UNIVERSITÉ DE BRETAGNE OCCIDENTALE

THÈSE DE DOCTORAT

Spécialité : **Océanologie Biologique**

par

Malika BEL HASSEN

Présentée pour obtenir le grade de **Docteur**

FONCTIONNEMENT DES MARAIS MARITIMES ATLANTIQUES: ÉCHANGES D'ÉNERGIE ET EFFETS DES FLUX AQUACOLES DIFFUS

Soutenue le 29 Février 2000 devant le jury composé de :

MM.	**J.C. Lefeuvre**	**Rapporteur**
	P. Mérot	**Rapporteur**
	M. Le Pennec	**Directeur de thèse**
	M. Héral	**Responsable Scientifique**
	A. El Abed	**Examinateur**
	P. Le Corre	**Examinateur**
Mme	**B. Guillaumont**	**Examinateur**

Avant propos

En premier lieu, je remercie Monsieur Maurice Héral, Ex-Directeur du CREMA, actuellement Directeur des Ressources Vivantes à IFREMER, de m'avoir accueillie au sein de son laboratoire. Je le remercie également pour son encadrement scientifique ainsi que pour l'ensemble de la logistique qu'il a mis à ma disposition pour accomplir ce travail.

Je remercie également Monsieur Amor Elabed, Directeur Général de l'Institut National Scientifique et Technologique de la Mer de Tunisie, d'avoir proposé ce sujet de thèse ainsi que pour sa participation à l'évaluation de ce travail.

Je tiens à remercier Jean Prou pour m'avoir initié aux Systèmes d'Information Géographique. Qu'il soit assuré de toute ma reconnaissance.

Pour avoir accepté de juger ce travail, je remercie vivement les membres du jury : Monsieurs Marcel Le Pennec, Jean-Claude Lefeuvre, Phillipe Mérot, Maurice Héral, Amor Elabed, Pierre Le Corre et Madame Brigitte Guillaumont. Je leur saurais gré des critiques constructives qu'ils ont bien voulu apporter à ce manuscrit.

Je ne saurais oublier toute la famille du CREMA pour son accueil et son aide, je remercie en particulier Cédric Bacher, Françoise et Jean Paul Lagardère, Alain Bodoy et Pierre Richard. Que l'ensemble des techniciens du laboratoire, Françoise, Martine, Lucette, Michel, Marcel et Céline sachent que je les remercie beaucoup pour leur aide précieuse et pour leur gentillesse. Un grand merci va à Evelyne Richard pour m'avoir initié aux secrets de la recherche bibliographique, mais également pour plein d'autres choses, notamment pour son sourire et ses jolis bouquets, qui ont souvent fait flâner la bonne humeur dans notre bureau.

Enfin, ces trois ans de thèse n'ont pas été qu'un dur labeur, et heureusement que la vie nous a réuni dans des circonstances pareilles, afin qu'on se soutienne et qu'on s'amuse ensemble. Ces moments je les dois à tous les thésards du CREMA, notamment à Florence, Cécile, Chang Keun, Ridha, Christel, Christine, Dorota, Seb, Stéphane, Francis, Bruno, claire…

Je ne saurais finir sans avoir une pensée toute particulière pour Flo, pour sa gentillese et son amitié...à toi Flo je te passe le flambeau...

Je dédie ce mémoire

à toute ma famille,

à mes parents pour tant d'amour pour ce 4ème enfant. Ce grand amour, ils ont toujours su partager entre ses 3 cadettes et ses 4 juniors,

à Mohamed pour sa présence, son soutien et sa grande patience…
à toi…à nous…

Résumé

De faibles biomasses phytoplanctoniques caractérisent les systèmes estuariens des côtes atlantiques jusqu'à la manche. La turbidité a été souvent considérée comme le facteur limitant pour le phytoplancton. De part leur situation géographique, à la charnière entre la terre et la mer, les écosystèmes estuariens sont sujet à des enrichissements en matières nutritives dissoutes provenant des systèmes amonts. Bien qu'on a déjà démontré qu'une partie de ces apports supporte une forte production benthique, la part qui échappe au réseau trophique estuarien aussi bien benthique que planctonique, et qui est exportée aux systèmes océaniques adjacents, reste toutefois indéterminée.

L'un des objectifs de ce travail est de quantifier les échanges de nutriments dissous et de matière particulaire entre la baie macrotidale du Fier d'Ars et l'océan côtier. L'approche utilisée a été basée sur la recherche de corrélations multiples à partir de paramètres hydrométéorologiques comme variables prédictives des flux instantanés. Le modèle de régression a pour but de déterminer un flux annuel net avec une estimation très fiable de la moyenne annuelle, tenant compte de la plus large variabilité, inhérente aux phénomènes périodiques tel que l'hydrodynamisme, ainsi qu'aux variations non périodiques relatives à des changements météorologiques.

La quantification des apports d'origine diffuse a été étudiée à l'aide d'un modèle stationnaire basé essentiellement sur le modèle spatial de la topographie et de l'utilisation du sol. Cette étude a fait appel à des outils d'analyses spatiales pour le développement de cartes thématiques, tels que les Systèmes d'Information Géographiques. La répartition spatiale des apports diffus aussi bien de l'azote que des phosphates a été réalisée à l'aide de l'interpolateur du krigeage. Un modèle d'écoulement de pente, en particulier une approche dérivée de ce modèle, le modèle d'accumulation pondérée, a permis d'estimer les flux diffus au niveau du réseau hydraulique.

L'étude de la capacité trophique du système a permis d'évaluer de faibles biomasses phytoplanctoniques. Une complexe succession de facteurs (lumière, azote, phosphore) assure le contrôle et la limitation de la production phytoplanctonique.

Il a été montré que le système étudié produit plus de matière inorganique azotée qu'il ne peut en consommer, l'excédant est donc exporté vers la zone côtière connexe. Une nette exportation de carbone organique dissous a été aussi mise en évidence. Le système importe de la matière particulaire dont plus de 70% est sous la forme minérale, engendrant ainsi une sédimentation progressive du système.

Le modèle de quantification des apports diffus au sein de l'écosystème a permis de mettre en évidence la contribution des activités intensives dans l'enrichissement de la zone côtière en charges azotées. Les apports atmosphériques ne représentent que de faibles proportions 4%-12% de ce que peuvent produire les activités aquacoles. Cependant, les processus internes au système, susceptibles de diminuer la charge exportée vers l'océan, telle que la dénitrification ou la fixation d'azote ne sont pas encore assez bien étudiés afin de dresser le bilan global du système. Des perspectives sont proposées pour y remédier, qui nécessitent la spatialisation des différents processus afin de déterminer leurs poids localement.

Mots clés : aquaculture, exportation, matière dissoute, Fier d'Ars, Systèmes d'information géographique, importation, usages, lumière, microphytobenthos, modélisation, nutriments, matière organique, matière particulaire, phytoplancton, sédimentation.

ABSTRACT

French estuaries, from the Atlantic coast to the English channel, have been characterized by a poor phytoplankton biomass. The turbidity has always been considered as the limiting factor for phytoplankton. Estuarine ecosystems are situated at the boundary between the land and the sea. Consequently, they are subject to an enrichment by nutrient loadings from upland ecosystems. Although it has been demonstrated that a part of the nutrient loadings support an important benthic production. The part, which escapes to the estuarine benthic and planktonic food web and is exported to adjacent estuarine ecosystems, still remains undetermined.

Quantifying the nutrient and particulate matter exchange between a macrotidal Bay, the Fier d'Ars, and the adjacent ocean was among the objectives of this study. The technique already performed was based on the regression estimation, using the hydrometeorological parameters as predictor variables of instantaneous nutrient fluxes. This method allowed us to establish the direction of net flux with tight confidence bounds on net annual flux. It also takes into account a large variability, inherent to the periodic phenomenon, such as hydrodynamics and also to nonperiodic variations, relative to the meteorological fluctuations.

The assessment of nonpoint source loadings was performed using a simple model that accounts for the spatial pattern in topography and land use. This model was based on spatial analyses using Geographic Information System (GIS) databases. The spatial distribution of nitrogen and phosphorus loadings was performed using the kriging interpolator. An 8-direction-model, particularly its derivative function, the weighted accumulation model, allowed the assessment of nutrient loading in each instream location.

The study of the phytoplankton standing crops, highlighted a low biomass throughout the year. Showing a pattern of recurring variations within factors regulating phytoplankton biomass, which incriminated both light and nutrient availability (nitrogen and phosphorus).

It was demonstrated that the study system produced more inorganic nitrogen than can be consumed, the excess was exported to the adjacent coastal ecosystem. A net export of inorganic dissolved carbon was also assessed. The system imported particulate matter, more than 70% was mineral form, which engendered a progressive sedimentation of the system.

The GIS- based assessment of nutrient loading highlighted the major contribution of intensive activities to the enrichment of the coastal area with nitrogen loading. Atmospheric loading represented low proportions (4% - 12%) of the potential aquacultural loading. Nevertheless, internal processes, such as denitrification and nitrogen fixation, susceptible to reduce the quantities exported to the ocean, are not enough well studied to establish the total balance of the system. Perspectives are proposed in this way, they concern the spatialisation of the different internal processes with the determination of their weight locally.

Key words : aquaculture, export, dissolved matter, Fier d'Ars Bay, Geographic Information Systems, import, land use, light, microphytobenthos, modelling, nutrients, organic matter, particulate matter, phytoplankton, sedimentation.

Liste des principles abréviations

Variables physico-chimiques :

- DIN : azote inorganique dissous
- PO_4 = OP : orthophosphates
- NN : nitrite et nitrate
- Si : silice
- DON : azote organique dissous
- POC : carbone organique particulaire
- MD: matière dissoute
- MP : matière particulaire
- DOC : carbone organique dissous
- NH_4 : ammonium
- N:P = NP : rapport azote inorganique dissous sur orthophosphates
- Si:P = SIP : rapport silice sur orthophosphates
- Si:N = SIN : rapport silice sur azote inorganique dissous
- TSS : matière totale en suspension
- MSS : matière minérale en suspension
- OSS : matière organique en suspension
- Temp : temperature
- Sal : salinité
- PAR : radiations photosynthétiquement actives
- urea : urée
- Phéo = Phae : phéopigments
- Chl = Chl a : chlorophylle a

Variables hydrométéorologiques :

- Vt : vitesse maximale du vent
- Vp : vitesse moyenne du vent
- Prmn : pression atmosphérique
- Tam : temperature de l'air
- Tom : temperature de l'eau
- Pn : précipitations
- Hn : hauteur maximale de l'eau
- E : jusant
- F : flot

Mois :

- Ja : janvier
- F : février
- Ma : mars
- A : avril
- M : may
- J : juin
- Jy : juillet
- Au : Août
- S : septembre
- O : octobre
- N : novembre
- D : decembre

Divers :

- GIS = SIG : systèmes d'information géographique
- NPS : charges diffuses

Glossaire

Les principales définitions des termes utilisés au cours de ce manuscrit sont reportées ci-dessous. Leurs significations dans le contexte du site étudié ont été également explicitées.

- **Les zones humides maritimes**, occupent des zones estuariennes ou lagunaires abritées, périodiquement recouvertes par la marée; elles évoluent généralement en marais saumâtres par fermeture naturelle des cordons littoraux. On y distingue principalement la slikke, le shorre et les marais endigués (Le détail de ces trois étagements est largement explicité dans la partie Synthèse des Chapitres, paragraphe le site étudié).

- **La zone intertidale**, c'est la zone de balancement des marées. Au niveau du site étudié cette zone correspond le plus souvent à la slikke, constituée de sédiment vaseux communément appelée *vasière* "mudflat". Cette zone s'étend durant les grands coefficients de marée (> 85), jusqu'à une partie du shorre, d'où l'appellation souvent au cours de ce manuscrit "intertidal mudflat" ou également "intertidal salt marshes".

- **Les marais maritimes**, "salt-marshes" dans les pays anglophones. Ils correspondent aux zones de shorre dans le site étudié. Suite à l'endiguement, une partie du shorre a cédé des superficies au profit des marais endigués. Ces derniers étant actuellement aménagés en bassins à poissons et /ou à huîtres mais également en marais salants.

INTRODUCTION GÉNÉRALE

SYNTHÈSE DES CHAPITRES

CHAPITRE I

CHAPITRE II

CHAPITRE III

INTRODUCTION GÉNÉRALE

- CADRE GÉNÉRAL
- LES OBJÉCTIFS DE LA THÈSE

CADRE GÉNÉRAL

La notion de zone côtière est relativement ambigue. En effet, la définition de la limite de cette zone suscite beaucoup de controverses. Selon Ray et Hayden (1992), elle inclue la plaine côtière, les bordures continentales, les eaux douces, saumâtres et marines, les baies, les estuaires, les systèmes dunaires et le plateau continental. Costanza *et al.* (1997) y incluent les estuaires, les herbiers marins, les récifs coralliens et la plaine continentale. Cette zone couvre une superficie de 8% du globe. Une intense activité biogéochimique caractérise la bande côtière, ce qui en fait l'un des écosystèmes les plus productifs au monde (Nixon *et al.*, 1986; Cloern *et al.*, 1995). La préservation de cette zone est donc une priorité mondiale.

Les échanges intersystèmes, entre zone côtière et mer ouverte, débouchent souvent sur une question principale: sont-ils une source de richesse ou une cause de dégradation ?
La compréhension écologique de ces processus a énormément progressé depuis que l'hypothèse du "outwelling " a été introduite par Teal (1962). Cet auteur a montré que 45% de la matière organique produite par les marais à *Spartina alterniflora* des côtes ouest atlantique, échappe à la consommation et au stockage et qu'elle est exportée vers les eaux marines littorales, où elle stimule une production secondaire, essentiellement constituée de bivalves, crustacés et poissons (Dame & Allen, 1996). Ce transfert d'énergie dépasse souvent le rôle bénéfique de l'enrichissement, et crée parfois un dysfonctionnement dans les systèmes marins. L'eutrophisation des eaux côtières, stimulée par l'augmentation de la charge en azote, est, incontestablement, la principale altération anthropogénique des écosystèmes côtiers dans le monde et la plus répandue (Nixon *et al.*, 1986; GESAMP 1990; Cloern *et al.*, 1995; Valeila *et al.*, 1997)

Le littoral français est classé en trois principaux faciès, les côtes rocheuses, les côtes sableuses et les marais et vasières. Ces derniers se répartissent sur presque 30% de la façade ouest de la France. Les marais salés ont été convoités depuis le moyen âge pour de multiples usages, marais salants, bassins à poisson, terres agricoles. Les XI, XII et XIII siècles correspondent aux plus grandes périodes d'endiguement jamais entreprises. La perte irréversible de superficie des zones humides naturelles a été un phénomène universel. En effet, ces écosystèmes sont profondément modifiés par l'homme pour le développement des activités salicoles, aquacoles ou agricoles. En Bretagne, par exemple plus 63% des marais littoraux ont été aménagés par l'homme.

L'aquaculture est en train de suivre le même essor qu'a connu l'agriculture il y a une cinquantaine d'années (Correll *et al.*, 1992). La production halieutique est dans l'incapacité actuellement de subvenir à la demande du marché mondial. La production en provenance de l'aquaculture devrait surpasser celle de la pêche en 2020, avec 50 10^6 T an^{-1} (FAO, 1991). Les marais endigués étaient souvent des sites préférentiels pour les pratiques salicoles, l'abandon progressif de la saliculture a libéré des terrains pour l'aquaculture. L'accessibilité du milieu et surtout la dynamique de la marée, assurant à la fois le remplissage des marais durant le flot et l'évacuation des eaux utilisées durant le jusant, induit une économie d'énergie et une nette baisse des charges pour les entreprises aquacoles, par conséquent un profit souvent assuré. Cependant, ces activités peuvent, à court terme, créer des perturbations locales liées à l'intensification des pratiques et provoquent alors des perturbations environnementales (Krom *et al.*, 1995; McCaig *et al.*, 1999). Le concept d'une aquaculture durable ou "Sustainable Aquaculture", est à l'heure actuelle devenu incontournable. Il permet de prendre en compte l'environnement parallèlement à la production et à la technologie, afin d'évaluer les réponses environnementales suite à des accroissements de la production (FAO, 1991).

LES OBJÉCTIFS DE LA THÈSE

Le transfert d'énergie entre la zone littorale marine et le reste de la zone côtière a fait l'objet de nombreuses études à différentes échelles d'analyse. Plus particulièrement, la quantification des échanges de nutriments dissous et de matière organique entre les marais salés, les estuaires et l'océan côtier a été conduite dans différents écosystèmes. Les causes et les implications de ces résultats ont été largement discutées (Wolaver *et al.*, 1985; Dame *et al.*, 1989; Childers *et al.*, 1993; Dame & Gardner, 1993; Boorman *et al.*, 1994; Dame & Allen, 1996).

Le concept généralement établi que les marais côtiers sont des systèmes très productifs et exportateurs de substances minérales et organiques (Odum, 1979), est basé sur des études réalisées aux Etats-Unis dans des écosystèmes peu anthropisés (Caroline du sud, Louisiane), et dans lesquels il existe une continuité entre les marais salés et la zone intertidale. Dans les marais européens, généralement endigués, nous avons voulu vérifier si ce schéma de forte production et d'exportation vers la zone côtière était respecté.

Bien que ces différentes études aient montré que les marais salés agissent généralement comme sources de nutriments (Jordan and Correll 1985, Jordan *et al.* 1986, Dame *et al.* 1991), l'aptitude de ces écosystèmes à piéger ou à libérer les nutriments dissous est vraisemblablement liée aux charges qu'ils reçoivent des écosystèmes amonts. Quantifier ces apports constitue donc une priorité, afin d'améliorer la compréhension de la dynamique des zones humides.

Pour tenter d'expliciter ces problématiques, deux questions principales ont été posées au cours de ce travail, auxquelles j'ai tenté de répondre :

➡ Les marais des côtes atlantiques françaises sont-ils des sources ou des puits de nutriments dissous et de matière particulaire?

➡ Existe-il une méthodologie générale qui permette d'établir le lien entre les caractéristiques des écosystèmes amonts, et les apports diffus au niveau des zones intertidales? Ceci afin de pouvoir prédire la réponse de l'écosystème côtier à des changements structuraux affectant les zones humides.

Le manuscrit est structuré sous la forme de trois chapitres.

De faibles biomasses phytoplanctoniques caractérisent le site d'étude (Barillé, 1996), contrairement à d'autres écosystèmes environnants tel que le bassin de Marennes-Oléron (Riera, 1995). Pour tenter d'expliquer ces différences, ce manuscrit débute par l'étude de la capacité trophique du système et l'étude des variations temporelles et spatiales de facteurs potentiellement limitants pour la biomasse phytoplanctonique (**Chapitre 1**).

Les possibilités de conclure quant au sens du bilan (importation ou exportation) de matière entre les marais et la zone côtière, se sont souvent heurtées à l'hétérogénéité des méthodes (Lefeuvre et Dame, 1994). La méthodologie utilisée au cours de ce travail, jusqu'alors inédite dans les écosystèmes européens, est largement présentée. Les résultats des bilans annuels estimés de nutriments dissous et de matière particulaire, ainsi que les implications de ces bilans sur la dynamique sédimentaire et trophique du système étudié, ont fait l'objet d'une analyse détaillée au cours d'un cycle annuel (**chapitre 2**).

Des outils d'analyses spatiales ont été développés et récemment appliqués à des problématiques environnementales, c'est la cas des Systèmes d'Informations Géographiques (SIG). Ces derniers présentent la grande originalité d'établir le lien entre la structure spatiale de l'écosystème et les processus biologiques et hydrologiques. Cette approche novatrice a été appliquée à notre site d'étude dans le cadre d'un modèle stationnaire développé à l'aide d'une interface (SIG) afin d'évaluer les charges diffuses d'azote inorganique dissous et des orthophosphates en provenance des marais endigués et afin d'étudier leurs répartitions spatiales au sein de la zone intertidale (**chapitre 3**).

Les trois chapitres de cette thèse sont présentés sous forme de publications rédigées en anglais. Afin de présenter le contexte général de cette étude, une synthèse a été rédigée en français. Elle illustre la démarche scientifique générale ainsi que les principaux résultats acquis au cours de ce travail.

SYNTHÈSE DES CHAPITRES

- **Le site étudié**
- **Conceptualisation du morphodynamisme des marais**
- **Variabilité hydrobiologique et météorologique: potentiel de production primaire**
- **Bilans annuels de flux de nutriments dissous de matière particulaire**
- **Les charges d'origine diffuse**
- **Discussions générales et perspectives**
- **Bibliographie**

I- LE SITE ÉTUDIÉ

L'île de Ré est comprise entre 46°15'31''N. à la pointe du Lizay et 46°08'29''O. à la pointe de Cheveau (Figure.1). Sa longeur maximale est de 24 km, de la pointe des Baleines (1°33'44''O) à la pointe de Sablanceaux (1°15'20''O). A l'est, elle est séparée par le coureau de la Pallice qui a 2500 m de large et qui est le siège de violents courants de marée.

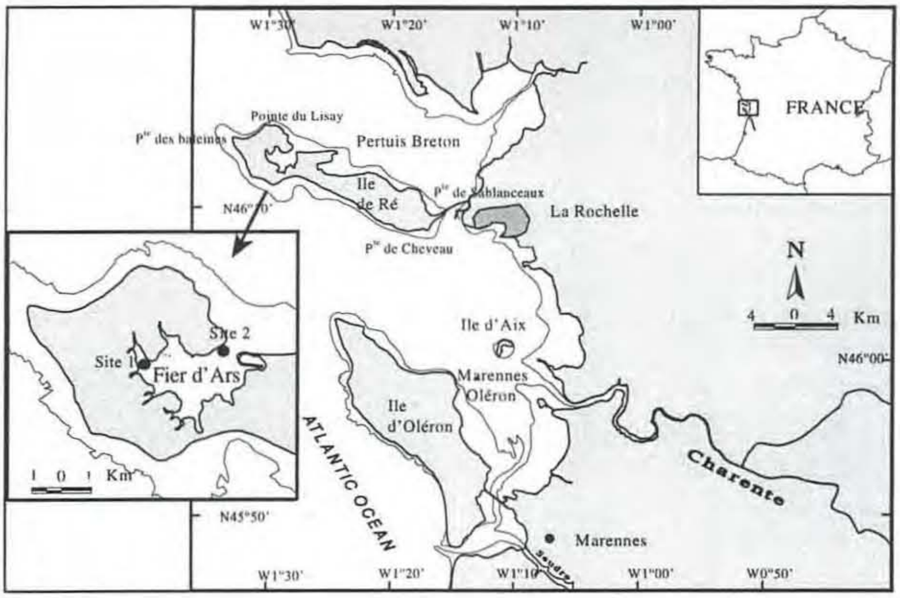


Figure 1 : Situation géographique de l'île de Ré et de la Baie du Fier d'Ars

L'île de Ré fait partie, avec l'île d'Oléron, d'un ensemble qui protège les côtes de Vendée et de Charente, entre la Baie de l'Aiguillon au nord et le Bassin de Marennes Oléron au sud (Germaneau, 1977). La côte méridionale de l'île de Ré est surtout exposée aux vents et houles de sud à ouest. La côte septentrionale est soumise à l'action des vents d'ouest à nord-ouest (Germaneau, 1977).

Le Fier d'Ars forme une mer intérieure constituée par des unités morphosédimentaires différentes. On peut ainsi distinguer essentiellement les marais endigués, les chenaux de marée et un estran vaseux (Figure.2).

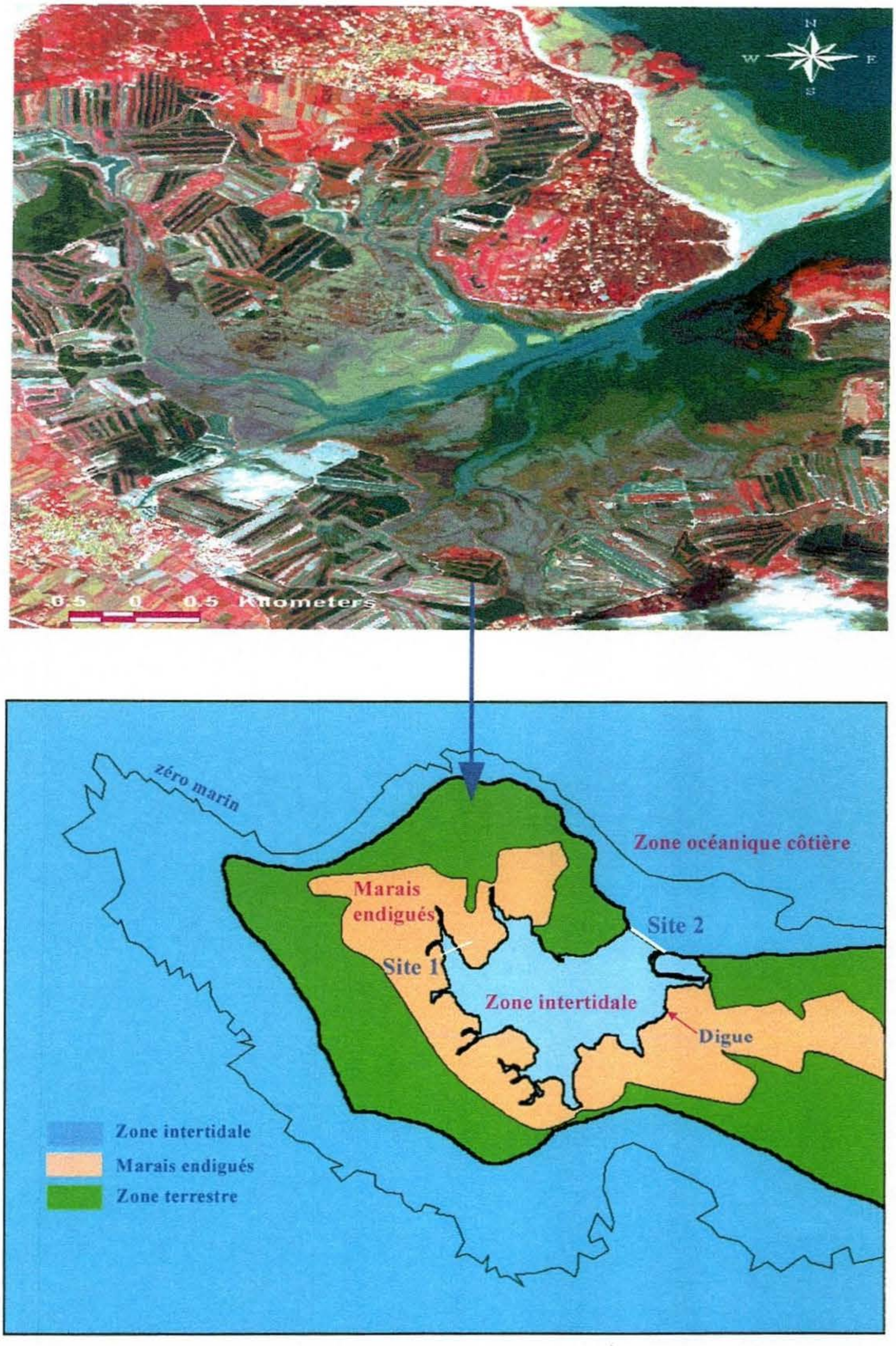


Figure 2 : Photographie aérienne de la baie du Fier d'Ars échelle 1/ 8 000 (photo du haut). Les différents compartiments morphologiques sont représentés schématiquement dans la figure du bas.

- Les marais endigués couvrent ~ 1200 ha. Ces terres gagnées par l'homme sur la mer, ont été reconquises par l'océan après la destruction des ouvrages de protection au cours des tempêtes. Une sédimentation marine s'est ensuite développée sur un sol continental. La vitesse de sédimentation montre une certaine variabilité, fluctuant entre 0.5 et 0.8 cm an^{-1} (Long, 1975).

- Les chenaux de marée correspondent aux axes de pénétration de l'eau dans le Fier d'Ars. Ils sont le siège d'importants courants tant durant le flot que durant le jusant.

- L'estran vaseux ou zone intertidale, couvre ~ 750 ha avec les chenaux de marée. La sédimentation est y assez homogène (Long, 1975). On y rencontre une faible couverture végétale.

Des critères de distinction hydrographique subdivisent ces entités morphosédimentaires en deux composantes distinctes: la slikke et le shorre.

- La slikke (Figure.3) (en néerlandais slijk=boue) est la partie inférieure et moyenne de la zone intertidale constituée de sédiment fins (vase, sable vaseux, tangue), inondée à chaque marée haute, même pendant les mortes-eaux. Elle est pauvre en végétation, sauf la partie la plus haute, qui est couverte de phanérogames marines, en particulier les zostères (l'espèce *Zostera noltii*).

La limite entre la slikke et le shorre est souvent floue. Cette transition dépend du rythme et de la durée de l'immersion par les hautes mers et le passage d'une zone à l'autre est donc contrasté si le marnage est important. En outre, certains auteurs font une distinction en se basant sur la colonisation végétale. Ainsi l'apparition d'un tapis végétal continu, constitué particulièrement des genres *Salicornia*, *Suaeda* et les graminées vivaces du genre *Spartina*, conjuguée à une variation de topographie (rupture de pente) annonce le passage au shorre (GEHU, 1976).

- Le shorre (en néerlandais schor=pré salé) correspond aux niveaux les plus élevés des marais salés; sa limite continentale est représentée le plus souvent par une digue dans le site étudié. Il se situe dans la zone inondable par les eaux marines lors des marées de vives-eaux, ou lors des tempêtes. Son couvert végétal est riche et dense, parfois interrompu par les enclaves de slikke remontant vers l'intérieur. Il se forme alors un réseau de chenaux (ou marigots) qui, tel un réseau hydrographique miniature, facilite la circulation des eaux. Les genres présents, *Halimione*, *Limonium*, *Suaeda* et *Artemisia* sont pourvus d'une souche ligneuse qui résiste au déchaussement de la plante et à l'érosion du substrat.

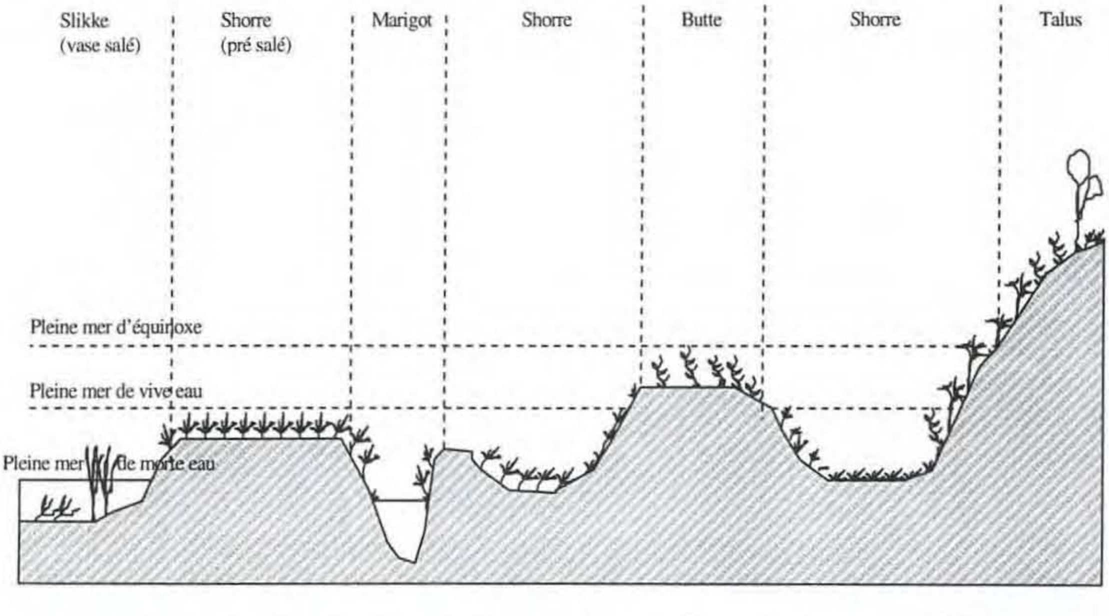


Figure 3 : Classification de l'estran d'après Bournieras *et al.* (1987)

L'action de la marée est prépondérante par rapport aux autres agents (vent, houle et clapots). Les courants de marée atteignent en quelques points des vitesses très élevées, ~ 4 noeuds à l'entrée du Fier. Seules certaines aires localisées, comme les parties internes de la vasière sont le siège d'un régime de courants très faibles (Long, 1975).

Les activités aquacoles se répartissent entre saliculture, ostréiculture et pisciculture (Figure.4):

La saliculture est la première activité installée dans le Fier d'Ars, le XIX ème siècle est celui de son apogée, avec une production annuelle moyenne de 31750 T. Depuis, la production n'a cessé de décroître, 18500 T en 1905 et seulement 1200 T en 1980. La crise économique de 1846 et la concurrence avec les sels exploités mécaniquement en méditerranée, sont les principales causes de ce déclin. A l'heure actuelle, la saliculture est en train de vivre un second souffle, 2200 T en 1997, grâce à une volonté régionale de promouvoir une activité ayant très peu de conséquences sur les écosystèmes environnants, et qui marque surtout l'identité culturelle de la région.

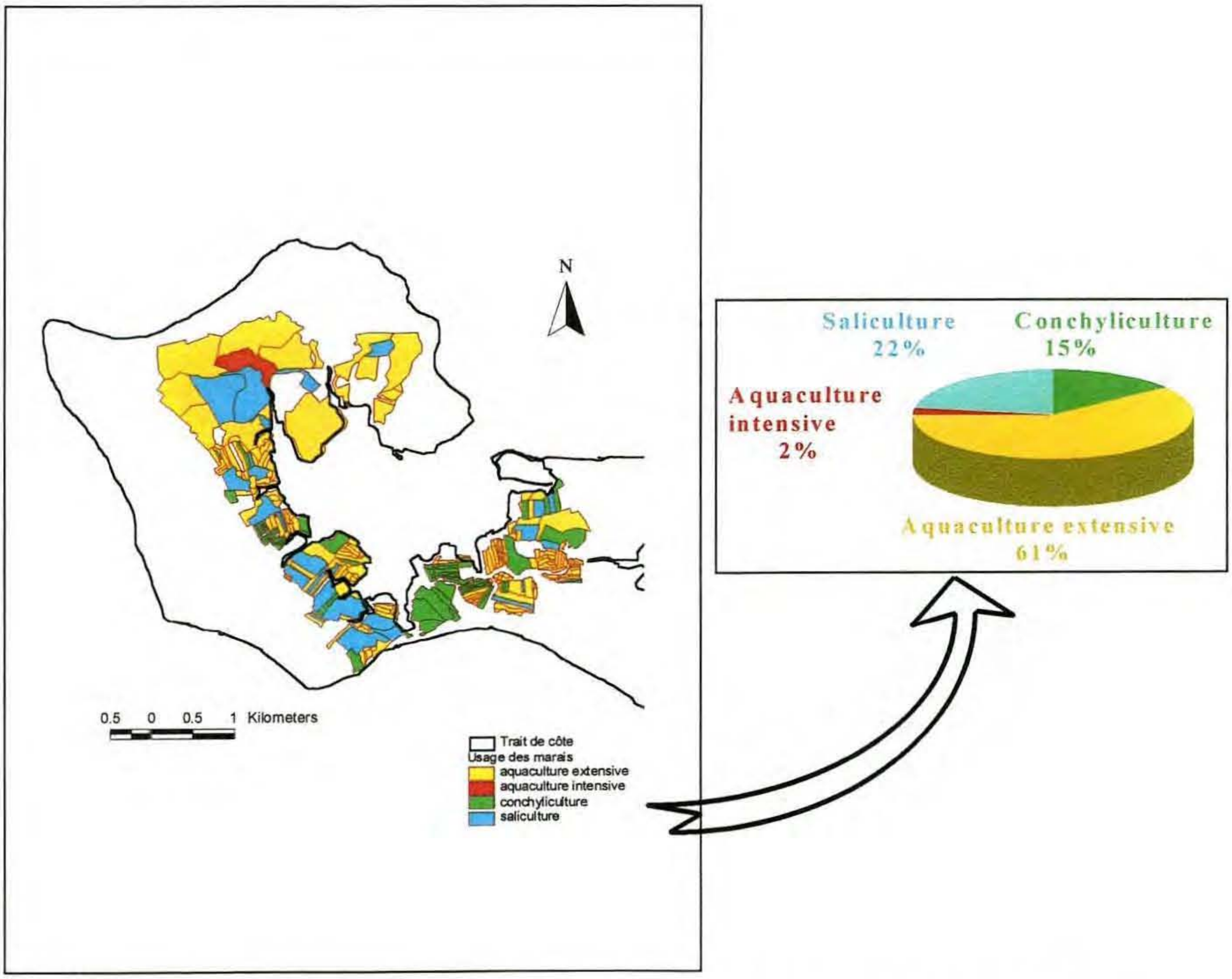


Figure 4 : Carte des usages du marais et la répartition des superficies allouées à chaque activité.

L'ostréiculture l'élevage des huîtres sur parcs situés sur l'estran découverts à marée basse, s'accompagne d'une phase d'engraissement et de verdissement appelée affinage. Ce dernier se pratique dans des claires à huîtres. Le Fier d'Ars en compte ~ 156 Ha.

La pisciculture extensive est souvent considérée comme une activité annexe à la saliculture. Les espèces élevées sont celles qui pénètrent naturellement dans les marais lors des procédures de remplissage. Il s'agit le plus souvent de l'anguille (*Anguilla anguilla*) ou du mulet (*Mugil sp*), rarement du bar (*Dicentrarcus labrax*) ou de la daurade (*Sparus aurata*). Le rendement annuel fluctue autour de 100 kg ha^{-1}.

La pisciculture intensive Elle concerne la Ferme Marine des Baleines. Récemment installée (1990), elle s'entend sur une superficie de 26 ha. Pour satisfaire des critères de conservation du littoral, la ferme a adopté une technique d'élevage à partir des bassins de terre. L'eau des bassins d'élevage est continuellement renouvelée depuis un bassin réservoir d'eau de mer. Le réseau d'alimentation hydraulique fonctionne en circuit ouvert durant les grands coefficients de marée, alors qu'un circuit fermé est mis en place durant les petits coefficients (< 63). La ferme produit quasi-exclusivement du bar (*Dicentrarcus labrax*) avec une spécialisation dans le grossissement d'alevins achetés à d'autres entreprises. La production est en continuelle augmentation, elle s'est élevée à 400 T en 1995 (Hussenot *et al.*, 1998) et a atteint récemment 500T par an.

Particularités du site étudié

Pourquoi avoir choisi le Fier d'Ars pour mener cette étude?

Le Fier est une Baie semi fermée. Malgré ses dimensions relativement réduites (2000 ha), elle présente une grande variabilité de faciès, entre une vasière intertidale (750 ha) et une zone de marais endigués (1250 ha). La cohabitation de différentes activités aquacoles en fait un modèle unique pour l'étude de l'influences de l'aquaculture sur les charges en nutriments et le transfert de matière entre zone côtière et mer ouverte.

De plus, la position géographique du Fier, partie intégrante de l'île de Ré, zone de grande production ostréicole et surtout à grande vocation touristique, a bénéficié d'un intérêt particulier pour des fins d'aménagements côtiers de la part des institutions locales (DDE) et a été également le siège de nombreux programmes de recherche (IFREMER) ainsi que de compagnes d'images satellitaires (CNES). Les supports imagiers aussi bien sous forme papier que numérique: photographies aériennes, images sattelitaires, banque de données topographique... étaient souvent des données disponibles. Les relevés bathymétriques sont aussi réactualisés (relevés en 1975, 1985, 1998).

