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THESE

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Spécialité : ECOLOGIE

Présentée et soutenue publiquement par

Louis de REDON de COLOMBIER

Le 12 décembre 2008

INTERETS ECOLOGIQUES DES BORDS DE ROUTE EN MILIEU AGRICOLE INTENSIF

Sous la co-direction de **: Nathalie MACHON, Professeur,**
et Frédéric JIGUET, Maître de conférences

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Soutenue le 12 décembre 2008 devant un jury composé de :

Thierry DUTOIT	Professeur	Université d'Avignon	Rapporteur
Françoise ROZE	Maître de conférences	Université de Rennes	Rapporteur
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Equipe d'accueil : U.M.R. 5173 « *Conservation des Espèces, Restauration et Suivi des Populations* »

Ecole doctorale : E.D. 227 « *Sciences de la Nature et de l'Homme* »

PH.D. THESIS
OF THE
FRENCH NATIONAL MUSEUM OF NATURAL HISTORY

DISCIPLINE : ECOLOGY

ECOLOGICAL INTERESTS OF ROADSIDE VERGES IN INTENSIVE AGRARIAN LANDSCAPES

Defended by
Louis de REDON de COLOMBIER

To obtain the academic degree of:
PHILOSOPHIÆ DOCTOR OF THE FRENCH NATIONAL MUSEUM OF NATURAL HISTORY

Defended on December 12th, 2008 in front of a scientific jury composed by:

Thierry DUTOIT	Professor	University of Avignon	Reviewer
Françoise ROZE	Assistant professor	University of Rennes	Reviewer
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Research Lab: U.M.R. 5173 « *Conservation des Espèces, Restauration et Suivi des Populations* »
College: E.D. 227 « *Sciences de la Nature et de l'Homme*

Gravures issues de Mitchell-Jones et al. (1999)

> *« Nous avouerons que notre héros était fort peu héros en ce moment. Toutefois, la peur ne venait chez lui qu'en seconde ligne ; il était surtout scandalisé de ce bruit qui lui faisait mal aux oreilles ».*
>
> **STENDHAL**
>
> LA CHARTREUSE DE PARME (1839)

PREAMBULE

Cette thèse de doctorat a été réalisée sous la codirection de Nathalie MACHON (Pr.) et de Frédéric JIGUET (Maître de conférences) au sein de l'U.M.R. 5173 M.N.H.N/C.N.R.S./ U.P.M.C. « *Conservation des Espèces, Restauration et Suivis des Populations* » dirigée par Denis Couvet (Pr.)

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SOMMAIRE

INTRODUCTION

“Two roads diverged in a wood, and I —
I took the one less travelled by,
And that has made all the difference”.

(*« Deux routes se séparaient dans un bois,*
Et moi, je pris la moins empruntée.
Cela fit toute la différence »).

ROBERT FROST
THE ROAD NOT TAKEN – 1916.

§ A. INFRASTRUCTURES ROUTIERES ET CRISE DE LA BIODIVERSITE

Pourquoi la biodiversité est-elle en crise ? Quelle est la part de responsabilité des routes dans cette crise et en quoi cela est-il un problème si important en 2008 ? En ce début de XXI$^{\text{ème}}$ siècle, la question des routes devient un sujet de plus en plus important en écologie du fait (a) de leur contribution majeure aux changements globaux qui affectent actuellement les écosystèmes terrestres, et (b) de la prise de conscience collective des enjeux écologiques matérialisée par l'intérêt croissant porté aux questions environnementales au sein de notre société (Angold 1997, Spellerberg 1998, Forman 2000, Ashenden et al. 2003). En effet, les infrastructures routières ont des impacts très importants sur l'environnement (Forman et Alexander 1998, Trombulak et Frissell 2000, Forman et al. 2003, Coffin 2007) et sont par là-même fortement impliquées dans les processus menant aux changements globaux, de manière générale, et à la crise actuelle de la biodiversité, plus particulièrement.

(1) Ampleur et réalité de la crise

La diversité biologique est composée de trois niveaux : diversité des écosystèmes, diversité spécifique et diversité génétique (ou intra-spécifique). La crise que traverse aujourd'hui la biodiversité touche durement ses trois niveaux :

- Au niveau des écosystèmes : une homogénéisation biotique, tant sur le plan taxonomiques que fonctionnel, induisant le remplacement progressif d'écosystèmes riches et complexes par des écosystèmes simplifiés pouvant rendre, entre autre, de plus faibles services écosystémiques (Chapin et al. 2000, Díaz et Cabido 2001) ;
- Au niveau des espèces : des taux d'extinction si élevés que devant l'ampleur du phénomène, certains auteurs ont été amenés à évoquer une sixième grande crise d'extinction (Leakey et Lewin 1995). Bien qu'un temps controversé (Mann 1991, Lomborg 2001), ce phénomène est désormais largement admis au sein de la communauté scientifique (Levin et Levin 2002) ;
- Au niveau génétique : une réduction des populations naturelles de nombreuses espèces (Lacy 1997) induisant une baisse de la variabilité intra-spécifique et la perte d'allèles qui pourraient jouer un rôle dans l'adaptation potentielle et future des espèces aux nouvelles donnes environnementales.

(2) Infrastructures routières et causes de la crise

Les modifications majeures apportées aux paysages au cours du XX^{ème} sont étroitement liées à la construction des routes, à leur utilisation et à leur gestion (Bennett 1991, Noss et Coorperrider 1994). Les impacts des routes sur la biodiversité sont nombreux (Sherwood et al. 2002) :

- Les routes participent aux mécanismes menant à la surexploitation des milieux car elles sont construites pour desservir ou promouvoir l'agriculture, les industries et les commerces qui nuisent à la flore et à la faune locales (Van Dyke et al. 1986, Karnefelt et Mattson 1989, Seibert 1993, Allan et Flecker 1993, Roth et al. 1996) ;
- Les routes participent à la dégradation des habitats naturels car elles consomment des espaces (Sherbune 1985, Reijnen et Foppen 1994, Angold 1997). Ces surfaces sont loin d'être négligeables à l'échelle du paysage. Les routes participent à la fragmentation des paysages qui a de nombreux effets sur la biodiversité notamment sur les dynamiques des populations végétales (Timmins et William 1990, Brothers et Spingarn 1992) et animales (Swihart et Slade 1984, Manserfgh et Scott 1989, Rich et al. 1994). Par ailleurs, les routes, soit suivent l'urbanisation des territoires, soit la permettent en améliorant l'accessibilité de certains espaces (Seibert 1993) qui sont alors consommés indirectement du fait du développement d'infrastructures de transports (Cramer et Hopkins 1982, Harris et al. 1996). Dans les espaces naturels, cette accessibilité des territoires améliorée permet aussi une augmentation de la pression des chasseurs (Bennett 1991, Robinson et Bodmer 1999) et des visiteurs qui endommagent les communautés végétales (Matlack 1993) et perturbent les animaux (Witmer et DeCalesta 1985, Pedevillano et Wright 1987, Del Campo et al. 1990, Czech 1991) ;
- Les routes favorisent les invasions biologiques car leurs accotements permettent la mise en place de continuités biologiques au sein des paysages ; continuités qui favorisent la dispersion d'espèces exotiques (Wester et Juvik 1983, Henderson et Wells 1986, Tyser et Worley 1992, Wein et al. 1992, Greenberg et al. 1997, Parendes et Jones 2000). Cela est aussi vrai pour certaines plantes maritimes qui se retrouvent au cœur des territoires à cause des salages hivernaux (Scott et Davinson 1982). Des expansions de maladies (Dawson et Weste 1985, Gad et al. 1986) et d'insectes (Pantaleoni 1989) sont aussi observées ;

- Les routes peuvent provoquer des extinctions locales car elles constituent de véritables barrières physiques à la dispersion de certaines espèces au sein des paysages (fragmentation) : insectes (Bhattacharya et al. 2003), petits mammifères (Oxley et al. 1974), grands mammifères (Nellemann et al. 2001) et oiseaux (Devely et Stouffer 2001). Cet effet barrière peut notamment trouver sa source dans les nombreuses collisions entre véhicules et faune (Harris et Scheck 1991, Mazerolle 2004). Les collisions peuvent être considérées la première cause de mortalité chez certaines espèces (Forman et Alexander 1998).

Par ailleurs, les routes sont aussi sources de rejets ayant des impacts sur la biodiversité comme les métaux lourds (Goldsmith et al. 1976, Dale et Freedman 1982, Leharne et al. 1992), les molécules organiques (Benfenati et al. 1992), l'ozone (Flueckiger et al. 1984), les nitrogènes (Carslaw 2005, Truscott et al. 2005) et les sels (Bogemans et al. 1989, Molles et Gosz 1980, Hoffman et al. 1981, Peters et Turk 1981, Mattson et Godfrey 1994).

Tous ces polluants se répandent ensuite à travers le paysage et se retrouvent à plusieurs niveaux dans les sols (Byrd et al. 1983, Indu et Choudhri 1991), dans les eaux (Gjessing et al. 1984) ainsi que dans les tissus des plantes (Datta et Gosh 1985, Beslaneev et Kuchmazonokova 1991) et des animaux (Collins 1984, Birdsall et al. 1986, Grue et al. 1986). Les routes entrainent aussi l'émissions de poussières (Farmer 1993, Auerbach et al. 1997) et de bruit (Brumm 2004), une compaction des sols (Helvey et Kochenderfer 1990) et une modification de conditions microclimatiques locales (Asaeda et Ca 1993).

Par ces modifications importantes de l'environnement, les routes perturbent la croissance des végétaux (Sarkar et al. 1986, Kammerbauer et al. 1986, Thompson et Rutter 1986, Spencer et Port 1988, Angold 1997, Viskari et Kerenlampi 2000, Singh et al. 2003) et les taux de germination de leurs graines (Spence 1988). Les routes induisent des changements comportementaux chez les animaux : insectes (Alstad 1982, Mader 1984), gastéropodes (Baur et Baur 1990), batraciens (Van der Zande et al. 1980), petits mammifères (Adams et Geis 1983, Mader 1984, Bakowski et Kozakiewicz 1988, Merriam et al. 1989, Korn 1991, Burnett 1992), grands mammifères (Murphy et Curatolo 1987, Brody et Pelton 1989) et oiseaux (Canaday 1996, Reijnen et al. 1997, Weiserbs et Jacob 2001, Forman et al. 2002).

Enfin, les effets de ces pollutions diminuent avec l'éloignement à la route (Gjessent et al. 1984, Thompson et al. 1986, Post et Beeby 1996, Truscott et al. 2005), la avec la réduction du trafic routier (Goldsmith et al. 1976, Dale et Freedman 1982, Leharne et al.1992) et avec le temps (Byrd et al. 1983, Tong 1990) mais de manière plus relative (Vora 1988).

§ B. INTERET DES BORDS DE ROUTE POUR LA BIOLOGIE DE LA CONSERVATION

« Quelle biodiversité conserver ? », « Comment la quantifier ? », sont des questions importantes sur lesquelles les écologues travaillent depuis de nombreuses années. En effet, la lutte contre la crise de la biodiversité est un impératif pressant compte tenu de l'urgence de la situation et de l'importance de sa conservation. Dans ce contexte, si la problématique liée à la conservation d'espèces menacées d'extinction est relativement connue, celle liée à la conservation de la « Nature ordinaire » est davantage ignorée alors que capitale en termes notament d'ingénierie écologique.

(1) Conservation de la biodiversité et Nature ordinaire

Dès le XIX^{ème} siècle, une prise de conscience de la nécessité de protéger la nature apparaît et, en 1932, le peintre américain George CATLIN propose la création d'un « parc contenant hommes et bêtes dans toute la beauté sauvage de leur nature ». Cette approche romantique de la Nature se traduit à deux niveaux : (a) par la création de parcs (le premier parc naturel est le *Yellowstone National Park* en 1872) et (b) par la mise en place de législations protectrices d'espèces considérée en danger. Les efforts de conservation se focaliseront donc sur certaines zones et certaines espèces. De nombreux débats scientifiques porteront sur ce sujet (Simberloff et Abele 1982, McNeill et Fairweather 1993, Virolainen et al. 1998).

Si les disparitions de milieux naturels et d'espèces constituent des pertes irréversibles, et que les empêcher doit être une priorité, la conservation de la Nature dans sa globalité doit aussi intégrer les préoccupations des scientifiques et des politiques. L'idée d'une nécessaire conservation des milieux naturels et des espèces menacées, à travers la mise « *sous cloche* » d'une partie seulement de la Nature, est aujourd'hui dépassée car trop étroite : changer les modes de gestion au sein de nos paysages devient un impératif sous peine de ne pouvoir régler les problèmes de la biodiversité à long terme dans un contexte où les changements globaux vont conduire à une importante, et désormais inéluctable, réorganisation (Génot et Barbault 2004) car des déplacements d'aire de répartition et des extinctions locales d'espèces sont à prévoir (Parmesan et Yohe 2003, Thuiller et al. 2005). Dans ce contexte, les zones protégées apparaissent comme des îlots au sein d'une matrice constituée d'écosystèmes anthropisés, incapables de protéger efficacement la biodiversité dans sa globalité (Franklin 1993), qui peuvent être regroupés sous le terme de « Nature ordinaire » (Abadie 2008).

(2) Développement de méthodes de suivis de la Nature ordinaire

La loi du 10 juillet 1976 relative à la protection de la Nature impose certes la conservation d'une nature « remarquable » mais, dans le même temps, elle invite aussi implicitement à une réflexion sur les moyens d'organiser une gestion rationnelle du patrimoine naturel dans sa globalité comprenant aussi la Nature ordinaire (Blandin 1986). *« Que mesurer ? »* et *« Comment mesurer ? »* (Yoccoz et al. 2001) sont donc deux questions actuelles capitales en écologie pour permettre une meilleure prise en compte de cette Nature ordinaire difficilement perceptible par la société civile. Dans ce contexte, la recherche d'indicateurs est devenue une des priorités des écologues (Pearson 1994, Ferris et Humphrey 1999, Lindenmayer 1999, Lindenmayer et al. 2000, Duelli et Obrist 2003, Dauber et al. 2003).

En effet, la biodiversité a souvent été le parent pauvre des politiques environnementales car regardée comme un concept trop vague pour être intégré au monde économique réel et aux processus de décision. Cependant ce problème peut être corrigé si de bons outils sont développés pour dépasser la complexité et les multiples niveaux de la biodiversité en offrant une information synthétique et de qualité. Il s'agit de trouver des indicateurs par l'étude de certains groupes taxonomiques bien définis (*ex :* oiseaux et projet STOC) dont les qualités essentielles doivent être (a) une bonne connaissance de leurs taxonomies, (b) une collecte facile des informations relatives à leurs communautés et populations, et (c) une diversité réelle existante en leur sein (Sutton and Collins 1991). Les indicateurs de biodiversité doivent fournir une information à la fois synthétique, scientifiquement rigoureuse et socialement transparente de l'état de la biodiversité (Couvet et al. 2005).

C. LES BORDS DE ROUTE, ZONES CREATRICES D'HABITAT ET DE CONINUITE POUR LA BIODIVERSITE ?

Au sein des paysages d'agriculture intensive, les routes peuvent-elles jouer un rôle dans la lutte contre l'érosion de la biodiversité à travers la création de zones d'habitats et de continuités pour la nature ordinaire ? L'intensification de l'agriculture, issue des années 50 et caractérisée par la suppression des haies, le remembrement, la mécanisation, l'utilisation de produits chimiques (Pain et Pienkowski 1997), a eu d'importants effets sur l'environnement (Burel et al. 1998) et la biodiversité a beaucoup reculé au sein des paysages agricoles (Altieri

1999, Le Cœur et al. 2002). Cela a pu se traduire, par exemple, par des crashs dans les populations d'oiseaux (Donald et al. 2000). Or le maintien de la biodiversité en paysage agricole permet le maintien de nombreux services écosystémiques (Altieri 1999) comme la stabilisation des populations d'insectes ravageurs (Andow 1991), le maintien de la fertilité et de la qualité des sols (Hendrix et al. 1990, Paoletti et al. 1994), ou encore la limitation des pollutions liées aux métaux lourds émis depuis les routes (Dochinger 1980, Greszta 1982).

(1) Bords de route et création d'habitats en paysage agricole intensif

Dans ce contexte, les bords de routes peuvent présenter une opportunité pour le maintien de la biodiversité. Les dépendances vertes des infrastructures de transport, dont les surfaces représentent des chiffres non négligeables à l'échelle du paysage, contiennent une biodiversité importante (Bennett 1991, Sherwood et al. 2002) et, avec les bords de champs (Marshall et Moonen 2002, Meek et al. 2002, Perkins et al. 2002, Vickery et al. 2002, Woodcock et al. 2005), peuvent qvoir un rôle à jouer en biologie de la conservation au sein des paysages agricoles. En effet il a été montré que les bords de route peuvent constituer des zones « habitat » ou « refuge » pour la biodiversité en paysage d'agriculture intensive (Akbar 1997, Spooner et al. 2004, O'Farrell et Milton 2006). Cela est le cas pour de nombreuses espèces de plantes (Way 1977, Hansen et Jensen 1972, Wester et Juvik 1983, Sykora et al. 1993), d'insectes (Samways 1989), de mammifères (Adams 1984) et d'oiseaux (Newbey et Newbey 1987).

Q1. Quelle est l'importance relative des bords de routes comme habitat ou zone refuge pour la faune et la flore au sein de la matrice agricole des quelques îlots d'habitats semi-naturels restant ?

Comme un unique taxon ne peut pas porter suffisamment de renseignements pour connaître l'état des autres (Lovell et al. 2007), il est intéressant de travailler sur plusieurs groupes taxonomiques (Pearman et Weber 2007) comme les communautés végétales (Schaffers et al. 2002, Gelbard et Belnap 2003, Rentch et al. 2004), les communautés carabiques (Day et al. 1993, Lovei et Sunderland 1996, Atlegrim 1997), les communautés d'araignées (Miyashita 1999, Le Viol et al. 2008), ou les communautés d'oiseaux (Canterbury et Blockstein 1997, Boulinier et al. 2001). Les approches multi-taxons sont par ailleurs rares alors que nécessaires (Kotze et Samways 1999). Ainsi, afin de répondre à cette question, l'étude de la biodiversité des bords de route et des milieux adjacents (1) a été réalisée à plusieurs niveaux d'organisation du Vivant (populations et communautés) et (2) a porté sur différents taxons : la flore vasculaire (*Plantae* : *Spermatophyta*), les carabes (*Animalia*, *Insecta*, *Coleoptera* : *Carabidae*) et les petits mammifères (*Animalia*, *Mammalia*, *Rodentia* : *Muridae*, et *Soricophorma* : *Soricadae*).

(2) Bords de route et création de continuités biologiques en paysage agricole intensif

Par ailleurs, si les risques liés aux bords de route comme corridors pour un certain nombre d'espèces exotiques et/ou invasives sont bien connus (Johnson et Johnson 2004), les connaissances scientifiques concernant le rôle des bords de route comme corridors pour les espèces natives restent très fragmentaires et que des études sont fortement espérées sur le sujet (Bailey 2007). Or il a été montré que dans des habitats naturels de taille réduite le maintien de certaines populations peut être compromis (Andrén 1994) sauf si la mise en place de continuités peut assurer l'induction de « *rescue effect* » (Burkey 1989) et la dispersion d'individus malgré la matrice agricole (Tischendorf et Farig 2000, Thomas 2000, Bullock et al. 2002).

Q2.A. Les bords de route permettent-ils la mise en place de continuités biologiques entre des éléments d'habitats semi-naturels isolés au sein de la matrice agricole et quelles espèces peuvent profiter de la mise en place d'une telle connectivité au sein du paysage ?

Si la mise en place de telles continuités a été montrée pour certaines espèces de plantes (Schmidt 1989, Wilcox 1989), d'insectes (Vermeulen 1994), de mammifères (Getz et al. 1978, Warner 1985) et de batraciens (Seabrook et Dettmann 1996), leurs impacts sur les milieux adjacents restent relativement méconnus.

Q2.B. Quels peuvent être les impacts des effets « bords de route » sur la biodiversité des milieux adjacents ? Les bords permettent-ils (1) un maintien et un enrichissement des communautés adjacentes, ou (2), au contraire, leur appauvrissement et leur homogénéisation ?

L'étude des infrastructures routières comme créatrices de continuités biologiques au sein du payage a aussi été menée à différents niveaux d'organisation (populations et communautés) et sur plusieurs taxons (plantes et petits mammifères). L'étude des impacts des bords de route sur les milieux adjacents a quant à elle été réalisée sur la végétation, et plus particulièrement sur la flore forestière à travers l'étude des communautés végétales de petits bois isolés au sein de la matrice agricole ou connectés à la route.

La première partie de la thèse, intitulée « Biodiversité des bords de route en milieu agricole intensif : Intérêts locaux et impacts sur les milieux adjacents », aura pour objectif de répondre à ces trois questions (Q1., Q2.A. et Q2.B.) à travers deux articles soumis à Acta Oecologia (*Road network in intensive agricultural landscape: Refuges, corridors or barriers for small mammals?*) et à Landscape Ecology (*Plant diversity in agrarian landscapes: Ecological roles of roadside verges as refugia and corridors*).

§ D. IMPACTS DES MODES DE GESTION ET DE LA STRUCTURE DES DES BORDS DE ROUTE SUR LA BIODIVERSITE

Les bords de routes sont des espaces fortement anthropisés. L'homme intervient régulièrement sur ces espaces pour les entrenir (ex : fauche de la végétation) ou les restructurer (travaux d'élargissement). Quels sont les impacts de telles mesures de gestion sur la biodiversité des dépendances vertes des bords de route ? En effet, beaucoup d'études sur la biodiversité se sont concentrées, jusqu'à un passé récent, sur des inventaires floristiques ou faunistiques en négligeant le développement de suivis annuels, multi-espèces et multi-sites sur le long terme (Rich et Woodruff 1992). Des suivis de végétation et de petits mammifères ont donc été mis en place en bords de route durant les trois années de thèse.

(1) Fauchage des bords de route et diversité végétale

Les bords de route sont gérés de manière intensive en Seine-et-Marne, *i.e.* la végétation y est fauchée trois fois par an.

Q3. Quels sont les impacts de tels modes de gestion sur la diversité végétale des bords de route ? Quelles seraient les conséquences d'un passage de trois fauches annuelles à une seule en termes de diversité spécifique et fonctionnelle pour les communautés végétales des bords de route ?

(2) Structure des bords de route et dynamique d’un campagnol (M. arvalis)

Les bords de route réprésentent des surfaces plus ou moins importante selon les sites étudiés. L’importance du réseau routier et de ses dépendances vertes n’est cependant pas sans conséquence sur la répartition et la dynamique de la biodiversité au sein du paysage agricole. Il a ainsi été montré que les bords de route pouvaient partiper à la stabilisation des populations de campagnols des champs (*Microtus arvalis*) (Meunier et al. 1997), petit rongeur connu pour ses explosion démographiques cycliques (Briner et al. 2007) préjudiciables à l’agriculture (Delattre et al. 1999).

Q4. Dans quelle mesure et à quelles conditions, les bords de route peuvent-ils participer à un contrôle des fluctuations démographiques cycliques de campagnols des champs au sein du paysage agricole ?

La deuxième partie de la thèse, intitulée « Structures et modes de gestion des accotements routiers : Impacts sur la biodiversité », aura pour objectif de répondre à ces deux dernière questions (Q3. et Q4.) à travers deux articles : un soumis à Journal of Environmental Management (*Effect of delayed single mowing on roadsides vegetation communities: A three years study*) et un autre en révision chez Mamalian Biology (*Possible effects of roadside verges on vole outbreaks in an intensive agrarian landscape*).

§ F. ANNEXES A LA THESE

Quelle peut-être la portée de la thèse située dans un contexte scientifique plus large et diffusée auprès de professionnels de la route ou de l’environnement ? La conservation de la biodiversité en milieu agricole est un vaste sujet qu’il est difficile de traiter à travers une simple thèse. Les annexes apportent quatre textes qui me semblent importants pour la compréhension des enjeux liés à cette thématique de la nature ordinaire au sein des espaces anthropisés : ouverture à la problématique des autoroutes et des bandes enherbées autour des champs ; transmission des savoirs au plus grand nombre et sensibilisation à la conservation de la biodiversité des non-scientifiques : professionnels de l’environnement, agriculteurs ou éco-citoyens.

1) Manuscrits complémentaires

Enfin et afin d'enrichir la discussion, trois manuscrits sont proposés en annexes : deux articles écrits à partir de mes travaux de Masters 2 Recherche, réalisés sein de l'U.M.R. 80 79 « Ecologie, Systématique et Evolution », sur la diversité végétale en milieu agricole et l'impact des bords de champs et de leur gestion (*Effects of local environment and human activities on plant communities of field marginsin an intensive agrarian landscape* et *A three-year study of weed control on plant community in field margins in an openfield landscape*) ; et un article d'Isabelle LE VIOL publié dans *Biological Conservation* (juin 2008), auquel j'ai été associé, sur l'impact de la plantation de haies dans les talus autoroutiers sur les communautés végétales et d'araignées (*Plant and spider communities benefit differently from the presence of planted hedgerows in highway verges*).

(2) Encadrés et Guide technique

Les travaux de thèse qui sont ici présentés ont été financés par la *Direction Générale des Routes* (http://www.route.equipement.gouv.fr) du *Ministère de l'Ecologie, de l'Energie, du Développement durable et de l'Aménagement du Territoire* (http://www.developpement-durable.gouv.fr) dans le cadre de la *Stratégie Nationale de la Biodiversité* (*S.N.B.*, *cf.* Encadré N°4) à travers une convention passée avec le *Muséum national d'Histoire naturelle* en 2005. Cette étude inscrite dans le *plan d'action infrastructures de transports terrestres* avec pour double-objectif de faire progresser la recherche scientifique sur la biodiversité (connaissance de l'écosystème « bords de route », impacts des routes sur les milieux adjacents, développement d'indicateurs et de méthodologies d'expertises et de suivis) et les politiques en faveur de la prise en compte de l'Environnement dans la gestion des infrastructures routières (construction et entretien plus respectueux de la biodiversité) conformément aux engagements pris par l'Etat dans le cadre de la *S.N.B.* (http://www.ecologie.gouv.fr/-Strategie-nationale-pour-la-.html).

Compte-tenu de ce financement, figure aussi en annexe, quatre *« encadrés »* complémentaires détaillant certains points de l'introduction.

MATERIELS & METHODES

« Dans la vie, rien n'est à craindre, tout est

à comprendre ».

MARIE CURIE

§ A. SITES D'ETUDE

L'étude de la biodiversité des bords de route a été menée en Ile-de-France où plus de trois cents sites ont été échantillonnés au sein de différents habitats présents en paysage agricole intensif (bords d'autoroute, bords de route, bords de champ, champs et îlots forestiers). L'échantillonnage des bords de route a été centré sur le département de Seine-et-Marne dans la mesure où le laboratoire CERSP était impliqué dans la création d'un Atlas de la Biodiversité financé par le Conseil Général de ce département. Chaque question scientifique posée nécessitant un protocole d'échantillonnage adapté, seul sera présentée ici la zone d'étude dans sa globalité. Les protocoles d'échantillonnage des sites d'étude sont donnés dans les articles.

(1) Occupation des sols en Ile-de-France (2003)

Malgré 11,5 millions d'habitants (I.N.S.E.E.) en Ile-de-France, seulement 22.1% des sols sont concernés par l'urbanisation avec des infrastructures routières représentant près de 10% de ces espaces urbanisés (*cf.* Figure N°1 et Table N°1). La région se composent essentiellement de grands ensembles agricoles (48.6%), ex : Brie et Beauce, et forestiers (23.4%), ex : vallée de Chevreuse et forêt Fontainebleau (*cf.* Figure N°2).

Figure N°1

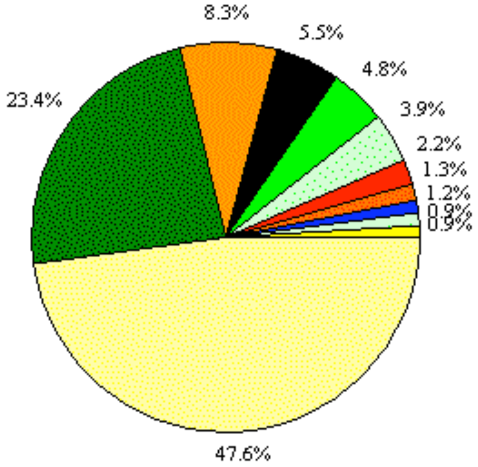
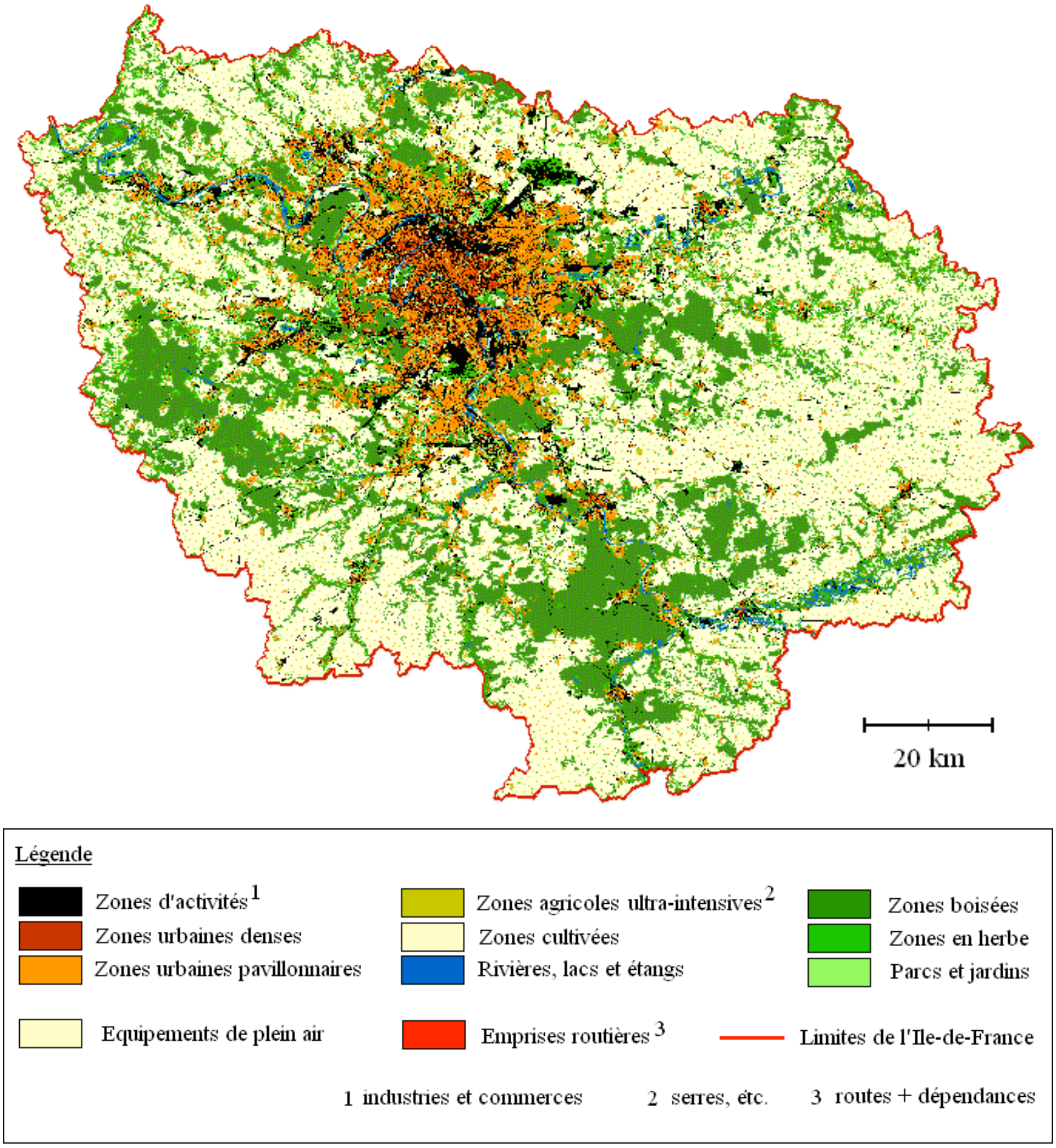


Table N°1

Occupation du sol	Surface (km^2)	Surface (%)
Zones cultivées	5,722.49	47.64
Zones boisées	2,806.26	23.36
Zones urbaines pavillonnaires	994	8.28
Zones d'activité	660.58	5.5
Zones en herbe	574.85	4.79
Parcs et jardins	469.77	3.91
Emprises routières *(hors chemins)*	**259.16**	**2.16**
Zones urbaines denses	156.1	1.3
Rivières, lacs et étangs	147.04	1.22
Equipements de plein air	111.25	0.93
Zones agricoles intensives	109.5	0.91

Occupation des sols en Ile-de-France : repartition en pourcenbtages (Figure N°1 et Table N°1) et en surface (Table N°1)

Figure N°2



Occupation des sols en Ile-de-France

(2) Sites d'étude en bords de route (Seine-et-Marne)

Un total de 68 sites a été échantillonné en bord de route en Seine-et-Marne (*cf.* Figure N°3) à l'aide du logiciel d'information géographique *ArcView 9.0* (SIG) :

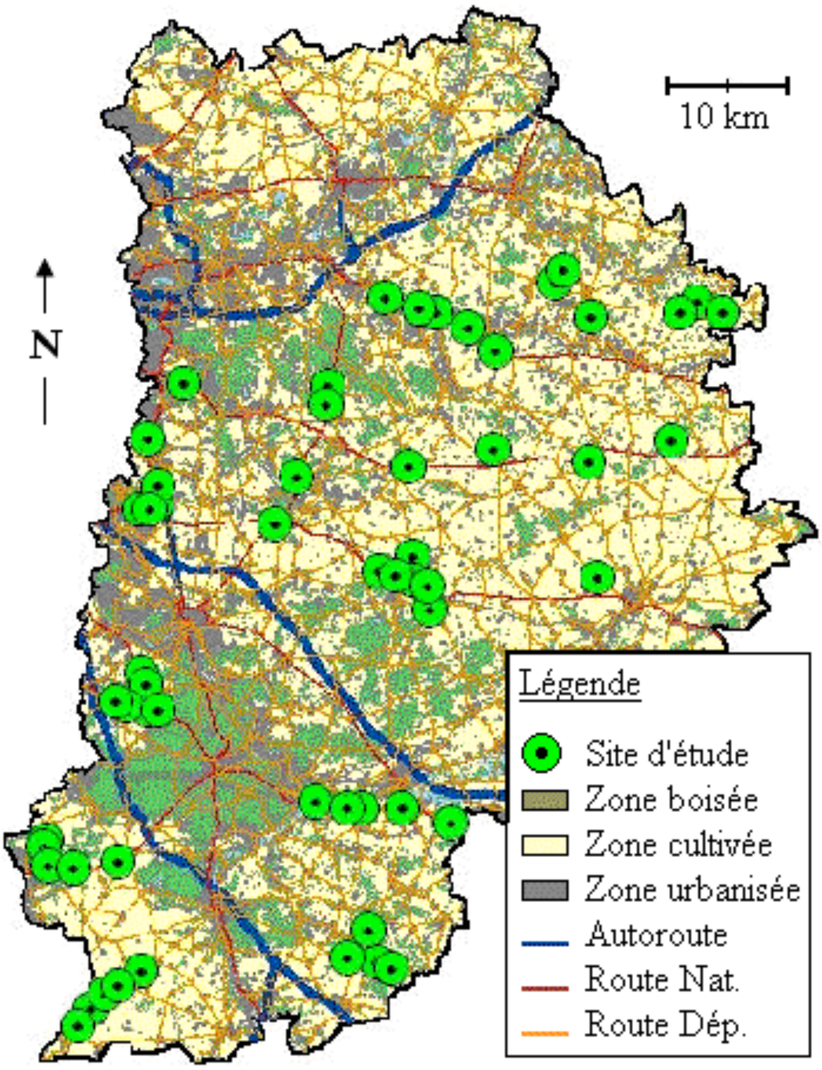
a- Pour l'étude des bords de route comme habitat et zones créatrices de continuité :

- 10 sites pour la flore à l'ouest du département (autour de Provins) ;
- 48 pour les petits mammifères sur l'ensemble du département ;

b- 10 sites pour l'étude des impacts de la fauche sur la diversité végétale ;

c- 45 sites pour l'étude des effets de l'importance de la densité des bords de route sur les fluctuations démographiques de *M. arvalis.*

Figure N°3



Carte de 68 sites d'étude échantillonnés en Seine-et-Marne (77)

NB : Toutes les données exploitées lors des différentes études ne sont pas issues de ces travaux de terrain, notamment en ce qui concerne les petits mammifères. Les données complémentaires ont été obtenues auprès de Christian KERBIRIOU (relevés en bords de champ et en plein champ), Ondine FILIPPI-CODACCIONI (idem), Olivier SCHER (relevés en milieu forestier) et Isabelle LE VIOl (relevés en bords d'autoroutes). Le protocole utilisé entre les différents acteurs était le même et issu d'une réflexion commune.

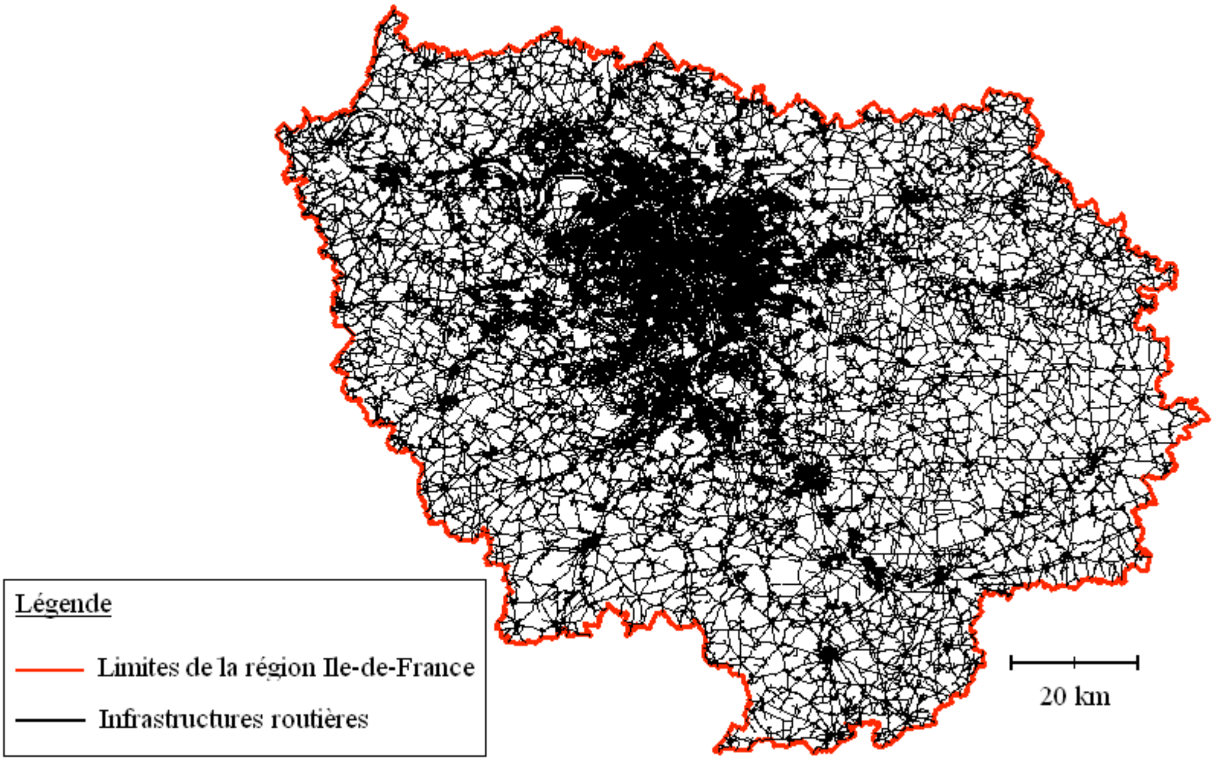
§ B. STRUCTURES ET EMPRISES DES ROUTES AU SEIN DES PAYSAGES AGRICOLES

Que sont les bords de route ? Que représentent-ils au sein des paysages agricoles ? Les infrastructures routières sont souvent accompagnées de dépendances dites « vertes ». Il s'agit d'une zone, plus ou moins importante, comprise dans l'emprise routière mais recouverte par de la végétation. Ces « dépendances vertes », ou « bords de route », ont été aménagées le long comme des zones « tampon » entre l'infrastructure et le milieu adjacent. Ces dépendances, dont l'importance a été longtemps ignorée, constituent aujourd'hui un véritable enjeu en termes d'écologie au sein des milieux agricoles.

(1) Importance du réseau routier en Ile-de-France

Les infrastructures routières sont très denses en Ile-de-France (*cf.* Figure N°4) et représentent plus 40,000 km (sans tenir-compte des chemins communaux et des voies privées représentant plus de 45,000 km) pour une région de 12,000 km². La densité de la voirie principale en Ile-de-France est donc 3.36 km de routes par km².

Figure N°4



Réseau routier principal d'Ile-de-France (2003)

Les infrastructures routières se répartissent ainsi :

- 887 km d'autoroutes, *i.e.* 2.2% du linéaire total pour une densité de 0.07 km.km^2 ;
- 1,568 km de routes nationales, *i.e.* 3.9% & 0.13 km.km^{-2} ;
- 8,518 km de routes départementales, *i.e.* 21.1% & 0.71 km.km^{-2} ;
- 29,311 km de routes communales, *i.e.* 72.8% & 2.44 km.km^{-2}.

Les impacts des routes ne se limitent cependant pas à leur consommation d'espaces en termes de linéaire et à leur environnement proche. En effet, c'est une grande partie des territoires qui sont impactés directement ou indirectement par les routes (*cf.* Encadré N°1).

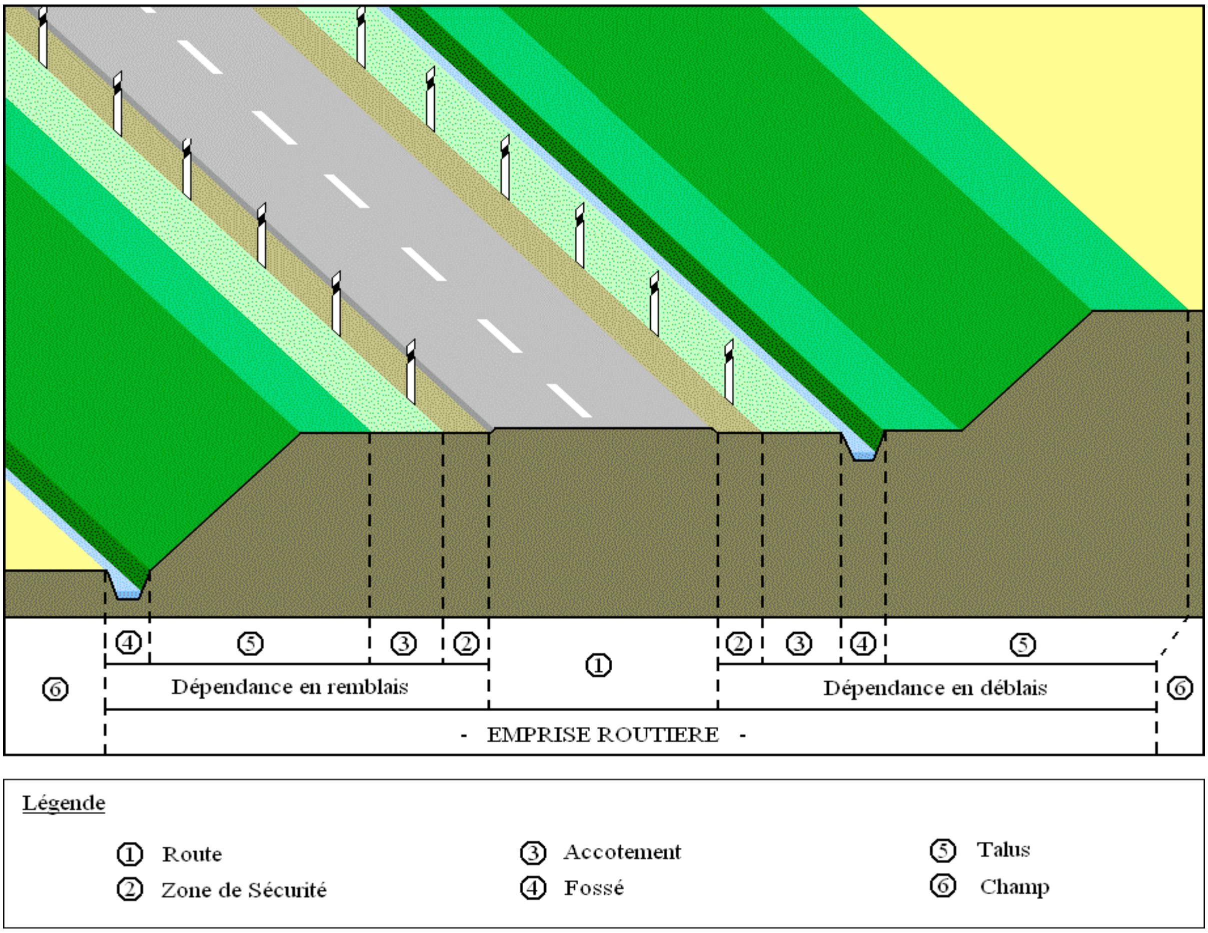
(2) Bords de route en Ile-de-France

En Ile-de-France, la structuration d'une dépendance verte routière est très variable d'une route à une autre en fonction de son importance (nombre de voies et trafic), de sa situation (zones urbaines, agricoles et boisées), de sa période de construction et de la nature même de la route (autoroutes, routes nationales, routes départementales, routes communales et chemins) ; faisant varier son emprise de rien du tout à plusieurs dizaines de mètres. Cependant, de manière générale, l'emprise est constituée de deux à quatre zones (*cf.* Figure N°5) :

- Une zone de sécurité large de 1 à 2 m, la gestion de la végétation y est intensive pour permettre les arrêts d'urgence de véhicules et une bonne visibilité ;
- Un accotement, large de 1 à 5 m, il est fauché plus ou moins intensivement, de une à trois fois par an ;
- Un fossé, large de 0.5 à 2 m, il est curé, *a priori*, de manière décennale ;
- Un talus, large de 1 m à plusieurs dizaines de mètres, il est généralement géré de manière extensive, *i.e.* une fauche par an en fin d'été.

Les dépendances vertes des différentes infrastructures routières représentent d'importantes surfaces (*cf.* Table N°2). Pour un département comme la Seine-et-Marne (5,915 km²), les bords de route représentent ainsi près de 130 km² (*i.e.* 2.2% de l'occupation des sols) quand, par exemple, les espaces naturels sensibles départementaux avoisinent seulement les 6.5 km² (0.11%). Les contributions des différentes infrastructures routières sont plus ou moins importantes et il est intéressant de noter que les bords de chemins et bords de routes communales, concernant des acteurs très locaux (maires et agriculteurs), représentent plus de la moitié de la surface totale des emprises (*NB :* sans même tenir compte des bords de champs).

