Onderzoek naar de gevolgen van het Sigmaplan, baggeractiviteiten en havenuitbreiding in de Zeeschelde op het milieu

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De bemanning van de schepen Scheldewacht II, Veremans, Parel en Scaldis I
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Inhoudstafel

Hoofdstuk 1. Inleiding

1.1. Omes: meer dan 10 jaar monitoring en onderzoek

Reeds meer dan 10 jaar volgt het monitorings- en onderzoeksprogramma Omes (Onderzoek Milieu-Effecten Sigmaplan) de waterkwaliteit van de Zeeschelde op. De Zeeschelde heeft immers grote ingrepen ondergaan, en het was duidelijk dat hieraan een uitgebreid monitoring programma moest gekoppeld worden. Het moet toelaten effecten van verschillende ingrepen te evalueren en kennis op te bouwen over het systeem om toekomstige ontwikkelingen in te schatten. Een estuarium is echter een zeer complex systeem dat door verschillende (menselijke) factoren wordt beïnvloed. Monitoring moet dan ook gebeuren vanuit verschillende onderzoeksdomeinen. Omes is een project waarin wetenschappers van verschillende onderzoeksdomeinen, van verschillende universiteiten, een gezamenlijk en geïntegreerd monitoringsprogramma uitvoeren.

Omes werd opgestart om de effecten van het Sigmaplan op het milieu in kaart te brengen. De stormvloed van 3 januari 1976 en de er op volgende overstromingen gaven aanleiding tot de uitvoering van dit omvangrijke plan dat het gehele Zeescheldebekken dient te beschermen tegen overstromingen. Na een nieuwe storm in 1994, met de hoogste waterstanden ooit gemeten op de Zeeschelde, werd hiertoe een nood- en urgentieprogramma goedgekeurd. De uitvoering hiervan diende wel te passen binnen een integrale visie op het beheer van de waterlopen. Dit leidde tot een Algemene Milieu-Impact studie voor het Sigmaplan en aansluitend tot een Onderzoek Milieu-Effecten Sigmaplan, OMES genaamd (Meire *et al.*, 1997; Van Damme *et al.*, 2001). Het OMES-project had tot doel de biogeochemische kennis van de Zeeschelde te actualiseren en in belangrijke mate uit te breiden. Als vervolg hierop werd een uitgebreid monitoringprogramma opgestart: “Onderzoek naar de gevolgen van het Sigmaplan, baggeractiviteiten en havenuitbreiding in de Zeeschelde op het milieu”, meestal ook kortweg Omes genaamd.

Deze Omes monitoring loopt nu meer dan 10 jaar. Verderzetting van deze monitoring is cruciaal voor het verder uitbouwen van de kennis van de Zeeschelde. Monitoring wordt pas echt waardevol als de metingen systematisch, over lange periode worden volgehouden. Dan pas kunnen langetermijneffecten, trends onderscheiden worden van kortetermijnvariaties. In de nieuwe visie waarbij veiligheid, toegankelijkheid en natuurlijkheid tezamen worden aangepakt, is een goede kennis van het estuariene ecosysteem van de Schelde onontbeerlijk. Omes is immers een belangrijk instrument geworden bij het wetenschappelijk onderzoek in het estuarium.

1.2. Moneos

Voorjaar 2008 werd Moneos voorgesteld: een programma voor geïntegreerde monitoring van het Schelde-estuarium, zowel in Nederland als Vlaanderen. Het Moneos programma geeft een omschrijving van een monitoring die wenselijk is om evoluties in de Schelde te beschrijven en oorzaak-gevolg relaties te achterhalen. Dat is essentieel om op een wetenschappelijk verantwoorde manier het estuarium te beheren.

Het plan omvat verschillende aspecten van het Schelde ecosysteem, gaande van hydro- en morfodynamiek, diversiteit habitats, fysico-chemie, ecologisch functioneren en diversiteit soorten. Veiligheid is als een afzonderlijk luik opgenomen. Het plan werd opgesteld op basis van informatie over lopende projecten, immers er wordt al heel veel gemeten in de Schelde, en overleg met verschillende betrokken instanties en experts. Er blijkt nog te weinig sprake van een goede integratie van de verschillende meetprogramma's en er bestaan nog aantal belangrijke hiaten.

In het kader van Moneos werd ook Omes onder de loep genomen. Er is een grote mate van overlap voor een aantal parameters tussen het Omes programma en de metingen van VMM. Echter hanteren Omes en VMM soms verschillende staalnametechnieken. Zo bemonsterd Omes tijonafhankelijk, VMM tijafhankelijk. Het was meer dan duidelijk dat voor een coherente, geïntegreerde monitoring, meer samenwerking tussen VMM en Omes nodig is.

Uit de Moneos studie kwam nogmaals pijnlijk aan het licht dat de Rupel ondervertegenwoordigd is in alle monitoringsprogramma's.

Om fytoplanktonbloei en bijgevolg het ecosysteemfunctioneren op te volgen is een maandelijkse monitoring ook onvoldoende. Met maandelijkse staalname worden bv pieken in algenbloei gemist. De frequentie van staalname moet bijgevolg beter afgestemd worden op de frequentie van te volgen fenomenen.

Vanaf 2009 is daarom het omes programma herschreven volgens de richtlijnen uit Moneos. Overleg met diverse actoren, waaronder VMM, hebben geleid tot een nieuw monitoringsvoorstel voor Omes.

1.3. Vernieuwde Omes programma

Omes beantwoordde reeds grotendeels aan de richtlijnen beschreven in Moneos. Ruimtelijk en temporeel schiet het programma echter af en toe te kort. Zo wordt de Rupel ondermaats bemonsterd en is de frequentie van staalname in de zomerperiode ontoereikend om algenbloei goed op te volgen. Daarom werden vanaf januari 2009 volgende aanpassingen doorgevoerd.

Elke monitoringscampagne wordt opgedeeld in 3 vaardagen, telkens een maandag, dinsdag en woensdag:
1: Beneden-Zeeschelde
2: Rupel
3: Boven-Zeeschelde.

Dag 1:
vertrek te Antwerpen, afvaart tot station Grens en vanaf daar opvarend meten tot Temse. Terugvaart naar Antwerpen.

Dag 2:
vertrek te Boom, opvaren tot Dijle en Netemonding. Afvarend meten tot Rupelmonde. Vanaf daar opvarend meten tot Baasrode en terugkeer per minibus naar Boom.

Dag 3:
Vertrek te Dendermonde, opvarend meten tot Melle. Terugvaart naar Dendermonde.

Een overzicht van de oude en nieuwe stations en de verdeling over de diverse vaardagen wordt gegeven in Tabel 1.1. Omwille van de samenwerking met VMM, werd het station Melsele wel verschoven. Dit punt wordt niet meer bemonsterd ter hoogte van Boei 105, maar iets meer opwaarts ter hoogte van het sluisje van de oude marine haven. Ook het punt Boom werd verschoven, een 100 tal meter opwaarts, zodat de staalname terug plaatsvindt ter hoogte van het ponton van de veerboot. Deze lokatie was een aantal jaar geleden stroomafwaarts verschoven tot net voor de spoorwegbrug, zodat het station steeds bereikbaar was per schip. Opwaarts de spoorbrug vormt nu geen probleem meer, aangezien de vaartohcten in het Rupelbekken nu steeds met Scaldis I worden uitgevoerd.

Tabel 1.1 Overzicht van stations en indeling in vaardagen voor 2008 en 2009

Station	Omes 2009	Omes 2008	description	km from mouth
mouth				0
Grens	day 1	day 1	Zandvliet, Noordzeeterminal	58
Liefkenshoek	day 1	day 1	Liefkenshoektunnel	63.5
Melsele	day 1	day 1	Punt van Melsele	71
Antwerpen	day 1	day 1	Steen	78
Kruibeke	day 1	day 1	veer	85
Bazel		day 1	veer	89
mouth Rupel				92
Steendorp	day 1	day 1	kerk	94
Temse	day 1	day 2	ponton	98.5
Lippenbroek	day 1	day 2	Hamme (Driegoten)	103.5
Mariekerke			veer	107
Baasrode	day 2	day 2	ponton	113
Vlassenbroek			kapel	118
Dendermonde	day 2	day 3	brug	121
St. Onolfs	day 2	day 3	Bocht v Damme	125
Appels	day 2	day 3	veer	128
Uitbergen	day 2	day 3	brug	138
Wetteren	day 2	day 3	baanbrug	145
Melle	day 2	day 3	brug	151
Niel		day 2		
Boom		day 2	spoorbrug	98
Rumst		day 2		
Nete		day 2		
Dijle		day 2		
Bovenschelde	boundary	boundary	boundary	
Durme	boundary	boundary	boundary	
Dender	boundary	boundary	boundary	
Haven	boundary	boundary	boundary, sluis, dokzijde	
Grote Nete		boundary		
Kleine Nete		boundary		
Dijle		boundary		
Zenne		boundary		

In de zomerperiode werd de frequentie van meten verhoogd van eens per maand naar tweemaal per maand, met als doel de algenbloei beter in kaart te brengen. Concreet betreft het de periode van (april) mei tot en met september. In Tabel 1.2 wordt de lijst voorgesteld met monitoringsdagen.

Tabel 1.2 Vaardagen Omes 2009

	Vaardag 1 **maandag**	**vaardag 2** **dinsdag**	**vaardag 3** **woensdag**
januari	12 jan	13 jan	14 jan
februari	9 feb	10 feb	11 feb
maart	2 mrt	3 mrt	4 mrt
april	6 apr	7 apr	8 apr
mei	4 mei	5 mei	6 mei
	25 mei	26 mei	27 mei
juni	8 jun	9 jun	10 jun
	22 jun	23 jun	24 jun
juli	6 jul	7 jul	8 jul
	27 jul	28 jul	29 jul
augustus	10 aug	11 aug	12 aug
	24 aug	25 aug	26 aug
september	7 sep	8 sep	9 sep
	21 sep	22 sep	23 sep
oktober	12 okt	13 okt	14 okt
november	16 nov	17 nov	18 nov
december	7 dec	8 dec	9 dec

Hoofdstuk 2. Basiswaterkwaliteit

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2.1. Inleiding

De basiswaterkwaliteit van de Zeeschelde heeft een spectaculaire verandering ondergaan. Van een quasi dode rivier die tot ver in de jaren negentig van vorige eeuw gekenmerkt werd door anoxie, is in het bijzonder het zoete deel van de Zeeschelde geëvolueerd naar een systeem dat nu oversaturatie kent in de zomermaanden. Dankzij meer dan 10 jaar monitoring in het kader van OMES, kunnen deze duidelijke langetermijntrends onderscheiden worden van kortstondige fluctuaties door bijvoorbeeld weersomstandigheden. De Zeeschelde heeft een belangrijke verschuiving ondergaan van een anoxisch systeem naar een systeem met oververzadiging. Het ecosysteemfunctioneren is fundamenteel gewijzigd.

De eerstvolgende paragraaf is een samenvattende paragraaf die de waargenomen verschuiving in het Schelde ecosysteem beschrijft. Deze beschrijving is gebaseerd op een wetenschappelijke publicatie in het vakblad Biogeosciences, waarin onderzoekers van de Universiteit Antwerpen en het Nederlands Instituut voor Ecologie, op basis van de resultaten uit het Omes programma, hun bevindingen presenteerden. Hierin wordt geconcludeerd dat pas vanaf halfweg de jaren negentig de waterkwaliteit van de zoete Zeeschelde merkbaar begon te verbeteren. In de Nederlandse Westerschelde en de zone rond Antwerpen was dat al sinds de vroege jaren tachtig het geval.

Nadien worden enkele belangrijke trends per parameter toegelicht in paragrafen 2.3 tot 2.11 en besproken aan de hand van “surface plots”. Deze werden opgesteld met de SURFER software, versie 5.1, met een lineaire Kriging interpolatie, anisotropie radius 1/100. Voor een selectie van parameters werden deze plots opgesteld voor de volledige Omes dataset (december 1995 – juni 2008). De plots geven de temporele en ruimtelijke trends weer van de Belgisch-Nederlandse grens (station Grens, km 58) tot nabij Gent (station Melle, km 151). De grijswaarden- of kleurschaal geeft de concentratie weer. Om belangrijke verschillen bij lage concentraties duidelijk weer te geven, is de schaal niet steeds lineair.

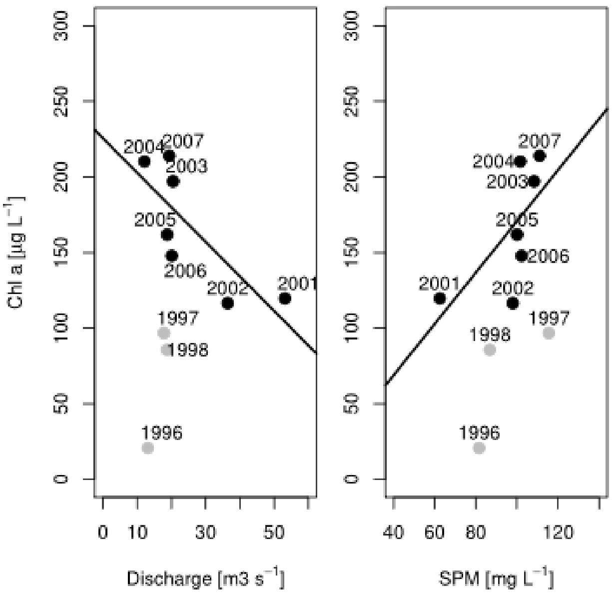
2.2. De Schelde: van oververvuild naar gewoon vervuild

2.2.1. Unieke vervuiling

De zoete Zeeschelde was zo sterk vervuild dat zelfs microscopische algen (fytoplankton) er niet meer konden groeien. De lage zuurstofconcentraties in het water zorgden ervoor dat de zoete

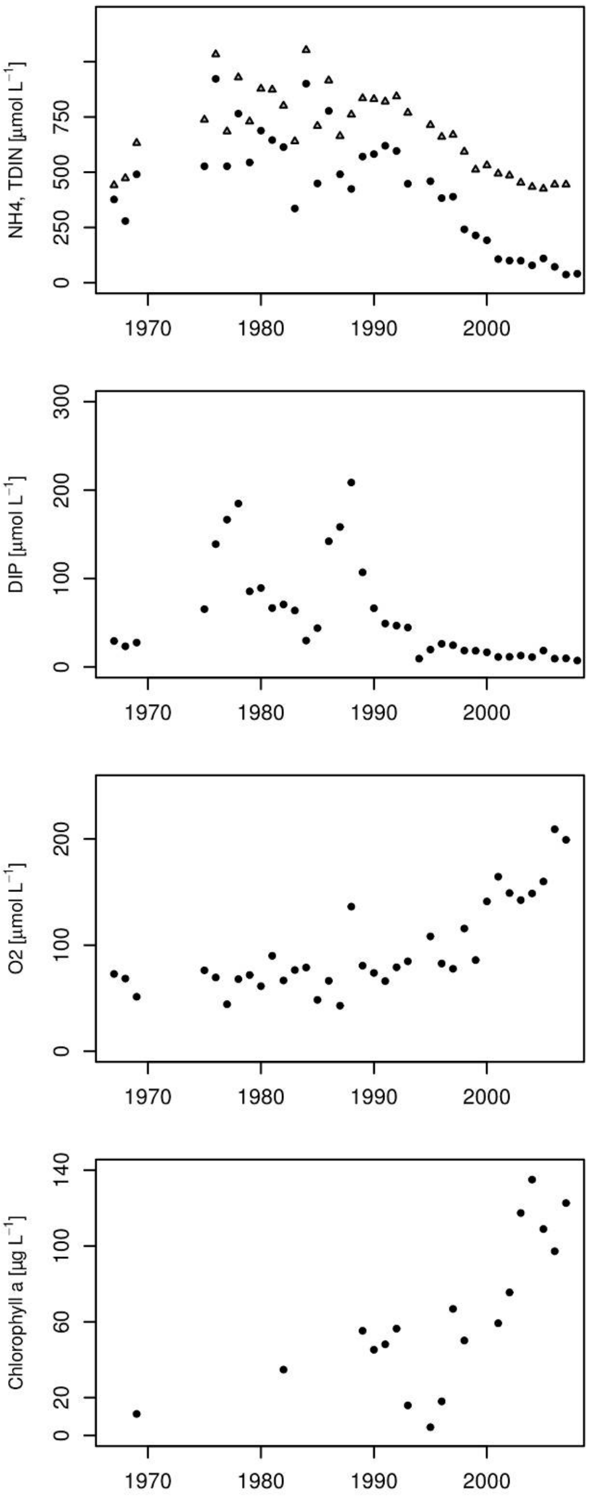
Zeeschelde niet alleen onleefbaar was voor de meeste hogere organismen (wormen, schelpdieren, vissen, etc.), maar dus ook voor het fytoplankton. Kortom, de Schelde was oververvuild.

Literatuuronderzoek en modelberekeningen ondersteunen de hypothese dat algen belemmerd waren in hun groei, hetzij rechtstreeks door de extreem lage zuurstofconcentraties, hetzij door andere kenmerken van de hypereutrofe toestand zoals hoge ammonium/ammonia concentraties en de aanwezigheid van sulfiden. Andere, klassieke verklarende factoren voor algenbiomassa, zoals licht- en nutrientenbeschikbaarheid en uitspoeling verklaren de langetermijntrend in de chlorofyl a concentraties niet (Figuur 2.1, Figuur 2.2). Hoewel een direct negatief effect van anoxia op fytoplankton op het eerste gezicht tegenstrijdig lijkt (fytoplankton produceert immers zelf zuurstof), zijn er verschillende bekende mechanismen bekend van negatieve effecten van anoxia op het fotosynthetisch apparaat en op het groeimetabolisme. Herstel van fotoschade en fotoinhibitie verloopt trager, onvolledig of niet in anoxische omstandigheden. En zo goed als alle organismen hebben een zuurstofsensor die ervoor zorgt dat bij lage zuurstofconcentraties celfuncties worden vertraagd of stilgelegd. In de Schelde, die zeer goed gemengd is, vertoeven algencellen slechts een beperkt gedeelte van de tijd in de bovenste waterlagen waar licht kan doordringen (ongeveer ½ meter). De rest van de tijd vertoeven ze in de donkere, diepere waterlagen. In die omstandigheden produceren algen geen zuurstof en zal ook binnen de cel anoxia optreden wanneer dat in het omliggende water zo is, met de hoger vermelde negatieve effecten. Het netto-effect is een al dan niet geheel belemmerde groei van de algenpopulatie.



Figuur 2.1: Mei–september en zoetwater gemiddelde Chlorofyl-a concentraties uitgezet t.o.v. het gemiddelde bovenstroomse debiet (Q) en gemiddelde zwevende stof concentraties (SPM). Er is geen correlatie voor de hele 1996-2007 dataset, en slechts een zwakke correlatie voor de data vanaf 2001 (zie regressielijn)

De waterkwaliteit van de zoete Zeeschelde is dan wel spectaculair verbeterd in de afgelopen tien jaar, maar in vergelijking met rivieren en estuaria over de hele wereld behoort ze nog steeds tot de meest ontwrichte ecosystemen. De vervuilde toestand waarin de zoete Zeeschelde tot halverwege de jaren negentig verkeerde, was uniek op wereldschaal. Deze hypereutrofe toestand werd veroorzaakt door hoge input van ammonium en allochtoon organisch materiaal, en gekenmerkt door (seizoenale) anoxia, hoge N- en P- concentraties en een beperkte algenbloei (Figuur 2.2).

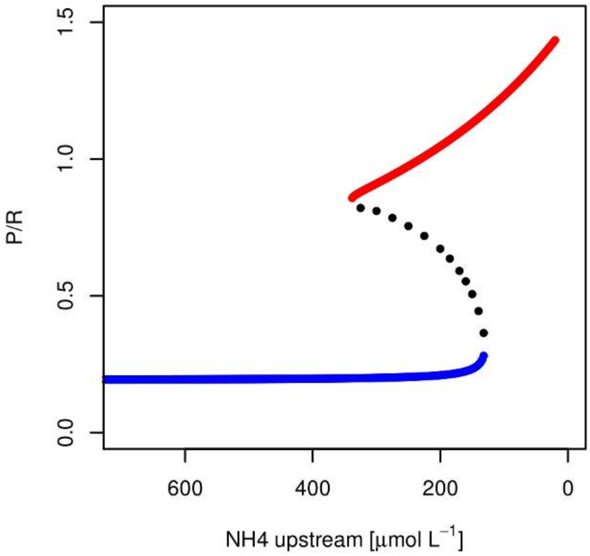


Figuur 2.2: Langetermijntrend van de jaargemiddelde en systeemgemiddelde NH4, TDIN, DIP, O2 , en Chlorofyl a concentraties voor de zoete Zeeschelde

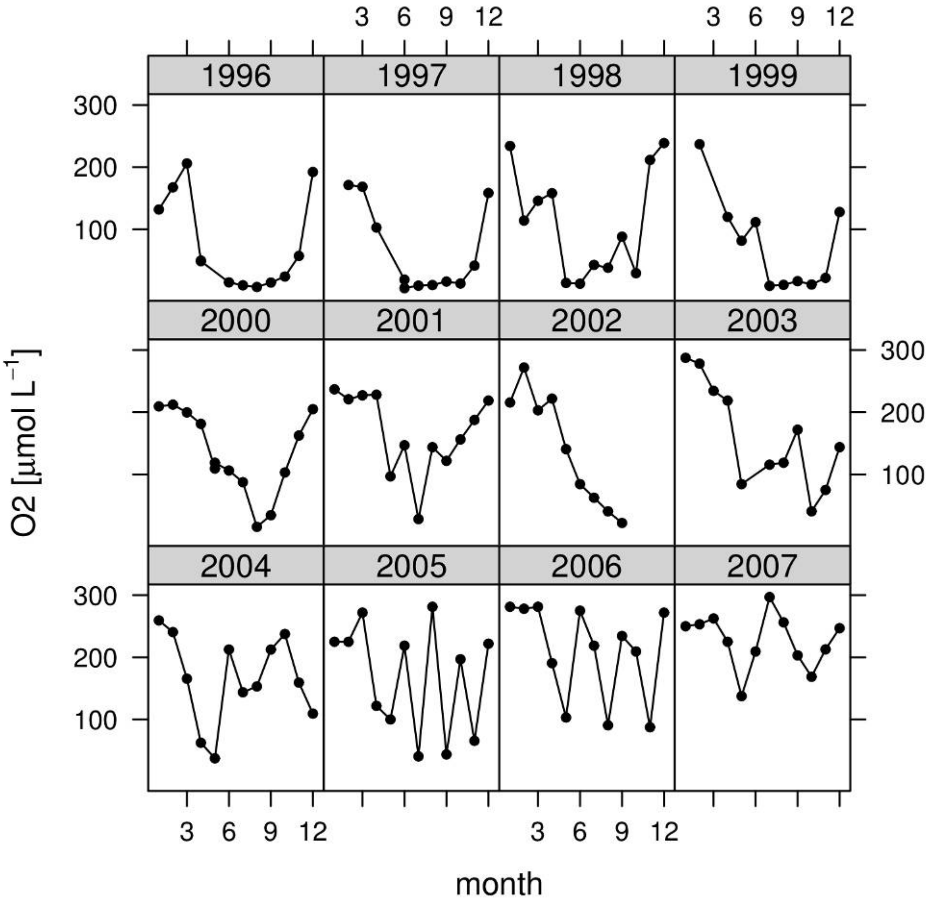
2.2.2. Relatief snelle verbetering

Het bijzondere aan het herstel van de zoete Zeeschelde is, dat de overgang van de oververvuilde (hypereutrofe) toestand naar de huidige situatie relatief snel is gebeurd. Na decennia van weinig verandering, zagen we op enkele jaren tijd (ongeveer tussen 2000 en 2005) een evolutie van een rivier met extreem lage zuurstofconcentraties in de zomer, naar een rivier waarin extreem hoge zuurstofconcentraties worden gemeten tijdens die zomermaanden. Dat is een gevolg van de herstelde algenbloei. Het evenwicht tussen de processen die zuurstof gebruiken (voornamelijk omzetting van ammonium naar nitraat en de afbraak van organisch materiaal) en de processen die zuurstof in het water brengen (fotosynthese en uitwisseling met de lucht) is in deze periode gekanteld. Vóór 2000 was het zuurstofverbruik door de hoge organische belasting van de Schelde zo hoog dat alle zuurstof in het water werd opgebruikt, waardoor ook algen niet meer konden groeien. Na 2000 was het zuurstofverbruik laag genoeg geworden zodat de algen opnieuw konden bloeien, en ook extra zuurstof in het water konden pompen.

Deze mechanismen zorgen ervoor dat de overgang van een hypereutrofe naar een klassieke eutrofe toestand als regime shift gekenmerkt kan worden. Met behulp van een eenvoudig maar realistisch model voor de zoete Zeeschelde bepaalden we de ratio van zuurstofproductie (primaire productie) over zuurstofconsumptie (nitrificatie en respiratie). Wanneer die uitgezet wordt ten opzichte van een de toevoer van ammonium, krijgen een klassieke bifurcatie-figuur (Figuur 2.3). Bij hoge ammonium-inputs consumeert het systeem netto zuurstof, en bij lage ammonium-inputs wordt netto zuurstof geproduceerd. Bij tussenliggende ammonium-inputs kunnen beide situaties voorkomen, afhankelijk van de randvoorwaarden (debiet, lichtinstraling, bovenstroomse input van algenbiomassa en zuurstof, etc.). Systemen in zulke toestand vertonen wat men noemd 'flickering': oscillaties van bepaalde variabelen. In de zoete Zeeschelde werd flickering vastgesteld in de gemeten zuurstofconcentraties aan Dendermonde (Figuur 2.4).



Figuur 2.3: Gemodelleerde ratio van zuurstofproductie over zuurstofconsumptie, uitgezet t.o.v. de bovenstroomse ammonium-inputs. Bij tussenliggende ammonium inputs vertoont het systeem twee alternatieve stabiele toestanden, een netto zuurstof producerende en een netto zuurstof consumerende toestand

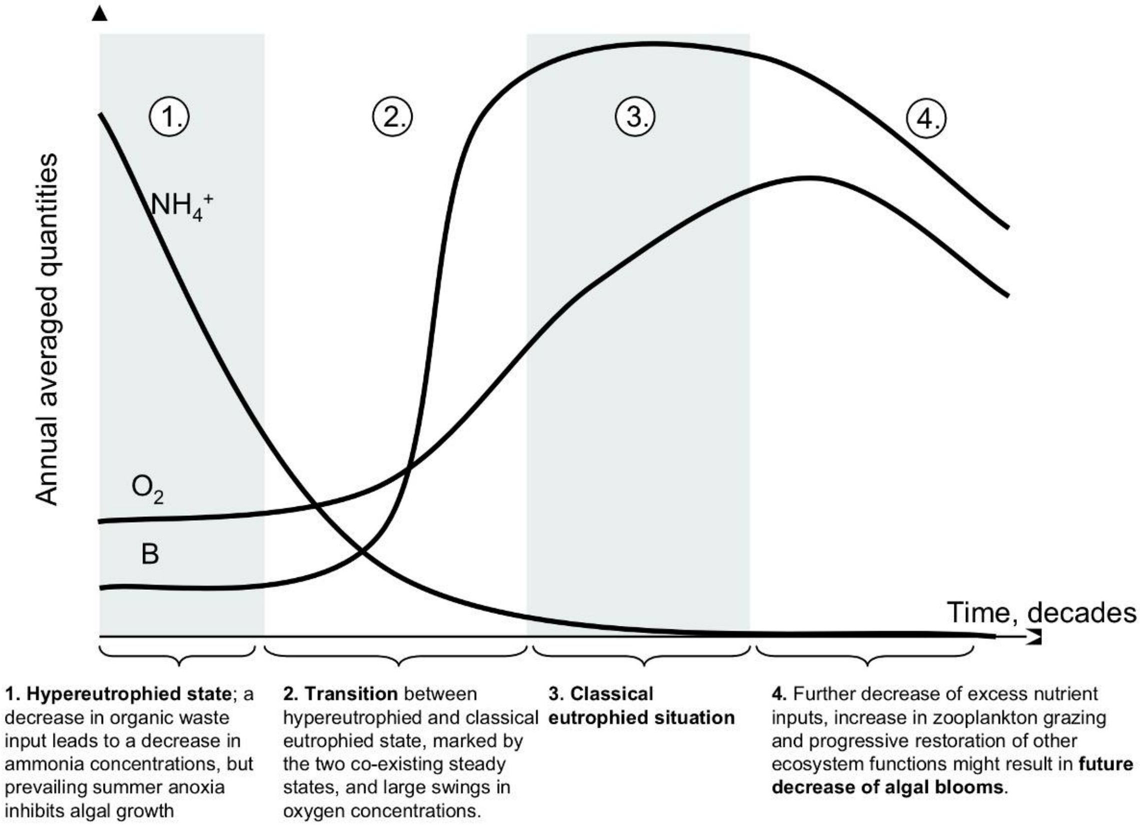


Figuur 2.4: Geobserveerd jaarlijks patroon van opgeloste zuurstofconcentraties aan Dendermonde. Tussen 1996 en 2002 vertoonde de zuurstofconcentratie een duidelijk seizoenaal patroon met hoge concentraties in de winter en lage in de zomer en de herfst. Vanaf 2003 wordt dit patroon steeds meer onregelmatig. In het bijzonder in 2004-2006 zien we grote fluctuaties in de zomermaanden. Dit soort 'flickering' is typisch voor systemen met twee stabiele toestanden.