La représentativité du site, l'accessibilité du terrain ainsi que la qualité et l'abondance des données ont été des critères déterminants pour le choix du site.

II- CONCEPTUALISATION DU MORPHODYNAMISME DES MARAIS

La conceptualisation du morphodynamisme des marais a été déjà développée par Childers *et al.* (1993), dont une version modifiée est proposée dans la figure (5). Elle implique comme variables de forçage:

↳ Les variables hydrodynamiques et hydrobiologiques dans la variation à **court terme des flux de matière.**

↳ Les caractéristiques sédimentaires, biotiques et topographiques dans la variation à **long-terme des données sédimentaires.**

-↳ Les charges d'origine diffuse sont à leur tour impliquées dans les deux processus précédants.

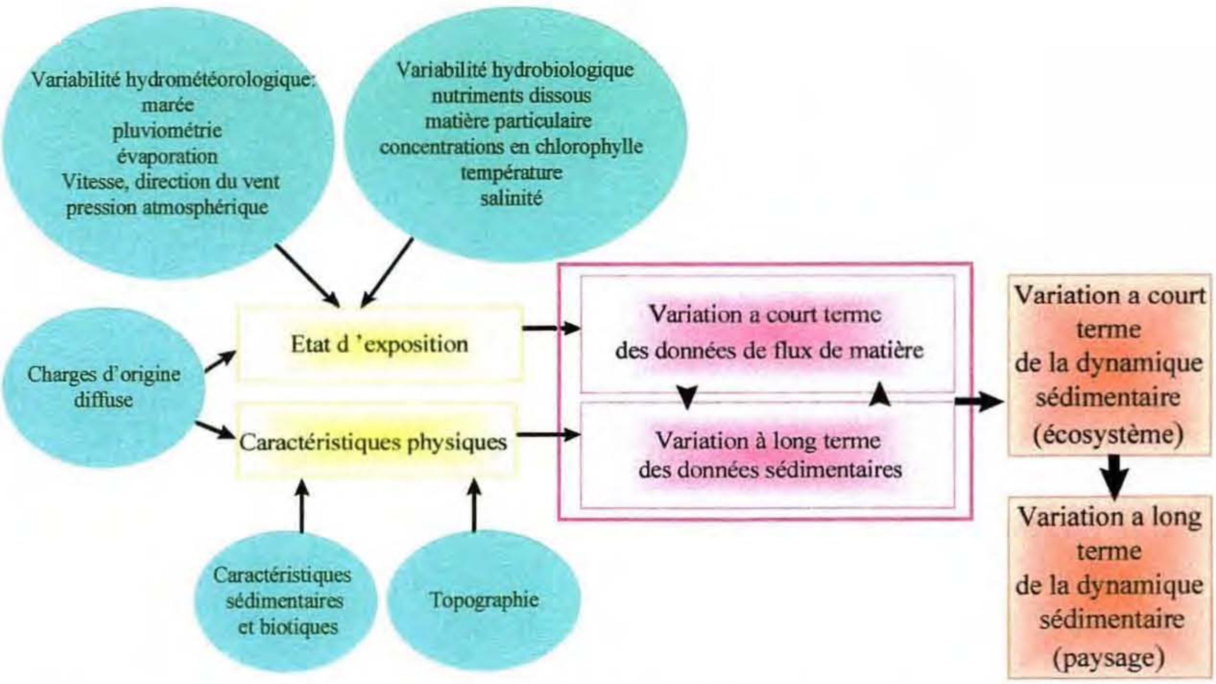


Figure 5 : Modèle conceptuel modifié du morphodynamisme des marais (d'après Childers *et al.*, 1993)

Les variables hydrométéorologiques concernent la vitesse et la direction du vent, l'amplitude de la marée, les temps d'immersion, la pluviométrie et la pression atmosphérique. Les variables hydrobiologiques regroupent les concentrations en sels nutritifs, la teneur en matière particulaire, les variables de production, à savoir teneur en chlorophylle a et phéopigments et des variables hydrologiques telles que température et salinité.

Les notions de quantité et de vulnérabilité ont été souvent évoquées dans la définition du risque. En particulier, la vulnérabilité a été exprimée dans le modèle conceptuel (Figure.5) par un état d'exposition, en d'autre termes il illustre la sensibilité du système aux actions des variables de forçage.

Les interactions entre ces variables déterminent l'aptitude du système à produire ou à consommer de l'énergie, selon son degré d'exposition, sa disponibilité à échanger cette dernière. Ces échanges se manifestent sous la forme de flux de matières.

La topographie ainsi que les caractéristiques sédimentaires et biotiques induisent les réponses physiques de l'écosystème qui ne sont perceptibles qu'à une plus ample échelle de variation à long-terme des données sédimentaires.

Les marais, aussi bien que l'ensemble des zones humides littorales, possèdent une position écologique très fragile. Ces biotopes, situés à la charnière entre terre et mer sont sujets à un enrichissement par des éléments dissous et particulaires. Les sources les mieux identifiées d'enrichissement externes sont les fertilisants d'origine agricole, les eaux urbaines, la déposition atmosphérique et les flux souterrains diffus. L'impact de ces charges diffuses sur les zones humides varie en fonction de la contribution relative de chacune de ces sources. Néanmoins, les caractéristiques physiques du système, principalement sa topographie et ses propriétés biotiques et sédimentaires, déterminent l'aptitude du système à contenir, transformer ou exporter ces apports exogènes.

Un feed-back entre les variations à court terme des flux de matière et les variations à long terme des données sédimentaires stimule une dynamique sédimentaire à court terme, perceptible à l'échelle de la station étudiée voire de l'écosystème étudié. Une variation à long terme se répercutera donc sur l'ensemble du paysage des zones humides, l'identifiant ainsi comme une entité géomorphologique et fonctionnelle à part entière. Ainsi cette stimulation à une échelle paysagère repose sur des interactions entre les flux de matière et l'état d'exposition, entre la sédimentation et la topographie et entre l'âge géologique et la géomorphologie. Elles intègrent l'ensemble des mesures spatiales, effectuées à différents cadres temporels, à l'intérieur de prédicteurs spatialisés, articulés entre eux, afin de définir le morphodynamisme des marais.

III- VARIABILITÉ HYDROBIOLOGIQUE ET MÉTÉOROLOGIQUE : POTENTIEL DE PRODUCTION PRIMAIRE

Les variables hydrobiologiques et météorologiques jouent un rôle essentiel dans la consommation ou la production d'énergie, souvent quantifiée en terme de biomasse phytoplanctonique. Cependant, ces variables peuvent agir à la fois simultanément ou marquer une transition temporelle (Pennock and Sharp, 1994) ou spatiale (Malone *et al.*, 1996) à l'échelle de l'écosystème.

Au cours de ce chapitre, on s'intéressera aux changements d'échelle temporelle et spatiale des facteurs (lumière et disponibilité en sels nutritifs) limitant potentiellement la biomasse phytoplanctonique dans la Baie macrotidale du Fier d'Ars.

1- Etude de la variabilité physico-chimique et hydrobiologique

1.1- Protocole expérimental

Deux sites expérimentaux, l'un situé à l'intérieur du Fier d'Ars et l'autre à l'interface avec la mer ouverte (Figure.1), ont été échantillonnés entre avril 1997 et mars 1998. Des prélèvements bimensuels ont été effectués (soit 28 cycles pour chaque site), au cours des vives eaux (coefficient de marée 85) et des mortes eaux (coefficient de marée 45). En raison de la forte variabilité journalière, l'ensemble du cycle de marée a été suivi à raison de 12 prélèvements en moyenne, 6 prélèvements pendant le flot et 6 durant le jusant. Les analyses ont été effectuées pour les sels nutritifs: nitrate et nitrite (NN), ammonium (NH4), silice (Si), orthophosphates (PO_4), matière en suspension (TSS), chlorophylle a (Chl), phéopigments (Phéo) et carbone organique dissous (DOC). Les variables tels que salinité (Sal), température

(Temp) et irradiance photosynthétiques (PAR) ont été simultanément enregistrées. Le détail de la méthodologie employée est décrit dans la publication jointe (Chapitre I).

1.2- Analyse en composante principale des variables physico-chimiques.

L'analyse en composante principale (ACP) permet de présenter une synthèse des résultats. Elle est réalisée sur un tableau croisé: relevés x variables, qui révèle que les deux premiers axes expliquent 60% de l'inertie totale au niveau du site 1 alors que pour le site 2 les deux axes factoriels F1 et F2 expliquent ~57% de la variance totale.

Le cercle de corrélation du plan factoriel F1 F2 (Figure.6 a) montre que pour le site 1, l'axe F1 est positivement corrélé à toutes les formes d'azote inorganique dissous, mais il est négativement corrélé à Sal, Temp et PAR. En revanche, l'axe F2 est positivement corrélé à PO_4 et négativement corrélé au rapport Si:P, indiquant clairement un gradient de concentration des phosphates.

L'ordination des relevés indique une large dispersion dans le sens de l'axe F2. Entre mai et septembre la dispersion des prélèvements reste assez limitée, ils se dispersent du mois de juin vers août autour des variables TSS et PAR, mettant en évidence l'importance aussi bien de la turbidité que de la lumière au cours de cette période. Une plus ample dispersion est cependant notée à partir du mois d'octobre et les scores se repartissent essentiellement autour du rapport Si:P.

Pour le site 2, le cercle de corrélation dans le plan factoriel F1 F2 (Figure.6 b) montre que l'axe F1 est négativement corrélé à Sal, Temp, PAR et le rapport Si:N, alors qu'il est positivement corrélé à l'azote inorganique dissous (DIN), Si et NN, exprimant ainsi la qualité physico-chimique de l'eau et une prédominance des éléments azotés. La corrélation de l'axe F2 avec les variables Si:P et N:P exprime de façon très nette la disponibilité proportionnelle des sels nutritifs, caractérisant la richesse de l'eau et conditionnant la productivité du milieu.

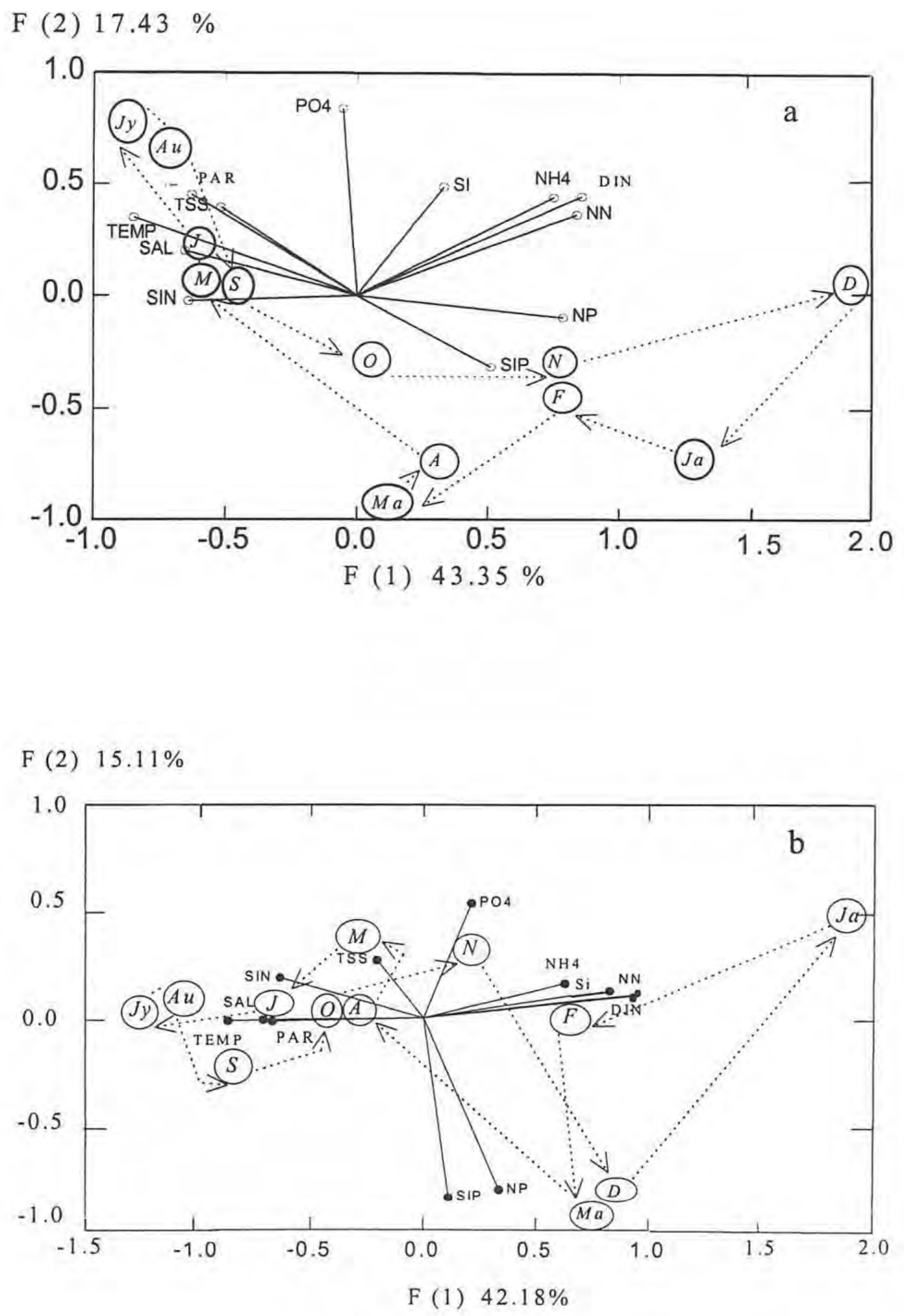


Figure 6 : Schéma vectoriel, du site 1 (a) et site 2 (b), indiquant la contribution aux Facteur 1 (F1) et Facteur 2 (F2) des 12 variables enregistrées. La dispersion de la moyenne mensuelle des relevés, représentée à l'intérieur des cercles, est aussi indiquée. La connexion entre les relevés mensuels est représentée par des flèches discontinues. Pour mieux identifier les mois une distinction est faite entre janvier (Ja), juin (J) et juillet (Jy), mars (Ma) est aussi à distinguer de mai (M) et avril (A) d'août (Au).

L'ordination des relevés montre qu'ils se dispersent d'avril à octobre autour de l'axe F1. Toutefois, d'avril jusqu'en juillet, ils se dirigent vers les valeurs négatives de l'axe F1, où ils s'approchent de TSS au mois de mai et de PAR en juin. A partir de juillet, ils changent de direction vers les coordonnées positives, essentiellement représentées par les concentrations azotées.

1.3- Dynamique annuelle de la biomasse phytoplanctonique.

L'analyse pigmentaire (Figure.7) a révélé de faibles valeurs de Chlorophylle, n'excédant pas 3.5 μgl^{-1} à l'aval du système (site 2), et $7\mu\ l^{-1}$ à l'amont (site 1). Les Phéopigments, résultant de la dégradation de la Chlorophylle fluctuent eux aussi autour de faibles valeurs, au dessous de 6 μgl^{-1} et de 15 μgl^{-1} au niveau des sites 2 et 1 respectivement.

Deux périodes de fortes concentrations se distinguent de la tendance saisonnière au niveau du site 2. La première, allant de mai à juillet, qu'on assimile à une efflorescence du printemps-été, est notée ″période 1″. La deuxième ″période 2″, un peu plus étendue, d'août jusqu'à novembre, est assimilée à une efflorescence automnale.

Pour l'amont de la baie (site 1), une distinction est faite entre une période de fortes teneurs en Chlorophylle ″période 1″, s'étendant de mai à septembre, et une deuxième d'octobre à mars ″période 2″caractérisée par de faibles valeurs.

De faibles productions planctoniques ont souvent été signalées le long des côtes atlantiques françaises, notamment à cause des fortes turbidités empêchant la pénétration de la lumière (Héral, 1985). Cependant, les valeurs de Chlorophylle enregistrées dans des écosystèmes fortement turbides tels que Marennes-Oléron ou l'Estuaire de la Gironde sont nettement plus élevées que celles enregistrées dans le Fiers d'Ars. (Tableau.1).

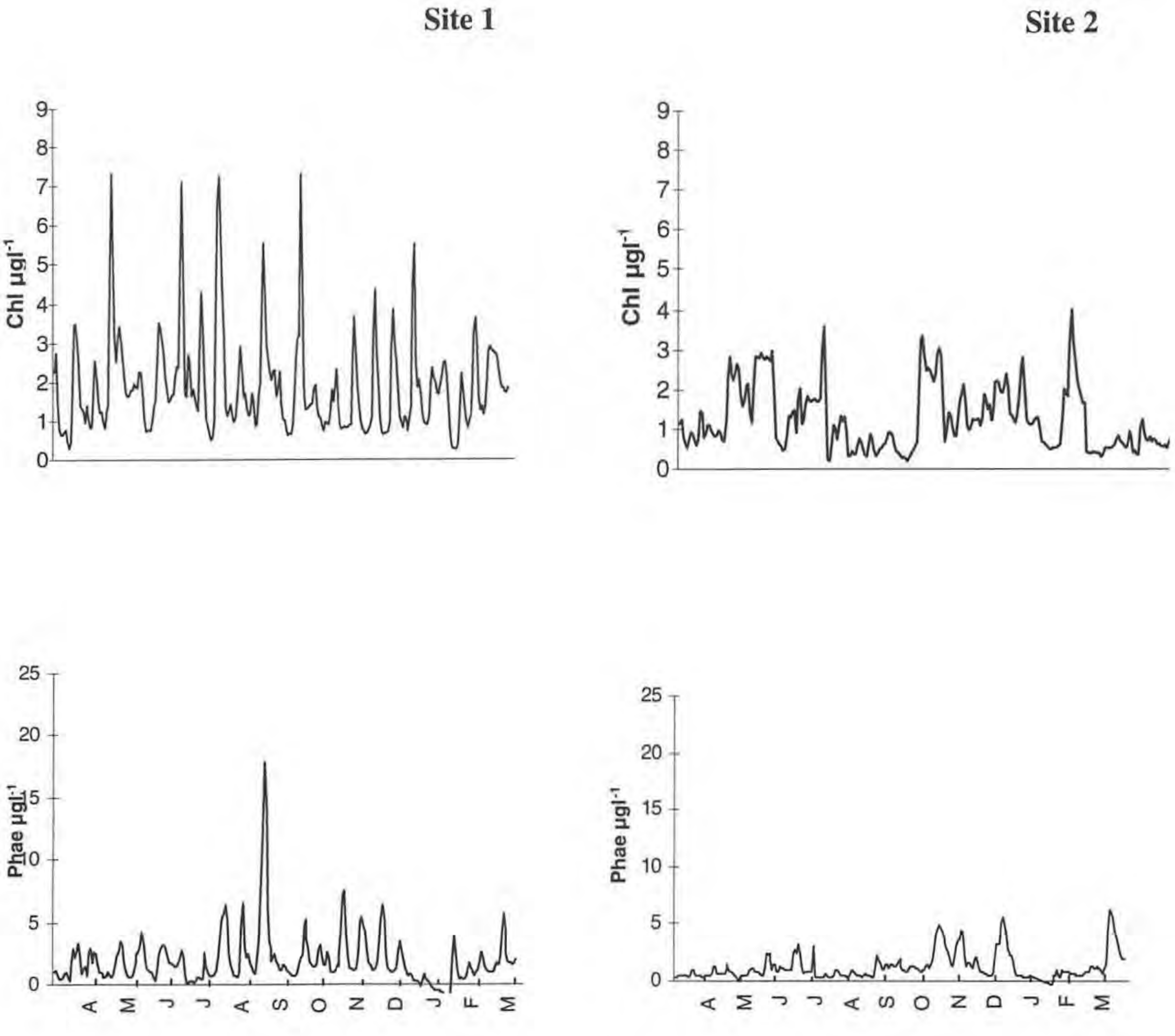


Figure 7 : Cycles mensuels de la Chlorophylle a (Chl) et des Phéopigments (Phae) au de site 1 (gauche) et site 2 (droite).

De ce fait, d'autres facteurs pourraient interférer voire remplacer la lumière dans le rôle de limitation de la production primaire au niveau de l'écosystème étudié.

Tableau 1 : Différentes valeurs de concentrations de chlorophylle et de matière en suspension dans des écosystèmes turbides avoisinant le site d'étude.

Printemps-été		Automne-hiver		Site	Référence
Chl µgl^{-1}	TSS gl^{-1}	Chl µgl^{-1}	TSS gl^{-1}		
1-25	0.1-3	0.1- 8	0.25-2	Gironde	Irigoien & Castel (1998)
5-90	0.2-0.5	1.5-5	0.2-0.7	Marennes-Oléron	Riera (1995)
1-4	0.03-0.1	0.5-1	0.01-0.2	Pertuis Breton	Barillé (1996)
0.2-3.5	0.05-0.5	0.4-4	0.01-0.05	Fier d'Ars	cette étude

1.4- Modèles de régression multiple de chlorophylle, Chlorophylle x PAR et phéopigments en fonction des scores de l'ACP (F1 et F2).

Les variables, Chlorophylle (Chl), Phéopigments (Phéo) et le produit Chlorophylle multipliée par les Radiations Photosynthétiquements Actives (PAR), Chl x PAR, (utilisé comme indicateur de la productivité phytoplanctonique) (Malone *et al.*, 1996), n'ont pas participé activement à l'analyse en composante principale, et par conséquent à la construction des axes. Elles ont été utilisées de ce fait comme **variables supplémentaires** ou **variables à expliquer**.

Les modèles de régressions multiples des variables supplémentaires en fonction des scores des axes F1 et F2 montrent que (Tableau.2):

- Au niveau du site 1: une forte corrélation en deuxième période entre les concentrations en Chl et le facteur (F2), ce dernier fortement corrélé aux concentrations en phosphates.

- Au niveau du site 2: de fortes corrélations entre la Chl, Chl x PAR et le facteur F1 au cours de la première période, qui deviennent nettement plus significatives en deuxième période. Ces résultats démontrent que la biomasse phytoplanctonique ainsi que sa productivité dépendent fortement des paramètres intervenant dans l'élaboration du premier axe de l'ACP, à savoir Temp, Sal et DIN.

Tableau 2 : Résumé des régressions multiples (r ajustés et valeurs de P) pour les variables indépendantes Chlorophylle a (Chl), le produit Chlorophylle x radiations photosynthétiquement actives (ChlxPAR) et Phéopigments (Phéo), corrélées aux scores de l'ACP. L'ACP est construite à partir de 12 variables caractérisant la colonne d'eau. L'étude a été réalisée pendant les périodes de fortes (1) et faibles (2) teneurs en Chl pour le site 1 et les deux périodes de fortes concentrations chlorophylliennes pour le site 2.
Le modèle est: Chl ou (Chl x PAR) ou Phae = constante + μ_1 Facteur 1+μ_2 Facteur 2. P significative (< 0.01) est indiquée en gras.

		n	r	Paramètres statistiques μ_i (valeur de P)	
Période				Facteur 1	Facteur 2
Site 1					
1	Chl	127	0.487	2.653 (0.013)	0.901 (0.049)
	Chl x PAR	127	0.349	0.109 (0.114)	0.042 (0.159)
	Phéo	127	0.520	0.99 (0.128)	0.924 **(0.001)**
2	Chl	105	0.502	0.210 (0.043)	0.642 **(<0.0001)**
	Chl x PAR	106	0.082	0.002 (0.712)	-0.005 (0.408)
	Phéo	105	0.320	-0.688 **(0.001)**	0.370 (0.230)
Site 2					
1	Chl	52	0.414	0.602 **(0.003)**	0.010 (0.410)
	Chl x PAR	52	0.426	0.075 **(0.003)**	-0.012 (0.421)
	Phéo	52	0.498	0.027 (0.882)	-0.450 **(<0.001)**
2	Chl	75	0.615	-0.929 **(<0.0001)**	-0.113 (0.725)
	Chl x PAR	76	0.759	-0.239 **(<0.0001)**	0.046 (0.322)
	Phéo	76	0.244	0.311 (0.439)	-0.674 (0.251)

2- Variations des échelles spatio-temporelles de facteurs limitant potentiellement la biomasse phytoplanctonique.

L'échelle spatiale va être appréhendée en comparant les variations observées respectivement au niveau des deux sites d'étude, alors que l'échelle temporelle le sera en comparant les deux périodes précédemment définies (période 1) et (période 2).

- Au niveau du site 1:

La décomposition de l'échelle annuelle en périodes de faibles et fortes concentrations de Chlorophylle, montre que le modèle de régression multiple est uniquement significatif durant la période des faibles teneurs en Chlorophylle, ce qui appuie l'hypothèse d'une limitation potentielle par les phosphates pour cette période. En revanche, durant le printemps et l'été, l'accumulation de biomasse phytoplanctonique est assurée par les apports de DIN en provenance des marais endigués ~ 78 μM de NH_4 (Hussenot *et al.*, 1998) et de la régénération benthique (Gouleau *et al.*, 1995). D'autre part une corrélation linéaire entre les concentrations en PO_4 et de matière en suspension (TSS) a été démontrée en période estivale ($r=0.53$, $P<0.001$) montrant l'existance d' un flux de PO_4 d'origine benthique. Ce processus a été déjà signalé dans des marais expérimentaux charentais (Crottereau, 1999) et dans d'autres écosystèmes américains, tels que la Baie de Chesapeake (Malone *et al.*,1996). Il a été expliqué par la dissolution du complexe Fe-P sous des conditions réductrices (Cornwell *et al.*, 1996).

En fin d'été et durant l'automne (Figure.8), de forts apports de DIN sont enregistrés en provenance du bassin versant (Barillé, 1996). Toutefois, les concentrations en PO_4 restent relativement faibles (0.1-0.5 μM) induisant des rapports atomiques N:P de l'ordre de 100-400, pouvant être limitants pour la productivité planctonique.

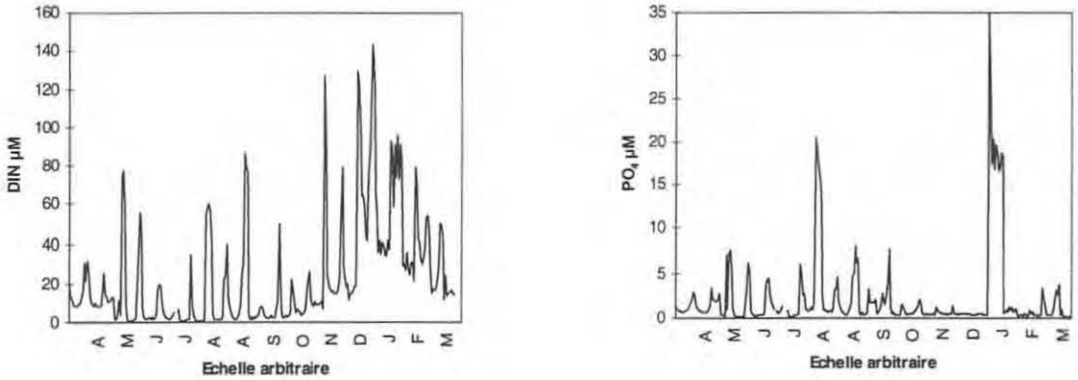


Figure 8 : Distributions mensuelles des deux variables DIN et PO_4 à différents coefficients de marée et à différents temps du cycle tidal au niveau du site 1.

- Au niveau du site 2:

L'ordination des relevés révèle une forte dépendance de paramètres tels que Sal, Temp, TSS et PAR. Néanmoins, l'absence de corrélation inverse entre Chl et TSS durant la période 1 va à l'encontre de l'hypothèse de la limitation par la lumière, résultante de l'absorption et de la réflexion des rayons incidents sur les particules solides (Cloern, 1987; Barillé, 1996; Irigoien & Castel, 1997). De fortes corrélations apparaissent entre Chl et Sal (Tableau.3) suggérant une importation de la Chlorophylle de régions dont les zones euphotiques sont plus profondes.

Tableau 3 : Corrélations entre Chl (y: $\mu g l^{-1}$), TSS ($mg l^{-1}$), salinité et température. Les probabilités significatives, P (< 0.01) sont en gras.

Période	Chl-TSS	r	Chl-Sal	r	Chl-Temp	r
1	y=-0.31x+1.58	0.089	y=-0.22x+9.3	0.522	y=-0.30x+7.16	0.533
	-0.45		**(<0.0001)**		**(<0.0001)**	
2	y=35.81x+0.62	0.683	y=1.16x-38.9	0.514	y=0.08x+0.04	0.42
	(<0.0001)		**(<0.0001)**		**(<0.0001)**	

Il apparaît donc que la population phytoplanctonique au niveau du Fier d'Ars ne serait pas d'origine autochtone. Une fois arrivée dans la baie, il est vraisemblable qu'elle est sujette à des chocs halins. Le mélange vertical (vitesse du courant atteignant $1 ms^{-1}$) peut aussi engendrer des variations rapides du niveau de pénétration de la lumière, de sorte que les cellules ne peuvent ajuster leurs physiologie à ces conditions changeantes (Demers *et al.*, 1986).

Durant la deuxième période, la corrélation entre Chl et TSS (Tableau.3) suggère une mise en resuspension du microphytobenthos, estimée à 15% de la population microbenthique de la Baie de Marennes Oléron (Guarini, 1998) alors qu'elle varie entre 17-32% dans l'estuaire de la Gironde (Santos, 1999). La productivité phytoplanctonique à ce stade semble être régulée par le broutage (Frikha, 1989; Pennock & Sharp, 1994; Malone *et al.*, 1996), comme illustré par l'augmentation des teneurs en Phéopigments (Figure.7). Un couplage étroit entre la biomasse Chlorophyllienne et la consommation de DIN a été illustré par la corrélation négative entre ces deux variables ($r=0.467$, $P<0.0001$) mettant en évidence un assimilation rapide de DIN, ce dernier finissant par être limitant dans le milieu (Figure.9).

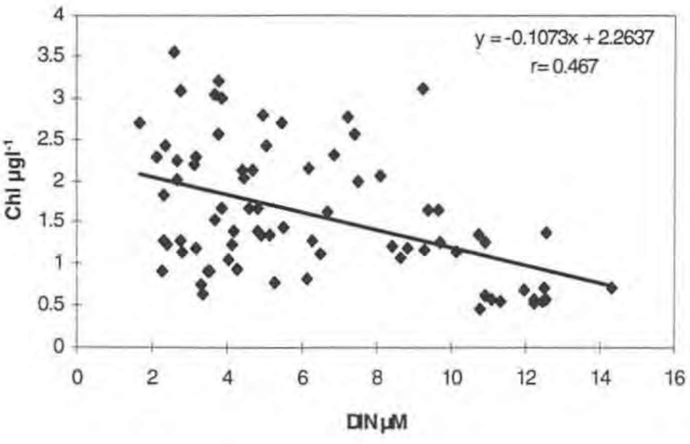


Figure 9 : Corrélation entre les concentrations de DIN et de Chl durant la période automnale de forte biomasse phytoplanctonique (période 2).

3- Conclusions

Les résultats discutés ci-dessus permettent de distinguer (Tableau.4) des variables ou des processus susceptibles de contrôler la biomasse phytoplanctonique, en contribuant, soit à son augmentation, tel est le cas de la ressuspension, soit à sa diminution, le broutage en est un exemple. Des variations de température provoquant quant à elles, des changements dans la composition spécifique de la communauté planctonique (Frikha, 1989; Dupuy, 1999). Parmi les facteurs de contrôle de la biomasse phytoplanctonique, quelques uns s'avèrent limitants, une diminution i.e. lumière, DIN et PO_4 de ces facteurs dans le milieu serait un facteur limitant pour la production phytopanctonique.

Tableau 4 : Résumé des facteurs contrôlant et ceux potentiellement limitant la biomasse phytoplanctonique durant les deux périodes d'efflorescence algale.

	Origine	Facteurs de contrôle		Facteurs limitants	
		Printemps-été	Automne	Printemps-été	Automne
Site 1	Autochtone	Lumière	PO_4 Ressuspension	Lumière	PO_4
Site 2	Allochtone	Salinité Lumière Température	Température Broutage Ressuspension DIN	Lumière	DIN

Ainsi selon le site et la période de l'année, une complexe succession de facteurs (lumière, azote et phosphore) assure le contrôle et la limitation de la production primaire. Ce schéma est non classique par rapport aux certitudes avancées par certains auteurs qui trouvent que l'azote et \ ou le phosphore sont constamment limitants pour un site donné (Barillé, 1996).

IV- BILANS ANNUELS DE FLUX DE NUTRIMENTS DISSOUS ET DE MATIÈRE PARTICULAIRE

1- Protocole expérimental

Deux transects bathymétriques ont été établis pour l'étude des flux de nutriments et de matière particulaire: à l'intérieur de la baie, à l'interface avec les marais salés endigués, et à l'extérieur de la baie, à l'interface avec la mer ouverte. L'estimation annuelle est basée sur l'échantillonnage de 28 cycles de marée, au sein d'une population annuelle finie de 704 cycles au cours de l'année 1997-1998. Le détail de la méthodologie employée est donnée dans le chapitre II.

L'approche utilisée a été basée sur la recherche de corrélations multiples (Tableau.5) à partir des paramètres hydrométéorologiques comme variables prédictrices des flux instantanés (Spurrier & Kjerfve, 1988). Nous avons anticipé l'existence de périodicités dans les flux nets, correspondant aux cycles annuels, bimensuels, diurnes et semidiurnes, en plus de variations non périodiques dues aux tempêtes, à des conditions de vents et à des fronts thermiques.
Cette méthode présente l'intérêt majeur d'estimer la variabilité associée à l'estimation de la moyenne annuelle des flux. Elle tient compte, aussi bien de la variabilité intracycle, que de la variabilité intercycle de marée (Dame *et al.*, 1989).

L'importation ou l'exportation seront considérées statistiquement significatives à un seuil de 95% quand l'intervalle de confiance de la moyenne ne contient pas la valeur zéro, de sorte que le flux annuel moyen garde le même sens de variation (positif pour l'importation et négatif pour l'exportation).

2- Flux à l'interface des marais endigués

Une importation significative de l'azote est observée avec une large contribution de l'ammonium qui s'élève à plus de 73% de l'azote inorganique dissous (Figure.7).

Tableau 5 : Résultats du modèle de régression multiple. Les variables prédictrices sont la vitesse moyenne et maximale du vent, respectivement Vt et Vp, la pression atmosphérique au cours du cycle de marée échantillonné et du cycle précédent respectivement, Prmn et Prmn1, la température de l'air et de l'eau respectivement, Tam et Tom, les précipitations au cours du cycle de marée échantillonné et du cycle précédent, respectivement Pn et Pn1 et la hauteur maximale de l'eau au cours du cycle échantillonné Hn et au cours du cycle précédant Hn1. E (ebb) correspond au jusant et F (flood) indique le flot. Le coefficient de détermination R^2 est reporté et l'étoile, *, indique que le modèle n'est pas significatif à un seuil de 5%.

	Urea		NN		NH_4		PO_4		Si		TSS		MSS		OOS		Chl a		DOC	
Transect	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2
Flood R^2	0,44	0,11	0,51	0,58	0,35	0,53	0,22	0,13	0,10	0,32	0,41	0,42	0,39	0,43	0,47	0,37	0,16	0,13	0,60	0,56
		*					*	*	*									*		
Vt.	E	E				E	F	E	E										F	E
Vp	E			E			F		FE	E				F					FE	
Prmn							E	E												F
Prmn1	F	F	FE	FE			FE			E				E					E	
Tam	F			F	F	E			E		FE		FE		F					
Tom	F	E	FE	E	FE	F		FE		FE		E			E	FE	FE		F	
Pn																		FE		
Pn1		FE				F				F										F
Hn			F		F		E							FE		E				E
Hn1			F		E		FE	E		F		FE	E					FE	E	F
Ebb R^2	0,21	0,34	0,76	0,59	0,33	0,24	0,15	0,47	0,19	0,36	0,37	0,45	0,48	0,37	0,36	0,34	0,26	0,31	0,38	0,58
	*						*		*											

L'urée montre aussi une tendance à l'importation. La matière particulaire est aussi significativement importée avec une nette dominance pour la fraction minérale ~ 73%. La silice et les phosphates montrent une tendance à l'exportation, toutefois seule l'exportation annuelle de la silice est statistiquement significative.

3- Flux à l'interface de l'océan

Une exportation d'azote minéral a été estimée avec une dominance pour les nitrites et nitrates alors que l'ammonium est importé dans le système (Figure.10). La matière particulaire est également importée avec une dominance de la fraction minérale qui représente 76% du total particulaire. Le carbone organique dissous et la Chl sont aussi exportés de la baie mais le modèle montre une tendance significative à un seuil de 95% que pour le premier.

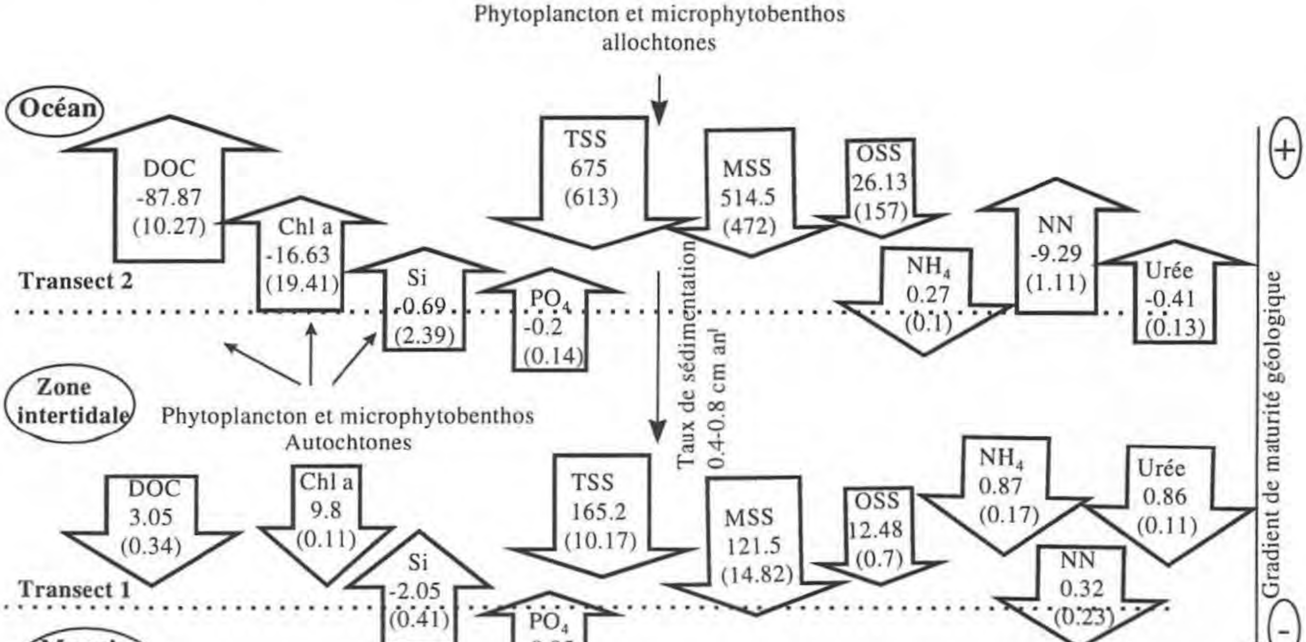


Figure 10 : Représentation schématique du budget annuel des nutriment dissous et de la matière particulaire, évalué à deux transects bathymétriques au niveau de la baie. La contribution de la production primaire allochtone et autochtone est indiquée. Le gradient de la maturité écologique est aussi reporté. Les bilans sont exprimés en g m^2 an^{-1} à l'exception de la Chl a exprimée en mg m^2 an^{-1}. L'importation est représentée par des flèches dirigées vers le bas alors que l'exportation par des flèches vers le haut.