Figure N°5



Structure des bords de route en France

Table N°2

Type d'infrastructure	Taille (km)	Densité des emprises ($km.km^{-2}$)	Emprise moyenne ($ha.km^{-1}$)	Emprise totale (km^2)	Occupation du sol (emprises) (%)
Autoroutes	*429*	*0.07*	*5.20*	*22.3*	*0.38*
Routes nationales	*699*	*0.12*	*0.90*	*6.3*	*0.11*
Routes départementales	*4,119*	*0.70*	*0.70*	*28.8*	*0.49*
Routes communales	*8,745*	*1.48*	*0.30*	*26.2*	*0.44*
Total/Moy. routes	13,992	2.37	0.60	83.6	1.41
Chemins	22,687	3.84	0.20	45.4	0.77
Total/*Moy.* **toutes infrastructures**	**36,679**	**6.21**	*0.35*	**129.5**	**2.18**

Importance du réseau routier et de ses dépendances vertes en Seine-et-Marne (2003)

§ C. RELEVES DE VEGETATION

Durant les trois années de thèse, un important travail de terrain a été réalisé avec plus de 2,500 relevés floristiques (*cf.* Table N°3). L'échantillonnage a été réparti entre différents terrains et mis en place autour de protocoles différents pour permettre d'apporter les réponses aux questions posées par la thèse : rôle « habitat » et effet « connectivité » des bords de route, impact sur la diversité végétale des habitats adjacents et effets de la fauche sur les communautés végétales des bords de route.

Table N°3

Intitulé du terrain	Groupes étudiés	Années de suivi	Nombre de sites d'étude	Nombre de relevés par site	Nombre total de relevés
PLANTES	Plantes	2007	10	25	250
PETITS MAMMIFERES	Petits mammifères	2006	48	5	240
FAUCHE	Plantes	2006, 2007 et 2008	10	5	150
DYNAMIQUE DES CAMPAGNOLS	Petits mammifères	2006 et 2007	45	5	450

Efforts de terrain 2006-2008

(1) Quadrats de relévé et réplication

Les relevés de végétation ont été réalisés de 2006 à 2008 de manière exhaustive au sein de quadrats de 1m² (0.5 x 2 m) à l'aide d'une échelle de Braun-Blanquet corrigée (*Cf.* Table N°4). Les quadrats ont été placés dans trois zones des bords de route : accotement, fossé et talus.

Table N°4

Recouvrement par espèce	Indice	Recouvrement moyen
Moins de 5%	1	2.5%
5 à 10%	2	7.5%
10 à 25%	3	17.5%
25 à 50%	4	37.5%
Plus de 50%	5	75.0%

Echelle de Braun-Blanquet corrigée

Cependant travailler à l'échelle des communautés peut se révéler difficile car les espèces peuvent avoir des probabilités de détection différentes : certaines espèces sont plus difficiles à détecter ou à déterminer que d'autres ce qui peut induire un biais dans les analyses. Celles-ci doivent donc pouvoir tenir compte de ce biais potentiel et, éventuellement, le corriger ou le contrôler à travers l'estimation de probabilités de détection associées aux espèces ou le calcul de richesse estimées (Boulinier et al). L'intégration aux protocoles de « *réplicats* » (5 par zone étudiée) permet d'obtenir de tels paramètres par des méthodes de « *capture-recapture* ». Les données ainsi corrigées par la prise en compte de probabilités de détection permettent des analyses fiables à l'échelle des communautés (*cf.* Encadré n°2).

(2) Base de donnée GALIUM

Travailler à l'échelle des communautés peut aussi se révéler difficile car certaines espèces peuvent avoir des fonctions écologiques équivalentes : certaines espèces présentes les mêmes traits fonctionnels et il devient alors difficile d'expliquer les compositions des communautés en se fondant strictement sur de la taxonomie. Le développement de nouvelles méthodologies d'étude des communautés, à travers les traits fonctionnels des espèces qui les composent, constitue un sujet de très forte actualité en recherche compte tenu des enjeux qui y sont liés en termes de politiques de conservation (nécessité de maintenir une diversité fonctionnelle au sein des écosystèmes) pour le maintien de services écosystémiques.

Ainsi, l'étude des communautés à travers les traits fonctionnels (*i.e.* « des traits morpho-physiologiques qui impactent indirectement la fitness d'un individu via leurs effets sur la croissance, la reproduction et la survie », Violle et al. 2007) est aujourd'hui une nécessité (McGill et al. 2006) car l'approche taxonomique ne suffit pas à décrire les fonctionnements des écosystèmes (Eviner 2004) et que les traits fonctionnels constituent de bons prédicteurs pour l'étude des communautés (Lavorel et Garbier 2002, Pywell et al. 2003). Les informations complémentaires apportées par les traits sont essentielles à la compréhension des dynamiques des communautés (Noble et Slatyer 1980, Fukami et al. 2005, Roy et Blois (de) 2006, Endels et al. 2007, Diaz et al. 2007, Aubin et al. 2007) ainsi qu'à leur éventuelle homogénéisation (Devictor et al. 2007). De plus l'analyse de la diversité fonctionnelle des communautés permet de mieux comprendre le fonctionnement des écosystèmes (Petchey et Gaston 2002), élément capital en termes de recherche pour la mise en valeur et la quantification des services écosystémiques.

Afin de travailler sur les traits de vie, un groupe s’est mis en place au laboratoire autour de Sophie GACHET afin de constituer une base de données sur le modèle BASECO (Gachet et al. 2005) pour les plantes de la région parisienne : GALIUM (*cf.* Figure N°6).

Figure N°6

GALIUM

Nombre d’espèces renseignées : 928 (avril 2008), espèces détectées lors des différents terrains des participants au projet.

Traits renseignés : 12, Ex : phénologie, modes de pollinisation, modes de dispersion, capacités, aptitude à la multiplication végétative, taille à floraison, etc.

Participants : Sophie Gachet (coordinatrice), Jean-Claude Abadie, Isabelle Le Viol, Audrey Muratey, Maëlle Rambaud, et Louis de Redon.

Sources : *Flore des champs cultivés* de Philippe JAUZEIN, *Flora Helvetica : Flore illustrée de Suisse* de Konrad LAUBER et Gerhart WAGNER, et *Nouvelle flore de la Belgique, du G.D. du Luxembourg, du Nord de la France et des régions voisines* de Jacques LAMBINON, Léon DELVOSALLE et Jacques DUVIGNEAUD.

Constitution de la base de données GALLIUM

§ D. RELEVES FAUNISTIQUES

Originellement pensé pour la capture de carabes, le protocole de piégeage a très fonctionné pour les petits mammifères dont l’étude a donné de nombreux résultats. Au total 480 pièges ont été posés entre 2006 et 2007 (*cf.* Table N°4).

(1) Pot-pièges « Barber »

Le piégeage daunistique a été effectué à l’aide de pot-pièges de type « *Barber* ». Chaque piège, mesurant 10.2 cm de haut pour 7.5 cm de diamètre, a été enfoncé dans le talus à 5 m de la route. Rempli d’environ 155 mL de solution (75 ML d’eau, 75 mL de monopropylene-glycol, conservateur non-attractif, 5mL de surfactant et 15g de sel), chaque piège est couvert par un toit en plastique carré de 15 cm de côté pour éviter que les précipitations ne le remplissent (*Cf.* Figure N°7).

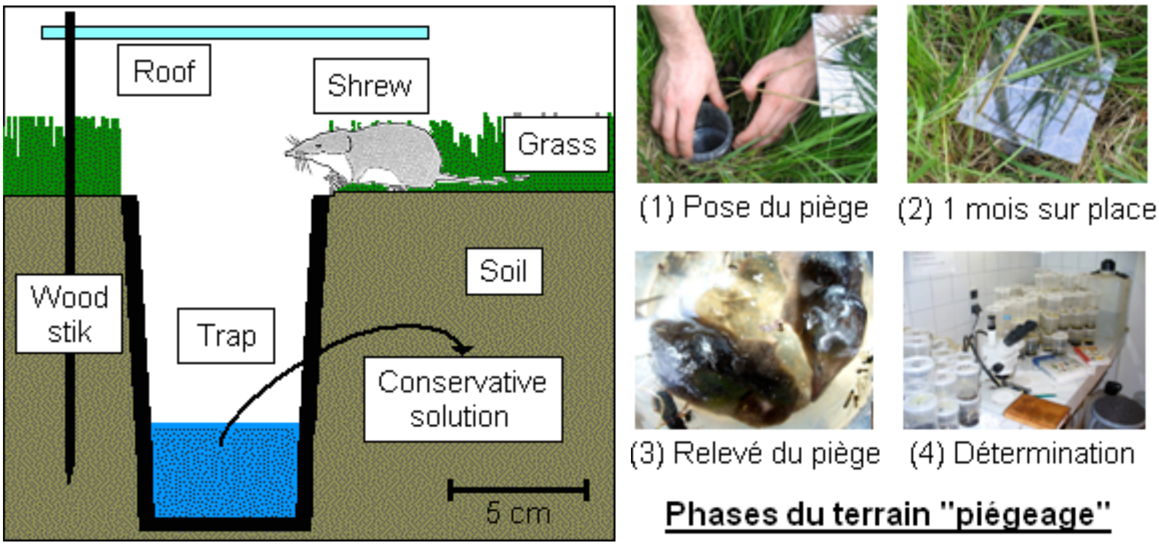
Le système est laissé sur place durant les quatre semaines de mai pour éviter un impact trop fort des conditions climatiques (un mois en mai permet de rencontrer pluies et beau temps).

Une fois les pièges récupérés, leur contenu a été transféré dans l'alcool pour permettre une meilleure conservation des animaux piégés en attendant leur détermination (l'alcool n'ayant pas été utilisé dès le piégeagé car très attractif). Les animaux ont été ensuite déterminés à la loupe binoculaire : Carabidae et petits mammifères (mulots, musaraignes et campagnols).

(2) Réplication des relevés

Comme pour les plantes, 5 replicats ont été posés sur chaque site d'étude chacun à une distance de 20m ; distance qui permettait l'indépendance des pièges les uns des autres pour le piégeage des insectes carabiques. Les pots-pièges ne peuvent cependant pas être considérés comme indépendants pour le piégeage des petits mammifères.

Figure 7



(1) Pose du piège (2) 1 mois sur place

(3) Relevé du piège (4) Détermination

Phases du terrain "piégeage"

Protocole « petits mammifères »

§ E. ANALYSES DE SOL

Les bords de route sont des milieux très pollués, des analyses de sol ont donc été effectuées sur les sites d'étude afin de les intégrer comme variables de contrôle lors des analyses. Les relevés ont été effectués en 2006 au niveau de l'accotement (à 1m de la chaussée) et du talus (à 5m de la chaussée) de 48 sites en Seine-et-Marne.

(1) Echantillonnage du sol

Pour analyses, 100g de terre était prélevés dans les deux zones de bords de route échantillonnées (accotement et talus) à 10cm de profondeur en 5 endroits distincts de chaque site d'étude (au niveau des 5 réplicats de relevé de végétation). Les 5 prélèvements étaient ensuite regroupés et mélangés afin d'obtenir un échantillon de terre de 500g par accotement et par talus de chaque site. Les 96 échantillons de terre de 500g on ensuite été envoyés pour analyses au Pôle d'Aspach de la SADEF (www.sadef.fr).

(2) Eléments analysés

Les analyses ont portées sur cinq paramètres : le pH (méthode NF ISO 10 390), les quantités de Na_2O échangeable (NFX 31-108), d'azote (NF ISO 13 878), de phosphore (P_2O_5, Méthode d'Olsen : NF ISO 11263) et de plomb (NFX 31-147).

PARTIE I

BIODIVERSITE DES BORDS DE ROUTE EN MILIEU AGRICOLE INTENSIF : INTERETS LOCAUX ET IMPACTS SUR LES MILIEUX ADJACENTS

LES MANGEUX D'TERRE (*extraits*)

[…] Si je m'souviens,
Voyons dans c'coin d'Beauce.
Y avait dans l'temps un bieau grand ch'min
[…]
A c't'heur' n'est pas pus grand qu'ma main…

[…] Mang'rint on n'sait quoué ces gas-là,
l's mang'rint d'la marde !
Le ch'min c'était, à leu' jugé
D'la bonn' terr' pardue:
A chaqu' labour i's l'ont mangé
D'un sillon d'charrue…

[…] Mais l'pauv' chemin en est d'venu
Minc' comme eun' couleuve.

[…] Z'ont semé du blé su l'terrain
Qu'i's r'tir'nt à ma route ; […]
Les épis baisser l'nez d'vant moué
Comm' s'i's avaient honte!...

[…] Et j'f'rai ben r'culer vos mouéssons,
Ah! les mangeux d'terre ! …

Y avait dans l'temps un beau grand ch'min,
[…]
A c't'heur' n'est pas pus grand qu'ma main…
J'pourrais bien l'élargir, demain !

Gaston COUTE (1980 – 1911)

MANUSCRIPT N°1

Submitted to Acta Oecologica

Plant diversity in intensive agrarian landscapes:
Ecological roles of roadside verges as refuge and dispersal corridors

Louis de REDON
Frédéric JIGUET
Isabelle LE VIOL
Sophie GACHET
Jeffrey B. JOY
&
Nathalie MACHON

PRESENTATION

TITRE

Diversité végétale au sein des paysages agricoles intensifs : Rôles écologiques des bords routes, Refuge et corridor pour la flore native.

RESUME

Au sein des paysages agricoles intensifs, les bords de route peuvent-ils constituer un réel habitat pour la flore native ainsi qu'un corridor pour la dispersion des plantes forestières entre différents fragments forestiers ?

Notre étude a porté sur une zone de 520 km² située en Seine-et-Marne (77) principalement composée de champs et de bois. Dix sites ont été échantillonnés afin d'évaluer la richesse et la diversité des communautés végétales dans différents milieux. Dans chaque site, des inventaires floristiques ont été menés dans cinq sous-sites : un champ, un bois isolé dans le champ, un bois adjacent à un bord de route, un bord de route adjacent au bois étudié et un bord de route adjacent au champ échantillonné.

Seulement 2.5% de la diversité végétale détectée a été relevée dans les champs et plus de 44% dans les bois. Plus de 65% des espèces détectées ont été observées en bords de route (soit environ 25% de la diversité végétale locale, *sources : C.B.N.B.P.*) Une majorité des espèces (51%) ont seulement été relevées en bords de route où un nombre important d'espèces forestières ont aussi été détectées (40). Il a aussi été montré une forte corrélation négative entre la proportion de plantes forestières au sein des communautés végétales des bords de route et les distances aux bois les plus proches (isolés et adjacents).

Ces résultats, couplés aux différences observées entre les communautés végétales des bois connectés et celles des bois isolés, ont permis de démontrer le rôle des bords de route comme corridor pour certaines espèces végétales entre les différents patchs d'habitats naturels répartis au sein de la matrice agricole.

Cette étude a donc permis de démontrer, que dans un paysage d'agriculture intensive, les bords de route peuvent offrir un refuge de qualité à la flore native et servir de corridor pour la dispersion des plantes entre des fragments de bois isolés.

MOTS CLEFS

Biodiversité, Bois, Communautés végétales, Diversité α et β, Forêt, Homogénéisation biotique, Impacts des routes.

PLANT DIVERSITY IN AGRARIAN LANDSCAPES: ECOLOGICAL ROLES OF ROADSIDE VERGES AS REFUGE AND CORRIDORS

Louis de REDON[1*]; Frédéric JIGUET[1]; Isabelle LE VIOL[1]; Sophie GACHET[2]; Jeffrey B. JOY[3] and Nathalie MACHON[1].

KEYWORDS

α-diversity
β-diversity
Biodiversity
Biotic homogenisation
Forest
Groves
Plant communities
Road impacts

ABSTRACT

We studied a 520 km^2 *area crossed by roads in central France to evaluate if roadside vegetation communities are valuable (1) as refuge for native plant diversity and (2) as corridors for dispersal of forest plant species in intensive agrarian landscapes. The area is primarily composed of agricultural fields and groves. Ten sites were sampled to evaluate species richness and diversity of roadside plant communities. At each site, we inventoried five subsites: agricultural field, grove adjacent to roads, grove isolated from roads and surrounded by fields, roadside adjacent to groves and roadside adjacent to agricultural fields. Only 2.5% of the plant diversity was found in crop field subsites and 44% in groves. More than 65% of plants were observed in roadsides (representing about 25% of the total regional plant diversity). A majority of plants (51%) were only detected in this habitat where we also found a large number of forest species (40). We showed strong negative correlations between percentages of forest plants in roadsides and distances to groves. Those relations coupled with the observation of enhanced plant diversity in groves adjacent to roadsides compared to isolated ones demonstrated the role of roadsides as corridors between natural habitats for plant communities. Taken together, the results of this study show that in intensive agricultural landscapes roadsides may often serve as refuge for native plant diversity and act as corridors for plant dispersal between isolated forest fragments.*

I. INTRODUCTION

Roads are known to have deleterious effects on plant diversity (Forman andAlexander, 1998): during construction they cause the destruction of natural habitats (Forman, 2000), they alter the surrounding environments through pollution produced by vehicles (Trombulak and Frissell, 2000), they promote the dispersion of exotic/invasive plant species (Parendes and Jones, 2000) and facilitate the penetration and use of natural zones by humans (Seibert 1993). Agriculture also detrimentally alters biodiversity through intensive cultivation of large areas of land (Burel et al., 1998). In agricultural landscapes, plant diversity is generally low consisting primarily of crop species growing in agricultural fields. In a large part of European agricultural landscapes, plant diversity is concentrated at field boundaries, groves and roadsides. For example, 45 % of the plant diversity was found along roadsides in the United Kingdom (Way, 1977), and 50% in the Netherlands (Sykora et al., 1993). Finally, in agricultural landscapes, despite the

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negative influence of roads on environment, roadsides could constitute one of the main refuges for wild plant species (Bennett, 1991; Akbar, 1997).

Roadsides, may also favour the dispersion of plant species (Ross, 1986; Parendes et al., 2000) and be considered as corridors (Corbit et al., 1999). They create connections between isolated natural patches of habitats (such as groves). Vehicle traffic on roads is known to increase seed dispersal (Schmidt, 1989; Lonsdale and Lane, 1994). Corridor effects of roadsides have been assessed for marine (Scott and Davinson, 1985; Wilcox, 1989) and meadow plants (Tikka et al., 2001), but they are generally difficult to highlight without the use of genetic markers (Machon et al. 2003) and studies are needed to assess the effects of connectivity on biodiversity in fragmented landscapes (Bailey, 2007).

In many countries, roadsides cover large areas. The French main and secondary road networks are approximately 1 000 000 km long (i.e. 1.82 km of roads per km^2, data from year 2005). On both sides, verges cover 5 100 km2 i.e. near 1% of the country area (540 000 km2), more than the total area of French National Parks (representing 3 400 km2). Thus, such large areas could play a significant role in plant conservation.

The aim of the present study is to assess the role of roadside verges as refuges and corridors for wild plant species in agricultural landscapes. We studied a typical intensive agrarian landscape in central France, about 60 km from Paris composed almost exclusively of agricultural fields and little woods noted as "groves" (usually smaller than 5 ha), the latter being fragmented and spread within the landscape. Some groves are isolated (i.e. completely surrounded by crop fields) while others border roadside verges. We performed plant inventories in groves, agricultural fields and roadsides to estimate (1) which plant species are harboured in roadsides; (2) if differences in species richness and composition of plant communities exist between isolated groves and groves adjacent to roadsides; and (3) the role of roadsides as corridor for forest plants between groves.

II. MATERIALS AND METHODS

Using the following design we evaluated three hypotheses:

(1) By comparing plant species in roadsides with those in adjacent habitats, we asked if roadside habitats harbour native plant species which are not found elsewhere in the agricultural landscape.

(2) We hypothesized that native plant diversity would be higher in groves adjacent to roads when compared with isolated groves thanks to corridor effects of roadsides.

(3) To evaluate the roles of roadside habitats as corridors we also assessed the percentage of forest species in roadside verges as a function of the distance to the nearest isolated or adjacent grove, and compared the linear trends obtained.

1. Study area (*Figure 1A*)

The study area is approximately 520 km2 and is located in the centre of France in the eastern part of the department of Seine-et-Marne, about 60 km east of Paris (48°47' North - 3°18' East, *Figure 1*). The area has a typical western European temperate climate. It is an intensive agricultural zone of the suburb of Paris. The area is mainly composed of agricultural fields (79.7 %), groves (12.5 %) and roadside verges (1.6 %). The remaining 5% is composed of villages, meadows, rivers, lakes, roadways, railways, etc. The road network is 852 km long i.e. 1.64 km of roads per km2. Road verges cover approximately 8.32 km2, namely 1.6 % of the total study area.

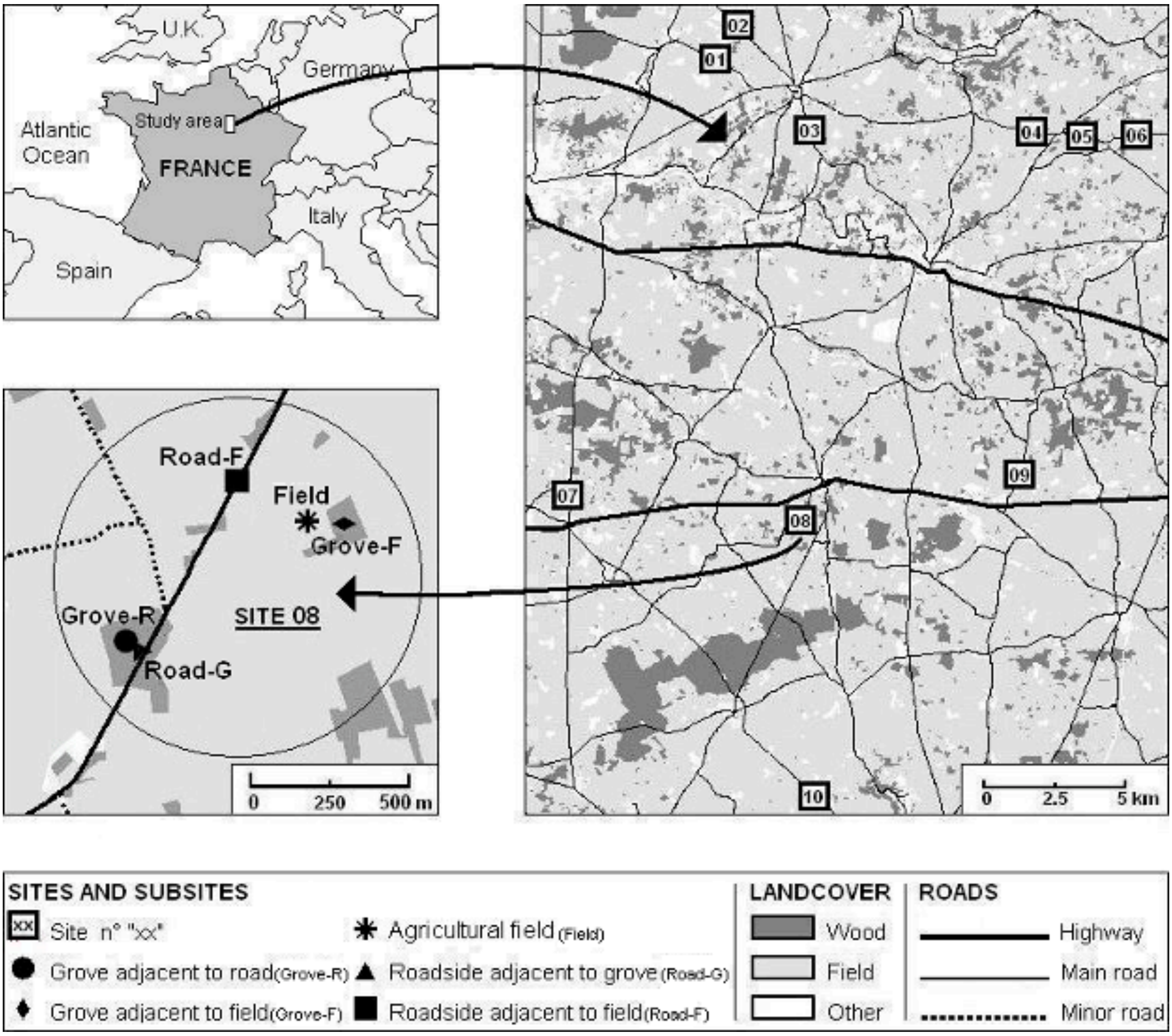
2. Sampling design (*Figure 1B*)

Our study comprised ten similar sites, each mainly composed of agricultural fields (> 80%), crossed by a road, containing at least two groves (woods from 1 to 4 ha and $\frac{Area}{Perimeter} > 20$), one adjacent to the roadside verge (noted Grove-R), the other isolated from roads (noted Grove-F) i.e. situated among the fields (more than 100 m and less

FIGURE 1:
SAMPLING DESIGN

The study area is composed of ten sites situated in central France. Each site is divided into five subsites representing major local habitats: one grove connected to a roadside (Grove-R), one grove isolated in a crop field (Grove-F), one crop field (Field), one roadside adjacent to the grove (Road-G) and one roadside adjacent to the crop field (Road-F). The five subsites are located in a circle 1,000 m in diameter and comprise five 1m^2 plots (2 x 0.5 m) where an exhaustive vegetation inventory has been conducted (using a corrected Braun-Blanquet scale).

A. Study area

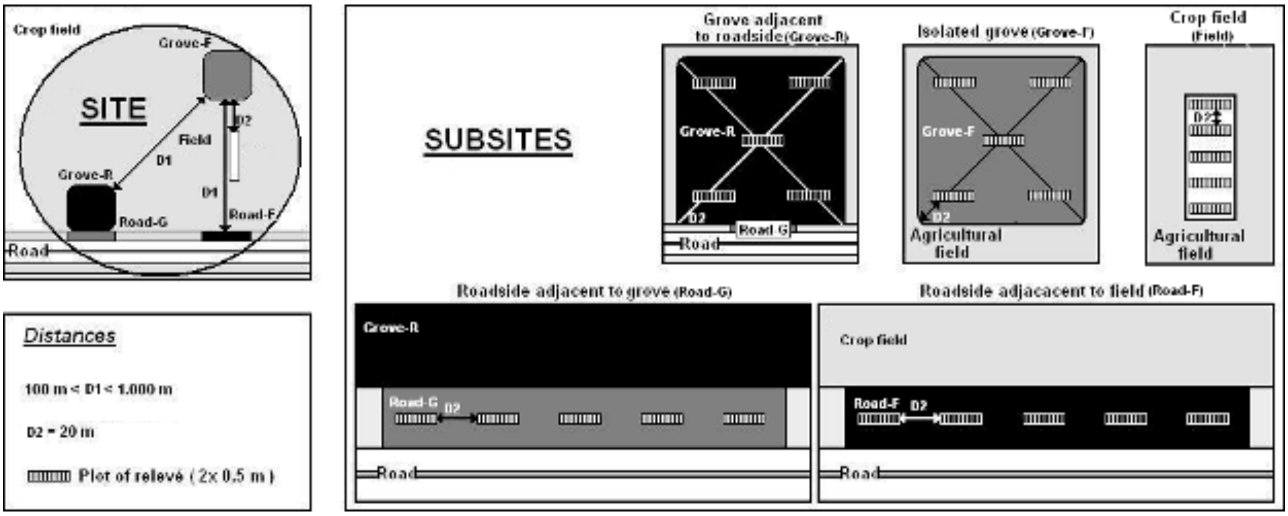


than 1 000 m from any road). The two groves at each site were at least 1 000 m apart. To avoid potential biases due to the influence of heterogeneity in local environmental conditions, the ten sites were selected within an area of less than 30 km diameter.

For each site, we performed vegetation inventories in five subsites: (1) the grove near the road (Grove-R), *i.e.* connected to roadside, (2) the grove among agricultural fields (Grove-F),), *i.e.* isolated from roadside, (3) the roadside adjacent to the connected grove (noted Road-G), (4) a roadside adjacent to the field containing the isolated grove (noted Road-F) and the crop field containing the isolated grove (noted Field). Each subsite will

FIGURE 1:
SAMPLING DESIGN

B. Subsites sampling design



be recognized thanks to its code: the habitat information (grove, road, field) followed by a letter describing the adjacent habitat (G for grove and F for field), except for the crop field noted just as "field".

At each subsite, vegetation inventories were performed in five systematically placed plots (2 m x 0.5 m). Their locations within a habitat were determined as follows:

(a) In the groves, two perpendicular transects were defined through the whole grove and plots were located, one at the intersection of the two transects, and the four others at 20 meters from the inside edge of the grove along transects.

(b) In the roadside habitat, five plots were regularly placed every 20 meters along the road in the bank perpendicular to the grove (1 m from the adjacent habitat).

(c) In the crop fields, the five plots were placed every 20 meters along a transect linking the Road-F subsite to the isolated grove (Grove-F). The first plot was located at 20 meters from the grove in the field.

3. Vegetation inventories

Vegetation sampling was performed during spring 2007 from April 3rd to 15th. The period was short enough to avoid temporal bias in vegetation development. In each plot, we listed all plant taxa detected and evaluated their abundance with a corrected Braun-Blanquet method (Braun-Blanquet, 1932; Westhoff and Van der Maarel, 1978); because vegetation could be layered, cover may exceed 100%. Almost all the taxa were identified to species level according to the International Plant Names Index (http://www.ipni.org). In some cases (*Crepis sp.* for example), taxa were identified to genus level due to difficulties in identification when roadside verges had been mowed. We did not distinguish the different species of moss and ferns (except *Equisetum arvense*), these taxa were recorded as "moss" and "ferns". We considered a species as a "forest species" if it was listed in the reference French forest flora (Rameau, 2005).

4. Data analysis

The floristic value of a community was evaluated through the calculation of its richness and diversity.

a. Diversity index

For each plot, we calculated a Hill diversity index (Hill, 1973) that we adapted to our plant sampling. We used the median recovery value

(Pyšek, 2004) of our Braun-Blanquet corrected scale given by the *Appendix 1*.

$$p_i' = \frac{R_n}{\sum_{n=1}^{S} R_n},$$

$$N_1' = -\sum_{i=1}^{S} p_i' \times \log_2(p_i'),$$

$$N_2' = \frac{1}{\sum_{i=1}^{S} p_i'^2},$$

$$E_{1,2}' = N_1' / N_2'.$$

S is the number of species detected in a plot.

R_n is the median Braun-Blanquet recovery value of the nth species of a plot.

p_i' is the relative abundance of ith species in a plot.

To have a more intuitive diversity index, we calculated $E = 1 - E_{1,2}'$ (*E* has a maximum value for a maximum of diversity, $E \in [0,1]$).

In subsequent analyses, we used an *exponential* transformation for Hill corrected index (*E*) to meet normality assumptions.

b. Species richness and similarities

First, to calculate species richness (*S*) we counted the number of species observed at each subsite. But considering the five plots as replicates, we also used a capture-recapture approach to account for heterogeneity in detection probability among plant species over the sampled sites. We ran the software ComDyn (Hines et al., 1999) using spatial replicates and we made the null hypothesis that vegetation should not differ from a plot to another in each subsite. We obtained estimated species richness *S** for each subsite and the estimated proportion of species found on pairs A and B of two subsites (noted $\varphi_{A,B}$). We could not run ComDyn for plots located in agricultural fields because the number of species found in that habitat was insufficiently large to do so. We used φ as a similarity index and further built a matrix of similarities. We also calculated estimated species richness and percentage (Q^*_{FOR}) of forest plants in each subsite.

In further analyses, we used a *log* transformation for observed species richness (*S*) to meet normality assumptions.

c. Indices equalities between subsites

To test for differences in species richness and similarity equalities between subsites, we performed student tests of mean equalities for paired observations between the six possible pairs of subsites at each site (two by two). To test for differences in Hill corrected index between subsites, we performed ANOVA with subsites as explaining factors. We used the F statistic for variance equality tests.

d. Species composition of subsites

We performed a PCA on subsites and plant species composition of subsites (presence/absence data). To avoid bias due to the presence of rare species, more difficult to detect, we removed from the dataset species recorded in less than seven subsites.

e. Road impacts on plant communities

To test for effects of connection to a roadside on plant communities, we created a variable "connection" to each grove, taking a value of 1 for Grove-R (i.e. subsites adjacent to roadsides) and 0 for Grove-F (i.e. groves isolated in fields). We then performed an ANOVA on *S* and *E* indices with controlled variables (grove dimensions: area and area/perimeter, land cover: percentages of groves, fields, meadows, urban, etc., and logarithm of the distance to road, *cf. Appendix 2*) and "connection" as explicative. For analysis of each subsite all replicates were pooled in the tests to increase the degrees of freedom. Though his method may introduce a bias into our analysis we assume that this bias will be constant across all subsites because we have exactly five replicates of each.

Because Grove-R are connected to roadside habitats, the presence of ruderal species could increase artificially the diversity index (*S** and *E*). To control for this possible artefact, we tested the difference of $Q*_{FOR}$ values between the two kinds of groves.

f. Roadsides as corridors for forest plant species

To test if roadsides function as corridors for forest plant species, we performed a linear regression between $Q*_{FOR}$ values of Road-F sites and distances to groves (Grove-R and Grove-F) according to the method given by Christen and Matlack (2006). We tested for slope equality with an ANOVA between $Q*_{FOR}$ values and the distances to connected wood, *Dc*, and the distances to unconnected wood, *Dnc* ($Q*_{FO\ R} \sim Dc + Dnc + Dc{:}Dnc$), looking at the interaction effect

All statistical analyses were performed using R (Ihaka and Gentleman, 1996).

III. RESULTS

1. Roadsides as habitat for plants

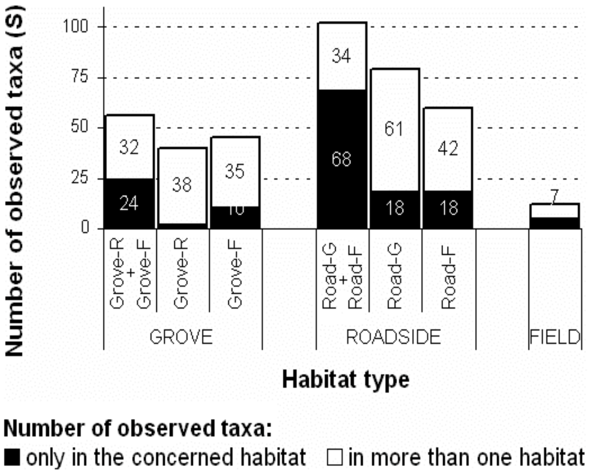
a. Species composition

Among the 250 vegetation plots, we detected 128 taxa (*Appendix 3*) i.e. 25.7 % of the total flora recorded within the study area by the National Botanical Conservatory of the Paris region (data downloaded from http://cbnbp.mnhn.fr/cbnbp/observatoire/coll TerrForm.jsp). We identified 103 taxa at the species level (80.5%) and 23 at the genus level only (18.0%). We also found moss and ferns that we did not identify more precisely (except *Equisetum arvense*). We detected no invasive species and only one non-endemic (*Veronica persica*) in our sampling sites.

Roadsides hosted the largest number of species, with 102 taxa detected among the 100 inventoried plots (79.7% of the observed diversity), while 68 of these taxa were exclusively found in this habitat. We detected 56 taxa in groves (100 plots, 43.8% of the diversity) and 24 of these exclusively in groves. In crop fields (50 plots), we detected 12 taxa (9.4% of the diversity) with five exclusive ones (*Figure 2*).

For each site and habitat, we compared estimated species richness (calculated using the software ComDyn) between habitats.

FIGURE 2:
HABITATS RICHNESS



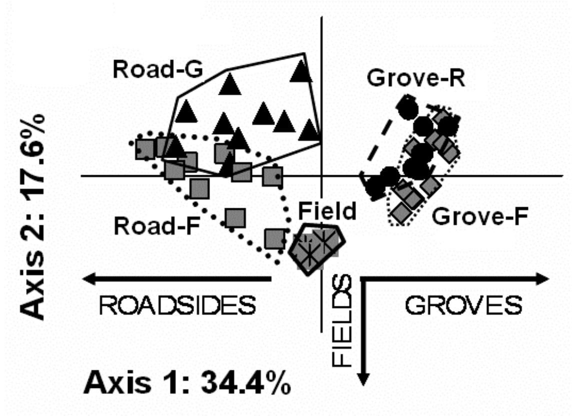
Species richness was recorded in each habitat. The highest species richness was found in roadside habitats (both roadside adjacent to grove (Road-G) and to field (Road-F) are richer than groves and field): 102 taxa (68 exclusive) representing 53.1% of the total diversity. Crop field was the most species poor habitat with only 12 taxa detected (5 exclusive). Grove habitats contained 56 taxa (24 exclusive). Isolated groves (Grove-F) contained 45 taxa (10 exclusive) and were richer than connected ones (Grove-R) (40 taxa, 2 exclusive).

Roadsides connected to groves ($S*_{Road\text{-}G}$ = 29.8 ± 9.85) have a higher species richness than roadsides adjacent to agricultural fields ($S*_{Road\text{-}F}$ = 20.9 ± 8.02, t_9 = 2.80, P = 0.021) and groves adjacent to roads ($S*_{Grove\text{-}R}$ = 20.5 ± 14.9, t_9 = 2.69, P = 0.025). Differences just failed to reach significance level with groves adjacent to agricultural fields ($S*_{Grove\text{-}F}$ = 20.3 ± 13.4, t_9 = 2.18, P = 0.057).

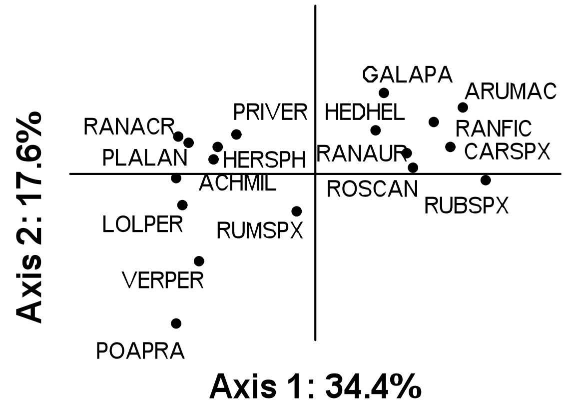
The floristic composition differed between these three habitats, as shown by the PCA (*Figure 3*) performed on the list of taxa detected within subsites. Axis 1 explained 34.4% of variance and discriminated between grove subsites and roadside subsites. Axis 2 explained 17.6% of variance and discriminated between groves and crop fields. Axis 1 identified grassland species found in roadsides subsites (e.g. *Achillea millefolium*, *Lolium perenne*, *Plantago lanceolata* or negative values of Axis 1) and forest species

FIGURE 3:
PRINCIPAL COMPONENT ANALYSIS ON SUBSITES FLORISTIC COMPOSITION

A. Subsites projection on Axis 1 and 2



B. Species projection on Axis 1 and 2



Their species composition shows that the five habitats have a different floristic composition. Axis 1 (34.4%) discriminates roadsides from crop fields and groves. Axis 2 (17.6%) separates subsites with their adjacent habitat: roadside, crop field or grove.

Ranunculus acris which are positioned on the found in groves (e.g. *Arum maculatum*, *edera helix*,*Ranunculus auricomus* or *Rosa canina* which can be observed on the right of the PCA: positive values). Axis 2 discriminated plants adapted to crop field habitats (e.g. *Poa pratensis* or *Veronica persica* are situated on negative values of Axis 2) and others (e.g. *Arum maculatum*, *Hedera helix*, *Primula veris* or *Ranonculus Acris* situated on the top of the PCA) which are totally excluded from agricultural fields and only present in roadsides or groves.

b. Diversity of communities

Comparing the diversity index *E* between habitats, roadsides adjacent to groves ($E_{Road\text{-}G} = 0.697 \pm 0.052$) constituted the most diverse habitats and agricultural fields ($E_{Field} = 0.194 \pm 0.185$) the least diverse. Roadsides adjacent to crop fields ($E_{Road\text{-}F} = 0.562 \pm 0.080$) were richer than groves and crop field subsites. There was no significant difference between the two types of groves ($E_{Grove\text{-}R} = 0.511 \pm 0.102$ and $E_{GF} = 0.496 \pm 0.160$) which were richer than crop field subsites ($F_{1,223} = 66.33$, $P = 2.10^{-16}$).

2. Road and roadside effects on plant communities

a. The effect of the distance to the next road

Among groves isolated within crop fields, the richest and most diverse flora is found in groves that are the largest distant from roads since a strong positive correlation was found between both the observed richness ($r_S = 0.523$, $F_{1,48} = 19.48$, $P = 5.738 \times 10^{-5}$) and the diversity index ($r_E = 0.548$, $F_{1,48} = 20.61$, $P = 3.790 \times 10^{-5}$) with the log-transformed distance to the nearest road (*Figures 4A and 4B*). The floristic richness and diversity of groves are correlated with the distance to the nearest road according to control variables (grove dimensions or landcover).

b. The effect of connexion to roadside on plant communities

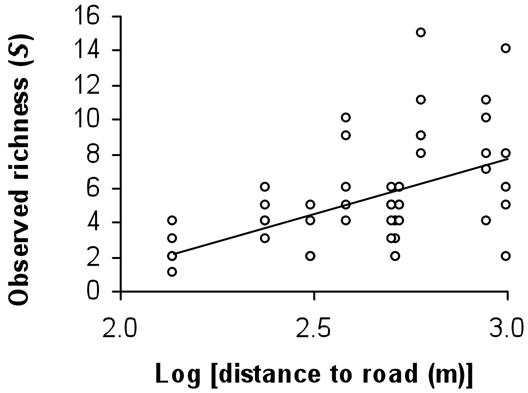
The previous linear trend was not maintained if groves adjacent to roads (Grove-R) were added because their richness and diversity were as high as in the most distant isolated groves.

ANOVAs performed on *S* and *E* showed that the connection of one grove to a roadside enhanced its floristic quality: for the richness (*S*: $F_{1,91} = 12.34$, $P = 7 \times 10^{-4}$) and for the community diversity (*E*: $F_{1,91} = 13.31$, $P = 4 \times 10^{-4}$).

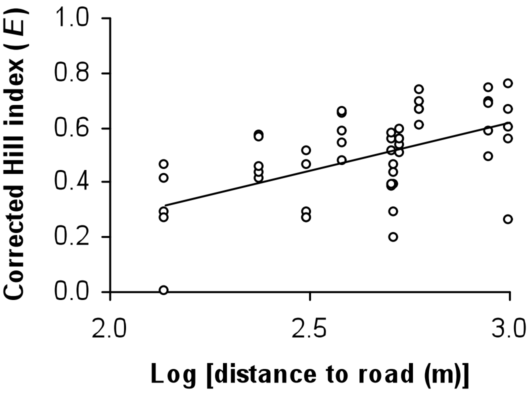
No significant differences of Q_F^* values were observed between the two types of groves ($Q_F^*{}_{\text{-}Grove\text{-}R} = 0.706 \pm 0.037$ & $Q_F^*{}_{\text{-}Grove\text{-}F} = 0.652 \pm 0.041$, $t = -0.321$ & $P = 0.749$).

FIGURE 4:
DIVERSITY INDICES VS. LOG [DIST TO ROAD (M)]

A. Observed species richness



B. Corrected Hill index



Diversity indices (S and E) in isolated groves (Grove-F) are strongly correlated with shortest distance from the grove to road. Observed richness (S: $r = 0.523$, $F_{1,48} = 19.48$, $P = 5.738 \times 10^{-5}$ and $f(x) = 6.39x - 11.43$) and corrected Hill Index (E: $r = 0.548$, $F_{1,48} = 20.61$, $P = 3.790 \times 10^{-5}$ and $f(x) = 0.34x - 0.43$) are positively correlated to this distance.

---

c. Effect of roads on plant communities:
Using *F-tests* we found significant differencesin variance of diversity indices for the two types of grove (isolated and connected to a roadside), specifically for species richness ($\sigma^2_{\text{Grove-R}} = 5.16$ and $\sigma^2_{\text{Grove-F}} = 5.48$, $F_{1,50} = 0.368$, $P = 3 \times 10^{-4}$) and corrected Hill index ($\sigma^2_{\text{Grove-R}} = 0.595$ and $\sigma^2_{\text{Grove-F}} = 0.608$, $F_{1,50} = 0.406$, $P = 0.001$) between Grove-F and Grove-R. Moreover, variances in species richness and corrected Hill index were higher in isolated groves (Grove-F) than in groves connected to roadsides (Grove-R).

Comparing similarity indices between paired sites, we found that connected groves were more similar to each other ($\varphi_{Grove-R} = 0.711 \pm 0.233$, φ is the similarity index) than they were to isolated ones ($\varphi_{Grove-F} = 0.641 \pm 0.250$): $t_{90} = 1.662$ and $P = 0.00680$.
These results could ensure from the heterogeneous distribution of some *rare* species (specialists) which are only present in isolated groves (e.g. *Convallaria majalis*, *Listeria ovata*, *Paris quadrifolia*, etc.) and to corridor effects between connected groves allowing dispersion of other (generalist) species.

3. Roadsides as corridors

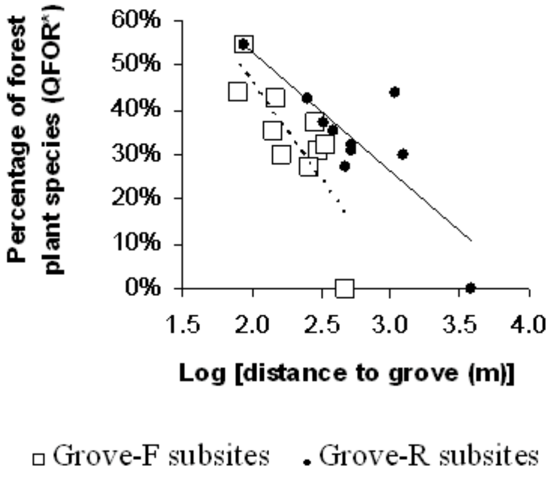
The latest published French forest flora (Rameau 2005) includes 56 of the 103 plants species detected in our inventories (i.e. 54.4 %). We ran ComDyn to estimate the proportion of forest species (Q^*_{FOR}) for each Road-F subsite. Performing linear regressions, we found significant negative correlations between (Q^*_{FOR}) in roadsides adjacent to agricultural fields (Road-F) and the distance to the nearest connected (Grove-R: $R^2_{\text{Road-F}\rightarrow\text{Grove-R}} = 0.581$, $P = 0.006$) and isolated (GF: $R^2_{\text{Road-F}\rightarrow\text{Grove-F}} = 0.648$, $P = 0.003$) groves (*Figure 5*). Slopes of these two regression models differed significantly ($F_{1,6} = 7.055$ and $P = 0.0377$). This seems to show a better dispersion of forest species to roadsides from connected groves than from unconnected ones ($\text{slope}_{(\text{Road-F}\rightarrow\text{Grove-R})} = -0.268 \pm 0.064 > \text{slope}_{(\text{Road-F}\rightarrow\text{Grove-F})} = -0.435 \pm 0.119$).

IV. DISCUSSION

1. Roadsides as habitat for biodiversity

In this study, we assessed the floristic diversity in an intensive agrarian landscape. We sampled subsites in the three most important habitats of the study zone: groves, roadsides and crop fields (representing more

FIGURE 5:
REGRESSION OF PERCENTAGE OF FOREST SPECIES AND THE DISTANCE FROM ROADSIDE TO CONNECTED AND UNCONNECTED WOODS



Significant negative correlations between the percentage of forest plant species in roadsides adjacent to crop fields (Road-F) and the distance to the nearest connected (Grove-R: $r^2_{Road\text{-}F \rightarrow Grove\text{-}R} = 0.581$, $P = 0.006$) and isolated (Grove-F: $R^2_{Road\text{-}F \rightarrow Grove\text{-}F} = 0.648$, $P = 0.003$) groves.

than 95 % of the landcover in the study area).

