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Bertrand Villeneuve

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Laboratoire d'Hydroécologie quantitative (LHQ)  
UR Milieux Aquatiques Écologie et Pollutions  
Irstea Lyon-Villeurbanne

# THÈSE

Pour l'obtention du titre de  
**DOCTEUR DE L'UNIVERSITÉ DE LORRAINE**  
Mention : Écotoxicologie, Biodiversité, Écosystèmes

## Modèles multi-stress et multi-échelles de l'état écologique : vers une analyse du risque d'altération des cours d'eau et des bassins versants.

Soutenue publiquement par  
**Bertrand VILLENEUVE**

Le 8 décembre 2016

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# Chapitre 1 - Introduction

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## 1 Contexte opérationnel et scientifique

Quelles limites ne pas franchir dans l'artificialisation des systèmes continentaux, dans les quantités et la nature des polluants qu'on y rejette, sous peine d'éroder encore plus sévèrement la biodiversité et d'obérer pour le futur différents services écosystémiques ?

Cette question générique traverse différents courants de la gestion des systèmes, qu'il s'agisse de la sphère « conservation de la nature », de la sphère gestion intégrée de l'eau, dont une déclinaison récente est la directive cadre européenne sur l'eau (2000), ou du plus récent « Millenium Ecosystem Assessment », qui inscrit la notion de services écosystémiques dans l'agenda international (Cook and Spray 2012). Les enjeux pour la société sont majeurs et les projections des changements futurs (Changements Globaux, Climatiques et « Land Use ») appellent des réponses urgentes et efficaces.

Les cours d'eau et leurs bassins versants sont des systèmes complexes et en équilibre dynamique (Amoros and Petts 1993). Si l'on connaît qualitativement assez bien la plupart des processus dont ils sont le siège (ex. transport de matière organique et de sédiments, transformation des litières, etc...), il est beaucoup plus difficile d'en avoir une représentation pertinente à la bonne maille compte tenu de l'emboîtement d'échelles des processus, des déterminants terrestres agissant sur les milieux aquatiques et surtout des différents facteurs de stress qui s'y combinent. Ce qui rend difficile l'appréciation de leur état, la prévision des états futurs et l'extraction d'une information pertinente en direction des gestionnaires (Page et al. 2012).

Par conséquent, dans l'optique de garantir à la fois une amélioration de l'état écologique des cours d'eau et les intérêts humains découlant de leur utilisation, la Directive Cadre Européenne sur l'eau (DCE) a mis en évidence le besoin d'outils d'évaluation et de diagnostic de l'état écologique des cours d'eau pour optimiser les mesures de gestion et de restauration en fonction des causes avérées d'altération. La crise économique renforce

l'exigence de l'identification des facteurs vraiment limitants pour que les investissements soient efficaces et s'accompagnent d'effets visibles.

Atteindre cet objectif nécessite bien sûr de maîtriser les connaissances sur le fonctionnement des hydrosystèmes et en particulier de savoir interpréter les signaux biologiques placés au cœur du processus d'évaluation. Ce qui implique que les liens entre ces signaux biologiques et les pressions qui les altèrent, leurs évolutions temporelles dans leur contexte géographique soient correctement décryptés.

Progresser dans cette direction suppose de lever un certain nombre d'écueils mis en avant par Smart et al. (2012).

Le premier écueil possible concerne l'importance de l'échelle des processus. Selon l'échelle considérée, on peut être confronté à des résultats divergents, alors qu'en réalité il ne s'agit que de la conséquence d'une échelle d'observation différente.

Par exemple, Carey and Mather (2008) ont souligné l'importance de l'échelle de prise en compte des processus en montrant une évolution différenciée d'un indice de disponibilité des nutriments pour les plantes du milieu par catégories d'occupation du sol (terres arables, forêts et espaces semi-naturels) sur une période de 30 ans alors que ce même indice n'évoluait pas significativement lorsqu'on l'analysait sur l'ensemble des parcelles sans tenir compte du type d'occupation du sol.

De même, Roth et al. (1996) ont mis en évidence des liens plus forts entre métriques du paysage et l'indice biologique poisson (IBI) qu'avec les métriques de la ripisylve, pour des cours d'eau du sud est Michigan (USA). D'autres auteurs, dont certains communs, Lammert and Allan (1999) concluent l'inverse. Ces derniers attribuent ces apparentes contradictions au plan d'expérience, avec dans le premier cas l'inclusion de nombreux bassins différents et dans le deuxième cas, un espace plus restreint avec de nombreux sites dans un nombre plus faible de bassins. Dans une autre étude de 38 cours d'eau en région agricole du Wisconsin (USA), Stewart et al. (2001) montrent qu'à l'échelle du bassin, le pourcentage de surface forestière influence la diversité piscicole, le pourcentage de poissons non tolérants et certaines métriques relatives aux macroinvertébrés. La fragmentation de la végétation riparienne affecte quant à elle des métriques relatives aux macroinvertébrés, et à l'échelle

stationnelle, le colmatage par sédiments fins influence également des métriques sur les invertébrés. Mais tous ces facteurs environnementaux sont fortement corrélés voire autocorrélés, rendant extrêmement délicates les conclusions.

Levin (1992), déclarait que compte tenu de toutes les variabilités temporelles, spatiales et des différentes organisations fonctionnelles au sein des écosystèmes, il n'existait probablement « pas une échelle unique pertinente » pour leur analyse.

L'article d'Allan (2004) permet déjà de bien baliser le travail restant à faire : les facteurs relatifs à la nature, à l'occupation du sol et à son organisation spatiale, réunis sous le terme anglo-saxon de *landscape*, sémantiquement mal rendu en français par « paysage », influencent les écosystèmes lotiques à travers un large éventail d'échelles spatiales. Il est difficile de séparer le rôle des échelles spatiales proches du cours d'eau (ex. ripisylves) de celui des échelles plus distantes (ex. imperméabilisation des sols d'un bassin versant),

Mais Smart et al. (2012) rappellent aussi l'importance des variables explicatives manquantes. Une variable pertinente peut-être omise des essais de mise en relation entre facteurs de stress et réponses biologiques (Platt 1964) permettant d'obtenir des résultats avec une significativité statistique forte, mais basés sur des liens avec des variables non causales ou trop éloignées des leviers d'actions possibles.

Et finalement, ils soulignent le manque de contrôle sur les variables de forçage, qui peut entraîner des biais dans l'analyse. Un gradient de pression peut en effet être plus ou moins intense et plus ou moins étendu entraînant des difficultés à analyser son effet sur le système étudié.

Friberg et al. (2009) ajoutent d'autres difficultés inhérentes à la donnée biologique elle-même, soit à cause d'une stratégie d'échantillonnage inappropriée par rapport à la question posée, soit à cause de métriques improprement sélectionnées.

De plus, la donnée de pression peut s'avérer décrite à une échelle inappropriée ou mal caractérisée (ex. renseignée à dire d'experts introduisant de la variabilité ou mesurée qualitativement), et ainsi ne pas rendre compte d'une altération de processus mais d'une résultante de plusieurs processus.

Enfin, il arrive parfois que l'on ne détecte pas d'effet, non parce qu'il n'y en a pas, mais possiblement parce que la pression observée est très peu variable et ne présente pas de gradient suffisamment net (Wenger et al. 2009). Ceci empêche par exemple de mettre en évidence un effet de l'agriculture dans une zone géographique où l'agriculture est très présente. Le manque de contraste empêche en effet les méthodes statistiques couramment utilisées de détecter une corrélation entre l'agriculture et la métrique observée (Wasson et al. 2010).

Cette thèse intervient à un moment favorable pour tenter d'apporter des solutions à certains de ces écueils tout en intégrant les avancées de la première phase de mise en œuvre de la DCE et ses limites.

En effet, pour répondre aux objectifs de la DCE, les recherches en Europe se sont d'abord focalisées sur le développement d'une nouvelle génération de bioindicateurs (macro-invertébrés, macrophytes, algues, poissons) prenant en compte un ensemble de pressions renseignables s'exerçant sur les cours d'eau. Ces bioindicateurs aident à faire une évaluation de l'état du cours d'eau au niveau du point échantillonné (Bonada et al. 2006; Furse et al. 2006; Hering et al. 2004; Marzin et al. 2012; Mondy et al. 2012; Pont et al. 2006). Mais ils sont encore des outils imparfaits. D'une part, ils ne permettent pas encore un diagnostic des causes de l'altération de l'état écologique et d'autre part, leur représentativité spatiale reste limitée et leur caractère ponctuel n'autorise pas une extension de l'évaluation à une plus large échelle d'observation. Un échantillonnage densifié spatialement est bien évidemment impossible sur l'ensemble des masses d'eau pour des raisons tant techniques qu'économiques. Quelques chiffres peuvent illustrer ce propos : le réseau national de contrôle et de surveillance (RCS) est constitué de 1500 stations pour 230 000 km de cours d'eau. Ceci n'est évidemment pas suffisant pour évaluer avec robustesse et précision l'état écologique des cours d'eau français. De plus, ces points sont suivis au mieux une fois par an pour les macro-invertébrés, un an sur deux pour les poissons et jusqu'à une fois tous les six ans pour les substances toxiques. Ceci plaide pour une évaluation qui prendrait mieux en compte le système pressions/réponses avec une description pertinente des pressions et de leur agencement spatial.



On retrouve dans la littérature un grand nombre d'études reliant les pressions et le fonctionnement biologique (Donohue et al. 2006; Kristensen et al. 2012; Marzin et al. 2013; Wasson et al. 2010). Ces modèles développés permettent d'extrapoler l'évaluation de l'état réalisée au niveau des stations à l'ensemble du réseau hydrographique.

Mais ces modèles, aussi sophistiqués soient-ils sur le plan statistique, sont incapables de s'adapter à des situations qu'ils ne connaissent pas et donc de restituer une prédiction (un état) correcte si on cherche à les appliquer à un type de cours d'eau ou à un corpus de pressions qui sont absents du jeu de données d'apprentissage (du réseau de suivi).

Il est donc essentiel de travailler sur des jeux de données les plus riches possibles couvrant la plus grande diversité de situations de pressions.

Enfin, par rapport à la grande variété d'indicateurs existants pour l'évaluation de l'état écologique, les capacités à prédire les réponses des écosystèmes au stress, à la restauration ou plus généralement à des mesures de gestion alternatives, sont sous-développées. En particulier, une plus grande connaissance sur les effets des expositions à des facteurs de stress multiples, sur les interactions entre stressseurs (synergiques, additifs ou antagonistes), est nécessaire mais reste encore peu visible dans la littérature scientifique (Hering et al. 2015).

L'exploration à mener dans la thèse s'apparente à des pratiques d'épidémiologie, le but étant de rechercher les causes souvent multiples de dégradation de l'état écologique, d'en évaluer l'étendue spatiale, la gravité, pour être en mesure de recommander des actions efficaces de restauration.

Elle se nourrit d'un questionnement multiple :

Comment les organismes aquatiques répondent-ils à des stress de différentes natures ? Ces réponses sont-elles dépendantes des contextes géographiques ou des compositions des communautés ? Les réactions biologiques négatives sont-elles temporaires ou durables, réversibles ou non ? Quels sont les niveaux de stress qui occasionnent des ruptures des équilibres écologiques ? D'où proviennent les différentes formes de stress, quelles sont les pressions qui en sont à l'origine et comment s'opèrent les combinaisons de ces stress au sein des bassins versants, des corridors rivulaires et des cours d'eau ?

Si ces questions posées individuellement ne sont pas nouvelles, par contre leur prise en compte simultanée soulève de nombreux défis méthodologiques.

## **2 La Directive Cadre Européenne sur l'Eau (DCE) et sa mise en œuvre**

### **2.1 Exigences de la DCE**

La directive du 23 octobre 2000 adoptée par le Conseil et par le Parlement européen définit un cadre pour la gestion et la protection des eaux par grand bassin hydrographique au plan européen. Cette directive joue un rôle stratégique et fondateur en matière de politique de l'eau. Elle fixe en effet des objectifs ambitieux pour la préservation et la restauration de l'état des eaux superficielles (eaux douces et eaux côtières) et pour les eaux souterraines.

Pour les eaux de surface, la Directive Cadre Européenne sur l'Eau (DCE) fixait pour objectif d'atteindre à l'horizon 2015 le « bon état » pour toutes les masses d'eau, de préserver celles qui sont en « très bon état », et d'atteindre le « bon potentiel » dans les masses d'eaux fortement artificialisées. Il s'y ajoutait également un objectif « zéro toxiques », non daté mais clairement affiché. Mais le point essentiel est ici une obligation de résultat dans le délai imparti (en réalité, dérogation jusqu'en 2027), et non plus seulement de moyens, la directive fixant seulement un catalogue des mesures possibles qui restent sous la responsabilité des états membres.

Le bon état est défini d'après la situation la plus déclassante entre un état chimique se rapportant à des normes de concentration de certaines substances particulièrement dangereuses (toxiques), et un état écologique qui repose sur une évaluation des « éléments de qualité » physico-chimiques (paramètres généraux et micro-polluants non inclus dans l'état chimique), et biologiques (peuplements végétaux, invertébrés et poissons). L'objectif de « bon état écologique » est défini comme un écart « léger » à une situation de référence, correspondant à des milieux non ou très faiblement impactés par l'homme.

Selon la définition de la DCE, l'état écologique se réfère « à la structure et au fonctionnement des écosystèmes aquatiques ». Mais son évaluation repose principalement

sur la bioindication : les peuplements aquatiques sont les juges de paix. En effet, les normes à appliquer pour les paramètres physico-chimiques généraux doivent être reliées à l'altération des peuplements, et pour les polluants toxiques, elles sont définies sur la base de tests écotoxicologiques. L'évaluation des altérations physiques (ou hydro-morphologiques) n'est explicitement requise que pour identifier les situations de référence et le « très bon état », mais elle est évidemment essentielle en tant qu'élément de diagnostic des causes d'altération.

## 2.2 Une cartographie de l'état des eaux

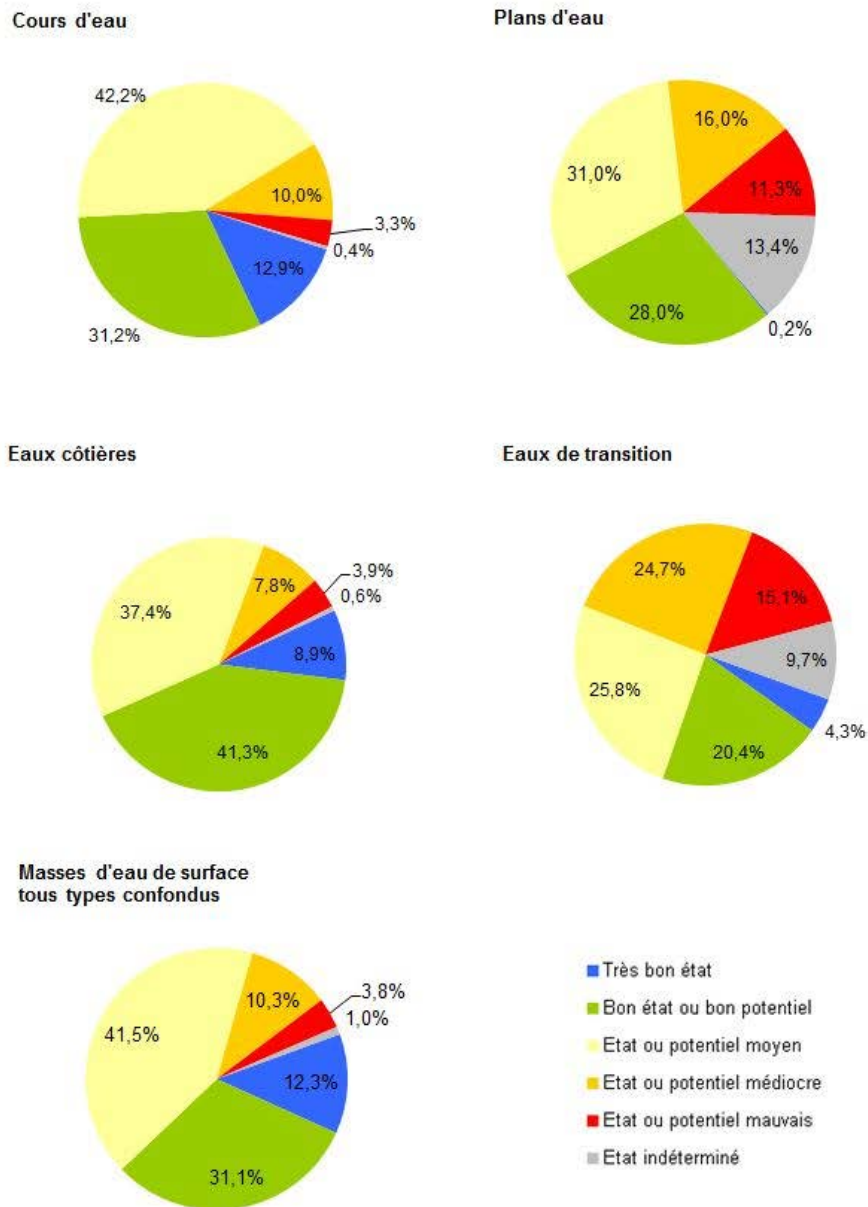
Pour la France, répondre à la DCE consiste avant tout à produire une cartographie de l'état écologique actuel et une cartographie de l'état chimique actuel de chacune des 11500 masses d'eau pour les eaux douces de surface (cours d'eau et plans d'eau) (dont 6000 hors très petites masses d'eau et masses d'eau fortement modifiées), selon les modalités suivantes :

- état écologique « agrégé » à partir des différents éléments de qualité, avec une représentation des cinq classes d'état écologique (Figure 1);
- état chimique « agrégé » à partir des 41 substances prioritaires et dangereuses prioritaires, avec une représentation des deux classes d'état chimique ;
- attribution de niveaux de confiance à l'état écologique et à l'état chimique évalués par masse d'eau.

Ces règles ont été reprises dans la loi française avec l'arrêté du 25 janvier 2010, arrêté d'application de l'article R.212-18 du code de l'environnement relatif aux méthodes et critères définissant l'état/le potentiel écologique et chimique des eaux douces de surface. Elles font partie des éléments à considérer pour déterminer et suivre les actions des programmes de mesure DCE et des autres dispositifs de planification dans le domaine de l'eau.

Ces cartographies de l'état écologique et de l'état chimique actuels des eaux douces de surface (cours d'eau et plans d'eau), sont incluses dans les schémas directeurs d'aménagement et de gestion des eaux (SDAGE) qui sont adoptés par les Comités de bassin depuis 2009 et sont rapportées au niveau européen en application de la DCE.

## État écologique des masses d'eau de surface en France métropolitaine et dans les DOM (état des lieux 2013)



Sources : agences et offices de l'Eau - Onema - ministère en charge de l'Environnement, état des lieux 2013.

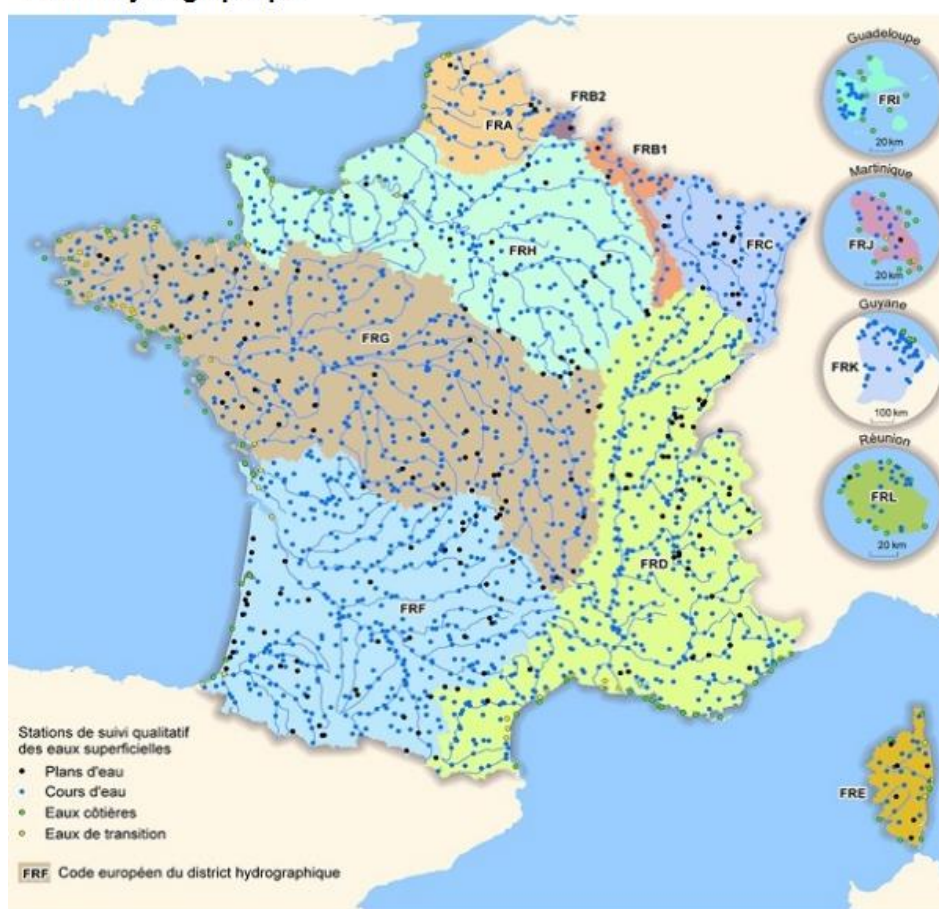
Figure 1. Etat écologique des masses d'eau de surface (état des lieux 2013)

## 2.3 Mise en place des réseaux de surveillance DCE

La directive-cadre sur l'eau (DCE) requiert la mise en œuvre de programmes de surveillance pour suivre au sein de chaque district hydrographique :

- l'état écologique et chimique des eaux superficielles, dont les eaux littorales et côtières ;
- l'état chimique et quantitatif des eaux souterraines.

### Répartition des stations de suivi qualitatif des eaux superficielles (RCS) par district hydrographique



**Note** : le rapportage ne comprenait pas le réseau de Mayotte.

**Sources** : rapportage à la Commission européenne d'octobre 2010 préparé par ministère chargé de l'Environnement – Onema – agences et offices de l'Eau – DIREN/DEAL (Dom)

Figure 2. Carte des stations du réseau de contrôle et de surveillance préparée par le ministère chargé de l'Environnement, l'Onema, Les agences et offices de l'Eau et les DIREN/DEAL pour le rapportage à la commission européenne d'octobre 2010.

Les différents réseaux de mesures, mis en place dans le cadre du programme de surveillance DCE, sont les suivants :

- le réseau de contrôle de surveillance (RCS) (Figure 2), mis en œuvre depuis janvier 2007. Il permet d'évaluer l'état général des eaux et son évolution au niveau d'un bassin avec 2007 comme année de référence. Le réseau est constitué de stations de mesures représentatives du fonctionnement global de la masse d'eau et a pour vocation d'être pérenne. On compte selon le dernier rapportage à la Commission européenne (Mayotte non compris) : pour le suivi de la qualité, 2 043 stations pour les eaux superficielles et 1 775 stations pour les eaux souterraines et pour le suivi du niveau des eaux souterraines (niveaux piézométriques), 1 674 stations.
- le réseau de contrôle opérationnel (RCO), dont le rôle est : (i) d'assurer le suivi de toutes les masses d'eau identifiées comme risquant de ne pas atteindre le bon état en 2015, (ii) d'assurer le suivi des améliorations des eaux, suite aux actions mises en place dans le cadre des programmes de mesures, (iii) et le cas échéant de préciser les raisons de la dégradation des eaux. Seuls les paramètres à l'origine du risque de non atteinte du bon état de la masse d'eau en 2015 sont suivis dans ce réseau, mis en place entre 2007 et 2009.

On compte selon les SDAGE publiés fin 2009 : 4 618 stations qualitatives RCO sur les eaux superficielles et 1 446 sur les eaux souterraines. Au contraire du RCS, ce réseau évolue au fil des cycles DCE en fonction de l'état des masses d'eau.

- le réseau de contrôle d'enquête : réseau de suivi de pollutions accidentelles, ou de dégradations d'origine mal connue, dans le but d'en déterminer l'ampleur et l'incidence (seulement pour les eaux de surface).
- les réseaux de contrôles additionnels : réseaux de contrôles portant sur les points de captage d'eau potable et sur les zones d'habitat et de protection d'espèces.

Le programme de surveillance de la DCE a été construit sur la base de réseaux de suivi qui existaient auparavant (réseau national de bassin (RNB) pour les eaux superficielles, réseau national de suivi des eaux souterraines (RNES) pour les eaux souterraines).

## 2.4 Principes généraux et rôles des différents éléments de qualité dans la classification de l'état écologique

Selon les termes de la DCE, lorsque les valeurs-seuils des différents éléments sont établies conformément aux prescriptions de la DCE, la règle d'agrégation qui s'impose est celle du principe de l'élément déclassant, au niveau de l'élément de qualité.

Le rôle des différents éléments de qualité (biologiques, physico-chimiques et hydromorphologiques) dans la classification de l'état écologique est différent pour la classification en état écologique très bon, bon, moyen, médiocre et mauvais (Figure 3).

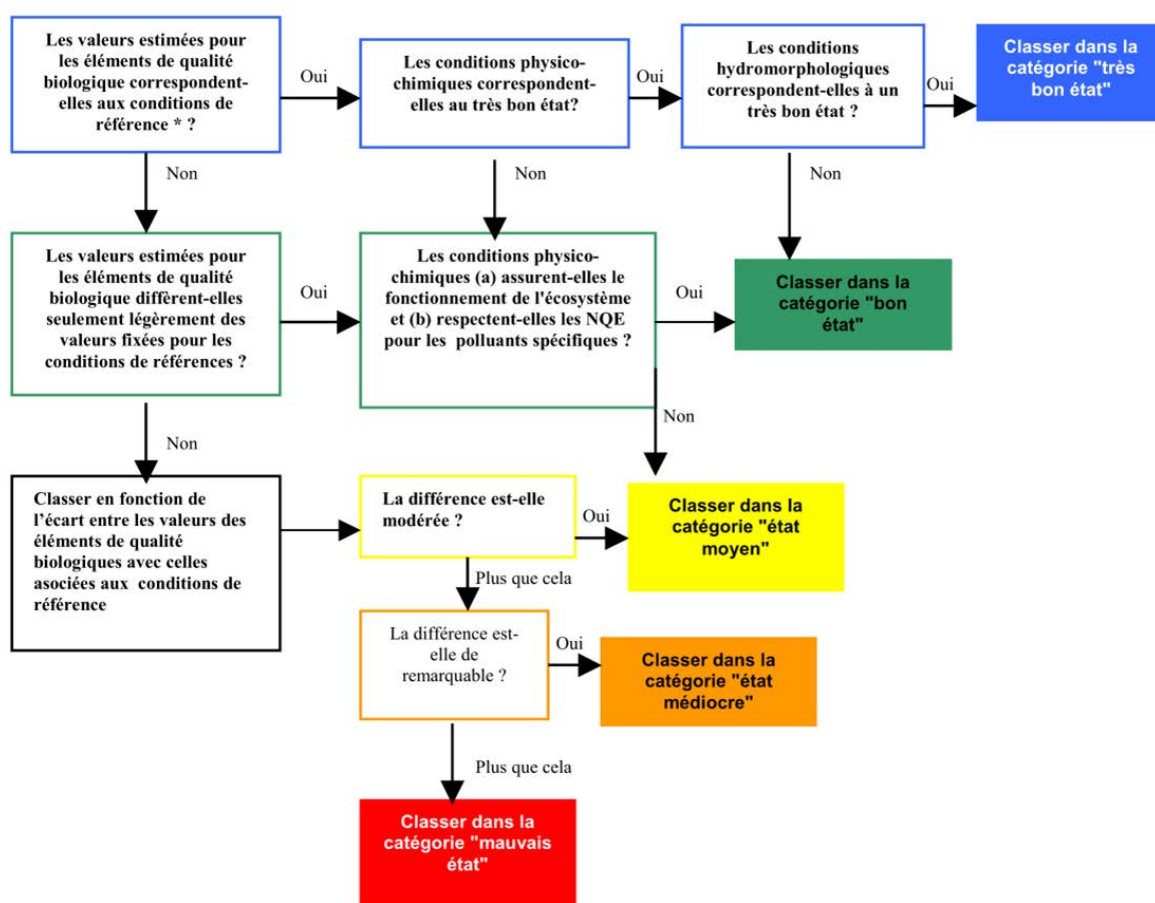


Figure 3. Rôles respectifs des éléments de qualité biologiques, physico-chimiques et hydromorphologiques dans la classification de l'état écologique, conformément aux termes de la DCE (définitions normatives de l'annexe V.1.2).

### **3 Des méthodes d'évaluation qui rendent compte des effets des pressions : développement d'un nouvel indice basé sur les macroinvertébrés benthiques : l'I<sub>2</sub>M<sub>2</sub> (Mondy et al. 2012)**

*L'article scientifique complet correspondant à cette partie est présenté en annexe 1 :*

*Mondy, C.P., Villeneuve, B., Archambault, V., Usseglio-Polatera, P., 2012. A new macroinvertebrate-based multimetric index (I2M2) to evaluate ecological quality of French wadeable streams fulfilling the WFD demands: A taxonomical and trait approach. Ecological Indicators 18, 452-467.*

L'indice Biologique Global Normalisé (IBGN) est la méthode normalisée nationale actuelle (NF T 90350 1992, 2004) visant à évaluer l'état écologique des cours d'eau, sur la base de l'élément de qualité biologique « faune benthique invertébrée », échantillonnable à pied (ou à l'aide d'embarcations légères) à l'aide d'un engin de type filet « Surber » ou filet « Haveneau ».

Un premier diagnostic de l'IBGN, au regard de la Directive Cadre sur l'Eau et par rapport aux méthodes les plus récentes développées au niveau européen, a montré ses faiblesses notamment dans :

- sa collecte de données non quantitatives, de niveau taxonomique limité à la famille, à l'aide d'un protocole d'échantillonnage spécifique n'assurant pas une représentativité optimale des sites ;
- sa non prise en compte de l'ensemble des paramètres biologiques inclus dans la « définition normative » des critères de bon état écologique au sens de la DCE ;
- sa faible capacité à diagnostiquer précisément les causes de la perturbation ;
- sa stratégie de bioévaluation de l'état écologique d'un cours d'eau non effectuée par comparaison avec une situation de référence objective et pertinente pour la masse d'eau considérée (même si une adaptation secondaire avait visé à modifier les limites de classes par type de cours d'eau pour tenir compte de l'hétérogénéité des masses



d'eau dans leur capacité à accueillir une faune riche et diversifiée, même en situation de référence) ;

- le peu de cohérence entre les objectifs justifiant les principes théoriques de la stratégie d'échantillonnage IBGN (i.e. l'évaluation des capacités biogènes d'un point de prélèvement) avec les objectifs qui sous-tendent la stratégie d'échantillonnage des invertébrés benthiques des méthodes de bioévaluation élaborées par certains de nos partenaires européens (e.g. l'obtention d'une « image moyenne » du peuplement benthique sur un point de prélèvement, protocole AQEM ; (Hering et al. 2003)).

Une étude a donc été lancée sur ce thème par le LIEC (Université de Lorraine - Metz) en collaboration avec Irstea avec pour objectifs principaux :

- (i) de permettre la mise en conformité de la méthode nationale de bioévaluation des cours d'eau, via la faune benthique invertébrée, aux exigences de la DCE,
- (ii) de transformer l'IBGN actuel, à considérer plutôt comme un « indicateur d'état », en un véritable « outil de diagnostic » (c'est-à-dire d'évaluation des causes d'altération de l'état écologique) et
- (iii) de faciliter l'intercalibration des méthodes européennes d'évaluation de l'état écologique (et les comparaisons internationales des résultats de bioévaluation) en recherchant une meilleure cohérence du protocole national normalisé avec les stratégies d'évaluation de nos partenaires européens.

C'est dans ce cadre que j'ai travaillé en collaboration avec Philippe Usseglio-Polatera et Cédric Mondy de l'université de Metz et Virginie Archambault de l'Irstea Antony (UR HBAN) au développement du nouvel indice macro-invertébrés national (I2M2). J'ai pour cela construit des modèles diagnostiques PLS reliant les pressions hydromorphologiques issues de SYRAH-CE (descriptif de la méthode p 51) aux métriques candidates à la construction du nouvel indice. Ceci a permis à Cédric Mondy de faire une sélection pertinente des métriques répondant aux pressions hydromorphologiques et ainsi, de construire un outil de bioindication répondant à ce type de pression. Ces travaux ont été valorisés par une publication dans *Ecological Indicators* (Mondy et al. 2012).

Nous avons ainsi développé un nouvel indice multimétrique (I2M2) pour l'évaluation écologique des cours d'eau français, en utilisant les assemblages de macroinvertébrés benthiques. Cet indice permet d'identifier les stations soumises à au moins un type de pression parmi les 17 relatifs à la qualité de l'eau ou la qualité physique de l'habitat que nous avons étudiés. En utilisant une base de données nationale, nous avons défini une procédure itérative pour sélectionner, parmi un grand nombre de métriques taxonomiques ou basées sur les traits bio-écologiques sélectionnées dans les assemblages d'invertébrés, la combinaison de métriques qui présente le meilleur compromis entre : (i) la plus forte capacité de discrimination possible, (ii) une faible spécificité et (iii) une grande stabilité dans les conditions de moindre impact anthropique (LIRR). L'I2M2 ainsi formé est une combinaison de cinq métriques : (i) l'indice de diversité de Shannon, (ii) l'indice biotique britannique ASPT (Average Score per Taxon), l'utilisation relative, au sein des communautés benthiques (iii) du polyvoltinisme et (iv) de l'ovoviviparité et enfin (v) la richesse taxonomique. Ces métriques ne se calculent pas à la même échelle d'observation. Si les trois dernières se calculent à l'échelle stationnelle, la première se calcule sur une fraction de l'échantillon prélevée dans les habitats présentant la meilleure habitabilité et la deuxième se calcule sur une fraction de l'échantillon prélevée dans les habitants dominants. L'I2M2 a été testé avec un jeu de données indépendant du jeu de données d'apprentissage. Il a présenté de bonnes et robustes relations pressions-impacts pour l'ensemble des types de pression étudiés, identifiant correctement en moyenne plus de 82 % des stations subissant des altérations significatives (i.e. au mieux «état moyen») de la qualité de l'eau ou de l'intégrité physique de l'habitat. L'I2M2 améliore de manière significative la détection des situations perturbées par rapport à la méthode de bioévaluation normalisée française actuelle, l'IBGN. Cette amélioration est d'au moins 17 % pour la pression «composés azotés» et va jusqu'à 35 % pour les « micropolluants organiques » et le « risque de colmatage ». L'I2M2 a été proposé pour une future utilisation dans la bio-évaluation des sites des réseaux nationaux dans un contexte de mise en place de la DCE.

## 4 Besoin d'outils prédictifs pour évaluer les masses d'eau non suivies (Reyjol et al. 2014)

L'article scientifique complet correspondant à cette partie est présenté en annexe 2 :

Reyjol, Y., C. Argillier, W. Bonne, A. Borja, A. D. Buijse, A. C. Cardoso, M. Daufresne, M. Kernan, M. T. Ferreira, S. Poikane, P. Narcís, A.-L. Solheim, S. Stroffek, P. Usseglio-Polatera, B. Villeneuve & W. Van de Bund, 2014. **Assessing the ecological status in the context of the European Water Framework Directive: Where do we go now?** *Science of the Total Environment* 497: 332-344.

### 4.1 La France est-elle en mesure d'évaluer l'état écologique de l'ensemble de ses cours d'eau ?

La directive cadre européenne sur l'eau demande que les états membres mettent en place des programmes de surveillance de l'état des eaux de surface dans le but d'évaluer de manière cohérente et transparente l'état des eaux de chaque district de bassin (article 8). Les réseaux de surveillance émanant de ces programmes doivent prendre en compte suffisamment de masses d'eau pour permettre une évaluation fiable de l'ensemble des masses d'eau de surface de chaque district. Enfin, les états membres doivent produire pour chacun des districts de bassin une carte illustrant la classification de l'état écologique de chaque masse d'eau de surface.

Malgré cela, la directive cadre n'impose pas un suivi systématique de chaque masse d'eau dans le cadre des réseaux de surveillance. Par exemple, le réseau de contrôle et de surveillance de la France couvre approximativement 15% des masses d'eau du territoire avec 1500 sites. Cela laisse par conséquent 8500 masses d'eau pour lesquelles l'évaluation de l'état écologique devra se faire sans données de suivi disponibles. Et cette situation n'est pas unique en Europe car les limitations techniques ainsi que les raisons économiques font que les états membres ont choisi de ne pas suivre l'ensemble de leurs masses d'eau (Kristensen and Christiansen 2012). D'autres pays comme le Royaume-Uni ont un réseau de suivi plus dense mais sur lequel l'échantillonnage n'est fait qu'une fois par station dans le but de couvrir un plus grand nombre de masses d'eau. Cette stratégie peut d'ailleurs donner

l'impression que l'effort d'échantillonnage est plus important que celui de la France ce qui n'est en fait pas le cas. Par contre, cette stratégie ne permet de suivre l'évolution temporelle de l'état d'une même masse d'eau.

Pour faire face à cette situation, la directive cadre permet de faire appel à des méthodes comme l'avis d'expert ou le regroupement de masses d'eau. Mais cela risque d'augmenter la subjectivité et l'hétérogénéité des évaluations écologiques à travers l'Europe. C'est pour cette raison que nous pensons qu'il est nécessaire d'utiliser une méthode fiable et reproductible pour prédire l'état écologique des masses d'eau non suivies de manière objective et homogène pour l'ensemble des masses d'eau d'un Etat membre.

Une des nouveautés de la directive était de placer le vivant au cœur de la gestion des milieux aquatiques : l'objectif central assigné par cette directive est le « bon état écologique » qui se réfère « à la structure et au fonctionnement des écosystèmes aquatiques ». L'évaluation de cet état repose principalement sur les besoins de la vie aquatique qui sont en définitive les « juges de paix » (Roche et al. 2005; Wasson 2001) de l'évaluation.

Mais la liste des éléments de qualité utilisés pour évaluer l'état écologique ou le potentiel écologique des masses d'eau (Figure 4), reflète le niveau de développement des systèmes d'évaluation nationaux décrits ci-dessous, et montre également que la proportion des masses d'eau surveillées est inférieure à la proportion classée pour la plupart des éléments de qualité.

Pour les rivières, les éléments de qualité les plus couramment utilisés sont les macroinvertébrés et les poissons, ainsi que les éléments de qualité « hydromorphologie », « physico-chimie » et « polluants spécifiques non prioritaires ». La proportion de masses d'eau classées est beaucoup plus grande que la proportion des masses d'eau réellement surveillées illustrant ainsi la pratique du regroupement de masses d'eau et/ou l'utilisation du jugement d'expert pour classer les masses d'eau non surveillées.

Ces résultats illustrent le manque de données de suivi nécessaires pour classer directement tous les plans d'eau ou rivières européennes. Il semble être très important, soit de renforcer la surveillance directe des masses d'eau en augmentant le nombre de points de contrôle (mais le coût sera très important) et/ou de développer des méthodes

complémentaires efficaces pour la classification des masses d'eau non surveillées sur la base de modèles pressions/impacts prédictifs robustes et homogènes.

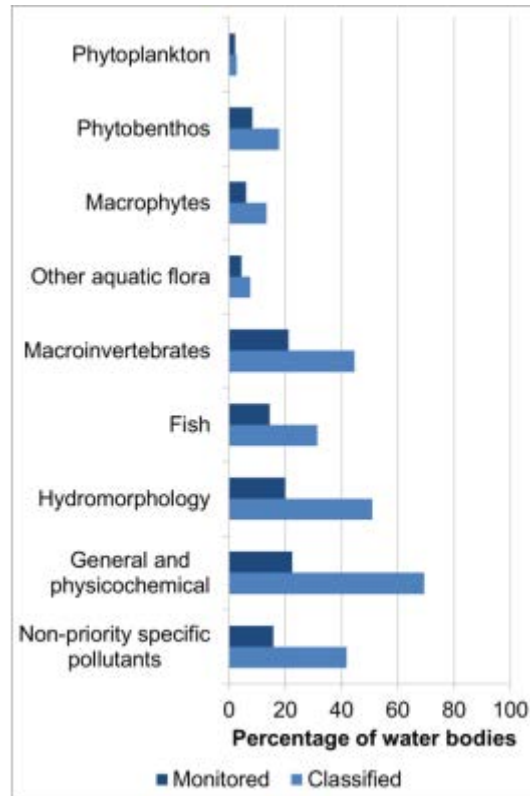


Figure 4. Présentation européenne des différents éléments de qualité rapportés par les états membres pour le contrôle et la classification des masses d'eau de rivière (Kristensen and Christiansen 2012).

Le renforcement de la surveillance directe se heurte à des limites techniques et économiques. En outre, il ne donne pas une compréhension suffisante des causes d'altération des peuplements aquatiques. Il ne permet pas aux gestionnaires de l'eau d'identifier les mesures de restauration, d'atténuation ou de protection de la rivière qui pourraient être mises à profit dans les programmes de mesures de la DCE.

Le développement de modèles pressions/impacts prédictifs devrait être très utile pour pallier le manque de données de surveillance locales. Même si ces modèles ne seront probablement pas capables de résoudre tous les problèmes liés au manque de données, ils peuvent aider les gestionnaires de l'eau à optimiser leurs programmes de surveillance. Ils

pourraient même aider à cibler les zones où il serait utile d'approfondir la connaissance de terrain afin d'améliorer d'une part les modèles et d'autre part l'évaluation de l'état.

Outre les fins d'extrapolation, le développement de tels modèles pourrait être particulièrement pertinent pour mettre en œuvre la procédure d'évaluation des risques de non atteinte des objectifs environnementaux des masses d'eau (article 4 et annexe II 1.5).

## 4.2 Modèles Pression impact en Europe? Un manque de connaissance

Très peu de travaux de recherche se sont intéressés au développement d'outils prédictifs permettant de faire une évaluation des masses d'eau non suivies. Donohue et al. (2006) ont construit un modèle permettant de prédire l'état écologique basé sur les peuplements macro-invertébrés benthiques à partir de données d'occupation du sol du bassin versant et d'indicateurs de la qualité chimique de l'eau en Irlande. Kristensen et al. (2012), au Danemark, ont développé un modèle capable de prédire la présence d'assemblages de poissons en fonction de l'occupation du sol dans le bassin versant et de données chimiques et morphologiques mesurées au niveau stationnel. On retrouve aussi ce type de travaux en dehors de l'union européenne. Aux USA en particulier ou quelques auteurs ont publié des travaux concernant le développement de modèles prédictifs de l'état écologique (toujours basé sur les peuplements de macroinvertébrés benthiques) à partir de données d'occupation du sol dans le bassin versant (Brown et al. 2012; Carlisle et al. 2009; Maloney et al. 2009; Waite et al. 2010).

Ces premiers outils prédictifs montrent qu'il est possible de prédire l'état écologique à partir de variables de pression. Mais ils tiennent peu compte des échelles multiples qui structurent à la fois les pressions et le fonctionnement longitudinal des cours d'eau. En effet, un cours d'eau est organisé de manière hiérarchique selon plusieurs échelles imbriquées : bassin versant, tronçon, chenal et microhabitat (Allan 2004; Allan and Johnson 1997; Frissell et al. 1986; Poff et al. 1997; Thorp 2014). De plus, l'altération de l'état écologique est souvent la résultante de plusieurs pressions qui peuvent agir simultanément : rejets ponctuels, pollutions diffuses, altération de l'écoulement solide et liquide, structures artificielles et rupture de connectivité (Borchardt and Richter 2003). D'après le concept DPSIR (Forces motrices, pressions, état, impact et réponse) (EEA 2003; Kristensen 2004), les

activités humaines (agriculture, urbanisation) génèrent des pressions combinées (rejets chimiques, altérations physiques) et constituent les forces motrices qui altèrent les composantes abiotiques de l'écosystème (physico-chimie, hydromorphologie). Ces altérations affectent ensuite les communautés biologiques et par conséquent l'état écologique (Figure 5).

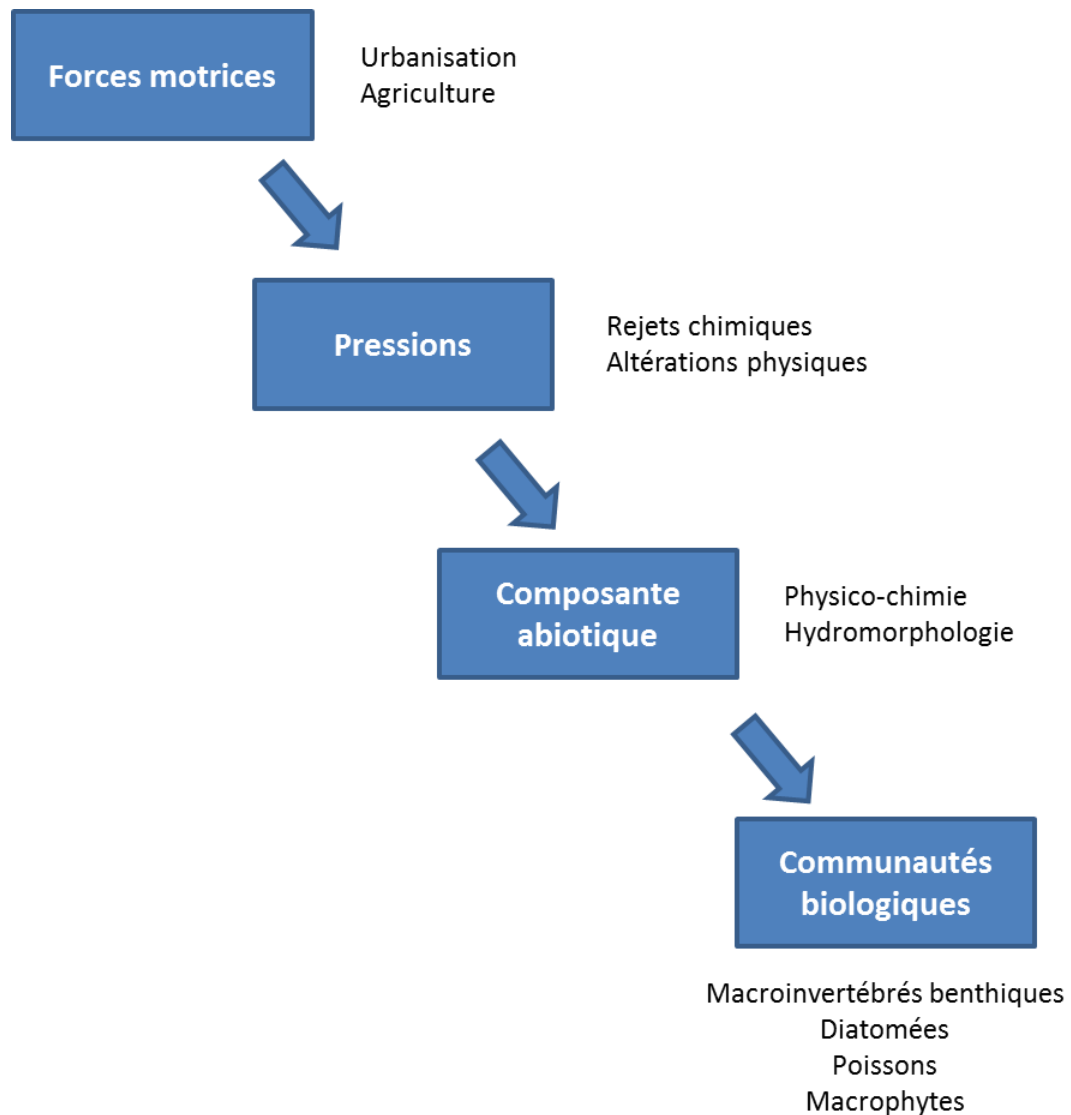


Figure 5. Schéma DPSIR représentant la chaîne de causalité de l'action humaine sur l'état écologique d'une masse d'eau

### 4.3 Quelles recommandations pour l'extrapolation de l'état écologique?

Le réseau de surveillance ne peut pas être utilisé pour évaluer l'état de toutes les masses d'eau, car sa résolution spatiale est insuffisante. Dans la mesure où ce réseau n'a pas été construit dans ce but, nous devons utiliser une autre approche que la surveillance directe pour évaluer l'état de toutes les masses d'eau qui ne sont pas surveillées.

La première solution envisagée était d'utiliser un avis d'expert ou des regroupements de masses d'eau (Kristensen and Christiansen 2012). Mais cette option pourrait produire des évaluations hétérogènes des masses d'eau à risque des différentes régions au sein d'un Etat membre ou à travers toute l'Europe.

Pour ces raisons, nous recommandons donc d'associer l'utilisation de modèles prédictifs à un avis d'experts afin de renforcer la robustesse de l'évaluation (Figure 6).

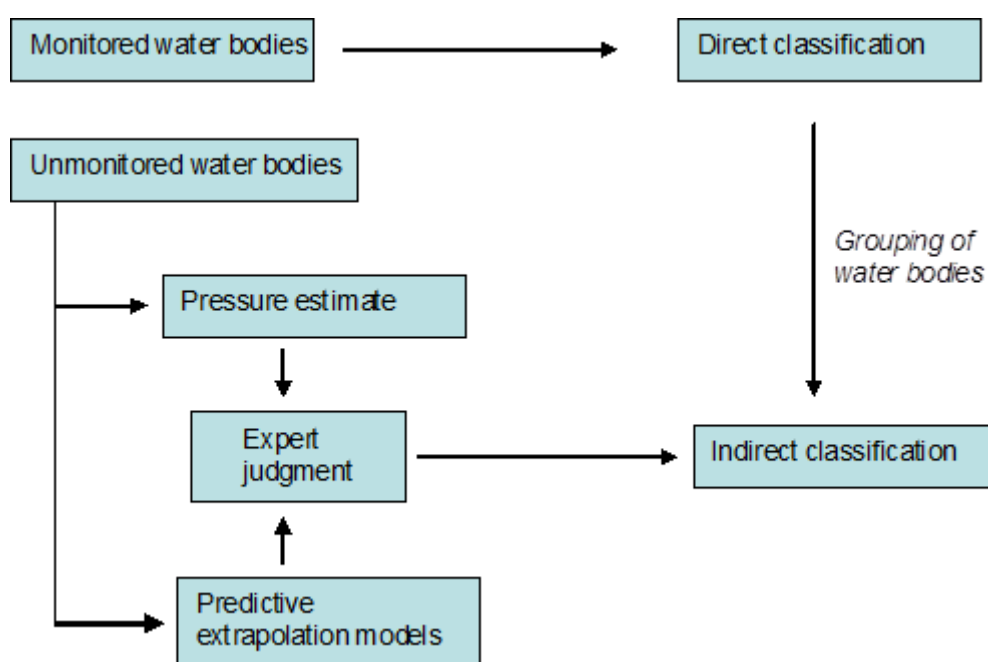


Figure 6. Recommandations pour l'utilisation de modèles prédictifs d'extrapolation pour le processus d'évaluation des masses d'eau non suivies.

Ainsi, les modèles prédictifs de l'état écologique seraient utilisés pour assurer une évaluation cohérente et robuste. Ces deux approches seraient ensuite confrontées afin d'optimiser le processus. Bien que l'utilisation d'un modèle s'inscrive dans une démarche de rationalisation, il peut s'avérer non approprié aux conditions locales spécifiques qui



n'auraient pas été prises en compte dans le jeu de données utilisé pour construire le modèle d'apprentissage. Au contraire, les jugements d'experts pourraient être hétérogènes. Cependant, l'expert aura la capacité de reconnaître des situations exceptionnelles contrairement au modèle d'extrapolation.

Enfin, l'approche que nous proposons doit être fondée sur des données complètes et homogènes à l'échelle de chaque Etat membre. Et pour cela, il sera nécessaire de produire et d'utiliser des données de pression homogènes afin de qualifier les paramètres écologiques (hydromorphologie, physico-chimie générale, polluants spécifiques) supportant des éléments de qualité biologiques. En effet, nous avons vu qu'il est essentiel d'associer l'occupation du sol, la physico-chimie et les altérations hydromorphologiques pour optimiser les capacités de prévision des modèles d'extrapolation de l'état écologique. Mais si ces modèles sont construits à partir de données mesurées à l'échelle des sites du réseau de surveillance, l'utilisation de ces modèles pour extrapoler l'état écologique des masses d'eau non suivies pourrait causer des problèmes du fait de l'indisponibilité d'un ensemble indispensable de données prédictives. Les données d'occupation du sol du bassin versant (Corine Land Cover 2006, (Büttner and Kosztra 2007)) sont disponibles pour l'ensemble du territoire de l'Europe, mais nous avons encore besoin de renforcer les connaissances sur la couverture des corridors rivulaires (Tormos et al. 2014), qui joue un rôle fondamental dans la production de sédiments fins, la pollution et le contrôle de la température et fournit du carbone organique aux cours d'eau (Naiman et Décamps, 2006). Les risques d'altérations hydromorphologiques peuvent être évalués pour l'ensemble du réseau hydrographique de France métropolitaine grâce aux données issues du projet SYRAH-CE (Chandesris et al., 2008), mais des efforts sont encore nécessaires afin de développer une donnée équivalente dans beaucoup d'états d'Europe. Les données physico-chimiques sont mesurées seulement dans les sites du réseau de surveillance. Des modèles capables de simuler des flux de nutriments et de matière organique dans les cours d'eau existent et sont utilisés par les gestionnaires (eg. Pegase, Seneque, Mosquiteau ou encore Nopolu). Ces modèles sont nécessaires à une extrapolation cohérente des paramètres physico-chimiques et constituent un support indispensable au déploiement de modèles prédictifs capables de simuler un état biologique à partir de pressions évaluées sur les masses d'eau non suivies.

## 5 Plan de travail de la thèse

Dans ce contexte, nous avons défini le plan de travail suivant :

Dans la première partie de la thèse, nous avons développé des modèles explicatifs de l'état écologique à large échelle capables de donner des réponses applicables aux échelles de décision et de gestion, à savoir européenne, nationale et régionale avec comme objectif principal de comparer les relations entre (i) les pressions combinées représentées par l'occupation du sol et (ii) l'état écologique des cours d'eau de quatre pays européens : la France, la Slovaquie, l'Estonie, et le Royaume-Uni (Angleterre et Pays de Galles). Nous avons développé des modèles pression-impact reliant les indices basés sur les invertébrés benthiques à l'occupation du sol du bassin versant et du corridor rivulaire. Les données des quatre pays ont en outre été stratifiées en ensembles géographiquement cohérents basés sur les hydro-écorégions d'Europe. Les objectifs plus spécifiques de l'étude étaient 1) d'établir une hiérarchie des catégories d'occupation du sol influant sur l'état écologique, en particulier l'agriculture et l'urbanisation, 2) d'identifier les tendances régionales dans ces relations pression-impact et 3) d'évaluer le poids relatif de la pression du bassin et des échelles de corridors riverains, et le possible effet tampon de la couverture des zones rivulaires.

Dans une deuxième partie, nous avons construit un corpus de modèles capables d'expliquer d'une part la variabilité des indices biologiques utilisés dans les réseaux et de prédire d'autre part l'état écologique des masses d'eau françaises non suivies par le réseau de contrôle et de surveillance (RCS). Les indices macro-invertébrés benthiques, diatomées et poissons ont été utilisés pour déterminer l'état écologique de 1200 sites du RCS répartis de manière homogène sur l'ensemble du territoire français. Les pressions prises en compte pour expliquer et prédire l'état écologique couvrent trois échelles spatiales : bassin versant, tronçon hydromorphologique, station. Le jeu de données prédictives recouvre trois types de pressions : les pressions d'occupation du sol, les pressions hydromorphologiques et les pressions physico-chimiques mesurées respectivement à l'échelle du bassin versant, du tronçon hydromorphologique et du site. Cela devrait permettre de répondre aux questions suivantes : (i) Quelles sont les pressions qui influencent le plus les outils de mesure de l'état

écologique ? (ii) Comment prédire l'état écologique des masses d'eau non suivies dans le cadre des réseaux ?

Dans une troisième partie, nous voulons tester si les relations entre les pressions à l'échelle locale et l'état écologique sont hiérarchiquement influencées par la typologie régionale (basée sur la physiographie naturelle et les forces motrices). Nous avons donc développé un modèle hiérarchique qui relie l'indice invertébrés multimétrique (I2M2) à des pressions hydromorphologiques et physico-chimiques selon des modèles hiérarchiques régionalisés. Pour ce faire, les bassins ont été regroupés en fonction de leurs caractéristiques naturelles et des gradients de forces motrices. Nous avons supposé que l'organisation hiérarchique des cours d'eau conduisait à différentes réponses des organismes à la pression, en fonction des caractéristiques du site étudié dépendant de l'échelle régionale.

Enfin, dans une quatrième et dernière partie, nous avons développé une approche nouvelle qui propose de prendre en compte ces connaissances sur l'organisation des échelles et sur les liens entre pressions et état écologique en construisant et en analysant un modèle structurel basé sur la méthode PLS path modelling qui permet de relier entre elles des variables latentes correspondant aux pressions d'occupation du sol, hydromorphologiques et physico-chimiques et d'analyser *in fine* leurs effets sur l'état écologique mesuré ici à travers l'indice invertébrés multimétrique : l'I2M2. Le développement de ce modèle a pour but de répondre aux questions suivantes : (i) Les différents niveaux d'échelles emboîtées jouent-ils un rôle différent sur la relation pressions-état écologique ? (ii) Les liens indirects entre pressions et état écologique modifient-t-ils la hiérarchie des pressions ? (iii) Ce modèle permet-il de mieux comprendre le rôle particulier de l'hydromorphologie sur l'état écologique ?

Chacune de ces quatre parties a fait l'objet d'un article scientifique (déjà parus pour les deux premiers). Trois articles complémentaires qui ont été réalisés dans le cadre de collaborations sont utilisés dans l'introduction et la discussion de ce mémoire. Ces articles n'ont pas été intégrés au corps du texte mais sont présentés dans les annexes A, B et C.



# Chapitre 2 - Concepts, données et méthodes

## 1 Organisation d'un cours d'eau, hiérarchie d'échelles et régionalisation

### 1.1 L'hydrosystème (Amoros et Petts, 1993)

Au cours des années 70, les réflexions scientifiques ont commencé à décrire les rivières comme des systèmes fluviaux dynamiques afin de mieux décrire leur fonctionnement en interaction avec leur plaine d'inondation et l'ensemble de leur bassin versant (Amoros and Bravard 1985; Hynes 1975; Schumm 1977; Vannote et al. 1980).

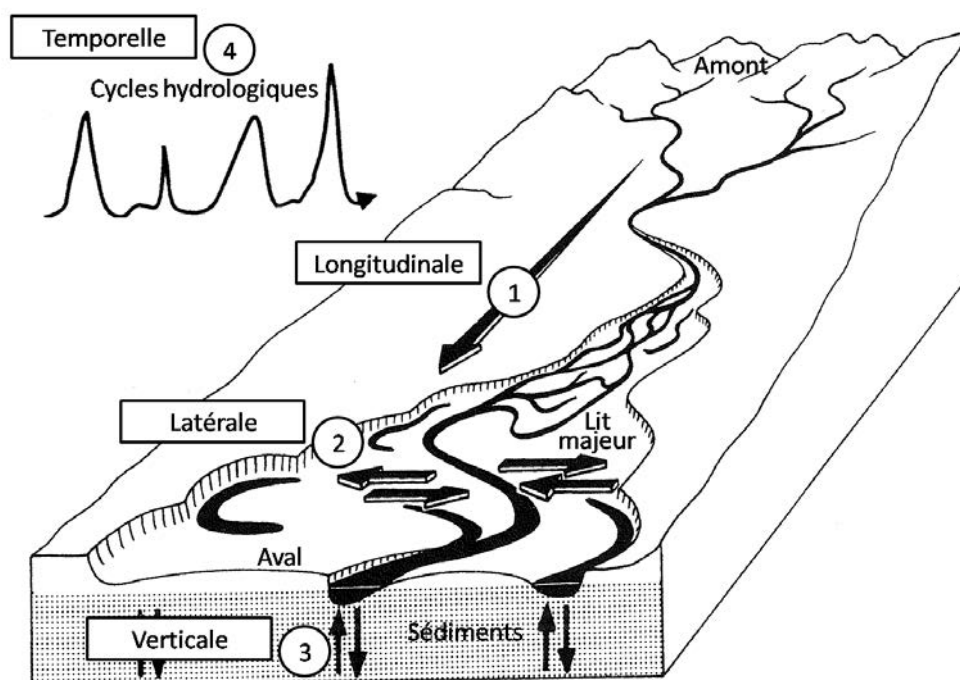


Figure 7. Schématisation des quatre dimensions de l'hydrosystème (d'après Amoros and Petts, 1993).

A la vision linéaire d'une rivière, dont le peuplement serait régi essentiellement par la qualité de son eau, s'est progressivement substituée celle d'un paysage aquatique : l'hydrosystème (Amoros and Petts 1993).

Cet hydrosystème est constitué par l'imbrication d'une mosaïque d'écosystèmes d'eau courante (chenal principal), d'eau plus stagnante (bras morts, plans d'eau), semi-aquatiques et terrestres (marais, forêt alluviales), aussi bien superficiels que souterrains, en interaction dans la plaine alluviale. Cet ensemble interactif de milieux naturels ou anthropiques est caractérisé par des échanges multidirectionnels de matières et d'énergie conceptualisés par Amoros et al. (1987) et Ward (1989) dans une structure à quatre dimensions (Figure 7) décrite par Amoros et Petts (1993) de la façon suivante :

- 1) La dimension longitudinale. Les rivières, les fleuves constituent des systèmes étirés dans l'espace, mais solidaires dans leur dimension longitudinale, dans la mesure où la dynamique des systèmes situés en aval dépend des processus physiques et chimiques qui se déroulent en amont. Cet axe longitudinal traduit l'évolution des principales caractéristiques hydrauliques, géomorphologiques et écologiques des cours d'eau suivant un gradient plus ou moins continu de la source à l'embouchure.
- 2) La dimension verticale. Il existe des relations étroites entre les eaux de surface et les eaux souterraines qui se manifestent, en particulier, par des échanges d'eau et d'éléments dissous : sels minéraux, nitrates, polluants... Les nappes souterraines assurent le maintien du débit en période d'étiage, alors qu'en période de crue les eaux de surface rechargent les nappes au niveau des zones d'inondations.
- 3) La dimension transversale. Le fleuve fonctionne avec sa plaine alluviale constituée d'une mosaïque d'écosystèmes. Les zones riveraines fournissent aux eaux courantes l'essentiel de leur alimentation en matière organique sous forme de détritus. Pour l'ensemble du bassin versant, l'utilisation des sols (boisements, cultures, urbanisation) a des conséquences sur le ruissellement et le fonctionnement hydrologique ainsi que sur les apports en substances particulières et dissoutes.
- 4) La dimension temporelle. Au-delà des cycles annuels, les systèmes aquatiques actuels sont un héritage tant sur le plan géomorphologique que sur le plan écologique, des changements climatiques et tectoniques qui ont eu lieu dans le passé. Pour comprendre leur structure et leur fonctionnement, il est nécessaire de connaître cette histoire ainsi

que les conditions de mise en place des flores et des faunes. Dans un passé plus récent, divers types d'aménagements (barrages, endiguements) ou d'activités (prélèvements de granulats) ont souvent modifié le lit des rivières. A chaque échelle de temps se déroulent des phénomènes qui ont une influence sur la dynamique actuelle.

## 1.2 L'organisation en échelles emboîtées

Frissell et al. (1986), repris ensuite par de nombreux auteurs (Allan 2004; Allan and Johnson 1997; Poff et al. 1997; Thorp 2014), ont décrit l'hydrosystème comme un système présentant une organisation en plusieurs niveaux d'échelles hiérarchiques emboîtés (Figure 8). Les déterminants primaires à l'échelle régionale (relief, climat, géologie) forment les variables de contrôle (régime hydro-sédimentaire, structure de la végétation rivulaire, connectivité latérale du cours d'eau et connectivité verticale). De celles-ci dépendent les facteurs clés du fonctionnement écologique des cours d'eau (habitat physique, climat aquatique et réseaux trophiques).

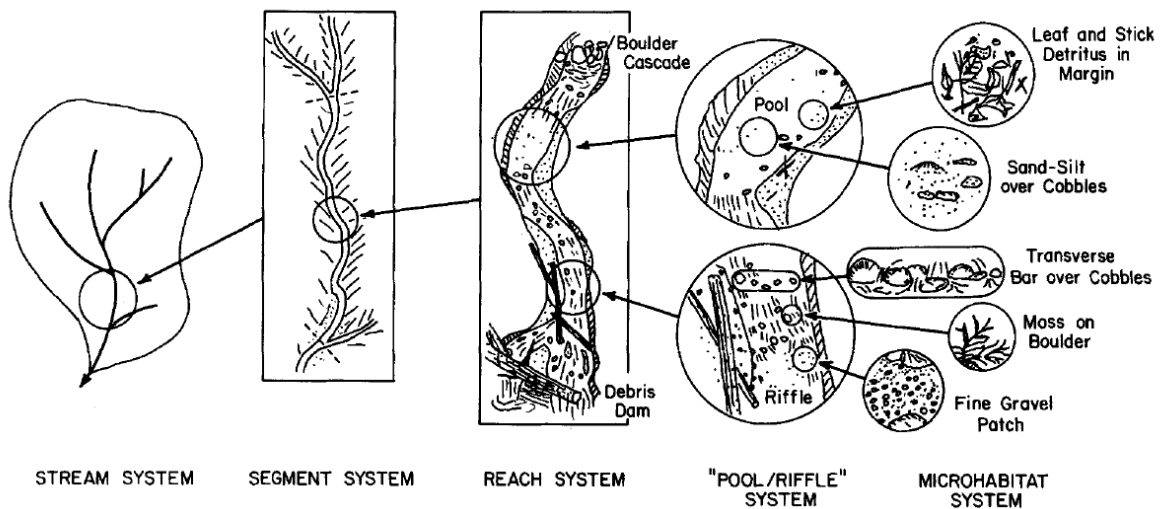


Figure 8. Relations hiérarchiques entre l'habitat et les caractéristiques du paysage des cours d'eau.

Les multiples unités de microhabitat se trouvent au sein de chaque unité de chenal, comme les radiers ou les mouilles; plusieurs unités radiers/mouilles constituent un tronçon; les tronçons sont contenus dans les segments de la rivière, qui font partie eux mêmes d'un bassin versant, qui est souvent un affluent dans un grand bassin de la rivière. (d'après Frissell et al., 1986).

On peut résumer comme suit l'enchaînement hiérarchique des facteurs qui déterminent le fonctionnement écologique des cours d'eau (Figure 9) : à l'échelle locale, dans le milieu

aquatique, la biodiversité et la productivité des communautés vivantes répondent aux « facteurs-clés » que sont l'habitat physique (hydraulique, substrat), le climat aquatique (lumière, température, gaz dissous, hydrochimie), et les ressources trophiques (production primaire endogène et matière organique exogène) ; toutefois l'influence de l'habitat sur des paramètres essentiels (oxygénation, ressources trophiques) conduit à assigner un rôle prépondérant aux facteurs physiques. A l'échelle du tronçon, la dynamique spatio-temporelle des facteurs-clés dépend du régime hydro-sédimentaire et de la géomorphologie de la vallée ; ces « facteurs de contrôle » définissent la morpho-dynamique fluviale (forme et stabilité du lit), la structure de la végétation rivulaire (qui influe significativement sur le milieu aquatique), et la connectivité du cours d'eau (interactions avec la zone inondable). Finalement, à l'échelle régionale, ces facteurs dépendent de « déterminants primaires » que sont la géologie (nature des roches), le relief (géomorphologie) et le climat (régime des températures et précipitations). Les sols et la végétation du bassin interviennent évidemment dans le bilan hydrique et sédimentologique, mais ils sont en dernier ressort déterminés par les caractéristiques géophysiques et climatiques.



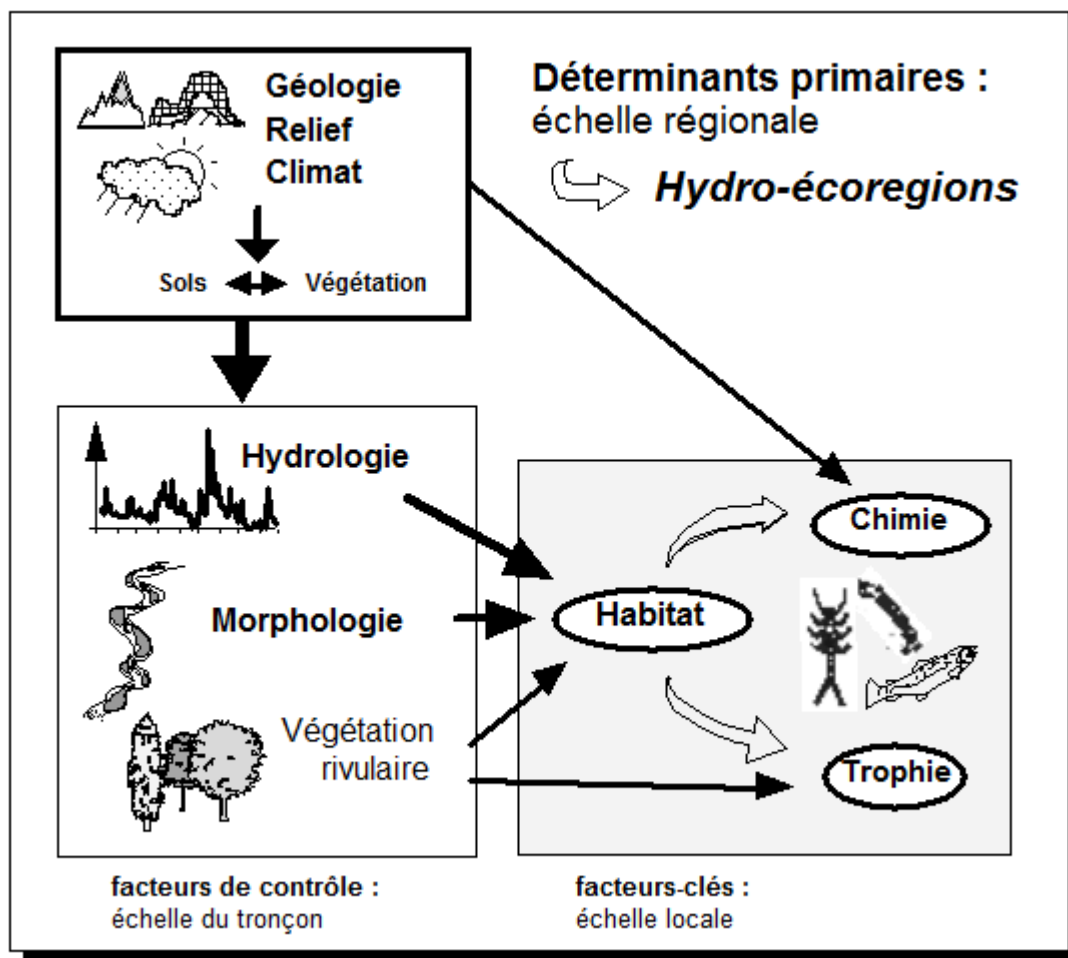


Figure 9. Cadre conceptuel de la régionalisation : représentation schématique de l’emboîtement hiérarchique des facteurs de contrôle des écosystèmes d’eau courante (d’après Wasson et al., 2002)

### 1.3 Les hydro-écorégions.

La démarche régionale développée sur le bassin de la Loire par Wasson et al. (1993) est basée sur une approche descendante dont le principe de base énoncé par Lotspeich (1980) et Lotspeich and Platts (1982) se résume ainsi : « il est plus pertinent de classifier les causes que les conséquences, et donc les facteurs de contrôle globaux plutôt que leurs conséquences au niveau local ». Ce concept s’est enrichi au contact des théories de contrôle hiérarchique des hydrosystèmes, et particulièrement celle de l’emboîtement des échelles physiques. Toutes ces théories reconnaissent comme déterminants primaires du

fonctionnement écologique des cours d'eau, à l'échelle du bassin, la géologie, le relief et le climat.

