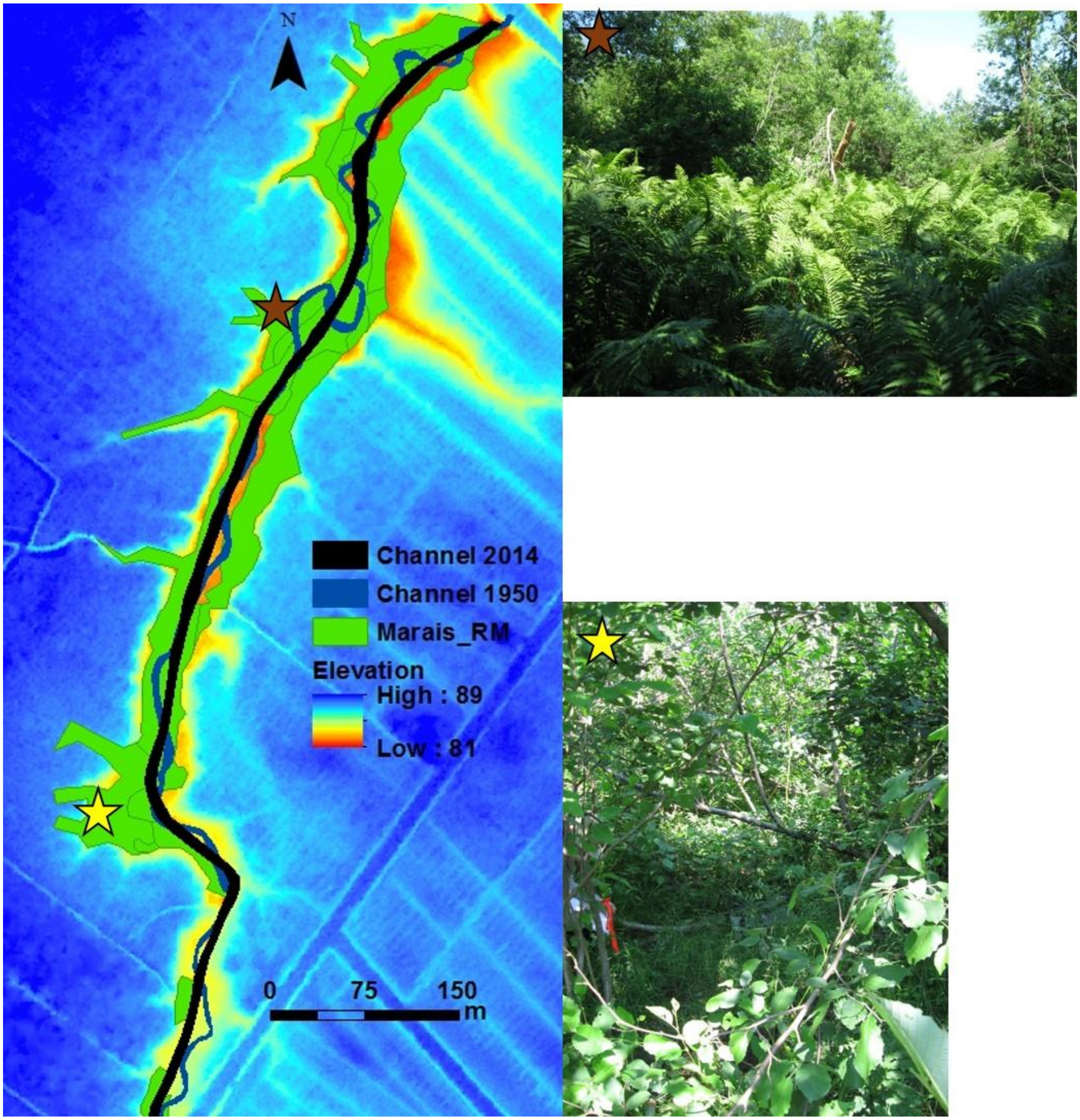


Figure 60: Vegetation units classified as wetland type for RM. Brown star photo of unit classified as a swamp, yellow star photo of unit classified as wooded swampland.

Table 7: Summary of floodplain loss to cropland and the percent of remaining vegetated floodplain classified as wetlands.

Site	Historical floodplain area (m^2)	Area assessed (m^2)	Area lost due to agriculture (%)	Wetland area (m^2)	Wetland portion of area assessed (%)	Wetland portion of historical floodplain (%)
DF	44,488	20,161	54.7	8,395	41.6	18.9
PPauB	32,292	14,634	54.7	8,558	58.4	26.5
RM	52,622	41,224.5	21.6	34,374	83.4	65.3

Wetland habitats have thus been identified at all three sites, mostly classified as swamp or wooded swampland. In general, assessment area is limited due to loss of riparian vegetation to cropland, with respect to the historical floodplain extent (Table 7). RM has maintained the largest portion of its historical floodplain as wetlands and had by far the greatest proportion of identified wetland habitats. DF and PPauB on the other hand both lost about 55% of their floodplain after channel straightening and are characterized by lower proportions of wetland habitat compared to RM. However, the vegetated patches that did remain at DF and PPauB also exhibited high species diversity and a clear presence of wetlands, suggesting that there is restoration potential here as wetlands persist despite the high degree of degradation of their respective watersheds.

4.3.6 Fish habitat

A fish habitat quality assessment was also conducted at the three study sites in the mid summer of year 2 under low flow conditions. The Qualitative Habitat Evaluation Index (QHEI) (Rankin, 2006) was adapted for use in anthropogenically disturbed rivers and in the Lowlands of Quebec (Massey et al., 2017). We focused on RM adding sections beyond the original study reach to include an upstream section which is heavily straightened and likely maintained by dredging, and a more natural downstream reach which had never been straightened. Results clearly show a low fish habitat quality in the anthropogenically-altered section while our study section which is classified as "recovering" shows good to excellent habitat quality similar to that of the downstream "reference" reach (Figure 61). It is interesting to note that the three ancient meanders at RM are characterized as excellent habitat. PPauB and DF on the other hand have mostly poor habitat quality, consistent with their degraded status (Figure 62). The lack of physical diversity and lack of riparian zone were the main contributors to low scores.

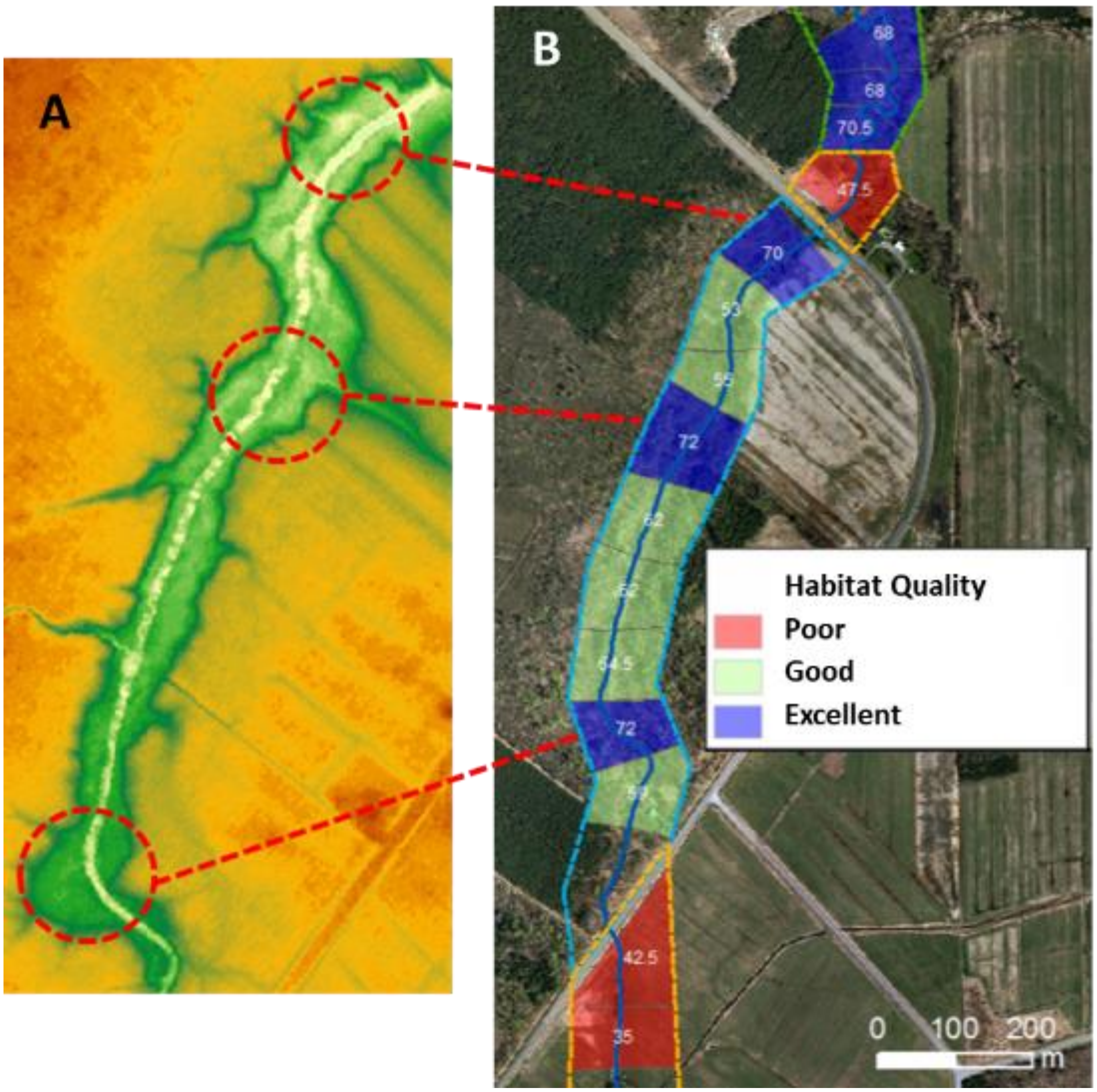


Figure 61: A) LiDAR showing general site locations and ancient meanders, B) results of habitat quality assessment for RM site.

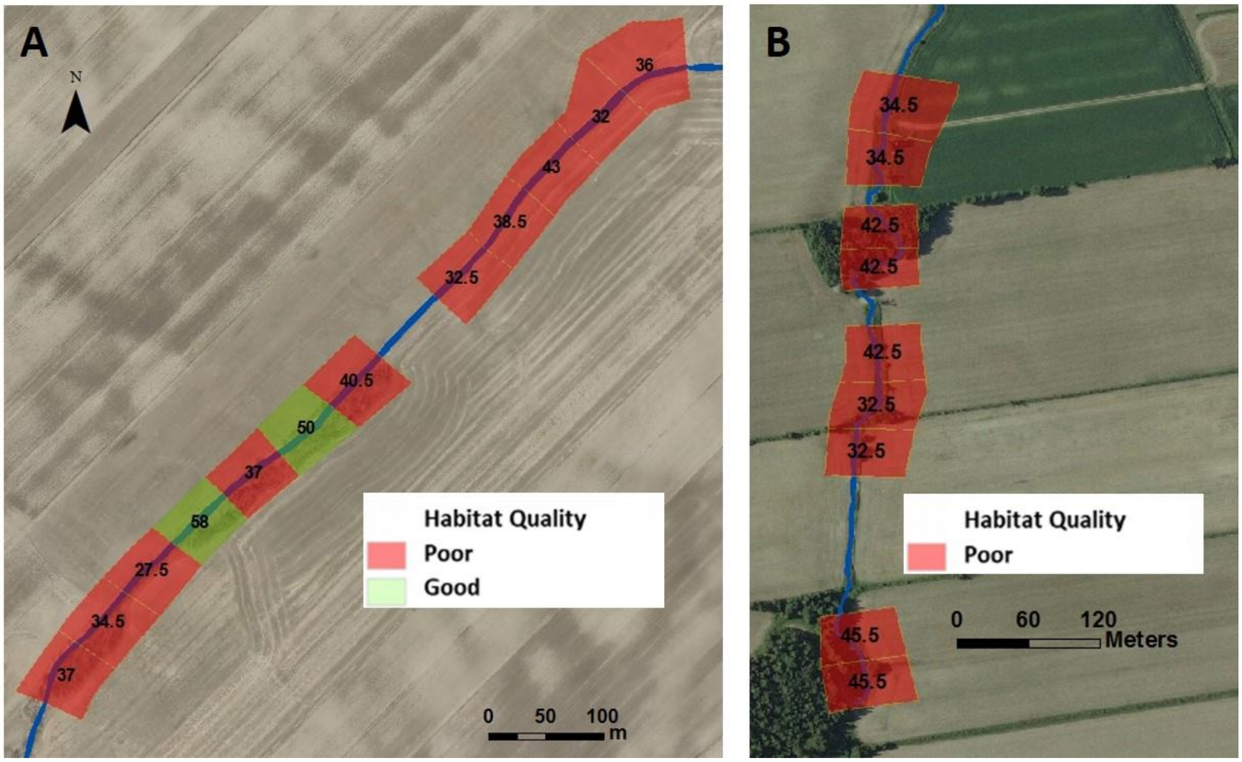


Figure 62: Results of habitat quality assessment for A) DF and B) PPauB.

4.4 Potential for wetland restoration

4.4.1 DF site

Overall, we evaluate DF as having an intermediate potential for wetland restoration compared to the other two sites PPauB and RM. In terms of hydrogeology, clear subsurface connections through multiple resurgence points bringing in fresh groundwater helps offset the negative impact that agricultural landscapes typically have on hydrology. Despite the presence of drainage ditches at the sites, the baseflow of the stream remains fairly constant and is not dry for months out of the year as it is at PPauB, which has positive implications for wetland restoration. The stratigraphic profile at DF is quite homogeneous and consists of either natural marine clay or agricultural fill. The presence of marine clay close to the surface in some areas is also a positive point for wetland restoration as it creates an impermeable layer upon which the surface water can pool. The topography of some ancient meanders also permits this as they are flat and at a similar elevation to the main channel. Cameras recording on site show that at least one of these ancient meanders is indeed connected to the main channel as it floods and pools water during peak flows (Figure 63A). Unfortunately, the landowner at DF is gradually filling up this ancient meander (Figure 63B) which demonstrates the important role that human interventions play when evaluating restoration potential and the necessity to include riparian owners in the restoration process. Furthermore, due to the lack of space provided to the stream and the resultant limited riparian zone, wetlands at this site are sparse and restricted to only a very small proportion of their historical floodplain. However, the moderating effect of the cooler groundwater on channel temperature from the resurgence points and the good fish habitat quality of some areas adjacent to vegetated riparian zone indicate that there is a potential for favorable physical conditions for fish and plant life.

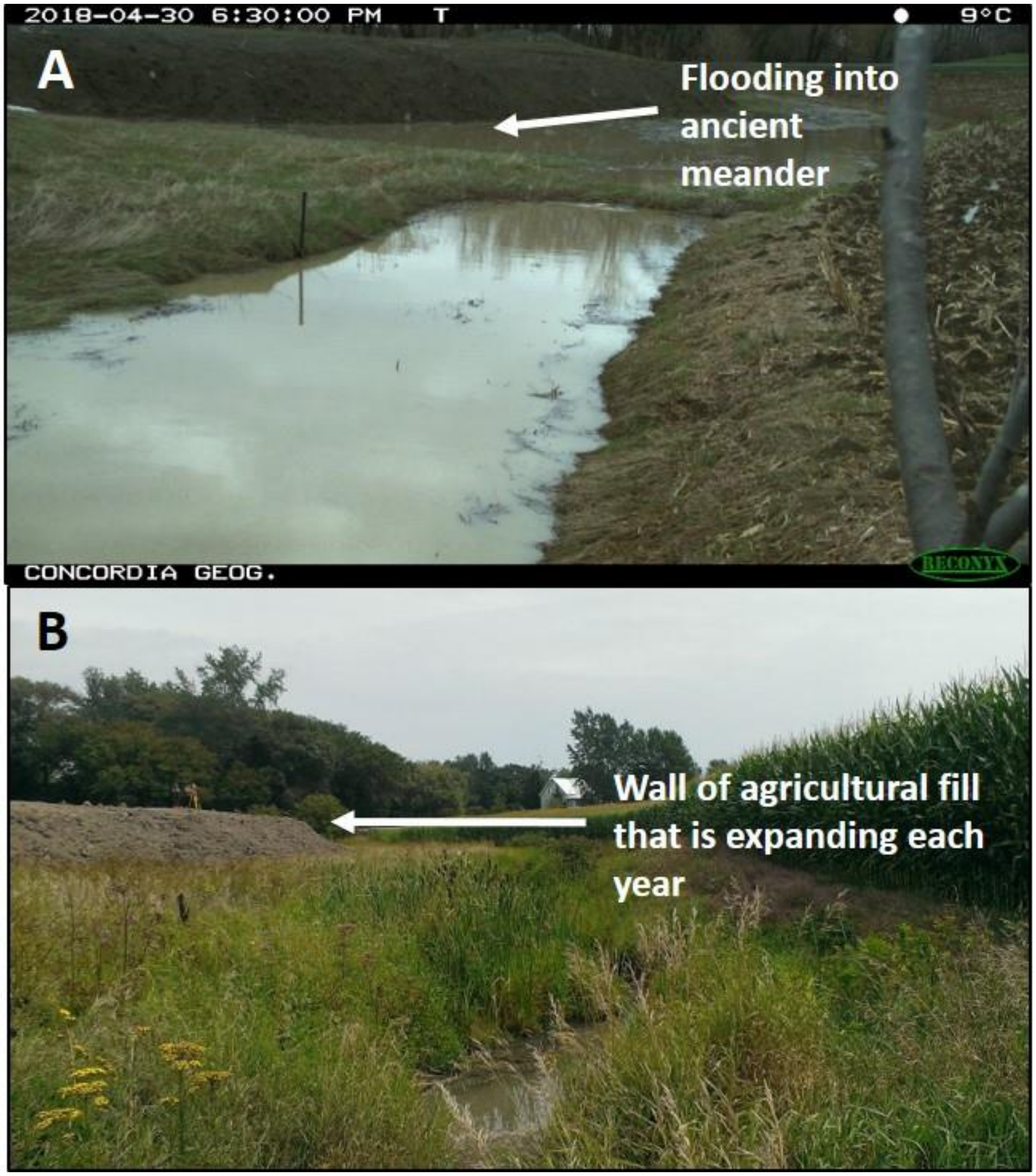


Figure 63: Topography of ancient meander at DF is favorable to maintain hydrological connections with the channel A); gradual filling of ancient meander greatly impedes restoration at this site B)

4.4.2 PPauB site

The restoration potential of wetlands at PPauB was evaluated as the lowest of the three study sites. Despite a dense network of piezometers in the ancient meander, which contained coarse sediments, no evidence was found that hydrological connections through subsurface processes are maintained here. Furthermore, the topography of the ancient meander that was artificially filled slopes toward the channel and is relatively hardpacked. As a result water cannot pool here and it is likely that little infiltration occurs. The low water table, influenced by the dense network of drainage ditches, contributes to the lack of water at this site. Indeed the stream is dry for several months during the summer. These conditions greatly lower the restoration potential of the site, at least through passive methods. However, awareness of the landowner has lead to interest about the restoration of wetlands on his property which has positive implications for restoration through active means. For example, this could be done through unburdening the ancient meander of the fine fill material that is clogging the alluvial sediments and providing a buffer zone which can become vegetated and help reduce sources of degradation from sediment accumulation. Furthermore, although the results of the vegetation assessment showed that wetlands are presently constricted to a narrow band in areas where agricultural fields are encroaching on the ancient floodplain, wetlands that do persist in vegetated patches between fields possess diverse assemblages of plant species which could be used as a source for future restoration projects.