2.2.3. Overmatige algenbloei

De zoete Zeeschelde (zoals zowat het ganse Schelde-estuarium) vertoont nu de symptomen van een klassieke eutrofe toestand (Figuur 2.5).. De huidige algenbloei is op zich een indicator voor overmatige beschikbaarheid van nutriënten. Zo'n overmatige algenbloei heeft bekende negatieve effecten, zoals schommelingen in zuurstofconcentraties tussen dag en nacht (overdag produceren de algen zuurstof, 's nachts gebruiken ze daar zelf weer een deel van) en het mogelijk voorkomen van schadelijke algensoorten. Op dit moment veroorzaakt de algenbloei geen acute problemen. Maar de evolutie naar een meer natuurlijke en gezond functionerende Schelde is slechts mogelijk wanneer inspanningen worden gedaan om de toevoer van nutriënten naar de Schelde verder te doen dalen.

Verder valt het af te wachten of de zich herstellende zooplanktonpopulatie een rol van betekenis kan spelen in het controleren van de algenbloei. Zooplankton voedt zich immers met phytoplankton, en in bv. het zoete gedeelte van het estuarium van de Hudson is aangetoond dat de algenpopulatie door een combinatie van fysische randvoorwaarden en begrazing door zooplankton bepaald wordt.



Figuur 2.5: Conceptueel schema dat het herstel vanuit de hypereutrofe toestand beschrijft. Tussen 2000 en 2005 evolueerde de zoete Zeeschelde van stadium 1 naar stadium 3.

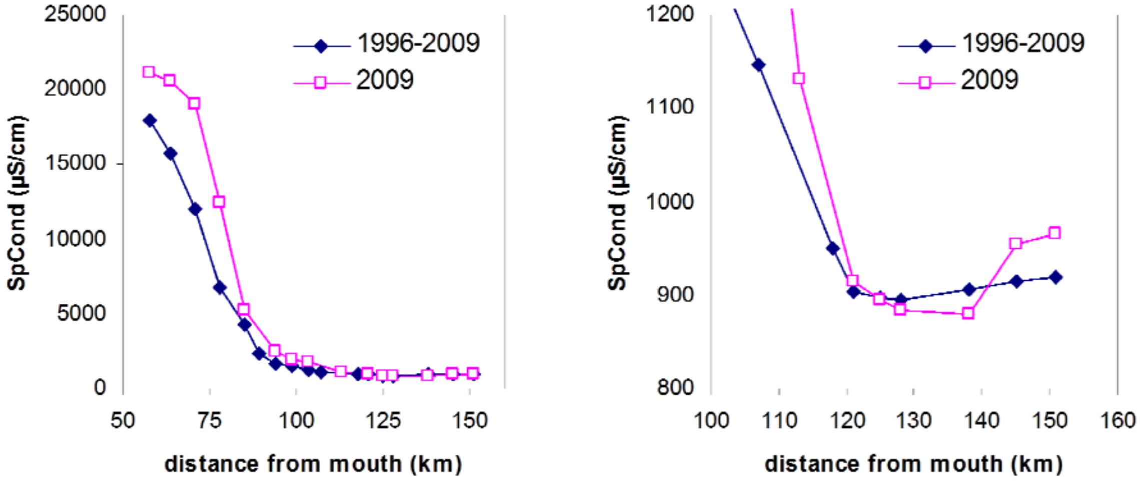
2.2.4. Rupelmonding blijft zorgenkind

Het herstel van de Schelde zal in de toekomst wellicht veel geleidelijker verlopen, al is dat grotendeels afhankelijk van beleidsbeslissingen. De verbeterde zuurstofconcentraties zorgen ervoor dat ook hogere organismen opnieuw in de Schelde kunnen overleven; er zwemmen bijvoorbeeld al een tijdje opnieuw vissen in de zoete Zeeschelde. De zone met de laagste zuurstofconcentraties van de Schelde bevindt zich nu ter hoogte van de Rupelmonding, waar het afvalwater van Brussel in de Schelde komt. Hoewel het inwerking stellen van de waterzuivering van Brussel (in 2006) onmiddellijk een positief effect had op de waterkwaliteit in deze zone, blijven we daar de laagste zuurstofconcentraties van de hele Schelde meten.

2.3. Specifieke geleidbaarheid en chloride

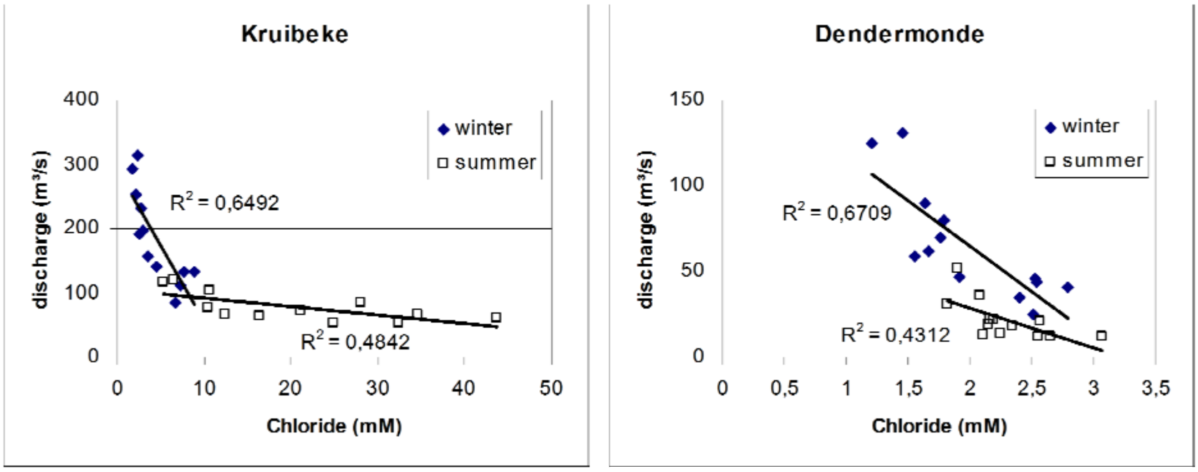
Saliniteit is in estuaria een determinerende factor voor het voorkomen van tal van organismen. Echte metingen van saliniteit zijn complex en gebeuren niet standaard binnen Omes. Vaak worden wel omrekeningen van geleidbaarheid naar saliniteit gemaakt, of van geleidbaarheid naar Chloride-gehalte. Deze omrekening maken wij hier niet, omdat zeker bij lage concentraties aan chloride, deze omrekening niet steeds nauwkeurig is. De geleidbaarheid wordt in hoofdzaak bepaald door de aanwezige Cl^{-}ionen, maar in het zoete vormen ook andere ionen een belangrijke bijdrage tot de geleidbaarheid.

Chloride en specifieke geleidbaarheid vormen, net als saliniteit, een goede maat voor de intrusie van zeewater. Specifieke geleidbaarheid (Surfer 2.1) kent duidelijk seizoenale trends: hogere winterafvoeren dringen de zoutinvloed terug. De steile geleidbaarheidsgradiënt schuift in de zomer ca 20 km verder stroomopwaarts in het estuarium. De invloed van natte en droge jaren is duidelijk waarneembaar: de jaren 2000, 2001 en 2002 kennen door de hoge afvoerdebieten een sterkere terugdringing van de geleidbaarheid. De intrusie van zouter water in de Zeeschelde lag in de zomer van 2009 iets hoger dan het gemiddelde (Figuur 2.6, links). Opvallend is wel dat in het zoete de geleidbaarheid daalt tot een minimum, om vervolgens weer toe te nemen richting Gent (Figuur 2.6, rechts). Deze trend was in 2009 nog iets sterker aanwezig, met een opwaartse verschuiving van het geleidbaarheidsminimum.



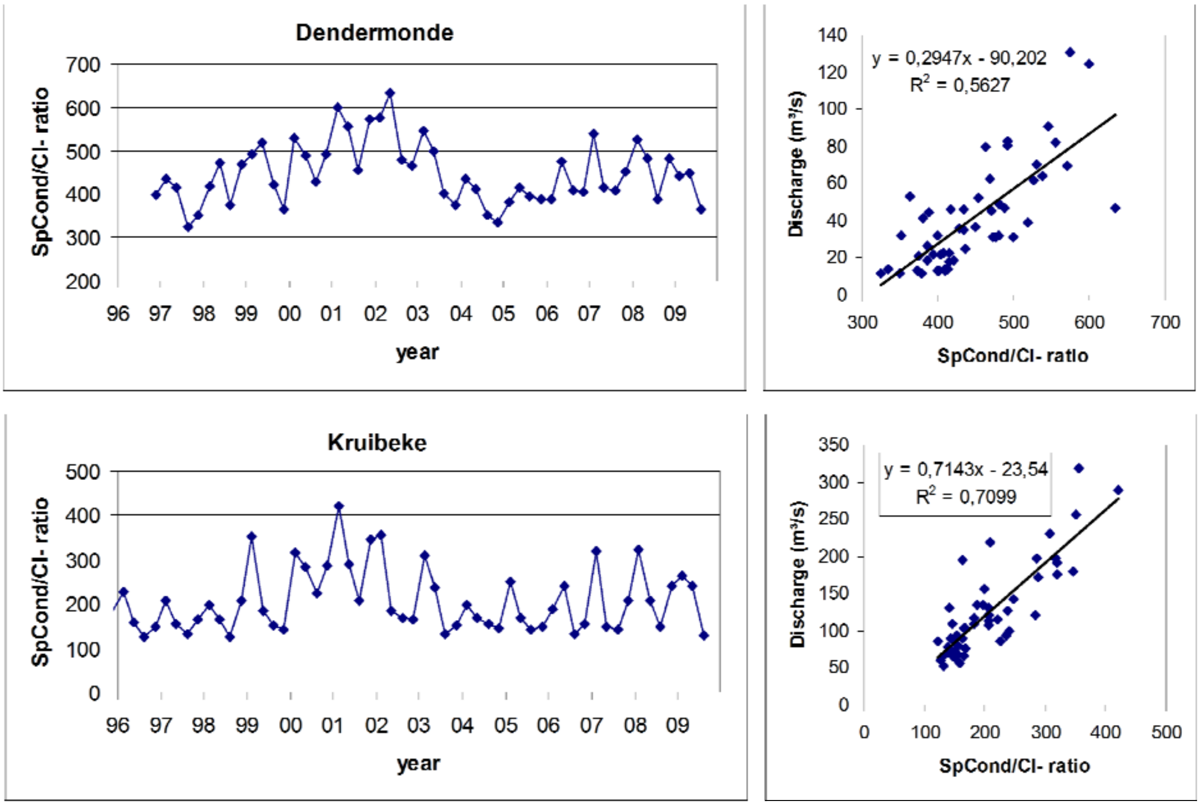
Figuur 2.6: Specifieke geleidbaarheid (µS/cm) tijdens de zomermaanden in functie van afstand tot monding voor de gehele Zeeschelde (links) en het zoete (rechts). De ruitjes geven de gemiddelde zomerwaarde weer over de periode 1996-2009; de open vierkantjes geven de gemiddelde zomerwaarde voor 2009.

Gelijkaardige patronen kunnen worden opgetekend voor chloride. Er bestaat ook een duidelijke correlatie tussen afvoerdebiet en chloridegehalte te Kruibeke, maar ook in het zoete te Dendermonde (Figuur 2.7).



Figuur 2.7: correlatie tussen debiet en chloridegehalte te Kruibeke en Dendermonde (periode 1996-2007).

Specifieke geleidbaarheid wordt in hoofdzaak bepaald door chloride-ionen. De ratio SpCond/Cl is echter niet constant, maar toont duidelijke seizonale schommelingen, wellicht gestuurd door de schommelende debieten (Figuur 2.8). In Dendermonde en Kruibeke is de ratio sterk debietsafhankelijk. Meer afwaarts verwatert deze relatie. De SpCond/Cl ratio ligt er ook lager, maw de geleidbaarheid wordt meer dan te Kruibeke en veel meer dan te Dendermonde bepaald door chlorides. Te Dendermonde spelen andere ionen dan Cl$^-$ ook een rol in de geleidbaarheid. Bij hogere afvoerdebieten verkleint het aandeel van Cl$^-$ in de geleidbaarheid in het zoete deel van de Schelde (positieve correlatie tussen SpCond/Cl ratio en afvoerdebiet). Mogelijks komen bij hogere debieten relatief meer andere ionen in het systeem. Dit illustreert nogmaals de noodzaak om geleidbaarheid te rapporteren als geleidbaarheid, chloride te rapporteren als chloride.



Figuur 2.8: Specifieke geleidbaarheid – Chloride ratio (SpCond/Cl$^-$ ratio in µS/cm / mm) en de correlatie tussen deze ratio en het afvoerdebiet. Voor Dendermonde werden de afvoerdebieten te Melle gehanteerd, voor Kruibeke en Grens de afvoefdebieten te Schelle (data WLH).

2.4. Zuurstof

In 2006 werd voor de eerste maal zuurstofoversaturatie gemeten tijdens een Omes campagne. Sindsdien werd dit sporadisch waargenomen (Surfer 2.2), tot in 2009: dan was er de ganse zomer lang op diverse lokaties oververzadiging. Dit staan in schril contrast met de quasi anoxische toestanden uit de jaren '90.

2.4.1. Brakke zone

Ter hoogte van het station Grens is deze stijging minder uitgesproken. De zuurstofwaarden zijn hier nooit dramatisch laag geweest (meestal boven 60% tijdens Omes metingen 1995 - 2007) ten gevolge van uitwisseling met de zuurstofrijke Westerschelde. Toch werd in ongeveer de helft van de metingen de norm van 5 mg/l niet gehaald (Tabel 2.1). De sterke seizoenale variatie met minima in de zomer en maxima in de winter die rond de Rupelmonding wordt waargenomen, is merkbaar tot aan de grens. Bij hoge afvoerdebieten wordt deze invloed verstrekt, zodat tijdens de zeer natte zomers van 2000, 2001 en 2002 de zuurstofwaarden aan de grens daalden, terwijl meer stroomopwaarts de hoge debieten resulteerden in een stijging van de zuurstofwaarden, mogelijks door verdunning van de vuilvracht.

Sinds 2003 wordt in ruim drie kwart van de metingen de norm gehaald.

2.4.2. Oligohaliene zone

Ter hoogte van de Rupelmonding (km 92, oligohaliene zone) is de seizonale schommeling van de zuurstofverzadiging zeer uitgesproken. Bacteriële afbraak van de vuilvracht tijdens de warmere zomermaanden zorgt voor een totale zuurstofdepletie in 1996 en 1997. Geleidelijk neemt echter de zone zuurstofdepletie af in ruimte en tijd. Gemiddeld is een stijging van het zuurstofgehalte waarneembaar. De beperkte jaargemiddelde stijging op het einde van de jaren '90 houdt gelijke tred met de stijging in de Rupel zelf en zou een gevolg kunnen zijn van toegenomen verdunning. Drogere jaren 2004 en 2005 kennen een terugval van de zuurstofverzadiging. 2003, een jaar met zeer lage afvoerdebieten, kent deze terugval in de Rupel niet. Vanaf 2006 stijgen de zuurstofwaarden weerom, en bereiken in 2008 een maximum in de oligohaliene zone. Wellicht is dit het gevolg van de waterzuivering op de Zenne, die sinds dan operationeel is. De zuurstofwaarden in de Zeeschelde in het mondinggebied van de Rupel zijn reeds sinds 2003 in stijgende lijn, en een gevolg van de spectaculaire zuurstoftoename in de zoetwaterzone. Vooral in de zomer zijn de zuurstofwaarden in het zoete sterk gestegen tussen 1996 en 2008. De waarden zijn echter nog ver van goed: in 35% van de metingen werd de zuurstofnorm nog niet gehaald in 2008 (Tabel 2.1).

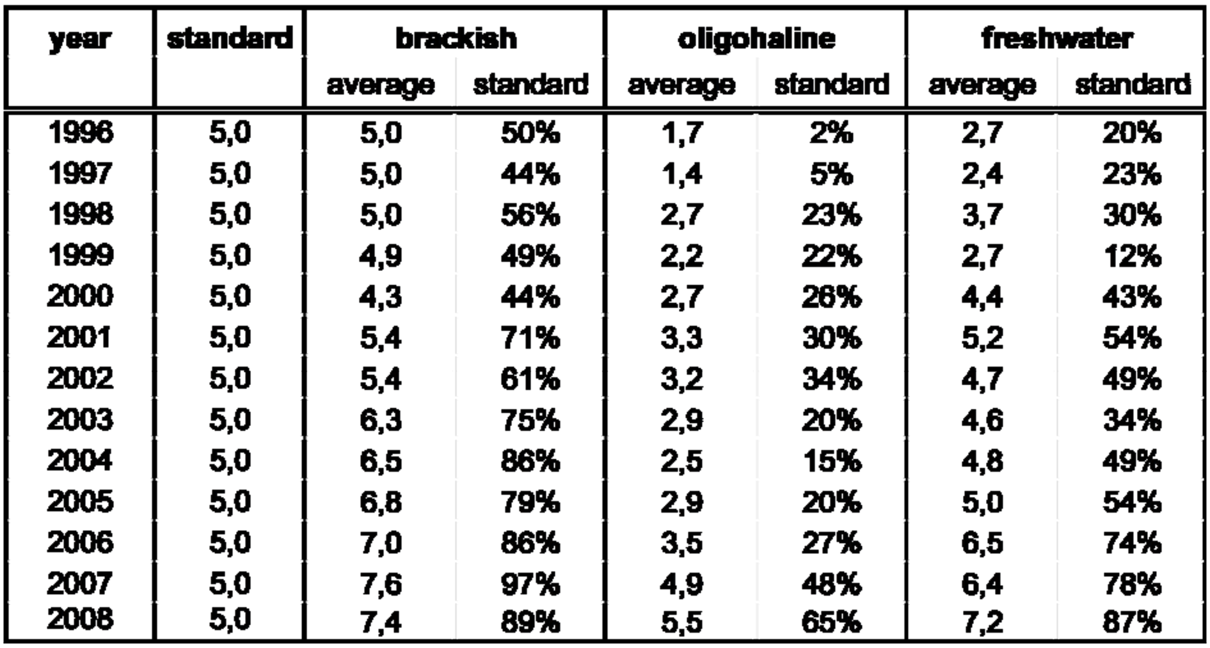
2.4.3. Zoete zone

In het zoete valt een spectaculaire verbetering van de zuurstofverzadiging op te tekenen, vooral in de zomermaanden. Tussen 1998 en 2002 verbetert vooral de zuurstofhuishouding in de Bovenschelde, wellicht een effect van toegenomen verdunning bij toegenomen debieten. Tussen 1998 en 2002 wordt immers een sterke daling van de concentratie aan biologische zuurstofvraag (BOD) waargenomen (Figuur 2.10). De totale BOD-vracht daarentegen vertoont deze daling niet (data niet weergegeven). Vanaf 2003 is er trendbreuk: de stijging van de zuurstofconcentratie in de zoetwaterzone kan niet aan verdunning worden toegeschreven. Tussen 2003 en 2007 wordt immers een stijging van de BOD-concentratie genoteerd in de Bovenschelde. In de Zeeschelde zelf is de BOD in dalende lijn.

Hoewel de verbetering van de zuurstofhuishouding reeds spectaculair is, voldoet de Zeeschelde nog niet aan de basiskwaliteitsnorm voor oppervlaktewater (Vlarem II) (Tabel 2.1).

Met de sterk toegenomen primaire productie moet wel een belangrijke kanttekening gemaakt worden bij de staalname. Door steeds 's ochtends te meten te Dendermonde en tegen de middag aan te komen te Melle, kan door primaire productie de zuurstofwaarde te Melle systematisch hoger liggen.

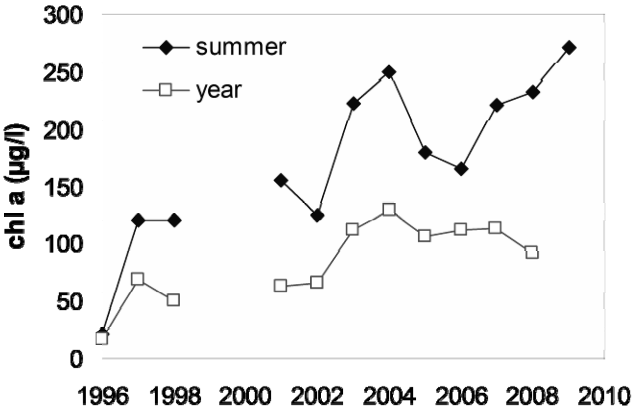
Tabel 2.1: Evolutie van de zuurstofgehaltes in het brakke (km 58 – 71), oligohaliene (km 78-98) en zoete deel (km 103-151) van de Zeeschelde. De kolom standard geeft de norm voor zuurstof weer (mg/l) uit Vlarem II. Voor elke zone wordt het jaargemiddelde weergegeven en het aantal metingen dat de norm haalt (%).



year	standard	brackish		oligohaline		freshwater	
		average	standard	average	standard	average	standard
1996	5,0	5,0	50%	1,7	2%	2,7	20%
1997	5,0	5,0	44%	1,4	5%	2,4	23%
1998	5,0	5,0	56%	2,7	23%	3,7	30%
1999	5,0	4,9	49%	2,2	22%	2,7	12%
2000	5,0	4,3	44%	2,7	26%	4,4	43%
2001	5,0	5,4	71%	3,3	30%	5,2	54%
2002	5,0	5,4	61%	3,2	34%	4,7	49%
2003	5,0	6,3	75%	2,9	20%	4,6	34%
2004	5,0	6,5	86%	2,5	15%	4,8	49%
2005	5,0	6,8	79%	2,9	20%	5,0	54%
2006	5,0	7,0	86%	3,5	27%	6,5	74%
2007	5,0	7,6	97%	4,9	48%	6,4	78%
2008	5,0	7,4	89%	5,5	65%	7,2	87%

2.5. Chlorofyl a

De stijging van het zuurstofgehalte wordt in de eerste plaats toegeschreven aan een sterke stijging van de primaire productie. Lokale zuurstofproductie door algen zorgt in het zoete deel van de Zeeschelde sinds 2003 voor een sterke toename van het zuurstofgehalte tijdens de zomermaanden. De chlorofylconcentraties stijgen immers sterk (Surfer 2.3), in het bijzonder in het zoete (Figuur 2.9). In de zomermaanden worden tijdens pieken van bloei concentraties tot boven 500µg/l waargenomen. De waargenomen trend strookt niet met het klassieke beeld van eutrofiëring: in de Zeeschelde zijn de voorbije 10 jaar de concentraties aan nutriënten afgenomen, de chl a concentraties daarentegen stijgen. Dit leidde tot de hypothese dat de zeer lage zuurstofwaarden een inhiberend effect kunnen hebben op algenbloei.



Figuur 2.9: Zomer- en jaargemiddelde chl a waarden in het zoete (km 103-151)

2.6. pH

De schommelingen in pH (Surfer 2.4) illustreren mooi de evolutie in de primaire productie. Bij elke algenbloei stijgen de pH waarden. De trendbreuk tussen de periode 1996-2002 en 2003-2008 wordt bijgevolg ook waargenomen in de pH, met gemiddeld hogere pH-waarden de laatste jaren. Rond de Rupelmonding is deze stijging sinds 2007 het sterkst voelbaar. Ter hoogte van de Rupelmonding, waar er sterke bacteriële afbraak plaatsvindt, situeert zich het pH minimum. In 1998 en 1999 daalde pH er nog sporadisch onder 7 (Vlarem II norm), nadien nooit meer. In het zoete daarentegen overschreed de pH in 2008 voor het eerst de bovengrens van de Vlarem II norm van 8.5.

2.7. Silicium

Opgelost Silicium (DSi) kent een duidelijk seizoenaal patroon, met zomerse minima tijdens pieken van consumptie door diatomeeën. Surfer 2.5 illustreert de trendbreuk tussen de periode 1996-2002 en 2003-2008. In de jaren '90 kwam DSi depletie slechts sporadisch voor, de pieken in algenbloei waren immers beperkt. In de zeer natte zomers van 2000, 2001 en 2002 was er wellicht ook een verhoogde aanvoer van silicium uit het bekken.

Vanaf 2003 is silicium depletie een jaarlijks terugkerend fenomeen, waarbij het probleem zich niet meer beperkt tot het zoete alleen. De beperkte DSi-depletie van de jaren '90 concentreerde zich nog tot de Boven-Zeeschelde. Eenmaal voorbij de Rupelmonding nam de concentratie DSi terug toe. De Rupel, waar algenbloei beperkt is, is een belangrijke bron aan DSi. Sinds 2003 worden, door de veel sterkere concumptie, de zones met lage DSi-concentratie echter groter en rijken tot aan de grens. Met de verbetering van de waterkwaliteit in de Rupel, zal de input van DSi vanuit de Rupel mogelijks afnemen door lokale consumptie. Het risico bestaat dat bij lage afvoerdebieten, de zomerse DSi-depletie niet meer beperkt zal blijven tot zoet, maar de ganse Zeeschelde kan bestrijken. In 2008 werd voor het eerst depletie waargenomen ter hoogte van station Grens.

2.8. Stikstof

Sinds de start van de Omes metingen, zijn de ammoniumconcentraties spectaculair gedaald Surfer 2.13). In de jaren '90 werd de grens van 5 mg/l (Vlarem II) nog frequent overschreden, sinds 2000 niet meer. De daling gaat gepaard met een stijging van de zuurstofwaarden. Hogere zuurstofwaarden laten een efficiënte nitrificatie toe, waarbij ammonium wordt omgezet in nitraat. Vermits bacteriële nitrificatie temperatuursafhankelijk is, kent de ammoniumconcentratie een sterk seizoenaal patroon met maxima in de winter (ondanks de hogere verdunning). De daling van NH_4 doet zich zowel in de zijrivieren als in het estuarium zelf voor. De jaargemiddelde waarden in het estuarium liggen wel systematisch lager dan deze in de Rupel en Bovenschelde.

De nitraatwaarden stijgen parallel met de daling van ammonium (Surfer 2.11). Door de toegenomen nitrificatie, is een opmerkelijke stijging van nitraat in de zomer meetbaar. In de zomer speelt echter nog een ander fenomeen: denitrificatie. In anoxische omstandigheden wordt NO3- omgezet in N2-gas en verdwijnt het uit het systeem. Vermoedelijk speelde deze denitrificatie een belangrijke rol in de zuurstofarme Zeeschelde in de jaren '90. Nitraat kende een seizoenaal patroon met duidelijke minima in de zomer wanneer de denitrificatieactiviteit het grootst is. Met de stijgende zuurstofwaarden is deze denitrificatie mogelijk afgenomen. Alleszins is met de stijgende zuurstofwaarden de nitrificatie toegenomen. Het wordt daarom interessant om naar de totale anorganische stikstof te kijken (TDIN = NH4+ + NO2- + NO3-). TDIN vertoont eind de jaren ’90 sterk dalende trend, die zich doorzet tot in 2002 (Surfer 2.9). Weerom kunnen debieten een verdunnend effect gehad hebben. Vanaf 2003 daalt TDIN echter langzaam verder, ondanks de dalende debieten. Wel is een duidelijk seizoenaal patroon zichtbaar, met de hoogste concentraties in de winter. De lagere concentraties in de zomer worden wellicht verklaard door denitrificatie en mogelijke opname van stikstof door algen.