We found that plant communities of roadside verges were at least as rich and diverse as those in the other habitats, though community compositions differed. The majority of plant species detected in roadsides (68 for a total of 128 detected taxas, 53.2 %) were not observed in groves and crop fields in the surrounding area. Thus, roadsides can be considered as refuges zones for many plant species in such intensive agricultural landscape where grasslands do not exist. Using estimated species richness, we supposed that more than 25% of all plant species inventoried in the area by the National Botanical Conservatory of the Paris region should be found in our 20 roadside subsites (i.e. an area representing 0.02% of the study area or 0.1% of local roadside verges). This proportion is very similar to results of Way (1977) and Sykora (1993) who found high plant diversity on roadside verges in U.K. and in the Netherlands. Furthermore, some portions of the roadsides, when adjacent to rich habitats (groves) can host a high floristic diversity.

It is perhaps surprising that we detected no invasive species in our sampling plots. This may be due to the placement of our sampled plots in the roadside bank which is the less disturbed zone of roadside verges. Further, Ulman et al. (1995) showed that the distribution of exotic plant species is spatially heterogeneous.

2. Impact of roads on plant communities

The presence of roads in a landscape appeared to have a negative impact on plant communities of surrounding areas, since we observed that species richness and diversity of groves increased with distance to roads. Causes of this negative trend could be either direct or indirect.

Direct causes could derive from pollution by vehicles with high volumes of car and truck traffic in our study area (average 420 vehicles?h-1 during the evening rushes). Another possible direct cause could lie in pressure of urban walkers (stamping and harvesting) that visit groves for recreational activities (the study area is approximately 60 km from the center of Paris, containing 11 million people). Their visitation rate is likely higher in groves situated closer to roads than in the groves completely isolated in fields. All those factors are known to lead to an impoverishment of plant communities (Forman and Alexander, 1998; Trombulak and Frissell, 2000).

Indirect causes could be the loss of many animal populations essential to plant life (pollinators, seed dispersers, and predators of phytophageous insects) through pollution or impacts with automobiles (Forman and Alexander 1998). Road casualties may have a strongly negative impact on wildlife in our study area because of the high traffic volume. Predators are known to be more affected by pollution (Morgan et al., 1983) and their local extirpation can lead to explosions of herbivore populations in roadside verges (Flückiger et al., 1978) and in adjacent habitats and thus affect plant communities.

3. Roadsides as corridors for plant species

Our results suggest a road corridor effect for plant species:

(1) Because forest plant species are found in roadsides for hundreds of meters with proportions negatively correlated within distances to groves, and because the slope of the regression with distance to isolated groves (Grove-F) is stronger than the one with the distance to adjacent groves (Grove-R), we assume that there is a facilitation of forest plant dispersal by roadsides. It was indeed probably the case for species such as the forest plants *Galium aparine*, *Ranunculus auricomus*, *Stellaria holostea* and *Veronica hederifolia* that have been found migrating the furthest along verges and found in roadsides adjacent to field (Road-F) subsites. This corridor effect probably depends on dispersal abilities of species (Noss, 1987; Beier and Noss, 1998) and could thus be different according to species considered. However, it is likely stronger for species such as *Heracleum sphondylium, Lamium purpureum* and *Stellaria holostea* that were found in connected groves (Grove-R), roadsides adjacent to groves (Road-G) and roadsides adjacent to fields (Road-F) but not in isolated groves (Grove-F). Those species seem to be only able to disperse along roadsides but not through fields. Hence, species that are able to migrate through roadsides may be favoured in such intensive agricultural landscapes.

(2) Because groves adjacent to roadsides (Grove-R) host particular, rich and diverse communities of plants with no significant difference of forest plant proportions with isolated groves, we also assume that plants from roadsides disperse in groves and impact them. According to metapopulation theory (Levins, 1968), population viability is enhanced when populations are connected to each other, leading to better community stability. We hypothesize that for this reason, the plant communities in groves adjacent to roadsides have a higher floristic quality. In isolated groves, the turn-over of species by stochastic extinction/recolonization could be more rapid and lead to more diverse but more fragile assemblages.

We conclude thanks to points (1) and (2) that roadsides act as corridors for forest plant species. This corridor function had been previously hypothesized (Zink et al., 1995; Corbit, 1999; Tikka et al., 2001) but never conclusively established (Sherwood et al., 2002).

Actually, all "forest" species detected in roadsides are not "strictly forest specialists" but "semi-forest species". But we can consider them as sufficiently generalist to disperse at short scale (few hundred meters) into roadsides. A future study should be carried out to investigate the possible effect of roadside with edges as corridor for strictly forest plants.

4. Roadsides and biotic homogenisation

Groves adjacent to roadsides had indeed more similar plant communities than isolated ones. This suggests a negative impact of roads on plant communities (loss of species and diversity) that could be partly mitigated by their role as corridors. We indeed observed a high alpha diversity in connected groves due to roadsides acting as corridors. However, we also found low beta diversity among them. This relationship between alpha and beta diversity is in accordance with Loreau (2000) proving the importance to work at different scales to correctly assess biodiversity: even if each connected grove is richer than most of the unconnected ones, the diversity they globally harbour is low because they share similar species. Increase in taxonomic similarity is typically scale dependent (Olden and Poff, 2003).

Moreover, specialist species of surrounding habitats seem to be negatively affected by roads. In roadside verges, we found 40 of the 56 forest plant species detected in the study but many of them do not strictly depend on forest conditions. They usually tolerate shade but in roadsides they also tolerate higher light conditions. Those plants can be defined as more generalist than other species, strictly found in forest habitats which are considered specialist. In contrast, species that have narrower ecological requirements were essentially detected in isolated groves and were totally absent from connected ones (*Anemone nemorosa*, *Convallaria majalis*, *Paris quadrifolia* or *Veronica montana* for example). Hence, connections to roadsides seem to lead to a homogenisation of plant community compositions of surrounding habitats. Often, an increase of local richness can lead to a decrease of global diversity through the increase of similarity among communities composition. It is usually the result of generalists or introduced species progressions that are more likely to invade disturbed ecosystems (Hobbs and Mooney, 1998). This non-random reshuffling of species pools leads to biotic homogenisation (McKinney and Lockwood, 1999) that is considered as one of the major causes of biodiversity crisis due the increase of human activities and disturbances (Olden and Rooney, 2006). While largely described for other human activities (McKinney, 2006, Smart et al., 2006), such impact of roads had never been mentioned. Given the area covered by roads and their potential influence on biodiversity at a large scale, we consider that this effect should be more studied to verify if the corridor function of roads do really mitigate their negative effects.

V. CONCLUSIONS

Presence of roadside verges doubles the local plant richness (53.2% of taxa were only detected in roadside verges) in intensive agrarian landscapes. Life history and dispersal mechanisms of plant species play an essential role in determining the efficiency of roadside verges as habitat (Godfree, 2004) and as corridors between habitats (Primack and Miao, 1992). A better understanding of how plant life history characteristics (i.e. dispersal and reproductive traits) influence the efficiency of roads as corridors (Roy and de Blois, 2006) should help to determine how roadside management (mowing planning, hedgerow plantation, etc.) could maintain and improve their role. Anthropogenic disturbances have strong negative impacts on the plant populations of roadside verges (Spooner, 2004). Therefore roadside management should also integrate environmental objectives to maintain verges as viable habitats and corridors for plant species, if it can be proved that the corridor effect do not threaten too much biodiversity at a large scale. We assume that the best way to protect local forest diversity is to avoid road constructions when it is possible. Else, if the road has to be established for numerous reasons, we assume that it will be better to build it as far as possible from groves. If this not possible to have more than a distance of 500 m between road and grove, then it should be better to build it very close to grove to connect them to roadside verges in order to mitigate as maximum as possible road impacts favouring roadside corridor effects.

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SUPPLEMENTARY DATA

APPENDIX 1:
CORRECTED BRAUN-BLANQUET SCALE

Taxa/species recovery	Index	Median recovery value
Less than 5%	1	2.50%
5 to 10%	2	7.50%
10 to 25%	3	17.50%
25 to 50%	4	37.50%
More than 50%	5	75.00%

APPENDIX 2:
SITES SAMPLED

Subsites	Aera	Area / Perimeter	Distance to road	Landcover *				
	(m²)	(m)	(m)	(A)	(B)	(C)	(D)	(E)
GR01	29728	38.8	0	7.9%	90.5%	0.0%	1.3%	0.3%
GR02	19620	28.1	0	5.8%	91.5%	0.0%	0.0%	2.7%
GR03	37268	28.6	0	6.5%	93.5%	0.0%	0.0%	0.0%
GR04	15289	28.9	0	9.7%	88.8%	0.0%	0.9%	0.5%
GR05	10686	22	0	3.4%	87.1%	0.0%	0.0%	9.4%
GR06	17174	28.8	0	3.7%	90.0%	0.0%	0.0%	6.4%
GR07	17933	26.4	0	5.0%	91.2%	0.2%	0.0%	3.6%
GR08	29221	43.2	0	9.3%	90.7%	0.0%	0.0%	0.0%
GR09	10958	22.8	0	4.3%	88.5%	0.0%	0.0%	7.3%
GR10	26192	23.9	0	5.3%	88.2%	0.0%	0.4%	6.2%
GF01	18123	25.4	385	4.2%	94.6%	0.0%	0.0%	1.2%
GF02	24631	24	600	3.6%	96.4%	0.0%	0.0%	0.0%
GF03	30163	39.7	517	6.5%	89.8%	0.0%	0.0%	3.7%
GF04	18925	31.7	137	2.9%	95.6%	0.0%	0.0%	1.5%
GF05	17841	31.4	888	8.6%	87.9%	1.4%	1.5%	0.6%
GF06	19840	33.2	311	2.6%	95.3%	0.0%	0.0%	2.1%
GF07	16735	30.7	238	5.7%	91.2%	0.0%	0.8%	2.3%
GF08	10435	24.5	531	7.0%	91.5%	0.0%	0.0%	1.4%
GF09	20201	29.7	999	3.3%	96.4%	0.0%	0.0%	0.3%
GF10	21070	22.8	507	6.3%	87.8%	0.0%	0.5%	5.4%

** Landcover: (A) Woods, (B) Fields, (C) Water, (D) Meadows and (E) Others (urban, road, etc.)*

APPENDIX 3:

TAXA IDENTIFIED

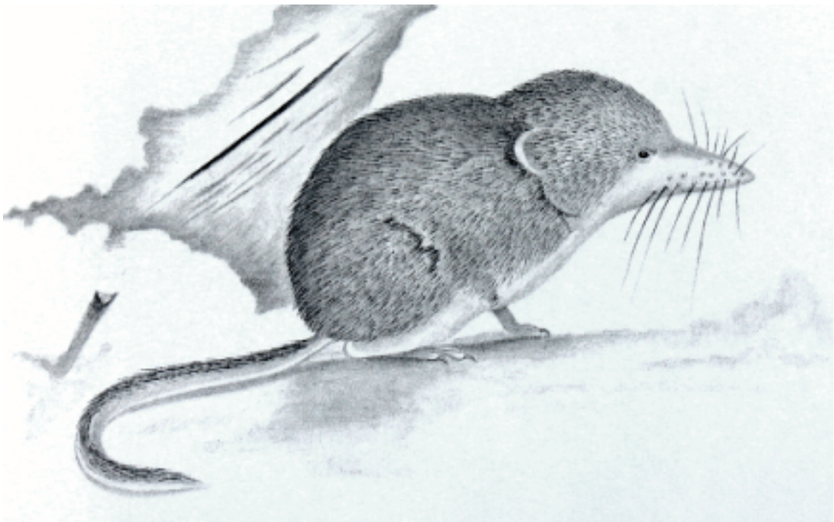
Taxas	Code
Acer campestre L.•	ACECAM
Adoxa moschatellina L.•	ADOMOS
Agrimonia eupatoria L.•	AGREUP
Agrostis sp.•	AGRSPX
Ajuga reptans L.	AJUREP
Alium sp.	ALISPX
Alopecurus pratensis L.	ALOPRA
Anemone nemorosa L.•	ANENEM
Anthriscus cerefolium Hoffm.	ANTCER
Anthriscus sylvestris (L.) Hoffm.•	ANTSYL
Arthemisia sp.	ARTSPX
Arum maculatum L.•	ARUMAC
Bellis perennis L.	BELPER
Brassica napus L.	BRANAP
Brassica rapa L.	BRARAP
Bromus arvensis L.	BROARV
Bromus erectus Huds.•	BROERE
Cardamine pratensis L.•	CARPRA
Carex sp.	CARSPX
Centaurea jacea L.	CENJAC
Centaurea nigra L.•	CENNIG
Cerastium fontanum Baumg.	CERFON
Cirsium sp.	CIRSPX
Clematis vitalba L.•	CLEVIT
Conium maculatum L.	CONMAC
Convallaria majalis L.•	CONMAJ
Convolvulus arvensis L.	CONARV
Crataegus laevigata DC.•	CRALAE
Crataegus monogyna Jacq.•	CRAMON
Crataegus sp.	CRASPX
Crepis sp.	CRESPX
Cruciata laevipes Opiz•	CRULAE
Dactylis glomerata L.•	DACGLO
Daucus carota L.	DAUCAR
Equisetum arvense L.•	EQUARV
Fagus sylvatica L.•	FAGSYL
Festuca arundinacea Schreb.	FESARU
Fragaria vesca L.•	FRAVES
Galium aparine L.•	GALAPA
Galium mollugo L.•	GALMOL
Galium verum L.•	GALVER
Geranium dissectum L.	GERDIS
Geranium molle L.	GERMOL
Geranium pusillum L.	GERPUS
Geranium robertianum L.•	GERROB
Geranium rotundifolium L.	GERROT
Geum urbanum L.•	GEUURB
Glechoma hederacea L.•	GLEHED
Hedera helix L.•	HEDHEL
Heracleum sphondylium L.•	HERSPH
Himantoglossum hircinum Spreng.	HIMHIR
Hypericum perforatum L.•	HYPPER
Lamium album L.	LAMALB
Lamium purpureum L.	LAMPUR
Leondonton sp.	LEOSPX
Leucanthemum vulgare Lam.•	LEUVUL
Ligustrum vulgare L.	LIGVUL
Listera ovata (L.) R.Br.•	LISOVA
Lolium perenne L.	LOLPER
Lonicera periclymenum L.•	LONPER
Lotus corniculatus L.	LOTCOR
Luzula campestris (L.) DC.•	LUZCAM
Matricaria sp.	MATSPX
Medicago lupulina L.	MEDLUP
Medicago minima (L.) L.	MEDMIN
Orchis mascula L.•	ORCMAS
Paris quadrifolia L.•	PARQUA
Picris echioides L.	PICECH
Picris hieracioides L.	PICHIE
Picris sp.	PICSPX
Plantago lanceolata L.	PLALAN
Plantago major L.	PLAMAJ
Plantago media L.	PLAMED
Platanthera bifolia (L.) Rich.	PLABIF
Poa annua L.	POAANN
Poa pratensis L.	POAPRA
Poa sp.	POASPX
Poa trivialis L.•	POATRI
Polygonatum multiflorum All.•	POLMUL
Populus sp.	POPSPX
Potentilla anserina L.	POTANS
Potentilla reptans L.•	POTREP
Potentilla sterilis Garcke•	POTSTE
Primula elatior Hill•	PRIELA
Primula veris L.•	PRIVER
Prunus sp.	PRUSPX
Quercus sp.	QUESPX
Ranunculus acris L.	RANACR
Ranunculus arvensis L.	RANARV
Ranunculus auricomus L.•	RANAUR
Ranunculus ficaria L.•	RANFIC
Ranunculus repens L.•	RANREP
Ranunculus sp.	RANSPX
Ribes rubrum L.•	RIBRUB
Rosa canina L.•	ROSCAN
Rubus sp.	RUBSPX
Rumex obtusifolius L.•	RUMOBT
Rumex sp.	RUMSPX

Sambucus nigra L.•	SAMNIG
Senecio jacobaea L.	SENJAC
Sonchus arvensis L.	SONARV
Sonchus oleraceus L.	SONOLE
Stachys sylvatica L.•	STASYL
Stellaria holostea L.•	STEHOL
Tamus communis L.•	TAMCOM
Taraxacum sp.	TARSPX
Teucrium scorodonia L.•	TEUSCO
Trifolium campestre Schreb.•	TRICAM
Trifolium dubium Sibth.	TRIDUB
Trifolium pratense L.	TRIPRA
Trifolium repens L.	TRIREP
Trifolium sp.	TRISPX
Urtica sp.	URTSPX
Veronica agrestis L.	VERAGR
Veronica arvensis L.	VERARV
Veronica hederifolia L.	VERHED
Veronica montana L.•	VERMON
Veronica officinalis L.•	VEROFF
Veronica persica Poir.*	VERPER
Veronica sp.	VERSPX
Vicia sativa L.	VICSAT
Vicia sp.	VICSPX
Vinca minor L.•	VINMIN
Viola reichenbachiana Jord. ex Bor.	VIOREI
Viola riviniana Rchb.•	VIORIV
Viola sp.	VIOSPX
Ferns	FERN
Moss	MOSS

Classification is according to the International Plant Names Index (http://www.ipni.org)

• Presence in the French forest flora (Rameau, 2000).

** Non-native plant species according to the Conservatoire botanique national du Bassin parisien (French flora protection Agency for Paris region), database: http://cbnbp.mnhn.fr.*



Sorex minutus

MANUSCRIPT N°2

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ROAD NETWORK IN INTENSIVE AGRICULTURAL LANDSCAPE: HABITAT, CORRIDOR OR BARRIER FOR TWO SMALL MAMMALS SPECIES?

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PRESENTATION

TITRE

Infrastructures routières en paysage agricole: Habitat, corridor ou barrière pour deux espèces de petits mammifères ?

RESUME

Nous avons étudié la répartition et l'abondance de petits mammifères dans une zone d'agriculture intensive en Ile-de-France. 176 sites on été échantillonnés dans différents milieux : zones marginales (bords d'autoroute, bords de route et bordures de champ), zones naturelles (bois) et zones de culture (champs). Des pot-pièges de type Barber (n = 864) ont été posés pour capturer des campagnols, mulots et musaraignes (418 individus identifiés).

Nous avons trouvé les abondances les plus élevés dans les bords de route et d'autoroute et avons ciblé notre étude sur deux espèces : *Microtus arvalis* (rongeur) et *Sorex coronatus* (insectivore). Nous avons montré que les campagnols et les musaraignes ont des patterns de répartition et des dynamiques très différents : les deux étaient très abondants dans les différents accotements mais les abondances de *Microtus arvalis* dans les champs étaient beaucoup plus importantes que celles de *Sorex coronatus* dont les quantités étaient essentiellement résiduelles.

Nous avons aussi montré que les dynamiques des musaraignes au sein des accotements étaient sensibles à la taille des emprises : les accotements les plus larges, *i.e.* ceux des bords d'autoroute, peuvent constituer un habitat de qualité pour *Sorex coronatus* alors que ceux plus étroits, *i.e.* ceux des bords de route, peuvent servir uniquement pour leur dispersion entre différents milieux naturels adjacents. Nos résultats ont aussi montré l'effet « barrière » de la route qui ne semble pas ou peu traversée par *Sorex coronatus*.

L'ensemble des résultats de cette étude montrent, dans des paysages d'agriculture intensive, que les dépendances vertes des réseaux routiers et autoroutiers peuvent être considérées comme des zones refuges, des habitats ou des corridors pour les petits mammifères en fonction des espèces considérées et des caractéristiques des emprises routières.

MOTS CLEFS

Bords de route, Campagnols, Dispersion, *Microtus arvalis*, Musaraignes, *Sorex coronatus*, Zones marginales.

ROAD NETWORK IN INTENSIVE AGRICULTURAL LANDSCAPE: HABITAT, CORRIDOR OR BARRIER FOR SMALL MAMMALS?

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KEYWORDS

Dispersal
Marginal areas
Mice
Microtus arvalis
Roadside
Road management
Shrew
Sorex coronatus
Vole

ABSTRACT

Road network have negative impacts on environment but their verges can act as a refuge and/or as ecological corridors in anthropogenic ecosystems. We studied the distribution and abundance of small mammals in intensive agrarian landscape in central France. 176 sites were sampled in different habitats: marginal zones (highway verges, roadside and field margin), natural areas (woods) and fields (at different distances from margins) using non-attractive pitfall traps (n= 864) to capture shrews, voles and mice (418 small mammals captured). We found higher abundances of small mammals in highway and road verges, and further focused on the two most-captured species: Microtus arvalis (rodent) and Sorex coronatus (insectivorous). We showed that voles and shrews had different dynamics and distribution patterns: both were very abundant in verges and margins and Microtus arvalis was more abundant in crop fields, a habitat where numbers of Sorex coronatus were residual and correlated to the distance to closest natural habitats. We showed that shrew dynamics in verges were function of margin widths: larger verges, constitute a habitat for Sorex coronatus while narrower verges serve only as corridor for their dispersion between natural connected habitats. Taken together, these results show that in intensive agricultural landscapes, roadside and highway verges serve as refuge, habitat or corridor for small mammals depending on species and margin characteristics.

I. INTRODUCTION

Road networks have expanded with human population over large areas (Watts et *al.* 2007). In France, main and secondary road networks are approximately 1,000,000 km long (i.e. 1.82 km of roads per km^2, data from 2005) and their verges cover approximately 5,100 km^2, *i.e.* nearly 1% of the country area. They are known to influence landscape structure and have to impact ecosystems dynamics through habitat destruction, alteration and fragmentation (Forman and Alexander 1998; Trombulak and Frissel, 2000).

However, the potential biological value of road verges has also long been recognised (Way 1977) with eventual contributions to conservation of indigenous flora (Spooner et *al.* 2004, O'Farrell and Milton 2006) and fauna (Meunier et *al.* 1999, Bellamy et *al.* 2000, Ries et *al.* 2001). In human dominated areas such as intensive agrarian landscapes, where non-agricultural habitats (e.g. edges) are critical to conservation of biological diversity and

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ecological processes (Burel 1996), road verges can play a crucial role as a refuge and/or as ecological corridors (Dawson 1994, 2002, Tikka et *al.* 2001). Road verges may present few interests in large natural areas but may be crucial in intensive agrarian landscapes where they represent the last semi-natural habitats (Davies and Pullin 2007). As linear areas of semi-natural vegetation, road verges may provide habitat and thus create refuges in hostile matrix and corridors for animals (Hansen & Jensen 1972, Merriam et *al.* 1990, Bennett 1990, Hodkinson & Thompson 1997).

Actually, small-mammals are supposed to be major natural factors of agrarian ecosystems because they regulate invertebrate populations in crop fields and grasslands (Churchfield et *al.* 1991). Shrews are well known as voracious predators (Churchfield and Brown 1987) and even small rodents predate some invertebrates (Parmenter and MacMahon 1988): for examples *Sorex araneus* populations can kill more than 100,000 moths per ha and per month (Buckner 1969) and *Apodemus sylvaticus* can destroy up to 50% of the overwintering cocoons of *Diprion pini* (Obrtel et *al.* 1978), a hymenoptera causing important damages to European pines forests (Pasquier-Barre 1999). Maintaining invertebrate predators is crucial for agricultural production (Schoener 1988, Spiller & Schoener 1990, Dial & Roughgarden 1995.) because it means low pest abundances (Maisonneuve & Rioux 2001) and thus lower inputs of pesticides. Small mammals are thus important for biological equilibriums in agrarian landscape.

Small mammals have been observed to spread tens of kilometers along highways and roads verges (Getz et al. 1978). Roadsides are thus considered as effective corridors for mammal species (Bennett 1990, Suckling 1984, Verkaar 1990). On the contrary, roads could act as significant barriers to dispersal for many animals (Rico et *al.* 2007), particularly in the case of high traffic rates (Harris & Silva-Lopez 1992). Roadside width certainly plays probably an important role in determining the role of edges as habitat, corridor or barrier, which we aim to study to promote biodiversity-friendly construction and management practices, in order to optimise the role of road verges as a refuge for wild fauna.

The aim of our study was therefore to assess the role of road verges as habitat, corridor and barrier for small mammal species in an agririan landscape. The study region is almost exclusively composed of crop fields. A few woods constitute the major natural areas. We sampled small mammals in highway and roadside verges, which are relative important marginal areas because they are remaining as quasi-unique open space habitats in the landscape. To obtain informations about the relative abundance of small mammals in verges compared to other habitats, we launch trapping with the same methodology in woods, crop fields and field margins.

Given the species we trapped, we decided to focus our study on two particular species: *Microtus arvali*s (vole, *i.e.* a rodent) and *Sorex coronatus* (shrew, *i.e.* insectivorous). We further inferred the ecological function of verges for those two species from their capture rates within and between habitats. By comparing small mammal species trapped in marginal areas (highway verges, road verges and field margins) to those trapped in adjacent habitats (fields and woods), we asked if those areas can be considered as habitats, corridors or/and barriers, and finally interpret these results in terms of roadside management.

II. MATERIALS AND METHODS

Using the following design we evaluated three hypotheses:

(1) By comparing plant species in roadsides with those in adjacent habitats, we asked if roadside habitats harbour native plant species which are not found elsewhere in the agricultural landscape.

(2) We hypothesized that native plant diversity would be higher in groves adjacent to roads when compared with isolated groves thanks to corridor effects of roadsides.

(3) To evaluate the roles of roadside habitats as corridors we also assessed the percentage of forest species in roadside verges as a

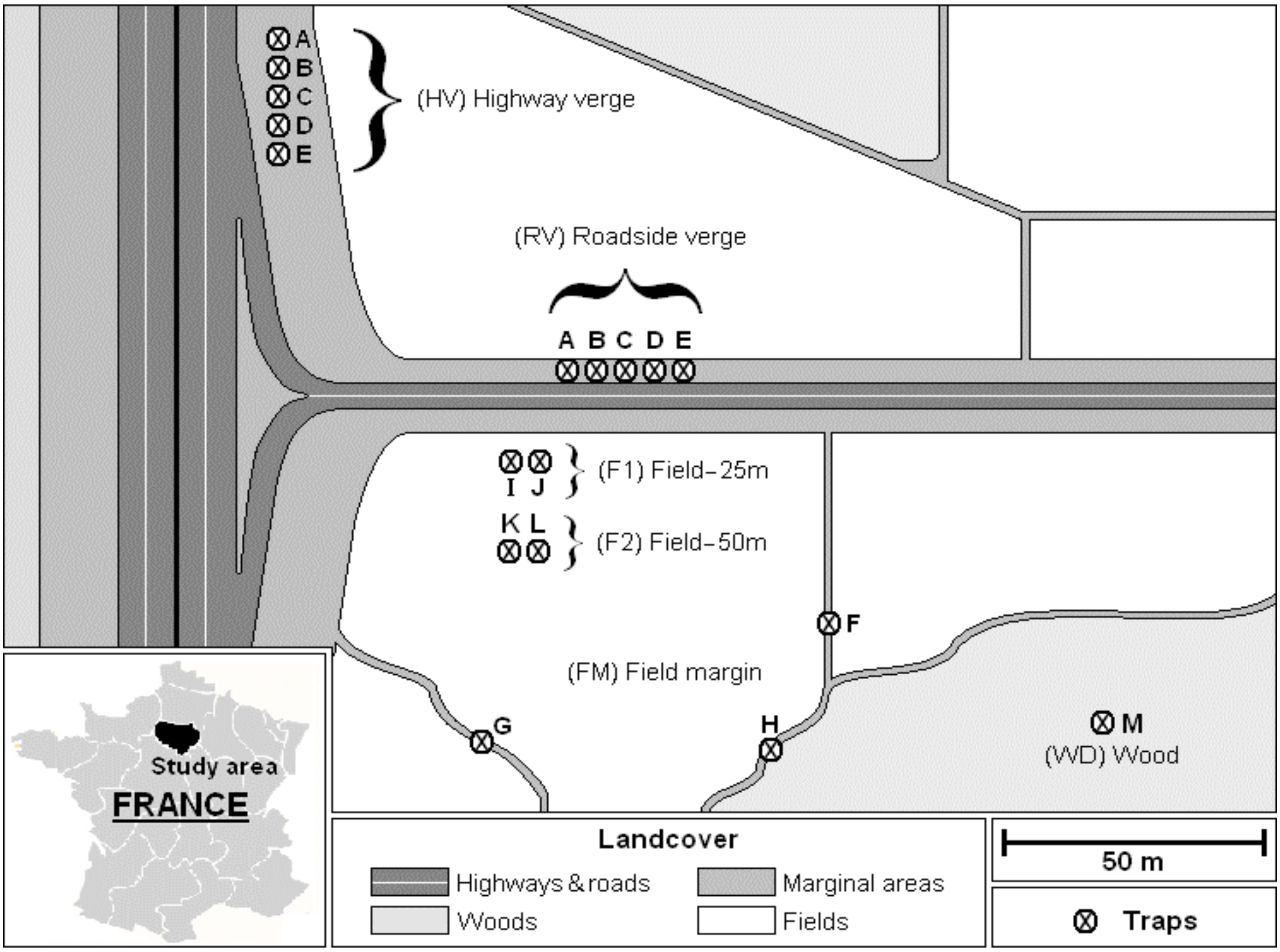
function of the distance to the nearest isolated or adjacent grove, and compared the linear trends obtained.

1. Study area (*Figure 1*)

The study area is in the Ile-de-France region located in the centre of France around Paris (48° 51′ North - 2° 21′ East). The study site is the intensive agricultural zone of the suburb of Paris. It is thus mainly composed of agricultural fields used for intensive crop: wheat, sugar beet, rapeseed (49.0%).

Urban areas represent 18.2% and the main "natural" habitat is woodland (representing about 23.7% of landcover; *Appendix 1*). Road network in the study area is 38,906 km long, *i.e.* 3,23 km of highways and roads per km^2. Verges of this network are narrow strips but represent 1.6% of the total area of the region (highway verges: 0.4%, road verges: 1.2%). In the same way, field margins (margins comprised between two fields) also represent 1.5% of the territory (estimations obtained using *GIS* data provided by the Institute for Urban Planning and Developpement of the Paris Ile-de-France Region, *I.A.U.R.I.F*). Road network verges (highway or roadside) and field margins are herbaceous strips mowed once or twice a year and differ mainly in width (15.7 ± 6.6 m [10.0 - 21.5] for highway verges, *HV*, 7.2 ± 3.4 m [4.3 - 14.6] for road verges, *RV*, and 4.0 ± 0.4 m for [2.0 – 10.0] for field margin, *FM*).

FIGURE 1:
SAMPLING DESIGN



The study area is located in central France in the Ile-de-France region (Paris area). Non-attractive traps (Barber pitfall traps) were installed in 297 sites of different habitats (marginal areas, woods and fields) to capture small mammals during one month in May 2006.

2. Sampling design (*Figure 1*)

To assess the potential habitat or corridor roles of road verges, we sampled the two main types of road network verges: 31 sites along highways (noted *HV*) and 48 sites along ordinary national roads (noted *RV*). On a site, a sampling unit is composed of five traps, linearly placed every 20m along highways and roads in the middle of banks. Each trap was further referenced with a letter according to its position in the linear design (A to E); given the small mammal movements, the five traps were not considered as independent sampling replicates in further analyses.

To measure the relative importance of verges for small mammals in the landscape, we also sampled field margins (65 sites, noted *FM*, with three traps along each field, and a distance between traps of at least 100 meters); field borders (65 sites, noted *F1*, with two traps placed 25m into the field), central fields (56 sites, noted *F2*, with two traps placed 50m into the field) and woods (32 sites, noted *WD*, with one trap randomly placed). A total of 864 traps were installed in May 2006 and collected one month later. Some of the traps were destroyed during the sampling period, so that a total of 813 traps were analysed from 297 sites (*Table 1*).

3. Small mammal captures and determinations

We used *Barber* pitfall traps, as 450 mL plastic pots (10.2 cm height & 7.5 cm diameter) inserted into the ground. Each trap contained a 155 mL solution composed with 75 mL of water, 75 mL of conservative (monopropylene glycol), 5 mL of surfactant (dishware soap) and 15 g of NaCl. Traps were protected from rain by plastic transparent covers (15 cm×15 cm) supported by wood sticks 10 cm above the ground. Sampling was carried out from 2 May to 4 June 2006. All captured mammals were dissected and identified from morphological cranial characters, such as teeth structures (Erome & Aulagnier 1982, Chaline et al. 1974).

4. Data analysis

a. Verges and margins as habitat

We tested the variation in the distribution of two main small mammals species across the sampled habitats (*HV*, *RV*, *FM*, *F1*, *F2* & *WD*), *Microtus arvalis* and *Sorex coronatus*, with an *ANOVA* on species abundances (average number of individuals by trap in each site)

b. Verges and margins as corridor

We ran *ANOVAs* to test two hypotheses possibly proving corridor function of verges and margins.

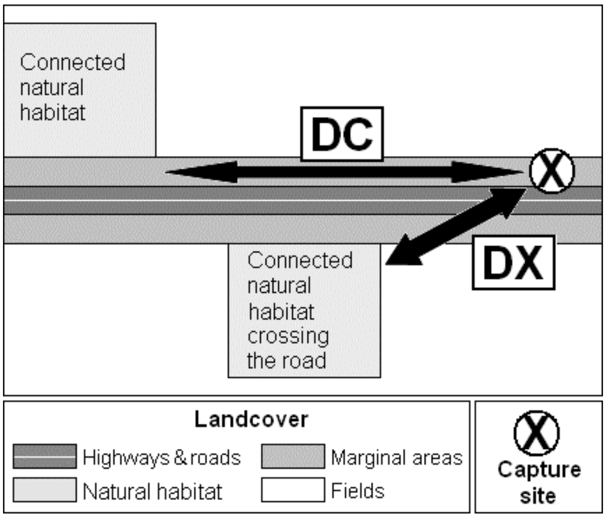
(1) If small mammals use verges as corridor, they preferentially fall in the two extreme traps (*A & E)*. We thus tested the trap position effect on the two species (for *HV* & *RV* sites).

(2) *S. coronatus* is a territorial species (Neet and Hausser 1990) and its populations densities are about 10.8 individuals per ha (Cantoni 1993), *i.e.* a territory can be approximately represented by a disk of 926m². Because of their territorial behaviour, if shrews move among their territories located in natural habitats (meadows or woods) through the verges, they should more often visit verge sites in the neighbourhood of natural sites connected to the verges. Thus, we calculated the distance between each verge site (*HV* & *RV* sites) and the nearest adjacent connected meadow or wood (distance noted *DC*, *Figure 2*), and further sorted verge sites in two categories: sites close to a natural habitat (*sites A*, *i.e.* closer than ten times a territory diameter – equivalent to 343m), and sites far from a natural habitat (sites *B*, *i.e.* further distant than ten times a territory diameter). We then ran *ANOVAs* to test differences of species abundance between the two categories of sites.

c. Highway and roadside verges as barrier for Sorex coronatus

If roads and highways act as barriers for *S. coronatus* and if the previous hypothesis is verified (*i.e.* significant effect of distance to nearest connected habitat, *DC*, on shrews abundances), we should not find any effect of distance to the nearest natural habitat located on the other side of the road on the abundance of *S. coronatus*. Thus we calculated the distance between each verge of *RV* and the next natural habitat of the other side of the road (distance noted *DX, Figure 2*). As previously, we sorted verges in two categories (*A*: <343m and *B*: >343m) and ran an *ANOVA* to test the difference in shrew abundances between sites of the two categories.

FIGURE 2:
DISTANCES CALCULATION



Distances between capture sites and nearest connected natural habitat (DC) or opposite nearest natural habitat (DX).

All statistical analyses were performed with the R software (Ihaka & Gentleman 1996) using *ANOVA*s and considering Poisson-distributed models.

III- RESULTS

A total of eight species were captured among the different habitats: three shrews (*Crocidura russula*, *Sorex coronatus* and *S. minutus*), three voles (*Clethrionomys glareolus*, *Microtus agrestis* and *Microtus arvalis*) and two mice (*Apodemus flavicollis* and *Apodemus sylvaticus*). Two of these species represented more than 78% of the collected specimen: a rodent, *M. arvalis* (141 trapped animals *i.e.* 34.6% of the total), and an insectivorous shrew, *S. coronatus* (178 trapped animals i.e. 43.6% of the total). Given the sampling sizes obtained, we decided to focus our analysis on these two species (*Table 1*).

1. Verges and margins as habitat

The *ANOVA* showed (Z = 1.058 and P = 0.290, *Figure 3A*) that *M. arvalis* is equally abundant in highway verges and roadside verges (*HV*: 0.24 ± 0.05 SE & *RV*: 0.30 ± 0.02 SE ind.trap^{-1}).

In field habitats, we found lower vole densities (*FM*: 0.10 ± 0.05 SE, *F1*: 008 ± 0.04 SE & *F2*: 0.07 ± 0.03 SE ind.trap^{-1}). Even if field margins are very similar habitats to verges, *M. arvalis* was less captured there than in highway verges (Z = 3.160 and P = 0.002) and roadside verges (Z = 4.255 and P < 10^{-3}). No differences were observed between field habitats (*FM vs. F1*: Z = -0.398 and P = 0.691; *FM vs. F2*: Z = -0.755 and P = 0.450; & *F1 vs. F2*: Z = -0.367 and P = 0.713). *M. arvalis* was not captured in woods (*WD*).

The distribution of *S. coronatus* (*Figure 3B*) varied significantly among habitats. This species was abundant in verges especially in highway verges (*HV*: 0.63 ± 0.08 SE) where *S. coronatus* was more captured than in roadside verges (Z = -5.444 and p < 10^{-3}, *RV*: 0.26 ± 0.05 SE ind.trap^{-1}). It was detected in woods (*WD*: 0.16 ± 0.08 SE ind.trap^{-1}) where its abundance did not differ from that in roadside verges (Z = 0.427 and P = 0.431).

In field habitats, shrew abundances were very low. *S. coronatus* abundances in field margins were less important than in roadside verges (Z = -3.337 and P < 10^{-3}) or in woods (Z = -2.802 and P =0.005). Abundances appeared as a weak function of distance to margins: a tendency was observed but no differences were significant (Z = 1.689 and P = 0.091) between abundances in field margin (*FM*: 0.05 ± 0.02 SE ind.trap^{-1}) which appeared to be higher and abundances in field-25 (*F1*: 0.01 ± 0.01 SE ind.trap^{-1}). No any *S. coronatus* was captured in field-50m.

TABLE 1:

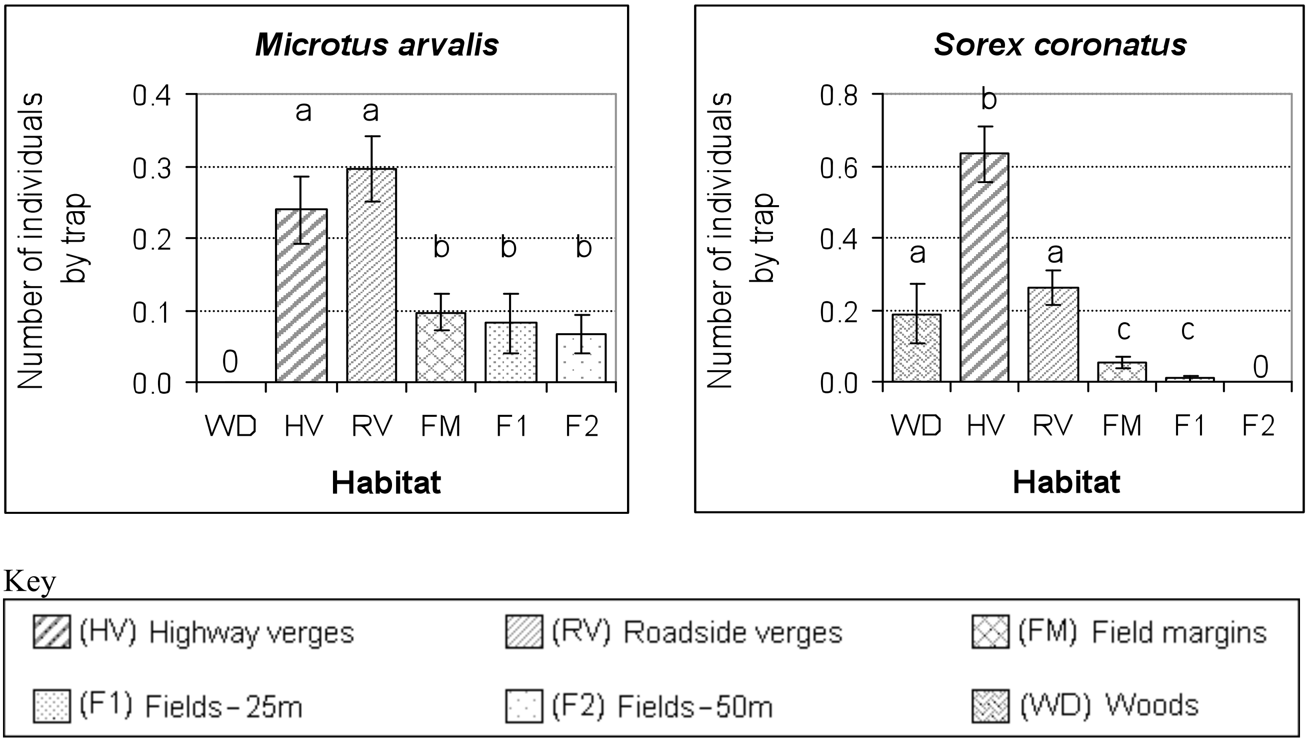
Species detected in the different habitats are distributed between shrews (*Croci1dura russula*, *S. coronatus* and *S. minutus*), voles (*Clethrionomys glareolus*, *M. agrestis* and *M. arvalis*) and mice (*Apodemus flavicollis* and *Apodemus sylvaticus*). The table presents the number of individuals by species captured in each habitat. Only recovered traps are considered (some traps were destroyed during the experiment).

Zones	Habitats	Sites	Trap by site	Traps recovered	mice		voles			shrews			Total
					(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	
Natural areas	*Woods*	32	1	32	-	1	2	2	-	2	6	0	**13**
Marginal areas	*Highway verges*	31	5	155	-	4	-	-	37	13	98	6	**158**
	Road verges	48	5	240	5	8	-	4	71	22	63	-	**173**
	Field margins	65	3	187	-	8	-	-	18	5	10	-	**41**
Fields	*Fields 25m*	65	2	110	-	5	-	-	9	1	1	-	**16**
	Fields 50m	56	2	89	-	3	-	-	6	-	-	-	**9**
Total	**6**	**297**		**813**	**5**	**29**	**2**	**6**	**141**	**43**	**178**	**6**	**410**

(1) Apodemus flavicolli, (2) Apodemus sylvaticus, (3) Clethrionomys glareolus, (4) Microtus agrestis, (5) Microtus arvalis, (6) Crossidura russula, (7) Sorex coronatus & (8) Sorex minutus.

FIGURE 3:

HABITAT ABUNDANCES



A. *M. arvalis* abundances by habitats

B. *S. coronatus* abundances by habitats

Variations in the habitat distribution of (A) M. arvalis and (B) S. coronatus. Groups labelled with different letters differed significantly.

2. Verges and margins as corridors

a. Trap position in verges

We found no effect of trap position on *M. arvalis* distribution in highway verges ($Z = 0.268$ and $P = 0.788$) and in roadside verges ($Z = 0.339$ & $P = 0.735$; *Figure 4A*).

For the shrew, we found no effect of trap position in highway verges ($Z = 1.681$ and $P = 0.093$; *Figure 4B*), but an effect was detected in roadside verges ($Z = -3.890$ and $P < 10^{-3}$; *Figure 4B*): more shrews were captured in traps located at the extremity of the trap lines (A & E: 0.43 ± 0.11 SE ind.trap^{-1}) compared to middle traps (C, D & E: 0.15 ± 0.03 SE ind.trap^{-1}).

b. Distance to connected natural habitats (DC) for S. coronatus

As previously, results differed if considering highway verges or road verges (*Figure 5A*). For highway verges, the distance to the nearest connected natural habitat had no significant effect on *S. coronatus* abundance ($Z = 0.892$ and $P = 0.372$). For road verges, abundance of *S. coronatus* was negatively correlated to the distance to the nearest connected natural habitat ($Z = -5.048$ and $P < 0.001$).

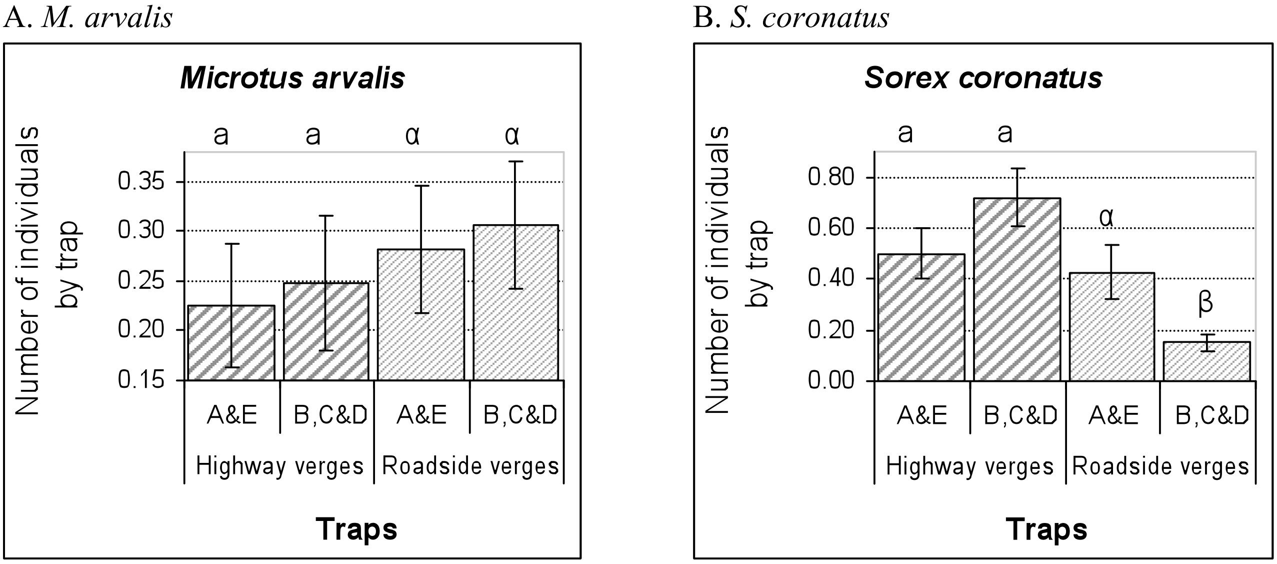
3. Roads as barrier for shrews

Because we did not found any effect of connected distance (*DC*) on shrew abundances in highway verges, we focused this analysis on road verges. We found no correlation between shrew densities and the distance to the nearest connected natural habitat situated on the opposite side of road ($Z = -0.708$ and $P = 0.479$, *Figure 5B*).

IV- DISCUSSION

First, confirming Lowell & Geis observations (1983), we found higher densities of small mammals in verge habitats compared to neighbouring habitats (woods, crop fields and field margins). This demonstrates the importance of these marginal zones for small mammals in intensive agrarian landscapes. Nevertheless, whereas frequented by voles (*M. arvalis*) and shrews (*S. coronatus*), such margins have not similar roles for the two

FIGURE 4:
TRAP POSITION EFFECTS


A. *M. arvalis*

B. *S. coronatus*

No effects of trap position on M. arvalis captures (RV: $Z = 0.339$ & $P = 0.735$, HV: $Z = 0.268$ & $P = 0.788$) and on S. coronatus captures in highway verges ($Z = 1.681$ & $P = 0.093$). Traps are not independent for the shrew in road verges ($Z = -3.890$ & $P < 10^{-3}$). Groups labelled with different letters differed significantly.

species. These results confirm that the ecological function of different elements of landscapes is strongly species dependant (Noss 1987, Beier & Noss 1998).