4.4.3 RM site

Of the three sites RM stands out as having the best potential for wetland restoration and provides an example that passive restoration is indeed possible even in degraded agricultural watersheds. After the initial straightening and widening of the main channel, land use practices at our study site changed from a dominance of agriculture (pre 1950) to that of forest and wetland habitats, illustrating the importance of providing space to rivers which will allow natural processes to occur. At RM we see that the topography of the ancient meanders and the presence of alluvial deposits have allowed surface water to drain and pool while the impermeable layer of till (or clay depending on the meander) encourages the persistence of wetlands (Figure 64). Our results indicate that there is a strong potential for hydrological connections between the regional water table and the channel through the ancient meanders. Ecological assessments showed that more than 80% of the area evaluated was wetland habitat, while fish habitat quality was good to excellent, in contrast with upstream sections where agricultural practices persist that showed poor habitat quality.

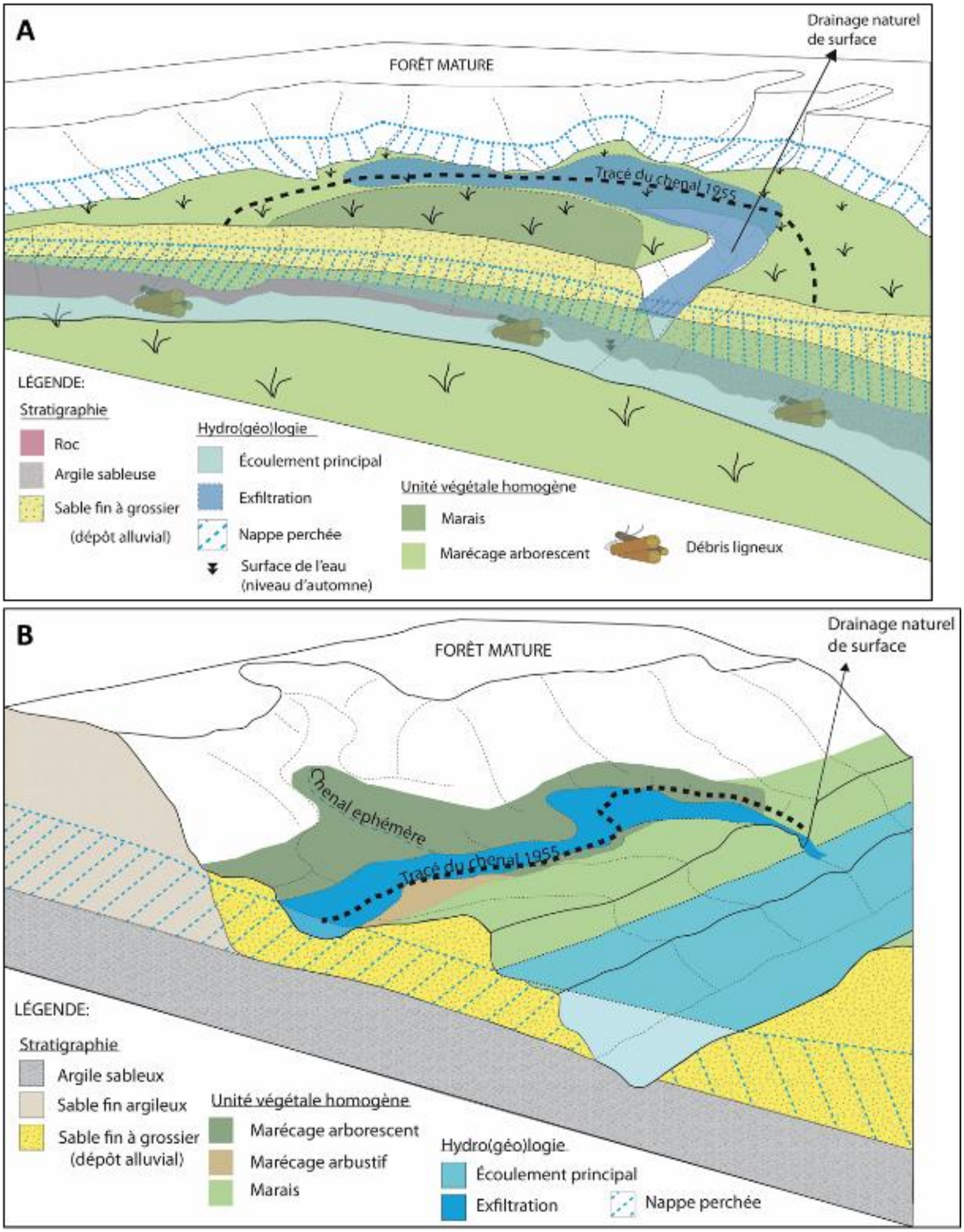


Figure credit: Jean-Philippe Marchand

Figure 64: 3D representation of ancient meander RM UP site A) and MID B).

5 PREDICTING RE-MEANDERING POTENTIAL WITH RVR MEANDER

In order to determine if passive or active restoration measures could be employed at the three study sites, our next objective is to predict the potential for each site to re-meander. Our approach was to use RVR Meander to simulate the long-term migration and meander development of straightened rivers. While this method has several limitations and is a gross simplification of the system, it does allow us to theoretically test how the current physical parameters of each site may affect its ability to passively re-meander as well as how the channel planform may change with respect to different restoration strategies. The results of this section will also help inform the role of channel migration and the creation of future meander oxbows on wetland restoration potential in degraded agricultural watersheds.

Modelling tools can help answer key research questions such as:

- Does the current hydro-morphological context of these small-straightened agricultural streams allow them to re-meander?

- How would physical changes in channel and floodplain conditions related to passive or active restoration practices affect their potential to re-meander?

5.1 RVR Meander, a tool for modeling long-term channel migration

RVR Meander is a toolbox originally developed by Abad and Garcia (2006) which has been used in several projects to model channel migration, providing insight into long-term stream planform evolution and helping to inform stream restoration projects (Langendoen et al., 2015). Initially RVR Meander used the classical migration approach (CMA) developed by Ikeda et al. (1981) to calculate local migration rates. The CMA essentially linearly relates channel migration to near bank velocity with an empirically derived dimensionless coefficient. Given that input parameters, including the derived coefficient, are treated as constant throughout the simulation, the CMA can be useful predominantly as a simple analytical model for streams with low planform and bed diversity. However, while the CMA may be good for improving our theoretical understanding of channel migration and re-meandering, it was criticized for its inability to be applied practically as it does not always reproduce observed patterns of meandering (Motta et al., 2012). The main reason for this disparity is due to the complexity of physical processes (including responses to anthropogenic influences) which makes calibrating the migration coefficient difficult. In response to this criticism, efforts were made by Motta et al. (2012) to upgrade RVR Meander with streambank erosion algorithms from the CONCEPTS model (Langendoen and Simon, 2008) to provide an alternative method for evaluating channel migration known as the physically-based meander-migration approach (PMA).

Clearly, the PMA is useful in relating channel migration to processes of bank erosion (both fluvial and mass wasting) which accounts in part the potential for heterogeneity in floodplain stratigraphic composition. However, while PMA is considered an improvement over the CMA in some cases, others consider that results are simply different and not necessarily better (Motta et al., 2012). For the purposes of this study, we chose to proceed with the CMA due in part to its simplicity as well as the more limited required information. Furthermore, our study sites as selected are anthropogenically straightened and widened channels exhibiting relatively low-sinuosity and bed diversity, conditions which favor the use of the CMA.

5.2 Determining rates of channel migration and coefficient

Following Ikeda et al. (1981), the CMA migration rates are calculated as:

$R = E_0 (U_b - U_{ch})$

Where R is the local migration rate (m/s), U_b and U_{ch} are the average velocities of the bank and channel respectively, and E_0 is the dimensionless migration coefficient.

As E_0 is typically determined by calibrating the model against past positions of the channel (Motta et al., 2012), historical positions (centerlines) were digitized for each site based on available air photography, along with the most recent known position of the channel and the valley centerline using orthoimages and LiDAR generated DEMs (Table 8).

Here, we are interested in determining past rates of natural channel migration, therefore only sections of the respective study areas where significant channel migration (greater than the 5-m georeferencing error) had occurred between historical and present channel positions were considered. Sections that were clearly influenced by meander cut-offs or anthropogenic influences such as bank stabilization or straightening were left out. Bank migration is then determined by dividing the area between past and present channel centerlines by 0.5 times its perimeter as described by Gaillot (2007) (Figure 65). This value is then divided by the number of years separating the two channel positions to obtain the "reference" migration rate (Table 8). It should be noted that even though we tried to select the most natural sections of the channel (ex: sinuous, riparian buffer present, no obvious culverts) the strong influence of agricultural activity surrounding out study sites, particularly PPauB and DF, limit our ability to reliably evaluate migration rates.

Table 8: Dates of channel centerlines available based on historical air photos, orthoimages and LiDAR generated DEMs along with estimated migration rates for each site.

Study site	Historical photo (year)	Most recent position (year)	Historical, reference migration rate (m/yr)
DF	1930	2008	0.147
PPauB	1935, 1964	2008	0.147
RM	1950, 1960	2014	0.122

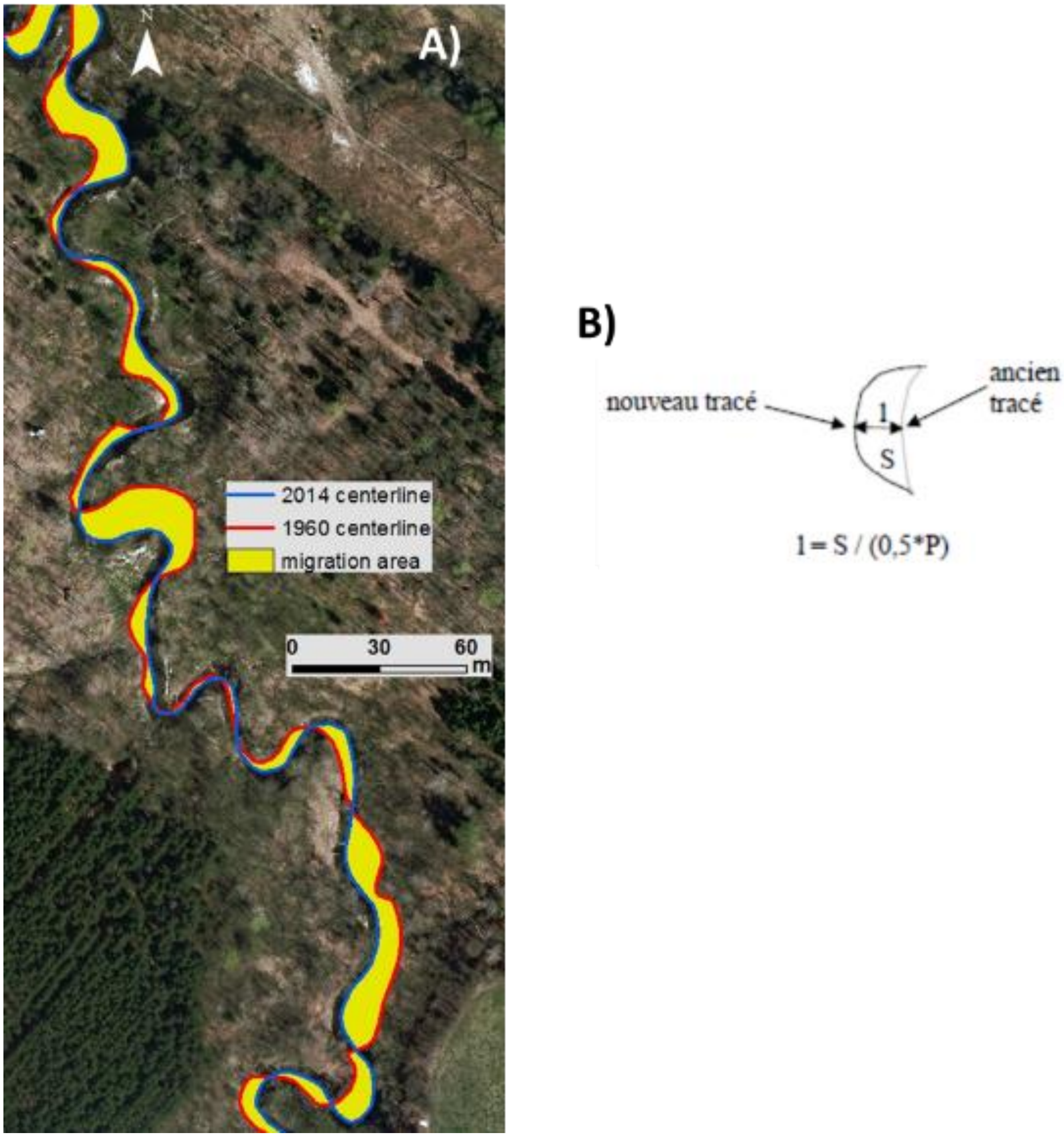


Figure 65: A) Example of determination of channel migration from historical channel position in 1960 to 2014 for Ruisseau Martin Site; B) calculation of average migration distances for a particular polygon (Gaillot, 2007)

Once the historical migration rate is determined, we can use it to calibrate the migration coefficient E_0. To accomplish this, RVR Meander is used to simulate the migration of the channel from its historical position to its current one using different values of E_0 until simulated migration rates match up within 10% to the reference historical rates (Table 9).

Table 9: Comparison of historical/reference migration rates to simulated rates

Study site	Historical, reference migration rate (m/yr)	Migration coefficient E_0 used	Simulated migration rate (m/yr)	Difference (%)
DF	[See Petite rivière Pot-au-Beurre]			
PPauB	0.147	$1.5*10^{-7}$	0.146	0.1
RM	0.122	$4*10^{-8}$	0.134	9.8

Note: The historical migration rate and coefficient E_0 was already calibrated for PPauB in a previous study using this methodology (Roux, 2012) and was therefore retained. These values were also applied to DF where the migration rate could not be determined due to significant channel straightening affecting historical channel positions. As noted by Motta et al. (2012), anthropogenic influences can make the calibration of E_0 quite difficult, however due to similarities in hydro-geomorphological conditions between the two sites we deemed it reasonable to use the same migration coefficient. All values of E_0 determined are small, ranging from 10^{-7} to 10^{-8}, as typically expected (Motta et al., 2012).

5.3 RVR meander input parameters

The calculation of other input parameters for RVR Meander (Table 10) (width, slope, discharge, roughness, sediment size) are briefly described below. It should be noted that while all these parameters will naturally vary throughout a stream reach they are treated as constant here.

Table 10: Summary of RVR Meander input Parameters for each site

Study site	Length (m)	Width (m)	Valley Slope	Discharge (m^3/s)	Manning	Sediment size (D_{50}) (mm)	Migration coefficient E_0
DF	1405	4.1	0.0024	0.052	0.045 (Paradis, 2015)	0.011	$1.5*10^{-7}$
PPauB	994	3.1	0.0027	0.082	0.05	0.004	$1.5*10^{-7}$
RM	1020	7.8	0.0021	0.855	0.07	0.25	$4*10^{-8}$

Width: Average bankfull width is used, calculated by first digitizing the bankfull area with the aid of orthophotos and LiDAR generated DEMs and dividing by the length of the channel centerline.

Slope: Valley slope is used (Motta et al., 2012), averaged across the respective study site using high precision elevation data from the LiDAR.

Discharge: the value of discharge typically used appears to vary from study to study with terms such as effective discharge or bankfull discharge (1.5-yr recurrence interval) being used interchangeably (Langendoen et al., 2015). One study used an input discharge value which lay between the average and maximum of the mean annual streamflow (Motta et al., 2012). For this study we used the mean annual discharge as we had at least one year of water level data collected for each site. Rating curves for each site were established between continuous water level measures and on site discharge readings at multiple flow stages (including spring flows).

Manning (roughness coefficient): generally this coefficient represents flow resistance within a stream, taking into account factors ranging from bank vegetation and boulders to channel sinuosity and bed formations (ex: deep pools). The value of this coefficient can be determined through comparison between channels, consulting tables (e.g. Chow, 1959) the description of which most closely resembles the field observations of our streams. It should be noted that anthropogenic influences (ex: bank stabilization, riparian vegetation removal, presence of culverts) can greatly affect local manning values. For example, the value of 0.02 reported by Roux (2012) for the PPauB appears to underestimate that of our site, which we determined to be closer to 0.05 due to the presence of several culverts (Figure 66D). Culverts can be a major restriction at low flow for incised/frequently dredged channels (Figure 66A) while also creating blockages at high flows as debris accumulate in the narrow openings (Figure 66B,C).

Sediment size: Determined with field sampling of bed and bank materials throughout the 1-km reach of each study site. Median grain size (D_{50})was calculated.

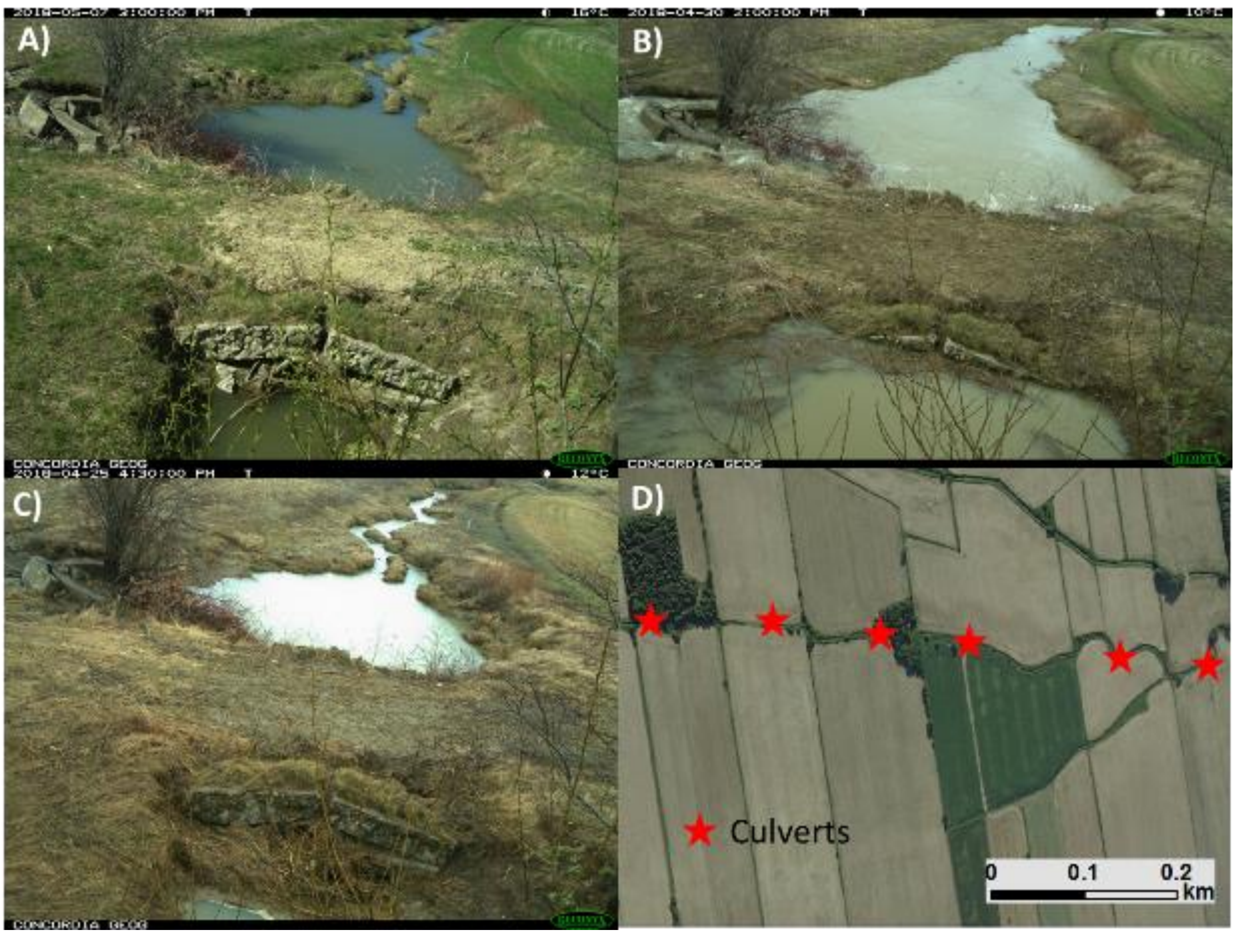


Figure 66: A) Example of a culvert at PPauB at low flow causing a restriction (narrowing) for incoming flow while also contributing to the formation of a relatively deep pool at the outlet of the outlet; B) flow is diverted around culvert at high flow (spring flood) due to it`s narrow opening; C) debris left over from previous flood further restricts flow through the culvert; D) 6 culverts identified over a 1.5km reach at the PPauB study site.