De aanvoer van TDIN vanuit Bovenschelde is de voorbije jaren beperkt afgenomen. De daling die in de Zeeschelde in de jaren ’90 wordt waargenomen, moet dus in hoofdzaak toegeschreven worden aan processen die zich in het estuarium zelf afspelen. De laatste jaren wordt wel een iets sterkere daling van TDIN waargenomen in de Bovenschelde en Rupel. Mogelijk werpt de inspanning naar tertiaire zuivering stilaan vruchten af, maar ook daling van de diffuse input onder impuls van maatregelen in het bekken kan een verklarende factor zijn. De gevolgen van de gedaalde input in de Zeeschelde, wegen echter niet zwaar door op de output naar de Westerschelde. De TDIN-concentraties ter hoogte van de Belgisch-Nederlandse grens zijn sinds 2003 tijdens de zomermaanden slechts beperkt gedaald. Mogelijk weegt het verlies aan denitrificatie zwaarder door dan een afname aan input in de Zeeschelde. In de wintermaanden wordt ter hoogte van de grens wel een daling van TDIN geregistreerd.

Wanneer enkel de organische stikstofcomponent wordt beschouwd (Kjehldahlstikstof – ammonium) valt zeker geen daling af te leiden (Surfer 2.10). De organische stikstoffractie vertoont een dalende trend tussen 1996 en 2002, mogelijks door verminderde input vanuit het bekken en verdunning. Vanaf 2003 neemt de concentratie terug toe. Lokale primaire en secundaire productie zorgen misschien voor deze verhoogde organische stikstofvracht.

2.9. Fosfaat en totaal fosfor

Fosfaat (Surfer 2.7) vertoont seizoenale schommelingen, wellicht gestuurd door schommelingen in debiet. Over de jaren heen is ook een duidelijke dalende trend waarneembaar. De zomer van 2005 vormt echter een uitschieter, met zeer hoge fosfaatwaarden in het zoete deel van de Zeeschelde. In 2006 zet de daling zich echter verder, met de laagste waarden in 2009. Toch wordt de norm (Vlarem II) nog niet steeds gehaald.

Ook totaal fosfor kent een duidelijk dalende trend, waarbij in 2005 een gelijkaardige uitschieter wordt waargenomen in het zoete (Surfer 2.6).

2.10. N/P/Si ratio’s

Bij dalende nutriëntconcentraties, wordt het interessant de nutriënt ratio’s te beschouwen. De molaire Redfield ratio voor N/P bedraagt 16/1. In de periode 1996-2001 is stikstof duidelijk het meest abundante element, behalve in de zomermaanden (Surfer 2.17). Dan wordt fosfor het limiterende element in het zoete (hoewel van limitatie niet echt sprake is). Zowel N als P kennen een duidelijk dalende trend. De daling van P is relatief gezien sterker dan die van N, zodat vanaf 2001 P steeds het limiterende element wordt. In het zoete lijkt de P limitatie nog toe te nemen, in het brakke neemt ze licht af. Een uitzondering hierop vormt de zomer van 2005, wanneer P bijzonder hoge waarden kent in het zoete.

De molaire Si/N ratio’s wijzen steeds op een duidelijke aan overmaat aan N realtief ten opzichte van Silicium (Surfer 2.16).

In 1996 en 1997 was, ten opzichte van fosfor, opgelost silicium steeds het limiterende element in het zomerhalfjaar (Surfer 2.15). In de winter, bij hoge aanvoer van DSi, werd fosfor het limiterende element in het brakke en oligohaliene deel van de Zeeschelde. Tussen 1997 en 2002, een periode met toenemende debieten, zien we de fosforlimitatie toenemen. In de droge zomers van 2003, 2004 en 2005 tekent zich terug duidelijk een silicium limitatie af. Deze limitatie lijkt terug af te nemen in de periode 2006-2008.

2.11. Biologische zuurstofvraag

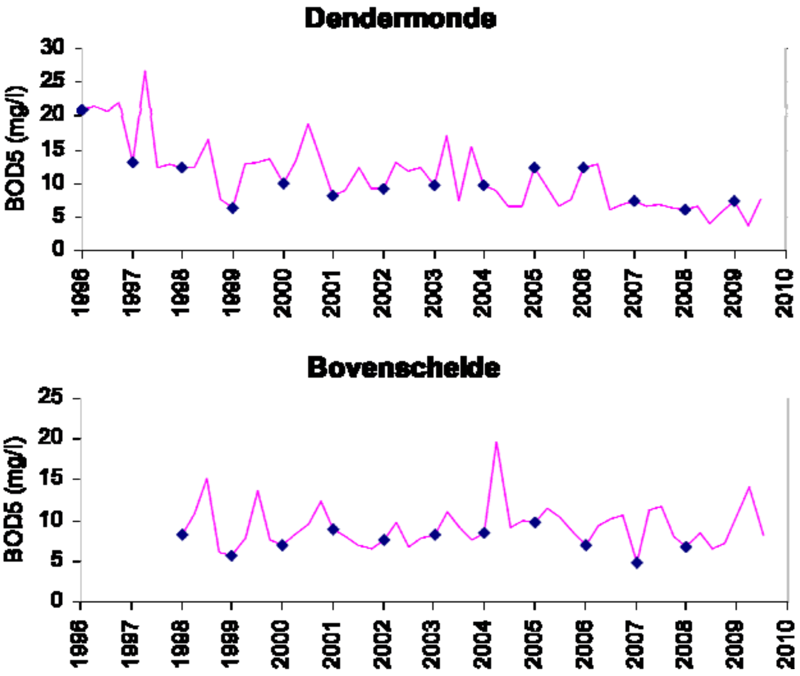
De biochemische zuurstofvraag (BOD5) is een complexe parameter en kent een grillig verloop (Surfer 2.8). BOD5 is en maat voor de organische belasting in het water, en wordt bepaald door een waterstaal gedurende 5 dagen donker te incuberen bij 20°C. De verbruikte zuurstof gedurende deze periode, is een maat voor de aanwezige hoeveelheid biologisch afbreekbaar materiaal.

De BOD5 bestaat in hoofdzaak uit een biochemisch afbreekbare koolstoffractie en een ammoniumfractie, welke bij nitrificatie ook zuurstof zal opeisen. Niet alle fracties breken echter even snel af, en deze afbraak is mede afhankelijk van de reeds aanwezige microbiële populaties in het Scheldewater. Daarom is interpretatie van de data niet eenvoudig. Toch is een duidelijke trend waarneembaar. De BOD5 concentraties zijn in dalende lijn (Surfer 2.8), in hoofdzaak door snellere afbraak van de vuilvracht in het estuarium zelf onder de verbeterde zuurstofcondities. De input van BOD5 vanuit de zijrivieren is immers quasi niet gedaald. In de Bovenschelde (Figuur 2.10) zijn duidelijk maxima in de zomer, minima door verdunning in de winter.

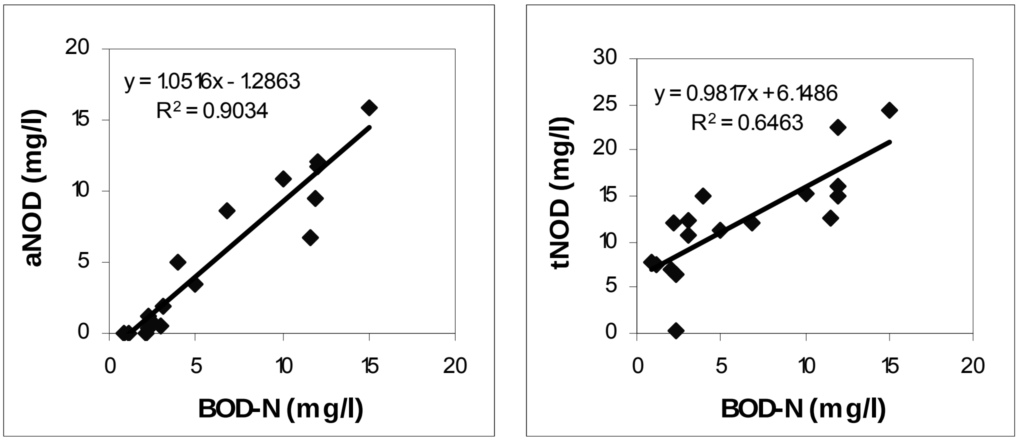
De daling in de Zeeschelde is het best meetbaar in het zoete tijdens de zomer, waar de zuurstofwaarden het meest zijn toegenomen. In de zomer zijn daar zuurstof en temperatuur hoog genoeg voor een efficiënte afbraak van de vuilvracht. Te Dendermonde kende BOD5 in de periode 1996-2002 minima in de winter, wellicht door verdunning (Figuur 2.10). Vanaf 2004 daalt de BOD sterk tijdens de zomermaanden zodat de zomerdata in 2004, 2005 en 2006 onder de winterwaarden liggen. In 2007, 2008 en 2009 dalen ook in de winter de BOD-waarden, waardoor de seizoenaliteit uit de data verdwenen is.

Dankzij de grote dataset van Omes wordt het mogelijk een inzicht te krijgen in welke fracties in de vuilvracht sneller afbreken en zorgen de zomerdaling in BOD5. Mineralisatie van de vuilvracht en nitrificatie van alle NH_4^+ zijn de belangrijkste zuurstofconsumerende processen. Experimenten met nitrificatieremmers (BOD N-serve) in het BOD-staal, laten toe de zuurstofvraag voor nitrificatie te onderscheiden van andere processen. Deze BOD N-serve metingen werden tussen 1998 en 2004 uitgevoerd.

De zuurstofvraag voor nitrificatie (gedurende 5 dagen bij 20°C) (BOD-N) wordt experimenteel bepaald door BOD N-serve data (bepaald door toevoegen van een nitrificatieremmer) af te trekken van BOD5 data. De zuurstofvraag voor nitrificatie kan ook berekend worden uitgaande van alle stikstofcomponenten aanwezig in het water. De anorganische stikstof-zuurstofvraag (aNOD) werd berekend uitgaande van het aanwezige NH_4^+ en NO_2^- en de zuurstof vereist om deze te oxideren tot NO_3^-. Figuur 2.11 toont voor de zomermaanden (juli-augustus-september) een goede correlatie tussen de experimenteel bepaalde BOD-N en de berekende aNOD te Dendermonde. Het verschil tussen de gemeten BOD5 en de berekende aNOD geeft de BOD-C: de biologische zuurstofvraag voor de afbraak van de koolstofverbindingen. De goede correlatie die tussen BOD-N en aNOD wordt gevonden in de zomermaanden te Dendermonde, bestaat ook voor de andere stations in het zoete, behalve voor de twee meest opwaartse stations. Te Melle en Wetteren is de correlatie zoek (data niet weergegeven). Misschien is tijdens de zomermaanden de populatie nitrificeerders hier nog niet op peil. In de Bovenschelde wordt ook geen goede correlatie gevonden.



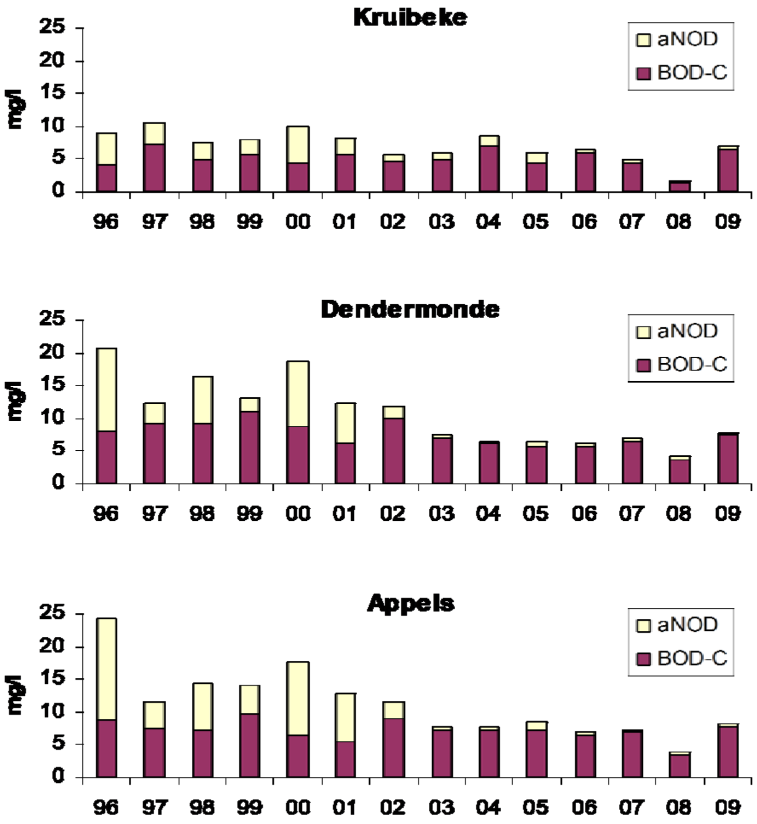
Figuur 2.10 Seizoensgemiddelde BOD5-waarden in de zoete Zeeschelde (Dendermonde) en in de Bovenschelde. De ruitjes markeren de winterperiode.



Figuur 2.11: correlaties tussen BOD-N (biologische zuurstofvraag voor nitrificatie) en aNOD (berekende anorganische stikstof-zuurstofvraag) en BOD-N en tNOD (berekende totale stikstof-zuurstofvraag) voor de zomermaanden (juli-aug-sept) te Dendermonde.

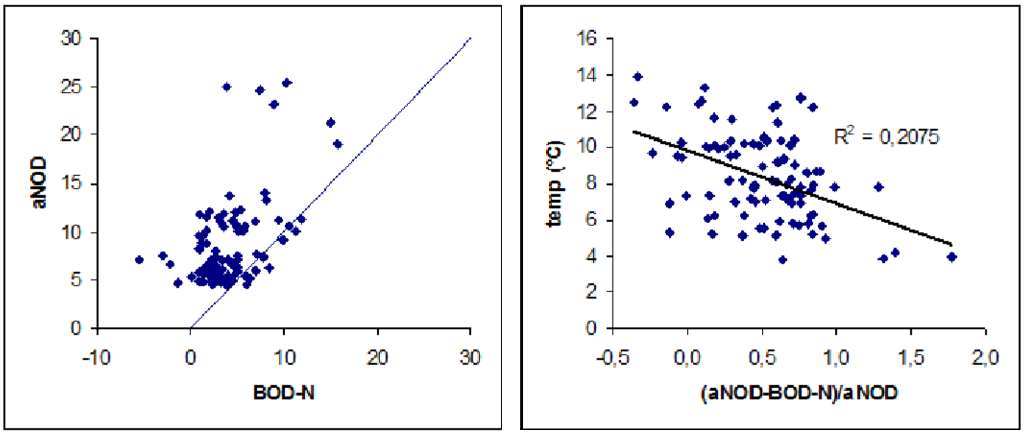
Op basis van de gevonden correlaties kan voor verschillende stations de bijdrage van nitrificatie aan de BOD5 ingeschat worden voor de zomermaanden (Figuur 2.12). Hieruit blijkt dat de dalende BOD5-trend in de zomer in hoofdzaak toe te schrijven is aan de sterke daling van de aNOD tussen de jaren 1996-2002, vanaf 2006 is de stikstoftrap quasi verdwenen. De BOD-C blijft echter constant. Naarmate men van Appels afwaarts gaat richting Dendermonden en Kruibeke is, vooral in de jaren ’90, een duidelijke afname van aNOD waarneembaar. Voor BOD-C is dit veel minder uitgesproken.

Voor de andere seizoenen geeft deze berekening geen goede resultaten. Enkel voor de zomermaanden vinden we een overeenkomst tussen de berekende aNOD en de gemeten BOD-N, in de wintermaanden is dit verband vaak zoek (Figuur 2.13). De verklaring is logisch: nitrificerende bacteriën groeien traag en zijn sterk afhankelijk van temperatuur. Enkel in de zomermaanden is een actieve populatie aanwezig. In de andere seizoenen is deze populatie wellicht minder of niet actief aanwezig. Bij incubatie bij 20°C gedurende 5 dagen tijdens de BOD5-meting, is het bijgevolg niet duidelijk welke de invloed van nitrificerende bacteriën zal zijn op de zuurstofvraag. In wintermaanden liggen de gemeten BOD5 en BOD N-serve waarden vaak zeer dicht bij elkaar, wat geenszins wijst op afwezigheid van een stikstof-zuurstofvraag, maar wel op de afwezigheid van nitrificeerders. De relatieve afwijking van de gemeten BOD-N ten opzichte van de berekende aNOD vertoont een negatieve correlatie met temperatuur (Figuur 2.13).



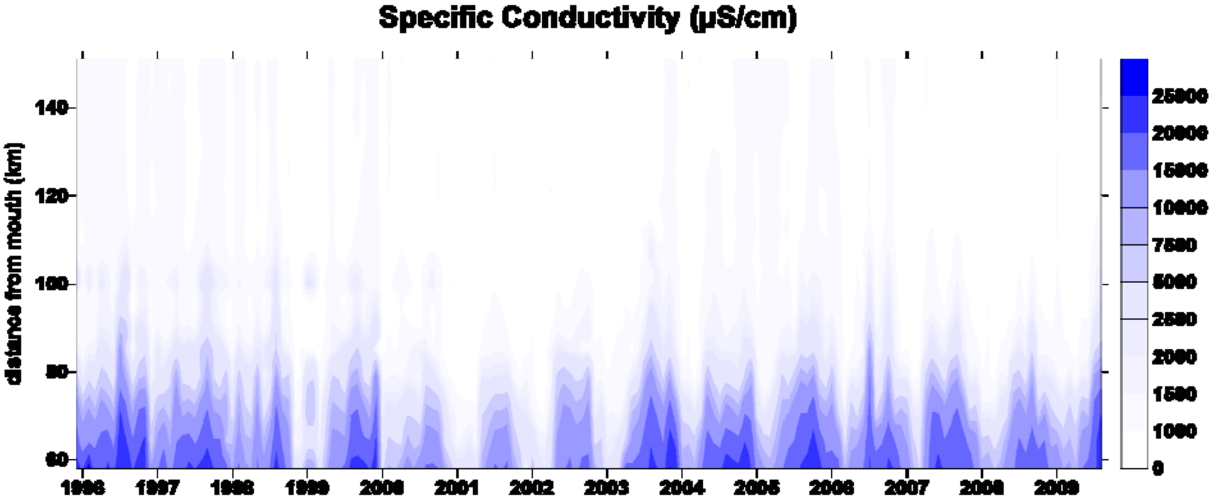
Figuur 2.12 Gemiddelde zomerwaarden voor de berekende anorganische stikstof-zuurstofvraag (aNOD) en de berekende koolstof-zuurstofvraag (BOD-C) te Kruibeke, Dendermonde en Appels.

Naast anorganische stikstof, zal ook de organische stikstof zuurstof onttrekken aan de waterkolom bij mineralisatie en nitrificatie. De som van deze zuurstofvraag en de aNOD is de totale stikstof-zuurstofvraag (tNOD). De correlatie tussen BOD-N en tNOD is echter minder goed dan tussen BOD-N en aNOD. Blijkbaar wordt gedurende de 5 dagen van de BOD5-meting in hoofdzaak alle anorganische stikstof genitrificeerd, en speelt de mineralisatie van de organische fractie een beperkte rol gedurende de duur van de BOD5-meting. De constante term uit de correlatievergelijking tussen BOD-N en aNOD (1.29 mg/l) kan een maat zijn voor de gemiddelde zuurstofvraag ten gevolge van afbraak van de organische stikstoffractie. De constante term uit de correlatievergelijking tussen BOD-N en tNOD (6.15 mg/l) kan een maat zijn voor de gemiddelde zuurstofvraag van de organische stikstoffractie die nog niet werd afgebroken gedurende de 5 dagen van de BOD5-metingen. Samen vormen deze termen een maat voor de gemiddelde totale organische stikstofzuurstofvraag.

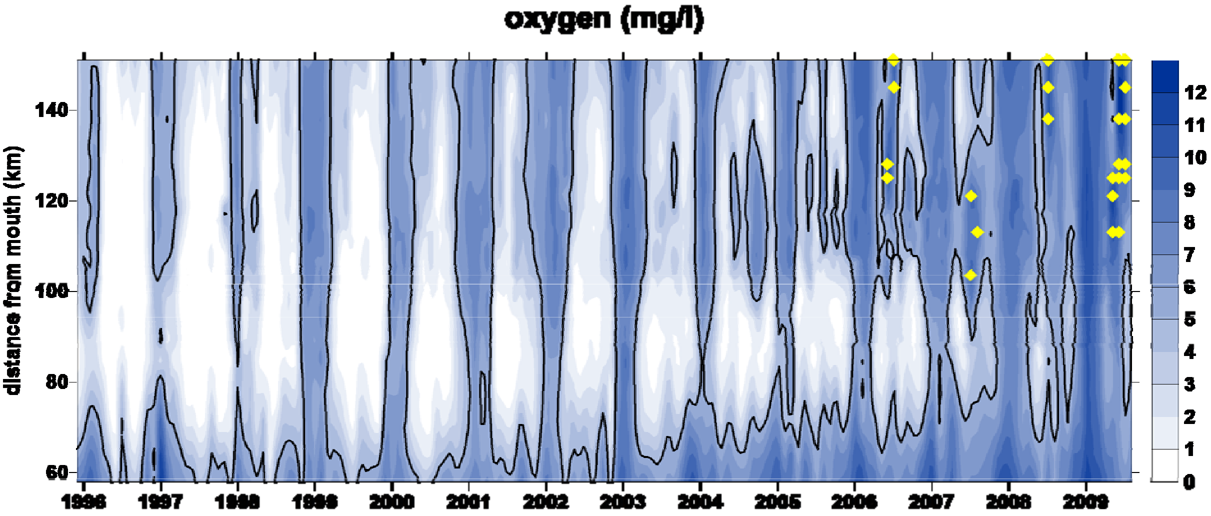


Figuur 2.13 Links: Correlatie tussen BOD-N (biochemische zuurstofvraag voor nitrificatie) en aNOD (berekende anorganische zuurstofvraag) tijdens winterhalfjaar (periode november – april). Rechts: relatieve afwijking van de gemeten BOD-N ten opzichte van de berekende aNOD ((aNOD - BOD-N)/aNOD) uitgezet tegen de watertemperatuur.

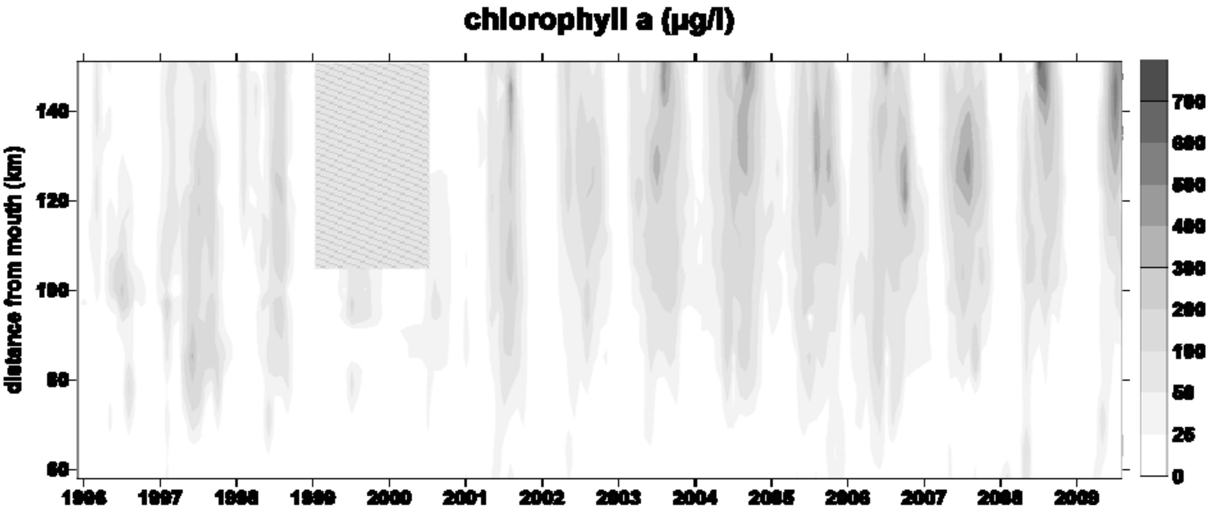
2.12. Surfers



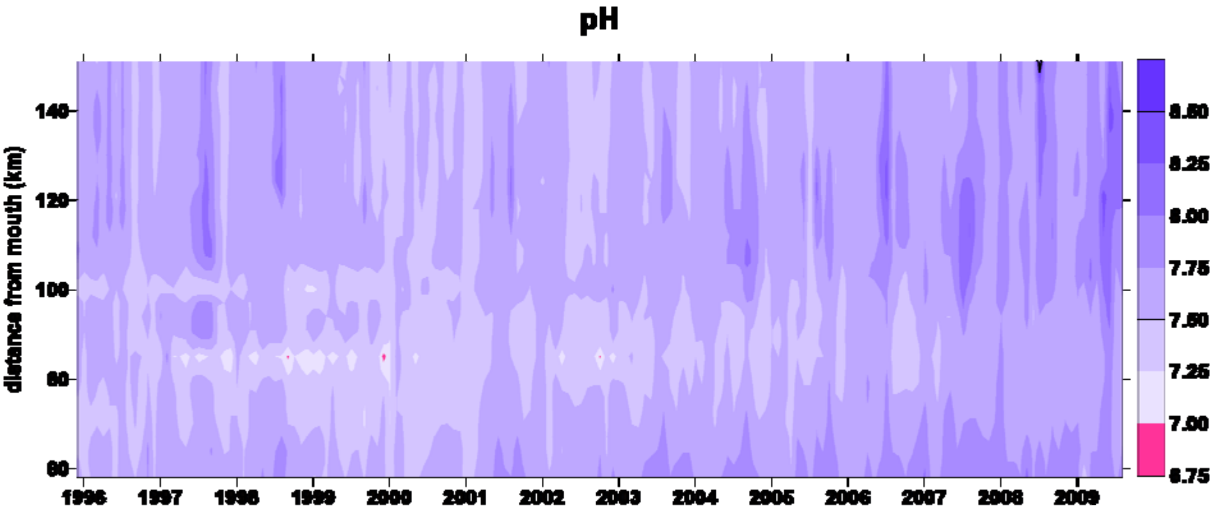
Surfer 2.1 Specifieke geleidbaarheid (µS/cm)



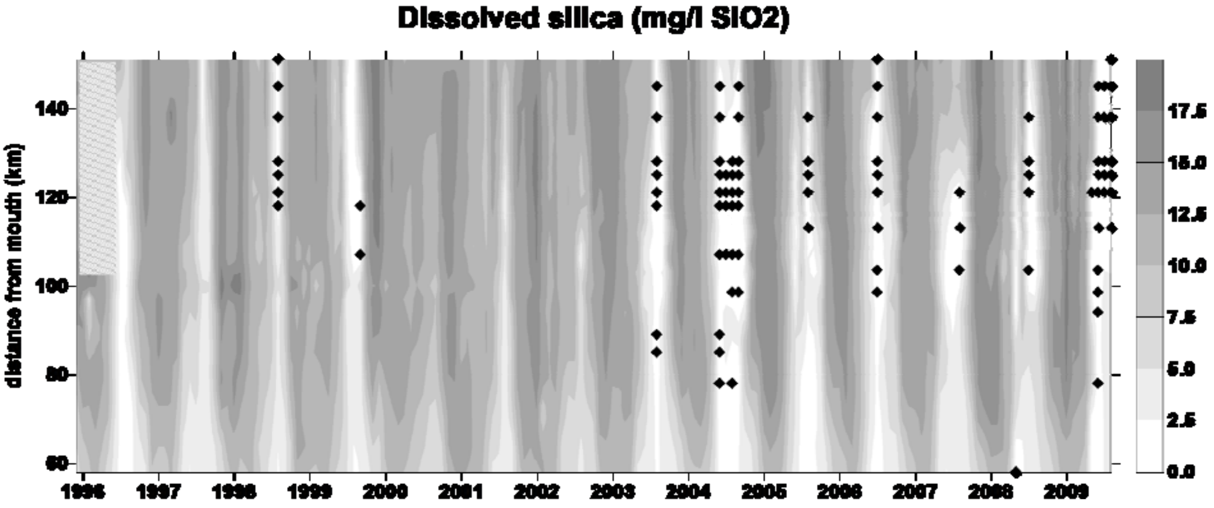
Surfer 2.2 Zuurstofconcentratie (mg/l). De zwarte lijn markeert de 5 mg/l norm (Vlarem II). De ruitjes duiden op momenten van oververzadiging (zuurstofsaturatie > 100%).