4- nteractions physico-chimiques des processus et évolution morpho-dynamique

Le modèle de corrélations multiples utilisé au cours de cette étude révèle une importation de la matière particulaire, dont plus de 70% est sous forme minérale. Ces résultats confirment la tendance générale d'exhaussement du sol sur une partie des marais salés des côtes ouest de la France, qui implicitement constituent des bassins de sédimentation. Ainsi les taux de sédimentation varient entre 0.4-0.8 cm an $^{-1}$ dans le bassin du Fier d'Ars (Long, 1975) et atteignent jusqu'à 5 cm an^{-1} dans la Baie Saint Michel (Berger & Caline, 1991).

Une différence dans la dynamique des nutriments dissous apparaît entre les deux subsytèmes étudiés. Les flux à l'interface des marais endigués agissent comme puits de nutriments alors que l'interface avec l'océan est plutôt une source d'enrichissement pour les zones côtières adjacentes. Un gradient nord-sud établi par Pignon (1975) montre que les formations géologiques les plus récentes se situent au sud alors que les anciennes sont plus au nord. Ceci va dans le sens de la classification donnée par Dame & Gardner (1993) (Figure.11), statuant que les écosystèmes géologiquement récents sont écologiquement immatures et se comportent comme importateurs de matière dissoute et particulaire. Réciproquement, les systèmes matures sont exportateurs de matière (Figure.11). Néanmoins, la morphologie du système avec l'existance des marais endigués et surtout une faible ouverture sur la mer est susceptible d'engendrer de fortes rétentions d'eau (Persson & Hakanson, 1994) qui s'accentuent à l'intérieur du système. Cette hypothèse a été vérifiée par les bilans hydriques qui montrent un déficit à l'amont du système (0.013 Km^3 an^{-1}) alors qu'un excédent a été enregistré à l'interface avec la mer ouverte (-0.09. Km^3 an^{-1}).

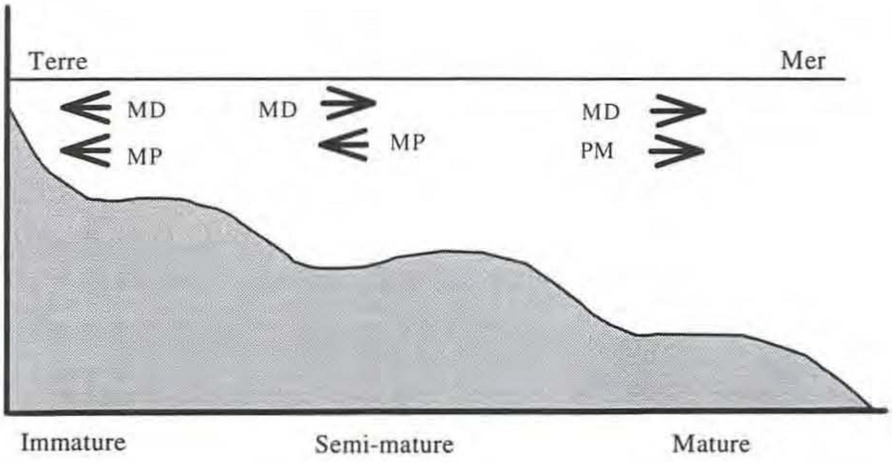


Figure 11 : Dynamiques des flux suivant le concept de la maturité géologique "Marsh-Estuarine Continuum" (Dame & Gardner, 1993).
MD= Matière Dissoute; PM=Matière Particulaire.

5- Fonctionnement des marais Européens par rapport aux écosystèmes Américains

L'importation de matière est particulièrement observée dans les marais européens alors que l'exportation est souvent la règle dans les systèmes américains (Tableau.6). Ces différences étaient souvent expliquées par la dynamique de la marée, dissymétrique en europe et par la différence dans la géomorphologie de chaque système (Dame & Allant, 1996). En effet une continuité entre le domaine intertidal et la bande côtière caractérise les systèmes américains rendant difficile la distinction entre le shorre et la slikke, contrairement aux systèmes européens, où très souvent l'endiguement provoque une rupture nette entre les deux compartiments modifiant ainsi les biocénoses et influençant les flux de matière.

Une différenciation basée sur l'origine de la matière organique et sa vitesse de décomposition a été proposée par Middelburg et al (1997): dans le système minéral (européen) la matière organique est adsorbée sur les particules minérales qui enveloppent aussi du phytoplancton et du microphytobenthos. Ces complexes organo-minéraux sont considérés comme une forme favorable à la sédimentation de la matière particulaire et aident, de ce fait, à immobiliser les substances riches en azote, tel le plancton, en minimisant les pertes dans les estuaires (Hedges & Keil, 1999). Ceci va à l'encontre du fonctionnement des marais américains où la production primaire est constituée surtout d'angiospermes, très riches en éléments ligneux réfractaires.

Tableau 6 : Synthèse des études de transport de matière dans les lagons côtiers américains et dans des estuaires européens. Pour les études effectuées en amérique les états, dans lesquels ces études ont été effectuées, sont aussi indiqués. Le transport est défini par un signe positif pour l'importation et négatif pour l'exportation.

Système	Localisation	Maturité	Particulaire	Dissous	Références
Bay of Fundy	USA, NS	Mature	-		Gordon & Cranford (1994)
Bedford Basin	USA, NS	Mature	-		Taguchi & Hargrave (1978)
Crommet Creek	USA, NH	Immature		+	Daly & Mathieson (1981)
Sippewissett	USA, MA	Mature	-	-	Valiela *et al.* (1978)
Flax Pond	USA, NY	Semi-mature	+	- +	Woodwell *et al.* (1978)
Gott's Marsh	USA, MD	Semi-mature	-	-	Heinle & Flemer (1976)
Ware Creek	USA, VA	Semi-mature	-	- +	Axelrad (1974)
Carter Creek	USA,VA	Semi-mature	-	- +	Axelrad (1974)
Bly Creek	USA, SC	Semi-mature	+	-	Dame et al (1991)
North Inlet	USA, SC	Mature	-	-	Dame *et al.* (1986)
Cumberland Island	USA,GA	Semi-mature	+ -	+ -	Childers (1994)
Sapelo Island	USA, GA	Mature	-	-	Chalmers *et al.* (1985)
Barataria Bay	USA, LA	Mature	-	-	Happ *et al.* (1983)
East Bay	USA, TX	Semi-mature	-	-	Borey *et al.* (1983)
Biscayne Bay	USA, FL	Mature	-		Roman *et al.* (1983)
Europe					
Himmerfjard	SW	Immature		+ -	Wilmot *et al.* (1985)
EMS-Dollar	NL	Semi-mature	+	-	Dankers *et al.* (1984)
Slufter	NL	Semi-mature	+	+ -	Asjes and Dankers (1994)
Stroodorpe	NL	Semi-mature	+		Wolff et al (1979)
Zwin	NL	Semi-mature	+		Hemminga *et al.* (1992)
Clone Point	UK	Semi-mature	+ -	+ -	Abd.Aziz & Nedwell (1986)
Tollesbury	UK	Mature	-	+ -	Boorman *et al.* (1994)
Mont St Michel	FR	Immature	+	+	Boorman *et al.* (1994)
Mira	PO	Semi-mature	+ -	+	Boorman *et al.* (1994)

V- LES APPORTS D'ORIGINE DIFFUSE

Outre l'influence côtière, qui est aussi sujette à des mélanges avec les eaux de ruissellement en provenance du bassin versant du pertuis Breton, la Baie du Fier D'Ars reçoit aussi des apports d'origine aquacoles, résultant des pratiques intensives ou extensives autour de la baie. Les apports atmosphériques (Paerl, 1997) et la diffusion des eaux souterraines (Moore, 1996) parfois chargés en nitrates et ammonium, peuvent aussi constituer des sources potentielles d'enrichissement dans les zones estuariennes. Déterminer la balance globale des apports exogènes transitant par la Baie du Fier d'Ars vers la bande littorale est l'un des objectifs de cette partie de l'étude.

La conceptualisation du modèle repose sur deux étapes fondamentales: la description de l'environnement en élaborant des cartes thématiques, et la représentation des processus hydrologiques en utilisant des équations mathématiques.

1- Elaboration de cartes thématiques

Les différentes étapes de genèse de cartes thématiques ainsi que les techniques utilisées sont schématisées (Figure.12) et détaillées dans le chapitre III.

La carte d'utilisation du sol (Figure.12), communément appelée plan d'occupation du sol (POS), est élaborée à partir de photographies aériennes (1/10 000), elle permet d'identifier les différentes activités pratiquées sur le marais et de déterminer les surfaces allouées à chacune d'elles.

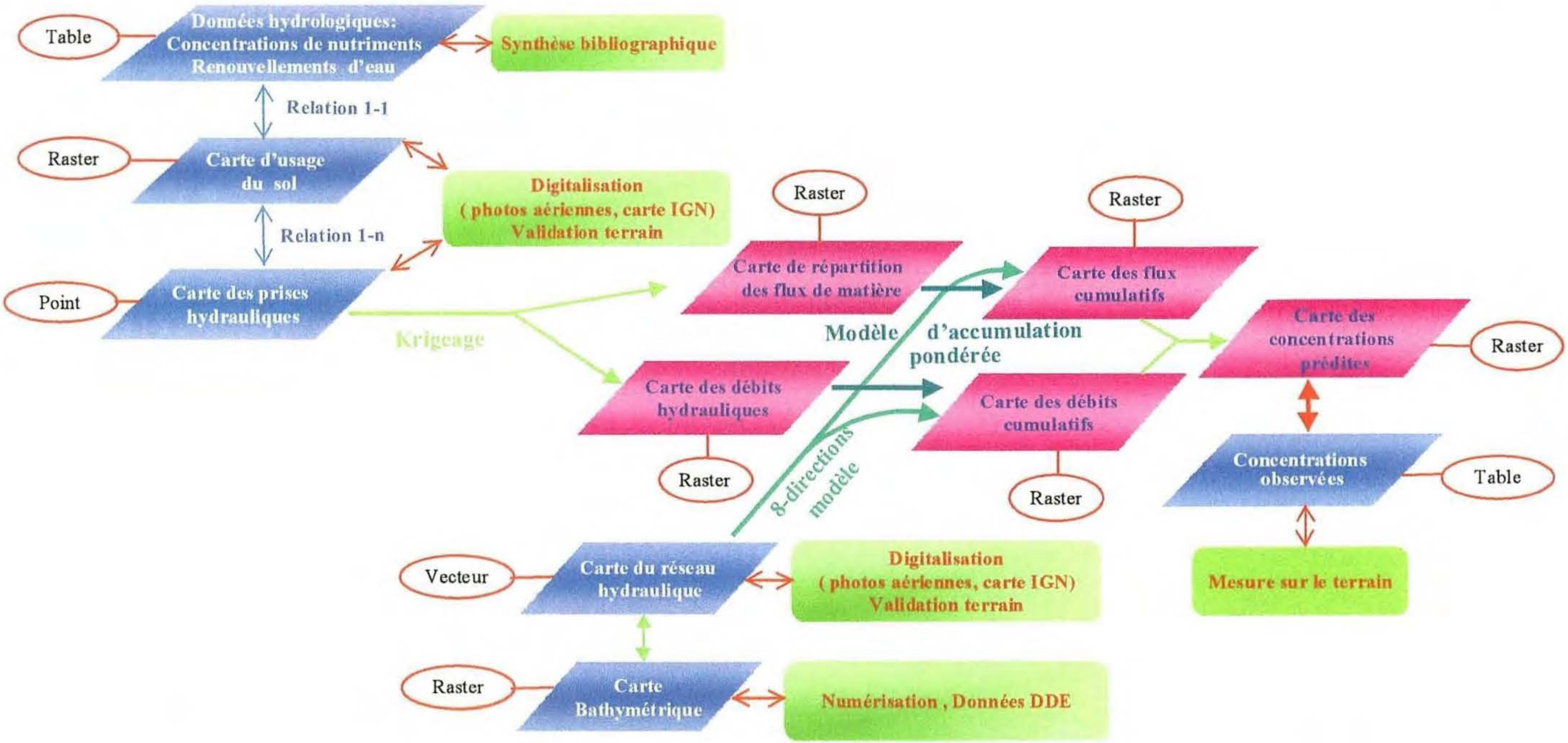


Figure 12 : Organisation des principales étapes suivies au cours de l'élaboration du modèle d'évaluation des flux aquacoles diffus. Les données rentrées (input data) sont représentées par des polygones bleus, celles élaborées (output data) sont représentées par des polygones roses. Les méthodes utilisées (polygones verts) et les types des objets (cercles rouges) sont aussi indiqués. Les relations entre les attributs (1-1 ou 1-n) sont également reportées.

Ainsi, la pisciculture extensive représente la plus large superficie exploitable (61%) suivie par la saliculture (22%), la conchyliculture (15%) et la pisciculture intensive(2%).

Les prises hydrauliques assurant les échanges entre les marais endigués et la zone intertidale ont été identifiées (Figure.12). Les subsytèmes, déterminant la zone de contribution de chaque prise hydraulique, sont déterminés en superposant la carte d'utilisation du sol avec celle du réseau hydraulique (Figure.12) et en attribuant un code à chaque polygone d'utilisation du sol correspondant à la prise se trouvant à l'aval du système. Cette technique, connue sous l'appellation "micro approche" (Ventura & Kim, 1993), permet d'agréger les charges en provenance de chaque polygone unitaire d'utilisation du sol vers le point d'évacuation des marais. Elle possède l'avantage de prendre en compte la large hétérogénéité des activités au sein de l'écosystème.

2- Représentation mathématique des processus hydrauliques

2.1- Au niveau des sources

La charge en nutriment est la variable à estimer à la sortie des marais endigués. Elle est obéit a la loi classique des flux de nutriment:

$$L_i = D_i.C_i$$

avec

- L_i flux journalier de nutriment
- D_i débit journalier d'eau
- C_i concentration de nutriment susceptible d'être générée par chaque type d'utilisation du sol.
- i fait référence au type d'activité pratiquée sur le marais.

Les variables D_i et C_i ont été extraites à partir de données bibliographiques. Pour les détails sur l'acquisition de C_i et D_i voir chapitre III section "water quality data".

En attribuant à chaque polygone d'utilisation du sol les valeurs appropriées de concentration et de débit journalier d'eau (Relation 1-1), des cartes de répartition des flux d'azote et des phosphates sont ainsi estimées au niveau des sources (marais). En utilisant la méthode de " micro approche" décrite ci-dessus, les flux seront agrégés au niveau des prises hydrauliques (Relation 1-n), qui constituent l'interface avec la zone intertidale.

2.2- Au niveau de la zone intertidale

2.2.1- Interpolation des données de flux de nutriments

Une approche géostatistique de la distribution spatiale des flux a été introduite pour étudier la structure spatiale des flux de nutriment à l'intérieur de la zone intertidale, en utilisant les flux estimés au niveau des prises hydrauliques comme sources connues. Un interpolateur spatial linéaire, le krigeage a été retenu pour cette opération.

$$Z^*_0 = \sum_{i=1}^{n} \alpha_i Z(x_i)$$

avec

• Z^*_0 un estimateur de la variable réelle Z_0.

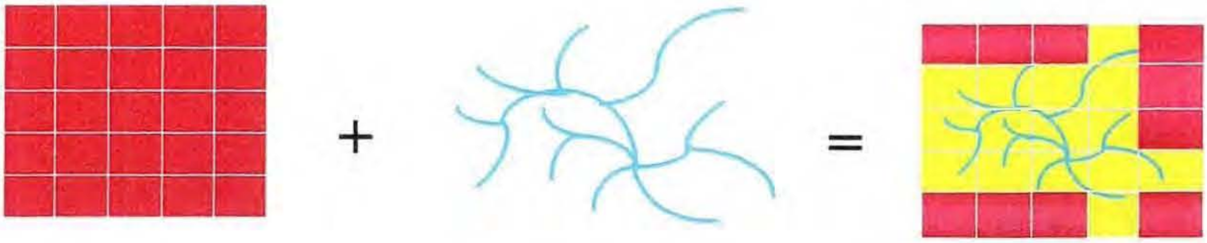
• α_i le poids attribué à la variable observée au niveau de la prise x_i, à l'intérieur de la Baie du Fier d'Ars.

Cette technique d'interpolation spatiale (Figure.12) présente en outre, un intérêt majeur qui réside dans l'utilisation de l'information fournie par la structure spatiale définie dans l'équation du semivariogramme (voir chapitre III section "Regional distribution of loading"), c'est aussi un interpolateur non biaisé qui permet à la fois de calculer l'erreur associée à l'estimation et de prendre en compte des distributions irrégulières d'échantillons (Chang & Sun, 1997; Daniellsson *et al.*, 1998).

2.3- Direction et accumulation des flux au niveau du réseau hydraulique

La distribution des flux de matière, élaborée par le krigeage, spatialise la donnée pour l'ensemble de la baie, sans tenir compte de la structure du réseau hydraulique. Celle ci évolue en suivant les gradients bathymétriques et influence de ce fait l'ensemble de la morphologie sédimentaire des zones intertidales (Crave, 1995). Pour palier à ce problème, une procédure spatiale ***"Burning In the stream"*** a été appliquée (Figure.13) (Saunder & Maidment, 1996). Elle consiste à attribuer les valeurs bathymétriques aux grilles correspondant au réseau hydraulique et à ajouter une forte valeur arbitraire aux cellules se situant en dehors du réseau. En procédant de la sorte, on "surélève" les cellules ne correspondant pas au réseau hydraulique, et on oblige les flux à circuler à l'intérieur du réseau une fois qu'ils y sont. En d'autres termes, cette procédure équivaut à approfondir le réseau hydraulique pour véhiculer toute l'eau à l'intérieur.

Afin de diriger les débits d'eau et les flux de nutriments à l'intérieur du réseau hydraulique, un modèle d'écoulement de pente, (8-direction-point modèle), proposé par Maidment (1993), permet de déterminer la direction du flux en fonction des données topographiques du milieu (Figure.14, étape.1).



Bathymétrie **Réseau hydraulique** **"Burning in streams" = Réseau hydraulique érodé**

• Attribuer les valeurs bathymétriques aux grilles correspondantes au réseau hydraulique.

• Surélever les cellules ne correspondant pas au réseau d'une forte valeur arbitraire.

•Produire un réseau hydraulique érodé "Burning In"= réseau cartographié.

Figure 13 : Les étapes explicatives de la procédure "Burning In the stream".

Le modèle est basé sur le concept suivant: chaque grille bathymétrique à huit directions possibles pour diriger le flux (Figure.14, étape.2) et c'est uniquement la direction de la plus grande pente qui sera retenue (Figure.14, étape.3). En dénombrant le nombre de grille se trouvant à l'amont de chaque grille, on arrive à retracer les zones d'accumulation des flux (Figure.14, étape.4).

Une approche améliorée de ce modèle, développée sous la forme d'un algorithme avec le langage Avenue du logiciel Arcview (annexe 1) consiste à dénombrer, non pas le nombre des grilles, mais plutôt leurs valeurs, correspondantes aux valeurs des flux de nutriment et de débits d'eau (Figure.14, étape.5). Les flux cumulatifs (Figure.14, étape.6), en provenance de l'amont du réseau, sont ainsi déterminés au niveau de chacune des grilles constituant le réseau hydraulique.

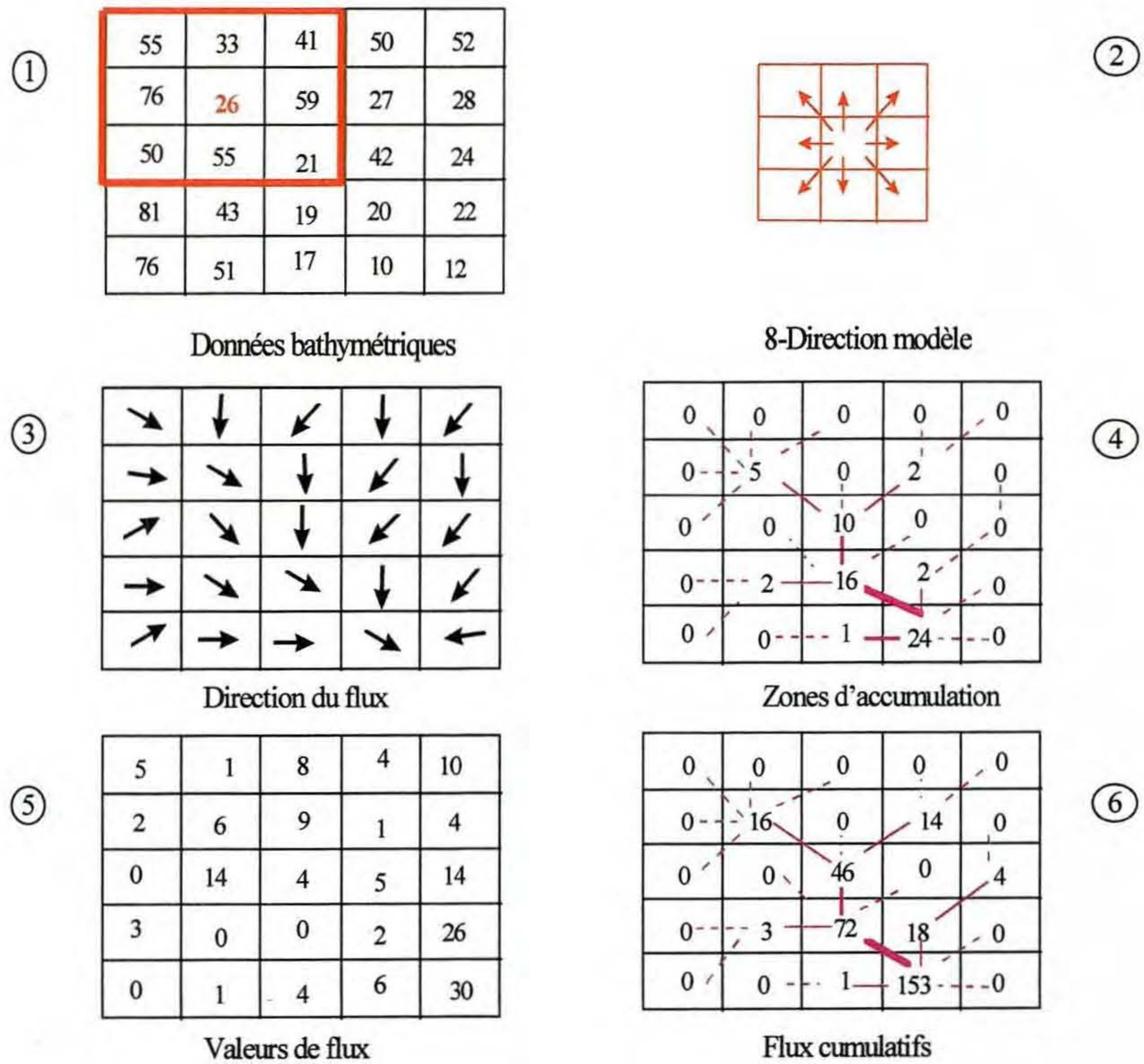


Figure 14 : Enchaînement du 8-direction-modèle et ses dérivées, modèle d'accumulation (4) et le modèle d'accumulation pondérée (6).

2.4- Prédiction des concentrations de l'azote inorganique dissous et des orthophosphates

2.4.1- Formulation mathématique

Les concentrations prédites (Figure.12) à différents points du réseau hydraulique sont la résultante des flux de nutriment se produisant à l'amont de chacune des grilles. Ce processus de mélange est approché en divisant les flux de matière par les débits d'eau cumulés dans chaque cellule du réseau.

$$C_j = L_j / D_j$$

avec

- C_j La concentration prédite au niveau de la j ^{ème} grille du réseau hydraulique.
- L_j Le flux cumulé au niveau de la j ^{ème} grille du réseau hydraulique.
- D_j Le débit d'eau cumulé au niveau de la j ^{ème} grille du réseau hydraulique.

2.4.2- Résultat

Les concentrations prédites de DIN montrent une décroissance de l'amont du système, où des valeurs de 30 μM ont été estimées, vers l'aval de la baie, à l'interface avec la mer ouverte. Les concentrations fluctuent à ce niveau entre 8 et 10 μM (Figure.15). Les phosphates montrent la même tendance que l'azote avec des valeurs entre 2 et 7 μM à l'amont, alors qu'une décroissance de l'ordre de 87% est notée entre les maxima de concentrations et les minima prédits à la sortie de la baie.

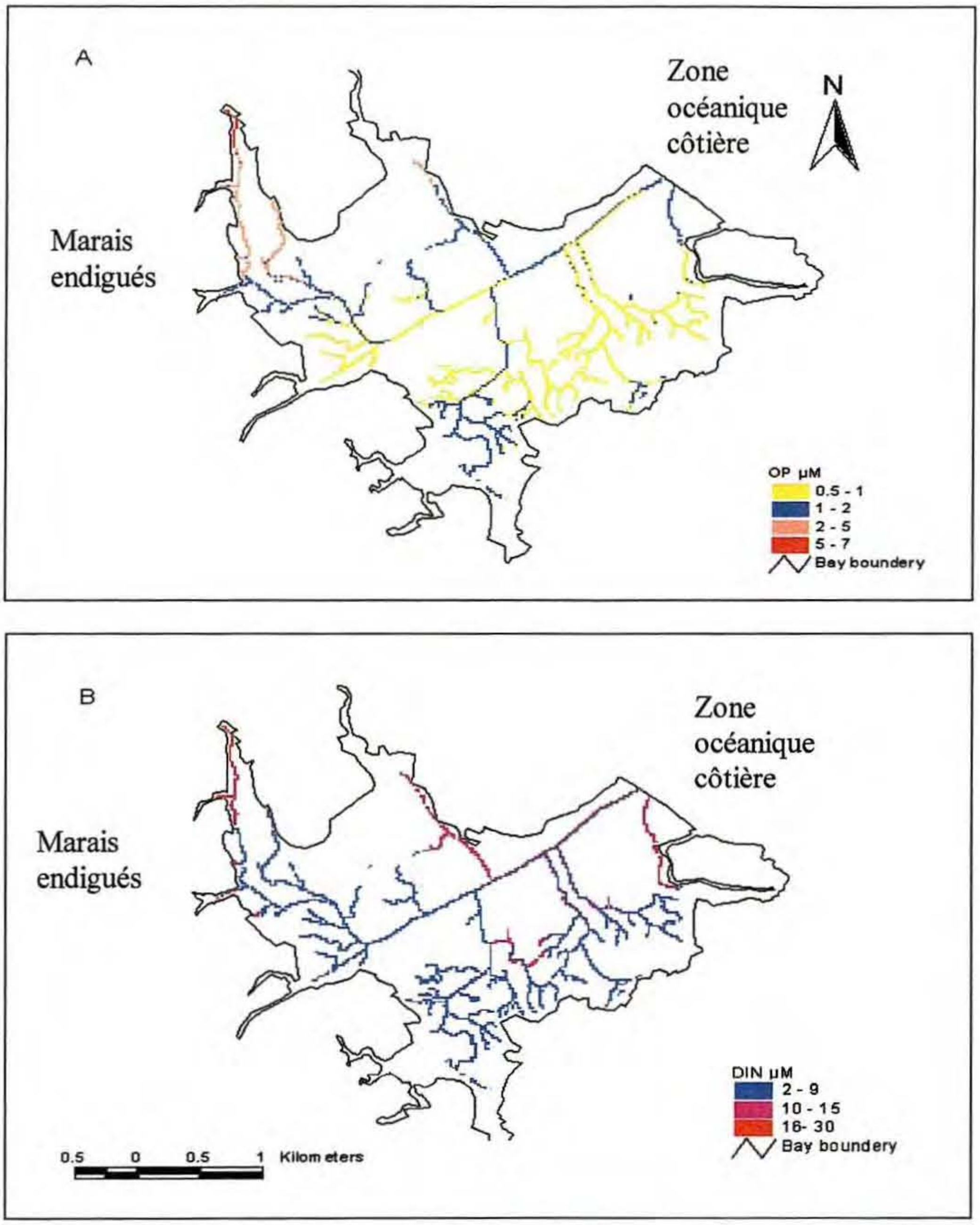


Figure 15 : Concentrations prédites des phosphates (A) et de l'azote inorganique dissous (B), estimées au niveau du réseau hydraulique. Les nuances de couleur indiquent des niveaux de concentrations.

3- Validation

3.1- Acquisition de données nécessaires à la validation

Un suivi bimensuel des flux d'azote et des phosphates au niveau de deux transects bathymétriques décrits dans la section III a été effectué au cours de l'année 1997-1998. Les flux instantanés sont enregistrés toutes les 30 mn au cours du jusant. Une concentration résultante est déterminée pour chaque cycle de marée comme étant la somme des masses de constituants sur la somme des volumes instantanés.

$$C_r = \frac{\sum_{t_0} c_t v_t}{\sum_{t_0} v_t}$$

avec

• c_t et v_t concentrations et volumes instantanés

• C_r concentration résultante pour un cycle de marée.

3.2- Résultats

La comparaison entre les valeurs prédites et celles observées (Figure.16), révèle que les observations des phosphates fluctuent fréquemment en dessous de 10 µM au niveau du premier transect (Transect 1), ce qui correspond aux valeurs prédites par le modèle. Pour L'azote les observations sont entre 20-30 µM alors que les prédictions fluctuent entre un peu moins de 20 µM, et jusqu'à 30 µM.

Au niveau du deuxième transect (Transect 2), les observations les plus fréquentes des phosphates varient entre 0.5 et 1.5 µM alors que les prédictions sont presque exclusivement autour de 1 µM. Les observations les plus fréquentes d'azote sont généralement comprises entre 10 et 20 µM alors que les prédictions fluctuent entre 10 et 15 µM.

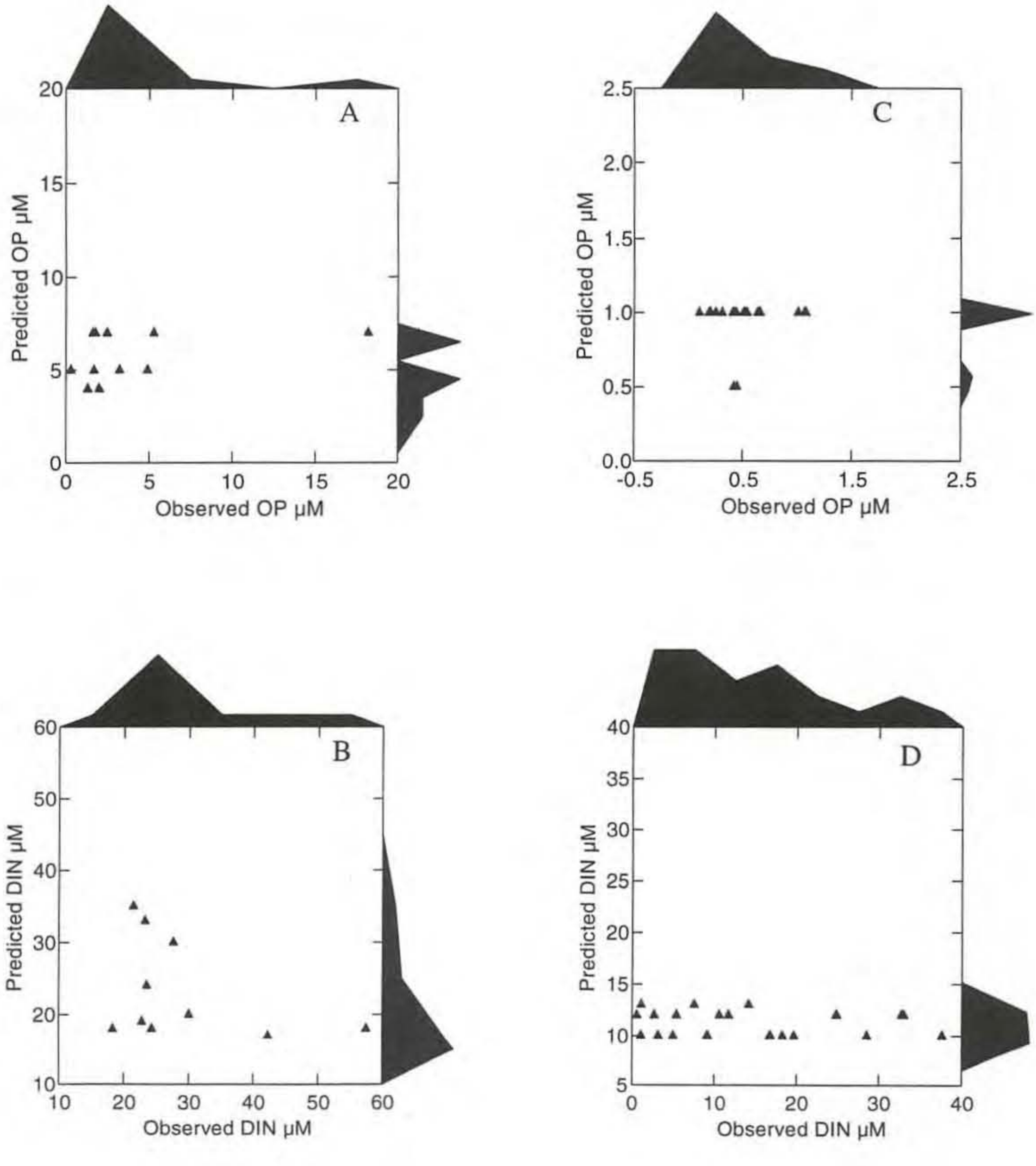


Figure 16 : Dispersions des concentrations prédites vs observées au niveau site 1 (A and B) et site 2 (C and D). Les polygones de fréquences de chaque concentration sont indiqués en face de chaque axe. Les concentrations prédites sont extraites des cellules correspondantes au segment échantillonné du réseau hydraulique.

4- Discussion

Bien que le modèle de prédiction de concentration de nutriment dissous puisse assez fidèlement reproduire les valeurs observées au niveau de la Baie du Fier d'Ars, une légère déviation est notée particulièrement à l'aval du système. Les sources d'erreur susceptibles d'interférer avec les résultats sont en premier lieu inhérentes aux données géographiques et spatiales (Burrough, 1986; Goodchild, 1993), à savoir des erreurs de cartographie, de spatialisation et de combinaison d'attributs à partir de différentes cartes thématiques (MacDougall, 1975). D'autres sources d'enrichissement externes telle que la déposition atmosphérique (Pearl, 1997) ou les apports souterrains diffus (Moore, 1996) pourraient aussi s'ajouter aux apports en provenance de l'aquaculture. Néanmoins, la quantification des apports atmosphériques au niveau du site étudié, au cours des 10 dernières années, montre qu'ils représentent moins de 12% des flux azotés annuellement fourni par l'aquaculture, cette proportion ne représentant que 4% en 1997. Les apports souterrains, particulièrement difficiles à quantifier (Sielgel, 1983; Shedlock *et al.*, 1993) ne sont pas suspects d'une large contribution aux apports azotés dans la baie. En effet, des mesures effectuées dans le bassin d'Arcachon, nettement plus urbanisé et supportant une plus grande pression agricole, montrent que ces apports ne dépassent pas 1% des flux externes d'azote inorganique dissous annuellement absorbés par le bassin (Rimmelin *et al.*, 1998).

Les processus internes tels que la dénitrification ou la fixation de l'azote peuvent diminuer la charge des apports externes et réduisent de ce fait les quantités exportées vers les zones côtières (Seitzinger, 1990). Des mesures effectuées dans la Baie de Marennes Oléron, présentant des caractéristiques sédimentaires et biologiques proches de celles de la Baie du Fiers d'Ars permettent d'estimer un taux annuel de dénitrification de l'ordre de 18.5 μmol m^{-2}

h^{-1}(Feuillet-Gérard, communication personnelle), à l'origine d'une réduction de seulement 13.2% des flux azotés.

5- Etude de sensibilité et applications pratiques du modèle

Les résultats présentés ci dessus, constituent un modèle stationnaire pour l'estimation des sources diffuses de nutriments dissous, particulièrement l'azote et les phosphates. Tel qu'il a été conçu, le modèle est conservatif, les processus se produisant sans perte ni retard. Pour vérifier cette hypothèse de conservativité, des variations de 10%, 25% et 50% des flux au niveau des sources ont été simulées. Un rapport, Ka, (Tableau.7) établi entre le flux cumulatif initial et celui après variations (±10%, ±25% et ± 50%) montre que cette hypothèse est bien vérifiée. En effet, Ka est sensiblement proche des taux de variations appliqués au niveau des sources de rejet. Ceci se vérifie aussi bien pour l'azote que pour les phosphates avec une légère déviation pour ces derniers (Tableau.7).

On a calculé deux rapports à partir de la différence entre concentration initiale et concentration finale après les variations (±10%, ±25% et ± 50) sur la concentration initiale. Le premier rapport, K_b, est basé sur des fluctuations des quantités drainées, le second, Kc, sur les fluctuations des concentrations.

Pour un égal taux de variation, Kc réagit plus intensivement que Kb, induisant une diminution plus significative des concentrations prédites. Ces différences s'accentuent surtout pour le plus fort pourcentage de variation (50%).

Tableau 7 : Sensibilité de DIN et PO_4 à des variations de flux de nutriments, $K_a = (L_{j\ nouveau} / L_{j\ initial})$ où L_j est le flux cumulatif dans la j^{th} cellule, à des variations du drainage et des concentrations sources, respectivement K_b et K_c, $K_b = [(C_{j\ nouveau} - C_{j\ initial}) / C_{j\ initial}]$, C_j est la concentration prédite dans la j^{th} cellule. K_b et K_c possèdent la même expression. AM indique la moyenne arithmétique et SD la déviation standard.

	DIN						PO_4					
	Ka		Kb		Kc		Ka		Kb		Kc	
	AM	*SD*	*AM*	*SD*	*AM*	*SD*	*AM*	*SD*	*AM*	*SD*	*AM*	*SD*
+10%	10	8,28	-11	6,26	8,8	19,33	7,36	9,62	-12,92	21,99	20,49	40
-10%	-10,4	6,37	10,23	7,13	-11,9	15,58	-12,8	8,13	32,99	45,88	-16,38	24,49
+25%	25,06	9,4	-21,87	5,55	24,13	21,85	21,48	6,55	-19,13	23,67	43,43	46,96
-25%	-25,25	5,71	33,57	8,75	-28,38	12,84	-27,41	3,64	57,39	44,45	-25,95	28,37
+50%	50,04	11,22	-36,09	4,7	49,94	25,89	46	7,7	-26,68	58	72,4	40
-50%	-50,3	3,67	103	12,2	-91,45	14,1	51,8	2,59	154	56,05	-83,08	23

Les tests de sensibilité impliquent qu'une gestion des concentrations au niveau des sources de rejet est plus efficace que la gestion des quantités d'eaux drainées. Le schéma ainsi proposé semble difficile à réaliser ou à intégrer dans un plan d'aménagement, car il suppose une gestion individuelle des concentrations produites de la part des différents exploitants des marais. Or le terme 'concentration' est souvent une notion abstraite pour la plupart d'entre eux. Pour diminuer les charges en nutriments dissous vers la bande côtière, l'installation de structures d'épuration est une autre solution, mais ces procédures sont souvent très coûteuses et lourdes à gérer. Par conséquent, la gestion des quantités drainées, quoique moins efficace, semble plus pratique et plus adaptable à une réalité du terrain.