5.4 Determining re-meandering potential

In order to determine (theoretically) if each of our study reaches have the potential to re-meander after having been disturbed from anthropogenic practices (channel straightening and channel widening), RVR Meander is used to simulate the respective channel centerlines given their current hydro-geomorphological context in the near (decades) and distant (centuries) future.

The first variable looked at is **time to significant migration**. Time to significant migration is defined as either the first time step where the simulation doesn't crash or a migration greater than 5m (georeferencing error) has occurred in at least one meander.

The second variable of interest is time to first year where the maximum meander amplitude (MMA) of the simulated centerline surpasses the historical MMA. Since these sites show evidence of anthropogenic disturbances (areas of straightening, removal of vegetation, presence of culverts) as far back as 1930, the historical channel positions (including MMA) are not necessarily indicative of the true potential of the system. Therefore it was decided that MMA would be estimated as roughly 0.5X the floodplain extent (digitized using LiDAR generated DEMs) at its widest section (to be conservative). The threshold values of MMA are 30m for DF, 35m for PPauB and 50m for RM.

The length of each simulated centerline is also used to compare between simulated results as well as past and present channel distances.

5.5 Understanding the influence of restoration efforts on re-meandering potential

Clearly, the restoration of the form and/or function of streams in agricultural settings, including the potential rehabilitation of wetland ecosystems, is a complex process and cannot be easily simulated with simple methods such as the migration coefficient in RVR Meander. However, certain input parameters can be modified to run sensitivity analyses and understand how the system may react to certain changes in the hydrogeomorphology context (Table 11).

The following three scenarios represent potential changes in RVR Meander input parameters related to passive and active restoration efforts as well as taking into account projected climate change driven discharge increases.

Scenario 1: Slope is decreased by 25% to represent potential reprofiling related to complete channel relocation or restoration through active re-meandering. In some cases small agricultural streams which have been straightened have been relocated quite significantly from their entire channel bed (not just the ancient meanders). Such reconstructed channels may have milder slopes (lengthening the channel would result in a decrease in slope) compared to their shorter (channelized) counterparts (Simon and Rinaldi, 2006). This scenario is also interesting as it allows us to theorise the potential to migrate naturally after a restoration effort.

Scenario 2: Discharge is increased proportionally to increases in total annual precipitation as projected by 12 climate change scenarios. For the purposes of this study we have taken the furthest available projection (2100) where total annual precipitation across the three sites is expected to be 14 – 16% greater than today's values (see section 6, Figure 80 for details). As we do not have a rainfall/discharge rating curve established we simply assumed a linear relationship.

Scenario 3: While channel widening generally accompanies practices of channel straightening, conducting simulations with even wider channels (ex: 25% wider) may help us to understand how the system will respond to passive restoration. The rational here is that as the stream begins to re-meander from its straightened and degraded morphology, changes in average cross-section morphology will also occur (i.e. width is entered as a constant in RVR Meander but we would expect this to change through time). Significant evidence exists that channelized streams in low-relief regions such as agricultural fields often experience aggradation (Simon and Rinaldi, 2006). This aggradation explains in part why these straightened streams such as DF and PPauB are regularly dredged in attempts to reduce flooding into adjacent crop fields. Indeed the aggradation and widening are part of the natural evolution of constructed channels (Stage V of Simon and Hupp, 1986), representing a potential intermediate stage of passive restoration.

Table 11: Input parameters for RVR Meander with adjustments for various restoration scenarios.

Study site	Scenario	Slope	Discharge (m^3/s)	Width (m)
DF	Actual	**0.0024**	**0.052**	**4.1**
	1: slope decreased (25%)	0.0018	0.052	4.1
	2: discharge increased (16%)	0.0024	0.06	4.1
	3: width increased (25%)	0.024	0.052	5.2
PPauB	Actual	**0.0027**	**0.082**	**3.1**
	1: slope decreased (25%)	0.002	0.082	3.1
	2: discharge increased (16%)	0.0027	0.095	3.1
	3: width increased (25%)	0.0027	0.082	3.9
RM	Actual	0.0021	0.855	7.8
	1: slope decreased (25%)	0.0016	0.855	7.8
	2: discharge increased (16%)	0.0021	0.975	7.8
	3: width increased (25%)	0.0021	0.855	9.7

5.6 Migration results

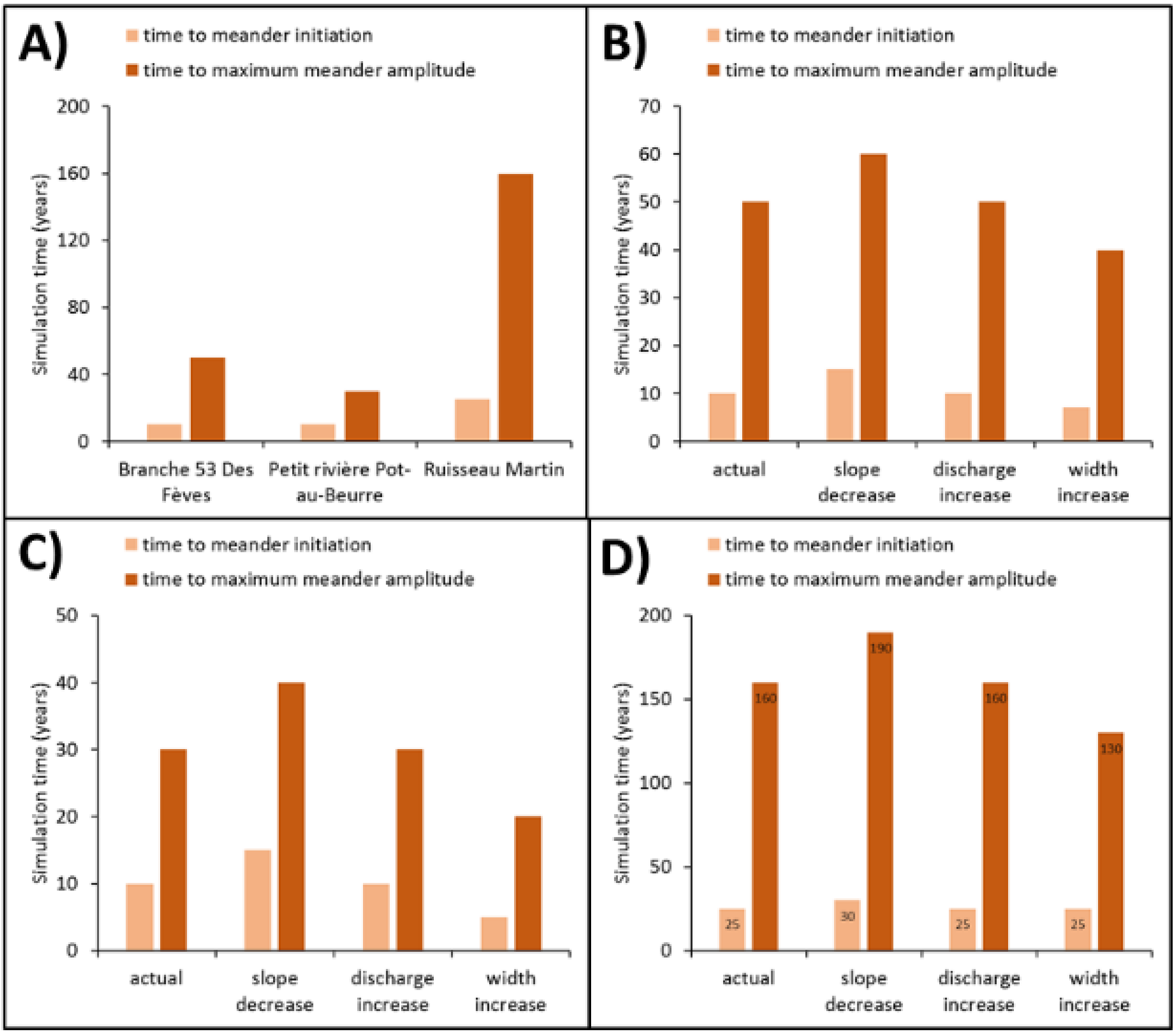


Figure 67: A) Simulation results of time to meander initiation and time to maximum meander amplitude for the three study streams, Actual and adjusted parameters for B) DF, C) PPauB and D) RM.

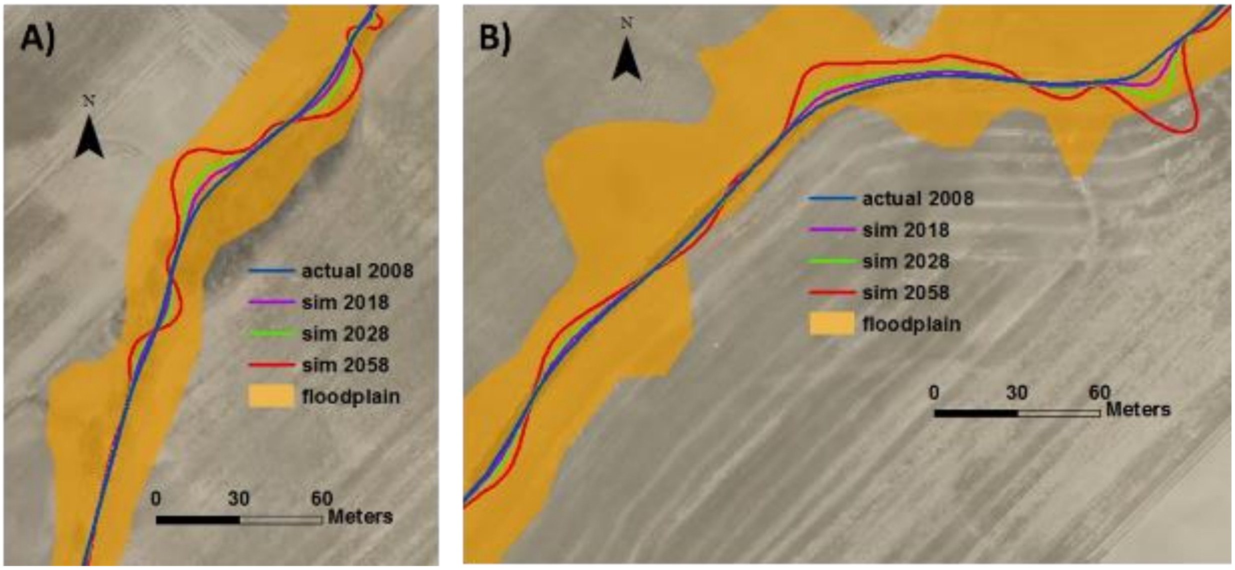


Figure 68: RVR Meander simulation results for DF using the actual input parameters for A) upstream section, B) downstream section

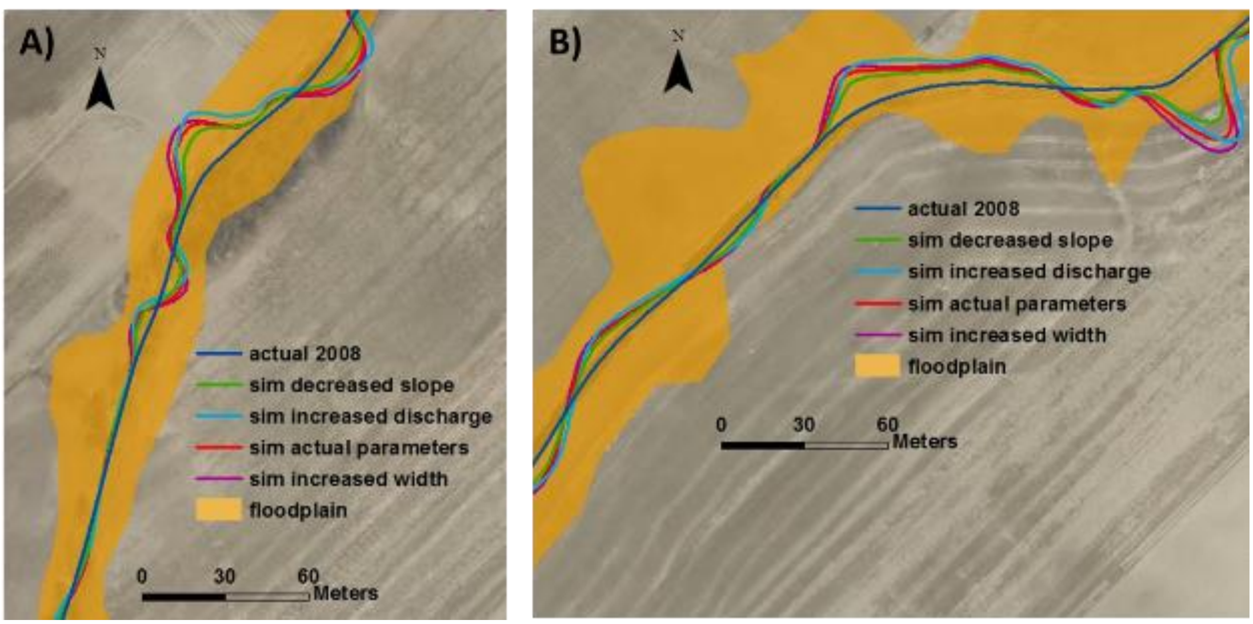


Figure 69: RVR Meander results for DF simulating actual and adjusted parameters over 50 years (2058) for A) upstream section, B) downstream section.

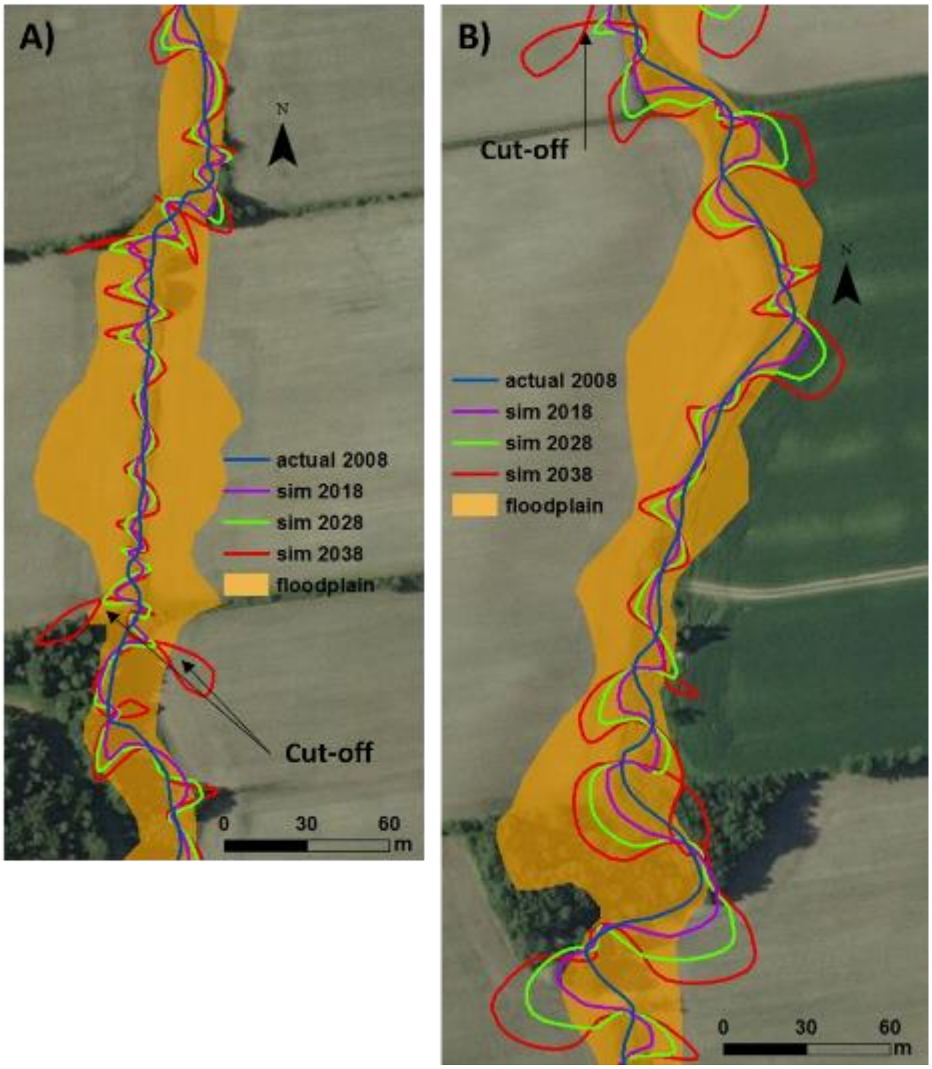


Figure 70: RVR Meander simulation results for PPauB using the actual input parameters for A) upstream section, B) downstream section.

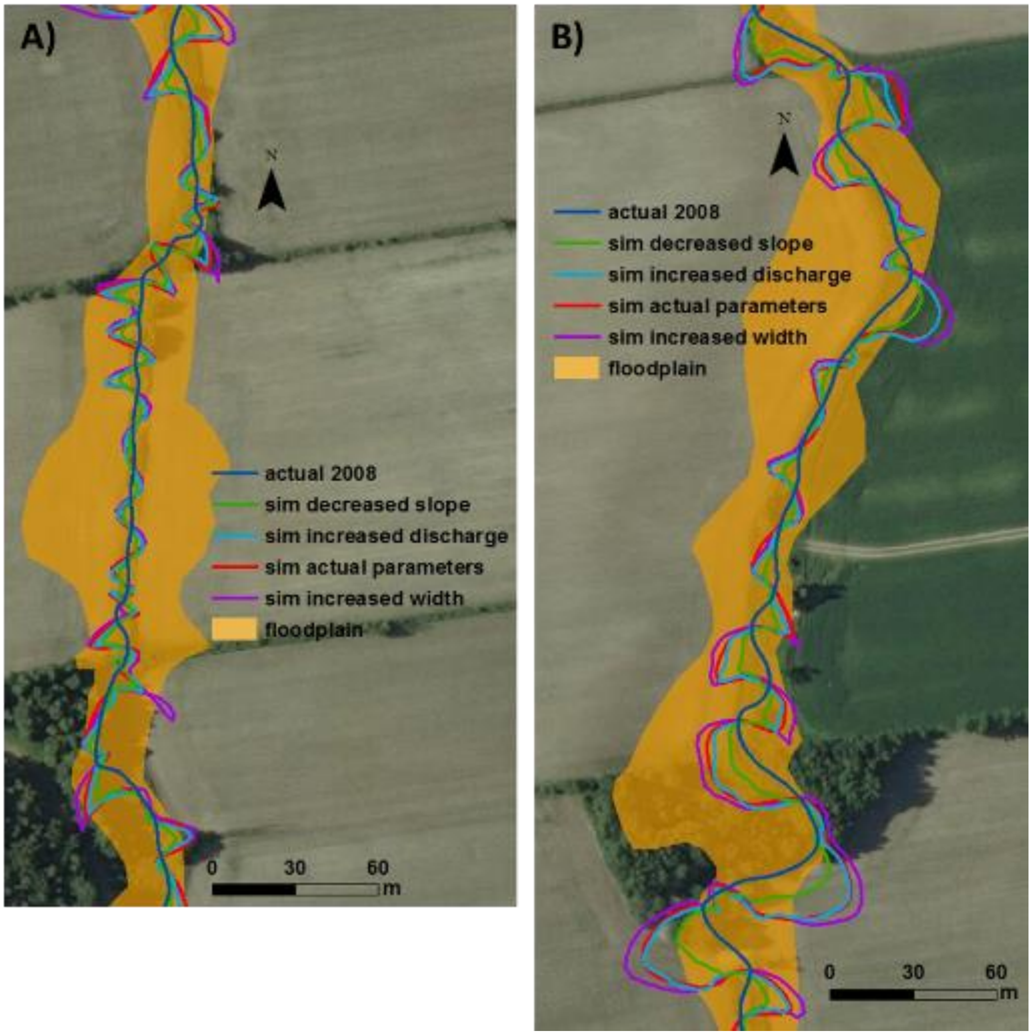


Figure 71: RVR Meander results for PPauB simulating actual and adjusted parameters over 20 years (2028) for A) upstream section, B) downstream section.