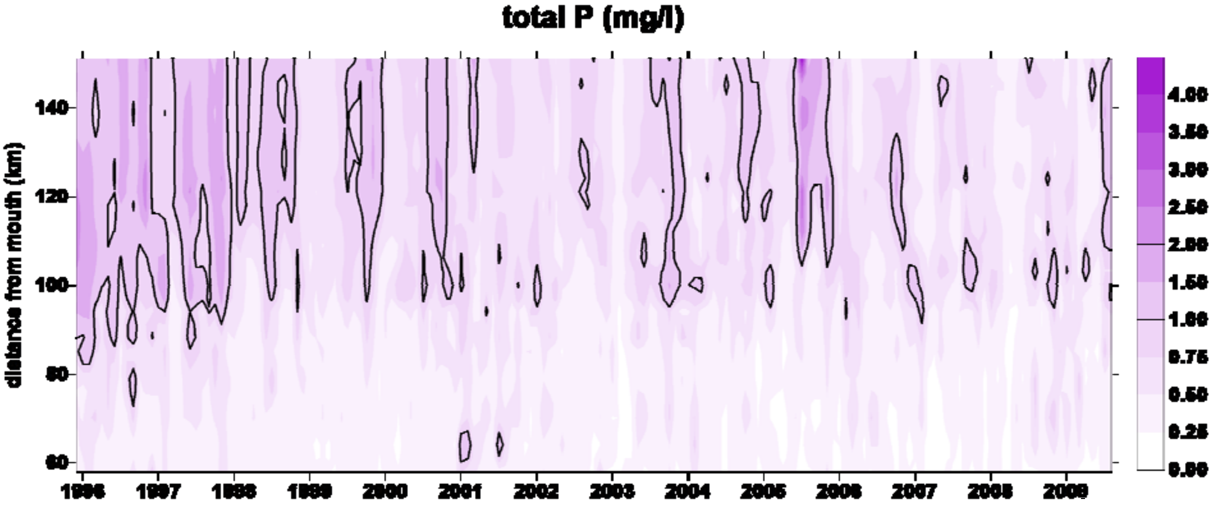
Surfer 2.3 Chlorofyl a concentratie (µg/l).



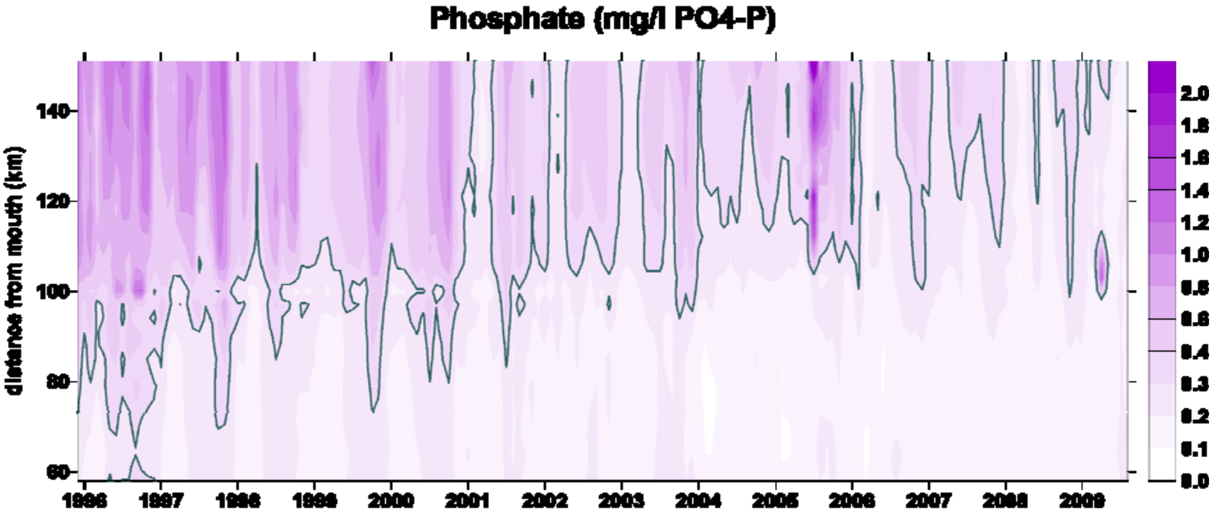
Surfer 2.4 Zuurtgraad (pH)



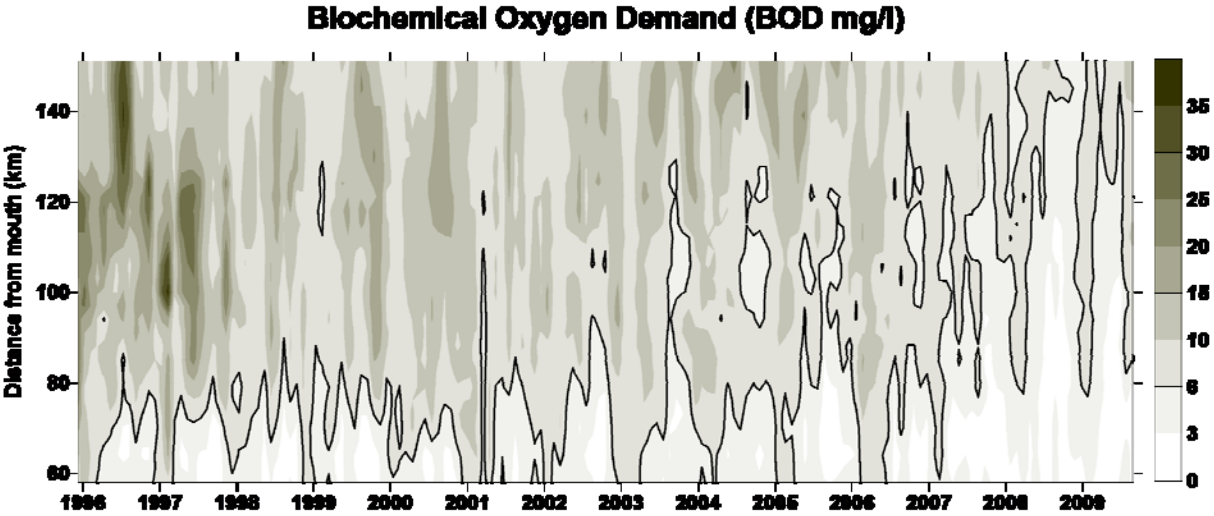
Surfer 2.5 Opgelost Silicium (mg/l SiO_2). de ruitjes duiden op momenten van Si depletie (SiO_2<0.01mM)



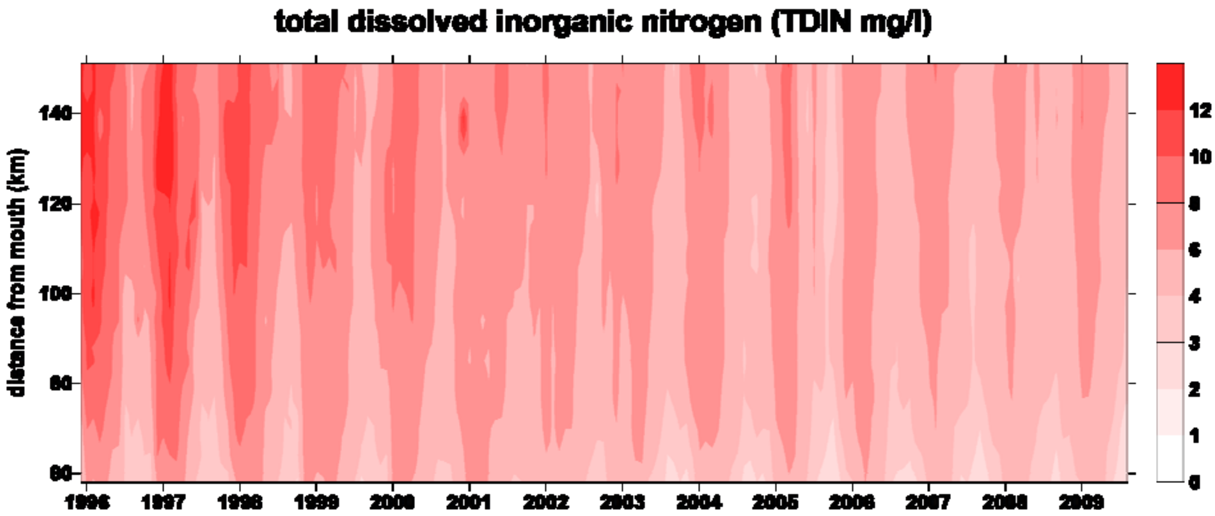
Surfer 2.6 totaal fosfor (mg/l). De zwarte lijn markeert de 1 mg/l norm (Vlarem II).



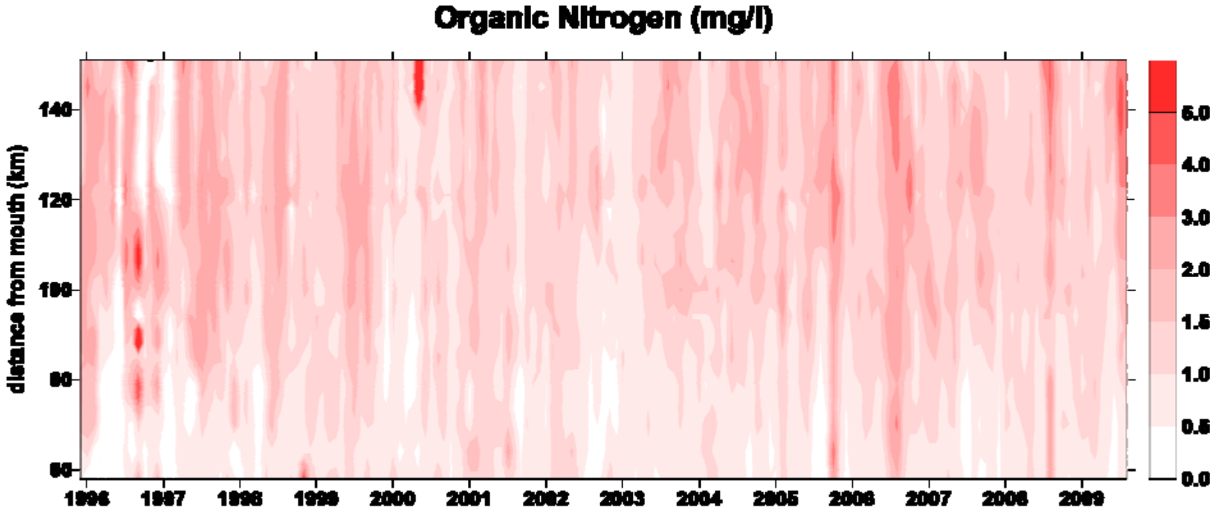
Surfer 2.7 Fosfaat (mg/l PO4-P). De zwarte lijn markeert de 0.3 mg/l norm (Vlarem II).



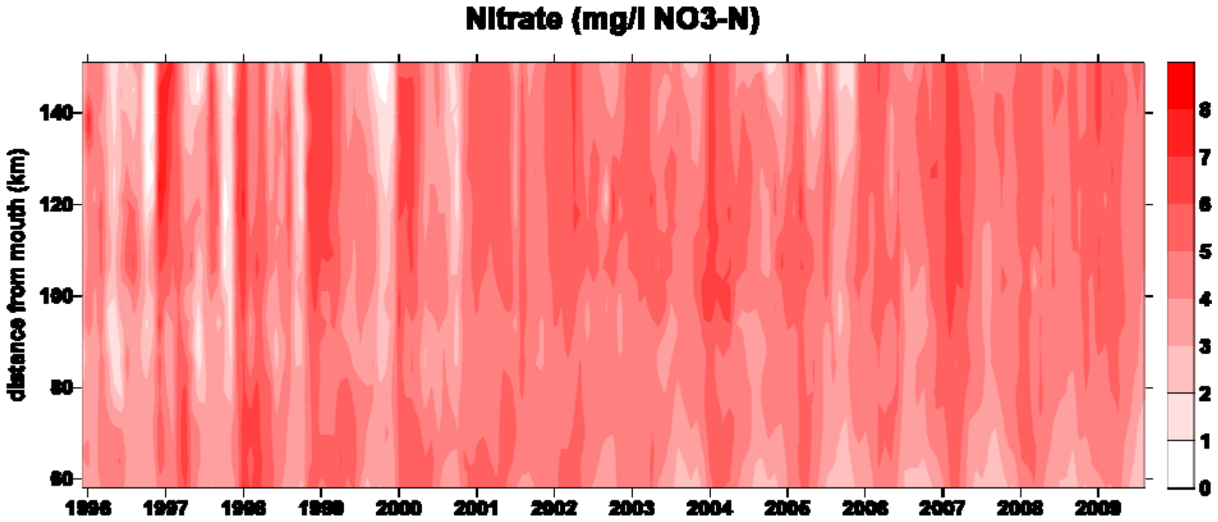
Surfer 2.8 Biochemische Zuurstofvraag (BOD mg/l). De zwarte lijn markeert de 6 mg/l norm (Vlarem II).



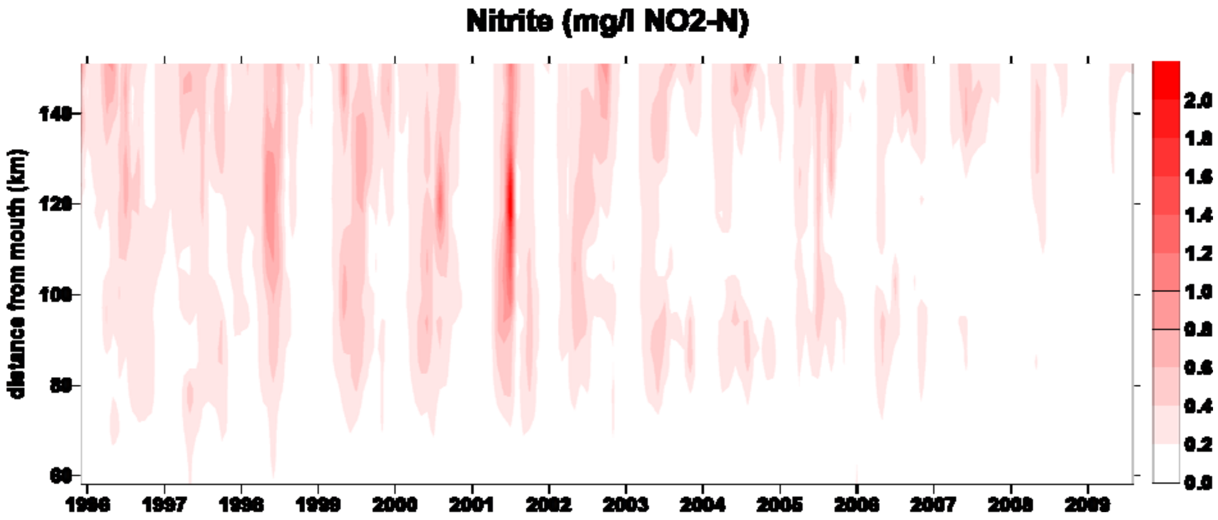
Surfer 2.9 Totaal opgeloste anorganische stikstof (mg/l) (TDIN = NH_4^+ + NO^{2-} + NO^{3-})



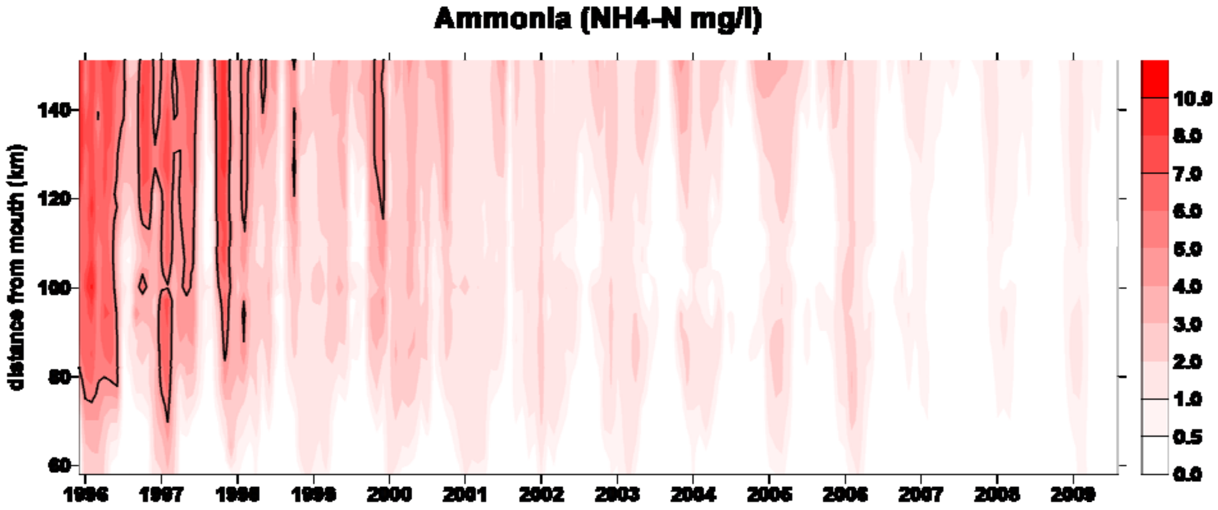
Surfer 2.10 Organische stikstof (mg/l) (Org N = Kjehldal N - NH_4^+-N)



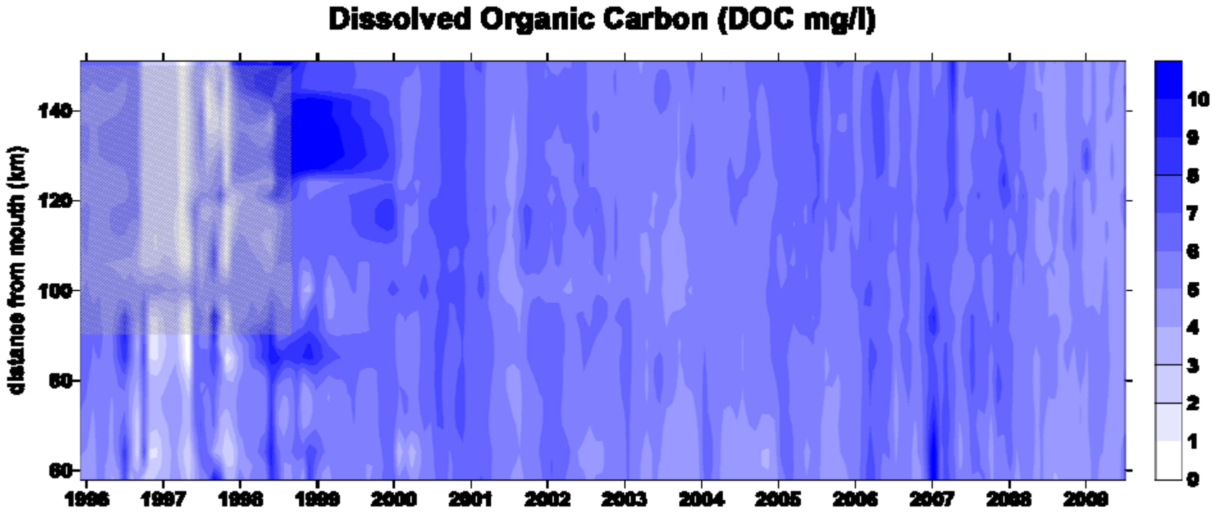
Surfer 2.11 Nitraat (mg/l NO^{3-}-N)



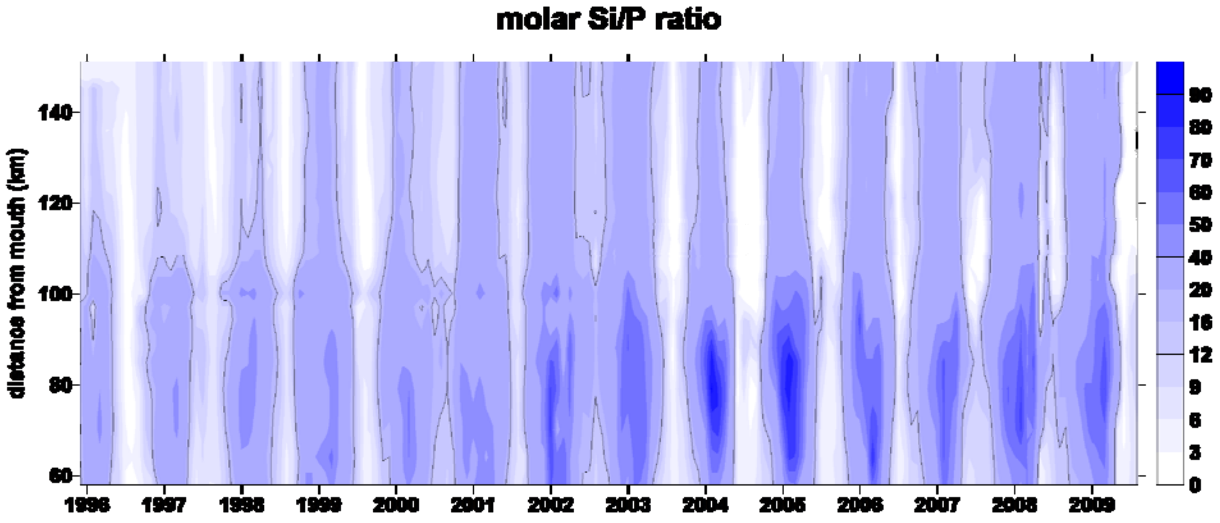
Surfer 2.12 Nitriet (mg/l NO^{2-}-N)



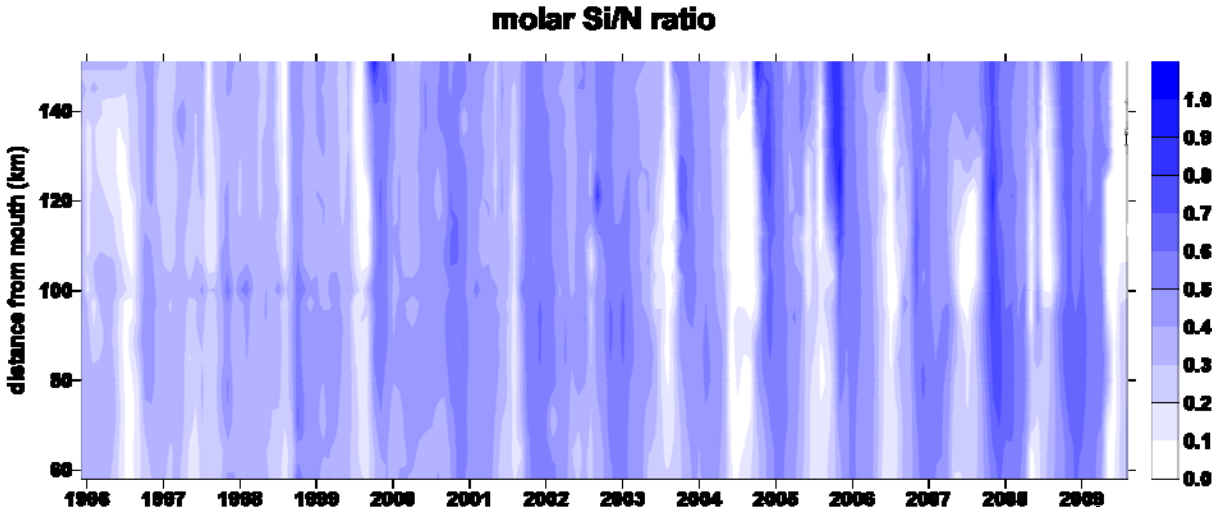
Surfer 2.13 Ammonium (mg/l NH^{4+}-N)



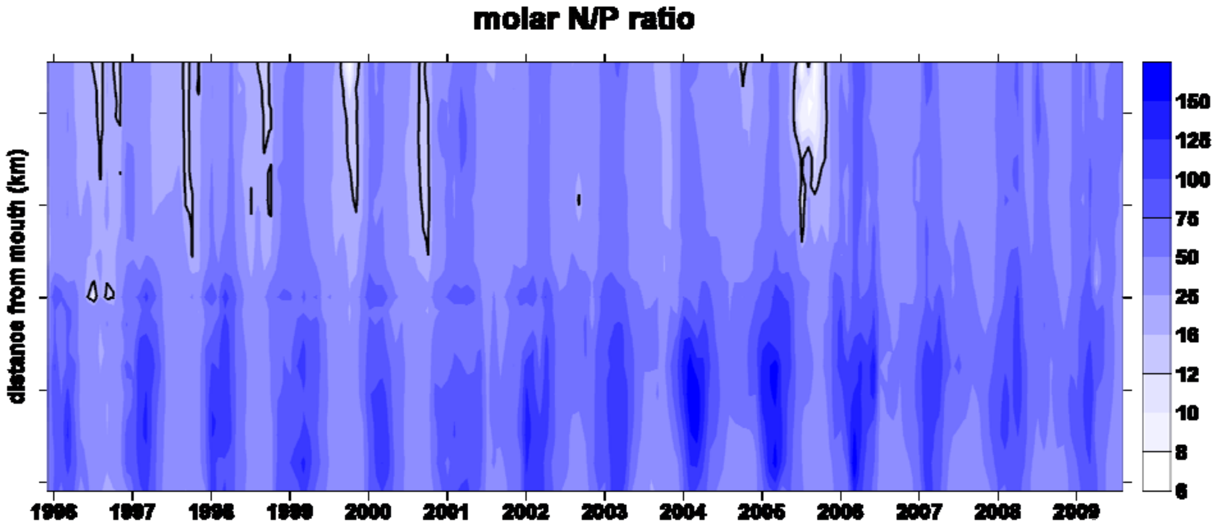
Surfer 2.14 Opgeloste organische koolstof (mg/l)



Surfer 2.15 Molaire Si-P ratio. De zwarte lijn markeert de theoretische Redfield ratio (16).



Surfer 2.16 Molaire Si-N ratio.



Surfer 2.17 Molaire N-P ratio. De zwarte lijn markeert de theoretische Redfield ratio (16).

Hoofdstuk 3. Koolstofcyclus in het Vlaams gedeelte van Schelde-estuarium

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Eindverslag voor deelstudie 2 (perceel 2), periode juli 2008-augustus 2009

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3.1. Introduction

This report presents the results obtained during the OMES campaigns from July 2008 to August 2009. Different parameters were measured during monthly campaigns along a longitudinal transect in the Zeeschelde between the Dutch-Belgian border and Gent as well as during three consecutive 13 hours campaigns in 3 stations.
The studied parameters include: particulate organic carbon (POC), particulate nitrogen (PN), total alkalinity (Talk) and the isotopic signature of dissolved inorganic carbon (DIC).

3.2. Materials and methods

3.2.1. Sampling

Sampling was carried out monthly at 15 stations in de Zeeschelde, as well as 5 stations in the mouth of the major tributaries being Rupel, Durme, Dender, Bovenschelde and Zandvliet from July 2008 to August 2009. Eight stations, located between km 58 and km 113 (i.e. distance from Vlissingen, = km 0), and the five tributaries were sampled day 1. Seven stations, located between km 118 and 151, were sampled day 2.
In 2009, the sampling planning changed and included two additional tributaries (Grote Nete and Kleine Nete) from January to August, and four new stations along the Rupel from April to August, increasing the number of sampling days to three.

Surface water was sampled with a clean bucket. Subsamples of 12 mL were taken in duplicate in glass vials for later determination of the isotopic signature ($\delta^{13}C$) of dissolved inorganic carbon. 15 µL of a saturated $HgCl_2$ solution were added to inhibit all further biological activities. The vials were then sealed with adapted stopper avoiding headspace formation. Subsamples were also taken in clean polyethylene (PE) containers for particulate organic carbon and nitrogen. Well homogenised fractions (2/station) of 150 to 500 mL were subsequently filtered on board using pre-ashed (450°C) GF/F filters (Sartorius, 47 mm diameter). Filters were packed in Millipore Petri-dishes and stored at -20°C till later on-shore analysis. A part of the filtrat was stored at 6°C in 100 mL PE vials with 20 µL $HgCl_2$ for later determination of total alkalinity.

3.2.2. Analysis

Total alkalinity was determined on 25 milliliter sample by automatic titration (Mettler-Toledo) using a standardised 0.01N HCl solution (Merck). The titration was conducted till the bicarbonate end-point. The dissolved inorganic carbon content (DIC = sum of bicarbonate, carbonate and dissolved CO_2) and the partial pressure of CO_2 can be calculated from these TAlk values and the pH data obtained on board, following Cai and Wang (1998).

For determination of the isotopic ratio of DIC, samples were prepared 12 hours before injection in a CN – Conflo – Isotopic Ratio Mass Spectrometer (IRMS) with the following method. Three milliliter headspace volume was created in the serum vials by replacing water by Helium injected through the vial septum. Then 0.5 mL of *ortho*-phosphoric acid (99% crystal) was injected and the sample was left overnight for CO_2 extraction in the headspace to proceed. 300 µL of the He-CO_2 mixture were injected through a GC sample port mounted between the reduction oven and the water trap of the Elemental Analyser (EA) IRMS system (single inlet Finnigan Mat Delta V Plus). Efficiency of the extraction was assessed via repeated analysis of aliquots of reference CO_2 gas. The $^{13}C/^{12}C$ isotopic ratio of DIC is presented in the results as a $\delta^{13}C_{DIC}$ value, obtained by the following equation:

$$\delta^{13}C_{DIC}\,(sample) = \frac{\frac{^{13}C}{^{12}C}sample - \frac{^{13}C}{^{12}C}std}{\frac{^{13}C}{^{12}C}std} \times 1000 = \pm \;\; x \;\; ‰\,,$$

where std is an international standard PDB (PeeDee Belemnite) for carbon.