VI- DISCUSSIONS GÉNÉRALES ET PERSPECTIVES

Le Fiers d'Ars est à la fois sous l'influence des apports en provenance du pertuis Breton, durant la période automne -hiver, et des apports en continu des marais endigués. Le système reçoit par conséquent des concentrations élevées en azote ~70 μM alors que celles des phosphates sont autour 0.5 μM. L'aspect le plus caractéristique du site d'étude est sa faible teneur en Chl a, 3.5 à 7 $\mu g l^{-1}$ respectivement au niveau du site 1 et site 2, témoignant d'une faible biomasse phytoplanctonique (Barillé, 1996).

La lumière, fréquemment considérée comme le facteur limitant de la biomasse phytoplanctonique sur les côtes atlantiques françaises (Héral, 1985; Barillé, 1996; Irigoien & Castel, 1997), semble n'intervenir dans la limitation qu'au cours de l'efflorescence printanière (Tableau.4). En effet, le phytoplancton printanier est suspecté d'être favorisé par les apports d'origine continentale, ainsi que le témoigne la corrélation négative entre la Chlorophylle et la salinité. Une fois qu'il atteint la zone intertidale, le phytoplancton serait l'objet de changements rapides des conditions de luminosité dues au fort mélange vertical. Ces fluctuations étant si rapides que les cellules phytoplanctoniques ne peuvent ajuster leur physiologie en fonction de ces changements (Demers, 1986), ceci induirait une baisse de la capacité photosynthétique due à une réduction des temps de résidence des cellules algales dans les zones euphotiques (Demers & Legendre, 1981).

Une deuxième efflorescence algale survient durant la période automnale où des remises en suspension du microphytobenthos ont été démontrées (Barillé, 1996; Guarini,1998; Sentos, 1997). L'analyse des comptages de microphytes dans le pertuis Breton montre, effectivement que le microphytobenthos représente plus d'un tiers des cellules algales phytoplanctoniques et que cette fraction devient dominante en automne (Barillé, 1996).

L'étude des facteurs responsables de la remise en suspension du microphytobenthos montre que le sédiment est érodé par les courants du flot. La resupension dépend donc de l'amplitude de la marée et par conséquent des surfaces érodées. De plus, le transport du microphytobenthos est accru par le déplacement engendré par les vents d'ouest et du nord-ouest, qui constituent les directions dominantes (Guarini,1998) (annexe 2). Durant cette période, la production phytoplanctonique semble régulée par le broutage zooplanctonique, suggérant ainsi que la production et la consommation du phytoplancton sont étroitement couplées à travers la chaîne trophique. Ce couplage est surtout illustré par la diminution de l'azote inorganique dissous, ce dernier finissant par manquer dans le milieu.

Ces résultats suggèrent que l'étude de la capacité trophique de cet écosystème doit donc être réalisée à l'aide d'un modèle de production phytoplanctonique et microphytobentique, couplé à un modèle hydrodynamique en tenant compte de l'impact potentiel des apports des cours d'eau alimentant le pertuis Breton et aussi des apports aquacoles qui alimentent l'amont du système.

Le pertuis Breton, qui assure en partie l'alimentation du Fier d'Ars, possède peu d'échanges avec l'océan. Un modèle hydrodynamique élaboré par Lazure (DEL- IFREMER Brest) indique des temps de résidence des masses d'eau relativement longs, 200 jours sont nécessaires pour renouveler 90% de la partie continentale du bassin (Barillé, 1996).

Le Fier d'Ars, bien que situé vers la partie océanique et à cause de sa morphologie (une faible ouverture sur l'océan) serait sujet aussi à de longs temps de résidence des masses d'eau. Dans cet écosystème, les conditions de lumière et de température durant la période hivernale ne permettent pas la photosynthèse phytoplanctonique. Par ailleurs, les nutriments apportés par la masse d'eau marine et ceux produits par les marais endigués s'accumulent dans le système. Au printemps, la température s'élève au-dessus de 15°C et l'efflorescence algale peut alors se développer. Cette production, freinée par la réduction de la pénétration de la lumière au sein de la colonne d'eau, reste inférieure à 7 $\mu g l^{-1}$ de chlorophylle.

La production primaire planctonique étant limitée, elle n'utilise pas la totalité des sels nutritifs accumulés dans l'écosystème. On peut donc penser qu'une partie non négligeable de ceux-ci est utilisée par le microphytobenthos après une accumulation dans l'eau interstitielle du sédiment. Ces sels nutritifs peuvent également être exportés vers l'océan. La quantification de ces échanges a particulièrement fait l'objet du chapitre 2 de la thèse.

Une estimation annuelle des flux de nutriments dissous et de matière particulaire a été établie à l'aide de la méthode des corrélations multiples (Spurrier & Kjerfve, 1988). Pour les deux sub-systèmes étudiés, 28 cycles de marée ont été suivis. Le but des modèles de régression, quoique nécessitant une base de données assez conséquente, est de procurer une estimation la plus fiable du flux annuel. Cette méthode conduit à une très **faible erreur standard**, comparée à la méthode classique de la moyenne des cycles observés extrapolée au nombre total des cycles sur l'année. Ainsi, **la variabilité est expliquée par la régression, ce qui permet de déterminer la direction du flux net avec un intervalle de confiance bien ajusté du flux annuel net** (Dame *et al.*, 1991). Outre ces avantages, la variance associée à l'estimation annuelle tient compte de la plus large variabilité se produisant aussi bien à l'intérieur d'un même cycle de marée que de la variabilité intercycle.

L'estimation a été basée sur une valeur constante des surfaces inondables. Or on sait que la surface varie au cours du cycle tidal et dépend étroitement de l'amplitude de la marée. Pour une meilleure estimation, **un modèle hydrodynamique de la marée devrait être couplé à une relation hypsométrique (aire inondée vs hauteur d'eau) afin de déterminer une valeur instantanée de l'aire inondable** (Childers *et al.*, 1993).

Au cours de cette étude 10 variables prédictives ont été retenues. Les corrélations obtenues sont satisfaisantes, indiquant des estimations fiables des flux instantanés et un bon déroulement de l'échantillonnage. On est arrivé, entre autre, à expliquer **plus de 75% de la variabilité des flux des nitrates et nitrites avec seulement 4 variables prédictives** (Tableau.5). Lors d'études similaires, jusqu'à 20 descripteurs ont été utilisés sans jamais dépasser un coefficient de détermination supérieur à 67% (Dame et al,. 1989, Dame et al,. 1991). Le modèle de régression linéaire a montré des relations significatives, essentiellement pour les éléments azotés, pour le carbone organique dissous et pour la matière particulaire, avec ses deux fractions minérale et organique (Tableau. 5). Les variables peu concentrées dans le milieu, telles que la Chlorophylle ou les phosphates, montrent des corrélations non significatives. D'autres descripteurs pourraient être intégrés dans le modèle afin de l'améliorer, à savoir, des mesures en continu de salinité, du pH ou encore des variables biologiques, à l'instar de la couverture végétale et des dénombrements algals ou bactériens.

A l'interface avec le marais endigué, le système se comporte comme un puits de nutriments dissous et de matière particulaire, alors qu'à l'interface avec la mer ouverte c'est plutôt une source de matière dissoute et un puits pour le particulaire (Figure.10). Bien que les marais endigués soient eux-même une source potentielle d'azote sous sa forme ammoniacale, comme en témoignent les différences de concentrations entre le flot et le jusant, le bilan annuel penche vers l'importation. La topographie, le régime de la marée dominant au cours du flot, suggère une forte rétention d'eau à ce niveau. Cette hypothèse a été vérifiée par les bilans hydrologiques. Pour appuyer encore plus ces conclusions, un modèle hydrodynamique (MARS) approprié au système (bathymétrie très précise) est en cours d'adaptation par J.C Salomon (DEL-IFREMER Brest) afin de déterminer précisément les temps de résidence des masses d'eau en chaque point du système.

La Baie du Fier d'Ars produit plus d'azote inorganique dissous qu'elle ne peut en consommer. L'excédent (-9.03 g $m^{-2}an^{-1}$) est donc exporté vers la zone océanique connexe. La quantité exportée représente le quart de ce que peut produire la Baie du mont St Michel (Boorman *et al.*, 1994). La même étude a démontré que la Baie du mont St Michel produit légèrement moins de carbone organique dissous (-74 mg $m^{-2}an^{-1}$) que ce que peut exporter le Fier (-87 mg m^{-2} an^{-1}). Sachant que cet écosystème n'est pas sous l'influence des apports détritiques continentaux (Riera & Richard, 1997), source majeure de carbone organique dissous (Morris *et al.*, 1995; Shindler *et al.*, 1996 a,b), on peut remarquer que l'exportation de ce paramètre au niveau du Fier d'Ars est relativement élevé par rapport à d'autres écosystèmes européens (Boorman *et al.*, 1994). Dans les marais maritimes, le carbone organique dissous provient essentiellement de la lyse des cellules algales (Crottereau, 1999) ce qui conduit à conclure que **le phytoplancton et le phytobenthos font l'objet d'un fort taux de broutage par la méiofaune. Des perturbations physiques par le fort mélange vertical, ou chimique par le biais des stress halins, peuvent également contribuer à la lyse des cellules algales et expliqueraient les faibles concentrations de chlorophylle enregistrées dans le milieu.**

L'importation de la matière particulaire caractérise les deux sub-sybstèmes étudiés. Une sédimentation progressive de la baie a été mise en évidence par Long (1975). Cet auteur a rapporté des taux de sédimentation moyen de l'ordre de 0.5 cm an^{-1} pour l'ensemble de la baie avec une tendance à l'accroissement vers les marais endigués (~0.8 cm an^{-1}). **Plus de 70% de la matière particulaire importée est sous forme minérale**. Cette proportion est approchée car le modèle produit une somme de la partie minérale et organique qui est à peu prés de 20% inférieure à la matière totale en suspension estimée par le modèle. Ceci est dû au fait que chaque variable est déterminée indépendamment à partir des variables prédictives différentes (Tableau.5). De plus, la méthode de la perte au feu, utilisée pour déterminer la proportion de la matière organique dans la matière particulaire totale, n'est pas très précise car il existe le risque de surestimation des valeurs, dû au fait qu'au delà d'une certaine température, quelques minéraux comme les carbonates et les argiles sont détruis.

Un modèle stationnaire a été développé afin de quantifier les apports aquacoles en provenance des marais endigués, de les cumuler au niveau des points de rejet et d'étudier leur répartition spatiale au sein de la zone intertidale. L'analyse a été basée essentiellement sur le modèle spatial de la topographie et de l'utilisation du sol.

Les charges aquacoles diffuses à travers les marais endigués, qualifiées de "Nonpoint source loading", ont été estimées à **132 T N an^{-1}** au niveau des exutoires des marais ou prises hydrauliques. Les apports atmosphériques azotés ne représentent que de faibles proportions, entre 4%-12%, comparé à de ce que peut produire l'aquaculture. Ces résultats confirment la tendance générale mondiale selon laquelle les charges diffuses sont les sources majeures d'enrichissement pour la bande côtière (Correll *et al.*, 1992 ; Poiani & Bedford, 1996; Lowrance *et al.*, 1998).

Le modèle développé présente **une logistique facilement adaptable à d'autres écosystèmes**. Une application sur un milieu plus complexe avec diverses occupations de l'espace, allant des forêts aux zones agricoles ou urbanisées, serait d'un intérêt majeur et consisterait un outil solide pour la gestion de la bande côtière. Une quantification des flux d'azote produit par différents types d'utilisation de l'espace (végétation naturelle, agriculture, zones résidentielles...) effectuée dans la Baie de Waqoit (Valiela *et al.*,1997), a constitué un travail précurseur de ce type de modèle dans les milieux estuariens. Ce modèle estime les pertes d'azote dans les différents compartiments de l'écosystème afin d'établir un bilan global au niveau de la baie. Aucune considération spatiale n'a été prise en compte dans l'établissement de ce modèle. Les bilans fournis, bien que tenant compte d'une large hétérogénéité des sources d'azote, sont des valeurs globales pour la baie et ne permettent pas d'identifier des zones critiques à l'intérieur du système.

Par ailleurs, le modèle développé au cours de cette étude est conservateur, ni retard ni dégradation ne sont susceptibles d'altérer les flux de nutriment. Cependant, divers processus peuvent interférer: les flux souterrains diffus (Sielgel, 1983; Shedlock *et al.*, 1993; Moore, 1996), les processus internes tels que dénitrification-fixation d'azote (Joye and Paerl 1994; Capone *et al.* 1997; Riou 1999), et la déposition atmosphérique (Paerl, 1997; Rimmelin *et al.*, 1998).

Pour intégrer ces différents processus, il est possible d'envisager de futurs travaux. Les perspectives du modèle seront améliorées avec une segmentation de la baie en différentes unités hydrologiques, matérialisées par des polygones adjacents correspondants à des unités topographiquement distinctes. Ceci sera accompagné par une estimation spatialisée de l'ensemble des processus internes au niveau de la baie.

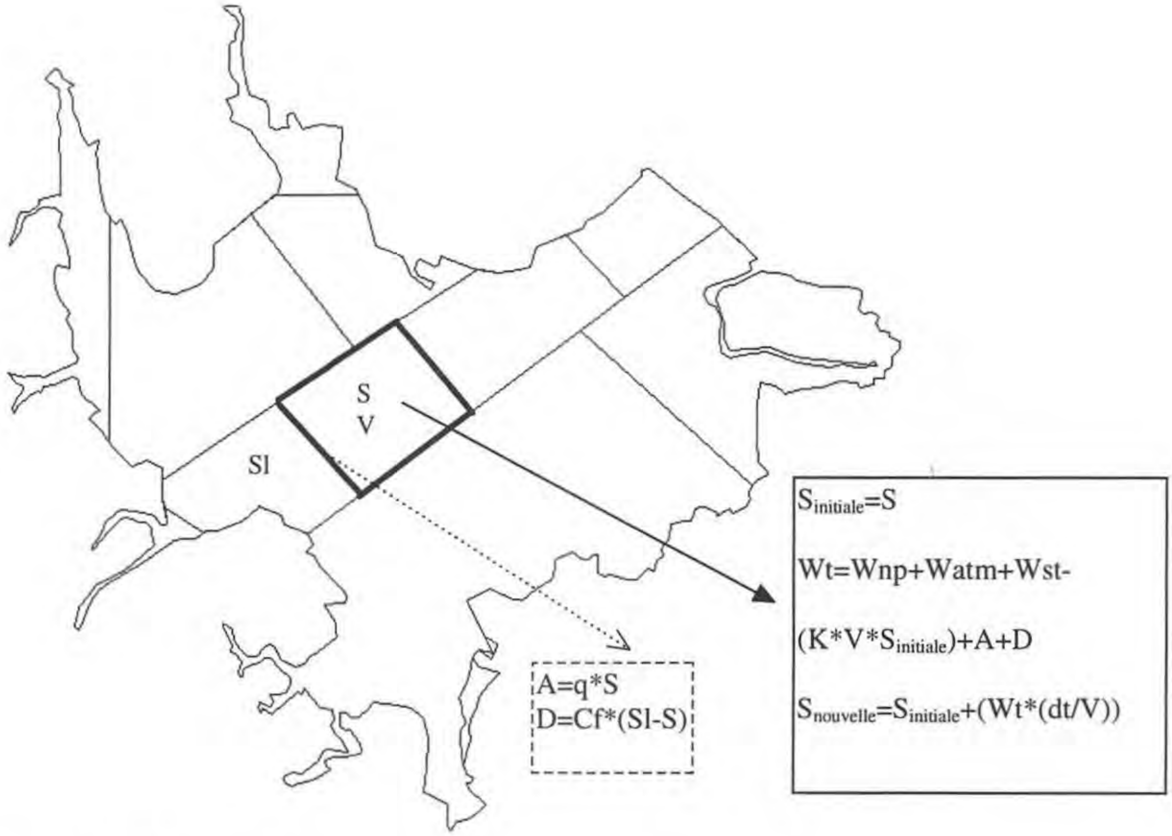
Un bilan du système pourrait être calculé au niveau de chaque polygone hydrologique comme étant la somme des flux aquacoles diffus, déjà estimés au cours de ce travail, et des sources internes au système provenant des processus ci-dessus discutés. Quoique considérés comme étant sans influence majeure sur le bilan global de la baie, ces processus pourraient avoir localement un poids non négligeable.

Les échanges entre les différentes unités ou polygones du modèle pourraient être envisagés à l'aide d'un modèle de transport d'advection-dispersion. Une conceptualisation de ce modèle est schématisée à la figure (17). Pour chaque unité hydrologique, matérialisée par le polygone en gras, la charge totale serait la somme des flux d'origine diffuse et des flux localisés. Les échanges entre les boites se géreraient avec les lignes de séparation, à travers lesquelles seraient déterminés les flux diffusifs et advectifs. La sortie du modèle serait les concentrations d'équilibre du système. Ces concentrations représentent les niveaux de valeurs atteints une fois que toutes les composantes du système sont équilibrées entre elles.

Ces concentrations seuils constitueraient une référence à un état donné, il suffirait de simuler un changement de scénario dans la répartition de la carte d'utilisation du sol, pour en déduire immédiatement les conséquences sur les niveaux de concentration.

Ce modèle ainsi conceptualisé, permettrait d'améliorer mon travail en terme de précision, en intégrant le maximum de processus susceptibles d'interagir. De par sa capacité à spatialiser les processus, ce modèle permettrait également de mettre en évidence ceux qui localement contribueraient à accroître les apports et ceux qui seraient les plus pertinents pour les réduire.

Ce modèle serait un nouveau concept pour la compréhension et la simulation du fonctionnement d'un écosystème. Ces propriétés intégratrices étant un atout majeur pour la représentativité de la complexité du système.



S= concentration en nutriment
q= flux
dt= pas de temps
Cf= Coefficient de dispersion
Sl= Concentration dans le polygone amont
V= volume du polygone
K= Coefficient du retard
A= flux advectif
D= flux diffusif
Wt= charge totale
Wnp= charge non-localisée (déjà estimée)
Watm= déposition atmosphérique
Wst= charge du sédiment

Figure 17 : Possibilité de conceptualisation de la balance cinétique du système en tenant compte des sources diffuses (déjà estimées) et d'autres sources internes au système.

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ANNEXES

ANNEXE 1

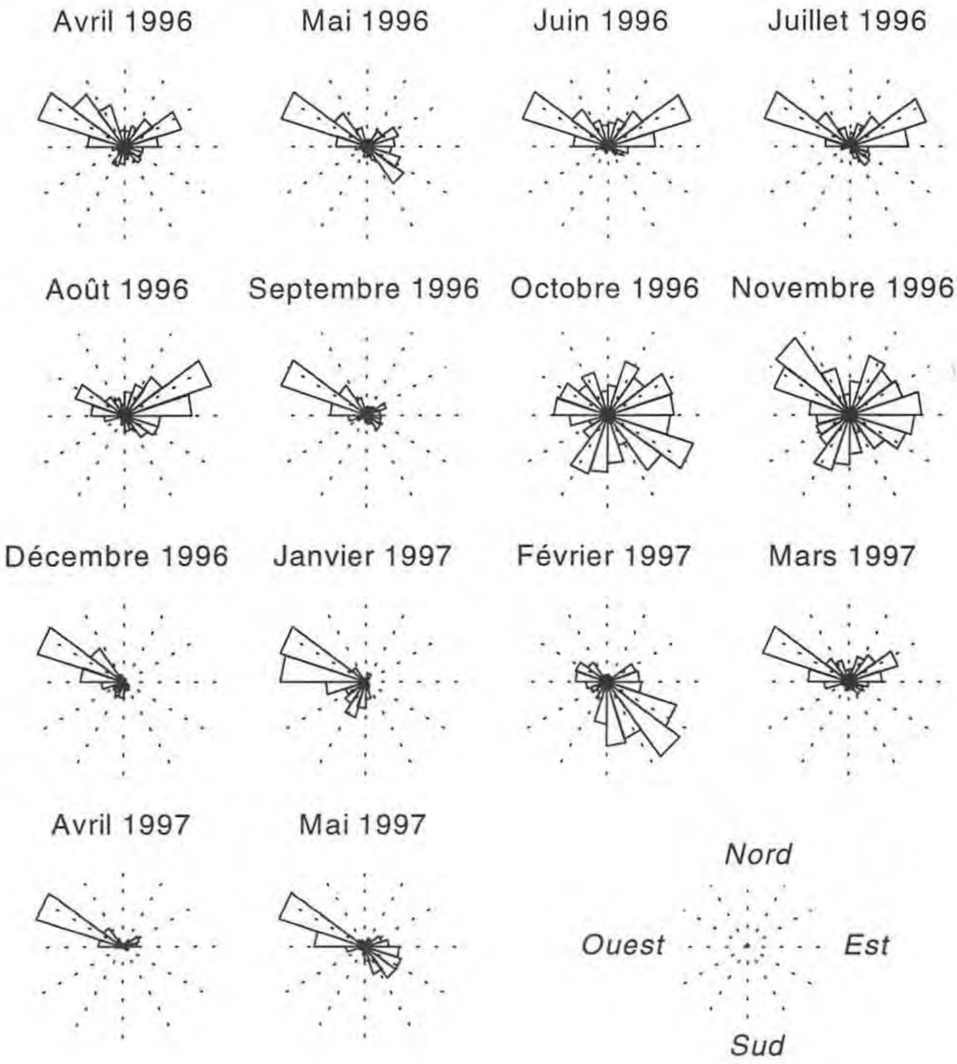
Algorithme développé en langage Avenue pour effectuer le modèle d'accumulation pondérée (weighted accumulation model)

```
'--------------------------
'--- Purpose/Description ---
'---------------------------
'
'Computes the weighted flow accumulation using the flow direction grid
'and the land surface loading grid.  The resulting grid produces
'a grid (wfacload) with the loadings to the bay in Kg/day.
'
'----------------
'--- Get view ---
'----------------
'
theview = av.getactivedoc
'
'------------------
'--- Get themes ---
'------------------
'
thethemes = theview.getthemes
if (thethemes.count = 0) then
  msgbox.error("No themes found", "WFAC GRID")
  exit
end
if (thethemes.count = 1) then
  msgbox.error("Only one theme found", "WFAC GRID")
  exit
end
thegthemes = list.make
for each thetheme in thethemes
  if (thetheme.getclass.getclassname = "gtheme") then
    thegthemes.add(thetheme)
  end
end
if (thegthemes.count = 0) then
  msgbox.error("No grid themes found", "WFAC GRID")
  exit
end
if (thethemes.count = 1) then
  msgbox.error("Only grid one theme found", "WFAC GRID")
  exit
end
```

```
fdrtheme = msgbox.listasstring(thegthemes, "Flow direction theme", "WFAC GRID")
if (fdrtheme = nil) then
  exit
end
ldtheme = msgbox.listasstring(thegthemes, "Load theme", "WFAC GRID")
if (ldtheme = nil) then
  exit
end
'
'-----------------
'--- Calculate ---
'-----------------
'
fdrgrid = fdrtheme.getgrid
ldgrid = ldtheme.getgrid
'
cellsize = number.makenull
dummmy = grid.getanalysiscellsize(cellsize)
extent = rect.makenull
dummy = grid.getanalysisextent(extent)
'
facgrid = (fdrgrid.flowaccumulation(ldgrid)).int
facfilename = av.getproject.makefilename("facgrid", "")
facgrid.savedataset("/home/ferdi/quenzer/trialrun/wfacload1".asfilename)
facgtheme = gtheme.make(facgrid)
theview.addtheme(facgtheme)
facgtheme.setvisible(true)
'
'final message to user
'
message = "Accumulated load grid calculated."
msgbox.info(message,"WFAC GRID")
'
'-----------
'--- End ---
'-----------
```

ANNEXE 2

Statistiques de direction du vent (occurrence) entre avril 1996 et mai 1997 inclus, enregistrées par le CREMA d'après Guarini (1998).



CHAPITRE I

Temporal and spatial scale variations of light- and nutrient-limitation of phytoplankton biomass in a macrotidal bay : Fier d'Ars (SW France)

M. BEL HASSEN

UMR 10 CNRS-IFREMER, Centre de Recherche en Ecologie Marine et Aquaculture, B.P. 5,

F-17137 L'Houmeau, France.

Ce chapitre a été soumis pour publication dans Estuarine, Coastal and Shelf Science

soumis le 17 Janvier 2000

ABSTRACT

In the Fier D'Ars Bay, Chlorophyll a (Chl) concentrations were rather low throughout the year, below 7µg l^{-1} in the upper bay and 4 µgl^{-1} in the lower bay. The potential of light- and nutrient-limitation of phytoplankton biomass accumulation was examined based on the ecosystem level of nutrient concentrations and stoichiometric ratios, total suspended sediment concentrations (TSS) and incident photosynthetically active radiation (PAR). The relative fluctuations of nutrient concentrations, particularly orthophosphorus (PO_4), in proportion to dissolved inorganic nitrogen (DIN) and silicate (Si), were found to be the predominant factor regulating the phytoplankton standing crops in the upper bay. The nutrient stoichiometry (N:P>20 and Si:P>20) and the significant correlation between Chl and the Principal Compound Analysis (PCA) factor represented by N:P and Si:P ratios favor the hypothesis of P-limitation during autumn winter, whereas a light- limitation was suspected during spring summer.

A temporal transition was carried out in the lower bay from the spring summer light limitation, a period of high turbidity (TSS = 50 mgl^{-1}) and disappearance of relationships between Chl and TSS, to a summer autumn high grazing pressure, illustrated by the hard coupling between Chl and DIN, which was ultimately depleted. From this data, it appears that DIN was potentially limiting to phytoplankton biomass during the second phytoplankton rise.

These temporal and spatial variabilities showed a pattern of recurring variations within factors regulating phytoplankton standing crops, which incriminated both light and nutrient availability- as controlled by tidal mixing and suspended sediment concentrations for the former and the anthropogenic allochtonous input and in situ biogeochemical mineralization for the latter .

Key words

Phytoplankton, Chlorophyll a, Light limitation, Nutrient limitation.

INTRODUCTION

The coastal ecosystems (including estuaries, bays and tidal rivers) are the most intensively fertilized environments on Earth (Nixon *et al.*, 1986; Cloern *et al.*, 1995). In many cases, they were open to overenrichment due to both buoyant surface water nutrient input (freshwater) and to residence time of freshwater nutrient-rich loadings that was longer than primary production growth rate, consumption and nutrient regeneration (Malone *et al.*, 1996). Although, under nutrient enrichment conditions, phytoplankton response can be nutrient limited (Fisher *et al.*, 1992; Pennock & Sharp, 1994), coastal eutrophication on the other hand, can occur during nutrient balanced conditions (Justic *et al.*, 1995). Much controversy was associated to nutrient limiting phytoplankton growth and/ or biomass and overenrichement, since tolerance to nutrient enrichment has been showen to depend closely on physical conditions within the system such as tidal mixing and periodic stratification-destratification of the water column, hypothesized to impoverish the phytoplankton standing crop (Monbet, 1992).

In general, N is considered to be the major regulator of estuarine phytoplankton production (Boynton *et al.*, 1982). However, deviation of this general pattern was numerous, such as light limitation in high turbid estuaries (Heral, 1985; Irigoien & Castel, 1997). Temporal variations in factors limiting phytoplankton production has been once more documented (Fisher *et al.*, 1992; Malone *et al.*, 1996) and a transition between limiting factors in relation to variations in major physical and biochemical processes, such as physical flushing and benthic recycling was also noticed (Pennock & Sharp, 1994).

In this study, we focused on phytoplankton production in relation to the ambient nutrients and light availability in terms of time and bay-space scale. A particular interest was also assigned to the functional role of transport processes as a regulator of nutrient supply, as well as a factor of plankton biomass accumulation. Seasonal transitions between light, ambient nutrients and food chain coupling regulation of phytoplankton production were assessed, based on ecosystem-level analyses of physical processes, nutrient and suspended sediment concentrations, stoichomitric ratios and Chl concentrations.

MATERIALS AND METHODS

Sampling site

The Fier d'Ars Bay is situated centrally on the French Atlantic coast, to the north of Marennes-Oléron Bay and belongs to the Pertuis Breton macrotidal estuary (Fig. 1), which is the most important mussel production site on the French Atlantic coast and is ranked, with a production of 450 $g.C.m^{-2}.yr^{-}1$, among the most macrophytobenthic productive sites in the world (Barillé, 1996).

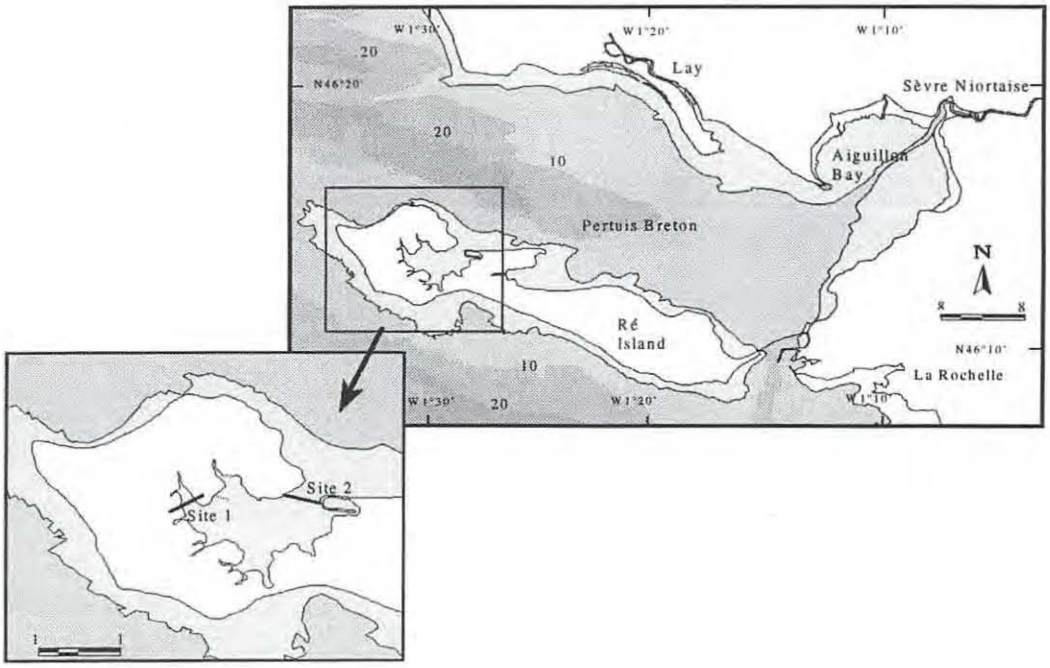


Figure 1 : Map showing the geographical position of the Fier D'Ars Bay and the location of the sampling stations. Lower site (site 1) is situated in the north of the bay. Salt marsh-mudflat, upper site (site 2) was just south-west of the lower station. The two *Sèvre Niortaise* and *Lay* rivers are also indicated. Shaded grey indicate different sea depths.

The Fier d'Ars Bay covers a total area of 750 ha of intertidal madflat, supporting a large area of oysters production and surrounded by 1258 ha of productive earthen ponds consisting of fish and/or oyster ponds. Besides aquacultural loading, water circulation within the Pertuis Breton estuary supplies the major terrestrial inputs into the bay through its north marine entrance. The two rivers, the *Sèvre Niortèse* and the *Lay*, draining respectively 2750 km^2 and 2000 km^2 of agricultural watersheds, are the dominant factors of inorganic water enrichment in the Pertuis Breton estuary and consequently in the Fier d'Ars Bay (Fig. 1). The flushing schedule has been identical for many years, particularly due to human control of the runoff stations. The mean freshwater residence time within the Pertuis Breton estuary is sufficiently long (~ 120 d) and is suspected to be longer in the Fier d'Ars Bay due to the semi-closed morphology of the bay.

The meteorological conditions exhibited a strong seasonality. The tide was semidiurnal, with a maximum range of about 5 m which led to general good mixing and high turbidity. Water velocity varied from 90 cm s^{-1} during spring tide to 50 cm s^{-1} during neap tide. The oxygen concentrations were close 6 mgl^{-1} during slack tide and reached 9 mgl^{-1} at maximum tide velocity.

Sampling strategy

The study was focused on two sampling stations. The upper bay, site 1, was in the junction between the upland and the extensive salt marshes (Fig. 1). It was surrounded by several earthen ponds, where many aquacultural activities were practiced, ranging from intensive to extensive aquaculture along with oyster cultures. This site also received little intermittent freshwater input during the wet season. The lower bay, site 2, marked the boundary of the bay with the coastal ocean. In addition to the coastal marine influence, this station received intensive freshwater inputs during autumn and winter from the surrounding agricultural watershed.

The sampling strategy consisted in investigating fine temporal scale variations. For this purpose, an intensive water sampling was conducted at both stations from April 1997 to March 1998. Water was sampled twice a month on average, covering the most frequent tidal ranges. The overall tidal cycle extend was sampled. A set of 12 samples in all, six at flood tide and six at ebb tide, were taken from a bridge at site 1 and from a pier at site 2.

Temperature and salinity were measured using a microprocessor conductivity meter (LF 196, WTW). Incident radiation, wind speed and atmospheric pressure data were supplied from the nearest meteorological station. The PAR was deduced from the linearly related equation between total incident radiations and PAR carried out by Jitts *et al.* (1976).

Discrete water samples for material concentration determination were collected in 2l acid-washed bottles and kept on ice at a temperature of about 5°C. They were then transported to the laboratory, which was 30 km far away. Samples were analyzed for dissolved organic carbon (DOC) using a spectrophotometer method (Pages & Gadel, 1990). Nitrate + nitrite (NN), ammonium (NH4), PO_4, Si and urea were measured on a Skalar continuous flow analyzer (Strickland & Parsons, 1968). Chl and phaeopigments (Phae) were estimated using the fluorometric method (Holm-Hansen & Riemann, 1978). TSS concentrations were based on dry weight filters

Criteria for stoichiometric and potential nutrient limitation

Based on nutrient requirements of diatoms, criteria for stoichiometric nutrient limitation was developed. The atomic ratio of Si:N:P of marine diatoms is about 16:16:1, when nutrient levels are sufficient (Redfield *et al.*, 1963). Deviations from this ratio indicated potential for N, P or Si limitation of phytoplankton growth. The combinations of N, P and Si ratios were established to investigate potential nutrient limitation based on nutrient uptake kinetics: N:P <10 and Si:N>1 indicated potential N limitation, whereas Si:N<1 and Si:P<3 indicated Si limitation (Levasseur & Therriault, 1987), N:P >20-30 stipulated P limitation (Goldman *et al.*, 1979).

We have to keep in mind that stoichiometric limitation does not indicate whether nutrient limitation is likely. However, potential limitation could be assessed by comparing the ambient nutrient concentrations with those likely to limit nutrient uptake (Justic *et al.*, 1995). In our assessment of stoichiometric limitations, we established two nutrient ratios for each ambient nutrient and applied the following criteria: (a) N limitation, if N:P<10 and Si:N<1; (b) Si limitation, if Si:P<10 and Si: N<1; (c) P limitation Si:P>20 and N:P>20. A similar approach was used by Justic *et al.*(1995).

Beside stoichiometric Criteria, which give clues on potential phytoplankton limitation, dissolved nutrient levels that limit phytoplankton growth were also established: N> 1.5µM, Si>5 µM and P>0.5 µM (Fisher *et al.*, 1988).

Phytoplankton productivity indicator

Phytoplankton growth and productivity are principally a function of light-limited growth. A strong positive correlation was seen in Harding et al (1986) results (r^2 = 0.71, n=19), between productivity (g C m^{-2} d^{-1}) and the product of incident radiation (Photosynthetically Active Radiation, PAR, E m^{-2} d^{-1}) and the euphotic zone Chl concentrations (mg m^{-2}). This relationship was once again checked by Malone et al (1996) using a larger data set (r^2 = 0.34, n=87). This product was established (Chl x PAR) during the present study to approach phytoplankton productivity within both sampling stations.

RESULTS

Watershed data

A twenty-two year river average cumulative flow (Fig. 2) indicated that the flushing period started in October and peaked during January-February, with a cumulative flow of 50 m^3s^{-1}. A rapid collapse was recorded during March-April, when the flow fell by half (~24 m^3s^{-1}). The DIN concentrations recorded at the outlet of the *Sèvre Niortèse* river exhibited a little increase during the last decade (Fig 2b), which came with a simultaneously sharp drop of PO_4 concentrations (Fig 2b). The Chl concentrations recorded at the same station during the spring summer bloom, May, June and July, showed rather high concentrations at the beginning of the decade,180μgl^{-1} in 1990, (Fig 2c) and dropped to less than 40 μgl^{-1} during the last three years.

Physical conditions

There were significant thermal variations of the water column throughout the year at both sampling stations (Fig 3). Two main periods were distinguished, the first from April to September, where temperatures shift between 15°C and 25°C and the second from October to March with temperatures < 15°C. From September to March and particularly in January, where they were most pronounced, plumes of fresh water marked by low salinity and low sediment loadings were observed at both stations (Fig 3). The absence of thermal and salinity stratifications was also noticed owing to the limited water column depth (5 m at maximum) and to the general good mixing.