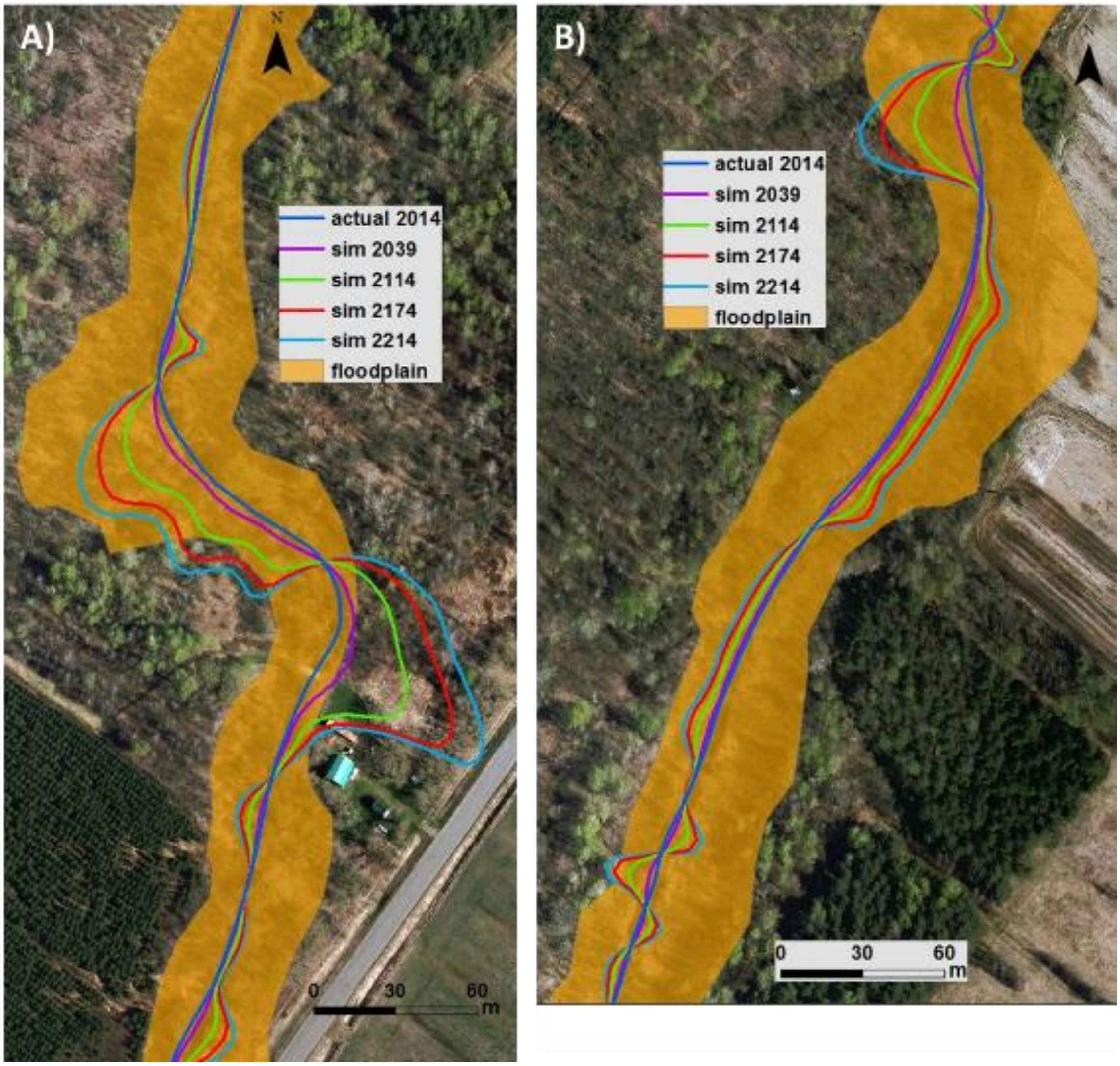


Figure 72: RVR Meander simulation results for RM using the actual input parameters for A) upstream section, B) downstream section.

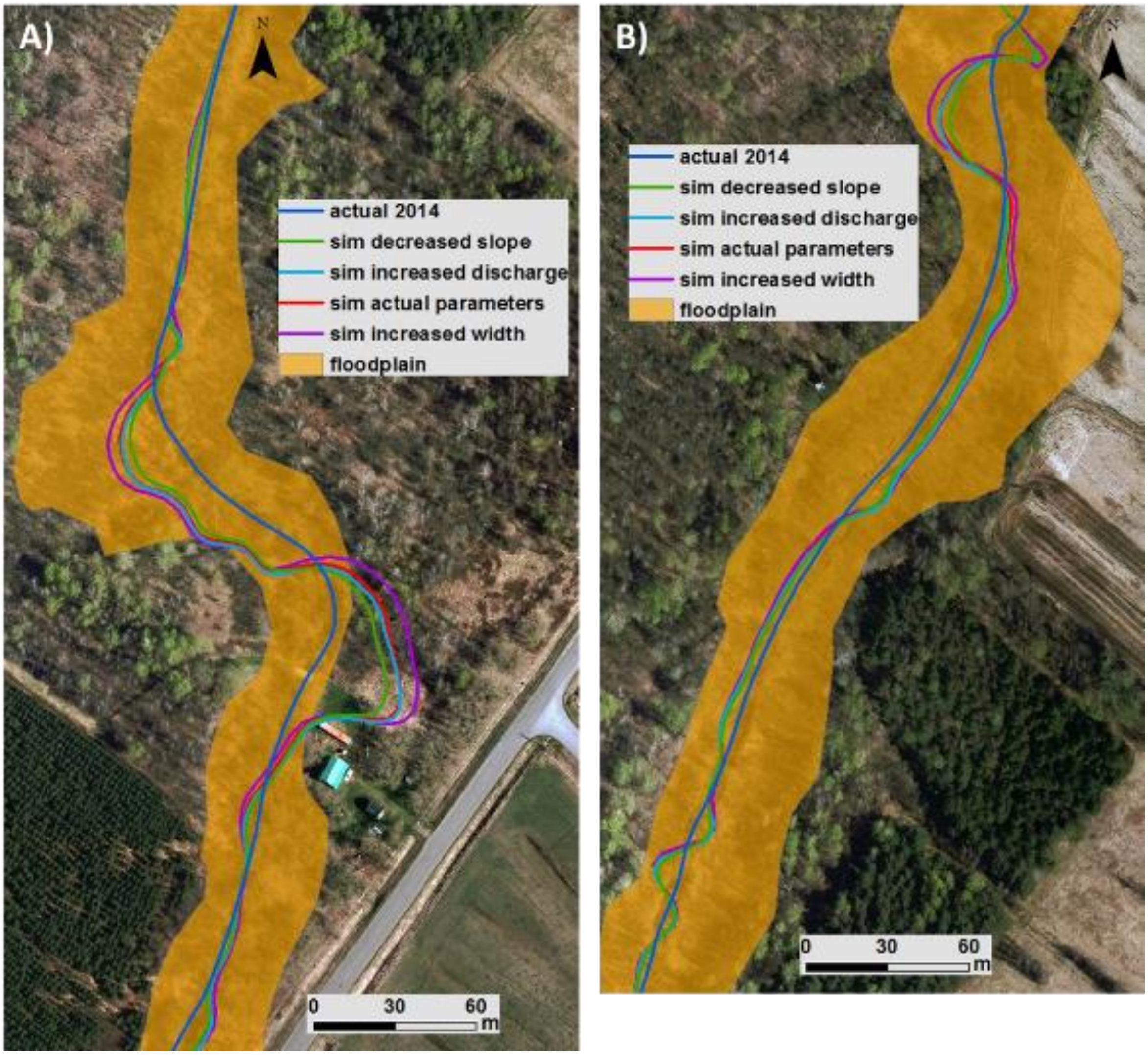


Figure 73: RVR Meander results for RM simulating actual and adjusted parameters over 100 years (2114) for A) upstream section, B) downstream section.

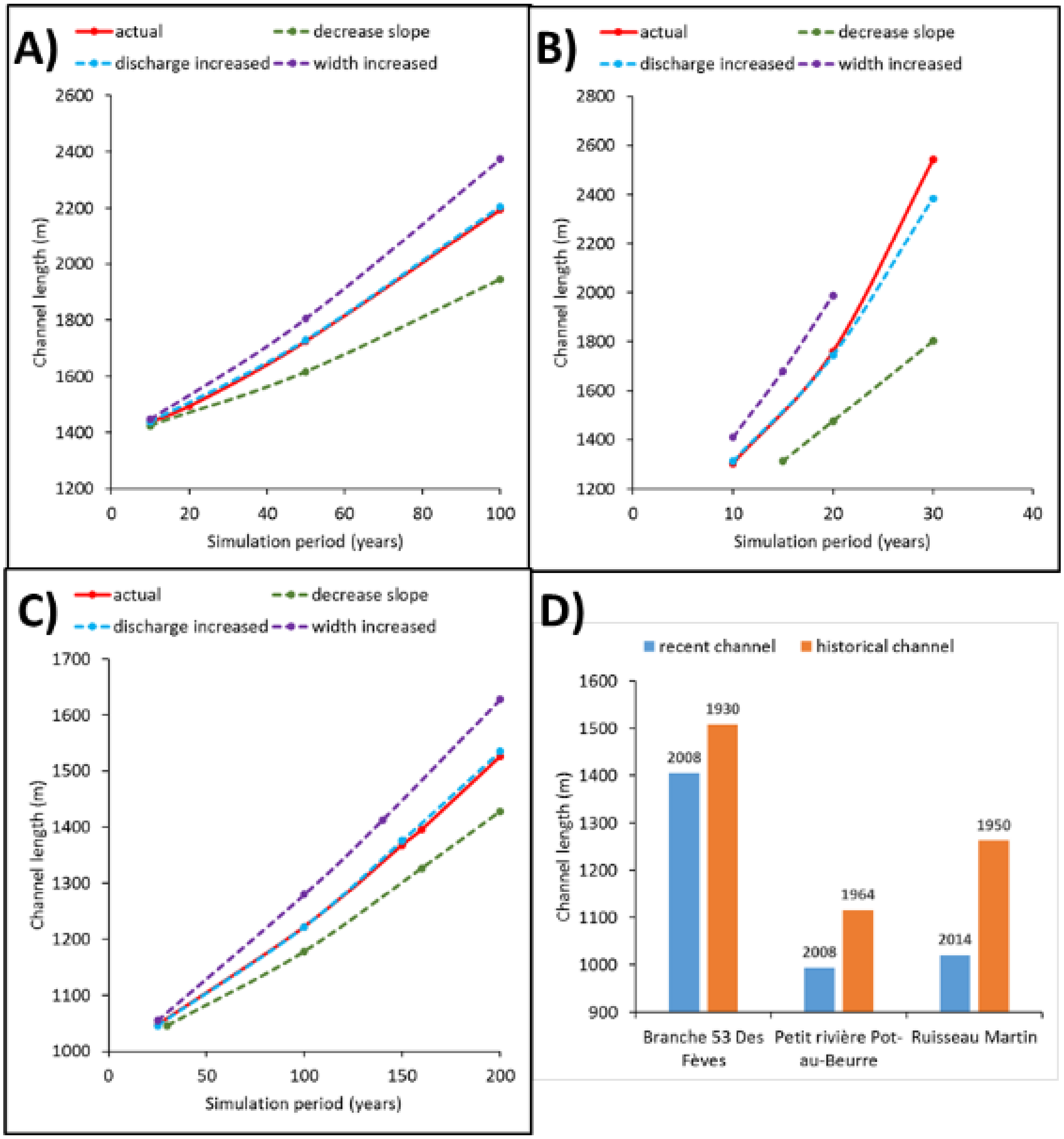


Figure 74: Comparison of channel lengths across different time periods and scenarios simulated in RVR meander for A) DF, B) PPauB, C) RM; as well as D) comparison of channel length between pre- and post-channel straightening intervention.

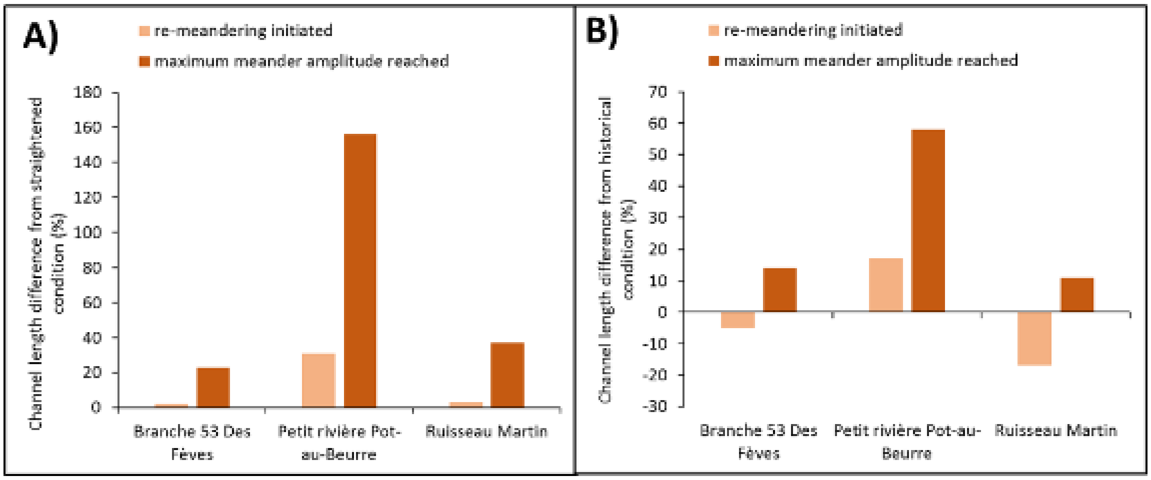


Figure 75: Comparison between sites of the magnitude of difference for total channel length when re-meandering was initiated and maximum meander amplitude was reached. A) Percent difference between simulated results and the most recent straightened condition of the site while B) Percent difference between simulated results and the historical, pre-straightened context.

5.7 Discussion

Results of simulations indicate that the current hydro-morphological context of the 3 study sites provides the potential to re-meander. For all simulations re-meandering was observed after 10 – 25 years depending on the stream. In the case of DF and PPauB re-meandering was observed sooner (10 years) compared to RM (25 years). Similarly, simulation results of time to maximum meander amplitude show that, theoretically, PPauB could recover the quickest (after 30 years) compared to DF (after 50 years) and RM (after 160 years)

This can be explained in part by differences in:

A) input parameters: RM greater Manning's n (0.07) due in part to the large amount of boulders, log jams and developed riparian vegetation - compared to DF (0.045) and PPauB (0.05). The migration coefficient as calibrated for RM was also much lower compared to the other sites ($4*10^{-8}$ compared to $1.5*10^{-7}$).
B) stream power: Stream power (energy available for erosion across a reach) calculated for each site (Table 12) explains why PPauB migrates more than DF. For similar channel dimensions, profiles and the exact same migration coefficient, PPauB has a unit stream power more than double that of DF (12.5 vs 5 Watts/m^2) due predominantly to the much smaller bankfull discharges observed at DF. Although unit stream power of RM (23 Watts/m^2) is much higher than PPauB and DF, its migration is slower, likely due to the smaller migration coefficient used.
C) The longer time to maximum meander amplitude simulated at RM compared to DF and PPauB may also be due to the possibility that the migration rates of DF and PPauB are overestimated because of how RVR Meander treats grain size (see below) or because RM simply has a wider floodplain (50m) compared to 35m at DF and 30m at PPauB.

However, results must be taken with some caution as there are many confounding issues. For instance, the presence of culverts which restrict mobility (Figure 66) may limit meandering potential. Also, the process of calibrating the migration coefficient is done on more "natural" sections of the channel and may therefore be overestimating the respective sites ability to re-meander. As noted by Motta et al. (2012) the migration coefficient E_0 is difficult to calibrate in areas experiencing high degrees of anthropogenic disturbance. Thus, our confidence in E_0 will be different for each site depending on the degree of anthropogenic influence. For example, we may be more confident in the migration rate coefficient calibrated at RM which has undergone passive restoration and a lessening in anthropogenic disturbances in the past 60 years. In comparison, the confidence of E_0 is less for PPauB and DF where persistent human influences continue to affect channel morphology, centerline position, and therefore, the establishment of reference migration rates.

It is also impossible to account for clay banks which are resistant to erosion using the CMA as grain size is treated as a roughness element. For instance, clay particle sizes of <0.002mm will produce simulation results with greater migration results compared to sand ~2mm. However, we know that hard-packed clay along the banks of many agricultural channels would actually reduce potential migration compared to sand. As such, simulation results likely overestimate migration potential of both PPauB and DF. Although it is not clear how this overestimation can be quantified, theoretically, reducing the migration coefficient E_0 can compensate for the bank cohesion resulting from the clay particles. This is a simple exercise as E_0 and the migration rate are linearly related. For example, dividing the coefficient by a factor of 2 essentially halves the migration rates. In this scenario, although simulation results show that re-meandering still occurs at PPauB and DF, migration rates become slower than those determined for RM. Furthermore, using the input E_0 from RM (i.e. $4*10^{-8}$) for PPauB and DF simulations rather than the original values used (i.e. $1.5*10^{-7}$) results in a simulation crash. This indicates that re-meadering is not likely to occur at PPauB or DF if applying the same migration coefficient as RM and demonstrates how sensitive the model is to input E_0 values.

Interestingly the unit stream power of all sites is lower than the critical threshold of 25-35 W/m^2 (Brooks, 1987) typically used to determine the initiation of morphological adjustments due to erosion. However, contradictory results have been reported before, particularly for RM where reaches that had a unit stream power < 25 W/m^2 showed signs of bank erosion or historical channel migration and vice versa (Olsen, 2012). This is not surprising as bank erosion is a complex process, particularly when anthropogenic influences are involved, and therefore cannot be described solely based on hydraulic variables such as unit stream power. Despite the low values of unit stream power recorded, all sites showed signs of erosion and deposition (Figures 76-78) with RM also exhibiting woody debris dynamics, suggesting channel migration is in progress there. However, in the cases of PPauB and DF, which are in close proximity to agricultural fields, potential re-meandering may be inhibited due to frequent dredging (both sites were was last dredged in 2013).