1. Verges as habitat for voles and shrews

Because (1) the vole *M. arvalis* was detected in margins and in fields and because (2) there was no evidence of dispersion along roads or highways (no effect of trap position), we consider that voles may use verges as habitats, which is in accordance with the biology of rodent species feeding in fields.

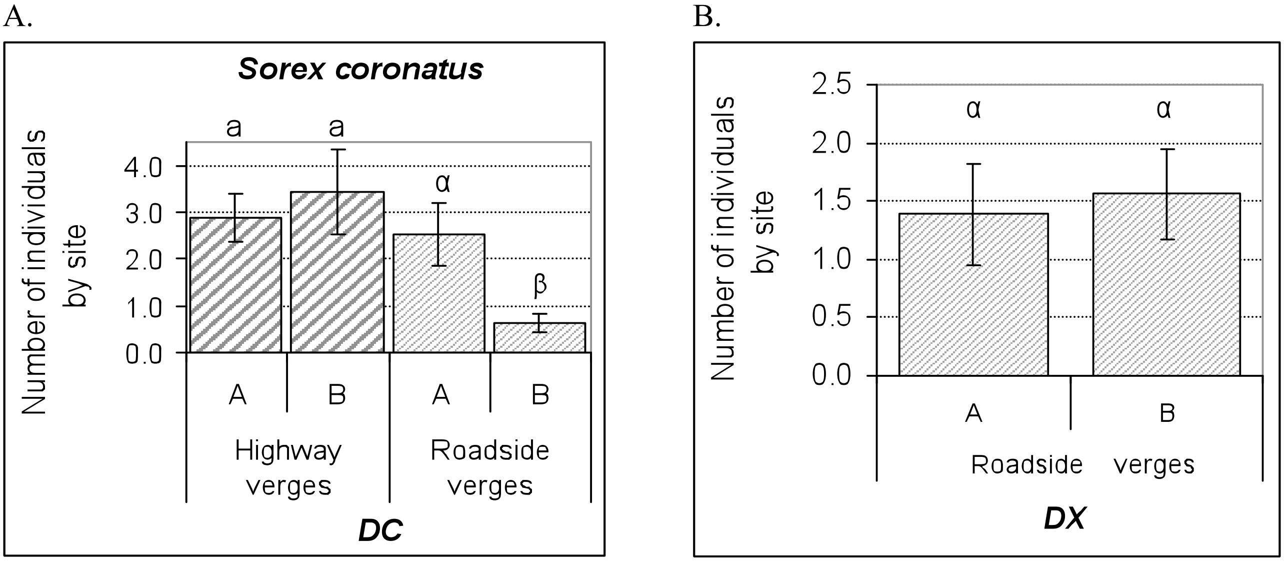
We assume that highway verges provide an effective habitat for *S. coronatus* because (1) we trapped a large number of individuals, and because (2) shrews were quasi-absent from crop fields. The suitability of highway verges as habitat for shrews is confirmed by our observations on the distribution of another shrew species, *Sorex minutus*: considering all traps, we only found this species in highway verges, while this shrew is considered as highly sensitive to habitat degradation and agriculture intensification (La Peña (de) et *al.* 2003).

On one hand, road verges appeared as habitats for shrews, species potentially important in term of ecosystem services for agriculture (regulation of pest insects), but on another hand, road verges appeared as habitats for voles, species considered as pests for agriculture. A future outlook of this study could be to study how simultaneously favour shrew populations and limit vole populations, through specific managements or structure of verges.

2. Verges as corridors for shrews

We observed differences in abundance patterns of *S. coronatus* in road verges. Actually their abundance was negatively correlated to the distance to the nearest adjacent natural habitat, and we found a further effect of trap position demonstrating linear movements of shrews along the road, while shrews were quasi-absent from crop fields. We rather assume that roadside verges

FIGURE 5:
DISTANCE TO NATURAL HABITATS EFFECTS ON *S. CORONATUS* ABUNDANCES



In highway verges, (A) we found no significant correlation between S. coronatus abundances and distance (DC) between capture sites and the nearest connected natural habitat (HV: Z = 0.892 & P = 0.372) but this correlation was significant for road verges (Z = -5.048 & P < 0.001) and (B) disappeared (Z = -0.708 & P = 0.479) considering the distance (DX) between capture sites and the nearest adjacent natural habitat. Groups labelled with different letters differed significantly.

could be used as corridor for dispersion and are lower habitat quality than highway verges, at least for *S. coronatus*. These results are in accordance with the biology of that species, a territorial animal which moves for various purposes. During the breeding season, males disperse to find mates (Cantoni 2002). And because *S. coronatus* is a very active shrew (Genoud 1984), its daily energy expenditure is important (Genoud 1985), which may push them to forage in roadside verges as access to food or food supply are increased on road networks (Oxley *et al.* 1974).

Because Tichendorf & Wissel (1997) found that corridor widths have strong effects on small mammals dispersion and because highway verges and road verges are very similar excepting for their structures (highway verges, 15.7 ± 6.6m, are larger than road verges, 7.2 ± 3.4m), we assume that those differences in verge uses between highways (habitat) and roads (corridor) by *S. coronatus* may be explained by their widths. In accordance with this hypothesis, we also noted that field margins have the smallest widths (4.0 ± 0.4m) and hold the lowest densities of the species (see *Figure 3*).

3. Roadside verges as barrier for shrews

In roadside verges, we found no relation between shrew abundance and the distance to the nearest natural habitat adjacent to the opposite side of the road, though such a correlation was positive when considering natural habitats on the same side of the road. We interpret these contrasting results as indicating some barrier effect of roads for shrews.

Kirby (1997) suggested that roads may be greater barriers for small mammals than their width may suggest, because of their particularly unfriendly habitat. This assumption has been verified in many studies which demonstrated that small mammals are not able to cross roads and tracks: Mader (1984) reported how rarely yellow necked mice *Sylvaemus flavicolis* and bank voles *Clethriomys glareolus* crossed a variety of roads from busy highways down to weakly used forests. For Swihart & Slade (1984), roads were crossed exclusively by medium size animals. They demonstrated that even a dirt track may act as a major barrier to prairie voles *Microtus ochrogaster* but that crossings of this track by cotton rats *Sigmodon hispidus* were density dependant and much lower than expected. Brehme (2003) found that different small mammals were running along roads rather than across and that only juveniles were observed to cross minor roads. Oxley et al. (1974) reported that the smallest road clearance (less than 3m) can inhibit some small mammal movements, a result further confirmed by Merriam et al. (1989) who showed that mice movements across roads were very infrequent although movements adjacent to roads were important and longer than road width.

4. Management policies and small mammals

With the new European *Common Agricultural Policy* (C.A.P.) from 2002 onwards, environmental objectives to preserve biodiversity were introduced into the process of agrarian subsidies distribution: farmers have to choose between different environmental practices to obtain some of the subsidies. One of those practices is the set-up of new field margins around crops. This practice is expected to increase biodiversity in agricultural landscape. Such policies should be maintained and strongly encouraged because they may also increased ecosystem services maintaining population of agriculture pest predators such as shrew which forage mainly on beetle, caterpillar and slug (Bellocq & Smith 1994, Churchfield 1982, Pernetta 1976, Maisonneuve & Rioux 2001) and stabilising vole dynamics by avoiding cycling outbreaks (Delattre et *al.* 1999).

Another actual policy is installation of road underground passages for the small fauna. Because (1) we showed that all roads are barriers for small mammals, because (2) we found that shrews densities are highly correlated to the distance to connected natural habitats, because (3) it has been proved that culverts have strong effects on shrew dispersal between the two sides of the roads (Yanes et al. 1995) and because (4) they limit small mammal road casualties (Lodé 2000). We therefore recommend to purchase such politics of underground passage installations in order to aid the maintenance of shrew communities in verges.

According to Meunier et al. (1999) and because verges represent important conservation areas for small mammals, it should be also interesting to encourage extensive policies of management of highway and roadside verges in intensive agrarian landscapes.

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APPENDIX:
LANDCOVER OF THE STUDY AREA

Landcover		**Area (km²)**	**Percentages**	
Urban areas	*Buildings*	2083	*17.3%*	**18.2%**
	Transports	104	*0.9%*	
Natural areas	*Woods*	2850	*23.7%*	**29.7%**
	Meadows	587	*4.9%*	
	Water	148	*1.2%*	
Marginal areas	*Highway verges*	46	*0.4%*	**3.1%**
	Road verges	144	*1.2%*	
	Field margins	180	*1.5%*	
Fields		5908	*49.0%*	**49.0%**

Ile-de-France region is mainly composed by fields, forests and urban areas. Marginal areas as field margins, roadside and highway verges represent non-negligible areas.

Sorex coronatus

PARTIE II

STRUCTURES ET MODES DE GESTION DES ACCOTEMENTS ROUTIERS : IMPACTS SUR LA BIODIVERSITE

CORRESPONDANCES

La Nature est un temple où de vivants piliers
Laissent parfois sortir de confuses paroles :
L'homme y passe à travers des forêts de symboles
Qui l'observent avec des regards familiers.

Comme de longs échos qui de loin se confondent
Dans une ténébreuse et profonde unité
Vaste comme la nuit et comme la clarté,
Les parfums, les couleurs et les sons se répondent.

Il est des parfums frais comme des chairs d'enfants,
Doux comme les hautbois, verts comme les prairies,
Et d'autres, corrompus, riches et triomphants,

Ayant l'expansion des choses infinies,
Comme l'ambre, la muse, le benjoin et l'encens,
Qui chantent les transports de l'esprit et des sens.

Charles BAUDELAIRE, LES FLEURS DU MAL (1857)

MANUSCRIPT N°3

Submitted to Journal of Environnemental Management

Effect of delayed single mowing on roadsides vegetation communities:
A three years study

Louis de REDON
Frédéric JIGUET
Jeffrey B. JOY
&
Nathalie MACHON

PRESENTATION

TITRE

Effets d'une fauche unique et tardive des accotements routiers sur les communautés végétales des bords de route: Trois ans d'étude

RESUME

Au sein des paysages agricoles, les bords de route peuvent constituer un des rares refuges pour la flore locale. Comme la gestion et l'entretien des dépendances vertes peuvent impacter les communautés végétales des bords de route, nous avons étudié les modifications en termes de diversité spécifique et fonctionnelle au sein de ces communautés durant trois années après un changement du nombre de fauches réalisées par les gestionnaires de routes en région parisienne (FRANCE).

La réduction à une fauche annuelle réalisée en fin de saison (septembre) a eu des effets très forts sur les communautés végétales des bords de route. Premièrement, la diversité végétale a augmenté fortement et rapidement. Deuxièmement, les communautés végétales ont fortement répondu à ces nouvelles pratiques par de profonds changements au niveau de leur organisation fonctionnelle : la gestion extensive des bords de route a par exemple favorisé les plantes bisannuelles et zoochores.

Avec cette expérimentation, nous avons pu montrer comment les politiques de gestion peuvent être conciliées avec des politiques de conservation de la diversité végétale tout en permettant des économies financières aux autorités publiques. Cette expérimentation nécessiteraient cependant d'être poursuivies afin de mesurer les effets à plus long terme de telles pratiques de gestion extensives des bords de route.

MOTS CLEFS

ACP, Gestion des bords de route, Indices de Hill, Politiques environnementales, Traits fonctionnels.

EFFECT OF DELAYED SINGLE MOWING ON ROADSIDES VEGETATION COMMUNITIES: A THREE YEARS STUDY

Louis de REDON[a*], Frédéric JIGUET[a,1], Jeffrey B. JOY[b,2] and Nathalie MACHON[a,3].

KEYWORDS

Environmental policies
Functional traits
Hill index
PCA
Road verges management

SUMMARY

In intensive agrarian landscape, roadside verges constitute one of few refuges for wild plant species. Because roadside management shapes plant communities on verges, we studied the species and functional diversity of roadside plant communities during the three years following a reduction in the number of times roadsides were mowed in the region of Paris (France). The reduction of mowing to one single late cut had a strong effect on roadside plant communities. First, plant diversity increased strongly and rapidly, though this effect was variable between years and thus requires long term monitoring to understand its significance to the plant community over time. Second, plant communities responded with deep changes in their structure with reduced mowing favouring apparition of bisannuals plants and species whose seeds are animal dispersed. With this experiment, we showed how management policies are able to conciliate plant diversity protection with public finance savings.

1. INTRODUCTION

In intensive agrarian landscape, plant diversity is low and mainly concentrated in field margins and roadsides (Way, 1977; Sỳkora *et al.*, 1993; Burel *et al.*, 1998). However, roads are known to have deleterious effects on nearby plant communities (Forman and Alexander, 1998). The metropolitan French main and secondary road networks comprise approximately 1,000,000 km long (1.82 km of roads per km^2) and roadside verges cover approximately 5,100 $km^{2,}$ an area equivalent to 1% of the area of France (544,000 km^2), more than the total area of French National Parks which represent 3,400 km^2. If roadside vegetation management combines visibility and environmental objectives, roadside verges could play a significant role in plant conservation. Roadsides are known toconstitute one of the main refuges for wild plant species in agrarian landscape (Bennett, 1991; Akbar, 1997). Usually, vegetated roadsides are only managed for traffic visibility and esthetical reasons; to fulfill those objectives, the vegetation on banks and verges are mowed regularly by the road services. The number of cuts generally depends on availability of manpower and machines (Genard *et al.*, 1994). In France, ordinary management of roadsides usually includes three cuts a year. We posit that roadside management and in particular the number of cuts per year has a strong effect on roadside plant communities (Parr and Way, 1988; Sỳkora *et al.*, 2002).

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In order to provide managers with the best advice to preserve biodiversity, we sought to investigate the potential impact of a single light and delayed yearly mowing on roadside communities which functional diversity of plant communities during three years with a reduction of the number of cuts in roadsides, compared to roadsides still subjected to three cuts. The experiment was conducted in the region of Paris (France), a typical intensive agrarian landscape composed almost exclusively of agricultural fields and groves.

2. MATERIALS AND METHODS

2.1. Study area

In France, roadside management followed a departmental council policy. Each department is divided into management units. Our study area is situated in the Fontainebleau management unit of the Seine-et-Marne department, central France about 70 km south-east from Paris (48° 24′ N, 2 42′ E). 51.2 % of the management unit area, 258.2 km², is located in a Natural Regional Park (*Parc naturel régional du Gâtinais français*). It is mainly composed of agricultural fields used for intensive crop production (33.6 % of landcover) and woods (54.0 %). Road networks (246 km long, *i.e.* 2.34 km.km^{-2}) represent important linear structures in the landscape along with their verges (5.23 km², 2.02 % of the total area of the department).

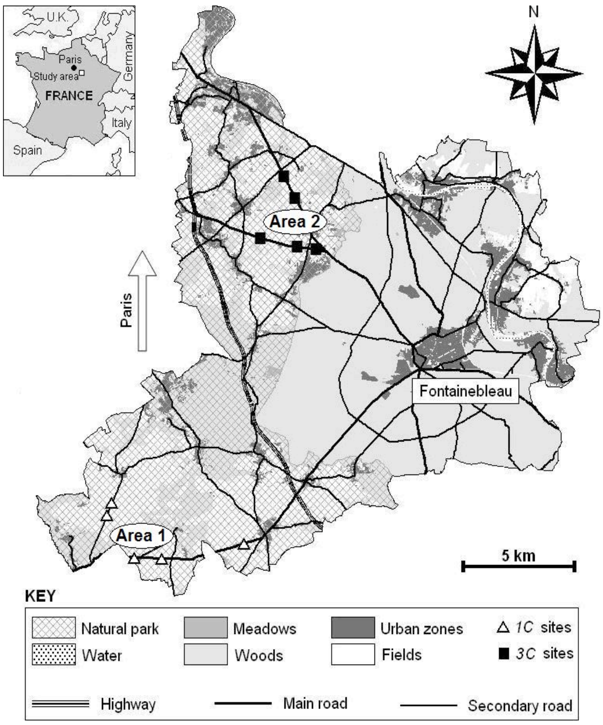
2.2. Sampling design

We sampled 10 sites: five roadsides in the southern part of the natural regional park (area 1) and five other roadsides in the northern part of the park (area 2, *Figure 1*). The distance between site 1 and site 2 is approximately 16 km, thus climate and local conditions are similar enough to permit a comparison between the different roadsides of the two areas.

Each roadside is divided into two zones: the berm and the bank. The berm, situated into the 2 to 3 first meters from the pavement was usually mown three times a year (March, May and September). The bank is situated between the berm and the adjacent habitat and is mown only once a year in September.

FIGURE 1

STUDY AREA



From 2006 to 2008, the park and the department council reduced the treatment of the berm to one single late cut in September (*1C*) on the southern surveyed sites. The other five (northern) sites were given the usual, three cuts (*3C*) a year; they are considered controls for the experiment.

For each site, five plots of 1 m² (0.5 x 2 m) were regularly placed every 20 meters along roads in the berms and in the banks. Vegetation was recorded in the plots in early May 2006, 2007 and 2008. We identified all plant taxa found in the plots and evaluated their abundance with a corrected Braun-Blanquet method (Braun-Blanquet, 1932; Westhoff and Van der Maarel, 1978). Almost all the taxa were identified to species level according to the International Plant Names Index (http://www.ipni.org). In some cases (*Crepis sp.* for example), taxa were identified to genus level due to difficulties in identification after mowing.

2.3. Data analysis

2.3.1. Changes of diversity index 2006-2008

For each plot, we calculated a corrected Hill diversity index (Hill, 1973) using the median recovery value (Pyšek 2004) of the Braun-Blanquet scale, noted as *E*:

$$p_i' = \frac{R_n}{\sum_{n=1}^{S} R_n},$$

$$N_1' = -\sum_{i=1}^{S} p_i' \times \log_2(p_i'),$$

$$N_2' = \frac{1}{\sum_{i=1}^{S} p_i'^2},$$

$$E_{1,2}' = \frac{N_1'}{N_2'}.$$

S is the number of species detected in a plot,
R_n is the median Braun-Blanquet recovery value of the nth species of a plot,
p_i' is the relative abundance of ith species in a plot.

To have a more intuitive diversity index, we calculated $E_y = 1 - E_{1,2}'$ (E_y has a maximum value for a maximum of diversity, $E_y \in [0,1]$ and $y \in [2006, 2007, 2008]$).

We then ran *ANOVA*s to test the differences of E_{2007} and E_{2008} values between the two categories of berms (1 cut *vs.* 3 cuts). Because northern and southern communities could not be considered completely similar, samples of 2006 are used as reference data with $E_{2006\ (\mathrm{BERM})}$ as control variables in each site (*Model*: $\mathrm{E}_{\mathrm{y}\ (\mathrm{BERM})} \sim \frac{\mathrm{E}_{\mathrm{y}\ (\mathrm{BANK})}}{\mathrm{E}_{2006\ (\mathrm{BANK})}} + \mathrm{E}_{2006\ (\mathrm{BERM})} +$ Treatments, $y \in [2007;2008]$). We also tested the differences of E_{2007} and E_{2008} values in banks between the two sites (*Model*: $\mathrm{E}_{\mathrm{y}\ (\mathrm{BANK})} \sim \mathrm{E}_{2006\ (\mathrm{BANK})} +$ Treatments, $y \in [2007;2008]$).

2.3.2. Changes in plant traits composition 2006-2008

Using data issued from three reference books (Jauzein, 1998; Lambinon et al., 2004; Lauber and Wagner, 2007), we ran an analysis on five life history traits with their different modalities to: flowering period (*FP<2*: flowering period shorter than two months; *2<FP<4*: flowering period comprised between 2 and 4 months; *FP>4*: flowering period longer than four months), number of cotyledons (*Monocots*; *Dicots*), phenology (*Annuals*; *Bisannuals*; *Perennials*), pollen dissemination (*Anemogamous*; *Autogamous*; *Entomogamous*) and seed dispersal (*Anemochorous*; *Autochorous*; *Zoochorous*) and we built a ***matrix A***: species *vs.* traits modalities. In each plot of berm zones, we calculated the total area covered by each species and we built a ***matrix B***: plots *vs.* species covering. We then calculated ***matrix B.matrix A*** to obtain a ***matrix C*** (plots *vs.* traits modalities covering) needed to perform a PCA on plots and their plant trait compositions (Aubin et *al.* 2007) on the three years.

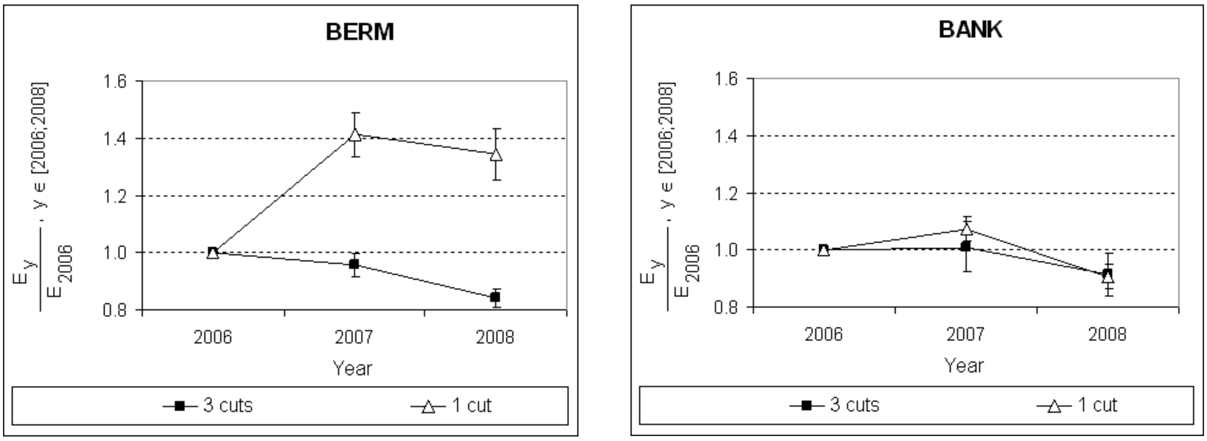
We tested the impacts of year (2006, 2007 and 2008) and treatment (*1C vs. 3C*) on plot coordinates on *Axis 1* and *Axis 2* of the PCA (*Model*: AXIScoordinates ~ Year*Treatments). Using this method, we studied plant community modifications following changes of mowing treatment in berms.

We also investigated in *1C* berms changes in the covering of some traits modalities (calculated for ***matrix C***) according to the PCA results (traits modalities which seemed to be favoured by delayed mowing). We tested the impacts of years on those changes (*Model*: $\mathrm{TraitModality}_{\mathrm{BERM}} \sim \mathrm{TraitModality}_{\mathrm{BANK}} +$ Year)

2.3.3. Changes in detection probalities 2006-2008

We calculated detection probabilities in all sites for the three years with ComDyn

FIGURE 2
CHANGES OF CORRECTED HILL INDEX (*E*) DURING THREE YEARS


A. In berms
B. In banks

software (Hines *et al.*, 1999). For this calculation, we used the five local plots as spatial replicates and we made the null hypothesis that vegetation did not differ among plots in each site.

We verified that the observed changes in plant communities (diversity and traits composition) were not due to variation in detection probalities by testing the variation in detection probabilities between years and mowing treatments.

All statistical analyses were performed with the statistical package R (Ihaka and Gentleman, 1996).

3. RESULTS

We detected a total of 133 taxa (106 at the species level; *cf. Appendix 1*in 300 vegetation sample plots during the three years of survey ($N_{E\ (2006)}$ = 86, $N_{E\ (2007)}$ = 73, $N_{E\ (2008)}$ = 85).

3.1. Changes in diversity index 2006-2008

In 2007, *E* values changed differently between the two categories of sites: they increased by 41 % in *1C* berms and decreased by 4% in *3C* berms ($F_{3,49}$ = 5.735 and P = 0.028; *Figure 2*).

In 2008, we observed a decrease of *E* values in berms (*3C* sites: -12%; *1C* sites: -5%). The difference of *E* values was significant between *3C* and *1C* berms ($F_{3,49}$ = 10.134 and P = 0.003).

3.2. Changes in plant traits composition 2006-2008

PCA showed clear changes in the traits of roadside plant communities according to their management during the three years of investigation (*Figure 3*). PCA 1 explained 33% of the total variance *Figure 3*) and sites along this axis were highly correlated with mowing treatments (t = -2.846 and P = 0.0051). Significant relationships were not found for PCA 2 (15.6%, t = 1.889 and P = 0.0609) or PCA 3 (13.4, t = 1.025 and P = 0.307).

Some traits modalities seemed to be favoured by the 1C treatment such as dicots, bisannuals, entomogamous and zoochorous species (*Table 1* and *Figure 4*). Plants with a long flowering period seemed also to benefit from the management changes.

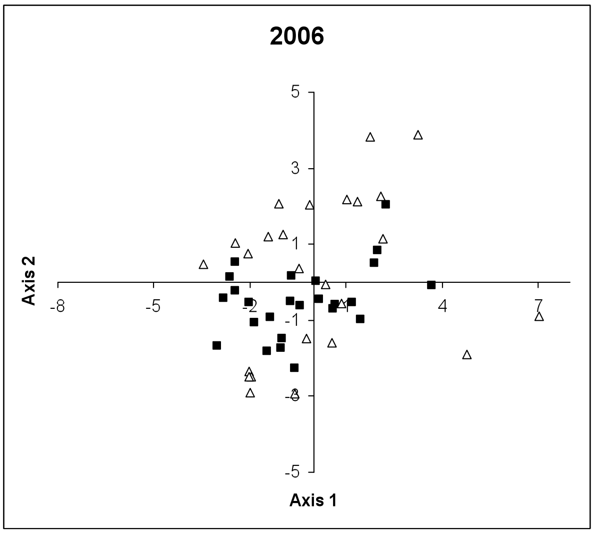
Traits modalities covering the *1C* berms changed significantly with year.(*Figure 5*). We found a strong increase of bisannuals species in berms (t = 2.491 and P = 0.0151), 2006: 11.9% ± 0.04, 2007: 15.8% ± 0.03 and 2008: 24.4% ± 0.04.

Zoochorous species also increased (t = 2.135 and P = 0.0362), 2006: 5.6% ± 0.02, 2007: 6.9% ± 0.02 and 2008: 13.7% ± 0.03).

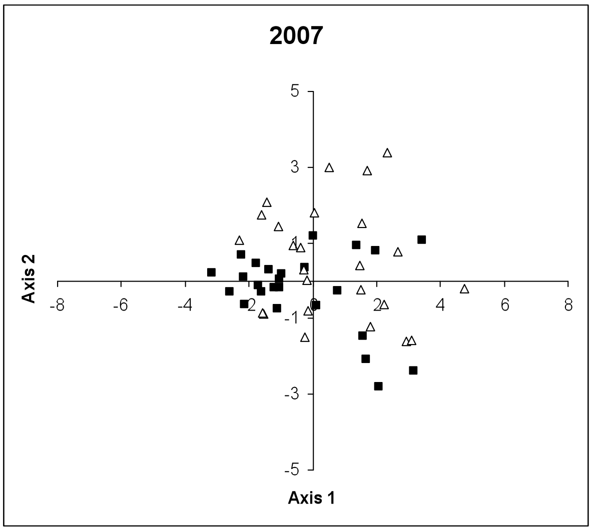
FIGURE 3

PCA ON SITES AND COMMUNITIES TRAITS RICHNESS, SITE PROJECTION (AXIS 1 X 2)

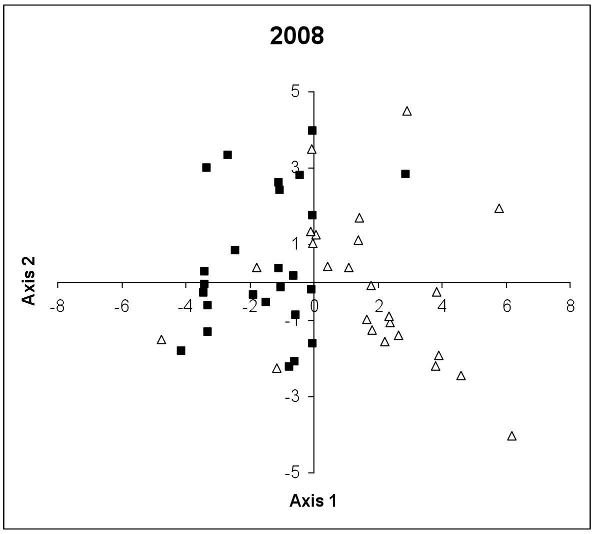
A. 2006



B. 2007



C. 2008



KEY: ■ 3 cuts sites △1 cut sites

No significant differences were observed for entomogamous (t = 0.027 and P = 0.979) or dicots (t = 1.398 and P = 0.166) species. Berm plants with long flowering periods were also not affected by the new mowing treatment (t = 0.060 and P = 0.953).

2.3. Changes in detection probalities 2006-2008

No changes in detection probabilities were observed between years and mowing treatments (t = -0.336 and P = 0.740).

TABLE 1

LIFE TRAITS COORDINATES ON PCA

A. Axis 1

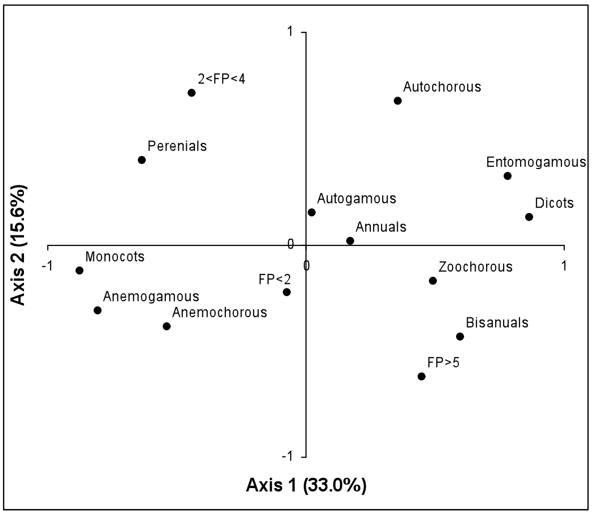
Axis 1 (33.0%)	
Coordinates	**Life traits**
0,870	Dicots
0,785	Entomogamous
0,599	Bisanuals
0,497	Zoochorous
0,451	FP>5
0,359	Autochorous
0,175	Annuals
0,026	Autogamous
-0,068	FP<2
-0,440	2<FP<4
-0,536	Anemochorous
-0,631	Perenials
-0,802	Anemogamous
-0,871	Monocots

B. Axis 2

Axis 2 (15.6%)	
Coordinates	**Life traits**
0,708	2<FP<4
0,675	Autochorous
0,395	Perenials
0,317	Entomogamous
0,150	Autogamous
0,124	Dicots
0,015	Annuals
-0,125	Monocots
-0,174	Zoochorous
-0,226	FP<2
-0,315	Anemogamous
-0,390	Anemochorous
-0,436	Bisanuals
-0,627	FP>5

FIGURE 4

PCA ON SITES AND COMMUNITY TRAIT RICHNESS, TRAITS PROJECTION: AXIS 1 (33.0%) VS. AXIS 2 (15.6%)



4. DISCUSSION

The reduction of mowing to one single late cut strongly influenced plant communities of the berms. First, plant diversity increased strongly and rapidly in one year (by more than 40%), representing a substantial gain in biodiversity in these habitats. Such strong increases in plant diversity maybe due to management changes or a result of plant community dynamics.

Further evaluation of the effects of the *1C* management regime will prove informative in determining the relative roles of management and plant community dynamics.

Second, plant communities responded with changes in their structure, and some trait modalities were favoured by the management changes. Notably, proportions of bisannuals and zoochorous plants within the communities increased strongly during the three-year experiment. The increase in bisannuals may be explained by the fact that a single late cut may allow them to finish their two year cycles, whereas more intensive mowing treatments (3 cuts a year) may not. Zoochorous species might be favoured because intensive mowing treatments have strong negative impacts on animal communities (Meunier *et al.*, 1999). This response is slower, continuous and very important because functional responses at the community level have a strong influence on ecosystem processes (Violle *et al.*, 2007).

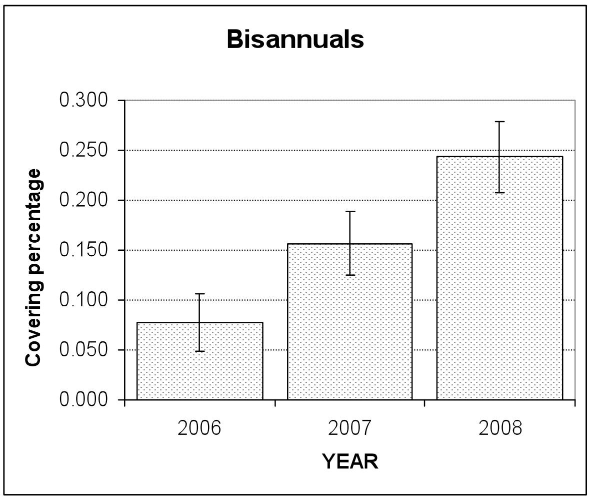
Some trait modalities, like *entomogamous*, *dicots* or *long flowering period*, appeared correlated to PCA axis 1 in a way suggests they are favoured by the light mowing treatments (*Table 1*). Because no significant changes were observed between years in *1C* berms, and because the observed changes are real and not due to variations in detection probability, those differences highlighted by the PCA may be a result of site specific differences between *1C* and *3C* sites which are separated by approximately 16km.

The power of this multiple trait approach could be limited because of the non-independence (colinearity) of some

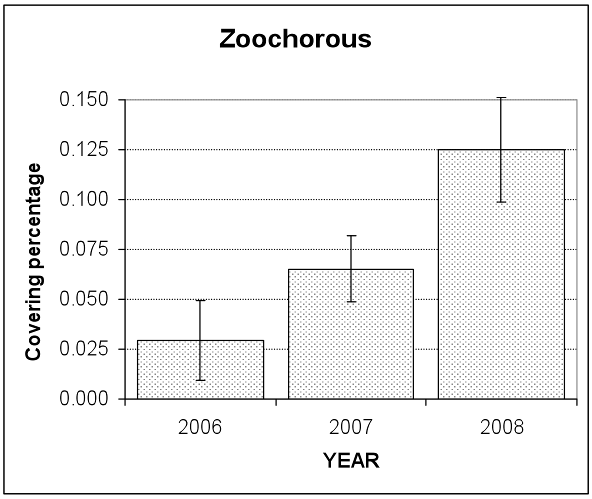
FIGURE 5

CHANGES IN TRAITS COMPOSITION OF PLANT COMMUNITIES OF ONE CUT BERMS (*1C*)

A. Bisannuals species



B. Zoochorous species



traits such as pollen dissemination and seed dispersal.

Nevertheless, the multiple trait approach is of value because (1) while some traits strongly covary among species, others traits can show no correlation (Wardele *et al.*, 1998) and (2) some correlations between traits are only seasonal (Eviner, 2004). Such experimental management changes should be carried out over a longer period to observe the long term effect of extensive mowing of roadsides on plant communities, as done for grazing experiments (Bullock *et al.*, 2001).

5. CONCLUSION

With this experiment, we highlighted the impact of management policies on local biodiversity even in areas usually considered as marginal for plant conservation. In intensive agrarian landscape, management of roadsides could lead to higher species and trait diversity (Le Viol *et al.* 2008)and thus better ecosystem services (Couvet *et al.,* 2006) for agriculture without additional cost. We thus strongly encourage public policies that are able to conciliate plant diversity conservation with public finance savings.

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APPENDIX 1

LIST OF DETECTED TAXAS

Achillea millefolium	*Agrostis stolonifera*
Agrimonia eupatoria	*Allium sp.*
Elymus repens	*Alopecurus myosuroides*

Anthriscus cerefolium
Anthriscus communis
Anthriscus vulgaris
Arhenaterum elatius
Artemisia vulgaris
Aster sp.
Bellis perennis
Bromus erectus
Bromus inermis
Bromus hordeaceus
Bromus sp.
Bromus sterilis
Capsella bursa-pastoris
Carex acutiformis
Carpinus betulus
Carex cuprina
Centaurea jacea
Centaurea scabiosa
Centaurea sp.
Cerastium fontanum
Cerastium sp.
Chaerophyllum temulum
Cichorium intybus
Circium vulgare
Convolvulus arvensis
Conyza sumatrensis
Crepis sp.
Dactylis glomerata
Daucus carotta
Dipsacus fullonum
Equisetum arvense
Erophila verna
Euphorbia sp.
Festuca arundinacea
Festuca filiformis
Festuca rubra
Festuca sp.
Ficaria sp.
Galium aparine
Galium molugo
Galium parisiense
Galium sp.
Galium verum
Geranium columbinum
Geranium dissectum
Geranium molle
Geranium rotundifolium

Geum urbanum
Glechoma hederacea
Hedera helix
Heracleum sphondylium
Hieracium sp.
Himenthoglossum hircinum
Holcus mollis
Hyacinthoides non-scripta
Lamium purpureum
Lamium sp.
Lathyrus pratensis
Lathyrus tuberosus
Ligustrum vulgare
Linaria vulgaris
Lolium perenne
Lotus corniculatus
Medicago lupulina
Mentha suaveolens
Mercurialis sp.
Ononis sp.
Papaver rhoeas
Petroselinum sp.
Phleum pratense
Phragmites sp.
Picris echioides
Picris hieracioides
Plantago coronopus
Plantago lanceolata
Plantago major
Poa chaixii
Poa pratensis
Poa sp.
Poa trivialis
Polygonatum multiflorum
Potentilla reptans
Primula veris
Prunus avium
Prunus spinosa
Prunella vulgaris
Ranunculus acris
Ranunculus arvensis
Ranunculus repens
brassicées hyb-----
Roripa amphibian
Rubus fruticosus
Rubia peregrina
Rubus sp.

Rumex acetosella
Rumex crispus
Rumex sp.
Senecio jacobaea
Senecio sp.
Senecio vulgaris
Silene vulgaris
Silene latifolia
Silene sp.
Solanum nigrum
Sonchus asper
Sonchus oleraceus
Tanacetum sp.
Tanacetum vulgare
Taraxacum officinalis
Taraxacum sp.
Teucrium scorodonia
Tragopogon pratensis

Trisetum flavecens
Trifolium pratense
Trifolium repens
Trifolium sp.
Tussilago sp.
Urtica sp.
Veronica arvensis
Verbena officinalis
Vicia cracca
Vicia hirsuta
Vicia sativa
Vicia sp.
Vicia tetrasperma
Viola arvensis
Vulpia myuros

Almost all the taxa were identified to species level according to the International Plant Names Index (http://www.ipni.org).

Apodemus sylvaticus

MANUSCRIPT N°4

In revision, Mammalian Biology

Possible effects of roadside verges on vole outbreaks in an intensive agrarian landscape

Louis de REDON
Christian KERBIRIOU
Nathalie MACHON
&
Frédéric JIGUET

PRESENTATION

TITRE

Effets potentiels des bords de route sur les explosions démographiques de campagnols dans les paysages d'agriculture intensive

RESUME

Notre étude a porté sur les populations de *Microtus arvalis* (campagnol des champs) en bord de route pendant deux saisons sur 45 sites. Entre les deux années de relevés, le nombre d'individus capturés a été multiplié par 2,6. Nous avons montré que l'explosion du nombre de *M. arvalis* n'a pas été homogène dans le paysage mais était fortement dépendante de sa structure, en particulier du maillage routier. Les populations les plus stables ont été observées dans les sites avec un important réseau de dépendances vertes des bords de route alors que les explosions ont été notées dans les sites avec les dépendances vertes les moins étendues.

MOTS CLEFS

Microtus arvalis; Route, Service écosystémique.

POSSIBLE EFFECTS OF ROADSIDE VERGES ON VOLE OUTBREAKS IN AN INTENSIVE AGRARIAN LANDSCAPE

Louis de REDON[1,2]; MACHON[2]; Christian KERBIRIOU[2] and Frédéric JIGUET[2].

KEYWORDS

Microtus arvalis
Population dynamics
Road

ABSTRACT

We sampled populations of Microtus arvalis (common vole) in 45 roadside verges during two consecutive years. The number captured increased by a factor of 2.6 between the two years. We showed that outbreak dynamics were not homogeneous over the landscape and were highly correlated to landscape structure, especially linked to the spatial extent of the roadsides. Populations were stable in sites where roadside cover large areas while population outbreaks occurred in sites with low roadside areas.

I. INTRODUCTION

Road verges are considered as refuge zone or habitat for a number of small mammals in intensive agrarian landscapes (Bellamy et al., 2000; Maisonneuve and Rioux, 2001; Pita et al. 2006) and also participate to the conservation of some endangered species (Santos et al.; 2007). This refuge zone effect has been verified for *Microtus arvalis* (La Peña (de) et al., 2003), a rodent known to display regular population fluctuations (Briner et al., 2007), local outbreaks (Murray, 1965, Delattre et al., 1992). Such variations in population size lead periodically to increased vole densities causing significant damages to crops, especially for vole densities over 200 individuals per hectare (Delattre et al., 1999).

We trapped voles in two successive springs (2006 and 2007) along roadsides to test and estimate the impact of roadside network densities on their inter-annual abundance variations. We found that if local voles densities were affected by fields network importance; their dynamics cycles between years were partly impacted by roadsides structures. Such marginal areas possibly provide habitats for vole predators, and therefore help with regulating vole populations and decreasing the intensity of occasional outbreaks (Delattre et al., 1999).

II. MATERIALS AND METHODS

Our study area was located in central France, Seine-et-Marne department, about 50 km east of Paris (48° 32′N, 2° 39′E). It was mainly composed of agricultural fields used for intensive crop production (60.7% of landcover). Road networks represented important linear structures in the landscape (13,992 km long, *i.e.* 2.4 km.km^{-2}) along with their verges (63.0 km², *i.e.* 1.07 % of the total area of the department). We sampled 45 roadside verges, each one with five traps placed linearly every 20 meters along the road in the middle of the bank (for a total of 225 traps each year). We used *Barber* pitfall traps: 450 mL plastic pot (10.2 cm height & 7.5 cm diameter) fully inserted into the ground and containing a 155 mL solution composed of 75 mL of water, 75 mL of conservative (ethylene glycol), 5 mL of surfactant (dishware soap) and 15 g of NaCl.

Each trap was protected from rain by a plastic transparent cover (15 cm × 15 cm) supported by wood sticks 10 cm above the ground. The traps were set up for 28 days twice during May 2006 and during May 2007. Not any pitfall was found empty or dried due to evaporation, and then all pitfalls were considered as efficient. This solution was considered as un-attractive solution for

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some animals as invertebrates (Darren et al. 2001).

We calculated (1) the area covered by crop fields and (2) the roadside network area in an area of 500 m around each trapping site using ArcGis 9.1^{TM} software on landcover and road network layers (buffers). We used a geographical layer classifying roads into three categories according to their status; R_N: national roads (0.9 ha of verges for 1 km of road), R_D: departmental roads (0.7 ha.km^{-1}), and R_L:local roads (0.3 ha.km^{-1}). We reported the numbers of captured *M. arvalis* per group of 5 traps, then we studied their inter-annual variations with an ANOVA, according to local roadside network area (noted as A_{RN}, $A_{RN} = 0.9 * R_N$ length + 0.7 * R_D length + 0.3 * R_L length). We further classified variations in four categories:

- ***ABS***, **Abs**ence: *M. arvalis* was not detected during either year of survey;

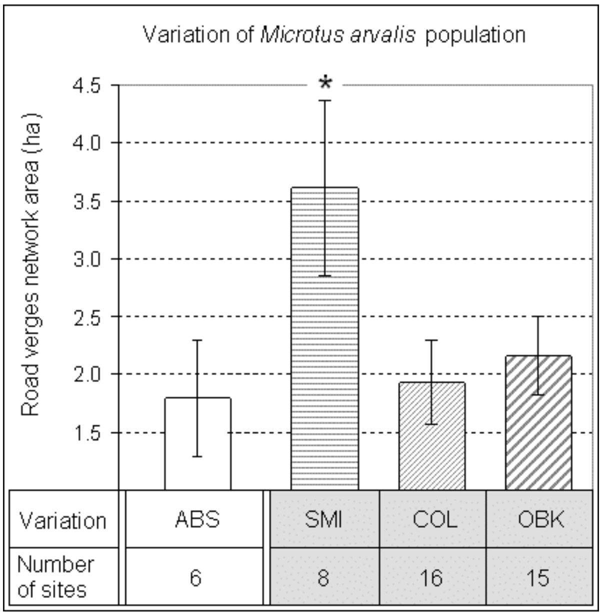
- ***SMI***, **S**table or **M**oderate **I**ncrease: the number of captures did not vary or increased by less than 75% between years;

- ***COL***, Local **Col**onization: *M. arvalis* was not detected in 2006 but was captured in 2007;

- ***OBK***, **O**ut**b**rea**k**: captures increased by more than 100% between the years.

We chose the level of 100% for outbreak classification according to Giraudoux et al. (1997) and Duhamel et al. (2000) studies. It should be added that we observed no local decrease and no increase between 75% and 100%. We also analysed the log ratio of local abundance of both years, as $\log(N_{2007}+2)/\log(N_{2006}+2)$, with a Generalized Linear Model, to test for linear relationships beyond potential differences between categories. Because *M. arvalis* is a rodent species adapted to field habitats, we also investigated the impact of crop local coverage on its distribution (total captured and variation classes). We also ran a GLM to test the possible correlation between the two variables used in this study: roadside network areas and local field coverage.

FIGURE:
IMPACT OF ROAD NETWORK ON VARIATIONS OF *M. ARVALIS* POPULATIONS



ABS: absence (sites excluded from analysis), SMI: no/moderate increase of captures (<75%), COL: colonization in 2007, OBK:

All statistical analyses were performed with R (Ihaka and Gentleman, 1996).

III. RESULTS

The trapping method worked very well without any full trap (0 to five individuals by trap) and 167 *M. arvalis* trapped (47 in 2006 and 120 in 2007). Excluding sites without any capture (6), the effect of roadside network areas on vole dynamics was significant ($F_{2,36} = 3.346$ & $P = 0.0465$): non-outbreaking persisting populations (*SMI*) were observed in sites with the most important roadside network densities (3.61 ha ± 0.75 SE). Populations outbreaks (*OBK*) occurred in sites with less developped roadside networks(2.16 km ± 0.33 SE), and local colonisation too (*COL*: 1.93 km ± 0.36 SE).

We found no effect of local crop cover on *M. arvalis* variation classes ($F_{2,36} = 0.8781$, $P = 0.4243$), though a positive effect was detected if considering the number of individuals trapped during the two years

($F_{1,43}$ = 4.919, P = 0.0319). We found weak slight non-significant linear relationship between yearly variation in local vole abundance and linear roadside length (the latter log-transformed; t = -1.91, DF = 37 & P = 0.063) or roadside network area (the latter square-root transformed; t = -1.99, DF = 37 & P = 0.054). We finally noticed that field cover and roadside areas were not correlated ($F_{1,45}$ = 2.658, P = 0.11 & R^2 = 0.036).

IV. DISCUSSION

We observed important differences in vole populations sizes between the two years of sampling in the studied agrarian landscape. Traps did not appear to be attractive to voles. However, as for all capture methods, a bias is possible. Neverless we assume that if it exists, it is probably the same for the two sampling years or the different study sites. Thus it should not have affected our results.

Because road verges are mowed intensively, i.e. three times a year, the results are in accordance with previous reports on the effects of such mowing treatments on vole population variations (Meunier et al., 1997). Moreover, we highlighted a positive effect of roadside areas on vole outbreak regulations: variations in population size were not spatially homogeneous, and large roadsides seemed to stabilize vole populations, while local colonization or outbreak occurred where roadside verges were smaller. The weak slight non-significant linear relationship between yearly variation in local vole abundance and linear roadside length may be due to the data set size which could be considered as not big enough important. Such results are in accordance with previous studies reporting that *M. arvalis* outbreaks in agrarian landscapes can be controlled by predation thanks to marginal areas providing habitats for predators such as cats (Hansson, 1988), polecats (Rondinini et al., 2006) or raptors (Meunier et al., 2000).