Cette démarche a été reprise par Wasson et al. (2002) pour construire les Hydro-écorégions de la France. A l'échelle du bassin, les déterminants primaires universellement reconnus du fonctionnement écologique des cours d'eau sont la géologie, le relief et le climat. Ce concept s'inspire des théories de contrôle hiérarchique des hydrosystèmes, et repose particulièrement sur l'emboîtement des échelles physiques, du bassin jusqu'au micro-habitat.

Ce cadre conceptuel justifie une régionalisation des écosystèmes d'eau courante à partir des déterminants primaires géophysiques et climatiques, qui permettent de délimiter des « hydro-écorégions ». Les hypothèses de base sont les suivantes : 1) à l'intérieur d'une même région les cours d'eau présenteront des caractéristiques physiques et biologiques similaires et un même gradient d'évolution longitudinale, et 2) les écosystèmes d'eau courante de différentes régions devraient se distinguer sur au moins un paramètre abiotique important, conduisant à des différences significatives et quantifiables au niveau des peuplements.

Des couches d'information spatialement homogènes sur les structures géophysiques et climatiques sont traitées par une approche géographique, au moyen d'un SIG. La délimitation des régions se fait visuellement, à partir d'une recherche des discontinuités naturelles de ces facteurs, et des concordances régionales de leur distribution.

Comme pour toute partition, le principe de base est de minimiser la variance intra-régionale et de maximiser les différences inter-régions. La cohérence de la partition est testée a posteriori, d'une part sur les paramètres qui ont servi à construire les régions, d'autre part sur des données totalement indépendantes recueillies à l'échelle locale.

Les points suivants peuvent être soulignés :

- Les données de base sont des caractéristiques physiques naturelles : géologie, géomorphologie, relief, climat (températures et précipitations). La végétation naturelle (potentielle), qui révèle la nature des roches et constitue un indicateur bioclimatique apporte également des informations complémentaires.

- Les paramètres utilisés ne sont pas affectés par l'anthropisation (sauf pour les dérives climatiques), ce qui est cohérent avec l'objectif de la DCE, qui consiste à définir des conditions naturelles de référence par types.
- L'approche descendante permet de définir deux niveaux hiérarchiques : le niveau 1 (HER-1) correspond aux grandes structures géophysiques et climatiques, et le niveau 2 (HER-2) correspond à des variations régionales à l'intérieur de ces types, ou dans certains cas à des « exceptions typologiques » dans des ensembles par ailleurs beaucoup plus homogènes.
- Les cours d'eau intègrent rapidement l'hétérogénéité locale de leur bassin, ce qui justifie la délimitation de régions d'une taille suffisante, identifiables à une échelle de l'ordre du 1/1.000.000ème. Il reste évidemment une hétérogénéité locale (à l'échelle des unités de paysage, de 1 à 100 km<sup>2</sup>) qui peut être considérée de manière statistique à l'intérieur des HER. A des échelles plus fines, l'hétérogénéité ne ressort plus de la régionalisation, mais doit être traitée par une analyse du linéaire.
- Il existe des frontières naturelles (géomorphologiques et climatiques) relativement nettes, mais aussi des gradients pluviométriques et altitudinaux. Pour ces derniers, nous avons privilégié une régionalisation par « massifs » géomorphologiques avec l'hypothèse d'une évolution longitudinale prévisible des cours d'eau dans ces massifs.
- Cette méthode de régionalisation présente de fortes similitudes avec celle d'Omernick (1987), largement utilisée par l'US-EPA pour la définition des conditions de référence et la bioindication. Elle s'en distingue cependant par une analyse séparée des structures naturelles et anthropiques, et des améliorations techniques grâce notamment à l'utilisation de données numériques.
- Les écorégions initialement proposées par la DCE comme support à la construction d'une typologie nationale étaient inadaptées au contexte français car établies d'après la seule distribution des invertébrés aquatiques (Illies 1978) et n'étaient donc pas représentatives du fonctionnement des cours d'eau.

Cette démarche a conduit Wasson et al. (2002) à définir pour la France 22 hydroécorégions de niveau 1 (Figure 10).



Figure 10. Carte des hydro-écorégions de niveau 1 de la France métropolitaine

## 1.4 La taille des cours d'eau

Les facteurs hydrologiques et morphologiques (facteurs de contrôle) conditionnent très largement la structure physique et le fonctionnement écologique à l'échelle du tronçon. Le gradient longitudinal (amont-aval) de ces facteurs a fait l'objet depuis plus d'un siècle de très nombreux travaux aboutissant à des propositions de zonations (Huet 1949; Illies and Botosaneanu 1963), ou des typologies longitudinales (Vannote et al. 1980; Verneaux 1976) plus ou moins fonctionnelles (Wasson 1989).

L'ordination de Strahler (1957) est une méthode simple, robuste, permettant de proposer des limites typologiques au niveau des discontinuités longitudinales réelles du réseau hydrographique correspondant à des changements de dimension (de taille des cours d'eau).

L'intérêt majeur de l'ordination de Strahler (Figure 11) est de prendre en compte la structure et la densité du réseau qui reflète, en première approche, la quantité d'eau transportée en surface durant les épisodes de forts débits. Ces épisodes constituent un des indices les plus significatifs en termes de processus hydrologiques au niveau du bassin (Gregory and Walling 1973). Ils sont un élément majeur dans la structuration morphologique du cours d'eau, en particulier la section à plein bord et les faciès morphodynamiques.

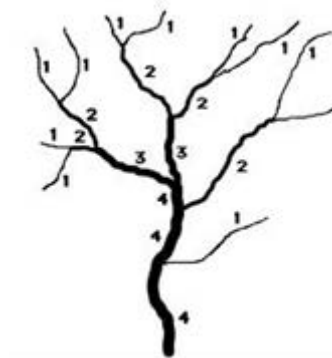


Figure 11. Méthode d'ordination des cours d'eau selon Strahler (1957). L'ordination consiste à attribuer le rang 1 aux cours d'eau en tête de bassin, puis, en progressant vers l'aval, chaque cours d'eau recevant un cours d'eau du même rang voit son rang augmenter d'une unité.

## 1.5 Concept DPSIR

Ce concept, mis au point par l'agence européenne de l'environnement (EEA) (EEA 2003; Kristensen 2004) se décompose de la manière suivante (Figure 12) : les forces motrices que constituent les activités humaines (agriculture, zones urbaines, industries, tourisme, ...) génèrent des pressions à l'échelle du bassin versant (pollution diffuse) et du tronçon (rejets directs, prélèvements et ouvrages). Ces pressions modifient dans l'espace et/ou dans le temps les processus fonctionnels au niveau hydro-morphologique (structure, hétérogénéité, connectivité, variabilité de l'habitat physique...), biogéochimique (pollution organique, nutriments, toxiques,...) et écologique, et déterminent *in fine* les états au niveau biologique (invertébrés, poissons ...) et chimique (substance toxiques et paramètres physico-chimiques).

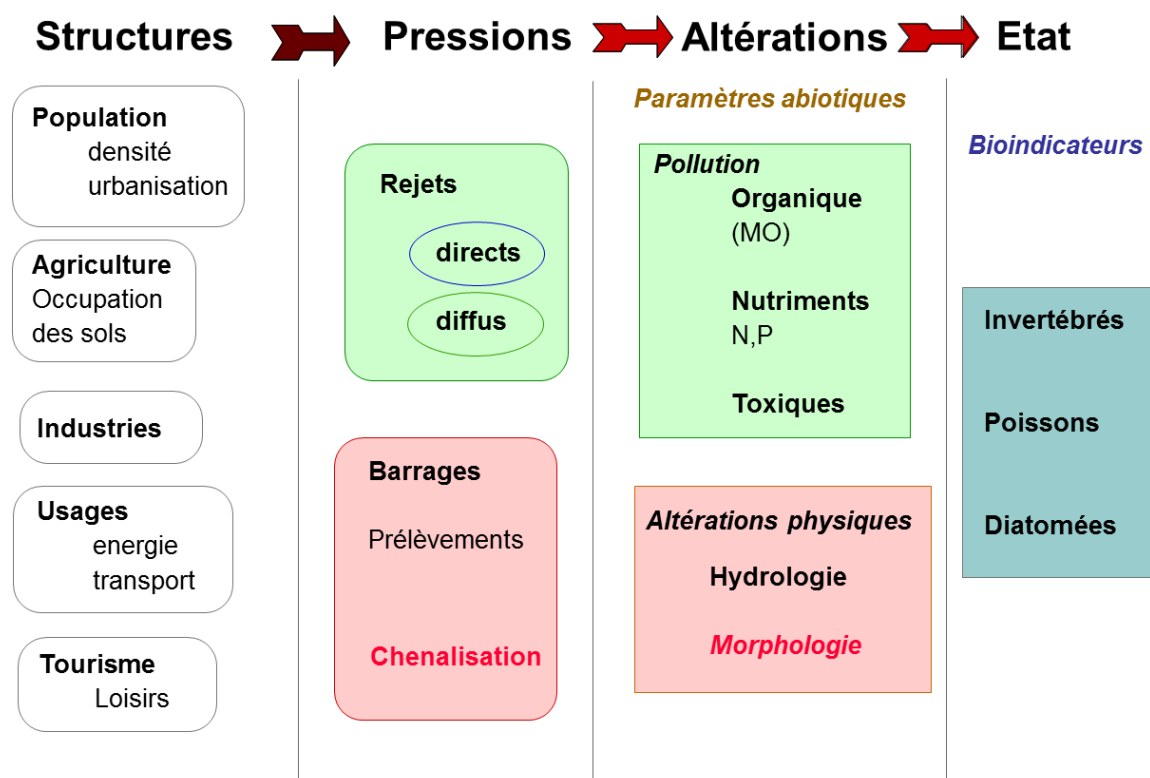


Figure 12. Adaptation du concept DPSIR pour la représentation de l'action humaine sur l'état écologique des cours d'eau

Ces altérations d'états engendrent alors des impacts sur la société au niveau environnemental (biodiversité, paysage...), économique (pêche, tourisme...) et sanitaire (qualité de l'eau, maladies...). Ces impacts, en fonction de leur gravité, de leur ampleur sur

les différents usages de l'eau, peuvent faire réagir la société qui apporte ainsi des réponses (5ème élément) regroupant l'ensemble des mesures et des instruments politiques.

Ces réponses globalisées et localisées portent sur les 4 premiers éléments : mesures préventives dirigées vers les forces motrices (agriculture raisonnée, sensibilisation, ...), mesures curatives vers les pressions et l'état (bandes enherbées, stations d'épurations, reméandrement ...) et mesures palliatives vers les états et les impacts (mélanges des eaux, usines de dénitrification,...).

Cette logique permet d'ordonner les principales composantes du système, et de regrouper différentes pressions issues de forces motrices variées dont l'impact s'exercera sur une masse d'eau donnée. Dans ce schéma, peut-être plus que les éléments, ce sont les liens de causalité qui s'avèrent importants et qui permettent de structurer les approches de modélisation.

## **2 Données utilisées**

### **2.1 Données biologiques**

Nous avons utilisé les données de 1200 sites du réseau de surveillance français suivis sur la période 2007-2012 (figure 1.). Ces sites sont répartis sur 22 hydroécocorégions et couvrent toutes les tailles de cours d'eau.

Nous avons utilisé les données de trois éléments de qualité biologique : les macroinvertébrés benthiques, les diatomées et les poissons.

Un élément de qualité biologique (EQB) est un groupe biologique servant de support à l'évaluation de l'état. Chaque élément de qualité biologique est évalué à partir d'une note d'indice. Un indice est un indicateur global d'évaluation de l'état du système calculé comme une métrique englobant toutes les fonctionnalités du système pour un groupe animal ou végétal donné. L'indice est exprimé en « Ecological Quality Ratio » (EQR). L'EQR, ou écart à la référence, est le rapport entre un état observé et l'état que « devrait » avoir le milieu en l'absence de perturbation anthropique., Son résultat est un ratio sur une échelle de 0 à 1.

L'expression de l'état en EQR est une exigence de compatibilité DCE des méthodes d'évaluation.

Les données macro-invertébrés benthiques ont été recueillies selon un protocole commun normalisé (AFNOR 2009). L'indice biologique calculé à partir de ces listes faunistiques est l'I2M2 (Mondy et al. 2012).

Les données diatomées ont été recueillies selon un protocole normalisé (AFNOR 2007) permettant de calculer l'indice biologique IBD 2007.

Les données poissons ont été collectées par pêche électrique selon un protocole normalisé (AFNOR 2004) et l'indice biologique IPR+ a été calculé (Marzin 2013). Un prélèvement est réalisé une fois par an pour chaque protocole et permet d'obtenir une valeur annuelle pour chaque indice.

Ces trois indices ont été transformés en EQR. Ceci permet de s'affranchir des différences typologiques entre les sites. Le type du site est déterminé comme le croisement d'une hydroécocorégion et d'une taille de cours d'eau (HER x rang) en tenant compte de l'héritage amont (Chandesris et al. 2006).

Les analyses menées dans le chapitre 3 ont été réalisées sur un jeu de données spécifique décrit en détail p 72.

## **2.2 Données d'occupation du sol**

La couche d'information géographique utilisée est CORINE Land Cover (CLC) 2006 (Büttner and Kosztra 2007). Elle couvre l'ensemble du territoire et repose sur une nomenclature standard hiérarchisée à 3 niveaux et 44 postes répartis selon 5 grands types d'occupation du territoire (territoires artificialisés, territoires agricoles, forêts et milieux semi-naturels, zones humides et surfaces en eau).

A partir de l'enveloppe du bassin versant préalablement délimitée d'après un modèle numérique de terrain, nous avons calculé pour chaque site le pourcentage de surface des différentes catégories d'occupation du sol CORINE Land Cover. Nous les avons ensuite regroupées en quatre catégories : urbanisation du bassin versant (zones urbanisées, zones industrielles ou commerciales et réseaux de communication, mines, décharges et chantiers,



espaces verts artificialisés, non agricoles), agriculture intensive (terres arables, cultures permanentes, vergers, vignes, oliveraies, cultures annuelles associées aux cultures permanentes, systèmes culturaux et parcellaires), agriculture non-intensive (prairies, territoires principalement occupés par l'agriculture avec présence de végétation naturelle importante, territoires agro-forestiers) et espaces non anthropisés (forêts et milieux semi-naturels, zones humides, surfaces en eau).

## 2.3 Données hydromorphologiques

### 2.3.1 Démarche SYRAH-CE

Les facteurs clés du fonctionnement écologique des milieux aquatiques : habitat physique, « climat » aquatique (physico-chimie et température), réseaux trophiques dépendent largement des variables régionalisées de contrôle de l'hydromorphologie (régime hydro-sédimentaire, largeur et pente des fonds de vallée), combinées à la structure du corridor rivulaire et au bon fonctionnement des connectivités latérales (berges et ripisylve), longitudinales et verticales (lien avec la nappe d'accompagnement) du cours d'eau (Valette et al. 2012)

Irstea a été mandaté en 2006 par le Ministère en charge de l'écologie et les Agences de l'eau, puis à partir de 2008 par l'Onema pour réaliser une démarche d'évaluation, cohérente à l'échelle nationale, du risque d'altération de l'hydromorphologie de l'ensemble des masses d'eau métropolitaines. L'objectif de cette démarche dénommée SYRAH\_CE (SYstème Relationnel d'Audit de l'Hydromorphologie des cours d'eau) était de mettre au point un outil de caractérisation de l'hydromorphologie des cours d'eau qui s'appuie sur les principes de fonctionnement morphologique des cours d'eau et qui en inventorie les altérations potentielles liées à différents usages.

Le schéma conceptuel de l'audit SYRAH-CE (Figure 13) retient trois niveaux hiérarchiques qui combinent échelles spatiales et mécanismes d'altération. La compréhension et le diagnostic des dysfonctionnements écologiques d'origine hydromorphologique doivent nécessairement intégrer cette organisation en échelles emboîtées du fonctionnement des hydrosystèmes. C'est cet emboîtement qui permet une

démarche de type descendante (top-down) qui renforce l'analyse là où des risques importants ont été identifiés au niveau supérieur.

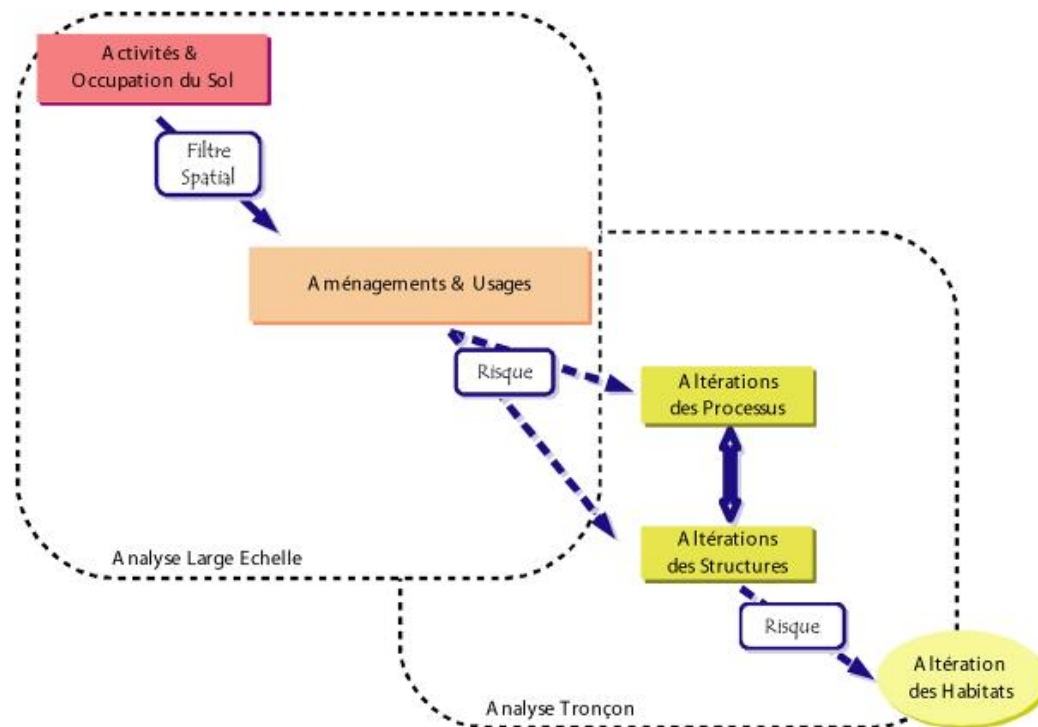


Figure 13. Schéma conceptuel de SYRAH-CE

Le premier niveau d'analyse concerne un compartiment supérieur nommé « Activités et Occupations des sols ». Ce compartiment interagit, selon sa nature, avec le fonctionnement des cours d'eau à plusieurs échelles spatiales latérales et longitudinales (le bassin versant, le lit majeur, le lit mineur). Il regroupe les grandes forces motrices qui sont à l'origine de pressions plus directes sur les cours d'eau.

A travers ce filtre spatial, on peut identifier et quantifier différents « Aménagements et Usages », constituant le deuxième niveau d'analyse, reflétant les activités anthropiques sous forme de « pressions » identifiables par des moyens directs ou indirects. Celles-ci ont des effets, directs et indirects, sur le fonctionnement des cours d'eau qui se traduisent par des « Altérations des processus hydromorphologiques » et des « Altérations des structures physiques ».

Ces altérations sont appréhendées à l'échelle du tronçon, défini par un fonctionnement hydromorphologique homogène (du point de vue des variables de contrôle). L'hypothèse

étant que si un tronçon homogène présente des portions de son linéaire manifestement différentes les unes des autres, on peut alors suspecter une ou plusieurs altérations hydromorphologiques. Ces altérations sont en réalité des ajustements dynamiques au changement de contexte géomorphologique impactant les formes naturelles des cours d'eau et par conséquent, les habitats des différents organismes du milieu aquatique.

Les altérations des structures morphologiques se traduisent généralement par une modification des « formes fluviales ». Les évaluer finement nécessiterait de recourir à des descriptions ou à des mesures de terrain et de définir des états de référence, ce qui est impossible à réaliser. Pour les altérations des processus (flux solides et liquides) s'ajoute une notion temporelle nécessitant le recours à des chroniques d'informations, peu disponibles et difficilement mobilisables.

### **2.3.2 Variables hydromorphologiques utilisées**

La donnée utilisée ici correspond à des descripteurs des pressions exercées sur le fonctionnement hydromorphologique du cours d'eau.

Deux types principaux de descripteurs ont été utilisés.

A l'échelle du bassin versant (zone hydrographique), nous avons utilisé des données de drainage, d'irrigation, d'érosion et de stockage afin de caractériser une éventuelle perturbation des flux solides et liquides naturels.

A l'échelle du tronçon hydromorphologique, nous avons utilisé les descripteurs d'altérations hydromorphologiques suivants : taux de seuils, taux de franchissements, taux de voies de communication à proximité du lit mineur, taux de voies de communication dans le lit majeur, taux de digues dans le lit mineur, taux de digues dans le lit majeur, taux de boisement sur les berges « rideau d'arbres », taux de boisement sur les berges « ripisylve », taux de boisement dans le lit majeur, taux d'occupation de sol de type « artificiel » à proximité du lit mineur, taux de tracé rectiligne, taux de surlargeur ainsi que le rapport profondeur/largeur.

Ces variables ont ensuite été rattachées à chacun des sites de suivi.

## 2.4 Données physico-chimiques

Les sites sur lesquels nous travaillons appartenant au réseau de surveillance national, nous avons pu obtenir les données nécessaires afin de caractériser la pression physico-chimique. Nous avons retenu les paramètres suivants :

- Concentrations en nutriments : ammonium, nitrites, nitrates, phosphore total

L'azote (N) et le phosphore (P) constituent des éléments nutritifs (nutriments) indispensables aux végétaux. Ces substances sont normalement générées par la minéralisation de la matière organique. Toutefois, présentes en trop grande quantité, elles favorisent la prolifération d'algues et de micro-organismes photosynthétiques (eutrophisation).

L'azote ammoniacal ( $\text{NH}_4^+$ ) résulte essentiellement de la dégradation aérobie de l'azote organique (protéines, acides aminés, urée...) lequel provient en grande partie, en milieu urbain, du rejet d'eaux usées non ou insuffisamment épurées. Le  $\text{NH}_4^+$  est dégradé en nitrites ( $\text{NO}_2^-$ ) puis en nitrates ( $\text{NO}_3^-$ ) via le processus de nitrification. Le  $\text{NH}_4^+$  en lui-même n'est pas nuisible mais peut se transformer sous certaines conditions en ammoniac ( $\text{NH}_3$ ), un gaz soluble dans l'eau et toxique pour la vie aquatique.

Le phosphore total (P total) correspond à la somme des organophosphates, des phosphates condensés et des formes organiques du phosphore présents dans l'eau.

- Dégradation de la matière organique : Demande biologique en oxygène (DBO)

La DBO constitue un indicateur de pollution par la matière organique biodégradable. Elle représente la quantité d'oxygène utilisée par les bactéries pour décomposer partiellement ou pour oxyder totalement les matières biochimiques oxydables présentes dans l'eau et qui constituent leur source de carbone (graisses, hydrates de carbone, tensioactifs, etc.). Ce prélèvement d'oxygène se fait au détriment des autres organismes vivants du milieu aquatique. Plus la DBO est élevée, plus la quantité de matières organiques présentes dans l'échantillon est élevée.

Ces paramètres sont mesurés avec une fréquence mensuelle.

## 2.5 Substrats de prélèvements

Les pourcentages de recouvrement des substrats constituant la station de prélèvement ont été utilisés pour qualifier l'habitat local des stations. Les substrats considérés (Tableau 1) sont les suivants : bryophytes, spermaphytes immergés, débris organiques grossiers, chevelus racinaires et supports ligneux, sédiments minéraux de grande taille, blocs, granulats grossiers, spermaphytes émergents, vases, sables et limons, algues, surfaces uniformes dures naturelles et artificielles.

Tableau 1. Substrats échantillonnés selon le protocole multi-habitats normalisé (d'après AFNOR 2009)

Définition du substrat	Ordre d'habitabilité
<b>Bryophytes</b>	11
<b>Spermaphytes immergés (hydrophytes)</b>	10
<b>Débris organiques grossiers (litières)</b>	9
<b>Chevelus racinaires, supports ligneux</b>	8
<b>Sédiments minéraux de grande taille (pierres, galets) (25 à 250 mm)</b>	7
<b>Blocs (&gt; 250 mm) inclus dans une matrice d'éléments minéraux de grande taille (25 à 250 mm)</b>	6
<b>Granulats grossiers (graviers) (2 à 25 mm).</b>	5
<b>Spermaphytes émergents de strate basse (hélophytes)</b>	4
<b>Vases : sédiments fins (&lt; 0,1 mm) avec débris organiques fins</b>	3
<b>Sables et limons (&lt; 2mm)</b>	2
<b>Algues</b>	1
<b>Surfaces uniformes dures naturelles et artificielles (roches, dalles, marnes et argiles compactes)</b>	0

### 3 Les méthodes de modélisation utilisées

#### 3.1 Méthode explicative : la régression Partial Least Squares (PLS)

La méthode de modélisation la plus fréquemment utilisée en écologie est sans aucun doute la régression linéaire multiple. C'est une méthode qui permet de mettre à disposition un outil dont le pouvoir de représentation est extrêmement large. Ce type de modèle est utilisé pour rendre compte des relations linéaires simples existant entre une variable dépendante et des prédicteurs. La régression PLS (Wold et al. 2001) est une adaptation de ce modèle linéaire classique. Son objectif est de pallier le principal défaut de cette dernière : l'instabilité des coefficients de régression due à la colinéarité des prédicteurs. En effet, lorsque la colinéarité devient forte au sein du jeu de variables prédictives, les estimations des coefficients de régression fluctuent énormément d'un échantillon à l'autre (Cramer et al. 1988). Aussi, lorsque l'on utilise la régression multiple pour évaluer l'importance relative des prédicteurs, l'interprétation devient aussi délicate que dangereuse. En effet, plus les variables prédictives sont inter-reliées, moins les coefficients de régression seront fiables pour évaluer leur importance relative. La régression PLS permet d'obtenir des coefficients des prédicteurs interprétables même en cas de corrélation forte entre les prédicteurs. L'équation finale s'interprète comme une équation de régression linéaire, les coefficients reflètent bien l'effet d'un prédicteur relativement aux autres et le  $R^2$  est un bon estimateur de l'efficacité du modèle. La significativité des coefficients est ensuite validée en utilisant la méthode du jack-knife (Efron and Gong 1983; Martens and Martens 2000). Wasson et al. (2010) ont montré l'intérêt d'utiliser cette méthode en écologie lorsqu'on utilise des prédicteurs qui comme ici sont fortement corrélés les uns aux autres. Nous avons illustré cette multicollinéarité en calculant le coefficient de discrimination ( $R^2$ ) de chacune des variables avec l'ensemble des autres prédicteurs.

#### 3.2 Méthode de modélisation prédictive : Les arbres d'inférence conditionnelle (CIT)

La méthode de modélisation utilisée est la méthode des arbres d'inférence conditionnelle (Hothorn et al. 2006). Par rapport à d'autres méthodes d'arbres de décision (comme les CART, (Breiman 1984) qui utilisent un critère d'inégalité), la méthode CIT

propose un test statistique (sur le principe des tests de permutations) permettant de sélectionner la variable la plus pertinente à chaque nœud et de définir un critère d'arrêt de la croissance de l'arbre. Cette méthode peut être décomposée en trois étapes : (1) l'hypothèse nulle d'indépendance entre chaque variable explicative et la variable dépendante est testée par un test de permutation, (2) si l'hypothèse nulle d'indépendance est rejetée, le groupe d'échantillons est divisé en deux sous-groupes définis par leurs valeurs pour la variable explicative assurant la meilleure discrimination en (1), (3) le processus est réitéré sur chacun des sous-groupes formés jusqu'à ce que l'hypothèse nulle d'indépendance ne puisse être rejetée.

L'évaluation de l'efficacité du modèle se fait à partir de la matrice de confusion entre les valeurs observées de la variable qualitative et les valeurs prédites par le modèle pour cette même variable. A partir de cette matrice, nous avons calculé le taux de mauvaise classification du modèle comme premier indicateur de l'efficacité du modèle. Nous avons ensuite calculé la sensibilité (proportion des vrais positifs parmi les sites en mauvais état) et la spécificité (proportion des vrais négatifs parmi les sites en bon état) afin d'évaluer la capacité du modèle à prédire à la fois les situations de bon et de mauvais état écologique. Enfin, nous avons calculé l'aire sous la courbe ROC (AUC) (Hanley and McNeil 1982) qui constitue un bon indicateur de l'efficacité prédictive du modèle (Swets 1988). Ces indicateurs ont été calculés à la fois pour les prédictions réalisées à partir du jeu de données d'apprentissage et à partir du jeu de données de test.

### **3.3 Méthode de modélisation : l'approche PLS**

L'approche PLS (Wold 1982) est une méthode statistique permettant de modéliser des relations complexes entre des variables observées et des variables latentes. Ce type de modèles est généralement appelé modèle d'équations structurelles à variables latentes. Depuis quelques années, cette approche est de plus en plus populaire dans des communautés scientifiques très variées (Vinzi 2008). Les modèles d'équations structurelles (Structural Equation Models) comprennent un grand nombre de méthodologies statistiques (dont l'approche PLS fait partie) qui permettent l'estimation de relations de causalité complexes entre des variables latentes mesurées elles-mêmes par des variables observées dites manifestes.

Un modèle structurel PLS est décrit par deux sous-modèles : (1) le modèle de mesure (ou modèle externe) reliant les variables manifestes (observées) aux variables latentes qui leur sont associées et (2) le modèle structurel (ou modèle interne) reliant des variables latentes dites endogènes à d'autres variables latentes.

On estime tout d'abord les variables latentes ( $\xi$ ) à partir du modèle externe. L'estimation externe  $Y_j$  de la variable latente  $\xi_j$  est construite comme une combinaison linéaire des variables manifestes  $x_{jh}$  :

$$Y_j = \sum_h w_{jh} x_{jh}$$

Où  $w_j$  est le vecteur colonne des coefficients  $w_{jh}$ . On impose à la variable  $Y_j$  d'être centrée-réduite. (Tenenhaus et al. 2005).

Puis on réalise l'estimation interne  $Z_j$  des variables latentes à partir des estimations externes  $Y_i$  des variables latentes  $\xi_i$  liées à  $\xi_j$  :

$$Z_j \propto \sum_{\{i|i \neq j, c_{ij} \neq 0\}} e_{ji} Y_i$$

Où le signe  $\propto$  signifie que la variable située à gauche de ce signe est obtenue par réduction de la variable située à droite. (Tenenhaus et al. 2005).

On répète ces deux étapes jusqu'à convergence, puis on estime les coefficients  $e_{ji}$  du modèle appelés « path coefficients » par des régressions PLS. L'hypothèse nulle de nullité de ces coefficients est ensuite testée en utilisant la méthode de ré-échantillonnage bootstrap.

Un avantage de cette méthode est de permettre d'évaluer les effets directs et indirects d'une variable latente A sur une autre variable latente B (Sanchez 2013). L'effet direct est donné par le « path coefficient » entre les deux variables A et B. L'effet indirect correspond à l'influence d'une variable latente A sur une variable latente B en utilisant un chemin indirect via une troisième variable C. L'effet indirect se calcule comme le produit des « path coefficients » entre les variables A et C et entre les variables C et B. L'effet total d'une



variable latente A sur une variable latente B correspond à la somme des effets directs et indirects.

Pour une interprétation plus aisée du modèle structurel, les effets directs et totaux ont été calculés et transformés en pourcentage.

Le détail de l'algorithme développé par Wold figure dans de nombreux travaux comme ceux de Jakobowicz (2007) et Tenenhaus et al. (2005).

Le modèle structurel est évalué sur la base de la pertinence prédictive des variables latentes. Il convient d'analyser les R2 multiples. Selon Croutsche (2002), trois seuils de R2 multiple peuvent être pris en compte. Si le R2 est supérieur à 0,1, le modèle est significatif. S'il est compris entre 0,05 et 0,1, alors le modèle est tangent. S'il est inférieur à 0,05, alors le modèle n'est pas significatif.



## Chapitre 3 - Relations large échelle entre l'occupation du sol du bassin versant et du corridor rivulaire et l'état écologique

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## 1 Synthèse de l'article

Les impacts écologiques sur les cours d'eau résultent de diverses pressions agissant simultanément : rejets ponctuels, pollution diffuse, modification des flux liquides et solides, structures artificielles, et ruptures de connectivité (Borchardt and Richter 2003). D'après le concept DPSIR (forces motrices, pression, état, impact, réponse) (EEA 2003; Kristensen 2004), les forces motrices sont des activités humaines (agriculture, urbanisation) qui génèrent une combinaison de pressions (rejets polluants, altérations physiques ...) qui modifient les éléments abiotiques de l'écosystème (physico-chimie, hydro-morphologie). Ces modifications affectent les communautés biologiques et l'état écologique. Il est difficile lorsque l'on travaille sur de grands jeux de données de réunir l'information relative aux pressions qui s'exercent sur les cours d'eau. L'occupation du sol représente une source d'information facilement mobilisable pour représenter les forces motrices s'exerçant sur le cours d'eau et il est donc très intéressant d'évaluer son lien avec l'état écologique.

L'étude des relations entre les pressions anthropiques et l'état écologique des cours d'eau et des écosystèmes fluviaux à grandes échelles géographiques pose à la fois des questions conceptuelles et méthodologiques. Les questions principales sont la quantification des réponses des indicateurs biologiques à toute la gamme de pressions ayant un impact d'une part, et la hiérarchisation des causes de ces impacts afin de déterminer leur importance d'autre part. Parmi ceux-ci, l'influence relative de l'agriculture et de l'urbanisation est une question cruciale et controversée (Burcher and Benfield 2006; Moore and Palmer 2005). Une question connexe concerne l'échelle spatiale qui est appropriée pour l'analyse de ces relations: l'impact de l'occupation du sol peut être différente au niveau du bassin ou au niveau du corridor rivulaire (Wang et al. 2001). En outre, les corridors rivulaires boisés peuvent atténuer les effets des pressions exercées par les zones agricoles ou urbaines dans le bassin grâce à leur effet tampon (Moore and Palmer 2005). Cependant, la variabilité régionale de ces relations pression-impact a rarement été analysée à une large échelle géographique. La répartition spatiale des activités humaines et les différences dans les réponses écologiques en fonction des caractéristiques naturelles des types de cours d'eau pourraient conduire à mettre en évidence des pathologies régionales distinctes. L'effet

protecteur des corridors rivulaires doit également être évalué à large échelle, afin de déterminer l'échelle de gestion la plus efficace (Dovciak and Perry 2002).

Dans le but de répondre à ces questions, nous avons développé des modèles explicatifs de l'état écologique à large échelle capables de donner des réponses applicables aux échelles de décision et de gestion, à savoir européenne, nationale et régionale. Les invertébrés benthiques sont de bons indicateurs de l'état écologique pour comparer les relations pression-impact dans différents pays européens pour deux raisons: i) ils répondent à un large éventail de pressions, y compris la dégradation hydro-morphologique (Buffagni and Furse 2006; Hering et al. 2006), et ii) des données sont disponibles à partir des réseaux de surveillance existants. En outre, les indices utilisés ont été soumis au processus d'inter-étalonnage pour assurer la comparabilité des évaluations de l'état écologique (Van de Bund et al. 2008).

L'objectif principal de notre étude était de comparer les relations entre les pressions combinées représentées par l'occupation du sol et l'état écologique des cours d'eau dans quatre pays européens: la France, la Slovaquie, l'Estonie, et au Royaume-Uni (Angleterre et Pays de Galles). Ces quatre pays ont construit une base de données commune dans le cadre du programme européen REBECCA (6<sup>ème</sup> programme cadre de recherche et développement de la commission européenne) qui nous a permis de développer des modèles pression-impact reliant les indices d'invertébrés benthiques à l'occupation du sol du bassin versant et du corridor rivulaire. Les données des quatre pays ont en outre été stratifiées en ensembles géographiquement cohérents basés sur les hydro-écorégions d'Europe (Wasson et al. 2007). Les objectifs plus spécifiques de l'étude étaient 1) établir la hiérarchie des catégories d'occupation du sol influant sur l'état écologique, en particulier l'agriculture et l'urbanisation, 2) identifier les tendances régionales dans ces relations pression-impact et 3) évaluer le poids relatif de la pression du bassin et de l'échelle du corridor riverain, et le possible effet tampon de la couverture des zones rivulaires.

Nos résultats montrent que dans 12 des 14 modèles développés, l'occupation du sol artificielle semble être le facteur négatif le plus important pour les indices basés sur les macroinvertébrés aux échelles nationale et régionale. Les zones artificielles et agricoles ont un poids similaire seulement en Slovaquie au niveau national et dans les Carpates au niveau

régional. Pour les modèles nationaux, l'effet des zones urbaines était d'environ deux fois celui des zones agricoles en France et en Estonie, et pour le Royaume-Uni aucun effet négatif de l'agriculture n'a pu être démontré.

Les zones urbaines ont un fort impact négatif dans toutes les régions analysées. A l'opposé, les forêts et la couverture des sols semi-naturels ont une influence positive (avec une exception pour le Royaume-Uni). Cependant, l'effet de l'occupation du sol agricole, y compris les terres arables, les pâturages et certaines prairies semi-naturelles est beaucoup plus variable. A l'échelle nationale, les zones agricoles et les terres arables en particulier, ont un effet négatif, sauf au Royaume-Uni. En général, l'intensité de cet impact est modérée, mais il constitue un impact potentiel dans une grande partie de l'Europe. Cependant, les tendances régionales sont très différentes et méritent une analyse plus détaillée pour identifier les impacts relatifs des différents types d'occupation du sol, surtout entre les terres arables et les pâturages.

En raison de la covariation du paysage anthropique et des caractéristiques naturelles (Allan 2004), les contraintes naturelles (pente, pluviométrie) de même que les pratiques d'utilisation des sols, pourraient déterminer la sensibilité des écosystèmes, conduisant à différentes réponses régionales. Afin de mieux comprendre les relations écologiques et pour mieux guider la prise de décision, une approche régionale pourrait être utile pour identifier les « pathologies régionales » des cours d'eau, qui pourraient ainsi être gérés différemment, même à l'intérieur d'un pays.

En Slovaquie, les données disponibles ne permettent pas de distinguer les effets des terres arables et des pâturages à l'échelle du bassin. Toutefois, nos résultats ont démontré que l'impact des zones rivulaires agricoles était plus grand dans les montagnes que dans les plaines.

En Estonie, les terres arables et les prairies naturelles ont un effet négatif similaire. Ces dernières sont utilisées comme pâturages, généralement avec une coupe à blanc de l'ensemble du corridor rivulaire qui mène à la croissance intensive des macrophytes dans les rivières à faible courant et à une accumulation de vase qui pourraient expliquer les changements dans la faune macrobenthique.

Au Royaume-Uni, nous avons observé des tendances régionales différentes. Dans le type « Uplands », les pâturages ont un effet positif. Ceci était attendu pour les zones de fougères supportant une intensité d'élevage ovin très faible, mais une pratique d'élevage plus intensive n'a pas produit d'effet négatif. Dans le type « Sedimentary », les forêts ont un effet positif attendu, mais les prairies ont un coefficient négatif. En dépit d'une faible corrélation entre les prairies et l'urbanisation, ce résultat pourrait correspondre à un véritable impact écologique lié à la pollution de l'eau par les déchets agricoles, reconnue comme un problème important dans les années 1990 (Skinner et al. 1997). L'effet positif des terres arables dans cette région est surprenant, mais s'explique probablement par une forte corrélation négative entre la proportion de zones urbaines et de terres arables dans les bassins. Ainsi, la plupart des bassins agricoles sont beaucoup moins touchés par la pression urbaine. On retrouve le même type de résultats chez Wang et al. (2000) qui ont observé une corrélation positive entre l'utilisation des terres agricoles et un indice poisson dans le Wisconsin, où des niveaux élevés d'agriculture sont associés à un faible niveau d'urbanisation.

En France aussi, les modèles régionaux ont montré différents patrons de réponse à l'occupation du sol agricole. L'impact élevé des vignes a été démontré dans le Massif Central et dans le Massif Armoricaïn, en raison de certains vignobles localisés, mais pas dans la région Méditerranée, même si les vignes sont rencontrées dans la moitié de ses bassins. Ici, les caractéristiques naturelles des HER pourraient probablement expliquer la différence dans les réponses régionales. Le substrat rocheux siliceux et les régimes d'écoulement pourraient conduire à un plus grand transfert de sédiments et de pesticides dans les vignes du Massif Armoricaïn et du Massif Central que dans le type Méditerranée, et ainsi à un impact biologique plus fort. Il est toutefois plus probable que l'homogénéité de la couverture de vignoble dans ce dernier type empêche la méthode de modélisation de faire le lien entre un gradient de pression (inexistant) et l'indice basé sur les macro-invertébrés. Les pâturages ont un effet positif dans la région méditerranéenne et dans les tables calcaires. Mais les pâturages ont un effet non-significatif dans les deux régions les plus dévolues à l'élevage: les Massifs Central et Armoricaïn. Cela peut être interprété comme une diminution de l'effet positif des pâturages lorsque la densité d'élevage augmente. De manière plus surprenante, les terres arables semblent n'avoir aucun effet dans les modèles français à l'échelle

régionale, même dans les tables calcaires où la plupart des bassins sont en grande partie consacrés aux cultures. Pourtant, l'effet négatif des terres arables est bien visible à l'échelle nationale. On retrouve probablement la même situation que précédemment avec le vignoble en zone méditerranéenne. L'absence de gradient d'agriculture lié au fait que les niveaux d'agriculture sont homogènes au sein des groupes d'HER (même dans les tables calcaires) empêche de détecter à cette échelle une corrélation entre agriculture et état écologique. Il faut remonter à l'échelle nationale pour retrouver de la variabilité dans le gradient agricole et par conséquent être capable de mettre en évidence l'effet de l'agriculture.

Bien que de nombreux articles traitent de la question des processus d'échelle et des relations inter-échelles, l'influence des pressions agissant à l'échelle des corridors et des bassins riverains nécessite encore une meilleure compréhension (Allan 2004). Gergel et al. (2002) ont conclu à partir d'une revue de la littérature que «de nombreux chercheurs ont tenté de déterminer l'étendue spatiale ou distance au cours d'eau, à partir de laquelle les caractéristiques du paysage influent sur la qualité de l'eau ou des organismes aquatiques, mais cette question reste en suspens".

Les modèles que nous avons construits pour la France, la Slovaquie et l'Estonie ont démontré, à large échelle, une relation négative très nette entre l'occupation du sol artificielle dans les corridors rivulaires et les indices basés sur les macroinvertébrés. Un autre résultat important a été la démonstration des effets positifs généralisés des forêts et des pâturages dans le corridor rivulaire, constatés dans tous les modèles à l'échelle nationale et régionale. Ce résultat indique une influence positive directe sur les invertébrés aquatiques, mais aussi met en évidence un effet tampon qui atténue les effets négatifs des zones artificielles ou des terres arables en dehors du corridor rivulaire. L'influence directe de la forêt rivulaire sur la structure des communautés d'invertébrés est largement reconnue (Maridet et al. 1998; Naiman et al. 2005). Dans nos résultats, le pourcentage de recouvrement des forêts rivulaires est corrélé positivement avec les valeurs des indices d'invertébrés.

L'effet tampon de la ripisylve a été testé dans trois régions françaises, dans les bassins avec soit une pression agricole dominante ou avec des pressions urbaines et agricoles



mixtes. Dans les zones rivulaires, la présence de forêts de feuillus, seuls ou combinés avec des pâturages, a eu un effet positif clair dans cinq des six ensembles de données testés. Dans la plupart des cas, l'augmentation des EQR-IBGN pourrait être suffisante pour passer de la classe moyenne au bon état.

Ces résultats appuient la conclusion que les caractéristiques des corridors rivulaires, avec notamment un mélange de zones boisées et de vastes pâturages, dans les bassins agricoles et urbains, peuvent avoir un impact significatif sur la santé des écosystèmes (Naiman et al. 2005). Alors que l'occupation du sol du bassin versant ne devrait pas changer dans un avenir proche, les corridors riverains sont des zones plus faciles à gérer et leur restauration peut être un moyen efficace pour améliorer l'état écologique des cours d'eau européens.

## 2 Large-scale relationships between basin and riparian land cover and ecological status of European rivers

### Abstract

1. The main objectives of our study were 1) to establish the relative impact of pressures that degrade ecological status, especially those caused by agriculture and urbanization, 2) to identify regional patterns in these pressure-impact relationships and 3) to evaluate the relative weight of the pressures acting at the basin and riparian corridor scales, and the possible buffering effect of riparian areas.

2. We developed large-scale models linking invertebrate indices of ecological quality to river basin and riparian land cover in France, Slovakia, Estonia and UK. Invertebrate indices, transformed to Ecological Quality Ratios (EQR), were taken from national monitoring networks. We based the models on Partial Least Squares (PLS) regressions at national and a hydro-ecoregion (HER) scales. The HERs provided a framework for grouping data in terms of natural river features and human activities.

3. The different national methods provided consistent results that indicated the hierarchy of pressures impacting river invertebrates at the European scale. Artificial land cover (e.g. urban and industrial sites) in the river basin represented the pressure with the most negative impact on invertebrate indices, in all countries and regions.

4. The impact of agricultural land cover was more variable. Arable land had a smaller impact than urban areas, and was insignificant in some models. The proportion of arable land in the river basin appeared to be a weak predictor of agricultural impacts by itself; the type of cultivation and intensity as well as the proximity to the river must be taken into account.

5. At the riparian corridor scale, the negative impact of artificial areas or arable land and the positive effects of forests and pastures were demonstrated in many regions. The protective effect of riparian forests against mixed agricultural and urban pressures was demonstrated in three regions in France. Riparian corridors appear to be manageable areas, and these results strongly support the idea of including their restoration in priority actions for achieving good ecological status.

## **2.1 Introduction**

The environmental objectives of the European Union Water Framework Directive (WFD, 2000/60/EC), to achieve good ecological status for all water bodies in a given time horizon (Article 4), raises serious problems for political decision-makers and water managers, and many questions for aquatic scientists relevant beyond Europe. Initially, most of the research for the directive was dedicated to develop methods to evaluate ecological status (Hering et al., 2004; Furse et al., 2006; Pont et al., 2006). However, scientific support for defining programmes of measures necessary to maintain or improve the ecological status of water bodies has been rather weak.

Ecological impacts in running waters result from various pressures acting simultaneously: point sources discharges, diffuse pollution, alteration of water and sediment regimes, artificial structures, and breaks in connectivity. (Borchardt & Richter, 2003). Following the DPSIR (driving forces, pressure, state, impact, response) concept (EEA, 2003), the driving forces are human activities (e.g. agriculture, urbanization) generating a combination of pressures (pollution discharges, physical alterations...) which alter the abiotic components of the ecosystem (physico-chemistry, hydro-morphology). These alterations affect biological communities and thus ecological status. Because of the lack of consistent datasets of pressures or abiotic parameters, land cover was the only useable source of spatially consistent information at the European scale to represent the driving forces.

The study of the relationships between anthropogenic pressures and ecological status in stream and river ecosystems at large geographical scales involves both conceptual and methodological issues. The main problem is to quantify the responses of biological indicators to the whole array of impacting pressures, and to rank the causes of impacts in order of their importance. Of these, the relative influence of agriculture and urbanisation is a crucial and controversial issue (EEA, 2003; Moore & Palmer, 2005; Burcher & Benfield, 2006). A related issue concerns spatial scale that is appropriate for analysing these relationships: the impact of a given land use can be different at the basin level compared to the riparian corridor level (Wang, Lyons & Kanehl, 2001). Moreover, the buffering capacity of forested riparian corridors can mitigate the effects of pressures from agricultural or urban areas in the basin (Moore & Palmer, 2005). However, regional variability of these pressure-impact

relationships has seldom been analysed at a large geographical scale. Both the spatial pattern of human activities and differences in ecological responses according to the natural characteristics of the river types could lead to distinct regional pathologies. The protective effect of riparian corridors also needs to be evaluated at large scales, to determine the most effective management scale (Dovciak & Perry, 2002).

There is a clear lack of knowledge about the relationships between combined pressures and the ecological status of rivers at large scales (Allan, 2004; Garcia, Villeneuve & Wasson, 2006). Except for the work of Donohue, McGarrigle & Mills (2006) in Ireland, we are not aware of models linking human activities within Europe (evaluated through land cover) to ecological status at a large or even a regional scale. In a pioneer work, Steedman (1988) demonstrated positive relationships between the condition of the fish community and both the percentage of forest in the basin and the proportion of channel with riparian forest, and a strong negative relationship with urbanization in the catchment. He also emphasized the need for a more detailed evaluation of agricultural land use and streamside vegetation to improve predictive models. All these issues remain unresolved (Allan, 2004), even though the answers are prerequisites for defining management policies. Decision-makers must prioritize their actions and define integrated policies tailored towards socio-economical structures, such as urban and rural development zones. For practical reasons, they need to know whether the same policies must be applied everywhere or adapted to regional contexts, and whether the management of the riparian areas can be used to improve ecological status.

We developed large-scale models, based on existing data, to give answers applicable to the decision and management scales, i.e. European, national and regional. Benthic invertebrates appear to be the best indicators of ecological status for comparing the pressure-impact relationships in different European countries for two reasons: i) they respond to a wide range of pressures, including hydro-morphological degradation (Buffagni et al., 2006; Hering et al., 2006), and ii) data were available from existing national monitoring networks. Moreover, the invertebrate indices were submitted to the WFD Intercalibration process, ensuring the comparability of the evaluations of ecological status (Van de Bund, 2008).

The main objective of our study was to compare the relationships between the combined pressures represented by land cover and the ecological status of running waters in four European countries: France, Slovakia, Estonia, and UK (England and Wales). We developed large-scale pressure-impact models that related comparable benthic invertebrate indices to land cover indicators calculated at the basin and riparian corridor scales. Data from the four countries were further stratified into regional datasets according to hydro-ecoregions. The more specific objectives of the study were 1) to establish the hierarchy of the major influences that impact ecological status, particularly agriculture and urbanisation, 2) to identify regional patterns in these pressure-impact relationships and 3) to evaluate the relative weight of the pressures acting at the basin and riparian corridor scales, and the possible buffering effect of riparian land cover.

## **2.2 Methods**

### **2.2.1 Evaluation of pressure**

For each biological monitoring site, land cover indicators were calculated at the basin scale. In France, Slovakia and Estonia, the CORINE Land Cover (CLC) 2000 map was used. This is a vector map, drawn from satellite imagery at a scale of 1:100 000, with a minimum polygon size of 25 ha (<http://terrestrial.eionet.europa.eu/CLC2000>). The CLC is based on a hierarchical standard nomenclature with 3 levels: 5 broad land cover categories at level 1 (1-artificial surfaces, 2-agricultural areas, 3-forests and semi-natural areas, 4-wetlands and 5-water bodies), 15 land cover classes (level 2) and 44 land cover types (level 3). The third level of CLC 2000 was used to describe the land cover in all the areas studied, except for Slovakia where only the level 1 was available at the basin scale. For UK, land cover indicators were from the Land Cover Map (LCM) 2000 (Fuller et al., 2002). Although not identical, the LCM 2000 categories are comparable to those used in the CLC 2000 classification.

Land cover indicators for the riparian corridors were also calculated from the CLC level 3 in France, Slovakia and Estonia. The corridors were delineated by sections 3 km long, 2.5 km upstream and 0.5 km downstream of each biological monitoring site. In France and Estonia, the width of the corridor was adjusted to the river width, according to the Strahler stream order: 100 m for the 1st to 3rd order streams, 140 m for 4th order, 250 m for 5th order, 600

m for order 6th order, 1200 m for order 7th order and 2400 m for 8th order (Pella, Chandesris & Wasson, 2004). In Slovakia, the width of the corridor was fixed at 100 m for all sites. Given the resolution of the CLC map, the riparian land cover indicators do not represent the narrow buffer strips along the river margins but only the proportion of the CLC types recognized in the delineated corridors (Tormos et al., 2006).

### **2.2.2 Biological Datasets**

Invertebrate datasets were extracted from the national monitoring networks. The sampling sites were well distributed spatially, and the coverage was roughly the same for France, Slovakia and Estonia (1 site for 140 to 270 km<sup>2</sup> of land area), but denser for UK (1 site for 33 km<sup>2</sup>). In Estonia, the index used was the Average Score Per Taxon (ASPT), which is based on family-level data (Armitage et al., 1983). The ASPT is a rather robust indicator able to reflect general degradation. The database provided by the Centre for Limnology (Institute of Agricultural and Environmental Science) comprised 168 stations with 179 samples from 1988-2004. In France, the biological stations were monitored by the Regional Environmental boards (DIREN) using the IBGN index (Indice Biologique Global Normalisé) (A.F.N.O.R., 1992). The IBGN is a combination of two metrics: the total number of taxa (14 classes at family level), and the faunistic indicator group representing the presence/absence of 39 indicator taxa, grouped into 9 classes of sensitivity. The index is sensitive to pollution (including toxics), and to general degradation (including habitat alteration). The national dataset, handled by Cemagref (Agricultural and environmental engineering research) currently comprises 3 662 stations and 12 682 samples covering the period 1992-2002. The mean IBGN value for each site was used in the models. In Slovakia, the Saprobic Index (SI) was used (Zelinka & Marvan, 1986). The SI works at the species level but recognizes only those taxa to which saprobic and indicative values have been assigned. It is a sensitive indicator of organic pollution, associated oxygen regime and nutrient concentrations. The dataset provided by The Slovak Hydrometeorological Institute from the Slovakian water quality database comprised 220 stations with 226 samples from 2003. In UK, the indices used in the national classification scheme were the ASPT and the Number of Taxa (N-Taxa), based on family-level data (Hemsley-Flint, 2000). The combination of both metrics reacts to a wide array of pressures. The RIVPACS model (Wright, Sutcliffe & Furse, 2000) was used to predict reference values of the two indices. The Ecological Quality Index (EQI) for ASPT and N-Taxa

at each site was calculated using observed values of each index and the expected values obtained from RIVPACS. The database provided by the Environment Agency comprised 4508 stations and results (as EQI-ASPT and EQI N-Taxa) from 4508 composite 'spring + autumn' RIVPACS samples collected in 1995.

The WFD requires ecological status to be evaluated as an Ecological Quality Ratio (EQR), i.e. a deviation from reference conditions. In Estonia, France and Slovakia, the indices were transformed to EQR values by dividing the observed values by reference values for each national river type. The reference values were derived from data observed at reference sites. The criteria used for the selection of reference sites in these countries complied with the criteria agreed in the WFD Intercalibration process (REFCOND, 2003), and the EQR values were equivalent to those used at the national level for determining the WFD ecological status. For UK, EQI values were used instead of EQRs. The biological quality of RIVPACS reference sites was the best available, which was sometimes poorer than the WFD reference state, particularly in lowlands. As a consequence, RIVPACS EQIs need adjustment to convert them to EQR values that meet the WFD's requirements (Davy-Bowker et al., 2008). For UK therefore, EQRs in lowland England will indicate poorer quality than the EQIs used in this study.

### **2.2.3 Hydro-ecoregion framework**

Ecoregions have been adapted to aquatic ecosystems by Omernik (1987) to define regional objectives for water quality and management (Hughes & Larsen, 1988; Warry & Hanau, 1993), and are still widely used in the USA (Omernik, 2004). In Europe, the 'WFD ecoregion' map has been adapted from Illies (1978) and has many weaknesses because it reflects biogeography more than river functioning (Verdonschot & Nijboer, 2004). The hydro-ecoregions (HER) are based on a top-down approach, allowing an a priori delimitation of geographical entities in which stream and river ecosystems should exhibit common characteristics. Geology, relief and climate are widely recognized as the primary large-scale determinants of running-water ecosystem functioning (Naiman et al., 1992), and HERs are delimited on the basis of these factors (Wasson, 1996; Wasson et al., 2002a; Wasson et al., 2002b). This method was used in France to define the national WFD river typology (Wasson et al., 2002c). The rationale and a description of the European HERs can be found in Wasson

et al. (2007). The HER map was used to stratify both river types and human activities in this study.



Table 1. General characteristics of the regions used in the analyses: hydro-ecoregion (HER), geology-relief and climate

Region	1	HER name	Geology- Relief	Climate
<b>France</b>				
Armorican		BRETAGNE NORMANDIE	Siliceous lowlands	oceanic
Armorican		MASSIF ARMORICAIN CENTRAL	Siliceous lowlands	temperate
Tables Calcaires		FRENCH TABLES CALCAIRES	Tabular calcareous	temperate
Massif Central		FRENCH MASSIF CENTRAL SOUTH	Siliceous mid-alt. mountain	temperate
Massif Central		FRENCH MASSIF CENTRAL NORTH	Siliceous hills	temperate
Mediterranean		LANGUEDOC	Alluvial plain	mediterranea
Mediterranean		PROVENCE	Heterogeneous relief	mediterranea
<b>Slovakia</b>				
Carpathians		INNER CARPATHIANS	Siliceous mid-altitude mountain	alpine
Carpathians		OUTER CARPATHIANS	Sedimentary mid-altitude	alpine
Pannonians		PANNONIAN PLAIN	Clay-rich plains	temperate
Pannonians		PANNONIAN FOOTHILLS	Detritic hills	temperate
Pannonians		TRANSDANUBIAN HILLS	Calcareous hills	temperate
<b>Estonia</b>				
Estonia		BALTIC PLAIN	Moraine plains	continental
<b>UK</b>				
English Uplands		ENGLISH UPLANDS	Calcareous hills	oceanic
Cornwall and		WALES	Siliceous hills	oceanic
Cornwall and		CORNWALL	Siliceous lowlands	oceanic
English Chalk		ENGLISH CHALK	Tabular calcareous	temperate
English		ENGLISH SEDIMENTARY	Tabular calcareous	temperate

#### 2.2.4 Study areas

We developed models at both the country level and for HERs or groups of HERs with similar characteristics (Fig. 1 and Table 1). The main land cover characteristics of these regions are summarized in Table 2.

In France, sites were stratified into 4 regions representing geographical diversity and covering 55% of the country. The Armorican region corresponds to Hercynian shield lowlands; most of the urban areas are located along the coasts, and the agriculture, described as 'complex cultivation patterns', is mainly intensive animal husbandry, with a high density of animals (>1.25 Livestock Units per hectare of catchment). The Tables Calcaires region covers 25% of France; the landscape comprises lowlands and plains, with low-energy rivers. The land cover is characterized by the strong contrast between the densely urbanized areas around Paris and the sparsely populated rural areas, by extensive agricultural land

cover mainly dedicated to crops (wheat, maize), to pastures near the Channel coast (Normandy), and locally to vineyards (Champagne). The Massif Central is a granitic region of hills and mid-altitude mountains, where land cover is mainly pasture, forest and semi-natural areas; population density is well below the national average, but there are some important industrial and urban concentrations in the main valleys. In the Mediterranean region, urbanisation has developed along the coast and in the plains, with a high tourist pressure in summer. The vineyards and orchards in the Rhône valley and coastal plains are surrounded by extensive hilly landscapes covered by forests and natural vegetation.

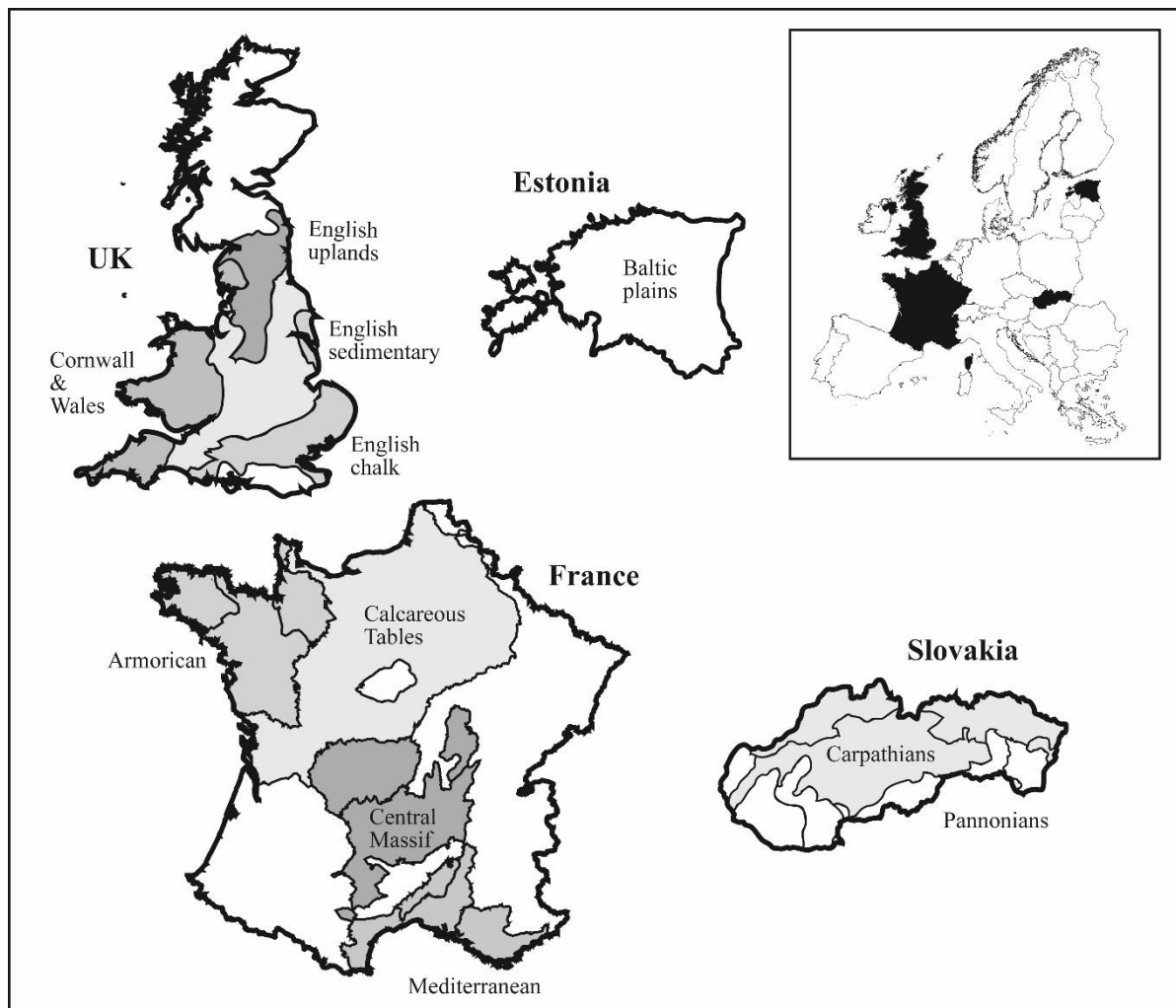


Fig 1. Geographical location of the countries and hydro-ecoregions analysed.

Table 2. Percentage of land cover classes (from CORINE Land Cover 2000) in the regions used in the analyses.

Country	Region	CLC code	Other high pressure			Other low pressure			Natural Areas	Wetlands and water
			Urbanized	Arable land	agriculture	Pastures	agriculture	Forests		
		<i>1</i>	<i>211</i>	<i>22, 24</i>	<i>231</i>	<i>243, 244</i>	<i>31</i>	<i>32, 33</i>	<i>4, 5</i>	
France	Armorican	4.8	33.6	24.3	25.1	1.8	7.7	1.1	1.6	
	Tables									
France	Calcaires	5.9	55.2	5.7	13.6	1.6	16.4	0.7	0.8	
France	Massif Central	2.3	6.5	15.1	34.9	3.8	32.5	4.5	0.5	
France	Mediterranean	8	5.6	34.7	0.4	3.3	25	18	4.9	
Slovakia	Carpathians	4.3	17.6	1.1	8.2	9.1	53.3	5.9	0.4	
Slovakia	Pannonians	7.4	65.9	2.2	1.9	4.9	14.8	1.5	1.4	
Estonia	Estonia	1.9	15	3.9	5.7	7.9	46.1	9.5	10	
	Cornwall and									
UK	Wales	4.1	5.4	7.6	47.5	3.6	8.9	20.7	2.3	
	English									
UK	Uplands	8.7	11.3	2.6	44	1	6	21.8	4.6	
	English									
UK	Sedimentary	12.1	55.1	4.9	20.9	1.3	3.2	1.2	1.3	
UK	English Chalk	14.2	63.1	2.5	8.8	1.9	6	1.2	2.3	

Slovakia was divided into two regions. The Carpathians, with tertiary mountains reaching 1500 m and mostly covered by forests, represents most of the country. Human settlements are scattered along small and middle-sized rivers. The remaining territory corresponds to the Pannonians, plains and lowlands bordered by hills with densely populated fertile lands and large rivers. The main sources of organic pollution and nutrients are municipal waste waters, agriculture and reservoirs.

Estonia was included in only one HER, the Baltic plain, composed of moraine overlying limestone and sandstone and with a relatively cold climate. Most rivers are rather short with a low slope. More than half the country is covered by woodland and natural areas, and a third is used for agriculture, including pastures. Wetlands (reeds, fens and bogs) and water bodies cover 10% of the country, but woodlands also include swampy forests, floodplains and bog pine forests. Urban and industrial areas, concentrated mainly in the north and northeast of the country (Meiner, 1999), cover 2% of the country. Many rivers were modified in 1950s-1970s by deepening and damming.

The UK was stratified into 4 regions. The first two correspond to hilly landscapes under an Atlantic climate: the English Uplands with a complex geology made of limestone bedrock overlain by base-poor millstone grit deposited during the last glaciation, and Cornwall and Wales with granitic and metamorphic bedrock. In both regions, most of the landscape is covered by pastures and semi-natural vegetation, but urbanisation is common in valleys in the English Uplands. The two lowland regions with temperate climates are separated by dominant geology: pre-Cretaceous sedimentary rocks for the English Sedimentary, and the characteristic English Chalk with areas of clay. Both regions have a very high coverage of urban and arable land, but pastures are also important in the former.

### **2.2.5 Statistical models**

We used the Partial Least Square (PLS) regression to model the influence of land cover (CLC types) on the invertebrate indices (EQR or EQI values). PLS regression is an extension of multiple linear regression (MLR) (Höskuldsson, 1988; Tenenhaus, Gauchi & Menardo, 1995; Tenenhaus, 1998; Wold, 1966; Wold, 1982; Wold et al., 2001). By handling numerous and co-linear X-variables, PLS regression allows us to investigate more complex problems than MLR and analyze all the available data in a more realistic way (Wold et al., 2001). The objective of PLS is to compensate for the main defect of MLR: the instability of the regression coefficients caused by co-linearity of the predictors. Indeed, when co-linearity is strong between the predictive X-variables (strong correlations, large number of predictors, etc.), the estimates of the regression coefficients fluctuate greatly from one sample to another (Cramer et al., 1988) and their interpretation becomes hazardous. In such cases, the MLR coefficients are misleading and un-interpretable. When the correlation between the X-

variables is high, MLR cannot assign 'correct' values to the individual coefficients, but only estimate their joint contribution to Y (Wold et al., 2001). In PLS regression, the coefficients of the predictors can be interpreted as degrees of correlation between each predictor and the dependent variable, even when predictors are strongly correlated. These coefficients are comparable (in sign and amplitude) to those of a simple correlation (Tenenhaus et al., 1995; Wold et al., 2001).

The NIPALS algorithm implemented in XLStat 2006 software (AddinSoft, France) examines both X and Y and extracts components that are directly relevant to both sets of variables. These are extracted in decreasing order of relevance. The number of components to be retained is then determined by cross validation according to an error minimisation criterion, the PRediction Error Sum of Squares (PRESS), which corresponds to the sum of all the prediction error squares calculated from the test sets. This gives a regression equation that is comparable to the equation of the conventional linear regression model. To evaluate the degree of relationship between each predictor and the response variable, we interpreted only the standardized PLS coefficients as giving the relative weights of the different predictors in the models.