Table 12: Summary of unit stream power

Study site	Bankful discharge (m^3/s)	Channel Slope	Bankful width (m)	Stream power (W/m)	Unit stream power (W/m^2)
DF	0.9	0.00235	4.1	20.72	5
PPauB	1.99	0.002	3.1	39	12.5
RM	5.8	0.00313	7.8	178.25	23

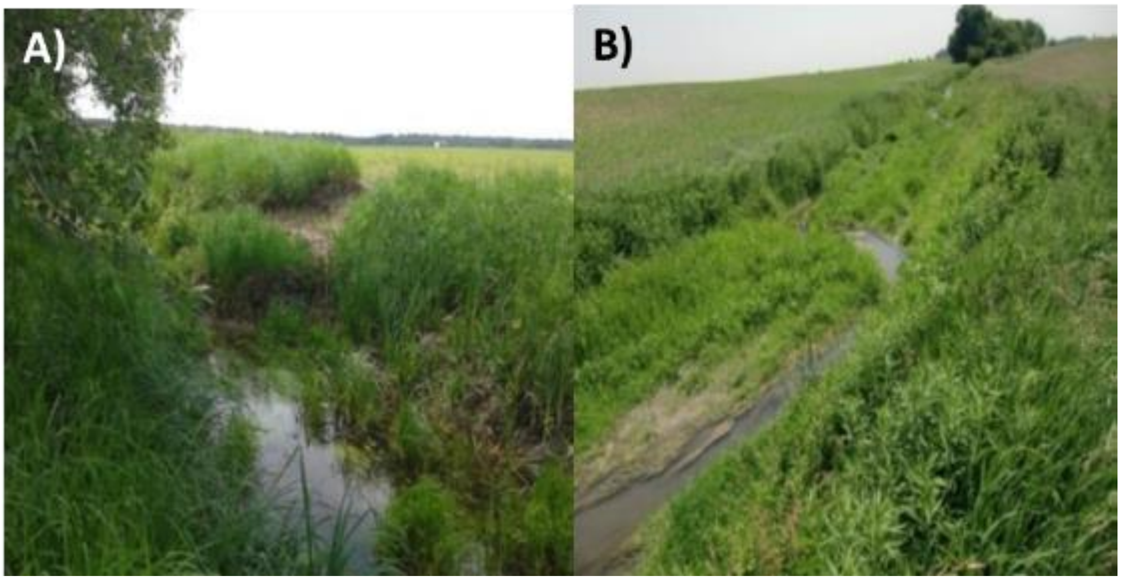


Figure 76: Field examples of erosion (bank failure) A) and deposition (side channel bar) B) for DF (photo: Paradis, 2015).

Figure 77: Field examples of bank erosion and deposition for PPauB A) upstream and B) downstream.

Figure 78: Field evidence of current channel migration for RM. A) vegetated mid channel bar, B) log jam, C) fluvial erosion along bank and D) channel spanning woody debris.

The question how would physical changes in channel and floodplain conditions related to passive or active restoration practices affect their potential to re-meander can be addressed by looking at the various tested scenarios.

Scenario 1: Decreasing slope by 25% increases both time to initiation of re-meandering and time to reach maximum meander amplitude (compared to actual input parameters) for all sites. This is not insignificant as it results in decades difference for time to reach maximum meander amplitude. Furthermore, according to results of differences in channel length (Figure 75) this disparity may increase for longer simulation runs. Similar results were reported by Roux (2012) who noted large decreases in channel migration following simulations with an experimentally decreased channel slope. While these findings are not surprising, as slope is directly related to stream power, they have important implications with regards to potential restoration efforts. For instance re-meandering of straightened Lowland streams is one of the more common restoration techniques employed around the world (Sear et al., 1998; Kondolf, 2006; Lorenz et al., 2009). While a variety of physical channel characteristics have been reported to increase following restoration by re-meandering, i.e increases in width and substrate diversity (Lorenz et al., 2009), slope on the other hand has been reported to decrease (Eekhout, 2014). Indeed, Eekhout (2014) noted consistent interannual decreases in channel slope for years after restoration by traditional meandering. Thus, despite the jump-start in meandering due to active manipulation of channel and profile, subsequent migration rates may be reduced. This could affect future cut-offs, oxbow formation and long-term rates of wetland formation. However, realistically many restoration project involving re-meandering also include efforts to lower the floodplain elevation and improve surface hydraulic connections with the channel, which if successful may improve wetland creation.

Scenario 2: There are no clear differences in time to initiation of re-meandering, time to maximum meander amplitude or channel length at the end of the simulation when discharge is increased compared to the actual mean annual discharge. Even when doubling the original discharge input, simulations revealed only differences in meander shape and overall position within the floodplain suggesting that understanding the effect of climate change on channel migration is not a simple task. Similarly, Choné (2012) simulated the effect of climate change-driven discharge increases (~10%) for two rivers with greatly different actual discharge values using RVR Meander, and concluded that while greater input discharges do not appear to affect the extent of the mobility corridor they could increase risks related to processes of erosion. In particular, Choné (2012) noted increased meander wavelength and a spatial redistribution of zones of erosion and deposition within the channel rather than any net increases in response to a higher discharge input. These results show how discharge may affect the shape and location of created meanders but not necessarily the rate of migration. However, depending on the location of erosion within the channel coupled with the greater potential erosional risks that can be expected with increased climate change driven discharge, more cut-offs may occur during the meandering process. Thus, while overall migration rates may not clearly be affected, the process of wetland formation through meander cut-off may potentially be increased where climate change leads to increases in runoff.

Scenario 3: Increasing width by 25% results in clear reductions in the time to initiation of re-meandering for DF (3 years less) and PPauB (5 years less) compared to actual width values. No difference was found in time to initiation of re-meandering for RM. However, increasing width decreased the time to maximum meander amplitude for all three sites, up to 30 years less for RM. Similar findings were reported by Roux (2012) who described a 14% increase in migration rate after a 30-yr simulation where input width was increased. Although understanding the theoretical links between width and channel migration/re-meandering may be less straightforward than other parameters, sensitivity analyses conducted by Motta et al. (2012) reveal that bed shear stress along the outer banks actually increases with channel width. Given that aggradation and channel widening are natural processes as part of the evolution of constructed channels (Simon and Hupp, 1986), not only do all three straightened sites show the potential to re-meander passively but recovery times could be reduced as the channel morphology adjusts and widens. The passive restoration of meandering patterns and potential wetland creation will, however, require that space be given to the stream (laterally) and that interventions such as dredging or bank stabilization cease.

Scenario 4: There is no obvious way to test reconnection of ancient meanders with channel in RVR Meander, but it remains useful to discuss the potential effects this could have on channel migration and time to maximum meander amplitude. In agricultural environments such as those of our sites (particularly DF and PPauB) the re-connection of ancient meanders entails the removal of the layer of agricultural fill “remblais” which is often blocking the hydraulic connections between the channel and the coarser sediments in the ancient meander. This would essentially lower the elevation of the floodplain and remove clay plugs, potentially increasing lateral and vertical hydraulic connections. In terms of effects on RVR meander parameters, these changes could give rise to increases in mean annual discharge due to the increased hydraulic connections and the provision of stored water during times of low flow in the summer. This would be particularly relevant for PPauB which is completely dry for much of the summer. Based on our results, slight increases in mean annual discharge related to bank storage would likely not have much impact of channel migration rates or the potential to re-meander although the increased hydraulic connectivity could benefit the establishment of wetland plant communities (Phillips, 2013).

A major consequence of the modification of the planform geometry of small agricultural streams is the rupture between the flow and its bedload sediment sources. In the Lowland environment, a large proportion of the sediments transported in the streams is carried in suspension and plays little part in the morphological dynamics of the bed and the migration of the channel (Brooks, 2003). The coarsest fraction, less important in terms of volume, is normally confined to the minor channel bed and is transported as bedload. This fraction of sediment is no less important, however, to the migration dynamics of the channel, whereas it actively participates in the spatial distribution of erosion zones via the formation of bars and the migration of the thalweg. By eliminating complete meander portions (through straightening) and deepening channels (in normally finer regional deposits), streams have become disconnected from significant sources of sediments that could have contributed to morphological changes. Reconnecting the old meanders and removing embankments could allow the mobilization of the coarsest sediments sequestered by human activities. It is possible to imagine that large floods succeed in mobilizing the coarser sediments of the old meanders and that they thus transit in the main channel. However, the mobilization potential of these coarse sediments may remain low if mechanical intervention is not carried out (e.g. modification of the main channel planform geometry, raising the bed, etc.). An alternative would be to mechanically add coarse sediments to the channel (in the form of side benches for example) to allow a bedload transport dynamics to be restarted (Bramard, 2015).

It is also important to consider that altering the migration coefficient has quite a large impact on re-meandering potential and migration/recovery rates. For example doubling the migration coefficient for RM from $4*10^{-8}$ to $8*10^{-8}$ effectively halves the time to re-meandering and from 25 years to 12.5 years as well as the time to reach maximum meander amplitude 80 years rather than 160 years. Thus, obtaining precise migration rates and reliably calibrating the migration coefficient E_0 is essential for accurately simulating the potential of these sites to re-meander. Given the limitation of the CMA approach and the difficulties in calibrating E_0 for areas experiencing high levels of anthropogenic influences, we stress again that these modelling results must be interpreted with caution.

5.8 Conclusion

Although there are many limitations to RVR Meander and the method used to simulate channel migration (CMA), notably the difficulty in calibrating the actual E_0 for each site due to high degrees of anthropogenic disturbances and the inability to predict exact meander positions, the results of the RVR Meander simulations are still relevant in theoretical discussions about the potential of these sites to re-meander and how they may respond to different restoration scenarios. All three sites demonstrated the ability to passively re-meander and potentially recover to historical patterns within reasonable timescales (decades). Furthermore, restoration practices involving either passive methods (giving space for example, buffer zone) or the active reconnection of ancient meanders could only benefit these systems and improve rates at which new wetlands are created and maintained.

6 EXAMINATION OF POTENTIAL HYDRAULIC RESPONSE TO WETLAND RESTORATION AND CLIMATE CHANGE IMPLICATIONS

The theoretical function of wetlands in terms of flood attenuation and moderating the overall hydraulic response as well as its potential benefits for both fluvial ecosystems and society are well known. For instance, wetlands can help decrease the impact of yearly peak floods, dissipating energy by spreading out and retaining excess water into the floodplain (DeLaney, 1995; Zhang and Song, 2014). Furthermore, wetlands established by vegetation tend to have a flat topography which may help reduce bank erosion (DeLaney, 1995). Indeed, the past several decades have presented many wetland restoration projects that have attempted to restore wetlands to reintroduce their influence on the systems hydraulic response (reducing flood levels for example) rather than continue to rely on structural solutions such as levees which often fail in the long-term (Hey and Philippi, 1995; Kristensen et al., 2014; Zhang and Song, 2014). Many of these restoration efforts involve identification of target areas to maximize the mitigation of flood risk using hydraulic models (Fisher and Stratford, 2008; Te Linde et al., 2010; Reckendeker et al., 2013), often focusing on the watershed scale. However, it remains unclear how successful wetland restoration projects are for long-term flood management, predominantly due to a lack of monitoring or consideration of process-based restoration (Beechie et al., 2008). For instance, while some have reported positive results in terms of reducing flood levels (Te Linde et al., 2010; Kristensen et al., 2014) following wetland and riparian restoration, others have noted that these benefits could be quickly lost if the influx of fine sediments is not properly managed (Reckendeker et al., 2013) particularly in watersheds dominated by agriculture (Gleason, 1996; Fisher and Stratford, 2008; Palmer et al., 2014). Indeed, it has been reported that riparian wetlands in agriculturally dominated watersheds without buffer zones can accumulate sediments twice as fast as those surrounded by vegetation (Adomaitis et al., 1967; Martin and Hartman, 1987). Ultimately the potential benefit of wetland restoration on mitigating extreme hydraulic responses will depend on the scale of the system and the size of the restoration project. For instance, restoring a small wetland patch will likely be short lived if surrounded by intensive agriculture where rapid runoff and high sediment loads can quickly add up (Gleason, 1996).

However, small-scale studies such as ours which focus on detailing the role ancient meanders can play in capturing and storing runoff and maintaining lateral hydrological connections between the floodplain and the channel, provide important information for understanding the hydraulic response of wetland restoration at larger scales. At RM for instance, much of the floodplain appears to be hydrologically connected to the channel through the multiple ancient meanders present. More importantly, there is a diversity of potential lateral hydrological connections through both surface and subsurface processes. For instance, the topography of the ancient meander allows precipitation to be captured at the surface while the presence of alluvial sediments, which are relatively free of fine particles, and a base layer of impermeable till create a physical environment which allows this water to be stored. Furthermore, the results of the vegetation assessment which revealed roughly 80% of the floodplain assessed is currently wetland habitat indicates that the current hydrological processes at RM are in a state where passive wetland restoration is possible. While the largest flood recorded during our two-year study did not result in overbank flow (Figure 79A) it is clear that the land use change (return to forest/wetland from previous agriculture) and lateral hydrological connections with the floodplain (100m at its widest) provides a significant amount of space for flood attenuation. Although there is still the presence of agriculture upstream from this site the large buffer of riparian trees and shrubs help reduce the impact of these anthropogenic pressures contributing to the potential long-term benefits of these passively restored wetlands on the hydraulic response to major precipitation events.

In contrast, the potential effect of wetland restoration on the hydraulic response is less clear at PPauB and DF due to the persistent dominance of agriculture on the watershed, lack of riparian buffer and presence of drainage ditches, which contribute to the lowering of the regional water table. For instance, while removing the overlying agricultural fill and reconnecting the ancient meander may help to mitigate flood impacts and bank erosion if vegetation is allowed to grow, these benefits may be offset due to high levels of sediment influx from adjacent land. Thus, any positive effect of restoration of riparian wetland ecosystem functions, to any benefit of society and local landowners could be only temporary if side measures (ex: riparian buffer, space) are not also implemented. There are also several confounding factors such as culverts and drainage ditches that are present in agriculturally dominated landscapes and can affect the hydraulic response of a stream. These structures ultimately limit infiltration and accelerate the movement of water through the system. At PPauB for instance culverts are clearly too narrow to accommodate the current hydraulic response of the system, often leading to inconvenient flooding of adjacent fields (Figure 79B). Even if wetlands were restored here, the presence of numerous ill-designed culverts will make flood management difficult. Furthermore, assuming the restoration of a large and well established wetland at PPauB, it is difficult to imagine that it would be able to persist in the long term due to a combination of extremely low flows during the summer and the risk of rapid sediment accumulation at this site.

Figure 79: A) Photo from camera installed at RM facing the ancient meander showing peak fall discharge reaching just below bankfull; B) Photo of narrow culvert at PPauB forcing peak flows to flood surrounding land; C) Example of surface water flooding on agricultural land at PPauB during the spring floods of 2018.

The role that wetlands can play in mitigating the hydraulic response of extreme precipitation events in degraded agricultural watersheds is further complicated by a changing climate. A global climate modeling report (IPCC, 2013) projects that global warming will induce changes in the atmospheric and hydrological regime at the regional scale. For Canada, it projects an increase in precipitation, in evaporation and in runoff. Climate change is thus expected to strongly impact streams and their riparian ecosystem (Capon et al., 2013; Taylor et al., 2013), but the direction, the magnitude and the location of changes remain largely uncertain. Model projections show a large range of possible futures due to the assumptions about the evolution of greenhouse gas concentrations in emission scenarios, the imperfection of climate models and the natural variability of the climate system (IPCC, 2013). Therefore, a streams response to climate change is expected to greatly vary in time and in space due to the influence of autogenic factors, such as geology, floodplain morphology as well as the distribution and type of riparian vegetation, which ultimately effects wetland hydrological regimes. Indeed, a changing climate will make the development of wetland restoration projects even more complicated and will likely decrease certainty of success in some cases.

Using PPauB as an example, results of 12 climate models (averaged) project a 14% increase in total annual precipitation for this watershed over the next 100 years (Figure 80A), (models project similar increases for DF Figure 80B and RM, Figure 80C). Such an increase in precipitation may in theory, actually benefit wetlands in agricultural watersheds such as PPauB which have very low summer flows due in part to artificially lowered water table and the disappearance of surface water storage, resulting from the drainage of agricultural fields, by increasing the frequency and duration of saturation. This of course depends on a restoration scenario where we assume that an ancient meander is actively restored (ex: flat topography, removal of fine sediments, and provision of riparian buffer), and that the nature of the precipitation events results in an increase of surface saturation (ex: mild and frequent as opposed to very large storm events). On the other hand, climate change is also expected to have an impact on air temperature as all scenarios show a trend of warming. Indeed, scenarios project shorter winters, less snow, more winter floods and greater evapotranspiration from May to October, conditions which could potentially counterbalance precipitation increases. However, even if the net effect is no change in precipitation, events are expected to be more intense, potentially leading to large discharges over short periods of time. Furthermore, the level of human interventions will also continue to change in response to climate change. Several studies suggest that increased disturbances in streams and watersheds as a result of climate change (ex: flooding, sediment regime change, and drought) will induce an increase in human interventions in these systems. For example, flood-prone areas (Figure 79C) will be subject to more mitigation measures (Sauri-Pujol et al., 2001), dredging interventions in agricultural streams will increase and more pumping and irrigation structures will be established in areas where groundwater recharge is declining (Taylor et al., 2013). These scenarios show the important role that socio-economic factors play in wetland restoration. In the province of Quebec, laws exist which set a maximum drainage density, however in practice these values are often doubled due to a lack of monitoring and accountability. It is therefore not unreasonable to expect that drainage density across agricultural fields would increase in response to climate change driven increases in precipitation. Furthermore, the increased runoff will bring with it, high sediment loads from the fields, which could quickly fill up any restored wetlands, reducing their lifespan and any influence they had on mitigating the hydraulic response to extreme flood events (Gleason, 1996).