We also found strong impacts of local landcover (field cover) on *M. arvalis* distribution. In fact, field habitats seemed to impact local abundances more than inter-annual variations because the number of individuals trapped in all sites during the two years and the local field cover are positively correlated. To conclude, we strongly encourage maintenance and development of large verges along existing roads to prevent vole outbreaks and avoid agrarian damages by stabilizing their population dynamics.

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Microtus arvalis

DISCUSSION ET CONCLUSION GENERALES

ANALOGIE DE LA GUEPE ET DU TRAMWAY ELECTRIQUE

Quelque chose de muet au repos et de chanteur en action. Quelque chose aussi d'un train court, avec premières et secondes, ou plutôt motrice et baladeuse. Et trolley grésilleur. Grésillante comme une friture, une chimie (effervescente).

Et si ça touche, ça pique. Autre chose qu'un choc mécanique : un contact électrique, une vibration venimeuse.

Mais son corps est plus mou – c'est-à-dire en somme plus finement articulé – son vol plus capricieux, imprévu, dangereux que la marche rectiligne des tramways déterminée par les rails.

Francis PONGE

LA RAGE DE L'EXPRESSION (1952)

§ A. EFFETS DES PROTOCOLES UTILISES ET DES GROUPES SUIVIS

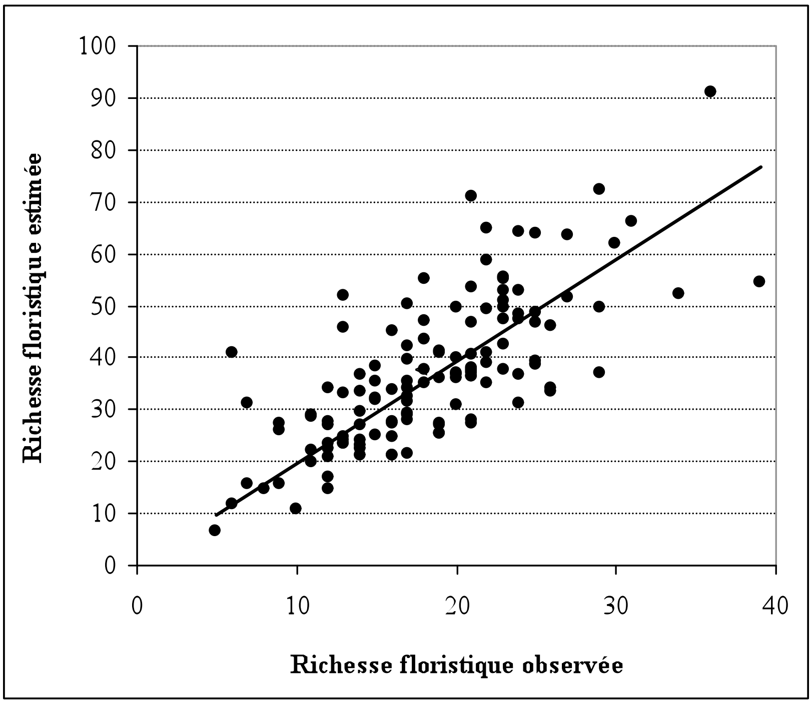
Cette étude de la biodiversité commune en milieu agricole intensif a été riche en résultats : sur la biodiversité des bords de route eux-mêmes (effet « habitat », effets des polluants) et sur la compréhension de l'écologie des milieux adjacents (homogénéisation biotique, effets « barrière », effets « corridor », maintien de services écosystémiques locaux). Ces résultats sont très intéressants car ils montrent l'intérêt de l'étude la biodiversité dans sa globalité, *i.e.* en tenant compte de la Nature ordinaire, riche en informations et permettant la mise en évidence de nombreux processus écologiques de nos paysages agricoles modernes. Cependant si, au début de la thèse, certains groupes ont été ciblés pour servir d'indicateurs, plantes et carabes, les différents protocoles n'ont pas tous eu le succès escompté.

1) Relevés floristiques

Le protocole de relevés floristiques mis en place en bords de route s'est révélé être assez bon permettant l'obtention de résultats probants (*cf.* Manuscrit N°1 et Manuscrit N°3) grâce à :

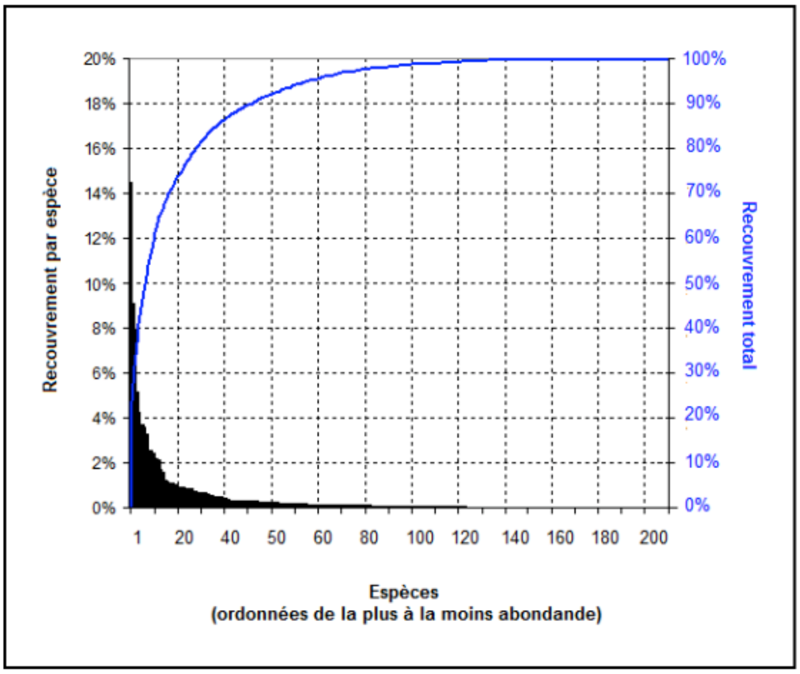
- Sa rapidité d'exécution (8 minutes en moyenne par relevé de 1m², temps d'installation compris, *i.e.* 2 heures sur site par station de 15 relevés), ce qui était capital pour un terrain important (48 stations à étudier en 2 semaines pour éviter un biais temporel lié au décalage de floraison des plantes) et un terrain dangereux (en bords de route, et parfois de routes très passantes !) ;
- Sa répliquabilité et sa simplicité, les quadras de 1 m² sont très faciles à placer et à replacer le long d'une structure linéaire et, par leur petite taille, permettent des relevés exhaustifs de la végétation quel que soit l'observateur ;
- Des taux de détection élevés (67.27% ± 1.10% ES en 2006) et une richesse estimée corrélée à la richesse observée ($F = 147.93$, $r^2 = 0.536$ & $P < 10^{-3}$, *cf.* Figure N°8) pour une erreur d'estimation associée acceptable (22.53% ± 0.48% ES < 25 %) ;
- Une flore des bords de route très commune centrée autour de quelques dizaines d'espèces seulement représentant une grande majorité du recouvrement total (seulement 80 des 212 espèces détectées représentent 95 % du recouvrement total et à peine 9 espèces en représentent plus de 50%, *cf.* Figure N°9) qui permet un apprentissage rapide par les observateurs.

Figure N°8



Richesse floristique observée Vs. Richesse floristique estimée (Ne)

Figure N°9



Répartition du recouvrement végétal en bords de route par espèce

Le protocole s'est révélé adapté aux analyses qui ont été réalisées et a permis la mise en évidence des effets de la fauche au sein des communautés végétales des bords de route, de l'existence d'une homogénéisation biotique des milieux adjacents liés à la flore des bords de route, ou encore des effets « corridor » (des analyses génétiques étant malgré tout nécessaires pour confirmation). Parallèlement, la construction de la base de données GALIUM s'est aussi révélée utile et adaptée aux analyses sur les traits fonctionnels réalisées.

Enfin les deux problèmes rencontrés ont été liés à des problèmes de détermination inhérents au milieu étudié : les bords de route. En effet, (a) certaines plantes avaient des morphologiques altérées (ex : gigantisme) pouvant être notamment dues à la pollution des sols (Angold 1977, Spencer et al. 1988), et (b) certains individus étaient devenus inidentifiables à cause de la fauche.

(2) Relevés faunistiques

Si le protocole pots-pièges de type « barber » pour invertébrés s'est révélé très intéressant pour l'étude des araignées (*cf.* Manuscrit N°7), l'étude des carabes en bords de route est apparue quant à elle décevante à plusieurs niveaux :

- La variabilité des captures entre 2006 et 2007 n'a pas permis de dégager des résultats d'une année sur l'autre même si d'autres études ont été menées avec succès sur les bordures de champ comme habitat (Woodcock et al. 2005, Griffiths et al. 2007) ou corridor (Martin et al. 2001), des bords de route comme habitat (Eversham et Telfer 1994, Noordijk 2008) et des pollutions aux métaux lourds auxquelles les carabes sont sensibles (Emets et Zhulidov 1983) ;
- Le nombre d'espèces (ou taxons) capturées s'est avéré décevant, 27 au total, apparaissant comme assez bas et en-dessous à ce que l'on pourrait s'attendre en bords de route (Pr. Michel BAGUETTE, communication personnelle). Ces résultats pourraient s'expliquer par le fait que certains auteurs considèrent parfois les bords de route comme un puits pour les carabes (Koivula 2005) ;
- La capture et la détermination des carabes se sont révélées longues et laborieuses : plusieurs passages sur sites nécessaires, gestion difficile de près de 250 pots avec des transferts d'individus dans l'alcool, et longue détermination à la loupe binoculaire de plus de 4.000 individus (1.823 en 2006 et 2.291 en 2007).

Le piégeage s'est cependant révélé très efficace pour les petits mammifères (*cf.* Manuscrit N°2 et Manuscrit N°4). Les analyses ont montré que les petits mammifères (musaraignes et campagnols dans notre étude) réagissent très fortement aux conditions écologiques du milieu (connectivité, fragmentation, occupation des sols, qualité des habitats) et pourraient donc constituer de bons indicateurs des milieux agricoles. Cependant et malgré cette réussite en terme de piégeage et d'analyses, certaines limites conviennent d'être apportées à l'utilisation d'un tel protocole pour des suivis de petits mammifères :

- Il existe une vraie question éthique compte tenu du fait que le piégeage est destructeur. Il conviendrait, à la vue des résultats probants obtenus, de développer un protocole non létal d'étude des petits mammifères ;
- L'attractivité des pièges n'a pas été testée rigoureusement et même si ceux-ci se veulent non attractifs (éthylène glycol employé à la place de l'alcool) ; il pourrait exister des effets non désirés comme l'accumulation d'insectes piégés dans les pièges (en place pendant un mois) sur la capture des musaraignes qui sont des insectivores.
- Il a tout de même été testé l'effet « nombre de carabes piégés » sur la variable « nombre de musaraignes piégées » au niveau des pots et les résultats se sont révélés non significatifs ($F_{1,240} = 0.001$ et $P = 0.977$) ; cependant les carabes, insectes déterminés et comptés, ne représentent qu'une partie des invertébrés piégés (nombreux autres insectes, araignées et gastéropodes présents dans les pièges) ;
- Le nombre d'espèces (sept en bord de route et d'autoroute) et le nombre d'individus piégés (souvent moins d'une dizaine au total par espèce sauf pour *Sorex coronatus*, *Crossidura russula* et *Microtus arvalis*) sont faibles, et même si ces chiffres enregistrés en bords de route sont plus importants que ceux obtenus dans les autres milieux, ils n'étaient pas suffisants pour le fonctionnement de ComDyn (pour lequel un minimum de sept espèces par site est nécessaire avec au moins un individu par répliqua) ; ce qui n'a pas permis (a) de calculer des taux de détectabilité pour les différentes espèces piégées et (b) de travailler sur des richesses corrigées (ce qui était l'objectif recherché avec les cinq répliquas par site).

Si les résultats obtenus sont prometteurs quant à l'utilisation des petits mammifères comme indicateur, la recherche d'un protocole plus adapté semble nécessaire : (a) plus respectueux des individus piégés, (b) avec des pièges dont l'attractivité potentielle déterminée, (c) avec des répliquas repensés par rapport à d'autres outils que ComDyn et (d) permettant une meilleure capture des mulots.

§ B. IMPACTS NEGATIFS DES ROUTES SUR L'ENVIRONNEMENT

Si les effets des bords de route étudiés lors de ces travaux de recherches ont essentiellement porté sur des aspects positifs, effets « habitat » et « corridor » pour les plantes et les petits mammifères, il ne faut cependant pas oublier la part de responsabilité importante des infrastructures de transport dans la crise de la biodiversité.

(1) Destruction et fragmentation des habitats

Les effets négatifs des routes en termes de destruction et de fragmentation des habitats ont peu été montrés lors de cette étude (sauf les effets « barrière » de la route pour les musaraignes et « appauvrissement » pour les bois). Cela peut être principalement dû à deux causes :

- L'étude s'est concentrée principalement en milieu agricole à cause des enjeux importants qui y sont associés en termes de conservation de la biodiversité (*cf.* Introduction). Or, depuis de nombreuses années, les haies, ou autres structures semi-naturelles, ont largement disparu de ces espaces sous l'effet de l'intensification des pratiques agricoles. Dans ce contexte, les constructions de route consomment peu d'espaces naturels et participent de manière très relative à la fragmentation de milieux qui le sont déjà fortement. Au contraire, les routes permettent la mise en place de zones refuges et de continuités. Les résultats de cette étude ne sont donc appréhendables que dans ce contexte et il est clair que les résultats seraient tout autres si l'étude avait été menée sur les effets des routes au sein d'espaces naturels (Forman et Alexander 1998, Coffin 2006) ;
- Les résultats présentés ne concernent que deux groupes : plantes et petits mammifères. Des effets négatifs des routes et de leurs accotements sur les communautés d'oiseaux ont été montrés (Van der Zande et al. 1980, Johnson 1996, Reijnen et Foppen 1994, Canaday 1996). Par ailleurs, les quelques résultats obtenus sur les carabes ont montré en 2006 : (a) l'existence d'une corrélation négative entre indices de diversité (*Hill*) et teneurs en plomb des sols des bords de route (t = 3.304 et P = 0.00193) avec quelques espèces malgré tout favorisées (peut-être par simple relâchement de compétition) comme *Pterostichus melanarius* et *Anchomenus dorsalis*, *cf.* rapport de stage d'Elise CONTAN, stagiaire de M1 (www2.mnhn.fr/cersp

/IMG/zip/Contan_2007.pdf.zip) ; et (b) l'existence d'un impact de la fragmentation du milieu sur la taille moyenne des carabes piégés (P = 0.002) avec les espèces de taille importante (*Carabus auratus*, *Carabus monilis*, etc.) disparaissant des milieux présentant des indices de fragmentation élevés, *cf.* rapport de stage de Johanna ROILLET, stagiaire de M1 (www2.mnhn.fr/cersp/IMG/zip/Roillet _2007.pdf.zip).

Ces résultats sur les carabes sont cependant à prendre avec prudence dans la mesure où, ces mêmes analyses réalisées sur les données de 2007 n'ont pas confirmé ces tendances. Ces différences pourraient s'expliquer par (a) la difficulté à étudier les insectes sur des pas de temps courts, comme deux années, alors que les effectifs des populations de carabes sont fluctuants d'une année sur l'autre et liés aux aléas climatiques (Baars et Van Dijk 1984, Wallin 1985, Honek 1997) ; 2006 ayant été très différente de 2007 notamment en début de printemps (mars « pluvieux », « froid » et « mal ensoleillé » en 2006 et « normalement arrosé », « doux » et « ensoleillé » en 2007 en Seine-et-Marne, sources *Météo France* : http://climatheque.meteo.fr); (b) les effets du protocole d'échantillonnage qui prélève par destruction de nombreux individus (*cf.* Table N°5) au sein d'un groupe, les Carabidae, aux capacités de dispersion limitées à court terme en milieu fragmenté, surtout au niveau des espèces ne disposant pas d'ailes importantes (Den Boer 1970, Frampton et al. 1995).

Il est intéressant de noter, par ailleurs, que si les bords de route permettent une augmentation locale de la biodiversité (diversité α), nous avons montré qu'elles induisaient aussi une homogénéisation biotique du milieu (baisse de la diversité β), *cf.* Manuscrit N°1.

Table N°5 :

Carabidae	**2006**		**2007**	
	Nombre	**Erreur standard**	**Nombre**	**Erreur standard**
Taxons	28.00	-	17.00	-
Taxons par pot	*1.97*	*0.10*	*1.82*	*0.09*
Individus	1823.00	-	2291.00	-
Individus par pot	*7.50*	*0.95*	*9.16*	*1.58*

Bilan des captures de carabes en bords de route (53 sites)

(2) Dispersion d'espèces invasives

La connectivité mise en place par les dépendances vertes des bords de route en milieu agricole permet certes la dispersion d'espèces invasives et éventuellement leur maintien au sein des paysages (métapopulations et métacommunautés) mais elle permet aussi la dispersion et la diffusion d'espèces exotiques et/ou invasives (*cf.* Introduction). Cette étude de la biodiversité des bords de route n'a cependant pas révélé la présence de telles espèces dans les relevés effectués, en particulier chez les plantes, malgré la présence avérée en Seine-et-Marne de certaines d'entre elles (*Sources :* http://cbnbp.mnhn.fr) comme l'ailanthe (*Ailanthus altissima)*, la renouée du Japon (*Reynoutria japonica*), la solidage du Canada (*Solidago canadensis*) et le séneçon du Cap (*Senecio inaequidens*). Cela pourrait être dû essentiellement à deux causes :

- Un protocole non-adapté à la détection de ces espèces dans la mesure où leur répartition se fait rarement sur un front continu mais plutôt sous forme de « tâches » éparses en bords de route (Pysek et Hulme 2005) et, donc, nécessitant un ciblage pour les échantillonner alors que nos relevés étaient tirés au hasard ;
- Une détermination difficile liée au fait que ces plantes invasives se trouvent souvent en toute bordure de route, dans la zone dite « de sécurité », là où la fauche est très intensive, précoce et systématique réduisant généralement les plantes à un certain état de mutilation et à une taille de quelques centimètres.

L'existence et les impacts de ces plantes en bords de route ne doivent cependant pas être oubliés à la lecture de ce manuscrit.

§ C. IMPORTANCE DES BORDS ROUTES EN MILIEU AGRICOLE INTENSIF

Si les routes ont des effets négatifs importants sur la biodiversité, cette étude a montré que leurs dépendances vertes permettaient la création (a) de zones refuges, habitat pour une biodiversité exclue de la matrice agricole, et (b) de continuités biologiques, tout en favorisant le maintien de services écosystémiques. Il ne s'agit pas de dire que ces effets peuvent en compenser d'autres. Il s'agit de montrer que, dans la mesure où les routes existent, un certain nombre de mesures peuvent être mises en place pour la biodiversité.

(1) Mise en place de zones habitats et refuges

Si les bords de route créent des zones « refuge » ou « habitat », ce ne sont pas les seuls éléments des paysages agricoles à offrir cela à la biodiversité. L'impact des bordures de champ est aussi loin d'être négligeable (bandes enherbées avec ou sans haies) : elles peuvent aussi représenter des surfaces importantes et constituer des zones « refuge » ou « habitat » pour la biodiversité, participant aussi à son maintien au sein des paysages agricoles intensifs (Marshall et Moonen 2002, Benton et al. 2003, Abadie 2008).

Des différences existent cependant entre bords de route et bordures de champ qui ne sont pas en bords de route. Cela a été montré pour les micro-mammifères (*cf.* Manuscrit N°2) mais cela est aussi vrai pour la flore. En effet, une étude (réalisée lors de mon stage de Master 2 sur la diversité végétale des bordures de champ) a montré que le premier facteur explicatif de la composition des communautés végétales était le facteur « bordure de champ en bords de route » (par rapport aux bordures situées entre deux champs et devant toute une série de modes de gestion opérées par les agriculteurs : fauche, broyage, herbicides, etc.) : l'ACP réalisée sur les stations et leur composition floristique a révélé une ségrégation entre bordures de champ à l'interface champ/chemin, et bordures de champ à l'interface champ/route ; les bordures à l'interface champ/route étant par ailleurs les plus riches en terme de diversité spécifique (*cf.* Manuscript N°5). Ces différences entre bords de route (interface route/champ) et bordures de chemin (interface champ/chemin) peuvent s'expliquer par un certain nombre de paramètres : impacts des routes (pollution, trafic, conditions microclimatiques, etc.), de la structure de l'emprise (largeur, présence de haies) et des modes de gestion (fauche, salage, etc.) et montrent l'intérêt de travailler sur les deux types de structures pour une meilleure appréhension des agro-écosystèmes.

(2) Mise en place de continuités biologiques

La mise en place de continuités biologiques par les bords de route a clairement été mise en évidence au cours de cette étude. Nos résultats ont confirmé que la notion de « corridor », ou de continuité biologique, était bien dépendante des espèces étudiées (Beier 1998) :

- L'homogénéisation biotique observée dans les bois connectés au niveau des communautés végétales (*cf.* Manuscrit N°1) a montré que seules certaines plantes bénéficiaient de la présence des bords de route pour leur dispersion (essentiellement des plantes semi-forestières non totalement inféodées au milieu forestier, *ex. : Stellaria holostea*, *Cruciata laevipes* ou *Ranunculus auricomus*). Afin d'affiner ces résultats, il pourrait être intéressant de poursuivre le travail de recherche sur le plan génétique au niveau de certaines populations de plantes au sein de bois connectés et non-connectés aux bords de route à l'aide de marqueurs génétiques afin de pouvoir affirmer avec certitude l'existence d'effets corridors ;

- Les patterns de piégeage différents entre micro-mammifères ont permis de montrer que les bords de route pouvaient permettre la dispersion des musaraignes mais pas celle des campagnols (*cf.* Manuscrit N°2). Un regret au sujet de cette étude est de ne pas avoir déterminé le sexe des individus piégés en bords de route. En effet, chez les musaraignes, animaux territoriaux, deux raisons peuvent provoquer le départ temporaire du territoire : la recherche de nourriture pour les mâles comme les femelles (Oxley *et al.* 1974), ou la recherche de partenaire sexuel pour le mâle uniquement (Cantoni 2002). Une étude du sexe ratio aurait donc pu nous renseigner sur les raisons pouvant expliquer la dispersion des musaraignes en bords de route. Enfin, une étude complémentaire, par piégeage au mois de septembre, permettrait de savoir si les jeunes musaraignes dispersent le long des bords de route à la recherche de nouveaux territoires et ne traversent pas la structure routière à cette même fin, comme cela est le cas pour les adultes.

La structuration des corridors est importante (Beier et Noss 1998) et cela a aussi pu être observé lors de l'étude : la largeur de l'emprise semble être un élément capital du fonctionnement de la structure pour la dispersion des musaraignes (*cf.* Manuscrit N°2).

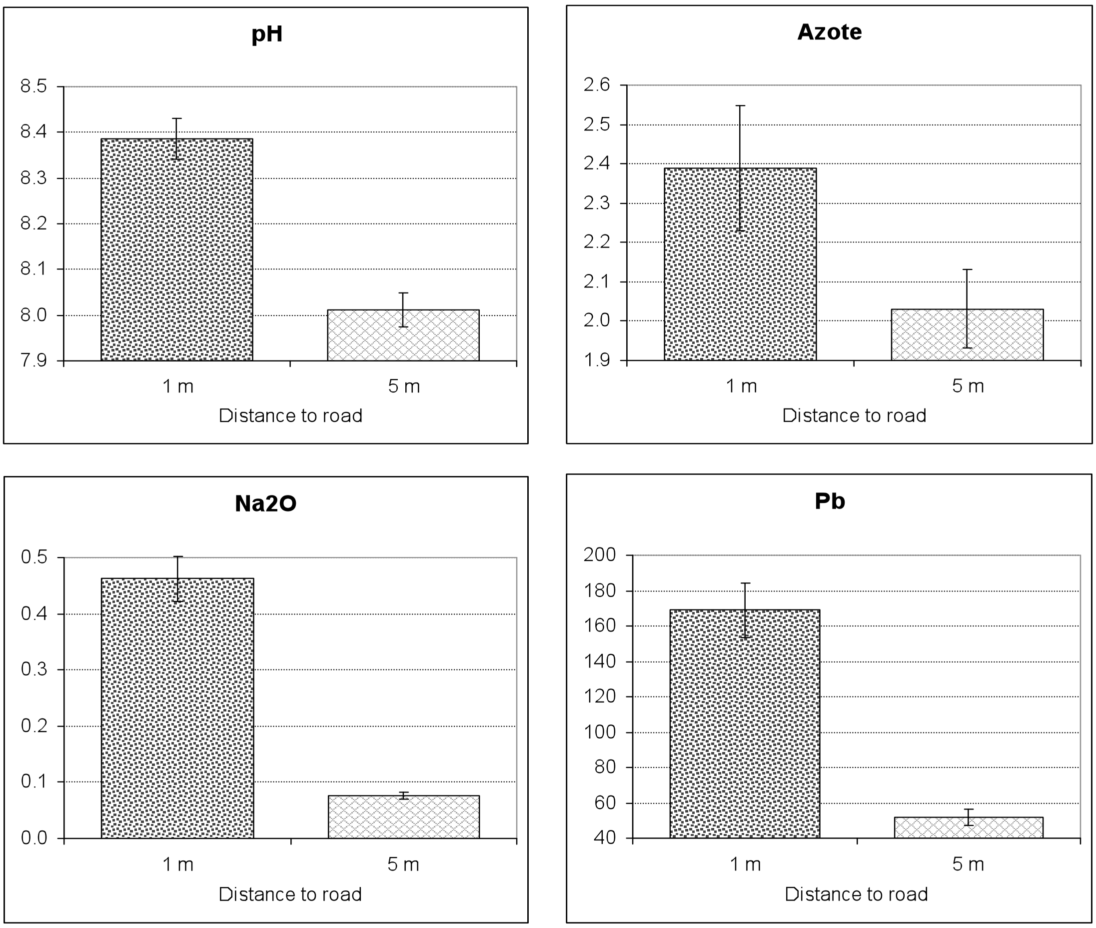
§ D. IMPACTS DES STRUCTURES ROUTIERES

Si la route a de nombreux effets négatifs sur l'environnement, il convient donc, dans la mesure où celles-ci existent, de favoriser les impacts potentiellement positifs de ses dépendances vertes. Il est tout d'abord possible de travailler sur la structure même de l'emprise. En effet, les différences observées pour la biodiversité au niveau des différents types d'emprises, bords d'autoroutes, bords de route et bordures de champ, montrent combien la structure de l'emprise est importante.

(1) Effets de la largeur de l'emprise

La largeur de l'emprise semble avoir un effet important sur l'environnement et la biodiversité : bien qu'une certaine hétérogénéité puisse exister entre les modes de gestion d'une emprise à une autre ; bords d'autoroutes (largeur, *l* = 15.7 ± 6.6 m), bords de route (*l* = 7.2 ± 3.4 m) et bords de champ (*l* = 4.0 ± 0.4 m) diffèrent fortement par la taille de leurs emprises respectives.

Figure N°10



Résultats des analyses de sols de bords de route en Seine-et-Marne (58 sites)

Notre étude a montré, notamment pour *Sorex coronatus*, que les bords d'autoroute pouvaient constituer un habitat suffisant, alors que les bords de route servaient davantage pour leur dispersion et que la musaraigne était quasi-absente des bords de champs (*cf.* Manuscrits N°2). Pour les populations de campagnols des champs, *Microtus arvalis*, l'importance des emprises semblent avoir un effet important sur la régulation des populations sujettes à des explosions démographiques préjudiciables aux cultures (*cf.* Manuscrit N°4). Enfin, les différences observées entre bords de route et bordures de champ, au niveau des communautés végétales (*cf.* Manuscrit N°5), pourraient peut-être s'expliquer par les pollutions émises depuis la route et par les modes de gestion différents des emprises (*cf.* Introduction) ; mais ces deux facteurs sont corrélés à la largeur de la structure.

En effet, premièrement, les deux premiers mètres des bords de route sont fauchés intensivement par mesure de sécurité (visibilité, arrêts d'urgence, etc.) ; il apparaît donc évident qu'un bord de route de moins de deux mètres de large ne sera pas de largeur suffisante pour une politique de gestion prenant en compte la biodiversité. Deuxièmement, la pollution des sols diminuent fortement avec l'éloignement à la route (*cf.* Introduction).

Nos résultats ont confirmé ces observations (*cf.* Figure N°10) : nos analyses de sols à 1 m et 5 m de la route ont révélé un fort effet de la distance à la route sur le pH (test t d'égalité des espérances, observations pairées, $t = 6.065$ et $P < 10^{-3}$), sur les teneurs en azote ($t = 2.423$ et $P = 0.020$), Na_2O ($t = 10.044$ et $P < 10^{-3}$) et en plomb ($t = 8.470$et& $P < 10^{-3}$) dans les sols.

La largeur de l'emprise est donc déterminante pour la biodiversité en bords de route. Il est important de le souligner car les récentes politiques prises en matière de développement de bandes enherbées dans le cadre des *Mesures Agri-Environnementales* (M.A.E.) de la *Politique Agricole Commune* (P.A.C.) ont imposé une largeur minimale de 5 mètres pour l'attribution de subventions depuis 2002. Cette largeur semble, à la vue des résultats obtenus au cours de cette thèse, une bonne base de travail car elle peut permettre une conservation effective de la biodiversité et le maintien de services écosystémiques par :

- Le maintien des populations de musaraignes qui sont des prédateurs d'insectes et de gastéropodes pouvant être des ravageurs des cultures (espèces listées d'intérêt communautaire dans l'annexe III de la Convention de Berne, *cf.* Manuscrit N°2) ;
- La stabilisation des populations de campagnols par le maintien de zones favorables à leurs prédateurs qui peuvent accèder aux campagnols dans les champs en utilisant les bords de route et les bordures de champ (*cf.* Manuscrit N°4) ;
- Le maintien d'un habitat de qualité pour les insectes pollinisateurs en bordure de champ : une gestion extensive des bords de route, possible au sein des emprises importantes, entraîne une augmentation des plantes zoochores (*cf.* Manuscrit N°3) pouvant permettre le maintien des communautés d'insectes grâce aux apports de ressources en pollen et en nectar (Kohler et al. 2006) ;

- La création d'une zone tampon entre la route et les cultures (à vocation essentiellement alimentaires) : notre étude a montré, par exemple, que les teneurs de plomb détectées en bords la route dépassent 100 $\mu g.g^{-1}$, valeur limite selon l'Arrêté du 8 janvier 1998 et entraînant une obligation de dépollution dans certains pays comme la Suisse (www.admin.ch/ch/f/rs/814_12/app1.html), dans 72,1% des prélèvements à 1 mètre de la route et dans seulement 7.0 % des prélèvements à 5 mètres.

La viabilité de telles mesures est assurée à travers un système de subventions complexes et que les attentes en terme d'évaluation de leurs impacts est un attendu fort et actuel compte tenu des coûts générés : 450 € par hectare de bande enherbée. Dans le même temps, les pertes en production liées à la réduction de l'espace cultivable sont d'environ 161 $€.ha^{-1}.an^{-1}$ pour l'agriculteur (*Sources :* Chambre d'Agriculture de Rhône-Alpes www.rhone-alpes.chambagri.fr/phytov3/pages/bande_herbe.htm), permettant ainsi en bout de chaîne un bénéfice annuel total de 95 € par hectare de bande enherbée pour l'exploitant agricole selon Desbois et Legris (2005), bénéfice financé par la *P.A.C.*

(2) Effets des plantations

L'étude a été centrée sur le réseau des routes nationales et départementales qui sont des infrastructures anciennes et construites à une époque où les enjeux environnementaux n'étaient pas encore connus, ou du moins, n'étaient pas une priorité. Aucun effort particulier n'avait donc été porté à la mise en place de leurs dépendances vertes dont la végétation est globalement issue de processus de colonisation depuis les milieux adjacents ou lds graines apportées par les voitures. Il s'agit d'une végétation qui pourrait être qualifiée de « spontanée ».

Cela n'est pas le cas des autoroutes dont les constructions ont commencé à la fin des années soixante et se poursuivent actuellement. Compte-tenu de leur importance (en termes d'emprises), de la rapidité de construction et des moyens financiers mis en place, les dépendances vertes font l'objet de véritables aménagements. Ces aménagements ont été, jusqu'à très récemment, dictés par des contraintes absolues d'économies financières (construire le « *moins* chère possible » au lieu de construire le « *mieux* chère possible ») et d'esthétisme (création de « paysages autoroutiers »). Les ensemencements à partir de cocktails de graines et les plantations de haies constituent la structure de ces aménagements. Il serait intéressant d'étudier leurs impacts sur la biodiversité afin de savoir s'ils peuvent aussi participer à son maintien et à quelles conditions : faut-il des haies de manière continue le long des routes ou laisser certains espaces ouverts ? Les cocktails de graines orientent-ils la structure de certaines communautés ou l'influence du milieu adjacent est-elle le facteur déterminant ?

Les réponses apportées à ces questions pourraient permettre de quantifier l'importance de ces investissements pour ensuite, peut-être, essayer de les mettre en place au niveau de tout le réseau routier s'il s'avère que les effets sont bénéfiques pour la biodiversité : plantation de haies le long des routes et ensemencement des accotements après chaque dérasement (le dérasement est une opération d'entretien décennale qui consiste à remettre les bords de route à niveau pour permettre le bon écoulement des eaux de la chaussée).

S'il n'a donc pas été possible d'étudier ces effets, effet « haies » et effet « semis », durant ma thèse, il convient néanmoins de rappeler quelques résultats connus. Au niveau de la végétation, si il a ainsi été prouvé que les semis avaient peu d'impacts, à terme, sur la composition des communautés (De Cauwer et al. 2005), il a en revanche été montré que les haies (a) permettent la conservation d'une flore plus diversifiée qu'en milieux ouverts sans toutefois défavoriser certaines espèces (*cf.* Manuscrit N° 7), et (b) constituent de véritables habitats pour les plantes forestières (McCollin et al. 2000).

Il pourrait être intéressant de réaliser les mêmes observations que celles effectuées dans la première étude (*cf.* Manuscrit N°1) en présence de haies en bords de route afin de savoir si l'effet « corridor » mis en évidence serait renforcé et limiterait, ou non, l'effet homogénéisant des routes ; des effets « corridor » ayant par ailleurs déjà été montrés pour les plantes forestières au niveau de haies (Davies et Pullin 2007)

Enfin, la présence de haies (a) assure la mise en place de communautés d'araignées originales en bords de route, différentes de celles observées en leur absence, et (b) permet ainsi la conservation de la diversité fonctionnelle au sein de l'écosystème (*cf.* Manuscrit N°7).

§ E. IMPACTS DES MODES DE GESTION

La structure de l'emprise n'est pas le seul paramètre ayant des effets sur la biodiversité, les modes de gestion sont aussi importants. Si les traitements chimiques semblent avoir disparus, du moins officiellement, la fauche intensive reste la règle et l'exportation des déchets n'est pas encore d'actualité. Les agriculteurs interviennent aussi sur les bords de route, même illégalement, puisque ceux-ci sont à l'interface de leurs champs.

(1) Impacts de la fauche

L'un des objectifs de cette thèse, sur le plan appliqué, était d'étudier les différentes modalités de gestion afin de trouver les plus respectueuses de la biodiversité. La majorité du terrain a été construit autour de cette question avec 43 stations échantillonnées à cet effet (calendriers de gestion différents, âges des routes variables, intensités de trafic distincts).

Les effets de ces paramètres ont été très difficiles à détecter au niveau des communautés végétales (*cf.* rapport de stage de Noémie VARET, stagiaire de M1 : www2.mnhn.fr/ Crsp/IMG/zip/Varet_2007.pdf.zip) comme carabiques (résultats obtenus en 2006 non vérifiés en 2007), *i.e.* au niveau des deux groupes étudiés.

Une importante variabilité interannuelle a été observée pour les deux groupes : une réduction importante du nombre d'espèces observées chez les végétaux (180 en 2006 contre seulement 126 en 2007) avec cependant une relative stabilité des plantes communes (sur les 97 espèces représentant 95% du recouvrement en 2006, 79 ont été détectées à nouveau en 2007 et représentaient encore 92% du recouvrement) ; et un piégeage carabique qui a semblé finalement assez aléatoire en 2007 malgré des résultats prometteurs en 2006 (effets négatifs de la « fauche de fin de printemps », *i.e.* la deuxième fauche de l'année située entre mai et juin selon les sites étudiés, sur les populations carabiques, nombre d'individus piégés, $F_{1,43} = 4.146$ et $P = 0.0481$, *cf.* rapport de M1 d'Elise CONTAN cité précédemment : www2.mnhn.fr/cersp/IMG/zip/Contan_2007.pdf.zip).

Pour la végétation, il a tout de même été détecté un léger effet non-significatif du calendrier de fauche au niveau des modes de dissémination du pollen avec un enrichissement des communautés végétales en plantes entomogames associé à l'allongement des délais entre aussi bien les deux premières fauches, *i.e.* entre celle de début de printemps en mars/avril et celle de fin de printemps en mai/juin ($F_{1,0} = 3.183$ et $P = 0.082$), qu'entre les deux dernières, *i.e.* entre celle de fin de printemps et celle d'été en août/septembre ($F_{1,40} = 2.964$ et $P = 0.093$). Le trait « dissémination du pollen » est le seul à avoir montré une telle tendance face aux modes de gestion des bords de route parmi tous ceux testés (« hauteur de floraison », « dispersion des graines », « mode de reproduction végétative », « durée de floraison », ou encore « phénologie »).

Cette légère tendance à l'enrichissement des communautés végétales en plantes entomogames liées à une extensification des pratiques de fauche est en accord avec les résultats obtenus lors de l'expérimentation du « fauchage tardif » où là cependant, les résultats deviennent significatifs (*cf.* Manuscrit N°3).

De manière générale, il semble donc que travailler sur le calendrier de fauche, sans changer les pratiques en profondeur, ne soit pas suffisant pour permettre une modification des communautés végétales vers des ensembles plus riches dotés d'une plus grande diversité fonctionnelle.

Cela a été confirmé par les travaux réalisés sur les bordures de champ où l'on peut voir (a) que le facteur déterminant pour les communautés végétales est la présence d'une route à l'interface (*cf.* Manuscrit N°5) et, et (b) que les modes de gestion ont aussi un impact relativement faible sur les communautés végétales quand ceux-ci sont finalement assez semblables, *i.e.* variabilité de la gestion dans un système restant malgré tout intensif (*cf.* Manuscrit N°6).

Ces résultats semblent en accord avec des études ayant montré le peu d'impact des mesures de gestion sur les communautés végétales des bords de route/bords de champ (Kleijn et Verbeek 2000) alors que finalement les dynamiques de plantes « adventices » sont considérées comme chaotiques par certains auteurs (Gonzalez-Andujar 1996) ; ce qui est par ailleurs critiqué (Freckleton et Watkison 2002).

Les résultats obtenus avec l'expérimentation « fauchage tardif » (*cf.* Manuscrit N°3) d'une part, et l'absence de résultats avérés avec l'étude des effets du calendrier de fauche et avec les études réalisées en bord de champs (*cf.* Manuscrits 5 et Manuscrit 6), d'autre part, montrent clairement qu'une meilleure prise en compte de la diversité des bords de route passe par un changement global des pratiques, comme le passage de trois à une fauche, et non par une adaptation à la marge de celles-ci, comme l'avancement de deux à trois semaines de la deuxième fauche de printemps, fauche qui semble être la plus critique.

Ces résultats sont malgré tout à prendre avec prudence et une poursuite de l'étude reste nécessaire dans la mesure où :

- Le pas de temps d'étude est faible (deux à trois ans) : (a) pour les insectes, il est trop court pour dégager de réelles tendances (résultats différents entre 2006 et 2007) ; et (b) pour la végétation, il a été montré que les communautés répondaient fortement à l'extensification des pratiques et étaient en changements en ne semblant pas stabilisées au bout de trois ans. Il faudrait voir si la réorganisation des communautés autour de nouveaux cortège floristiques n'entraîne pas, à terme, des communautés différentes mais pas nécessairement plus riches (Parr et Way 1988, Sykora et al. 2002) ; un fauchage différencié pourrait être alors une solution au maintien de différentes communautés en bords de route ;

- Seuls deux groupes ont été étudiés (plantes et carabes) alors que d'autres sont aussi susceptibles de subir les changements des modes de gestion des bords de routes : araignées, petits-mammifères, oiseaux, etc. (*cf.* Introduction).

(2) Impacts de l'exportation des déchets

Cette opportunité de travail peut s'organiser autour de l'étude d'autres facteurs qui n'ont pu, faute de moyens, être étudiés au cours de ces travaux de recherche :

- L'exportation des déchets est désormais envisageable en Seine-et-Marne ce qui ne l'était pas il y a trois ans alors qu'il s'agit d'un point critique pour l'étude de la diversité végétale des bords de route (De Cauwer et al. 2005). En effet, les bords de route constituent un lieu d'emmagasinement de nutriments notamment azotés, et l'exportation des déchets organiques issus de la fauche peut participer au rééquilibrage des sols (Berense et al. 1994, Schaffers et al. 1998) ;
- L'excavation des sols des bords de route serait aussi à étudier mais certainement moins envisageable pour le moment compte-tenu des coûts très importants d'une telle opération. Il n'en demeure pas moins que les sols des bords de route se sont révélés être hautement pollués (*cf.* Partie § C. (1) de la Discussion) ; en accord avec les études déjà menées sur le sujet (Thompson et al. 1986, Rutter et Thompson 1986, Carslaw 2005). De telles concentrations en azote, sels et métaux lourds ne sont pas sans conséquences sur la biodiversité : microbes du sol (Post et Beeby 1996), plantes (Akbar 2003, Singh et al. 2003, Johnston et Johnston 2004, Truscott 2005) et insectes (Alstad et Edmunds 1982, Spencer et al. 1988, Throop et Lerdeau 2004).

La question des sols a aussi été abordée lors de notre étude mais malgré des premiers résultats encourageants (*cf.* le rapport de M1 d'Amélie DELERUE pour les plantes : www2.mnhn.fr/cersp/IMG/zip/Delerue_2008.pdf.zip, et *cf.* le rapport de M1 d'Elise CONTAN cité précédemment pour les carabes :, www2.mnhn.fr/cersp/IMG/zip/ Contan_2007.pdf.zip), il n'a pas été possible de pousser des investigations plus loin : tous les sols étant très pollués, il était difficile de dégager des résultats plus intéressants.

PERSPECTIVES

« *Le gain de la recherche, c'est la recherche elle-même* ».

GREGOIRE DE NYSSE

L'étude des effets écologiques des bords de route s'est révélée très intéressante. Elle implique le travail avec de nombreux acteurs : politiques (Ministère et Conseil général), gestionnaires des routes (Direction Départementale de l'Equipement, Service « Route » du C.G., bureaux d'étude et agriculteurs *de fait*) et encore d'autres acteurs (associations naturalistes, citoyens et scientifiques). L'attente est forte auprès de ces différentes personnes au niveau du développement (a) de méthodes d'évaluation des impacts de la construction et de la gestion des infrastructures, et (b) de méthodes de suivis sur le long terme des mesures mises en place pour la biodiversité.

§ A. NECESSAIRE POURSUITE DE L'ETUDE DES EFFETS DES ROUTES SUR LA BIODIVERSITE

Il s'agit du moment difficile où l'impression du travail inachevé domine (études à poursuivre ou à lancer, données à analyser, résultats à valoriser et articles à écrire) alors qu'il faut mettre un point final à trois ans de travaux et passer à autre chose. Alors qu'il reste beaucoup à chercher en écologie des routes.

En effet, de manière générale en écologie, et plus particulièrement au sein de l'U.M.R. 51 73, les travaux sur les routes sont nouveaux. Cette thèse, ainsi que celle d'Isabelle LE VIOL, constituent une première étape dans ce champ d'étude de l'écologie assez proche de l'écologie urbaine (il s'agit de milieux fortement anthropisés, perturbés et pollués). De nombreuses questions se posent encore sur le sujet, et comme les enjeux liés (a) aux espaces marginaux tels que les bords de route au sein de paysages agricoles intensifs, et (b) à la compréhension des impacts des actions anthropiques sur la biodiversité, constituent des enjeux majeurs en terme de recherche pour les années à venir ; la poursuite de ces travaux semblent être une nécessité.

§ B. NECESSAIRE COMMUNICATION ET VULGARISATION DES SAVOIRS SCIENTIFIQUES

Si davantage de recherches sont nécessaires, il n'en reste pas moins que les connaissances scientifiques sont aujourd'hui suffisantes pour affirmer qu'il existe des impacts très forts des routes sur la biodiversité et qu'il est possible de les réduire par des politiques de construction et de gestion appropriées. La prise en compte de la biodiversité dans toute politique est désormais un enjeu majeur pour les années à venir.

Ce défi ne pourra être relevé qu'à travers la sensibilisation de la société à la conservation de Nature, et de toute la Nature : Nature ordinaire comprise. Si la prise de conscience sociétale autour de la question de la Nature protégée semble désormais en bonne voie, la question de la Nature ordinaire est encore largement ignorée. Il est vrai qu'un sujet sur les bords de route, *« Intérêts écologiques des bords de route… »,* peut prêter à sourire, malheureusement, chez beaucoup de personnes pourtant sensibilisés à la nécessaire conservation de la biodiversité. Le sourire est à la mesure par lui-même du travail à accomplir sur ce sujet.

Pour conclure ces travaux de thèse, je me permets de dire que cette situation m'a fait penser à celle de Francis PONGE (1899-1988). Poète du *parti pris des choses*, il n'a eu de cesse d'essayer et d'opérer un retournement complet de la poésie par rapport à une conception romantique alors à la mode. Un texte, tout ce qui peut paraître de plus ordinaire à la première lecture (à tel point que le lecteur s'en demande franchement l'utilité), lorsqu'il est décrypté révèle sous une comparaison incongrue entre un tramway et la guêpe, toute l'utilité de la Nature. Poète de l'ordinaire, loin de tout sentimentalisme, Francis PONGE révèle l'utilité, et par là-même la beauté, des objets et des mots les plus ordinaires par leurs qualités physiques et recourt volontiers au vocabulaire technique des sciences expérimentales ; signalant même à plusieurs reprises sa dette envers Buffon. Une reconnaissance qui peut sembler prémonitoire à l'heure où ce sont des *disciples* de Buffon, écologues de la rue du même nom, qui tentent désormais de faire reconnaître à leurs contemporains la nature ordinaire comme utile, nécessaire et donc belle ; bien que semblant sans intérêt au premier abord.

L'étude de la Nature ordinaire est indispensable à l'écologie et à l'Homme tout comme les mots ordinaires le sont à la pensée et à l'expression comme Francis PONGE le montra à travers sa poésie prosaïque.

ANNEXES

MANUSCRITS COMPLÉMENTAIRES

&

GUIDE TECHNIQUE A DESTINATION
DU M.E.E.D.D.A.T.