Filters were thawed and dried at 50°C till constant weight over night. To assess the organic C and N contents and isotopic ratios, portions of 12 mm diameter discs were cut-out from these filters and left 4 hours in contact with HCl vapour in a evacuated container to remove carbonates. These filters discs were subsequently dried at 50°C for few minutes and then, packed in tin cups. Blank filters were subjected to the same treatment. Weights amounts of acetanilide, nitrogen standard and carbon standard were used for further determination of POC/PN concentrations, $\delta^{13}C_{POC}$ and $\delta^{15}N_{PN}$. The tin cups containing either the samples, blanks or standards are then ready for analysis on the EA-IRMS system.

3.3. Results

To better view the spatial and/or temporal evolutions of the different parameters, some results were regrouped by zones and/or seasons when they showed the same profile. Table 1 shows the abbreviations used to name the sampled stations.

Table.1 Abbreviations of the stations sampled in the Schelde

S1	Boei 87
S2	Boei 92
S3	Boei 105
S4	Antwerpen
S5	Kruibeke
S7	Steendorp
S8	Temse

S9	StAmands
S10	Vlassenbroek
S11	Dendermonde
S12	StOnolfs
S13	Appels
S14	Uitbergen
S15	Wetteren
S16	Melle

3.3.1. Total alkalinity (Talk)

Figure 1 represents the temporal evolution of TAlk in the longitudinal transect studied from July 2008 to May 2009. For most campaigns, the more we are in the freshwater zone, the higher the Talk values are. For each zone, the maxima are obtained in winter, but no clear seasonal trend can be observed. The comparison of these data with the measurements of pH and anion / cation concentrations (data from other OMES partners) would be helpful to interpret these results.

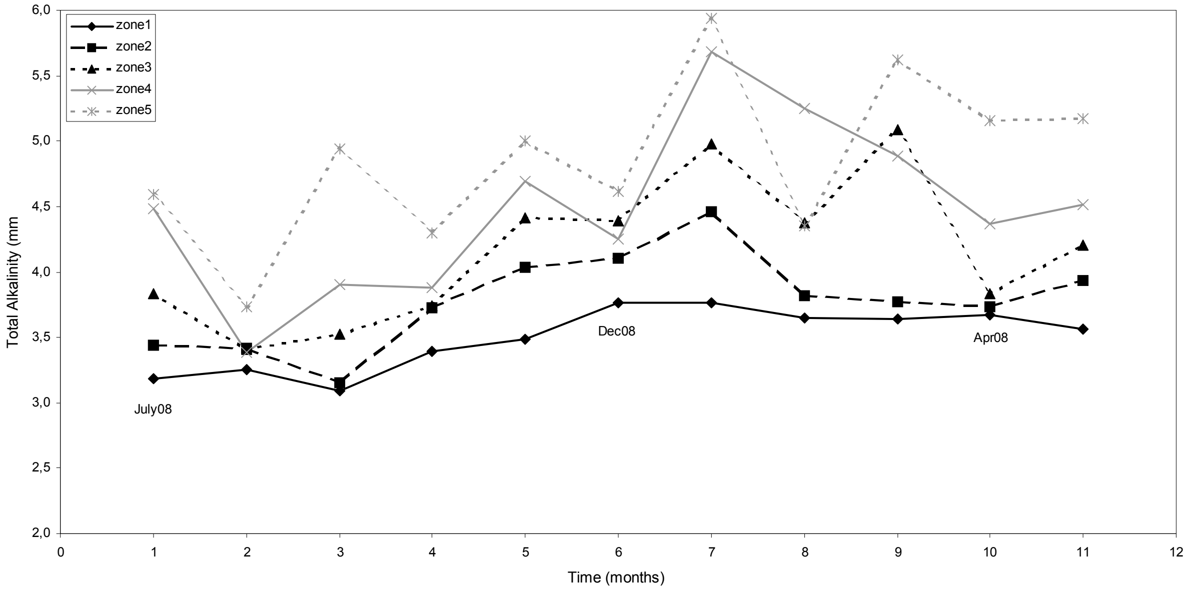

Figure 1. Temporal evolution of Talk (mmol L^{-1}) from July 2008 to May 2009 in five different zones of the longitudinal transect in the Scheldt. Zone 1 regroups S1 and S2, zone 2 regroups S3 and S4, zone 3 regroups S5 and S7, zone 4 regroups S8 and S9, zone 5 regroups S10 to S16.

3.3.2. Isotopic signature of dissolved inorganic carbon ($\delta^{13}C_{DIC}$)

Figure 2 shows the evolution of $\delta^{13}C_{DIC}$ along the Scheldt at different periods in 2008-2009. We can observe two specific trends corresponding to two seasons: the winter and the summer, and also two zones, the first one from S1 to S5 (the oligohaline zone) and the second one, from S7 to S16 (the freshwater zone). In winter, in freshwater, the mean value obtained (-13,2 ± 0,5) approaches the one from the groundwater that has its DIC system control in equal parts by carbonate dissolution (isotopic signature close to 0‰) and by heterotrophic respiration on soil C-3 organic matter (isotopic signature close to -27‰). The more we sample in the oligohaline zone, the higher the $\delta^{13}C_{DIC}$ value is.

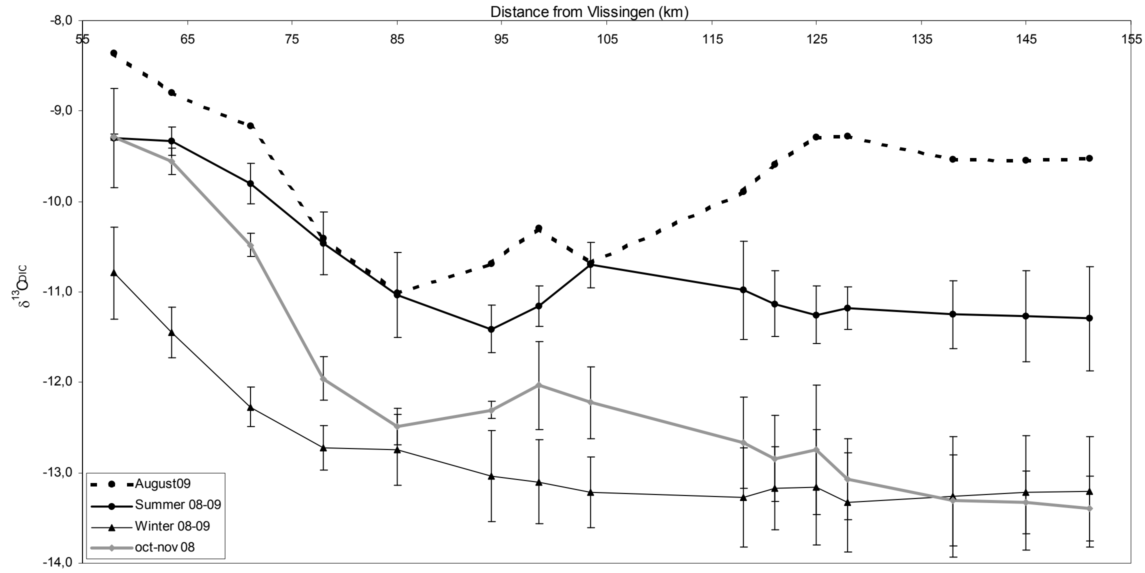

Figure 2. Spatial evolution of $\delta^{13}C_{DIC}$ (‰) along the Scheldt transect at six different periods. Winter 08-09 regroups December 2008, January, February, March and April 2009. Summer 08-09 regroups July, August and September 2008, and July 2009.

In summer, the longitudinal profile keeps the same trend, but the values are higher all along the transect, mainly due to a major biological process occurring in summer: the photosynthesis. Indeed, during the C-fixation by plants, $^{12}C\text{-}CO_2$ is preferentially incorporate. As a result, the DIC pool in the water becomes relatively enriched in ^{13}C and the $\delta^{13}C_{DIC}$ observed in summer show less negative values. The period of October-November 2008 is intermediary between the winter curve and the summer curve, with higher values in the oligohaline zone and lower ones in the freshwater zone. We can also note that in August 2009, the $\delta^{13}C_{DIC}$ values are much higher in freshwater from Vlassenbroek to Melle. For both periods, the inter-comparison with the Chlorophyll *a* data (from the OMES partners) would be necessary to explain if these specific profiles are due to phytoplanktonic blooms or not.

In the Rupel, the $\delta^{13}C_{DIC}$ values are often lower than in the Scheldt. This result is also confirmed by the values obtained in the Rupel river in the new sampled stations (since April 2009). There, the $\delta^{13}C_{DIC}$ values are equal or lower t the ones measured at the 'Rupel' station.

The values obtained in Bovenschelde are very close to the one measured in the Scheldt, in the same zone. In Dender, the $\delta^{13}C_{DIC}$ is lower than in the Scheldt, and show no clear seasonal trend.

3.3.3. Particulate organic carbon and nitrogen (POC/PN)

Figures 3 and 4 show the temporal evolution of POC and PN respectively in six different zones of the studied transect. These two parameters follow the same trend. The maxima are obtained in summer (mainly July 2008) and in freshwater. The minima are observed in winter (November and December 2008, January 2009) and in the oligohaline zone.

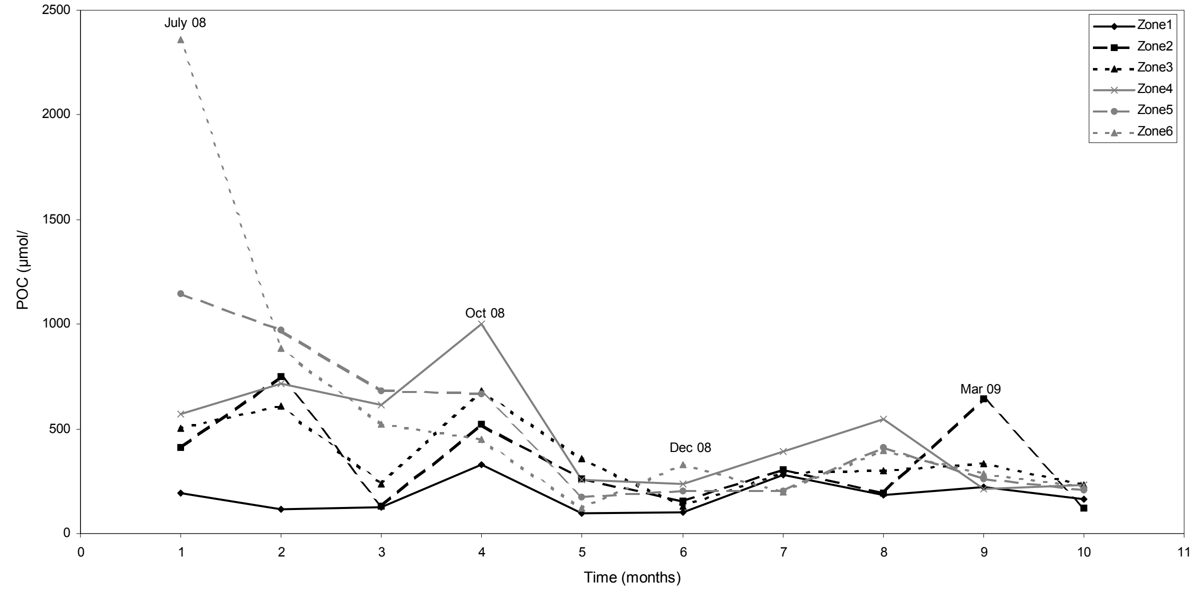

Figure 3. Temporal evolution of POC (µmol L^{-1}) in six different zones of the Scheldt. Zone1: S1; zone2: S2, S3 and S4; zone3: S5, S7 and S8; zone4: S9 to S13; zone5: S14 and S15; zone6: S16.

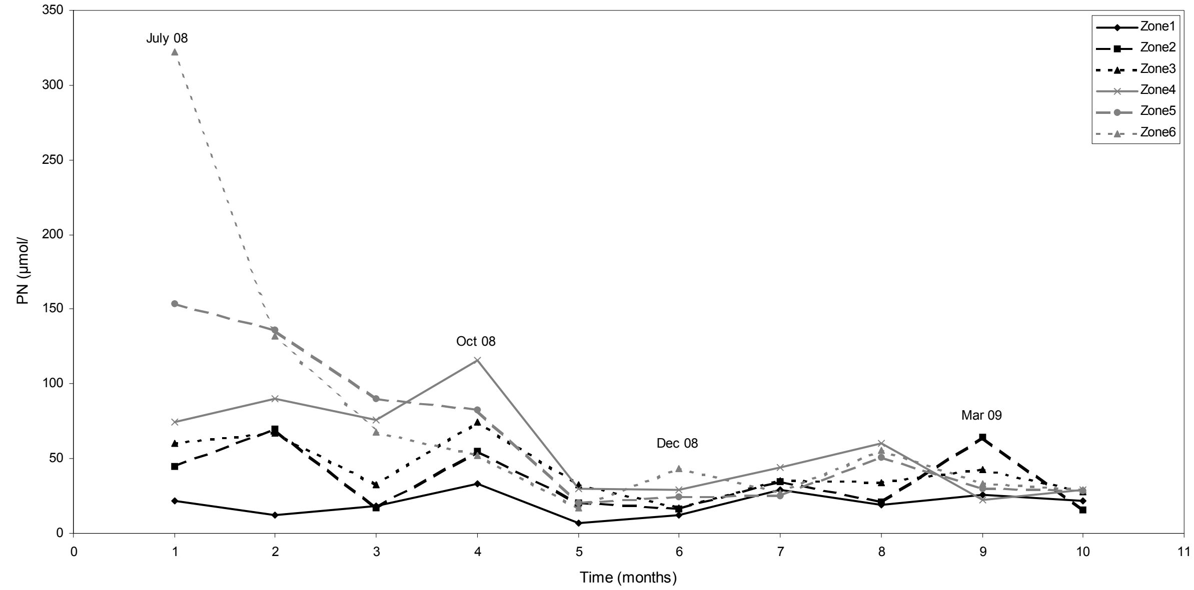

Figure 4. Temporal evolution of PN (µmol L^{-1}) in six different zones of the Scheldt. Zone1: S1; zone2: S2, S3 and S4; zone3: S5, S7 and S8; zone4: S9 to S13; zone5: S14 and S15; zone6: S16.

We can hypothesize that the high values of July 2008 and October 2008 come from, respectively, a summer bloom and an autumnal bloom of phytoplankton. During these periods, the particular organic matter (POM) pool is enriched by alive organisms. The data of suspended particular matter (SPM) and Chlorophyll *a* are necessary to see the correlations existing or not between these three parameters over the period 2008-2009.

3.4. 13hrs campaigns

3.4.1. Sampling and analyses

The 13hrs experiments were carried out during three consecutive days respectively in Schellebelle, Boom and Kruibeke, the 02, 03 and 04th September 2008. During these experiments, water samples were taken every hour during 13hrs at the same place to measure several parameters including $\delta^{13}C_{DIC}$, POC, PN and Talk.

The sampling procedure for each parameter is the same as the one described in 3.2.1. The analyses of the different parameters are described in the following paragraph.

3.4.2. Results

The temporal evolution of TAlk during these 13hrs experiments is shown in Figure 5. In Schellebelle and Kruibeke, no significant variation is observed during the sampling at different steps of the tide. For both stations, the variability observed during the day is not different to the variability obtained on replicates. In Schellebelle, the mean value for the whole day of 13hrs measurements was 5,02 ± 0,07 whereas the value obtained in the same station during the monthly campaigns was 4,96 ± 0,04. In Kruibeke, for the same parameter, we obtained respectively 3,35 ± 0,07 and 3,40 ± 0,12.

In Boom, even if, visually, the Talk seems to slighty fluctuate during the day, no significant difference was found between the values measured at different steps of the tide and the values obtained in replicates at the same step (t-test).

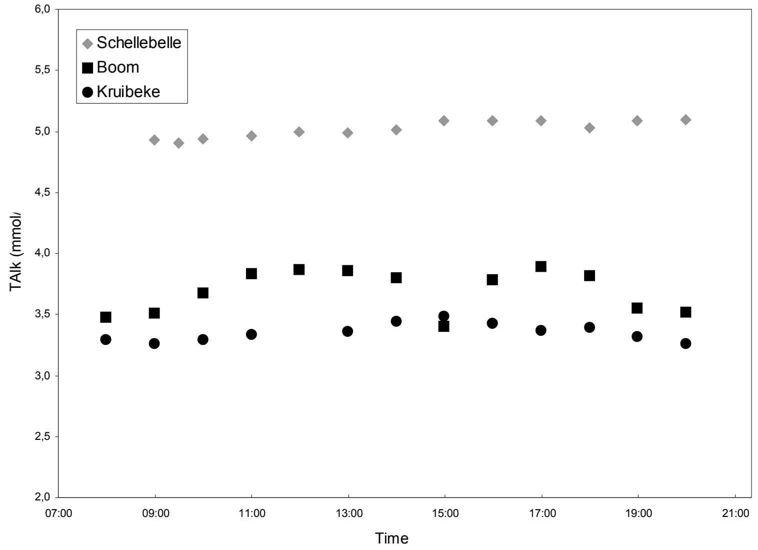

Figure 5. Temporal evolution of Talk (mmol L^{-1}) during the 13 hours experiments in Schellebelle, Boom and Kruibeke.

During the 13hrs experiments, the $\delta^{13}C_{DIC}$ varies with the tide (Figure 6). At high tide, the water entering in the estuary and river has a higher isotopic signature (characteristic of marine water). The mixing between the marine water and the freshwater is visible at 8:00 h and 20:00 h for the three stations. However, in Schellebelle and Kruibeke, the magnitude is smaller than in Boom. Indeed, the marine water has lower effect in Schellebelle, which is far from the marine zone, as well as in Kruibeke, where the high water volume (compare to the two other stations) limits the dilution effect.

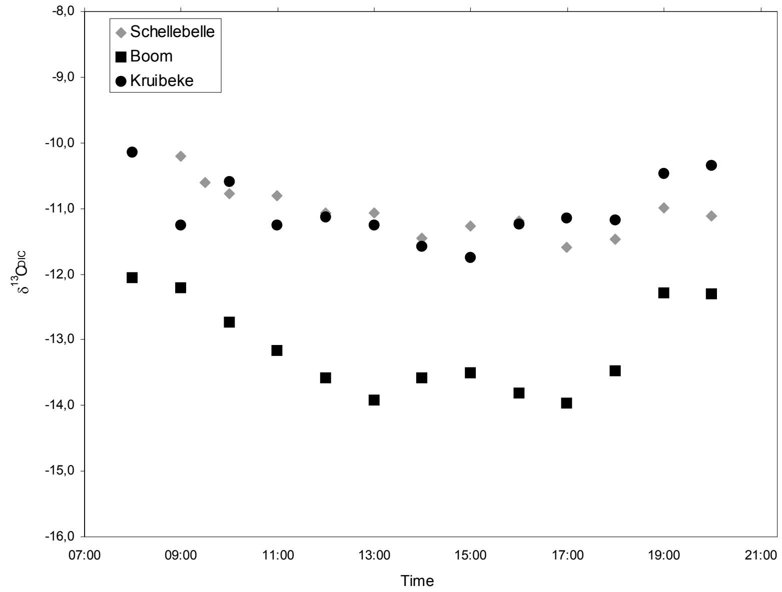

Figure 6. Temporal evolution of $\delta^{13}C_{DIC}$ (‰) during the 13 hours experiments in Schellebelle, Boom and Kruibeke.

During the 13hrs experiments, the POC and PN concentrations follow the same trend (Figure 7). In Kruibeke and Boom, the C/N ratio is close to 10, which may indicate that the POM pool is predominated by terrestrial organic matter and/or domestic and industrial wastes (allochthonous source).

In Schellebelle, the C/N ratio is around 7,5, which may indicate that the POM pool is, there, dominated by autochthonous sources (i.e. phytoplankton and derivates). To confirm these trends, the Chlorophyll *a* concentrations should be compared in each station.

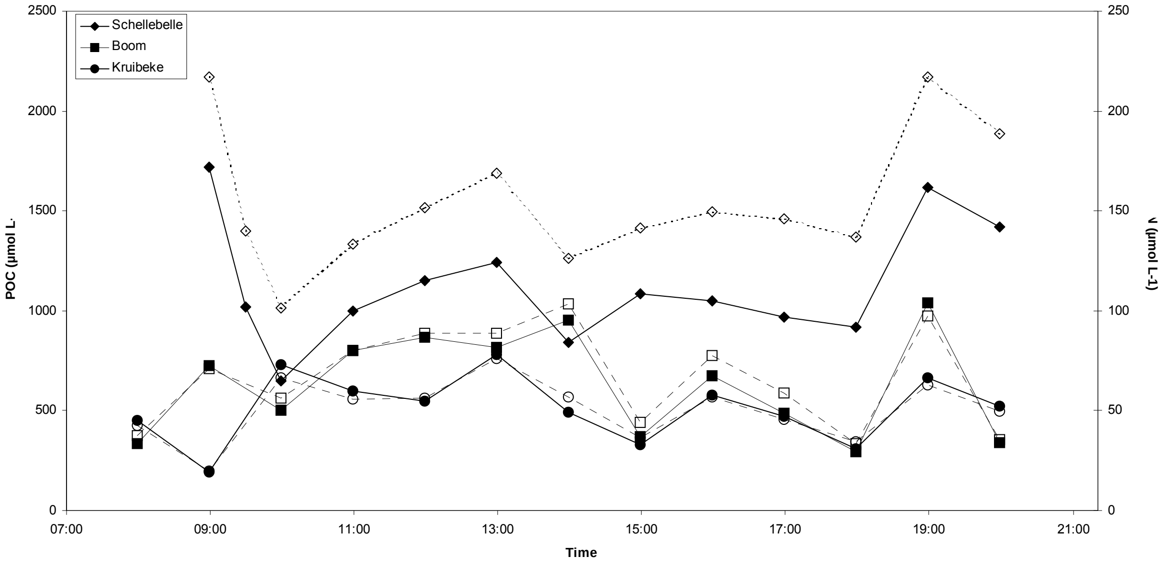

Figure 7. Temporal evolution of POC an PN (µmol L^{-1}) during the 13 hours experiments in Schellebelle, Boom and Kruibeke.

3.5. Reference

Cai W.J. and Y. Wang, 1998. The chemistry, fluxes and sources of carbon dioxide in estuarine waters of the Satille and Altamaha Rivers, Georgia, *Limnology and Oceanography*, 43, 657-668.

Hoofdstuk 4. Zwevende stof en sedimenten in de Zeeschelde

Margaret Chen
Florimond de Smedt
Stanislas Wartel
Edward Van den Storme

Eindverslag voor deelstudie 1 (perceel 1), periode juli 2008 - augustus 2009

Perceel 3 – Partim: the Schelde Estuary
Vakgroep Hydrologie en Waterbouwkunde
Vrije Universiteit Brussel
(HYDR-VUB)

4.1. Introduction

This report describes the observations and measurements performed in fulfillment of the contract: *Onderzoek naar de gevolgen van het Sigmaplan, baggeractiviteiten en havenuitbreiding in de Zeeschelde op het milieu – Studie: Sediment en Zwevende Stof.*

Next to the monthly observations several complete tide measurements were performed near Schellebelle, Lippenbroek, Kruibeke and Boom. The measured parameters were suspended matter concentration, current velocity and salinity.

4.2. Methodology

4.2.1. Field sampling of suspended matter

Samples for the measurement of the suspended matter concentration were taken by pumping water from 5 predefined depths (10%, 25%, 50%, 75% and 90% of the actual water depth) and collecting water in PVC bottles of 250 ml. In order to minimize biochemical processes the bottles were immediately stored in a cooler box and, after transport to the laboratory, stored in a dark room at about 4°C till further analyses. Total suspended sediment was collected on pre-weighed 0.45 μm sterile membrane filters (Whatman WCN type filters, Cat. No 7141-114) and dried at 105°C before weighing. Salts were removed by rinsing the filters for at least 3 times with at least 50 ml of demineralised water.

The measured suspended matter concentrations in the Schelde estuary are far from homogeneous. It varies with the tide along the estuary in depth as well as over the cross section. Therefore knowing the accuracy of the measuring method is important. The accuracy of the sampling technique was tested on 52 samples taken in twofold on different occasions and at several sampling sites. The suspended matter concentration of these test samples varied from a few milligrams to 476 mg per liter covering the range of concentrations commonly observed in the river (Figure 1). It could be calculated from these data that the average standard deviation is around 9%. The largest relative deviations (up to 40%) are observed in samples containing less than 25 g m^{-3}.

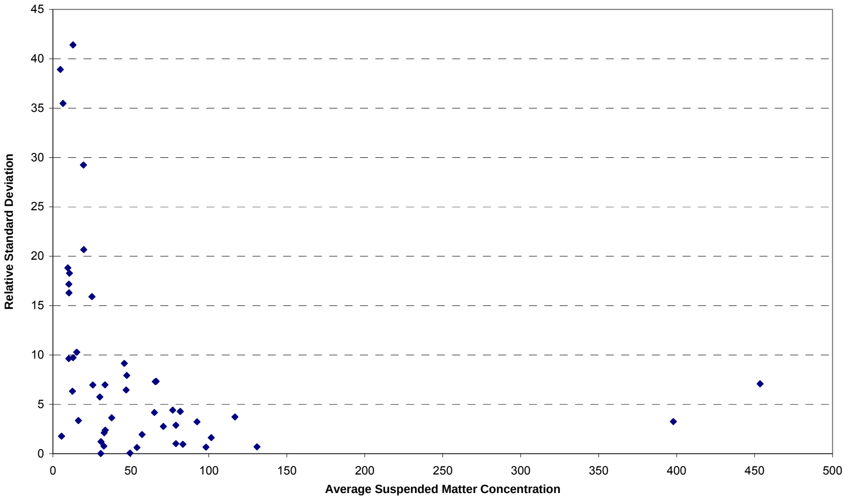

Figure 1. Percent standard deviation observed in test samples as a function of concentration of suspended particulate matter in g m^{-3}.

4.2.2. Field measurement of turbidity

The turbidity of the water is an indicative indirect method to measure the amount of suspended matter available in the water column. The turbidity is obtained by measuring the light reflection of a red laser beam. When the reflected light hits the surface of a detector a small electrical current is generated. This current, expressed in millivolt, is recorded together with the date, the time and the water depth. In practice two sensors are used: one for low SM concentrations (below 0.5 gram per liter) and one for high SM concentrations (up to several gram per liter). The data recording is set at an interval of every 5 seconds, and each recording is the average of 100 readings. For every campaign both sensors are calibrated with the SM concentration obtained from water samples as described in the pervious section "Field sampling of suspended matter". Figure 2 illustrates an example of calibration of two OBS sensors.

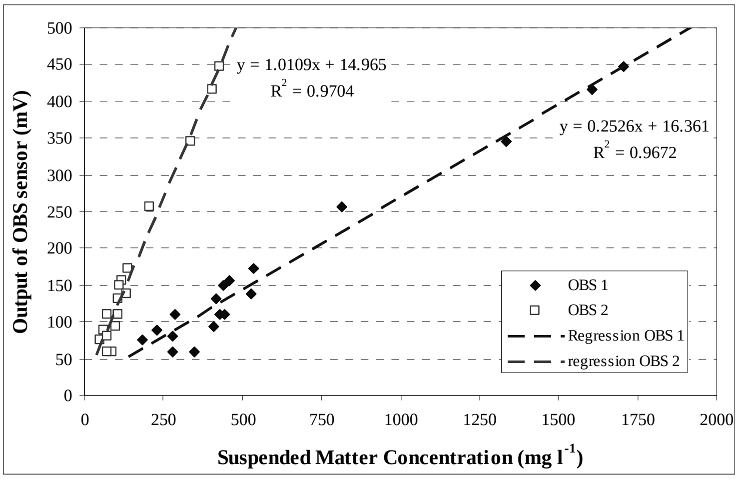

Figure 2. Calibration curves of the OBS output and suspended matter concentration.