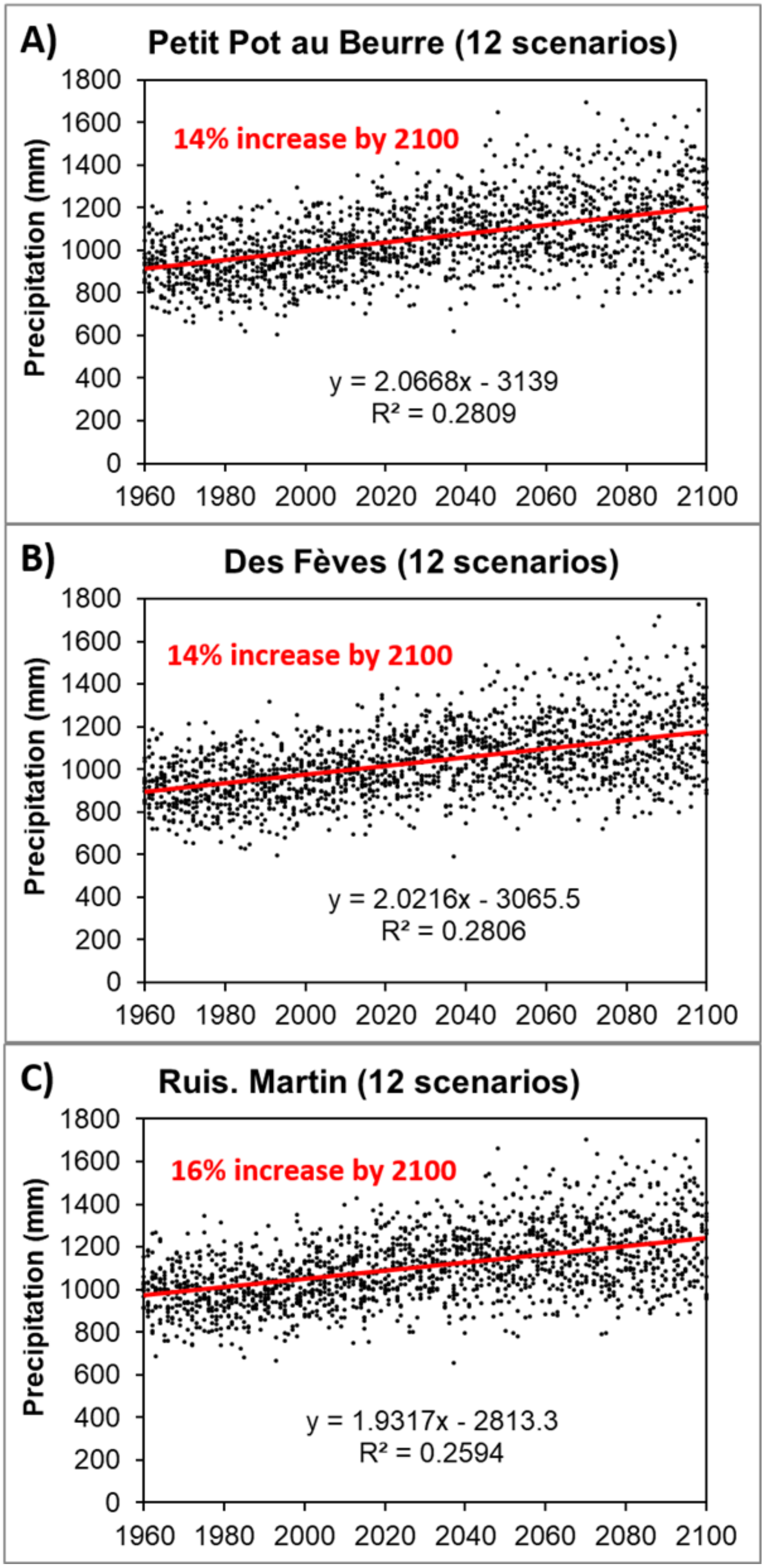


Figure 80: Results of 12 climate change models averaged to show projected increase in precipitation from 1960 to 2100. A) PPaub; B) DF; C) RM

7 DETERMINING CRITERIA FOR EVALUATING POTENTIAL RESTORATION OF WETLANDS

In this section we propose criteria for evaluating the restoration potential of riparian wetlands in the straightened agricultural rivers of Quebec. Our approach is to combine criteria established in the literature with what we evaluated experimentally, drawing on qualitative and quantitative data collected at our sites in order to provide a list (Table 13) of the most important criteria to consider.

We define restoration potential as the likelihood that restoration efforts at a site will return functional ecosystem services that are both long-term and in equilibrium with the surrounding environment as it is today rather than trying to recreate past conditions which are difficult to define. Thus, although many restoration projects target specific services (ex: habitat, water quality) or focus on partial restoration when anthropogenic activities limit full recovery (ex: Marshy Hype Creek, Maryland (Interagency Workgroup on Wetland Restoration (IWWR) 2019), efforts which target the restoration of natural physical, biological and/or chemical processes will ultimately be the most successful in the long term (Beechie et al., 2010 Wohl et al., 2015). When attempting to successfully restore processes, passive approaches (ex: limiting source of degradation, providing space – buffer zone) are preferred over active restoration approaches (physical intervention, adjusting topography, controlling water flow, planting etc.) as they are not only less expensive but increase the potential that the wetland will be sustained long-term as they do not require speculation over historical reference conditions.

Furthermore, while specific wetland types, (swamp, wooded swampland etc.) are typically defined by the species present (vegetation) as well as the local soil characteristics (discussed in Appendix 1), these ecological elements are ultimately dependent on hydrology and hydrogeology (Reid et al., 2015). We therefore use the definition of riparian wetlands in agricultural lands from the 1985 Food security Act (United States): land that is “inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and under normal circumstances does support, a prevalence of hydrophytic vegetation typically adapted for life in saturated soil conditions”

The criteria we identified for evaluating the restoration potential of wetlands in degraded agricultural watersheds are summarized as eight main factors to consider (Table 13): availability of historical data, hydrology and hydrogeology, topography, surface geology, vegetation, channel migration, scale of project, and socio-economic context. The following section will briefly expand on and discuss the rationale behind the chosen criteria.

Historical wetlands:

One of the initial factors that should be considered when evaluating the potential for wetland restoration is the possible extent of wetlands at a site before the stream/river was channelized. Ancient meanders for instance are often clearly visible as depressions in the LiDAR indicating zones adjacent to the stream which were likely frequently flooded. In regions where ancient meanders were artificially filled however, these depressions are less obvious. In these cases, although National databases of historical aerial photos are often spatially incomplete and limited to the 20th and 21st century (Fuchs et al., 2012; NRCAN, 2016; USGS, 2015), historical air photos can be quite useful for identifying past channel positions and meander oxbows.

Hydrology/hydrogeology:

While the existence of past wetlands does not guarantee that restoration efforts will be met with success if processes involved with their creation and maintenance are not restored, sites that demonstrate lateral hydrological connectivity, either by surface or subsurface processes are well known to offer a greater potential for long-term restoration of many wetland functions (Vidon et al., 2010; Reid et al., 2015). The importance of lateral hydrological connections in wetland restoration potential, particularly in agricultural watersheds, can be further illustrated when considering that one of the more common active restoration practices involves some type of water control structure which often succeeds in increasing water retention but does not necessarily promote exchanges between surface water and groundwater (Steven and Gramling, 2011). In this context, although the deeper water may be beneficial to wildlife it could also have a dampening effect on nutrient filtering (IWWR, 2019). Furthermore, Steven and Gramling (2011) reported that 11-15% of projects employing water retention measures in the Piedmont-Atlantic coastal region of Southern USA were drier than expected, which could have implications for long term wetland persistence. At RM the presence of groundwater storage in the ancient meander and groundwater flowing from the floodplain to the channel help explain why this site was able to passively restore much of its wetland habitat. At PPauB the low water table in the floodplain and the lack of evidence for lateral hydrological connections suggest limited storage within the floodplain and thus a lower potential for wetland restoration. It is therefore more likely that it is possible to restore wetland functions at sites such as RM where lateral hydrological connections are maintained as opposed to a scenario of active restoration through water retention at PPauB where artificial embankment may have permanently modified the potential to restore lateral hydrological connections. Unfortunately, most restoration projects display a limited integration of exchanged fluxes between surface water and groundwater (Boulton, 2007; Lehr et al., 2015; Morén et al., 2017). If no prior assessment was done about surface water-groundwater interactions, restoration projects might not reach their goals and the project may not be sustainable. For instance, the reforestation of a riparian area along an incised channel to create riparian habitats (including wetlands) and to increase nitrogen uptake processes could fail if water table conditions are not accounted for in the choice of vegetation species. For example, low water table conditions in the floodplain, which can be a consequence of man-made channel incision (Schilling et al., 2004) could result in most subsurface nutrients flowing below the range of the root system if species with very deep roots are not used (Dosskey et al., 2010). Restoration projects must therefore include consideration of the floodplain storage capacity and groundwater flows, and not strictly with the goal of restoring initial conditions.

Topography and surface geology:

Topography and surface geology are two other important factors to consider when evaluating wetland restoration potential. For instance, the presence of depressions in the landscape adjacent to the channel and an impermeable layer in the stratigraphic profile may facilitate the pooling of surface water and a potentially viable saturated zone for wetland habitat. Capturing surface runoff can increase the duration of flooding and initiate germination of some wetland vegetation species (Reid et al., 2015). Furthermore, the presence of alluvial deposits can encourage fluxes between surface and groundwater (if they are not clogged by fine particles), potentially improving nutrient exchange (Kasahara and Hill, 2008). There is also evidence to suggest that the depth of the wetland can play a role in maintaining biodiversity as shallower wetlands tend towards monodominance. For instance, Linz et al., (1996) reported that wetlands with shallow depths (<60cm) were dominated by cattails which are also known to serve as roosting sites for blackbirds that can be a nuisance for cereal crops. Drawing on a comparison between study sites, at RM wetland habitats were able to form in the topographic depressions of ancient meanders which captured surface runoff, while the stratigraphic profile of organic material at the surface followed by layers of fine to medium sand above an impermeable layer of till or clay helped maintain the saturation. At PPauB on the other hand agricultural fill has mostly flattened the past depression of the ancient meander with a mild slope towards the channel. This hard packed material buries the ancient alluvial sediments below approximately 2m of silt and clay, likely further reducing infiltration and underground storage.

Vegetation:

The presence of wetland vegetation nearby is another criteria to consider which contributes to a sites potential for wetland restoration. In highly degraded agricultural watersheds it is common that much of the native vegetation will have been disturbed. However, as we saw in the results of our vegetation assessment for PPauB and DF, wetland plant species often persist in small, often isolated patches between farmland plots (wetland plant species were abundant throughout the RM reach). These patches provide a reference site which can provide information about species compositions of native wetland habitats and identify the potential of invasive species which could hinder restoration (when considering species diversity as a goal). They also serve as a source for plants which can be used to colonize the site to be restored although there is still a lot of uncertainty regarding how long recolonization will take. Monitoring of potential colonization by exotic species will also be important for restoration success and time to success. This is particularly important in agricultural watersheds where drastic changes in wetland hydrology can favor the dominance of non-native species over native ones (Catford et al., 2011).

Channel migration:

Channel migration is a key contributor to the potential for passive wetland restoration as ongoing and natural processes of erosion and deposition may eventually produce meander cut-offs. These frequently flooded areas adjacent to the main channel can be established by wetland vegetation species and therefore have a great potential to become future wetland habitats (Philips, 2013). Indeed, oxbows and associated wetlands are classified as part of the minimum space a watercourse needs for natural fluvial processes to occur as described in the Freedom space concept (Biron et al., 2014). Criteria used to evaluate lateral channel migration include evidence of natural fluvial bank erosion (i.e. the stream has energy to migrate and the banks are erodible) and the presence of large woody debris. The link between woody debris input and channel migration is well established in the literature (Ireneusz, 2005) with the added benefit of introducing physical habitat complexity for a range of aquatic biota (Massey et al., 2017).

Scale of project:

Worldwide, there is an important gap between the scale of stream alteration and the scale of restoration (Wohl, 2017). Most restoration projects have focused on the reach scale instead of coordinating measures at the watershed scale (Beechie et al., 2010). Individual reach-scale restoration projects can hardly restore “natural” conditions and processes in a stream because of the great level of interdependence that exists between the different components of a watershed (Lorenz et al., 2009). This doesn’t mean that one reach-scale project cannot attain specific restoration goals. For instance, reconnecting a channelized stream to an abandoned meander can help improve in-stream and riparian habitats (Haines, 2017), as well as partly store floods and improve downstream water quality. However, the loss of most ecological functions in streams including wetlands results from the accumulation of point to non-point human disturbances through out the watershed that cannot be addressed without at least a certain number of coordinated reach scale projects. Thus, focusing many restoration efforts in a target watershed rather than spatially distant and isolated projects may offer greater potential for wetland restoration in degraded agricultural watersheds.

Socio-economic context:

No less important to consider when evaluating the potential for wetland restoration is the consideration of socio-economic factors such as ownership, cost, and monitoring management which can slow down or inhibit initiatives (Wohl et al., 2015). For instance, it is important to consider if the project benefits from cooperation of the landowner and if there is overall interest in wetland restoration from the community. These conditions can have an obvious influence on mitigating or eliminating the original source of degradation. As an example, at RM anthropogenic activities were greatly reduced along the floodplain following the initial straightening and widening of the channel, contributing to the passive restoration observed at this site. Sites such as DF and PPauB on the other hand are continuously maintained (most recently dredged in 2013) and the dense network of anthropogenic drainage which influences local hydrogeology will compromise restoration efforts. The level of awareness of landowners can be an important step not only for implementing the project but also increasing the potential for long-term process-based restoration. For example, one of the many ecosystem functions of wetlands is the filtering of fine sediments which has positive implications for water quality and can be of great value for reducing the impact of high sediment loads in agricultural watersheds (Kuenzler, 1990). However, the influx of fine sediments is also a major pollutant of wetlands (Baker, 1992) which can greatly reduce their long-term persistence in the landscape (Gleason, 1996). Sediment loads in agricultural watersheds can be quite high causing wetlands to fill faster than those adjacent to natural grasslands (Gleason, 1996), Furthermore, several studies have shown that wetlands that were not contoured by vegetation filled at twice the rate as those surrounded by vegetation (Adomaitis et al., 1967; Martin and Hartman, 1987). Thus, we can consider the ability to create space for a buffer zone as an important criteria for the long-term restoration of ecosystem function, one which demonstrates the need to work with the community to determine the best management practices that will benefit both agricultural and wetland interests.

Ancient meanders as the unit of restoration:

In summary, gauging the restoration potential of wetlands in landscapes degraded by agricultural processes can at first seem a highly complex task due to the wide range of factors and complications that must be considered. Focusing restoration efforts (either passive or active) on ancient meanders as priority targets however offers a clear potential for long-term success and return of wetland ecosystem functions. Ancient meanders are easy to identify with LiDAR data and historical photos, and possess several properties (natural depression, alluvial sediments) that can promote the basic hydrological connections upon which a thriving ecosystem can be restored. Furthermore, ancient meanders often occupy a relatively small area and as such could be more easily considered by landowners who may be reluctant to concede land. These areas also demonstrate a good potential for passive restoration measures, particularly if natural processes of channel migration can freely occur as they could over time produce more wetlands through meander cut-offs. Ideally, as ancient meanders are restored throughout the watershed this would create a positive influence for the large scale, long-term restoration of wetland habitats.

Table 13: Summary of criteria to consider for evaluating the restoration potential of wetlands in degraded agricultural watersheds. Note: criteria are not listed in any order of importance.

Factor to consider	Rationale	Criteria for higher restoration potential	Possible data source
Historical wetlands	Provides evidence that wetlands previously existed while allowing to identify potential restoration sites	• Presence of ancient meanders • Extent of past flooding events • Extent of vegetation before agricultural disturbance	• LiDAR • Orthophotos • Historical air photos

Hydrology/ Hydrogeology	Sufficient water and frequency of saturation	• Presence of lateral hydrological connections • Frequent flooding/saturation • Water table close to ground surface in the floodplain	• Field instrumentation using pressure transducers • Gauging station • Wells/piezometers
Topography	Landscape controls on runoff	• Presence of sub-surface drainage • Presence of depressions that can store water	• LiDAR • Field measurements (Total station / dGPS) • Camera installed on site
Surface geology	Landscape controls on groundwater flux	• Presence of impermeable layer (ex: till/clay) • Presence of unburdened (not clogged with silt/clay) alluvial deposits	• Maps of surface geology • On site exploration via boreholes (auger)
Vegetation	Potential for recolonization	• Native wetland species nearby • Monitoring of invasive species cover	On site assessment by expert in identifying wetland species
Channel migration	Passive restoration through creation of new meander oxbows when migration rates are high enough	• Evidence of natural fluvial bank erosion • Presence of large woody debris in channel • Presence of bedload transport	• On site investigation of stream reach • Orthophotos • Historical air photos
Scale of project	Match scale of restoration with scale of project	• Ability to conduct multiple restoration projects throughout the watershed rather than sparse isolated restorations	• Assessment to identify extent of degradation • Identify degradation sources
Socio-economic context	Willingness of riparian landowners to participate will improve chance of success	• Landowner participation • Community support • Reduce impact of adjacent land-use • Space for buffer zone	• Communication with local landowners • Orthophotos

8 CONCLUSIONS

This study has highlighted that it is worth working on the restoration potential of straightened agricultural streams as they can provide important ecosystem services if natural hydrogeological processes are able to operate. This was particularly obvious at the RM site, which has not been maintained after being straightened in the late 1950s, and which is now providing excellent habitat to diverse aquatic species, mainly through woody debris which are ubiquitous in the studied reach. Although this work provides original and useful new knowledge about the restoration potential of wetlands in small headwater streams located in agricultural landscapes, many questions still need to be addressed.

The three sites (DF, PPauB and RM) were originally chosen to be as similar as possible, but another key finding of this study is that the local geological context results in significant variability in the hydrogeological responses. It is therefore key for successful restoration to avoid a “one size fits all” approach, and to choose the most appropriate approach based on local variables. More studies in different geological and land use conditions will be necessary to provide specific guidelines for restoration projects.

Repeated human interventions which have resulted in the loss of local depressions associated with old meanders, such as at PPauB, are clearly a serious constraint when considering passive restoration options. Nevertheless, even in these conditions, coarse fluvial deposits still remain below compacted fill material, and more work is needed to assess the potential for subsurface process restoration in such cases, perhaps with active restoration approaches.

For a variety of reasons, passive or active restoration of wetlands will probably not be feasible everywhere on a given stream or river. It thus remains a necessity to further understand how many wetlands need to be restored for them to contribute significantly to increase ecosystem services (plant and wildlife habitats), improve hydrological functions (reducing downstream floods) and improve stream and river water quality. Field-scale and watershed-scale studies, combined with model applications will be necessary to provide answers to these questions.

Lastly, the problem of drain outlets was not examined in this study, but clearly more work is needed to understand the impact of tile drainage on the water table and on the potential for restoration of subsurface processes. The impacts of agricultural drainage on water resources, and the possible solutions to attenuate these impacts are wide-reaching issues that will require further research investments.

In a context where the role of riparian wetlands is increasingly recognized as critical in integrated watershed management, this study provides a unique database to further explore the restoration potential of the ancient meanders of straightened streams. Favouring approaches such as easement or purchase of land that would facilitate passive restoration should be seen as a priority in degraded agricultural watersheds while being the most challenging issue. Targeting sites where active reconnection of former meanders could be tested as a pilot project should also be considered in future pilot projects.

REFERENCES

Abad, J. D., and Garcia, M. H. (2006). RVR Meander: A toolbox for re-meandering of channelized streams. *Computer & Geosciences*, 32(1), 92-101.

Addy, S., Wilkinson, M. E. (2019). Geomorphic and retention responses following the restoration of a sand-gravel bed stream. *Ecological Engineering*, 130, 131-146.

Adomaitis, V. A, Kantrud, H. A., Shoesmith, J. A. (1967). Some chemical characteristics of aeolian deposits of snow-soil on prairie wetlands. Proceedings of the North Dakota Academy of Science 21, 65-69.

Amoros, C., and Roux, A. (1988). Interaction between water bodies within the floodplains of large rivers: Function and development of connectivity. *Münstersche Geographische Arbeiten*, 29(1), 125-130.

Baker, L. A. (1992). Introduction to nonpoint source pollution in the United States and prospects for wetland use. *Ecological Engineering,* 1, 1-26.

Beechie, T. J., Sear, D. A., Olden, J. D. et al. (2010). Process-based Principles for Restoring River Ecosystems. *BioScience*, 60(3), 209–222.

Biron, P. M., Buffin-Bélanger, T., Larocque, M., et al. (2014). Freedom space for rivers: a sustainable management approach to enhance river resilience. *Environmental Management*, 54(5), 1056-1073.

Biron, P. M., Buffin ‐ Bélanger, T., & Massé, S. (2017). The need for river management and stream restoration practices to integrate hydrogeomorphology. *The Canadian Geographer/Le Géographe Canadien*, 62(2), 288-195.

Boulton, A. J. (2007). Hyporheic rehabilitation in rivers: Restoring vertical connectivity. *Freshwater Biology*, 52(4), 632-650.

Brookes A., (1987). River channel adjustments down-stream from channelisation works in England and Wales. *Earth Surface Processes and Landforms*, 12, 337–351.