> *« Quel peut donc être notre intérêt à courir si nous nous trouvons sur la mauvaise route ? »*
>
> PROVERBE

MANUSCRIPT N°5

In preparation

Effects of local environment and human activities on plant communities of field margins in an intensive agrarian landscape

Louis de REDON
Aurélie GARNIER
Jane LECOMTE
Robert HAICOURT
Jean-Michel DREILLAUX
Sandrine PIVARD
&
Agnès RICROCH

PRESENTATION

TITRE

Impacts des conditions locales et des activités humaines sur les communautés végétales des bords champ en paysage agricole intensif

RESUME

Nous avons étudié les impacts de l'organisation des exploitations agricoles et des pratiques des agriculteurs qur les communautés végétales des bords de champ au sein d'un paysage agricole intensif français. Des inventaires de végétation exhaustifs ont été menés au sein de 83 bordures de champ et un total de 187 taxons identifiés. Nous avons sélectionné 11 paramètres pour expliquer la richese et la diversité fonctionnelle des communautés végétales des bords de champ : des caratéristiques de l'exploitation (taille, nombre d'employés, remembrement), la localisation des bordures (adjacentes ou pas à des routes) et des pratiques de traitements des bordures (type de traitements et calendrier). Il est apparu que les 16 agriculteurs contactés, lors d'une enquête que nous avons menée, avaient des modes de gestion intensifs de leurs bordures de champ et que peu de variabilité existait en fait entre les différentes modalités de traitement. Malgré ce, nos résultats ont montré que les richesses spécifiques et fonctionnelles des communautés végétales des bords de champ étaient affectées par quelques paramètres : les bordures adjacentes aux routes étaient plus riches et le fauchage intensif favorisait les plantes à floraison tardive (automne).

MOTS CLEFS

ACP, Agriculteurs, Biodiversité, Bords de route, Politiques environnementales, Traits fonctionnels.

EFFECTS OF LOCAL ENVIRONMENT AND HUMAN ACTIVITIES ON PLANT COMMUNITIES OF FIELD MARGINS IN AN INTENSIVE AGRARIAN LANDSCAPE

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KEYWORDS

Biodiversity
Environmental policies
Farmers
Functional traits
PCA
Road verges

ABSTRACT

We studied the impacts of farm structure and farmer practices on field margins plant communities in a French intensive agrarian region. An exhaustive evaluation of species richness was carried out in 83 field margins and a total of 187 taxa were identified. We selected 11 parameters to explain richness and traits composition of plant communities in field margins: some farm characteristics (size, number of workers, land regrouping), field margin–localisation (adjacent to a road or a track) and weed control practices (weed control type and calendar). It appeared that almost all the 16 farmers contacted during the survey were using very intensive weed control on their field margins and that only few variations could be detected between different types of intensive weed control. However, our results showed that plant traits and community richness were affected by some parameters: field margin adjacent to roads were richer and mowing favoured flowering in autumn.

I. INTRODUCTION

Biodiversity is becoming rarer within intensive agrarian landscapes despite of its major roles in agro-ecosystem functioning (Altieri 1999; Le Coeur *et al.* 2002). Field margins (FM) are crucial habitats for plant conservation in intensive agrarian landscapes in such context. Indeed, as permanent non crop habitats, they are generally considered to play a larger role than arable land in maintaining wildlife diversity (Benton *et al.* 2003). Plant diversity is low and mainly concentrated in field margins (FM) (Burel *et al.* 1998; Le Coeur *et al.* 2002) and they are known to constitute one of the main refuges for wild plant species in agrarian landscapes (Marshall & Moonen 2002).

Farm characteristics and farmer practices are very likely to affect plant communities of FM. The number of mowing events generally depends on the availability of manpower, machines, farm structure and farmer sensibility to biodiversity protection. In France, ordinary weed control practices in FM usually includes from two to three mowings a year and/or herbicide spraying with glyphosate. In order to provide farmers with the best advice to preserve biodiversity in this paper, we sought to investigate the potential impact of different parameters on richness and composition of plant communities.

Because analysis of plant communities via their traits composition is now a priority (McGill 2006) and because this allows a good understanding of plant communities composition (Lavorel & Garnier 2002; Pywell *et al.* 2003), sampled FM were studied according to some chosen life traits as flowering period, pollen and seed dispersal, light tolerance, etc. We estimated the species

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richness and functional diversity of plant communities during spring 2005 in different farms with different management practices. The experiment was carried out in the region of Beauce (France), a typical intensive agrarian landscape composed almost exclusively of agricultural fields.

II. MATERIALS AND METHODS

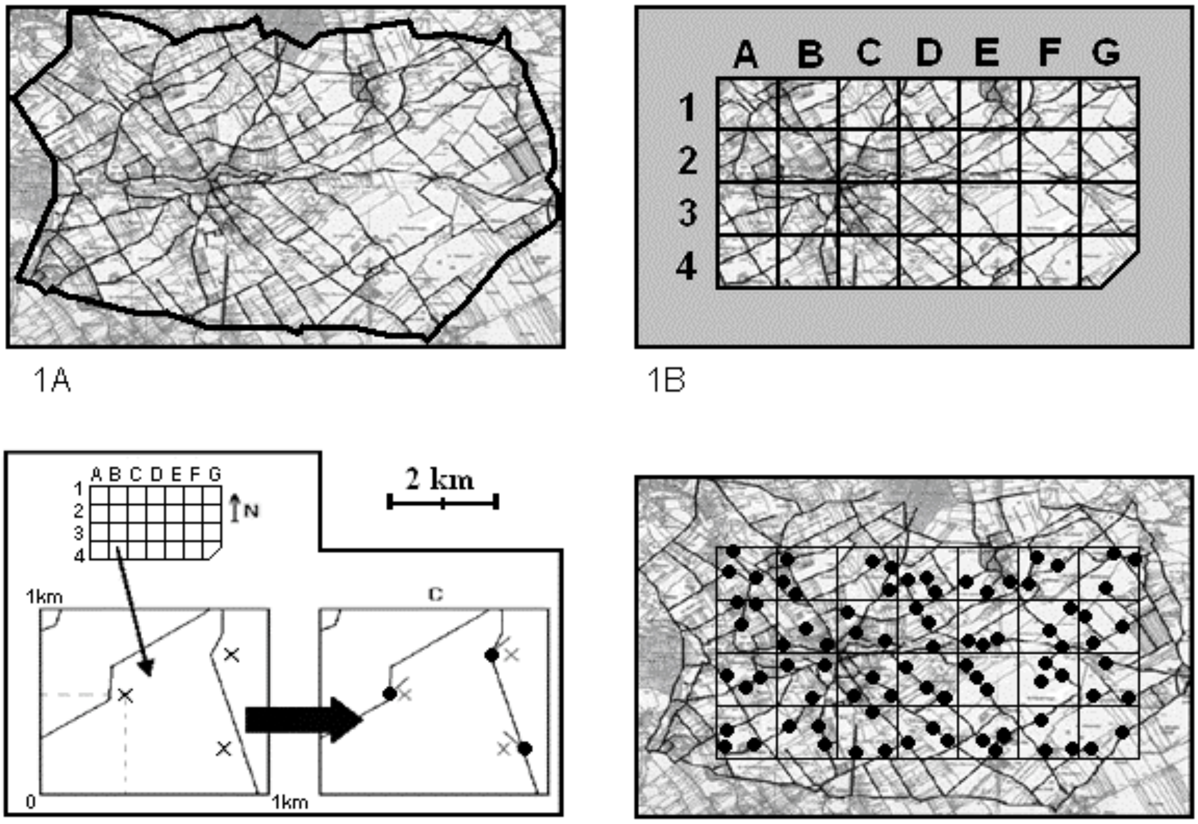
Here, the term "field margin" (FM) is adapted from Greaves and Marshall (1987) and defined as the whole of the crop edge, any margin strip present and the semi-natural habitat associated with the boundary. In this study, they are all adjacent to a road or to a track.

1. Study area and site sampling

The study area is located in the central region of Beauce which is a very intensive agrarian zone of France.Thanks to other studies (Pivard *et al.* 2008), informations about landcover and farmers have been previously registered within an area of 46 km² (Fig. 1A) around the village of Selommes (47°45′ N, 1°11′ E). This area is mainly composed by crop fields and their FM, by roads and tracks, and by small isolated farms.

We focused our study on the central part of this area and divided it into 28 1 km × 1 km plots (Fig. 1B). A small part of the last plot (plot G4) was not in the previous studied area of 46 km² and its

FIGURE 1
STUDY AREA AND SAMPLING PROTOCOL



1A

1B

1C

1D

1A- Study area of 46 km² (roads and tracks are painted in black)
1B- Study area divided in to 28 (7x4) plots of 1 km²
1C- FM randomly chosen (3 by plot)
1D- Location of the 83 FM studied (sites are represented by a black point)

dimensions were reduced to 0.7 km². In each plot, 3 points were placed (excepting for plot G4 where only 2 points weresampled to take its size into account) according to randomly chosen coordinates (*x*, *y*). The nearest FM to the point was then selected as a study site (Fig. 1C). According to this sampling, 83 FMs have been randomly selected into a 27.7 km² study area (Fig. 1D).

We examined 30 FMs adjacent to an asphalt road (*i.e.* road in on side and field in the other side of the FM) and 53 FMs adjacent to a private grassy track (*i.e.* track in on side and field in the other side of the FM). This point is very important because the State road agency may also take part in weed control performed in FM for those adjacent to a roads.

2. Vegetation inventories

In each FM a 1 m × 25 m plot was positioned along the FM. Vegetation sampling (restricted to Angiosperm taxa) was performed during one week in May 2005. The period was short enough to avoid temporal bias in vegetation development. In each plot, we listed all plant taxa detected. Almost all the taxa were identified to species level according to the International Plant Names Index (http://www.ipni.org). In some cases (*Crepis spp.* for example), taxa were identified to genus level due to difficulties in identification when FMs had been mown or sprayed.

We then established a file for six life traits (Phenology, Vegetative reproduction, Flowering period, Prefered soil, Shadow affinity & Raunkiaer type) and their different modalities (18 in total) for all the species detected (Table 1). For each species traits modalities were found in Fitter (1986).

3. Management practices of farmers and farm structures

We contacted the 16 farmers who owned the fields adjacent to the studied FM to collect the following informations:

- Farm attributes: farm area (in hectares), work time unit (number of workers into the barn corrected by their time presence), land regrouping (i.e. a value between 1 and 5: 1 if fields where all scattered around the study area and 5 if all fields where adjacent to the farm) and distance to farm buildings for each FM (in meters);
- Weed control calendar: number of events (between 1 and 4) and period of application (March/April, and/or May/June, and/or July/August);
- Type of weed control: vegetation mowing, and/or herbicide spraying.

All FM were described by those ten variables and their localisation (i.e. adjacent to a road vs. to a track). Those eleven variables are referenced as “FM characteristics”.

4. Data analysis

To test the impacts of local environment and management on FM plant communities, we made two groups of tests:

(1) Richness (number of taxa, noted *N*) was recorded. We then performed ANOVAs on *N* with the eleven FM characteristics as explicative variables. Because those analyses represented a quite large amount of tests we reduced the significant probability to 0.027 (instead of 0.05) according to *False Discovery Rate* method (Benjamini *et al.* 1995): $\alpha_m = \alpha_1 \frac{m+1}{2m}$ with $\alpha_1 = 0.05$ and $m = 11$;

(2) We then calculated for each FM the percentage of plants of each trait modalities for the six life traits and we performed a PCA on sites and their traits composition (percentages). We then ran ANOVAs on the coordinates of sites on the more important axes of the PCA (each axis representing more than 10% of the total variance) with the eleven FM characteristics as explicative. We used the significant probably of 0.027 as previously for the same reasons.

TABLE 1:
LIST OF THE 18 TRAITS MODALITIES CHARACTERISTICS DESCRIBING PLANT COMMUNITIES

Traits	Trait modalities	Symbol
Phenology	Annual	A1
	Bisannual	A2
	Perennial	A3
Vegetative reproduction	Yes	B1
	No	B2
Flowering period	Spring	C1
	Summer	C2
	Autumn	C3
Prefered soil	Poor	D1
	Rich	D2
	Fertilized	D3
Shadow affinity	Yes	E1
	No	E2
Raunkiaer type	Cryptophyte	F1
	Geophyte	F2
	Hemicryptophyte	F3
	Nanophanerophyte	F4

Traits modalities from Fitter (1986).

All statistical analyses were performed using R (Ihaka &Gentleman 1996).

TABLE 2
RESULTS OF ANOVAS PERFORMED ON LOCAL PLANT RICHNESS (*N*) AND 11 VARIABLES DESCRIBING FIELD MARGINS ENVIRONMENT AND MANAGEMENT

Variables describing field margin (FM)		Plant richness (N)		
		t-value	P	Effect
Farm attributes	Farm area	-0.724	0.471	
	Work time unit	*2.057*	*0.043*	+
	Land Regrouping	-0.930	0.355	
	Distance to farm buildings	1.094	0.278	
Weed control calendar	Nb of weed control events	*1.945*	*0.055*	+
	March - April	0.151	0.880	
	May - June	0.583	0.561	
	July - August	*1.872*	*0.065*	+
Type of weed control	Mowing	1.150	0.254	
	Spraying	-0.224	0.824	
Field Margin Localisation	Adjacent to road	**5.609**	**$<10^{-3}$**	+

III. RESULTS

1. Management practices of farmers and farm structures

To control weeds in field margins (FM), 68% of farmers (corresponding to 62 % of FM) used mowing to counterpart disadvantages of chemical treatments. Cutting height varied from 0 to 50 cm and the most cited reasons for performing mowing was preventing the colonisation by perennial plants of crop fields when the soil is nude. The second disadvantage of herbicide spraying was the cost of chemical products, then the social pressure for not using such products. Only 12.5% of farmers (corresponding to 7.6% of FM) performed chemical treatments and 100 % of them used glyphosate. If applied once, spraying presented the advantage of a unique application per year (time savings). Farmers did not discriminate weed control in FM according to the type of adjacent crop. The period of herbicide application depended on vegetation height, *Gramineae* state, time schedule, and equipment availability.

In order to avoid seed production of FM weeds (and thereby seed dispersal), farmers mowed FM in spring in March-April or May-June (when the vegetation development is maximal), then during the summer in July-August (when plants grow again). Some farmers mowed in July (when plants are well developed), then in August if necessary. Some farmers mowed twice before and after harvesting (in July-

August). Some other farmers mowed once in spring (in March-April or May-June).

2. Impacts of local environment and human activities on plant richness (*N*) of FM plant communities

A total of 187 different taxa (Appendix 1) were observed including 150 plants identified at the species level (80.2%). Only 30 taxa were recorded in more than in 10% of the FM (Appendix 2).

The presence of a road adjacent to the FM was the only characteristic with a significant positive effect on FM plant richness ($t = 5.609$; $P < 10^{-3}$ – Table 2). We observed a positive tendency, (although insignificant), of three other characteristics: “work time unit”, “number of weed control events” and “July/August weed control” (Table 2).

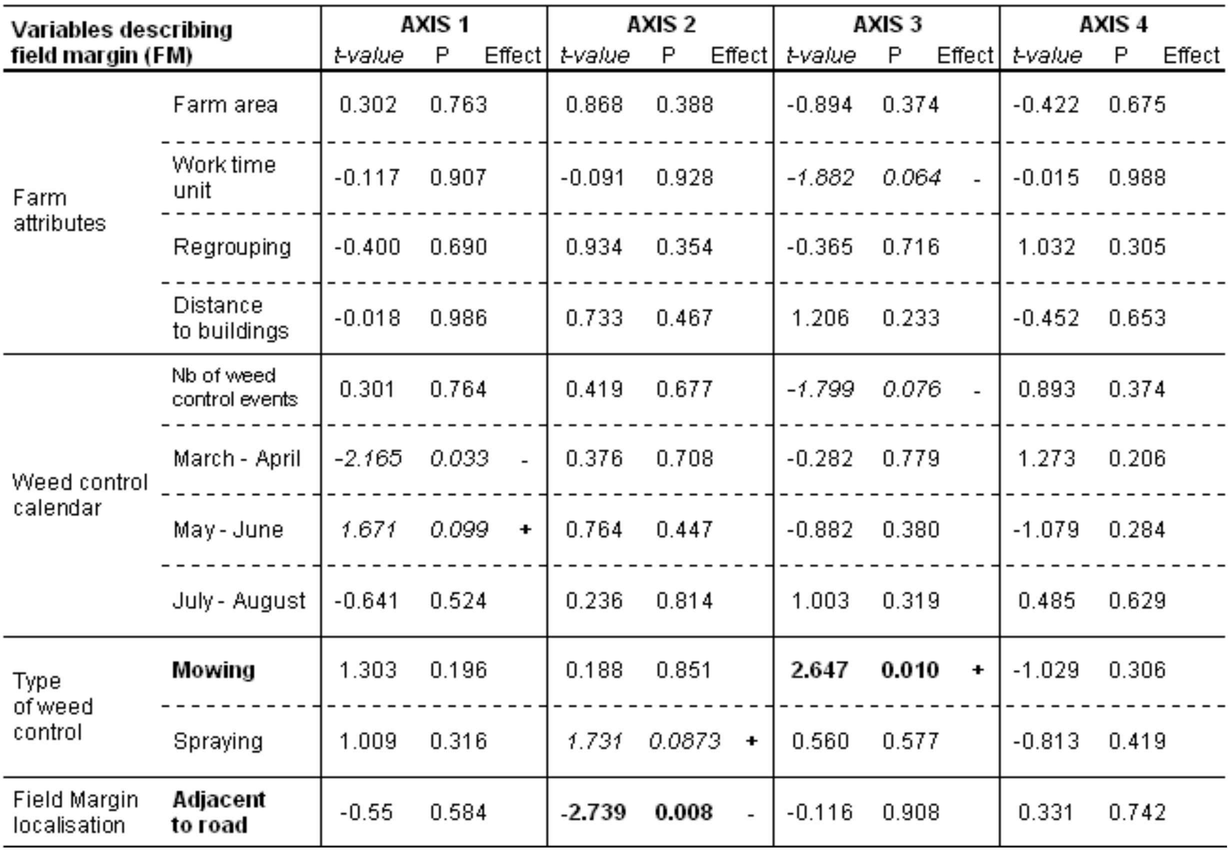
3. Impacts of local environment and human management on plant life traits in FM plant communities

The PCA performed on sites and their composition in plant life traits showed that four axes represented each more than 10% of the total variance (and altogether 68.9% of the total variance). These PCA axes established strong discriminations between FM according to their of plant communities composition so we focused our tests on them (Fig. 2A and 2B).

We did not detect any significant effects of FM characteristics on the coordinates of FM along the first PCA axis 1 (Table 3). Some modalities of weed control calendar had little influence on those coordinates (insignificant tendencies): negative effect of “number of weed control events” ($t = -1.799$; $P = 0.076$) and of “March/April weed control”

TABLE 3

RESULTS OF ANOVAS PERFORMED ON COORDINATES OF STUDIED SITES ON MAIN PCA AXIS (AXIS 1 TO AXIS 4) AND 11 VARIABLES DESCRIBING FIELD MARGINS ENVIRONMENT AND MANAGEMENT.



Variables describing field margin (FM)		**AXIS 1**			**AXIS 2**			**AXIS 3**			**AXIS 4**		
		t-value	P	Effect	*t-value*	P	Effect	*t-value*	P	Effect	*t-value*	P	Effect
Farm attributes	Farm area	0.302	0.763		0.868	0.388		-0.894	0.374		-0.422	0.675	
	Work time unit	-0.117	0.907		-0.091	0.928		*-1.882*	*0.064*	-	-0.015	0.988	
	Regrouping	-0.400	0.690		0.934	0.354		-0.365	0.716		1.032	0.305	
	Distance to buildings	-0.018	0.986		0.733	0.467		1.206	0.233		-0.452	0.653	
Weed control calendar	Nb of weed control events	0.301	0.764		0.419	0.677		*-1.799*	*0.076*	-	0.893	0.374	
	March - April	*-2.165*	*0.033*	-	0.376	0.708		-0.282	0.779		1.273	0.206	
	May - June	*1.671*	*0.099*	+	0.764	0.447		-0.882	0.380		-1.079	0.284	
	July - August	-0.641	0.524		0.236	0.814		1.003	0.319		0.485	0.629	
Type of weed control	**Mowing**	1.303	0.196		0.188	0.851		**2.647**	**0.010**	+	-1.029	0.306	
	Spraying	1.009	0.316		*1.731*	*0.0873*	+	0.560	0.577		-0.813	0.419	
Field Margin localisation	**Adjacent to road**	-0.55	0.584		**-2.739**	**0.008**	-	-0.116	0.908		0.331	0.742	

(*t* = -2.165; *P* = 0.033) and positive effects of "May/June weed control" (*t* = 1.671; *P* = 0.099).

We detected a strong and significant negative effect of the presence an adjacent road to FM (t = -2.739; P = 0.008) and a tendency for a positive effect of spraying (*t* = 1.730; *P* = 0.087) on the coordinates of FM along Axis 2 (Table 3).

We only found a significant positive effect (*t* = 2.647; *P* = 0.01) of mowing on the coordinates of FM along Axis 3 (Table 3), but no significant effect on the coordinates of FM along Axis 4 (Table 3).

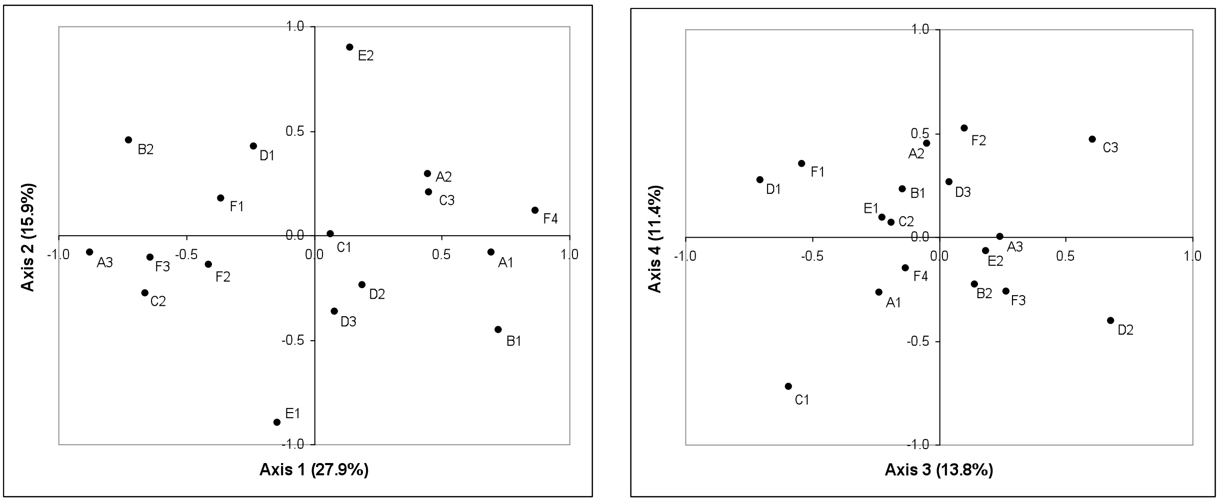
IV. DISCUSSION

Field margins (FM) are composed by semi-natural vegetation and constitute an important habitat area for plant population persistence in landscapes dominated by intensive agriculture. Decreases in plant diversity in FM is generally linked to the intensification of agricultural land use, causing a reduction of FM area, an increase of disturbances and input of agro-chemicals from neighbouring arable fields (Kleijn & Snoeijing 1997; De Snoo 1999). FM play an important role in the agricultural landscapes since they provide a habitat or corridor for a large range of plants and animals (Klejin & Snoeijing 1997). The species richness in FM can be influenced by their structure and weed control performed by farmers in FM (Le Coeur *et al.* 1997). Because these authors found 224 species in hedgerow network landscapes also located in western part of France (16 km²), while we found 187 species in an openfield landscape; we assume that diversity in FM of an openfield landscape is comparable to that of FM in a hedgerow landscape is thus not negligible.

1. Farmer's surveys

Surveys we carried out allowed a better understanding of how farmers act on landscapes in a practical manner, and could help to design future farming systems, integrating environmental goals. We had a pretty good return from our contacts to farmers and most of them accepted to answer all our questions. The fact that only few explicative factors had a significant impact on plant diversity in FM could be partially due to the two following facts:

FIGURE 2
PROJECTION OF THE 18 LIFE TRAITS ON AXIS N°1 TO N°4 OF THE PCA



A. Projection on Axis 1 vs. Axis 2

B. Projection on Axis 3 vs. Axis 4

(1) All farmers were practicing a strong FM management because they want to control plants they describe as “weeds”. According to this observation, the type of weed control actually performed by farmers in the study area was generally intensive and therefore we did not have contrasted weed control scenarios in the studied FM.

(2) The number of people working in the farm and the existence of spraying treatment performed on FM may have been underestimated in the data collected during the farmers’ survey because a part of the manpower might be undeclared workers and it is generally unpopular to admit using chemical treatments.

2. Impacts of local environment

The strongest impact of local environment on FM plant communities was the location of the FM: adjacent to an asphalt road or to a grassy. This could be explained by the fact that roads have strong direct impacts on plants communities: pollution, fragmentation, light conditions, seed dispersal by cars, etc. , which are known to affect plant diversity and plant growth rates in roadside verges (Formal & Alexander 1998). FMs adjacent to a road are also managed by the local State agency for roads in addition to farmers (?) and are significantly wider than FM adjacent to a track (respective mean widths are 2.23 ± 0.13m and = 1.25 ± 0.05m, $t = 8.223$ & $P < 10^{-3}$). Plant richness is larger in FM adjacent to a road and plants tolerating shadow conditions to flower appeared to be favoured there (Axis 2 – Fig. 2A). This result could be explained by a vegetation more developed in FM adjacent to a road because (1) these FM are more extensively managed (they are very wide and farmer’s mowing device only covers 1 to 1.5 m); and (2) microenvironmental conditions in FM adjacent to a road are very different than that of FM adjacent to a track, according to the soil composition that is known to impacts plant growth (Angold 1997).

3. Impacts of field margin management

Mowing seemed to favour plants flowering in autumn and developing in rich soil (Axis 3 – Fig. 2B & Table 3). This could be explained by the fact that mowing occurs during spring and summer periods, and because mown material is not removed: its decomposition in FM results in a very rich soil potentially affecting plant communities (De Cauwer *et al.* 2005).

Management calendar seemed to slightly affect the composition of plant communities but this effect was insignificant (Axis 1– Fig 2A & Table 3). Plant richness seemed to be positively affected by three parameters “work time unit”, “number of weed control events” and “July – August weed control” (non significant effect). In fact those three parameters were not independent: “work time unit” was strongly correlated to “number of weed control events” ($t = 7.580$; $P < 10^{-3}$; $R^2 = 0.436$) because when the farm hires more workers, additional weed control events can be easily performed.

4. Conclusion

This work showed the importance of field margins as habitat for plant species of farming landscape (187 plant taxa detected) and the relatively poor impact of variations of weed control in a context of intensive farming practices. The results proved that considering different field margins adjacent to roads, which are wide and where weed control is less intensive and generally performed by State employees, it is possible to favour different and rich plant communities. Small modifications in farmer practices in terms of weed control would have little impact in the attempt to make field margins become a better refuge area for plant diversity in intensive agrarian landscapes.

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SUPPLEMENTARY MATERIALS

APPENDIX 1

LIST OF THE 187 TAXA OBSERVED IN FIELD MARGINS

Taxa	**Family**
Acer platanoides	Aceraceae
Achillea millefolium	Asteraceae
Adenostyles alliariae	Asteraceae
Agrimonia eupatoria	Rosaceae
Agropyrum repens	Poaceae
Agrostis ssp.	Poaceae
Agrostis stolonifera	Poaceae
Agrostis tenuis	Poaceae
Ajuga reptans	Lamiaceae
Allium ssp.	Lamiaceae
Alopecurus agrestis	Poaceae
Alopecurus ssp.	Poaceae
Amaranthus retroflexus	Amaranthaceae
Anagallis arvensis	Primulaceae
Anthriscus sylvestris	Apiaceae
Anthriscus vulgaris	Apiaceae
Aphenes arvensis	Rosaceae
Arabidopsis thaliana	Brassicaceae
Arctium lappa	Asteraceae
Arenaria serpyllifolia	Caryophyllaceae
Arrhenaterum elatius	Poaceae
Artemisia vulgaris	Poaceae
Atriplex ssp.	Asteraceae
Avena sativa	Chenopodiaceae
Bellis perennis	Asteraceae
Brachypodium pinnatum	Chenopodiaceae
Brachypodium sylvaticum	Poaceae
Brassica napus	Brassicaceae
Brassica nigra	Brassicaceae
Brassica ssp.	Brassicaceae
Bromus arvensis	Poaceae
Bromus maximus	Poaceae
Bromus mollis	Poaceae
Bromus sterilis	Cucurbitaceae
Bryonia dioica	Lamianae
Capsella bursa-pastoris	Brassicaceae
Carduus nutans	Asteraceae
Carex divulsa	Cyperaceae
Carex muricata	Cyperaceae
Centaurea jacea	Asteraceae
Centaurea scabiosa	Asteraceae
Centaurea ssp.	Asteraceae
Ceorynephorus canesciens	Poaceae
Cerastium ssp.	Caryophyllaceae
Cerastium triviale	Caryophyllaceae
Chenopodium album	Chenopodiaceae
Chrysantenum segetum	Asteraceae
Chrysanthemum leucanthemum	Asteraceae
Circium lanceolata	Asteraceae
Cirsium arvense	Asteraceae
Cirsium vulgare	Asteraceae
Convolvulus arvensis	Convolvulaceae
Conyza canadensis	Asteraceae
Crataegus oxyacantha	Rosaceae
Crepis setosa	Asteraceae
Crepis ssp.	Asteraceae
Dactylis glomerata	Poaceae
Daucus Carota	Apiaceae
Epilobium ssp.	Onagraceae
Erigeron cancolensis	Asteraceae
Erodium cicutarium	Geraniaceae
Erophila verna	Brassicaceae
Ervum lens	Fabaceae
Eryngium campestre	Apiaceae
Euonymus europaeus	Celastraceae
Euphorbia aparisias	Euphobiaceae
Euphorbia helioscopia	Euphobiaceae
Fallopia convolvulus	Polygonaceae
Festuca arumdinacea	Poaceae
Festuca pratense	Poaceae
Festuca rubra	Poaceae

Galium aparine	Rubiaceae
Galium cruciata	Rubiaceae
Galium mollugo	Rubiaceae
Gallium ssp.	Rubiaceae
Geranium dissectum	Geraniaceae
Geranium molle	Geraniaceae
Geranium pirenaicum	Geraniaceae
Geranium robertianum	Geraniaceae
Geranium rotundifolium	Geraniaceae
Geranium ssp.	Geraniaceae
Hedera helix	Araliaceae
Heracleum ssp.ondylium	Apiaceae
Hordeum murinum	Poaceae
Hypericum ssp.	Hypericaceae
Juncus conglomeratus	Juncaceae
Kickxia ssp.uria	Plantaginaceae
Knautia arvensis	Dipsacaceae
Lactuca scariola	Asteraceae
Lamium purpureum	Labiaceae
Lamium ssp.	Lamiaceae
Lampsana communis	Asteraceae
Lathyrus pratensis	Fabaceae
Lathyrus ssp.	Fabaceae
Lathyrus tuberosus	Asteraceae
Ligustrum vulgare	Oleaceae
Linaria supinia	Plantaginaceae
Linaria vulgaris	Plantaginaceae
Lolium perenne	Poaceae
Lotus corniculata	Fabaceae
Lychnis dioica	Caryophyllaceae
Malva neglecta	Malvaceae
Malva rotondifolia	Malvaceae
Matricaria chamomilla	Asteraceae
Matricaria discoidea	Asteraceae
Matricaria inodora	Asteraceae
Matricaria ssp.	Asteraceae
Medicago lupulina	Fabaceae
Mercurialis annua	Euphobiaceae
Myosotis arvensis	Boraginaceae
Mysosotis ssp.	Boraginaceae
Ononis repens	Fabaceae
Origanum vulgaris	Lamiaceae
Orobanche ssp.	Orobachaceae
Papaver rhaeas	Pavaveraceae
Papaver ssp.	Papaveraceae
Pastinaca sativa	Apiaceae
Phleum pratense	Poaceae
Picris heracioides	Asteraceae
Picris ssp.	Asteraceae
Plantago coronopus	Plantaginaceae
Plantago lanceolata	Plantaginaceae
Plantago major	Plantaginaceae
Poa annua	Poaceae
Poa nemoralis	Poaceae

Poa pratensis	Poaceae
Poa ssp.	Poaceae
Poa trivialis	Poaceae
Polygonatum multiflorum	Convallariaceae
Polygonum aviculare	Polygonaceae
Polygonum convolvulus	Polygonaceae
Polygonum lapatifolium	Polygonaceae
Polygonum persicaria	Polygonaceae
Portulaca oleracea	Portulacaceae
Potentilla fragaria	Rosaceae
Potentilla reptans	Rosaceae
Prunus avium	Rosaceae
Prunus spinosa	Rosaceae
Pulmonaria sp	Borraginaceae
Quercus robur	Fagaceae
Ranunculus acris	Renonculaceae
Ranunculus bulbusus	Renonculaceae
Ranunculus repens	Renonculaceae
Ranunculus sp	Renonculaceae
Raphanus raphanistrum	Brassicaceae
Renunculus sardus	Renunculaceae
Rosa sp.	Rosaceae
Rubus sp	Rosaceae
Rumex acetosa	Polygonaceae
Rumex crispus	Polygonaceae
rumex obtusifolius	Polygonaceae
Rumex sanguineus	Polygonaceae
Rumex sp	Polygonaceae
Sagina sp	Caryophyllaceae
Sanguisorba sp	Crassulaceae
Sedum telephium	Crassulaceae
Senecio vulgaris	Asteraceae
Silene inflata	Caryophyllaceae
Silene latifolia	Caryophyllaceae
Sinapis arvensis	Brassicaceae
Sisymbrium officinale	Brassicaceae
Solanum nigrum	Solanaceae
Sonchus asper	Asteraceae
Sonchus oleraceus	Asteraceae
Stachys betonica	Labiaceae
Stellaria graminea	Caryophyllaceae
Stellaria media	Caryophyllaceae
Tamus communis	Dioscoreaceae
Taraxacum dens-leonis	Asteraceae
Taraxacum sp	Asteraceae
Tragopogon dubius	Asteraceae
Tricatum flavecens	Fabaceae
Trifolium minus	Fabaceae
Trifolium repens	Fabaceae
Ulmus campestris	Ulmaceae
Urtica dioica	Urticaceae
Verbena officinialis	Verbenaceae
Veronica arvensis	Plantaginaceae
Veronica chamaedris	Plantaginaceae
Veronica hederifolia	Plantaginaceae

Veronica persica	Plantaginaceae
Vicia cracca	Fabaceae
Vicia hirsuta	Fabaceae
Vicia sativa	Fabaceae
Viola arvensis	Violaceae
Viola sp	Violaceae
Vulpia myuros	Poaceae

1

APPENDIX 2

LIST OF 30 SPECIES PRESENT ON MORE THAN 10% OF FIELD MARGINS

Taxa	**(%)**
Convolvulus arvensis	67%
Bromus mollis	59%
Bromus sterilis	57%
Poa annua	56%
Poa pratensis	53%
Geranium dissectum	51%
Dactylis glomerata	47%
Lolium perenne	39%
Galium aparine	38%
Plantago major	33%
Geranium molle	31%
Taraxacum dens-leonis	28%
Cirsium arvense	27%
Artemisia vulgaris	27%
Achillea millefolium	25%
Agropyrum repens	25%
Veronica persica	25%
Polygonum aviculare	24%
Plantago lanceolata	23%
Potentilla reptans	22%
Papaver rhaeas	21%
Senecio vulgaris	21%
Heracleum spondylium	20%
Brassica napus	18%
Arrhenaterum elatius	17%
Sonchus asper	17%
Veronica arvensis	17%
Polygonum convolvulus	16%
Capsella bursa-pastoris	14%
Matricaria discoidea	11%

(%): Percentage of field margins where species were observed.

MANUSCRIPT N°6

Submitted to Weed Research

A three-year study of weed management on plant community in field margins in an openfield landscape

Nadia MICHEL
Aurélie GARNIER
Robert HAICOURT
Louis de REDON
Aude SOURISSEAU
&
Agnès RICROCH

PRESENTATION

TITRE

Effets du contrôle des adventices sur les communautés végétales des bordures de champs dans un paysage agricole intensif, trois ans d'étude.

RESUME

Nous avons étudié durant trois ans les effets du contrôle des adventices (par fauche et produits chimiques) sur la composition et la richesse des communautés végétales de 27 bordures de champ au sein d'un paysage d'agriculture intensive. Onze paramètres, incluant les modalités de pollinisation, ont été définis pour décrire la flore des bordures de champ et leurs variations ont été analysées à l'aide d'analyses multivariées et de tests d'égalité sur les observations pairées. Les paramètres testés se sont révélés affectés les systèmes de gestion des agriculteurs ; dépendant des modes de contrôle des populations d'adventices. La richesse spécifique observée a augmenté entre 2005 (102 espèces) et 2007 (134 espèces) coïncidant avec une extensification des pratiques de gestion. Les récents changements de pratique de gestion, abandonnant la volonté de contrôle des populations d'adventices dans les bordures de champ adjacents aux routes (à l'initiative des nouvelles politiques environnementales françaises), a montré l'effectivité de certaines mesures pour protéger la biodiversité et augmenter la qualité de espaces marginaux comme les bordures de champ comme habitats. Dans ce contexte, l'augmentation observée d'espèces végétales entomophiles peut procurer des ressources en pollen et en nectar aux populations de pollinisateurs permettant leur maintien au sein des paysages agricoles intensifs. Le nombre d'espèces non-entomophiles a aussi augmenté pendant les trois années d'étude ce qui a montré qu'elles pouvaient aussi se maintenir au sein de bordures de champ perturbées dans les agro-écosystèmes.

MOTS CLEFS

Contôle des adventices, Zones non-cultivées, Pratiques de gestion, Agro-écologie & Plantes entomophiles.

A THREE-YEAR STUDY OF WEED CONTROL ON PLANT COMMUNITY IN FIELD MARGINS IN AN OPENFIELD LANDSCAPE

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KEYWORDS

Weed control
Uncultivated areas
Farming practices
Agro-ecology
Insect-pollinated species

ABSTRACT

This study examined the effects of three years of weed control (mowing and spraying) on the composition and richness of plant communities in 27 field margins (FM) in an openfield landscape. Eleven synthetic parameters, including pollination modes and life-form types, were defined to describe the FM flora and their variations were analysed using multivariate analyses and pair-wise comparisons. The synthetic parameters were affected by some components of the farming system. They all depended on weed control, but differently according to the year. The species richness increased from 2005 (102 species) to 2007 (134 species). This increase in species richness coincided with a decrease in the number of mowed or sprayed FM. The recent adoption of farming practices avoiding weed control in FM adjacent to roads (validated by the French Departmental Commission for Agricultural Policy) proved effective to preserve plant communities and rapidly enhance the quality of FM habitats. For insect-pollinated species, their persistence and their ability to flower provided pollen and nectar resources for pollinators that could help ensure the persistence of the pollinator community. Species that are not insect-pollinated increased also, which showed that they were able to persist in the disturbed habitats of the FM in an agro-ecosystem..

I. INTRODUCTION

At the local scales of the field and the farm, improvements in crop management techniques have been adopted and they now involve weed control by mowing and herbicide spraying, increased use of fertilizer, simplification of crop rotations, and improvements in seed-cleaning techniques (Benton et al., 2003). Arable plant communities are thus exposed to a frequently changing environment, which is in turn likely to generate great community variability in space and time (Harper, 1977). Field margins, as permanent non crop habitats, are generally considered to play a larger role than arable land in maintaining wildlife diversity in agricultural landscapes (Benton, Vickery and Wilson 2003). Here, the term "field margin" (FM) is adapted from Greaves and Marshall (1987) and defined as the whole of the crop edge, any margin strip present and the semi-natural habitat associated with the boundary. Any plant species that occurs in a FM is

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generally considered as a weed species by farmers because it can cause major crop losses. From a point of view of biodiversity preservation these weed species are however important components of the agro-ecosystem (Marshall et al., 2003). Indeed, reduced weed densities may result in fewer weed seeds being available as food for wintering birds (Watkinson et al., 2000) and may reduce the number of invertebrate herbivores, together with their predators. The management of FM is complex as FM are biodiversity refuges, and also potential relays for the spread of transgenes since they are the main habitat of feral populations of OSR. GM feral plants can introduce transgenes into wild and weed populations in FM.

Since 1998 we studied the dynamics of feral populations of oilseed rape (OSR; *Brassica napus* L.) in an open field landscape in France (Pivard et al. 2008). Feral OSR populations mainly result from the escape of OSR from cultivated crops and are mainly located in FM. These feral populations as a source for further transgene flow by pollen and/or seed dispersal can persist several years (Hüsken & Dietz-Pfeilstetter 2007). To limit transgene escape via feral populations FM spraying and/or mowing should be more intense (Garnier, Deville and Lecomte, 2006).

Spraying herbicide onto or mowing FM may affect the biodiversity in the local agro-ecosystem in three main ways by: (i) a direct impact on plant community composition, (ii) a direct impact on pollinators due to herbicide toxicity or mowing of flowering FM, and (iii) an indirect impact on pollinators if nectariferous and/or polleniferous plants are affected by weed control.

Pollinator population dynamics and persistence of plants according to their reproductive systems should be better documented to understand the complex interactions between cropped and non-crop areas. It is also important to understand the pollination processes that generate landscape-scale gene dispersal in plants, particularly in crop plants with genetically modified varieties. At the landscape scale in particular, gene flow among populations maintains the cohesion of a species' gene pool, thereby diminishing the potential for local adaptation and speciation. Indeed, modifications of the available local nectar resources could influence the spatial patterns of insect-mediated gene flow at the landscape scale because a lack of insect-pollinated species as local food resources would make the pollinators travel over longer distances, thereby favouring long-distance pollen dispersal (Kohler et al., 2007). Biesmeijer et al. (2006) showed that linked elements in biological communities (i.e. specialist pollinators and the obligatory outcrossed plants that they pollinate) are declining in tandem in Britain and in the Netherlands.

In this paper we present the first French openfield assessment of the impacts of chemical and mechanical control options on the floristic composition in FM. The main purpose of this study was the evaluation of (1) the spatio-temporal variability of plant community compositions in relation to its pollination mode in FM and (2) the impact of weed control on this variability in an openfield farming landscape.

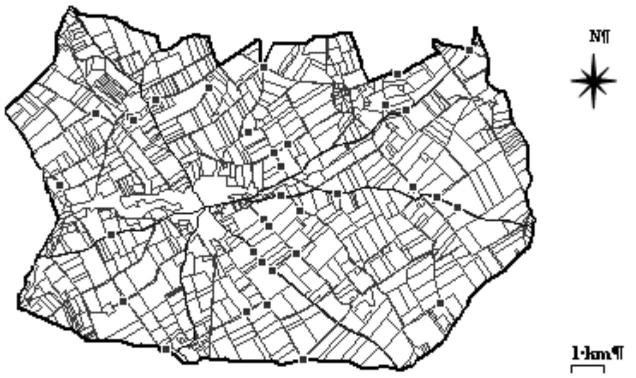
We thus performed a three-year field survey in 27 FM within a 46 km² openfield landscape in France (2005-2006-2007). We used several synthetic parameters describing richness and diversity of plant communities in these FM to examine the impacts of mowing and herbicide spraying on plant community composition. We addressed three main questions: (i) Does plant community composition in FM vary over time? (ii) Can these variations be related to the type of weed control? (iii) Do these variations differ among categories of plant species, especially concerning insect-pollinated vs. not insect-pollinated species?

2. MATERIAL AND METHODS

2.1. Sampling design of FM plant communities

The field study was conducted in an intensive openfield agricultural region of 46 km2 around Selommes (47° 45′ 24″ N; 1° 11' 34''E) in Loir-et-Cher region (France). This study site is located within the temperate marine climate zone. The selection of field

FIGURE 1:
LOCATION OF THE 27 FIELD MARGINS IN THE STUDY AREA



The black points represent the FM that were surveyed during the field study. Roads and paths are painted in black.

margins (FM) for the survey was performed was first divided into 1 km x 1 km quadrats using a stratified sampling: the 46 km2 area and then locations of FM were randomly chosen within each quadrat, with the constraint that FM were equally distributed between those adjacent to a road and those adjacent to a path. A total of n = 27 FM were thus selected and located with a Trimble® GPS: 14 were adjacent to a path (52 %), 8 to a two-way road (30 %), and 5 to a one-way road (18 %) (Fig. 1).

To allow inter-annual comparisons between the FM plant communities, the floristic composition (restricted to angiosperms) was surveyed in the 27 FM in May (i.e. the peak bloom period of plant species in FM) in 2005, 2006 and 2007. Two complementary sampling methods were carried out in each FM: 1) an exhaustive determination of all the taxa that were present along a 25 m × 1 m-strip and 2) the measure of cover percentage (using the Braun-Blanquet index - 1932) for each taxa observed in two (resp. five) 1 m^2-quadrats randomly selected in the 25 m2 strip in 2005 (resp. 2006 and 2007). The total plant cover was also recorded in each FM. The 25 m2 strip is assumed to be representative of the FM since the weed control unit is the same in the whole FM (Baudry et al., 2000). The identification of species was completed according to Lambinon et al. (2004). Then we checked the species names on the Germplasm Resources Information Network (GRIN) Taxonomy for Plants (http://www.ars-grin.gov). We referred to plant taxa for either a species (for plants identified at the species scale) or a genus (for plants identified at the genus scale). Indeed, the external damage resulting from mowing and/or spraying prevented the identification at the species scale of some plant samples.

Along with floristic surveys, the external damage to plants was recorded by eye to detect whether the plants had been sprayed with herbicide or mown before the sampling session.

2.2. Statistical analyses

2.2.a. Definition of synthetic parameters describing plant community diversity

Eleven synthetic parameters were computed from the floristic data to describe the composition of the plant community and to evaluate its variations among years and among the FM. These synthetic parameters are the following: species richness, number of botanical families, Shannon index, Simpson index, number of Eudicotyledons and Monocotyledons species, number of Annuals, Biennials and Perennials species and number of species that are known to be insect-pollinated or not. Not insect-pollinated species designates either autogamous or wind-pollinated species. Data about life form and type of pollination are only available for the taxa identified at the species scale.

The Shannon and Simpson diversity indices were calculated using the ‘diversity’ function of the ‘vegan’ library (Oksanen et al., 2007) of the R software (R Development Core Team, 2004). In 2007, Shannon and Simpson diversity indices calculated with two quadrats were significantly smaller than the same indices calculated with five quadrats. The same result was obtained in 2006 for the Shannon index. As a consequence, we chose randomly two quadrats among the five quadrats for 2006 and 2007 so that diversity indices were calculated with two quadrats for all years.

Both ubiquitous and rares species were counted each year of the study. Ubiquitous species (resp. rare species) are defined by Gabriel, Thies and Tscharntke (2005) as species occurring in more than 50 % (resp. in less than 5 %) of the quadrats surveyed, that is to say in more than 13 FM (resp. in less than 2 FM) in our study.

2.2.b. Multivariate analyses: global view and evolution of the FM plant communities

We performed a co-inertia analysis (Dolédec et al., 1996) on three annual synthetic tables that were build with the eleven synthetic parameters measured in the 27 FM (one table for each year studied). These analyses were not directly performed at the taxa level, because the number of columns of matrices, i.e. the number of taxa observed (185 – see Results) would have been too high. This type of multivariate analysis is a two-table ordination method based on a covariance matrix that is calculated for pairs of (contingency) matrices (i.e. pairs of annual synthetic tables here). The correlation between pairs of tables was measured by the RV coefficient (Robert and Escoufier, 1976) computed with ADE-4 SoftwareTM (Thioulouse et al., 1997). A Monte-Carlo permutation test was then performed for each pair of tables to test whether the percentage of co-variation between the two tables was significant (number of permutations: 1000).