Jack-knifing (Efron & Gong, 1983) is a convenient way to estimate the standard errors and confidence intervals of the coefficients directly from the data. Wold (1982) recommended this in his original PLS work, and it has been revived by Martens & Martens (2000). The variation in the parameters of the various sub-models obtained during cross-validation of the PLS model is used to derive their standard deviations, and then the t-distribution is used to give confidence intervals. Since all PLS parameters are linear combinations of the original data, these parameters are close to being normally distributed, and hence jack-knifing works well. We used a 95% confidence interval. PLS coefficients are significant at the 5% level when the confidence interval does not cover zero. Only those land cover categories that had a significant relationship with the biological indices according to the jack-knife test were retained in the models.

The presence of different influential land cover categories in the basins, such as urban and agricultural, can blur the relationships and make them difficult to interpret (Meador & Goldstein, 2003). To discriminate the relative impact of these two major driving forces

better, we tested the effect of agricultural land cover independently of its correlation with the percentage of urban land cover in the basins. For each region, we analysed the Pearson correlation between these two land cover categories. First we produced a linear regression to explain the EQR (or EQI) by the percentage of urban land cover in the basins. Then, we made a multiple linear regression (classical MLR) with the percentage of agricultural land cover added as a second predictor, to evaluate the part of the EQR variation caused by agriculture. Urban land cover was considered to be the whole CORINE artificial surfaces category (CLC 1) in Estonia, France and Slovakia, and the sum of Land Cover 2000 urban and suburban land cover categories (LCM 171 + 172) in UK. Agricultural land cover was evaluated as arable land for Estonia, France (CLC 211) and UK (LCM 4), and as the whole agricultural category for Slovakia (CLC 2). The urban and agricultural models were compared using an extra sum of square F-test (Sokal & Rohlf, 1995).

Finally, in three French regions, we tested the effect of the presence of potentially protective land cover, pastures (CLC 231) and forests (CLC 311 and 312), in the riparian corridors. We regressed EQR-IBGN against the proportion of these land cover types in the corridors, and reported the slope (and its confidence interval) for that regression. The slope is significant when the confidence interval does not cover zero. The model was run for subsets of sites having predominantly agricultural or mixed (urban and agricultural) pressures in their catchments: agricultural basins had < 2% of artificial areas (CLC 1) and > 30% of cultivation (CLC 211, 221, 222, 241, and 242); mixed pressure basins had > 2% of urban areas and < 80% of cultivation.

All statistical analyses were undertaken with XLStat 2006 software (AddinSoft, France).

Table 3. Results of the PLS regression for France: national model (France-all), and Armorican, Tables Calcaires, Massif Central and Mediterranean regions. We give standardized PLS regression coefficients between EQR-IBGN and significant land cover variables (from CORINE Land Cover 2000) at both basin and riparian scales. Negative coefficients in bold characters and positive coefficients in italics. R<sup>2</sup>: coefficient of determination of the models. Only coefficients for variables that were significant at p=0.05 are given.

France		France – all (n=3662)		Armorican (n=116)		Tables Calcaires (n=378)		Massif Central (n=458)		Mediterranean (n=182)	
		R <sup>2</sup> = 18 %		R <sup>2</sup> = 20%		R <sup>2</sup> = 15 %		R <sup>2</sup> = 31 %		R <sup>2</sup> = 38 %	
CLC code	CLC name	basin	riparian	basin	riparian	basin	riparian	basin	riparian	Basin	riparian
111	Continuous urban fabric	<b>-0.09</b>		<b>-0.07</b>		<b>-0.07</b>		<b>-0.13</b>		<b>-0.12</b>	<b>-0.05</b>
112	Discontinuous urban fabric	<b>-0.16</b>	<b>-0.08</b>	<b>-0.19</b>	<b>-0.09</b>	<b>-0.18</b>	<b>-0.09</b>	<b>-0.18</b>	<b>-0.12</b>	<b>-0.21</b>	<b>-0.07</b>
121	Industrial-commercial units	<b>-0.10</b>	<b>-0.09</b>	<b>-0.12</b>		<b>-0.10</b>		<b>-0.16</b>	<b>-0.15</b>	<b>-0.16</b>	
132	Dump sites	<b>-0.04</b>									
133	Construction sites	<b>-0.04</b>									
211	Non-irrigated arable land	<b>-0.07</b>	<b>-0.05</b>	<b>-0.09</b>							
221	Vineyards	<b>-0.06</b>		<b>-0.07</b>				<b>-0.09</b>	<b>-0.08</b>		
222	Fruit trees	<b>-0.04</b>		<b>-0.04</b>	<b>-0.03</b>					<b>-0.11</b>	
231	Pastures					<i>0.06</i>	<i>0.10</i>			<i>0.10</i>	<i>0.06</i>
311	Broad-leaved forest	<i>0.05</i>	<i>0.05</i>	<i>0.11</i>	<i>0.11</i>					<i>0.12</i>	
312	Coniferous forest	<i>0.07</i>						<i>0.10</i>	<i>0.08</i>		
313	Mixed forest	<i>0.05</i>								<i>0.08</i>	

## 2.3 Results

### 2.3.1 Land cover effect at the basin and riparian scale

The coefficient of determination ( $R^2$ ) of the PLS regression models in France was lower in the lowland regions than in the mountains, with the highest value in the Mediterranean region (Table 3). There was a clear distinction between the CLC categories showing a negative relationship with the EQR-IBGN and those with a positive relationship. Artificial areas, particularly continuous urban areas, discontinuous urban areas, and industrial-commercial units had a clear and strong negative effect at both the basin and riparian scales, and at the national and regional levels. The effect of dump and construction sites appeared only at the national level. Among the agricultural areas, a distinction must be made between arable land, vineyards and fruit trees which had a negative effect, and pastures which had a positive effect. The negative effects of arable land, vineyards and fruit trees at the national level were about half those of artificial areas. Arable land had a negative effect at both the basin and riparian scales at the national level, but at the regional level it was significant only in the Armorican model. Vineyards had a clear negative effect at the national level, but this effect appeared only in the Massif Central and Armorican regions. Fruit trees had a strong negative effect in the Mediterranean region, and a weaker one in the Armorican region. The positive effect of pastures was evident only in the Tables Calcaires and Mediterranean regions, at both the basin and riparian scales. All the categories of forests and semi-natural areas exhibited positive relationships. The positive effect of forests was clear, at both the basin and riparian scales, at the national level and in all regions except the Tables Calcaires. Other semi-natural land cover categories appeared only in some models (national and Mediterranean).

The  $R^2$  of the Slovakia models was greatest at the national level, and indicated better relationships in the mountains than in the lowlands (Table 4). At the basin level, artificial surfaces and agricultural areas had a clear negative relationship with the EQR-SI; both categories had a similar effect at the national level and in the mountains, but in the lowlands the relative effect of artificial surfaces was much greater. Forests and semi-natural areas had a similar positive effect. At the riparian level, in the national model, industrial and commercial areas and arable land had a negative effect, while all forest categories had a



positive effect. However, the regional patterns were clearly different: the same CLC categories appeared with comparable weights in the Carpathians, while in the Pannonians, only the broad-leaved forest had a significant positive effect.

Table 4. Results of the PLS regression for Slovakia: national model (Slovakia-all), and Carpathians and Pannonians regions. We give standardized PLS regression coefficients between EQR-SI and significant land cover variables (from CORINE Land Cover 2000) at both basin and riparian scale. Negative coefficients in bold characters and positive coefficients in italics.  $R^2$ : coefficient of determination of the models. Only coefficients for variables that were significant at  $p=0.05$  are given.

Slovakia		<i>Slovakia – all</i>		<i>Carpathians</i>		<i>Pannonians</i>	
		<i>(n=193)</i>		<i>(n=137)</i>		<i>(n=56)</i>	
		<b><math>R^2 = 44\%</math></b>		<b><math>R^2 = 38\%</math></b>		<b><math>R^2 = 29\%</math></b>	
CLC code	CLC name	<i>basin</i>	<i>riparian</i>	<i>basin</i>	<i>riparian</i>	<i>basin</i>	<i>riparian</i>
1	Artificial surfaces	<b>-0.17</b>		<b>-0.15</b>		<b>-0.24</b>	
2	Agricultural areas	<b>-0.17</b>		<b>-0.16</b>		<b>-0.14</b>	
3	Forests and semi-natural areas	<i>0.18</i>		<i>0.17</i>		<i>0.17</i>	
121	Industrial-commercial units		<b>-0.07</b>		<b>-0.09</b>		
211	Non-irrigated arable land		<b>-0.11</b>		<b>-0.09</b>		
311	Broad-leaved forest		<i>0.08</i>		<i>0.07</i>		<i>0.14</i>
312	Coniferous forest		<i>0.12</i>		<i>0.13</i>		
313	Mixed forest		<i>0.06</i>		<i>0.07</i>		

1

Discontinuous urban areas together with industrial and commercial units had the highest negative relationships with the EQR-ASPT at the basin level in the Estonia models, with a significant riparian effect for the first category (Table 5). The regression coefficient for arable land was half that of artificial areas. Among the natural land cover categories, we observed positive relationships for mixed forest and water bodies in the basins, and coniferous forest in the riparian corridor, but natural grassland had an unexpected negative regression coefficient.

Table 5. Results of the PLS regression for Estonia. We give standardized PLS regression coefficients between EQR-ASPT and significant land cover variables (from CORINE Land Cover 2000) at both basin and riparian scale. Negative coefficients in bold characters and positive coefficients in italics.  $R^2$ : coefficient of determination of the model. Only coefficients for variables that were significant at  $p=0.05$  are given.

<i>Estonia – all</i>			
<b>Estonia</b>			
<i>(n=168)</i>			
<b><math>R^2 = 39\%</math></b>			
CLC code	CLC name	<i>basin</i>	<i>riparian</i>
112	Discontinuous urban fabric	<b>-0.24</b>	<b>-0.09</b>
121	Industrial-commercial units	<b>-0.28</b>	
211	Non-irrigated arable land	<b>-0.13</b>	
312	Coniferous forest		<i>0.08</i>
313	Mixed forest	<i>0.15</i>	
321	Natural grassland	<b>-0.10</b>	
512	Water bodies	<i>0.09</i>	

1

Models in the UK were developed with basin scale LCM indicators for N-Taxa and ASPT expressed as EQIs. As a whole, the results for both indices were very similar. The  $R^2$  was generally higher for ASPT, but the patterns and relative weights of the variables entering the models were identical (Table 6). Urban and suburban-rural developed land cover always had the greatest negative effects. Two agricultural land cover categories appeared only in the English Sedimentary model, but surprisingly as a positive factor for arable land and negative for neutral grass. In all other models, improved grassland had a positive relationship, as did acid grass in the English Uplands region. Natural and semi-natural areas, including broad-leaved and coniferous woodland together with bracken and heath, generally had positive effects in the national and regional models. Broad-leaved woodland was a notable exception, appearing as a negative factor in the English Uplands. These natural areas had no significant effects in the two regions, Cornwall and Wales and English Chalk. All the regional models differed with regard to the effects of agricultural and natural areas.

Table 6. Results of the PLS regression for UK: national model (UK-all), and English Uplands, Cornwall and Wales, English Sedimentary and English Chalk regions. We give standardized PLS regression coefficients between EQI-N-Taxa or EQI-ASPT and significant land cover variables (from Land Cover Map 2000) at basin scale. Negative coefficients in bold characters and positive coefficients in italics.  $R^2$ : coefficient of determination of the models. Only coefficients for variables that were significant at  $p=0.05$  are given.

UK		UK – all (n=4508)		English Uplands (n=639)		Cornwall and Wales (n=1077)		English Sedimentary (n=1788)		English Chalk (n=634)	
		<i>N-Taxa</i>	<i>ASPT</i>	<i>N-Taxa</i>	<i>ASPT</i>	<i>N-Taxa</i>	<i>ASPT</i>	<i>N-Taxa</i>	<i>ASPT</i>	<i>N-Taxa</i>	<i>ASPT</i>
		$R^2 = 28\%$	$R^2 = 34\%$	$R^2 = 39\%$	$R^2 = 40\%$	$R^2 = 17\%$	$R^2 = 17\%$	$R^2 = 34\%$	$R^2 = 39\%$	$R^2 = 27\%$	$R^2 = 35\%$
LCM code	LCM name										
172	Urban	<b>-0.24</b>	<b>-0.26</b>	<b>-0.22</b>	<b>-0.22</b>	<b>-0.17</b>	<b>-0.17</b>	<b>-0.21</b>	<b>-0.23</b>	<b>-0.24</b>	<b>-0.27</b>
171	Suburban-rural developed	<b>-0.28</b>	<b>-0.30</b>	<b>-0.24</b>	<b>-0.24</b>	<b>-0.26</b>	<b>-0.26</b>	<b>-0.26</b>	<b>-0.28</b>	<b>-0.28</b>	<b>-0.32</b>
4	Arable land							<i>0.15</i>	<i>0.16</i>		
61	Neutral grass							<b>-0.13</b>	<b>-0.13</b>		
51	Improved grassland	<i>0.12</i>	<i>0.13</i>	<i>0.08</i>	<i>0.08</i>	<i>0.16</i>	<i>0.16</i>			<i>0.18</i>	<i>0.20</i>
81	Acid grass			<i>0.09</i>	<i>0.09</i>						
11	Broad-leaved woodland	<i>0.05</i>	<i>0.06</i>	<b>-0.11</b>	<b>-0.11</b>			<i>0.07</i>	<i>0.08</i>		
21	Coniferous woodland	<i>0.08</i>	<i>0.09</i>	<i>0.12</i>	<i>0.12</i>			<i>0.07</i>	<i>0.08</i>		
91	Bracken	<i>0.05</i>	<i>0.05</i>	<i>0.10</i>	<i>0.10</i>						
10	Dwarf shrub heath			<i>0.09</i>	<i>0.09</i>						

### 2.3.2 Urban vs. agricultural effect

As a whole, the negative effect of artificial land cover was predominant in all the models, but some relationships for the agricultural land cover categories were unexpected. The correlation between the occurrence of urban and agricultural land cover within the basins was different in the different regions tested (Table 7), with a subsequent confounding effect in the models.

Table 7. Test of the effect of agricultural land cover independent of its correlation with the percentage of urban land cover within the basins for each region of France, Slovakia, Estonia and United Kingdom. We give Pearson's correlation between urban and agricultural land cover in the basins; linear regression between EQR (or EQI) and urban land cover; multiple linear regression (MLR) between EQR (or EQI) and urban + agricultural land cover; difference in  $R^2$  of the two regressions and extra sum of square F-test p-value between both models.

Country	Region	Pearson's correlation	Linear	Multiple Linear	Difference	Extra sum of square F-test
		urban / agricultural land cover	regression EQR = f(urban)	regression EQR = f(urban+agricultural)		
			$R^2$	$R^2$	$dR^2$	p-value
France	Armorican	0.035	14.7%	14.7%	0.0%	0.86
France	Tables Calcaires	-0.076	12.7%	14.2%	1.5%	<b>0.01</b>
France	Massif Central	-0.092	21.0%	22.0%	1.0%	<b>0.009</b>
France	Mediterranean	-0.1	33.0%	33.6%	0.6%	0.54
Slovakia	Carpathians	0.56	23.0%	31.7%	8.7%	<b>&lt;0.001</b>
Slovakia	Pannonians	0.53	32.0%	32.5%	0.5%	0.67
Estonia	Estonia	0.06	26.0%	30.5%	4.5%	<b>0.001</b>
UK	Cornwall and Wales	0.013	16.0%	16.4%	0.4%	<b>0.003</b>
UK	English Uplands	-0.03	37.0%	38.0%	1.0%	<b>0.004</b>
UK	English Sedimentary	-0.54	37.7%	37.7%	0.0%	0.88
UK	English Chalk	-0.74	29.0%	33.9%	4.9%	<b>&lt;0.001</b>

In all the regions, the regressions with urban areas had  $R^2$  between 12.7% and 37.7% (Table 7); the highest  $R^2$  were encountered in the Mediterranean, Pannonians, English Uplands and English Sedimentary regions. In the second regression, with urban + agricultural

land cover as predictors, the R<sup>2</sup> were only slightly higher. The addition of agricultural land cover as predictor increased the R<sup>2</sup> noticeably, by between 4.5% and 8.7%, in only in 3 of the 11 regions tested (Carpathians, Estonia and English Chalk).

### **2.3.3 Test of a protective riparian effect**

In all the models that included the riparian scale (France, Slovakia and Estonia), the presence of pastures or forests in the river corridors had a significant positive effect on invertebrate indices (Tables 3-5). This could be interpreted as a protective effect of these land cover categories against pressures coming from arable land or artificial areas at the basin scale (Moore & Palmer, 2005). To validate this hypothesis, we tested the effects that could be related to the presence in the riparian corridor of pastures and forests, alone or combined, in three French regions (Armorican, Tables Calcaires and Massif Central). In each region, we selected two datasets that corresponded at the basin scale to a dominant agricultural pressure or to mixed pressures (agricultural and urban). In each dataset, we regressed the EQR-IBGN against the proportion of the supposed protective land cover in the riparian corridors. All the significant relationships indicated a positive effect on the EQR-IBGN (Table 8).

The presence of pastures alone in the riparian corridors had a significant effect in the mixed pressure basins of the Massif Central and Tables Calcaires, but not in the Armorican region (Table 8). Broad-leaved forests alone had a significant effect in the mixed pressure basins of the Armorican region (Table 8), but not in the Tables Calcaires or the Massif Central. Broad-leaved forests also had a significant effect in the agricultural basins in the Massif Central, and coniferous forests had a strong effect in the mixed pressure basins in the Massif Central. The combination of pastures and forests in the riparian corridors (Table 8) had a significant effect in the agricultural basins of the Tables Calcaires; in the mixed pressure basins, the effect was significant in both the Tables Calcaires and Massif Central regions.

Table 8 Test of the protective effect of riparian pastures in basins subject to predominantly "agricultural" or "mixed" (agricultural and urban) pressures, in three regions of France (Armorican, Tables Calcaires, Massif Central). We give slopes and confidence intervals of linear regressions between EQR-IBGN and the percentage of pastures, forest or combined land cover in the riparian corridors. Slopes significantly different from zero are in bold characters.

	<i>HER</i>	<i>Basins</i>	<i>Slope (95% confidence interval)</i>
<b>Pastures CLC 231</b>	Armorican	agricultural	-0.074 (-0.26 – 0.11)
	Armorican	mixed	-0.076 (-0.2 – 0.05)
	Tables calcaires	agricultural	0.1 (-0.03 – 0.23)
	Tables calcaires	mixed	<b>0.1</b> (0.02– 0.19)
	Massif central	agricultural	-0.09 (-0.25 – 0.06)
	Massif central	mixed	<b>0.15</b> (0.01 – 0.29)
<b>Broad-leaves forest CLC 311</b>	Armorican	agricultural	0.12 (-0.08 – 0.3)
	Armorican	mixed	<b>0.3</b> (0.08– 0.47)
	Tables calcaires	agricultural	0.16 (-0.03 – 0.48)
	Tables calcaires	mixed	0.003 (-0.09 – 0.09)
	Massif Central	agricultural	<b>0.23</b> (0.05 – 0.41)
	Massif Central	mixed	0.13 (-0.04 – 0.3)
<b>Coniferous forest CLC 312</b>	Massif Central	agricultural	0.57 (-0.37 – 1.52)
	Massif Central	mixed	<b>0.49</b> (0.08 – 0.9)
<b>Combined CLC 231+311</b>	Armorican	agricultural	0.03 (-0.21 – 0.28)
	Armorican	mixed	-0.04 (-0.09 – 0.16)
	Tables calcaires	agricultural	<b>0.2</b> (0.06 – 0.35)
	Tables calcaires	mixed	<b>0.11</b> (0.02 – 0.2)
	Massif Central	agricultural	0.11(-0.07 – 0.29)
	Massif Central	mixed	<b>0.23</b> (0.09 – 0.37)
<b>Combined CLC 231+312</b>	Massif Central	agricultural	-0.08 (-0.24 – 0.08)
	Massif Central	mixed	<b>0.19</b> (0.05 – 0.32)

## 3 Discussion

### 3.1 Invertebrate indices responses

Despite their differences, the invertebrate indices used in the four countries generally gave consistent results and responded with similar patterns to the land cover indicators.

In all countries, some models accounted for R<sup>2</sup> around 40%. Such R<sup>2</sup> values reflect good relationships considering 1) the distance, in terms of ecological processes, between the land cover indicators evaluated from satellite imagery and the biological response evaluated through a synthetic index, 2) the noise associated with monitoring data collected by different laboratories and sometimes covering different years, and 3) the fact that only a part of the variability of the predictors is retained in the PLS regression. Consequently, lower R<sup>2</sup> can be interpreted as weaker relationships in the models. In this case, two explanations are possible: either the land cover indicators do not represent the actual pressures acting in the region well, or the invertebrate index does not react in the same way to the pressures. In France and Slovakia, the regional models had higher R<sup>2</sup> in mountains than in lowlands. However, in the lowlands of Estonia, the model was good, and in UK, only the Cornwall and Wales model had a low R<sup>2</sup>. Thus, both explanations are plausible but the hypothesis that the French and Slovakian invertebrate indices are less sensitive in the lowland rivers cannot be ignored. Whether these different responses reflect a true difference in the sensitivity of ecosystems to pressures remains an open question. For this reason, we shall interpret only the relative weights of the factors in the different models.

### 3.2 Hierarchy of impacts: urbanization vs. agriculture

The relative impact of agriculture and urbanization on aquatic biota is not well established. Various studies have demonstrated negative relationships between biological indices and the percentage of agricultural land in the catchment or gradients of agricultural intensity (Roth, Allan & Erickson, 1996; Walser & Bart, 1999; Cuffney et al., 2000). These impacts depend on the different categories of production: many authors designate intensive agriculture as a cause of degradation in stream habitats (Allan & Johnson, 1997; Allan, 2004). Moreover, physical alterations caused by agriculture can generate biological impacts over

decades (Harding et al., 1998). Other authors have found less negative effects (Meador & Goldstein, 2003), or even positive relationships between agricultural land cover and invertebrate diversity (Moore & Palmer, 2005).

The impacts of urbanisation are mainly related to the direct effects of point sources of pollution. However, many other pressures are generated by urban areas, such as canalization, untreated storm-water and altered hydrological regimes (Cormier et al., 2000), and these cause both hydro-morphological and biological impacts (Paul & Meyer, 2001; Stepenuck, Crunkilton & Wang, 2002; Roy et al., 2003; Wang & Kanehl, 2003). When comparing the impacts of different forms of land cover, some authors found agriculture to be the most damaging activity, while others found the strongest negative correlations with urban areas. The hierarchy of relationships between these driving forces may vary according to the human or ecoregional context, although some of the differences observed could have been caused by the methodology (bioindicators, models, scales).

Our results show that in 12 of the 14 PLS models, urban or artificial land cover (sensu CORINE nomenclature) appeared to be the most important negative factor on invertebrate indices at the national and regional scales. Artificial and agricultural areas had a similar weight only in Slovakia at the national level and in the Carpathians at the regional level. For the national models, the effect of the urban areas was about twice that of the influential agricultural land cover categories in France and Estonia, and in UK agriculture appeared to have no negative effect. This picture was validated and even strengthened by the direct linear regressions (Table 7), correcting a possible confounding effect introduced by the correlation between the percentage of urban and agricultural land cover within the basins. For instance, in the Carpathians, the apparently stronger negative effect of agriculture could be caused by a positive correlation between urban and agricultural land cover: cultivated basins are also more impacted by urbanization. The very low increase in R<sup>2</sup> when adding the agricultural land cover as a predictor in the linear regressions demonstrated that, in all cases, the effect of urban areas was much stronger than the effect of arable land. In the Carpathians, for instance, the urban effect was 3 times greater than the effect of agricultural land cover, and in the English lowlands the effect of arable land was very low.



Urban areas had a strong negative impact in all the regions analysed. On the other hand, forests and semi-natural land cover (such as moors and heath land, bracken, and transitional scrub) appeared to be positive influences (with one exception in UK). However, the effect of agricultural land cover, including arable land, pastures and some semi-natural grassland, was much more variable. Seen from the largest scale, the national level, agricultural areas, and particularly arable lands, had a negative effect, except in UK. In general, the intensity of this impact remained moderate, but it is a potential concern in much of Europe. However, the regional patterns were very different and deserve a more detailed analysis to identify the relative impacts of different agricultural land covers better, especially between arable land and pastures.

### **3.3 Regional variability of agricultural impacts**

Because of the “covariation of anthropogenic and natural landscape features” (Allan, 2004), natural constraints (e.g. terrain slope, rainfall regime) could determine the sensitivity of the ecosystems, as well as the land use practices, leading to different regional responses. In testing the effectiveness of two landscape classifications (watershed vs. agro-ecoregion), Dovciak & Perry (2002) observed that local habitat conditions, which strongly influence invertebrate assemblages, were determined by regional-scale landscape factors. In order to understand ecological relationships better and to help decision-making, a regional approach could be useful to identify regional river ‘pathologies’ that should be managed differently, even within a country.

In Slovakia, the available data did not allow the effects of arable land and pastures to be separated at the basin scale. However, both PLS models and linear regressions demonstrated that the impact of agricultural areas was greater in the mountains than in the lowlands. In Estonia, both arable lands and natural grasslands had a similar negative effect. The latter are used as pastures, usually with the clear-cutting of the whole riparian corridor that leads to intensive growth of macrophytes in low-gradient rivers, with accumulation of mud that could explain the changes in the fauna.

In UK, we observed three different regional patterns. In the English Uplands, the categories of open land used as pastures had a positive effect; this was expected for bracken

(*Pteridium aquilinum*) supporting a very low intensity sheep-farming, but the more intensive agriculture for dairy and livestock generally associated with improved grassland did not produce a negative effect. The unexpected negative PLS coefficient for broad-leaved woodland was explained by a positive correlation between deciduous forests and urbanisation (including historical industrialisation) in the lowest parts of the Pennine valleys; thus, this relationship cannot be interpreted as an impact. In the English Sedimentary PLS model, forests had the expected positive effect, but neutral grass had a negative coefficient. Despite a weak correlation between neutral grass and urbanisation, this result could correspond to a true ecological impact, as water pollution from farm wastes was recognized as a significant problem in the 1990s (Skinner et al., 1997). The positive effect of arable land in this region was surprising, explained by a strong negative correlation between the proportion of urban areas and arable land within the basins: so, the most agricultural basins are much less impacted by urban pressures. This was corroborated by the null effect of arable land in the direct linear regression (Table 7). Similarly, Wang et al. (2000) observed a positive correlation between agricultural land use and fish biotic integrity in Wisconsin, where high levels of agriculture were associated with low level of urbanization. Therefore, we cannot demonstrate any impact on invertebrate indices related to the percentage of arable land in the basins of this English Sedimentary region.

In France also, the regional models showed different patterns in the impacts of agricultural land cover. The high impact of vineyards was demonstrated only in the Massif Central and in the Armorican regions, because of some localized vineyards, but not in the Mediterranean region even though vineyards are encountered in half of its basins. Here, the natural characteristics of the HERs could probably explain the difference in the regional responses. The siliceous bedrock and the flow regimes could lead to a greater transfer of sediment and pesticides in the Armorican and Massif Central vineyards than in the Mediterranean ones, and so to a stronger biological impact. Pastures had a positive effect in the Mediterranean region with low-intensity sheep farming, and in the Tables Calcaires, but pastures did not appear in the PLS models of the two regions most dedicated to cattle rearing: the Massif Central and Armorican. This can be interpreted as a decrease in the positive effect of pastures as the cattle stocking intensity increased.

More surprisingly, arable land appeared to have no effect in the French regional PLS models, even in the Tables Calcaires region where most basins are largely dedicated to crops. This was confirmed by the very small effect of arable land in the linear regression (Table 7). In fact, a wide range of WFD ecological status conditions, from high to bad, can be encountered in basins with a high proportion of arable land (up to 80%) and low urbanization, but on average, the EQR-IBGN values were between the good and moderate status. Thus, in France, the results seemed contradictory: arable land had a negative effect at the national level, but this effect disappeared at the regional scale. This could be interpreted in the following way: in general, cultivated lowland regions have more degraded ecological conditions, but in these regions, the impact of agricultural activities is not directly linked to the percentage of arable land in the basins. In other words, the land cover alone is only a general descriptor of the probability of agricultural impacts. In cultivated areas, local factors such as natural characteristics (hill slopes, soil properties), the intensity of agricultural practices, and physical alterations to the landscape generated by agricultural development determine the actual impact on running water ecosystems (Meador & Goldstein, 2003). Among these factors, land use in the riparian areas is expected to exert a strong influence.

### **3.4 Basin vs. riparian scales**

Although many papers address the issue of scaling processes and relationships, the influence of pressures acting at the riparian corridor and basin scales needs better understanding (Allan, 2004). Gergel et al. (2002) concluded from a literature review that "a variety of investigators have tried to determine the spatial extent, or distance from the water body, over which landscape patterns influence water quality or aquatic biota, yet this question remains unresolved". Some studies identify landscape-level factors as the main variables for predicting biological indicators, while others find no relationships beyond the riparian or local scale (Garcia et al., 2006). Meador & Goldstein (2003) emphasized "the universal importance of riparian zones to the maintenance and restoration of diverse fish communities in streams", and the restoration of river corridors is seen as a key action to improve ecological conditions (Naiman, Décamps & McClain, 2005; Palmer et al., 2005; Walsh et al., 2007). Recently, Johnson et al. (2007) proposed a model taking into account the

spatial pattern of land cover within catchments to explain the biological response and develop riparian restoration strategies.

The actual influence of riparian areas on river biota, and particularly the invertebrate indices, has been evaluated mainly in meso-scale studies (Lammert & Allan, 1999; Sponseller, Benfield & Valett, 2001; Stewart et al., 2001). The PLS models used in France, Slovakia and Estonia demonstrated, at large scale, a very clear negative relationship between artificial land cover in the riparian corridors and invertebrate indices. In all the models (except in the Pannonians) discontinuous urban fabric and industrial-commercial units appeared as negative factors. Thus, the causes of impacts from artificial land cover in river corridors must be sought in physical alterations and diffuse contamination generated by the presence of urban and industrial areas close to the rivers, as well as in the effects of domestic pollution. Arable land cover in the riparian corridor was an important negative factor at the national level in France and in Slovakia, but not in Estonia. At the regional level, this category appeared only in the Carpathian model. Vineyards had negative impacts in the Massif Central and fruit trees in the Armorican region. In general, the presence of cultivation in the riparian corridors correlated with a biological impairment in the rivers, but the actual impact depended on local conditions. Among these, the presence of narrow vegetated buffer strips, not detected by CORINE Land Cover, could alter the relationship.

Another important result was the demonstration of the widespread positive effects of forests and pastures in the riparian corridor, shown in all the models at the national and regional scales. This result could indicate a direct positive influence on aquatic invertebrates, but also to a buffer effect that mitigated the negative impacts of arable land or artificial areas outside the riparian corridor. In this respect, forests and pastures must be considered separately.

The direct influence of riparian forest on invertebrate community structure is widely recognized (Maridet et al., 1998; Naiman et al., 2005), but it is not often evaluated in terms of ecological status. In our results, the effect of riparian forests, when significant, was always associated with an increase in the values of invertebrate indices. The buffer effect was tested in three French regions, in basins with either a dominant agricultural pressure or mixed urban and agricultural pressures. The presence of broad-leaved forests in the riparian

areas, alone or combined with pastures, had a clear positive effect in five of the six datasets tested: the mixed basins of the Armorican region, and both agricultural and mixed basins of the Tables Calcaires and Massif Central (Table 8). In most cases, the increase in EQR-IBGN could be sufficient to shift from the WFD moderate class to good status. In the Armorican and Massif Central regions, the positive effect of broad-leaved or coniferous riparian forests was more pronounced in the mixed pressure basins than in the agricultural ones, suggesting that it was effective against urban pressures. This is consistent with the results of Moore & Palmer (2005), but contrary to the findings of Walsh et al. (2007), who stated that in Australian urban catchments, "riparian revegetation is unlikely to have an effect on indicators of stream biological integrity". However, the effect of riparian forests was also subject to some regional variability, as it appeared weaker in the Tables Calcaires. Nevertheless, in general, riparian forests can be very effective for mitigating the impacts from both agricultural and urban land cover at the basin level.

Evaluating the buffering effect of pastures is more complex, because they can be either a buffer intercepting fine sediments, phosphorus and pesticides coming from cropland, or a pressure generating organic and nutrient loads, and erosion from cattle trampling when they support intensive livestock. The direct effect of riparian pasture on river invertebrate communities is probably related to changes in the trophic structure caused by increased light availability, as in other open agricultural areas. In the Armorican region, dedicated to intensive animal husbandry, the effect of pastures alone in the riparian corridor was not apparent (Table 8). The same occurred in the agricultural basins of the Massif Central, where cattle rearing, although less intensive, is also a dominant activity. But in the Massif Central, riparian pasture had a positive effect in mixed pressure basins. These facts suggest that the actual use of the riparian pastures, particularly the intensity of the livestock that they support, must be taken into account when evaluating their ecological effect. Conversely, in the Tables Calcaires region mainly dedicated to crops, the effect of pastures was positive in the mixed basins, and the combination of pastures and woodland was very effective against pressures coming from arable land (Table 8).

These results support the conclusion that the appropriate management of riparian corridors, mixing wooded areas and extensive pastures, in agricultural as well as in urban

catchments, “may have significant impact on ecosystem health” (Naiman et al., 2005). While the basin land cover is unlikely to change in the near future, riparian corridors are more manageable areas and their restoration can be an effective way to improve the ecological status of European rivers.

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# Chapitre 4 - Peut-on prédire l'état biologique des cours d'eau ? Une approche multipressions reliant trois indices biotiques à la physico-chimie, à l'hydromorphologie et à l'occupation du sol.

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## **1 Synthèse de l'article**

Une des principales attentes sociétale vis-à-vis de l'écologie concerne sa capacité à déterminer l'état des milieux et surtout à diagnostiquer les causes d'altérations de ce dernier lorsqu'il est dégradé. Comment définit-on cet état ? Où le mesure-t-on ? A partir de quels paramètres biologiques, chimiques ou hydromorphologiques ? Une forte ambition sociétale et politique est apparue ces dernières années et s'est focalisée sur l'état des eaux depuis le Clean Water Act aux Etats-unis en 1972 jusqu'à la Directive Cadre Européenne sur l'eau (2000). L'objectif est d'enrayer la dégradation continue des milieux et d'enclencher une nouvelle ère de gestion ; le bon état écologique des eaux, souvent décrié (voir réf citées supra), étant institué en objectif à atteindre. Pour répondre à ces questions, les états ont mis en place des programmes de surveillance de l'état des eaux de surface dans le but d'évaluer de manière cohérente et transparente l'état des eaux continentales.

Mais aucune des réglementations existantes n'impose un suivi systématique de chaque masse d'eau dans le cadre des réseaux de surveillance. Par exemple, le réseau de contrôle et de surveillance de la France couvre approximativement 25% des masses d'eau du territoire avec 1500 sites. Cela laisse par conséquent 4500 masses d'eau pour lesquelles la classification de l'état écologique devra se faire sans données de suivi disponibles. Et comme nous l'avons vu dans l'introduction, cette situation n'est pas unique en Europe car les limitations techniques ainsi que les raisons économiques font que les Etats membres ont choisi de ne pas suivre l'ensemble de leurs masses d'eau (Kristensen and Christiansen 2012).

Pour faire face à ces questionnements, il est nécessaire d'utiliser des méthodes fiables et reproductibles pour prédire l'état écologique des masses d'eau non suivies de manière objective et homogène pour l'ensemble des masses d'eau d'un Etat. Cependant, l'étendue spatiale de la plupart des recherches en écologie est relativement limitée, ce qui implique de fait le besoin de développer des outils capables d'extrapoler les connaissances à des échelles plus larges ou à de nouveaux territoires (Miller et al. 2004).

Des travaux de modélisation d'indicateurs d'état sont apparus au début des années 2000 dans les pays qui ont basé l'évaluation de l'état écologique sur le système RIVPACS comme au Royaume-Uni (Clarke et al. 2003; Davy-Bowker and Furse 2006; Wright et al.



1998). Les conditions de référence ont été modélisées en fonction de descripteurs environnementaux naturels. Ce système a été repris par de nombreux pays : MEDPACS en Espagne (Poquet et al. 2009), Mondego model au Portugal (Feio and Poquet 2011), PERLA en République Tchèque (Kokeš et al. 2006), SWEPAC DRI en Suède (Johnson and Sandin 2001) et AusRIVPACS en Australie (Smith et al. 1999).

Par contre, très peu de travaux de recherche se sont intéressés au développement d'outils prédictifs permettant de faire une évaluation des masses d'eau non suivies : Donohue et al. (2006) et Kristensen et al. (2012) en Europe et Brown et al. (2012), Carlisle et al. (2009), Maloney et al. (2009) et Waite et al. (2010) aux USA.

Ces premiers outils prédictifs montrent qu'il est possible de prédire efficacement l'état écologique à partir de variables de pression. Mais ils tiennent peu compte des échelles spatiales multiples qui structurent à la fois les pressions et le fonctionnement longitudinal des cours d'eau. En effet, un cours d'eau est organisé de manière hiérarchique selon plusieurs échelles imbriquées : bassin versant, tronçon, faciès et meso/microhabitat (Frissell et al. 1986). De plus, l'altération de l'état écologique est la résultante de plusieurs pressions qui agissent simultanément mais avec des importances relatives éventuellement variables en fonction du temps et un impact également variable en fonction des conditions environnementales saisonnières : rejets ponctuels (plus ou moins épisodiques), pollutions diffuses, altération des flux solides et liquides, structures artificielles et rupture de connectivité (Borchardt and Richter 2003). D'après le concept DPSIR (Forces motrices, pressions, état, impact et réponse) (EEA 2003; Kristensen 2004), les activités humaines (agriculture, urbanisation) génèrent des pressions combinées (rejets chimiques, altérations physiques) et constituent les forces motrices qui altèrent les composantes abiotiques de l'écosystème (physico-chimie, hydromorphologie). Ces altérations affectent ensuite les communautés biologiques et par conséquent l'état écologique.

En parallèle, un certain nombre de travaux se sont intéressés à la compréhension des liens entre l'état biologique et les pressions dans ce contexte multi-échelles. Occupation du sol, hydromorphologie et physico-chimie ont été reliées tour à tour à des indices biotiques basés sur les communautés de macrophytes (Feld 2013), de diatomées (Dahm et al. 2013), de poissons (Feld 2013; Marzin et al. 2012; Marzin et al. 2013) et de macro-invertébrés (Feld

2013; Marzin et al. 2013; Sponseller et al. 2001; Sundermann et al. 2013; Wasson et al. 2010).

De manière plus générale, ces études ont montré les liens existant entre chacune de ces pressions et les indices biologiques. Certains ont aussi montré que les liens entre pressions et indices biologiques étaient conditionnés par l'échelle de prise en compte de la pression. Ainsi, bassin versant, tronçon hydromorphologique et corridor rivulaire sont des échelles qui structurent fortement l'effet des pressions sur l'état écologique.

En développant les modèles de ce chapitre, nous avons pour objectif de prendre en compte ces connaissances sur l'organisation des échelles spatiales et sur les liens entre pressions et état écologique dans la détermination de l'état écologique des masses d'eau non suivies (non-monitored).

Dans ce but, nous avons construit un corpus de modèles capables d'expliquer d'une part la variabilité des indices biologiques utilisés dans les réseaux et de prédire d'autre part l'état écologique des masses d'eau françaises métropolitaines non suivies par le réseau de contrôle et de surveillance (RCS). Les indices basés sur les macro-invertébrés benthiques, diatomées et poissons ont été utilisés pour déterminer l'état écologique de 1200 sites du RCS répartis de manière homogène sur l'ensemble du territoire français. Les pressions prises en compte pour expliquer et prédire l'état écologique couvrent trois échelles spatiales : bassin versant, tronçon hydromorphologique, station. Le jeu de données prédictives recouvre trois types de pressions : les pressions d'occupation du sol, les pressions hydromorphologiques et les pressions physico-chimiques mesurées respectivement à l'échelle du bassin versant, du tronçon hydromorphologique et du site.

Cela permet de répondre aux questions suivantes :

- Quelles sont les pressions qui influencent le plus les outils de mesure de l'état écologique ? Dans quel sens ? Avec quelle intensité ?
- Comment prédire l'état écologique des masses d'eau non suivies dans le cadre des réseaux ?

Les modèles explicatifs ont montré l'importance majeure des paramètres physico-chimiques. En effet, Le phosphore total, les nitrites, l'ammonium et dans une moindre mesure la DBO5 sont les facteurs de pressions qui sont les plus influents dans les modèles, quel que soit l'indice biotique pris en compte.

Ces résultats concordent avec les études réalisées ailleurs en Europe. La concentration en nutriments est un facteur qui influence négativement les indices biotiques qu'ils soient basés sur les macroinvertébrés benthiques, les diatomées ou les poissons (Dahm et al. 2013; Donohue et al. 2006; Johnson and Hering 2009).

Nous avons aussi pu voir que les trois indices utilisés (I2M2, IBD2007 et IPR+) montrent des profils de réponse similaires avec un impact prépondérant des paramètres physico-chimiques (nutriments et matière organique) mesurés à l'échelle du site suivis en deuxième position par l'impact de l'occupation du sol à l'échelle du bassin versant et en troisième position, par les facteurs d'altération hydromorphologiques avec un effet beaucoup plus difficile à mettre en évidence. Cependant, nous avons pu montrer que les réponses des différents indices variaient sensiblement, ce qui correspond à ce que l'on peut retrouver chez d'autres auteurs (Dahm et al. 2013; Johnson and Hering 2009). Par exemple, l'indice basé sur les macro-invertébrés (I2M2) est plus particulièrement sensible aux paramètres signant un ralentissement de l'écoulement et une modification des faciès menant à un style fluvial plus lentique. L'indice diatomées (IBD2007) est plus sensible aux paramètres signant une modification de l'amplitude des crues. L'indice poissons (IPR+) est lui influencé par les paramètres signant un risque de colmatage du substrat par les sédiments fins et par les paramètres indiquant une rupture de la continuité écologique. Ceci soutient le fait que ces indices sont complémentaires et doivent être utilisés ensemble afin de pouvoir mener une analyse complète des pressions qui s'exercent sur le cours d'eau.

Globalement, les 3 modèles prédictifs développés montrent de bonnes performances. Ces résultats sont cohérents avec les résultats obtenus avec les modèles explicatifs. Les modèles les plus performants correspondent en effet aux indices qui répondent à la plus grande diversité de stressseurs avec dans l'ordre : I2M2 puis IBD2007 et enfin IPR+.

Ce type de modèles est d'un intérêt majeur pour les gestionnaires. Ils permettent en effet à partir d'un jeu de données de pressions de prédire avec un bon taux de réussite l'état de masses d'eau pour lesquelles ils ne disposent pas de données de suivi biologique.

La méthode des arbres d'inférence conditionnelle utilisée ici est un atout pour le transfert aux gestionnaires, elle permet en effet de communiquer directement aux gestionnaires le schéma de l'arbre de décision qu'ils peuvent ensuite utiliser directement sans compétence particulière en modélisation.

## **2 Can we predict biological condition of stream ecosystems? A multi-stressors approach linking three biological indices to physico-chemistry, hydromorphology and land use.**

### **Abstract**

We built a corpus of models capable of explaining the variability of the biological indices used in the French surveillance monitoring network and also predict the ecological status of non-monitored water bodies. Benthic macroinvertebrates, diatoms and fish indices have been used to determine the ecological status of 1100 sites of the monitoring network distributed homogeneously over national territory.

The pressures taken into account to explain and predict ecological status cover three spatial scales: catchment, reach, site. The set of predictive data cover three types of pressure: land use pressure, hydromorphological pressure and physico-chemical pressure measured at catchment, reach and site scale, respectively.

We showed that the parameters characterising the load of nutrients and organic matter had a predominant effect on the three biological compartments, and that land use variables played an integrating role of the different pressures acting on rivers and explained a major part of their degradation. On the contrary, we also showed that it was more difficult to characterise the role of the hydromorphological descriptors measured at the intermediate scale of the reach due to the difficulty of characterising the links between scales.

The three predictive models developed demonstrated good performances to evaluate biological condition and are of great interest for managers as it permits using a set of pressure data to successively predict the status of water bodies for which biological monitoring data are unavailable.

## **2.1 Introduction**

One of the main expectations of society regarding ecology is the capacity to determine the status of environments and above all to diagnose the causes of the degradations that affect them. How can this status be defined? What should be measured? What biological, chemical and hydromorphological parameters should be taken into account? Over recent years, strong societal and political ambitions have developed, manifested for example in attention being given to the status of water through the Clean Water Act in the United States in 1972 and the European Framework Directive on Water in 2000. The objective is to ensure the good ecological condition of water. To respond to these questions, governments have set up programs for the monitoring of water status in order to establish a coherent and comprehensive overview of water status within each river basin district.

However, none of the existing regulations require systematic monitoring of each water body in the framework of systematic observation networks. For example, the monitoring network in France covers approximately 25% of the water bodies of the territory with 1,500 sites. This leaves 4,500 water bodies for which the classification of ecological status must be done without monitoring data being available. What is more, this situation is by no means unique in Europe since both technical constraints and economic reasons have led the member states to opt for only partial monitoring of their water bodies (Kristensen and Christiansen, 2012).

In order to answer the above questions, reliable and reproducible methods are required to predict the ecological status of non-monitored water bodies objectively and uniformly for all the water bodies of a country. However, the spatial coverage of most ecological research is relatively limited, which implies the need to develop tools capable of extrapolating knowledge to larger scales and new territories (Miller et al., 2004).

Works to model status indicators emerged at the beginning of the 2000s in countries that base the evaluation of ecological status according to the RIVPACS system, such as the United Kingdom (Wright et al. 1998, Clarke et al. 2003, Davy-Bowker et al. 2006). The reference conditions have been modelled as a function of natural environmental descriptors. Many other countries have adopted this system: MEDPACS in Spain (Poquet et al., 2009), the

Mondego model in Portugal (Feio and Poquet, 2011), PERLA in the Czech Republic (Kokeš et al., 2006), SWEPAC DRI in Sweden (Johnson and Sandin, 2001) and AusRIVPACS in Australia (Smith et al., 1999).

On the contrary, very few research has focused on the development of predictive tools that permit evaluating non monitored water bodies. Donohue et al. (2006) built a model to predict ecological status based on populations of benthic macroinvertebrates using catchment land use data and water chemical quality indicators of 797 sites in Ireland. In Denmark, Kristensen et al. (2012) developed a model capable of predicting the presence of fish assemblages as a function of catchment land use and chemical and morphological data measured at site level on 335 sites.

This type of research has also been conducted outside the European Union. In the USA in particular, where several authors have published works on the development of models to predict ecological status (again based on benthic macroinvertebrate populations) on the basis of catchment land use (Brown et al., 2012; Carlisle et al., 2009; Maloney et al., 2009; Waite et al., 2010). Brown et al. (2012) developed a predictive model using the Boosted Regression Tree (BRT) method to predict ecological status based on macroinvertebrate populations using catchment land use data, and river corridor and environmental descriptors. They worked on a dataset consisting of 159 sites distributed in Southern Coastal California. Maloney et al. (2009) compared five types of predictive method (classification and regression trees, random forest, conditional random forest and ordinal logistic regression) to predict ecological status based on the macroinvertebrates of 1561 sites of Chesapeake Bay (Maryland). The predictive variables were composed of land use variables and environmental descriptors. Carlisle et al. (2009) used Random Forests to predict the biological status of 920 sites distributed over the whole of the USA, again using macroinvertebrates as status indicators. The predictive variables were also based on land use data and other general environmental descriptors. Waite et al. (2010) used multiple linear regression to model several metrics based on macroinvertebrates, measuring the ecological status (biological condition) for 299 sites in Oregon and Southern Coastal California. The predictive variables were land use variables measured on the scale of the catchment, the river corridor and environmental descriptors.

These initial predictive tools demonstrate that it is possible to predict ecological status efficiently on the basis of pressure variables. Nonetheless, they take little account of the multiple spatial scales that structure both the pressure and longitudinal functioning of rivers. Indeed, a river is organised hierarchically as a function of several interlinked scales: catchment, reach, mesohabitat and microhabitat (Allan, 2004; Allan and Johnson, 1997; Frissell et al., 1986; Poff et al., 1997). In addition, the degradation of ecological status results from several types of pressure: punctual discharges (more or less episodic), diffused pollutions, changes of solid and liquid flows, artificial structures and breaks in connectivity (Borchardt and Richter, 2003). According to the DPSIR concept (driving forces, pressures, status, impact and response) (Kristensen, 2004), human activities (agriculture, urbanisation) generate combined pressures (chemical discharges, physical changes) and constitute driving forces that change the abiotic components of the ecosystem (physico-chemistry, hydromorphology). These changes then affect biological communities and thus ecological status.

In this framework of analysis, some works have focused on understanding the links between biological condition and pressures in a multiscale context. Land use, hydromorphology and physico-chemistry have been linked one after the other to biotic indices based on macrophyte communities (Feld, 2013), diatoms (Dahm et al., 2013), fish (Feld, 2013; Marzin et al., 2012; Marzin et al., 2013) and macro-invertebrates (Feld, 2013; Marzin et al., 2012; Sponseller et al., 2001; Sundermann et al., 2013; Wasson et al., 2010).

Feld (2013) showed that not all biological groups respond in the same way to pressures and that lotic fauna was more correlated than lotic flora to the land use indicators measured on the scale of a catchment such as riparian corridor, with a more marked correlation for mountain rivers than for plain rivers. In line with Sponseller et al. (2001) and Wasson et al. (2010), it has also been shown that the riparian corridor plays a major protective role for the ecological functioning of a river and that taking into account specific regional geographical characteristics, in particular geographical entities such as eco-regions, is essential for good river management. Dahm et al. (2013) and Sundermann et al. (2013) used more categories of pressure variables at different spatial scales to explain ecological status and showed that



physico-chemistry and land use had a greater effect than hydromorphology on the status indicators of all the biological groups.

More generally, these studies have highlighted the links that exist between each of these pressures and biological indices. Some have shown that the links between pressures and biological indices are influenced by the scale at which the pressure is taken into account. Thus a catchment, reach and riparian corridor are scales that have a considerable structural effect on the pressures affecting ecological status (Marzin et al., 2013; Wasson et al., 2010).

Our objective is to take into account this knowledge on the organisation of spatial scales and on the links between pressures and ecological status in determining the ecological status of non-monitored water bodies.

To this end, we built a corpus of models capable of explaining the variability of the biological indices used in the survey network and also predict the ecological status of non-monitored water bodies in France. Benthic macroinvertebrates, diatoms and fish indices have been used to determine the ecological status of 1100 sites of the monitoring network distributed homogeneously over French territory. The pressures taken into account to explain and predict ecological status cover three spatial scales: catchment, reach, site. The set of predictive data cover three types of pressure: land use pressure, hydromorphological pressure and physico-chemical pressure measured at catchment, reach and site scale, respectively.

This permits answering the following questions:

- What are the pressures with the greatest influence on ecological status measurement tools? In what way? With what intensity?
- How can the ecological status of non-monitored water bodies be predicted?

## 2.2 Materials and methods

### 2.2.1 Biological data

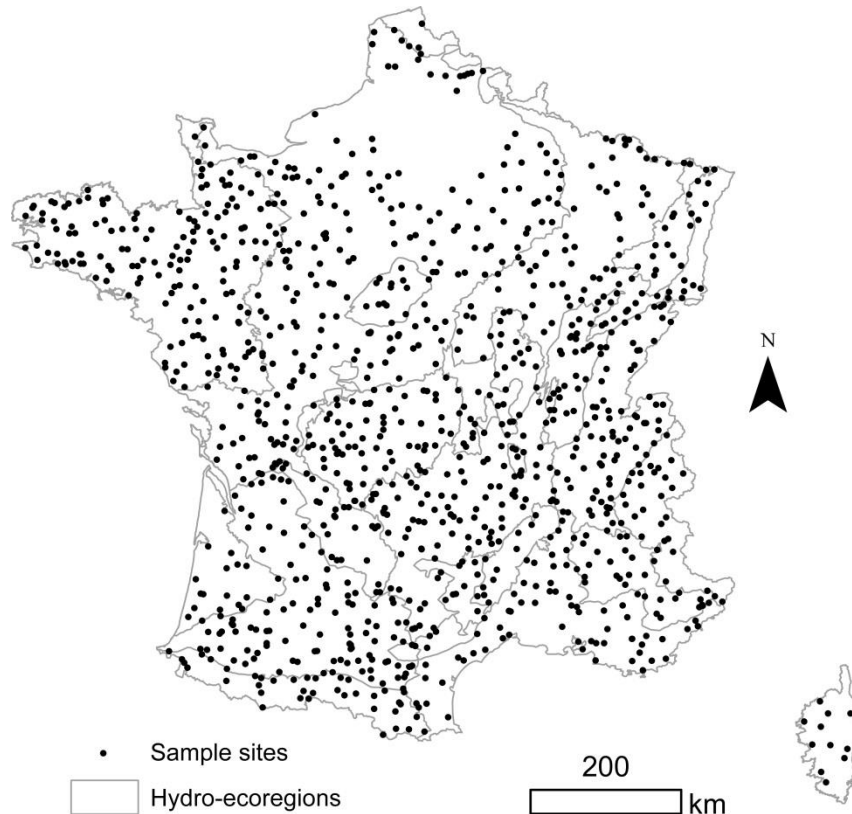


Figure 1. Map of the 1100 selected sites of the monitoring network of France. Limits of hydro-ecoregions are represented by grey lines.

We used the data of 1100 sites from the French monitoring network for the period 2008-2009 (Fig. 1). These sites are distributed in 22 hydro-ecoregions and cover every size of river. The benthic macro-invertebrate data were collected in conformity with a standardised common protocol (AFNOR, 2009). The biological index calculated on the basis of these fauna lists is the I2M2 (Mondy et al., 2012). The diatom data were collected according to a standardised protocol (AFNOR, 2007) used to calculate the biological index IBD 2007. The fish data were collected by electric fishing according to a standardised protocol (AFNOR, 2004). Sampling was performed once a year for each protocol.

These three indices are expressed in EQR (Ecological Quality Ratio - Annex V, 1.4.1, WFD 2000). The values of the indices were calculated as the ratio with the reference value of the

index in the type of site where it was measured. This made it possible to neglect the typological differences between the sites. The mean of the index values for 2008 and 2009 was then calculated and each of these three indices was then converted into a ranking of "Good" or "Bad" by using the limit values of good status used by French managers.

## **2.2.2 Pressure data**

We evaluated the pressures to which the monitoring sites are subjected by using the spatial framework of longitudinal organisation originally described by Frissell et al. (1986). To do this we chose three main spatial scales: catchment, reach and site. We crossed these scales with three types of driving force and pressure: land use, hydromorphological alterations and water physico-chemistry. The pressures are listed and described in table 1.

### **2.2.2.1 Land use data**

The layer of geographical information used is CORINE Land Cover (CLC) 2006 (Büttner and Kosztra, 2007). It covers the whole territory and is based on a standard nomenclature ranked in 3 levels and 44 categories of land use distributed according to 5 main types of environment (artificial environments, agricultural environments, forests and semi-natural environments, wetlands and bodies of water).

On the basis of the catchment basin delimited beforehand by a digital terrain model, we calculated for each site the percentage of the surface of different categories of land use using CORINE Land Cover. We then grouped them into four categories.

### **2.2.2.2 Hydromorphological data**

The data used here corresponds to descriptors of pressure subjected on the hydromorphological functioning of a river. Two main types of descriptor were used. At catchment scale (hydrographic area), we used data to characterise possible disturbance of natural solid and liquid flows. At reach scale, we used descriptors of hydromorphological changes. These variables were then assigned to each of the monitoring sites.

Table 1. Description of pressure variables.

<i>Scale</i>	<i>Variable</i>	<i>range of average values (min-max)</i>	<i>Source</i>
Catchment	urbanisation of catchment: urbanised zones, industrial and commercial zones, transport networks, mines, dumps and construction sites, artificial green spaces, non agricultural spaces	3.3 (0-58.7)	Corine Land Cover (CLC) 2000 <a href="http://terrestrial.eionet.europa.eu/CLC2000">http://terrestrial.eionet.europa.eu/CLC2000</a>
Catchment	intensive agriculture: arable land, permanent crops, orchards, vineyards, olive orchards, annual crops linked to permanent crops, crop and field systems	35.5(0-100)	Corine Land Cover (CLC) 2000 <a href="http://terrestrial.eionet.europa.eu/CLC2000">http://terrestrial.eionet.europa.eu/CLC2000</a>
Catchment	non-intensive agriculture: meadows, land mainly devoted to agriculture with considerable presence of natural vegetation, agro-forestry land	20.3(0-89.9)	Corine Land Cover (CLC) 2000 <a href="http://terrestrial.eionet.europa.eu/CLC2000">http://terrestrial.eionet.europa.eu/CLC2000</a>
Catchment	non anthropised space: forests and semi-natural environments, wetlands, ponds	40.8(0-100)	Corine Land Cover (CLC) 2000 <a href="http://terrestrial.eionet.europa.eu/CLC2000">http://terrestrial.eionet.europa.eu/CLC2000</a>
Catchment	Erosion	0.6(0-3.4)	alea erosion des sols INRA (Montier et al. 1998)
Catchment	Drainage ratio	0.03(0-1)	Recensement general agricole (RGA) 2000
Catchment	Irrigation ratio	0.03(0-0.47)	Recensement general agricole (RGA) 2000
Reach	Rate of roads/railways near the riverbed	7.8(0-262)	BD Topo IGN
Reach	Rate of roads/railways in the flood plain	111(0-3087)	BD Topo IGN
Reach	Rate of dykes in the riverbed	5(0-393)	BD Topo IGN
Reach	Rate of dikes in the flood plain	17(0-764)	BD Topo IGN
Reach	Rate of crossings	0.8(0-8.5)	BD Topo IGN
Reach	Rate of trees on banks "screen of trees"	77(0-100)	BD Topo IGN
Reach	Rate of vegetation on banks "riparian vegetation"	59(0-100)	BD Topo IGN
Reach	Rate of vegetation in riverbed	48(0-100)	BD Topo IGN
Reach	Rate of "artificial" type land use near riverbed	9.8(0-100)	BD Topo IGN
Reach	Rate of straightness	27.8(0-100)	BD Topo IGN
Reach	Rate of ponds in riverbed	0.7(0-33)	BD Topo IGN
Reach	Overwidth	41.4(0-15742)	BD Topo IGN
Site	Suspended matter (mg/l)	20.5(1.5-1347)	RCS 2008-2009
Site	Dissolved oxygen (mg/l)	10.1(6.1-13)	RCS 2008-2009
Site	BOD5 at 20°C (mg/l)	1.6(0.5-4.7)	RCS 2008-2009
Site	Ammonium (mg/l)	0.08(0-0.47)	RCS 2008-2009
Site	Nitrites (mg/l)	0.006(0.01-0.98)	RCS 2008-2009
Site	Nitrates (mg/l)	13.8(0.3-70.4)	RCS 2008-2009
Site	Total phosphorous (mg/l)	0.08(0.01-0.48)	RCS 2008-2009

### 2.2.2.3 Physico-chemical data

The sites on which we worked belong to the national monitoring network. We were able to obtain the data necessary in order to characterise physico-chemical. These parameters were measured every month. We chose as pressure indicator the mean value of concentrations measured during the period 2008-2009.

## 2.2.3 Modelling methods

### 2.2.3.1 Modelling plan

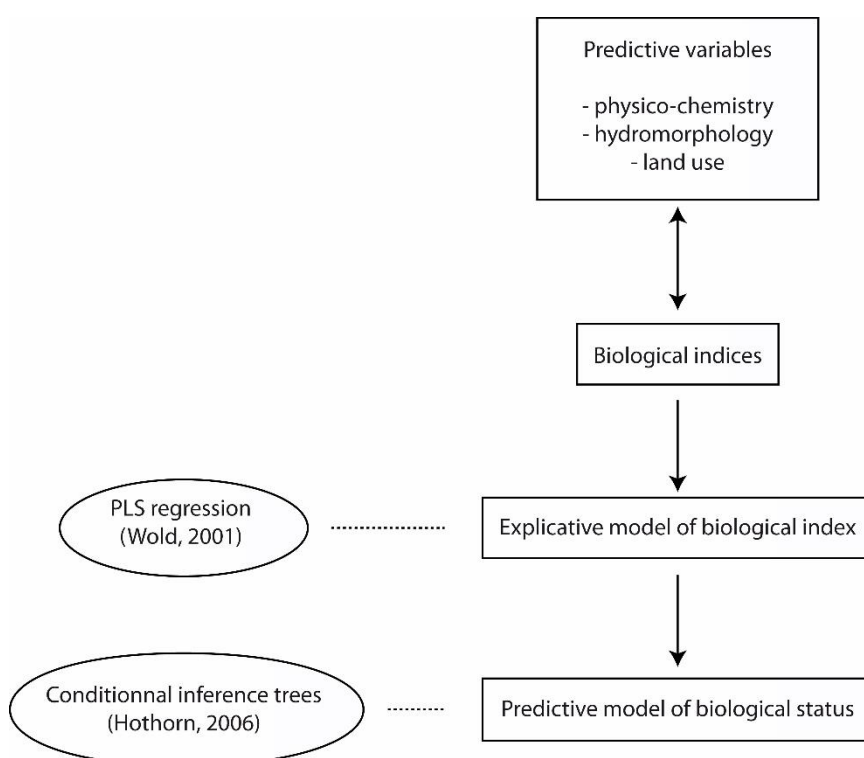


Figure 2. Modelling plan

An initial explanatory model was developed using the PLS regression method (Wold et al., 2001) to link all the pressures selected to the macroinvertebrates (I2M2), diatoms (IBD2007) and fish (IPR+) biological indices. This model was used to quantify the effect of all the predictors on each index and quantify the relative effect of each predictor on the same index. The significant variables were then used as input variables for a second model aimed at predicting the ecological status. This predictive model was developed by using the

conditional inference tree method (Hothorn et al., 2006) to efficiently predict the ecological status (good or bad) determined on the basis of the three biological indices. Using a set of pressures evaluated for the non-monitored water bodies, this model permits extrapolating their probable ecological status.

This modelling plan is summarised in Fig. 2.

### **2.2.3.2 Explicative method**

PLS (Wold et al., 2001) is an adaptation of classical linear model. Its objective is to mitigate the main fault of the latter: the instability of the regression coefficients due to the collinearity of the predictors. Indeed, when collinearity becomes high in a set of predictive variables, the estimations of the regression coefficients fluctuate from one sample to the other (Cramer et al., 1988). PLS regression permits obtaining interpretable prediction coefficients even in the case of strong correlation between the predictors. The final equation is interpreted as a linear regression equation, the coefficients clearly reflect the effect of a predictor relative to others and the  $R^2$  is a good estimator of the model's efficiency. The significance of the coefficients is then validated using the jack-knife method (Efron and Gong, 1983; Martens and Martens, 2000). Wasson et al. (2010) demonstrated the interest of using this method in ecology when using predictors which, as in this case, are strongly correlated with each other. We illustrated this multicollinearity by calculating the discrimination coefficient ( $R^2$ ) of each variable with all the other predictors.

### **2.2.3.3 Predictive modelling method**

The modelling method used is that of conditional inference trees (Hothorn et al., 2006). In comparison to other decision tree methods (such as CART (Breiman, 1984) which uses an inequality criterion), the CIT method proposes a statistical test (on the basis of permutation tests) to select the most pertinent variable at each node and to define a tree growth stopping criterion. This method can be broken down into three steps: (1) null hypothesis of independence between each explanatory variable and the dependent variable is tested by a permutation test; (2) if the null hypothesis of independence is rejected, the group of samples is divided into two sub-groups defined by their values for the explanatory variable ensuring

the best discrimination in (1); (3) the process is reiterated on each of the subgroups formed until the null hypothesis of independence cannot be rejected.

Table 2: Multicollinearity statistics of predictive variables. The value of the multicollinearity coefficient expresses the percentage of variance of a variable explained by all the other variables.

<i>variable</i>	<i>multicolinéarité (R<sup>2</sup>)</i>	<i>Scale</i>
erosion	0.391	Catchment
irrigation	0.208	Catchment
drainage	0.256	Catchment
artificial land cover	1.000	Catchment
high impact agriculture land cover	1.000	Catchment
low impact agriculture land cover	1.000	Catchment
natural land cover	1.000	Catchment
dams density	0.363	Reach
roads in riverbed	0.463	Reach
roads in floodplain	0.763	Reach
dykes in riverbed	0.790	Reach
dykes in floodplain	0.851	Reach
urbanisation of 100m buffer	0.476	Reach
overwidth	0.048	Reach
pounds in floodplain	0.143	Reach
vegetation in 10m buffer	0.743	Reach
vegetation in 30m buffer	0.881	Reach
vegetation in floodplain	0.769	Reach
straightness	0.201	Reach
suspended matter	0.082	Site
dissolved oxygen	0.388	Site
BOD	0.453	Site
ammonium	0.615	Site
nitrites	0.804	Site
nitrates	0.724	Site
total phosphorous	0.662	Site

The model was built on the basis of a learning dataset of 800 sites and then tested on a test dataset of 300 sites.

The evaluation of the model's efficiency is done using the confusion matrix between the observed values and predicted values of ecological status. Using this matrix, we calculated the rate of poor classifications of the model as the first indicator of its efficiency. We then calculated the sensitivity (proportion of true positives among the sites with bad status) and

the specificity (proportion of true negatives among the sites with good status), in order to evaluate the model's capacity to predict situations of both good and bad ecological status. Lastly, we calculated the area under the curve (AUC) (Hanley and McNeil, 1982) which is a good indicator of the model's predictive efficiency (Swets, 1988). These indicators were calculated for both the predictions made using the learning dataset and the test dataset.

## **2.3 Results**

### **2.3.1 Explanatory models of biological indices**

The analysis of multicollinearity coefficients (table 2) shows that the pressure variables are strongly intercorrelated with the coefficients having values up to 1. This fully justifies the use of the PLS regression to build explanatory models of indices I2M2, IBD2007 and IPR+.

The results of the PLS (Fig. 3) show that the set of pressure variables explains 41% of the variability of I2M2, 26% of the variability of IBD2007 and 24% of the variability of IPR+.

The profiles of the responses to the variables are similar for the three indices. In terms of intensity of response, the physico-chemical variables have the highest coefficients, followed by the land use variables and, lastly, the hydromorphological variables. The variables with a negative effect on these three indices are: concentrations in nutrients and organic matter, urbanisation and the proportion of intensive agriculture in the catchment.

The variables with a positive effect on these three indices are: the concentration in dissolved oxygen and the proportion of vegetation in the 10 and 30 m buffers of reaches.

The other variables present more specific responses for each index. For I2M2, the model shows the negative effect of the straightness of the course of the river, irrigation, dam density, the presence of dykes in both the riverbed and in the flood plain of the river, the presence of roads in the flood plain and suspended matter. The same model shows a positive effect of low impact agriculture. For IBD 2007, the model shows a negative effect of low impact agriculture, drainage, the presence of ponds in the riverbed, the presence of dykes in the riverbeds and flood plains of rivers and roads in the riverbed. This model shows



Variables	I2M2			IBD			IPR+					
	coefficient	Normalised PLS regression coefficients	p-value	coefficient	Normalised PLS regression coefficients	p-value	coefficient	Normalised PLS regression coefficients	p-value			
total phosphorous	-0.02462		0.00000	-0.01966		0.00000	-0.01651		0.00001			
nitrates	-0.00140		0.64546	-0.00130		0.09317	-0.00648		0.00001			
nitrites	-0.03173		0.00000	-0.01572		0.00000	-0.01634		0.00000			
ammonium	-0.03069		0.00000	-0.01769		0.00000	-0.01426		0.00002			
BOD	-0.00690		0.02156	-0.01249		0.00002	-0.01324		0.00000			
dissolved oxygen	0.01412		0.00001	0.01010		0.00000	0.01580		0.00006			
suspended matter	-0.00637		0.00037	-0.00021		0.77436	-0.00271		0.11451			
natural land cover	0.00242		0.05064	0.01351		0.00000	0.01216		0.00000			
low impact agriculture land cover	0.01719		0.00002	-0.01236		0.00024	0.00366		0.00745			
high impact agriculture land cover	-0.01019		0.00021	-0.00521		0.00036	-0.01355		0.00000			
artificial land cover	-0.02377		0.00003	-0.01166		0.00000	-0.01055		0.00134			
drainage	-0.00111		0.69572	-0.00662		0.00007	-0.00768		0.00018			
irrigation	-0.01183		0.00725	-0.00221		0.00650	-0.00721		0.00000			
erosion	-0.00205		0.45775	-0.00001		0.99253	-0.00384		0.03669			
straightness	-0.01817		0.00042	0.00021		0.87218	-0.00656		0.00245			
vegetation in floodplain	0.00238		0.36530	0.00966		0.00001	0.00549		0.00002			
vegetation in 30m buffer	0.00388		0.15682	0.00643		0.00225	0.00709		0.00010			
vegetation in 10m buffer	0.00737		0.02154	0.00441		0.02344	0.00574		0.00159			
ponds in floodplain	-0.00265		0.37405	-0.00772		0.00750	-0.00385		0.00137			
overwidth	-0.00147		0.71351	-0.00359		0.10467	-0.00022		0.91031			
urbanisation of 100m buffer	-0.01634		0.00003	-0.00442		0.04600	-0.00205		0.05932			
dykes in floodplain	-0.01092		0.00090	-0.00307		0.08741	-0.00186		0.12930			
dykes in riverbed	-0.00877		0.00285	-0.00446		0.00542	-0.00234		0.16697			
roads in floodplain	-0.00999		0.00007	-0.00219		0.18547	0.00005		0.95338			
roads in riverbed	-0.00551		0.08920	-0.00344		0.00595	-0.00067		0.48495			
dams density	-0.01396		0.00043	-0.00001		0.99732	-0.00249		0.04774			
		Total deviance explained by model = 41%				Total deviance explained by model = 26%				Total deviance explained by model = 24%		

Figure 3 : Results of regression models for indices I2M2, IBD2007 and IPR+. The normalised regression coefficients and the p-value of the jack-knife test are presented for each model. Each histogram represents the values of the normalised coefficients: the variables that have a negative effect on the biological index are shown in dark grey; the variables that have positive effect on the biological index are shown in light grey; that variables that have a non-significant effect are shown in white.

a positive effect of the percentage of natural spaces in the catchment. For IPR+, the model shows the negative effect of drainage, irrigation, straightness, the presence of ponds in the riverbed, erosion and dam density. This model shows the positive effect of the percentage of natural spaces in the catchment.