Brown, A. G., Lespez, L., Sear, D. A., Macaire, J., Houben, P., Klimek, K., . . . Pears, B. (2018). Natural vs anthropogenic streams in europe: History, ecology and implications for restoration, river-rewilding and riverine ecosystem services. *Earth-Science Reviews*, 180, 185-205.

Brunland, G., Hanchey, M., Richardson, C. (2003). Effects of agriculture and wetland restoration on hydrology, soils, and water quality of a Carolina Bay complex. *Wetlands Ecology and Management*, 11, 141-156.

Cabezas, A., Gonzalez, E., Gallardo, B., Garcia, M., Gonzalez, M., & Comin, F.A. (2008). Effects of hydrological connectivity on the substrate and understory structure of riparian wetlands in the middle ebro river (NE spain): Implications for restoration and management. *Aquatic Sciences*, 70(4), 361-376.

Catford, J. A., Downes, B. J., Gippel, C. J. and Vesk, P. A. (2011). Flow regulation reduces native plant cover and facilitates exotic invasion in riparian wetlands. *Journal of Applied Ecology*, 48, 432-442.

Carrier, M.-A., Lefebvre, R., Rivard, C., Parent, M., Ballard, J.-M., Benoit, N., Vigneault, H., Beaudry, C., Malet, X., Laurencelle, M., Gosselin, J.-S., Ladevèze, P., Thériault, R., Beaudin, I., Michaud, A., Pugin, A., Morin, R., Crow, H., Gloaguen, E., Bleser, J., Martin, A., Lavoie, D. (2013) Portrait des ressources en eau souterraine en Montérégie Est, Québec, Canada. Projet réalisé conjointement par l'INRS, la CGC, l'OBV Yamaska et l'IRDA dans le cadre du Programme d'acquisition de connaissances sur les eaux souterraines, rapport final INRS R1433, soumis en juin 2013.

Choné, G. (2013). L'espace de mobilité des rivières du sud du Québec : implications pour les écosystèmes et la gestion des rivières dans une perspective de changements climatiques. MSc thesis, Concordia University, Montreal, Qc., 138 pages.

Chow, V. T. (1959). Open-channel hydraulics: New York, McGraw-Hill, 680 pages.

DeLaney, T. A. (1995). Benefits to downstream flood attenuation and water quality as a result of constructed wetlands in agricultural landscapes. *Journal of Soil and Water Conservation*, 50(6), 620-626

Dépret, T., Riquier, J., and Piégay, H. (2017). Evolution of abandoned channels: Insights on controlling factors in a multi-pressure river system. *Geomorphology*, 294, 99-118.

Dosskey, M. G., Vidon, P., Gurwick, N. P., Allan, C. J., Duval, T. P., & Lowrance, R. (2010). The role of riparian vegetation in protecting and improving chemical water quality in Streams1. *JAWRA Journal of the American Water Resources Association*, 46(2), 261-277.

Eekhout, J. P. C. (2014). Morphological Processes in Lowland Streams - Implications for Stream Restoration, PhD thesis, Wageningen University, Wageningen, NL, 178 pages.

Eekhout, J. P., Hoitink, A. J., Brouwer, J. H., Verdonschot, P. F. (2015). Morphological assessment of reconstructed lowland streams in the Netherlands. *Advances in Water Resources*, 81, 161-171.

Fisher, J., & Stratford, C. (2008). Does reconnection mean restoration for an oxbow lake, hungary? *International Journal of River Basin Management*, 6(3), 201-211.

Fischer, J. R., Bakevich, B. D., Shea, C. P., Pierce, C. L., and Quist, M. C. (2018). Floods, drying, habitat connectivity, and fish occupancy dynamics in restored and unrestored oxbows of west central iowa, *USA. Aquatic Conservation: Marine and Freshwater Ecosystems*, 28(3), 630-640.

Flávio, H. M., Ferreira, P., Formigo, N., and Svendsen, J. C. (2017). Reconciling agriculture and stream restoration in Europe: A review relating to the EU Water Framework Directive. *Science of the Total Environment*, 596, 378-395.

Fondriest Environmental, Inc. “Streamflow Measurements.” Fundamentals of Environmental Measurements. 17 Sept. 2015. Web. < https://www.fondriest.com/environmental-measurements/measurements/hydrological-measurements/streamflow-measurements/ >.

Fourcier, L., Mingelbier, M., Brodeur, P., Morin, J. (2007). Le marais aménagé de la Baie de Lavallière : état de situation, enjeux agroenvironnementaux et recommendations de gestion. Québec : MRNF, Direction de la recherche sur la Faune, 69 pages.

Fuchs, R., Herold, M., Verburg, P. H., & Clevers, J. (2012). A high-resolution and harmonized model approach for reconstructing and analyzing historic land changes in europe. *Biogeosciences*, 10(3), 1543-1559.

Gaillot, N. (2007). La dynamique géomorphologique du méandrage de la basse vallée de la Bruche (Bas-Rhin) : approche historique. 59 pages.

Gallardo, B., Cabezas, Á, Gonzalez, E., & Comín, F. A. (2012). Effectiveness of a newly created oxbow lake to mitigate habitat loss and increase biodiversity in a regulated floodplain. *Restoration Ecology*, 20(3), 387-394.

Gleason, R. A. and Euliss, N. H. (1996). Sedimentation of prairie wetlands. *Great Plains Research*, 8, 97-112.

Haines, B. J. (2017). Characterization of hydrology and water quality at a restored oxbow: Ecosystem services achieved in year one. MS (Master of Science) thesis, University of Iowa, 109 pages.

Hansen, A., Dolph, C., Foufoula-Georgiou, E., Finlay, J. (2018). Contribution of wetlands to nitrate removal at the watershed scale. *Nature Geoscience*, 11, 127-132.

Hammer, Ø., Harper, D.A.T., Ryan, P.D. (2001). PAST: Paleontological statistics software package for education and data analysis. Palaeontologia Electronica 4(1), 9 pages. http://palaeo-electronica.org/2001_1/past/issue1_01.htm

Harrison, M. D., Groffman, P. M., Mayer, P. M., & Kaushal, S. S. (2012). Nitrate removal in two relict oxbow urban wetlands: A 15 N mass-balance approach. *Biogeochemistry*, 111(1-3), 647-660.

Harrison, M. D., Miller, A. J., Groffman, P. M., Mayer, P. M., & Kaushal, S. S. (2014). Hydrologic controls on nitrogen and phosphorous dynamics in relict oxbow wetlands adjacent to an urban restored stream. *JAWRA Journal of the American Water Resources Association*, 50(6), 1365-1382.

Hauer, C., Mandlburger, G., Schober, B., & Habersack, H. (2014). Morphologically related integrative management concept for reconnecting abandoned channels based on airborne LiDAR data and habitat modelling. *River Research and Applications*, 30(5), 537-556.

Hein, T., Baranyi, C., Reckendorfer, W., & Schiemer, F. (2004). The impact of surface water exchange on the nutrient and particle dynamics in side-arms along the river danube, austria. *Science of the Total Environment*, 328(1-3), 207-218.

Hester, E. T., Hammond, B., & Scott, D. T. (2016). Effects of inset floodplains and hyporheic exchange induced by in-stream structures on nitrate removal in a headwater stream. *Ecological Engineering*, 97, 452-464.

Ikeda, S., Parker, G., and Sawai, K. (1981). Bend theory of river meanders. Part 1. Linear development. *J. Fluid Mech*., 112, 363-377.

Interagency Workgroup on Wetland Restoration (IWWR). An introduction and user's guide to wetland restoration, creation and enhancement. Developed by the IWWR : National Oceanic and Atmospheric Administration, Environmental Protection Agency,Army Corps of Engineers, Fish and Wildlife Service and National Resources Conservation Service (United States). Last accessed (March 26, 2019) from : https://dec.vermont.gov/sites/dec/files/wsm/wetlands/docs/pub_wetlands_restore_guide.pdf

Ireneusz M. (2005). Rates of lateral channel migration along the Mala Panew River (southern Poland) based on dating riparian trees and Coarse Woody Debris. *Dendrochronologia*, 23(1), 29-38.

Jones, C. S., Kult, K., & Laubach, S. A. (2015). Restored oxbows reduce nutrient runoff and improve fish habitat. *Journal of Soil and Water Conservation*, 70(3), 69A-72A.

Kasahara, T., & Hill, A. R. (2008). Modeling the effects of lowland stream restoration projects on stream–subsurface water exchange. *Ecological Engineering*, 32(4), 310-319.

Kondolf, G.M. (2006). River restoration and meanders. *Ecology and Society*, 11(2), 42.

Kristensen, E. A., Baattrup-Pedersen, A., Jensen, P. N., Wiberg-Larsen, P., & Friberg, N. (2012). Selection, implementation and cost of restorations in lowland streams: A basis for identifying restoration priorities. *Environmental Science & Policy*, 23, 1-11.

Kristensen, E. A., Kronvang, B., Wiberg-Larsen, P., Thodsen, H., Nielsen, C., Amor, E., . . . Baattrup-Pedersen, A. (2014). 10 years after the largest river restoration project in northern europe: Hydromorphological changes on multiple scales in river skjern. *Ecological Engineering*, 66, 141-149.

Kuenzler, E. J. (1990). Wetlands as sediment and nutrient traps for lakes. In Enhancing the States'Lake and Wetland Management Programs, ed. J. Taggart, 105-12. Chicago: U.S. Environmental Protection Agency, NorthAmerican LakeManagementSociety, Northeastern Illinois Planning Commission, Proceedings of a National Conference.

Lamouroux, N., Gore, J. A., Lepori, F., & Statzner, B. (2015). The ecological restoration of large rivers needs science ‐ based, predictive tools meeting public expectations: An overview of the R hône project. *Freshwater Biology*, 60(6), 1069-1084.

Langendoen, E.J., and Simon, A. (2008). Modeling the evolution of incised streams. II: Streambank erosion. *Journal of Hydraulic Engineering*, 134(7), 905-915.

Langendoen, E.J., Motta, D., Abad, J.D., Garcia, M.H., Fernandez, R., Oberg, N. 2015. RVR meander - a toolbox for river meander planform design and evaluation. 3rd Joint Federal Interagency Sedimentation Conference Proceedings. April 19-23, 2015, Reno, NV, pp. 1823-1830.

Larocque, M., Gagné, S., Barnetche, D., Meyzonnat, G, Graveline, M. H. et Ouellet, M. A. 2015. Projet de connaissance des eaux souterraines du bassin versant de la zone Nicolet et de la partie basse de la zone Saint-François - Rapport final. Rapport déposé au

Ministère du Développement durable, de l’Environnement et de la Lutte contre les changements climatiques. 258 p.

Larocque, M., Biron, P.M., Buffin-Bélanger, T., et al. (2016). Role of the geomorphic setting in controlling groundwater–surface water exchanges in riverine wetlands: A case study from two southern Quebec rivers (Canada). *Canadian Water Resources Journal*, 41(4), 528-542.

Lehr, C., Pöschke, F., Lewandowski, J., & Lischeid, G. (2015). A novel method to evaluate the effect of a stream restoration on the spatial pattern of hydraulic connection of stream and groundwater. *Journal of Hydrology*, 527, 394-401.

Lewandowski, J., & Nützmann, G. (2011). Geochemical processes in the aquifer of a floodplain before and after re-opening of a meander. Proceedings of Symposiom H01 Held during IUGG 2011 in Melbourne

Linz, G. M., R. A. Dolbeer, J. J. Hanzel, and L. E. Huffman. (1996). Controlling blackbird damage to sunflower and grain crops in the northern Great Plains. Agriculture Information Bulletin No. 679. Washington, DC: United States Department of Agriculture, Animal and Plant Health Inspection Service.

Lorenz, A. W., Jähnig, S. C., & Hering, D. (2009). Re-meandering german lowland streams: qualitative and quantitative effects of restoration measures on hydromorphology and macroinvertebrates. *Environmental Management*, 44(4), 745-754.

Lüderitz, V., Speierl, T., Langheinrich, U., Völkl, W., & Gersberg, R. M. (2011). Restoration of the upper main and rodach rivers–the success and its measurement. Ecological Engineering, 37(12), 2044-2055.

Massey, W., Biron, P. M., and Choné, G. (2017). Impacts of river bank stabilization using riprap on fish habitat in two contrasting environments. *Earth Surf. Process. Landforms*, 42, 635– 646.

Michaud, A.R., N.-R. Rocha Medrano, R. Lagacé et A. Drouin. (2014). Développement et validation de méthodes de prédiction du ruissellement et des débits de pointe en support à l'aménagement hydro-agricole. Rapport final. Projet réalisé dans le cadre du Programme canadien d’adaptation agricole (PCAA) en collaboration avec le Conseil pour le développement de l’agriculture du Québec (CDAQ). Institut de recherche et de développement en agroenvironnement inc. (IRDA), Québec, Canada, 142 pages.

Morén, I., Wörman, A., & Riml, J. (2017). Design of remediation actions for nutrient mitigation in the hyporheic zone. *Water Resources Research*, 53(11), 8872-8899.

Motta, D., Abad, J. D., Langendoen, E. J., and Garcia, M. H. (2012). A simplified 2D model for meander migration with physically-based bank evolution. *Geomorphology*, 163-164, pp 10-25.

NRCAN (2016, March). Aerial Survey Data Base. Retrieved from https://www.nrcan.gc.ca/earth-sciences/geomatics/satellite-imagery-air-photos/air-photos/9691#photom

Obolewski, K., & Glińska ‐ Lewczuk, K. (2011). Effects of oxbow reconnection based on the distribution and structure of benthic macroinvertebrates. *Clean‐soil, Air, Water*, 39(9), 853-862.

Obolewski, K., Gliska-Lewczuk, K., Ogo, M., & Astel, A. (2016). Connectivity restoration of floodplain lakes: An assessment based on macroinvertebrate communities. *Hydrobiologia*, 774(1), 23-37.

Obolewski, K., Glińska-Lewczuk, K., & Bąkowska, M. (2018). From isolation to connectivity: The effect of floodplain lake restoration on sediments as habitats for macroinvertebrate communities. *Aquatic Sciences*, 80(1), 4.

Oldenborg, K. A., & Steinman, A. D. (2019). Impact of sediment dredging on sediment phosphorus flux in a restored riparian wetland. *Science of the Total Environment*, 650, 1969-1979.

Olsen, Taylor et Buffin-Bélanger, Thomas. (2012). Caractérisation hydrogéomorphologique du ruisseau Martin et de son bassin versant. Rapport présenté au MDDEP et à COPERNIC. Laboratoire de géomorphologie et dynamique fluviale. LGDF-003-UQAR. 43 pages.

Paganelli, D., & Sconfietti, R. (2013). Biodiversity loss in a small riverine wetland of the ticino river (lombardia, northern italy). *Journal of Limnology*, 72(3), 48.

Pander, J., Mueller, M., & Geist, J. (2018). Habitat diversity and connectivity govern the conservation value of restored aquatic floodplain habitats. *Biological Conservation*, 217, 1-10.

Paradis, A. (2015). Integrating hydrogeomorphological concepts in management approaches of lowland agricultural streams in Quebec: Perspectives, problems and prospects, MSc thesis, Concordia University, Montreal, Qc, 129 pages.

Paradis, A., Biron, P. M. (2017). Integrating hydrogeomorphological concepts in management approaches of lowland agricultural streams: Perspectives, problems and prospects based on case studies in Quebec. *Canadian Water Resources Journal*, 42(1), 54-69.

Pedersen, M. L., Kristensen, K. K., & Friberg, N. (2014). Re-meandering of lowland streams: Will disobeying the laws of geomorphology have ecological consequences? *PLoS ONE*, 9(9).

Phillips, J. D. (2013). Hydrological connectivity of abandoned channel water bodies on a coastal plain river. *River Research and Applications*, 29, 149–160.

Piégay, H., Darby, S. E., Mosselman, E., Surian, N. (2005). A review of techniques available for delimiting the erodible river corridor: A sustainable approach to managing bank erosion. *River Research and Applications*, 21(7), 773–789.

Pierce, S. C., Kröger, R., & Pezeshki, R. (2012). Managing artificially drained low-gradient agricultural headwaters for enhanced ecosystem functions. *Biology*, 1(3), 794-856.

Pilařová, P., Bláhová, K. (2014a). Nature park "the meander of the Ohře river" an example of connection between recreation and nature protection. In Fialová, J. & Pernicová, D. (Eds), Conference proceeding on Public recreation and landscape protection – with man hand in hand. Brno: Mendel university, p.9.

Pilařová, P., Bláhová, K. (2014b). Restoration of abandoned meanders – the view of public recreation and landscape protection. In Fialová, J. & Pernicová, D. (Eds), Conference proceeding on Public recreation and landscape protection – with man hand in hand. Brno: Mendel university, p.220.

Reckendorfer, W., Funk, A., Gschöpf, C., Hein, T., & Schiemer, F. (2013). Aquatic ecosystem functions of an isolated floodplain and their implications for flood retention and management. *Journal of Applied Ecology*, 50(1), 119-128.

Riley, W. D., Potter, E. C. E., Biggs, J., Collins, A. L., Jarvie, H. P., Jones, J. I., Quinn, MK., Ormerod, S. J., Sear, D. A., Wilby, R. L., Broadmeadow, S., Brown, C. D., Chanin, P., Copp, G. H., Cowx, I. G., Grogan, A., Hornby, D. D., Huggett, D., Kelly, M. G., Naura, M., Newman, J. R., Siriwardena, G. M. (2018). Small Water Bodies in Great Britain and Ireland: Ecosystem function, human-generated degradation, and options for restorative action. *Science of The Total Environment*, 645, 1598-1616.

Riquier, J., Piégay, H., Lamouroux, N., & Vaudor, L. (2017). Are restored side channels sustainable aquatic habitat features? predicting the potential persistence of side channels as aquatic habitats based on their fine sedimentation dynamics. *Geomorphology*, 295, 507-528.

Roux, C. (2012). Portrait hydrogéomorphologique d'un bassin versant agricole et processus de re-méandrisation : Le cas de la Baie de Lavallière au Québec. Department of Fluvial Geomorphology of Concordia University, Institut de Recherche et de Développement en Agroenvironnement (IRDA) et Département de Mécanique de l'Université de Lyon 1. Montréal. 50p + annexes.

Schilling, K., Zhang, Y., & Drobney, P. (2004). Water table fluctuations near an incised stream, walnut creek, iowa. *Journal of Hydrology*, 286(1), 236-248.

Sear, D. A., Briggs, A., Brookes, A. (1998). A preliminary analysis of the morphological adjustment within and downstream of a lowland river subject to river restoration*. Aquatic Conservation: Marine and Freshwater Ecosystems*, 8, 167–183.

Seer, F. K., Brunke, M., & Schrautzer, J. (2018). Mesoscale river restoration enhances the diversity of floodplain vegetation. *River Research and Applications*, 34(8), 1013-1023.

Seidel, M., Voigt, M., Langheinrich, U., Hoge-Becker, A., Gersberg, R. M., Arevalo, J. R., & Luderitz, V. (2017). Re-connection of oxbow lakes as an effective measure of river restoration. *Clean - Soil, Air, Water*, 45(3), pp 16.

Sidle, W. C., Roose, D. L., & Yzerman, V. T. (2000). Isotope evaluation of nitrate attenuation in restored and native riparian zones in the kankakee watershed, indiana. *Wetlands*, 20(2), 333-345.

Simon, A., Hupp, C.R. (1986). Channel evolution in modified Tennessee channels. Proceedings, Fourth Federal Interagency Sedimentation Conference, Las Vegas, March 24–27, 1986, vol. 2, pp. 5–82.

Simon, A., Rinaldi, M. (2006). Disturbance, stream incision, and channel evolution: The roles of excess transport capacity and boundary materials in controlling channel response. *Geomorphology*, 79, 361–383.