2.2.c. Inter-annual comparisons of the synthetic parameters

We performed paired-samples Wilcoxon tests of the eleven synthetic parameters using StatisticaTM Software (StatSoft, Inc.) to draw inter-annual comparisons of FM plant community richness and diversity.

2.2.d. Impact of weed control on community composition in FM

We evaluated the impact of weed control on the composition of plant communities in FM by comparing the plant community composition according to the different types of control performed in FM. Thus for each year, we performed Mann-Whitney tests with Statistica^{TM} to compare the synthetic parameters according to the type of weed control (No Control, Mowing, and Spraying)

To test whether the different types of weed control would favour some reproductive strategies, we particularly focussed on five of the synthetic parameters that are associated with two life traits: (1) type of life form (annual or biennial species vs. perennial species) and (2) type of pollination (insect-pollinated vs. not insect-pollinated species).

3. RESULTS

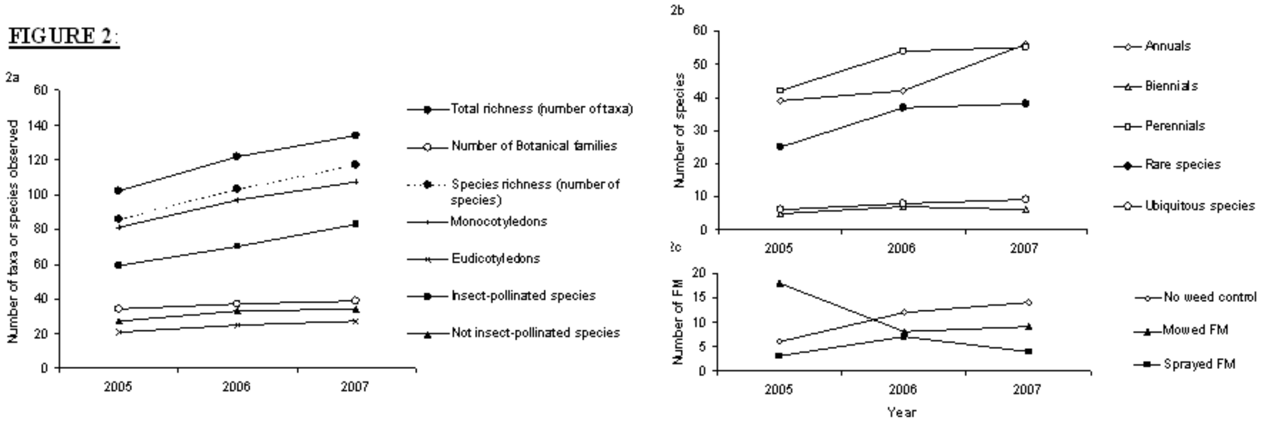
3.1. General view of the plant community

The land-use in fields adjacent to the FM followed crop rotations (including bare fallows and fallows) and the more common crops were winter cereals, conventional oilseed rape, and pea

A total of 185 plant taxa were observed in the 27 FM including 154 taxa identified at the species scale and 29 at the genus scale (“spp.”) (150 Eudicotyledons; 35 Monocotyledons; 37 botanical families; 112 genus - Appendix S1). The large majority of the taxa identified at the species scale were equally distributed between Annuals (70 species, 45.5 %) and Perennials (75 species, 48.7 %); Biennials were less numerous (9 species, 12.0 %). The ratio insect-/wind-pollinated species was 2.6 (108 vs. 41 species, i.e. 70.1 % vs. 26.6%). The remaining five species were strictly autogamous (3.2 %). All Perennials we recorded had the ability to reproduce sexually (Fitter, 1986). All plants observed in FM were herbs except seedlings of Crataegus laevigata, Ligustrum vulgare and Prunus spinosa (tree species), Hedera helix (liana), and Avena sativa, Brassica napus and Triticum aestivum (feral crops plants). We observed the presence of one feral OSR plant in 7 FM, 4 FM and 1 FM in 2005, 2006 and 2007 respectively.

Among the total richness (185 taxa; 37 families), 60 taxa (19 families) were present the three successive years, 53 taxa (25 families) two of the three years and 72 taxa (25 families) only one year.

FIGURE 2:



Number of mowed or sprayed field margins (FM) and FM with no weed control for each of the 3 years of the study, and number and proportion of species according to life forms and types of pollination.

The total richness in the 27 FM increased from 2005 to 2007. Indeed a total of 102 taxa were found in 2005, 122 in 2006, and 134 in 2007 (Fig. 2). The number of botanical families slightly increased over years, from 34 to 39 families. Interestingly we noted that this increase in richness coincided with the decrease of the number of mowed or sprayed FM: 21 FM in 2005 (78 % of FM), 15 in 2006 (56 %), and only 13 in 2007 (48 % - Fig. 2). Conversely, the relative proportions of Monocotyledons vs. Eudicotyledons (79 % vs. 21 %), Annuals vs. Perennials (45 % for both), and insect-pollinated vs. not insect-pollinated species (70 % vs. 30 %) were all stable from year to year. As well as total Richness, numbers of ubiquitous species, rare species, insect-pollinated species and not insect-pollinated species all increased over years.

3.2. Synthetic parameters of the plant community and their evolution over years

Co-inertia analysis showed that the synthetic tables (11 parameters) of 2006 and 2007 were significantly correlated (RV = 36.85 %; Monte-Carlo test: $p = 0.01$) whereas no correlations were found between the tables of 2005 and 2006 (RV = 17.10 %; Monte-Carlo test: $p = 0.10$), and between 2005 and 2007 (RV = 9.66 %; Monte-Carlo test: $p = 0.44$).

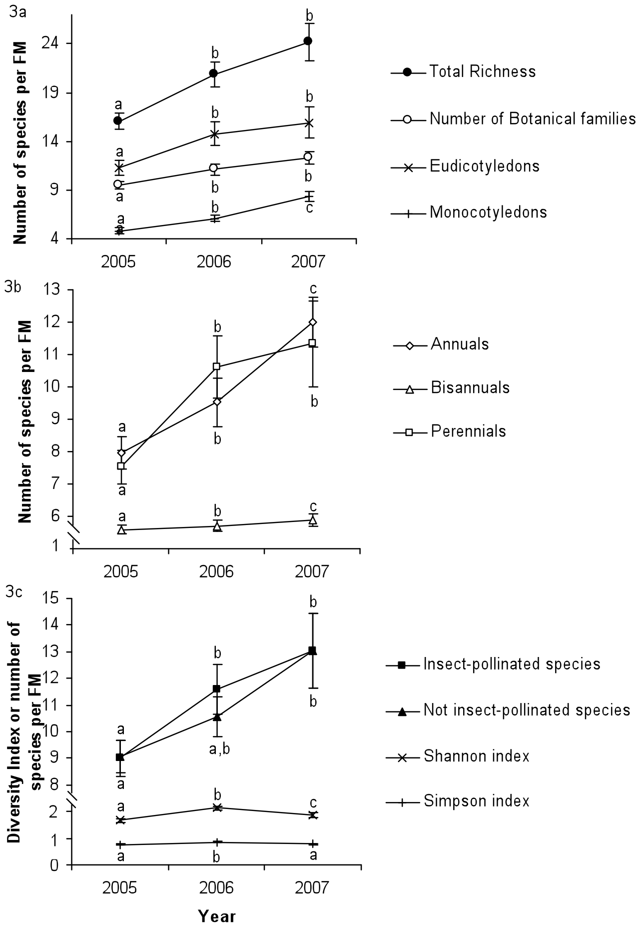
All the eight parameters of richness (Total Richness; Number of Eudicotyledons, Monocotyledons, Annuals, Biennials, Perennials, Insect-pollinated species and Not insect-pollinated species) showed a trend to increase over years (Fig. 3). This increase in parameters of richness over years was statistically significant for all parameters at least between two years (Fig. 3), except for Biennials due to a very low number of species that prevented detection of any significant trend. Evolution of diversity indices was quite different: as richness increase, both Shannon and Simpson diversity indices increased from 2005 to 2006 whereas they decreased in 2007 (Fig. 3c).

The major global differences were observed between 2005 and 2006 (8 parameters were significantly different - Fig. 3), and between 2005 and 2007 (9 significantly different parameters). The differences between 2006 and 2007 were minor (4 significantly different parameters).

3.3. Effects of weed control on the plant community

Some of the studied FM were not submitted to any weed control (no mowing nor spraying): 6 FM (22 % of 27 FM), 12 FM (44 %), and 14 FM (52 %) in 2005, 2006, and 2007 respectively. The remaining FM were either mown or sprayed. In 2007 the 8 FM adjacent to a two-way road (ca. 30 % FM) were not submitted to any control by the State agents who adopted farming practices avoiding weed control in FM adjacent to roads (validated by the French Departmental Commission for Agricultural Policy). Our results showed that the different types of

FIGURE 3:
CHANGES OVER THE THREE YEARS OF THE SYNTHETIC PARAMETERS DESCRIBING THE PLANT COMMUNITY COMPOSITION.



For each of the eleven synthetic parameters and for each pair-wise comparison, significant differences (i.e. p < 0.05) were indicated using different letters (a, b, c).

weed control had no impact on total richness in 2005 whereas it had a significant negative effect on total richness in 2006 and 2007 (Fig. 4).

In 2006 mowing had a significant effect on several synthetic parameters: plant communities in mowed FM indeed showed lower values in Richness, Shannon and Simpson Diversity, Insect-pollinated species, and Not insect-pollinated species than in FM with no weed control. Conversely, no effect of spraying in FM was detected in 2006 (Fig. 4).

The effects of weed control however differed largely from year to year. Indeed, the type of control (either mowing or spraying) that had a significant effect and the synthetic parameters that were affected a given year sometimes differ from those of the other two years. For example, the pattern observed in 2007 (i.e. a significant effect of spraying - Fig. 4) clearly differed from the pattern observed in 2006 and described above (i.e. a significant impact of mowing). More precisely, in 2007 Total Richness and numbers of Eudicotyledons, Monocotyledons and Perennials were significantly lower in sprayed FM compared to FM submitted to any control (no significant difference was observed between mowed FM and FM with no weed control).

3.4. Type of crops adjacent to FM surveyed and effects on the plant community

Land-use in fields that were adjacent to the studied FM was mostly cereal crops: 20 FM (74 % of FM), 18 (67 %), and 18 (67 %) were adjacent to a cereal crop in 2005, 2006 and 2007, respectively. Oilseed rape crops and pea crops represented a smaller proportion of adjacent crops (a few FM each year, about 10-20% of FM). Bare fallow and fallows were even rarer and only concerned one or two FM each year. We did not detect any effect of adjacent land-use on community composition. Moreover no correlation was observed between the type of weed control of FM and the land-use in field.

4. DISCUSSION

In this three-year study, we highlighted the high inter-annual variability in plant community composition in field margins (FM) in an openfield farming landscape and showed that this variability was mediated by weed control (mowing and herbicide spraying) performed in FM. The adjacent land-use (i.e. the nature of the adjacent crop and indirectly the associated farming practices) however did not have any impact of FM plant composition in our study, in opposition to field studies carried out in hedgerows farming landscapes (Baudry et al. 2000).

FIGURE 4:
SIGNIFICANT EFFECTS OF WEED CONTROLS ON THE SYNTHETIC PARAMETERS OF THE PLANT COMMUNITY

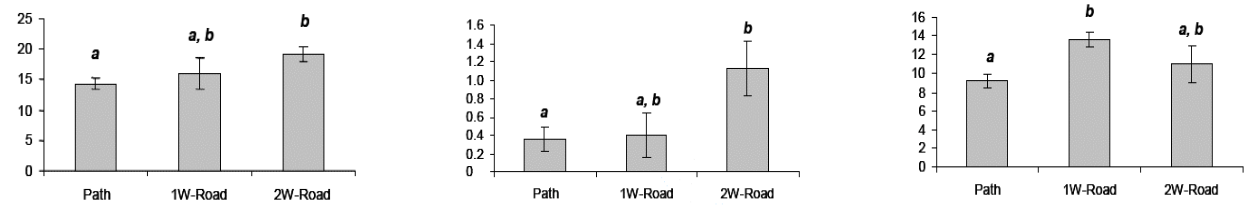


Fig. 4A: Richness in 2005
(Path *vs.* 2W-Road; W= 17.5; p= 0.0092)

Fig.4D: Number of Biennials in 2005
(Path *vs.* 2W-Road; W= 26.5; p= 0.0294)

Fig. 4G: Number of Not-Insect-pollinated species in 2006
(Path *vs.* 1W-Road; W= 6.5; p= 0.009)

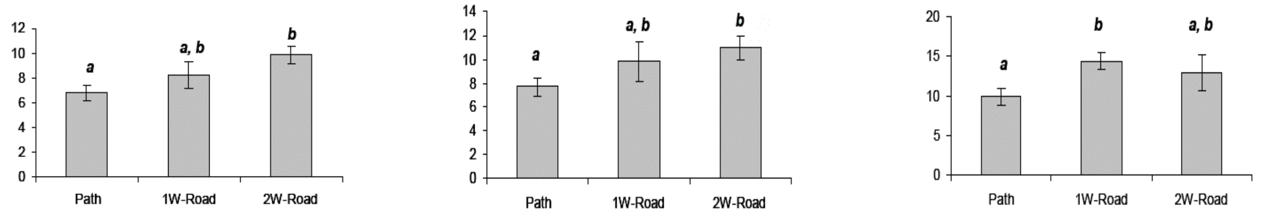


Fig. 4C: Number of Annuals in 2005
(Path *vs.* 2W-Road; W= 17; p= 0.0080)

Fig. 4E: Number of Not insect-pollinated species in 2005 (Path *vs.* 2W-Road; W= 23; p=0.0257).

Fig. 4H: Number of Insect-pollinated species in 2006
(Path *vs.* 1W-Road; W= 10; p= 0.0211)

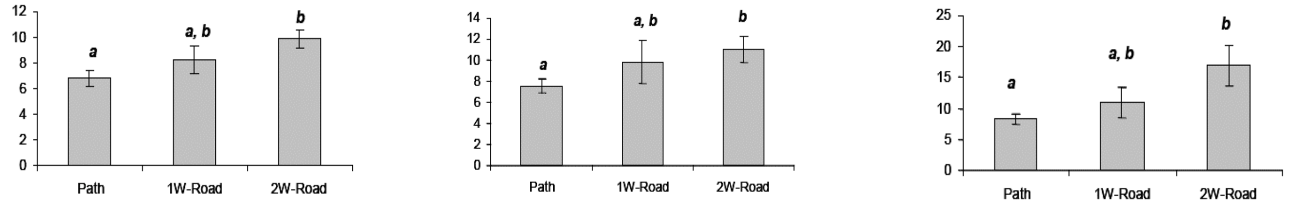


Fig. 4C: Number of Annuals in 2005
(Path *vs.* 2W-Road; W= 17; p= 0.0080)

Fig. 4F: Number of Insect-pollinated species in 2005
(Path *vs.* 2W-Road; W= 22; p= 0.0203).

Fig. 4I: Number of Perennials in 2007
(Path *vs.* 2W-Road; W=23.5; p= 0.0282)

For each year separately, results were presented only for the synthetic parameters for which at least one Mann-Whitney test was significant. For those synthetic parameters, results were however presented for all the three types of controls (NoWC, Mowing and Spraying) for better clarity. For each pair-wise comparison, significant differences (i.e. $p < 0.05$) were indicated using different letters (a, b, c). The numbers of FM that were mown, sprayed or not submitted to any weed control each year are indicated on the top of the figures ("n=").

4.1. Variability of plant community

Our results indicated a high variability in space and time of all the synthetic parameters that were chosen to describe the richness and diversity of the FM plant community within a 46-km² openfield farming landscape in western France. Indeed, these synthetic parameters presented each year a large range of variation and therefore pointed out the disparity between FM (i.e. spatial variability). Moreover, we underlined inter-annual variations of these synthetic parameters: they all generally increased with time (globally: 2005 < 2006 ≤ 2007). This increase also concerned rare species, which were present in larger proportions in species-rich FM. Rarer species may have smaller niche width and thereby an increased sensitivity to changing weed control, which would explain the high inter-annual heterogeneity in plant communities observed.

Variability of plant species richness and diversity are traditionally related to local site conditions such as nutrient and water availability (Gabriel, Thies and Tscharntke, 2005) and on factors determining the regional species pool. These factors concern approaches based on Ellenberg species indicator values and approaches based on the occurrence of species in different syntaxa in the framework of the Braun-Blanquet system – for example the number of species available for colonization (Huston 1999). Plant populations occurring within disturbed habitats (particularly numerous in farming landscapes) such as arable weeds in crop fields or plant species in FM are exposed to a frequently changing environment, i.e. in local management, abiotic site conditions, climate and historical changes on arable plant communities. Such environmental heterogeneity is in turn likely to generate great variability in community structure in space and time.

4.2. Effects of weed control on community composition

In our study, farmers mowed or sprayed only the FM that were adjacent to paths. State agents were in charge of weed control of FM adjacent to a road until 2006. The management practices probably played a large role in the evolution of the plant community structure and composition. State agents and farmers all preferentially mowed the FM they were in charge of, but farmers also occasionally sprayed FM when mowing did not eradicate all weeds (De Redon pers. comm.). Paths adjacent to FM were disturbed habitats due to intensive agricultural practices (weed control, pesticide use, and passage of tractor).

Our results showed that from 2005 to 2007 all richness parameters increased. This increase probably mainly resulted from the decrease in the proportion of FM that were mown or sprayed from year to year. Chiverton and Sotherton (1991) showed that biomass, density and diversity of plant species were affected by applications of herbicides, particularly non-selective herbicides such as glyphosate. Moreover inter-annual variations appeared to be mediated by weed control practices. Indeed since 2007, 30 % of the FM in our studied area were not submitted to any weed control according to the new application of usual good agricultural practices (GDA) validated by the French Departmental Commission for Agricultural Policy. These new practices occurring since 2007 aim at improving the quality of FM for fauna and plant preservation. The effect of practical solution was immediate in our study as seven synthetic parameters describing FM richness displayed higher values in FM adjacent to two-way roads (i.e. no weed control) than in FM adjacent to path or adjacent to one-way road (i.e. possibly mown or sprayed). Insect-pollinated plants (in addition to other parameters of richness and diversity) benefited from this policy in 2007.

Ubiquitous species were able to be maintained in species-poor FM under drastic conditions of weed control. For example, Bromus hordeaceus subsp. Hordeaceus, B. sterilis, Convolvulus arvensis, Geranium dissectum, Lolium perenne, and Poa annua were found in the three FM characterized by a very low vegetation cover (< 10 %) that probably resulted from intensive and repeated weed controls (herbicide sprayings and/or mowing).

De Cauwer et al. (2005) showed that early succession of newly created sown/unsown FM on ex-arable land, mown twice a year (with or without removal of mown material), was characterised by the replacement of Annuals in favour of Perennials, a steady increase in the importance of Monocotyledons and a decrease in non-nitrogen-fixing Eudicotyledons. We did not found any similar trend in this study: the increase of total richness did not favour any particular botanical group of plants, nor did it induce the replacement of any group in favour of another group. Numbers of Eudicotyledons and Monocotyledons, Annuals and Perennials as well as Insect-pollinated species, and Not insect-pollinated species all increased with time, simultaneously with the global increase of the total richness. Our data were thus not in accordance with the general observation that the perennial vegetation most often prevents the establishment of annual species in the FM and their spread into the crop (Marshall and Moonen, 2002).

Diversity indices mathematically increased from 2005 to 2006 then decreased in 2007 because species frequencies in 2005 and 2006 were more balanced than in 2007 and the total number of species observed increased from 2005 to 2007. In 2007 ubiquitous species were indeed more dominant than during the two previous years and rare species were less represented relatively to other species (although their numbers were greater compared to previous years).

According to Wilson (1992), a potential factor for the decline of plant species in farming landscapes is the application of chemicals. Comparisons between FM that were submitted to different types of weed control (none, mowing or spraying) confirmed this effect of weed control practices on the composition of the plant community in FM. Indeed, most of the richness and diversity parameters were affected, at least one year. Nevertheless these effects varied among years and each type of weed control had generally a significant effect only one year but non-significant effects the other two years of the study. An experimental trial with controlled parameters (number of applications of mowing and/or herbicide use, height of mowing, concentration of herbicides, and timing of application of weed control) would help to understand more precisely the effect of these agricultural practices on plant community composition.

4.3. Persistence of plants according to their reproductive strategies

The global increase in plant species richness concerned equally insect-pollinated species and those that are not: the numbers of both types increased while their relative proportions remained constant. The apparent persistence (and even the increase) of both types of species resulted from the combined effects of local recruitment (via sexual reproduction and/or vegetative multiplication), plant survival (via specific vegetative forms adapted to over winter for perennials) and seed survival (for seed-bankers).

For insect-pollinated species, their persistence and their ability to flower provided pollen and nectar resources for pollinators that could help ensure the persistence of the pollinator community (Kohler et al. 2007). The activity of honey-bees (Apis mellifera L.) and bumble-bees (Bombus terrestris L.) appears indeed to be intense in the FM of the study area (Chifflet pers. comm.). The presence of local sources of pollen and nectar is likely to prevent long-range foraging of pollinators and thereby to limit long-distance gene flow for FM insect-pollinated species in the agro-ecosystem studied here. Species that are not insect-pollinated increased also, which showed that they were able to persist in the disturbed habitats of the FM in an agro-ecosystem (Biesmeijer et al. 2006).

Concerning not-pollinated species (i.e. autogamous species and wind-pollinated species), Regal (1982) showed that plant community structure of disturbed habitats is characterised by higher proportions of self-pollinated plants compared to natural and semi-natural systems. Our studied area could not be considered as a disturbed habitat in the sense of Regal (1982) since we observed only 3.2 % of strictly autogamous species.

4.4. Effects of local characteristics of the field margins

Many studies have demonstrated the importance of local site characteristics and management practices for the occurrence of single species, community composition and species richness (see reviews in Gabriel, Thies and Tscharntke, 2005). The structure and heterogeneity of FM as disturbed areas were characterized by the width of FM and the type of weed control. In Auestad et al. (1999), FM that shelter the richest plant communities are often wide, regularly mown, adjacent to permanent grassland, and located in extensively managed agricultural landscapes. In our study area, the FM that were adjacent to a path were narrow and therefore received larger amounts of herbicide from the field edge than wider FM did. Given that weeds within fields compete with the crop for space and resources, farmers regulate their densities by using herbicides, mechanical weeding and soil tillage within the field. In FM adjacent to path, individual species may decline as a result of habitat degradation and interactions of species, as well as additional, stochastic threats. Local plant species composition was influenced by landscape complexity (aggregation of intensively managed arable fields) together with farming system (increased use of pesticides and mineral fertilizers).

4.5. Impacts on pollinators and gene flow in agro-ecosystems

Studies on pollinators and persistence of plants according to their reproductive systems should be more documented to understand complex interactions between cropped and non-crop areas. It is also important to understand the pollination processes that generate landscape-scale gene dispersal in plants, particularly in crop plants with genetically modified varieties (Gallais and Ricroch 2006). At the landscape scale in particular, gene flow among populations maintains the cohesion of a species' gene pool, thereby diminishing the potential for local adaptation and speciation.

A better understanding of the impact of weed control on pollinators and plants in openfield farming landscapes may allow an optimal design of field and margins Thus it may allow the selection of management measures that preserves the biodiversity in FM adjacent to crops and conserve farmland wildlife. These studies help to suggest farming practices that ensure a sustainable protection of plant biodiversity and pollinator community in agro-ecosystems.

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APPENDIX:

LIST OF THE 185 PLANT TAXA RECORDED IN THE 27 FIELD MARGINS SURVEYED FROM 2005 TO 2007.

Taxa code number	Genus and species	Family	E. vs. M
ACMM	*Achillea millefolium* L.	Asteraceae	E
AGEE	*Agrimonia eupatoria* L.	Rosaceae	E
AGCA	*Agrostis capillaris* L.[1]	Poaceae	M
AGSP	*Agrostis* spp.	Poaceae	M
AJRE	*Ajuga reptans* L. [a]	Lamiaceae	E
ALVI	*Allium vineale* L.	Alliaceae	M
ALMY	*Alopecurus myosuroides* Huds.[2]	Poaceae	M
ALSP	*Alopecurus* spp.	Poaceae	M
AMRE	*Amaranthus retroflexus* L. [b]	Amaranthaceae	E
AMSP	*Amaranthus* spp. [c]	Amaranthaceae	E
ANAR	*Anagallis arvensis* L.	Myrsinaceae	E
ANSP	*Anthriscus* spp.	Apiaceae	E
ANSX	*Anthriscus sylvestris* (L.) Hoffmann[3]	Apiaceae	E
ATSP	*Anthyllis* spp. [b, c]	Fabaceae	E
ARLA	*Arctium lappa* L. [4]	Asteraceae	E
AREL	*Arrhenatherum elatius* (L.) P. Beauv.	Poaceae	M
ARVU	*Artemisia vulgaris* L.	Asteraceae	E
ATSP	*Atriplex* spp.	Chenopodiaceae	E
AVSA	*Avena sativa* L.	Poaceae	M
BEPX	*Bellis perennis* L.	Asteraceae	E
BRPI	*Brachypodium pinnatum* (L.) P. Beauv. [b]	Poaceae	M
BASY	*Brachypodium sylvaticum* (Huds.) P. Beauv. [a]	Poaceae	M
BRNO	*Brassica napus* L.	Brassicaceae	E
BRNI	*Brassica nigra* (L.) W. D. J. Koch	Brassicaceae	E
BRSP	*Brassica* spp. [b]	Brassicaceae	E
BRAV	*Bromus arvensis* L.	Poaceae	M
BRER	*Bromus erectus* Huds. [b]	Poaceae	M
BRHO	*Bromus hordeaceus* L. subsp. *hordeaceus* [5, d]	Poaceae	M
BRRI	*Bromus rigidus* Roth [6, c]	Poaceae	M
BRST	*Bromus sterilis* L. [d, e, f]	Poaceae	M
CASP	*Calendula* spp. [b]	Asteraceae	E
CABU	*Capsella bursa-pastoris* (L.) Medik.	Brassicaceae	E
CANU	*Carduus* spp. [c]	Asteraceae	E
CAEC	*Carex echinata* Murray [c]	Cyperaceae	M
CAML	*Carex muricata* L.	Cyperaceae	M
CEJA	*Centaurea jacea* L.	Asteraceae	E
CENI	*Centaurea nigra* L.	Asteraceae	E
CESP	*Centaurea* spp.	Asteraceae	E
CEFT	*Cerastium arvense* L. ssp. *strictum* (L.) Ugborogho	Caryophyllaceae	E
CESE	*Cerastium semidecandrum* L.	Caryophyllaceae	E
CERS	*Cerastium* spp.	Caryophyllaceae	E
CHTE	*Chaerophyllum temulum* L.	Apiaceae	E
CHAA	*Chenopodium album* L.	Chenopodiaceae	E
CIAR	*Cirsium arvense* (L.) Scop.	Asteraceae	E
COAR	*Cirsium vulgare* (Savi) Ten. [7]	Asteraceae	E
CORC	*Convolvulus arvensis* L. [d, e, f]	Convolvulaceae	E
COCA	*Conyza canadensis* (L.) Cronquist [8]	Asteraceae	E
COSP	*Coronopus squamatus* (Forsk.) Ascherson [c]	Brassicaceae	E
CRMO	*Crataegus monogyna* Jacq. [b]	Rosaceae	E
CRCA	*Crepis capillaris* (L.) Wallr. [9]	Asteraceae	E
CRSE	*Crepis setosa* Haller f.	Asteraceae	E
CRSP	*Crepis* spp.	Asteraceae	E
CRLA	*Cruciata laevipes* Opiz [10, b]	Rubiaceae	E
DAGG	*Dactylis glomerata* L.	Poaceae	M
DACA	*Daucus carota* L.	Apiaceae	E
ELRE	*Elytrigia repens* (L.) Desv. ex Nevski [11, f]	Poaceae	M
ELSP	*Elytrigia* spp.	Poaceae	M
EPSP	*Epilobium* spp.	Onagraceae	E
ERCI	*Erodium cicutarium* (L.) L'Her.	Geraniaceae	E
ERVE	*Eryngium campestre* L.	Apiaceae	E
EUCY	*Euphorbia cyparissias* L. [c]	Euphobiaceae	E
EUHE	*Euphorbia helioscopia* L.	Euphobiaceae	E
FACO	*Fallopia convolvulus* (L.) A. Love [12]	Polygonaceae	E
FEAR	*Festuca arundinacea* Schreb.	Poaceae	M
FEOV	*Festuca ovina* L. [c]	Poaceae	M
FEPP	*Festuca pratensis* Huds.	Poaceae	M
FERR	*Festuca rubra* L.	Poaceae	M
FESP	*Festuca* spp.	Poaceae	M
GAAP	*Galium aparine* L. [e, f]	Rubiaceae	E
GAMO	*Galium mollugo* L.	Rubiaceae	E
GASP	*Galium* spp.	Rubiaceae	E
GAVE	*Galium verum* L.	Rubiaceae	E
GECO	*Geranium columbinum* L.	Geraniaceae	E
GEDI	*Geranium dissectum* L. [d, e, f]	Geraniaceae	E
GEMO	*Geranium molle* L.	Geraniaceae	E
GEPU	*Geranium pusillum* L.	Geraniaceae	E
GEPY	*Geranium pyrenaicum* Burm. F.	Geraniaceae	E
GERT	*Geranium rotundifolium* L.	Geraniaceae	E

GESP	*Geranium* spp.	Geraniaceae	E
GLHE	*Glechoma hederacea* L.	Lamiaceae	E
HEHH	*Hedera helix* L. [a]	Araliaceae	E
HEAN	*Helianthus annuus* L.	Asteraceae	E
HYSP	*Heracleum sphondylium* L.	Apiaceae	E
HOMU	*Hordeum murinum* L. [c]	Poaceae	M
HOVU	*Hordeum vulgare* L. subsp. *vulgare* [13, a]	Poaceae	M
HYPE	*Hypericum perforatum* L.	Hypericaceae	E
HYSP	*Hypericum* spp. [a]	Hypericaceae	E
HYRA	*Hypochaeris radicata* L. [c]	Asteraceae	E
JUCO	*Juncus conglomeratus* L. [a]	Juncaceae	M
KISS	*Kickxia spuria* (L.) Dumort. [14, c]	Plantaginaceae	E
KNAR	*Knautia arvensis* (L.) Coult.	Dipsacaceae	E
LASE	*Lactuca serriola* L. [15]	Asteraceae	E
LAAM	*Lamium amplexicaule* L. [a]	Lamiaceae	E
LAHY	*Lamium hybridum* Vill.	Lamiaceae	E
LAPU	*Lamium purpureum* L.	Lamiaceae	E
LACO	*Lapsana communis* L. [c]	Asteraceae	E
LAPR	*Lathyrus pratensis* L.	Fabaceae	E
LASP	*Lathyrus* spp.	Fabaceae	E
LEVU	*Leucanthemum vulgare* Lam. [16, c]	Asteraceae	E
LIVU	*Ligustrum vulgare* L. [b]	Oleaceae	E
LIAR	*Lithospermum arvense* L. [17]	Boraginaceae	E
LOPE	*Lolium perenne* L. [d, e]	Poaceae	M
LOCO	*Lotus corniculatus* L.	Fabaceae	E
MANE	*Malva neglecta* Wallr. [18]	Malvaceae	E
MACH	*Matricaria chamomilla* L.	Asteraceae	E
MADI	*Matricaria discoidea* DC	Asteraceae	E
MAPE	*Matricaria perforata* Mérat [19]	Asteraceae	E
MASP	*Matricaria* spp.	Asteraceae	E
MESP	*Medicago* spp. [c]	Fabaceae	E
MEAN	*Mercurialis annua* L.	Euphobiaceae	E
MYSP	*Mysosotis* spp.	Boraginaceae	E
ONRE	*Ononis repens* L.	Fabaceae	E
PARH	*Papaver rhoeas* L.	Papaveraceae	E
PASP	*Papaver* spp.	Papaveraceae	E
PASA	*Pastinaca sativa* L. [b]	Apiaceae	E
PEMA	*Persicaria maculosa* Gray [20, b]	Polygonaceae	E
PIHI	*Picris hieracioides* L.	Asteraceae	E
PISP	*Picris* spp.	Asteraceae	E
PISA	*Pisum sativum* L. [c]	Fabaceae	E
PLCO	*Plantago coronopus* L. [a]	Plantaginaceae	E
PLLA	*Plantago lanceolata* L. [d, e]	Plantaginaceae	E
PLMM	*Plantago major* L.	Plantaginaceae	E
POAN	*Poa annua* L. [d, e, f]	Poaceae	M
POPR	*Poa pratensis* L. [d, e, f]	Poaceae	M
POSP	*Poa* spp. [b]	Poaceae	M
POTR	*Poa trivialis* L.	Poaceae	M
POAV	*Polygonum aviculare* L.	Polygonaceae	E
POoL	*Portulaca oleracea* L. [a]	Portulacaceae	E
PORE	*Potentilla reptans* L.	Rosaceae	E
POSP	*Potentilla* spp.	Rosaceae	E
PRSP	*Prunus spinosa* L.	Rosaceae	E
RAAA	*Ranunculus acris* L.	Ranunculaceae	E
RAFI	*Ranunculus ficaria* L. [21, b]	Ranunculaceae	E
RARE	*Ranunculus repens* L.	Ranunculaceae	E
RASA	*Ranunculus sardous* Crantz	Ranunculaceae	E
RASP	*Ranunculus* spp.	Ranunculaceae	E
RARA	*Raphanus raphanistrum* L.	Brassicaceae	E
RELU	*Reseda lutea* L. [c]	Resedaceae	E
ROSP	*Rosa* spp.	Rosaceae	E
RUFR	*Rubus fruticosus* L.	Rosaceae	E
RUAC	*Rumex acetosa* L.	Polygonaceae	E
RUOO	*Rumex acetosella* L. [c]	Polygonaceae	E
RUCR	*Rumex crispus* L.	Polygonaceae	E
RUSA	*Rumex sanguineus* L. [a]	Polygonaceae	E
RUSP	*Rumex* spp.	Polygonaceae	E
SASP	*Sagina* spp. [a]	Caryophyllaceae	E
SAPA	*Salvia pratensis* L.	Lamiaceae	E
SEJA	*Senecio jacobaea* L.	Asteraceae	E
SEVU	*Senecio vulgaris* L.	Asteraceae	E
SEMO	*Seseli montanum* L. [c]	Apiaceae	E
SHAR	*Sherardia arvensis* L.	Rubiaceae	E
SIDI	*Silene dioica* (L.) Clauv. [22]	Caryophyllaceae	E
SIVU	*Silene vulgaris* (Moench.) Garcke [23]	Caryophyllaceae	E
SIAR	*Sinapis arvensis* L. [a]	Brassicaceae	E
SYOF	*Sisymbrium officinale* (L.) Scop.	Brassicaceae	E
SONI	*Solanum nigrum* L. [c]	Solanaceae	E
SOAS	*Sonchus asper* (L.) Hill	Asteraceae	E
SOOL	*Sonchus oleraceus* L.	Asteraceae	E
STGR	*Stellaria graminea* L.	Caryophyllaceae	E
STMM	*Stellaria media* (L.) Vill. [a]	Caryophyllaceae	E
TAOF	*Taraxacum officinale* aggr. [24]	Asteraceae	E
TOAN	*Torilis anthriscus* (L.) C. C. Gmel. [25]	Apiaceae	E
TRDU	*Tragopogon dubius* Scop.	Asteraceae	E
TRAR	*Trifolium arvense* L.	Fabaceae	E
TRCA	*Trifolium campestre* Schreb.	Fabaceae	E

TRDU	*Trifolium dubium* Sibth. [26]	Fabaceae	E
TRIN	*Trifolium incarnatum* L.	Fabaceae	E
TRRE	*Trifolium repens* L.	Fabaceae	E
TRFL	*Trisetum flavescens* (L.) P. Beauv. [c]	Poaceae	M
TRAE	*Triticum aestivum* L.	Poaceae	M
ULMI	*Ulmus minor* Mill. [27]	Ulmaceae	E
URDI	*Urtica dioica* L.	Urticaceae	E
VALO	*Valerianella locusta* (L.) Laterr. [28]	Valerianaceae	E
VBOF	*Verbena officinalis* L.	Verbenaceae	E
VEAR	*Veronica arvensis* L.	Plantaginaceae	E
VECH	*Veronica chamaedrys* L.	Plantaginaceae	E
VEHH	*Veronica hederaefolia* L.	Plantaginaceae	E
VEOF	*Veronica officinalis* L.	Plantaginaceae	E
VEPE	*Veronica persica* Poir.	Plantaginaceae	E
VICR	*Vicia cracca* L. [c]	Fabaceae	E
VISA	*Vicia sativa* L.	Fabaceae	E
VISP	*Vicia spp.* [29]	Fabaceae	E
VIAR	*Viola arvensis* Murray	Violaceae	E
VUMY	*Vulpia myuros* (L.) C. C. Gmel.	Poaceae	M

NB 1:

E = Eudicotyledon & M = Monocotyledon

NB 2:

The taxa code number follows Flora Europaea (Turin et al. 1980). [a] rare species observed in 2005, [b] in 2006, and [c] in 2007 ; [d] ubiquitous species observed in 2005, [e] in 2006 and [f] in 2007.

NB 3:

[1] *syn. A. tenuis* Sibth.; [2] *syn. A. agrestis* L.; [3] *syn. Chaerophylum silvestre* L.; [4] *syn. Lappa major* L.; [5] *syn. B. mollis* L.; [6] *syn. Bromus maximus* Desf.; [7] *syn. Cirsium lanceolatum* (L.) Scop.; [8] *syn. Erigeron canadensis* L.; [9] *syn. C. virens* L.; [10] *syn. Galium cruciata* Scop.; [11] *syn. Agropyron repens* (L.) P. Beauv.; [12] *syn. Polygonum convolvulus* L.; [13] *syn. Hordeum hexastichon* L.; [14] *syn. Linaria spuria* L.; [15] *syn. Lactuca scariola* L.; [16] *syn. Chrysanthemum leucanthemum* L.; [17] *syn. Buglossoides arvensis* L.*;* [18] *syn. Malva rotundifolia* auct. non L.*;* [19] *syn. Matricaria inodora* L.; [20] *syn. Polygonum persicaria* L.; [21] *syn. Ficaria ranonculoides* Roth; [22] *syn. Lychnis dioica;* [23] *syn. S. inflata* Sm. var. *prostrata* Gaudin; [24] *syn. T. dens-leonis* Desf.; [25] *syn. T. japonica* (Houtt.) DC. or *Anthriscus caucalis* M. Bieb.; [26] *syn. T. minus* Sm.; [27] *syn. Ulmus campestris* auct. ; [28] *syn. V. olitoria* (L.) Pollich; [29] *syn. Ervum* spp.

Microtus agrestis

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Plant and spider communities benefit differently from the presence of planted hedgerows in highway

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PRESENTATION

TITRE

Communautés de plantes et communautés d'araignées bénéficient différemment de la présence des plantations de haies au sein des dépendances vertes des bords d'autoroutes.

RESUME

Les bords de routes peuvent jouer un rôle crucial comme refuge pour la flore et la faune native au sein des paysages anthropisés. Il apparaît que l'influence des choix de construction, comme les plantations d'espèces forestières, sur la biodiversité des bords de route a peu été étudiée alors même que la présence de haies, par exemple, pourrait jouer un rôle d'habitat pour les espèces forestières. A l'aide d'une méthodologie standardisée, nous avons étudié l'impact des plantations de haies sur deux groupes taxonomiques (les plantes et les araignées) établis en bords de route au sein des paysages d'agriculture intensive. Nous avons examiné la richesse spécifique et fonctionnelle des communautés au sein de sites avec et sans haies. Au niveau des sites, la réponse des communautés de plantes et d'araignées ont répondu très différemment : les haies été associées à une augmentation significative de la richesse en plantes (plus grande diversité α) et à une même richesse en araignées. Les communautés de plantes des sites sans haies sont apparues comme partie des communautés de plantes des sites avec haies ; alors que pour les communautés d'araignées, celles des sites sans haies étaient complémentaire de celles des sites avec haies (plus grande diversité β). La présence de haies étaient aussi associée à une augmentation de la diversité taxonomique et des traits fonctionnels au niveau du paysage (diversité γ) grâce à une augmentation de la diversité β pour les plantes comme pour les araignées. Nos résultats suggèrent qu'un mélange d'habitats avec haies et d'habitats ouverts est crucial pour le maintien de la biodiversité à l'échelle des paysages. En procurant des informations pour les professionnels de la route et aux décideurs politiques quant à leur influence potentielle sur la biodiversité, ces résultats ont des implications directes pour la gestion des réseaux routiers et de leurs dépendances vertes.

MOTS CLEFS

Diversité α, diversité β, biodiversité, traits fonctionnels, gestion des bords de route, revégétalisation & analyses RLQ.

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Plant and spider communities benefit differently from the presence of planted hedgerows in highway verges

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ABSTRACT

Road verges should play a crucial role as a refuge for native flora and fauna in human dominated landscapes. However, the influence of construction choices, such as plantation of woody species, on the biodiversity supported by roadsides has received little attention, although the presence of hedgerows in roadsides is likely to enhance their role as a refuge, notably for woodland species. Using standardised methods, we assessed the impact of planted hedgerows on two taxonomic groups (plants and spiders) inhabiting highway verges within an intensive agricultural landscape. We examined community richness, taxonomic and functional composition in sites with and without planted hedgerows. At the site level, the response of plant and spider communities to the presence of planted hedgerows differed markedly: hedgerows were associated with significantly higher plant richness (higher α-diversity), but similar spider richness. Plant communities in sites without hedgerows appeared as a subset of communities in sites with hedgerows, whereas spider communities in non-planted sites were complementary to that of planted sites (increased β-diversity). The presence of planted hedgerows was also associated with increased taxonomic and functional trait diversity at the landscape level (γ-diversity), through an increased β-diversity in both plants and spiders. Our results thus suggest that a mosaic of planted hedgerows and grassland habitats is crucial for the maintenance of biodiversity at a landscape scale. By providing information for road practitioners and policy makers regarding their potential impact on biodiversity, these results have important direct implications for the management of road networks.

1. Introduction

Road networks, which have expanded over large areas with human population growth (Watts et al., 2007), are known to influence landscape structure and have major negative impacts on ecosystems dynamics (Trombulak and Frissell, 2000). However, the potential biological value of road verges has also long been recognised (Way, 1977) with possible contribution to the conservation of indigenous flora (Spooner et al., 2004; O'Farrell and Milton, 2006) and fauna (Meunier et al., 2000; Ries et al., 2001). This role as a refuge depends on the surrounding landscape: in natural habitats, generally supporting a high species diversity, road verges may not serve as a refuge (O'Farrell and Milton, 2006) and even have negative

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effects, notably by promoting invasion by non-indigenous species (Hansen and Clevenger, 2005; Brown et al., 2006). In contrast, in human dominated areas such as intensive agricultural landscapes, where non-agricultural habitats (e.g. edges) are critical to the conservation of biological diversity and ecological processes (Burel, 1996), road verges should play a crucial role as a refuge and as ecological corridors (Tikka et al., 2001; Smart et al., 2006).

Road verges support intensively managed habitats, which are artificially created on bare soil after road construction (restoration) and subsequently experience frequent mowing. This has direct consequences on the communities they can support and there is a need to identify and promote biodiversity-friendly construction and management practices, in order to optimise the role of road verges as a refuge for wild flora and fauna. As in any habitat, community composition in road verges is controlled by an array of distinct factors acting as filters at both landscape and local scales (Bochet et al., 2007), few of which can easily be manipulated to optimise biodiversity conservation. There have been extensive studies addressing the influence of management on roadside biodiversity, notably mowing frequency and timing (Schaffers, 2002; Jantunen et al., 2007). In contrast, construction choices have received much less attention, with the exception of revegetation with herbaceous species (Matesanz et al., 2006; Leps et al., 2007). In particular, the role of planted trees in the maintenance of diversity in road verges was never considered to our knowledge, even though the presence of hedgerows along roadsides is likely to enhance their role as a refuge, by increasing the diversity of microhabitats. This role of hedgerows as a refuge, however, depends on their ability to receive a significant proportion of the local species pool and of the functional diversity they shelter (Roy and de Blois, 2006). Outside of roadsides, hedgerows appear to be a sustainable habitat and even functional corridors (flora: Corbit et al., 1999; Roy and de Blois, 2008; fauna: Maudsley, 2000; Griffiths et al., 2007; Davies and Pullin, 2007 for a review) even though other authors consider hedgerows as sink habitats (McCollin et al., 2000; Sitzia, 2007).

The aim of the present study is to evaluate the effect of hedgerows on the surrounding biodiversity of highway verges at different spatial scales, and on the distribution of biodiversity within and among sites in the context of an intensive agricultural landscape. Highway verges offer ideal settings to study the influence of hedgerows, because, in contrast to other habitats, the influence of history on community composition is strongly reduced, due to common history of all study sites. To address the impact of hedgerows on biodiversity, we compared the community richness and composition in sites with and without planted hedgerows for two taxonomic groups (plants and spiders), because no single taxon can inform on the ecological behaviour of other groups (Lovell et al., 2007); In addition, in the context of the search for biodiversity indicators, it is of great importance to better assess the characteristics of different taxa (Pearman and Weber, 2007). These two groups differ in their ecological requirements and dispersal capabilities and are likely differentially affected by the presence of hedgerows: plant community composition is expected to be controlled mostly by soil and light characteristics (Schaffers, 2002) whereas spider communities are known to be highly sensitive to habitat structure and microclimatic conditions (Marc et al., 1999; Entling et al., 2007; see also Beals, 2006) and may exhibit variation on much smaller time scales than plants. By comparing similar grassland habitats in sites with and without planted hedgerows, we specifically examined (1) whether the presence of planted hedgerows influenced the surrounding specific and functional trait diversity, (2) whether plant and spider communities exhibited different environmental patterns, and (3) whether artificial hedgerows resulted in a higher overall biodiversity supported by highway verges.

2. Methods

2.1. *Study area*

This study was conducted within a 50 km section of highway A11 (1°39′30″E–48°29′15″N to 1°03′E–48°14′N) in the Beauce region, an intensive agricultural landscape, West of Paris, France. The Beauce covers about 6000 km^2 and is characterised by a temperate climate and high base status soils. This study area is a fairly typical modern open-field landscape with a complete absence of hedgerows and very few semi-natural habitats, such as remnant woodland patches. Agricultural practices consist mostly of intensive cereal production.

A11 Highway was built 30 years ago and enlarged in 1992. At this time, the newly created verges, devoid of vegetation, were sown with the same seed mixture, including mainly *Leguminosae* and *Poaceae* species (*Festuca rubra*, *Festuca ovina*, *Lolium perenne*, *Festuca arundinacea*, *Bromus inermis*, *but also Medicago lupulina*, *Lotus corniculatus*, *Trifolium arvense* and *Achillea millefolium*), along the whole highway section studied, and were planted with trees at discrete locations (*Cofiroute* construction company, personal communication). In the following, we define hedgerows as linear patches of planted shrubs and trees longer than 150 m, following the definition of Baudry et al. (2000). Sites with and without planted hedgerows are subsequently referred to as "WT" sites and "WO" sites, respectively. Management practices of highway verges depend on distance from the carriageway edge: all verges receive a visibility cut up to 6 m from the road edge twice a year, but areas behind this safety zone are only cut once a year, whereas vegetation growing under planted trees is seldom, if ever, cut.