4.2.3. Field measurement of water velocity

Water velocities were measured at every sampling depth using either a calibrated OTT current meter or a VALEPORT model 802 two-axis electromagnetic current meter with underwater housing. The VALEPORT current meter measures the flow by an electromagnetic sensor that uses the Faraday principle to measure the flow past the sensor in two orthogonal axes. The magnetic field is generated within the sensor by a coil, and the electronics detect the signal generated across two pair of electrodes, one pair for each axis. The VALEPORT current meter is set at a data rate of 16 records per second. Each measurement is averaged over a time interval of 1 minute.

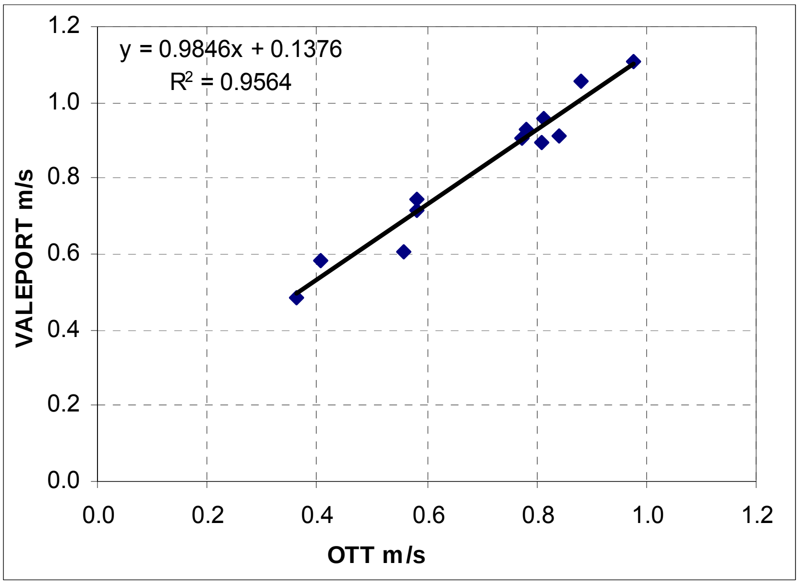

Figure 3. Comparison of the current velocity simultaneously recorded by an electromagnetic VALEPORT- (x-axis) and a propeller type OTT (y-axis) current meter. The full line is the linear regression of the data.

A series of measurements was performed with the OTT current meter and the VALEPORT model 802 current-meter mounted next to each other on the same underwater frame, ballasted with a streamlined weight of 100 kg. The VALEPORT current meter was set at a data rate of 16 records per second. The recording time for both meters was set at 1 minute.

Figure 3 shows the output from both meters. The data suggests that both instruments give very comparable outputs.

4.2.4. Field measurement of salinity and conductivity

Salinity was calculated using the electrical conductivity of the water measured at every sampling depth using a WTW LF325 conductivity meter. The salinity is expressed in Practical Salinity Units according to the UNESCO technical paper n°36 (Unesco, 1981).

4.3. Flow velocity and salinity distributions

A depth-integrated flow velocity was calculated for every station for the period January 2002 to August 2009, the results along with measured maximum flow values are given in the Table 1.

Table 1. Depth-integrated flow velocity and measured maximum flow values at every monitoring station from January 2002 to August 2009.

stations	**distance (km)**	**depth-integrated (m/s)**	**maximum (m/s)**
Zandvliet	58	0.51	0.92
Liefkensoek	64	0.62	1.32
Melsele	71	0.87	1.44
Antwerpen	78	0.84	1.50
Kruibeke LO	85	0.81	1.58
Kruibeke RO	85	0.69	1.07
Bazel	89	0.95	1.37
Rupel (Boom)	98	0.60	0.89
Steendorp	94	0.80	1.09
Temse	99	0.70	1.25
Lippenbroek LO	104	0.69	1.45
Lippenbroek RO	104	0.81	1.26
Baasrode	118	0.76	1.35
Dendermonde	121	0.73	1.46
St Onolfs	125	0.75	1.53
Appels	128	0.88	1.59
Uitbergen	138	0.66	1.07
Wetteren	145	0.67	1.18
Melle	151	0.57	1.25

Salinity distributions near-surface layer or the uppermost 10% of the water column and near bottom layer or the lowermost 10% of the water column between the Belgian-Dutch border (Zandvliet) and Kruibeke are listed in the Table 2.

Table 2. Average, minimum and maximum salinity values at the stations between the Belgian Dutch border (Zandvliet) and Kruibeke from January 2002 to August 2009.

	Surface-layer			Bottom-layer		
	average	minimum	maximum	average	minimum	maximum
Zandvliet	9	1	16	10	1	18
Liefkenshoek	7.6	0.1	13.7	8.0	0.1	15.1
Melsele	5.1	0.4	11.9	5.4	0.5	12.1
Antwerpen	2.3	0.1	10.2	2.5	0.1	10.3
Kruibeke LO	1.0	0.1	6.6	1.1	0.1	7.5
Kruibeke RO	0.8	0.3	2.2	0.9	0.3	2.2

4.4. Concentration and behavior of suspended matter

4.4.1. Surface plots of monthly observations

An overview of the monthly SMC distributions is given as surface plots. It shall be noted that the data presented in the distance versus monthly SMC surface plots were obtained at different moments of the tide. The surface plot of SMC monthly observations does not give an image of the suspended matter concentration at a given time in the estuary but it gives the suspended matter concentration at the time of sampling. One of the advantages of this type of plot is that it projects somehow the distributions and the variations of large data sets collected through the years.

The surface plots of SMC are shown for two depth-layers, the near bottom layer or the lowermost 10% of the water column and the near-surface layer or the uppermost 10% of the water column. These two depth-layers are selected because SMC at the near-bottom layer illustrates possible resuspension of recently deposited sediment, and SMC at the near-surface layer is important for many biological and chemical aspects of water and it may play an important role with respect to inundation areas.

The monthly suspended matter concentrations in the lowermost 10% of the water column for the period 2002 to August 2009 versus distance from Vlissingen (NL) are given in the Figure 4. The SMC varies from less than 100 g m^{-3} to more than 2000 g m^{-3}. It can be observed that at a given location the SMC shows an alternation of periods with low concentration and periods with high concentration. These periods do not exactly match the seasonal variation. This is to some extent due to the sampling program which does not consider tidal moment. In spite of this shortcoming the distance-time-SMC plot for the period 2002-2009 shows some important information. A zone with recurring high SMC existed between 58 km (Belgian-Dutch border) and 85 km (Kruibeke) where SMC exceeding 500 g m^{-3} was frequently observed. A second zone with high SMC occurred between 90 km (around the confluence of the Rupel-Schelde) and 100 km (around Lippenbroek). A third zone of high SMC occurred in the vicinity of 135 km (ca. Uitbergen).

The monthly suspended matter concentrations in the uppermost 10% of the water column for the period 2002 to 2009 versus distance from Vlissingen (NL) are given in the Figure 5. The observed SMC ranged from less than 100 g m^{-3} to more than 800 g m^{-3}. The reoccurrence of high SMC can be observed between 90 km (the Rupel mouth) to 135 km (ca. Uitbergen).

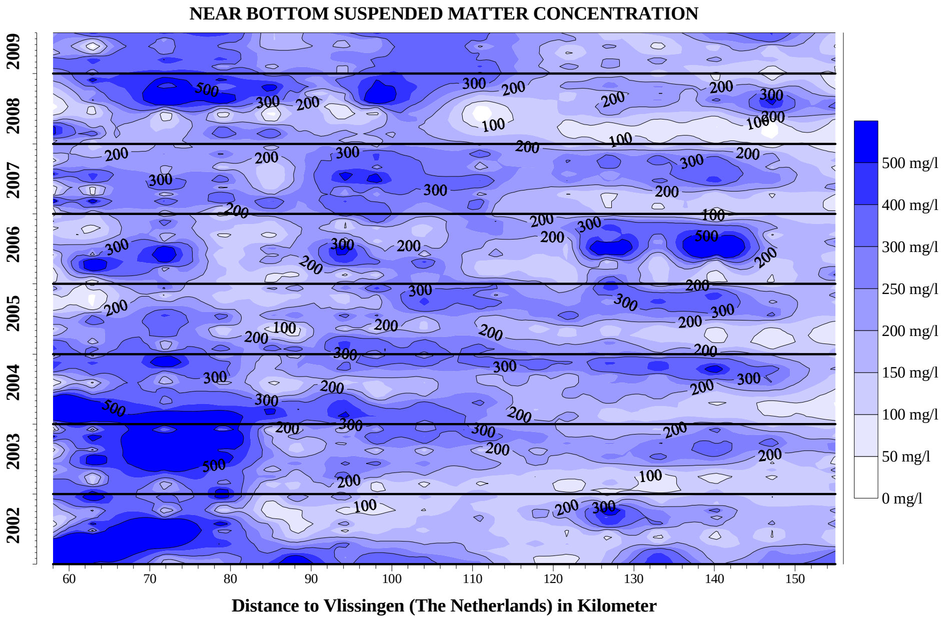

Figure 4. Monthly observation of suspended matter concentrations in the lower 10% of the water column.

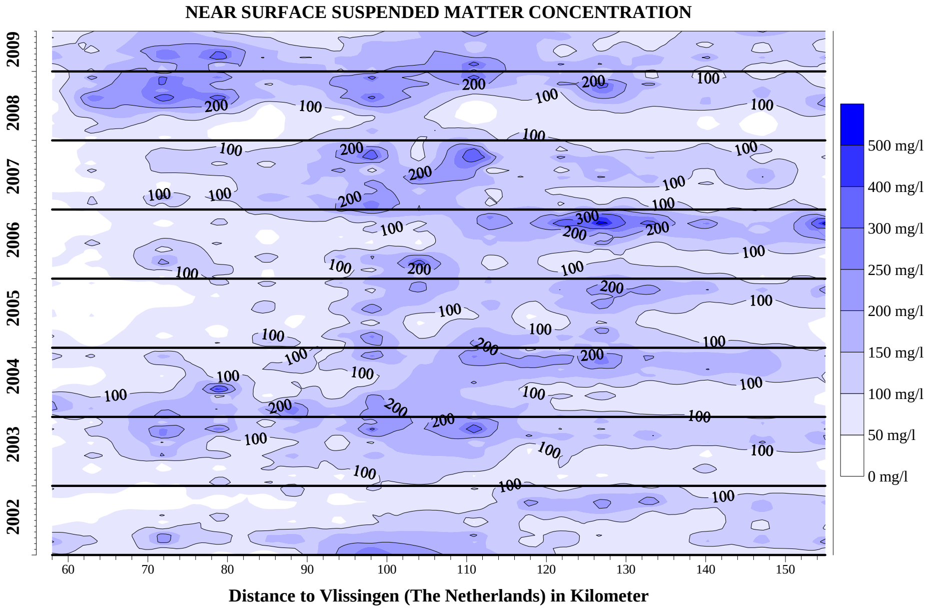

Figure 5. Monthly observation of suspended matter concentrations in the uppermost 10% of the water column.

The total suspended matter flux (2002-2009), based on the depth integrated suspended matter concentration and the current velocity, gives the amount of suspended matter that is passing by a given section per unit surface and time calculated from the monthly observations. They vary form less than 1 kg m^{-1} s^{-1} to more than 4 kg m^{-1} s^{-1} (Figure 6). A surface plot of the total suspended matter flux as a function of distance to Vlissingen is given in the figure 6. The largest SM fluxes (exceeding 2 kg m^{-1} s^{-1}) occurred in a zone between 50 km and 100 km, except for the second half of 2004 and in 2005 when SM fluxes were quite low. In general, between 100 km and 155 km, SM fluxes were low (below 2 kg m^{-1} s^{-1}), except for the winter 2003 to 2004 and the mid year of 2006, when SM fluxes exceeding 2 kg m^{-1} s^{-1} occurred.

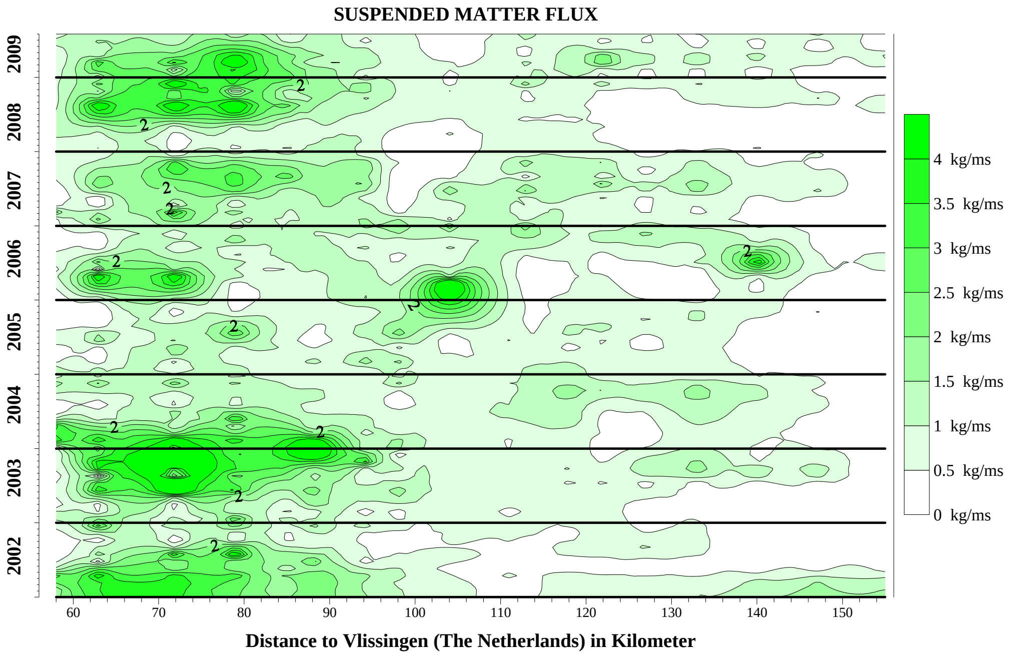

Figure 6. Depth integrated suspended matter flux (kg m^{-1} s^{-1}) in whole water column.

4.4.2. Longitudinal average concentrations of suspended matter

Monthly measurements of suspended matter concentration between June 2008 and August 2009 were analysed. The average suspended matter concentrations at each station in 2008 are summarized in the tables 3 and 4 comparing to the averages of the values obtained from the previous monitoring measurements from 2002 to 2007. The average values obtained from January-August 2009 were not used because they did not cover the whole year.

Table 3. Average suspended matter concentrations in g m^{-3} at the near-surface layer for 2008 and for the period 2002-2007. Stations Bazel, Mariekerke and Vlassenbroek were not sampled in 2008.

station #	station	km	2008	2002-2007
9.1	Belgian-Dutch border	58	58	64
10.1	Liefkenshoek	64	137	69
11.2	Melsele	71	172	106
12.1	Antwerpen	78	145	90
13.1	Kruibeke	85	94	98
13.2	Bazel	89	-	109
14.1	Steendorp	94	135	115
14.2	Temse	98	188	162
15.1	Lippenbroek	104	130	163
15.2	Mariekerke	107	-	155
15.3	Baasrode	113	120	152
16.1	Vlassenbroek	118	-	132
16.2	Dendermonde	121	114	136
17.1	St. Onolfs	125	133	143
17.2	Appels	128	109	121
18.1	Uitbergen	138	100	111
19.1	Wetteren	145	103	114
20.1	Melle	151	107	99

SMC average values near surface in 2008 between Liefkenshoek and Temse, except for Kruibeke, were much higher than the period 2002-2007, while opposite was observed from Lippenbroek to Melle where SMC average values were relatively somewhat lower in 2008 than the period 2002-2007.

SMC average values near bottom showed a similar trend comparing to the near surface values (Tables 5 and 6). Comparing to the period 2002-2007, the SMC average values in 2008 were a bit higher between Melsele and Steendorp with the exception at Kruibeke, while in the upper part of the estuary from Lippenbroek to Melle, the SMC in 2008 were a bit lower with the exception at Wetteren.

The average suspended matter concentrations for the year 2008 were also compared to the average values obtained from the monitoring campaign in 2002, for both near-surface layer (Figure 7) and near-bottom one (Figure 8). Some notable changes in SMC at a few locations can be observed upstream of Rupel mouth, but SMC showed quite similar trend towards the upstream.

Table 4. Average suspended matter concentrations in g m^{-3} at the near-bottom layer for 2008 and for the period 2002-2007. Stations Bazel, Mariekerke and Vlassenbroek were not sampled in 2008.

station #	station	km	2008	2002-2007
9.1	Belgian-Dutch border	58	270	291
10.1	Liefkenshoek	64	223	360
11.1	Melsele	71	432	409
12.1	Antwerpen	78	419	310
13.1	Kruibeke	85	311	210
13.2	Bazel	89		208
14.1	Steendorp	94	217	256
14.2	Temse	99	363	268
15.1	Lippenbroek	104	171	252
15.2	Mariekerke	107		232
15.3	Baasrode	113	152	227
16.1	Vlassenbroek	118		180
16.2	Dendermonde	121	141	188
17.1	St. Onolfs	125	185	226
17.2	Appels	128	185	208
18.1	Uitbergen	138	185	235
19.1	Wetteren	145	214	176
20.1	Melle	151	137	146

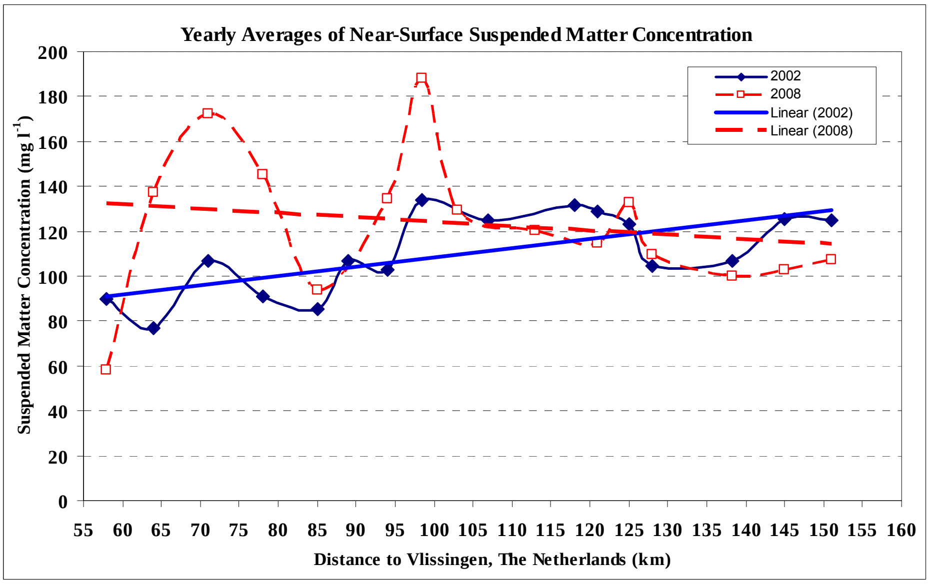

Figure 7. Near-surface average suspended matter concentration 2008 versus 2002.

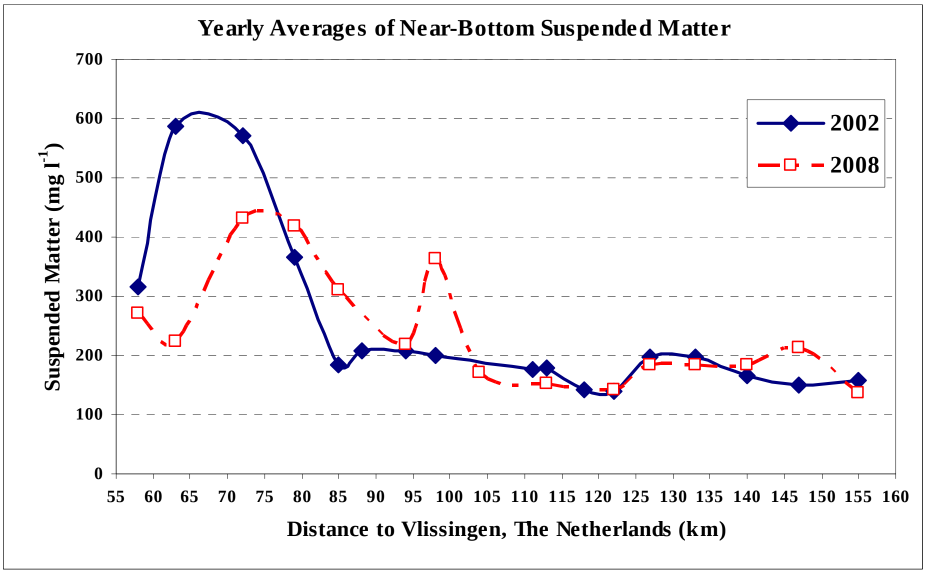

Figure 8. Near-bottom suspended matter concentration 2008 versus 2002.

4.4.3. Suspended matter concentration in the Rupel, Dijle and Nete

Monthly monitoring of suspended matter has been performed only near Boom in the Rupel since 2006 (Figure 9). From January 2009 to August 2009, the OMES program adapted to the MONEOS framework. Five new stations were added into monthly monitoring program of suspended matter, i.e., three stations in the Rupel (Terhagen, Boom and Niel), one in the Dijle and one in the Nete (Figure 10).

The monthly suspended matter concentrations in the uppermost 10% of the water column ranges from a few milligrams per litre up to 200 mg l^{-1} (Figure 9). SMC near Boom has a tendency to increase since 2006.

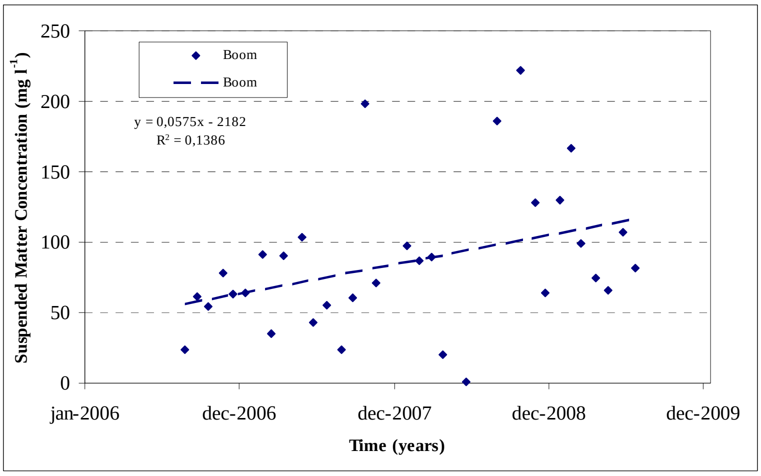

Figure 9. Monthly observation of suspended matter concentrations in the uppermost 10% of the water column near Boom in the Rupel from June 2006 to August 2009.

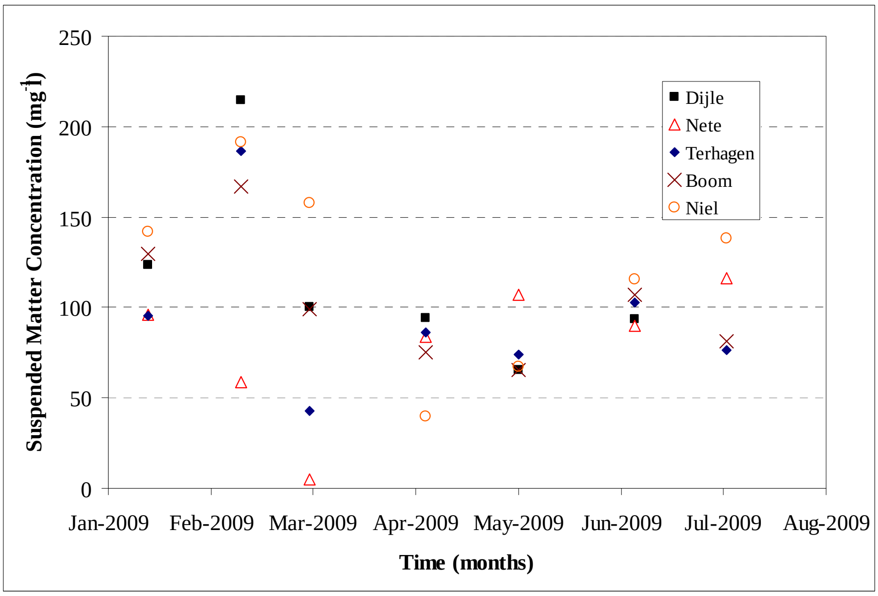

Figure 10. Monthly observation of suspended matter concentrations in the uppermost 10% of the water column at all five (5) stations in the Rupel, Dijle and Nete, January - August 2009.

The monthly suspended matter concentrations in the lowermost 10% of the water column are plotted in the Figures 11 and 12. The SMC near bottom was quite close to the ones near water surface with only a few exceptions at Boom where SMC was close to 500 mg l^{-1}.

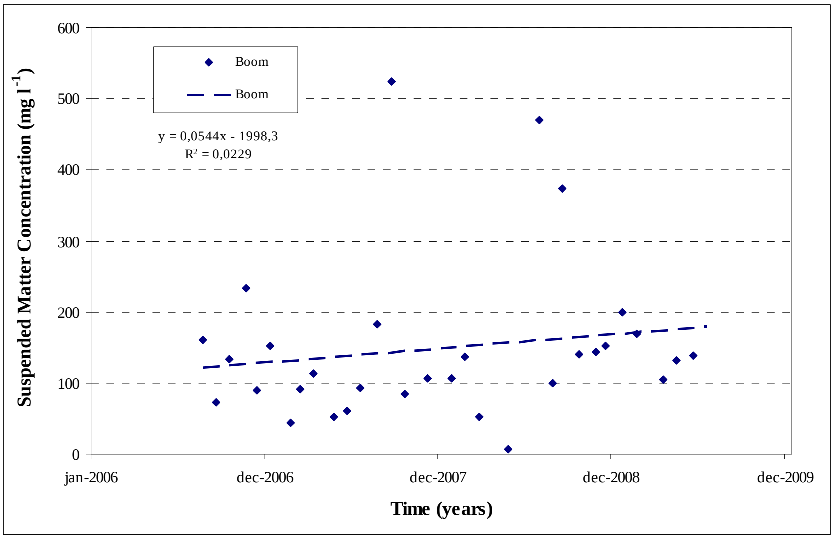

Figure 11. Monthly observation of suspended matter concentrations in the lowermost 10% of the water column near Boom in the Rupel from June 2006 to August 2009.

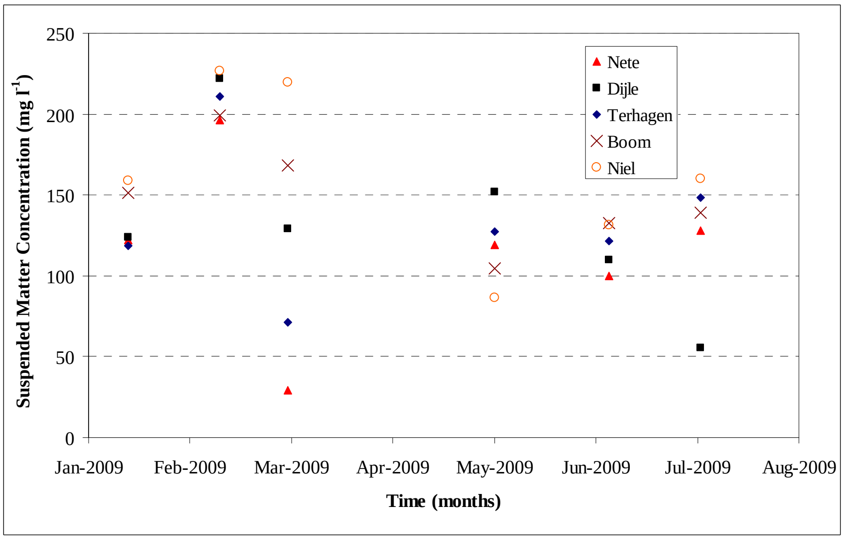

Figure 12. Monthly observation of suspended matter concentrations in the lowermost 10% of the water column at all five (5) stations in the Rupel, Dijle and Nete, January - August 2009.

4.4.4. Relationship between surface suspended matter concentration and depth integrated suspended matter flux

The depth integrated suspended matter flux at a given moment and at a given location is calculated based on the depth integrated SMC and the depth integrated water velocity. A low correlation (r^2 = 0.19) was obtained. It shows that there is a very poor or no correlation between surface suspended matter concentration and depth integrated suspended matter flux at a given moment and at a given location between the Belgian-Dutch border and Gent in the Scheldt estuary (Figure 13).

Therefore, in order to understand overall suspended matter behavior, vertical profiles of suspended matter concentration need to be considered and analysed. The relationship between suspended matter flux and water velocity is presented in the next section.