Stammel, B., Cyffka, B., Geist, J., Müller, M., Pander, J., Blasch, G., . . . Kilg, M. (2012). Floodplain restoration on the upper danube (germany) by re-establishing water and sediment dynamics: A scientific monitoring as part of the implementation. *River Systems*, 20(1-2), 55-70.

Stefanik, K. C., & Mitsch, W. J. (2017). Vegetation productivity of planted and unplanted created riverine wetlands in years 15–17. *Ecological Engineering*, 108, 425-434.

Steven, D. D. and Gramling, J. M. (2011). Assessing wetland restoration practices on southern agricultural lands. United States Department of Agriculture, Natural Resources Conservation Service, Conservation Effects Assessment Project. Last accessed (March 26, 2019) from : https://www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcseprd1401639.pdf

Striz, E. A., & Mayer, P. M. (2008). Assessment of near-stream ground water-surface water interaction (GSI) of a degraded stream before restoration US Environmental Protection Agency, Office of Research and Development ….

Surridge, B. W., Heathwaite, A. L., & Baird, A. J. (2012). Phosphorus mobilisation and transport within a long-restored floodplain wetland. *Ecological Engineering*, 44, 348-359.

Te Linde, A., Aerts, J., & Kwadijk, J. (2010). Effectiveness of flood management measures on peak discharges in the rhine basin under climate change. *Journal of Flood Risk Management*, 3(4), 248-269.

Thoms, Martin & Reid, Munique & Reid, Michael. (2015). Ecological significance of hydrological connectivity for wetland plant communities on a dryland floodplain river, MacIntyre River, Australia. *Aquatic Sciences*, 78(1).

Tremblay, T. (2008). Hydrostratigraphie et géologie du quaternaire dans le bassin-versant de la rivière Châteauguay, Québec. Masters Thesis. Université du Québec à Montréal.

USGS (2015, January). National Aerial Photography Program (NAPP). Retrieved from https://lta.cr.usgs.gov/NAPP

Vidon, P., Allan, C., Burns, D., Duval, T. P., Gurwick, N., Inamdar, S., Sebestyen, S. (2010). Hot spots and hot moments in riparian zones: Potential for improved water quality management1. *JAWRA Journal of the American Water Resources Association*, 46(2), 278-298.

Vlad, L., & Toma, D. (2017). Restoring wetlands and natural floodplain along the lower baçeu river in romania. *Environmental Engineering & Management Journal (EEMJ)*, 16(10), 2337-2346.

Wetlands Regulatory Assistance Program. (2000). Installing monitoring wells/piezometers in wetlands. Technical document from the Washington State Department of Transportation. www.wsdot.wa.gov/sites/default/files/2017/07/24/Env-Wet-InstallMonWellsPiezometers.pdf Last access: 02/16/2019.

Wohl, E. (2015). *Rivers in the landscape: Science and management.* John Wiley & Sons, 344 pages.

Zhang, X., & Song, Y. (2014). Optimization of wetland restoration siting and zoning in flood retention areas of river basins in China: A case study in Mengwa, Huaihe River Basin. *Journal of hydrology*, 519, 80-93.

APPENDIX 1 CARACTÉRISATION DE LA VÉGÉTATION AUTOUR DES RUISSEAUX AGRICOLES

Mise en contexte

La présente étude a été produite dans le cadre d'un projet mené par Environnement Canada visant à caractériser trois ruisseaux agricoles dans une optique de restauration. Des travaux en hydrologie et en géomorphologie ont préalablement été menés dans la zone alluviale des trois sites d'études, impliquant notamment la pose de piézomètres. À la demande des dirigeants du projet, des travaux de terrain sur la végétation présente dans la zone alluviale des sites ont été réalisés à l'été 2018. Ces travaux avaient pour principal objectif de délimiter et de caractériser les unités de végétation homogènes dans la zone d'étude.

Description sommaire des sites d'étude

Ruisseau des Fèves

Le ruisseau des Fèves est situé dans la municipalité de Sainte-Martine, qui se trouve dans la MRC de Beauharnois-Salaberry (Figure 1). L'aire d'étude couvre la branche #53 de la rivière des Fèves, qui s'écoule de manière très linéaire entre deux champs agricoles de maïs et de soya. L'aire d'échantillonnage est généralement très étroite (~ 5m), contrainte par la proximité des cultures, et présente une forte pente sur la majorité de sa superficie. Quelques sections plus larges (25-30m) et davantage boisées bordent la rive Sud-Est.

Ruisseau Pot au Beurre

Le ruisseau Pot au Beurre est situé dans la municipalité de Yamaska, qui se trouve dans la MRC de Pierre-De Saurel. Il représente une branche de la Petite Rivière Pot au Beurre, plus sinueuse que le ruisseau des Fève, mais également entouré de champs agricoles de maïs et de soya. L'aire d'échantillonnage couvre une portion d'environ 1km du ruisseau et est généralement très étroite (~ 5m), contrainte ici aussi par la proximité des cultures. Elle présente également une forte pente sur la majorité de sa superficie. Quelques sections plus larges (25-60m) et davantage boisées se retrouvent aux intersections entre différents plus petits ruisseaux perpendiculaires au site d'étude.

Ruisseau Martin

Le ruisseau Martin est situé dans la municipalité de Saint-Samuel, qui se trouve dans la MRC d'Arthabaska. L'aire d'étude couvre la branche du ruisseau qui traverse en deux endroits la rue Sainte-Hélène. Ce tronçon s'écoule de manière assez linéaire sur environ 1km, mais la présence d'anciens méandres a été détectée à l'aide d'analyses géomorphologiques. Une petite portion sur la rive Est du ruisseau est bordée par un champ de maïs, alors qu'une petite portion de la rive Ouest est bordée par une plantation de pins blancs. Le reste du ruisseau est bordé par de larges bandes de forêts et de prairies. La zone alluviale du ruisseau Martin est généralement plus large que celle des deux autres sites d'étude (5-75m) et présente en quelques endroits d'étroites branches s'étirant sur une distance allant jusqu'à 100m du lit du ruisseau.

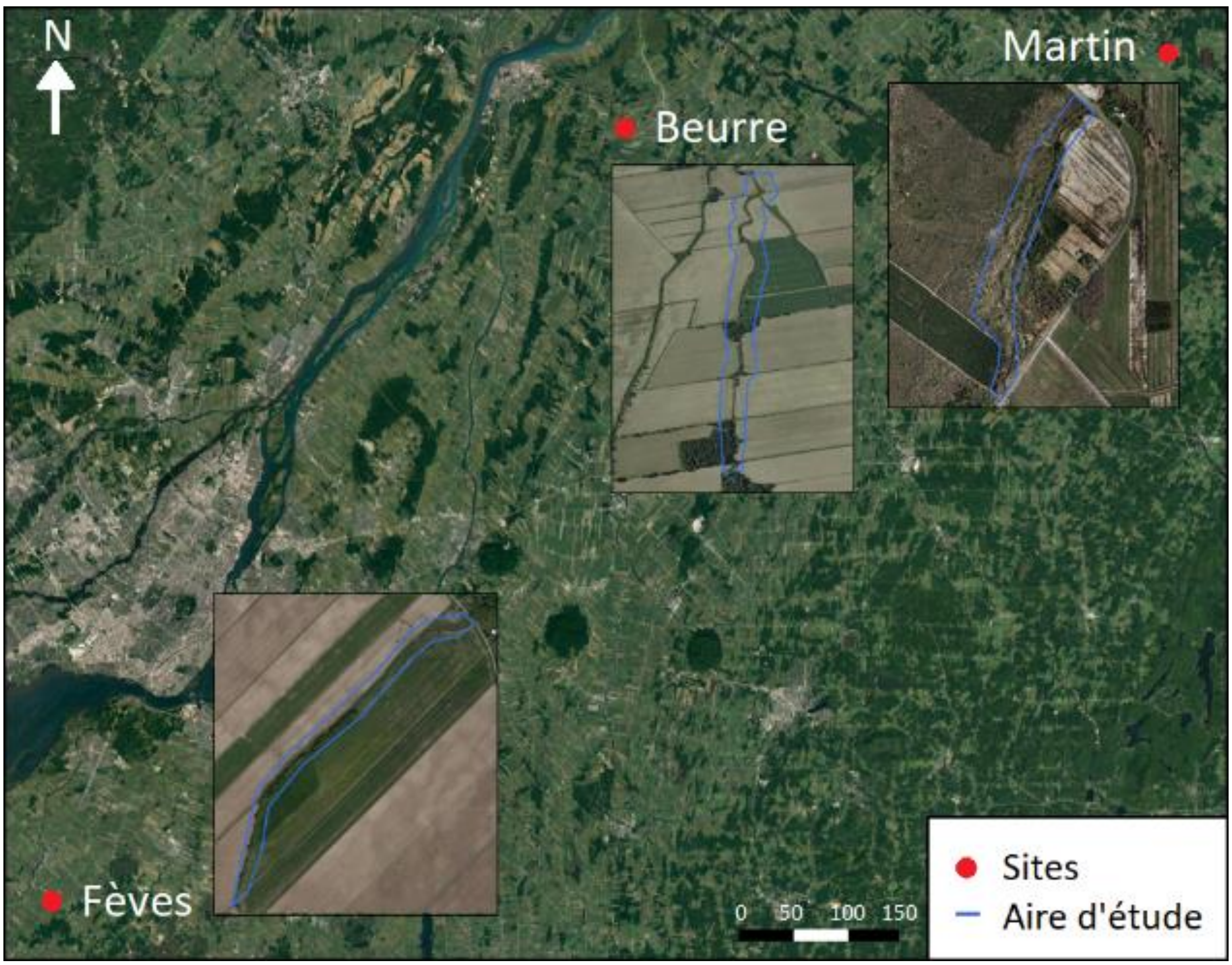


Figure 1 : Carte des sites échantillonnés à l'été 2018.

Méthodologie

Délimitation des unités homogènes

Les travaux de terrain ont été réalisés en cinq jours, entre le mardi 3 juillet et le mardi 10 juillet. La végétation présente dans la zone alluviale aux abords de chaque site a été séparée en unités de végétation homogènes. Une unité représente une portion de terrain dominée par la ou les mêmes espèces. La distinction entre les unités repose donc sur les différences en termes de dominances des espèces présentes dans chacune des strates (herbacée, arbustive et arborée). Chaque unité présente entre le lit du ruisseau et le haut du talus de la zone alluviale a été délimitée à l'aide d'un GPS. Dans le cas des unités en bordure de champs agricoles, la limite extérieure se situe là où la culture commence. Lorsqu'une unité homogène était traversée par un ponceau où la végétation était entretenue, deux unités distinctes ont plutôt été créées. Ce genre de division est seulement retrouvé au ruisseau Pot au Beurre.

Caractérisation

Chaque unité a été caractérisée en fonction de plusieurs variables. Certaines ont été prises directement sur le terrain, alors que d'autres ont été retravaillées suite à l'échantillonnage. Par exemple, le couvert agricole adjacent et l'ouverture de la canopée ont été notés sur le terrain, alors que le type d'unité et le groupement forestier ont été corrigés avec les données de végétation.

1) Type d'unité

Le type d'unité représente une classification très générale en six grandes catégories : Prairie, Marais, Arbustaie, Forêt, Marécage arbustif et Marécage arboré (Tableau 1). Cette typologie dépend d'abord du statu du milieu : humide ou terrestre. Pour le déterminer, la méthode experte de délimitation des milieux humides a été utilisée (Bazoge *et al.* 2014). Cette méthode se base à la fois sur la végétation et sur la nature du sol (Tableau 1).

Tableau 1 : Méthode de détermination du statu des unités échantillonnées.

Végétation typique?	Sol typique?	Statu de l'unité
Oui	Oui	Humide
Oui	Non	Humide
Non	Oui	Humide

Non	Non	Terrestre

Ensuite, le couvert de la végétation dans les différentes strates a été utilisé pour raffiner la classification (Tableau 2). Lorsque le recouvrement de la végétation dans les strates arbustive et/ou arborescente dépassait 30%, les unités ont été considérées comme des arbustaies, des forêts ou des marécages (Zoltai & Vitt 1995; Groupe de travail national sur les terres humides 1997). Autrement, elles étaient classées comme des prairies ou des marais.

Tableau 2 : Méthode de classification des unités en fonction du type de végétation et du statu du milieu (humide ou terrestre)

Type d'unité	Strate déterminante	Humide ou terrestre
Prairie	Herbacée	Humide
Marais	Herbacée	Terrestre
Arbustaie	Arbustive	Humide
Marécage arbustif	Arbustive	Terrestre
Forêt	Arborescente	Humide
Marécage arboré	Arborescente	Terrestre

Sur le terrain, une classification sommaire a été effectuée pour les besoins de la cause. Cette classification a ensuite été raffinée en analysant correctement les données de végétation.

2) Unité détaillée

Puisqu'il arrivait souvent de retrouver deux unités de même type côte à côte, mais où les espèces dominantes étaient différentes, des détails supplémentaires ont été répertoriés. Par exemple, les unités M16 et M17 du ruisseau Martin sont toutes deux catégorisées comme des forêts. Par contre, M16 est dominée par l'érable à Giguère (*Acer negundo*) et le framboisier rouge (*Rubus idaeus*), alors que M17 est dominée par le peuplier faux-tremble (*Populus tremuloides*) et la verge d'or (*Solidago sp*.). Cette classification spécifie donc l'espèce dominante ou les espèces codominantes, toutes strates confondues. D'autres détails pertinents ont également été notés, tels que la présence d'un point d'eau libre, ou d'une portion de terrain récemment exondée, donc dénudé de végétation.

3) Couvert agricole

Pour chaque unité, le couvert agricole adjacent a été noté. Lorsque l'unité n'était pas directement en contact avec un champ agricole, mais plutôt bordée par une autre unité ou un groupement forestier, la mention « Aucun » a été ajoutée. Enfin, lorsque l'unité était adjacente à un pré agricole sans culture définie, la mention « Pré » a été ajoutée.

4) Ouverture de la canopée

L'ouverture de la canopée a été évaluée de façon sommaire en fonction du recouvrement des espèces dans les strates arbustive et arborescente. Cette donnée ne correspond pas à un point spécifique échantillonné dans l'unité, mais représente une moyenne pour l'ensemble de l'unité, afin d'être davantage représentative.

5) Groupement forestier au site

Pour les unités comportant un couvert de plus de 30% dans la strate arborescente (Forêt et Marécage arboré), le groupement forestier a été déterminé en fonction des essences d'arbres et/ou de grands arbustes dominants.

6) Groupement forestier adjacent

Lorsque l'unité était bordée par une section boisée (unité adjacente boisée ou forêt adjacente présente sur le talus), le groupement forestier de cette section a été déterminé en fonction des essences d'arbres et/ou de grands arbustes dominants.

7) Pédon

À chaque unité, un pédon a été creusé afin de déterminer la nature du sol (sableux, silteux, argileux, organique, etc.) et ses caractéristiques (humidité, texture et couleur). La présence de mouchetures a également été notée, puisque leur présence représente une indicateur fiable de la présence d'un milieu humide. En général, le pédon a été creusé dans une section plane au centre de l'unité. Dans le cas où l'unité se trouvait dans une forte pente, le pédon a été creusé dans la portion la plus élevée de la pente, dans une zone où la végétation était représentative de l'unité.

8) Photographie

Chaque unité a été photographiée de manière à ce que la photo prise soit représentative de la végétation retrouvée au sein de l'unité. L'angle de la photographie a également été noté. Une ou plusieurs photos du pédon ont également été prises afin de confirmer la présence de mouchetures ou d'autres caractéristiques importantes, tel que la présence d'eau.

9) Échantillonnage de la végétation

La végétation présente dans chaque strate a été échantillonnée de façon sommaire, c'est-à-dire qu'une recherche exhaustive n'a pas été faite. L'inventaire a été réalisé dans une portion de l'unité représentative de son ensemble. Le recouvrement des espèces présentes dans chaque strate a été noté dans un rayon de 10m en utilisant les classes de recouvrement de Braun-Blanquet : <1% = *, 1-5% = 1, 6-25% = 2, 26-50% = 3, 51-75% = 3 ; 76-100% = 5. Un temps de recherche de deux à cinq minutes a ensuite été alloué à la recherche d'autres espèces à travers le reste de l'unité. Le temps passé à chercher d'autres espèces dépend à la fois de la difficulté de déplacement au sein de l'unité et de sa taille. Une grande unité à forte densité étaient explorée pendant cinq minutes, alors qu'une petite unité très ouverte était explorée pendant deux minutes. Lorsque l'unité s'étendait sur une très grande distance, un second point d'échantillonnage était réalisé et toutes les autres mesures étaient également reprises (Ouverture de la canopée, pédon, photo, etc.). Des spécimens inconnus ont été récoltés à des fins d'identification ultérieure. La nomenclature des espèces suit celle de VASCAN (Brouillet et al. 2010+).

Description des résultats

Un total de 92 unités homogènes a été échantillonné : 17 au ruisseau des Fèves, 33 au ruisseau Pot au Beurre et 42 au ruisseau Martin. Un second point d'échantillonnage a été réalisé dans seulement trois de ces unités, pour un total de 95 points d'échantillonnage. Il est à noter que l'unité M20 du ruisseau Martin a été éliminée, puisqu'elle recoupait en fait une autre unité déjà échantillonnée.

Au total, 177 espèces ont été recensées aux différents sites. 131 ont été recensées dans la strate herbacée, 40 dans le strate arbustive et 15 dans la strate arborescente. Le tableau 3 présente les espèces qui ont été observées le plus grand nombre de fois (présence / absence) dans chacune des strates de la végétation, ainsi que celle qui totalisent le plus grand recouvrement total, calculé en utilisant les médianes des classes de Braun-Blanquet.

Tableau 3 :

Strate	Présence / absence	Recouvrement total
Herbacées	*Thalictrum pubescens* *Vicia cracca* *Impatiens capensis*	*Bromus inermis* *Phalaris arundinacea* *Solidago altissima*
Arbustive	*Alnus incana subsp. rugosa* *Rubus idaeus* *Rhamnus cathartica*	*Alnus incana subsp. rugosa* *Rubus idaeus* *Cornus sericea*
Arborescente	*Acer negundo* *Alnus incana subsp. rugosa* *Salix alba*	*Acer negundo* *Alnus incana subsp. rugosa* *Salix alba*

Les fichiers de données comprennent trois éléments différents :

1) Fichier de données Excel

Le fichier de données Excel comprend quatre onglets. Dans l'onglet « Caractérisation », on retrouve toutes les données des variables de caractérisation pour les unités échantillonnées, ainsi qu'une liste des espèces échantillonnées dans chacune des strates avec leur recouvrement. Les trois autres onglets comprennent seulement les données de la végétation dans une matrice espèce x site pour chacune des strates de la végétation.

2) Couches GIS

La couche shapefile remise contient l'identité et les délimitations de toutes les unités homogènes échantillonnées.

3) Photos

Le dossier de photos comprend toutes les photographies qui ont été prises sur le terrain des unités et des pédons. Chaque image est identifiée en fonction du site (M = Martin; B = Beurre ; F = Fèves) et du numéro de l'unité.

Références

Bazoge, A., Lachance, D., Villeneuve, C., Bérubé, D., Ducruc, J. P., & Lavoie, G. (2014). *Identification et délimitation des milieux humides du Québec méridional*. Développement durable, environnement et lutte contre les changements climatiques Québec.

Groupe de travail national sur les terres humides (1997). *Système de classification des terres humides du Canada*. Centre de recherche sur les terres humides, Université de Waterloo. Ontario, Canada.

Zoltai, S. C., & Vitt, D. H. 1995. Canadian wetlands: environmental gradients and classification. *Vegetation* 118: 131-137.