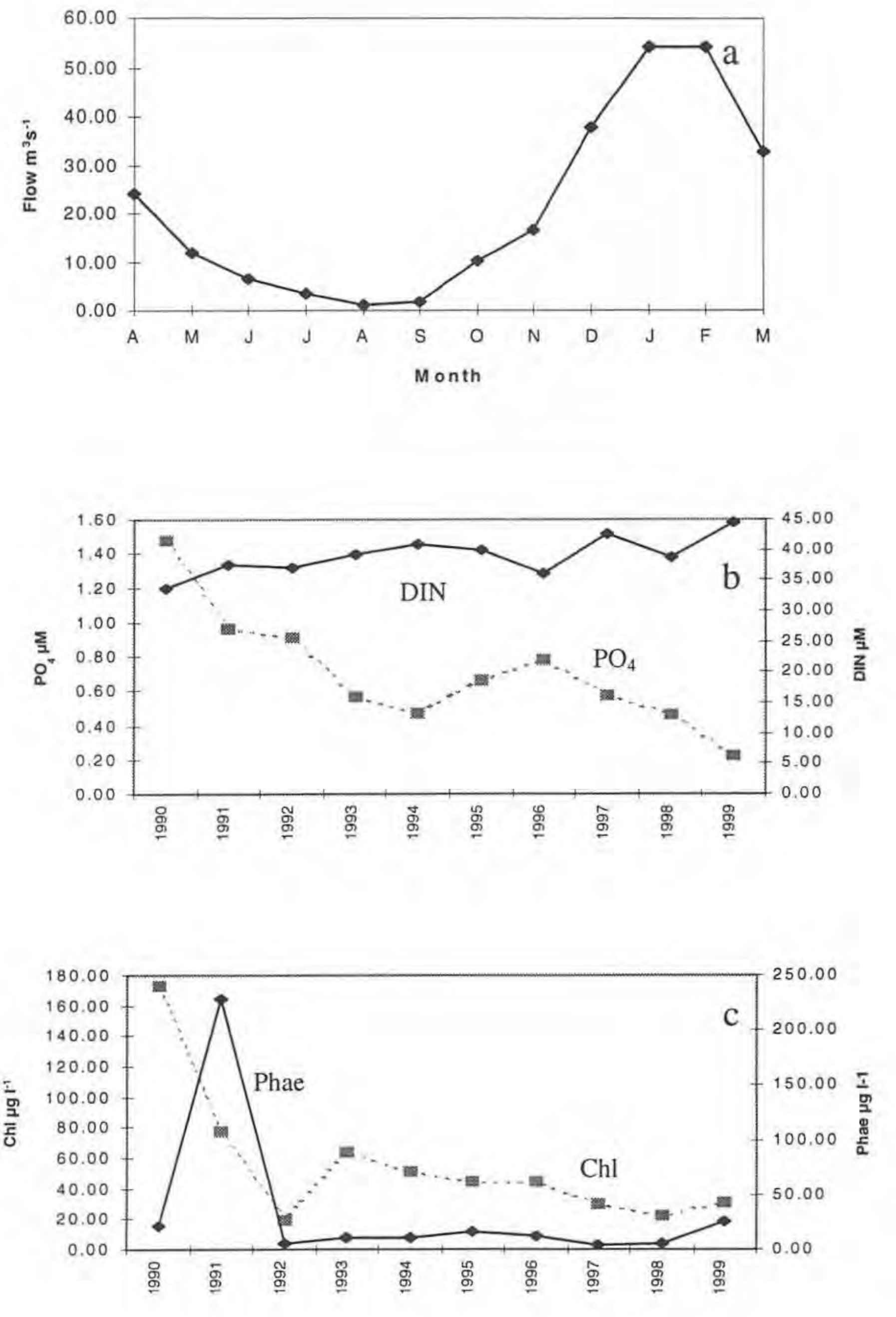


Figure 2 : A twenty-two year average (1969-1996, missing data are from 1970 to 1972 and from 1991 to 1992) monthly cumulative river flows (a). Annual the *Sèvre Niortèse* river DIN and PO4 -concentrations are shown for the last decade (1990-1999) (b). Spring summer bloom Chl and Phae pigment concentrations are also illustrated (c). Data provided by l'Agence de l'Eau Bretagne Loire Atlantique.

Total suspended sediment

Two main periods emerged from recorded TSS concentrations. From April to September concentrations were relatively high fluctuating between 5 and 30 mgl^{-1} and between 3 and 10 mgl^{-1} in the upper and lower bay respectively (Fig 4). There was a sharp drop in September, where concentrations were maintained at less than 6 mgl^{-1}. The mineral compound (result not shown) occupied a great part in TSS concentrations varying mostly between 70 and 80% throughout the year.

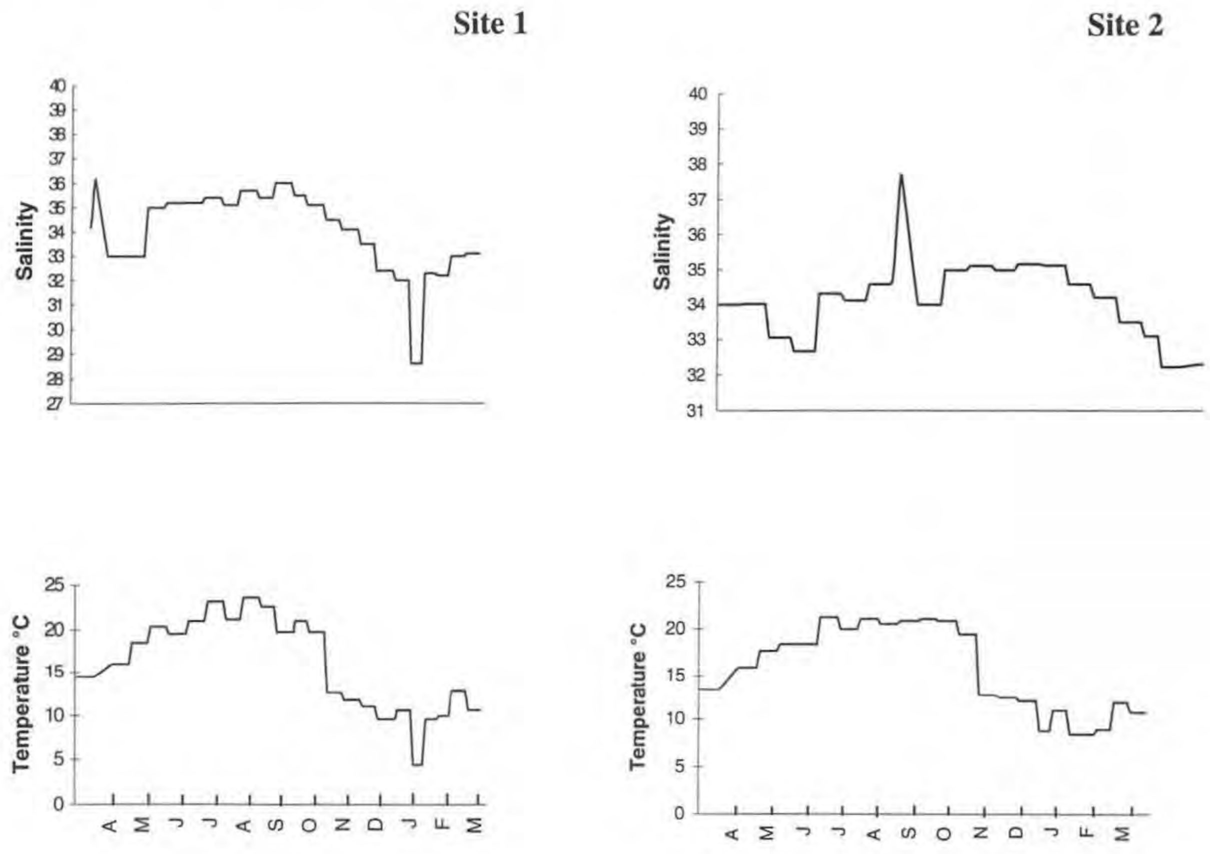


Figure 3 : Monthly salinity and temperature distribution in site 1, left curves; site 2, right curves.

Dissolved nutrients

There was a strong seasonality of NN concentrations (Fig 5), varying less than 1µM during the summer at the lower bay. These concentrations showed a rapid increase in November and

reached a peak of about 60-70µM in January. A similar pattern was noticed in the upper bay with more fluctuations during spring and summer before the rapid increase in November. Particularly variable NH_4 concentrations were observed within the bay: they varied in a limited range in the lower part (< 7µM), whereas they were rather high in the upper bay, reaching more than 70 µM in December. An increase of NN, as well as NH_4, was shown with salinity decrease reflecting the influence of freshwater input, which supplied an important amount of dissolved nitrogen into the adjacent coastal zones.

Monthly PO_4 distribution showed two periods (Fig 5): the spring summer where concentrations varied from 0.1 to 20µM and from 0.1 to 6µM in the upper and lower bay respectively. There was a sharp drop of PO_4 during the summer-autumn transition, except for a peak in January (25 and 6 µM), concentrations were maintained below 0.5 µM.

A similar pattern as NN was exhibited by Si (Fig 5) with concentrations increases coupled to salinity drop. There was a more important fluctuation within Si concentrations in the upper part of the bay (5-50µM), whereas at the lower bay concentrations were comprised between 5 and 10µM during spring and summer, showing a sharp rise in November and peaking up to 25µM in January.

There was no clear tendency of DOC concentrations (Fig 5): they fluctuated below 3.5 mgl^{-1} and 2.5 mgl^{-1} in the upper and the lower stations respectively. An exceptional pronounced peak was however recorded in August in the low bay. Links between DOC concentrations and dissolved nutrient concentrations, as well as Chl and TSS, were attempted but no significant relationships were found. Correlations between DOC and hydrometerological variables, including salinity, temperature, wind speed and air pressure were also investigated. Only a weak, but highly significant negative correlation was found at the lower bay between the discrete values of DOC and the maximum wind speed recorded during the sampling tidal cycles ($r = 0.327$, $P < 0.001$, $n=212$).

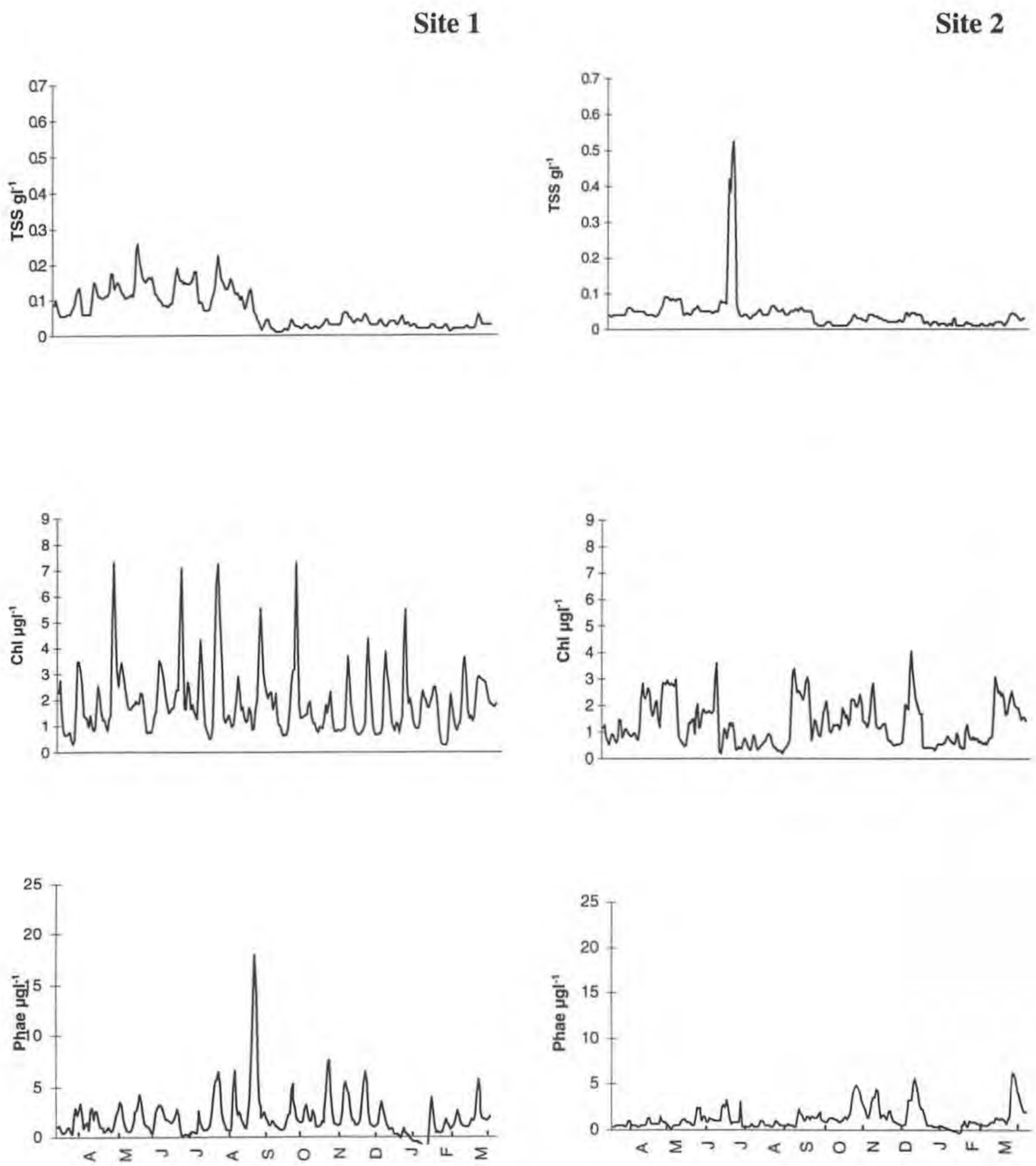


Figure 4 : Monthly cycles of TSS, Chl and Phae. site 1 (left) and site 2 (right).

Dissolved nutrient ratios

The upper part of the bay was primarily N limited, 38% of the cases (Fig 6) occurring from May to September, however DIN concentrations were infrequently below $1.5\mu M$. The Si limitation observed in 27.1% of the cases, was evenly distributed throughout the year, frequently noticeable from March to April in August and from December to January .

The P limitation, observed from November to February, was observed in 18.6% of the cases. The limit of 0.5µM, demonstrated by Fisher *et al.* (1988) as the level that limit the phytoplankton growth, was frequently reached particularly from September to February.

The lower bay was particularly N and P limited (26.3% and 26% respectively) (Fig 6). N limitation occurred from May to September, where DIN concentrations were almost exclusively below 1.5µM, whereas P depletion was observed from September to March. A slight Si limitation was also seen (15.9%), but only occurred during April where concentrations decreased below 4µM.

Chlorophyll a and Phaeopigments

In surface water, Chl ranged from 0.2 to 3.5$\mu g l^{-1}$ in the lower bay (Fig 4). There was no identified Chl bloom within this station. Nevertheless, we could distinguished two periods of high concentrations, the first in May, June and July; the second took place during a much longer period from August to November and was followed by a short peak in December. In the upper bay, Chl concentrations were particularly variable, fluctuating in a very short time scale (less than a month). However, four peaks of about 7 $\mu g l^{-1}$ were recorded during spring and summer. Late autumn and winter concentrations did not show any significant drop and fluctuated from 1 to 3 $\mu g l^{-1}$ on average.

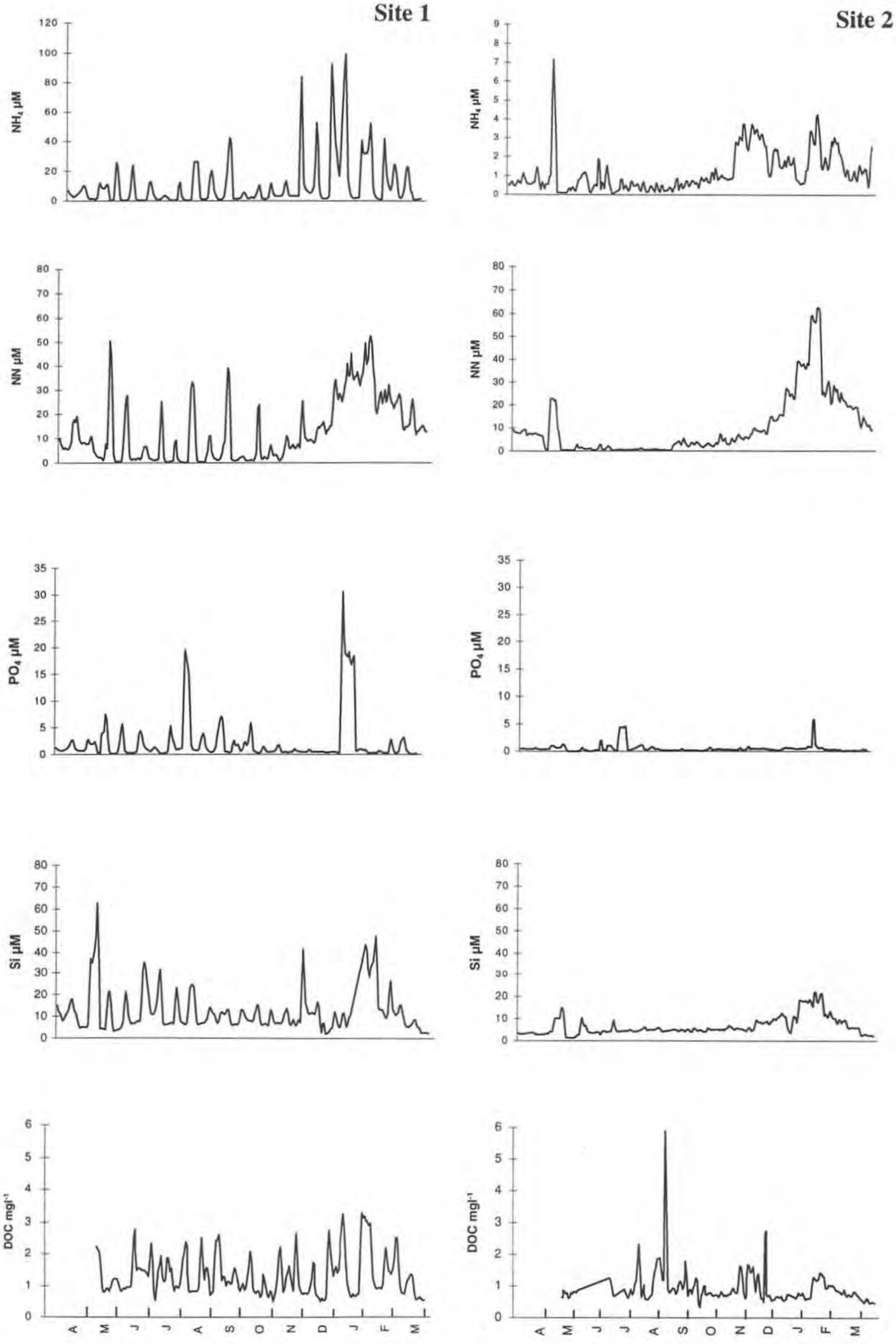


Figure 5 : NH_4, NN, PO_4, Si and DOC surface water distributions, site 1 (left), site 2 (right).

Phae which reflect the phytoplankton degradation, fluctuated between 0.1 and 20 $\mu g l^{-1}$ in the upper bay with a rapid peak in August (Fig 4).

In the lower bay, concentrations ranged from 0.2 to 6.8 $\mu g l^{-1}$ and showed two peaks, the first of 4 $\mu g l^{-1}$ was observed during late autumn and early winter and the second, more pronounced, (6.8$\mu g l^{-1}$) was recorded in March.

Principal compound analysis

Principal compound analyses (PCA) were generated for each site separately (Fig 7). In the upper bay, the first factor, factor 1, accounted for 43.35%, whereas factor 2 accounted for 17.43%, giving a total of ~60% of the variance expressed by the first two factor axes.
In the lower bay factor 1 accounted for 42.18%, whereas factor 2 accounted for 15.11%. These first two axes accounted for ~57% of the total variance.

The first principal compound in the upper bay (Fig 7a) was dominated by negative loadings for salinity, temperature and Si:N ratio and positive loadings for all inorganic nitrogen forms (NH_4, NN and DIN) and N:P ratio. This axis expressed the physio-chemical water quality based particularly on nitrogen loadings. On the other hand factor 2 was correlated to PO_4 and clearly expressed a gradient of dissolved inorganic phosphorus concentrations. The scatter plot of PCA scores (Fig 7a) indicated a large dispersion along factor 2. Between May and September, there was no large shift in PCA scores. October scores become more negative along factor 2 and shifted close to Si:P ratio.

In the lower bay (Fig 7b), factor 1 was positively correlated to DIN, Si and NN and negatively correlated to salinity, temperature, PAR radiation and Si:N. However factor 2 was dominated by Si:P and N:P ratios and positively correlated to PO_4, expressing a gradient of PO_4 concentrations.

Site 1 **Site 2**

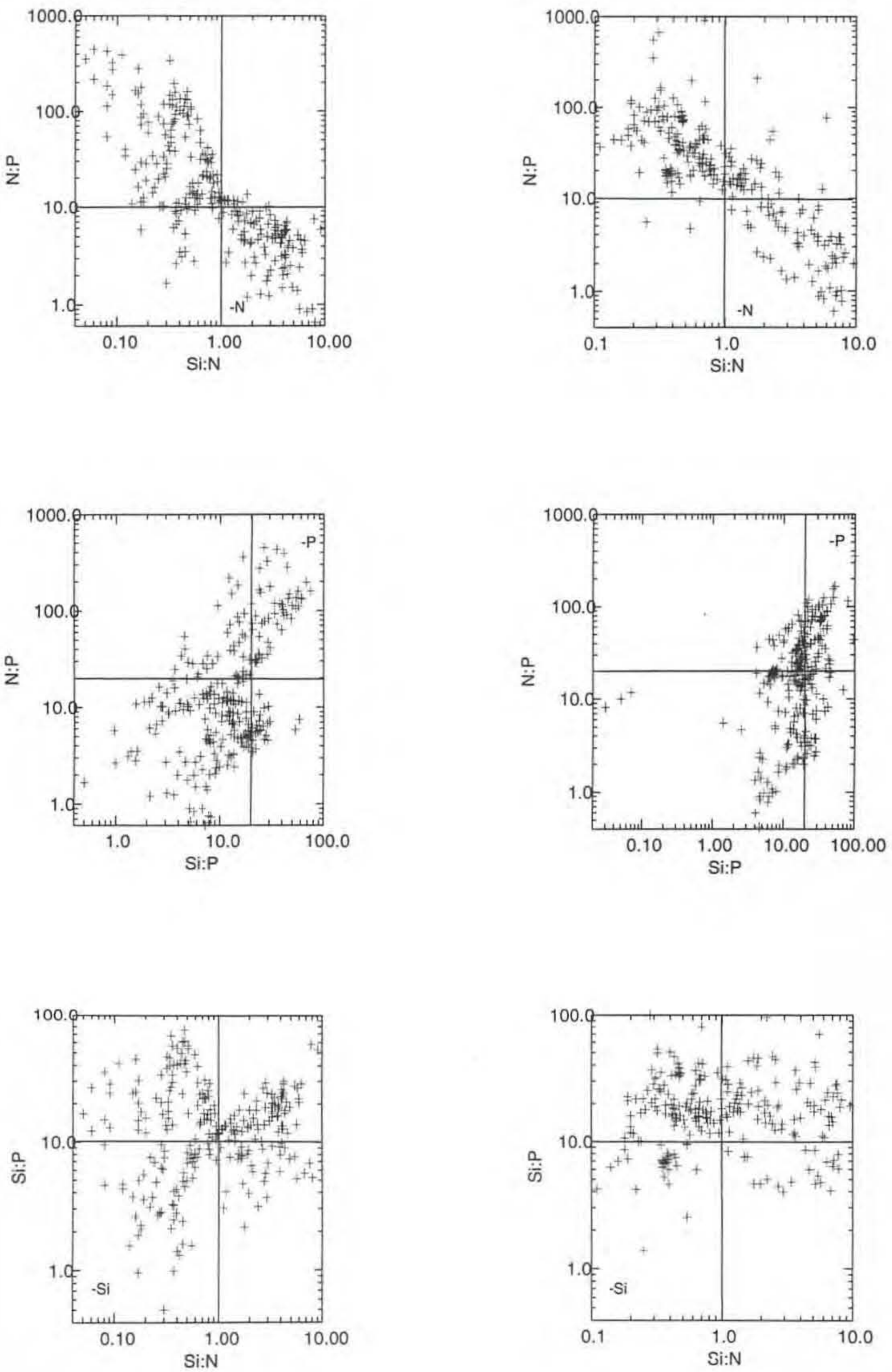


Figure 6 : Scatter diagrams of atomic nutrient ratios in surface water of site 1 (left) and site 2 (right). Stoichiomatric (= potential) limitation is indicated by -N, -P and -Si.

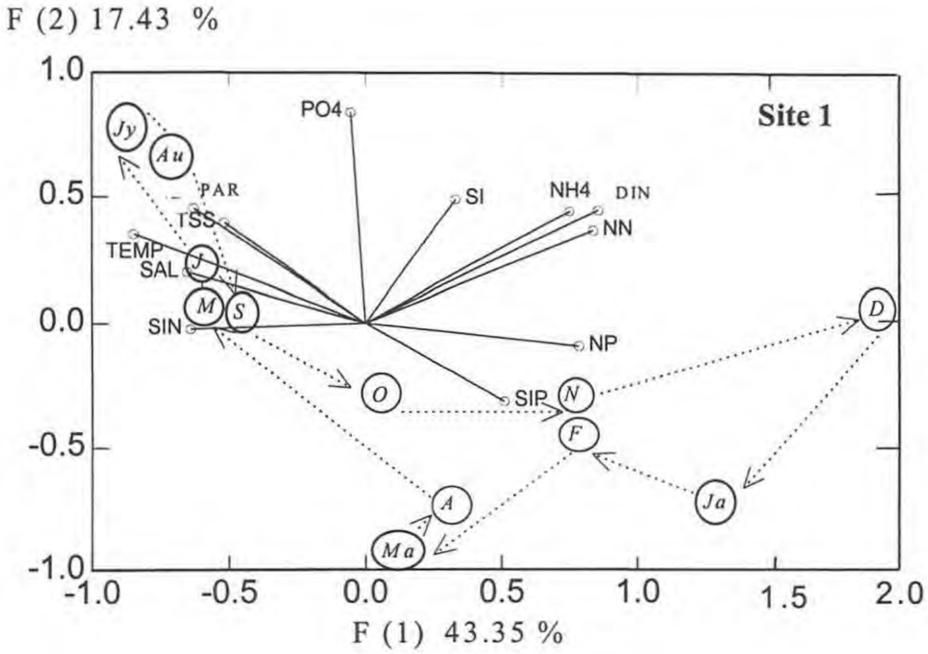
The PCA scores were scattered from April to October along the first axis: they shifted straightforward negative values from April to July, but changed their direction from August to October (Fig 7b). There was a large distribution from January to March limited in Factor1 positive loadings.

Multiple regression model

Multiple regression of Chl concentrations against the PCA scores (Factor 1 and Factor 2) over an annual scale (Table 1) indicated Chl was positively related to factor 2 in the upper bay ($r=0.473$, $P_2<0.0001$). A rather weak significant negative correlation was also carried out between Chl concentrations and factor 2 scores in the lower bay ($r= 0.188$, $P_2=0.007$).
Phae concentrations in the upper bay were positively correlated to factor 2 ($r=0.403$ $P_2 = <0.0001$). Significant negative correlation was also found in the lower bay between Phae and factor 2: however, it was rather weak ($r=0.185$, $P_2 = 0.004$).

The product of euphotic Chl and photosynthetically active radiation (Chl x PAR), which provides clues on phytopalnkton productivity, showed a significantly positive correlation with factor 2 in the upper bay ($r=0.368$, $P_2 = <0.0001$), but a significantly negative correlation with factor 1 in the lower bay ($r= 0.419$, $P_1 < 0.0001$).

Scaling down the overall annual extend on periods of high Chl concentrations allowed us to identify two periods in the lower bay, fluctuating between May and July for the first and from August to November for the second. As for the upper bay, May-September period characterized by high Chl concentrations, whereas October-March was slightly less Chl concentrated. As described above, a similar principal of multiple regression was performed for each period as identified previously.



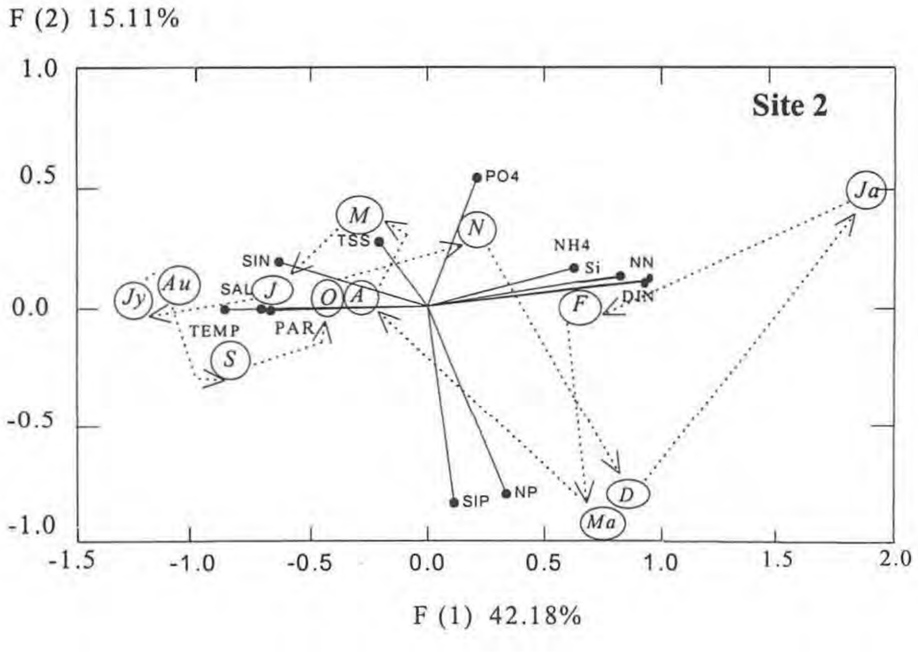


Figure 7 : Vector diagram, site 1 and site 2, indicating the loadings on Factor 1 (F1) and Factor 2 (F2) of the 12 recorded variables, scatter plot of average monthly PCA scores is also indicated in circles, connection between monthly scores is made with dotted arrows. Distinction between January (Ja), June (J) and July (Jy) is made, also March (Ma) is to be distinguished from May (M) and April (A) from August (Au).

The model performed only showed a significantly positive correlation between Phae and factor 2 ($r= 0.52$, $P_2 = 0.001$). during the first period (Table. 2), in the upper bay. During the second period, Chl was positively correlated to factor 2 ($r= 0.502$, $P_2< 0.0001$), whereas Phae to factor 1 ($r= 0.320$, $P_1= 0.001$).

Table 1 : Annual multiple regression summary (adjusted r and P values) for the dependent variables Chl, (Chl x PAR), Phae regressed against PCA axis scores, constructed from 12 water column variables from the two field sampling stations.
The model was: Chl or (Chl x PAR) or Phae = intercept + μ_1 Facto1+μ_2 Factor 2.
Significant P-value (P < 0.01) are in blod type.

	n	r	Parameter statistics μ_i (P value)	
			Factor 1	Factor 2
Site 1				
Chl	245	0.473	0.173	1.54
			(0.352)	**(<0.0001)**
Chl x PAR	246	0.419	-0.029	0.080
			(0.014)	**(<0.0001)**
Phae	245	0.403	-0.090	0.931
			(0.509)	**(<0.0001)**
Site 2				
Chl	250	0.188	-0.073	-0.183
			(0.277)	**(0.007)**
Chl x PAR	251	0.368	-0.038	-0.017
			(<0.0001)	(0.014)
Phae	239	0.185	0.018	-0.233
			(0.832)	**(0.004)**

In lower bay (site 2) and during the first period, Chl, as well as the (Chl x PAR) product, were significantly correlated to factor 1 ($r= 0.414$, $P_1= 0.003$; $r= 0.426$, $P_1= 0.003$)

respectively (Table. 2). As for Phae, negative correlation was found with factor 2 (r= 0.498, P_2<0.001). High significant correlations were found during the second period between Chl and Chl x PAR product and factor 1 (r=0.615, P_1< 0.0001; r=0.759, P_1< 0.0001).

Table 2 : Periodic multiple regression summary (adjusted r and P values) for the dependent variables Chl, (Chl x PAR), Phae regressed against PCA axis scores, constructed from 12 water column variables. During periods of high (1) and low (2) Chl concentrations at site 1 and the two high Chl concentrations at site 2.
The model was: Chl or (Chl x PAR) or Phae = intercept + μ_1 Facto1+μ_2 Factor 2. Significant P-value (< 0.01) are in bold type.

		n	r	Parameter statistics μ_i (P value)	
Period				Factor 1	Factor 2
	Site 1				
1	Chl	127	0.487	2.653	0.901
				(0.013)	(0.049)
	Chl x PAR	127	0.349	0.109	0.042
				(0.114)	(0.159)
	Phae	127	0.520	0.99	0.924
				(0.128)	**(0.001)**
2	Chl	105	0.502	0.210	0.642
				(0.043)	**(<0.0001)**
	Chl x PAR	106	0.082	0.002	-0.005
				(0.712)	(0.408)
	Phae	105	0.320	-0.688	0.370
				(0.001)	(0.230)
	Site 2				
1	Chl	52	0.414	0.602	0.010
				(0.003)	(0.410)
	Chl x PAR	52	0.426	0.075	-0.012
				(0.003)	(0.421)
	Phae	52	0.498	0.027	-0.450
				(0.882)	**(<0.001)**
2	Chl	75	0.615	-0.929	-0.113
				(<0.0001)	(0.725)
	Chl x PAR	76	0.759	-0.239	0.046
				(<0.0001)	(0.322)
	Phae	76	0.244	0.311	-0.674
				(0.439)	(0.251)

DISCUSSION

Nutrients sources and sinks

Most of the dissolved nitrogen enters the bay as DIN 80% to 90% (Barillé, 1996). The annual cycling of DIN in surface water exhibited an autumn winter maximum and a summer minimum, with an increasing amplitude and concentration with distance up to the axis of the bay; a similar pattern was observed for Si. These concentrations were conservative with respect to salinity (Fig 3) and (Fig 5) suggesting a great influence of the surrounding agricultural watershed. Barillé (1996) found up to 400 T of monthly nitrate input during the wet season from the two main rivers close to Fier d'Ars Bay. In addition to the fresh water input, the upper bay received intermittent aquacultural loadings (NH_4 ~78 μM , TSS = 30 mgl^{-1}) (Hussenot *et al.*, 1998) which were reflected by increasing both the amplitude and the concentrations of NH4, TSS and Si (Fig 5). In contrast to both DIN and Si, the annual cycle of riverine input was not reflected in the distribution of PO_4 which, except for a very rapid peak in January, tended to show high concentrations during summer.

Comparison of dry and wet season surface water nutrient distribution showed that, contrary to autumn and winter, where DIN and Si exhibited the same pattern, DIN was completely depleted in summer (Fig 5). However Si showed other timid peaks in both the lower and upper bay. This trend may be explained by the effects of benthic diatoms recycling (Conley & Malone, 1992).

The PO_4 distribution, less dependent on riverine input than DIN (Fig 2) and (Fig 5) was higher during summer, suggesting other potential source of PO_4 than freshwater loading.

TSS exhibited a similar pattern (Fig 4), moreover, a peak of TSS noticed in the lower bay during June- July transition correspond to a peak of PO_4. The link between these two constituents was attended, revealing a significant positive correlation ($r=0.53$, $P<0.001$). The increase of TSS during summer was probably the consequence of high aquacultural loadings (Hussenot *et al.*, 1998) and the resuspension of the bottom sediment due to high navigational traffic within the bay. Thus PO_4 seems to have two potential sources, during winter allochtonous riverine and coastal input and during summer autochtounous benthic source. This sediment delivery of dissolved inorganic phosphorus under summer conditions is explained as a result of dissolution of the Fe-P complexes within the sediment (Cornwell *et al.*, 1996). A similar pattern of phosphorus delivery was advanced in Cheasapeake Bay (Malone *et al.* 1996) and in an experimantal earthen pond close to the Fier d'Ars bay (Crottereau, 1999) .

A link between DOC and maximum wind speed indicated that the build-up and decrease of DOC, particularly at the ocean interface could be caused by mixing processes. Thus, the absence of significant correlation with TSS ($r=0.047$; $P = 0.49$) eliminated the hypothesis of DOC benthic resuspension and should favor an allochtonous coastal origin of DOC. In an aquatic environment, DOC results from plankton cellular lysis and phytoplankton exudations (Jumars *et al.*, 1989, Williamson & Morris, 1999), which characterize a rather labile organic matter, C:N~6 (Romankevich, 1984). Thus, the DOC loadings into the Fier d'Ars Bay will contribute, with their rapid decomposition, to the enrichment of the bay.

Nutrient limitations at time and bay-space scale

Nutrient limitation phytoplankton can be apprehended in two ways: kinetic experiments based on relationships between phytoplankton nutrient uptake and nutrient concentrations and response to nutrient enrichment (Goldman & Gilbert, 1983; Fisher *et al.*, 1992). This approach deals with time and scale variations depending on phytoplankton physiological responses, which were shorter compared to ecosystem scale variations. Field measurements of nutrient distribution and phytoplankton on seasonal time scales of water input and turnover at the level of the specified ecosystem fall in the second category. This approach has the drawback of taking into account a time scale greater than the phytoplankton biologically response (Malone *et al.*, 1996).

Potential DIN depletion occurred more frequently than that of Si and PO_4 (Fig 6). However, we expected more PO_4 limitation due to the high DIN and Si input relative to PO_4. Nevertheless, when we closely examined the nutrient depletion timing, we noticed that Si was firstly depleted followed by late spring and summer DIN depletion and late autumn and winter PO_4 depletion. In the upper bay, a similar pattern was noticed for DIN and PO_4 ,whereas Si showed short, yet frequent depletion.

The regression estimation of the annual data set (Table. 1) showed that Chl, as well as (Chl x PAR) product, and Phae in the upper site were closely related to factor (2), itself strongly correlated to PO_4, suggesting a possible PO_4 limitation of both phytoplankton biomass and productivity. This site received both upstream and downstream high DIN loadings and was more suspected of Si limitation due to the frequent characteristic of this constituent depletion. However no evident relationship was reported between Chl and Si concentrations. It seemed that a prevention against Si depletion was set off every time that this constituent attempted to lack in the system. A rapid regeneration of dissolved silicate was demonstrated to maintain sufficient Si levels to prevent silicate limitation (Conley *et al.*, 1993; Rabalais *et al.*, 1996).

The disintegration of the annual time scale over two periods of high (period 1) and low (period 2) Chl concentrations (Table 2) showed that the model was only significant during low Chl concentrations, strengthening the idea of possible PO_4 limiting the phytoplankton biomass accumulation. Nevertheless, phytoplankton Chl productivity (approached with Chl x PAR product) did not seem to be limited by any nutrient supply. This shift in the temporal scale illustrated more clearly the dynamics of phytoplankton biomass and allowed us to identify more precisely its controlling factors. Thus, during the spring/ summer (period 1), Chl biomass accumulation was supplied by both DIN aquacultural loading and benthic regeneration of ammonium. As for PO4, it was supplied from TSS resuspension. From late summer to autumn, when riverine input became significant, the phytoplankton Chl productivity was also maintained. However the depletion of PO_4 with regard to DIN and Si have suppressed a second autumnal maximum biomass.

In the less anthropogenic influenced site, the lower bay, the first period showed positive correlation of Chl and (Chl x PAR) with factor (1) (Table 2). A strong dependence on salinity, temperature (< 20°C), PAR and TSS was also reported from the scatter of PCA scores and a lesser dependence on DIN loadings (Fig. 7b). Moreover, DIN depletion started to be observed, suggesting that primary production may not be sensitive to nitrogen supply and that other factors were susceptible to regulate the phytoplankton spring summer biomass. We should notice that in this area, the tide is semidiurnal, with a maximum range of about 5 m, which leads to a general good mixing and a high turbidity, particularity during spring and summer (Fig 4). This leads to a limitation of sunlight penetration through the water column, resulting from the absorption and the reflection of incident layers by mineral particles (Cloern, 1987; Barillé, 1996; Irigoien & Castel, 1997). The lack of significant correlation between Chl and total suspended sediment (TSS) during the spring phytoplankton rise (Table 3), a period of high turbidity and low river flow, may be explained by two hypotheses: (1) The importation of phytoplankton to the bay from areas with high euphotic depth (Cloern, 1987; Irigoien & Castel, 1997); or (2) a strong adaptation of phytoplankton to light limitation. The significant negative correlations of Chl concentrations with both salinity and temperature during this period (Table. 3) favors the retention of the first hypothesis.