These results can be summarised in the following way: (1) the three indices respond to the variables characterizing nutrients, organic matter, land use of the catchment and hydromorphological pressures; (2) the profile of the responses of these three indices is similar but particularities appear with differentiated responses mainly for hydromorphology.

### **2.3.2 Predictive models of biological status**

We selected for each of the models described previously the variables with a p-value lower than 0.05. These variables were used to build the predictive models.

The decision tree predicting the status determined on the basis of index I2M2 (Fig. 4) was built using the following variables: nitrites, artificial land cover, irrigation, total phosphorous, dissolved oxygen and dam density.

The predictive model of status I2M2 shows a low error rate of classification (19% for the learning dataset and 26% for the test dataset). It has a good predictive capacity (AUC=0.86 for the learning dataset and AUC= 0.78 for the test dataset) and a satisfactory capacity to predict situations of good status (specificity of 0.92 for the learning dataset and 0.86 for the test dataset). Its capacity to predict situations of bad status is less efficient with relatively poor sensitivity (0.57 for the learning dataset and 0.16 for the test dataset).

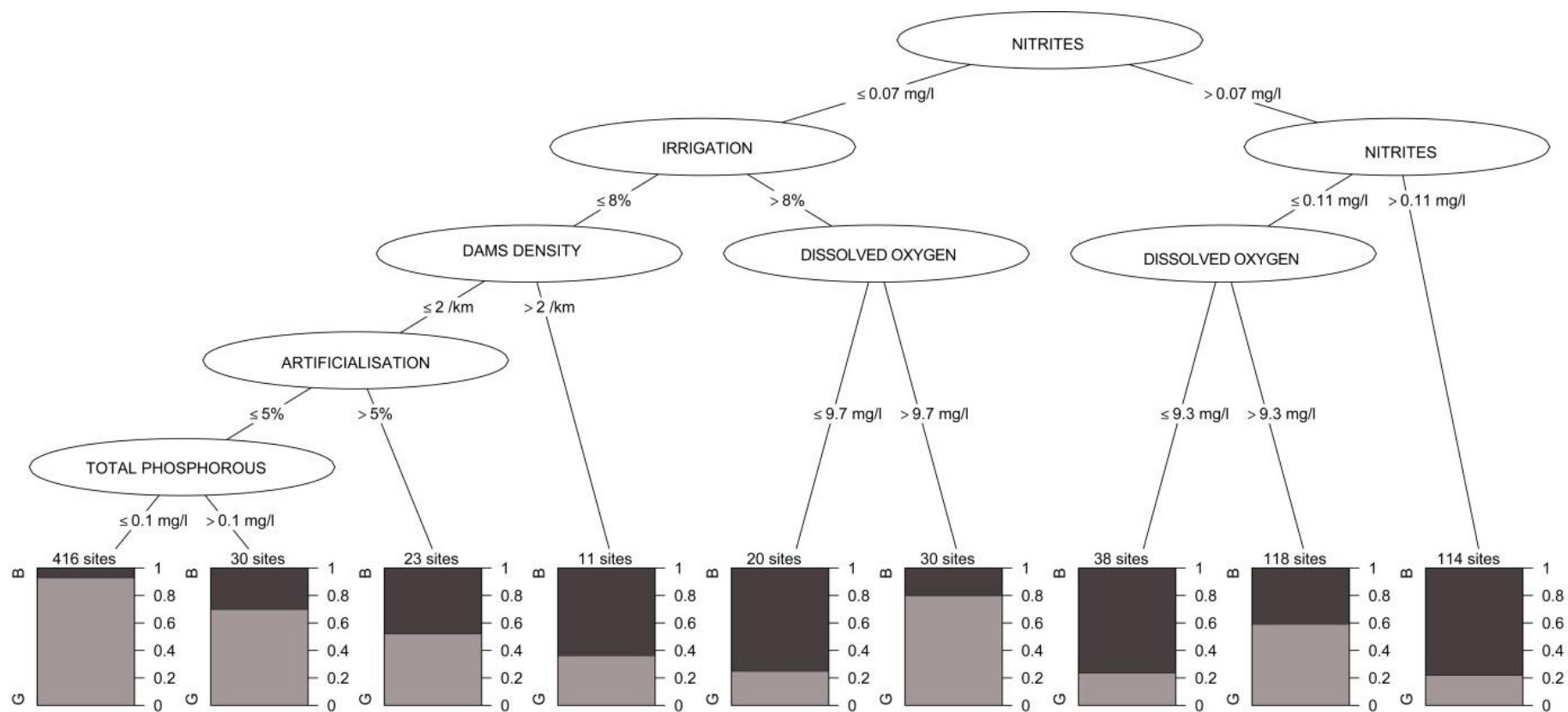


Figure 4. Conditional inference tree of macroinvertebrate index I2M2. This flow chart describes the decision tree that permits determining the proportion of stations with good and bad status in the learning dataset on the basis of a succession of threshold values for the variables selected. This value is expressed by the probability of having a good or bad status. B=bad status. G=good status.

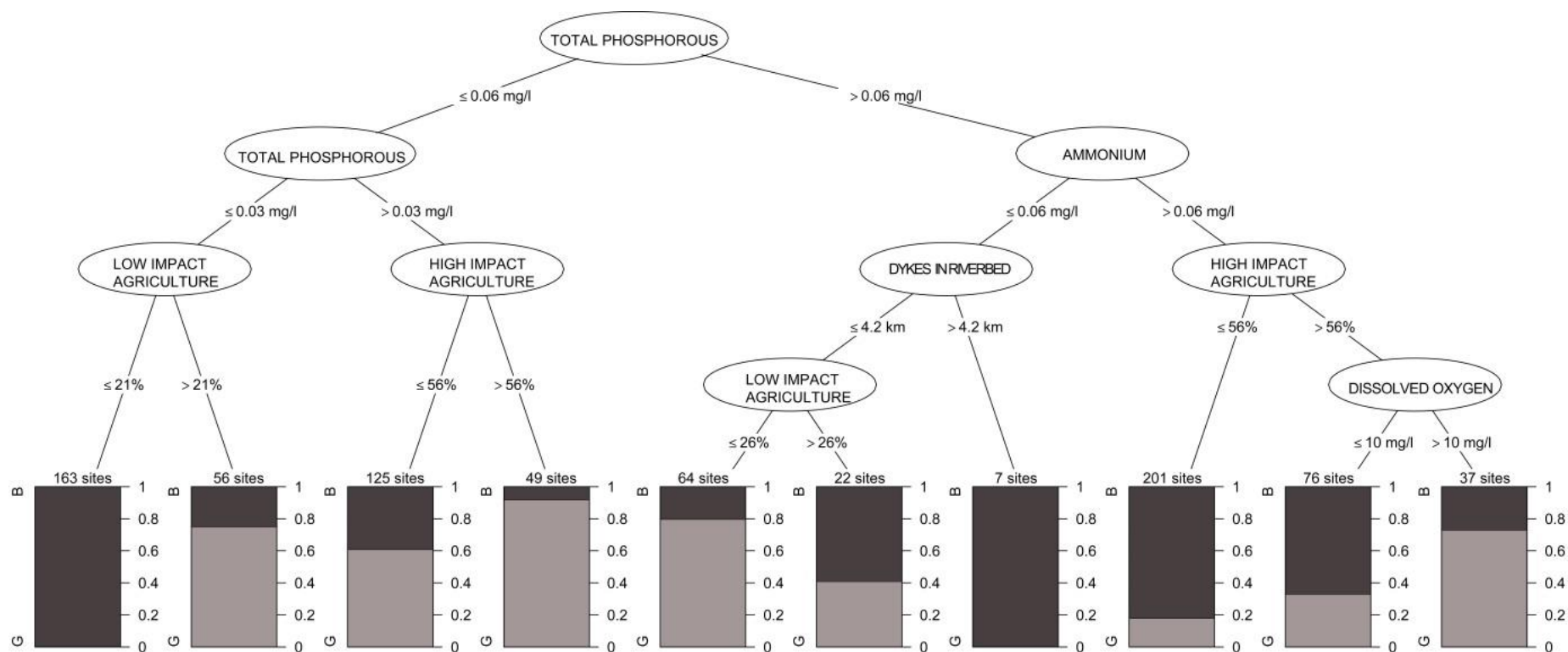


Figure 5. Conditional inference tree of diatom index IBD2007. This flow chart describes the decision tree that permits determining the proportion of stations with good and bad status in the learning dataset on the basis of a succession of threshold values for the variables selected. This value is expressed by the probability of having a good or bad status. B=bad status. G=good status.

The decision tree predicting the status determined on the basis of index IBD2007 (Fig. 5) was built using the following variables: total phosphorous, ammonium, dykes in the riverbed, intensive agriculture, low impact agriculture and dissolved oxygen.

The predictive model of status IBD 2007 showed a low error rate of classification (21% for the learning dataset and 26% for the test dataset). It had a good predictive capacity (AUC=0.85 for the learning dataset and AUC= 0.80 for the test dataset) and a satisfactory capacity to predict situations of good status (specificity of 0.84 for the learning dataset and 0.83 for the test dataset). Its capacity to predict situations of bad status was slightly less good with relatively poor sensitivity (0.70 for the learning dataset and 0.61 for the test dataset).

The decision tree predicting status determined on the basis of index IPR+ (Fig. 6) was built using the following variables: nitrites, irrigation, dissolved oxygen, low impact agriculture, ammonium and BOD5.

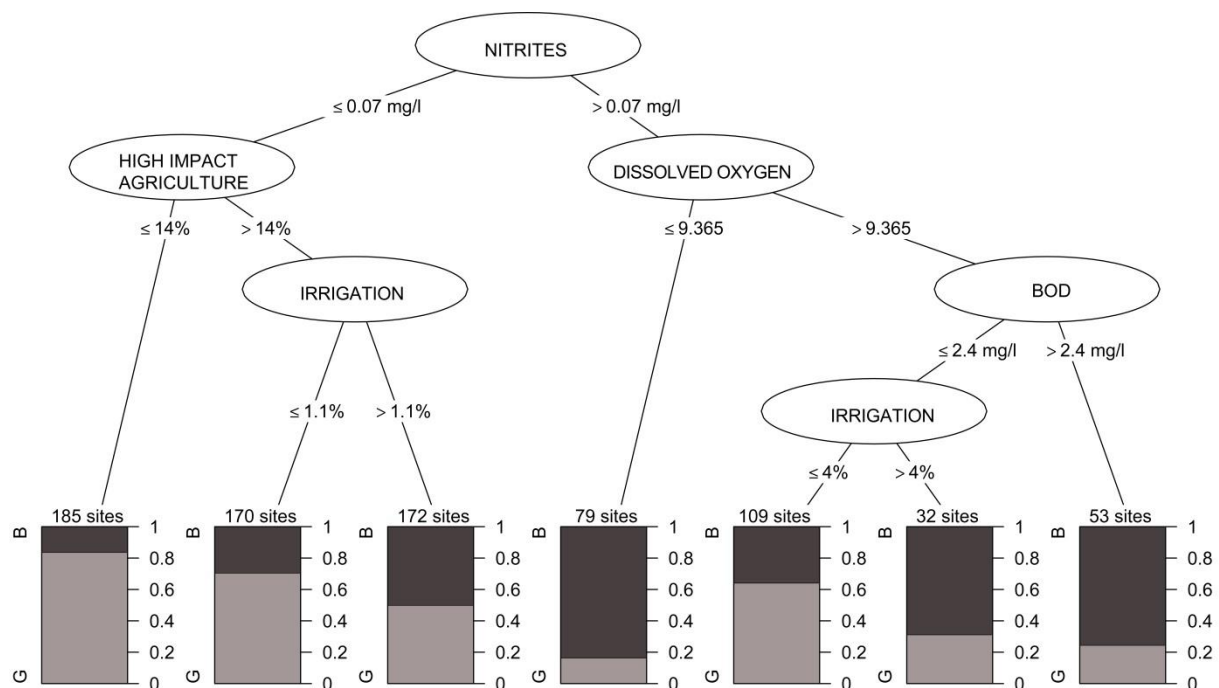


Figure 6. Conditional inference tree of fish index IPR+. This flow chart describes the decision tree that permits determining the proportion of stations with good and bad status in the learning dataset on the basis of a succession of threshold values for the variables selected. This value is expressed by the probability of having a good or bad status. B=bad status. G=good status.

The predictive model of status IPR+ presented a higher error rate of classification (30% for the learning dataset and for the test dataset). Its predictive capacity was average (AUC=0.74 for the learning dataset and for the test dataset) and more efficient for predicting situations of good status (specificity of 0.73 for the learning dataset and 0.76 for the test dataset) than the situations of bad status (sensitivity of 0.64 for the learning dataset and 0.61 for the test dataset).

## **2.4 Discussion**

The aim of this study was above all to explain and predict the ecological status measured on the basis of macroinvertebrate, diatom and fish biological quality elements using pressure factors measured at every scale of a river catchment.

### **2.4.1 Importance of physico-chemistry in a context of reducing pollution**

The explanatory models highlighted the major importance of physico-chemical parameters. Indeed, total phosphorous, nitrites, ammonia and to a lesser extent BOD5 are pressure factors that had the highest impact in the models whatever the biotic index taken into account.

These results agree with the studies performed elsewhere in Europe. The concentration in nutrients is a factor with a negative influence on biotic indices whether they are based on benthic macroinvertebrates, diatoms or fishes (Dahm et al., 2013; Donohue et al., 2006; Johnson and Hering, 2009).

Studies that attempt to discover how aquatic organisms respond specifically to the presence of organic matter or eutrophication remain rare, especially in the context of multiple pressures (Floury et al., 2013). Although certain direct toxic effects of nitrogen are known (Camargo et al., 2005; Charvet et al., 1998; Lecerf et al., 2006; Mondy and Usseglio-Polatera, 2014), it appears that the effects of nutrients and eutrophication on benthic fauna occur more through indirect processes (Friberg et al., 2010). The depletion of dissolved oxygen is the most probable indirect factor, due to the increase in consumption caused by algal and microbial proliferations linked to excess nutrients (Biggs et al., 2000). This effect is accentuated still further by oxygen consumption linked to the degradation of organic matter

during senescence. The nutritive enrichment of rivers leads to the degradation of trophic resources and a subsequent loss of biodiversity, with assemblages composed of taxa more resistant to pollution (Allan, 2004; Dahl et al., 2004; Smith et al., 1999).

The importance of these physico-chemical variables as a predominant impact factor on the value of biotic indices whatever the biological compartment considered is surprising in a context of reduced emissions of organic matter and nutrients. Indeed, since the 1990s, improvements in water treatment processes linked to stricter regulations on the use of phosphates has considerably reduced punctual inputs of phosphorous and, to a lesser degree, nitrogen in the hydrosystem (Billen et al., 2007; Larroudé et al., 2013). However, diffuse sources of contributions of nitrogen from agricultural soils and contaminated aquifers are more than ever present due to the absence of appropriate agro-environmental measures (Billen et al., 2007). Eutrophication persists in the network and is sufficient to have an impact on aquatic communities.

#### **2.4.2 Role of hydromorphology**

The role of the hydromorphological variables used was to take into account changes of non natural origin that could be correlated clearly to a degradation of ecological status. Thus different developments and uses of the flood plain (agriculture, urbanisation, transport) and the riverbed (transport, energy, tourism) whose effects resulted in a change of hydromorphological structures and processes were taken into account.

Biotic indices respond in different ways to changes in solid and liquid flows. Agricultural drainage causes changes in habitat and nutrient cycling resulting from stronger floods and the contribution of fine sediment (Blann et al., 2009). Irrigation results in severer low water levels or levels kept low by dam. Both should affect macroinvertebrates, diatoms and fishes. This point suggests that these pressures variables do not reflect precisely the field alterations.

Regarding the change of morphological processes, synergy is again apparent between the different indices. I2M2 is influenced negatively by variables expressing a change of sinuosity, change of bank erosion, the slowing of flows and a change in the succession of facies (straightening, urbanisation of the 100m buffer, dykes and roads in the riverbed and

flood plain, dam density). This can be understood insofar as they are factors that will act either on the diversity of the meso-scale habitat pattern, especially in riparian zones (straightening, slower flow rate, dykes, roads), or on the dissemination process (change of successions of facies, change of bank erosion). The diatom index responded with sensitivity to the same factors but much less obviously (very low regression coefficients). Lastly, the fish index responded only to the change in sinuosity and to dam density indicating specific sensitivity to the homogenisation of the flow facies and to the break in ecological continuity.

Lastly, these three indices exhibited a positive response to the presence of riparian forest in the river corridor. Although riparian vegetation plays a complex role (shade, input of organic matter, habitat and bank stabilisation, and an effect on bank erosion) (Naiman et al., 2010; Shields et al., 2003), it is above all evidence of a certain level of river quality.

Considered synthetically, factors of hydromorphological change come in third position after physico-chemical factors and land use. In her study, Dahm et al. (2013) concluded that the effect of hydromorphological pressures was measurable but their impacts on the biological metrics she had used were relatively low. This points towards the use of modern and efficient indicators such as I2M2 (Mondy et al., 2012) that can take more detailed account of the impact of different hydromorphological stress factors.

### **2.4.3 Land use remains a compromise representing an “average effect”**

Different human activities (urbanisation, agriculture, industry, transport, etc.) generate pressures on hydrosystems. At site level, these pressures result in physico-chemical and morphological changes. Ecological quality at site level is strongly influenced by land use within the surrounding valley at multiple scales. (Allan, 2004).

Both artificial and agricultural land use is a major impact factor for biology (Allan, 2004; Clapcott et al., 2012; Dahm et al., 2013; Marzin et al., 2012; Marzin et al., 2013) and this result stands out clearly in our models.

Anthropisation occurring on artificial land concerns land use for urbanisation, industry and transport infrastructures. Although spatially fairly limited (50% of catchments have less than 2% of artificial surface, often situated near the river), artificial land use has the greatest impact in the models after nutrients (Allan, 2004; Paul and Meyer, 2001). On the one hand,



this variable is the direct reflection of the presence of punctual sources of pollution and, on the other, of sources of diffuse pollution and hydromorphological changes.

The presence of strongly anthropised land use has three main types of consequence on rivers (Allan, 2004; Effenberger et al., 2006; Paul and Meyer, 2001; Stepenuck et al., 2002): (1) increased runoff and thus erosion and diffuse pollution; (2) chemical pollution; and (3) a change in the hydromorphological functioning of the river (elimination of riparian vegetation, straightening and change of width).

Intensive agriculture appears to be the land use variable having the most negative effect on biological indices after artificial land use. It is a source of chemical and physical pressures on rivers. Indeed, the presence of agricultural land use in a catchment leads to an increase of the risk of erosion and thus risks of clogging the benthic substrate by suspended matter and a change in the river's hydromorphology. Lastly, intensive agriculture is a major source of nutrients (nitrogen and phosphorous) and phytosanitary products, leading to eutrophication and diffuse chemical pollution of the river.

These elements make land use an efficient indirect descriptor of pressures exerted on rivers that incorporates the different functional scales structuring the hydrosystem (Allan, 2004).

#### **2.4.4 Three complementary indices**

The three indices used (I2M2, IBD2007 and IPR+) show similar response profiles with a predominant impact for physico-chemical parameters (nutrients and organic matter) measured at the scale of a site, followed in second place by the impact of land use at catchment scale and, in third place, by factors of hydromorphological change with an effect far more difficult to highlight. However, we were able to show that the responses of the different indices varied considerably, corresponding to what has been observed by other authors (Dahm et al., 2013; Johnson and Hering, 2009). For example, the macroinvertebrate index (I2M2) is more specifically sensitive to parameters signalling slower flows and the modification of the facies, leading to a more lentic river regime. The diatom index (IBD2007) was more sensitive to parameters signalling a modification of the magnitudes of floods. The fish index (IPR+) was influenced by parameters signalling a risk of clogging of the substrate

by fine sediments and parameters indicating a break in ecological continuity. This supports the fact that these indices are complementary and must be used together in order to perform a complete analysis of the pressures exerted on rivers.

#### **2.4.5 Predictive models**

Globally, the 3 predictive models developed demonstrated good performances. The best model was I2M2 with a rate of correct classification of ecological status of 81%. This was followed in order by model IBD2007, with a rate of correct classification of 79%, and model IPR+ with 70%.

These results are consistent with the results obtained using explanatory models. The most efficient models corresponded to the indices that respond to the widest diversity of stressors with, in order, I2M2 then IBD2007 and, finally, IPR+.

The variables used to build the model cover the three spatial scales taken into account: the catchment, reach and site. However, these scales are not representative in an equivalent way. Indeed, in the three models, the catchment variables represent, as a proportion of the number of variables selected to build the model, 25% of the variables, the reach variables represent 25% of the variables and the site variables represent 50% of the variables.

In terms of predictive performance, it should be emphasised that these predictive models are capable of predicting good status efficiently, since they highlight values of high specificity. Their performances are less good when it comes to predicting bad status. The conditions of bad status are difficult to define because of multiplicity of pressures types influencing biological indices. Conditions of good status are easier to define.

This type of model is of great interest for managers as it permits using a set of pressure data to successfully predict the status of water bodies for which biological monitoring data are unavailable.

The conditional inference tree method used here is an advantage for making information available to managers. The literature describes methods such as the random forest (Liaw and Wiener, 2002) and their conditional equivalent, the conditional random forest (Hothorn et al., 2006; Maloney et al., 2009) capable of improving the predictive

capacities of models. We made the choice of using a simpler method with the specific aim of providing managers with a tool that can be easily integrated in a decision algorithm of the regulatory chain of ecological status evaluation. The method chosen allows us to communicate a decision tree flowchart directly to decision-makers so that they can then use it directly without having any specific modelling know-how.

#### **2.4.6 Taking into account spatial scales**

Since Allan and Johnson (1997); Frissell et al. (1986); Parsons et al. (2003, 2004); Thorp et al. (2006); Townsend et al. (2003), there has been consensus that understanding the functioning of river ecosystems requires taking into account the organisation of its spatial hierarchy. This is what we have done here by taking pressures into account at catchment, reach and site scales. The results of the explanatory models show the importance of the site scale (physico-chemical variables) to explain the variability of biotic indices. But these results also show that the variables measured at catchment scale (in this case land use) integrate every type of pressure at smaller scales located in the downstream catchment. Lastly, the intermediate scale of reach is less well represented in these models. The parameters measured at this scale do not have a direct effect on biological organisms though they allow highlighting the structural modification of their living conditions, which accentuates the effect of the physical and chemical pressures of the environment.

Marzin et al. (2012) analysed inter-scale relations and showed that the physico-chemical quality of water at the scale of a station was better correlated with the land use of the catchment than with hydromorphological parameters. This seems to confirm what we saw with our results. Indeed, the indices were less sensitive to hydromorphological changes. This may imply a problem in taking into account the effect of pressures exerted on a river at the intermediate scale of the reach. The processes linking the changes of the hydromorphological functioning of a river and the habitat of biological organisms are many and complex and it appears difficult to apprehend them exhaustively with the data available to us in this study. Aquatic ecosystems are influenced by the consequences of anthropic actions whatever the scales (Allan, 2004). But these impacts are numerous, direct and indirect, and complex, meaning that it is very difficult to determine the relative roles of these different scales.

## **2.5 Conclusion**

This study allowed us to show that it is possible to build models capable of predicting ecological status that are efficient and easily transferable, using data of different scales that influence the functioning of hydrosystems. Another aim was to determine the impact factors that influence ecological status determined on the basis of macroinvertebrates, diatoms and fishes.

We showed that the parameters characterising the load of nutrients and organic matter had a predominant effect on the three compartments, and that land use variables played an integrating role of the different pressures acting on rivers and explained a major part of their degradation. On the contrary, we also showed that it was more difficult to characterise the role of the hydromorphological descriptors measured at the intermediate scale of the reach due to the difficulty of characterising the links between scales.

These results open the path to the development of a model that specifically takes into account the links between the ecological status and hydromorphological functioning of rivers. To be complete, this model must take into account catchment, reach, river corridor and site scales, and place the links between these different scales in perspective. This work should focus on both the functioning of the biological compartment and on the trophic functioning of the environment. The different compartments exerting pressures on the environment should be studied hierarchically and specific attention will be given to their combined effects.

## **2.6 Acknowledgements**

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## **Chapitre 5 - Prise en compte de la hiérarchie d'échelles dans les modèles multi-stress : vers une meilleure évaluation des impacts anthropiques sur l'écosystème rivière.**

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## **1 Synthèse de l'article**

Les cours d'eau sont constamment soumis à des pressions qui structurent à la fois les communautés aquatiques et leur environnement local. Récemment, (Thorp 2014) a décrit les paysages fluviaux comme une mosaïque de « patchs » longitudinaux, latéraux et verticaux. Les structures et les fonctions écologiques sont fortement influencées par les attributs du patch considéré à des échelles immédiatement au-dessus et au-dessous de l'échelle d'intérêt. Il a donc souligné la nécessité d'introduire la nature hiérarchique des rivières dans les modèles pression-impact et d'évaluer la façon dont les réponses des pressions diffèrent par rapport aux modèles non-hiérarchiques ou ne prenant en compte qu'une seule échelle spatiale.

En supposant que la structure et la dynamique de l'habitat du cours d'eau sont déterminées par le bassin versant environnant, (Frissell et al. 1986; Thorp 2014) décrit le cours d'eau comme un système hiérarchique, variable dans l'espace et le temps et hiérarchisé successivement en segments, tronçons, radiers/mouilles et micro-habitats. Une échelle spécifique de cette organisation constitue donc un sous-système avec une structure homogène et une fonction spécifique dans l'écosystème cours d'eau. Depuis lors, il y a un consensus sur le fait que la compréhension du fonctionnement des écosystèmes fluviaux nécessite de prendre en compte l'organisation hiérarchique en échelles spatiales imbriquées (Allan and Johnson 1997; Parsons et al. 2003; Parsons et al. 2004; Poff et al. 1997; Poff et al. 2006; Thorp 2014; Thorp et al. 2006; Townsend et al. 2003).

La dégradation du fonctionnement écologique de l'écosystème cours d'eau résulte d'une combinaison de plusieurs types de pressions potentielles : les rejets ponctuels, les pollutions diffuses, les altérations des flux solides et liquides, des structures artificielles et des ruptures dans la connectivité (Borchardt and Richter 2003). Selon le concept DPSIR (EEA 2003; Kristensen 2004), les activités humaines et l'utilisation des terres (agriculture, urbanisation) constituent des forces motrices qui modifient les composantes abiotiques de l'écosystème (physico-chimie, hydromorphologie) en générant des pressions combinées (rejets de produits chimiques, changements physiques, etc.). Ces changements affectent alors les communautés biologiques et l'état écologique des cours d'eau.

Les forces motrices et les pressions s'organisent en suivant un modèle régional, ce qui, considérant les activités humaines (les structures bâties, l'agriculture, l'eau utilisée) comme régies par l'environnement naturel (Gallant et al. 2004), permet d'identifier des « pathologies régionales » significatives de la rivière. Par conséquent, la corrélation entre une typologie naturelle des masses d'eau et la répartition spatiale des pressions humaines permettrait de constituer des unités d'évaluation et de gestion cohérentes.

En Europe, la directive cadre sur l'eau demande de maintenir et de restaurer le bon état écologique des cours d'eau et de mettre en place des programmes de surveillance. Pour répondre à ces exigences, une meilleure compréhension des relations entre les pressions générant des modifications du fonctionnement des cours d'eau et l'état écologique est nécessaire (Reyjol et al. 2014).

Les études qui ont tenté de prendre en compte différentes échelles spatiales pour étudier la réponse des éléments de qualité biologique à des pressions et des forces motrices, telles que (Dahm et al. 2013; Marzin et al. 2013; Villeneuve et al. 2015), n'ont pas réellement réussi à rendre compte de la structure hiérarchique des données de pressions. Quelle que soit la méthode de modélisation ou le nombre de sites étudiés, ils ont démontré que les facteurs de changement hydromorphologiques arrivaient en troisième position après l'occupation du sol et la physico-chimie parmi les variables explicatives qui influent sur l'état écologique. Dans leur étude, (Dahm et al. 2013) ont conclu que les effets des pressions hydromorphologiques étaient mesurables mais que leurs impacts sur les paramètres biologiques utilisés étaient relativement faibles. (Villeneuve et al. 2015) ont souligné que l'utilisation d'indicateurs modernes et efficaces tels que l'I2M2 (Mondy et al. 2012), pouvait améliorer la compréhension des effets hydromorphologiques sur la faune macroinvertébrée.

Il apparaît donc nécessaire d'introduire plus explicitement la structure régionale du fonctionnement du cours d'eau et son effet sur les relations hiérarchiques entre les forces motrices et les différentes pressions qu'elles génèrent dans le cadre de la modélisation pression-état. Nous avons proposé pour cela d'utiliser le concept statistique des modèles hiérarchiques (ou multi-niveaux) qui sont utiles lorsque l'information est disponible sur plusieurs niveaux différents d'unités d'observation.

Notre objectif est donc d'introduire cette connaissance dans le processus de modélisation afin de mieux comprendre les relations pression-impact dans le système cours d'eau. Plus précisément, nous voulons tester si les relations entre les pressions à l'échelle locale et l'état écologique sont hiérarchiquement influencées par la typologie régionale (basée sur la physiographie naturelle et les forces motrices).

Dans cet article, nous avons développé un modèle hiérarchique qui relie l'indice multimétrique basé sur les invertébrés benthiques (I2M2) à des pressions hydromorphologiques et physico-chimiques selon des modèles hiérarchiques régionalisés. Pour ce faire, les bassins ont été regroupés en fonction de leurs caractéristiques naturelles et des gradients de forces motrices. Nous avons supposé que l'organisation hiérarchique des cours d'eau conduisait à différentes réponses des organismes à la pression, en fonction des caractéristiques du site étudié, dépendant de l'échelle régionale.

La différence de variance expliquée entre le modèle non-hiérarchique et les modèles hiérarchiques (38% vs 45%) montre que la prise en compte d'une typologie des bassins versants en fonction des caractéristiques physiographiques et des forces motrices donne une plus grande puissance explicative à la relation entre les pressions hydromorphologiques et physico-chimiques et l'I2M2.

Par ailleurs, les résultats montrent un meilleur ajustement du modèle hiérarchique qui présente des coefficients sensiblement différents de ceux obtenus avec le modèle non hiérarchique.

L'analyse des modèles hiérarchiques par type ne permet pas d'identifier une tendance générale. Par conséquent, l'étude des caractéristiques des bassins versants seules semble être insuffisante pour permettre de comprendre comment les assemblages d'invertébrés sont affectés par les pressions. Les différences peuvent être expliquées par les caractéristiques naturelles (à savoir la géologie, le climat) représentées par les hydro-écorégions. Il peut également y avoir un effet d'interaction des conditions naturelles et des pressions sur la réponse de l'organisme. (Marzin 2013) montre que les variables physiographiques peuvent représenter jusqu'à la moitié de la variabilité expliquée pour les indices biologiques qu'elle considère (poissons et macroinvertébrés).



Quel que soit le modèle considéré les communautés de macroinvertébrés sont altérées de façon similaire par les mêmes variables de pression. Certaines d'entre-elles ont un effet positif, telles que la présence et la continuité de la végétation rivulaire dans les zones rivulaires d'une largeur de 10 et 30 mètres ou le niveau de la concentration en oxygène dissous. Certaines autres variables présentent une influence négative globale sur le score I2M2, surtout la DBO5, les concentrations en ions ammonium, nitrites, nitrates et les concentrations en phosphore total. Ces résultats sont cohérents avec ceux de (Dahm et al. 2013; Donohue et al. 2006; Hering et al. 2006; Johnson and Hering 2009; Villeneuve et al. 2015).

En plus des paramètres physico-chimiques, les résultats montrent que les assemblages biotiques sont également affectés par des altérations hydromorphologiques qui représentent à la fois l'homogénéisation potentielle de l'habitat (seuils, rectification, surlargeur, suppression de la ripisylve, endiguements et artificialisation des berges) et les conditions hydrauliques (drainage, irrigation, stockage).

Les indices biotiques réagissent de différentes façons aux altérations des flux solides et liquides. En ce qui concerne les processus morphologiques, l'I2M2 est influencé négativement par des variables décrivant la perte de la sinuosité, l'augmentation de l'érosion des berges, la réduction du débit et une altération de la succession radiers/mouilles (évaluées indirectement par des proxys mesurables sur les couches géographiques numérisées disponibles). Cela peut se comprendre dans la mesure où ce sont des facteurs qui agissent sur la diversité structurale de l'habitat à méso-échelle, en particulier dans les zones rivulaires.

L'indice I2M2 est plus fortement affecté par des altérations physico-chimiques (concentrations en nutriments) que par les altérations hydromorphologiques. Les macroinvertébrés semblent être plus sensibles aux facteurs qui maintiennent la qualité de l'eau, en particulier ceux liés à l'eutrophisation (azote total et les concentrations en phosphore), qu'aux pressions hydromorphologiques agissant sur le débit et la diversité de l'habitat dans les cours d'eau.

Cette tendance n'est pas spécifique à une zone géographique donnée, mais semble être convergente avec les résultats mis en avant par les études de (Dahm et al. 2013; Johnson

and Hering 2009; Villeneuve et al. 2015), qui ont également observé une plus grande sensibilité des organismes aquatiques à la pollution chimique qu'aux altérations hydromorphologiques. Sur ce point, notre hypothèse était que la prise en compte de la typologie des bassins versants comme paramètre hiérarchique dans le modèle fournirait une meilleure compréhension du rôle relatif des altérations hydromorphologiques. Nos résultats ne diffèrent pas des résultats publiés précédemment, ce qui renforce l'hypothèse du rôle probablement secondaire des altérations hydromorphologiques (appréciées ici majoritairement par des proxies) sur la qualité biologique des cours d'eau par rapport à celui joué par les pressions physico-chimiques.

## **2 Introducing scale hierarchy in multi-stress models: towards better assessment of human impacts on river ecosystems**

### **Abstract :**

We developed a hierarchical model that links the benthic invertebrate index (I2M2) to hydromorphological and physico-chemical pressures according to a regional hierarchical patterns.

To do this, catchments were grouped according to their natural characteristics and driving forces gradients.

Introducing hierarchical factors in multi-stress models allows one to consider slight variations in responses of organisms to pressures according to their natural characteristics and their driving forces acting at catchment scale and therefore, better understanding of macroinvertebrate response to human pressures in well-defined river types, (i.e. specific geographic areas with specific natural conditions and catchment-scale pressures) was provided. The I2M2 index is more strongly impaired by physico-chemical alterations (nutrient concentrations) than by hydromorphological ones.

Taking into account regional geographical and driving force patterns improves the capacity to distinguish the responses of biological quality elements to local environmental pressures. Consequently, pressures-state relationship studies should integrate the hierarchical links between scales. Moreover, it would be interesting to differentiate reach and site scales, and apply an analytical method able to quantify the effects of the various pressures and natural characteristics on various biological compartments according to the spatial scale in which they evolve.

## **2.1 Introduction**

Streams are constantly subjected to pressures that structure both aquatic communities and their local environment. Recently, (Thorp 2014) pointed out that riverine landscapes are composed of a mosaic of longitudinal, lateral and vertical patches such that ecological structures and functions are greatly influenced by patch attributes considered at scales immediately above and below the initial scale of interest. He therefore highlighted the need to introduce the hierarchical nature of rivers in pressure-state models and assess how it contrasts these responses in comparison to non-hierarchical site- or reach-based models relying on a single spatial scale.

Assuming that the structure and dynamics of stream habitat are determined by the surrounding catchment area, (Frissell et al. 1986) described stream structure as a hierarchical system, variable in both space and time. Stream systems can be defined as hierarchically organized systems incorporating, on successively lower levels, stream segment, reach, pool/riffle and microhabitat subsystems. A specific organization scale thus forms a subsystem with a homogeneous structure and a specific function in the stream ecosystem. Since then, there has been consensus that understanding the functioning of river ecosystems requires taking into account the organization of their spatial nested hierarchies (Allan and Johnson 1997; Parsons et al. 2003; Parsons et al. 2004; Thorp et al. 2006; Townsend et al. 2003).

For (Allan 2004), aquatic ecosystems are influenced by anthropic actions whatever the scales. But these impacts are not only numerous, and direct or indirect, they are complex and often linked with each other, meaning that it is very difficult to determine the relative roles of the processes taking place at these different scales.

The degradation of the ecological functioning of stream systems results from a combination of several types of potential pressures: punctual discharges (more or less episodic), diffused pollutions, changes of solid and liquid flows, artificial structures and breaks in connectivity (Borchardt and Richter 2003). According to the DPSIR concept (driving forces, pressures, state, impact and response) (Kristensen 2004), human activities and land-use (agriculture, urbanisation) constitute driving forces that change the abiotic components

of the ecosystem (physico-chemistry, hydromorphology) by generating combined pressures (chemical discharges, physical changes, etc.). These changes then affect biological communities and thus ecological status.

The driving forces and pressures themselves frequently follow a regional pattern, making it possible to identify meaningful regional “river typical pathologies”, considering that human activities (human settlement, agriculture, water uses) are governed by the natural environment (Gallant et al. 2004). Consequently, congruence between natural types and the spatialization of human pressures would allow grouping water bodies into coherent evaluation and management units.

The assessment of human impacts on rivers therefore requires a hierarchical system that could be used at a scale adapted to the tasks of evaluating different quality elements, guiding management and policy, and communicating to the public.

In Europe, the Water Framework Directive (WFD) requires the protection of water bodies, the restoration of their good ecological status and the establishment of monitoring programs. To fulfil these requirements (WFD, appendix II), better understanding is needed of the relationships between pressures generating stream functioning alterations and the ecological status of streams (Reyjol et al. 2014).

According to the WFD, impacts on the ecological status of streams are measured by four different biological indices based on four distinct biological quality elements (diatoms, macrophytes, macroinvertebrates, fishes, WFD appendix V) impacted by functioning alterations and whose community structures can be linked to environmental quality. Thus these indices have been developed to be sensitive to different gradients of pressures, such as pollution and habitat degradations, and are complementary as the different quality elements do not express the same alterations and pressures. It is possible to assess the impacts of different pressures acting on river ecosystems by studying the composition of stream communities on the basis of these indices.

On the basis of this analytical framework, some works have already focused on understanding the links between biological conditions and pressures in a multi-scale context. Land-use, hydromorphology and physico-chemistry have been linked to biotic indices based

on macrophyte communities (Feld 2013), diatoms (Dahm et al. 2013; Villeneuve et al. 2015), fish (Feld 2013; Marzin 2013; Marzin et al. 2012; Villeneuve et al. 2015) and macro-invertebrates (Feld 2013; Marzin et al. 2012; Sponseller et al. 2001; Sundermann et al. 2013; Villeneuve et al. 2015; Wasson et al. 2010). However, these studies failed to explicitly take into account spatial scale hierarchy in their models.

(Feld 2013) showed that not all biological groups respond in the same way to the same pressures and that lotic fauna were more correlated than lotic flora to the land-use indicators measured on the catchment scale, such as riparian corridor, with a more marked correlation for mountain rivers than for plain rivers. In line with (Sponseller et al. 2001) and (Wasson et al. 2010), it has also been shown that the riparian corridor plays a major protective role in the ecological functioning of a river and that taking into account regionally specific geographical characteristics, in particular geographical entities such as eco-regions, is essential for good river management. (Dahm et al. 2013; Sundermann et al. 2013; Villeneuve et al. 2015) used a wider range of categories of pressure variables at different spatial scales to explain ecological status and showed that physico-chemistry and land-use had a greater effect than hydromorphological alterations on the biotic indices of all the biological groups.

More generally, these studies have highlighted the links that exist between each of these pressures and biological indices. Some have shown that the links between pressures and biological quality elements are influenced by the scale at which the pressure is considered. Thus catchment, reach and riparian corridor are scales that have a considerable structural effect on the pressures affecting ecological status (Marzin 2013; Wasson et al. 2010).

Recent studies suggest that natural descriptors can interfere with the response of organisms to pressures and should be considered in the analysis, such as in (Marzin et al. 2012; Marzin et al. 2013) who included natural physiographic variables (altitude, stream power, mean slope, catchment area, mean air temperature, temperature range, distance to source, hydrological regime, geological type, ecoregions) in their analyses and showed that they interact with human pressures at different spatial scales.

The few studies that have attempted to introduce influential variables considered at different spatial scales to study the response of biological quality elements to pressures and driving forces, such as (Dahm et al. 2013; Marzin et al. 2013; Villeneuve et al. 2015), did not succeed in hierarchically structuring the pressure data. Whatever the method or the number of studied sites, they demonstrated that hydromorphology change factors only take third place after physico-chemical and land-use factors among the three impacting variables explaining ecological status. In their study, (Dahm et al. 2013) concluded that the effects of hydromorphological alterations were measurable but that their impacts on the biological metrics they had used were relatively low. (Villeneuve et al. 2015) pointed that the use of modern and efficient indicators such as the  $I_2M_2$  (Mondy et al. 2012), which can take more detailed account of the impact of different hydromorphological stress factors, may improve understanding of hydromorphological alterations effects on macroinvertebrates.

It now appears necessary to explicitly introduce the regional structure of stream functioning along with the hierarchical relations between driving forces and the different pressures they generate in a pressure-state modelling framework. An interesting way of doing this relies in the statistical concept of hierarchical (or multilevel) models which are useful when information is available on several different levels of observational units (Gelman et al. 2004).

Our objective is therefore to introduce this knowledge on the organization of spatial scales and on the links between pressures and ecological status to better understand the pressure-impact processes occurring in stream systems. More precisely, we want to test if the relationships between local scale pressures and ecological status are hierarchically influenced by regional patterns (such as natural physiographic and/or driving forces).

In this paper, we have developed a hierarchical model that links the benthic invertebrate index ( $I_2M_2$ ) to hydromorphological alterations and physico-chemical pressures according to regional hierarchical patterns. To do this, catchments were grouped according to their natural characteristics and driving force gradients. We assumed that stream hierarchical organization leads to different organism responses to types of pressure, according to the broader scale characteristics of the studied site.

## 2.2 Materials and Methods

### 2.2.1 Data

This study focuses on 1,015 sites linked to the French monitoring network in the twenty-two hydro-ecoregions (Wasson et al. 2002) (Fig. 1). Hydro-ecoregions (HER) are based on a top-down approach, assuming an *a priori* delimitation of geographical entities in which stream and river ecosystems should exhibit common characteristics. The concept is derived from ecoregions originally adapted to aquatic ecosystems by (Omernik 1987). Geology, relief and climate are widely recognized as the primary large-scale determinants of the functioning of running-water ecosystem (Naiman 1992), and HER are delimited on the basis of these factors (Wasson et al. 2002). The 1015 sites are homogeneously distributed between the 22 HER ensuring good representation of the different physiographic conditions in the dataset.

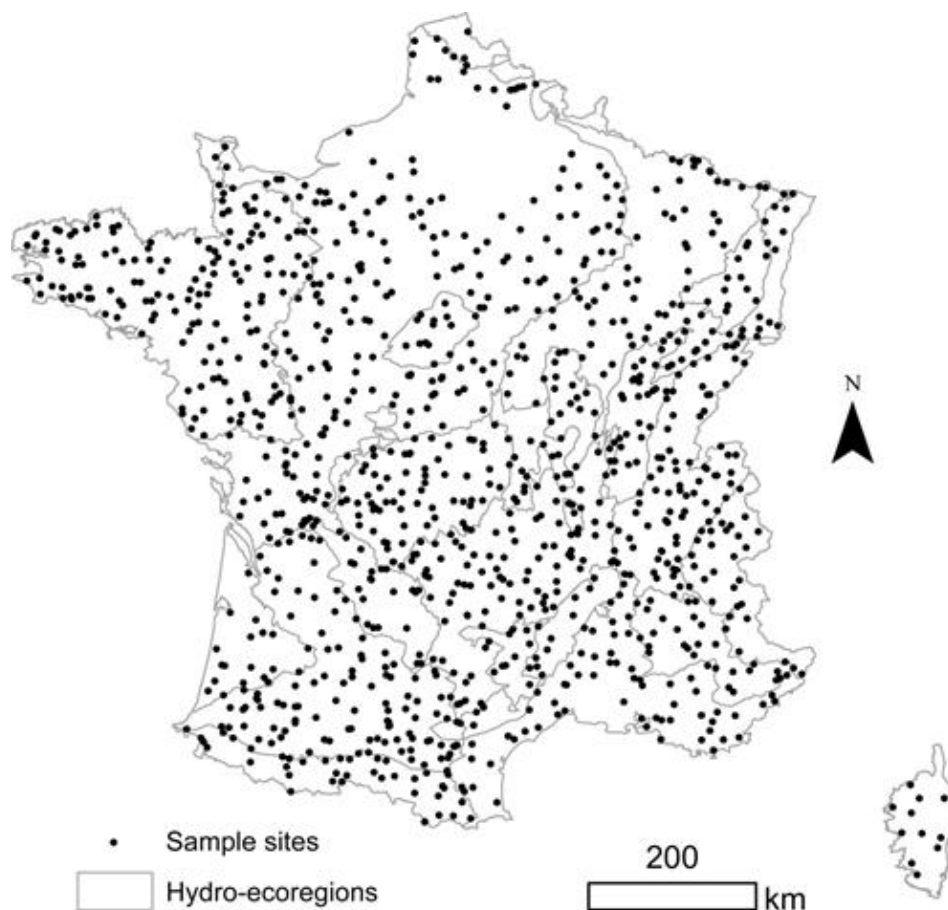


Fig. 1. Location of the 1,015 study sites and the limits of the hydro-ecoregions of France



### **2.2.1.1 Land-use data**

Catchment land cover percentage data are used to describe driving forces patterns (Kristensen 2004). They were compiled from CORINE Land Cover 2006 (CLC) geographical data (Büttner and Kosztra 2007) that inventories land-use in 44 categories. Land-use data were finally aggregated into four land cover percentage variables (Table 1):

- Urban: urban land, industrial or commercial and communication networks, mines, landfills and construction sites, artificial green spaces, non-agricultural areas;
- Intensive agriculture: arable land, permanent crops, orchards, vineyards, olive groves, natural crops associated with permanent crops, cropping and fragmented systems;
- Non-intensive agriculture: meadows, land mainly used for agriculture with considerable natural vegetation and agro-forestry land;
- Natural land: forests and semi-natural land, wetlands, and water surfaces.

In addition to these catchment land cover variables, total catchment area, and the covering percentage and mean number of patches per kilometer of riparian forest in a 30-meter buffer zone on the entire catchment were also integrated in the analysis.

### **2.2.1.2 Hydromorphological data**

Within catchment areas, the hydromorphological variables considered at a reach-scale are gathered from the SYRAH\_CE project (Chandesris et al. 2008) (Table 1). These data are used either as proxies for a risk of hydromorphological alteration (rate of dams, rate of roads, rate of dikes, rate of urban land-use, rate of over-width, rate of lakes and ponds, rate of vegetation, rate of straightening), or as a direct description of disturbance of natural solid and liquid flows (erosion).

### **2.2.1.3 Physico-chemical data**

Site-scale physico-chemical data (Table 1) are measured monthly at each station of the monitoring network. This includes: suspended matter rate, dissolved oxygen rate, biochemical oxygen demand at 20°C (BOD5), and concentrations of ammonium, nitrites, nitrates and total phosphorus. We chose as pressure indicator the mean value of concentrations measured during the period 2008-2009.

#### **2.2.1.4 Biological data**

The biological compartment is represented by the multimetric macroinvertebrate index,  $I_2M_2$ . This index is sensitive to pressures acting on the stream environment such as nutrient concentration and urbanization (Mondy et al. 2012). As recommended in the Water Framework Directive (WFD), we transformed  $I_2M_2$  values into an equivalent quality ratio (EQR) that expresses the ratio from the calculated score to reference scores. Therefore the equivalent quality ratio represents a deviation from the reference score. We averaged the scores based on data collected once a year from 2008 and 2009 to decrease the model's sensitivity to index observation.

Table 1. Description of data set: variables, land-scale and range.

Scale	Variable	range of values (min-max)	Source
Catchment	urbanization of catchment (%)	0 - 58.7	Corine Land Cover (CLC) 2000 <a href="http://terrestrial.eionet.europa.eu/CLC2000">http://terrestrial.eionet.europa.eu/CLC2000</a>
Catchment	intensive agriculture (%)	0 - 100	Corine Land Cover (CLC) 2000 <a href="http://terrestrial.eionet.europa.eu/CLC2000">http://terrestrial.eionet.europa.eu/CLC2000</a>
Catchment	non-intensive agriculture (%)	0 - 89.9	Corine Land Cover (CLC) 2000 <a href="http://terrestrial.eionet.europa.eu/CLC2000">http://terrestrial.eionet.europa.eu/CLC2000</a>
Catchment	Non-anthropized space (%)	0 - 100	Corine Land Cover (CLC) 2000 <a href="http://terrestrial.eionet.europa.eu/CLC2000">http://terrestrial.eionet.europa.eu/CLC2000</a>
Catchment	Presence of riparian vegetation (% of stream length)	6.8 - 100	% of total stream length
Catchment	Structure of riparian vegetation (patches/m)	0.5 - 42.5	Number of patches/m
Reach	Erosion	0-3.4	alea erosion des sols INRA (Montier et al. 1998)
Reach	Rate of roads/railways near the riverbed	0-262.6	BD Topo IGN
Reach	Rate of roads/railways in the flood plain	0-3087.5	BD Topo IGN
Reach	Rate of dikes in the riverbed	0-221.6	BD Topo IGN
Reach	Rate of dikes in the flood plain	0-764.4	BD Topo IGN
Reach	Rate of dams	0-6.5	BD Topo IGN
Reach	Rate of trees on banks "screen of trees"	0-100	BD Topo IGN
Reach	Rate of vegetation on banks "riparian vegetation"	0-100	BD Topo IGN
Reach	Rate of vegetation in riverbed	0-100	BD Topo IGN
Reach	Rate of "artificial" type land use near riverbed	0-106	BD Topo IGN
Reach	Rate of straightness	0-100	BD Topo IGN
Reach	Rate of ponds in riverbed	0-34	BD Topo IGN
Reach	Over-width	0-15742.1	BD Topo IGN
Site	Suspended matter (mg/l)	1.5-864.9	RCS
Site	Dissolved oxygen (mg/l)	6.5-13	RCS
Site	BOD5 at 20°C (mg/l)	0.5-4.8	RCS
Site	Ammonium (mg/l)	0-0.48	RCS
Site	Nitrites (mg/l)	0-0.36	RCS
Site	Nitrates (mg/l)	0.34-65.33	RCS
Site	Total phosphorous (mg/l)	0-0.93	RCS

### **2.2.2 Spatial typology**

To represent the hierarchical links between scales, we first depicted a typology to assign sites to a type describing its physiographic and driving force characteristics. Hydrocoregions were grouped at regional scale by mapping expertise in six distinct groups: high mountains, highlands, detritic hills, crystalline hills, sedimentary lowlands and detritic lowlands (Fig. 2).

Then, an independent principal component analysis (PCA) was performed for each of these groups using the catchment-scale variables (land-use, catchment area, presence and structure of riparian vegetation) of the sites. The first two axes of each PCA explained between 52 and 68 % of the total variability of the dataset, depending on the HER group considered.

Using these PCA outputs for each HER group, a Euclidian distance was computed and introduced into a hierarchical ascendant clustering method using the Ward aggregation criterion, as proposed in (Feio and Dolédec 2012; Piscart et al. 2009), to distribute the 1015 study sites into homogeneous types. In view to applying statistical models, we introduced the constraint of keeping at least 30 sites per type.

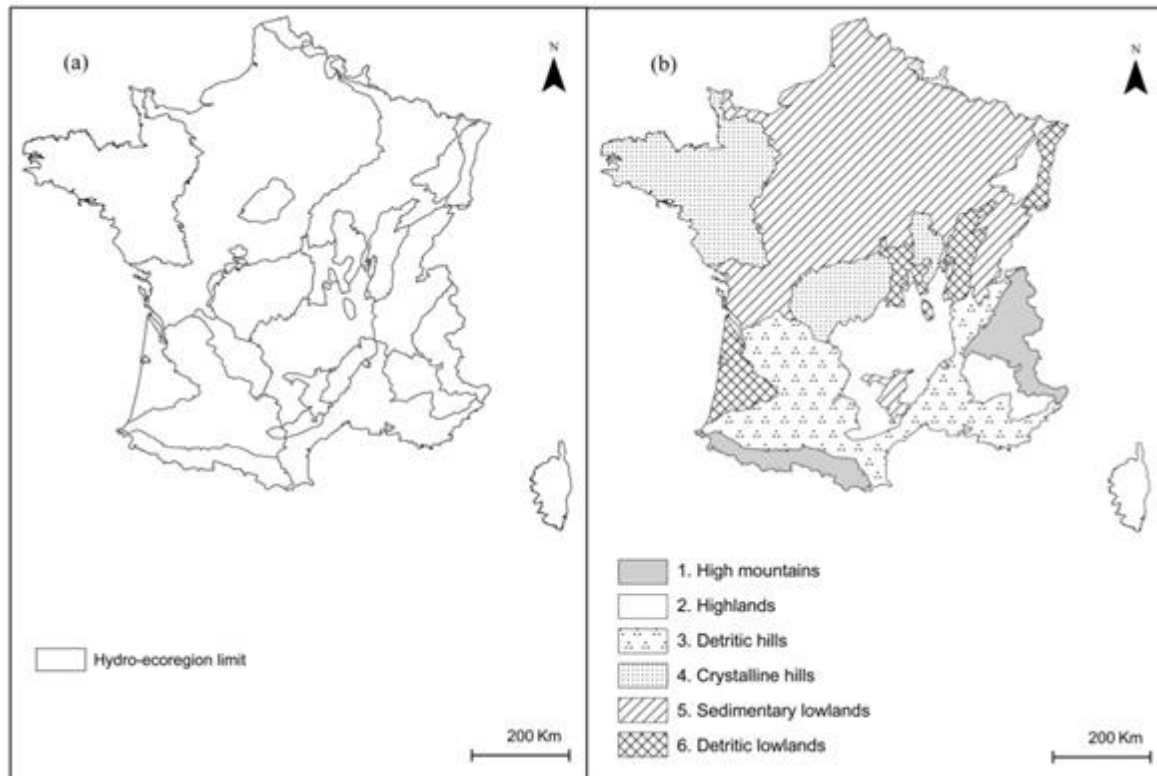


Fig. 2. Limits of French hydro-ecoregions (a) and of the six groups created after geographical analysis (b).

### 2.2.3 Statistical modelling

The response of the macroinvertebrate biological index to both hydromorphological and physico-chemistry pressures at reach and site scales (Table 1), was analyzed using partial least square regression (PLS) (Wold et al. 2001). PLS is an adaptation of the classical linear model. Its objective is to mitigate the main drawback of the latter: the instability of the regression coefficients due to the collinearity of the predictors. Indeed, when collinearity becomes high in a set of predictive variables, the estimations of the regression coefficients fluctuate from one sample to the other (Cramer et al. 1988). PLS regression permits obtaining interpretable prediction coefficients even in the case of strong correlation between the predictors. The final equation is interpreted as a linear regression equation, the coefficients clearly reflecting the effect of a predictor relative to others; moreover  $R^2$  is a good estimator of the model's efficiency. The significance of the coefficients is then validated using the jack-knife method (Efron and Gong 1983; Martens and Martens 2000). (Wasson et al. 2010) demonstrated the interest of using this method in ecology when using predictors which, as in this case, are strongly correlated with each other.

In order to highlight the benefits of the hierarchy introduced we developed both a non-hierarchical and a hierarchical model.

The hierarchical PLS model is developed including the typology effect by inserting interaction terms between each variable and the vector containing type membership information, thus considering a two-level hierarchy. The non-hierarchical PLS model was built using sites across all regions in France to obtain a reference model for comparison with the hierarchical one.

The final equation of the hierarchical model is written for the  $i^{\text{th}}$  site belonging to the  $j^{\text{th}}$  group:

$$Y_{i,j} = A_j + \sum_{k=1}^n B_{jk} * X_{ik} + \varepsilon_{i,j}$$

Where A represents variables taken into account without the interaction term, B is the k-vector of regression coefficients according to type j, X is the k-vector of covariates values for site i, and  $\varepsilon$  is the model residuals matrix.

The models are interpreted according to their coefficient of determination  $R^2$ , reflecting the percentage of explained variance of the biological index, and their estimated normalized regression coefficients, indicating the relative contribution of each variable to the models.

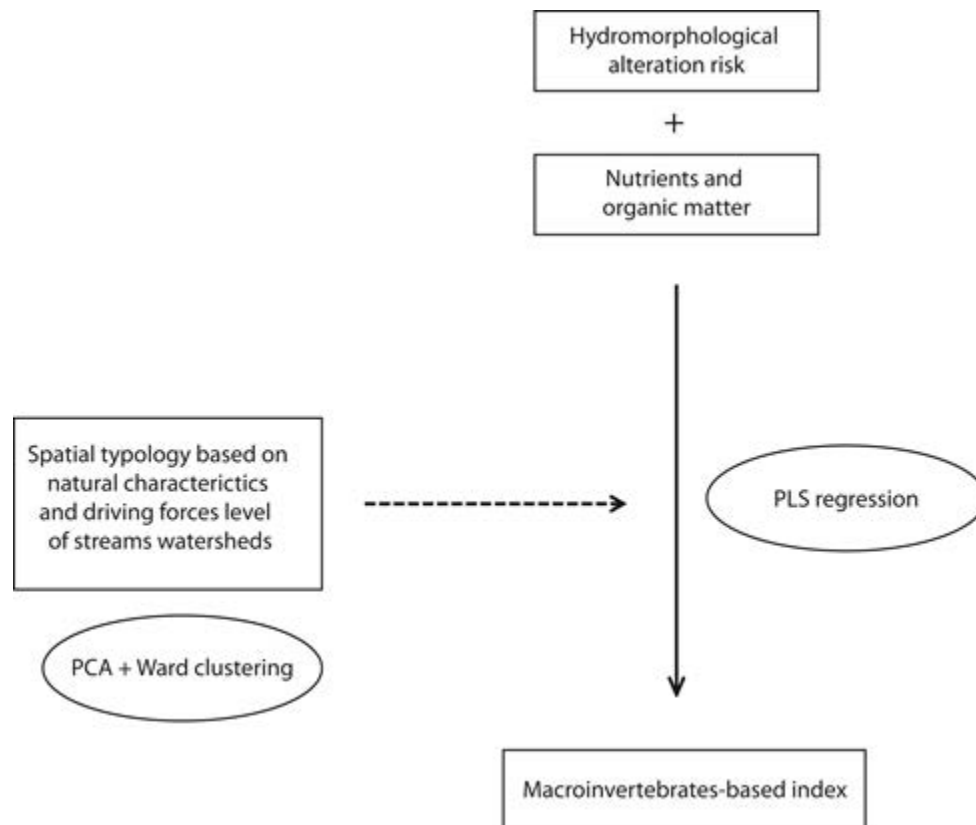


Fig. 3. Modelling plan. Arrow in solid lines represent the main statistical relationship that we wish to study. The dotted line arrow represents the hierarchical effect we introduced in the statistical relationship. Ellipses represent the statistical methods used.

## 2.3 Results

### 2.3.1 1. Typology

Thirteen types were defined in which sites are distributed (Table 2).

HER group no.1 was divided into two types following the automatic classification. Type 1a presents larger, more urbanized and more agricultural catchments. However, their riparian forests are relatively similar in terms of extent of coverage and structure. Type 1a had 43 stations and group 1b 45 stations.

The classification of the second group of HER, the highlands, permitted the determination of two types. The first type, 2a, comprised 61 stations, had a high percentage of natural territory, continuous riparian forest coverage and relatively small catchments. The second type of this group, 2b, also had a lower percentage of natural territory, less though

continuous riparian forest, but also large catchments and more agricultural territory. It contained 112 stations.

The third group characterized by a warm climate was divided into two types. The first type, called 3a, was determined by landscape dominated by intensive agriculture and a high rate of urbanization. It contained 97 stations. Type 3 was also subject to not inconsiderable urbanization though was not impacted by agriculture. The 59 catchment areas presented a large proportion of natural territory, with abundant and continuous riparian forest.

The group corresponding to crystalline hills was divided into 3 types after the automatic classification. The first, 4a composed of 30 stations, was characterized by agriculture with low impact and coverage by continuous riparian forest. The second type with 74 stations called 4b, contained catchments subjected to agricultural pressures of high and low impact. The last type of this fourth group, called 4c, with 86 stations, was subjected to heavy pressure from intensive agriculture and the rate of non-natural land.

The classification of the penultimate HER group dealt with here, which we have defined as sedimentary lowlands, described three types. 5a, the first of these, was heavily urbanized, subjected to both agriculture with low impact and intensive agriculture, and included 125 stations. The second type presented a territory heavily covered by agriculture with high impact, non-natural land, and large catchments. This group, 5b, comprised 141 catchments. The last type of the fifth group, type 5c, comprised 92 less artificial catchments, with less agriculture and a high proportion of natural land.

The sixth and last group, the detritic plains, was not subdivided. Indeed, it only had 50 stations. This type was subjected to intensive agriculture and non-natural land use, but retained a large proportion of natural land.



Table 2. Catchment characteristics of the 13 types obtained by the classification algorithm based on the results of the PCA applied to land-use driving force data, catchment area, and the presence and structure of riparian forest in a 30 meter buffer zone around rivers.

For each type, the median value is indicated. First and third quartiles are given in brackets.

Type	No. of sites	Urban	Intensive agriculture	Non-intensive agriculture	Natural	Catchment area	Presence of riparian forest	Structure of riparian forest
High mountains								
1a	43	2.1 [1.3;3.8]	8.9 [6.3;16.8]	13 [7.3;16.4]	75.6 [63.3;81.9]	733 [125;2795]	67.1 [61.8;72.1]	5 [3.99;6]
1b	45	0.6 [0;1.3]	1 [0;2.9]	3.3 [0.9;6.1]	93.9 [89;97.4]	175 [109;448]	71 [50;88.4]	3.9 [2.8;4.9]
Highlands								
2a	61	0.6 [0; 2.5]	3.6 [0;12.6]	5.1 [3.2;7.3]	87.8 [75.7;94.7]	135 [135;436]	78.4 [70.7;90.3]	5.3 [3.9;6.3]
2b	112	2 [0.8;4]	16.4 [7.3;29.9]	27.1 [16.5;36.7]	47.5 [38.8;60.5]	198 [77;1028]	57.1 [47.6;63.6]	6.8 [5.3;7.8]
Detritic hills								
3a	97	1.7 [0.6; 5.4]	60 [46.6;71.6]	11.5 [5.7;15.8]	21.3 [14.8;37.3]	98 [48;324]	43.7 [36.4;54.3]	6.5 [4.9;8.3]
3b	59	2 [0.6;4.3]	22.5 [13.1;32.7]	9.7 [3.5;18.3]	63.4 [48.7;73.2]	167 [90;328]	74.1 [66.8;84]	4.4 [3.5;5.2]
Crystalline hills								
4a	30	1.4 [0.8;2.2]	16.5 [5.2;22.1]	30.1 [24.4;36.4]	51.7 [40.4;65.8]	119 [45;878]	59.1 [55.5;64.3]	6 [5.2;6.7]
4b	74	1.7 [0.9;2.4]	28.8 [18.3;36.3]	54.3 [47.1;64.2]	12.3 [5.8;20.3]	136 [48;366]	38.6 [32;48.3]	8.8 [7.3;10.8]
4c	86	3.2 [2;4.2]	66.7 [56.8;74.7]	19.7 [12.8;28.6]	6.5 [3.1;16.4]	123 [63;334]	47.7 [39;55.5]	7.4 [6.1;9]
Sedimentary lowlands								
5a	125	4 [1.8;6.6]	47 [38.3;66.1]	23.1 [14.8;35.7]	14.6 [6.3;24.5]	242 [90;558]	33.3 [26.7;40.1]	9.4 [7.6;12.6]
5b	141	3 [1.9;4.3]	67 [57.8;79.4]	6.3 [2.9;14.8]	18.5 [9.8;27.2]	319 [153;881]	47.1 [41.1;53.6]	5.9 [4.8;6.8]
5c	92	2.3 [1.1;3.5]	24 [15.9;34.1]	22 [12.8;28.4]	48.8 [40.1;55.9]	188 [88;626]	54.7 [44.1;65.7]	6.1 [4.9;7.3]
Detritic lowlands								
6	50	3.3 [1.6;5.7]	36.8 [16.7;50.5]	6.2 [2.1;20.3]	40.5 [21.2;74.7]	141 [50;430]	47.8 [35.1;86.6]	5 [2.5;7.2]

### 2.3.2 PLS analysis

Table 3 presents the national mean pattern of the biological index response to local pressures. Table 4 presents the mean trends and specific patterns of the biological index response to local pressures according to the different types.

Table 3. Significant normalized PLS coefficients of the non-hierarchical model based on reach and site scale pressure variables.

Studied variable	PLS coefficient
Dam density	-0.068
Roads in riverbed	
Roads in floodplain	
Dykes in riverbed	
Dikes in floodplain	
Urbanization of 100m buffer zone	-0.065
Over-width	
Ponds in floodplain	
Vegetation in floodplain	
Vegetation in 10m buffer zone	0.040
Vegetation in 30m buffer zone	0.055
Straightening	-0.072
Erosion	-0.055
Suspended matter	
Dissolved oxygen	0.093
BOD5	-0.083
Ammonium	-0.132
Nitrites	-0.168
Nitrates	-0.069
Total phosphorous	-0.135
Total explained variance (R <sup>2</sup> ): 38%	

Table 4. Significant normalized PLS coefficients of the hierarchical model based on reach and site scale pressure variables. Significant differences between type coefficient and mean trend coefficient are in bold.

Variable	Mean trend coefficient (b)	Estimated normalized deviation coefficient by type (b')												
		1a	1b	2a	2b	3a	3b	4a	4b	4c	5a	5b	5c	6
Dam density	-0.031	-0.031	-0.031	-0.031	<b>-0.043</b>	-0.031	-0.031	<b>-0.042</b>	<b>-0.039</b>	-0.031	<b>-0.043</b>	<b>-0.044</b>	-0.031	-0.031
Roads in riverbed											<b>-0.004</b>			
Roads in floodplain													<b>0.007</b>	
Dikes in riverbed					<b>-0.008</b>									
Dikes in floodplain					<b>-0.010</b>									
Urbanization of 100m buffer	-0.029	-0.029	-0.029	-0.029	-0.029	-0.029	-0.029	-0.029	<b>-0.042</b>	-0.029	-0.029	-0.029	-0.029	-0.029
Over-width										<b>-0.010</b>		<b>0.010</b>		
Ponds in floodplain									<b>-0.013</b>					
Vegetation in floodplain				<b>0.008</b>										
Vegetation in 10m buffer	0.018	0.018	0.018	0.018	0.018	0.018	0.018	<b>0.026</b>	0.018	0.018	<b>0.035</b>	0.018	0.018	0.018
Vegetation in 30m buffer	0.024	0.024	0.024	0.024	0.024	0.024	0.024	<b>0.035</b>	0.024	0.024	0.024	0.024	0.024	0.024
Straightening	-0.032	-0.032	-0.032	-0.032	-0.032	-0.032	-0.032	<b>-0.041</b>	<b>-0.047</b>	-0.032	<b>-0.050</b>	-0.032	-0.032	-0.032
Erosion	-0.025	-0.025	-0.025	-0.025	<b>-0.041</b>	<b>-0.032</b>	-0.025	<b>-0.034</b>	-0.025	-0.025	-0.025	-0.025	-0.025	-0.025
Suspended matter					<b>-0.019</b>	<b>-0.009</b>				<b>-0.010</b>				<b>-0.014</b>
Dissolved oxygen	0.042	0.042	0.042	<b>0.057</b>	<b>0.053</b>	0.042	<b>0.051</b>	0.042	0.042	0.042	<b>0.060</b>	<b>0.054</b>	0.042	0.042
BOD5	-0.037	-0.037	-0.037	-0.037	<b>-0.043</b>	<b>-0.045</b>	-0.037	-0.037	<b>-0.045</b>	-0.037	-0.037	-0.037	<b>-0.056</b>	-0.037
Ammonium	-0.059	-0.059	-0.059	-0.059	<b>-0.077</b>	-0.059	-0.059	<b>-0.074</b>	-0.059	-0.059	<b>-0.089</b>	<b>-0.073</b>	<b>-0.080</b>	-0.059
Nitrites	-0.075	-0.075	-0.075	<b>-0.099</b>	<b>-0.101</b>	-0.075	-0.075	<b>-0.092</b>	-0.075	-0.075	<b>-0.111</b>	<b>-0.103</b>	<b>-0.100</b>	-0.075
Nitrates	-0.031	-0.031	-0.031	-0.031	<b>-0.047</b>	-0.031	-0.031	<b>-0.046</b>	-0.031	-0.031	<b>-0.052</b>	-0.031	-0.031	-0.031
Total phosphorous	-0.061	-0.061	-0.061	-0.061	<b>-0.082</b>	-0.061	-0.061	<b>-0.075</b>	-0.061	-0.061	-0.061	<b>-0.077</b>	<b>-0.082</b>	-0.061

Total explained variance (R<sup>2</sup>): 45%

The I<sub>2</sub>M<sub>2</sub> response to stress variables applied at reach and site scales was studied by PLS regression. Of the two models tested, one did not consider the typology. The effects refer to all sites without distinction, while the model incorporating the typology introduces deviation by type to a global situation.

The determination coefficients ( $R^2$ ) indicated that the proportion of variance of the I<sub>2</sub>M<sub>2</sub> explained by the model increased from 38% for the non-hierarchical model to 45% for the hierarchical model (Tables 4 and 5). The inclusion of the typology in the model therefore allows explaining a larger proportion of variance of the I<sub>2</sub>M<sub>2</sub>.