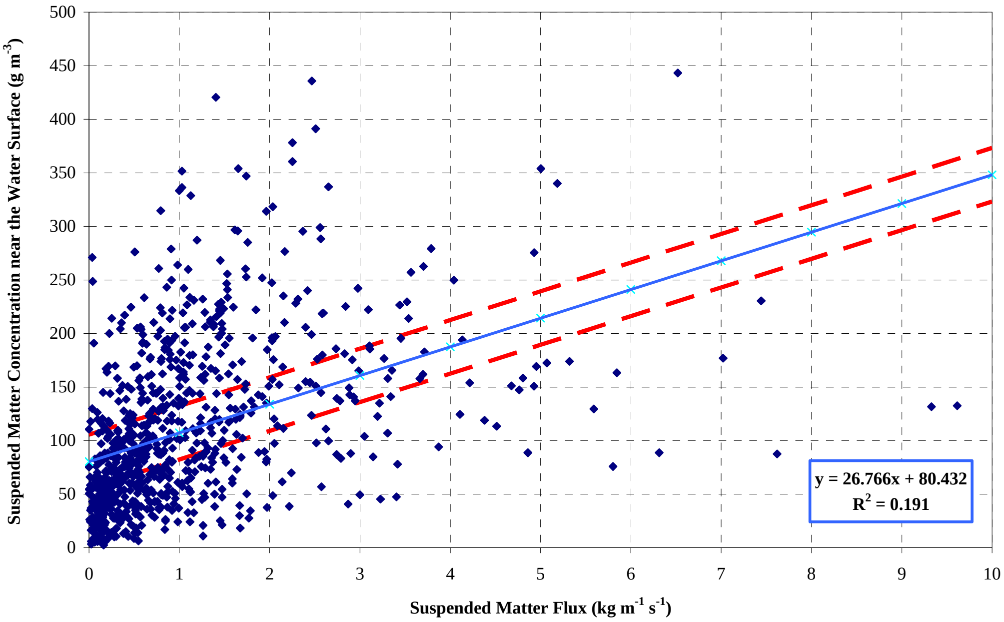

Figure 13. Relationship between the suspended matter flux and the suspended matter concentration in the upper part of the water column.

4.4.5. Suspended matter flux and water velocity

In the previous sections an overview of the monthly SMC observations is given as surface plots. As discussed earlier, the surface plot of SMC monthly observations does not give an image of the suspended matter concentration at a given time in the estuary but it gives the suspended matter concentration at the time of sampling. A better correlation is shown when the suspended matter flux, expressed in kg m^{-1} s^{-1} is given as a function of the depth averaged water velocity expressed in m s^{-1}. Since the SMC and the water velocity vary with depth and time, a relation between depth averaged SMC and depth averaged velocity can give valuable information to understand the sediment behavior at a given locality.

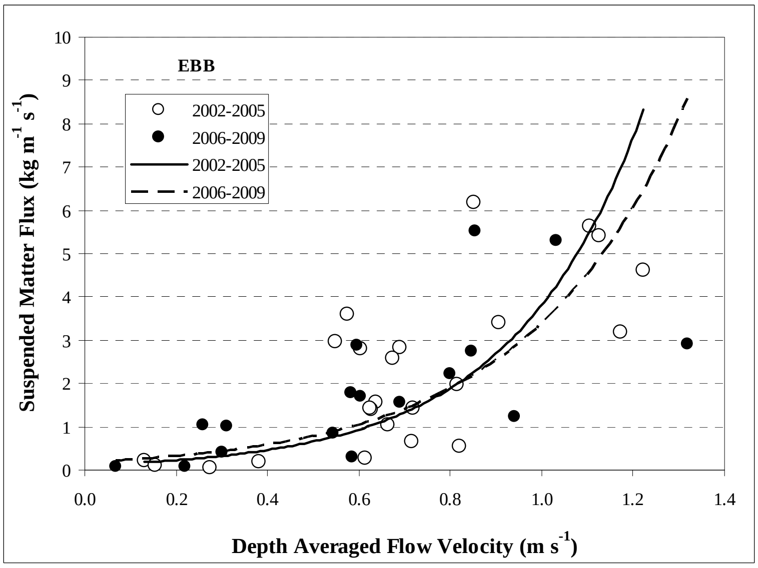

Figure 14. Evolution of the suspended matter flux (kg m^{-1} s^{-1}) with the depth averaged water velocity (m s^{-1}) during ebb at Liefkenshoek (station 10.1) for the period 2002-2005 and 2006-2009.

From the observations at the sixteen stations it was found that the relation between the measured suspended matter concentrations and the velocity is well represented by an exponential equation, given that the water velocity does not exceed the maximum observed value of 1.4 m s^{-1}. The results are obtained based on observations between 2002 and 2009 and are represented at three selected stations (Figures 14 to 16).

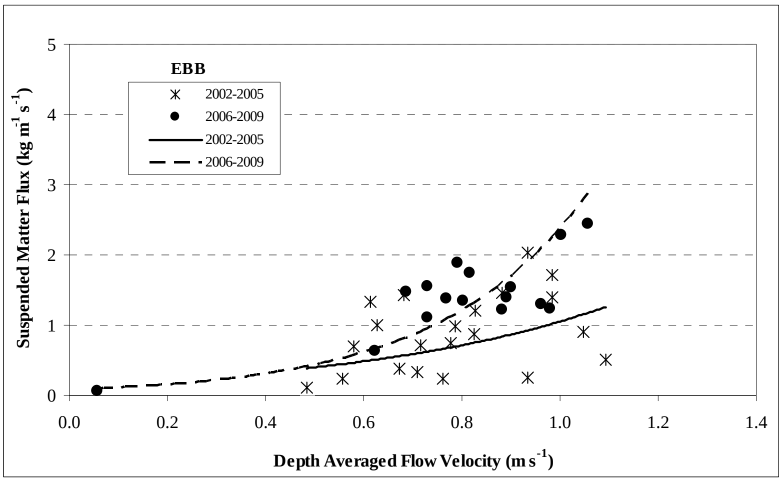

Figure 15. Evolution of the suspended matter flux (kg m^{-1} s^{-1}) with the depth-averaged water velocity (m s^{-1}) during ebb at Steendorp (station 14.1) for the period 2002-2005 and 2006-2009.

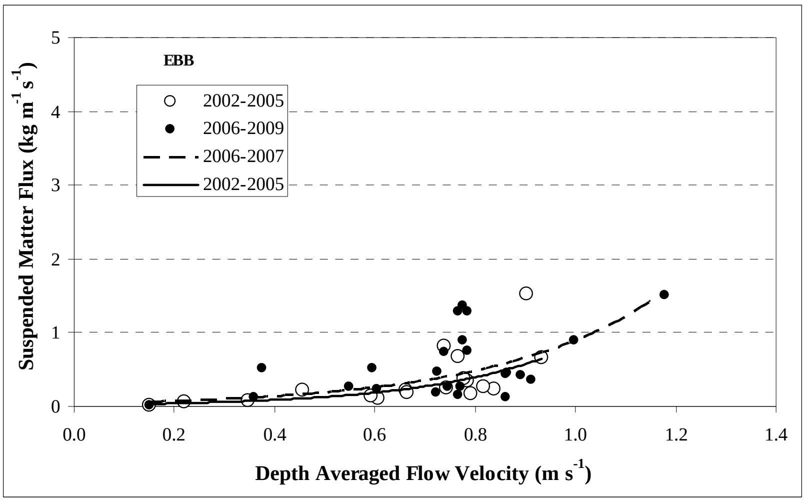

Figure 16. Evolution of the suspended matter flux (kg m^{-1} s^{-1}) with the depth-averaged water velocity (m s^{-1}) during ebb at Wetteren (station 19.1) for the period 2002-2005 and 2006-2009.

The results for all the measurements showed that between Liefkenshoek and Kruibeke, suspended matter flux observed in 2006-2009 were close to or lower than the period 2002-2005, but the relative higher values were observed from Steendorp to St. Onolfs, and remained relatively stable between Appels and Melle.

The relation between the measured suspended matter concentrations and the velocity is best represented by an exponential equation of $y = a\ e^{b.x}$. The parameter 'a', in kg m^{-1} s^{-1} and at the slack of tide ($u_t = 0$), in the exponential regression equation stands for the initial suspended matter load. The parameter 'b' is a constant that expresses the rate of increase of the suspended matter concentration with increasing water velocity. It is likely to assimilate 'b' with the degree of resuspension of bottom sediments.

The change of 'a' and 'b' with the distance from Vlissingen is given in the Table 7 and plotted in the Figures 17 and 18. It can be observed that the suspended load during ebb evolves quite differently from its evolution during flood. During ebb the initial suspended matter load (a) at slack water reaches a maximum around 80 km (2002-2009 in Figure 17). During flood the maximum initial load was located between 100 and 120 km (2002-20009 in Figure 18). The shift of the maximum initial load from downstream at ebb to upstream at flood was of the order of 20 to 40 km. For comparative purposes the values for the period 2006-2009 are also shown. It can be noticed that, during ebb phase, there were two maxima initial loads in the vicinity of 80 km and of 110 km, and the downstream one close to Melsele had shifted toward Antwerp (Figure 17). During flood phase, at the confluence of the Rupel and Scheldt around 92 km, the period 2002-2009 showed higher initial load upstream of 92 km, while opposite trend was shown in the period 2006-2009 with relatively lower initial load upstream of 92 km (Figure 18).

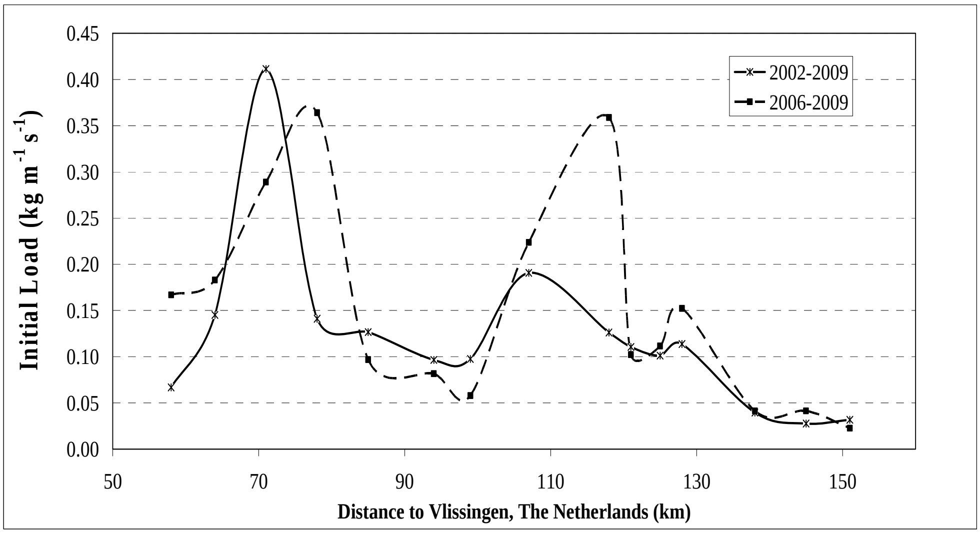

Figure 17. Variation of initial load (a-parameter) during ebb tide.

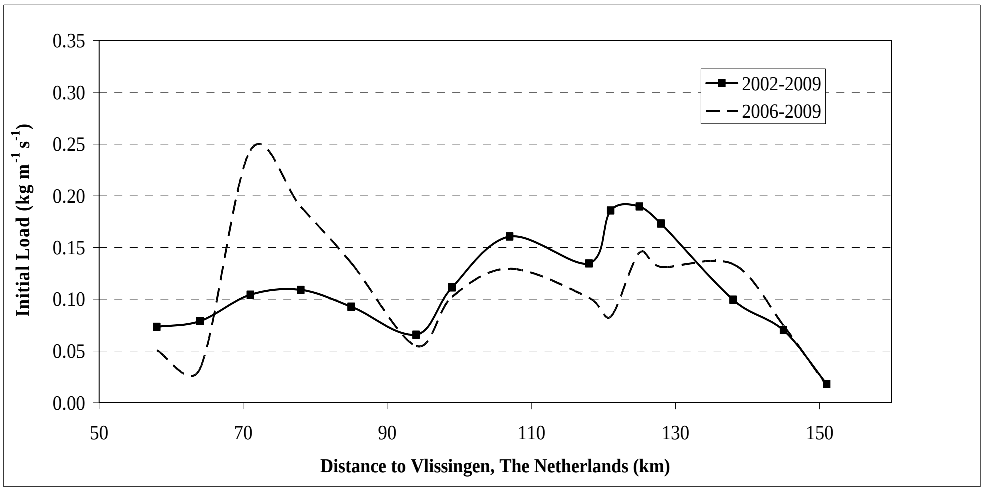

Figure 18. Variation of initial load (a-parameter) during flood tide.

Table 5. Correlation coefficient (R^2) and parameters (a and b) for both ebb and flood. Data of the 16 stations obtained from the measurements between 2002 and 2009.

station number	**station**	**km**	**R^2 ebb**	**a ebb**	**b ebb**	**R^2 flood**	**a flood**	**b flood**
9.1	Belgian-Dutch Border	58	0.47	0,05	4,27	0,42	0,06	3,54
10.1	Liefkenshoek	64	0.58	0,12	3,31	0,29	0,14	2,73
11.1	Melsele	71	0.33	0,33	1,67	0,57	0,08	2,86
12.1	Antwerpen	78	0.55	0,25	1,86	0,61	0,09	2,79
13.1	Kruibeke	85	0.68	0,13	2,38	0,42	0,15	1,75
14.1	Steendorp	94	0.41	0.10	2.79	0.65	0.07	3.34
14.2	Temse	99	0.55	0.10	2.75	0.52	0.11	2.78
15.2	Mariekerke	107	0.36	0.19	1.88	0.66	0.16	2.34
16.1	Vlassenbroek	118	0.46	0.13	2.24	0.60	0.13	2.12
16.2	Dendermonde	121	0.58	0.11	2.28	0.43	0.19	1.61
17.1	St.-Onolfs	125	0.44	0.10	2.05	0.49	0.19	1.57
17.2	Appels	128	0.34	0.11	1.85	0.47	0.17	1.65
18.1	Uitbergen	138	0.48	0.04	3.37	0.43	0.10	2.23
19.1	Wetteren	145	0.54	0.03	3.47	0.58	0.07	2.40
20.1	Melle	151	0.55	0.03	2.39	0.77	0.02	4.85
summary	***averages***		**0,49**	***0.12***	***2.59***	***0.51***	***0.11***	***2.63***
	standard deviations		**0,10**	***0.09***	***0.69***	***0.17***	***0.05***	***0.84***

$$L_t = a.e^{b.u_t}$$

L_t : total suspended load at time t in kgm^{-1}s^{-1}
a_t : initial load at time in kg.m^{-1}s^{-1}
b: constant
u_t: depth averaged water velocity at time t

The b-factor, which to certain extent reflect degree of resuspension of bottom sediments, for both ebb and flood phase, showed high values at three zones, around downstream of Liefkenshoek, around Steendorp, and around upstream of Uitbergen (Table 5). These three (3) zones may explain the observed monthly high suspended matter concentrations in the lowermost 10% of the water column for the period 2002 to August 2009 (Figure 4).

4.4.6. Total suspended matter flux, load and water velocity for full tide measurements

The data obtained from full tide measurements in 2008-2009 are compared to the previously obtained data from the monitoring program. The total suspended matter flux at a given moment of tide is compared to depth averaged water velocity at that moment (Table 6 and Figures 19 to 24).

It has been shown that the suspended matter load and the depth averaged water velocity can be best described by an exponential equation. For every full-tide measurement, without discriminating ebb and flood data, the correlation coefficient (r^2) and the parameters a (initial suspended matter load at slack of tide) and b (a constant) are given in Table 6. The initial load (parameter a) varies between 0.07 kg m^{-1} s^{-1} and 0.13 kg m^{-1} s^{-1}. The b-factor, which to certain extent reflect degree of resuspension of bottom sediments, showed high values in the upper esturay, and relatively low values in the Rupel (Table 6).

In the upper estuary and at Kruibeke, during ebb tide, the suspended matter flux in 2006-2009 was generally higher than the period 2002-2005 and often exceeded the maxima observed during this period. During flood tide, the suspended matter flux remained relatively stable, except at Kruibeke higher values were observed (Figures 19 to 22). Since in the Rupel measurements started in 2006, a correlation with earlier data is not available. An effort was made to compare the Rupel data to the data of the upper estuary (Figures 23 and 24). It can be observed that for a given velocity the suspended matter flux in the Rupel was lower than those in the upper estuary during ebb. During flood period, however, the difference between Ruple data and the upper estuary was rather little.

Table 6. Correlation coefficient (R^2) and value of parameters a and b for all data from the full-tide measurements.

station	date	R^2	a	b
	BELGIAN-DUTCH BORDER			
Zandvliet	12/09/2002	0.70	0.08	2.880
Zandvliet	28/04/2003	0.89	0.13	2.246
Pas van Rilland	02/09/2003	0.41	0.37	1.598
Pas van Rilland	30/06/2004	0.75	0.03	5.125
Pas van Rilland	22/09/2004	0.57	0.06	3.106
	averages	***0.66***	***0.13***	***2.991***
	KRUIBEKE			
Kruibeke	16/04/2002	0.75	0.19	2.187
Kruibeke	07/05/2003	0.63	0.11	2.116
Kruibeke	29/06/2004	0.87	0.08	3.062
Kruibeke	21/09/2004	0.81	0.09	3.091
Kruibeke	02/06/2005	0.55	0.11	1.713
Kruibeke	17/10/2005 (2)			
Kruibeke	12/10/2006	0.71	0.08	4.457
Kruibeke	30/08/2007	0.88	0.19	4.724
Kruibeke	04/09/2008	0.31		1.464
	averages	***0.69***	***0.12***	***2.852***
	UPPER ESTUARY			
Baasrode	22/08/2002	0.93	0.03	4.859
Baasrode	01/09/2003	0.62	0.22	2.113
Lippenbroek	28/06/2004	0.93	0.07	3.536
Lippenbroek	20/09/2004	0.66	0.44	1.341
Schellebelle	31/05/2005	0.57	0.07	2.192
Schellebelle	20/10/2005	0.80	0.16	2.465
Schellebelle	10/10/2006	0.88	0.04	5.716
Lippenbroek	03/07/2006	0.50	0.16	1.534
Lippenbroek	25/10/2006 (1)			

Schellebelle	28/08/2007	0.83	0.01	8.417
Schellebelle	02/09/2008	0.86	0.09	4.102
Schellebelle	30/06/2009	0.92	0.01	7.202
	averages	***0.77***	***0.12***	***3.952***
	RUPEL			
Rupel (Walem)	01/06/2005	0.21	0.12	1.001
Rupel (Walem)	18/10/2005	0.64	0.02	2.641
Rupel (Boom)	11/10/2006	0.70	0.02	5.264
Rupel (Boom)	28/08/2007	0.73	0.07	2.357
Rupel (Boom)	03/09/2008	0.47	0.14	1.988
	averages	***0.55***	***0.07***	***2.650***
	SUMMARY OF ALL DATA			
	averages	***0.69***	***0.21***	***3.259***
	standard deviation	***0.19***	***0.50***	***1.816***

$$L_t = a.e^{b.u_t}$$

L_t : suspended matter flux at time t in kg.m^{-1}.s^{-1}
a_t : initial suspended matter flux at slack water in kg.m^{-1}.s^{-1}
b: constant
u_t: depth-averaged water velocity at time t

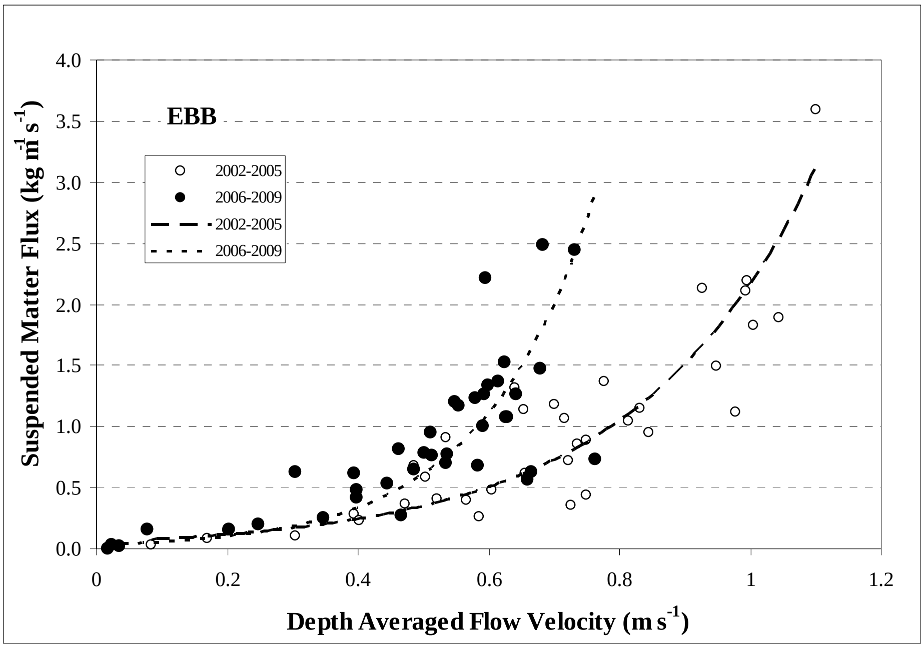

Figure 19. Evolution of the suspended matter flux (kg m^{-1} s^{-1}) with the depth-averaged flow velocity during ebb at the upper-estuary stations: Baasrode, Schellebelle and Lippenbroek.

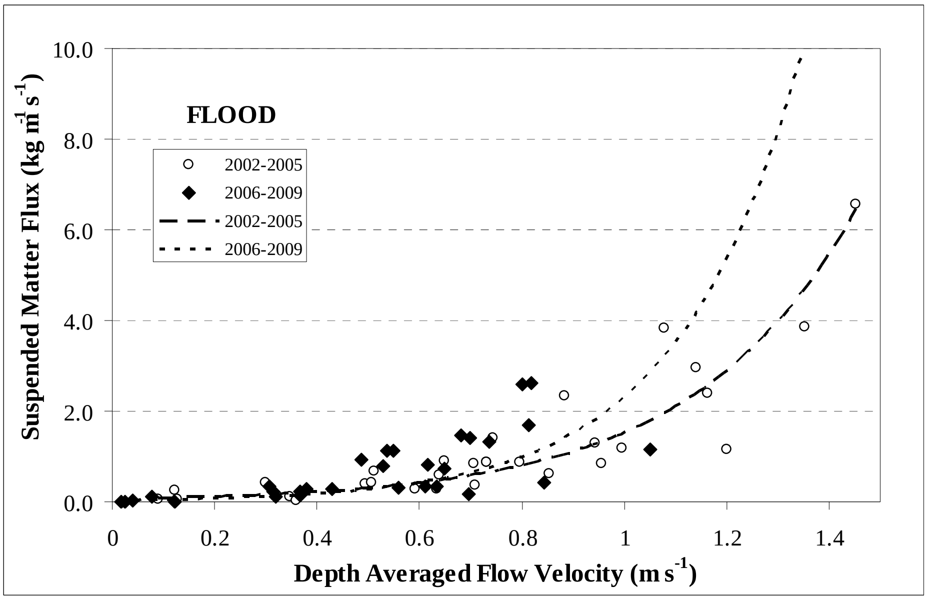

Figure 20. Evolution of the suspended matter flux (kg m^{-1} s^{-1}) with the depth-averaged water velocity during flood at the upper-estuary stations: Baasrode, Schellebelle and Lippenbroek.

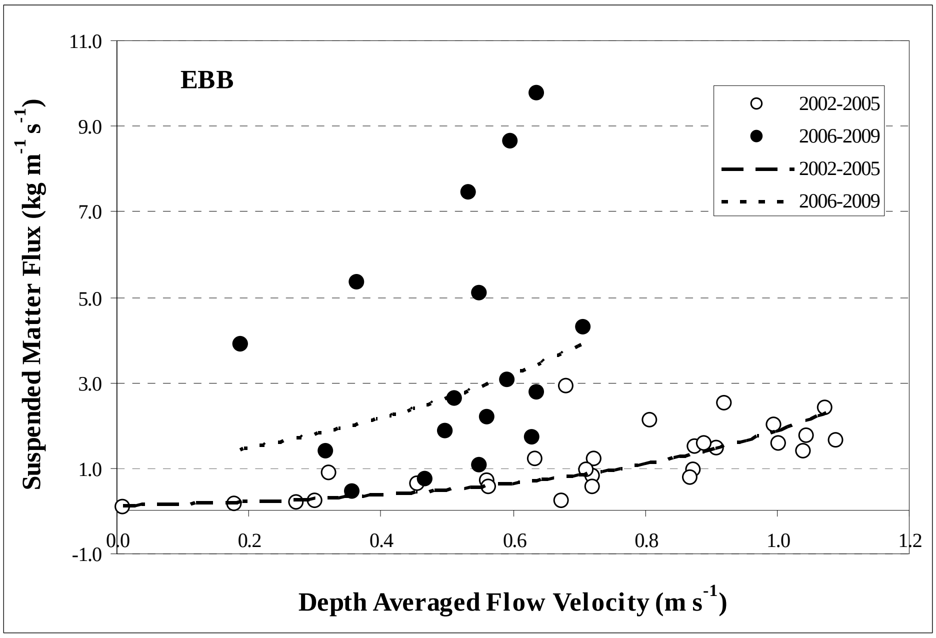

Figure 2. Evolution of the suspended matter flux (kg m^{-1} s^{-1}) with the depth-averaged flow velocity during ebb at the mid-estuary station Kruibeke.

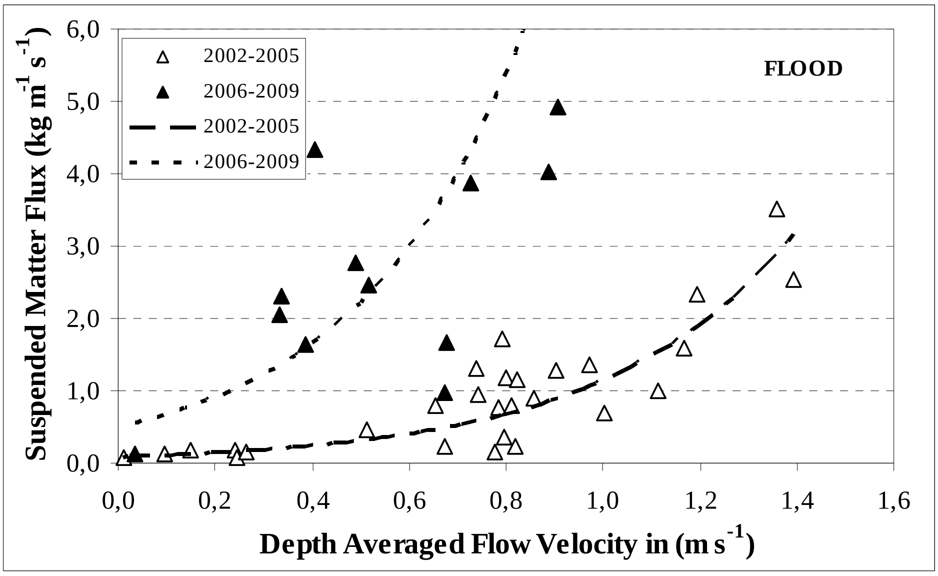

Figure 3. Evolution of the suspended matter flux (kg m^{-1} s^{-1}) with the depth-averaged flow velocity during flood at the mid-estuary station Kruibeke.

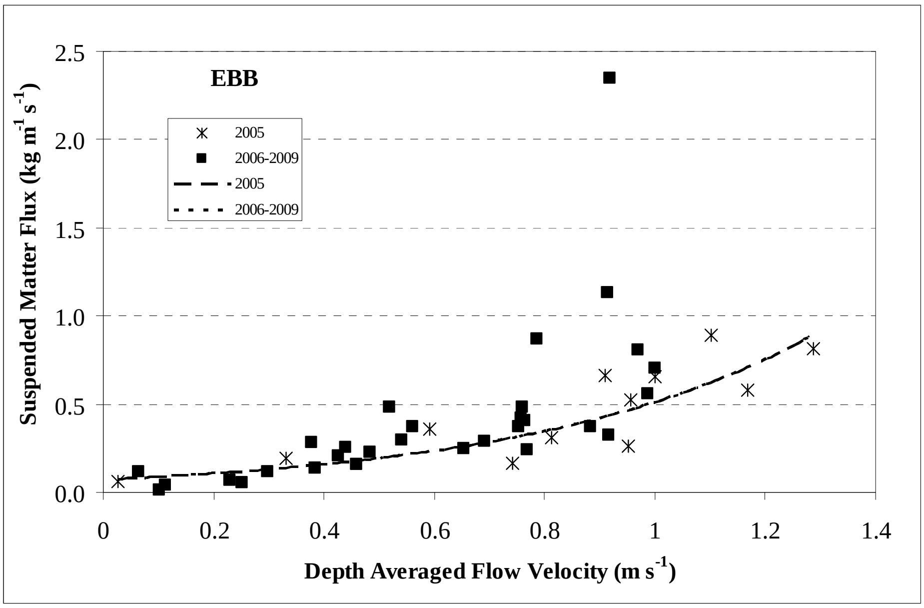

Figure 4. Evolution of the suspended matter flux (kg m^{-1} s^{-1}) with the depth-averaged flow velocity during ebb at the Rupel stations, Walem and Boom.