In view of these observations we could suggest that the spring phytoplankton rise observed in the lower Fier d'Ars Bay does not probably have an autochtonous origin and is derived from the riverine and / or mixed seaward areas. A similar pattern of entering offshore rich water, varying in proportion to the amount of input freshwater, was demonstrated in Narragansett Bay (Nixon, 1997). However, two relevant questions still need to be asked: firstly, why was there not a high phytoplankton biomasss accumulation?. And secondly, which factors set off the phytopalnkton bloom mechanic collapse?.

Three main hypotheses may answer these questions: the first two could act separately, the last may co-occur with each of them. (1) As soon as they reached the bay, phytoplankton cells were likely to be submitted to salinity stress, which could limit their growing capacity (2). The intensive vertical mixing can produce changes in light conditions which fluctuate faster than cells adjusting their physiology (Demers *et al.*, 1986). This leads to a decrease in photosynthetic activity, due to a reduction in the residence time of algae in photic zones (Demers & Legendre, 1981) (3) . The chronology of allochtonous riverine input, mostly occurring during the autumn and winter, which was outside of the spring summer phytoplankton biomasss accumulation period, as a consequence phytoplankton rise was set off during a DIN-limited condition. This possibility of nitrogen limiting primary production may not by itself, basically influence the spring bloom; however combined with the two previous hypotheses it may accelerate bloom collapse.

During the second phytoplankton rise, PO4 depletion was reported, while a starting allochtounous riverine DIN and Si input was noticed. However, neither Chl nor (Chl x PAR) seemed to be affected (Table. 2). Chl concentrations and specific Chl productivity were negatively correlated to factor 1. A hard coupling between DIN concentrations and specific Chl productivity emerged (r = 0.467, P<0.0001), showing a rapid nitrogen uptake (Fig 8). During this period, Chl also seemed to be salinity- and temperature-dependent (Fig 7). Temperature-dependent summer phytoplankton productivity is well documented and seems to be associated to a transition in phytoplankton assemblage (Levasseur *et al.*, 1984; Wasmund, 1994; Malone *et al.*, 1996) .

Table 3 : Regression between Chl (y: $\mu g l^{-1}$), TSS ($mg l^{-1}$), salinity and temperature. Significant P-value (< 0.01) are in bold type.

Period	Site	Chl-TSS	r	Chl-Sal	r	Chl-Temp	r
	Site 1						
1		y=6.79x+2.44	0.075	y=0.25x-5.84	0.04	y=0.26x-2.25	0.105
		(0.379)		(0.57)		(0.22)	
2		y=23.25x+0.91	0.263	y=-0.13x+6.18	0.193	y=-0.05x-2.24	0.159
		(0.004)		(0.038)		(0.08)	
	Site 2						
1		y=-0.31x+1.58	0.089	y=-0.22x+9.3	0.522	y=-0.30x+7.16	0.533
		(0.45)		**(<0.0001)**		**(<0.0001)**	
2		y=35.81x+0.62	0.683	y=1.16x-38.9	0.514	y=0.08x+0.04	0.42
		(<0.0001)		**(<0.0001)**		**(<0.0001)**	

This second late summer Chl rise, characterized by a high phytoplankton productivity, with biomass nearly equivalent to that of the first spring bloom, was the response of both summer nutrient regeneration and starting allochtonous nutrient input. Factor 1 accounted for ~ 20% ($(0.759)^2$ x 42.18) of total specific Chl productivity variance, which was rather weak.

Thus, a second potential Chl biomass source was suspected of enriching the phytoplankton standing corp. This phytoplankton rise was also characterized by the strong relationship between Chl and TSS concentrations (Table 3) which suggested the influence of the resuspension on the Chl distribution. The contribution of the microphytobenthos to the enrichment of the water column with Chl was noted in the Gironde estuary, indicating that 17-32% of microphytobenthos biomass may be resuspended in the water column (Santos, 1995). This estimate was close to the value of 15 % found by Guarini (1998) in the Marennes Oléron Bay, using a microphytobenthos dynamic model. Moreover, the positive correlation of Chl with temperature (Table 3) contrasted the hypothesis of thermo-inhibition of microphytobenthos production during high temperature conditions (~ 25°C), as described by Guarini (1998). However, a summer shift of phytoplankton composition was noticed by Frikha (1989), with transition from diatoms and dinoflagellates to small-sized (2- 3µm) autotrophic flagellates. A similar pattern was pointed out by Dupuy (1999) with spring summer phytoplankton shifting from 11-124µm cell-sized diatoms to 3- 19 µm small cell flagellates. Further factors may also regulate the phytoplankton productivity during this period, such as high phytoplankton grazing rates (Frikha, 1989; Pennock & Sharp, 1994; Malone *et al.*, 1996), suggesting that phytoplankton production and consumption are closely coupled through the food chain, such coupling being illustrated by the high DIN uptake following Chl production.

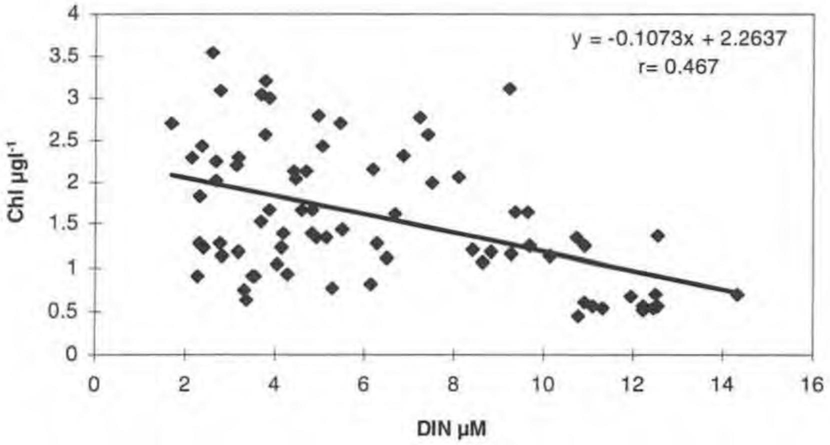


Figure 8 : Correlation between DIN and Chl concentrations in the lower bay during the summer autumn phytoplankton rise (period 2).

The phytoplankton exudation during high productivity, as well as an increase of phytoplankton cell lysis due to high grazing rates, were the precursor of DOC accumulation in the water column (Jumars *et al.*, 1989; Williamson & Morris, 1999) and may explain its increase during August and September (Fig 5). Furthermore, the increase of DOC was demonstrated to decrease the penetration of light (Shindler *et al.*, 1996 a,b) and, consequently, the inhabitable area for phytoplankton. Thus, DOC plays a major role in regulating the transmission of the solar radiation in freshwater and marine ecosystems. Morris et al (1995) found that 68% of the variation of PAR was explained by, DOC and Chl concentrations. Hence DOC seems to be a serious competitor of phytoplankton for PAR absorption and may contribute, for a part, to the lack of large biomass responses during the summer.

The shift of space and time scales revealed that the system was reacting differently. The input ratio of DIN and PO_4 imparted an important influence on the regulation of phytoplankton production. It is clear that the variations in N:P stoichiometry during autumn winter (100- 400) with frequently low PO_4 concentrations < 0.5µM strengthened the conclusion of P-limitation in the upper bay. The disintegration of the annual scale phytoplankton response in the lower bay allowed us to highlight factors controlling plankton biomass accumulation and carry out a temporal transition within these factors. Such transition was illustrated by the fluctuation from a spring summer light-limitation to a summer autumn enrichment with microphytobenthos, coupled to a high grazing pressure, the whole possibly co-occurring with ultimately N-limitation.

CONCLUSIONS

Nixon (1997), comparing prehistoric and recent nutrient inputs into Narragansett Bay, pointed out a five-fold increase of DIN and a two-fold increase of DIP input, which increased the primary productivity of the bay by a factor of two. In the Fier d'Ars Bay, there was evidence of increasing nitrogen loadings (Fig. 2). As for phosphorus, riverine input still decrease, explained by a high retention by the sediment of the upstream watershed (Barillé, 1996)). The sediment supply of PO_4 seems to prevent a P-limitation during spring summer phytoplankton rise. Nevertheless, there are still low levels of Chl within the bay ($<7\mu g\ l^{-1}$). This is possibly due to high water turbidity and to the timing of allochtonous nutrient inputs which were outside the period of phytoplankon accumulation. As a consequence, the system absorbed a large amount of nitrogen-containing nutrient which is not profitable to primary microproducers.

Although macrotidal estuaries have been shown to have a greater tolerance to pollution by nitrogen rich nutrients due to high tidal mixing (Monbet, 1992), we nevertheless know little about the dynamics of macroalgae populations, their tolerance to nutrient loadings and light limitation, since a shift in algae species has been recorded in response to an alteration of light penetration, which has favored algae with growth forms, keeping them up into the water column (McComb & Humphries, 1992).

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CHAPITRE II

Spatial and Temporal Variability in Nutrients and Suspended Material Processing in the Fier d'Ars Bay (France)

M. BEL HASSEN

UMR 10 CNRS-IFREMER, Centre de Recherche en Ecologie Marine et Aquaculture, B.P. 5, F-17137 L'Houmeau, France.

Ce chapitre a été soumis pour publication dans *Estuarine, Coastal and Shelf Science*

soumis le 12 octobre 1999

accepté le 21 Janvier 2000

Réf: N73/99

ABSTRACT

A net material flux study in an estuarine mudflat along the French Atlantic coast, confirmed the general hypothesis already shown in European salt marshes. Despite its small area, the system was not as homogenous as first thought. The upper part of the bay imported both dissolved and particulate matter. Whereas, the lower part exported dissolved and imported particulate matter. Tidal dynamics and water residence time were suspected of enhancing these differences.

Particulate mineral import represented more than 70% of the total suspended sediment, this came with an export of Silicate, Chlorophyll a and Dissolved Organic Carbon. We assumed that phytoplankton and microphytobenthos were exported from the bay in a degraded form. A wide synthesis was conducted integrating the tidal physical dynamics, with the geomorphology of the system, to explain material fluxes differences between the two sampling sites.

Differences of particulate material transport between European and North American systems were once more verified. New hypotheses relative to the functional role of phytoplankton and microphytobenthos in coastal food chains in European salt marshes were also investigated.

Showing that, in contrast to American systems, mineral particulate import in European ecosystems were prevalent, enhancing small cells growth, such as phytoplankton and micophytobenthos. These latter, being sorbed into mineral particules, were preferential for sedimentation, thus limiting loses from estuaries and would explain the major particulate import in these systems.

Key words

Export, Dissolved matter, Fier d'Ars Bay, Import, Microphytobenthos, Nutrients, Organic matter, Particulate matter, Phytoplankton, Sedimentation rate.

INTRODUCTION

Understanding ecological characteristics of estuaries including their biological and chemical variabilities has always been a challenge to estuarine ecologists. Their large productivity and particularly, their critical position at the boundary between terrestrial and marine systems, make them vulnerable to global changes and to direct human activities (Nixon et al., 1986; Cloern et al., 1995; Valeila et al., 1997). Quantification of nutrient and organic matter exchanges between salt marshes, estuaries and the coastal ocean have been conducted in several ecosystems: the causes and the implications of these results have been widely discussed (Wolaver et al., 1985; Dame et al., 1989; Dame et al., 1991; Childers et al., 1993a, b; Dame & Gardner, 1993; Boorman et al., 1994; Dame & Allen, 1996).

In the light of these studies, many ecosystem classifications emerged, those inherent to the geomorphology of the site (Odum et al., 1979), to the ecological maturity (Dame et al., 1992) and to the intercontinental geographic position (Lefeuvre and Dame, 1994). These classifications were based on the "Outwelling " or the " inwelling" hypothesis. The first expressing the movement of materials from the estuary to the sea (Odum, 1980) and the second from the sea to the estuary. Since most salt marshes appear to import organic and inorganic suspended sediment and produce more organic material than can be degraded or stored within the system and that excess is exported to the coastal ocean.

Comparative studies have always faced unstandardized methodology (Lefeuvre and Dame, 1994). Hypotheses already obtained have to be reviewed using equivalent methodologies to prove their soundness. The aim of this paper is to establish an annual budget in a French salt marsh-estuary and to investigate the relationships between its functioning and those of other European ecosystems. A detailed comparison with a North American estuary, the Bly creek, was conducted and a synthesis was attempted to integrate tide physical dynamics with the geomorphology of the system and the origin of the matter, to explain materials fluxes.

MATERIALS AND METHODS

1- Study area

The Fier d'Ars is an estuarine semi-closed bay situated centrally on the French Atlantic coast (Fig. 1), in the north of the Marennes Oléron Bay . It covers a total area of 750 ha of intertidal mudflats and salt marshes, surrounded by a large area (1258 ha) of productive earthen ponds. Unlike the Eastern Atlantic coast of the U.S., where many coastal areas are dominated by angiosperm marsh plants, such as *Spartina alterniflora*, the salt marshes lining the Bay of the Fier d'Ars are better managed now than at the beginning of the century, consisting of fish and/or oyster ponds.

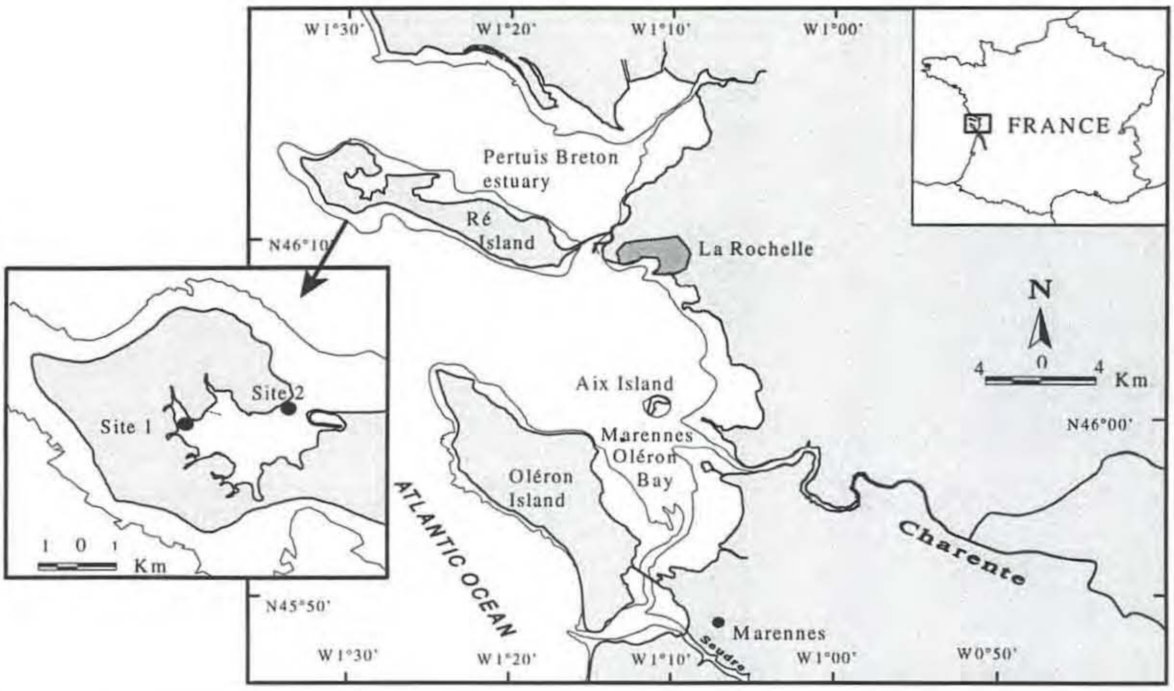

Figure 1 : Map showing the geography of the Fier D'Ars Bay and the location of the sampling transects. The bay-ocean transect is situated in the north of the bay, the salt marsh-mudflat transect was just south-west of Transect 2. The thin line indicats the zero sea level.

The marsh grass grows only in the upper intertidal zone. This area was flooded mainly during the spring tide (tidal constant > 90), which occurred during the sampling period, in about 30% of the cases. *Halimione portulacoïdes* was the most dominant marsh grass.

The meteorological conditions exhibited a strong seasonality.The tide is semidiurnal, with a maximum range of about 5 m which, leads to general good mixing and a high turbidity due to sediment resuspension. The maximum water velocity reaches 0.9 m s^{-1} during the spring tide to 0.5 m s^{-1} in the neap tide. The bathymetry, which expresses the topographic elevation with regard to sea level, ranges from about 3m, it tends to decrease with distance down to the axis of the bay and reaches 1m near the ocean boundary.

Marine waters enter the bay from the North through a narrow entrance: this flow supplies water to the earthen ponds. After a residence time within the ponds, varying between a few hours to 15 days, which depends on activities, ranging from intensive aquaculture, extensive aquaculture along with shell fish culture, water leaves the ponds with the ebbing tide.

The Fier d'Ars Bay exemplifies a simple functioning pattern: the sedimentary morphology of this area shows typical drain structures (Crave, 1995), where stream network is easily identified. Moreover, a single entrance allows water exchange with the open sea, which simplifies and makes easier the study of materiel transfer between the bay and the coastal area. Beside the earthen ponds loadings, the bay receives, from the upland systems, a few intermittent rainwater inputs during wet seasons. These attest the reduced number of external water loading sources. The small area and relatively easy access of the Fier d'Ars Bay, allow its use as a model to study nutrient processes between the estuary and the coastal ocean.

2- Sampling strategy

The most dominant morphological-sedimentary structures characterising the site were the earthen ponds, built up as a resut of the increasing sedimentation rate (Long 1975) and the extensive mudflat (intertidal area separating the earthen pond from the coastal area). In order to investigate both subsystems, two bathymetric transects were established to estimate the material budget (Fig.1). The transect 1 (T 1), situated in the upper part, was the conjunction between the earthen pond system and the extensive mudflat, which supplied the major anthropogenic input to the bay. The second, transect 2 (T 2) was from the mudflat to the Atlantic Ocean.

Fluxes were measured over 28 tides, twice a month on average (about every 13.02 d), from April 1997 to March 1998. The sampling strategy was planned to cover the most frequently occurring tidal ranges, tidal constant about 85 for the spring tide and about 45 for the neap tide, which led us to sample 14 spring tides and 14 neap tides for each station. As described above, the tide is semidiurnal, it lasts, depending on the tidal range, for approximately 12 hours. Time intervals between 0.5 h to 1 h separated successive samples, generally leading to 12 samples per tide, which we carefully divided into two groups of 6 samples during the flood tide and 6 on the ebb tide. This strategy permits an annual total number of 336 samples (28x12) for each station. The two stations were sampled during successive days to reduce all likely variations of meteorological and hydrological conditions, which further ensured an accurate comparison between the two subsystems.

Discrete water samples for material concentration determinations were collected, from a bridge (T1) and from a pier (T2), using a (WILDCOC instrument) sampling bottle at mid-depth in the middle of the bathymetric transect, which adequately represented fluxes through the transect. Samples were collected in 2l acid-washed bottles, they were filtred for the estimation of total suspended sediment (TSS) (prerinsed, pre-ashed Watman GF/F filters) and of Chlorophyll a (Chl a) (Watman GF/F filters). Both the filter of Chl a estimation and the filtrate were stored on ice at a temperature of about 5°C, in the dark, before being frozen in a field laboratory. Once arrived in the laboratory, filters for TSS estimation were kept at 50°C in a thermoregulated dryer, for at least 48 h, before being weighed.

Samples were analysed for dissolved organic carbon (DOC) using the spectrophotometer method (Pages & Gadel, 1990). Nitrate + nitrite (NN), ammonium (NH_4), orthophosphate (PO_4), silicate (Si) and urea were measured on a Skalar continuous flow analyzer (Strickland & Parsons 1968). Chlorophyll a (Chl a) was estimated using the fluorometric method (Holm-Hansen et al., 1965). Total suspended sediment (TSS) and organic suspended sediment (OSS) concentrations were based on the dry weight and ashed weight of each filter respectively. Mineral suspended sediment (MSS) was thus deduced as the difference between (TSS) and (OSS) concentrations.

At any given sampling time, instantaneous water fluxes were calculated as a function of water level, water velocity (measured using multi-parameter water quality monitoring instrument (EMP 2000, Martec)) and the microtopography of the transect. Instantaneous water flux multiplied by constituent concentrations from the water sample was thus instantaneous flux.

Fluxes, calculated as mg of constituent per m^{-2} of drained marsh and h^{-1} of inundation time, required the knowledge of the total area drained at each sampling site. Numerical methods using a GIS (Geographic Information System) were performed to determine the drainage area at each sampling station. It used the topographic elevation of the area (20 x 20 m grid cell) as input data and then carried out the direction of the flow, with the assumption that water will follow the steepest elevation (Maidment, 1993). By making out the flow path within the sampling zone, it then calculated the area that flows upstream of each sampling station and assumed it to be the drainage area of this particular location.

3- Statistical design

The statistical design involved sampling 28 tidal cycles out of the 704 cycles during the year. As the number of 30 cycles of an annual finite population (704 in our case) was retained as an arbitrary limit number to ensure the conformity of the observed distribution to a Normal distribution (Scherrer, 1984). This constrain, i.e. the Normality of the distribution, was requisite before applying the "Regression Estimation" (Spurrier and Kjerfve, 1988). Thus, in our case, we were below the minimum number required (30), this was why the Normality of the sampling distribution was tested using the Kolmogorov-Smirnov test (Systat 7.0 package). This procedure was avoided in Spurrier and Kjerfve (1988) by sampling 34 of the 707 annual tidal cycles and assuming that this is a sufficiently large number of cycles to ensure statistical significance.

The technique of "Regression Estimation" was then applied to estimate the annual net material fluxes. A set of 10 hydrometeorological predictor variables were measured for each of the 704 total cycles (Table1). They were used to predict the material fluxes of dissolved nutrient, particulate matter, chlorophyll and dissolved organic carbon. This set of predictor variables was refined by running all possible regressions and selecting the model which retained significant variables with the maximum coefficient of determination. The regression estimate of net annual flux was thus equal to the sum of the predicted flux values for the 704 tidal cycles. This method has a small standard error because the variability was explained by the regression estimation, this allowed us to establish the direction of net flux with tight confidence bounds on net annual flux.

Table1 : Predictors variables used in regression estimation for annual flux budgets production. Parameters were measured from the nearest meteorological station during the year-long sampling period. Units are indicated in brackets

Predictor variables	Descriptions
Vt.	Mean wind speed ($m.s^{-1}$)
Vp	Maximum wind speed ($m.s^{-1}$)
Prmn	Atmospheric pressure during the sampling cycle minus average atmospheric pressure for all sampling cycles (hPa)
Prmn1	Atmospheric pressure during the previous sampling cycle minus average atmospheric pressure for all sampling cycles (hPa)
Tam	Water temperature minus average water temperature for all sampling cycles (°C)
Tom	Air temperature minus average air temperature for all sampling cycles (°C)
Pn	Precipitation during sampling cycle ($mm.m^{-2}$)
Pn1	Precipitation during previous sampling cycle ($mm.m^{-2}$)
Hn	Maximum column depth during sampling cycle (m)
Hn1	Maximum column depth during previous sampling cycle (m)

The models of import or export of constituents were considered statistically significant at 5% level when the estimated mean annual flux confidence intervals did not contain zero. Thus the mean annual fluxes fluctuated in the same direction, positive for the import and negative for the export.

RESULTS

1- Water velocity and water level

During neap tide and at T1, water velocity fluctuated in a large range between 0.1 to 0.6 m s^{-1}(Fig. 2A). The mean flood velocity exceeded that of the ebb. With spring tide (Fig. 2B), the mean ebb velocity was higher. The ebb tide showed a relatively slow build-up followed by a constant velocity from 0.65 to 0.70 m s^{-1}.

For T2, the mean ebb velocity exceeded that of the flood for both neap and spring tide (Fig. 2C,2D). The maximum range was about 0.5 m s^{-1} in the neap tide and 0.9 m s^{-1} in the spring tide. Unlike T1, there was a rapid build-up and a relatively short peak. Tidal flow was asymmetric in this area mainly during the spring tide. Outflow of the water during the ebb tide lasts much longer with a slow decrease of the water level.

2- Water budget

The volume of ebb and flood water was calculated for each tidal cycle (Table 2). The mean difference between the flood and the ebb percentage showed that there was an import of 9 % of the water to T1 and a slight export of -0.7 % for T2. The difference of water budget assessed at T1 could be explained as the water used to feed earthen ponds with renewed flood water, whereas the excess at T2 may take origin from precipitation or also from intermittent rainwater inputs.

Table 2 : Water budget found at the two sampling transects.

	flood tide		ebb tide		%flood-%ebb
	min	max	min	max	
Transect 1	44596	167207	43305	132890	9
Transect 2	3303561	8651144	2440184	10128217	-0,7

3- Nutrient concentrations

The inorganic nitrogen occurs in salt marshes as NN and NH_4. Nitrate tended to be the most important form contributing to nitrogen fluxes in T2 (Table 3). Nitrate concentrations were generally between 0.33 and 61.5 $\mu mol.l^{-1}$ and represented between 46% and 98% of the total inorganic nitrogen . Ammonium concentrations were particularly variable: relatively low at T2, they could reach more than 70% of the sum of NH4 and NN at T1.

Table 3 : Concentration range of constituents assessed at both transect. Units are μml^{-1} for Urea, NH_4, NN, PO_4 and SI, mgl^{-1} for TSS, OSS, MSS and DOC and μgl^{-1} for CHL a.

	Transect1		Transect 2	
	flood	ebb	flood	ebb
UREA	2.85±2.04	2.75±1.85	1.62±1.21	1.36±0.9
NH_4	12.13±18.03	9.4±17.39	1.16±1.02	1.32±1.3
NN	13.57±14.21	11.7±12.34	10.02±13.11	10.27±13.37
PO_4	2.44±4.91	2.12±4.3	0.47±0.57	0.67±1.19
SI	13.82±11.48	10.42±8.09	6.64±4.19	6.86±4.61
TSS	75.43±57.5	65.92±50.33	54.94±151.69	57.96±144.83
OSS	15.46±13.16	14.7±13.91	8.76±5.55	8.42±5.5
MSS	60.1±46.53	51.49±40.16	28.23±15.08	27.45±15.22
CHL a	2.63±3.95	2.07±3.65	1.35±0.96	1.43±1.2
DOC	1.39±0.72	1.22±0.65	0.88±0.55	0.94±0.58

Total suspended sediment concentrations ranged between 15-196 mg. l^{-1} at T1, but were rather low at T2 (10,29 - 86,72). Nevertheless, the proportion of the organic matter was nearly equivalent between the two sampling sites and varied between 10% and 34% at T1 and 12% to 34% at T2.

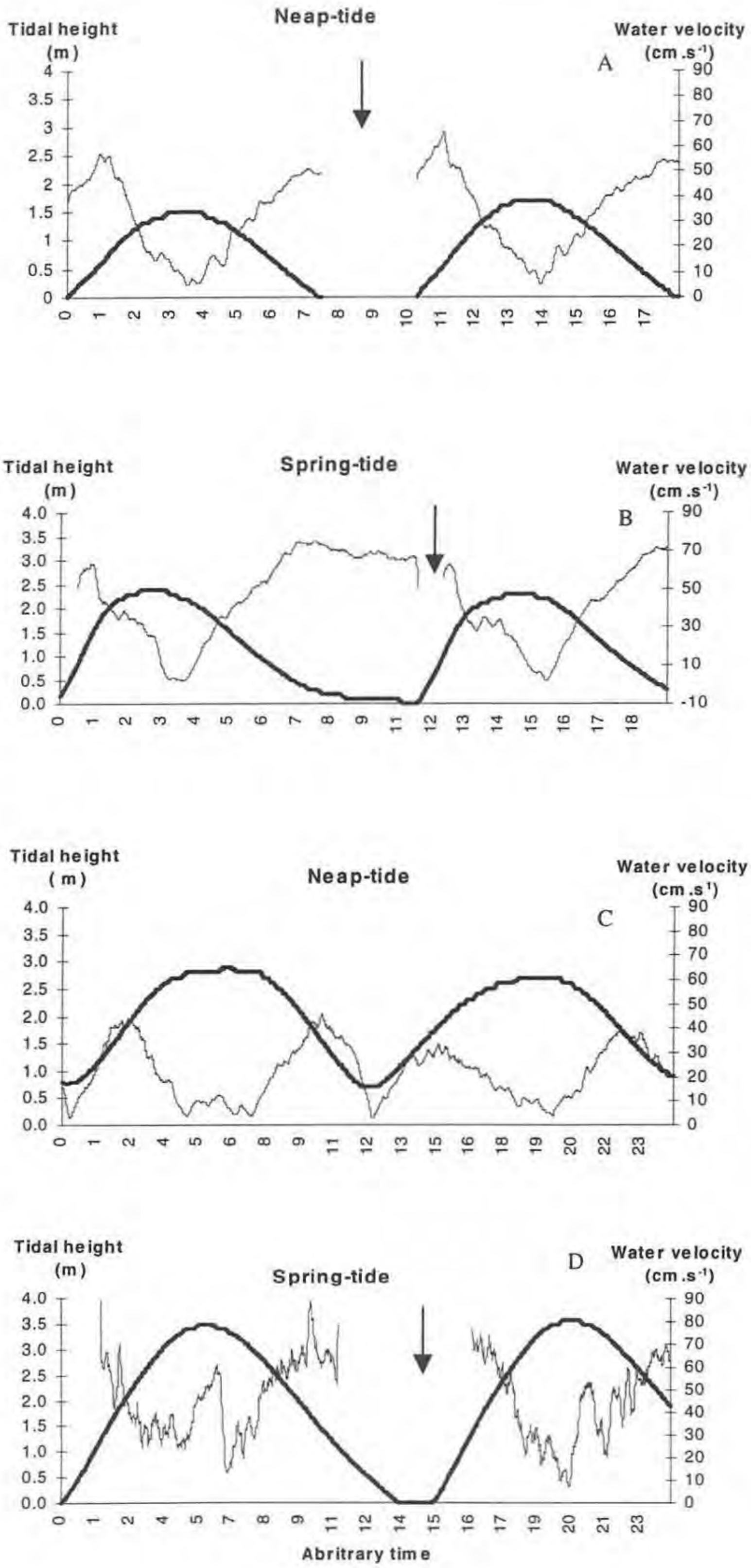

Figure 2 : Continuous Tidal height and water velocity recorded at neap and spring tide for both transect 1, figure A and B and transect 2 figure C and D. Black arrows indicate the interruption of the current sensor record when the water level was near zero. Missing velocity data were obtained from the plotted curve of the water velocity function of the water level.

There were slight differences between the mean concentrations recorded at both flood and ebb tides, T1 showed higher concentrations levels and amplitude mainly for nitrogen-ammonium and suspended sediment with the two mineral and organic compounds. As for T2, there was no significant change in mean nutrient concentrations between flood and ebb tides, the water instantaneous flow will better govern the volume of the nutrient transport between the ebb and the flood tides.

4- Regression estimation

The Kolmogorov-Smirnov test (Table 4) showed a rather good conformity to a Normal distribution for the major part of constituents ($P<0.001$ and $P<0.05$). This probability increased ($P<0.08$) for DOC distribution at both transects 1 and 2 and for urea, Si and NN at T1. Although the latter exhibited a deviation from Normal distribution slightly more than the other constituents ($P<0.001$ and $P<0.05$), by concern for the homogeneity of the method for all the constituents, we also applied the regression estimation method (Spurrier and Kjerfve, 1988) to estimate their annual budget.

Table 4 : Kolmogorov-Smirnov probability test.Three significant levels were indicated ($p< 0.001$, 0.05 and 0.08).

P	< 0,001		< 0,05		< 0,08	
	ebb	flood	ebb	flood	ebb	flood
Transect1						
	UREA	PO_4	CHL a	NH_4	NN	UREA
	NH_4	TSS			DOC	NN
	PO_4	OSS				DOC
	SI	MSS				SI
	TSS	CHL a				
	OSS					
	MSS					
Transect2						
	SI	NN	NH_4	NH_4	MSS	UREA
	CHL a	PO_4	NN	SI	DOC	DOC
		TSS	PO_4	CHL a		
		MSS	TSS	OSS		
			OSS			
			UREA			

The estimation models (Table 5) showed a high coefficient of determination for NN, where more than 50% of the variability was explained mainly by the temperature and the atmospheric pressure. The model incorporated the wind speed, the maximum water column depth, the rain and the atmospheric pressure to estimate the net annual DOC flux with a high coefficient of determination of about 60% at the flood tide for T1 and 56% for T2. TSS, OSS and MSS fluxes were explained essentially by the water temperature and air temperature, Tam and Tom (Table 5).

4.1- Nitrogen

The regression estimation of the annual fluxes of nitrogen yielded statistically significant correlations of NN with predictor variables at T2 (Table 5) and a net export of -9.29 ± 1.11 $g.m^{-2}.yr^{-1}$. However, there was a net import of 0.32 ± 0.23 $g.m^{-2}.yr^{-1}$at T1. There was an import of NH_4 at both sites with a higher amount at T1 (0.87 ± 0.17 $g.m^{-2}.yr^{-1}$) than at T2 (0.27 ± 0.1 $g.m^{-2}.yr^{-1}$). These mean annual values fluctuated within tight confidence intervals, which did not contain zero. A net import of urea was estimated at T1, whereas an net export was estimated at T2 (Table 6).

4.2- Carbon

The annual estimated dissolved organic carbon fluxes (Table 6) exhibited a net export of -87,87 ± 10.27 $g.m^{-2}.yr^{-1}$ at the transect between the bay and the ocean, whereas an import of 3.05 ± 0.34 $g.m^{-2}.yr^{-1}$ was estimated at T1. These values were significantly different from zero.

4.3- Total suspended sediment and organic suspended sediment

There were statistically significant net imports of TSS at both sites (Table 6). The organic matter presented a slight contribution, about 7%, of the total TSS imported at T1, whereas this

Table 5 : The predictor variables used in the annual regression estimation for given constituent fluxes, dependent variables are material constituents. Coefficient of determination R^2, significance level are also indicated. * Regression estimation not significant at 5% level. E (ebb) or F (flood): indicated predictor variables used in regression estimation for a given phase of the tide.

	Urea		NN		NH_4		PO_4		Si		TSS		MSS		OOS		Chl a		DOC	
Transect	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2	1	2
Flood R^2	0,44	0,11	0,51	0,58	0,35	0,53	0,22	0,13	0,10	0,32	0,41	0,42	0,39	0,43	0,47	0,37	0,16	0,13	0,60	0,56
		*					*	*	*									*		
Vt.	E	E				E	F	E	E										F	E
Vp	E			E			F		FE	E				F					FE	
Prmn							E	E												F
Prmn1	F	F	FE	FE			FE			E				E					E	
Tam	F			F	F	E			E		FE		FE		F					
Tom	F	E	FE	E	FE	F		FE		FE		E			E	FE	FE		F	
Pn																		FE		
Pn1		FE				F				F										F
Hn			F		F		E							FE		E				E
Hn1			F		E		FE	E		F		FE	E					FE	E	F
Ebb R^2	0,21	0,34	0,76	0,59	0,33	0,24	0,15	0,47	0,19	0,36	0,37	0,45	0,48	0,37	0,36	0,34	0,26	0,31	0,38	0,58
	*						*		*											

proportion was only about 3% from the ocean to the bay. Although OSS yielded significant correlations at both stations (Table 5), the mean annual value at T2 was not significantly different from zero (Table 6).

Table 6 : Net annual constituent budgets for both transects. Standard errors are indicated. * Model not significant at 5% level.
Units are Units are g m^{-2} yr^{-1}, except for Chl a unit is mg m^{-2} yr^{-1}. Export = -and import = + (no sign shown)

	transect1		transect 2	
	Mean	Standard error	Mean	Standard error
urea	0,86	0,11	-0,41	0,13
NN	0,32	0,23	-9,29	1,11
NH_4	0,87	0,17	0,27	0,1
PO_4	-0,05 *	0,37	-0,20	0,14
SI	-2,05	0,41	-0,69*	2,39
TSS	165,2	10,17	675,05	613,64
MSS	121,58	14,82	514,59	472,33
OSS	12,48	0,7	26,13*	157,54
Chl a	9.8	0.11	-16.63 *	19.41
DOC	3,05	0,34	-87,87	10,27

4.4- Phosphorus and silicate

Phosphorus and silicate exhibited the same patterns of export for the two transects (Table 6). Only the net annual values of silicate at T1 and phosphorus at T2 were significantly different from zero at a level of 5%.

4.5- Chlorophyll a

The regression estimate of the annual fluxes of Chl a at T2 yielded statistically non-significant correlations (Table 5) and a net export which was not significantly different from zero (Table 6), whereas a statistically significant import was estimated at T1.

DISCUSSION

1- Tidal dynamics and bay morphology

The transect at the salt marsh-mudflat interface showed a major import including both particulate and dissolved matter, whereas the bay-ocean interface exported dissolved constituents and imported particulate matter. This heterogeneity within the same area suggested the existence of two functionally different systems. Furthermore, ammonium concentrations were very high within T1 for both the flooding and the ebbing tides, whereas they were rather low at the bay-ocean transect (T2). The ammonium import at T1 exceeded that at T2, which suggested its production occurring mainly at T1 level. The differences of ammonium concentration levels were picked up between the flood and the ebb waters at T1 (Table. 3). At this station, flood waters were more concentrated than ebb waters, whereas they were quite similar at T2 (Table. 3). As both sites were expected to be receiving the same flood water, reaching T1 last, the difference between ammonium concentrations could be explained by the fact that the earthen ponds were producers of ammonium, which left the system with the ebbing tide. Moreover, the high concentrations during flood tide at T1 could be interpreted as some of the ebbing water, which probably did not completely leave the bay and was forced back with the flooding tide to the upper part of the bay. This led to a longer residence time of water constitutents within the system. The hypothesis of significant water residence time at T1 was supported by the water budget (Table 2), which showed a flooding dominated system at T1, whereas a slight ebbing dominance was observed at the bay-ocean transect.

The morphology of the bay with its relatively narrow entrance was susceptible to influence the water budget and consequently the nutrient balance. The asymmetry of the tide may explain these dynamics. In fact, contrary to the rapid incoming flood tide, 3 hours at T1 and 5 hours at T2, outflow during the ebb tide lasts much longer : about 9 hours at T1 and 10 hours at T2 (Fig. 2). Hakanson et al. (1986) suggested that water retention or residence time was hardly linked to the morphology of the system. Furthermore, Persson and Hakanson (1994) demonstrated that more than 90% of the variation of the empirical values of water surface turnover could be statistically explained by the topographic openness. The latter describes the exposure of the coastal area towards the open sea or adjacent coastal area.