In the hierarchical model, some variables are significant whereas their term of interaction with type is not. Therefore these variables describe a global effect for all the sites. These are variables such as dam density, rate of urbanization in a buffer zone of 100 meters, rate of vegetation in a buffer zone of 10 to 30 meters, rate of straightening, rate of erosion, concentration in dissolved oxygen, BOD<sub>5</sub>, and concentrations of ammonium, nitrites, nitrates and total phosphorous (Table 5). These variables are the same as in the model without typology and their relative influence on the I<sub>2</sub>M<sub>2</sub> values are similar in both models. The relative values between the coefficients are the same in both models meaning that it is always the same variables that affect the I<sub>2</sub>M<sub>2</sub> values more or less although the effects are lower in the non-hierarchical model (tables 4 and 5). The pressures with the highest impact on the biological index are physico-chemical variables which have higher coefficients in absolute values than the hydromorphological pressures in both the first and the second model.

The two types of the first group, along with type 6 have no significant interaction between any variable and type. The types of groups 2, 3, 4 and 5 are all different from the mean trend. Type 2a has few differences with the common model, contrary to type 2b which presents a higher deviation. The main effects correspond to physico-chemical variables. The two types of group 3 do not present a strong deviation in comparison to the mean trend. As with type 2b, type 4a presents a strong deviation in comparison to the mean trend. The only type that presents a greater effect for hydromorphological variables than physicochemical variables is type 4b. Dam density, urbanization, the presence of ponds in the flood plain, straightening and BOD<sub>5</sub> have effects different to those observed in the mean model. The last

type of group 4, type 4c, does not present a large difference with the mean trend model. Only over-width and suspended matter have a significant interaction term. All the types in group 5 have several differences with the mean trend model. Type 5a presents the strongest effects for the hydromorphological variables. The two other types, 5b and 5c, have higher deviations for the physico-chemical variables.

## **2.4 Discussion**

The difference of explained variance between the non-hierarchical and the hierarchical models (38% vs 45%) shows that taking into account a typology of catchments based on physiographic characteristics and driving forces provides greater explanatory power to the relation of hydromorphological and physico-chemical pressures with the I2M2.

Moreover, the results show a better fit of the hierarchical model within the types presenting coefficients significantly different from those obtained with the non-hierarchical model.

For the sake of simplicity, we find three main patterns of response:

The types for which the coefficients are not statistically different from the mean model. These types are 1a, 1b and 6.

The types for which few coefficients differ from the mean model. These are types 2a, 3b, 4c.

The types that differ markedly from the mean model with a larger number of statistically different coefficients. These are types 2b, 3a, 4a, 4b, 5a, 5b, 5c.

It appears that the types subject to the least pressure are those that differ least from the mean model. 1a and 1b are high mountain catchments with low percentages of urbanization and agriculture and with well-developed and only slightly fragmented riparian forest. The sites belonging to these types are concentrated in a small part of the total pressure gradient, preventing them to show a particular response pattern. On the other hand, group 6 is subjected to average pressures and therefore does not stand out much from the general trend.

On the contrary, the higher the pressure at catchment area level, the more the models differ from the mean non-hierarchical model (2b, 3a, 4b, 4c, 5a, 5b).

Types 2a and 2b present two different patterns of response of the macroinvertebrate index to local pressures. The I2M2 responses are stronger for the type 2B sites than for the type 2a ones. These two types mainly differ by the fact that type 2b presents more urbanized basins (0.7% of urban land cover for 2a vs. 2% for 2b), fewer natural ones (90% of natural land cover for 2a vs. 50% for 2b) and generally more impacted riparian vegetation (80% of riparian forest present for 2a vs. 58% for 2b). Its basins are therefore more altered and its communities tend to be more sensitive or less resilient to any variation in physico-chemical pressure.

Types 3a and 3b show relatively small variations from the mean trend. The urbanization levels of these two types are close to the national mean (5%, 3.3% and 3.5% of urban land cover for types 3a, 3b and national, respectively). It has been shown that urban land cover has a strong influence on macroinvertebrate community variations (Allan 2004; Villeneuve et al. 2015; Wasson et al. 2010), and thus it appears logical to observe two close response patterns. However, type 3a exhibits higher intensive agriculture catchment land cover than type 3b (57.5% vs. 22.5%), which may explain the response of corresponding sites to higher erosion, a higher concentration of suspended matter and an increase of BOD5, probably due to the lack of water treatment in rural territories.

The gradients of driving forces for type 4a are similar to those for type 2b. The effects of local pressures observed on macroinvertebrates reinforce the mean trend effects (as the positive/negative ones become more positive/negative, respectively). As observed for type 2b, the catchments are urbanized (1.7%); they have on average 15% of intensive agriculture land cover and 60% of riparian cover. Type 4b presents more agricultural land cover (either intensive or extensive) and both less covering and more fragmented riparian cover. This exacerbates the impacts of local hydromorphological alterations (dam density, straightening, urbanization in 100m buffer zones and pond rates in floodplains). Type 4c shows high percentages of urban and intensive agricultural land cover, as well as even more degraded riparian vegetation. Therefore there are few contrasted conditions in its catchments, and the relationship between the I2M2 and local pressures does not vary from the mean trend.

The type 5a shows a high level of urban land cover, associated with a high percentage of intensive agricultural land cover. The riparian vegetation does not cover a large surface area of the corridor and is also very fragmented. Type 5b is less urbanized but intensive agriculture covers a larger amount of the catchment area. Its riparian vegetation appears less altered. Type 5c tends to be more natural than the two other types with less agricultural cover. Consequently, type 5a exhibits stronger responses to hydromorphological pressure variables, while the three other types appear to be strongly affected by physico-chemical variations. Therefore, it appears that hydromorphological alterations can play a primary role in determining invertebrates communities, while physico-chemical pressures drive the I<sub>2</sub>M<sub>2</sub> answers in spaces where hydromorphology is less impacted.

The comparison of the benthic macroinvertebrate response in basins exhibiting similar driving forces does not allow identifying a general trend. Therefore the study of catchment characteristics alone seems to be insufficient to allow understanding how invertebrate assemblages are impaired by pressures. Differences can be explained by natural characteristics (i.e. geology, climate) represented by hydro-ecoregions. There may also be an interaction effect of natural conditions on organism response to pressures. Although physiographic data are not often taken into account in analyses, it seems important to incorporate them, as was done in (Marzin et al. 2013).

Whatever the model we consider (either hierarchical or not), macroinvertebrate communities are impaired similarly by the same pressure variables. Some of them have a positive effect, such as the presence and the continuity of riparian vegetation in the 10 and 30 meter buffer zones and the level of dissolved oxygen concentration. (Dosskey et al. 2010) demonstrated the positive role of the riparian forest in improving the chemical water quality, for example by neutralizing nitrogen in the river. Moreover, the shade effect of riparian forests also plays an important role in preventing water temperatures from increasing (Abell and Allan 2002), thus maximizing the amount of dissolved oxygen. Oxygen is obviously an essential chemical element to aquatic life. Conversely, hypoxia can cause severe biological impoverishment in the environment (Lim et al. 2006). Certain other variables exhibited an overall negative influence on the I<sub>2</sub>M<sub>2</sub> score, especially BOD<sub>5</sub>, ammonium, nitrite, nitrate and total phosphorus concentrations. These results are

consistent with those of (Dahm et al. 2013; Donohue et al. 2006; Hering et al. 2006; Hering et al. 2004; Johnson and Hering 2009; Villeneuve et al. 2015). Ecotoxicological studies such as those of (Camargo et al. 2005) have shown that high nitrogen concentrations can be toxic to organisms. Furthermore, their accumulation in the environment can indirectly induce a decrease of available oxygen in streams. Indeed, high nitrogen and phosphorus concentrations mainly increase the biomass of primary producers that are also oxygen consumers (Biggs et al. 2000; Dodds and Welch 2000). Moreover, the final degradation of this organic matter increases the oxygen deficit (Villeneuve et al. 2015).

In addition to physico-chemical parameters, biotic assemblages are also impaired by hydromorphological alterations that include variations of habitat homogenization and hydraulic conditions, in turn leading to non-optimal or unfavorable environmental conditions.

Biotic indices respond in different ways to changes in solid and liquid flows. Regarding the change of morphological processes, the  $I_2M_2$  is negatively influenced by variables describing loss of sinuosity, increased bank erosion, flow reduction and an alteration of pool/riffle succession (straightening, urbanization of the 100m buffer, dikes and roads in the riverbed and flood plain, dam density). This can be understood insofar as they are factors that act either on the diversity of the meso-scale habitat pattern, especially in riparian zones (straightening, slower flow rate, dykes, roads), or on the dissemination process (change of pool/riffle successions, change of bank erosion).

The  $I_2M_2$  index is generally stronger impaired by physico-chemical pressures (nutrient concentrations) than by hydromorphological alterations. Macroinvertebrates seem to be more sensitive to factors maintaining water quality, specifically those linked to eutrophication (total nitrogen and phosphorous concentrations), than to hydromorphological pressures acting on flow and habitat diversity in streams. This trend is not specific to a given geographic area but seems to have a positive effect on macroinvertebrates since it is redundant in both models, as noted in the studies of (Dahm et al. 2013; Johnson and Hering 2009; Villeneuve et al. 2015), who also observed higher organism sensitivity to physico-chemical parameters than to hydromorphology, although they focused on macroinvertebrates, fish and diatoms. Regarding this point, our hypothesis



was that taking into account the typology of streams as a hierarchical parameter in the model would provide us with better understanding of the relative role of hydromorphological alterations. Our results do not differ from those of works published previously, therefore reinforcing the hypothesis that hydromorphological alterations probably play a minor direct role on the biological quality of streams when compared to physico-chemical pressures. Nevertheless, hydromorphology could play an interactive role along with physico-chemical pressures, which was not taken into account in these models. For example, the hypothesis of invertebrate communities better resisting to physico-chemical pressures in the less hydromorphologically altered context could be tested using a more detailed hierarchical model.

## **2.5 Conclusion**

Introducing hierarchical factors in multi-stress models can lead to slight variations in responses of organisms to pressures according to their natural characteristics and their driving forces acting at catchment scale. These results enhance the information previously provided by (Marzin et al. 2013; Villeneuve et al. 2015).

Better understanding of macroinvertebrate response to human pressures in well-defined river types, (i.e. specific geographic areas with specific natural conditions and catchment-scale driving forces) was provided.

Taking into account regional geographical and driving force patterns improves the capacity to distinguish the responses of biological quality elements to local environmental pressures. Consequently, pressures-state relationship studies should integrate the links provided by hierarchical scales. Moreover, it would be interesting to differentiate reach and site scales, and apply an analytical method able to quantify the effects of the various pressures and natural characteristics on various biological compartments according to the spatial scale in which they evolve.

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## **Chapitre 6 - Effets directs et indirects de pressions multiples sur la faune des cours d'eau à travers les échelles bassin versant, tronçon et station : mise en évidence du rôle de l'hydromorphologie.**

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## 1 Synthèse de l'article

Dans un contexte de nature anthropisée, les activités humaines ont altéré, et continuent d'altérer, les capacités du système à s'équilibrer, tant sur les aspects de dynamique physique qu'écologique. Si les forces motrices et pressions qui s'exercent sur les cours d'eau sont nombreuses, Stendera et al. (2012) ont identifié dans leur revue bibliographique de 2012 que l'occupation du sol, l'eutrophisation et la destruction d'habitat était clairement identifiées comme les stressseurs les plus impactants.

De même que pour le fonctionnement du cours d'eau, les pressions anthropiques s'organisent selon une hiérarchie d'échelles emboîtées décrite par Frissel et al. en 1986 et pèsent sur les différents compartiments physiques et biologiques de ces systèmes (Allan, 2004) :

- l'occupation des sols et les grands items d'activités humaines (agriculture, urbanisation, industries) vont influencer les caractéristiques physico-chimiques, hydromorphologiques et hydrologiques des cours d'eau (altération du transport sédimentaire, enrichissement en nutriments, pollution toxique, altérations hydrologiques, altération de la ripisylve, perte d'habitats),

- la gestion des débits pour la prévention des crues, l'hydroélectricité ou l'irrigation va modifier les régimes hydrauliques, avec des conséquences possibles sur l'hydromorphologie (rupture de la continuité sédimentaire, incision) et thermiques (réchauffement des eaux) qui vont à leur tour modifier les conditions locales d'habitat *sensu lato*,

- les modifications locales de géométrie du cours d'eau (redimensionnements, recalibrages, protections de berge) vont également modifier les habitats et les processus biogéochimiques (ruptures de connectivités latérales, perte de connexions avec les milieux terrestres associés et perte de leur action épuratrice),

- les rejets ponctuels ou diffus de substances potentiellement toxiques vont avoir un effet toxique direct sur la faune aquatique et un effet sur l'équilibre trophique du site.

Mesurer l'impact relatif des différentes pressions et altérations sur les communautés biologiques aquatiques ne peut pas s'envisager sans prendre en compte le fonctionnement



hiérarchique des cours d'eau, de façon systémique. Les cours d'eau sont des systèmes ouverts, et la nature et le fonctionnement des biocénoses qu'ils abritent sont étroitement liés aux activités humaines. Comme dans tout écosystème, les processus fonctionnels interagissent à des échelles multiples dans l'espace et dans le temps (Lévêque 2001). Ainsi, le rôle de collecteur d'un bassin versant fait que toute altération des écosystèmes terrestres a des répercussions potentielles, à grande distance et à long terme, sur les flux d'eau et d'éléments qui alimentent les rivières et donc sur les écosystèmes aquatiques situés à l'aval. La structure linéaire des cours d'eau les rend particulièrement vulnérables aux structures transversales qui interrompent les flux, mais aussi aux contraintes longitudinales exercées par leur environnement (Wasson et al. 1993).

Il est depuis longtemps reconnu que le fonctionnement d'un cours d'eau s'organise selon une hiérarchie d'échelles (Frissell et al. 1986) depuis l'échelle régionale jusqu'à l'échelle du microhabitat, et que les processus et structures observées aux échelles les plus larges influencent les processus et structures observés aux échelles inférieures. Ce sont donc des systèmes dynamiques complexes, résultant de l'adaptation en continu de compartiments sans cesse en interaction :

- les contextes géologique et géographique couplés aux régimes climatique et hydrologique déterminent les caractéristiques hydromorphologiques,
- les caractéristiques hydromorphologiques, hydrauliques et thermiques déterminent les habitats disponibles,
- les conditions d'habitat et le pool potentiel d'espèces disponibles, leurs dynamiques individuelles, spécifiques et d'interaction vont déterminer la communauté d'organismes présents, tant dans sa composition que dans sa structure.

A cette vision schématique s'ajoute la variabilité naturelle du milieu (régimes thermiques et hydrologiques interannuels) à laquelle les communautés biologiques sont naturellement adaptées.

L'approche développée dans ce chapitre propose de prendre en compte ces connaissances sur l'organisation des échelles et sur les liens entre pressions et état écologique en construisant et en analysant un modèle structurel basé sur la méthode PLS

path modelling (Jakobowicz 2007; Tenenhaus et al. 2005; Vinzi 2008; Wold 1982) qui permet de relier entre elles des variables latentes correspondant aux pressions d'occupation du sol, hydromorphologiques et physico-chimiques et d'analyser in fine leur effet sur l'état écologique mesuré ici à travers l'indice biologique basé sur les macro-invertébrés : l'I2M2.

1. Les liens indirects entre pressions et état écologique modifient-ils la hiérarchie des pressions ?

2. Les différents niveaux d'échelles emboîtées jouent-ils un rôle différent sur la relation pressions-état écologique ?

3. Ce modèle permet-il de mieux comprendre le rôle particulier de l'hydromorphologie sur l'état écologique ?

La méthode de modélisation utilisée, l'approche PLS (ou « PLS path modelling ») nous a permis de développer un modèle prenant réellement en compte l'organisation hiérarchique en trois échelles emboîtées (bassin versant, tronçon et site de mesure) et d'évaluer les effets directs et les effets indirects des différentes variables de pressions entre elles et sur les assemblages de macroinvertébrés. Les méthodes plus classiques utilisées précédemment (Dahm et al. 2013; Feld 2013; Villeneuve et al. 2015) ne permettaient pas de prendre en compte la complexité des relations étudiées ici.

Nous avons pu mettre en évidence que l'ordre d'impact des pressions pouvait différer en fonction du type de cours d'eau considéré. Ainsi, dans les systèmes d'eau calcaire, l'hydromorphologie et l'occupation du sol ont un impact considérable sur les assemblages de macroinvertébrés, parfois plus élevé que la physico-chimie alors que dans les eaux non calcaires, la physico-chimie est le premier facteur d'impact sur les petits cours d'eau.

Les modèles de la littérature montrent des profils de réponse avec un impact prépondérant des paramètres physico-chimiques (nutriments et matière organique) mesurés à l'échelle du site, suivis en deuxième position par l'impact de l'occupation du sol à l'échelle du bassin versant et en troisième position, par les facteurs d'altération hydromorphologiques avec un effet beaucoup plus difficile à mettre en évidence (Dahm et al. 2013; Hering et al. 2006; Villeneuve et al. 2015).

Les résultats de notre étude sont cohérents avec les travaux précédents montrant que les caractéristiques physico-chimiques et de la mosaïque d'habitats ont un effet prépondérant qui s'exerce directement à l'échelle du site. Par contre l'analyse des effets indirects liés à l'occupation du sol montre clairement qu'exceptés pour les petits cours d'eau non calcaires, l'effet de l'occupation du sol est plus important que celui des nutriments et de la matière organique.

L'occupation du sol aussi bien artificielle qu'agricole est un facteur d'impact direct important pour la biologie et de nombreux auteurs ont montré l'existence de ce lien (Allan 2004; Clapcott et al. 2012; Dahm et al. 2013; Marzin et al. 2012; Marzin et al. 2013; Villeneuve et al. 2015). Ce résultat apparaît encore plus clairement dans nos modèles du fait de la séparation des effets directs et indirects ce qui n'avait pas été fait jusqu'à présent.

Le fait que les macroinvertébrés benthiques répondent aux altérations hydromorphologiques est déjà rapporté dans de nombreuses études précédentes (Buffagni et al. 2004; Hering et al. 2004; Lorenz et al. 2004). Cependant, d'autres études (Dahm et al. 2013; Hering et al. 2006; Villeneuve et al. 2015) ont montré que l'hydromorphologie n'intervenait qu'en troisième position après les effets de la physico-chimie et de l'occupation du sol avec la conclusion que les effets liés aux altérations hydromorphologiques avaient un impact relativement faible sur le compartiment biologique des cours d'eau.

La structure hiérarchique de notre modèle nous a permis de réévaluer l'ordre de grandeur de l'impact de la pression hydromorphologique sur les macroinvertébrés en prenant en compte plusieurs niveaux d'échelle de mesure du risque d'altération hydromorphologique (bassin et tronçon) et en analysant les effets indirect de la pression hydromorphologique via la physico-chimie et l'habitat.

La somme des contributions de la pression hydromorphologique aux échelles bassin et tronçon correspond à la contribution de l'occupation du sol à la fois pour les effets directs et indirects. Ceci montre qu'une meilleure description de la pression hydromorphologique permet de mieux mettre en évidence ses effets sur les assemblages de macroinvertébrés.

Enfin, nous avons pu démontrer que l'hydromorphologie avait un effet important sur le compartiment nutriments et matière organique. Ceci implique notamment que l'effet total

de l'hydromorphologie sur les macroinvertébrés est plus fort car la prise en compte de cet effet indirect augmente considérablement la proportion de la variance de l'I2M2 expliquée par la pression hydromorphologique.

En conclusion, nous avons pu montrer :

- l'effet important de l'occupation du sol sur l'hydromorphologie et sur la physico-chimie et l'effet indirect qui en découle sur les assemblages de macroinvertébrés ;
- l'effet de l'hydromorphologie sur la mosaïque de substrats, les nutriments et la matière organique justifiant l'effet indirect majeur de l'hydromorphologie sur les assemblages de macroinvertébrés ;
- les nutriments et la matière organique ont un effet plus faible qu'attendu en regard des résultats précédents car la prise en compte des effets indirects montre qu'ils sont en grande partie la résultante de l'effet des forces motrices qui dirigent le système.

## **2 Direct and indirect effects of multiples stressors on stream fauna across watershed, reach and site scale: a path modelling analysis revealing the role of hydromorphology.**

### **Abstract**

The purpose of our approach was to take account of the nested scales of stream functioning on the links between pressures and ecological status by building and analysing a hierarchical model. The development of this model aimed at answering the following questions: Does the indirect links between pressures and ecological status modify the impact hierarchy of pressures? Do the different nested scales play a different role in the pressures-ecological status relationship? Does the model lead to better understanding of the specific role of hydromorphology in ecological status evaluation?

To achieve that goal, we used the PLS path modelling method for developing a structural model linking the latent variables of (i) land use, and hydromorphological alterations at watershed scale, (ii) hydromorphological alterations at reach scale, (iii) nutrients-organic matter contamination levels, and (iv) substrate samples to explain macroinvertebrate based index  $I_2M_2$  value variation.

As a first important result, we have highlighted the importance of land use effect exerted on both hydromorphology and physico-chemistry and their translation as an indirect effect on biological condition of streams. We have also demonstrated that hydromorphological alterations had an effect on substrates structure and nutrients and organic matter concentrations. This result implies that hydromorphology has a major indirect effect on macroinvertebrates and that the ranking of pressures previously described in literature is now modified. And finally, the nutrients and organic matter effect on macroinvertebrates remains lower than expected since we take into account all the indirect effects of land use and hydromorphological alterations. A large part of physico-chemistry variance is the result of driving forces expressed at higher scales (watershed and reach).

Aquatic ecosystems have long been affected by the consequences of anthropic activities whatever the scale, and the corresponding impairments, both direct and indirect, are various. We have highlighted the importance of the site scale (physico-chemical and substrates mosaic variables) for explaining the biological condition of streams. However, these results have also shown the important role of the variables measured at the reach scale, both directly via their contribution to habitat feature modifications, and indirectly via their confounding effect on the relationship between physico-chemistry and macroinvertebrates. Lastly, we have also shown that the variables measured on the scale of the watershed (in this case: land use and large scale hydromorphology) integrate all types of pressure acting at lower scales located on the upstream watershed and play a strong structural role through their indirect effects on all the pressure characteristics.

## **2.1 Introduction**

### **2.1.1 Anthropisation and ranking pressures**

Against a background of nature subjected to increasing anthropisation, human activities have disturbed and continue to alter the capacities of ecosystems to balance themselves, with respect to their physical and ecological dynamics. Although streams are subject to a larger number of significant driving forces and pressures, in their bibliographical review of 2012, (Stendera, Adrian et al. 2012) showed that land use, eutrophication and habitat destruction were clearly identified as the stressors with the greatest impacts.

Moreover, in the stream functioning context, anthropic pressures can be organised hierarchically as a function of several nested scales described by (Frissell, Liss et al. 1986) and have serious impacts on the different physical and biological compartments of streams (Allan 2004):

- land use and large sectors of dense human activity (e.g. agriculture, urbanization, industries) deeply influence the physico-chemical, hydromorphological and hydrological characteristics of streams (e.g. via modifications in sediment transport, nutrient enrichment, toxic pollution, hydrological modifications, riparian clearing or habitat loss) (Paul and Meyer 2001, Novotny, Bedoya et al. 2009);
- flow rate management for flood prevention, hydroelectricity production and irrigation modify hydraulic regimes, with possible hydromorphological (e.g. breaks in sediment continuity, incision) and thermal (warming of water) drawbacks which in turn modify local habitat conditions (Poff, Allan et al. 1997, Villeneuve, Souchon et al. 2015, Verdonschot, Kail et al. 2016);
- local modifications of stream geometry (e.g. stream re-dimensioning, re-gauging, bank protection) also modify habitats and biogeochemical processes (e.g. disruptions of lateral connectivity, loss of connection with neighbouring terrestrial habitats and loss of effective cleaning action) (Baker, Bledsoe et al. 2012, Weigelhofer, Welti et al. 2013);
- point and diffuse discharges of toxic substances potentially impair stream water quality and resident biotic communities (Novotny 2004, Archambault, Usseglio Polatera et al. 2010).

### **2.1.2 Impacts of human activities and regulations**

The management of streams in conformity with the Water Framework Directive (WFD) in force in Europe does not only involve the observation of ecological status degradations of water bodies. It obliges governments to diagnose the impacts of human activities on water bodies to provide guidelines to aquatic ecosystem managers for the design of efficient restoration strategies at the scale of coherent management units (e.g. watershed and reach scales for streams). In terms of pressure the WFD demands “the identification of the significant anthropogenic pressures and the evaluation of the risk incurred if the WFD environmental objectives are not complied with”. In other words, it is necessary to identify and hierarchize the causes of physical, biogeochemical and ecological alterations of the functional processes in aquatic ecosystems.

### **2.1.3 Evaluation of existing models and their shortcomings**

In this multi-scale context, several studies have focused on how relating the biological or ecological status of rivers to cocktails of pressures. Land use, hydromorphology alterations and physico-chemical changes have already been linked individually to biotic indices based on assemblages of macrophytes, diatoms, fish and macroinvertebrates (Sponseller, Benfield et al. 2001, Wasson, Villeneuve et al. 2010, Marzin, Archambault et al. 2012, Dahm, Hering et al. 2013, Feld 2013, Sundermann, Gerhardt et al. 2013, Villeneuve, Souchon et al. 2015). These studies showed that some links exist between each of these pressure types and biological indices. They have also demonstrated that the links between pressures and biological indices are influenced by the scale on which the pressure is taken into account (Roth, Allan et al. 1996, Allan, Erickson et al. 1997, Lammert and Allan 1999, Allan 2004, King, Baker et al. 2005). Thus, watershed, hydromorphological reach and riparian corridor are scales that considerably structure the effect of pressures on the ecological status of river sites (Wasson, Villeneuve et al. 2010, Marzin, Archambault et al. 2012).

The results of these studies can be shortly summarised as follows: (i) the biotic indices respond to environmental variables characterizing nutrients, organic matter, land use in the watershed and hydromorphological pressures; (ii) the response profile of all these indices is similar but specificities can be observed with differentiated responses mainly regarding hydromorphology. However, in every case hydromorphological alteration related factors are

ranked in third place after both physico-chemical and land-use factors. Both (Dahm, Hering et al. 2013) and (Villeneuve, Souchon et al. 2015) showed that the effects of hydromorphological pressures on biotic assemblages could be measured but that their impact on the set of biological metrics considered in these studies was relatively low.

#### **2.1.4 Watershed and stream**

These existing models showed that it is possible to predict and clearly explain the ecological status on the basis of pressure descriptors. But these models took little account of the multiple scales that structure both the pressures and the longitudinal functioning of streams. In addition, the alteration of the ecological status is the result of several simultaneously co-occurring pressures: e.g. punctual discharges, diffuse pollutions, alterations of sediment and water flows, impacts of various artificial structures and breaks in river connectivity (Borchardt and Richter 2003). According to the DPSIR concept (Driving forces, Pressures, State, Impact and Response) (Kristensen 2004), human activities (agriculture, urbanization) generate combined pressures (chemical discharges, physical alterations) and create driving forces that alter the abiotic components of the ecosystem (physico-chemistry, hydromorphology). These alterations then affect biological communities and thus the ecological status.

Measuring the impacts of the different pressures and alterations on local stream biological communities cannot be considered without systematically taking into account how the stream functions. Streams are open systems, and the nature and functioning of the biocenoses they host are closely linked to human activities. Like in any ecosystem, functional processes interact at multiple scales in space and time (Hynes 1975, Lévêque 2001). Thus the collecting role of a watershed means that any alteration of neighbouring terrestrial ecosystems has potential impacts, over a longer distance and at long-term, on water flows and fluxes of elements feeding the stream, and thus on the aquatic ecosystems located downstream. The linear structure of streams makes them particularly vulnerable to both transversal structures that disrupt water flow, and longitudinal constraints brought to bear by the environment (Wasson, Bethemont et al. 1993).

It has long been recognized that (i) stream functioning is organised according to a hierarchy of scales (Frissell, Liss et al. 1986, Roth, Allan et al. 1996, Poff, Allan et al. 1997,



Allan 2004, Thorp, Thoms et al. 2006, Thorp 2014) from the regional scale to the microhabitat scale, and (ii) the processes and structures observed at the largest scales influence the processes and structures observed at smaller ones. These are therefore complex dynamic systems, resulting from continuous adaptation of compartments in constant interaction; e.g.:

- the geological and geographic contexts coupled with climatic and hydrological regimes determine hydromorphological characteristics;
- hydromorphological, hydraulic and thermal characteristics determine available habitats, *sensu lato*
- habitat conditions and the potential pool of colonists, their specific dynamics and species interactions will determine the local species assemblage, in terms of both composition and structure.

In addition to this schematic view, account must be taken of the habitat natural variability (inter-annual thermal and hydrological regimes) to which biological communities are naturally adapted.

### 2.1.5 Objectives of the study

The purpose of our approach was to take into account the nested scale organisation and the links between anthropogenic pressures and river ecological status by building - and analysing the results of - a model based on the PLS path modelling method (Wold 1982, Tenenhaus, Vinzi et al. 2005, Jakobowicz 2007). This method can be used to simultaneously analyse the effects of latent variables corresponding to land-use, hydromorphological and physico-chemical pressures on the ecological status of rivers, synthetically measured in this study by the macroinvertebrate-based French biotic index for wadeable rivers ( $I_2M_2$ ; (Mondy, Villeneuve et al. 2012)), as an example of one of the multimetric index already in use (see (Birk, Bonne et al. 2012) for a review)

The development of this model should allow us to answer the following questions:

Does the indirect links between pressures and ecological status modify the impact hierarchy of pressures?

Do the different nested scales play a different role in the pressures-ecological status relationship?

Does the model lead to better understanding of the specific role of hydromorphology in ecological status evaluation?

## **2.2 Materials and methods**

### **2.2.1 Typological classification of sites**

The concept of “ecoregion” was adapted to aquatic ecosystems by (Omernik 1987) to define water quality and regionalised management objectives and has been widely used in the USA (Omernik 2004). The concept of hydro-ecoregions (HER) developed in France by (Wasson, Chandesris et al. 2002) is based on a top-down approach whose basic principle relies on global control factors for classification rather than their consequences at local level. On the watershed scale, the universally recognized primary determinants of stream ecological functioning are geology, relief and climate. This concept based on the theories of hydrosystem hierarchical control, in particular on nested physical scales from the watershed to the micro-habitat (Frissell, Liss et al. 1986). (Wasson, Villeneuve et al. 2010), has demonstrated that hydro-ecoregions has the capacity to provide typological support for better analyzing the causal links between land use and benthic macroinvertebrates.

For this study, we chose to focus on the hydro-ecoregions of lowlands and hills (altitude lower than 450 m) we have grouped into two categories on the basis of their geology: i.e. ‘limestone’ and ‘non-limestone’. Conversely, we have eliminated the hydro-ecoregions liable to give rise to non-generalizable constraints or to types of over-typical populations such as mountain and Mediterranean regions.

Each of these two groups was then divided into two size-based sub-groups to analyse separately small streams (ranks 1 to 3) and medium-sized streams (ranks 4 to 6).

### **2.2.2 Biological data**

We used benthic macroinvertebrate data from a database of 1200 sites of the French monitoring network achieved during the 2007-2012 period (figure 1).

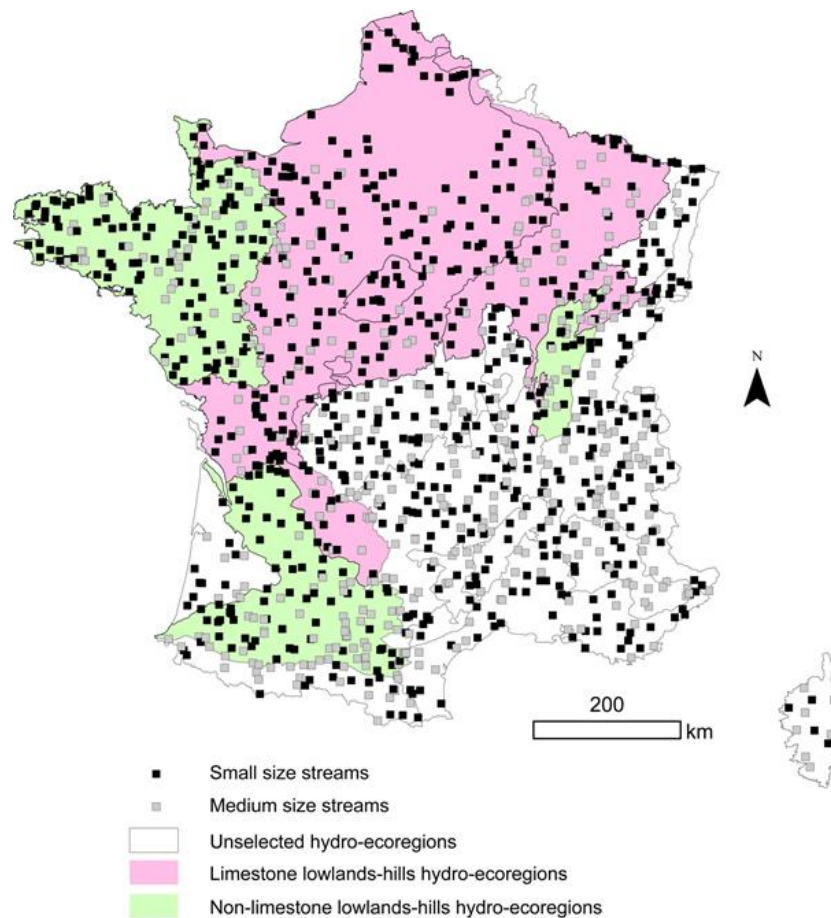


Figure 1. Spatial distribution of the 1200 sites of the French control and monitoring network involved in this study.

This database made it possible to select four groups of sites. The first group represented small non-limestone streams and consisted of 638 samples performed on 160 sites. The second group represented medium-sized non-limestone streams and included 492 samples from 127 sites. The third group represented small limestone streams and was composed of 817 samples from 228 sites. The last group represented medium-sized limestone streams and had 460 samples from 128 sites.

The benthic macroinvertebrate data were collected according to a standardized common protocol (AFNOR 2009, 2010). The biological index derived from these faunal lists was the  $I_2M_2$  (Mondy, Villeneuve et al. 2012).

### **2.2.3 Land use data**

The layer of geographic data used was CORINE Land Cover (CLC) 2006 (Büttner and Kosztra 2007). It covers the entire territory and is based on a standardized nomenclature ranked according to three levels and 44 items distributed in five main types of territorial cover (urban, agricultural land, forests and semi-natural habitats, wetlands and bodies of water).

On the basis of the catchment basin delimited beforehand by a digital field model, we calculated for each stream site the percentage of the surface of four category groups of land use based on CORINE Land Cover nomenclature: urbanization of the watershed (gathering urban areas, industrial and commercial areas, roads and highways, mines, dumps and construction sites, non-agricultural artificial green spaces), intensive agriculture (arable lands, permanent crops, orchards, vineyards, annual crops associated with permanent crops, cropping and field systems), non-intensive agriculture (meadows, territories mainly occupied by agriculture with considerable natural vegetation, agro-forestry territories) and non-anthropized spaces (forests and semi-natural habitats, wetlands, ponds).

### **2.2.4 Hydromorphological data**

Hydromorphological data corresponded to descriptors of the pressures exerted on the hydromorphological functioning of streams. Two main types of descriptors were used. On the watershed scale, we used information on drainage, irrigation, erosion and ponds to characterize possible disturbance of natural sediment and water flows. On the hydromorphological reach scale, we used several descriptors of hydromorphological alterations risk including the rate of dams, the rate of stream crossings, the rate of roads in the stream vicinity, the rate of roads in the flood plain, the rate of dikes in the stream vicinity, the rate of dikes in the flood plain, the rate of "wind break" afforestation on the banks, the rate of riparian forest on the banks, the rate of afforestation in the stream vicinity, the rate of "artificial" land use close to the stream vicinity, the rate of straight channel, the rate of surplus width and the depth/width ratio. These variables were assessed for each of the selected sites from the database at the relevant scales, watershed or reach.

### **2.2.5 Physico-chemical data**

Information on the physico-chemical characteristics of sites were obtained from the physico-chemical survey of these sites belonging to the National monitoring network. We selected the following parameters: ammonium, nitrites, nitrates, total phosphorous concentrations in water and BOD5.

These parameters were measured monthly. We chose as indicators of the nutrient and organic matter pressures, the mean concentrations of each of these elements during the period covering the 12 months prior to the date of macroinvertebrate assemblage sampling.

### **2.2.6 Substrates mosaïc**

The percentages of substrate coverage at the sampling sites were used to qualify the local habitats of the stations. The twelve pre-defined substrate types considered (AFNOR, 2009) were bryophytes, hydrophytes, litter, branches and root mats, pebbles and stones, rocks, gravel, helophytes, mud, sand and silt, algae and uniform or artificial area.

Table 1. Description of pressure variables.

Scale	Variable	Average values (min-max range)	Source
<b>Watershed</b>	urbanization of catchment: urbanized zones, industrial and commercial zones, transport networks, mines, dumps and construction sites, artificial green spaces, non agricultural spaces	3.3 (0-58.7)	Corine Land Cover (CLC) 2006
<b>Watershed</b>	intensive agriculture: arable land, permanent crops, orchards, vineyards, olive orchards, annual crops linked to permanent crops, crop and field systems	35.5 (0-100)	Corine Land Cover (CLC) 2006
<b>Watershed</b>	non-intensive agriculture: meadows, land mainly devoted to agriculture with considerable presence of natural vegetation, agro-forestry land	20.3 (0-89.9)	Corine Land Cover (CLC) 2006
<b>Watershed</b>	non anthropized space: forests and semi-natural environments, wetlands, ponds	40.8 (0-100)	Corine Land Cover (CLC) 2006
<b>Watershed</b>	Erosion	0.6 (0-3.4)	aléa érosion des sols INRA (Montier, Le Bissonais et al. 1998)
<b>Watershed</b>	Drainage ratio	0.03 (0-1)	Recensement général agricole (RGA) 2000
<b>Watershed</b>	Irrigation ratio	0.03 (0-0.47)	Recensement général agricole (RGA) 2000
<b>Watershed</b>	Ponds storage	0.03 (0-2.24)	Recensement général agricole (RGA) 2000
<b>Reach</b>	Rate of roads/railways near the riverbed	7.8 (0-262)	BD Topo IGN
<b>Reach</b>	Rate of roads/railways in the flood plain	111 (0-3087)	BD Topo IGN
<b>Reach</b>	Rate of dykes in the riverbed	5 (0-393)	BD Topo IGN
<b>Reach</b>	Rate of dikes in the flood plain	17 (0-764)	BD Topo IGN
<b>Reach</b>	Rate of stream crossings	0.8 (0-8.5)	BD Topo IGN
<b>Reach</b>	Rate of trees on banks "screen of trees"	77 (0-100)	BD Topo IGN
<b>Reach</b>	Rate of vegetation on banks "riparian vegetation"	59 (0-100)	BD Topo IGN
<b>Reach</b>	Rate of vegetation in riverbed	48 (0-100)	BD Topo IGN
<b>Reach</b>	Rate of "artificial" type land use near riverbed	9.8 (0-100)	BD Topo IGN
<b>Reach</b>	Rate of straightness	27.8(0-100)	BD Topo IGN
<b>Reach</b>	Rate of ponds in riverbed	0.7 (0-33)	BD Topo IGN
<b>Reach</b>	Overwidth	41.4 (0-15742)	BD Topo IGN
<b>Reach</b>	Anomaly index of Width/Depth ratio	3 (1-5)	BD Topo IGN
<b>Site</b>	Suspended matter (mg/L)	20.5 (1.5-1347.0)	RCS 2007-2012
<b>Site</b>	Dissolved oxygen (mg/L)	10.1 (6.1-13.0)	RCS 2007-2012
<b>Site</b>	BOD5 at 20°C (mg/L)	1.6 (0.5-4.7)	RCS 2007-2012
<b>Site</b>	Ammonium (mg/L)	0.08 (0-0.47)	RCS 2007-2012
<b>Site</b>	Nitrites (mg/L)	0.01 (0.01-0.98)	RCS 2007-2012
<b>Site</b>	Nitrates (mg/L)	13.8 (0.3-70.4)	RCS 2007-2012
<b>Site</b>	Total Phosphorous (mg/L)	0.08 (0.01-0.48)	RCS 2007-2012
<b>Site</b>	Substrates mosaïc : bryophytes, hydrophytes, litter, branches and root mats, pebbles and stones, rocks, gravel, helophytes, mud, sand and silt, algae and uniform or artificial area	-	RCS 2007-2012

### 2.2.7 Modelling method: the PLS approach

The PLS path modelling method (Wold 1982) is a statistical method used to model complex relations between observed and latent variables. This method have been used by (Riseng, Wiley et al. 2011) in a recent studies using comparable data. This type of model is generally known as a latent variable structural equation. In recent years, this approach has become increasingly popular in very different scientific communities (Vinzi 2008). Structural Equation Models comprise a large number of statistical methodologies (including the PLS approach) used to estimate complex causal relations between latent variables that are measured by observed variables called manifest variables.

A structural PLS model is described by two sub-models: (1) the measurement model (or external model) linking the manifest (observed) variables to the latent variables associated with them; (2) the structural model (or internal model), linking the latent or endogenous variables to other latent variables.

We first estimated the latent variables with the external model. The external estimation  $Y_j$  of latent variable  $\xi_j$  was built as a linear combination of manifest variables  $x_{jh}$ :

$$Y_j = \sum_h w_{jh} x_{jh}$$

where  $w_j$  was the column vector of coefficients  $w_{jh}$ . Variable  $Y_j$  was forced by standardization. (Tenenhaus, Vinzi et al. 2005).

Then the internal estimation  $Z_j$  of the latent variables was performed using external estimations  $Y_i$  of latent variables  $\xi_i$  linked to  $\xi_j$ :

$$Z_j \propto \sum_i e_{ji} Y_i$$

Where the  $\propto$  sign means that the variable positioned to the left of this sign is obtained by reducing the variable positioned on the right (Tenenhaus, Vinzi et al. 2005).

These two steps were repeated until reaching convergence, then we estimated the coefficients  $e_{ji}$  of the model called path coefficients by PLS regressions. The null hypothesis of the nullity of these coefficients was then tested using the bootstrap method for resampling. All the coefficients presented in this work were significant.

The advantage of this method is that it can evaluate the direct and indirect effects of a latent variable A on another latent variable B (Sanchez 2013). The direct effect is given by the path coefficient between the two variables A and B. The indirect effect corresponds to the influence of a latent variable A on a latent variable B by using an indirect path via a third variable C. The indirect effect is calculated as the product of path coefficients between variables A and C and between variables C and B. The total effect of a latent variable A on a latent variable B corresponds to the sum of the direct and indirect effects.

For an easier interpretation of the structural model, the direct and total effects were calculated and converted into percentages to better represent the contribution of the latent variable to the variance explained by the model.

The details of the algorithm developed by Wold (1982) is available in many works such as those of (Tenenhaus, Vinzi et al. 2005, Jakobowicz 2007).

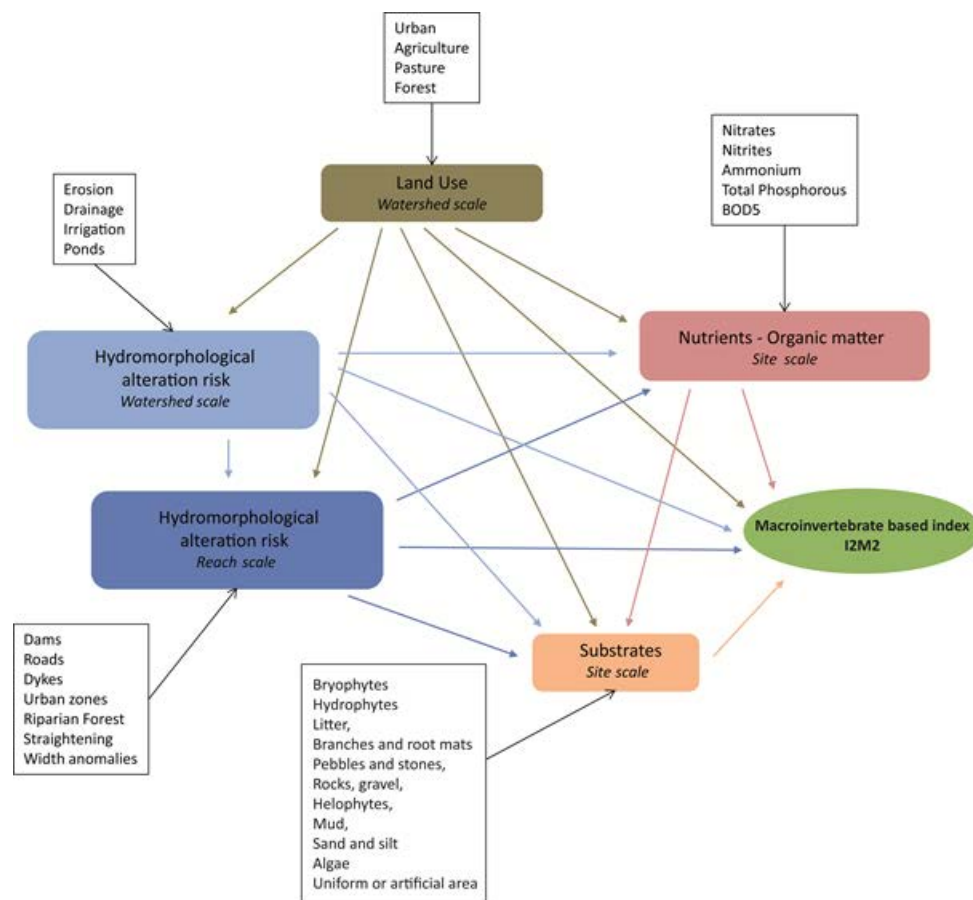
The structural model was evaluated on the basis of the predictive pertinence of the latent variables. It is advisable to analyse the multiple  $R^2$ . According (Croutsche 2002), three thresholds of multiple  $R^2$  can be taken into account. The model is significant if  $R^2$  is higher than 0.1. If it is between 0.05 and 0.1, then the model is tangent. If it is lower than 0.05, then the model is not significant.

Lastly, each structural equation was evaluated by Stone-Geisser's coefficient  $Q^2$ , also called cross-validated redundancy index by Tenenhaus et al. (2005). It comprises a cross-validated  $R^2$  test between the manifest variables of an endogenous latent variable and all the manifest variables associated with the latent variables explaining the endogenous latent variable, using the structural model estimated. If the  $Q^2$  value is positive, then the model presents predictive validity. If it is negative, then the absence of predictive validity is observed (Tenenhaus, Vinzi et al. 2005).

Using this methodology, we developed a structural model linking the latent variables of land use, hydromorphological alterations at watershed scale, hydromorphological alterations at reach scale, nutrients-organic matter, and substrate samples in view to explain variation in the  $I_2M_2$  macroinvertebrate-based index.

This model is illustrated in figure 2.





**Figure 2:** Design of the multiscale and multi-stressors structural model.

## 2.3 Results

We developed a structural model linking the latent variables of land use, hydromorphological alterations at watershed scale, hydromorphological alterations at reach scale, nutrients-organic matter, and substrate samples in view to explain variation in the I<sub>2</sub>M<sub>2</sub> macroinvertebrate-based index for four geographical types (figure 3).

### 2.3.1 Analysis and validation of the structural model

Analysis of the validation index of this structural model shows (table 1) that it is capable of satisfactorily explaining and predicting the I<sub>2</sub>M<sub>2</sub> values for the four geographic types studied with a R<sup>2</sup> of 33% for non limestone small streams, 50% for type non limestone small streams, 44% for limestone medium streams and 40% for limestone medium streams. All the

$Q^2$  (table 1) were positive for the  $I_2M_2$ , demonstrating the good predictive capacities of these four models with respect to  $I_2M_2$ .

Table 1. Validation indices of the structural model.

	Non limestone small streams		Non limestone medium streams		Limestone small streams		Limestone medium streams	
	$R^2$	$Q^2$	$R^2$	$Q^2$	$R^2$	$Q^2$	$R^2$	$Q^2$
<b>Watershed HM alteration risk</b>	1.6%	-0.14	4.4%	-0.11	1.4%	-0.16	0.8%	-0.09
<b>Reach HM alteration risk</b>	10.4%	-0.07	13%	-0.05	19.3%	-0.01	11.2%	-0.05
<b>Nutrients and organic matter</b>	27%	0.08	57%	0.36	42%	0.13	40%	0.15
<b>Substrates mosaïc</b>	9.90%	-0.04	51%	0.07	15.2%	-0.03	28.5%	0.02
<b><math>I_2M_2</math></b>	33.00%	0.24	50%	0.5	44%	0.32	40%	0.34

The hydromorphological alterations at watershed scale was very poorly correlated with the land use of watershed ( $R^2$  from 0.08% to 4.4%) (figure 3).

The hydromorphological alterations at the reach scale was poorly explained by land use and hydromorphological alterations at watershed scale with an  $R^2$  from 10 to 19% (figure 3). However, most of the information was provided by land use (from 48 % to 93% of the  $R^2$  value – figure 3) underlining the fact that the hydromorphological alterations risk at watershed and reach scale do not indicate the same processes.

The physico-chemical parameters were explained by the land use and the risk of hydromorphological alterations with a  $R^2$  from 27% to 57% and with always positive  $Q^2$  values (figure 3).

The substrates mosaic of sample site was explained more or less successfully with an  $R^2$  from 10 to 51% according to river types (figure 3). It was better explained for streams of ranks 4, 5 and 6 (51% and 28% for 'non limestone' and 'limestone' streams, respectively) than for streams of ranks 1, 2 and 3 (10% and 15%). The  $Q^2$  values were positive only for streams of ranks 4, 5 and 6.

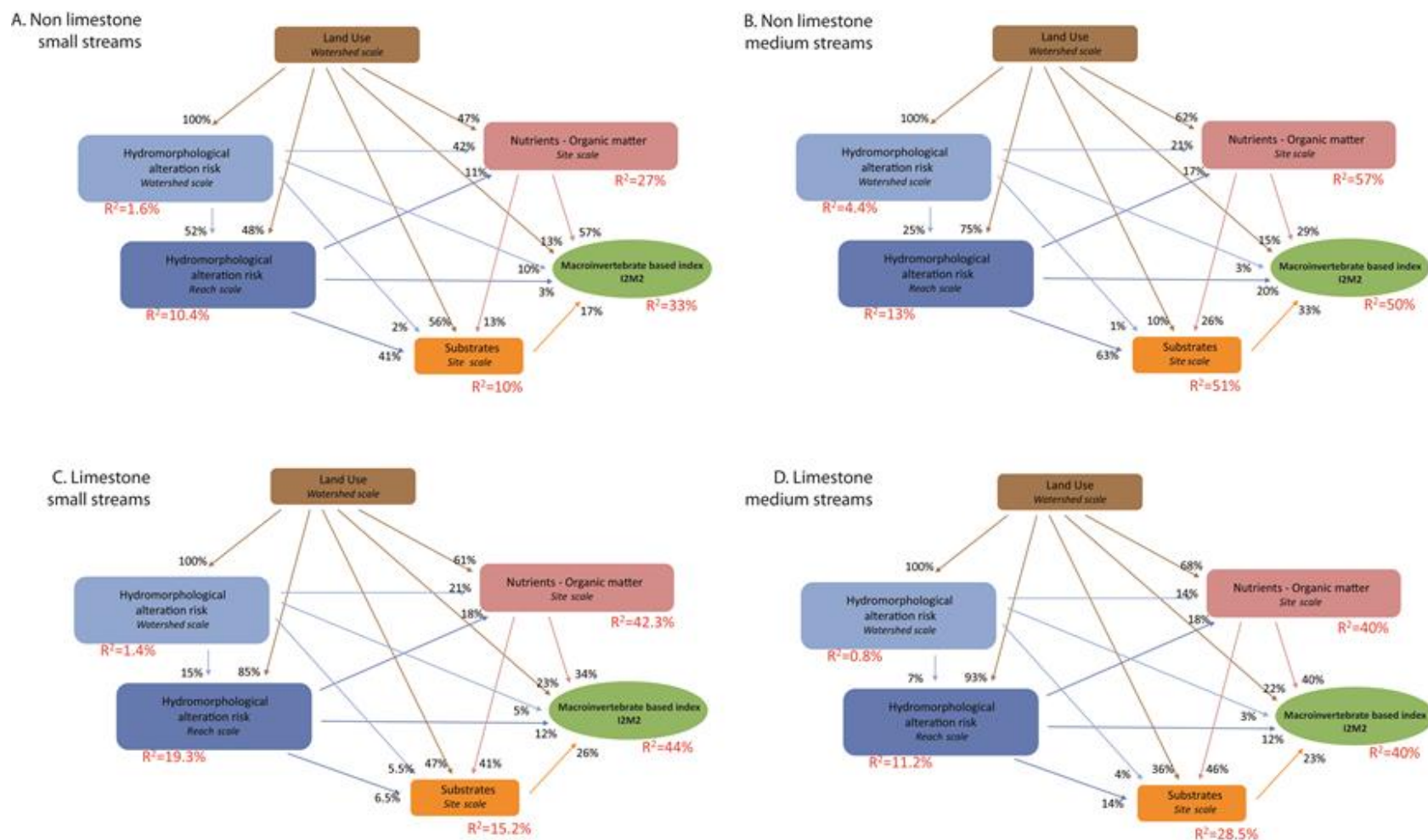


Figure 3. Models derived from PLS path modelling for non limestone small streams (A), non limestone medium streams (B), limestone small streams (C) and limestone medium streams (C). Each latent variable is represented by a coloured box and each direct effect from this latent variable to another is represented by an arrow of the same colour. R<sup>2</sup> of each internal model are represented in red. Contributions of latent variables to the deviance explained by a model (in percentages of the model R<sup>2</sup>) are represented in black.

### 2.3.2 Direct effects

The latent variables with the greatest effects on the  $I_2M_2$  were the nutrient and organic matter concentrations whatever the river type. However the response pattern differed as a function of stream type (figure 3).

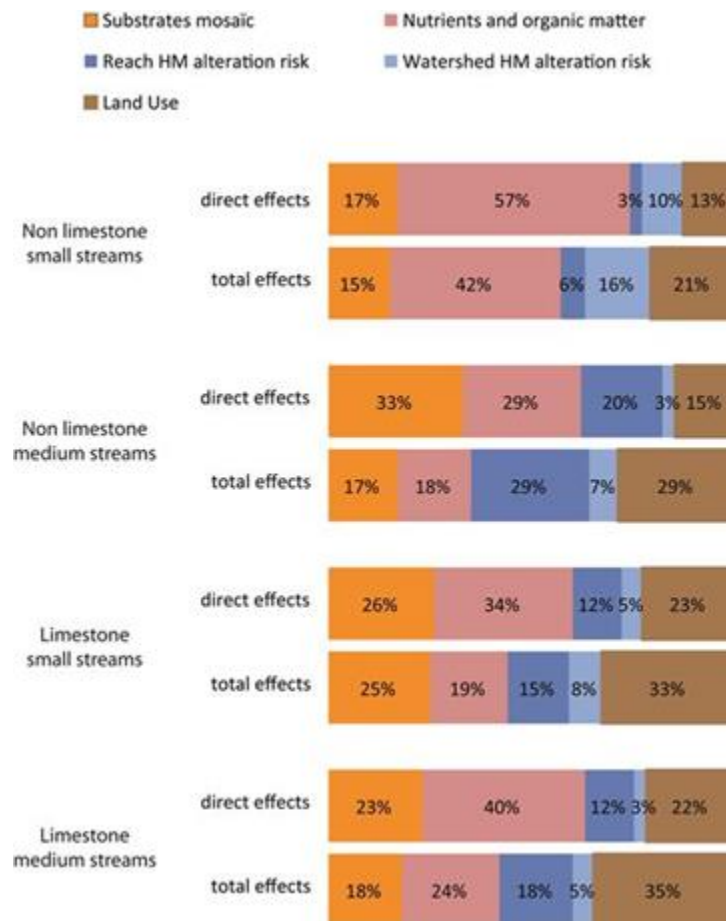


Figure 4. Relative contribution of latent variables to the explained deviance of  $I_2M_2$  taking into account direct effects only or total effects (directs + indirects) for the four stream types.

For small non limestone streams, the order of decreasing contribution to the deviance of  $I_2M_2$  explained by the model was: nutrients and organic matter (57%), substrates mosaic (17%), watershed land use (13%), hydromorphological alterations at watershed scale (10%) and hydromorphological alterations at the reach scale (3%). It was: substrates mosaic (33%), nutrients and organic matter (29%), hydromorphological alterations at the reach scale (20%),

watershed land use (15%) and hydromorphological alterations at watershed scale (3%), for medium-sized non limestone streams.

For small limestone streams, the order of decreasing impact importance was: nutrients and organic matter (34%), substrates mosaic (26%), watershed land use (23%), hydromorphological alterations at watershed scale (12%), and hydromorphological alterations at the reach scale (5%). It was: nutrients and organic matter (40%), substrates mosaic (23%), watershed land use (22%), hydromorphological alterations at the reach scale (12%) and hydromorphological alterations at watershed scale (3%) for medium-sized limestone streams.

### **2.3.3 Total effects (direct + indirect)**

When focusing on the total effects (direct + indirect) of latent variables on the  $I_2M_2$  values, it can be seen that although the variable with the most impact remained nutrients and organic matter for small non limestone streams, the contribution of direct effects considerably modified the order of impact of the other latent variables for the other stream types (Figure 4).

For small non limestone streams, the order of decreasing contribution to the variance of  $I_2M_2$  explained by the model was: nutrients and organic matter (42%), land use (21%), hydromorphological alterations at the watershed scale (16%), substrates mosaic (15%), hydromorphological alterations at the reach scale (6%). It was: land use (29%), hydromorphological alterations at the reach scale (29%), nutrients and organic matter (18%), substrates mosaic (17%) and hydromorphological alterations at the watershed scale (7%) for medium-sized non limestone streams.

For small limestone streams, the order of decreasing impact importance was: land use (33%), substrates mosaic (25%), nutrients and organic matter (19%), hydromorphological alterations at the reach scale (15%), and hydromorphological alterations at the watershed scale (8%). It was: land use (35%), nutrients and organic matter (24%), substrates mosaic (18%), hydromorphological alterations at the reach scale (18%) and hydromorphological alterations at the watershed scale (5%) for medium-sized limestone streams.

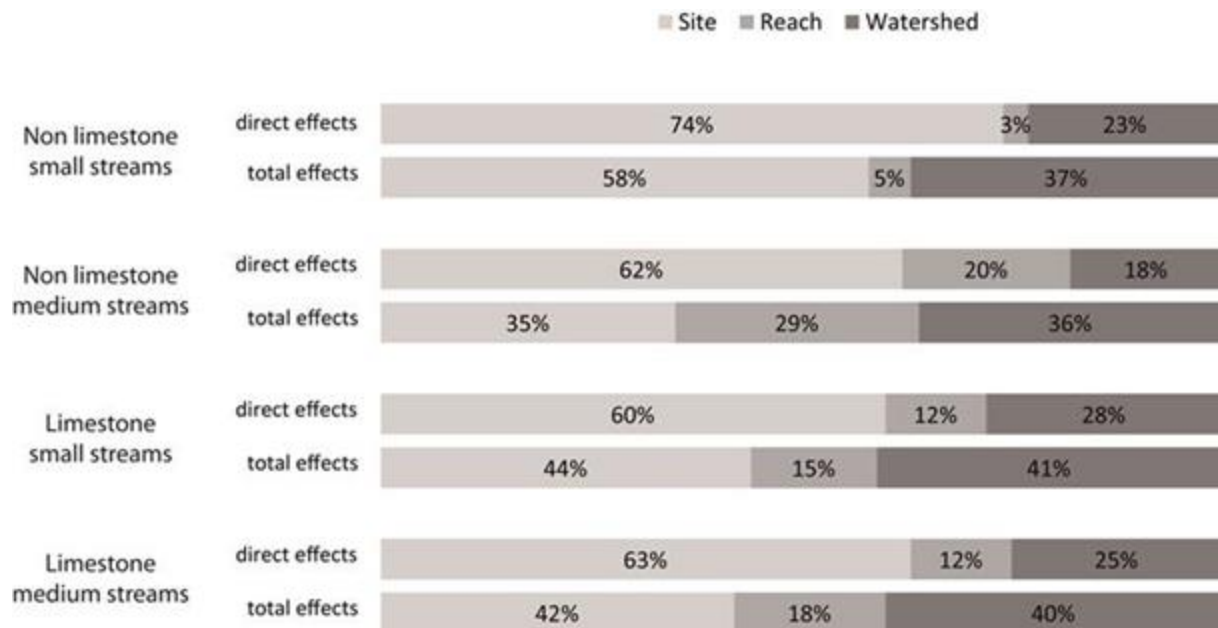


Figure 5. Relative contribution of scales levels to the explained deviance of  $I_2M_2$  taking into account direct effects only or total effects (directs + indirects) for the four stream types.

### 2.3.4 Synthesis of the effects by scale size

Another potential way to consider the contributions of latent variables was to group them by level of scale (watershed, reach, site) (figure 5). Land use and hydromorphological alterations at the watershed scale represented the watershed level. Hydromorphological alterations at the reach scale represented an intermediate level covering pressures at the scale of sub-watershed or hydromorphological reach. Substrates mosaïc and nutrient and organic matter pressures represented explicative parameters expressed directly at the site level.

When adding the % of direct contributions to the  $R^2$  of the latent variables by level of scale, the site level exhibited the highest contributions to the variance of  $I_2M_2$  explained by the model, from 60% to 74% of the  $R^2$ . The remaining contributions were shared by the reach scale (from 3.3% to 19.7% of the  $R^2$ ) and the watershed scale (from 18.4% to 28% of the  $R^2$ ), nonetheless exhibiting a great effect of watershed scale.

When grouping the direct and indirect effects (total effects), the watershed and reach scale effects increased considerably, representing up to 65% of the total contribution to the  $R^2$ .

### 2.3.5 Focusing on the effect of Hydromorphology

Lastly, the total of the direct contributions of the watershed and reach hydromorphological variables represented from 13% to 23% of  $R^2$  whereas they increased from 8% to 14% when taking into account the indirect effects expressed via the impact of watershed and reach hydromorphology on physico-chemical variables and habitat mosaic, providing total contributions from 21% to 37% of the  $R^2$ .

## 2.4 Discussion

### 2.4.1 Structural model advantages

In a previous article, (Riseng, Wiley et al. 2011) highlight the ability of structural equation methods to allow to work in a first time on a *conceptual causal model* (a construct that represents the hypothesized causal linkages between variables in the system being analyzed) and then, in a second time, to developed from that conceptual model a *generalized structural model* (in the form of a specific causal path hypothesis) constrained by the availability of variables. And we have seen in our study that the PLS Path modelling method which belong to the same family of structural equation modelling (SEM) allowed to develop a model taking into account the hierarchical organization of streams with three scales (watershed, reach and site) and allowing to evaluate both direct and indirect effects of pressures variables on macroinvertebrates. Using more classical methods, previous studies (Dahm, Hering et al. 2013, Feld 2013, Villeneuve, Souchon et al. 2015) highlighted the difficulty to take into account complex relationships and multiscale organization in their models. Structural equation modelling should be use more often to evaluate complex causal relationships in aquatic ecology.

### 2.4.2 Regionalisation, typology and ranking of pressures

At the scale of the watershed, the primary, universally recognized, determinants of the ecological functioning of streams are geology, relief and climate. This concept is based on the theory of a hierarchical control of aquatic ecosystems, especially with nested physical scales, from the watershed to the meso-habitat.

The following summary can be provided about the hierarchical sequence of factors determining the ecological functioning of streams (Wasson, Chandesris et al. 2002): at local scale, in the aquatic environment, the biodiversity and productivity of living communities respond to the “key factors” of the physical habitat (hydraulics, substrate), the aquatic climate (light, temperature, dissolved gases, hydrochemistry), and the trophic resources (endogenous primary production and exogenous organic matter). Moreover, the influence of habitat characteristics on major parameters like oxygenation and trophic resources led to assigning a preponderant role to physical factors of supporting biotic conditions of streams. At the reach scale, the spatio-temporal dynamics of key factors depends on the hydro-sedimentological regime and the geomorphology of the valley. These “control factors” define the fluvial morpho-dynamics (bed shape and stability), the structure of riverine vegetation (which significantly influences the aquatic habitat), and the connectivity within the stream (e.g. the interactions with the floodable zone). Finally, at the regional scale, these factors depend on the “primary determinants” of geology (nature of rocks), relief (geomorphology) and climate (temperatures and precipitations). The soils and vegetation of the watershed are obviously components of the water balance and sediment dynamics, though they are determined finally by geophysical and climatic characteristics.

In our study we have shown that the hierarchy of pressures can differ according to the type of stream studied. Thus in a hard water streams, hydromorphology and land use have a considerable impact on macroinvertebrates, sometimes greater than physico-chemistry. However, in a soft-water habitat (non-limestone), physico-chemistry is the primary impact factor for small streams. Hard-water habitats are naturally more productive than soft-water ones (Hill and Webster 1982, Hill 1992, Jin and Ward 2007). Thus low primary production limits the secondary production under natural conditions and the habitat will be less buffered and more vulnerable to imbalance during physico-chemical stress. However, in the case of hard-water habitats, since primary production is naturally higher and thus a non-limiting factor, secondary production would therefore be more stable and more effectively buffered when a physico-chemical stress occurs.

Finally, ecological and geomorphic responses to human alteration of land cover will have to be calibrated to the regional hydroclimatological, geologic, and historical context in



which the streams occur, in order to determine the degree to which stream responses are region-specific versus geographically independent and broadly transferable (Poff, Bledsoe et al. 2006) .

### **2.4.3 Direct and indirect effects of land use**

The models examining the responses of biotic assemblages to various stressors acting at different scales, have often shown, in the literature, a preponderant impact of physico-chemical parameters (nutrients and organic matter) measured at site scale followed, in order of decreasing impact importance, by land use at watershed scale and then by hydromorphological alteration factors, with an effect far more difficult to evidence (Dahm, Hering et al. 2013, Villeneuve, Souchon et al. 2015). More precisely, (Hering, Johnson et al. 2006) show that that the macroinvertebrates but also diatoms, macrophytes and fishes responded less strongly to land use gradients than to nutrient enrichment. But their study did not take into account the indirect effect of land use despite highlighting the potential affect of the community through numerous cause-effect relationships acting singly or in concert like sedimentation and hydromorphological alteration, diffuse pollution as well as direct inputs of toxic substances (metals, pesticides).

Our results were consistent with these works that had shown the preponderant effect of physico-chemistry on land use effect, exerted directly at site level. However, the analyses of indirect effects of land use have shown clearly that, except for non limestone small streams, the land use effect was more important than the nutrients and organic matter effect. Different human activities (urbanization, agriculture, industry, transport, etc.) generate pressures on ecosystems. These pressures are expressed at site level by metrics measuring physico-chemical and hydromorphological alterations. At larger scale, the land use data has the potential to highlight the presence in the watershed of stressor sources altering the ecological status of rivers at the site scale (Allan 2004). And more than that, some authors have demonstrated that land use have an effect on macroinvertebrates at multiple scales (Wang and Kanehl 2003, Weigel, Wang et al. 2003, Wang, Robertson et al. 2007). Consequently, we can consider that whatever the scale of measurement applied to the variables (watershed, reach or site), the direct effect measures the impact of individual

variables at site scale whereas the indirect effect measures the impact of individual variables at larger scale.

Both artificial and agricultural land use constitute a major direct impact factor for biotic assemblages and many authors have demonstrated the existence of this link (Allan 2004, Clapcott, Collier et al. 2012, Marzin, Archambault et al. 2012, Dahm, Hering et al. 2013, Marzin, Verdonschot et al. 2013). This result appeared still more clearly in our models due to the separation of the direct and indirect effects, something which had not been done up to now. The presence of heavily anthropized catchments has three types of impact on streams (Paul and Meyer 2001, Stepenuck, Crunkilton et al. 2002, Allan 2004, Effenberger, Sailer et al. 2006): it increases (i) runoff and thus erosion and diffuse pollution; (ii) chemical pollution; and (iii) the alteration of the stream hydromorphological functioning (via eradication of riparian forest, straightening of the water course and re-gauging, incision). As a result, land use is an efficient indirect descriptor of the pressures exerted on a stream, on different functional scales structuring the hydrosystem (Allan 2004). We have found similar results with our models with significant contributions of land use on the hydromorphological reach, physico-chemistry and the substrates mosaic. Furthermore, when measuring the direct effect of land use at site scale, we omitted all the indirect processes (increasing of N and P concentrations or clogging for example) that govern this effect and partially omitted its nature of "driving force" for considering it as a pressure.

#### **2.4.4 Focus on the role of hydromorphology**

The purpose of the hydromorphological variables used in this work was to take into account alterations of non-natural origin that could be clearly correlated with the degradation of ecological status. This strategy allowed taking into account different developments and usages of the watershed (agriculture, urbanization, transport) and the low flow channel (transport, energy, tourism) whose effects are resulting in the alteration of structures and hydromorphological processes. The finding that benthic macroinvertebrates respond to changes in hydromorphology (reach scale) supports a number of earlier studies (Buffagni, Erba et al. 2004, Hering, Meier et al. 2004, Lorenz, Hering et al. 2004). But in previous studies (Hering, Johnson et al. 2006, Dahm, Hering et al. 2013, Villeneuve, Souchon et al. 2015), hydromorphological alterations were ranked in third position in order of

decreasing impact importance, after physico-chemical factors and land use with the conclusion that the effects of hydromorphological pressures could be measured but their impact could be relatively weak on biological metrics. Based on our modelling design, two aspects have advocated for re-evaluating hydromorphological pressures in terms of magnitude of impact on the macroinvertebrates: (i) taking into account two levels of scale (watershed and reach) and (ii) analyzing the indirect effects through physico-chemistry and habitats mosaic attributes.