2.2. *Field sampling and data collection*

2.2.1. *Field sampling*

We selected 25 field sites (Fig. 1) based on the following structural and biological criteria: highway sections were "road-cuts" (resulting from excavation), with edges of the same shape (width = 16.1 m ± 0.7 SE, height = 4 m ± 0.3 SE), slope (30° ± 2.4 SE) and aspect, and were bordered by crop fields. Within the 50 km study area, we selected 11 sites with planted hedgerows (WT) along these criteria, using geomorphic maps, aerial photographs, and field surveys. Hedgerows consisted of mature deciduous trees in the canopy layer, and were on average above 3 m tall and 5 m wide (5.2 ± 0.5 SE, varying between 3 and 8 m). They were generally pluri-spe-

Fig. 1 – Location of the 25 study sites. ▲: sites without hedgerows (WO), O: sites with planted hedgerows (WT); line: A11 highway; grey patches: woodland patches within a 2000 m radius buffer.

cific, including most frequently the following planted species: *Acer campestre*, *Acer platanoides*, *Acer pseudoplatanus*, *Alnus cordata*, *Betula verrucosa*, *Betula pendula*, *Carpinus betulus*, *Cornus alba*, *Cornus mas*, *Cornus sanguinea*, *Corylus avellana*, *Cotinus coggygria*, *Fraxinus excelsior*, *Prunus avium*, *Prunus mahaleb*, *Prunus padus*, *Prunus spinosa*, *Quercus sessiliflora*, *Quercus robur*, *Robinia pseudo-acacia*, *Salix caprea*, *Sambucus nigra*, *Sorbus aucuparia*, *Sorbus torminalis*, *Viburnum lantana*, *Viburnum opulus*. When possible, one site without hedgerows (WO) was chosen within 1.5 km of a given site with hedgerows (WT). We thus selected seven pairs of geographically close sites with and without hedgerows and completed this sample with isolated WT ($n = 4$) and WO sites ($n = 7$), for a total of 25 sampled sites (11 WT–14 WO). WO sites were always located at least 50 m from the closest tree.

2.2.2. *Data collection*

All inventories were performed through a 60 m transect running parallel to the carriageway, located at least 7.5 m from the carriageway edge and at 8.2 ± 0.6 SE from the neighbouring crop field. Such location corresponded to extensive management practices. In WT sites, transects were also located 0.5–1 m from the canopy of planted trees (vertical projection of lower branches), and started and ended at least 50 m away from the end of the hedgerow, to avoid border effects.

For flora, sites were surveyed from May 10th to June 10th 2006. In each site, all vascular plant species were inventoried once in five 1 m^2 quadrats regularly distributed over the 60 m transect. Almost all taxa were identified to species level according to the International Plant Names Index. A few species that were difficult to distinguish when not in flower or fruit (*Vicia gr tetrasperma-hirsuta* for example) were grouped.

Spiders were sampled using five pitfall traps per site. Traps were placed within 30 cm of each vegetation quadrat on the 60 m transect and were left in place from May 10 to June 10 2006. Each trap was 8.5 cm wide in diameter and 10 cm deep, and contained ethylene glycol, a non-attractive preservative. Two pitfall traps were destroyed and were omitted in the analyses ($n = 123$). Adult spiders were identified to species level following the nomenclature of Platnick (2007). We failed to identify three individuals to species level; nevertheless, because they were obviously different from other species, they were conserved for analyses. Spider juveniles were generally not identifiable to species level and were not considered.

Sites characteristics: sampling design was defined to test impact of hedgerows, so that the variation of other environmental variables was minimised. We recorded sites characteristics and checked the absence of correlation between presence/absence of hedgerows and other environmental factors using Pearson correlations, to check that the effects of hedgerows were not confounded with other environmental factors. For each site, we recorded geographic coordinates, verge width (final variation: 10–20 m), adjacent land use type (crop vs. temporary fallow), slope, distance from crop edge, aspect, and soil nutrient status. The latter was characterised via determination of soil pH, as well as phosphorus (P_2O_5, Olsen%), salt (Na_2O), and total nitrogen (%) concentration in a bulked representative soil sample, taken to a depth of 10 cm in each of five plots and subsequently mixed. For each site, we also recorded the distance from the nearest woodland patch (411.6 m ± 70.4 SE), as well as the total woodland area within 250 m, 500 m and 1000 m radius buffers, using aerial photographs and the Geographical Information System package ARCGIS 8 (ESRI, 2000). In this open-field system, woodland patches can be considered as the main natural habitat in Beauce. Consequently, we considered that they could be important reservoirs of species compared to others habitats, providing a pool of both plants and spiders from which species were able to disperse and potentially colonise highway verges.

2.3. *Species attributes*

To address the influence of hedgerows on community functional composition, we characterised species based on readily available functional traits. As such data were analysed using multivariate methods that are highly sensitive to "rare" species or classes, we chose to group some classes to minimise the resulting bias (see caption of Fig. 2). For plants, these functional traits include Raunkier types, maximum adult height, and seed dispersal mode, scored according to a literature review (Supplementary material 1). For spiders, we retained hunting strategies, body size and aerial dispersal behaviour (ballooning vs. non-ballooning species). Hunting strategies (diurnal cursorial hunters, nocturnal cursorial hunters, ambush hunters, and web-builders) and species-specific body sizes were derived from Roberts (1985, 1987). As body sizes usually differ between sexes, the mean value of male and female median sizes was retained. Species exhibiting aerial dispersal using silk (ballooning) are considered species with great dispersal capabilities (Bell et al., 2005 for review) and were defined using the world species list of ballooning Araneomorphae established by Bell et al. (2005). All species that did not appear in this list were considered "non ballooners", which may be incorrect because the list is not exhaustive. However, as we worked with common species frequently found in agricultural landscapes and whose biology is generally well-known, we expect that this bias should be small. For both plants and spiders, a given taxon was assigned attributes only when it was identified to species level and observed in more than four instances (quadrats or traps), which yielded a total of 46 plant species (100% of species observed in more than four quadrats) and 56 spider species (84% of species detected in more than four pitfalls) for which all traits were scored.

2.4. *Data analysis*

To remove the obvious increase in plant species richness due to the simple presence of planted species and examine solely indirect effects of planted hedgerows on plant community composition, we chose to work with spontaneous plant species only. Hence, unless otherwise stated, shrub or tree species that were observed in the herbaceous strata in a site where they had been planted (as inferred from historical plantation map data, *Cofiroute–Vegetude*) were discarded in subsequent analyses, i.e. they were excluded of this particular site; not, however, that these species were retained in other sites. In addition, for spider species, we chose to work with presence/absence data only, so that a range of different analyses, some of which valid with presence/absence data only, could be used on the same dataset. With few exceptions, including abundance data yielded very similar results. All analyses were conducted with (R Development Core Team, 2007).

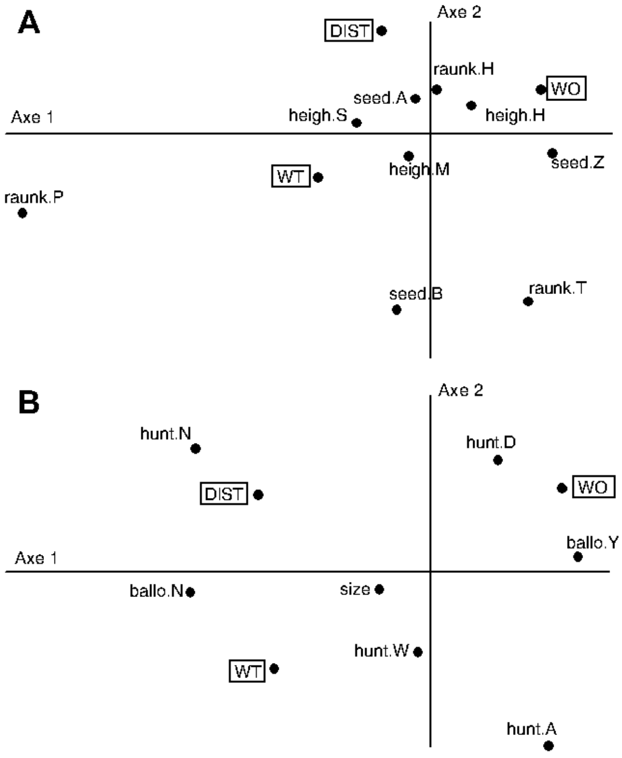

Fig. 2 – RLQ analysis ordination biplot illustrating relationships among environmental and plant species attributes (A) and spider species attributes (B). The analysis is based on presence/absence of species with known attributes that were observed in more than four samples (46 plant species and 56 spider species). (1) *Environmental parameters*. WT: sites with planted hedgerows, WO: sites without hedgerows; dist: distance to the distance to the nearest woodland patch. (2) *Plant traits* (A). rank: raunkier type: H: Hemicryptophytes and Geophytes (28 species), P: Phanerophytes and Chamaephytes (6), T: Therophytes (12); heigh: maximum adult height: S: small (0–45 cm: 9), M: medium (46–100 cm: 19), H: high (100 and more: 18); seed: Seed dispersal: A: anemochory (30), B: barochory and autochory (9), Z: zoochory (7). (2) *Spider traits* (B). ballo: ballooning behaviour: Y: Yes (30), N: No (26); size; hunt: hunting strategies: W: diurnal wandering (18); N: nocturnal wandering (10); W: web-builders (21); A: ambush hunters (7).

2.4.1. *Species richness*

To assess the effect of hedgerows at different spatial scales, we considered species richness at the quadrat scale (S_{local}, plants only), at the site scale (S_{site}, in a transect of 60 m, plants and spiders) and at the highway section scale, within each type of site (S_{type}, within WT or WO types of sites, plants and spiders). For plants, we considered local species richness (S_{local}) within quadrats because we assumed that all species were detected, whereas for spiders, pitfall trapping is known to be influenced by the activity-density of species (Melbourne, 1999). In addition, analysis on local spider richness showed no effect of any environmental variable. Comparing observed species richness across sites and habitats might not be valid in case of unequal species detection probabilities (Boulinier et al., 1998; Nichols et al., 1998). We, therefore, used statistical methods derived from capture–recapture approaches in population or community dynamics to estimate species richness and changes in community composition As in recent studies (Lekve et al., 2002; Doherty et al., 2003; Selmi and Boulinier, 2003; Devictor et al., 2007; Kerbiriou et al., 2008) addressing richness estimation and differences in community composition from species count data, we used the program COMDYN (Hines et al., 1999), based on the jackknife estimator of Burnham and Overton (1979). To estimate species richness at the site level (S_{site}), we used quadrat or pitfall captures as replicates. To estimate richness at type of site level (S_{type} within WT or WO type of sites), we used sites as replicates ($n = 11$): analyses were based on presence–absence data at the site level, pooled across the five quadrats for plants and the five pitfall traps for spiders. As 11 replicates only were available for WT sites, we sampled randomly without replacement 11 sites out of 14 in the WO group; this sampling procedure was performed 15 times. Each of 15 pairs of (1) 11 sampled WO sites plus (2) the 11 WT sites was input into program COMDYN to obtain estimated richness per type of site. The estimated richness in each type of site was then characterised using the mean and standard error of the 15 outputs.

We analysed the combined influence of hedgerows and environmental variables on both local species richness for plants (S_{local}) and estimated richness per site for plants and spiders (S_{site}), using linear models and analyses of variance, as follows: for each taxonomic group (plants and spiders) and dependent variable (S_{local} and S_{site}), we selected the most informative model using a stepwise algorithm based on Akaike Information Criterion (StepAIC, library MASS in R) starting from the most complete model including all environmental variables likely to influence species richness (listed above: see “site characteristics”). Plant richness was included as an additional environmental variable for spiders. Data were log-transformed when necessary to meet normality assumptions; the effects of variables retained by the stepwise algorithm were tested adjusted to others and the nested structure of the data was taken into account (local richness per quadrat nested within site: lme, library nlme in R).

2.4.2. *Species composition*

We investigated a possible spatial autocorrelation in community composition by performing mantel tests (mantel.randtest, library ade4 in R) to assess the correlation between matrices of geographic vs. specific distances among sites. For plants and spiders, matrices of specific distances across sites were computed with the Jaccard index on presence/absence matrices. This effect was also tested separately within both types of edges (WT and WO sites). For these analyses, all species were retained.

We also examined the effect of hedgerows on the composition of plant and spider communities in the herbaceous strata using two types of multivariate analyses: Constrained Analysis of Principal Coordinates (CAP) and partial Constrained Correspondence Analysis (pCCA), followed-up by ANOVA-like permutation tests ($n = 999$) to assess significance of effects. CAP (capscale, library vegan in R, see Anderson and Willis, 2003) is an ordination method similar to Redundancy Analysis (RDA), i.e. exploring the relationship between two sets of variables, but allowing non-Euclidean dissimilarity indices. It was used to examine whether species similarity among sites depended on the presence/absence of hedgerows, and on the distance to nearest woodland patch, which was used as a covariable. It was performed on presence/absence matrices across sites, using the Jaccard index. pCCA (cca, library vegan in R, ter Braak, 1986) is an ordination method based on CCA, a constrained ordination method developed to relate community composition to known variation in the environment. It is considered as a good choice if the user has clear and strong a priori hypotheses on constraints. Here, we used pCCA to test whether the presence of some species was associated with the presence or absence of planted hedgerows. We chose pCCA instead of a simple CCA to remove the effect of the distance to the nearest woodland patch, used as a covariable, as in CAP. As both analyses are sensitive to rare species, we only retained species observed in more than four samples, i.e. 46 plant species and 67 spider species.

Third, we examined the complementarity of communities (Nichols et al., 1998) in each type of site (WO type and WT type) using COMDYN with sites as replicates ($n = 11$). In addition to estimating richness within each type of site (S_{type} within WO and WT type of sites), we obtained the following estimators: complement of “extinction probability” between the two types, Φ (i.e. proportion of WT species present in WO), estimated complement of species “turnover”, γ (i.e. proportion of WO species present in WT), and “estimated local colonising” species, B (i.e. number of species not present in WT but present in WO). As for richness, we compared the mean and standard error (over 15 re-sampling events) of these estimators between WT and WO types.

2.4.3. *Functional trait composition*

We investigated whether some species attributes were more likely associated with one type of site and distance to neighbouring woody patch (environmental variables). To this end, we used a multivariate ordination analysis, RLQ, which is an extension of coinertia analysis. RLQ performs a double inertia analysis of two arrays (R, here the environmental array and Q, here the trait array), i.e. measures the adequacy between the two arrays, with a link expressed by a contingency table (L, here the species array: matrix of presence–absence of species across sites) (rlq, library ade4 in R, complete description: see Doledec et al., 1996; for an example: Ribera et al., 2001). The general significance of the relationship between environmental variables and species traits was tested using random permutations test ($n = 999$). We also used only species observed in more than four samples to limit the potentially strong effects of rare species.

3. Results

Across 25 highway verge sites, we observed a total of 85 plant species and 123 different spider species. For spiders, we captured 8300 individuals, 7862 of which were adults that could be identified to species level. Only eight of the 85 plant species (*A. campestre*, *A. pseudoplatanus*, *B. pendula*, *F. excelsior*, *P. avium*, *P. padus*, *P. spinosa*, *V. opulus*) were identical to those planted in the immediately neighbouring hedgerow in WT sites and were excluded from most analyses; note, however, that including them had no effect on the following results.

3.1. Species richness

3.1.1. Influence of planted hedgerows on plant communities

We observed a significant positive effect of the presence of hedgerows (Table 1) on plant richness at the local scale (quadrat level, $n = 125$, $S_{\mathrm{localWT}} = 10.20 \pm 0.46$ SE; $S_{\mathrm{localWO}} = 8.20 \pm 0.47$ SE, $F_{1,20} = 5.79$, $p = 0.026$). We also found a negative effect of distance from crop edge ($F_{1,20} = 5.75$, $p = 0.026$), as well as tendencies for effects of adjacent land use (higher richness with temporary fallow, $F_{1,20} = 4.15$, $p = 0.055$) and phosphorous rate (negative: $F_{1,20} = 4.00$, $p = 0.059$) on local plant richness. In contrast, no effect of hedgerows or any environmental factor was detected on plant richness at the site level ($n = 25$, $S_{\mathrm{siteWT}} = 25.45 \pm 1.66$ SE; $S_{\mathrm{siteWO}} = 23.75 \pm 3.12$ SE) and there was no difference in detection probabilities between the two types of sites. However, at the section level ($n = 11$), the estimated plant richness was 139.16 over all WT sites (S_{typeWT}, ±29.20 SE; detection probability dp = 0.47 ± 12.42 SE) vs. 87.51 over all WO sites (S_{typeWO}, ±19.43 SE; detection probability dp = 0.68 ± SE = 0.12).

3.1.2. Influence of planted hedgerows on spider communities

We found no effect of the presence of planted hedgerows on spider richness at the site level. Spider richness was significantly affected by distance to the nearest woodland patch (negative effect, $F_{1,18} = 5.94$, $p = 0.025$), site slope (positive effect, $F_{1,18} = 4.65$, $p = 0.044$) and plant richness (positive effect, $F_{1,18} = 6.81$, $p = 0.018$). There was no difference in detection probabilities between the two types of sites. At the section level, ($n = 11$), estimated spider richness was 119.3 over all WT sites (S_{typeWT}, ±19.6 SE; detection probability dp = 0.80 ± 0.10 SE) and 142.0 over all WO sites (S_{typeWO}, ±27.15 SE; detection probability dp = 0.64 ± 0.11 SE).

Table 1 – Change in species composition in the herbaceous strata of highway verges in sites without planted hedgerows (WO) vs. sites with planted hedgerows (WT)

	Φ	γ
Plants	0.86 ± 0.11	0.99 ± 0.10
Spiders	0.77 ± 0.09	0.83 ± 0.08

Φ is the proportion of WT sites species still present in WO sites and γ is the proportion of WO sites species still present in WT sites.

3.2. Community composition

3.2.1. Spatial autocorrelation

Using Mantel tests on the full dataset, we found significant spatial autocorrelation in plant communities ($p = 0.001$), but not in spider communities ($p = 0.066$). In contrast, within types of site, spatial autocorrelation was observed in plants (WT: $p = 0.006$; WO: $p = 0.001$) and spiders (WT: $p = 0.007$; WO: $p = 0.028$). However, as WO and WT sites were randomly distributed in space (Fig. 1, autocorrelation should not interfere with our main purpose, i.e. to assess the influence of hedgerows on community composition.

3.2.2. The presence of planted hedgerows influence community composition

Sites with and without planted hedgerows supported significantly different plant and spider communities, as shown by CAP (ANOVA-like permutation tests: plants: $F_{1,22} = 2.26$, $p < 0.001$, inertia explained by hedgerows = 9% and by distance to the nearest woodland patch = 4%; spiders: $F_{1,22} = 1.90$, $p = 0.006$, inertia explained = 7% and 6%, respectively). This effect of hedgerows on plant and spider community composition was confirmed by the pCCA analysis (ANOVA-like permutation tests: plants: $F_{1,22} = 2.13$, $p < 0.01$, inertia explained = 8% and 4% respectively; spiders: $F_{1,22} = 1.81$, $p < 0.005$, inertia explained = 7% and 5%, respectively). This method also identified species that seemed preferably associated with WT sites. For plants, these were *A. campestre*, *B. pendula*, *Galium mollugo*, *Centaurium erythraea*, *Trifolium repens*, and for spiders, *Panamomops sulcifrons*, *Maso sundevalli*, *Walckenaera dysderoides*, *Ceratinella brevis*, *and Drassodes cupreus*. In contrast, the spider species *Tegenaria picta*, *Ero furcata*, and *Pallidupohantes ericaeus*, were preferably associated with WO sites.

3.2.3. Nested vs. complementary communities

We found that the plant community observed in WO sites was a subset of the plant community in WT sites (Table 1). WO sites did not exhibit any private species: we found 99% of WO sites species in WT sites, so that all WO species were expected in WT sites (COMDYN estimator B = 0). The reverse was not true: WT exhibited several private species (14%), with only 86% of WT sites species also observed in WO sites. In contrast, spider communities appeared complementary. Respectively 83% (WT) and 77% (WO) of spider species were present in the other type of site, so that each type of sites exhibited private species.

3.3. Relationship between species attributes and presence of hedgerows

For both plants and spiders, there was a significant relationship between species attributes and environmental variables (Fig. 2). RLQ randtest, $p = 0.001$ and $p = 0.001$). For the two taxa, the presence/absence of planted hedgerows was associated with the first axis, which explained respectively 81% and 80% of the total inertia, whereas the distance to the nearest woodland patch tended to be associated with axis 2 (respectively, 19% and 20% of inertia). In spontaneous plant species, Raunkier type was influenced by the presence of hedgerows, with Phanerophytes and Chamaephytes (Rank: P) positively associated with WT sites (Fig. 2). In contrast, adult height was not significantly associated with site type. Finally, species with no specific means of dispersal (barochory, autochory) tended to be more present in WT sites than in WO sites, whereas animal-dispersed species (zoochory) were more common in WO than in WT sites; wind-dispersed species were not affected by the presence of hedgerows. In spiders, the mode of dispersal (ballooning vs. non-ballooning) was strongly affected by the presence of hedgerows: ballooning species were more likely found in WO sites, and non-ballooning species in WT sites. In addition, web-builders were more common in WT than in WO sites, and diurnal wandering hunters in WO sites. Finally, larger species tended to be more common in WT sites than in WO sites.

4. Discussion

We addressed the effect of the presence of planted hedgerows on neighbouring plant and spider communities of highway verges. We showed that (1) planted hedgerows were associated with higher species richness, i.e. a higher α-diversity ($\alpha_{WO} < \alpha_{WT}$), for plants, but not for spiders. This was due to the fact that plant communities in sites without hedgerows (WO) were a subset of communities in sites with hedgerows (WT), whereas spider communities in WO sites were complementary to communities in WT sites. Below, we argue that such differences are attributable to different ecological characteristics of species, which is supported by the distribution of functional traits, and derive the consequences in terms of roadside management.

4.1. *The influence of propagule source vs. local conditions on plant and spider community composition*

Following the environmental filter model (Keddy, 1992), two non-exclusive hypotheses may explain the effect of hedgerows on community composition, depending on taxonomic group. Hedgerows could either be a source of dispersers ("source" hypothesis) or could provide microclimatic conditions that affect the presence of some species due to their environmental requirements ("local conditions" hypothesis).

4.1.1. *"Source" hypothesis and the effect of hedgerows on plant communities*

The dispersal of some species from neighbouring hedgerows into a suboptimal habitat (grassland) would affect community composition in the suboptimal habitat. This phenomenon, known as the edge effect, has been observed in several taxa: contrasted adjacent habitats supporting distinct species assemblages often mix over a few meters on either side of the interface (see for example, Bedford and Usher, 1994). For example, Downie et al. (1996) studying spider community composition across a plantation–pasture ecotone, found that most of the community (70% of species) was formed by the overlap of both pasture and plantation species encroaching across the interface through dispersal or foraging processes. Interface-specialist species were present but contributed less to the interface community content than the other species categories (23% of individuals and 25% of species).

Under the "source" hypothesis, communities are, therefore, expected to exhibit higher species richness in WT vs. WO sites. This was the case for plant communities ($\alpha_{WO} < \alpha_{WT}$), at the quadrat scale, but not at the site level. Another prediction of the source hypothesis is that WO communities should be a subset of WT communities. This was true again for plants: in WT sites, we observed 'private' species which were not supported by WO sites, whereas the reverse was not true. Finally, in the case of a higher effect of dispersal vs. local conditions on species distribution within similar habitats, a distance effect can be expected independently of presence/absence of hedgerows. In plant communities, spatial autocorrelation was in fact detected regardless of the presence of hedgerows. These three observations converge to suggest that dispersal is a major driver of community composition in this group.

The distribution of plant functional traits is consistent with this role of dispersal. We found that Phanerophytes and Chamaephytes were more represented in WT sites, corresponding mostly to the occurrence of the following species: *A. campestre*, *B. pendula*, *Cytisus scoparius*, *F. excelsior and P. spinosa*. These species observed in the herbaceous strata most likely originated from seed-bearer trees that grow spontaneously within the otherwise planted hedgerow. Thus, hedgerows offer a sustainable habitat where particular species, and notably woody plants, can grow and reproduce, whereas such species cannot reach maturity within WO sites, due to frequent mowing. The fact that zoochory was more represented in WO sites is, however, not easily interpreted in this framework.

4.1.2. *"Local conditions" hypothesis and the effect of hedgerows on spider communities*

Microclimatic conditions (influenced, among others, by the neighbouring hedgerows) might be responsible for observed community composition, especially in spiders. For example, spider richness within sites was positively correlated with local plant richness; plant community composition and consequently, the architecture of habitat are well-known predictors of spider community composition (e.g. Beals, 2006). Moreover, woody hedgerows are known to provide particular microclimate conditions in their neighbourhood (Forman and Baudry, 1984) and most authors agree that those are a major factor controlling spider distribution (see, Marc et al., 1999). For example, Downie et al. (1996), examining spider community composition, proposed that close to an interface, the role of vegetation structure is low compared to other factors, including changes in microclimate. Finally, perennial non-crop habitats are known to be used by spiders as overwintering sites and refuges (Oberg et al., 2008). Grassland habitats located next to hedgerows maybe, therefore, provide better conditions than sites without hedgerows for overwintering, with direct consequences on spider community composition. Thus, neighbouring hedgerows may have an environmental filter effect, which leads to the selection of species with particular ecological requirements.

Under the 'local conditions' hypothesis, we expect that sites with and without planted hedgerows support different species. This was the case with spiders: WT and WO communities were complementary instead of nested, thus yielding similar species richness with or without hedgerows. This implies that some spider species occurring in WO sites were absent from planted sites. As the two types of sites offered the same habitat (grassland with similar percentage cover and dominance of graminoïds), the observed differences are likely be explained by specific microclimatic conditions created by the neighbouring hedgerows. Finally, the role of hedgerow-induced local conditions in spider community composition is also supported by patterns of spatial autocorrelation: the dominant impact of neighbouring hedgerows on spider community composition removed spatial autocorrelation patterns across all sites, whereas spatial autocorrelation remained significant within WT and WO sites.

Despite the significant role of local conditions, spider community composition is also likely affected by dispersal. For example, spider richness within sites was negatively correlated with the distance to the nearest woodland patch. Such effect of distance is well known from the theory of island biogeography (MacArthur and Wilson, 1967) and consistent with results from other field studies: spider species richness in wheat fields has been shown to increase with the proportion of non-crop habitats in the surrounding landscape (Schmidt and Tscharntke, 2005; see also Bianchi et al., 2006 for other groups).

Although the observed distribution of species between site types suggests a major role of local conditions, trait analyses outlined that aerial dispersal capabilities are an important factor explaining the interaction between the presence of hedgerows and spider community composition: species exhibiting ballooning behaviour were more represented in WO sites and the reverse was true for non-ballooning species. This result is comparable to that of Bonte et al. (2004), who found that the distribution of 29 spider species inhabiting fragmented grey dunes depended on their aerial dispersal potential, as well as on the interaction between patch connectivity and area. Ballooning is thought to be more common in species from unstable and ephemeral habitats (Bonte et al., 2003; Bell et al., 2005); moreover, high dispersal capabilities are generally dominant in disturbed habitats (e.g. Nyffeler and Sunderland, 2003 for spiders; Ribera et al., 2001 for beetles), allowing the maintenance of populations in ephemeral agricultural ecosystems (Weyman et al., 1995). We, therefore, suggest that the significantly higher proportion of ballooning species in WO sites may be due to the absence of a neighbouring hedgerow, i.e. of a stable habitat. In contrast, species that do not exhibit this behaviour should be more common in WT sites because hedgerow likely provides more stable conditions, both in terms of management (no mowing) and microclimatic variables. Hedgerows would thus be overwintering refuges (see also, Pywell et al., 2005), from which species colonise the surrounding habitats in spring. Sites without planted hedgerows, which receive a cut, generally in autumn, would be poor refuges that are partly recolonised from more distant sources in spring. Hence, they support a higher proportion of species exhibiting large dispersal capabilities, i.e. “ballooners”. Dispersal and local conditions, therefore, appear to drive spider community composition at different temporal scales.

Finally, note that taxonomic differences in composition of spider communities were also associated with differences in hunting strategies: web-builder were more represented in WT sites possibly due to the greatest heterogeneity of support provided by the presence of neighbouring hedgerow and the higher richness of plant whereas diurnal wandering hunter were more abundant in WO sites.

4.1.3. *Conclusions*

Planting generally increased the γ-diversity of highway verges, via an increase in both taxonomic and functional trait β-diversity between sites with and without hedgerows, but this positive effect was different in the two taxa considered. For plants, it was mostly due to an increase in taxonomic α-diversity within sites with hedgerows, whereas for spiders it was related to a direct increase of taxonomic β-diversity between the two types of sites. Note that this partition of diversity is likely to vary through time as well (e.g. due to overwintering dispersal patterns in spiders), with different temporal scales in the two taxa, but this was not the focus of the present study. From a theoretical viewpoint, the fact that the partitioning of diversity between α- and β-diversity differed between the two taxa is probably due to the different spatial and temporal scales at which the heterogeneity of interactions between organisms and their environment can be found (see, Loreau, 2000). According to Loreau (2000), “the challenge now is to understand the relationship between α- and β-diversity at multiple scales, and the processes that determine it”. In our opinion, comparing the partitioning of diversity among several taxa and functional groups at different spatial scales is, therefore, a promising way to explore the processes that control the distribution of biodiversity.

4.2. *Consequences for the management of highway verges*

Our findings have direct consequences for the management and conservation of highway verges in agricultural landscapes, where most biodiversity is aggregated in more stable non-crop habitats and field edges (Bianchi et al., 2006). Increasing the botanical and structural heterogeneity of hedgerows by planting is likely to have significant beneficial effects on native flora and fauna (Maudsley, 2000). Obviously, plantation increases species richness both directly (introduction of new species) and indirectly, by providing particular environmental conditions within woody hedgerows and thus contributing to the maintenance of some species (e.g. woody plants) in landscapes where they are poorly represented. But our results suggest that planted hedges may also affect community composition of nearby herbaceous strata, enhancing both taxonomic and functional trait γ-diversity along highway verges.

In practice, our results highlight the importance of creating a mosaic of planted sites alternating with non-planted sites in highly disturbed landscapes. Of course, “natural” dynamics could yield the same results and probably a greater biodiversity as demonstrated by Boutin et al. (2002), who compared natural vs. planted hedgerows. However, in the case of highway verges, natural hedgerows would only arise by recolonisation of the newly created habitats, which involves successional dynamics with intermediate states, and the presence of pioneer species often considered as weeds. In the context of intensive agricultural landscape, where farmers wish to limit weeds in field boundaries as much as possible, planting trees along highways verges appears as a compromise between biodiversity conservation and anthropogenic pressures. The issues of the identity of planted species (which are usually exotic species), of management of hedgerows (Croxton et al., 2004) and of their spatial arrangement along highway verges (to provide effective corridors) remain to be addressed. Given the urge to conserve biodiversity, especially in the context of climate change, road practitioners should consider plantation not only for esthetical reasons but also as a possibility to increase the role of verges as a refuge and, consequently, landscape connectivity.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.biocon.2008.04.003.

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ENCADRES

« Délaisse les grandes routes, prends les sentiers ».

PYTHAGORE (-569/-494)

ENCADRE N°1

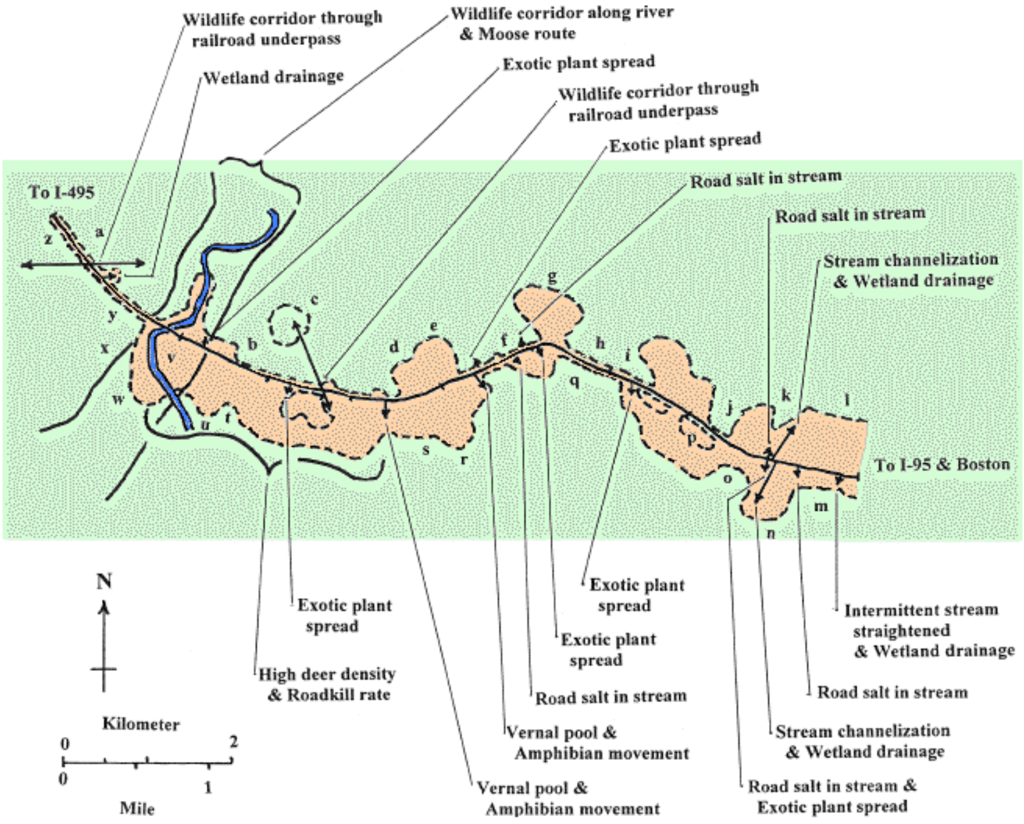
ETENDUE DE L'IMPACT ECOLOGIQUE DES ROUTES

Les effets des routes ne se limitent pas à leurs emprises (destruction d'habitat et mise en place de bordures vertes), 22% de la surface totale des U.S.A. est ainsi affectée directement par le réseau routier, *i.e.* 1,73 millions de km² (Forman 2000).

Les effets des routes sont asymétriques et dépendent des milieux traversés (*cf. Figure N°6*, zone orangée), du type de route considérée (autoroute, route principale, route secondaire, etc.) et du trafic (Forman 2000, Forman et Deblinger 2000).

Figure N°6

Zones d'impact de la route (10 km Massachusetts, U.S.A.)



(a), (b), (i), (j), (m), (p) & (z) zones construites ; (c) couloir de vent ; (d) zone basse ; (e), (g), (l), (n), (r) &(t) forêts ; (f), (h), (o) & (q) talus routiers ; (k) zones de collision ; (s) & (y) collines ; (u) prairie ; (v) rivière ; (w) zone humide et (x) golf (Forman 2000).

ENCADRE N°2

COMDYN LOGICIEL DE DYNAMIQUE DES COMMUNAUTES

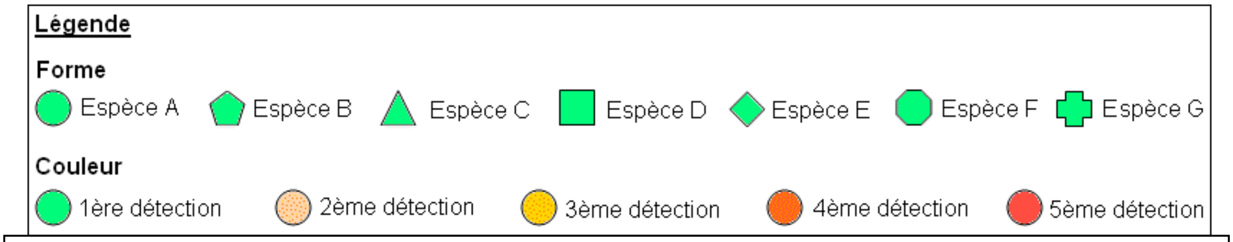
L’estimation de paramètres, tels que la richesse spécifique d’une communauté ou la similarité entre deux communautés, peut poser problème dans la mesure où ils sont issus d’observations souvent incomplètes ou réalisées par différents observateurs ayant des aptitudes différentes (connaissances, concentration, etc.). Il convient donc, non pas d’éliminer ces aléas qui sont partie intégrante de tout travaux de terrain, mais de quantifier les erreurs pour les corriger. L’utilisation de réplicats, spatiaux ou temporels, permet l’estimation de probabilités de détection permettant la correction des données brutes.

Ainsi deux communautés composées, en partie, d’espèces différentes dont les probabilités de détection sont faibles seront considérées comme relativement plus similaires entre elles que deux autres communautés présentant les mêmes particularités mais dont les espèces auront des probabilités de détection élevées.

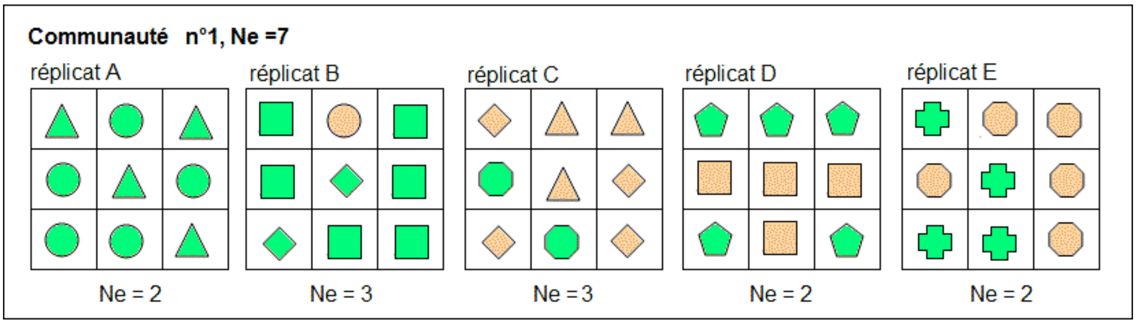
Nous avons ainsi travaillé avec des protocoles intégrant nécessairement la mise en place systématique de réplicats spatiaux (cinq) et avec un logiciel de capture-recapture ComDyn (Hines et al. 1999) permettant l’estimation d’un certains nombre de paramètres et de leurs erreurs associées : richesses estimées, probabilités de détection, similarités estimées, etc.

Exemple :

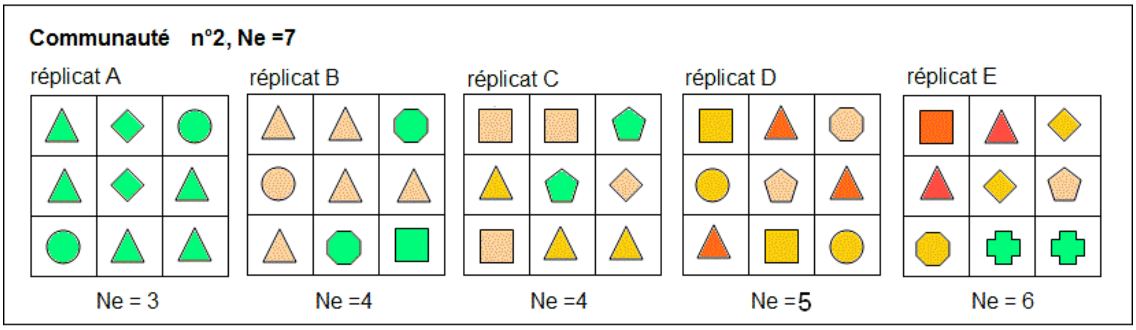
Trois communautés sont échantillonnées à partir de cinq réplicats et, bien qu’en termes de richesse brute, ces communautés peuvent apparaître comme similaires (sept espèces détectées pour chacune d’elles), elles apparaissent sont néanmoins très différentes.



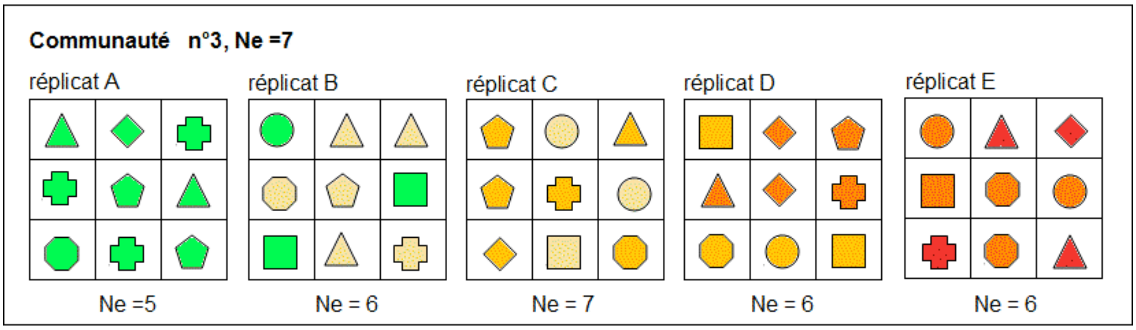
(1) Dans la première communauté, aucune espèce n'est détectée plus de deux fois seulement et de nouvelles espèces sont détectées à chacun des réplicats, la richesse des réplicats est assez faible : Ne = 2 ou 3 ;



(2) Dans la deuxième communauté, certaines espèces sont détectées quasiment dans chaque réplicat alors que d'autres ne sont détectées qu'une seule fois et progressivement au cours des réplicats, la richesse des réplicats est moyenne : Ne = 3 à 6 ;



(3) Dans la troisième communauté, toutes les espèces sont détectées au moins quatre fois et toute la richesse est échantillonnée avec uniquement les deux premiers relevés, la richesse des quadrats est importante : Ne = 5 à 7.



Il apparaît donc évident que ces différentes communautés, ultra-simplifiées et schématiques, ne sont pas équivalentes. Si l'ensemble des espèces composant la communauté n°3 semblent avoir été détecté, cela ne semble pas être le cas pour la communauté n°2 et encore moins pour la communauté n°1.

Table N°3

Résultats obtenus avec ComDyn

Communauté	NeO	NeE	NeE-se	NeE-se / NeE	Pdetec	Pdetec-se	Pdetec-se / Pdetec
n°1	7	8.6	1.24	14.4%	0.81	0.10	12.7%
n°2	7	7.8	1.37	17.6%	0.90	0.11	11.7%
n°3	7	7.0	0.00	0.0%	1.00	0.00	0.0%

Clef : ***NeO :*** *Nombre d'espèce Observé –* ***NeE :*** *Nombre d'espèce Estimé –* ***Pdetec :*** *Probibilté de Détection –* ***se :*** *Standard Error (erreur standard).*

Les résultats obtenus avec ComDyn confirment ce sentiment avec des richesses corrigées ($NeE_3<NeE_2<NeE_1$) et des probabilités de détections finalement différentes d'une communauté à l'autre (*cf.* Table N°3). Les rapports *Valeurs estimées/Erreurs standards* sont acceptables (toujours inférieurs à 20%).

ENCADRE N°3

STRATEGIE NATIONALE POUR LA BIODIVERSITE

Convention on
Biological Diversity

De part sa signature, au *Sommet de la Terre* à Rio de Janeiro (Brésil, 1992), du traité fondateur des politiques de conservation de la biodiversité, la *Convention sur la Diversité Biologique* (http://www.cbd.int/convention/convention.shtml), et de part l'engagement de l'Union européenne qui s'en est suivi au *Sommet de Göteborg* (Suède, 2001), la France a pris l'engagement d'arrêter l'érosion de la biodiversité sur son territoire avant 2010 à travers le *« Strategic Plan for the Convention on Biological Diversity »* (http://www.cbd.int/decisions/?dec=VI/26), plus connu du grand public sous l'appellation *« Countdown 2010 »*.

COUNTDOWN 2010
Halt the loss of biodiversity

Afin de d'honorer ses engagements internationaux, la France s'est dotée en 2005 d'une *Stratégie Nationale pour la Biodiversité* (*SNB*, http://www.ecologie.gouv.fr/-Strategie-nationale-pour-la-.html).

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La recherche d'indicateurs et la mise en place de suivis sont des éléments essentiels à la mise en place des plans d'action de la stratégie française.

Ministère
de l'Écologie, de l'Énergie,
du Développement durable
et de l'Aménagement
du territoire

Le financement de travaux de recherche sur le sujet, et en particulier de thèses, sont aussi lancés.

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« Si vous possédez une bibliothèque et un jardin, vous avez tout ce qu'il vous faut ».

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RESUME / SUMMARY

Les routes sont des infrastructures essentielles de nos paysages et ont de nombreux impacts sur l'environnement. Plusieurs études ont cependant montré que leurs dépendances vertes pouvaient jouer un rôle bénéfique pour la biodiversité au sein des paysages agricoles où l'intensification des pratiques depuis soixante ans a mené à sa forte raréfaction. Dans ce contexte, nous avons étudié les rôles « habitat » et « corridor » des bords de route pour les communautés de petits mammifères et de plantes. Nous avons tout d'abord retrouvé un certains nombres d'effets négatifs associés aux routes, comme une homogénéisation biotique des milieux adjacents, une pollution importante de l'environnement proche et un effet barrière pour certaines espèces animales. Nous avons surtout montré que les effets « refuge pour la biodiversité » et « corridors de continuités biologiques » étaient bien une réalité associée aux dépendances vertes des bords de route en milieu agricole. Nous avons aussi montré que ces effets dépendaient de paramètres comme les modes de gestion (fauche), les dimensions de « l'emprise verte », ou encore, la présence de haies ; et qu'ils permettaient le maintien de services écosystémiques locaux. L'intérêt écologique des bords de route en milieu agricole intensif a donc été mis en évidence tout en montrant que certaines mesures politiques pouvaient être instaurées pour une meilleure prise en compte de la biodiversité originale de ces espaces marginaux représentant 2% de l'occupation de sols. Un document pratique de conseils de gestion a été élaboré à destination des gestionnaires de routes et de leurs dépendances vertes.

MOTS-CLEF : *Agriculture intensive*, *Anthropisation des milieux*, *Biologie de la Conservation*, *Changements globaux*, *Communautés végétales*, *Corridors*, *Homogénéisation biotique*, *Nature ordinaire*, *Politiques de gestion*, *Traits fonctionnels*, *Services écosystémiques*.

Roads are essential components of our landscapes and they have important negative effects on biodiversity. However studies have shown that their verges may provide a positive effect for biodiversity in agrarian landscape where management intensification of the last sixty years leaded to the high rarefaction of biodiversity. We studied the possible functions of "habitat" and "corridor" of roadside verges for small mammals and plant communities in this agrarian context. First we rediscovered numbers of negative road effects as biotic homogenization of adjacent habitats, pollution of environment and barrier effects for some mammal species. We then showed that "refuge" and "corridor" effects associated to verges were a reality in agrarian landscapes. We also showed that number of anthropogenic parameters could impact those effects: roadside management (cutting), verge widths and hedgerows. We found that biodiversity of roadsides was able to maintain some local ecosystem services. Ecological interests of roadsides have been demonstrated with evidences that management policies could protect biodiversity of such marginal areas representing more than 2% of the landcover.

KEY-WORDS: *Anthropization*; *Biotic homogenization*; *Common nature*; *Conservation biology*; *Corridors*; *Ecosystem services*; *Functional traits*; *Global changes*; *Intensive agriculture*; *Management policies*; *Plant communities*; *Reconciliation ecology*.