2- System maturity

The geohydrologic continuum classification, discussed for East American Atlantic coasts (Dame et al., 1992) and for European systems (Boorman et al., 1994) demonstrated that immature systems import both particulate and dissolved matter, whereas mid-age systems import particulate matter and export dissolved elements. In the extreme case, mature systems export both dissolved and particulate matter. According to this classification, T1 situated at the upper part of the bay, just beyond the land, exemplified an immature stage of development and acted as a sink of constituents, whereas, the bay-ocean transect showed a mid-mature stage with a net export of DOC and DIN. This classification was supported by a geological study already performed in the Fier d'Ars Bay, (Pignon, 1975), which pointed out a north-south gradient where the most recent geological formations were situated in the south and the oldest in the north. However, geohydrologic continuum classification was contrasted by the particulate matter, which was imported in a higher amount at T2, assumed to be ecologically more mature. Nevertheless, the import at T2 fluctuated in a large range: the standard error associated to the annual estimation was approximately equivalent to the mean annual import, which suggested that particulate matter showed high fluctuations over the year. The comparison on an annual scale with the particulate import at T1 may be misleading. Thus, the form of transported constituents involved within the ecological maturity concept is not yet clear. Particulate matter seemed to defy this rule. Moreover, the determination of the degree position in the estuary, hypothesized to approximately the degree of ecological maturity and susceptible to affect subsystem interactions, still remain a relevant question (Childers et al,. 1993b)

3- Particulate matter dynamics

The bay acts as a sink for both mineral, as well as organic particulate matter. Nevertheless, the mineral import reached 76% of the total particulate import at T2 and 73% at T1 (Table 6). The import of the suspended matter with a high proportion of mineral form was associated with an import of NH_4 for both sites. The sorption of such nitrogenous substances into mineral grains is a well known phenomenon resulting from attraction between positively charged NH_4 and negatively charged minerals characterizing fine clay substrate (Hedges and Keil, 1999). According to Allen (1995) this bay belongs to the minerogenic deposition facies, which are distinguished from organogenic mode characterizing *Spartina*-dominated marshes. These two modes of vertical accretion have direct and indirect effect on primary producer communities : since mineral matter accumulation is already available to the autotrophic organisms and will enhance a rapid cell growth characterizing small cell organisms. This effect of the mineral material accumulation was verified by the important communities of microphytobenthos assessed in the Marennes-Oléron Bay (Cariou-LeGall and Blanchard, 1995), which is very close to the Fier d'Ars Bay. Guarini (1998) also pointed out that the autotroph microphytobenthos in this area possesses the same photosynthetic capacity as phytoplankton (1-20 mg C (mg chl a $)^{-1}$ h^{-1}). A net export of Chl a was observed (Table 6), which suggested a net production within the bay. However, the phytoplankton production was limited by the generally high turbidity (Héral, 1985) and could probably not explain the global biomass production. A significant contribution of the microphytobenthos was also demonstrated, 15% of the microphytobenthos biomass contained in the first five centimetres of the sediment was transferred to the water column and contributed to the fluxes from the intertidal zone (Guarini, 1998). The addition of benthic algae to the seston was also noticed in the Zwin salt marsh after the chemical characterization of the flood and ebb organic particulate matter using in-source pyrolysis mass spectrometry (Hemminga et al., 1992). The export of Chl a was coupled with that of silicate, since diatoms were the most significant class of microphytobenthos in this area (Cariou-LeGall, 1995). This trend may be explained by the effects of benthic diatoms recycling within the sediment.

4- Organic matter processing

Import of particulate material was observed in European systems. Unlike those in American ecosystems, where particulate material export was the rule (Lefeuvre and Dame, 1994). This conclusion was once more verified under direct net-flux measurement techniques and using regression estimations. These results have been usually attributed to the tidal dynamics and to the geomorphology of the system. However, the characteristics of the matter may play a key role in the transfer between the coastal zone to the ocean, mainly in which form, dissolved or particulate, mineral or organic, the transport action may occur. Stable isotopic analyses provides clues about the origin and processing of the organic matter. Using this technique, Riera and Richard (1996) demonstrated that phytoplankton and detritus produced by micophytobenthos were the major compounds of the particulate organic carbon (POC) found in the entrance of the Fier d'Ars Bay. Also, the benthic diatoms seemed to be an important source of food in the Mont Saint Michel salt marshes (Lefeuvre et al., 1994).

The C:N ratio probably has a significant bearing on the degradability of the organic matter, that of *Spartina* sp in the Marennes Oléron Bay was about 16.56 (Richard unpublished), which was very high in comparison with those of phytoplankton or microphytobenthos (Table 7). The two latter characterize a more labile, relatively heavy substance: thus, the enrichment of particulate organic matter with phytoplankton strongly increased its degradability (Hemminga et al., 1993) and enhanced its rapid mineralization.

Table 7 : Comparison of the C:N ratio of different sources of organic matter in estuarine salt marshes.

	C:N	Source
Spartina senescent stems	55,3	Middelburg *et al.*, 1997
Spartina anglica	54.52	Creach, 1995
Spartina Litter	37,3	Middelburg *et al.*, 1997
Halimione portulacoïdes	34.78	Creach, 1995
Zostera noltii	11,48	Richard (unpublished)
Diatoms, Peridineans	6,25	Romankevich, 1984

The comparison of nutrient transfers of a recently studied American creek, the Bly creek, characterised by an import of particulate and an export of dissolved material and exemplifying a mid-aged system, with other European systems, generally importing particulate and exporting dissolved matter and illustrating a different stage of geological maturity (Table 8). Showed that, for the Bly creek, the import of NN was less significant than the import of NH4. Moreover, the export of DON was rather higher than that of DIN. These results agree with the observations of Jackson and Williams (1985), which demonstrated a dominance of the dissolved fraction in nitrate poor water, where DON was the dominant nitrogen form. The export of organic matter of both DOC and DON was very significant in the Bly creek in comparison to European systems. These kind of exports characterized a historical poor-nitrogen input and a minimal exposure to mineral surfaces, in that case organic matter was released as dissolved form (Hedges et al,. 1994).

The DOC:DON ratio in the Bly creek exceeded those observed in all other European ecosystems (Table 8). The organic matter produced in Bly Creek took its origin from the degradation of the Spartina altaniflora sea grass bed, as it has been generally assumed that the ebb water leaving the *Spartina alterniflora* dominated the salt marshes of the North American Atlantic coasts was enriched with material from this plant (Teal, 1962). In the Fier d'Ars Bay and in the Mont Saint Michel Bay, phytoplankton and microphytobentos supplied the major organic matter to the water column.

Table 8 : Constituent budgets derived from net transport data of a *Spartina alterniflora* American creek, Dame *et al.* (1991) and of some European systems, Boorman *et al.* (1994) and the present study. Units are g m^{-2} yr^{-1}. Export = -and import = + (no sign shown)

	NN	NH4	DIN	DON	PO4	DOC	POC	C:N	TSS	OSS
Bly creek	0,24	-0,66	-0,42	-11,77	-0,30	-272,30	30,30	23,14	-	-
England	-54	39	-15	-	2.2	-	-	-	-	-
Netherlands	0,30	-0,20	0,10	-1,37	0,20	-20,40	19,50	14,89	347	17,8
Fier d'Ars	-9,29	0,27	-9,03	-	-0,20	-87,87	-	-	675,05	26,13
Mont Saint Michel	-30,00	-4,00	-35,60	-14,80	-	-74,00	15,90	5,00	29970	-

In addition to preconceived ideas concerning the export of dissolved material in all American systems and in the light of the present data, supported by other European studies, particularly in the Bay of Saint Michel and in England (Table 8). The DIN export was particularly assessed in most of the European systems and reached a rather high amount in most of them. A significant contribution of NN was coupled to the DIN export from European systems, whereas NH_4 fluxes influenced DIN export in the *Spartina* dominating creeks. We need to keep in mind that during complete organic matter mineralization, nitrogen is liberated in the form of ammonium salts, which undergo further oxidation to free nitrogen, nitrite and nitrate ions . Thus, in European systems, the salt marshes characterized by a high tidal range (5 m in this area) and an asymmetric tide, were subject to a relatively longer period of exposure, during which NH_4 gradually build up in the sediment (Gouleau et al., 1995; Rocha, 1998) and in the near bottom water (Falcão and Vale, 1995). Rocha (1998) advanced that NH_4 increased by a factor of two during tidal exposure in intertidal sediment. At high tide, a gradient of temperature was established between sediment and water enhancing the export of NH_4 to the water column. This export was estimated at 75% of the total NH_4 pool of the surface sediment (Rocha, 1998). This contributed to a rise in NH_4 concentrations during the flood. Once in the water column, NH_4 was subject to a rapid oxygenation and left the system during ebb tide under NN form.

During the last decade, the sedimentation recording of salt marshes has received considerable attention (Allen, 1990; Van de Plassche, 1991). In most of intertidal European systems the sedimentation rate exceeded those observed on the East American coasts (Table 9). The relationship between the sedimentation rate and the organic matter has been frequently evoked but not yet fully understood. However, many speculations about this relationship have been given, such as the decrease of the residence time of the organic matter at the active sediment-water interface, which is followed by an increase in the deposition rate. The high rates of sedimentation lead to a dilution of the organic content by mineral matter (Stein, 1990). A qualitative and quantitative change of the particulate organic matter was measured after tidal inundation: Consequently, the sedimentation of seston particules during tidal coverage possibly results in selective removal of organic compounds (Hemminga et al., 1992).

Estimation of salt marshes accretion based on $\delta^{13}C$ isotopic analysis performed in both mineral (European) and peaty (American) salt marshes (Middelburg et al., 1997) demonstrated a more important accumulation rate within the mineral systems. Where mechanisms affecting sedimentary $\delta^{13}C$ involved the input of allochtonous organic matter, which included organic matter sorbed on mineral matter, as well as phytoplankton, microphytobenthos and non-local macrophytes. With reference to this data, the major factor governing the $\delta^{13}C$ of sedimentary organic matter in salt marshes is therefore the relative contribution of local plants and other carbon input.

In the same way, the decomposition of the organic matter depends on the age of material deposition. Fresh material, where decomposition is dominated by the the labile fraction, has an age of only a few days to one week old (Van Raaphorst et al., 1992). Further, nitrougous-rich substances in the manner of plankton were prefentially retained on soil minerals owing to the sorption of organic matter into mineral matter. This mechanism helps to immobilize precious fixed nitrogen and also should minimize loss from estuaries (Hedges and Keil, 1999). The present data with a rapid tidal dynamic in addition to a marine labile material input, both combined with a high import of mineral suspended material, seemed to support these conclusions.

Table 9 : Sedimentation rate derived from different salt marsh ecosystems, sources of data are indicated. Unit is mm per year.

	Sedimentation rate	Source
Cape Cod, USA	1	Valiela, 1983
Long Island, USA	2 - 4,3	Valiela, 1983
North Inlet, USA	2	Dame & Gardner, 1993
Fier D'Ars, France	4 - 8	Long, 1975
Marennes Oléron, France	7	Gouleau *et al.*, 1999
Mont Saint Michel, France	50	Berger & Caline, 1991

The present synthesis based on statistical estimation of total annual nutrient fluxes provides an important insight into the relationships between salt marsh processes, subsystem input and estuary-ocean coupling on an annual scale. This technique was statistically sound, with particular emphasis on highlighting water column nutrient processes and water-sediment interactions. However, these relationships are more dynamic over a much shorter time scale than a year, in the manner of the inundation or drainage area, assumed to be constant during our study, whereas its extension depends closely on the hydrometeorological conditions.

As explained above, the origin of the organic matter has a strong effect on sedimentation rate. In the Fier d'Ars Bay, its origin comes from phytoplankton and microphytobenthos. As they are seasonally dependent, a more precise scale than a year should be investigated. Daily resolution introduced by Childers et al. (1993b), consisted of a tidal hydrology model to generate daily areas of each inundated intertidal habitat in the system. Such a procedure accounted for nutrient variability at fine temporal resolution for each subsystem and also indicated the shortcomings of any site-specific extrapolation to the whole estuary conclusions, where a homogenous ecosystem has to be assumed.

CONCLUSIONS

This paper is a new attempt to understand the nutrient exchange between the estuary and the coastal ocean. It provides relevant explanations about the different forms of material transfers in different ecosystems and the factors that may influence these processes. Multi-disciplinary studies, involving the tidal dynamics, the geomorphology of the system as well as the characterization of the food sources, will provide a good insight into the understanding of constituents processing within these ecosystems.

Isotopic analysis is suitable for the characterization of the organic matter and its degradation. It should be performed simultaneously as the nutrient and the material sampling to investigate the probable differences between input and output organic matter from salt marshes.

The residence time of water mass in estuarine areas was correlated to the biological productivity. Areas with slow turn over are often enclosed and generally have a high biological production capacity. The determination of water residence time is of great interest for the comparison of nutrient processing in different ecosystems. The morphology of the system has a strong impact on the water turnover in intertidal areas (Boon and Bryne, 1981). Nevertheless, this parameter was very difficult to determine and generally required hydrodynamic modelling. The relationship between residence time and the morphology system was already conducted in non-affected tidal European areas (Persson and Hakanson, 1994). More attention should be paid in this way in intertidal zones, since digital techniques, using geographic information systems (GIS), for transferring information from standard charts into morphometric parameters expressing various characteristics of the coast, have been developed.

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CHAPITRE III

A GIS-BASED ASSESSMENT OF POTENTIAL AQUACULTURAL NONPOINT SOURCE LOADING IN THE FIER D'ARS BAY, ATLANTIC COAST (FRANCE)

M. BEL HASSEN, J. PROU[1]

UMR CNRS-IFREMER, Centre de Recherche en Ecologie Marine et Aquaculture, B.P. 5, F-17137 L'Houmeau, France.

Ce chapitre a été soumis pour publication dans *Ecological Applications*

soumis le 7 Mai 1999

sous presse

Réf: 99-5130

[1] Present address: IFREMER, Laboratoire d'Aquaculture Tropicale, Unité Hîutre Perlière, B. P. 7004, Taravao-Tahiti

ABSTRACT

Spatial-explicit quantitative dissolved inorganic nitrogen (DIN) and orthophosphorus (OP) loadings were developed from aquatic earthen ponds to the adjacent salt marshes. These loadings were based on water renewal, the concentration of constituents available for all land use and the spatial positions of constituent sources. Potential loading was assessed at every subbasin gauge station by adding the individual contributions of every land use polygon. Kriging was also used to spatialize loading from gauge stations to salt marshes. A grid of concentration values was established specific to the bay stream network using an 8-direction-point model.

Analyses showed a more important concentration near intensive activities from about 15 µM to 30 µM of (DIN) and 2 µM to 7 µM of (OP). These values decreased rapidly near the Ocean, where a dilution of about 75% was noticed for (DIN) and 87% for (OP). Constituent concentrations were quite sensitive to parameter changes and the system reacted more intensively to point source concentration changes than to water drain. The use of regional parameters to assess (DIN) and (OP) loadings from earthen ponds to salt marshes based on land use and topography may be useful when working on a regional scale. Modeling nonpoint sources (DIN) and (OP) from aquacultural land to surface water can be both complex and data intensive. Our goal was to develop a simple model that would account for spatial pattern in topography and land use using Geographic Information System (GIS) databases.

Key words: aquaculture; earthen ponds; GIS; kriging; land use; network; nitrogen loading; nonpoint source; modeling; phosphorus loading; streams; subbasin.

Key phrases: Outlining methods using spatial pattern of land use and topography; land use polygon as unit of analysis: accurate estimation; important constituent concentrations near intensive activities; local effect of intensive aquaculture loading; kriging as geostatistical interpolator: low variance associated to estimations; more sensitivity of the area to point source concentrations than to water drains; predicted concentrations showed realistic values; change of observation scale: conservation of the existing loading signal; method with logistic advantages: adaptation to other study areas.

INTRODUCTION

Major changes have taken place in agriculture over the past 50 years. Many techniques have transformed agriculture from a way of life into a business. Most of these changes occurred during an era when the public took the environment for granted, accepting the practices of agriculture. With the sudden awareness of its effects on the environment, the concept of sustainable agriculture has became indispensable. Aquaculture has entered a similar period of rapid expansion over the last decade. Science and technology have opportunities to address environmental and institutional constraints associated with marine aquaculture, including the interface between the environment and production, and technology to assist with problems of production and environmental impacts.

The Landscape along the West Atlantic French coast is dominated by several aquaculture earthen ponds with a mosaic of different activities ranging, from intensive to extensive aquaculture along with shellfish culture. These hydrological ecosystems are coupled with the downstream aquatic system, i.e. intertidal zone.

Many studies of nutrient cycling through salt marshes demonstrate that they act as nutrient sinks (Jordan and Correll 1985, Jordan *et al.* 1986, Dame *et al.* 1991). Whether these ecosystems trap or release nutrients may depend on how many nutrients they receive from upland ecosystems. Other effects on salt marsh species and ecosystem functioning can be expected with long-term nonpoint source (NPS) loading (Aerts and Berendse 1988, Moore et al. 1989, Morris 1991).

Estimating NPS loading rates to the coastal zone, using detailed hydrologic and nutrient budgets that are based on extensive field samplings, is a laborious task (Winter 1981, Dame *et al.* 1991). This is why there is a need to develop a general method linking upland characteristics to (NPS) nutrient loading, which allows for evaluation of land use change consequences and management practices.

New tools of spatial analysis have been developed and made sensitive to these kinds of problems, e.g. (GIS), which provides a useful interface to link the spatial structure to the hydrological and biological processes and to emerge the spatial pattern that controls the functioning of an identified area. The use of (GIS) can provide a general approach to assess NPS loading. It is of particular interest because of its capacity to :

1- Store information about locations, topology and attributes of spatially referenced objects.(Tsihrintzis *et al.* 1996).

2- Integrate and display different spatial information (Ventura and Kim 1993)

3-Analyze spatially-explicit information and link nutrient loading processes to the spatial patterns of factors in landscapes that control such processes (Levine *et al.* 1993) and to identify critical areas in a very efficient manner.

This paper concentrates on outlining the method used to assess surface (DIN) and (OP) loadings to intertidal zone, based on the spatial pattern of land use and topography.

METHODS

1- Study area

Fier d'Ars Bay (Charente-Maritime, France) is located along the Atlantic coast (Fig. 1). It covers a total of 1258 ha of productive coastal marine ponds connected to the Atlantic Ocean with 750 ha of intertidal mudflat. This area is among the leading sites in France for oyster production. The study area has little topographic relief: the bathymetry, which express the topographic elevation with regard to sea level, ranges from about 3m, it tends to decrease with distance down to the axis of the bay and reachs 1m near the ocean boundary. The sedimentary morphology of this area shows typical drain structures (Crave, 1995), where stream network is easily identified.

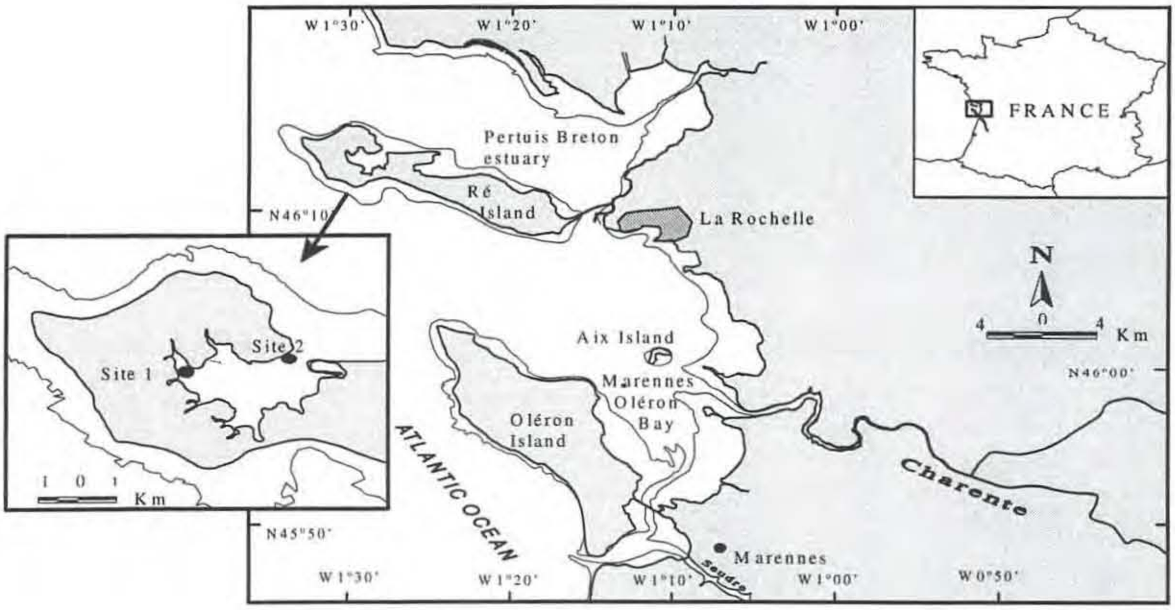

Figure 1 : The Fier d'Ars Bay situated along the French Atlantic coast, belongs to the Pertuis Breton estuary, at the north east of Marennes Oléron Bay. Experimental studies were carried out at the *La Rochelle* experimental laboratory. The position of sampling site 1 and 2 is indicated.

The tide is semidiurnal, with a maximum range of about 5m. Water velocity varies from 90 cm s^{-1} during spring tide to 50 cm s^{-1} in neap tide. Marsh grass only grows in the upper intertidal zone with dominance of the cordgrass *Spartina* spp and the elgrass *Zostera noltii*.

The intertidal bay is subject to different kinds of loading (Fig. 2), which significantly affect its balance. The first component is marine influence carried with tidal flood, the latter suppling water to aquatic ponds. After a residence time within the ponds, which depends on the type of water use, water leaves the ponds at ebb tide.

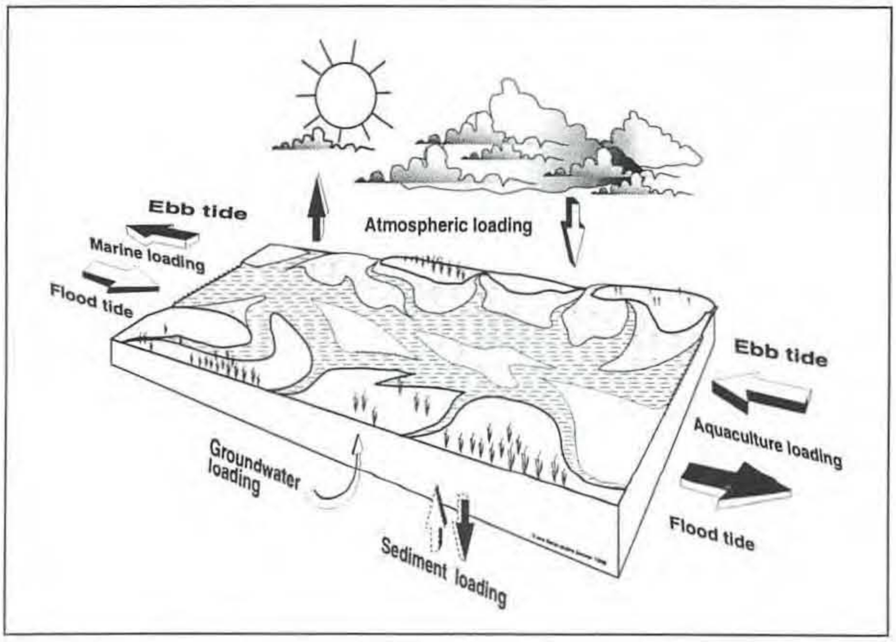

Figure 2 : Diagrammatic representation of the total balance of the Fier D'Ars Bay. The main components of loading exchange are symbolized by arrows, white arrows indicated external inputs to the Bay, black arrows are outputs. Dotted arrows are internal water-sediment process pathways.

Once released into the intertidal zone, the aquacultural loading was mixed, as well as the flood water remaining in the system, with other external loading sources, such as atmospheric and groundwater loadings and loading from internal origin, such as fluxes released at the water-sediment interface. This functioning diagram, with three potential external sources, shows the difficulty to assess separately each source once it enters into the system.

French Atlantic aquaculture ponds are man made structures, built up as a result of the increasing of land height due to the high sedimentation rate (Long 1975). They were first used for saltculture, but later this activity was orientated to fish and shellfish production due to the accessibility of the site and to the tidal dynamics, which allowed the evacuation of wastewater from aquaculture ponds to the coastal zone. Shellfish culture is exclusively a single species oyster culture (*Crassostrea gigas*), the density being limited, according to French standards to 20 individuals per square meter. The extensive aquaculture practiced, as well as shellfish culture in 1m deep earthen ponds, is multispecies with a density of about 100 kg ha^{-1}. The intensive aquaculture is a single species seabass culture (*Dicentrarchus labrax*), characterized by a rather high density of 10 kg m^{-3}. Unlike extensive aquaculture, intensive aquaculture needs fewer extended rearing surfaces, which necessarily require more water renewal within the system, food supply and intensive control of environmental parameters, such as oxygen and temperature. Although extensive culture effort was homogenous throughout the year, intensive aquaculture was more pronounced during spring and summer, whereas shellfish culture marked a break during summer, a period of pond drying which enhanced the mineralization of organic matter within the pond sediment (Gouleau *et al.* 1995).

2- Atmospheric loading

The direct wet DIN and OP deposition flux to surface waters was estimated from the four nearest Fier d'Ars Bay meteorological monitoring stations. The 21-old yr atmospheric input data 1977-1998, were obtained from La Rochelle department public health laboratory (Laboratoire d'Hygiène Départementale de La Rochelle). The direct wet deposition was estimated throughout the year from monthly fluxes, calculated from precipitation chemistry data reported at the four sampling sites near the bay. Using rainfall quantities also recorded, we estimated the total DIN and OP input by interpolating the data to the 750 ha of the studied area.

3- Aquacultural loading

3.1- Water quality data

Nonpoint source loading, as previously defined by Novotony and Chesters (1981), describes the degradation of water quality by materials which are of diffuse origin. During our study, the earthen pond outlet locations were these we associated to the point source loading, whereas instream ungauged locations were nonpoint sources. Water quality data were monitored for each aquacultural activity, which we classified as different "land use" categories. Although different categories precisely consist of distinguished "uses of water", they may also occupy different geographic land locations and therefore influence and orientated them differently. Consequently, they also merit "land use" qualification. The expected mean concentrations and nutrient loading data were previously computed, being further associated to geographical spatial structures to generate source thematic maps.

3.3.1- Expected mean concentrations

The measurement of nutrient levels that occurred during an earthen pond water discharge event was the expected mean concentration. It was the concentration which every land use category was susceptible to discharge out of the ponds. The expected mean concentrations were derived from time series studies, as part of the pond-ecosystem carrying capacity program (IFREMER, 1987; Frikha, 1989; Sornin *et al.*, 1990; IFREMER, 1995). They were already performed in laboratory experimental ponds, close to the studied area and fed with the same *Pertuis Breton* estuary water (Fig 1). Thus, extensive and shellfish culture concentrations were monitored from residual water during discharge events. For intensive aquaculture, measurements were carried out from fish rearing basins water quality survey, as part of a pisciculture wastewater purification program (Hussenot *et al.*, 1995). The time accuracy of the measurement differed within the different studies: monthly to seasonally temporal resolutions were generally practiced (Table 1). The mean concentrations and the exact number of measurements "n" performed in one pond for each time-period specific study are indicated in (Table 1). Water quality was monitored for DIN as nitrate (NO_3), nitrite (NO_2) and ammonium (NH_4): NH_4 concentrations hold a great part of the DIN concentrations, shifting between 17%-95%. The NH_4 contribution to DIN concentrations was particularly pronounced for intensive aquaculture varying between 50 % and 92% (Table 1). This is due mainly to the biodegradation of the lost food and from the decomposition of fish feces.

Different time-period measurements yielded different concentration values (Table 1). A one-way ANOVA analysis performed to test whether time-period study affected expected mean concentration, revealed that only mean OP shellfish culture concentration differed significantly between study "1" and study "2" and "3" (P=0.002) and (P=0.014) respectively.

Table 1 : Averaged annual OP and DIN concentrations (µM) computed from experimental ponds during different time-series studies. n indicate the sample number performed during each time-period study, NH_4 indicate the percentage of NH_4 contribution to DIN concentrations, the daily percentage of water renewal *Rr*i was specified for each activity. Monthly and seasonal accuracy estimation was generally practiced. Sources from each dataset where extracted are also indicated. Further explanations on data derivation are given in the text (see 'Material and methods')
AM indicated arithmetic mean, SD was standard deviation.

	DIN			OP		*Rr*i	Period	Source
	Concentration (AM±SD)	n	% NH_4	Concentration (AM±SD)	n			
Shellfish culture								
1	21.9 ± 18.4	10	-	3.1± 1.51	9	4	Month	Sornin *et al.* 1990
2	14 ± 6.49	10	17-46	0.7 ± 0.48	7	4	Month	IFREMER 1995
3	5.06 ± 1.9	4	42-86	3.9 ± 2.9	4	4	season	Frihka 1989
Extensive aquacultutre								
1	4 ± 4.8	13	24-89	4.9 ± 4.15	13	1.6	Month	IFREMER 1987
2	4.05 ± 1.9	5	52-82	5.45 ± 4.8	5	1.6	season	Frihka 1989
Intensive aquaculture								
1	150 ± 142(**)	9	50-92	11 ± 10.9	9	600	Month	Hussenot *et al.* 1995

Although differences were noticed among specific land use study concentrations, which was the case of shellfish culture, an average value was established for each land use category from the different study yielded concentrations and was assumed to be the expected mean concentration for the specific constituent. This assumption has the merit to consider the maximum time domain extent for the computed variable. Smoothing the time-series concentrations eliminates time as a dimension by restricting the computation to the mean concentrations, such an assumption, qualified by "time average " spatial modeling, as widely discussed by Maidment (1996), where annual resolution was retained as a minimum time interval and was particularly recommended for spatial modeling.

3.3.2- Nutrient loading

A well-defined water exchange between the ponds and the sea characterized these systems. Their large biological productivity and easy access allowed many biological studies and the ponds were often used as models to study biological rate processes (Sornin *et al.* 1990).

Drain was the quantity of water that leaves the ponds at ebb tide. It depends closely on the land use, intensification of rearing (intensive or extensive culture), rearing species and water renewal. The water renewal practiced in this area occurs with agreement to an identified perpetual schedule, which was identical for each land use category. For intensive aquaculture, water was renewed daily. However, for shellfish culture, it was renewed only during high tidal range (tidal constant above 85). To simplify the system, the drain volume was evaluated daily for each land use type. For this purpose, the volume of the drained water occurring during the pond discharge was weighed according to the total volume of the pond. The daily percentage of the renewal volume was derived from the same studies as the expected mean concentrations and then established for each land use category (Table 1). Daily volume of water leaving the ponds was then calculated using the following equation:

$$D_i = Rr_i . H . S \qquad (1)$$

D_i and Rr_i were respectively: daily water volume discharged and daily percentage of renewal volume for land use *i*, *H* was water level within ponds, for extensive and shellfish culture *H* is a constant (1m), for intensive culture *H* refers to the level of rearing basins, this data was derived from the subject matter administration (Hussenot *et al.*, 1995). *S* was the area of each land use polygon, it was derived from the land use map discussed below.

Daily water drain calculated using (Eq. 1) was then multiplied by appropriate expected mean concentration to obtain daily nutrient discharge equations (Eq. 2):.

$$L_i = D_i . C_i \qquad (2)$$

L_i was the daily nutrient loading, D_i the daily water drain, and C_i the expected mean concentration for land use *i*.

3.2- Spatial data

Several digital layers were generated to provide (GIS) data input including, i) land use, ii) point sources or gauge stations, iii) stream network and iv) bathymetry map.

Land use was digitized from (1/10 000) aerial photographs. Its interpretation was verified and updated by field investigations. Results of land use inventory are shown in (Fig. 3). Their attributes were established involving areas for each land use polygon (*S*) and its appropriate gauge station. Attributes, like expected mean concentrations (C_i), percentage of renewal volume (R_i) and mean water levels (*H*) were also linked to the land use thematic map. Spatial calculatoins allowed us to determine daily water drain and nutrient loading based on (Eq. 1) and (Eq. 2).

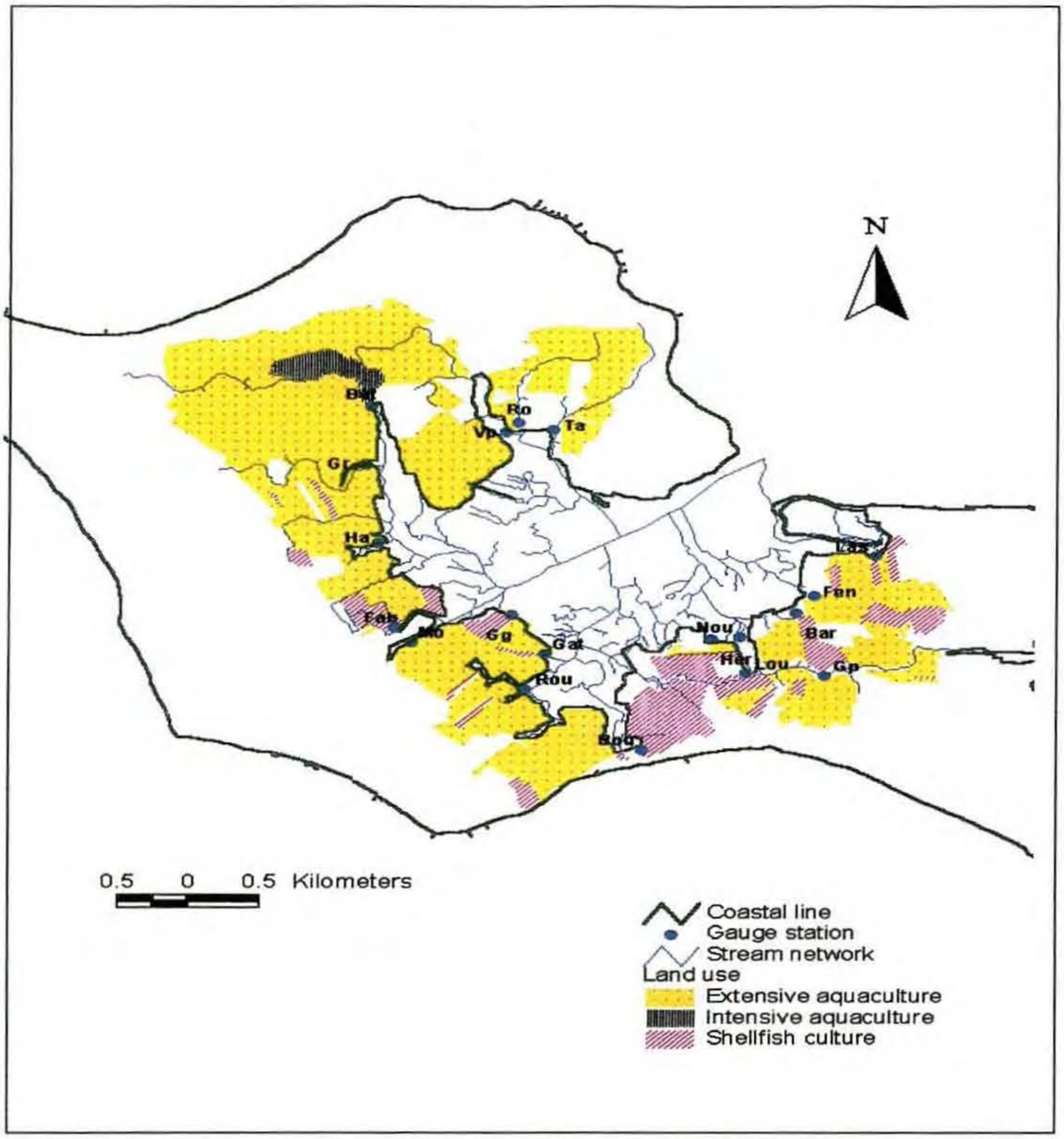

Figure 3 : Land use map distribution within the Fier d'Ars Bay. Labeled points are gauge stations, hypothesized to be point sources. Stream network is represented with thin lines, thick lines delimit the terrestrial band from the coast.

Stream network and gauge stations were also digitized from (1/10 000) aerial photographs. The bathymetry of the Bay was obtained from the DDE (Direction Départementale de l'Equipement) : it is a 20 m x 20 m grid cell Lambert II projected map (projection already used in Middle West France).

The subbasin layer, which showed the zones of contribution of each gauge station, was determined from polygon land use in conjunction with stream network and gauge station geographic locations . It was carried out by assigning a gauge station code to each land use polygon unit, while stream network and gauge station layers were superimposed.

Using aggregation of loading from small homogenous areas, total loading could be obtained to estimate overall water quantity impacts at earthen outfull stations. This approach called "Micro approach" (Ventura and Kim 1993) was used to aggregate nutrient loading and water discharge from the land use polygon unit to the appropriate outfull stations. Water drain and nutrient loading were then stored as gauge station attributes (Table 2).

3.3- Regional distribution of loading

An interpolating technique was used to obtain the regional distribution of nutrient loading and stream flows within the bay using gauge stations as known sources. The geostatistical kriging method based on the linear minimum variance unbiased estimation was introduced (Kitanidis 1983). It seems to be a suitable technique already used in hydrological studies. It is of particular interest because of its ability to estimate stream flow values using existing stations and its capacity to deal with irregular distributed samples

Table 2 : Gauge station attribute table. Area (ha) is total earthen pond area drained into the lower gauge station, Drain was expressed in m^3d^{-1}, DIN and OP in mole d^{-1}. Values were the sum of land use polygon unit attributes for each subbasin as calculated from the "aggregation method" (see 'Material and methods'). Total loading is also indicated (last line).

Gauge station	Area	Drain	DIN	OP
Ta	58.2	4517.5	220.0	22.1
Ro	55.4	2577.3	14.0	12.6
Vp	59.3	5814.1	16.4	14.8
Bat	303.7	184387.4	25312.2	1907.1
Gr	134.1	10138.2	61.8	46.7
Ha	59.5	2553.5	19.9	9.9
Fab	46.1	7137.2	77.1	8.6
Mo	29.4	621.8	3.4	3.1
Gg	8.1	1621.2	24.2	1.1
Gat	12.0	2468.0	17.0	10.5
Rou	145.7	4831.9	44.7	15.5
Bou	50.5	6132.2	91.4	4.3
Her	30.9	4140.4	58.1	4.4
Nou	21.8	1271.7	6.9	6.2
Lou	17.4	2703.7	35.4	4.1
Gp	32.6	2429.7	17.3	10.1
Bar	15.4	606.9	3.3	3.0
Fen	13.0	2592.8	38.6	1.8
Las	42.2	2661.4	33.6	4.5
Total	1135	249207	26095	2090

(Chang and Sun 1997, Danielsson *et al.* 1998). Contrary to other methods, it gives an estimation of the variance for any point in the kriged domain.

Given n measurements of potential water drain or nutrient loading, at spatial locations $x_1 \ldots x_n$; an estimator using a linear combination of n variables was considered:

$$Z^*_0 = \sum_{i=1}^{n} \alpha_i Z(x_i) \tag{3}$$

Z^*_0 was an estimation of the true value Z_0 and α_i was the weight of the observed value at location x_i.in the Fier d'Ars Bay. The aim was to find a set of weights that gave the best estimation, so that the estimation was unbiased with a minimum variance. The first condition can be expressed as follows:

$$\sum_{i=1}^{n} \alpha_i - 1 = 0 \tag{4}$$

and the minimum variance condition:

$$E\{[\sum_{i=1}^{n} \alpha_i Z(x_i) - Z_0]^2\} = \min \tag{5}$$

The unknown weights αi (i=1....n) are identified with respect to the two constraints 4 et 5, conditionally to a given model of a semi-variogram γ (h), which was the expectation of $(z(x_i) - z(x_i+h))^2$ given by

$$\hat{\gamma}(h) = \frac{1}{2n} \sum_{i=1}^{n} (z(x_i) - z(x_i + h))^2 \tag{6}$$

where n(h) is the number of pairs observations $(z(x_i) - z(x_i+h))$. The graph of $\hat{\gamma}$ (h) versus h is a powerful tool for the analysis of the random function Z(x): it provides information of the short scale heterogeneity and on the distance beyond which spatial correlations between samples become negligible. The experimental semi-variogram $\widehat{\gamma}$ (h) has to fit the classical analytical model, ('spherical', 'gaussian' or 'exponential'), ensuring the positiveness of the variance ($[\sum_{i=1}^{n}\alpha_i Z(x_i) - Z_0]^2$). This latter constraint stresses the importance of semi-variogram model choice, consequently the following (*R* and *M*) criteria derived from cross-validation, are used to assess the quality of the fit of $\widehat{\gamma}$ (h) to the theoretical function γ(h).

$$R = \frac{1}{n\sigma^{*2}}\left(\sum_{i=1}^{n}(z(x_i) - z^*_{-i})\right)^2 \quad \text{and} \quad M = \frac{1}{n}\sum_{i=1}^{n}(z(x_i) - z^*_{-i}) \qquad (7)$$

$z(x_i)$ is observed value and z^*_{-i} prediction for this value after removing the i^{th} sample, σ^{*2} is the mean kriging variance. Clearly, *R* should be close to 1 whereas *M* close to 0. We should stress that these criteria are not designed to perform statistical tests, nevertheless they help to identify lack -of- fit causes when checking semivariogram models.