The sum of the contributions to macroinvertebrates explained deviance of both hydromorphological at watershed and reach scales corresponded to an amount of impact of nearly the same level as land use considering either direct or indirect effects (figure 4). Logically, a better description of hydromorphological pressure have provided a better appraisal of the link between hydromorphology and stream biological status.

The reach hydromorphology expresses parameters that can be used to measure a risk of alteration of the river hydromorphological functioning which can be linked to pressures whose effects are best detectable at the reach scale. Obviously, the alteration of hydromorphological processes at the reach scale represented a direct pressure on macroinvertebrates by degrading the macroinvertebrate habitats. The presence of weirs, dikes, bank modifications, the alteration of the riparian forest and of the bed geometry will, in particular, lead to the modification of the facies which become more lentic, thereby directly modifying the macro-invertebrate substrates mosaic. Previous studies have addressed the role of spatial scales in hydromorphological alterations–community relationships (Lamouroux, Dolédec et al. 2004, Hering, Johnson et al. 2006, Feld and Hering 2007, Dahm, Hering et al. 2013) and highlighted the important role of the reach scale in this relation. This is consistent with our results that show the important role of hydromorphological variable in the environment-biological condition relationship. Moreover, our analysis reinforce the representativity of the results using a large data set of 1200 sites where others studies use only one or two hundred of sites.

Taking into account alterations linked to the erosion of agricultural surfaces, irrigation, drainage and storage at an appropriate scale allows the inclusion of processes acting on the watershed scale even if they can also act directly at the reach and site scales. The direct

effect of these variables could be evidenced by an increase of suspended (and deposited) fine sediments and clogging risk, and more marked low water levels. Our results have also shown a hydromorphological watershed effect on the other types of variables making it possible to identify an indirect effect of around 50% of the direct effect.

Moreover, we have demonstrated that hydromorphology had a direct effect on physico-chemistry. A modification of the morphological facies in the reach can potentially modify the residence time of molecules, potentially the infiltration rate in down welling areas, the self-purification capacity of streams and other internal physico-chemical processes, which depend on hydromorphological conditions (Baker, Bledsoe et al. 2012). And consequently, taking into account this indirect effect via the nutrients and organic matters increased the total effect of hydromorphology (at watershed and reach scales) on macroinvertebrates.

## **2.5 Conclusion**

Since (Hynes 1975, Frissell, Liss et al. 1986, Allan and Johnson 1997, Parsons, Thoms et al. 2003, Townsend, Dolédec et al. 2003, Parsons, Thoms et al. 2004, Thorp, Thoms et al. 2006), it has been acknowledged that understanding the functioning of a stream ecosystem requires taking its hierarchical spatial organisation into account, as done in this study by taking into account pressures at three nested scales, i.e. watershed, reach and site.

As a first important result, we have highlighted the importance of land use effect exerted on both hydromorphology and physico-chemistry and their translation as an indirect effect on biological condition of streams. We have also demonstrated that hydromorphological alterations had an effect on substrates structure and nutrients and organic matter concentrations. This result implies that hydromorphology has a major indirect effect on macroinvertebrates and that the ranking of pressures previously described in literature is now modified. And finally, the nutrients an organic matter effect on macroinvertebrates remains lower than expected since we take into account all the indirect effects of land use and hydromorphological alterations. A large part of physico-chemistry variance is the resultant of driving forces expressed at higher scales (watershed and reach).

Aquatic ecosystems have long been affected by the consequences of anthropic activities whatever the scale (Allan 2004), and the corresponding impairments, both direct and

indirect, are various. We have highlighted the importance of the site scale (physico-chemical and substrates mosaic variables) for explaining the biological condition of streams. However, these results have also shown the important role of the variables measured at the reach scale, both directly via their contribution to habitat feature modifications, and indirectly via their confounding effect on the relationship between physico-chemistry and macroinvertebrates. Lastly, we have also shown that the variables measured on the scale of the watershed (in this case: land use and large scale hydromorphology) integrate all types of pressure acting at lower scales located on the upstream watershed and play a strong structural role through their indirect effects on all the pressure characteristics.

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# Chapitre 7 - Discussion et prospective

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## 1 Synthèse des principaux résultats de la thèse

Comme nous l'avons souligné dans le chapitre d'introduction, les cours d'eau et leurs bassins versants sont des systèmes complexes et en équilibre dynamique. Si l'on connaît qualitativement assez bien la plupart des processus dont ils sont le siège (ex. transport de matière organique et de sédiments, transformation des litières, etc...), il est beaucoup plus difficile d'en avoir une représentation pertinente à la bonne maille compte tenu de l'emboîtement d'échelles des processus, des déterminants terrestres agissant sur les milieux aquatiques et surtout des différents facteurs de stress qui s'y combinent. Ce qui rend difficile l'appréciation de leur état, la prévision des états futurs et l'extraction d'une information pertinente en direction des gestionnaires.

Par conséquent, dans l'optique de garantir à la fois une amélioration de l'état écologique des cours d'eau et les intérêts humains découlant de leur utilisation, la Directive Cadre Européenne sur l'eau (DCE) a mis en évidence le besoin d'outils d'évaluation et de diagnostic de l'état écologique des cours d'eau pour optimiser les mesures de gestion et de restauration en fonction des causes avérées et correctement hiérarchisées d'altération.

Encore une fois, atteindre cet objectif nécessite bien sûr de maîtriser les connaissances sur le fonctionnement des hydrosystèmes mais aussi en particulier de savoir interpréter les signaux biologiques placés au cœur du processus d'évaluation. Ceci implique que les liens entre ces signaux biologiques et les pressions qui les altèrent, leurs évolutions temporelles dans leur contexte géographique soient correctement décryptés.

L'exploration que nous avons menée dans la thèse s'apparente aux pratiques d'épidémiologie, le but étant de rechercher les causes souvent multiples de dégradation de l'état écologique, d'en connaître l'étendue spatiale, leur gravité, pour être en mesure de recommander des actions efficaces de restauration.

Dans la première partie de la thèse, nous avons développé des modèles explicatifs de l'état écologique à large échelle capables de donner des réponses applicables aux échelles

de décision et de gestion, à savoir européenne, nationale et régionale avec comme objectif principal de comparer les relations entre les pressions combinées représentées par l'occupation du sol et l'état écologique des cours d'eau de quatre pays européens : la France, la Slovaquie, l'Estonie, et au Royaume-Uni (Angleterre et Pays de Galles). Nous avons développé des modèles pression-impact reliant les indices basés sur les invertébrés benthiques à l'occupation du sol du bassin versant et du corridor rivulaire. Les données des quatre pays ont en outre été stratifiées en ensembles géographiquement cohérents basés sur les hydro-écorégions d'Europe. Cette étude a permis (i) d'établir la hiérarchie des catégories d'occupation du sol influant sur l'état écologique, en particulier l'agriculture et l'urbanisation, (ii) d'identifier des tendances régionales dans ces relations pression-impact et (iii) d'évaluer le poids relatif des pressions du bassin et des échelles de corridors riverains, et l'effet tampon de la couverture des zones rivulaires.

Dans une deuxième partie, nous avons construit un corpus de modèles capables d'expliquer d'une part la variabilité des indices biologiques utilisés dans les réseaux et de prédire d'autre part l'état écologique des masses d'eau françaises non suivies par le réseau de contrôle et de surveillance (RCS). Les indices basés sur les macroinvertébrés benthiques, diatomées et poissons ont été utilisés pour déterminer l'état écologique de 1200 sites du RCS répartis de manière homogène sur l'ensemble du territoire français. Les pressions prises en compte pour expliquer et prédire l'état écologique couvrent trois échelles spatiales : le bassin versant, le tronçon hydromorphologique et la station. Le jeu de données prédictives recouvre trois types de pressions : les pressions d'occupation du sol, les pressions hydromorphologiques et les pressions physico-chimiques mesurées respectivement à l'échelle du bassin versant, du tronçon hydromorphologique et du site. Cette étude a permis de montrer : (i) que le facteur le plus impactant pour l'état biologique qu'il soit mesuré à partir des macroinvertébrés, des poissons ou des diatomées était la physico-chimie, puis l'occupation du sol et enfin l'hydromorphologie et (ii) que l'utilisation de ces trois types de pressions permettait de développer des modèles prédictifs de l'état biologique robustes et fiables.

Dans une troisième partie, nous avons voulu tester si les relations entre les pressions à l'échelle locale et l'état écologique étaient hiérarchiquement influencées par la typologie régionale (basée sur la physiographie naturelle et les forces motrices). Nous avons donc

développé un modèle hiérarchique qui relie l'indice invertébrés multimétrique (I2M2) à des pressions hydromorphologiques et physico-chimiques selon des modèles hiérarchiques régionalisés. Pour ce faire, les bassins ont été regroupés en fonction de leurs caractéristiques naturelles et des gradients de forces motrices. Nous avons supposé que l'organisation hiérarchique des cours d'eau conduisait à différentes réponses des organismes à la pression, en fonction des caractéristiques du site étudié dépendant de l'échelle régionale. Ceci nous a permis de montrer que la prise en compte d'une typologie basée sur les forces motrices des bassins versants ne remettait pas en cause la hiérarchie des effets des pressions sur la biologie.

Enfin, dans une quatrième et dernière partie, nous avons développé une approche nouvelle qui propose de prendre en compte les connaissances sur l'organisation des échelles et sur les liens entre pressions et état écologique en construisant et en analysant un modèle structurel basé sur la méthode PLS path modelling qui permet de relier entre elles des variables latentes correspondant aux pressions d'occupation du sol, hydromorphologiques et physico-chimiques et d'analyser *in fine* leur effet sur l'état écologique mesuré ici à travers l'indice invertébrés multimétrique : l'I2M2. Le développement de ce modèle nous a permis de montrer : (i) l'effet important de l'occupation du sol sur l'hydromorphologie et sur la physico-chimie et l'effet indirect qui en découle sur les macroinvertébrés, (ii) l'effet de l'hydromorphologie sur la mosaïque de substrats, les nutriments et la matière organique impliquant un effet indirect majeur de l'hydromorphologie sur les assemblages de macroinvertébrés, et (iii) un effet plus faible qu'attendu des nutriments et de la matière organique (en regard des résultats précédents), car la prise en compte des effets indirects montre qu'ils sont en grande partie la résultante de l'effet des forces motrices qui dirigent le système.

Ces résultats ont conforté nos hypothèses initiales concernant le rôle de l'occupation du sol et des paramètres physico-chimiques et leur impact sur le compartiment biologique. Ils jouent un rôle important dans le fonctionnement écologique de l'hydrosystème et permettent de prédire et expliquer assez efficacement l'état biologique des cours d'eau. Comme nous l'avons dit plusieurs fois au fil de ce mémoire, ces résultats sont cohérents avec ce que l'on retrouve dans la littérature scientifique récente. Cependant, si de nombreux auteurs ont décrit le fonctionnement en échelles emboîtées du cours d'eau, aucune étude à

notre connaissance n'avait réellement mis en évidence les liens inter-échelles structurant les relations pressions-état comme nous l'avons fait ici.

De plus, si l'hydromorphologie est toujours décrite comme un facteur structurant majeur du cours d'eau, elle n'est que rarement, voire jamais, associée à un effet majeur sur l'état biologique du cours d'eau à une échelle plus large que l'échelle du site. Nous pensions bien sûr améliorer notre connaissance sur le rôle de l'hydromorphologie en prenant en compte dans nos modèles les différentes échelles emboîtées du cours d'eau et leur organisation hiérarchique. Mais nous pensions surtout révéler un effet direct de l'hydromorphologie sur la biologie s'exerçant via l'habitat. Mais ces effets sont restés en retrait face aux effets de l'occupation du sol et de la physico-chimie. Par contre, nous avons pu enfin démontrer, en prenant en compte les effets indirects inter-pressions, que le rôle important de l'hydromorphologie se faisait en grande partie via son influence sur les relations entre physico-chimie et état biologique. C'est à notre sens le résultat le plus intéressant de cette thèse.

## **2 Importance du choix des méthodes de modélisation**

Le développement des moyens informatiques et de calcul ont permis le stockage, le traitement et l'analyse d'ensembles de données toujours plus volumineux. Cette évolution, ainsi que la popularisation de nouvelles techniques algorithmiques a conduit au développement de logiciels intégrant un sous-ensemble de méthodes statistiques et algorithmiques utilisées sous la terminologie de Data Mining (fouille de données).

Ces méthodes sont intéressantes et ont permis de populariser la plupart des techniques de modélisation utilisées aujourd'hui en sciences (et en particulier celles que nous avons utilisées dans ces travaux). Mais elles peuvent conduire à penser, qu'à condition d'avoir à disposition une grande base de données, si possible géoréférencées, on serait en mesure de faire émerger des hypothèses nouvelles.

La critique principale que nous pourrions faire à ce type d'approche est de reposer sur de grandes bases de données. Or, c'est justement le point faible de beaucoup d'études en écologie des cours d'eau. Il est très difficile et surtout très coûteux de réunir des jeux de



données biologiques conséquents. Et il est encore plus compliqué de réunir des jeux de données de pressions cohérents et pertinents associés à ces jeux de données biologiques.

A titre d'illustration, constituer une base de données biologiques comme la base Pandore du Laboratoire d'Hydroécologie Quantitative (LHQ) d'Irstea représente trois années de travail pour un ingénieur à plein temps pour la phase initiale de construction de la base de données et la bancarisation des données existantes. Continuer à l'alimenter en continu avec les données provenant des réseaux DCE a nécessité le travail d'un technicien à mi-temps pour assurer une bancarisation correcte. Et ceci ne prends pas en compte le coût lié à la validation de la qualité de la donnée initiale (liées aux défauts de pratique des méthodes normalisées d'échantillonnage, de traitement des échantillons, de détermination des organismes, de saisie des données, etc...). Prenons comme autre exemple le projet SYRAH qui a permis de produire une donnée de pression hydromorphologique homogène à l'échelle de la France entière. Il a nécessité 5 ans de travail pour deux ingénieurs pour arriver à une base de données des risques d'altération hydromorphologique des tronçons de cours d'eau.

Cela nous montre à quel point il est difficile de constituer des bases de données suffisamment conséquentes et surtout suffisamment homogènes à l'échelle du territoire national pour pouvoir utiliser de manière optimale des algorithmes de fouille de données non dirigés.

Nous avons choisi dans cette thèse d'adopter une démarche analytique plus classique reposant sur des méthodes performantes adaptées aux impératifs de nos jeux de données.

Nous avons affaire à des variables explicatives très souvent fortement corrélées entre elles et c'est pour cette raison que nous avons utilisé la régression PLS comme outil explicatif de base. En effet, l'utilisation de la régression multiple classique aurait conduit à des erreurs d'interprétation à cause de l'influence des corrélations entre variables sur les coefficients de régression (Tormos et al. 2014; Villeneuve et al. 2015; Wasson et al. 2010).

Concernant le choix d'une méthode prédictive, nous avons choisi la méthode des arbres d'inférence conditionnelle (CIT, (Hothorn et al. 2006)). Par rapport à d'autres méthodes d'arbres de décision (comme les CART (Breiman 1984) qui utilisent le critère de Gini), la méthode CIT propose un test statistique (sur le principe des tests de permutations)

permettant de sélectionner la variable  $X_i$  la plus pertinente à chaque nœud et de définir un critère d'arrêt de la croissance de l'arbre. Cette méthode avait déjà été utilisée avec succès par Mondy et al. (2012).

Cependant, d'autres méthodes basées sur les arbres de décision sont fréquemment utilisées en écologie. Par exemple, Mondy and Usseglio-Polatera (2013) utilisent les forêts d'arbres (ou random forests) qui sont constituées d'un grand nombre d'arbres (de type CART souvent ou CIT dans leur cas). Chacun de ces arbres est alors construit sur un sous-ensemble aléatoire du jeu de données d'apprentissage. Et chaque individu du jeu de données se verra alors attribuer la valeur moyenne des prédictions de l'ensemble des arbres.

Une autre méthode très utilisée récemment (Dahm et al. 2013; Feld 2013; Feld et al. 2016a) est la méthode des « Boosting Regression Trees » (BRT) (Elith et al. 2008). Il s'agit d'une technique d'optimisation numérique qui cherche à minimiser l'erreur en ajoutant à chaque pas un nouvel arbre construit à partir des résidus du modèle précédent. Le modèle final est une combinaison linéaire d'un grand nombre d'arbres qui peut être vu comme un modèle de régression dans lequel chaque terme est un arbre.

Ces deux types de méthodes ont montré des résultats prédictifs plus performants que les arbres de décision simples que nous avons utilisés. Mais notre choix a été guidé par une préoccupation principale : construire un modèle qui pourrait être utilisé directement par les gestionnaires sous forme d'un algorithme simple. En ce sens, ces méthodes ne pouvaient convenir. Bien sûr, nous aurions pu fournir avec chacune de ces méthodes un outil intégré permettant le calcul de prédictions de manière simple pour l'utilisateur. Mais plus qu'un outil, nous souhaitons transmettre une méthode d'interprétation des niveaux de pressions.

L'approche PLS, utilisée dans le chapitre 6, fait partie des modèles d'équations structurelles à variables latentes, « méthodes de modélisation de phénomènes apte à bien définir des systèmes complexes en interaction » (Jakobowicz 2007). Qualifiés de méthodes de seconde génération (par comparaison aux méthodes de première génération comme les analyses en composantes principales ou les analyses multidimensionnelles), les modèles d'équations structurelles offrent au chercheur une plus grande flexibilité dans l'interprétation entre théorie et données (Chin 1998). Plus précisément, ces méthodes permettent d'introduire « des variables latentes ou non observables, de spécifier la nature

des relations entre ces variables latentes et leurs mesures, de préciser le type de relations envisagées entre les variables latentes ou encore d'analyser des inférences causales entre plusieurs ensembles de variables explicatives et expliquées » (Croutsche 2002). Cette méthode a été dès son développement présentée comme adaptée aux analyses causales prédictives dans des situations de forte complexité et d'information théorique faible (Jöreskog and Wold 1982).

Depuis quelques années, cette approche est de plus en plus populaire dans des communautés scientifiques très variées (Vinzi 2008) et plus particulièrement en écologie aquatique (Musseau et al. 2015; Riseng et al. 2011; Selim et al. 2016). Et nous ne pouvons qu'approuver cet engouement à la lumière de nos résultats. En effet, la possibilité de différencier les effets directs et indirects des variables latentes nous a permis de revoir la hiérarchie des pressions telle que les méthodes utilisées jusqu'à présent en rendait compte.

### **3 Liens avec la thématique multi-stresseurs**

En 2006, Munns soulignait les besoins en termes de recherche pour évaluer les risques, encourus par les populations sauvages, qui résultent de l'exposition à des facteurs de stress multiples. Mais il soulignait aussi l'incertitude considérable qui persiste en ce qui concerne la façon dont ces évaluations devaient être menées. Et cette incertitude est aggravée par les questions concernant les effets d'interaction des facteurs de stress concomitants, la nécessité de prendre en compte les échelles spatiales appropriées et l'utilisation d'ensembles de données limitées.

En 2010, dans l'article de présentation du numéro spécial de *Freshwater Biology* sur les multiples stresseurs dans les écosystèmes d'eau douce, Ormerod et al. (2010) rappelaient que la gestion des écosystèmes aquatiques continentaux implique presque toujours des questions simultanées, soit parce que les pressions anthropiques modifient généralement plus d'un facteur environnemental (par exemple l'urbanisation affecte quantitativement le ruissellement et son patron temporel, la qualité de l'eau, les régimes thermiques, la disponibilité en habitats, la dispersion des espèces invasives), soit parce que les pressions exercées par plusieurs sources distinctes coïncident souvent.

Et si cela ne suffisait pas pour adopter un nouveau regard sur les multiples stressseurs, le changement climatique est maintenant exacerbé et entraîne une confusion qui complique les problèmes existants. Il n'entraîne pas seulement des modifications des régimes thermiques et hydrologiques mais modifie aussi les effets des autres pressions existantes en les aggravant (Ormerod et al. 2010).

De plus, il reste assez délicat d'étudier l'effet de plusieurs stressseurs présents en même temps dans un système complexe comme l'hydrosystème. En effet, les multiples stressseurs peuvent produire plusieurs effets possibles. Tout d'abord, l'effet des multiples stressseurs peut être de nature additive ou ces effets combinés peuvent être synergiques (i.e. plus forts que la somme des effets individuels) ou plus faibles à cause des interactions antagonistes entre stressseurs (Folt et al. 1999; Townsend et al. 2008; Vinebrooke et al. 2004).

Mais parce que les stressseurs d'origine anthropique agissent généralement de manière simultanée, les gestionnaires ont besoin de connaître leurs modes d'interaction. Cependant, les études sur ce sujet restent rares pour les cours d'eau (Matthaei et al. 2010).

En nous basant sur ces constatations, nous avons décrit l'écosystème cours d'eau comme un système dynamique complexe, résultant de l'adaptation en continu de compartiments hiérarchiquement emboîtés sans cesse en interaction, soumis en continu à des contraintes naturelles ou à des pressions d'origine anthropique. Nos résultats l'ont clairement mis en évidence en soulignant les effets directs et indirects des forces motrices et des pressions sur l'état des cours d'eau.

L'analyse de l'impact des activités humaines sur le fonctionnement écologique, via l'observation des communautés écologiques, doit désormais tenir compte de cette complexité des pressions, et s'attacher à en analyser les déterminants, les interactions (synergies et antagonismes) et les échelles d'impacts au sein d'approches systémiques (Boldt et al. 2014).

Dans un article récent, Feld et al. (2016b) proposent une définition des terminologies « pressions » et « stressseurs ». Le terme de pression étant issu du DPSIR (Kristensen 2004), il fait de fait référence à la résultante directe d'une force motrice. Au contraire, le terme de stressseur est beaucoup plus spécifique et représente une variable environnementale

mesurable qui, comme résultante d'une pression anthropique, affecte et modifie l'intégrité biologique et écologique du système (Matthaei et al. 2010). Ces termes ne sont donc pas interchangeables : une seule pression (par exemple, la pollution diffuse) peut en fait représenter plusieurs stressseurs (par exemple, augmentations des concentrations en nitrates, nitrites, ammoniac, phosphore) qui agissent simultanément. Ainsi Feld et al. (2016b) soulignent le fait que si l'interprétation des effets de multiples stressseurs sur la biologie peut se faire de manière directe, car la relation mécanistique directe peut être décrite (Poff et al. 1997), il n'en est pas de même pour l'effet des pressions qui sont définies à une maille plus grossière qui ne permet pas de modéliser les relations de cause à effet de manière mécanistique.

Ainsi, analyser des pressions et des stressseurs dans le même modèle peut s'avérer limitant, surtout si les deux sont fortement corrélés. Aussi, une sélection appropriée des pressions et des stressseurs est capitale pour la construction d'une analyse de multiples stressseurs (Feld et al. 2016b). L'avantage principal du modèle PLS-path que nous avons développé dans le chapitre 6 est de distinguer ces deux concepts avec des variables latentes représentant des pressions et constituées par des combinaisons linéaires de stressseurs environnementaux individuels. Dès lors, nous avons pu montrer l'effet de la pression hydromorphologique sur les assemblages de macroinvertébrés alors que de nombreux auteurs (y compris nous dans les chapitres 4 et 5) n'y étaient pas parvenus.

Comme nous nous sommes attachés à le montrer tout au long de ce travail, les impacts sur le fonctionnement écologique des cours d'eau ne sont pas seulement déterminés par des sources de pollution ponctuelles, mais aussi par l'ensemble des influences anthropiques plus complexes comme les pollutions diffuses, les dysfonctionnements hydrologiques, les variations de flux de sédiments, les modifications hydromorphologiques et les ruptures de connectivité. Dans la plupart des cas, ces pressions agissent simultanément (pressions combinées) et sont réparties irrégulièrement sur le territoire. De plus, sous des pressions équivalentes, des cours d'eau peuvent réagir différemment en fonction de leur environnement local et de leur géologie. La recherche des causes d'altération de l'état écologique implique donc bien une analyse multi-échelles régionalisée des processus impactants.

Mais le besoin de tels outils est-il toujours d'actualité ? Oui, et c'est ce que soulignent Hering et al. dans leur revue bibliographique de 2015. Par rapport à la grande variété d'indicateurs existants pour l'évaluation de l'état écologique, les capacités à prédire les réponses des écosystèmes au stress, à la restauration ou plus généralement à des mesures de gestion alternative sont sous-développées. En particulier, une plus grande connaissance sur les effets des expositions à des facteurs de stress multiples, les interactions entre stressseurs (synergiques, additifs ou antagonistes), est nécessaire mais reste encore peu visible dans la littérature scientifique.

De manière complémentaire, la revue de Nöges et al. (2016) souligne que paradoxalement, malgré l'importance des problèmes causés par la combinaison de multiples stressseurs à la fois aux scientifiques et aux gestionnaires, ce domaine est effectivement très mal représenté dans les recherches actuelles. Downes (2010) et Ormerod et al. (2010) avaient déjà souligné la capacité limitée de prédire correctement les effets des activités humaines sur les écosystèmes aquatiques. Ce constat s'avère encore plus pessimiste si on cherche des études qui s'aventurent à démêler les effets des multiples stressseurs (Van Looy et al. 2015).

À notre avis, les raisons pour lesquelles des effets multi-contraintes n'ont pas été évalués de manière adéquate jusqu'à présent sont multiples: (i) les descriptions de stress ne sont toujours pas entièrement harmonisées empêchant des analyses performantes à grande échelle, (ii) les informations biologiques recueillies ne conviennent pas (insensibilité) pour analyser les effets de facteurs de stress combinés et (iii) les données sont recueillies à une résolution temporelle ou spatiale insuffisante pour détecter les effets des facteurs de stress pertinents (par exemple, une distribution de taille granulométrique des composantes du substrat ne dit rien sur le colmatage profond ... ou le dit mal !).

## **4 Vers une approche risque intégrée de l'analyse des altérations des cours d'eau (Van Looy et al. 2015)**

*L'article scientifique complet correspondant à cette partie est présenté en annexe 3 :*

Van Looy, K., Piffady, J., Tormos, T., Villeneuve, B., Valette, L., Chandesris, A., & Souchon, Y. (2015). *Unravelling River System Impairments in Stream Networks with an Integrated Risk Approach*. *Environmental management*, 55(6), 1343-1353.

Grâce aux améliorations techniques et méthodologiques dans la production, la collecte, l'interprétation de données sur de grands territoires (progrès en télédétection et en SIG), des analyses holistiques modélisant les relations entre occupation du sol et état écologique aux différentes échelles emboîtées de l'écosystème rivière ont été menées (Allan et al. 2012; Esselman et al. 2013; Tormos et al. 2014; Villeneuve et al. 2015; Wasson et al. 2010). L'intérêt de ces approches holistiques a été montré notamment pour identifier (i) les grandes structures de paysage générant les pressions sur l'état écologique et (ii) l'échelle de gestion la mieux adaptée (Forbes et al. 2008; Thoms et al. 2007; Vorosmarty et al. 2010).

Toutefois, ces approches holistiques n'offrent pas encore un diagnostic précis des causes d'altérations. Elles ne décrivent pas la chaîne de causalité allant de la structure du paysage et des activités humaines jusqu'à l'altération des processus naturels aux différentes échelles de fonctionnement de l'écosystème rivière. Appréhender ce chainage n'est pas une tâche aisée compte tenu de sa complexité (fonctionnement issu de processus multi-échelles altérés par un paysage contemporain de pressions cumulées, altérations encore observables mais résultant de pressions historiques, etc.). Très clairement, les connaissances actuelles ne permettent pas de quantifier précisément l'ensemble des mécanismes en jeu (Downes 2010; Norris et al. 2012). Certains mécanismes sont encore méconnus, ou difficiles à caractériser par manque de données.

C'est pourquoi nous avons proposé d'établir un cadre général permettant de mieux comprendre et appréhender l'ensemble de la chaîne de causalité en tirant parti des connaissances actuelles et en participant à leur progression. Les gestionnaires ont besoin dès

aujourd'hui d'outils les appuyant sur cette thématique afin d'agir de façon protectrice ou réparatrice sur l'écosystème rivière que ce soit pour des problématiques de biodiversité/conservation ou des problématiques de restauration.

Compte tenu des connaissances actuelles, nous avons défini une démarche méthodologique qui intègre la notion de risque d'altération, ou plus précisément de probabilité d'observer un niveau d'altération. Bien évidemment, plus les connaissances seront importantes pour le mécanisme d'intérêt, plus l'incertitude associée à la relation diminuera. Cette démarche peut par exemple se baser sur les approches hiérarchiques bayésiennes qui présentent l'avantage de pouvoir s'adapter à la complexité des systèmes multistress étudiés (Allan et al. 2012; Stewart-Koster et al. 2010). De par leur structure causale, et leur capacité à intégrer des connaissances provenant tant de l'expertise que de l'expérience, elles représentent un cadre souple pour l'investigation des relations causales, mais aussi pour des tests de scénarios. La description des distributions de probabilités permet d'une part de représenter explicitement les incertitudes associées aux données en entrée et sur les relations entre les variables. De plus, les propriétés des probabilités conditionnelles, sur lesquelles les modèles bayésiens reposent, autorisent à décomposer les modèles systémiques complexes en sous-modèles indépendants, plus faciles à étudier et à relier ensuite les uns aux autres.

Dans la continuité des travaux de cette thèse, nous proposons donc d'inscrire nos travaux futurs dans une démarche méthodologique appliquée à tout ou partie de la chaîne de causalité responsable de l'altération du bon fonctionnement des cours d'eau (Figure 14) et de prendre ainsi en compte trois grands types de processus : (i) physiques, qui déterminent la morphodynamique fluviale et l'habitat aquatique, (ii) biogéochimiques, qui régissent les flux de matière organique et d'énergie métabolique et les paramètres physico-chimiques essentiels et (iii) écologiques, qui conditionnent les possibilités de dispersion et de recolonisation des espèces et la structure des peuplements. Cette méthodologie représente bien sûr le projet d'une équipe de recherche entière.

Cette méthodologie s'organisera en 3 étapes :

1. l'agrégation des descripteurs de conditions naturelles et de pressions humaines



Les conditions naturelles de fonctionnement de l'écosystème rivière et les multiples pressions les altérant dépendent de la structure du paysage et des activités humaines aux différentes échelles de fonctionnement (bassin versant, sous bassin, tronçon de rivière). C'est dans cette étape que nous allons chercher à caractériser ces structures et activités pour chaque condition naturelle et pression humaine d'intérêt (variable responsable d'une altération d'un processus).

## 2. Intégration du risque d'altération d'un processus de fonctionnement

Pour chaque processus de fonctionnement, l'objectif est d'analyser leur risque d'altération en considérant leurs conditions naturelles de fonctionnement et les multiples pressions les altérant aux différentes échelles de fonctionnement (descripteurs de l'étape 1).

## 3. Modélisation intégrée du risque d'altération d'un état physico-chimique, hydromorphologique, biologique ou écologique

L'objectif aussi est d'étudier les relations entre chaque processus. C'est aussi un thème où différents types de modélisations innovantes seront testés et à partir desquels pourront être construites des démarches d'évaluation du risque environnemental intégrées à l'échelle des bassins versants, mixant à moyen terme, physico-chimie, hydromorphologie et biologie, mais pouvant aussi se combiner avec des scénarios de restauration.

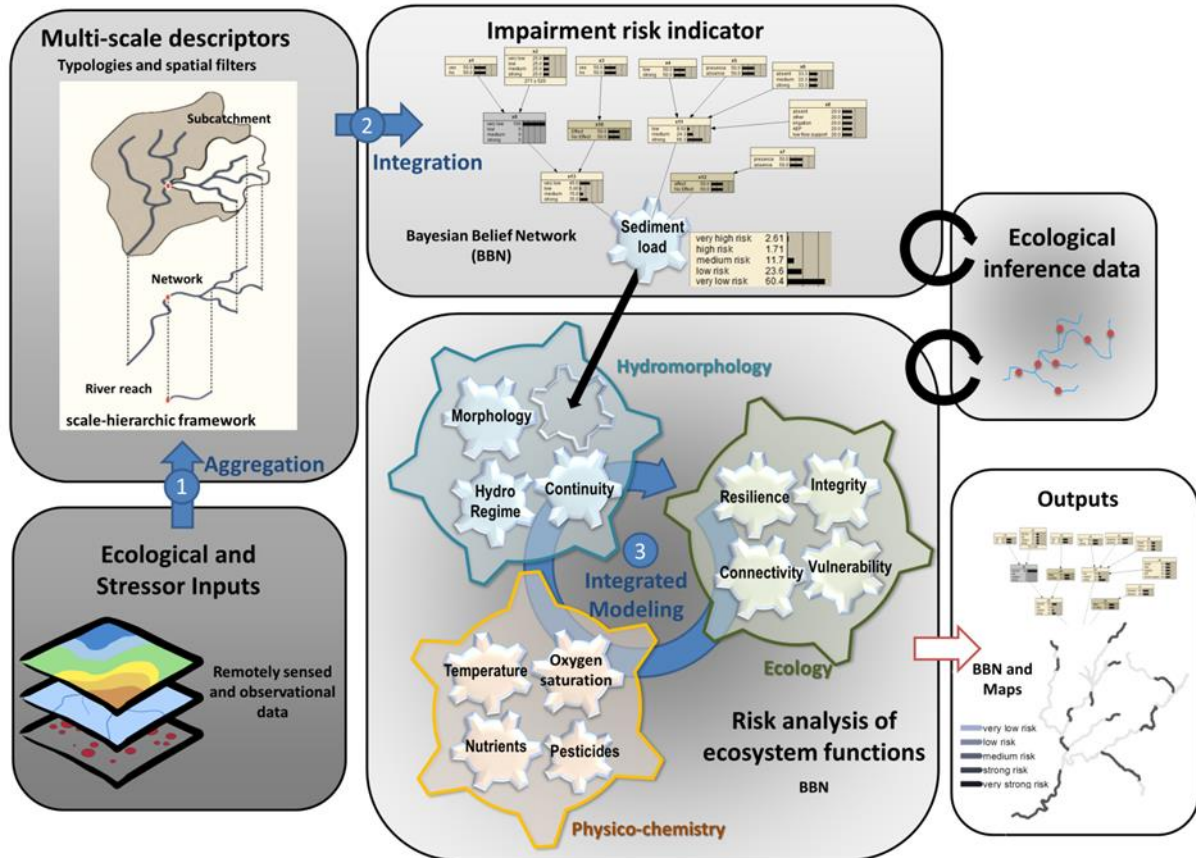


Figure 14. Présentation de l'approche holistique d'analyse du risque d'altérations du fonctionnement de l'écosystème (Van Looy et al. 2015).

## 5 Prospective

L'approche holistique d'analyse du risque d'altération du fonctionnement de l'écosystème cours d'eau que nous venons de développer dans le paragraphe précédent représente le projet collectif du laboratoire d'hydroécologie quantitative (LHQ). Il s'agit d'associer des compétences en écologie aquatique, en écologie du paysage, en géomatique et télédétection, en hydromorphologie et en modélisation afin de nous donner les moyens de répondre aux questions qui restent en suspens après ce travail de thèse.

Nous avons d'abord montré tout au long de ce travail de thèse que les concentrations en nutriments et matière organique jouaient un rôle prépondérant sur l'état écologique des cours d'eau. Or, les excès de nutriments d'origine anthropique, en particulier ceux relatifs à des apports ponctuels ou diffus de phosphore et d'azote engendrent le phénomène

d'eutrophisation qui est la manifestation d'un déséquilibre dans le fonctionnement trophique (dystrophie) des milieux aquatiques.

Pour les cours d'eau, ce phénomène est régi par les concentrations locales de nutriments, qui résultent, outre des rejets de STEP ou des pratiques d'épandages, de facteurs de forçages généraux comme la typologie naturelle (la sensibilité du système n'est pas la même en cours d'eau calcaires ou cristallins), de la propagation de flux (vitesse et distribution spatiale), de la morphologie du cours d'eau (pente, berges et échanges de nappes), et de paramètres thermiques comme la température et le degré d'éclairement. Les cours d'eau présentant des faciès lenticules (ralentissement des flux et donc concentration des nutriments) exposés à la lumière (favorisant la photosynthèse) sont ainsi beaucoup plus sujets aux phénomènes d'eutrophisation (Hilton et al. 2006).

Nous proposons donc de développer un modèle d'analyse du risque d'eutrophisation qui tiendra compte du rôle de ces facteurs de forçages à la lumière des relations multi-échelles directes et indirectes que nous avons mis en évidence dans la thèse.

D'autre part, nous avons montré que l'hydromorphologie jouait un rôle indirect sur les relations entre pressions et état écologique. Cette relation est encore mal connue et demande à être mieux étudiée afin de déterminer les processus mis en jeu dans ce lien. Cela présente une grande importance notamment pour soutenir les actions de restauration physique des milieux. En effet, des altérations telles que la rectification, le recalibrage, l'endiguement, la présence de seuils ou de barrages, l'extraction de matériaux et la suppression de la ripisylve sont des types d'altération fréquemment rencontrés et qui sont de nature à impacter fortement le fonctionnement hydromorphologique du cours d'eau. Nous savons que les opérations de restauration menées dans les cours d'eau impactent à la fois leur compartiment biologique et leur fonctionnement hydromorphologique (Lorenz et al. 2009). Maintenant que nous avons la certitude que l'hydromorphologie a un effet déterminant sur le compartiment biologique des cours d'eau, nous proposons d'étudier plus en détail la nature de cette relation.

De plus, la DCE demande de déterminer des « concentrations seuils » (au sens de point de rupture) pour les paramètres physico-chimiques soutenant la biologie. Aujourd'hui, ces seuils sont déterminés sans tenir compte au préalable des effets inter-échelles entre forces

motrices et pressions que nous avons démontrés dans cette thèse (Bougon and Ferréol 2012). Et la détermination de ces seuils ne tient pas non plus compte des effets indirects de l'occupation du sol et de l'hydromorphologie sur le compartiment biologique. Nous proposons donc de ré-évaluer ces seuils en tenant compte des nouvelles connaissances apportées pendant la thèse, c'est-à-dire en les replaçant dans des typologies de pressions différentes.

Enfin, un facteur de stress important a volontairement été mis de côté dans la thèse : les toxiques. Nous envisageons dans un avenir à moyen terme d'explorer le rôle des toxiques comme les métaux, les HAP, les pesticides et les PCB sur le compartiment biologique dans le système multi-échelles et multi-pressions que nous avons décrit dans cette thèse. Bien entendu, la maturation de ce sujet de recherche devra se faire en collaboration étroite avec des thématiciens en géochimie et en écotoxicologie.

## Chapitre 8 – Conclusion

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Dans cette thèse, nous souhaitons répondre à des questions aussi diverses que :

Comment les organismes aquatiques répondent-ils à des stress de différentes natures ? Ces réponses sont-elles dépendantes des contextes géographiques ou des compositions des communautés ? D'où proviennent les différentes formes de stress, quelles sont les pressions qui en sont à l'origine et comment s'opèrent les combinaisons de ces stress au sein des bassins versants, des corridors rivulaires et des cours d'eau ?

Nous avons pu montrer le rôle de l'occupation du bassin versant et du corridor rivulaire sur l'état écologique des cours d'eau. Nous avons pu montrer l'effet prépondérant des paramètres physico-chimiques et en particulier des nutriments et de la matière organique sur les compartiments biologiques macroinvertébrés, poissons et diatomées, à l'échelle de l'assemblage d'espèces. Nous avons aussi pu révéler l'effet indirect important de l'hydromorphologie sur les assemblages de macroinvertébrés. Et enfin, nous avons montré que ces relations étaient dépendantes à la fois des conditions typologiques naturelles et du contexte de pressions du bassin versant.

Ces résultats posent les bases de connaissances indispensables à la prise en compte du fonctionnement global de l'écosystème cours d'eau dans une analyse de risque d'altération holistique. Cela ouvre la porte à de nombreux développements à visée opérationnelle.



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# Annexe 1 - A new macroinvertebrate-based multimetric index (I2M2) to evaluate ecological quality of French wadeable streams fulfilling the WFD demands: a taxonomical and trait approach.

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## **A new macroinvertebrate-based multimetric index (I2M2) to evaluate ecological quality of French wadeable streams fulfilling the WFD demands: a taxonomical and trait approach**

### **Abstract**

Following the Water Framework Directive (WFD) requirements, we designed a new multimetric index (I2M2) for the invertebrate-based ecological assessment of French wadeable streams. This index should be able to identify impaired reaches for 17 anthropogenic pressure categories potentially leading to water quality alteration or habitat degradation. Based on a national data base, we defined an iterative procedure to select taxonomy- and trait-based metrics exhibiting the best trade-off between (i) high discrimination efficiency, (ii) low specificity and (iii) high stability in least impaired conditions. The I2M2, defined as the best combination of such metrics, has been composed by: (i) Shannon diversity index, (ii) original ASPT score, (iii) the relative abundance of polyvoltine taxa, (iv) the relative abundance of ovoviviparous taxa and (v) taxonomic richness. The I2M2 was tested against an independent data set. It exhibited good and robust pressure-impact relationship for all the pressure categories, correctly identifying in average 82% of reaches impaired by water quality alterations or habitat degradation. The I2M2 significantly improved the detection of impaired reaches by at least 17% for nitrogen compounds and up to 35% for organic micropollutants and clogging risk, when compared to the normalized French biotic index (IBGN). The I2M2 has been proposed for future use in the national biomonitoring of wadeable reaches in the context of the WFD implementation

Keywords: Macroinvertebrates – WFD – Multimetric index – Pressure-impact relationship – Taxonomic metrics – Bio-ecological traits



## **1 Introduction**

The European Water Framework Directive (European Council, 2000) put for the first time the ecological quality in the very heart of the environmental policies of European member states. It requires that countries evaluate the quality of their water bodies using Biological Quality Elements (BQE): i.e. fish, invertebrates, diatoms, plants and phytoplankton. Among BQEs, invertebrates have a long history as part of biomonitoring tools (Hellowell, 1986; Rosenberg & Resh, 1993; Bonada et al., 2006), being the most widely used biological group in freshwater bioassessment of human impact (Norris & Thorns, 1999; Hering et al., 2006b).

The WFD requires that bioassessment methods implicitly evaluate the ecological status of water bodies, by comparing BQEs between an observed versus a reference situation. The reference situation should be representative of near natural conditions. Moreover, this comparison has to take into account the typology of water bodies and the metrics selected to evaluate the ecological status of water bodies, and has to regard abundance, diversity and pollution sensitivity of taxa (See annex 5 in European Council, 2000).

In France, the IBGN method (Indice Biologique Global Normalisé) has been used at the national scale and normalized since 1992 (revised in 2004, norm NF T 90-350 in AFNOR, 2004) but is no longer satisfying due to severe inconsistencies with WFD, e.g. the IBGN index is not type specific: the same scoring system and quality class boundaries are used for all types of rivers without considering “reference conditions”. Moreover the IBGN sub-metrics [i.e. the faunal indicator group (FIG) and taxonomic richness] did not take into account taxon abundances.

To overcome the technical shortcomings of the French biotic index in the WFD implementation framework, the development of a new biotic index, i.e. the MultiMetric Invertebrate Index (I2M2), was decided by the French Ministry of Environment (MEDDLT).

Multimetric indices were first included in biomonitoring approaches with fish communities (Karr, 1981). They have been increasingly used (e.g. Kerans & Karr, 1994; Thorne & Williams, 1997; Buffagni et al., 2004; Böhmer et al., 2004b; Ofenböck et al., 2004; Gabriels et al., 2010) and have become major tools in macroinvertebrate-based biomonitoring within the European WFD framework (e.g. Hering et al., 2004b; Lücke & Johnson, 2009). Indeed a multimetric index has the potential to simultaneously and efficiently evaluate the responses of benthic communities to

different categories of pressure because its individual metrics could consider different attributes of communities that specifically respond to different categories of pressure (Karr & Chu, 1997). Several authors have searched for individual metrics that significantly respond to toxic contamination (Archambault et al., 2010) or hydromorphological alteration (Lorenz et al., 2004) and have included such metrics in biomonitoring tools. Nevertheless, few works have been done to combine in a single index, metrics able to detect a wide range of anthropogenic pressures at large spatial scale [but see, for example, Ofenböck et al., 2004; Buffagni et al., 2004 (organic contamination + hydromorphological alteration) or Böhmer et al., 2004b (organic contamination + acidification + hydromorphological alteration)].

In this work, we aimed at identifying biological metrics (based on taxonomy or life history traits) that significantly respond to 17 pressure categories potentially leading to water quality or habitat degradation. We selected metrics exhibiting the best trade-off between (i) high mean discrimination efficiency, (ii) low specificity and (iii) high stability in reference conditions. We searched for combinations of those metrics that could be relevant for pressure-impact identification in French wadeable stream-types and selected the best metric combination to build the new multimetric index (I2M2). We tested the discrimination efficiency, stability and robustness of this new index on a test data set. In the future intercalibration exercises the new French multimetric index will be compared and intercalibrated to European standards, e.g. with the commonly used European intercalibration multimetric index ICMStar (Buffagni et al., 2006). We already tested the correlation of the I2M2 with the ICMStar. and compared its discrimination efficiency with those of the ICMStar and the former French biotic index (IBGN).

## **2 Material & Methods**

### **2.1 Data collection**

Fieldwork was performed between 2004 and 2009 by 22 regional environmental agencies on a national network. Selected reaches were representative of 57 stream types of the French hydroecoregion-based typology (Wasson et al., 2002; MEDD, 2005; Chandesris et al., 2006) gathering most of the French wadeable rivers (1305 streams, 1725 reaches and 4132 sampling events; cf. Fig.1 and Appendix A).

A 'development' data set was formed by randomly selecting 75% of the reaches (1293 reaches, 3112 samples) from the whole data set, while the remainder (i.e. 432 reaches; 1020 samples) was used as a 'test' data set.

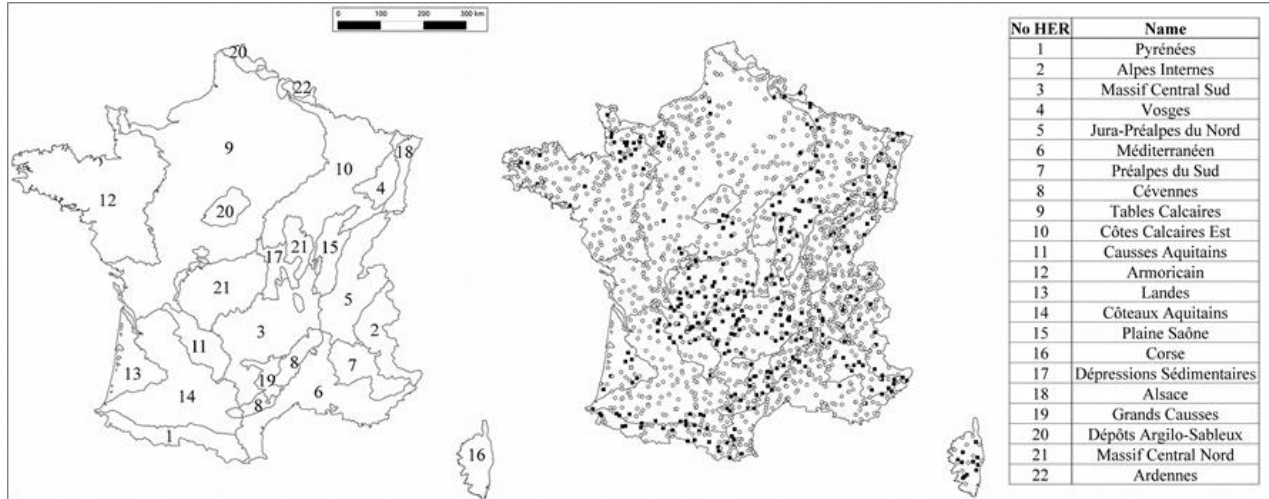


Fig. 1. Map of the French hydroecoregions and location of the sampling sites. Black squares represent least impaired river reaches and open circles impaired river reaches.

Macroinvertebrate communities were sampled in all reaches with a common normalized protocol (Multi-Habitat Sampling, norm XP T 90-333 in AFNOR, 2009). During low flow conditions, twelve sample units per reach were performed on pre-defined mesohabitat types with a normalized Surber net (sampling area 0.05m<sup>2</sup>, mesh size 500µm). Four sample units from 'marginal habitats' (i.e. with an individual share of less than 5% coverage) were selected according to their hosting capacity ('B1' group) and eight sample units were taken from 'major habitats' (i.e. with an individual share of at least 5% coverage). Four of these samples were selected according to their hosting capacity ('B2' group). The last four sample unit were proportionally selected according to the relative coverage of major mesohabitats within the sampling reach ('B3' group), taking into account mesohabitats already sampled in group 'B2' (AFNOR, 2009). Sample units from the same group (B1, B2 or B3) were preserved together with formalin (4% final concentration). In the laboratory, invertebrates were sorted, counted and identified at the normalized taxonomic level [i.e. genus level except for Oligochaeta, some Diptera (mainly family), Trichoptera Limnephilidae, Coleoptera Dytiscidae and Hydrophilidae (sub family); norm XP T 90-388 in AFNOR, 2010; cf. Appendix B].

## 2.2 Reach characterization

Water quality characterization of reaches was performed considering a variable number of parameters among 173 parameters distributed in ten chemical pressure categories (cf. Table 1 & Supplementary material SI) for which information was available in the French water quality assessment system (i.e. Water Quality Evaluation System or SEQ-Eau; Oudin & Maupas, 2003). The water quality status of a given reach at the macroinvertebrate sampling date was estimated by averaging chemical measures from this reach during the six months before faunal sampling [i.e. 4.16 ( $\pm$  2.17) measures available, in average, on this period; a number varying according to the parameter and the reach taken into account].

Land use and hydromorphological characterization was performed considering ten parameters distributed in seven habitat degradation pressure categories (cf. Table 1 & Supplementary material SII). Individual habitat degradation parameters were measured using ESRI's ArcGis 9.2 software (ESRI, 2006). Used geographic data are given in supplementary material SII.

For each available parameter, pressure level was assessed by comparing the parameter measure with the threshold delimiting 'low' to 'moderate' pressure levels (thresholds are given in supplementary materials SI and SII). The pressure level allocated to a given reach for a given pressure category was the worst pressure level allocated to this reach by individual parameters from this pressure category.

Table 1. Water quality and habitat degradation pressure categories taken into account in this study

<b>Water quality</b>	<b>Habitat degradation</b>
<i>Organic matter</i>	<i>Transportation facilities</i>
<i>Nitrogen compounds, except Nitrates</i>	<i>Riverine vegetation</i>
<i>Nitrates</i>	<i>Urbanization</i>
<i>Phosphorous compounds</i>	<i>Clogging risk</i>
<i>Suspended matter</i>	<i>Hydrological instability</i>
<i>Acidification</i>	<i>Catchment anthropization</i>
<i>Mineral micropollutants</i>	<i>Straightening</i>
<i>Pesticides</i>	
<i>PAH</i>	
<i>Organic micropollutants</i>	

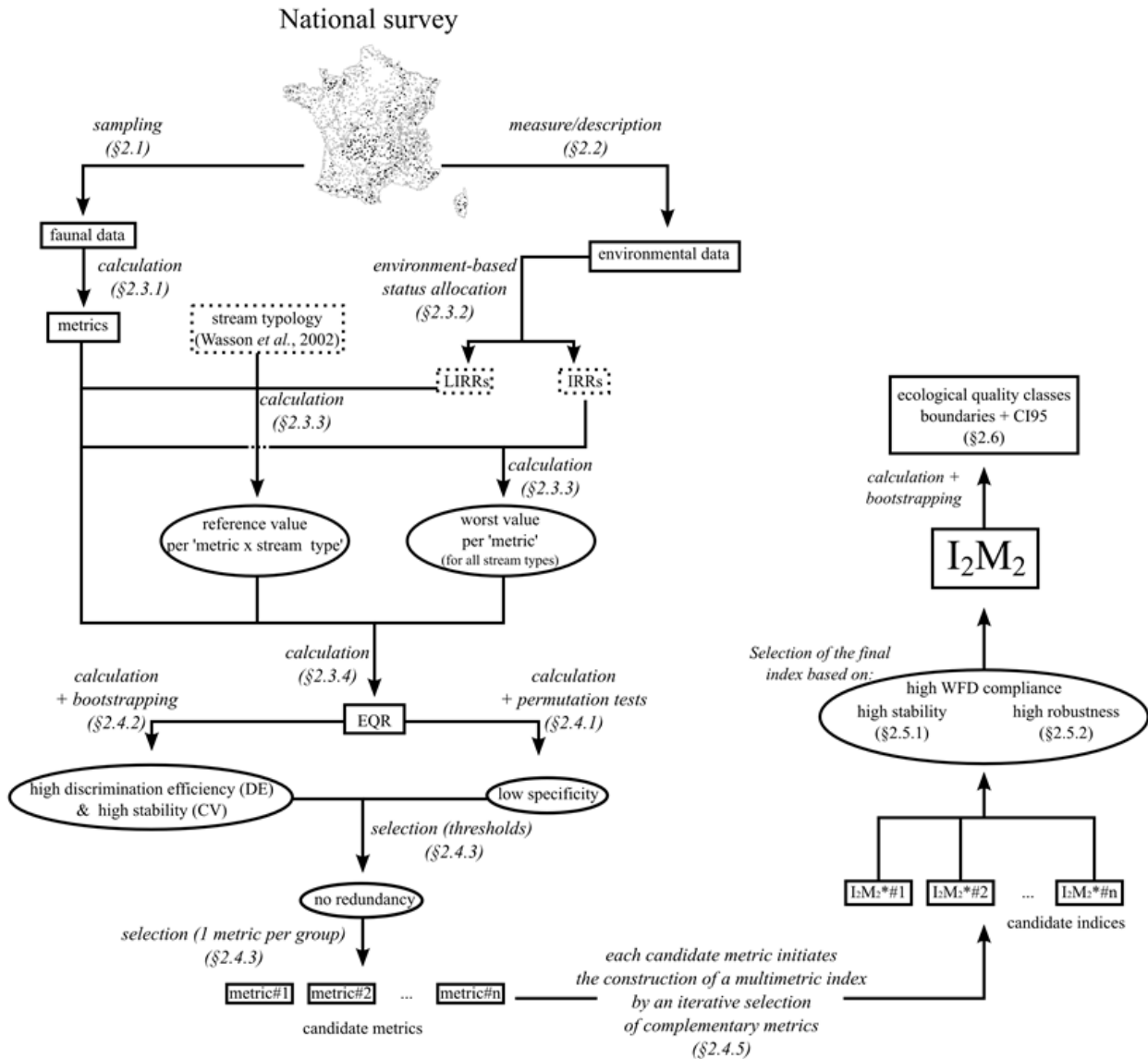


Fig. 2. Flow diagram giving the main steps of the I<sub>2</sub>M<sub>2</sub> development strategy. Literature references and sections from this paper precisely describing each step of the index development design were given into brackets.

To define the new WFD-compliant French biomonitoring tool we followed the recommendations of Barbour et al. (1999) for the development of multimetric assessment methods. Barbour decomposed this process in four main steps: 1) stream classification (cf. Wasson et al.; 2002), 2) metric identification (cf. §2.3), 3) metric normalization (cf. §2.3) and 4) index development (cf. §2.4). In Figure 2, the main steps of the applied design were summarized from data collection to the final index and ecological class boundary definition.

## **2.3 Metric identification and normalization**

### **2.3.1 Metric set**

418 biological metrics (see Supplementary material SIII for a detailed list) were calculated considering sample units from (i) all habitats (i.e. 'reach'), (ii) only 'marginal habitats' (i.e. 'B1'), (iii) only major habitats sampled according to hosting capacity (i.e. 'B2') or relative coverage (i.e. 'B3'), (iv) all major habitats (i.e. 'B2+B3') and (v) all habitats sampled according to hosting capacity (i.e. 'B1+B2'). Three supplementary metrics corresponding to the French biotic index (IBGN) and its two sub-indices were only calculated at the 'B1+B2' level (because best corresponding to the combination of habitats sampled when applying the IBGN sampling protocol). Fourteen metrics were also specifically calculated at the 'reach' level including the Flemish MMIF and its six sub-indices, three metrics measuring the taxonomic specificity of 'B1', 'B2' or 'B3' within the reach, one between-group (i.e. B1, B2, B3) beta diversity measure and three alien species-related metrics. Then, the 2525 'metric x calculation level' (= metrics hereafter) were allocated to 199 groups (cf. Supplementary material SIII), each group being composed of metrics bringing the same [but calculated at different levels, i.e. (i), (ii), (iii), (iv) or (v)] or very similar biological or ecological informations.

In contrast with Barbour et al. (1999), we normalized metrics before selecting the more convenient ones. Indeed, following the WFD requirements, new biomonitoring tools have to be expressed in Ecological Quality Ratio (EQR). This ratio is a number between zero and one, with values from 'reference' reaches close to one and values from reaches with 'bad' ecological status close to zero.

### **2.3.2 'Least impaired' and 'impaired' river reaches**

To define 'reference' conditions, we selected least impaired river reaches (LIRRs, e.g. Statzner et al., 2005; Dolédec & Statzner, 2008) using first, available data on water quality and habitat degradation, then validating reach status evaluation with a reduced set of biological metrics (including IBGN, ASPT, Shannon diversity, relative richness in Ephemeroptera, Plecoptera and Trichoptera taxa, relative utilization frequency of 'oligotrophic' and 'oligosaprobic' trait categories in reach communities). If not matching the criteria for integrating the LIRRs, river reaches were considered as impaired (IRRs).

### 2.3.3 Reference and worst metric values

In a previous work (Mondy & Usseglio-Polatera, 2009) we have already demonstrated that, in LIRRs, the inter-annual variability was negligible when compared to the spatial variability in metric values. As a result, data from all the reach sampling dates were simultaneously analysed, spatial variability being taken into account through the normalization of metrics (cf. §2.3.4).

Depending on pressure category, a given metric could exhibit three major response patterns: (i) not simply and/or significantly responding to the pressure (type I), (ii) significantly decreasing in impaired conditions (i.e. pressure level being at least 'moderate'; type II) or (iii) significantly increasing in impaired conditions (type III).

We identified the response pattern of metrics (i.e. the sense of the deviation from values in LIRRs) by transforming metric values into normalized deviations (SES; cf [1] and Gotelli & McCabe, 2002). SES normalization allowed us to directly compare metric values obtained from different stream types, at large spatial scale.

$$SES = (\text{Obstype} - \text{Mtype}) / \text{sdtype} \quad [1]$$

with: Obstype the observed value of the metric in a given reach, Mtype and sdtype being respectively the mean and the standard deviation of the metric value distribution in LIRRs from the same stream type.

Then, the discrimination efficiencies (DE; e.g. Ofenböck et al., 2004) of metrics were calculated. For a given metric and a given pressure category, DESES corresponds to the proportion of samples pre-assigned to IRRs with (i) smaller values than the first quartile of the LIRRs value distribution (DESES(25), type II) or (ii) higher values than the third quartile of the LIRRs value distribution (DESES(75), type III) (Fig. 3).

Metrics for which neither DESES(25) nor DESES(75) were higher than 0.25 (i.e. the distribution of values from IRRs assemblages was not different from the distribution of values from LIRRs assemblages) corresponded to type I. Metrics for which DESES(25) was higher than both 0.25 and DESES(75) or for which DESES(75) was higher than both 0.25 and DESES(25) corresponded to type II and III, respectively.

Last, we identified the 'best' and 'worst' values of each metric. The 'best' value corresponded to the highest (type I or II) or the lowest (type III) value this metric could take in the LIRRs from a given stream type. The 'worst' metric value corresponded to the lowest (type I or II) or the highest (type III) value a metric could take in the IRRs from the whole data set. The 5th and 95th percentile of the distribution of values for a given metric, were used as 'reference' or 'true worst' (= 'worst' hereafter) values instead of the highest/lowest values to discard metric values of outliers (Ofenböck et al., 2004).

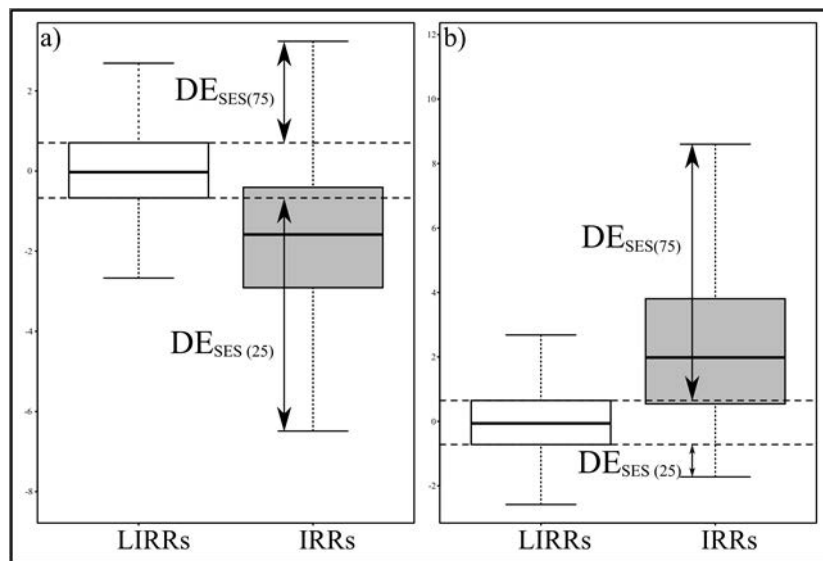


Fig. 3. Discrimination efficiency of normalized metric ( $DE_{SES}$ ) decreasing (a) or increasing (b) with increasing anthropogenic pressure. Boxplots represent SES value distribution of metrics in least impaired river reaches (LIRRs, white box) and impaired river reaches (IRRs, grey box). The boxes range from the 25<sup>th</sup> percentile to the 75<sup>th</sup> percentile. The thick line represents the median and the whiskers extend to extremes values. Black dashed lines represent the 25<sup>th</sup> and 75<sup>th</sup> percentiles of SES distributions in LIRRs

### 2.3.4 Metric normalization

Following Hering et al. (2006a), EQR were calculated using equation [2] for metrics of types I and II and equation [3] for metrics of type III.

$$EQR = \frac{Obs - Lower}{Upper - Lower} \quad [2]$$

$$EQR = 1 - \frac{Obs - Lower}{Upper - Lower} \quad [3]$$



with: 'Obs' the metric value for a given sample. In equation [2], 'Upper' and 'Lower' correspond to the 'reference' and 'worst' metric values, respectively; whereas in equation [3], 'Upper' and 'Lower' correspond to the 'worst' and 'reference' metric values, respectively.

As stipulated in the WFD, EQR values should be bounded between 0 and 1. If observed reaches exhibited metrics values out of the 'reference' - 'worst' interval for the same stream type, the EQR values were arbitrarily fixed as 1 (if higher quality than the reference value) and 0 (if lower quality than the worst value), respectively. This EQR normalization allowed interpreting metric values from a given reach, regarding their deviation from reference conditions associated to the corresponding stream type.

## **2.4 Index development**

Candidate metrics were selected taking into account four criteria: (i) low specificity, (ii) high discrimination efficiency (DE), (iii) high stability in LIRRs, and (iv) no redundancy. As the estimation of the three first criteria could depend on the development data set composition, we limited this bias using i) permutation tests for specificity and ii) bootstrap sub-sampling for robust estimation of DE and stability.

### **2.4.1 Specificity**

A metric was considered as 'specific' if it significantly responded to a low number of pressure categories. We searched for metrics with low specificity, i.e. metrics exhibiting significant difference in the distribution of values in LIRRs vs. IRRs for a high number of pressure categories. These differences were tested for each pressure category and each metric, with a conditional tree approach (Hothorn et al., 2006), i.e. a dichotomic classification method using Monte Carlo permutation tests ( $\alpha=0.01$ , 9999 permutations).

### **2.4.2 Discrimination efficiency and stability in LIRRs**

The DE of a metric for a given pressure category was calculated as the proportion of IRR assemblages with lower EQR values than the first quartile of the LIRR value distribution. The stability of a metric in LIRRs was evaluated using the coefficient of variation (CV) of its EQR value distribution from LIRR assemblages. The calculations of DE and CV were repeated 100 times based on randomly selected sub-samples of 60% of the reach data included in the development data set.

A robust estimation of DE and CV for each metric was obtained by averaging the 100 estimations from corresponding sub-sampled data sets.

### 2.4.3 Selection of candidate metrics

Selected metrics simultaneously exhibited (i) low specificity (significant responses for at least seven from ten 'water quality' and five from seven 'habitat degradation' pressure categories), (ii) high DE (mean robust DE  $\geq 0.6$ ) and (iii) high stability in LIRRs (mean robust CV  $\leq 1/3$ ).

To avoid redundancy, for each of the groups of metrics giving the same biological or ecological information (cf. §2.3.1) only the metric with the highest DE was kept for potential inclusion in the multimetric index.

### 2.4.4 I2M2 calculation rationale

For each pressure category, a sub-index was calculated by averaging the EQR of the selected metrics, each EQR being weighted by its DE for this pressure category, as illustrated in equation [4] for PAH contamination.

$$i2m2PAH = \frac{\sum(DE_{mPAH} \times EQR_{mPAH})}{\sum DE_{mPAH}} \quad [4]$$

with:  $i2m2PAH$ : the sub-index for PAH contamination,  $EQR_{mPAH}$ : the EQR value of the metric 'm' for PAH contamination and  $DE_{mPAH}$ : the robust discrimination efficiency of the metric 'm' for PAH contamination.

The final I2M2 score was obtained by averaging the seventeen sub-indices ( $i2m2_{pressure}$ ).

### 2.4.5 Construction of potential index metric combinations

Each of the 'n' candidate metrics (cf. §2.4.3) initiates the construction of a 'potential' multimetric index (I2M2\*) by an iterative selection of complementary metrics performed on the development data set. To reduce the potential bias of the development data set composition in metric selection, the iterative process includes bootstrap sub-sampling of the development data set.

1. One of the candidate metric was selected as the first metric,
2. a smaller data set was obtained by sub-sampling 60% of the reaches from the development data set,

3. in this sub-set, the I2M2\*(i) and DE(i) values for each pressure category were calculated with the pre-selected metric(s) at step i (i = 1, for the first iterative process of metric selection),

4. the I2M2\*(i+1) and DE(i+1) values were calculated (corresponding to each potential metric combination obtained by adding one of the (n-i) candidate metrics to the metric(s) pre-selected at the beginning of the ith iterative step),

5. the relative increase in DE ( $\Delta$ ) for each pressure category (p) when including an additional metric to the I2M2\* was calculated (equation [5]).

$$\Delta = \Sigma[(DEp(i+1) - DEp(i))/DEp(i)] \quad [5]$$

with: DEp(i+1) and DEp(i) the discrimination efficiency of the I2M2\* related to pressure category p calculated with the selected metrics after (i) and (i-1) iterative metric selections

6. the significance of the increase in DE (considering both water quality and habitat degradation) was statistically tested with unilateral paired Wilcoxon rank sum tests for each additional metric included in the index calculation.

7. the following procedure was applied to select a metric from the set of candidate metrics:

a) if only one candidate metric gave a significant increase in DE, this metric was selected,

b) if more than one candidate metric gave a significant increase in DE, the metric with the highest  $\Delta$  [5] was selected,

8. steps 2 to 7 were repeated one hundred times on randomly selected sub-sets of the development data set and the candidate metric which was the more often selected was included in the I2M2\* index,

9. additional metrics were successively included in the I2M2\*, following steps 2) to 8) as long as the increase in DE, calculated on the development data set, was statistically significant ( $\alpha < 0.05$ ).

10. steps 1 to 9 were repeated using as first metric, each metric from the candidate metric pool.

## **2.5 Final selection of the I2M2 metric composition**

The  $n$  I2M2\* metric combinations were compared considering their (i) mean DE (the highest is the best), (ii) stability (i.e. no significant differences of index scores in LIRRs between the development and the test data sets), (iii) robustness (i.e. no significant differences in DE between the development and the test data sets) and (iv) compliance with WFD requirements.

### **2.5.1 Stability of I2M2\* values in least impaired conditions**

The Kolmogorov-Smirnov test was used to search for significant difference in the distributions of I2M2\* values from LIRRs between the test and the development data sets.

### **2.5.2 Robustness of I2M2\* discrimination efficiency**

The DEs respectively obtained with the development and the test data sets were compared with a bilateral paired Wilcoxon rank sum test.

## **2.6 Ecological quality class boundaries**

As recommended by the WFD, we defined ecological quality classes boundaries (i.e. delimiting 'high', 'good', 'moderate', 'poor' and 'bad' classes). Class boundary identification was based on the distribution of the I2M2 scores from the LIRRs of the development data set. To limit the influence of the development data set composition on the distribution of I2M2 values, a bootstrap sub-sampling approach was used. For each of the one hundred sub-sets (corresponding to 60% of the reaches from the development data set), the 75th and 25th percentiles of the I2M2 distribution in LIRRs were calculated. These values were considered as the 'high-good' and the 'good-moderate' boundaries, respectively. For defining the 'moderate-poor' and the 'poor-bad' boundaries, we divided the I2M2 scoring range between the minimal value (0) and the 'good-moderate' boundary in three equal classes. The robust estimate of each class boundary was calculated as the median of the one hundred estimates, and the corresponding 95% confidence interval was calculated (Fig. 4).

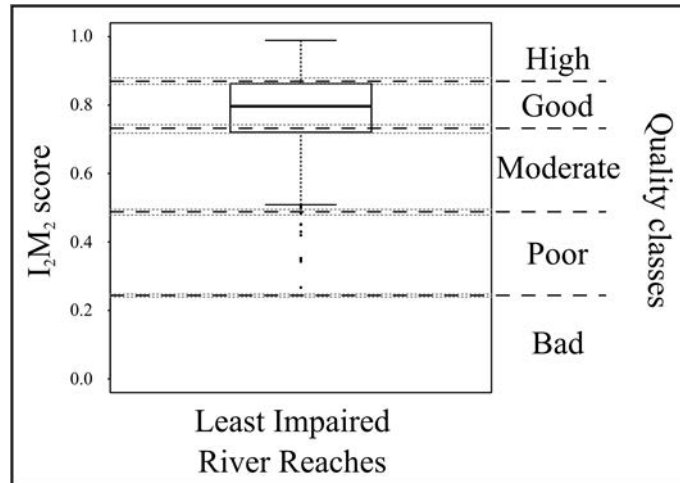


Fig. 4. Ecological quality class boundaries (black dashed lines) and their respective 95% confidence interval (grey dotted lines). The boxplot represents the I2M2 score distribution in least impaired river reaches, ranging from the 25th to the 75th percentile of the distribution. The thick line represents the median. The whiskers extend to the extreme data points but no more than 1.5 times the interquartile range from this box. Black dots represent outliers.

## 2.7 Comparison of the I2M2 with other indices

### 2.7.1 Correlation with ICMStar

The ICMStar was calculated following equation [6]:

$$\text{ICMStar} = 0.167 \times \text{Sfam} + 0.083 \times \text{EPTfam} + 0.083 \times \text{H'fam} + 0.334 \times \text{ASPT} + 0.067 \times (1 - \text{GOLD}) + 0.266 \times \log_{10}(\text{sel\_EPTD} + 1) \quad [6]$$

with Sfam and EPTfam being respectively the total number of families and the number of families within the EPT orders, H'fam: the Shannon diversity index calculated at the family level, GOLD: the relative abundance of Gastropoda, Oligochaeta and Diptera and  $\log_{10}(\text{sel\_EPTD} + 1)$ : the log-transformed sum of Heptageniidae, Ephemeridae, Leptophlebiidae, Brachycentridae, Goeridae, Polycentropodidae, Limnephilidae, Odontoceridae, Dolichopodidae, Stratiomyidae, Dixidae, Empididae, Athericidae and Nemouridae abundances. All these metrics were expressed in EQR (cf. §2.3.4) before their aggregation.

We tested if the I2M2 was correlated with the ICMStar with a Pearson correlation test.

## **2.7.2 Comparison of the I2M2, ICMStar and IBGN discrimination efficiency**

Similarly to I2M2, ICMStar and IBGN DEs were calculated for each of the 17 investigated pressure categories as the proportion of IRRs providing scores lower than the first quartile of the score distribution in LIRRs. We tested the null hypothesis asserting there was no differences in DE between methods (i.e I2M2, ICMStar and IBGN) with a Friedman rank sum test for unreplicated blocked data. This test was followed by a multiple comparison test (Siegel & Castellan, 1988) to localize the potential significant differences between the three methods.

All metric calculations and statistical procedures were performed with R software (R Development Core Team, 2009), using packages 'party' (Hothorn et al., 2006) for conditional trees and 'pgirmess' (Giraudoux, 2011) for multiple comparison test after Friedman test.

## **3 Results**

### **3.1 Index development**

#### **3.1.1 Specificity of metrics**

Among 2525 tested metrics, 475 metrics significantly responded at least to 7 of the 10 water quality pressure types and 5 of the 7 habitat degradation pressure types.

#### **3.1.2 Discrimination efficiency of metrics**

The mean discrimination efficiency of individual metrics ranged from 0.0000 for the relative abundance/richness of several rare groups (taxa, trait categories or bio-ecological groups) to 0.7603 for the richness of the reach EPT taxa identified following the recommendations of Gabriels et al. (2010). Three hundred and ninety three metrics exhibited mean DE greater than 0.6.

#### **3.1.3 Stability of metrics in LIRRs**

In LIRRs, the mean CV ranged from 0.099 (“Belgium Biotic Index” calculated from reach faunal assemblages) to 3.132 (relative abundance of Planipennia in marginal habitat assemblages). Eight hundred and seventy two metrics exhibited rather low variability (CV < 1/3).

### **3.1.4 Candidate metrics**

One hundred and eighty-six metrics fulfilled all the selection criteria. These metrics belonged to thirty groups gathering metrics giving similar bio-ecological information. As a result, the final set of candidate metrics (Table 2) was composed of thirty metrics, each selected metric exhibiting the highest mean DE of its group.

Table 2. Metrics candidate to inclusion in the I<sub>2</sub>M<sub>2</sub>. The (i) full label, (ii) code, (iii & iv) numbers of pressure categories (water quality and habitat degradation) with significant response, (v) mean discrimination efficiency (DE) and (vi) mean coefficient of variation in least impaired river reaches (LIRRs) are given. S = taxonomic richness

Candidate metrics [calculation level]	Code	Nb of pressure categories with significant responses		Mean DE	Mean CV
		Water quality	Habitat degradation		
TAX (number of taxa*) [B1+B2+B3]	C1	8	7	0.6560	0.2861
Shannon Diversity index [B2+B3]	C2	9	7	0.6395	0.2741
Crustacea (%S) [B2+B3]	C3	7	6	0.6379	0.2335
EPT (number of taxa*) [B1+B2+B3]	C4	8	6	0.7603	0.2353
log10(sel_EPTD+1) [B1+B2+B3]	C5	8	6	0.6830	0.1992
revised BMWP [B1+B2]	C6	8	7	0.7303	0.2582
original ASPT [B2+B3]	C7	9	6	0.7479	0.1641
BBI [B1+B2+B3]	C8	7	6	0.6173	0.0995
IBGN [B1+B2]	C9	8	6	0.6440	0.1777
adult, aquatic stage (%) [B1+B2+B3]	C10	7	5	0.6458	0.2457
aerial, active dispersion (%) [B2]	C11	8	6	0.6882	0.1912
Crawler (%) [B1+B2+B3]	C12	8	6	0.6108	0.2495
microphytes as 'substrate' (%) [B1]	C13	7	6	0.6260	0.2528
ovoviviparity – trait 'reproduction technique' (%) [B3]	C14	7	5	0.6749	0.2160
polyvoltinism – trait 'number of cycles per year' (%) [B2+B3]	C15	7	6	0.7300	0.2261
oligotrophic – trait 'trophic status' (%) [B1+B2+B3]	C16	7	6	0.6368	0.2476
a-mesosaprobic – trait 'saprobity' (%) [B2]	C17	7	6	0.6742	0.2244
brackish water preferendum (%) [B1+B2]	C18	7	6	0.7018	0.2139
temporary water preferendum (%) [B2+B3]	C19	8	7	0.6326	0.2469
biological group b (%S) [B3]	C20	7	5	0.6557	0.2152
biological group f (%S) [B1+B2]	C21	7	6	0.6517	0.2619
ecological group B (%S) [B1]	C22	8	6	0.6204	0.2886
bio-ecological group γ2 (%S) [B1+B2+B3]	C23	7	6	0.6292	0.2008
SPEARmetallic [B1+B2+B3]	C24	7	7	0.6336	0.1324
SPEARpesticide I (number of taxa) [B1+B2]	C25	8	6	0.6967	0.2996
redundancy (dispersal) [B1+B2]	C26	7	7	0.6723	0.2420
specialization (maximal potential size) [B3]	C27	8	7	0.6273	0.2208
specialization (salinity preferendum) [B1+B2+B3]	C28	7	6	0.6917	0.1833
specialization (transversal distribution) [B1+B2]	C29	7	6	0.6779	0.2870
specialization (trophic status preferendum) [B1+B2]	C30	9	6	0.6345	0.3106

\* metrics integrated in the Flemish MMIF (Gabriels et al., 2010)  
%S : relative richness



### Final selection of I2M2 metrics

The iterative metric selection process provided thirty indices composed of two to eight metrics. Their mean DE on the whole data set ranged between 0.8107 and 0.8442 (cf. Table 3). The stability of indices in LIRRs was estimated with the Kolmogorov-Smirnov p-value and ranged from 0.0006 to 0.2479. Robustness was evaluated with the Wilcoxon p-value, and ranged from 0.0348 to 1.000.

Five of the 30 metric combinations (i.e. combinations #1, #2, #9, #25 and #30 in Table 3) displayed no significant ( $\alpha > 0.10$ ) stability or robustness differences between the development and the test data sets. Among these five metric combinations, only combination #2 fulfilled all the WFD requirements. As a result, the new multimetric index (I2M2) was finally composed of five metrics (Table 4): (i) Shannon diversity index, (ii) original ASPT score and (iii) the relative abundance of polyvoltine species in the assemblage, all of them calculated at the major habitat scale (i.e. B2+B3), (iv) the relative abundance of ovoviviparous species calculated at the B3 level and (v) a measure of taxonomic richness ('TAX') calculated at the reach level following taxonomic identification levels recommended by Gabriels et al. (2010). These five metrics had homogeneous response patterns for all the pressure categories, three were decreasing [i.e. (i), (ii) and (v); type II] and two were increasing [i.e. (iii) and (iv); type III] with increasing pressure gradient. Reference values for each combination of 'metric x stream type' were given in Appendix C.

Table 3. Metric composition, discrimination efficiency (DE), stability and robustness of the 30 tested metric combinations. See Table 2 for full labels of metrics

Combination	Metric								mean DE	Stability	Robustness
	1	2	3	4	5	6	7	8			
#1	C1	C17	C8						0.8201	0.2057	0.7467
#2	C2	C7	C15	C14	C1				0.8243	0.1617	0.4874
#3	C3	C6	C18						0.8319	0.0855	0.4586
#4	C4	C15	C14	C8					0.8317	0.0246	0.9265
#5	C5	C15	C4						0.8191	0.0542	0.8536
#6	C6	C18							0.8212	0.0846	0.5477
#7	C7	C6	C20	C17					0.8442	0.0213	0.2842
#8	C8	C6	C17	C15					0.8373	0.0069	0.8536
#9	C9	C18	C7						0.8231	0.1145	0.2842
#10	C10	C6	C15	C18	C8				0.8232	0.0300	1.0000
#11	C11	C6	C20	C8	C15				0.8274	0.0374	0.2435
#12	C12	C6	C14	C15	C7	C27	C8		0.8398	0.0179	0.5477
#13	C13	C6	C15						0.8190	0.0359	0.7819
#14	C14	C6	C28						0.8247	0.0120	0.2842
#15	C15	C6	C4	C17	C14	C19	C3	C8	0.8427	0.0176	0.7467
#16	C16	C6	C20						0.8309	0.0478	1.0000
#17	C17	C6	C15	C8					0.8385	0.0069	0.7119
#18	C18	C6							0.8195	0.0846	0.6112
#19	C19	C4	C15	C7	C14				0.8247	0.0259	0.5477
#20	C20	C6	C4	C11	C23				0.8337	0.0189	0.7467
#21	C21	C6	C15						0.8111	0.0362	0.2435
#22	C22	C6	C20	C7	C15				0.8435	0.0006	0.0448
#23	C23	C6	C15	C11	C8				0.8237	0.0304	0.4038
#24	C24	C4	C15	C27	C8	C14	C6		0.8344	0.0190	0.8900
#25	C25	C18	C20						0.8107	0.2051	0.4586
#26	C26	C9	C8	C15					0.8156	0.0073	0.0348
#27	C27	C15	C4						0.8196	0.0388	0.5791
#28	C28	C6	C14	C8					0.8273	0.0095	0.5477
#29	C29	C6	C20	C8					0.8283	0.0509	0.4307
#30	C30	C6	C20	C8					0.8202	0.2479	0.4038

Table 4. Response patterns and discrimination efficiency of the I<sub>2</sub>M<sub>2</sub> individual metrics, I<sub>2</sub>M<sub>2</sub>, ICM<sub>Star</sub> and IBGN. See § 3.1.5 for a full description of I<sub>2</sub>M<sub>2</sub> metrics

	Shannon (B2+B3)	Original ASPT (B2+B3)	Polyvoltinism (B2+B3)	Ovoviviparity (B3)	TAX (B1+B2+B3)	I <sub>2</sub> M <sub>2</sub>	ICM <sub>Star</sub>	IBGN
<i>Response pattern</i>	-	-	+	+	-	-	-	-
<i>Organic matter</i>	0.6743	0.7736	0.7658	0.7580	0.6718	0.8345	0.7931	0.6690
<i>Nitrogen compounds (except Nitrates)</i>	0.7493	0.8542	0.7983	0.7246	0.7512	0.8641	0.8252	0.7379
<i>Nitrates</i>	0.5994	0.7829	0.7430	0.7562	0.6218	0.8276	0.7658	0.6087
<i>Phosphorous compounds</i>	0.6952	0.8557	0.8261	0.7886	0.7061	0.8900	0.8272	0.7225
<i>Suspended matter</i>	0.6691	0.7432	0.8106	0.6806	0.7681	0.8864	0.8011	0.6989
<i>Acidification</i>	0.5229	0.5372	0.6191	0.4924	0.5645	0.7018	0.5965	0.5789
<i>Mineral micropollutants</i>	0.6105	0.7027	0.6494	0.5928	0.6108	0.7577	0.7357	0.5977
<i>Pesticides</i>	0.6911	0.8838	0.8120	0.7309	0.6903	0.9155	0.8592	0.7711
<i>PAH</i>	0.6757	0.7922	0.7869	0.7020	0.7014	0.8864	0.8288	0.7024
<i>Other organic micropollutants</i>	0.5592	0.7608	0.6918	0.6733	0.5844	0.7867	0.7243	0.5826
<i>Transportation facilities</i>	0.6288	0.6513	0.6910	0.5722	0.6718	0.7853	0.7435	0.6675
<i>Riverine vegetation</i>	0.6004	0.6972	0.6600	0.6233	0.6006	0.7547	0.7120	0.5829
<i>Urbanization</i>	0.7109	0.8047	0.7709	0.7094	0.7065	0.8703	0.8388	0.7015
<i>Clogging risk</i>	0.6380	0.8080	0.7747	0.7662	0.6345	0.8618	0.7985	0.6363
<i>Hydrological instability</i>	0.6233	0.6510	0.6579	0.6230	0.6384	0.7609	0.7040	0.6025
<i>Catchment anthropization</i>	0.6257	0.7644	0.7289	0.6897	0.6288	0.8186	0.7748	0.6277
<i>Straightening</i>	0.5984	0.6515	0.6230	0.5901	0.6006	0.7321	0.6917	0.5736

### 3.2 Ecological quality class boundaries

The calculated values of the 'high-good', 'good-moderate', 'moderate-poor' and 'poor-bad' boundaries were: 0.8696 (CI95 = [0.8603; 0.8796]), 0.7327 (CI95 = [0.7195; 0.7411]), 0.4885 (CI95 = [0.4797; 0.4941]) and 0.2442 (CI95 = [0.2398; 0.2470]), respectively.

### 3.3 Test of the I<sub>2</sub>M<sub>2</sub>

#### 3.3.1 I<sub>2</sub>M<sub>2</sub> values in least impaired conditions

The distribution of I<sub>2</sub>M<sub>2</sub> scores from LIRR assemblages in the development and the test data sets exhibited no significant difference (Kolmogorov-Smirnov test:  $D = 0.0805$ ,  $p = 0.1617$ ; Fig. 5).

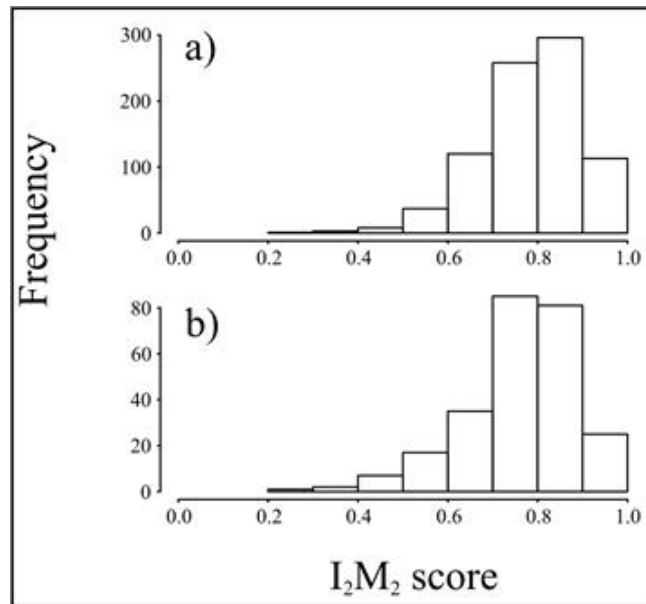


Fig. 5. Histograms of I<sub>2</sub>M<sub>2</sub> score frequency distribution in the development (a) and in the test (b) data sets.

#### 3.3.2 Discrimination efficiency

The difference in I<sub>2</sub>M<sub>2</sub> discrimination efficiency between the development and the test data sets was not significant (bilateral paired Wilcoxon signed rank test:  $W = 61$ ,  $p = 0.4874$ ; Figs 6 & 7).

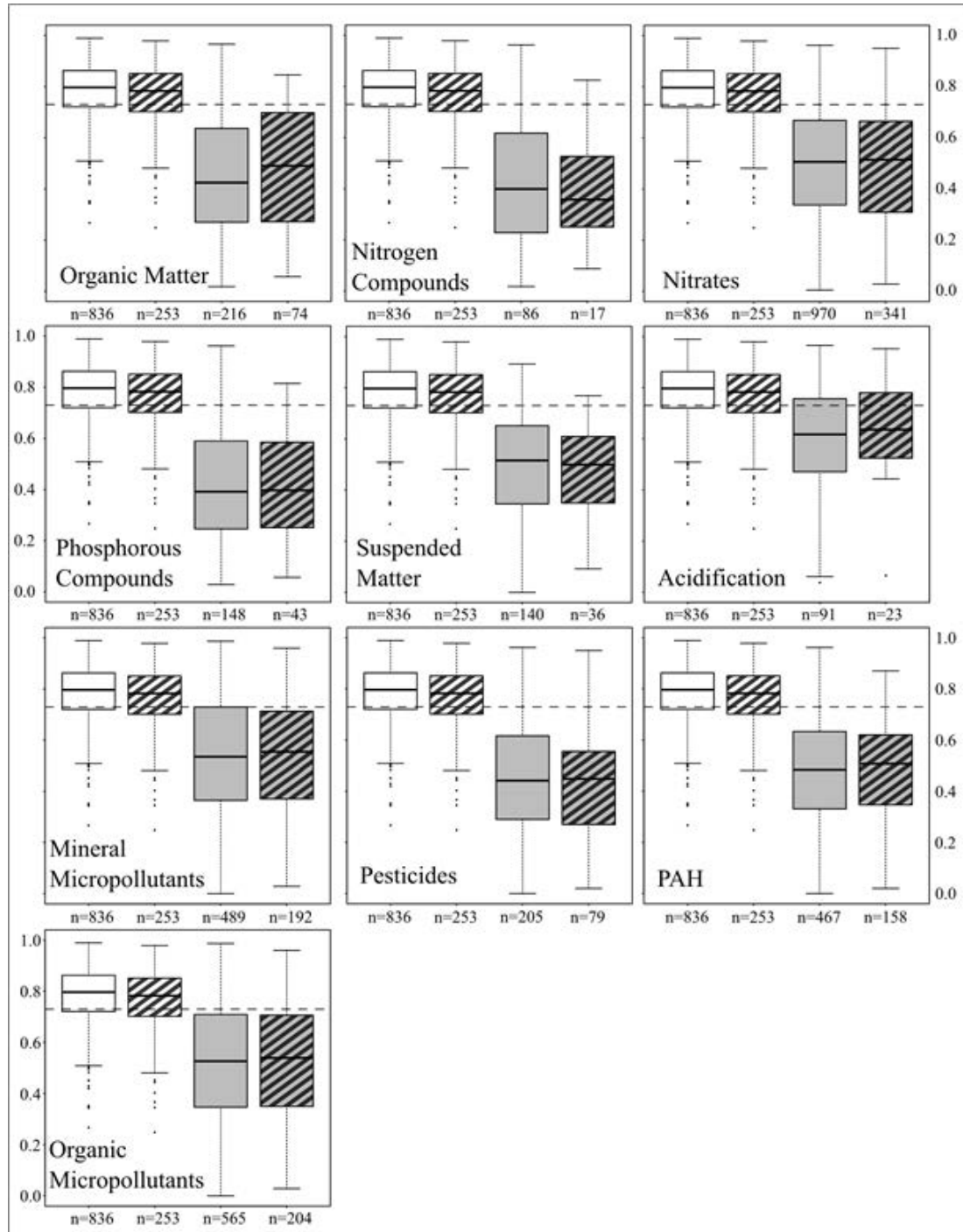


Fig. 6. I<sub>2</sub>M<sub>2</sub> score distributions in least impaired river reaches (white boxes) and impaired river reaches (grey boxes) for 10 different water quality pressure categories. Solid boxes represent the I<sub>2</sub>M<sub>2</sub> score distribution in the development data set whereas stripped boxes represent the I<sub>2</sub>M<sub>2</sub> score distribution in the test data set. Black dashed lines represent the 'low-moderate' pressure level boundary. 'n' represents the number of faunal samples considered in each group. For further details, see Fig. 4 legend.

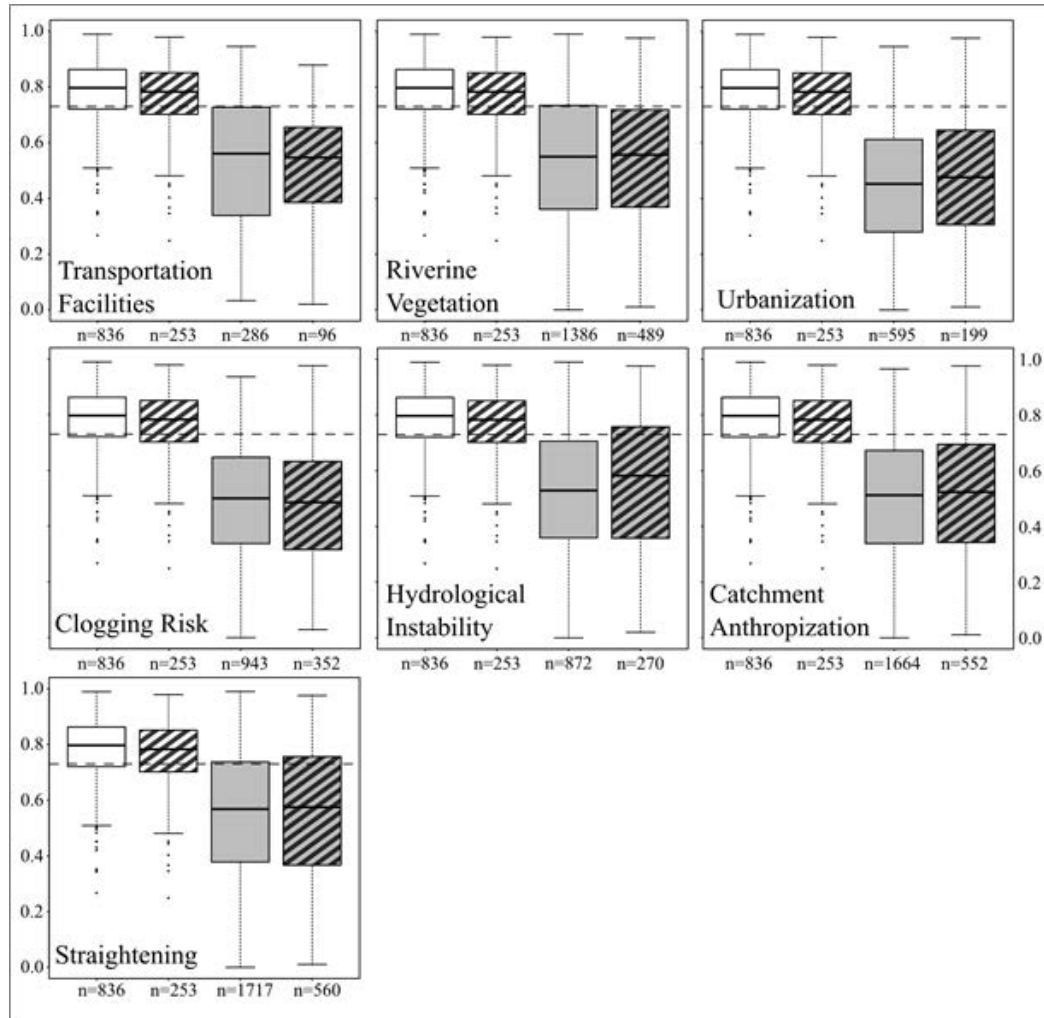


Fig. 7. I<sub>2</sub>M<sub>2</sub> score distributions in least impaired river reaches (white boxes) and impaired river reaches (grey boxes) for 7 different habitat degradation pressure categories. For further details, see Fig. 4 & 6 legends.

### 3.4 Correlation of the I2M2 with the European intercalibration ICMStar index

The I2M2 was strongly and significantly correlated with the ICMStar (Pearson's product moment correlation = 0.9095, p-value < 2.2 10<sup>-16</sup>)

### 3.5 Comparison of the I2M2, ICMStar and IBGN discrimination efficiency

The three indices exhibited significant differences in discrimination efficiency (Friedman rank sum test:  $\chi^2 = 34$ , d.f. = 2, p-value = 4.14 10<sup>-8</sup>; Table 4); the I2M2 (mean DE = 0.820 ±

0.064) better performing than the ICMStar (mean DE =  $0.766 \pm 0.067$ ) and the IBGN (mean DE =  $0.651 \pm 0.063$ ) (cf. Fig. 8; multiple comparison test after Friedman test,  $\alpha = 0.05$ ).

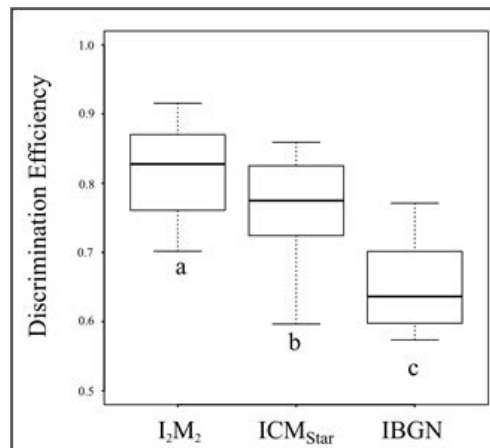


Fig. 8. Distribution of the 17 DEs of I2M2, ICMStar and IBGN. The boxes range from the 25th to the 75th percentile. The median is represented by the black thick line. The whiskers extend to the extreme data points. Different letters indicate significant differences in DE (multiple comparison test after Friedman test, Siegel & Castellan 1988)

## 4 Discussion

### 4.1 Typology specificity

The Water Framework Directive has focused on the need to take into account the specific characteristics of streams from different regions and natural contexts. Many countries using the AQEM approach have defined stream type-specific multimetric indices considering only a low number of stream types: e.g. four in Austria (Ofenböck et al., 2004), three in Portugal (Pinto et al., 2004) and Greece (Skoulikidis et al., 2004) or two in Netherlands (Vlek et al., 2004). Even if 24 stream types have been defined in Germany, type-specific indices have been developed only for five of them (Lorenz et al., 2004).

This strategy seemed quite unsuitable for French streams because of their high environmental diversity (125 stream types have been defined by Chandesris et al., 2006). A not type-specific approach was necessary, as already developed in Germany (Böhmer et al., 2004a; Hering et al., 2004a) or Flemish Belgium (Gabriels et al., 2010). Even after stream typology simplification (57 stream types), defining one specific index per stream type would

be highly difficult due to the low number of available data on reference or least impaired river reaches for several stream types (cf. appendix A). Moreover, large scale (i.e. between-stream types) comparisons of index values would be difficult due to the potential differences in metric composition of stream type-specific indices.

As a result, we preferred to evaluate the ecological status of rivers using a single common set of metrics for all the stream types, taking into account stream type characteristics when normalizing metrics into Ecological Quality Ratios (EQRs).

## **4.2 Pressure specificity**

In their cook book, Hering et al. (2006a) suggested to develop either generalist or pressure-specific multimetric indices. In contrast with several European countries (Böhmer et al., 2004b; Ofenböck et al., 2004; Sandin et al., 2004), we aimed at building a generalist index, usable for a wide spectrum of environmental conditions (not only for a unique combination of ‘bioregion x stressor type’; e.g. Ofenböck et al., 2004), by selecting metrics which discriminate anthropogenic pressure from natural variability for a large number of stream types and pressure categories. This choice resulted from several considerations:

1. Pressure-specific index development needs the selection of sampling sites to ensure that: “environmental stressor gradient[s] is [are] ideally represented by a set of sites of one freshwater ecosystem type covering the whole range [...] of the environmental stressor that is to be targeted by the Multimetric System” (Hering et al., 2006a). The French survey network was not designed to fulfil this requirement. Indeed, in the two main French National survey networks, reaches were selected to be either least impaired (i.e. Reference Reach National survey) or simply representative of the mean quality of the water body they belonged to (i.e. RCS National survey).

2. The search for pressure category specific metrics, would require that reaches included in the development data set be individually impaired by one or a low number of pressure categories. As a result, complex pressure combinations, which often impair river reaches, would not – or not optimally - be taken into account in the development of pressure-specific indices. As an illustration, the 1725 river reaches included in our data base were significantly impaired, in average, by 4.23 ( $\pm$  2.29) of the 17 pre-defined pressure



categories (see also Comte et al., 2010). A generalist index seemed to be much more suitable for identifying multiple pressure scenarii.

3. The concentration of many toxicants still remains very difficult (at low level) and expensive to quantify in water and/or sediment, making them scarcely included in routine survey networks (Kolpin et al., 2002). As an example, mineral micropollutants and PAH were respectively measured for only 39.23% and 35.70% of the samples in the used data base. We hypothesized that, if selected biological metrics significantly respond to a high number of different pressure categories, the multimetric index would have higher chance to identify not targeted (or unexpected) pressure categories.

4. Metrics and indices considered as pressure-specific were, in fact, not so specific. For example, Lorenz et al. (2004) demonstrated that the German Fauna Index, based on taxonomic metrics, designed to specifically identify hydromorphological alterations, significantly responded also to organic contamination. SPEAR indices were developed to identify the biological impact of some specific categories of toxic pollutants: e.g. pesticides (Liess & von der Ohe, 2005); metals (von der Ohe & Liess, 2004), organic micro-pollutants (Beketov & Liess, 2008). Even if they were designed to respond to specific toxic pressure, these indices significantly responded to a more diverse combination of pressure categories, i.e. in average  $6.6 \pm 1.4$  of the 10 predefined water quality pressure categories and  $5.3 \pm 0.8$  of the 7 pre-defined habitat degradation risks, based on our whole data base.

5. Last, we found that metric DE for the different pressure categories were strongly correlated (Pearson's coefficient  $r = 0.92 \pm 0.04$ ). In other words, metrics tended to have similar DE for the different pressure categories. As a consequence, it seemed extremely difficult to identify 'truly specific' metrics able to efficiently discriminate only one or a small group of pressure categories.

### **4.3 Selected metrics**

The final I2M2 was composed of only five metrics. Three of them are taxonomic metrics that have been widely used in biotic indices: (i) the Shannon's diversity index (Shannon, 1948) included in several European multimetric indices, e.g. in Germany (Böhmer et al., 2004b) or in Belgium (Gabriels et al., 2010); (ii) the "Average Score Per Taxon" (ASPT,

Armitage et al., 1983) already involved - sometimes with regional adaptations - in several European multimetric indices developed in the AQEM context, e.g. in Italy (Buffagni et al., 2004), Czech Republic (Brabec et al., 2004), southern Sweden (Dahl & Johnson, 2004) or Portugal (Pinto et al., 2004); and (iii) taxonomic richness, considered as the simplest measure of diversity (Stirling & Wilsey, 2001; Mendes et al., 2008). This metric has been already taken into account in several biotic indices, e.g. the IBGN (France, AFNOR, 2004) and the BBI (Belgium, Pauw & Vanhooren, 1983) and has been included in several multimetric indices (e.g. Royer et al., 2001; Ofenböck et al., 2004; Vlek et al., 2004; Gabriels et al., 2010).

The two last metrics were biological traits, i.e. fuzzy-coded variables (Chevenet et al., 1994) describing various biological attributes of species (Resh et al., 1994, Usseglio-Polatera et al., 2000, Statzner & Bêche, 2010). During the twenty last years, these traits have been increasingly used, first to elucidate the filtering role of habitat on species attributes at various spatial scales (e.g. Townsend & Hildrew, 1994; Townsend et al., 1997; Poff, 1997), then to study the additional filtering role of human activities on biological traits of stream assemblages in a biomonitoring perspective (e.g. Dolédec et al., 1999; Charvet et al., 2000; Usseglio-Polatera & Beisel, 2002; Gayraud et al., 2003; Liess & von der Ohe, 2005; Dolédec & Statzner, 2008; Townsend et al., 2008; Archaimbault et al., 2010; Statzner & Bêche, 2010), but were still rarely included in the composition of multimetric indices. Two trait categories regarding reproduction have been included in the I2M2: (iv) 'polyvoltinism' and (v) 'ovoviviparity'. Polyvoltinism - supposed to ensure a higher resilience capacity - is a reproductive strategy expected to occur with higher frequency in unstable conditions compared to a 'reference' situation (Townsend & Hildrew, 1994), whereas 'ovoviviparity' as parental care strategy, would prevent high mortality at egg stage in harsh environmental conditions. An increase in polyvoltine and/or ovoviviparous species frequency in benthic assemblages has been already observed with different pressure categories (e.g. Usseglio-Polatera & Beisel, 2002; Archaimbault, 2003; Piscart et al., 2006; Dolédec & Statzner, 2008).

#### **4.4 Ecological quality class boundaries**

Because using LIRRs instead of 'true references', we have not divided the 0-1 range of EQR values in five classes of equal range (e.g. Böhmer et al., 2004b; Gabriels et al., 2010) or used the 25th percentile of the reference value distribution as the 'high-good' boundary

before defining the other class boundaries with equal bands (e.g. Munné & Prat, 2009; Poquet et al., 2009). We assigned the 75th and the 25th percentiles of the I2M2 scores in LIRRs to the 'high-good' and 'good-moderate' boundaries respectively; equal bands only defining the three other quality classes. Moreover, in the establishment of between-class boundary, we used a bootstrap sub-sampling method that provided (i) a robust estimate of boundary and (ii) the 95% confidence interval associated with each class boundary (Fig. 4), in agreement with the uncertainty measure around class boundaries required by the WFD (European Commission, 2003).

#### **4.5 I2M2 efficiency**

Testing the I2M2 with an independent data set has demonstrated its stability in LIRRs and its robustness regarding discrimination efficiency. The I2M2 could also be considered as a highly sensitive index, since there is nearly no overlap between LIRRs and IRRs inter-quartiles of index score distribution (Royer et al., 2001). In average more than 81% of the reaches pre-classified as 'impaired' on environmental criteria were also considered as 'impaired' by the I2M2. This high efficiency of detecting a large panel of pressures (even at moderate level) allows considering the I2M2 as a robust and efficient biomonitoring tool (Sandin & Johnson, 2000). Compared to the IBGN, the I2M2 significantly improved the detection of impaired reaches by at least 17% for nitrogen compounds and up to 35% for organic micropollutants and clogging risk.

A reasonable proxy for global anthropogenic pressure on reaches could be the anthropization level of their catchment, evaluated with the addition of the relative surfaces respectively used by urbanization, agriculture and industry. The distribution of I2M2 reach scores among the different quality classes, closely matches the distribution of corresponding pressure levels of catchment anthropization (Fig. 9), then validating the procedure establishing ecological boundaries.

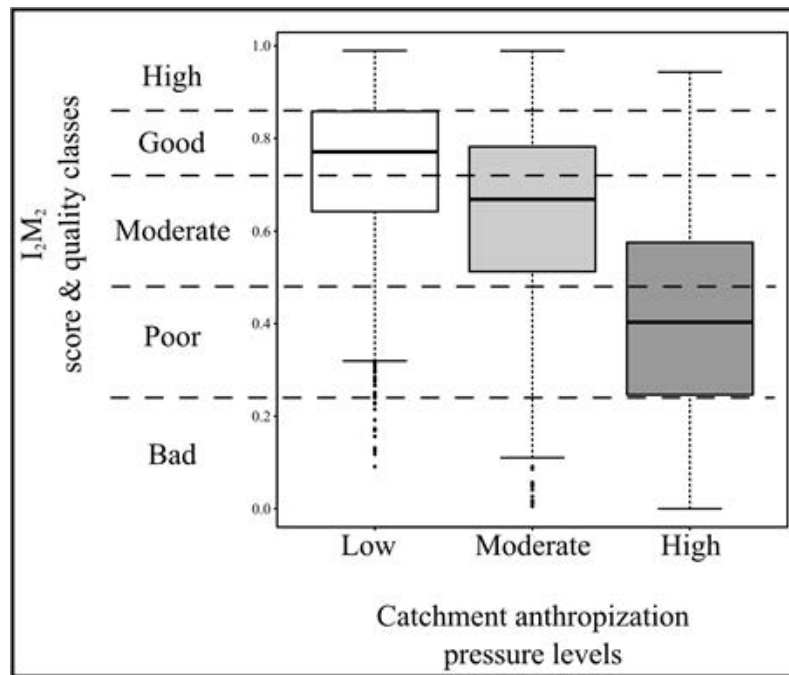


Fig. 9. Boxplot of I2M2 scores in relation to the pre-defined reach pressure levels concerning catchment anthropization. Black dashed lines represent ecological quality class boundaries. For further details, see Fig. 6 legend. Conclusion

The proposed multimetric index (I2M2) (i) completely fulfils the WFD requirements, (ii) significantly improves the detection of impaired reaches when compared to the former French IBGN, (iii) is one of the very first biomonitoring tool designed - from a large national data base - to take into account pressure-impact relationships for a high number of pressure categories (including both water quality and habitat degradation of reaches) and considering both taxonomic characteristics and biological traits of benthic macroinvertebrate assemblages. The I2M2 has been proposed for future use in the national biomonitoring of wadeable reaches in the WFD implementation framework and for integration in the future French online system 'SEEE' (Système d'Evaluation de l'Etat des Eaux = Water Status Evaluation System) that will provide to managers a simple way to calculate this index (among other metrics describing BQE assemblages) after uploading reach invertebrate assemblage abundance distribution. Moreover, the I2M2 is (better performing than and) highly correlated to the European intercalibration multimetric index (ICMStar), which is very promising regarding the future integration of the I2M2 in the European pool of WFD-compliant biotic indices.

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## Annexe 2 - Assessing the ecological status in the context of the European Water Framework Directive: where do we go now?

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## **Assessing the ecological status in the context of the European Water Framework Directive: where do we go now?**

### **Abstract**

The Water Framework Directive (WFD) is now well established as the key management imperative in river basins across Europe. However, there remain significant concerns with the way WFD is implemented and there is now a need for water managers and scientists to communicate better in order to find solutions to these concerns. To address this, a Science-Policy Interface (SPI) activity was launched in 2010 led by Directorate-General for Research and Innovation and Onema (the French national agency for water and aquatic ecosystems), which provided an interactive forum to connect scientists and WFD end-users. One major aim of the SPI activity was to establish a list of the most crucial research and development needs for enhancing WFD implementation. This paper synthesises the recommendations from this event highlighting 10 priority issues relating to ecological status. For lakes, temporary streams and transitional and coastal waters, WFD implementation still suffers from a lack of WFD-compliant bioassessment methods. For rivers, special attention is required to assess the ecological impacts of hydromorphological alterations on biological communities, notably those affecting river continuity and riparian covering. Spatial extrapolation tools are needed in order to evaluate ecological status for water bodies for which no data are available. The need for more functional bioassessment tools as complements to usual WFD-compliant tools, and to connect clearly good ecological state, biodiversity and ecosystem services when implementing WFD were also identified as crucial issues.

Key-words: aquatic ecosystems; bioassessment; science-policy interface; state-of-the-art; WFD.



## **1 European context**

The European Water Framework Directive (WFD), which was adopted in 2000, changed the way European Union (EU) Member States (MS) considered water management by putting ecosystem integrity at the base of management decisions (European Commission 2000). Since then, all MS expended considerable time and resources to collect appropriate biological, environmental and pressure data and to develop operative tools in order to elaborate river basin management plans (Birk et al. 2012). As the magnitude and difficulties of this large-scale endeavour became evident, both the European community and individual MS have funded a large number of research projects, particularly in the areas of ecological assessment and catchment modelling (e.g. Hering et al. 2013).

The WFD was welcomed by many for its innovativeness and the radical shift towards measuring the status of all surface waters using a range of biological communities rather than the more limited aspects of chemical quality or targeted biological components. A WFD-compliant method necessarily complies with the requirement to include all the biological parameters listed in the normative definitions (Annex V), define the reference conditions i.e. pristine conditions and express results as an Ecological Quality Ratio, a relative and comparable measure of quality. Recent years have been pivotal for ecological assessment of water quality in Europe (Nõges et al. 2009, Hering et al. 2010). To this aim, after several years of scientific and technical work as well as important financial contributions from MS, the second round of intercalibration for biological methods was achieved, greatly improving homogeneity in the assessment of ecological status throughout Europe (Birk et al. 2013), for all surface water body categories (rivers, lakes, transitional and coastal waters). A major step was achieved at the end of 2012 with the intercalibration of 230 methods from 28 countries (EC 2013). Nevertheless, further effort is still required as around 100 methods (around 30% of the total) are not developed and/or not intercalibrated (Poikane and van de Bund, personal communication).

A key stage in the implementation of the WFD has therefore been reached, whereby a state-of-the-art evaluation can be undertaken and the scientific outputs required to move forward in water management identified. More and more tools are becoming available for

integrative management of hydrographic basins (Hering et al. 2013). Several recently completed European research projects specifically designed to support water body / catchment management: WISER: “Water bodies in Europe – Integrative Systems to assess Ecological Status and Recovery” (<http://www.wiser.eu/>), MIRAGE: “Mediterranean Intermittent River ManAGEment” (<http://www.igb-berlin.de/mirage.html>), REFRESH “Adaptive Strategies to Mitigate the Impacts of Climate Change on European Freshwater Ecosystems” (<http://www.refresh.ucl.ac.uk/>), etc.), as well as the contributions of national scientific programs have been important to this aim. An efficient strategy for communication and transfer of knowledge has been established, in order to make water managers aware of the availability of these tools and how to use them. If this knowledge and these tools are not taken up, it will constitute an enormous loss of scientific profit for the water community (scientists, managers, end-users, and consumers in a broad sense) as a whole, and will inevitably cause major delays in WFD implementation. Likewise, water managers have identified bottlenecks and barriers to implementation that need to be tackled as a matter of priority.

In this context, a Science-Policy Interface (SPI) activity was launched in 2010 led by Directorate-General for Research and Innovation and Onema (the French national agency for water and aquatic ecosystems), for the period 2010-2012. The Common Implementation Strategy (CIS)-SPI activity aimed to support the river basin management planning process by linking the research needs of end users with scientific research outputs and promoting partnership relations between researchers and policy makers. The first ‘water science meets policy’ event (SPI event) was held on 30 September 2010 and was attended by 150 participants from 15 MS and Switzerland as well as Non-Governmental Organisations, stakeholder groups, the Joint Research Centre, DG Research and DG Environment. The present work represents a summary of the discussion, outcomes and recommendations from the event by the working group A (WG A) of the SPI activity, devoted to ecological status evaluation, and is intended to identify the way forward for ecological assessment over the next decade. To do so, a state-of-the-art of the available scientific knowledge for each issue raised during the SPI activity was achieved, as well as identification of research needs and means of making progress. This study complements the work of Hering et al. (2010) which sought to make recommendations for enhancing WFD implementation 10 years after

its adoption. Three issues were addressed by Hering et al. (2010): (i) the development of assessment methods (including reference conditions, typologies and intercalibration); (ii) the implementation of assessment systems in monitoring programmes; and (iii) the consequences for river basin management plans (such as the design, monitoring and success of restoration measures). As with Hering et al. (2010), this work will be of particular interest for scientists working in the field of aquatic ecology and assessment of anthropogenic pressures impacts, as well as for water managers directly concerned by WFD implementation.

## **2 Outline of the SPI process**

The first SPI event took place on 30th September 2010 in Brussels (Belgium). During this event, there was a relatively balanced representation between researchers and WFD end-users (35% of representatives were from the scientific community and 65% were end-users). End-users category covered all the three main levels of implementers or decision makers in relation to WFD, from the European level to national and river basin levels. They represented different points of view but were all invested in WFD implementation, even if in different ways i.e. from general issues to very specific local ones such as the definition of restoration measures. Research needs were discussed at parallel round tables. The round table themes were aligned with the CIS groups (ecological status, chemical aspects, groundwater, floods, water scarcity and droughts, WFD and agriculture, hydromorphology) together with a number of cross-cutting issues (socio-economics, integrated river basin management plans / management and dissemination). In total, 59 research areas (encompassing about 180 specific research issues) were discussed (the full “Water Science meets Policy” report can be downloaded at: <http://www.onema.fr/IMG/EV/cat1a-13.html>).

For working group A (ecological status), 16 research issues were prioritised and discussed. After the first SPI event was completed, a formal consultation of the ECOSTAT working group was undertaken. ECOSTAT led the intercalibration exercise and brings together stakeholders from every MS, from different interest groups (water managers, scientific experts and representatives of ministries). The SPI process was presented to ECOSTAT members and comments/identification of additional issues were invited. This

resulted in a consolidated shortlist of 10 priority research issues which was finalized at the beginning of 2012. Some discussion points on the SPI process were then made at each ECOSTAT meeting.

From April to June 2012, scientific experts working in different research fields identified were solicited and asked to contribute a state-of-the-art summary, including knowledge gaps for each research issue. These elements were compiled into a synthesis document in late 2012 and transmitted to the Strategic Coordination Group at the end of 2012, as part of the CIS-SPI progress report for transmission to the Water Directors group. Discussion among the different working groups of SPI (chemical aspects, groundwater, etc.) was facilitated during the whole process (notably, the ways to validate the different steps of the process in each group were compared), and the CIS-SPI final activity report was then published (European Commission 2013b; <http://bookshop.europa.eu/en/science-policy-interface-in-support-of-the-water-framework-directive-pbKI3112744/>), bringing together the information collected from the SPI activity.

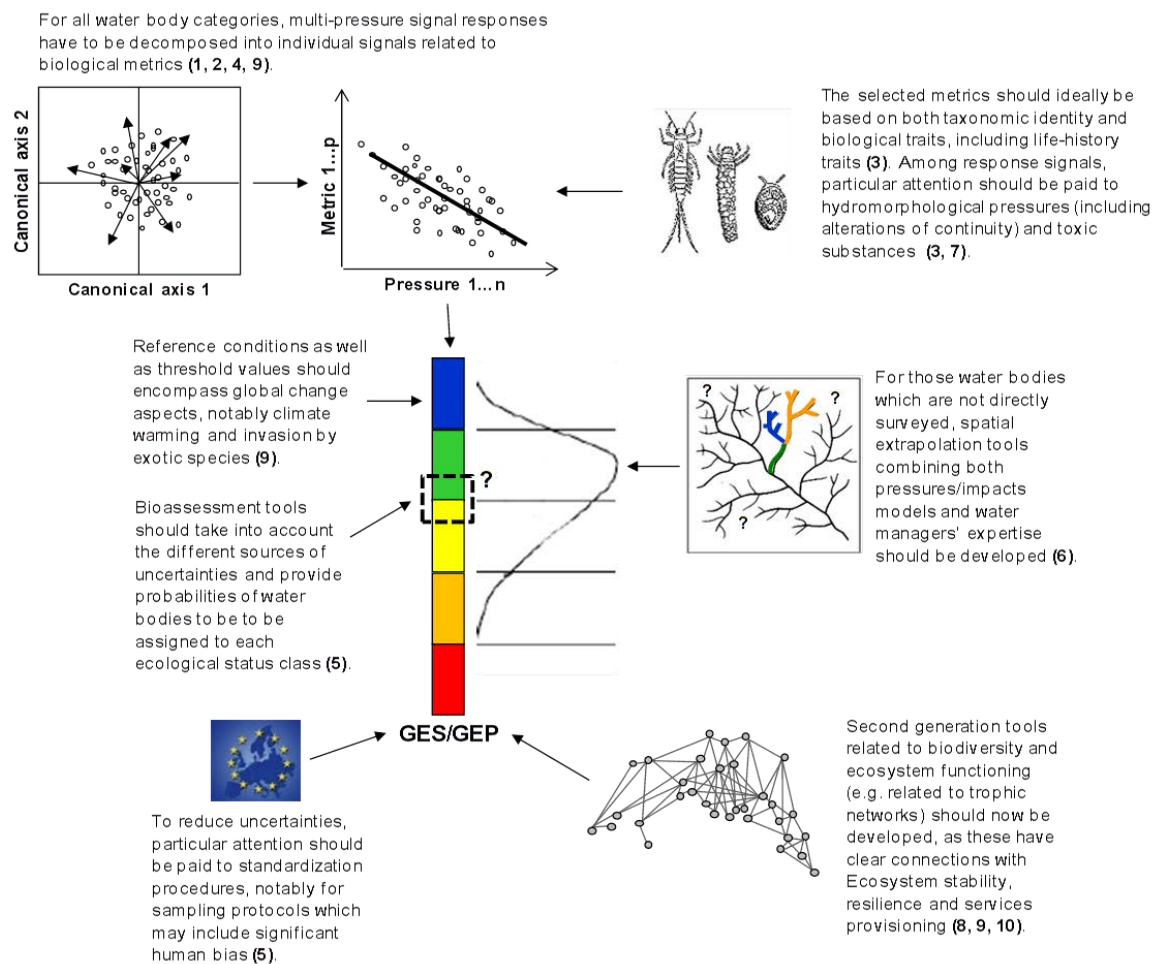


Figure 1. Conceptual scheme illustrating a proposed way forward with WFD ecological status/potential assessment, following the SPI activity. Numbers in brackets refer to the sections in this paper where the corresponding issues are addressed.

### **3 Top-10 Issues**

Following the first SPI event and formal consultation with CIS-ECOSTAT members, the following research issues were identified as of primary concern for enhancing WFD implementation.

#### **3.1 To overcome knowledge gaps for transitional waters**

Transitional waters (i.e. estuaries and lagoons) are usually viewed as complex ecosystems as they constitute ecotones between freshwater and saltwater (they therefore exhibit characteristics from these environments as well as specific ones). Moreover, they are of particular concern for management and conservation issues as they usually shelter very abundant and diversified biological communities (algae, phanerogams, invertebrates, fish, birds, etc.). Within this context, we suggest that the following issues are given consideration in the future to enhance ecological status evaluation and WFD implementation of this water body type.

##### **3.1.1 Multi-pressure context**

Among transitional waters, estuaries are the water bodies most affected by pressures resulting from activities such as dredging, land reclamation, harbor and industrial development, as well as recreational and tourism development have induced major alterations to the original hydromorphological characteristics (EEA 2012, Fehér et al. 2012). The water quality of these environments is also affected by important of pollutants from domestic and industrial effluents (Borja et al. 2011), and is strongly dependent on the chemical fluxes coming from the upstream areas of the drainage basins (Masson et al. 2006). At last, the ecology of estuaries has been subjected to intense human influence through commercial harvesting for many years, from intensive commercial harvesting and aquaculture (Sousa Leitao and Gaspar, 2007). One of the challenges in transitional waters, and given the difficulty of distinguish between the effects of natural and anthropogenic stressors in these environments (the so called “Estuarine Quality Paradox”; Elliott and Quintino, 2007), is therefore to underpin decision making, risk assessment and management of these systems under complex multiple stress conditions. Research should enhance the

understanding of multiple stressor interactions (Thrush et al. 2008, Ban et al. 2010). In this context, some biological quality elements (BQEs) may be more sensitive to some of the pressures than to others (e.g. phytoplankton and macroalgae to eutrophication; seagrasses and fish to habitat loss or hydromorphological changes).

### **3.1.2 Reference conditions**

Estuaries are very dynamic water bodies because of their location at the interface between freshwater and saltwater. They are therefore prone to considerable physical and chemical variation, notably water depth and velocity and associated chemical parameters (salinity, conductivity, etc.). Besides, the transitional waters fish component comprises marine, estuarine and freshwater species, and estuarine phytoplankton and nutrients are influenced by catchment run-off and marine/tidal flushing (Elliott and Whitfield, 2011). Hence, it is difficult to detect the anthropogenic influence against a background of natural variation in these very dynamic environments (Elliott and Whitfield 2011). These natural variations involve some difficulties to define reference conditions for estuaries, all the more that pristine or minimally disturbed estuaries are rare at the European scale. This leaves 'Best Professional Judgment' as the most practical tool for setting reference conditions in transitional waters (Basset et al. 2013). Stoddard et al. (2006) defined Best Professional Judgment as judgment based on “experienced aquatic biologists, with perhaps decades of experience sampling and examining physical, chemical, and biological attributes across wide ranges of severity and types of human disturbance” who developed “an empirical understanding of condition in the absence of significant human disturbance”. BPJ has proven to be very useful in assessing the status of areas across United States and Europe, with a common set of criteria among different experts (Borja et al. 2012). Hence, exploring the use of these alternative options in setting reference conditions should be encouraged.

### **3.1.3 Estuaries as a part of the river basin management**

We know that climate, oceanic, riverine and catchment factors control a hierarchy of processes and broadly determine the physical and biological characteristics of estuaries (Hume et al. 2007). Hence, when taking measures to lessen or remove human pressures, these factors must be taken into account in order to ensure successful recovery (Borja et al. 2010). A better knowledge of the environmental and biological relationships across the river-

estuary-coastal continuum is therefore needed for a better understanding of the response of estuarine systems to the measures and the recovery processes. Truly integrated coastal management, where terrestrial and marine managements are considered together should be a priority for understanding the impacts and the effects of mitigation strategies (Meiner 2010).

## **3.2 To overcome knowledge gaps for lakes**

Over the last decade, WFD-compliant assessment methods for lakes have been developed for most BQEs, supported by major European projects, e.g. REBECCA and WISER (Lyche-Solheim et al. 2008, 2013). There are now close to 100 different national assessment methods for lakes across Europe (Brucet et al. 2013). Despite this indubitable progress, the following recommendations can be made to enhance WFD implementation.

### **3.2.1 Improving indicators**

Different BQEs in lake assessment are needed to address different anthropogenic pressures (e.g. phytoplankton usually addresses eutrophication while benthic invertebrates are good indicators of acidification and morphological degradation), different habitats (e.g. benthic invertebrates and macrophytes are used to assess littoral habitat, while phytoplankton is used for pelagic areas), different timescales (e.g. phytoplankton may react immediately to the modifications of the environment while macrophytes respond more slowly to the changes in pressures) and different ecosystem services (e.g. harmful algal blooms are crucial indicators for recreation and provision of drinking water while condition of alimentary resources is mainly related to fish). Most methods available to date mainly address eutrophication (or general degradation), whereas lakes are generally affected by multiple stressors, climate change including browning of the water, biological manipulation (e.g. fish management and macrophytes removal) and hydromorphological alterations (European Environment Agency 2012). Lake assessment tools should now be developed to address pressures other than nutrient enrichment and to disentangle the effects of multi-stressors. Special attention should be paid to cyanobacterial blooms as they threaten ecosystem services and ultimately human health (Carvalho et al. 2013), as well as to macrophytes and macroinvertebrates growing and living in the littoral areas as they are



sensitive to water level fluctuations and alteration of the littoral habitats (Mjelde and Hellsten 2013).

Some of the pressure/impact relationships established to develop the national BQE methods are not yet sufficiently robust, and often defined using small datasets. Moreover, the large diversity of methods and strategies used (Brucet et al. 2013) and the large number of national lake types (i.e. 673) where the majority is not clearly linked to the common intercalibration types restricts comparability of class boundaries, as well as performance of methods in different situations. Thus, better and more harmonized metrics for all BQEs must be developed allowing more robust pressure/impact relationships to be defined. This is vitally important in the context of revising the current nutrient standards and to ensure that these are correctly linked to the intercalibrated good/moderate class boundaries, as nutrient standards are the benchmark used to plan the programmes of measures.

Management practices for fish have not been fully considered in the development of bioassessment methods based on fish communities, despite the possibility that this could be a major source of bias and/or variability in the pressures/impacts models (Argillier et al. 2002). It is necessary to distinguish the impacts of these activities from those of other human pressures on fish-based indicators to improve the predictive power of the indices for both European natural lakes and reservoirs.

### **3.2.2 Revisiting European lake typology**

There is an urgent need to harmonise lake typologies across Europe, to ensure more comparable reference conditions and to establish environmental target values. The difficulty of achieving 'type-specific' assessment in some cases was already emphasized by Hering et al. (2010) in their critical review of the WFD. A first step in this process could be to identify a small number of broad lake types across Europe to allow grouping of existing national types into a less detailed classification, while a second step could be to harmonise the ranges used for the most commonly factors used (e.g. alkalinity, colour, mean depth, altitude, surface area) to establish the typologies.

### **3.2.3 Ecological assessment methods in the Eastern Continental and Mediterranean regions**

There have been additional difficulties with the development and intercalibration of ecological assessment methods for the Eastern Continental and Mediterranean lakes, notably naturally eutrophic lakes in the continental lowlands (Borics et al. 2013). As a result, the nutrient content of these lakes, even in a relatively natural state, can be high (TP > 100 µg L<sup>-1</sup>) but no significant relationships may be found between phosphorus, chlorophyll, and macrophyte coverage (Krasznai et al. 2010). As well as different targets, these lakes may require different management measures as a large enough reduction of nutrient loading to result in an expected response may not be feasible within a realistic time scale (Borics et al. 2013). Identifying functionally-different lake types and setting eutrophication targets for these lakes is an important area that requires further research.

### **3.3 To analyse the links between ecotoxicology and bioassessment tools**

The implementation of several European new regulations (e.g. WFD, REACH) still requires important efforts to develop ecotoxicological risk and biological assessment tools for water bodies. Ecotoxicological Risk Assessment (ERA) is primarily a predictive discipline that attempts to predict the future consequences of water or sediment contamination by chemicals from agriculture, industry, or any other uses. On the other hand, bioassessment in a WFD context mainly focuses on patterns and processes that occurred in the past or are occurring in the present, which resulted in the current ecological status of the water bodies. However, there are predictive approaches in bioassessment as well as retrospective methods in ERA where these two research-and-practice disciplines merge (Verberk et al. 2013). An international consensus now supports the need to move from a fundamentally toxicology-based to a more ecology-based risk assessment of chemicals (Artigas et al. 2012). In this context, the following recommendations can be made.

#### **3.3.1 Pushing forward the development of trait-based bioassessment tools**

Biological traits are well defined, measurable properties of organisms that strongly influence organismal performance, and constitute the causal mechanisms underlying the relationships between the species and their environment. As a result, trait-based

approaches have clear potential in environmental diagnoses and prognoses (Dolédec and Stutzner 2008, Mondy et al. 2012), notably by (i) providing mechanistic understanding allowing cause-effect relationships between stressors and biological impairments to be inferred; (ii) enabling the evaluation and prediction of anthropogenic impacts at large spatial and temporal scales, and (iii) establishing links between community organization and ecosystem goods and services. Biological traits (e.g. life-forms and cell-sizes for diatoms, relative abundance of invertebrates from “plurivoltine” species or species with “aquatic passive dispersal” or using “ovoviviparity” as reproduction technique) offer the potential to resolve the effects of multiple stressors (Mondy and Usseglio-Polatera 2013), given the diversity and specificity of trait responses (e.g. Schäfer et al. 2011). However, they have been rarely included in WFD compliant methods, except feeding habits or guilds based on food preferences for invertebrates and fish (e.g. Böhmer et al. 2004), or longevity, age structure and reproductive or migration behaviour for fish (e.g. Pont et al. 2006). Besides, multi-metric tools facilitate the diagnosis of the most probable cause(s) of ecological status degradation at a given site or water body, and therefore relevant selection of appropriate restoration measures (Hering et al. 2010). For instance, if a biological metric sensitive to toxics is impacted on a given water body, it can be assessed with the knowledge of pressures likely to explain failure of the environmental targets in mind, and restoration measures selected accordingly. Multi-metric tools based on biological traits are therefore a promising research avenue for the ecological evaluation of water bodies in a multi-pressure context.

### **3.3.2 Reconnecting chemical and ecological evaluations**

There is a disconnection between the evaluation of ecological status on the one hand and chemical status on the other hand. Chemical status does not provide water managers with a straightforward means to efficiently reduce the discharges of harmful substances. This is especially true regarding sediment contamination, for which BQEs such as macroinvertebrates which are directly in contact with the sediment would be useful. The reason for this disconnection is that the procedure for chemical status evaluation is often based on an ERA which cannot be compared with the observed values of ecological indices calculated on the basis of data gathered from surveys. Thus, one drawback in the WFD as it stands is the difficulty with defining relevant environmental quality standards for chemical substances that are truly related to the ecological assessment results. To overcome this, a

potential approach could be to compare the reaction of both biological indicators and biomarkers with a common set of toxics at different levels, from large-scale statistical data analyses to in situ comparative studies and mesocosm experiments. This would provide important fundamental and applied knowledge that could facilitate the diagnosis of water status.

The ecological vulnerability of ecosystems may be viewed as the combination of potential exposure risk, trait-based sensitivity and recovery capacity, and be predicted using auto-ecological information on internal metabolism, regulation capacity, toxicological sensitivity of individuals and traits (DeLange et al. 2010). The development of integrated methods enabling the assessment of ecosystem vulnerability and resilience to toxic substances, and increasing understanding on ways to reduce vulnerability, is challenging but offers considerable potential for surface water management. Within this context, it is important to consider life-history theory issues and associated specific traits (fecundity, egg size, triglyceride contents, etc.), which determine both resource allocation and energy pathways in a dynamic evolutionary context, as these govern individual fitness and, ultimately, demographic issues (see Stearns 1992). It would be also important to consider the concept of Pollution Induced Community Tolerance (the PICT approach), taking into account the potential replacement of sensitive species by more tolerant ones (especially in photoautotrophic communities) following chronic contamination for providing ecologically relevant predictions of toxic effects in the environment (Pesce et al. 2010).

### **3.4 To overcome difficulties in assessing ecological status in temporary streams**

Temporary streams comprise half the global river network and this proportion is predicted to increase due to global change (Carlisle et al. 2010). The recurrent cessation of water flow of temporary rivers influences biotic communities as well as nutrient and organic matter processing (Larned et al. 2010). Several studies have focused on the highly adapted biological communities that live in these streams (reviewed in Lake 2011). However, until now they have not been fully integrated into water regulations because most water managers apply perennial river management principles when making decisions related to

temporary streams. Based on this, we recommend the following issues are given further consideration.

### **3.5 Developing specific bioassessment methods**

The definition of six aquatic states (hyperrheic (floods), eurheic (continuous flow with riffles), oligorheic (connected pools), arheic- (disconnected pools), hyporheic (no surface water, alluvium saturated) and edaphic (alluvium not saturated)) by Gallart et al. (2012) summarized the set of aquatic mesohabitats which occurs on a given stream reach at a particular moment (flood conditions, riffles, connected or disconnected pools and dry-bed), depending on the hydrological conditions. In this context, while a myriad of methods are available to establish the ecological status of permanent streams (Birk et al. 2012), no methods are defined for ephemeral streams and very few for intermittent streams. In certain cases, the same methodologies used for permanent streams could be of use, notably when the stream has been in a eurheic or oligorheic state for a sufficiently long period (Garcia-Roger et al. 2011). For other cases there is an urgent need to develop robust and specific bioassessment tools, as well as to incorporate expert knowledge in the diagnosis (European Environment Agency 2012). Other promising approaches would consider hyporheic communities in the bioassessment tools developed for this type of streams (notably phytobenthos and invertebrates) as it is well-known that these organisms may survive in the substratum even during long dry events, as well as the use of molecular methods (Biomonitoring 2.0; Baird and Hajibabaei 2012). Given the particularly high natural variability of temporary streams, it is of particular importance to ensure biological data are adequately collected and reliable information about pressures is available before making any development, whatever the method being favoured.

#### **3.5.1 Optimizing ecological assessment relevance**

The innate variability that characterises temporary streams provides challenges for the assessment of their ecological status because reference conditions may vary between seasons, dry and wet periods, and after hydrological events in the same river type (Munné and Prat 2011). Ecological status assessment in temporary streams requires preliminary analysis of the hydrological regime (permanent, intermittent with pools, intermittent dry

and ephemeral) as well as knowledge of the aquatic state of the stream at least three months before sampling (Gallart et al. 2012). Once this analysis has been done, the sampling calendar is crucial as temporary streams should be sampled during the flowing phase but at least one month after the spring floods. In this case the standard ecological assessment methods for permanent streams may be used. Under other circumstances (e.g., floods or droughts), the aquatic organisms are subjected to pressures and comparisons with ecological status in permanent streams are less reliable. It is important therefore to determine whether or not the intermittence of a given stream is natural or not, as subsequent interpretations and recommendations would differ from one case to another one. In this context, the hydrological status concept has been introduced (Cazemier et al. 2011) allowing reconstruction of the natural hydrological regime of temporary streams and predictions about changes in the future using rainfall-runoff parameters. Another key issue is the requirement for improved knowledge of ecological status during the arheic (only isolated pools) or dry states and there are currently methods in development for both cases (Steward et al. 2011). In this context, the use of functional measures has some promise but in all three cases we are still far from implementation and more research is needed.

### **3.5.2 Defining a temporary stream typology for Europe**

The presence of temporary streams in the hydrographical network of drainage basins is a characteristic shared by numerous MS across Europe, not only in Mediterranean areas. Even if temporary rivers are very frequent and abundant in Southern Europe (e.g. in SE Spain even some large catchments > 50 Km length are intermittent or ephemeral), there are also many temporary streams in non-Mediterranean or mountain areas across Europe. For instance, it has been shown by Datry et al. (2011) that the density of temporary streams in the Mediterranean areas of France (a dry climatic zone) is comparable with Brittany (a climatic zone with high levels of precipitations). Within this context, it would be very useful to map the temporary streams at the European scale and to propose a European typology for this type of streams (The RM5 river type envisaged in the intercalibration exercise is a mixture of river types and was not intercalibrated for such reason; Feio et al. 2014). This would greatly facilitate the definition of reference conditions for temporary streams and subsequently ecological status thresholds.

### **3.5.3 To take into account uncertainties in ecological evaluation**

The importance to deal with uncertainties was already pointed out by Hering et al. (2010) in their review of the difficulties met when implementing WFD. Uncertainties in ecological state assessment may have many sources. As an illustration, guidance 13 of the Common Implementation Strategy for the WFD listed several potential sources of variation for a given water body (European commission 2005): daily and seasonal patterns, longer temporal trends e.g. climate warming, variation with sampling location, bias from equipment, etc. These different types of variation can be clustered into three different categories, i.e. natural temporal variability, natural spatial variability and human bias. The following ways to deal with these sources of variability could be explored.

### **3.5.4 Improving skills**

The best way to limit human bias is to improve skills in applying both the sampling protocols and evaluation tools. This can be achieved by standardizing methods, both at the European level (i.e. European Committee for Standardization) and at national levels (national standardization committees), and by producing best practice documents for several key steps, from selection of sampling sites to protocol application and taxonomic determination (in laboratories or directly onto the field depending on the BQE). Once the protocols have been standardized, training people in the application of field or laboratory protocols as well as quality assurance procedures are crucial to enhance confidence in ecological evaluations.

### **3.5.5 Developing “stable” bioassessment tools**

Once protocols have been standardized, people adequately trained and quality assurance procedures adopted, it is clearly needed that water managers rely on “stable” bioassessment tools, i.e. tools which take into account the natural variability of communities and do not assign bad status for wrong reasons. Typically for rivers, a high flow event can drastically affect communities but be considered as natural as long as it fits with the natural hydrological cycle observed at the (sub-)basin scale (i.e. the so-called “Natural flow regime”, Poff et al. 1997). It should therefore not result in a water body being declassified for bad reasons (some species might be favoured by high flow events while other might collapsed). These natural abiotic events (e.g. high flow events, “hot” years) are likely to have

direct and indirect effects, the latter by creating conditions which modify biotic interactions, which may in turn affect population dynamics and ultimately ecological quality class. It is therefore important that bioassessment tools integrate natural variability in their development (e.g. Marzin et al. 2014), so that water managers can rely confidently on the ecological status evaluation being performed. In this context, it is becoming increasingly important to consider uncertainties associated with global warming and their influence in the shift of reference conditions (Logez and Pont 2013).

### **3.5.6 Developing “probabilistic” tools**

The final output of WFD biological assessment is a 'quality class' (from high to bad) for each individual BQE. Within this context, and because of the different sources of uncertainty in ecological evaluation, it is now very important to provide water managers with tools providing probabilities for the different status classes rather than single 'simplistic' assignments. For instance, for a water body the outcome of a tool providing the probabilities (from high to bad) of: 30–50–10–10–0% would be very different from another water body where the same tool gives probabilities of: 20–30–20–20–10%. In both cases the water body would be classified in good status, but while in the first case the diagnosis (80% chance of at least good status) will be very reliable, the second provides only a 50-50% chance to be either in at least good status or less than good status. The use of probabilistic tools for one (Marzin et al. 2014) or, ideally, all BQEs of a given water body (Marzin et al. 2012) provides a promising approach as it would give much more confidence to water managers when designing their programmes of measures, therefore limiting some of the drawbacks associated by the one-out, all out principle (Hering et al. 2010).

### **3.6 To develop models for the spatial extrapolation of ecological status**

During the past 10 years, most of the WFD oriented research has focused on the development of bioassessment tools for the different BQEs using data from national monitoring networks. However, a large number of water bodies are not directly monitored across Europe. Within this context, it is of crucial importance to develop efficient extrapolation methods which enable a diagnosis to be made without directly measured



biological and chemical data, for all water body type categories. The following approaches can be recommended.

### **3.6.1 Developing models relating biota to pressures at different spatial scales**

There is a clear lack of predictive tools linking pressures to ecological status at large or regional scales. Recently developed tools offer both scientists and water managers a means to do this. For instance, Donohue et al. (2006) showed that urbanisation, arable farming and extent of pasturelands are the main factors affecting ecological status of streams in Ireland, and that the probability of a river site achieving good status can be predicted using simple models requiring widely available landcover data or chemical data. In Denmark, Kristensen et al. (2012) developed models capable of predicting the presence of different stream fish based on land use data at different scales, showing that the presence of assemblages can be predicted with high accuracy. In France, Villeneuve (2010) developed models predicting the ecological status of rivers based on biological indices calculated using macroinvertebrates, phytobenthos and fish data using variables related to land use, hydromorphological features and physico-chemical conditions. The whole of these examples illustrates the types of predictive models that should be pushed forward as they could be particularly useful for implementing the risk assessment procedure and identify water bodies which are likely to fail environmental objectives (Article 4 and annex II 1.5 of the WFD).