The variance of the kriging estimation is given by

$$\sigma^2 = -\sum_{i=1}^{n}\sum_{j=1}^{n}\alpha_i\alpha_j\gamma(x_i - x_j) + 2\sum_{i=1}^{n}\alpha_i\gamma(x_i - x_0) \qquad (8)$$

Where variance can be interpreted as the measure of the precision associated with the kriging estimation. When x_0 coincides with a data location at x_i, the system solution yields $\alpha_i=1$, which produces minimum variance close to known locations.

An Avenue script (Arcview language) was developed to perform this task. This program presented advantages to run on an Arcview interface (software already used to perform GIS data input). Thus no data transfer was required and it also overcame projected data problems. For phosphorus and nitrogen loading, kriging was performed on the Log_{10} of the original value. Kriging the logarithm also had the advantage that predicted values of the original variable could never be negative.

Kriging spatialized hydrological data for the whole of the bay, in-stream and off-stream locations, while preferential paths of water existed along the stream network and depended closely on the bathymery of the area, thus attempting comparison with measured values at this stage of the analysis will therefore be a misleading estimation. Nevertheless, we could appreciate the accuracy of the kriging interpolation with the associated error estimation.

The Burning-in process (Saunder and Maidment 1996) was first used, followed by an 8-direction-point model and its derivative function, i.e. weight accumulation (Maidment 1993). These procedures, discussed below, had the merit to cumulate loading within the stream network and to direct the water movement following the steepest slope.

3.4- "Burning-in" streams process

This process consists of raising the elevation of all the cells, except those that coincide with the digitized streams, thus water is forced to remain in the streams once arrives there. However, the water is not forced to flow towards the streams. This process consisted in storing a bathymetry value in the stream cells and adding an arbitrary elevation increment only to the off-stream bathymetry cells, while saving the in-stream cells. Such a process was equivalent to digging a channel within the stream network where loading was constrained to remain.

3.5- Accumulation loading in the stream network

A standard methodology was applied to accumulate loading within the stream network. It consisted first of all in filling the sink of the *"Burning-in"* bathymetry, i.e. increasing the elevation of the points that are fictitious pits, secondly the flow direction, following the 8-direction-point model (Maidment 1993), was determined and the flow accumulation was calculated using the flow direction grid to count the number of cells upstream of each particular cell, which was then stored as the value for that cell. A derivative function of the flow accumulation, i.e. the weighted flow accumulation, was finally performed and the sum of the values (in this case, nutrient loading and water drain), relative to the cells upstream of a particular cell, was stored as the value of that cell.

By dividing both weight accumulation loading and weight accumulation water discharge by the flow accumulation grid, referring to the following equations, we estimated nutrient and water loading that took into account both the spatial distribution of stream flow value and the network design.

$$l_j = L_j / N_j \qquad (9a)$$

$$d_j = D_j / N_j \qquad (9b)$$

l_j was the nutrient loading and d_j water discharge drain within the j^{th} cell, L_j and D_j were respectively, the cumulative nutrient loading and water drain. N_j was the number of cells upstream the j^{th} cell.

3.6- Predicted concentrations

Predicted constituent concentrations at various in-stream locations resulted from the mixing of all constituent flow, estimated from upstream of the particular location. This mixing process was approximated by dividing the accumulated loading at each cell by the accumulated water drain also co-occurring at this location. We can mathematically represent this idea using the following equation.

$$C_j = L_j / D_j \quad (10)$$

L_j and D_j were respectively daily cumulative nutrient and water loading, C_j was the average concentration expected at the j^{th} cell.

3.7- Sensitivity analysis

Sensitivity analysis was carried out to examine the dynamics of the predicted concentrations to a change of different rates (±10%, ±25% and ±50%) of point source water drain volumes and nutrient concentrations. The ratio of new cumulative nutrient loading under initial cumulative loading, k_a, was calculated to test whether fluctuation of source loading was conservative. Difference between initial and new predicted concentration under initial concentration, K_b and K_c, relative to water drain and concentration changes, were also established as criteria of analysis to test the sensitivity of the predicted concentration to a shift of these control parameters.

4- Validation

In order to compare constituent predicted concentrations to field observed concentrations, two sampling sites were investigated monthly from April 1997 to March 1998. The first (Site 1) was situated in the upper part of the bay, whereas the second station was close the outlet boundary of the bay (Site 2), (Fig. 1). Discrete water samples for material concentration measurements were collected in 2l acid-washed bottles, every 30mn during the ebb tide. Samples were analyzed for nitrate + nitrite, ammonium and orthophosphate on a Skalar continuous flow analyzer (Strickland & Parsons 1968). At any given sampling time instantaneous water fluxes were calculated as a function of water level, water velocity (measured using multi-parameter water quality monitoring instrument (EMP 2000, Martec)) and the microtopography of the sampling section. Instantaneous water flux (Kjerfve 1982) multiplied by constituent concentrations from the water sample was thus instantaneous flux (Eq 11).

$$F = \int_A UCdA \tag{11}$$

F is flux per unit time, *U* is water velocity, *C* is nutrient concentration and *A* is the area of the cross section. Let c_t and v_t be respectively instantaneous water concentration and volume estimated through the *A* section. The estimation of the resultant observed concentration, C_r was thus the sum of the mass constituent under the sum of water volume (Eq 12).

$$C_r = \frac{\sum_{t_0} c_t v_t}{\sum_{t_0} v_t} \tag{12}$$

The observed resultant concentration C_r, was thus the concentration assumed to be representative of the sampled month.

RESULTS

1- Estimated aquacultural loading

Spatial distributions of daily nutrient loading and water drain are given in (Fig. 4). For the both constituents, a more important loading was observed near intensive activities. These values decreased rapidly near the Ocean boundaries. Interpolation variance (Fig. 4B, 4D, and 4F) showed low values within the upper part of the bay and tented to decrease near the ocean. This variance, which decreased with distance up to the axis of the bay, was particularly due to the decrease of the gauge station numbers toward the ocean: as usual, the highest accuracy is achieved in the neighborhood of the known sources, owing to the variance proper definition (Eq 8). The result of cross validation given in (Table 3) showed that M is near its expected values ($0.04 < M < 0.09$), which illustrates an accurate estimation for all of the constituents. As for DIN, $R = 2.02$ which attests to the poor results from the fit to DIN data. This is particularly due to the high DIN loading recorded in one station, station *Bat,* (Table 2) compared to the others.

Table 3 : The cross-validation criteria results. *M* should be close to 0 and *R* close to 1 (see 'Material and method'), σ^{*2} is the mean kriging variance also used for *M* and *R* calculation. R=2.02 attest to the weak result from the fit to DIN data (see 'Results').

	Drain	DIN	OP
M	0.09	0.04	0.04
R	0.94	2.02	0.78
σ^{*2}	0.31	0.6	0.4

The grid of concentration values was established corresponding to the bay stream network (Fig. 5). Values from the predicted total constituent concentration grid are fitted into these cells, which correspond to locations along the stream network, where all loading was cumulated. Concentrations showed a decrease from the bottom to the outlet of the bay. DIN ranged from $15\mu M$ to $30\mu M$ near (site1) and were established between $8\mu M$ to $10\mu M$ at the end of the bay. OP fluctuated between $2\mu M$ to $7\mu M$ near intensive activities and dropped immediately between $2\mu M$ and $1\mu M$ to establish around $0.5\mu M$ in the main channel leading to the Ocean.

Local effects of intensive aquaculture loading were rapidly attenuated through the stream network, where a decrease of about 75% of the DIN concentration was observed between the highest values, carried out in the bottom of the bay and the lowest concentrations estimated at the end of the bay. A similar pattern was exhibited by OP concentrations with a decrease of about 87% following the concentration path between the site1 location to the Ocean boundaries. As previously explained in the "sampling site" section, when discharged into the intertidale zone, the aquacultural loading was mixed with flood water remaining in the system which probably contributed to the shift of constituent concentrations. However, 1 to 2 hours after the slack tide, constituent concentrations became more sensitive to earthen pond loading (Fig. 6). As for sites near the ocean boundary, NPS loading were more sensitive to dilution and became difficult to distinguish from flood waters.

Sensitivity results (Table 4) showed that the estimated NPS loading varied in the same range as the point source loading. K_a expressing the ratio between new (after increase or decrease of point source loading) and initial cumulative loading, was calculated to test

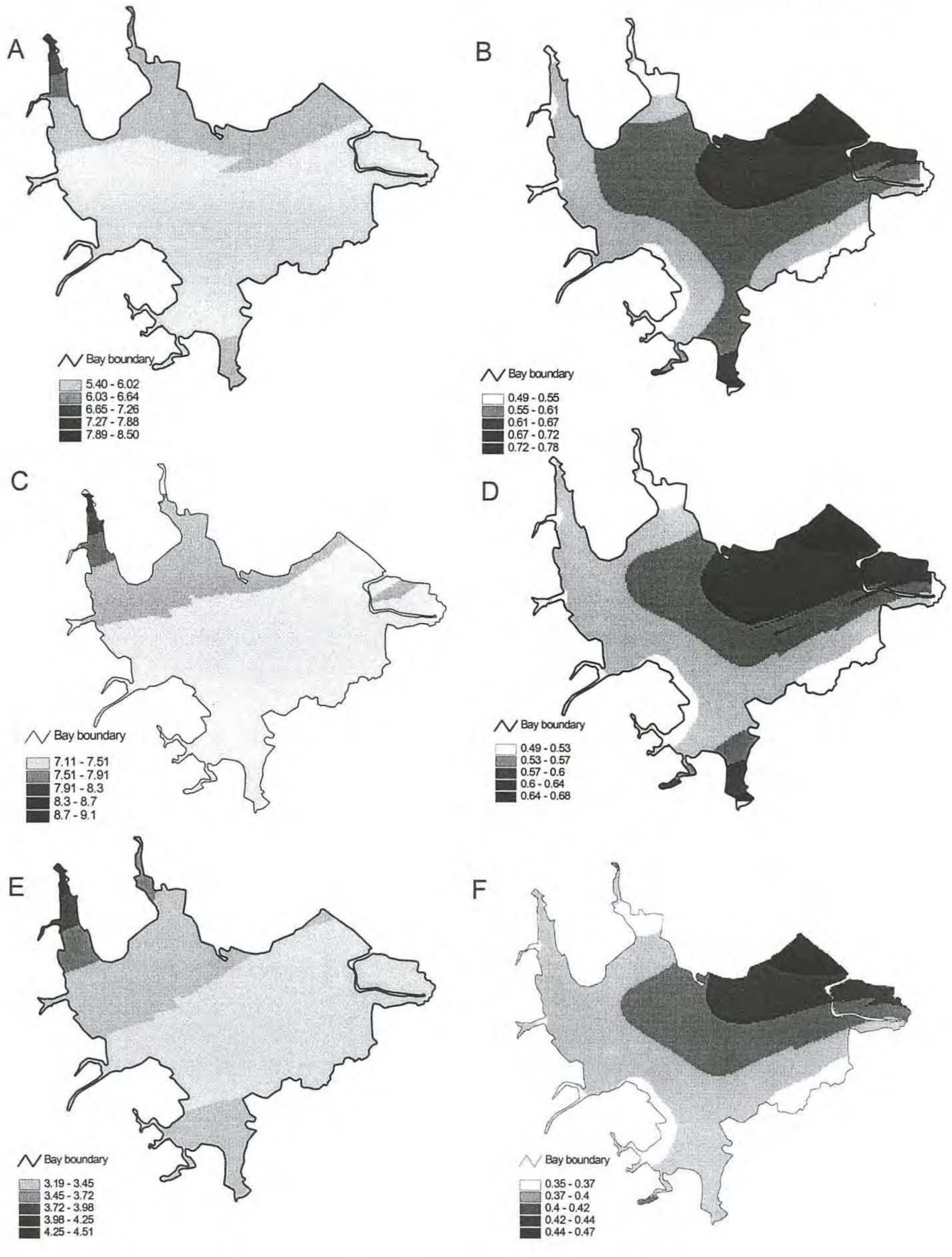

Figure.4. Kriged maps of potential OP and DIN loading (A and C) and water drain (E) with the estimation of their respective standard deviations (B, D and F), variables were Log10 of daily loading, computation was performed over a 20x20 m regular interpolation grid covering the whole bay.

whether the loading variations were conservative. This means that a variation of point source loading would be entirely transmitted to the NPS loading without any decay . K_a showed values close to the original variation rates (10%, 25% and 50%) (see table 4 columns K_a, *AM*). This presumed that this assessment method, basically not considering loading decay, occurred without any bias.

Table 4 : Sensitivity of DIN and OP to a change of constituent loadings, $K_a = (L_{j\ new} / L_{j\ initial})$, where L_j was the cumulative nutrient loading in the j^{th} cell, to a change of drain and concentration sources, respectively K_b and K_c, $K_b = [(C_{j\ new}-C_{j\ initial})/ C_{j\ initial}]$, C_j was the predicted concentration within the j^{th} cell. K_b and K_c have the same expression. *AM* indicated arithmetic mean and *SD* the standard deviation.

	DIN						OP					
	Ka		Kb		Kc		Ka		Kb		Kc	
	AM	*SD*	*AM*	*SD*	*AM*	*SD*	*AM*	*SD*	*AM*	*SD*	*AM*	*SD*
+10%	10	8,28	-11	6,26	8,8	19,33	7,36	9,62	-12,92	21,99	20,49	40
-10%	-10,4	6,37	10,23	7,13	-11,9	15,58	-12,8	8,13	32,99	45,88	-16,38	24,49
+25%	25,06	9,4	-21,87	5,55	24,13	21,85	21,48	6,55	-19,13	23,67	43,43	46,96
-25%	-25,25	5,71	33,57	8,75	-28,38	12,84	-27,41	3,64	57,39	44,45	-25,95	28,37
+50%	50,04	11,22	-36,09	4,7	49,94	25,89	46	7,7	-26,68	58	72,4	40
-50%	-50,3	3,67	103	12,2	-91,45	14,1	51,8	2,59	154	56,05	-83,08	23

Predicted concentration variations, as assessed using the ratio between the difference between new and initial concentrations, under initial concentrations, following on drain and concentration source fluctuations, respectively K_b and K_c, were very sensitive to water drain

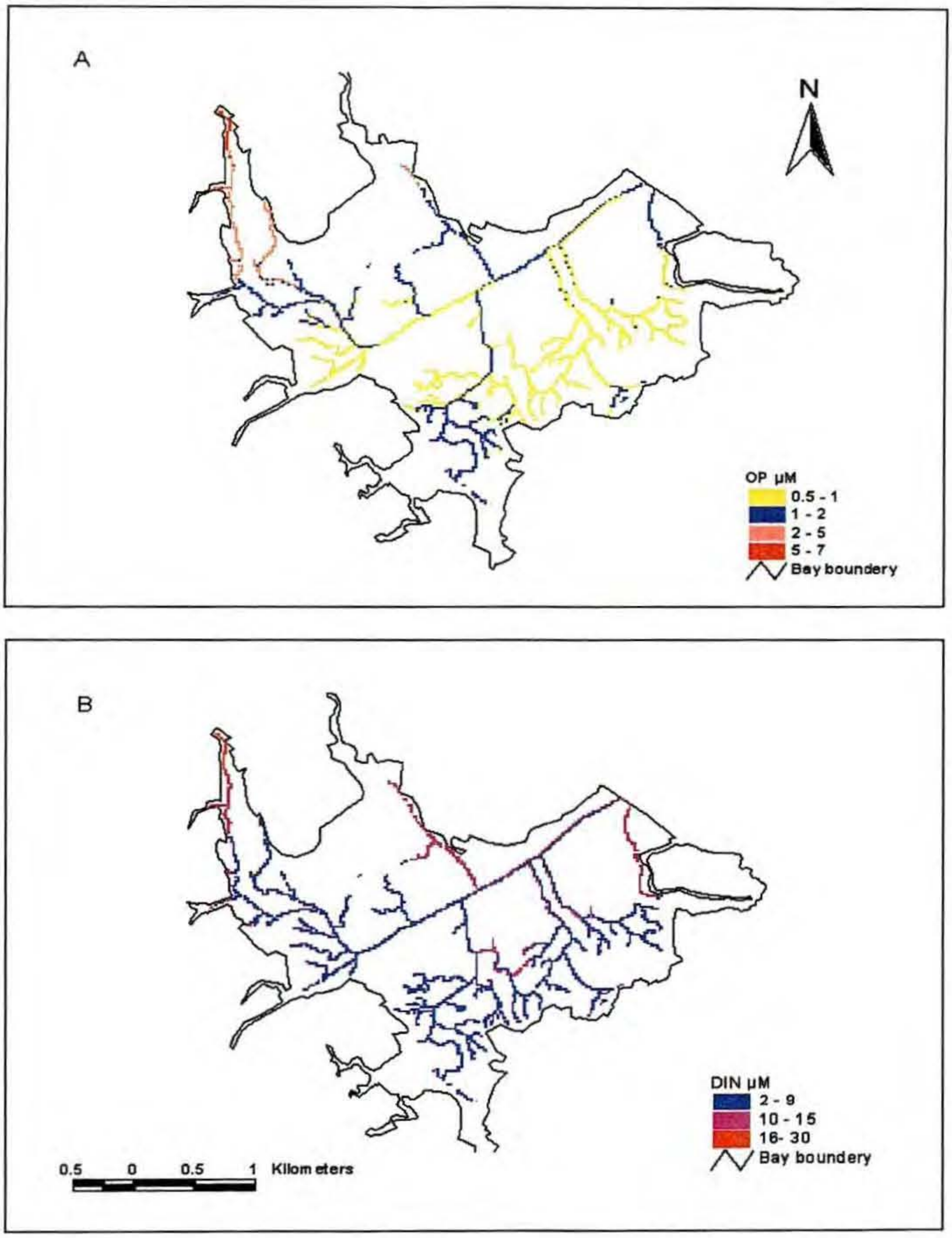

Figure 5 : Predicted concentrations of OP (A) and DIN (B) estimated within the stream network. Shade of colors indicate the concentration levels.

Table 4, columns K_b, *AM)*. Change in source concentrations do not imply an equal variation rate of DIN predicted concentrations, the latter being more sensitive to a diminution than to an increase of source concentrations (see table 4, column DIN, K_c, *AM*).

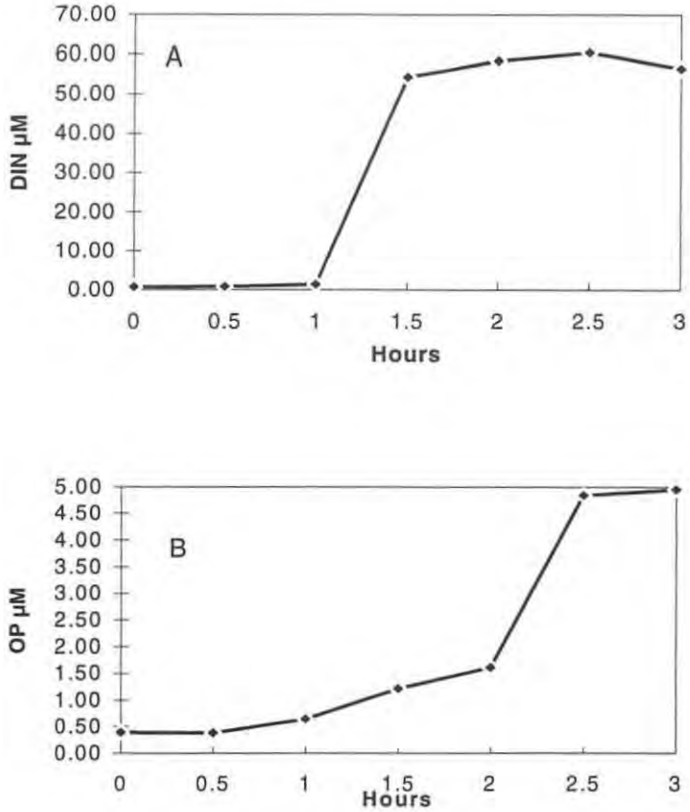

Figure 6 : Concentrations survey close to one gauge station, measurements are for DIN and OP, carried out every 30 mn at the ebb tide. The results show fairly homogeneous water concentrations when drained out of the aquacultural ponds.

2- Recorded atmospheric loading

There was a pronounced peak of DIN atmospheric deposition during 1983-1985 (Fig. 7), a sharp drop was recorded afterwards. Although little increase was shown during 1991-1993, DIN fluxes exhibited a continuous decrease from 1993 to 1997, followed by a slight increase in 1998. Four periods characterize OP atmospheric depositions: from 1977 to 1980 they peaked up to 4 T yr^{-1} followed by a sharp drop from 1980 to 1985. After a small increase until 1988, a continuous decrease was noticed where depositions shifted under 0.5 T yr^{-1}. The DIN atmospheric input into the Fier d'Ars Bay (varying between ~ 12 T yr^{-1} and less than 1 T yr^{-1} in the last eight years (Fig.7), shifted in a rather small range (1.4% to 9.1%) according to aquacultural loading (132 T yr^{-1}) (Table 2). The contribution of atmospheric deposition to the external DIN input had severely decreased since 1987 and represented less than 6% of potential aquacultural input.

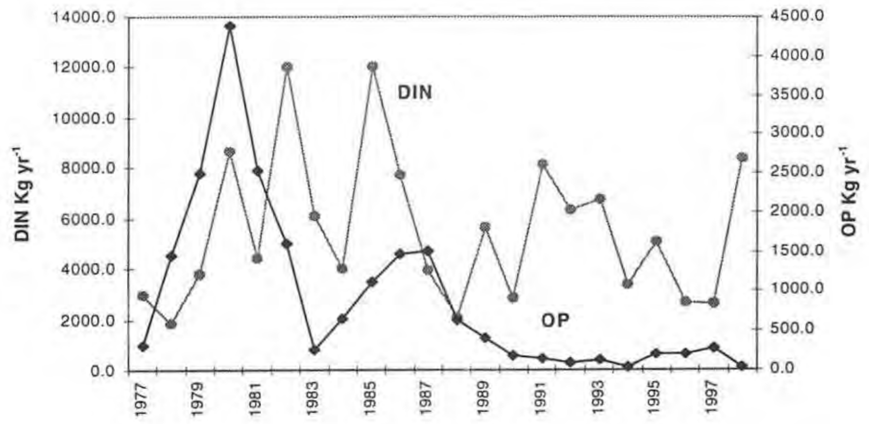

Figure 7 : Wet atmospheric deposition of DIN and OP onto the 750 ha Fier d'Ars Bay. Data, over the last 21-yr, were from the four nearest monitoring stations.

DISCUSSION

1- External loading sources.

Surficial, atmospheric and groundwater loading were identified as the three potential sources of external nitrogen input along the French Atlantic coasts (Rimmelin *et al.*, 1998). Surficial loading origins differ according to the studied site, the most frequently assessed were wastewater and fertilizer, mainly carried by rivers (Jordan *et al.* 1991, Scudlark and Church 1993, McClellan *et al.* 1997, Valiela *et al.* 1997, Rimmelin *et al.* 1998, Carpenter *et al.* 1998). Atmospheric deposition in the Fier D'Ars Bay exhibited a decreasing tendency during the last decade. There are also indications of decreasing precipitation nitrogen concentrations after the mid- 1980 in certain sites of north America and Europe (Dillon *et al.* 1988, Fricke and Beilke 1992). Stressing that atmospheric deposition recorded here was only wet deposition, dry deposition was unfortunately difficult to assess, thus generally assumed to be twice as much as wet deposition (Valiela *et al.* 1997).

The Fier d'Ars Bay, receiving between 5.62 and 21.75 Kg N ha^{-1} yr^{-1} during the last decade. As well as the major anthropogenic component in the European Atlantic coast, presenting atmospheric DIN fluxes of about 15-25 Kg N ha^{-1} yr^{-1} (Rimmelin *et al.* 1998), atmospheric fluxes are lower than in regions affected by heavy acid rains. Although moderate, the bulk atmospheric deposition (wet and dry) has fluctuated, according to aquacultural loading, less than 12 % since 1987, and represented only 4 % in 1997. This was below the range of 20% to 40% of DIN input to coastal water from atmospheric origin computed from different ecosystems (Paerl 1997). There was much less concern about phosphorus atmospheric inputs, comparing to nitrogen atmospheric deposition: they were rather variable between years, representing only 0.2% to 6% of phosphorus NPS aquaculural loading since 1987.

Groundwater nitrogen inputs, not assessed during the present study, have not been extensively characterized and quantified throughout the world. Nevertheless large groundwater inputs were recently noted (Moore 1996) and may be comparable to atmospheric loading in certain regions (Paerl 1997). The lack of data unables us to reach conclusions on groundwater loading, however recent work in the Arcachon lagoon, just south of Fier d'Ars Bay, more urbanized and subject to intensive agricultural activities than the Fier d'Ars Bay, showed that groundwater yields a slight contribution (less than 1%) to the global DIN lagoon inputs (Rimmelin *et al.* 1998).

2- Nonpoint aquacultural loading

The previous paragraph stresses the significant contribution of NPS aquacultural loading as a main source of both nitrogen and phosphorus loading to the Fier d'Ars Bay. Their quantification and particularly their spatially different impacts are therefore of major concern.

This study is a steady state model to estimate spatial distribution of nonpoint source loading from different aquatic ponds to the adjacent system. For this, three main steps were performed. "Micro approach" was used to move water and nutrient loading through the landscape and cumulate them into gauge stations. The geostatistical method was also used to interpolate loading from gauge stations to the adjacent bay. Finally, an 8-direction model and its derivative function, i.e. weight accumulation, were performed to cumulate water within the stream network, where constituent concentrations were assessed.

"Micro approach" based on individual land use polygon as the unit of analyses, took into consideration the high degree of land use heterogeneity within the study area. Aggregation of nutrient loading from the land use polygon unit to gauge station outlets may occur without decay or deposition, mainly for the reduced flow network length within each subbassin (Fig. 3).

Spatial distribution of water flow, DIN and OP loading, performed using kriging, gives an accurate estimation of the distribution of these parameters, mainly for the low variance associated with the estimation. This approach should normally be applied because the spatial structure of the data, as represented by semi-variogram modeling (Eq 6), will be integrated into the estimation (Corwin *et al.* 1997). Moreover, it was rapid and helped to verify the precision of the approximation at any unknown points (Villeneuve *et al.* 1979), as shown with the Kriging error maps (Fig. 4)

Sensitivity results revealed that the shift of predicted concentrations to a change of parameters, already used to express nutrient loading and water drain, lends support to the use of generalized parameters, when working on a landscape scale. It also demonstrated that constituent concentrations were sensitive to drain changes, with different answers depending on the kind of the constituent and on the direction of the change (increase or decrease). Controlling point source concentrations, as described by K_c, seem to be more efficient than water drain volume, illustrated by K_b variation: in fact it engenders a more important decrease of predicted NPS constituent concentrations (Table. 4). Nevertheless, this scenario appears to be less practical and not suitable to field work, because of the lack of continuous concentrations survey and to the excessive cost of appropriate water treatment installations. Controlling drain quantities is more realistic and can be integrated more easily into a managing plan.

In many case studies, no validation was associated to the estimation of concentration models, mainly for wetland, due to difficulties with field measurements (Poiani *et al.* 1996). In fact, wetlands often do not have an integrating body of water, so no analogy with stream outlets can exist. Internal geochemical and biological processes along the paths, such as nitrification, denitrification and nitrogen fixation (Joye and Paerl 1994, Capone *et al.* 1997) can have profound effects on water quality (Pionke and Lowrance. 1991). Nevertheless, these internal processes imparted low contribution to the nitrogen cycle along the French Atlantic Arcachon lagoon (Riou 1999). Inputs of groundwater can also enter the salt marshes through diffuse flows, which are very difficult to measure (Sielgel 1983, Shedlock *et al.* 1993). Furthermore, mixing processes due to other source loading, such as tidal loading, can alternatively increase or decrease observed concentrations depending on flood water levels. In our study, straightforward adaptation of techniques was also used to validate NPS loading from earthen ponds to the adjacent intertidal area. We should notice that the topography of the area showed clear stream network design, where drain was conducted through identified channels, so flux calculation can normally be performed.

Denitrification, demonstrated to decrease the transport of nitrogen to downstream or offshore waters, was considered in many ecosystems as a significant sink for external nitrogen inputs. In estuaries and nearshore coastal marine systems, denitrification often removes between 40% to 50% of external DIN input (Seitzinger 1990). The availability of NO_3 is an important factor controlling the denitrification rate in many ecosystems. In our case, the main DIN loading occurs as NH_4 (Table 1), thus nitrification must previously happened to ensure further denitrification. Nevertheless, in contrast of freshwater systems, where almost 80% to 100% of the NH_4 production is nitrified and subsequently denitrified, in coastal marine sediment only a portion of 40% to 60% of NH_4 is nitrified and denitrified

(Seitzinger 1990). This was mainly explained by the reduced availability of exchangeable NH_4 in salt sediment, because of competition for cation exchange and ion pairing (Gardner *et al.* 1991, Seitzinger *et al.* 1991). Although denitrification has not been particularly assessed in the Fier d'Ars Bay and considering the above explanation, combined with the tidal flux, which rapidly evacuates NPS loading out of the area, we do not expect high denitrification values. The estimation we hold was from the Marennes Oléron Bay, south east of the Fier d'Ars Bay and processing close sediment characteristics (Fig.1), where a rate of 37 μmol m^{-2} h^{-1} was assessed during spring, whereas it was almost zero during the summer (Feuillet-Gérard, unpublished manuscript). Applying an average annual denitrification rate (18.5 μmol m^{-2} h^{-1}) to the NPS loading (143.5 μmol m^{-2} h^{-1}), yields a removal of 13.2% of DIN loading by denitrification, which was rather low compared to the range of 40% to 50% advanced by Seitzinger (1990).

Comparison between observed and predicted concentrations, the latter were captured from the observed location stream segment cell values, showed that OP observed concentrations matched predicted values in site1. The most frequent observed values varied less than 10 μM, which was the case of those predicted (Fig. 8A). The same trend could be observed for DIN, i. e. frequent observed values fluctuated between 20 and 30μM, whereas predicted concentrations shifted slightly less than 20 μM to 30 μM (Fig. 8B). As for site 2, observed OP values frequently fluctuated between 0.5 μM and 1.5 μM, whereas predictions were almost exclusively close to 1 μM (Fig. 8C). For DIN (Fig. 8D), observed concentrations have a large scatter, frequently observed less than 20μM, whereas predictions were close to 10μM to 15 μM.

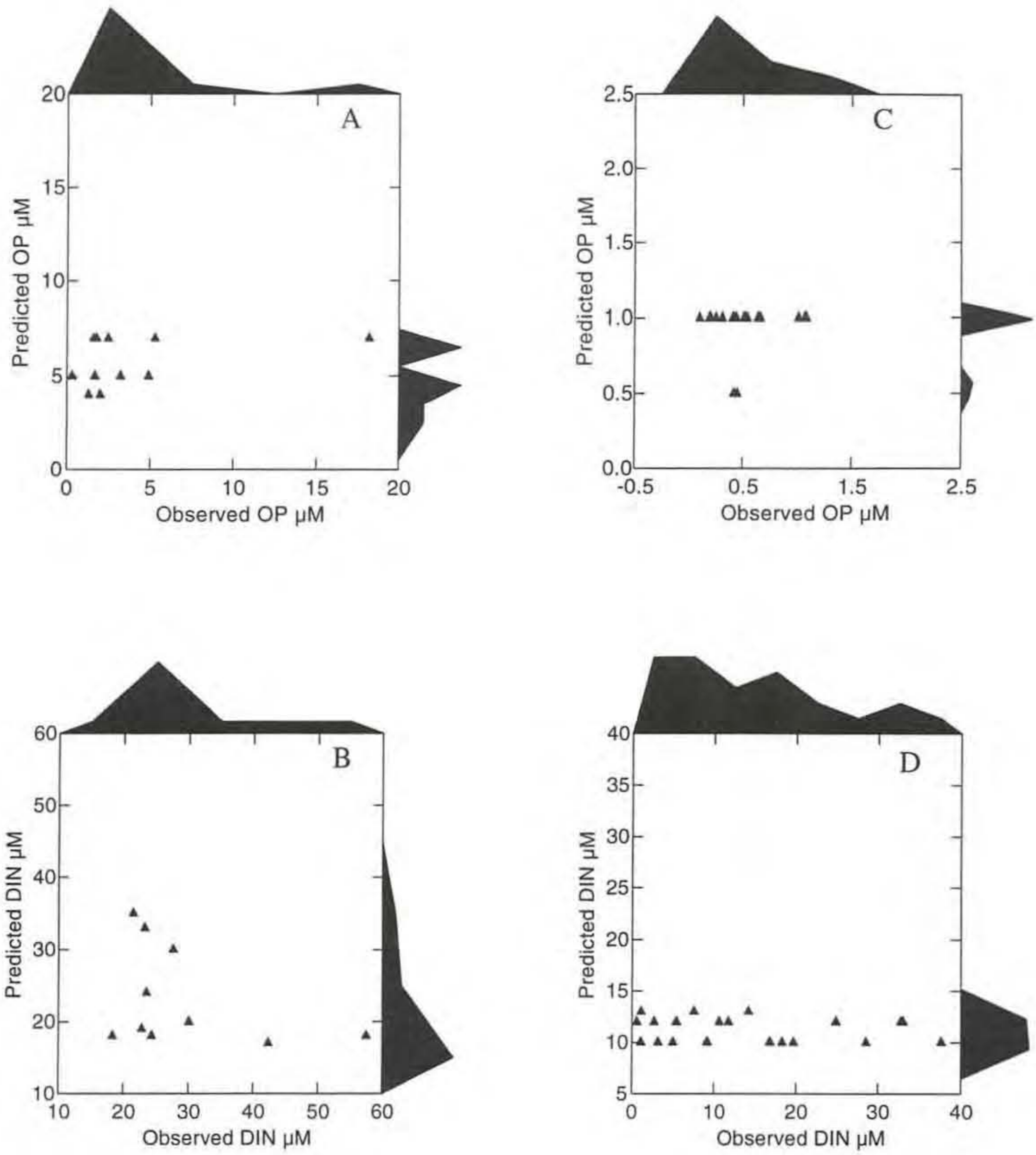

Figure 8 : Scatter plot of predicted vs. observed concentrations at site 1 (A and B) and site 2 (C and D). The frequently recorded range of both observed and predicted concentrations are respectively indicated by frequency polygon in front of each axis. Predicted concentrations were captured from the sampled stream cell values.

Although observations were in the same range as predictions at site 1, a slight deviation was noticed at site 2. This validation result with more accuracy close to site 1, was expected due to the proximity to point sources, where interpolation generated fewer errors (Fig. 4). This drawback can be also explained by the mixing with remaining flood water in the system. Such phenomena was more pronounced near the ocean boundary, where marine influence was likely to affect NPS loading.

The loading transfer between aquatic ponds and the adjacent system was rapid, due to the relatively reduced stream network within the study area, associated with the lack of fluctuations of the characteristics of the point source loading. In fact, water leaving earthen ponds was fairly homogeneous thanks to the relatively long residence time, which permitted the establishment of the concentration within the ponds (Fig. 6). The changes in the observation scales between the earthen ponds and the adjacent salt marshes do not have profound effect on estimated parameters. The signal of existing NPS loading was significant. The predicted concentrations showed realistic spatial distribution, with high concentrations near locations of intensive loading and a decrease with distance down to the axis of the bay.

This assessment method provides an efficient way to identify specific locations where elevated levels of nutrient concentrations may be expected. It used simple, yet powerful, empirical ²uations to represent loading dynamics within the system. This may be possible through a well-defined water exchange between ponds and the adjacent bay and a rather good knowledge of the general parameters governing its functioning.

The values of Table 1, which are used in our model are for the Fier d'Ars Bay example. If the model was to be used in other sites, local values should be inserted.

Nevertheless, the concept of such parameters to integrate in the model is interesting and may be easily adaptable to other sites. Moreover, methods outlined in the present study provide logistic advantages which allow for adaptation to other study areas, particularly the different geostatistical approach and spatial procedures. Such an approach is important for understanding the contribution of the land use distribution and physical factors to the nutrient enrichment of coastal waters, since the development of nutrient criteria at regional and local level has been among the recent preoccupations of several organizations (U.S.EPA 1998). General methods for estimation nutrient input levels, which depend on the characteristics of each ecosystem integrating the topography and the land use distribution, has became therefore pressing.

Besides the advantages, a number of limitations may be listed. The first was inherent to the parameter errors: constant value of expected mean constituent concentrations were associated with each land use and were not considered to vary from event to event. The same observation can be made for water renewal, which was calculated daily for every land use and was assumed to occur simultaneously and without decay. Secondly, the numerous source of errors from geographic and spatial data (Burrough 1986, Goodchilds 1993), inherent to mapping errors, i.e. errors associated with original maps which depend on factors such as the type of data accuracy and precision in the original measurement. Data processing techniques (Burrough 1986), errors at this stage include rasterisation (conversion of digitized line to grid cell) and overlay mapping (combining attributes from different maps)(Mac Dougall 1975). To avoid all accumulation of errors, the most original maps used in the present work were generated from a single support (aerial photographs), only bathymetric data were obtained from other authorities.

The GIS-based NPS pollutants simulation model has several aims, such as assessing existing conditions from generalized parameters purchased in the region, predicting future impacts from changing scenarios of existing activities or finally constituting a new concept for understanding and simulating the functioning of an ecosystem. Several dilemmas were generated around the capacity of GIS-based NPS pollutant simulation models to accomplish their expectations. However, Corwin *et al.* (1997) admitted its fantastic potential to provide useful insights into the assessment and prevention of increasingly visible regional-scale contamination problems. Maidment (1996) qualified them as quick analyses which showed, where and when pollution was at or near expected levels, where point sources were raising the concentration well above the levels expected from NPS alone, providing a spatial picture of the variation of observed pollutant levels.

Integrating other external sources such as, groundwater input and tidal loading, are our future aims. Addressing the internal water-sediment processes likely to reduce the flow of nitrogen to downstream ecosystems, are also among our major concerns. These will be concentrated towards establishing the total water balance of the studied area.

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