### **3.6.2 Integrating expert judgment in diagnosis tools**

Another approach to assess the ecological status of non-surveyed water bodies involves using expert judgment in the diagnosis (European Environment Agency 2012). Models may provide water managers with objective predicted data/outputs for a given sampling site or water body, but these results may be very variable in terms of usefulness depending on the resolution of data used to build the model (the “calibration” dataset). For example, some models may provide insights for nutrient contents or degree of river chenalisation, but if they were developed using data collected at large spatial scale (i.e. at catchment scale or even national scale) or with relatively low resolution, they may be inappropriate for delivering appropriate information at the local scale where diagnosis needs to be made. Taking into account expert judgment could reduce this bias, by directly injecting knowledge on local context/pressure in the diagnosis parallel to model outputs. However, this option

may lead to a heterogeneous assessment across MS as practices may differ among water managers, so that specifying the way expertise applies should be addressed through normative documents or technical European guidances. Bayesian modelling, which enables to directly inject expertise in the model outputs, would also be a promising research avenue (Marzin et al. 2014).

### **3.6.3 Improving (sub)basin-scale nutrient models**

Physico-chemical data are often measured at the monitoring network site level. Yet, physico-chemical parameters, as well as hydromorphology, must be considered in the WFD as supporting elements governing the development of the biological communities. In this context, models capable of simulating physico-chemical variables at the river basin scale (notably nutrients) based on basin-specific hydrological features as well as land use and population density are now required (e.g. INCA model; Whitehead et al. 2002, MONERIS model; Behrendt et al. 2003). This is a prerequisite for implementation of real basin-integrated approaches in the MS, greatly facilitating comparisons from one country to another. Furthermore, these models should be reliably linked to the biological elements of status, to generate efficient programmes of measures.

## **3.7 To better understand hydromorphological impacts**

River hydrology and habitats have been substantially altered during the last century, while eutrophication, toxic substances and emerging stressors contribute to the complex set of pressures affecting our rivers. The benefits of wastewater treatment have been thoroughly documented over the last 30 years. In contrast, the response to hydromorphological restoration was shown to be more complex and less predictable (Vaughan et al. 2009). Thus, there is a great need to better understand and predict the costs and benefits of future river hydromorphological restoration projects (Buijse et al. 2005a, Lake et al. 2007), as there is a lack of knowledge in several key areas including: (i) the characterisation of hydromorphological status focuses on pattern not processes; (ii) data collected represent small spatial scales, with larger spatial scales or long term impacts often neglected; and (iii) the way hydromorphological change affects BQE and ecological functioning is inadequately understood. The Heavily Modified Water Bodies (HMWB), which

represent a significant proportion of water bodies across Europe, are particularly concerned by these issues (European Environment Agency 2012). For HMWB, a “good ecological potential” must be reached in the same way as good ecological status for natural water bodies but only few member states actually do have approaches to achieve it. Based on these concerns, the following recommendations can be made.

### **3.7.1 Restoring continuity: sediment transport, water flows and fish migration**

There is a need to understand more thoroughly the processes and dynamics of sediment transport, hydraulic connectivity and flow regimes within river systems (Gurnell 2014). The use of demonstration projects to improve or restore sediment transport would provide the framework to improve existing knowledge. This should be fully evaluated and shared among MS. There is also a need to assess the impacts and effectiveness of the removal of in-stream structures in terms of fish migration, especially downstream in large rivers (e.g. for sturgeon, eels and salmon smolts), which remain poorly studied. While the ecohydraulics of fish passes for large migratory fish and salmonids are well known, these remain unclear for smaller species (Santos et al. 2012). Rivers present a multitude of small obstacles and the effect of these on the ecohydraulics of migration upstream remains poorly assessed for every kind of species (salmonids and non-salmonids). Moreover, most migratory species are strongly threatened at the European scale, and restoring continuity therefore constitutes an urgent and imperative need to keep these species from becoming extinct.

### **3.7.2 Linking geomorphology, habitats and biology**

There is an urgent need to gather scientific evidence to illustrate how geomorphology supports biota and to improve understanding the links between morphology, habitats and biology. It is important to examine closely the geomorphological functioning of systems beyond a simple description of the geomorphological conditions. Many of the existing tools only give a description of condition rather than an understanding of functioning. There is a crucial need to understand the hydromorphological and biological responses to new modifications of the water environment and future environmental change. Examples of such recent and upcoming modifications encompass modern interventions for flood protection (e.g. 'room for the rivers'), land use changes affecting sediment budgets in streams and rivers (e.g. reforestation of mountain regions resulting from depopulation) and

rehabilitation measures to regain dynamic processes (e.g. dam removal, remeandering and naturalizing riparian zones). Again, demonstration projects provide an excellent way to fully understand these links (Verdonschot et al. 2013).

### **3.7.3 Terrestrial and aquatic interface and catchment management**

There is a need to understand the linkage between the terrestrial catchment and the aquatic ecosystems, including the impact of different land management practices on the water environment, which could give insights into the restoration of the different parts of a catchment (e.g. the headwaters, floodplains and riparian corridors). Some water bodies may depend upon the chemical or ecological, as well as hydromorphological status of other water bodies situated upstream (and possibly downstream, e.g. regressive erosion). There is a need to fully understand the three dimensions of connectivity of river systems (i.e. upstream-downstream connectivity, floodplain connectivity and links to groundwater) to better inform WFD assessments and select mitigation measures.

### **3.7.4 Rehabilitation in degraded ecosystems**

In heavily degraded and multi-stressed systems, there is a need for a specific restoration approach, a ‘reasonable’ objective being more an ‘ecological improvement’ rather than a ‘complete restoration’ of the water body (e.g. in systems exhibiting heavy flow regulation due to several dams) (Buijse et al. 2002; Buijse et al. 2005b). In particular, there is a research gap in our technical understanding of how to rehabilitate very large rivers, especially heavily incised river systems. This has relevance to the definition of “good ecological potential” for HMWB as the different practices in the different MS need to be compared and discussed.

## **3.8 To develop functional assessment tools**

Many bioassessment tools based on metrics reflecting community attributes (taxonomical composition, traits occurrence, etc.) have been developed under the WFD (Birk et al. 2012). These methods generally reveal organism stress along gradients of environmental quality and deviation from reference conditions. Even if these tools are widely recognized as useful in a management context, they only partially highlight specific changes in ecosystem functioning and processes, even if some promising multimetric tools

have sometimes been developed taking into account biological traits related to functions (e.g. reproductive strategies, feeding, migratory capacities; Dolédec and Statzner 2008, Mondy et al. 2012). However, when trying to restore ecosystem integrity, understanding the mechanisms that drive ecosystem functioning and stability/resilience can certainly help to identify the specific pressures responsible for the observed patterns. Here we suggest different ways to evaluate and quantify ecosystem functioning.

### **3.8.1 Focusing on ecological fluxes and trophic networks**

One of the most conventional ways to assess ecosystem functioning is to focus on trophic networks and fluxes of matter and energy through the system. These fluxes are generally characterised by the levels of primary and secondary production, the efficiency of matter and energy transfers from one trophic level to another, and the biomass of the organisms belonging to the different trophic levels. For instance, the carbon flux can be characterised by the Lindeman efficiency  $\alpha$ , which is the ratio of total metabolic energy fluxes at trophic level 1 to those at level 0 (Lindeman 1942). The  $\alpha$  ratio is generally calculated between the biomass of adjacent trophic levels. A more sophisticated way to proceed is to analyse global photosynthesis and metabolism of the ecosystem via the evaluation of gross and net primary production, ecosystem respiration, and CH<sub>4</sub> efflux (Yvon-Durocher et al. 2010). These variables are known to be sensitive to anthropogenic pressures and particularly to organic pollution and global warming (Young and Collier 2009, Yvon-Durocher et al. 2011), and have potential as functional indicators. The use of stable isotope analyses to characterize the general structure of trophic networks is now also acknowledged to be a relevant indicator ecosystem stability and resilience (Arreguín-Sánchez 2014). At last, the rate of litter decomposition, as it reflects processes acting at a very small spatial and temporal scales (i.e. microbial activity), may bring very early warning signals of increasing anthropogenic pressures (Gessner & Chauvet, 2002).

### **3.8.2 Integrating size as a functional metric**

Size is a key biological characteristic in ecology, and is required as a normative condition in WFD (i.e. for fish-based bioassessment tools). Many ecological properties are governed by body size or body mass, from individuals (e.g. fecundity) to populations (e.g. population growth rate), or communities (e.g. inter-specific competition). Consequently, size structures

have been used to evaluate ecosystem functioning (San Martin 2005), and may be particularly useful for situations with low taxonomic richness (e.g. fish in some Alpine rivers). Evaluation using size may be based on the relationship between size classes and normalised abundances, with the slope of the linear relationship being considered to account for the efficiency of biomass transfer across the different trophic levels, while the intercept is believed to give an idea of the total abundance of organisms (San Martin 2005). With increasing productivity, the determination coefficient usually tends to decrease (indicating that the system is far from a stable state) as well as the slope, and conversely for the intercept. Methods based on allometric relationships (i.e. changes in organisms in relation to proportional changes in body size) and size analysis also provide an interesting framework to assess ecosystem functioning, as they offer opportunities to estimate the ecological impacts of various anthropogenic drivers (Basset et al. 2012). The conjunction of allometric approaches and size structure analysis offers a potential way to develop functional bioassessment tools related to ecosystem resilience and stability.

### **3.8.3 Going deeper into thermodynamic concepts**

Another way to assess ecosystem functioning which has been given some consideration recently is the use of thermodynamic concepts. Ecosystems can be considered as open and dissipative systems as defined by Prigogine. As a consequence, there is a conceptual framework where thermodynamics and ecology are linked. Most of the research has focused entropy (Margalef 1996) and more recently on exergy (Jørgensen 1992). For instance, it has been shown that the eco-exergy tends to decrease with increasing chemical pollution (Xu et al. 1999, 2011). However, if approaches based on thermodynamic principles are attractive, they suffer from a complicated theoretical background the difficulty involved in associating a decrease in eco-exergy to a specific pollution or disturbance. Nevertheless, it is important to develop these research approaches as physical and biological systems are intrinsically connected and thermodynamic concepts can be related to restoration efforts and programmes of measures.

### **3.9 To clarify the links between global change and ecosystem functioning**

Unprecedented global changes resulting from societal use of natural resources will profoundly impact on aquatic ecosystems over the next century with major consequences for ecosystem structure and function (Sala et al. 2000). Climate change, water resource requirements, land use change, connectivity losses, alterations in global circulation of nitrogen and carbon, breaching of biogeographical barriers and biotic exchanges will result in major stress affecting biodiversity (including freshwater, estuarine and marine components) and ecological status at a global scale, despite policy-driven efforts to minimize these. These changes are likely to affect the implementation of the WFD quality assessments (Wilby et al. 2006, Hering et al. 2010) and the maintenance of good status. To tackle these changes, the following key research areas should be investigated in the very near future.

#### **3.9.1 Clarifying the nature and combination of effects from global stressors interactions**

The nature of interactions between global stressors is still mostly speculative (Kernan et al. 2010). Some global changes may even be beneficial and compensatory, while in other cases primary global changes will trigger secondary stresses, as human society adapts. A major response to the predicted changes in seasonal variability and decreases in precipitation in Southern Europe will be water storage in existing and newly built reservoirs with subsequent alteration of flow regimes. The success of biotic introductions will vary according to environmental conditions and will be related to the degree of human activity but also to secondary pressures resulting from global change (Jeppesen et al. 2011). Global changes may trigger a series of stressor feedbacks and additive, synergistic or antagonistic interactions propagating across ecosystem compartments. Increased understanding of such dynamic stressor networks is required in order to increase our predictive capabilities at a global scale (Feld et al. 2011).

#### **3.9.2 Identifying best available conditions and developing indicators for global pressures**

WFD Planning depends on the use of reliable indicators of quality, measured against reference conditions (Hughes 1995) likely to evolve following global changes. Thus, the decoupling of effects due to natural temporal variability (e.g. demographic issues related to

high flow events or “hot” years) from those resulting for anthropogenic changes on a longer time period (e.g. inter-annual trend due to climate warming) requires urgent attention (Nestler et al. 2010). We should be able to predict the ways and degree by which reference communities are affected by global stressors, how these alterations affect the ecological assessment of status, and how robust our assessment systems are. Long-time series of biological change in near-natural ecosystems used with modelling forecasts are invaluable tools to understand how global changes interact with natural variability. Developments with palaeo-limnological methods mean this approach can now be used to assess alternative recovery targets, notably for lakes (Bennion et al. 2011).

### **3.9.3 Decoupling global non-biological and biological disturbances**

Anthropogenic introduction is the main cause of non-native fish species invasions (Leprieur et al. 2008). Worldwide, barrier removal has resulted in a general homogenization of freshwater faunas (Rahel 2007). We need to understand cause-effect relationships between alien invasive species and global changes, potential conflicts (for example, barrier breaching as opposed to damming) and interactions with local scale disturbances (Strayer 2010). In many European bioassessment methods, non-natives are usually included in community metrics and not considered as aliens likely to degrade ecological status (Birk et al. 2012), despite Kennard et al. (2005) demonstrated they can be seen as a reliable ‘first cut’ indicator of river health. We need to understand when biotic exchanges are a primary symptom of global change, or rather resulting from ecosystem cascading effects. Finally, we need to allow for the potential for species migration as a result of climate change, by encouraging habitat protection and connectivity and to understand and accommodate the difference between benign migration and real invasions.

### **3.9.4 Incorporating global changes uncertainties into restoration planning**

Some global change mechanisms remain difficult to forecast, notably variations in precipitation, runoff and phreatic levels (Johnson and Weaver 2009). The impacts will depend on interactions among drivers, and there is a need to quantify the uncertainties inherent to global change (Adger 2006). Ecological assessments result in uncertainties related to the assessment method, sampling strategy, and seasonal variability (Johnson et al. 2006). Nonetheless, uncertainty will determine the comparability band for the



intercalibration of quality assessments (Bennet et al. 2011). Therefore, we need to disentangle uncertainties resulting from current monitoring approaches and those related to global change (Lempert et al. 2004), and to investigate the usefulness of top-down prediction-based assessments for local decision-making (i.e. large-scale and remotely obtained indicators and patterns, enabling us to be less dependent from local monitoring data which usually incorporate a large ecological noise resulting from sampling and local biological variability).

### **3.10 To reinforce the knowledge on relationships between Good Ecological Status, biodiversity and ecosystem's services**

The concept of ecosystem services (ES) is currently recognized as having great potential for examining the interaction between ecosystems and human well-being (National Research Council 2005, Cardinale et al. 2012). Following the Millennium Ecosystem Assessment (MA 2005) and the The Economics of Ecosystems and Biodiversity (TEEB 2010) report, the ES concept has been taken up more widely in environmental planning and in national and international policy obligations (i.e. the MAES initiative, Maes et al. 2012).

The European Biodiversity Strategy to 2020 is clearly a key driver for using the ES approach but other key policy areas such as WFD also provides opportunities for that purpose. The Blueprint to Safeguard Europe's Water Resources (European Commission 2012a) describes actions to improve the implementation of Europe's water policies, including the need for cross-cutting problem solving. In this context, the integration of an ecosystem services approach (ESA) in WFD implementation is envisaged to improve the effectiveness of its implementation, as the ESA could help frame the WFD objective of "good status" under a broader social and economic context. Additionally, there are strong links between the concept of ES and several aspects of the WFD, namely the economic analysis, exemptions, concept of water services, pricing and cost-effectiveness of the programmes of measures.

Based on this, the following recommendations can be made.

### **3.10.1 Enhancing understanding of ecological processes**

Biodiversity is considered a key component of ecosystem structure which is essential to maintaining basic ecosystem processes and supporting ecosystem functions. There is a need to develop further research in particular on the links between good ecological status, biodiversity and ecosystem functioning, from both preservation and restoration perspectives. Within this context, developing functional indicators based on fluxes of matters and energy, trophic networks and taking into account biological metrics related to size and other life-history traits are promising paths (see issue 8 related to the development of functional assessment tools). It is increasingly obvious that functional ecology is becoming a fundamental step between the analysis and understanding of biodiversity patterns and ESAs, as it is believed to make more straightforward inferences on ecosystems' resilience and stability compared to classical taxonomical approaches. Research is also needed to provide knowledge on the impact of multiple drivers on the functional capacity of aquatic systems (such as biodiversity and tipping points) and on the impacts on quality and provision of different ES.

### **3.10.2 Raising water managers awareness**

The concept of ES is still relatively new particularly amongst water policy makers and managers. Simple and concrete guidelines for operational use by water managers are needed to facilitate the application of ESA. In the context of the Biodiversity Strategy 2020 implementation, the MAES (Mapping and Assessment of Ecosystem Services) working group developed an analytical framework for ecosystem assessments (Maes et al. 2013). This work sets out a conceptual framework for mapping and assessment of ES by linking biodiversity to human well-being. The framework builds on the important and commonly accepted conclusions of the MA (2005) and the TEEB (2010) but refocuses the debate on biodiversity as pivotal to delivery of ecosystem services and benefits. The MAES conceptual framework links ecosystems to socio-economic systems via the flow of ES, and through the drivers of change that affect ecosystems either as consequence of using the services or as indirect impacts due to human activities in general. In this context, it must be kept in mind that scale issues are important components when performing an ESA, as any ecosystem assessment should be bounded by spatio-temporal scales that are appropriate to the objectives of local

policy makers and natural resource managers. Different types of ecosystem services are valued differently as the spatial or temporal scale of the analysis varies, and the spatial fit between the geographical extent of the water resource and the territorial scope of its management institutions is frequently mismatched (Moss 2012).

## **4 Synthesis and recommendations**

The outcomes of this SPI activity are timely in that they can feed into the revision of the WFD which anticipated in 2019 ahead of planning for the future management cycles (i.e. post 2027 as stated in the directive), and highlight the main issues which need to be addressed during this process. Some urgent concerns were already identified in particular the need to evaluate uncertainties and to better quantify the good ecological potential of HMWB (Hering et al. 2010). In this context, the European Commission working group ECOSTAT has recently launched an activity focused on the evaluation of good ecological potential, the objective of which is to compare and harmonize the various approaches to the evaluation of HMWB which currently co-exist at the European scale. ECOSTAT also plans to prioritize uncertainty in classification, as this is a recurrent topic raised by several MS. A key aspect of this is the need to take into account the uncertainties associated with the natural variability as this is pivotal for encouraging water managers to adopt ecological rather than chemical evaluation, which is often perceived as more straightforward and therefore more reliable.

An important step in WFD implementation in terms of ecological status evaluation has been taken recently with the intercalibration of 230 methods from 28 countries (European Commission 2013a). However, further efforts are still required as a significant proportion of methods are still not intercalibrated or remain under development (Birk et al. 2013), and it is therefore essential to complete intercalibration of the remaining methods as soon as possible (the new round of the intercalibration exercise runs from 2014 to 2016). This gap is particularly important for transitional waters, which suffer from multiple stresses, indefinite reference conditions and lack of historical data (e.g. Ban et al. 2010). These limitations also apply for lakes, although historical data may often be more available for these water bodies, facilitating the definition of reference conditions. Significant gaps also exist for countries in

Southern and Eastern Europe and specific ecosystem types such as ephemeral streams or very large rivers (Birk et al. 2012).

Multi-pressure conditions, which are much more often the rule than the exception at the European scale, constitute another major issue that is faced by both scientists and water managers. This is especially true for very large rivers, transitional waters and lakes as they usually constitute the focal point of catchment processes (either by their location downstream of hydrographical basins or because they constitute closed or semi-closed environmental systems in the case of lakes) and are affected by intense hydromorphological alterations (bank alteration by channelization and/or urbanisation, impoundments, etc.). In this context, specific assessment tools are needed to disentangle the effects of the different pressures acting, and to help water managers prioritizing programmes of measures. This constitutes a major scientific challenge as the effects of these pressures are often difficult to differentiate, and although classical methodologies such as variation partitioning may be useful (Marzin et al. 2012), innovative statistical and experimental approaches may be pushed forward (e.g. Mondy and Usseglio-Polatera 2013). Developing “hybrid” Bayesian tools combining classical statistical modelling and expert judgment based on water managers’ knowledge offers a useful approach in this context. Future integrative tools will also have to be developed as part of an integrated management framework and take into account the fluxes of nutrients (N, P) at the hydrographical basin scale, rather than observed concentrations at an individual site at a given time, increasing the value of the tools for water managers. The outputs of the ongoing MARS European research project (Managing Aquatic ecosystems and water Resources under multiple Stress; <http://www.mars-project.eu>) are expected to be useful for these purposes.

During the last 10 years and parallel with WFD implementation, 'ecosystem functioning' has become a priority research avenue (e.g. Naeem et al. 2009). In this context, the development of trait-based multi-metric tools constitutes a promising research area. These tools consider the functional attributes of the flora/fauna under consideration, and are specifically designed to assess multi-pressure conditions (Dolédec and Statzner 2008, Mondy et al. 2012). They will allow ecological status to be related more easily to ecosystem functioning and not only to biological community structures which has often been the case

to date, despite the WFD explicitly including both terms (“Ecological status is an expression of the quality of the structure and functioning of aquatic ecosystems associated with surface waters”). Integrating size as a functional metric in future bioassessment tools may be a useful first step in this way (San Martin 2005), especially as it is already required by the WFD for fish, but seldom put into practice. In parallel with these trait-based approaches, it is now increasingly recognised that ecosystem functioning, stability and resilience in the face of external pressures can notably be assessed through the study of trophic networks (Arreguín-Sánchez 2014). This may give birth to a second generation of bioassessment tools likely to provide information about ecosystem functioning as a whole (and not BQE by BQE), with a more integrated perspective, complementing classical taxonomical tools currently used and already intercalibrated. It may be difficult to prescribe these kinds of second generation tools on a regulatory basis in the forthcoming revision of the directive, but MS should be encouraged to use them. Functional indicators such as leaf litter decomposition rate (Gessner & Chauvet 2002) may also be pushed forward as they may help detecting early signals of anthropogenic degradation, or, conversely, restoration following measures.

Most bioassessment tools developed to date by MS enable water managers to focus on general degradation or organic pollution (e.g. tools based on macroinvertebrates in streams and rivers) or nutrient issues (e.g. tools based on phytoplankton in lakes). Regardless of the water body category under study, more emphasis on the study of the impacts of hydromorphological alterations (bank degradation, density of riparian areas, alteration of natural continuity both in its lateral and longitudinal dimensions, etc.) is urgently needed. In particular, special attention should be paid by all MS to the alteration of longitudinal continuity by the succession of weirs and dams along river courses, as it is acknowledged that these have significant impacts on natural hydrological and thermal regimes, cause deficit of coarse sediments in downstream areas, and participate to the decline of migratory species already at great risk of extinction at the European scale. An important limit of the WFD as it stands is that a river water body may be classified in good ecological status even if the populations of migratory species historically present in the water body have been drastically impacted. The lateral connectivity of fluvial systems, which are usually multi-impacted and whose ecological functioning is known to be complex (Buijse et al. 2005) should also be taken into account in future, particularly as the intercalibration exercise for

very large rivers currently being carried by MS focuses only on main channel. Contamination by toxic substances is another pressure category which now urgently needs to be addressed as it is now acknowledged that all aquatic systems are contaminated to some degree at the continental scale (Malaj et al. 2014). The development of related tools will clearly necessitate greater collaborative work as this constitutes the boundary between ecological and chemical status evaluation in the sense of WFD, and ecologists and ecotoxicologists remain unintegrated (Artigas et al. 2012). The requirement to collect robust data regarding sediment contamination by toxic substances, for all water body categories, is a crucial element to support this aim. This is a major challenge as these substances are often present at very low concentrations, but are likely to interact in an additive way (i.e. producing a "cocktail" of effects).

Of equal importance in terms of river basin management is the adoption of the Ecosystem Services Approach (ESA). This provides a unique opportunity to bring together scientists working across different disciplines together (e.g. ecologists and economists), and provides a very practical and useful common language that can be used independently by politicians, scientists, water managers and citizens, in a sustainable development perspective. In this context, the new research framework for Europe (Horizon 2020 – The Framework Programme for Research and Innovation) clearly places emphasis on the necessity to protect biodiversity and ecosystem services, from a global change perspective (European commission 2012b). To date, ESA is not clearly used by water managers when designing their programmes of measures, even if some examples do exist (see Wallis et al. 2012 for details). This is especially relevant for wetlands, which are only marginally taken into account in the WFD despite their major importance in purifying water and for biodiversity conservation (flora, fauna). More generally, it is important that water managers are able to identify all the main ecosystem services provided by aquatic environments at the basin, sub-basin and water body scales, as this will encourage and facilitate dialogue and compromises among end-users and allow a more integrated management of resources, with both ecological and economical perspectives (Wallis et al. 2012). This offers the opportunity to overcome one of the key barriers to implementation of the WFD and thus also merits special attention in the upcoming revision of the directive.

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## Annexe 3 - Unravelling river system impairments in stream networks with an integrated risk approach

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## **Unravelling river system impairments in stream networks with an integrated risk approach**

### **Abstract**

Rivers are complex systems for which it is hard to make reliable assessments of causes and responses to impairments. We present a holistic risk-based framework for river ecosystem assessment integrating all potential intervening processes and functions. Risk approaches allow us to deal with uncertainty both in the construction of indicators for magnitude of stressors, and in the inference of environmental processes and their impairment. Yet, here we go further than simply replacing uncertainty by a risk factor. We introduce a more accurate and rigorous notion of risk with a transcription of uncertainty in causal relationships in probability distributions for the magnitude of impairment and the weight of different descriptors, with an associated confidence in the diagnostic. We discuss how Bayesian Belief Networks and Bayesian hierarchical inference allows us to deal with this risk concept to predict impairments and potential recovery of river ecosystems.

We developed a comprehensive approach for river ecosystem assessment, which offers an appealing tool to facilitate diagnosis of the likely causes of impairment and predict future conditions. The ability of the risk approach to integrate multi-scale quantitative and qualitative descriptors in the identification of multiple stressor sources and pathways in the stream network, and their impairment of specific processes and structures is illustrated for the national level risk analysis for hydromorphology and pesticide pollution. Not only does the risk-based framework provide a more complete picture of environmental impairments, it also offers a comprehensive, user-friendly tool to instruct the decision process.

## **1 Introduction**

Ecological assessment methods are increasingly being used to determine the presence and degree of environmental impairment. In most cases, however, assessment results cannot be used to confidently identify suitable management options unless the causes of impairment have been identified explicitly (Suter 1993). Survey-based assessment methods typically include more integrated measures – so-called integrity indices that are designed to reduce complex arrays of ecosystem responses to various disturbances into a single number, but they do not unravel the causalities (Dyer et al. 2000). This challenges scientists to develop new tools to structure and analyse information and uncertainties from different levels of river ecosystem functioning (Page et al. 2012; Ocampo-Duque et al. 2013).

During the last decade, scientific progress has been characterized by a tendency to use ever more complex models built on relationships observed locally, neglecting many important aspects of ecosystem functioning that occur at larger temporal and spatial scales (Wiens 2002). The lack of scientific information gathered at appropriate scales has prevented management actions from being effective (Palmer et al. 2007). Thanks to technical and methodological improvements in data gathering, interpretation and extrapolation, new perspectives of risk-based approaches for holistic scale-hierarchic ecosystem analyses have recently been explored (Jones 2006; Fock 2011). However, even these advanced methods have limited diagnostic power to quantify environmental risks with multiple causalities, which is the prevailing situation in multi-stressor stream contexts.

The effects of human-induced alterations in hydrological regimes, water quality and physical structure on the functional attributes of riverine systems are multiple and complex (Ormerod et al. 2010; Leigh et al. 2012). Causal relationships between ecological integrity and the multitude of river system impairments and stressors in catchments are often difficult or even impossible to identify (Downes et al. 2010; Norris et al. 2012). Hence, there is a need for methods to assess the combined and relative risks of multiple stressors to ecosystem functioning at multiple scales (Camargo and Alonso 2006; Dudgeon 2010; Ormerod et al. 2010; Rasmussen et al. 2013). Furthermore there is need for improvement in ecological information, in particular in determining landscape features that can have a

profound influence on the estimated exposure to stressors, and that can therefore significantly improve ecological risk analysis (Heathwaite 2010; Kapustka 2005).

Here, we adopt the broad definition of ‘ecological assessment’, as it is used in environmental management in general and more specifically in the European Water framework Directive, which we will illustrate for two of its components: the hydromorphology and the physical-chemical water quality assessment. We illustrate the novelty and strength of the integrated, scale-hierarchic framework and Bayesian Network approach developed for the French river network that is now nationally in use for the assessment of both hydromorphological impairment and pesticide pollution. Assessments from this framework infer the impairment risks for specific ecosystem functions and processes, using a diagnostic approach to determine the magnitude of stressors and impairments. We first introduce the principles of scale-hierarchy and the notion of risk in the probabilistic framework, as introduction to the integrated risk assessment framework that we illustrate with the assessment of hydromorphological impairment of riparian zones and pesticide contamination risks.

## **2 General framework and methodology**

### **2.1 The scale-hierarchic audit framework**

A scale-hierarchic approach is essential for ecological risk assessment that views integrity in terms of dynamic states influenced by factors at multiple levels (Suter 1993; Kapustka 2005). The scale-hierarchic framework is also crucial for inference methods that model hierarchically structured spatial data. Hierarchic structuring is a basic step in eliminating a large fraction of the uncertainty in causal relationships, as it can discriminate effects at different scales and determine their relative contribution to ecological impairment (Page et al. 2012). Instead of focusing on local causes, we attribute weight to the driving forces at higher (e.g. regional) scales, given that risks associated with factors acting at these scales can often be quantified more straightforwardly (Van Sickle and Johnson 2008; Norris et al. 2012).

The scale-hierarchic river audit system for France (Chandesris et al. 2009) is based on a spatial classification and attribution of multi-scale descriptors of human-induced disturbances. At the highest level, we distinguish freshwater ecoregions ( $10^3$ - $10^4$  km<sup>2</sup>) as large climatic geological entities. For France, this freshwater ecoregion classification comprises 22 geographical entities in which stream ecosystems should exhibit common characteristics (Wasson *et al.*, 2002). Geology, relief and climate are the determinants for freshwater ecoregion delimitation. The ecoregions provide a framework for grouping data in terms of natural river features and human activities. The ecoregions are divided in Hydrological Units (1093 entities of  $10^2$ - $10^3$  km<sup>2</sup>) of connected sub-catchments, in order to enable gathering relevant data on land cover and land use for upstream catchments. Finally, a systematic segmenting of the 230 000 km French river network identified 69500 reaches, ranging from 1 km on average for small streams to up to 10 km on average for large rivers. The segmenting into hydro-morphologically homogeneous reaches is based on a semi-automatic Gis-process that distinguishes confluences, geomorphological boundaries, and changes in channel form, sinuosity and valley floor width. For each of these river reaches natural and stressor descriptors are aggregated from two spatial scales; catchment land cover information is gathered from the Hydrological Units, and locally, information on land cover and hydromorphology is extracted for the individual river segment for summer and winter bed and over different dimensions (i.e. buffer sizes) for the riparian corridor (valley floor, floodplain, 100 m, 30 m and 10 m).

This smallest spatial unit of the river reach has unique local and network catchment boundaries, and unique descriptors of river network position and connectivity. River reach characteristics are defined as a function of influences of natural elements (climate, elevation, geology, soil, land cover, river network position and connectivity), human activities in local and network contexts and other interactions across spatial hierarchical units (Thorp et al. 2006). The descriptors of these characteristics are gathered in a uniform way from national databases, geospatial datasets and GIS derived features. The nested structure reflects the notion that factors operating at lower hierarchical spatial levels may be influenced by factors at a higher spatial level. Information gathered at different scale levels will be integrated in multi-scale descriptors in order to describe the occurrence and impact of human alterations to river functions.

Not only has this approach the strength of a full length assessment of the river network (not biased by sampling strategy) it furthermore allows working with an oriented, connected network structure (Grant et al. 2007). Upstream stressors do not only operate on a different scale level than downstream stressors, they can also be arranged according to stream flow, for their effects strongly depend on distance and quality aspects of the corridor (Van Sickle and Johnson 2008). Ecosystem impairments and pressures are propagated through the network mediated by the riparian corridor (Cormier et al. 2000). Moreover, as river functions and structure are tightly linked and strongly related to dendritic network structure, not only is a scale-hierarchic approach to the river basin context required, but a truly spatially interconnected network to identify functional relations is called for (Grant et al. 2007; Peterson et al. 2011). For this purpose, a topological structure of the river network, based on graph theory and stream flow model approaches, is constructed in which all segments along the branches of the dendritic network structure are embedded both in the downstream and upstream directions, so as to enable weighting connection by distance, and weighting segments as nodes in the network (Van Looy et al. 2013).

## **2.2 Causal relationship structuring in Bayesian Belief Networks**

Bayesian Belief Networks (BBNs) are emerging as tools to aid in the conservation, restoration, and management of ecosystems (see McCann et al. 2006; Allan et al. 2012; Shenton et al. 2011). They consist of two components: a qualitative graphical structure that describes the causal relationships between the different considered variables, and a quantitative description of the strength of these relationships that relies on conditional probability tables (Naïm et al. 2007). Since the nodes are modelled by means of probability distributions, uncertainty can be estimated more accurately than in models where only mean values are taken into account. In this way, BBNs allow scientists to combine measured responses to environmental change with a conceptual understanding of the ecosystem in question within a probabilistic framework that depicts the chain of hypothesized causal relationships and quantifies the relative influence of individual linkages with explicit uncertainty (Borsuk et al. 2004). This enables ecosystem responses to potential, human-induced, change in environmental drivers to be predicted (Leigh et al. 2011; Stewart-Koster et al. 2011). In this regard they are especially suitable to risk approaches in order to detect



injuries and predict responses in river systems. Here, we present an assessment for quality elements that are not measured over the river network, which we evaluate based on proxies that are either aspects higher up the causal chain (land cover in the catchment) or local proxies like the riparian forest cover that allows to presume an unprotected, natural bank structure.

The strength of BBNs lies firstly in their ability to handle data sources of different origin. Multi-scale and multi-source descriptors, including both direct sources and influential factors for impairment risks, can be included in the model. Secondly, the probabilistic framework allows both empirically-based parameters and non-parametric expert knowledge-based probabilities to be entered as priors in the inference process. Prior probability distributions can be obtained through expert elicitation of probabilities, which can subsequently be updated with the information contained in an empirical dataset. The result of such an inference is a new, more accurate assessment of the relationships between the considered variables, described by a Bayesian *posterior* probability distribution.

BBNs are particularly well adapted to deal with complex systems, for the state of a node only depends on the states of its immediate parent nodes, thus allowing one to work on smaller sub-models, which can then be linked to each other and integrated into the complete model. This property also allows for the modification of a model part without having to modify the entire model. This flexibility in structuring a model through smaller sub-models that integrate variability in the explanatory variables or in the modelled processes (Biggs et al. 2009) makes BBNs attractive tools both for informing on knowledge gaps and uncertainties, and for the construction of scenarios and presentation of possible futures.

Finally, the graphical structure of BBNs allows for the visualisation of functional links and relationships between variables and scales. The underlying graphical structure makes it easier for experts from different fields and stakeholders or decision-makers to discuss the various hypotheses depicted in the BBN and define a common language.

## 3 Resulting river impairment assessment over the river network

### 3.1 The integrated risk-based framework

The risk assessment framework (Fig. 1) simultaneously considers ecosystem functions and impairments in physico-chemistry, hydromorphology and ecology. Both within and in-between these components, functional relationships can be modelled with hierarchic Bayesian frameworks, by either quantitatively or qualitatively modelling causal relationships. The modelling framework therefore relies on the identification of natural baselines and the analysis of stressor influence on the processes and structures of river systems. Three successive steps are distinguished in our risk assessment approach (Fig. 1): 1) the *aggregation* of relevant information from different scale levels into multi-scale descriptors using typological and hierarchical spatial filtering; 2) the *integration* of descriptors into impairment indicators for processes and structures, which are implemented through BBNs (under Netica) to assess the relative contributions of the descriptors to the impairment magnitude; and 3) the *integrated modelling* of relationships between specific ecological processes and structures and their combined influences on ecosystem functioning. Uncertainties in impairment levels described at step 2 are included so as to provide a complete risk analysis. It is important to note that two kinds of statistics are provided for each node in the BBN: the most probable impairment level and the diagnostic confidence in the causal relationships for the impairment, given in the probability distribution.

Spatial filtering based on typologies can serve to better characterize and qualify environmental conditions and descriptors and thus adjust probability distributions of the impairment levels. For this purpose a chemical, hydromorphological and ecoregional typology was developed that frames natural gradients and climatic, geographical variance. For example, Van Looy et al. (2013) looked amongst chemical river types to determine relationships between nutrients and riparian conditions based on type-specific responses. When ecological data are available, the previously elicited prior conditional probability tables in the BBNs can be empirically updated using Bayesian inference. In this manner, the inferred process-based relationships can return information to descriptors and the characterisation of the river system. These feedback loops are an intrinsic part of the hierarchic inference process. As an example, Piffady et al. (2010; 2013) assessed the impacts

of inter-annual variations in temperature and hydrologic regimes on the fish community using a Bayesian hierarchical generalized linear model.

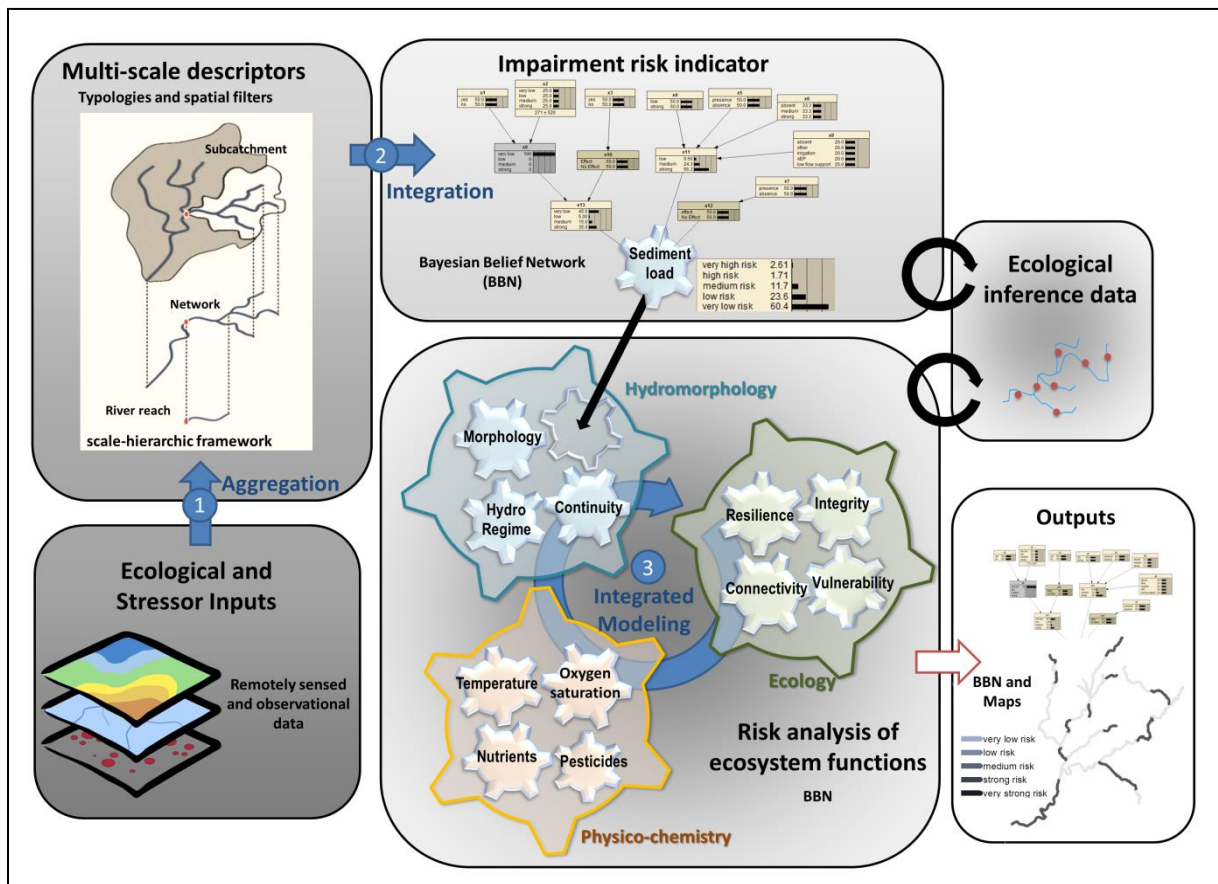


Fig. 1. Scheme of the risk-based framework of river system impairment assessment. Main steps in the framework concern the aggregation of relevant data into multi-scale descriptors, the integration in impairment indicators for ecosystem processes and structures (for which risks are assessed in the framework) and the integrated modelling of ecosystem function impairment. The central part of the scheme shows how the three ecosystem compartments interact to determine ecosystem functioning.

Finally, outcomes of the analyses can be: a nationwide homogeneous identification of impairment risks covering the entire network, a spatial interpretation of risks of specific impairment, and a weighting or comparison of different stressors for specific functions or regions. The construction of BBNs can therefore result or evolve in accordance to specific questions for management or strategic purposes. We illustrate the framework in the following paragraphs through the hydromorphological sub-model and the pesticide pollution risks.

## 3.2 Risk assessment to hydromorphological impairments

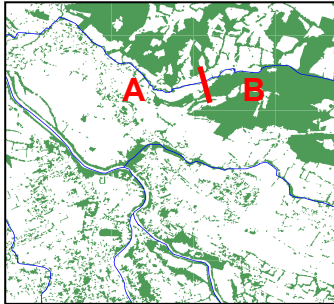
Rivers have long been modified by human activities, which have impacted on three main physical components: (i) flow regime (flow quantity, dynamics and groundwater connectivity), (ii) river continuity (connectivity) and (iii) morphological conditions (width-depth variance, bed structure and substrate, river bank structure). As human impacts can act at different scales and through different pathways upon a river's hydromorphology, the advent of the European Water Framework Directive challenged the scientific community and Member States to nationally address these issues, and to synthesize elements of hydrology, geomorphology and riparian zone characteristics whilst linking them to human pressures (Gottardo et al. 2011).

BBN's are constructed for each hydromorphological indicator, with causal relationships that were primarily quantified with expert elicited prior conditional probability tables by an expert panel. These constructed causal chains are based on both natural and stressor multi-scale descriptors, taking into account natural filters (i.e. altitude, climatic-hydrologic regime) and the spatial network context (i.e. presence of dams or weirs in the upstream or downstream parts of an analysed reach). For example for *the impairment indicator of river bed structure*, the BBN is based on alteration probabilities of channel straightening (causing increase in flow velocity and shear stress) and local damming (causing upstream sedimentation and downstream erosion), combined with upstream risk of sediment load blocking and the network position of large dams upstream. This indicator construction illustrates both the use of an integrative approach to stressors of different origin and scales (local and upstream network) and their interpretation in the river network. Two different and complementary statistics are provided: the magnitude of the river bed alteration depicted by the discretized state within the Bayesian network node taken by a variable and a probability distribution for the variable to take this state.

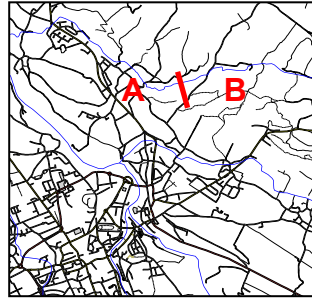
### **Confidence in the causal networks**

Confidence in the judgment is depicted through the probability distribution among states: the more one state tends to 100% probability; the more confident we can be in assigning this state to the variable. In this work, we insist on the importance of providing two statistics; the most probable impairment and the level of confidence in this judgment. The confidence expressed in the probability distributions is given by the expert panel, based on the strength of causal relationships and the weighting of different causalities. The complexity and difficulty of this choice is then apparent in the number of parental linkages in the BBN. For this reason, the discretization of intermediate latent variables that have better identified causal links can add to the overall confidence in the BBN. So, confidence in causalities can be ensured high up in the network with the availability and accuracy of relevant information (data), or lower down in the BBN with increased knowledge for weighting the (multiple) causalities – ideally with inferred posterior distributions.

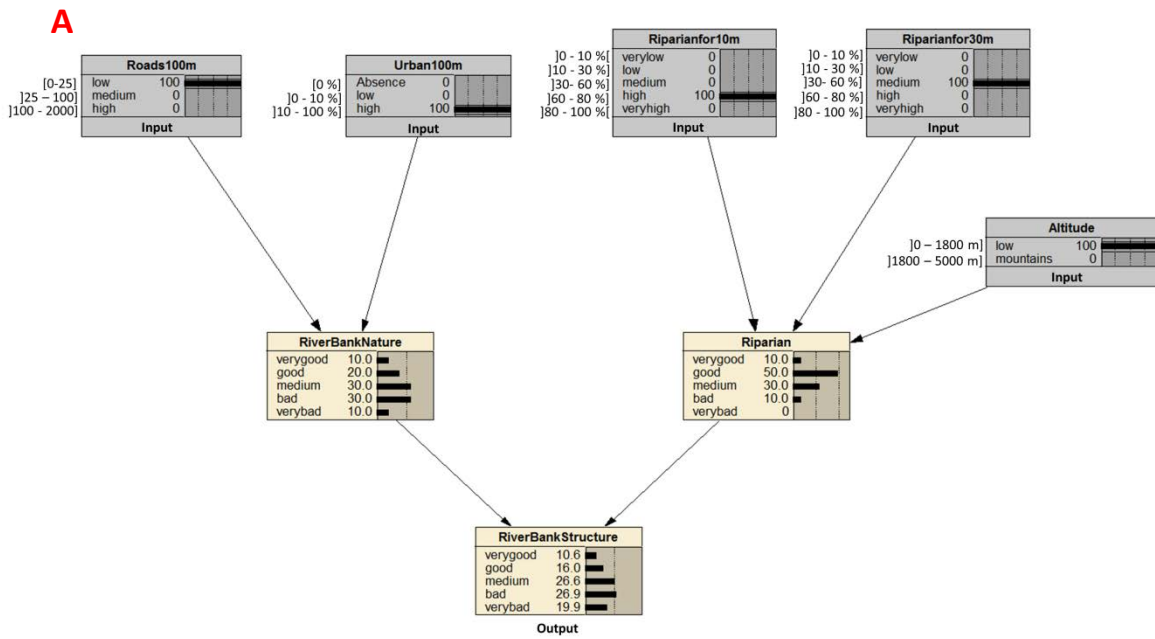
This aspect is illustrated in figure 2 with the assessment of impairment risk to river bank structure. The impairment risk indicator has two contributing latent variables; the nature of the river bank and riparian forest cover. Presence of urbanization and roads in the riparian zone are identified as causes of river bank stabilization, whereas river bank functionality is represented by riparian forest cover. Two descriptors for the riparian forest cover were retained: riparian forest cover within a 10m buffer is considered an indicator of potential shading and organic matter provision to the aquatic environment; whereas the 30m buffer riparian forest cover informs on buffering capacity (nutrient retention, diminishing impact of adjacent land use). The descriptors of riparian forest cover give good confidence with narrow probability distributions of the riparian forest latent variable in the two cases (A and B of fig. 2), whereas for the nature of the river bank, the information at hand gives less confidence, leading to more uniform probability distributions. The descriptors of roads and urbanization in the corridor give less confidence to the impairment risk to the nature of the river bank, resulting in more uniform probability distributions. Still, for the overall river bank structure we can reach quite high confidence in the case of an established riparian forest cover.



**Riparian forest**  
**A:** 72% 10m buffer, 32% 30m buffer  
**B:** 79% 10m buffer, 81% 30m buffer



**Urbanisation**  
**A:** 7% roads 26% Urb100m  
**B:** 3% roads 5% Urb100m



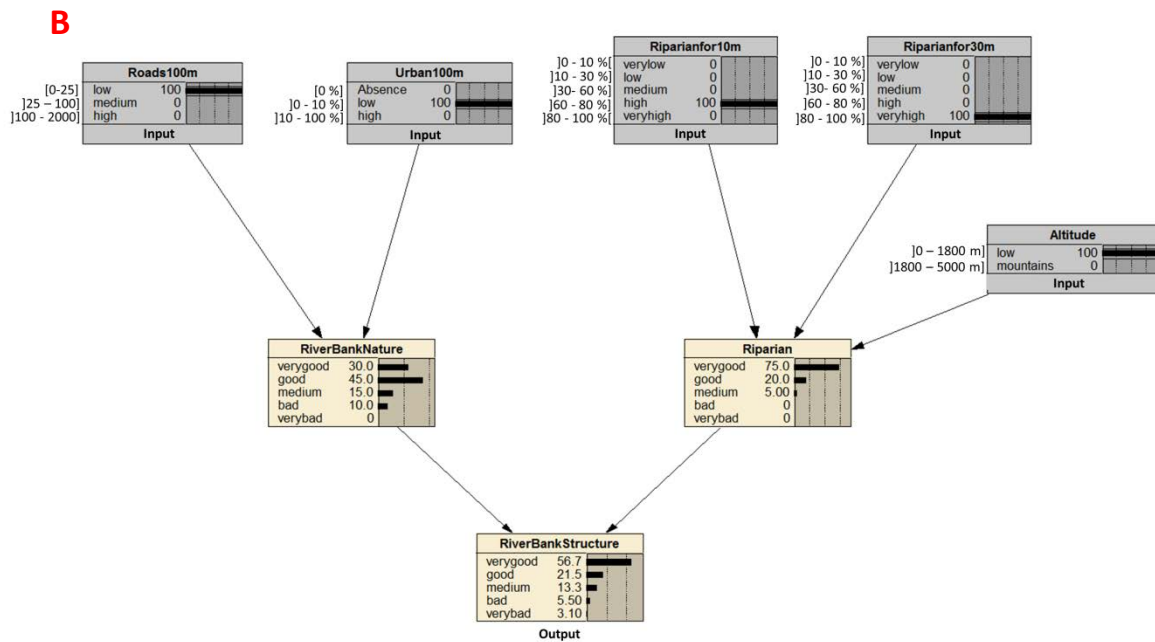


Figure 2. illustration of confidence in the BBN. The network for the indicator of river bank structure shows causal links to two latent variables for the nature of the river bank and for the riparian forest cover. The position on the map is shown with the segment percentages of the descriptors, which are interpreted in classes in the BBN (e.g. values between 60-80% riparian forest cover for the 10m buffer are classified as high). The confidence in risk judgment is present in the probability distributions for the latent variables and for the final integration in the indicator. A rather uniform probability distribution and thus little confidence in the judgment is present in case A. High confidence in the judgment is present in case B, with a highly skewed probability distribution.

Based on the observed relationship between riparian forest cover descriptors and the physico-chemical and biological quality of the river reach, boundaries for a sufficient (i.e. functional) riparian forest cover were determined (Van Looy et al. 2013). These boundary values were entered as descriptor classes into the BBN, resulting in higher confidence for the riparian forest cover latent variable in the integration and thus in a stronger foundation to the risk mapping of impairment to the riparian zone structure (Fig. 3). Mapping these associated confidences in judgments furthermore allows identification of knowledge gaps linked to geographical context (see inset in figure 3).



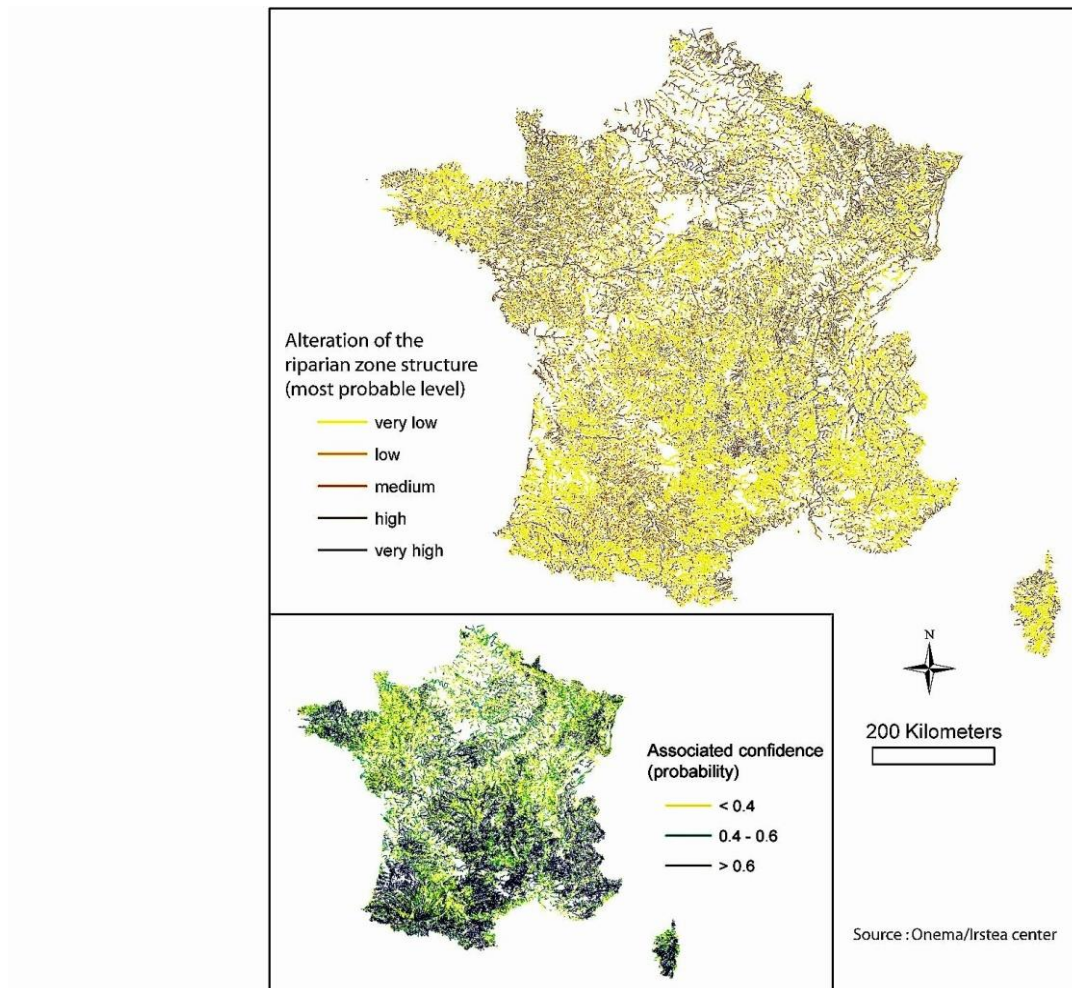


Figure 3. Impairment risks to the riparian zone structure for the French river network. The risk assessment is presented in terms of the most probable level of impairment of the riparian zone structure in France. The inset figure map shows the associated confidence to this level of impairment, for the 230 000 km French river network.

### 3.3 Risk assessment for diffuse pollutions

The potential impacts of agricultural pesticides on watersheds' ecological status are a further important concern in the European water strategies. There is a major interest in developing models that can reliably assess the risk for rivers of being contaminated in view of the complexity of multiple stressors in the watershed (Harris and Heathwaite 2012; Heathwaite 2010). At first, such approach demands a diagnosis of physical properties of soils at the basin scale to determine the main way of transfer from the field to the river. Existing methods did not account for the different chemical component properties in the pollution transfer, such as persistence or water affinity that can either positively or negatively interact



with the contamination risk. The ARPEGES (Risk Analysis of Pesticides for Surface Water management) model ambitions taking the soil and the molecule properties into account to provide a unified model of contamination risk over the French national territory (Gauroy et al. 2015). It provides a robust tool for the French Water Authorities in assessing the status and the risk of failing to meet the environmental objectives.

The model is developed within the risk assessment framework. It combines catchment-scale descriptors into indicators of seasonal acute and chronic pollution magnitudes. Descriptors, related to main ways of pollution transfer, chemical component properties and pesticide use, are derived from national GIS databases.

In the BBN, the pollution process was decomposed according to the three identified main ways of transfer to the river (surface run-off, leaching of water-soluble pesticides through permeable soils, and tile drainage) that were described for their intrinsic vulnerabilities. The intrinsic vulnerabilities refer to the physical soil properties, aquifers, hydrographical network and drainage density, and potential interactions with biological characteristics of the riparian corridor that can slow down water transfer and actively extract solutes. The considered properties of pesticides are the persistence (half-life duration), the soil organic carbon-water partitioning coefficient (water affinity) and the solubility. By crossing the intrinsic vulnerabilities with chemical properties of molecules, the specific vulnerabilities are determined. Two specific vulnerabilities were considered: vulnerability to chronic and acute pollution.

In addition to the vulnerability, information on pesticide use is entered in the model. Two sources of data are exploited to describe respectively the qualitative and quantitative aspects of application. At the local scale, exhaustive quantitative data is available of farmer's pesticide consumption. The qualitative data of the molecules is aggregated at sub-catchment scale from substance quantities sold at distribution points. Both the processes of transfer as the application practices have been considered as being influenced by climatological variables. These are principally the frequency and quantity of rainfall, which were retained as factors to a spatial and temporal hierarchic approach: based on a geoclimatic zone delimitation (Champeaux and Tamburini 1996) and time periods for the seasonality (spring/summer and autumn/winter). Finally, the information on the quantity of pesticides

used is crossed with the specific vulnerabilities to give the resulting seasonal acute and chronic pollution risk level. Class boundaries and conditional probability tables have been set up accordingly with expert judgments for presentation at a national scale level. See Figure 3 for an example of model result over the French territory (for the 1093 Hydrological Units).

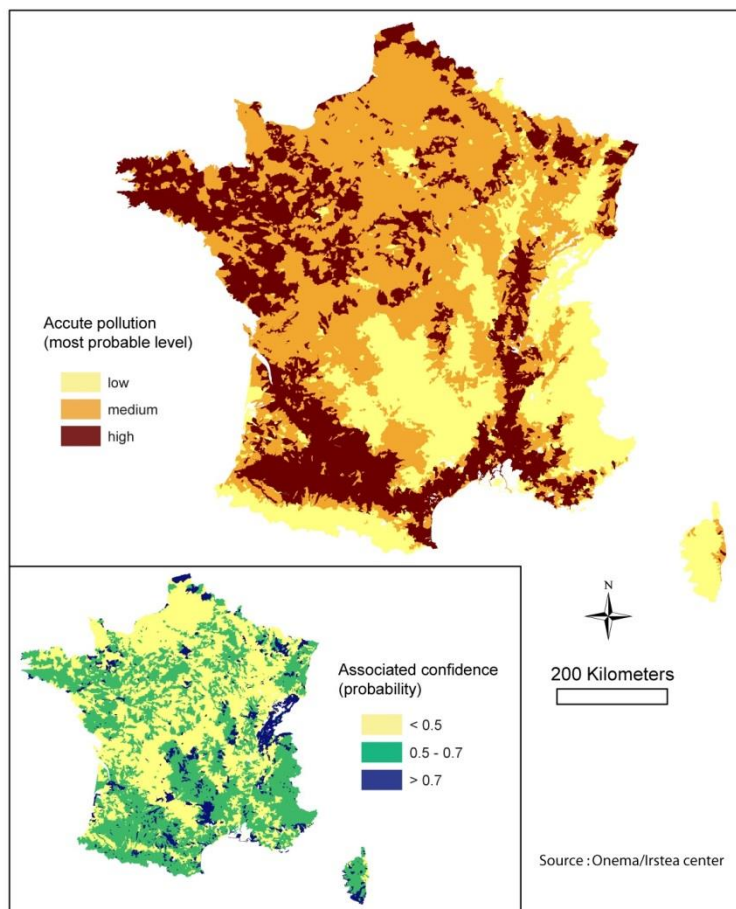


Figure 3. Risk assessment for acute pollution of pesticides with presentation of the most probable level of pollution and the associated confidence for the French national territory.

## **4 Discussion**

The examples described above demonstrate the potentials of the integrated risk-based framework to better frame and determine the relative contributions of stressors to the impairments of river systems. The presented hydromorphology assessment covers the entire stream network of France and was used by the river management authorities in elaborating the 2013 status report under the Water Framework Directive. Local validation by the water authorities' experts confirmed the assessment in 80% of the cases. To our knowledge, few similar comprehensive assessment systems exist at such large scales using spatial hierarchic frameworks (see Davies et al. 2010; Davies et al. 2012; Wang et al. 2011; Esselman et al. 2013). Those examples are habitat-oriented audit frameworks assembling relevant natural factors and human disturbances in order to assess ecosystem types. Still, it is difficult to bring all the relevant elements into the picture, both due to gaps in information, and to the complexity of interactions. We believe the proposed risk approach provides a promising additional field for most comprehensive assessment systems, as it allows the inclusion of all environmental insights and at the same time highlights shortcomings in both information and understanding of relationships.

This is also illustrated by the ARPEGES model that brings a comprehensive pesticide pollution risk assessment, combining strengths of different existing model types. The aggregation and integration of the information entered in this model is quite unique; most diffuse pollution models are small-scale pathway models or catchment scale models at best combining hydrological and river quality models (Arheimer and Olsson 2003). The ARPEGES model is oriented at sub-catchment scale risks while integrating individual molecule pathways and both local soil biogeochemical processing aspects as large scale climatic and geologic-geographic factors. Thus we follow the recommendations for diffuse pollution modelling to combine the approaches and results of chemometric data analysis methods that are based on observation and analysis of experimental data, and deterministic models that are defined as numerical methods based on theoretical principles (Terrado et al. 2009).

The risk approach allows weighting of the different sources of evidence in the integrated assessment model, where drivers may operate at different scales. The innovative elements in the proposed framework consist firstly in the presentation of both the confidence for the appraisal and for the understanding of processes and causal relationships. Secondly, the integrative character of the BBN structure translates the complexity of the relationships studied. They can be associated with evidence weighting to determine the incidence, distribution, and causes of impairment (Cormier and Suter 2008; Norris et al. 2012; Webb et al. 2012), and probabilistic frameworks that allow modelling and predicting possible futures for the relationships between environmental stressors and stream condition (Stewart-Koster et al. 2010; Allan et al. 2012; Sheldon et al. 2012). In this formalized risk model structure, expert judgment can lead to a better understanding of causal relationships between environmental stressors and ecological responses (Allan et al. 2012). Inferring posterior values to prior estimates in integrated modelling of ecosystem processes is a necessary step to ensure a rigorous weighting of conditional probabilities. However, observational data are not always available, and not for all descriptors. This can be due to many factors, such as costs of field campaigns in a national network survey or the difficulty of developing a coherent experimental protocol. The main strength of this risk approach is that it can reach conclusions (albeit with less certainty) without having full data. Furthermore, in the context of very complex systems, it can be significantly helpful in identifying knowledge gaps for a full understanding of the studied process.

The second strength of the proposed framework is the elucidation through BBNs of the pathways of impairments to ecosystem processes. All existing knowledge and expertise can be integrated in a network that depicts the chain of causal relationships, while at the same time quantifying the relative influence of individual linkages with explicit uncertainty (Borsuk et al. 2004), and taking into account complexity through a multitude of links between stressors and causalities (Biggs et al. 2009). Although relationships that can be identified in this way are not necessarily causal, the ability to predict aspects most at risk using broad scale predictors should serve as a useful management tool (Cormier and Suter 2008; McGinnis and Kerans 2013). With such a model, permanently taking the uncertainties associated with the processes into account, the effects of different management options for stream systems can be assessed fairly, as future possible conditions and their probabilities of

occurrence can be predicted based on forecasts of land-use change or implementation of best management practices.

In conclusion, the framework with Bayesian estimation of risks presented here provides a comprehensive and rigorous assessment approach to river impairments. Particular attention has been paid to the development of visually attractive, user-friendly, flexible and evolving approaches to improve their transfer. Both indicator selection and restoration decision are founded on a balance of environmental and social principles (Pedroli et al. 2002; Bouleau et al. 2009), and effective management solutions are only achieved when based on thorough and sound science that evolves with the changing environment.

## 5 Acknowledgments

The authors gratefully acknowledge the financial and structural support for the SYRAH hydromorphology audit and ARPEGES model project of the ministry of ecology, the water authorities and the French National Agency for Water and Aquatic Environments (ONEMA). The international collaboration and the scientific harmonisation of the public water policies to the European WFD since 2000 started the flow of ideas at the basis of the presented framework. We would like to thank Ricardo Simon, Angus Webb, Catherine Leigh and Martin Thoms for their valuable comments and suggestions to earlier versions of the manuscript.

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## RÉSUMÉ

Les cours d'eau et leurs bassins versants sont des systèmes complexes et en équilibre dynamique. Si l'on connaît qualitativement assez bien la plupart des processus dont ils sont le siège (ex. transport de matière organique et de sédiments, transformation des litières, etc...), il est beaucoup plus difficile d'en avoir une représentation pertinente à la bonne maille compte tenu de l'emboîtement d'échelles des processus, des déterminants terrestres agissant sur les milieux aquatiques et surtout des différents facteurs de stress qui s'y combinent. Nous avons mené une exploration dont le but était de rechercher les causes souvent multiples de dégradation de l'état écologique, d'en connaître l'étendue spatiale et la gravité, pour être en mesure de recommander des actions efficaces de restauration.

Dans la première partie de cette thèse, nous avons développé des modèles explicatifs de l'état écologique à large échelle capables de donner des réponses applicables aux échelles de décision et de gestion, à savoir européenne, nationale et régionale avec comme objectif principal de comparer les relations entre les pressions combinées représentées par l'occupation du sol et l'état écologique des cours d'eau de quatre pays européens: la France, la Slovaquie, l'Estonie, et le Royaume-Uni (Angleterre et Pays de Galles). Cette étude a permis (i) d'établir une hiérarchie des catégories d'occupation du sol influant sur l'état écologique, en particulier l'agriculture et l'urbanisation, (ii) d'identifier des tendances régionales dans ces relations pression-impact et (iii) d'évaluer le poids relatif de la pression à l'échelle du bassin et du corridor rivulaire, et l'effet tampon des zones rivulaires suivant la nature de leur couverture.

Dans une deuxième partie, nous avons construit un corpus de modèles capables d'expliquer d'une part la variabilité des indices biologiques utilisés dans les réseaux et de prédire d'autre part l'état écologique des masses d'eau françaises non suivies par le réseau de contrôle et de surveillance (RCS). Cette étude a permis de montrer que : (i) le facteur le plus impactant pour l'état biologique qu'il soit mesuré à partir des macroinvertébrés, des poissons ou des diatomées était la physico-chimie, suivie de l'occupation du sol et de l'hydromorphologie et (ii) l'utilisation de ces trois types de pressions permettait de développer des modèles prédictifs de l'état biologique robustes et fiables.

Dans une troisième partie, nous avons voulu tester si les relations entre les pressions à l'échelle locale et l'état écologique étaient hiérarchiquement influencées par la typologie régionale (basée sur les caractéristiques géographiques naturelles et les forces motrices). Ceci nous a permis de montrer que la prise en compte d'une typologie basée sur les forces motrices des bassins versants ne remettait pas en cause la hiérarchie des effets des pressions sur la biologie.

Enfin, dans une quatrième et dernière partie, nous avons développé une approche nouvelle qui propose de prendre en compte les connaissances sur l'organisation des échelles et sur les liens entre pressions et état écologique, en construisant et en analysant un modèle structurel qui a permis de relier entre elles des variables latentes correspondant aux pressions d'occupation du sol, hydromorphologiques et physico-chimiques et d'analyser in fine leur effet sur l'état écologique mesuré ici à travers l'I2M2. Le développement de ce modèle a montré : (i) l'effet important de l'occupation du sol sur l'hydromorphologie et sur la physico-chimie et l'effet indirect qui en découle sur les macroinvertébrés, (ii) l'effet de l'hydromorphologie sur la mosaïque de substrats, la dynamique des nutriments et de la matière organique, impliquant un effet indirect majeur sur les macroinvertébrés.

## ABSTRACT

The rivers and their watersheds are complex systems in dynamic balance. If processes acting in stream are relatively well known (organic matter and sediment transport, litter degradation, etc ...), it is much more difficult to have a relevant representation of this functioning considering the hierarchy of scales, land determinants affecting aquatic environments and combined multiple stressors. We conducted an exploration whose purpose was to seek the multiple causes of degradation of the ecological status, to know the spatial extent and severity and to be able to recommend effective restoration actions.

In the first part of the thesis, we developed large-scale models, based on existing data, to give answers applicable to the decision and management scales, i.e. European, national and regional. The main was to compare the relationships between the combined pressures represented by land cover and the ecological status of running waters in four European countries: France, Slovakia, Estonia, and UK (England and Wales). This first study allowed us i) to establish the hierarchy of the major influences that impact ecological status, particularly agriculture and urbanisation, ii) to identify regional patterns in these pressure-impact relationships and iii) to evaluate the relative weight of the pressures acting at the basin and riparian corridor scales, and the possible buffering effect of riparian land cover.

In the second part of the thesis, we built a corpus of models capable of explaining the variability of the biological indices used in the survey network and also predict the ecological status of non-monitored water bodies in France. This permits to demonstrate that : (i) the parameters characterizing the load of nutrients and organic matter had a predominant effect on the three compartments, followed by land use and hydromorphology, (ii) it is possible to build models capable of predicting ecological status that are efficient and easily transferable, using data of different scales that influence the functioning of hydrosystems.

In the third part, we tested if the relationships between local scale pressures and ecological status are hierarchically influenced by regional patterns (such as natural physiographic and/or driving forces). Introducing hierarchical factors in multi-stress models can lead to slight variations in responses of organisms to pressures according to their natural characteristics and their driving forces acting at catchment scale.

And in the last part, the purpose of our approach was to take into account the nested scale organisation and the links between anthropogenic pressures and river ecological status by building - and analysing the results of - a model based on the PLS path modelling method. This method can be used to simultaneously analyse the effects of latent variables corresponding to land-use, hydromorphological and physico-chemical pressures on the ecological status of rivers, synthetically measured in this study by the macroinvertebrate-based French biotic index for wadeable rivers (I2M2). The development of this model should allow us to demonstrate : (i) the importance of land use effect exerted on both hydromorphology and physico-chemistry and their translation as an indirect effect on biological condition of streams, (ii) that hydromorphological alterations had an effect on substrates structure and nutrients and organic matter concentrations implying that hydromorphology has a major indirect effect on macroinverteb