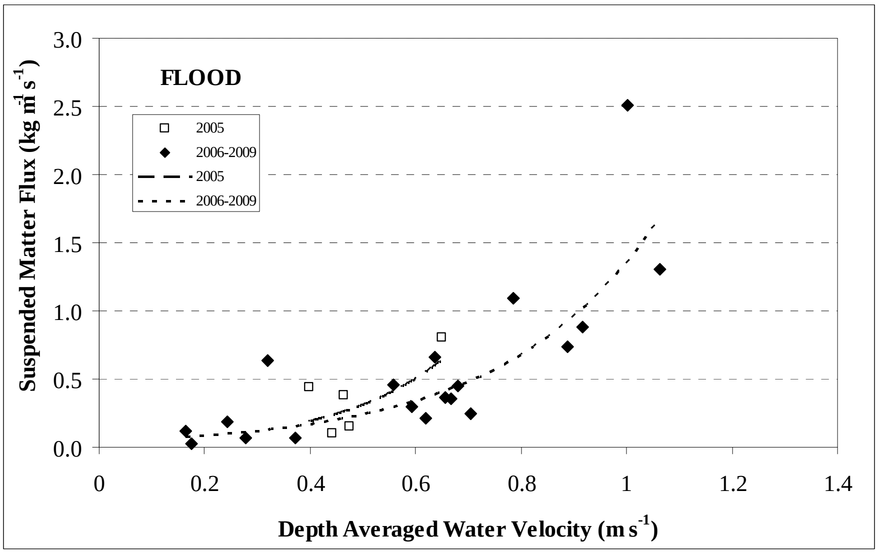

Figure 24. Evolution of the suspended matter flux (kg m-1 s-1) with the depth-averaged flow velocity during flood at the Rupel stations, Walem and Boom.

4.4.7. Influence of tidal range and river discharge on suspended matter behaviour

Evolution of suspended matter behaviour (2002-2009) discussed above demonstrated some worth noting phenomena. Monthly monitoring of suspended matter exhibited three zones with high turbidity, a zone existed between Belgian-Dutch border and 80 km (around Kruibeke), a second zone occurred between 90 km (around the confluence of the Rupel-Schelde) and 100 km (around Lippenbroek), and a third zone occurred in the vicinity of 135 km (ca. Uitbergen). Full tidal measurements showed that in the upper estuary and at Kruibeke, during ebb tide, the suspended matter flux in 2006-2009 was generally higher than the period 2002-2005 and often exceeded the maxima observed during this period. Moreover, depth integrated suspended matter flux and water velocity showed two maxima initial suspended loads in the vicinity of 80 km and of 110 km, and the downstream one close to Melsele (ca. 70 km) had shifted toward Antwerp (ca. 78 km).

An explanation for the observed important phenomena can be related to the influence of tidal range and river discharge (Figure 25). Taverniers and Mostaert (2009) reported that the tidal range at Antwerp decreased in 2000-2005 but increased from 2006 to 2009. The river discharge showed similar trends of reducing in 2002-2005 and increasing after 2006 (Figure 25). The combination of increased river discharge and tidal range may lead to the observed upstream shift of high initial suspended load which consequently influences suspended matter behavior and water turbidity.

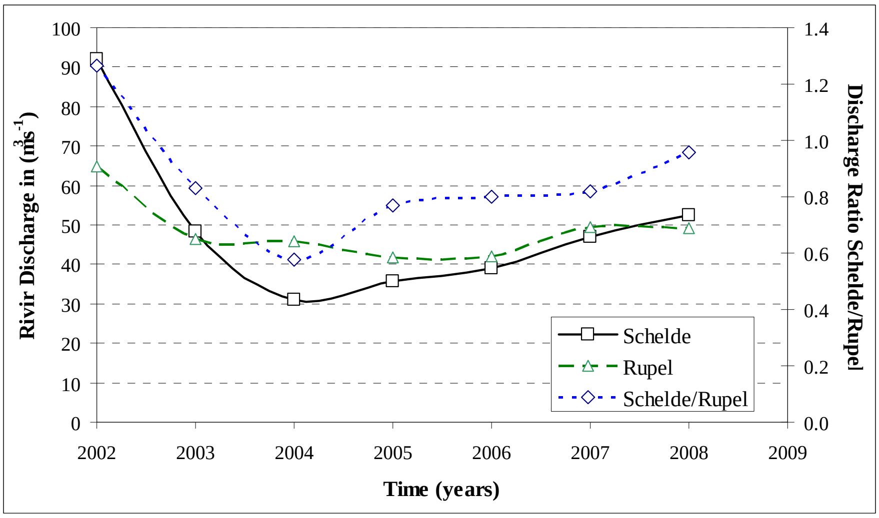

Figure 25. River discharges in the Schelde (upstream of the Schelde-Rupel confluence), Rupel, and the discharge ratio Schelde to Rupel.

4.5. References

Taverniers, E. and Mostaert, F. 2009. MONEOS, jaarboek monitoring WL 2008. WL2009R833_07rev4_0, Waterbouwkundig Laboratorium, Borgerhout, Antwerpen.

Hoofdstuk 5. Fytoplankton en fytobenthos

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Abstract

From July 2008 to August 2009 **phytoplankton** was monitored monthly on 15 different stations along the Scheldt continuum from Ghent to the Belgian-Dutch border and in 5 main tributaries of the Scheldt estuary. Compared to previous years a number of small tributaries was added in 2009. It concerns the Netes (Grote Nete, Kleine Nete and Beneden Nete) and the Dijle river. Those rivers flow into the Rupel, which is monitored at different stations in 2009. Also the frequency of sampling increased during summer. Between May and September 2009 two monitoring campaigns were organized, one at the beginning and one at the end of the month. Like in previous year, one summer bloom is observed in 2008 from July, when an extremely high chlorophyll a concentration is observed (822 µg chl a l^{-1}) at Melle, till September (243 µg chl a l^{-1}). These high chlorophyll a values are comparable with the maxima observed in the exceptional warm, sunny and dry summers of 2003 and 2004 and are much higher than the concentrations detected in 2007. The year 2009 also shows one summer bloom with a maximum of 550 µg chl a l^{-1} in July at Wetteren. Like in previous years, centric diatoms dominate the phytoplankton community in the estuary and the tributaries exposed to the tidal cycle. *Actinocyclus normanii, Cyclotella scaldensis* and *Stephanodiscus hantzschii* are the most important species. Chlorophytes are important in spring and summer, mainly in the river Bovenschelde.

Phytobenthos was also monitored monthly from July 2008 to August 2009 along a transect at 5 intertidal flats in the Scheldt estuary between Ghent and the Belgian-Dutch border. Maximal chlorophyll a concentrations were observed in summer at the intertidal flat in Appels. A general pattern is observed in the distribution of the phytobenthos. With a few exceptions, chlorophyll a concentration decreases from the high water line to the low water line. Diatoms were dominant at all flats in the brackish zone, while more diverse communities with green algae and euglenophytes as well were important in the freshwater zone.

5.1. Monitoring phytoplankton biomass and community composition

5.1.1. Methodology

Each month, a 50 ml water sample was taken on each station along the Schelde continuum and fixed with acid Lugol's solution. To ensure long-term storage, samples were post-fixed with formalin (5% final concentration) in the laboratory within 2 days of sampling. The phytoplankton species composition was analyzed using inverted microscopy. Therefore, a 5 to 10 ml subsample was transferred to a sedimentation chamber and phytoplankton cells were allowed to settle for one day. The sedimentation chamber was inspected by means of an inverted microscope (Zeiss Axiovert) at a magnification of 200 to 400 times. Identification was carried out to species level or, where identification was impossible by means of light microscopy, up to genus level. A fixed number of 100 phytoplankton 'units' were enumerated in each sample. A 'unit' corresponded to a phytoplankton cell, a coenobium or a colony. For each phytoplankton taxon, 15 units were measured to estimate the biovolume. The biovolume of each taxon was then multiplied by its abundance to estimate the total biovolume in the sample. Using published conversion factors (Menden-Duer & Lessard, 2000), biovolume was converted to biomass (in μg C l^{-1}).

Moreover, phytoplankton from each station was collected by filtering a known volume of water over a glass fibre filter (GF/F). This filter was immediately frozen and stored in the laboratory at a very low temperature (-80°C) to avoid degradation of pigments until analysis. Phytoplankton pigments were extracted in acetone 90 %. Sonnication with the use of a tip-sonnicator was used to destroy the phytoplankton cells and to make sure all pigments present on the filter were extracted in the solvent. The pigment extract was then injected into a Gilson High Performance Liquid Chromatography system to separate the different pigments present in the pigment extract. The time at which a peak in absorbance or fluorescence is observed (the retention time), can be used to identify the pigment associated with this peak. The surface of the peak can be used to quantify the amount of pigment that was present in the extract. By comparing retention times, peak surfaces and absorption spectra with analysis of pure pigment extracts of known concentration, the pigment associated with the peak can be identified and its concentration calculated. The HPLC system was set up, according to the method of Wright & Jeffrey (1997). Since the pigment 'chlorophyll a' is present in all phytoplankton groups, it is therefore used as measure for total phytoplankton biomass. Other pigments are specific for one or only a few phytoplankton groups. To calculate the phytoplankton class abundances from measurements of chlorophyll and carotenoid pigments determined by HPLC, the program Chemtax will be applied, (Mackey et al., 1996) which divide the total concentration of chlorophyll a over the different phytoplankton classes. The results will be expressed as amounts of chlorophyll a equivalent for each phytoplankton class.

5.1.2. Phytoplankton biomass in the Scheldt estuary in 2008-2009

One phytoplankton summer bloom was observed in the Scheldt estuary in 2008 (figure 1.1). It extended from July (822 μg chl a l^{-1}) to September (322 μg chl a l^{-1}) and occurred in the upstream reaches of the estuary. Maximal chlorophyll a concentration was detected at Melle where it reached 822 μg chl a l^{-1}, an exceptionally high value compared to previous years.

Like in 2008, the first chlorophyll a maximum of 2009 was found in May (185 μg chl a l^{-1}). In contrast with 2008 it was observed at Uitbergen instead of Dendermonde. The high chlorophyll a value close to the Belgian-Dutch border at boei 92 (140 μg chl a l^{-1}) is exceptional. Microscopic analysis revealed a bloom of *Thallasiosira* sp.. 2009 was, similar to the year 2008, characterized by one summer bloom. It extended from May (185 µg chl a l^{-1}) to August (352 µg chl a l^{-1}), as August 2009 was the last studied month in 2009. Maxima were observed in July 2009 (551 µg chl a l^{-1}) at Wetteren. Those values are less high than the ones observed in 2008 but also similar to the chlorophyll a concentrations of the summer of 2003-2004 (figure 1.4).

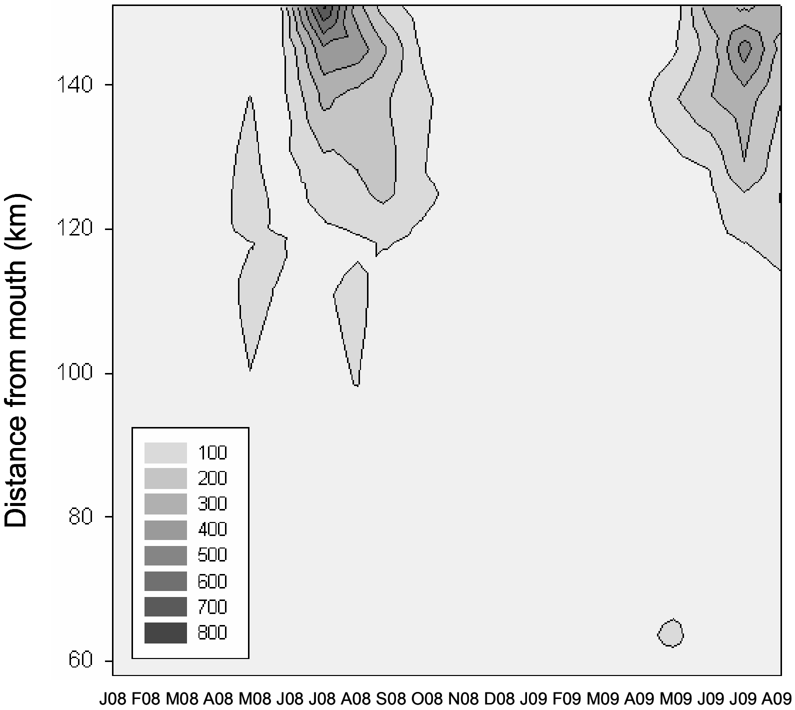

Figure 1.1: Chlorophyll a concentrations (µg l^{-1}) between January 2008 and August 2009 along the Scheldt estuary from the freshwater reaches towards the city of Ghent (at about 160 km) to the brackish water at the Belgian-Dutch border (at about 60km).

As seen in figure 1.2, higher discharge induces a decrease in retention time in the estuary which leads to a lower primary production and a weaker build-up of the phytoplankton.

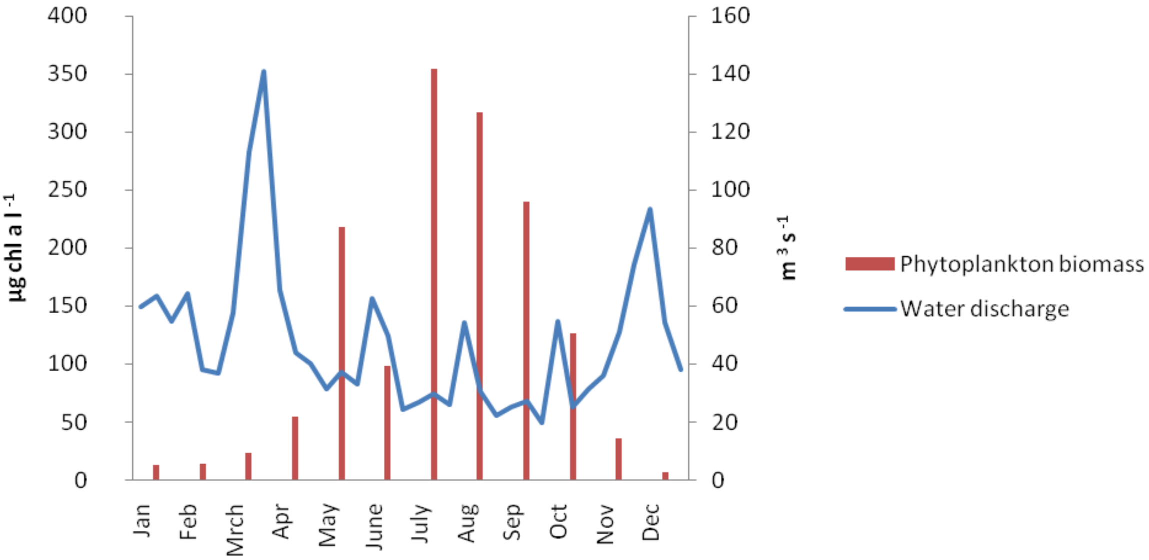

Figure 1.2: 10 days averages of water discharge from the Scheldt at Schelle compared to the monthly averages of phytoplankton biomass (chlorophyll a) from the Scheldt continuum between Ghent and the Belgian-Dutch border for 2008.

A decrease in chlorophyll a concentration is detected towards the brackish sampling stations all over the year (figure 1.3). This was also observed in previous years. It might be explained by salinity stress and stronger light limitation. The rapid increase in salinity in the brackish zones can lead to an osmotic shock in phytoplankton cells. Stronger light limitation in this zone is mainly due to a combination of a higher water column depth and a higher turbidity. In contrast to the average chlorophyll a values between July 2007 and June 2008, those between July 2008 and August 2009 do not show a maximum in the middle of the freshwater zone. Maximal phytoplankton biomass is found between Wetteren and Melle, two sampling stations that are situated in the most upstream part of the estuary.

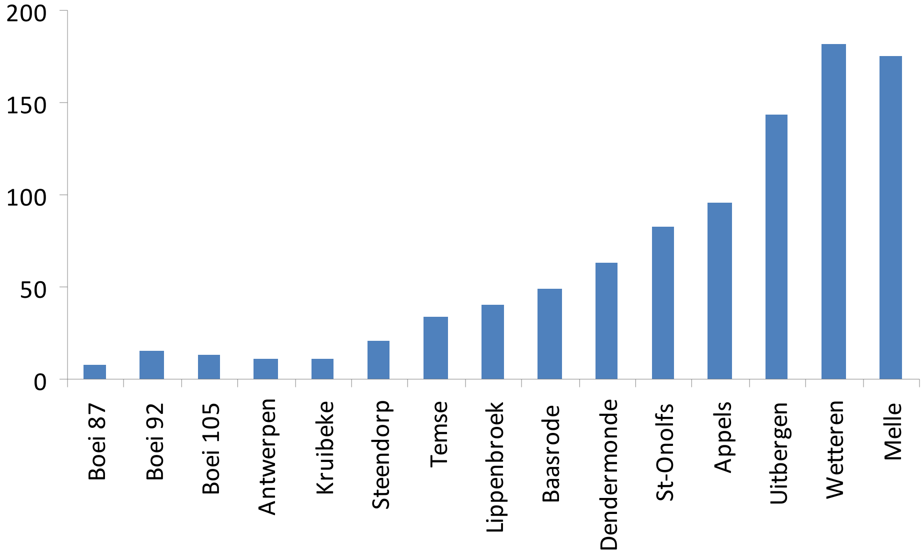

Figure 1.3: Averaged phytoplankton biomass (µg chl a l^{-1}) for each monitoring station in the Belgian part of the Scheldt estuary between July 2008 and August 2009.

5.1.3. Phytoplankton biomass in the Scheldt estuary between 1996-2009

Figure 1.4 shows an increase in chlorophyll a concentration between 1996 and 2009. Also a shift is observed in chlorophyll a maximum towards the more upstream parts of the estuary.

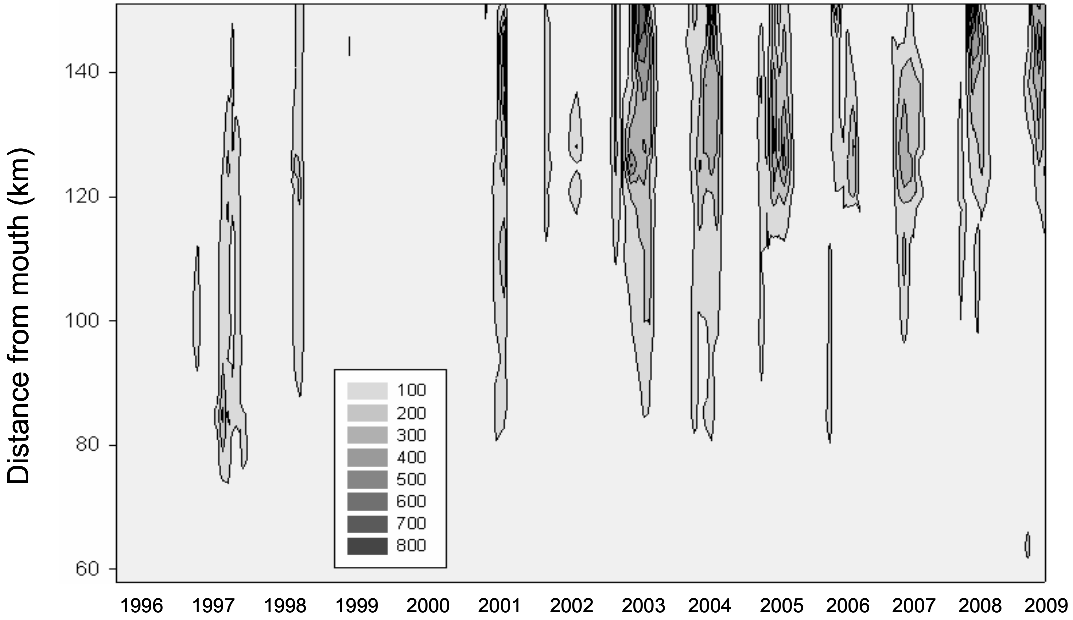

Figure 1.4: Phytoplankton bloom development in the Belgian part of the Scheldt estuary over the last 13 years (no data in 2000) expressed as µg chlorophyll a l^{-1}.

5.1.4. Comparison between estuary and tributaries

The comparison between the estuary and its tributaries is important to gain a better understanding of the contribution of phytoplankton communities from the tributaries to the phytoplankton community in the Scheldt estuary. Figure 1.5 represents the amount of chlorophyll a in the tributaries of the Scheldt estuary and in the upstream and downstream stations between July 2008 and August 2009.

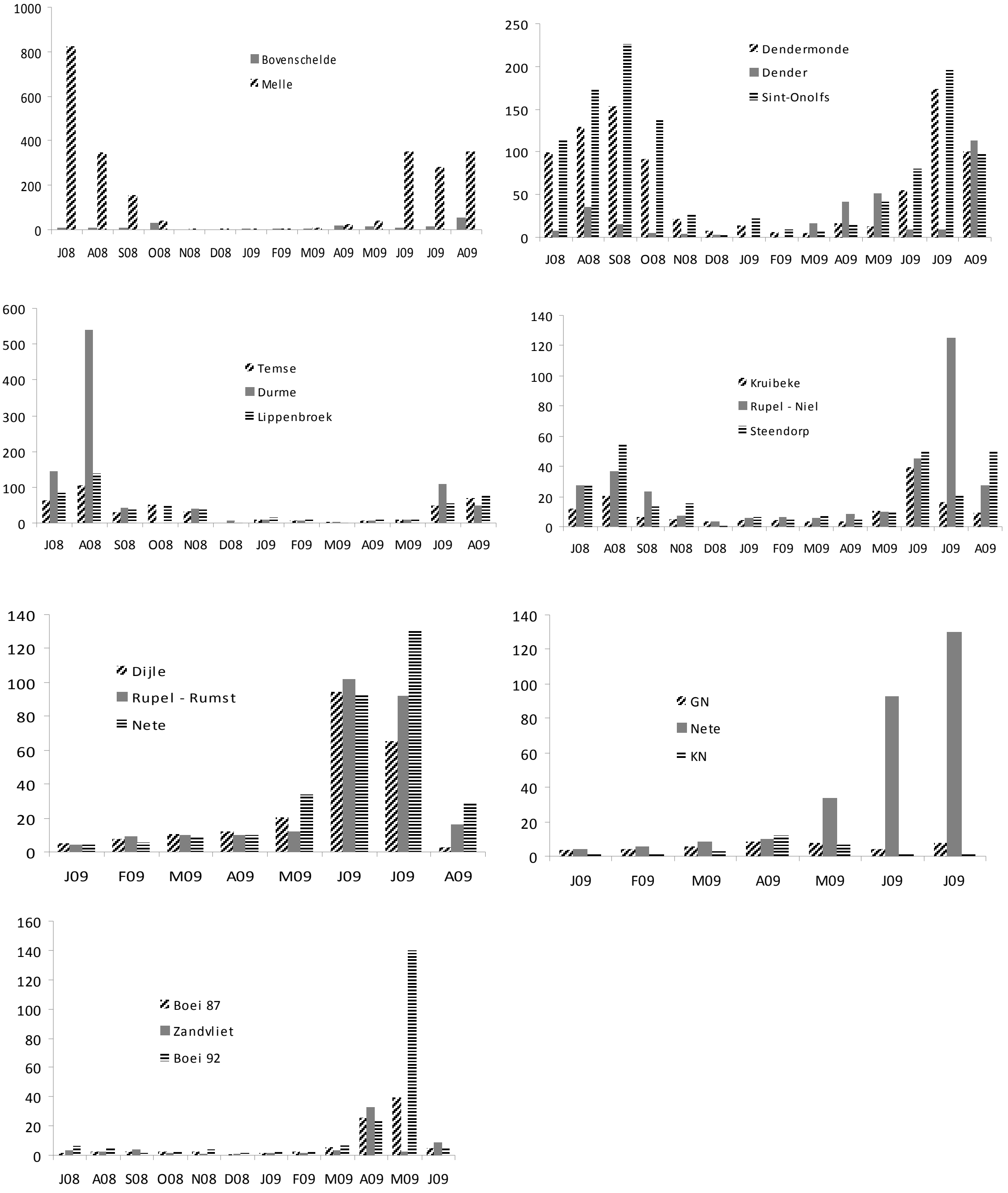

Figure 1.5: Monthly chlorophyll a concentrations (in $\mu g\ l^{-1}$) from the tributary rivers of the Scheldt estuary compared to adjacent monitoring stations in the estuary in 2008-2009.

Phytoplankton biomass in the river **Bovenschelde** was significantly lower than the biomass in the estuary. Maximal chlorophyll a concentrations in the river reached 32 µg chl a l^{-1} in October 2008 and 56 µg chl a l^{-1} in August 2009, while in the estuary a maximal biomass was observed of 822 µg chl a l^{-1} in July 2008 and 352 µg chl a l^{-1} in August 2009. The river could thus not have been a source of phytoplankton for the Scheldt estuary.

Chlorophyll a concentrations in the river **Dender** were in general much lower than in the Scheldt estuary, except in spring 2009 when the Dender could have been a phytoplankton source for the estuary and in August 2009. In spring 2008, a similar pattern was observed (Maris et al., 2008).

In the **Durme** river, exceptionally high phytoplankton biomass was observed in August 2008 (538 µg chl a l^{-1}). In general, phytoplankton biomass is substantially higher in the Durme than in the Scheldt estuary. The Durme river is a shallow river and phytoplankton can benefit from a better light climate, especially in spring and summer, which explains the higher concentrations. These higher concentrations make the Durme a possible source of phytoplankton for the Scheldt estuary.

Phytoplankton biomass in the **Rupel** river at Niel was always similar to the biomass in the estuary, except in July 2009, where biomass reached high values in the Rupel. Compared to previous years, phytoplankton biomass was much higher in the summer of 2009. This could be caused by the improved water quality in the Rupel, due to the water treatment at the Zenne river, with a better light climate as a consequence. When this trend continues and phytoplankton biomass remains high in the Rupel river, it may become a phytoplankton source for the Scheldt estuary.

At the confluence of the **Nete** and **Dijle** river, the Rupel arises. Chlorophyll a concentrations in both of the rivers are similar to the values in the Rupel at Rumst. Nete and Dijle could be phytoplankton sources for the Rupel. The Nete river, also called the Beneden Nete, is composed by the confluence of the **Kleine Nete** and the **Grote Nete**. Biomass in the Nete is always higher than the one in the two little rivers. From May till July 2009 phytoplankton biomass in Nete river is much higher. The Kleine and Grote Nete could thus not be a phytoplankton source for the Beneden Nete.

In general, phytoplankton biomass at **Zandvliet** is lower than the biomass in the estuary. At those brackish sampling stations biomass is much lower than in the rest of the estuary. In April 2009 high biomass is observed at Zandvliet. One month later, in May 2009, exceptionally high values were reported in the estuary, especially at boei 92. Possible causes for this high chlorophyll a concentrations should be investigated.

5.1.5. Phytoplankton community composition in the Scheldt estuary and tributaries

Like in previous years, pigment analysis revealed that phytoplankton biomass in 2008-2009 was dominated by diatoms along the estuary, as was shown by the amount of fucoxanthin (figure 1.6). Most abundant diatom species was *Actinocyclus normanii.* With a number of exceptions it dominated the estuary the whole year round. Beside *Actinocyclus normanii* also *Cyclotella* spp. and *Stephanodiscus hantzschii* were important. In Antwerp, most important species was, beside *Actinocyclus normanii*, *Thallasiosira* sp. *Stephanodiscus hantzschii* became more important in spring in the freshwater parts of the estuary. Other diatoms like *Navicula* sp., *Aulacoseira granulata* and *Asterionella formosa* contributed significantly to the total phytoplankton biomass at times while diatom species never attained significant biomass. Like in previous years, chlorophytes were only of importance in spring and summer in the estuary, especially in the freshwater part. *Desmodesmus* spp. and *Scenedesmus* spp. were the dominant genera. The cryptophytes *Cryptomonas* sp. was at times of minor importance while cyanobacteria, euglenophytes and dinophytes were observed but never attained a significant biomass.

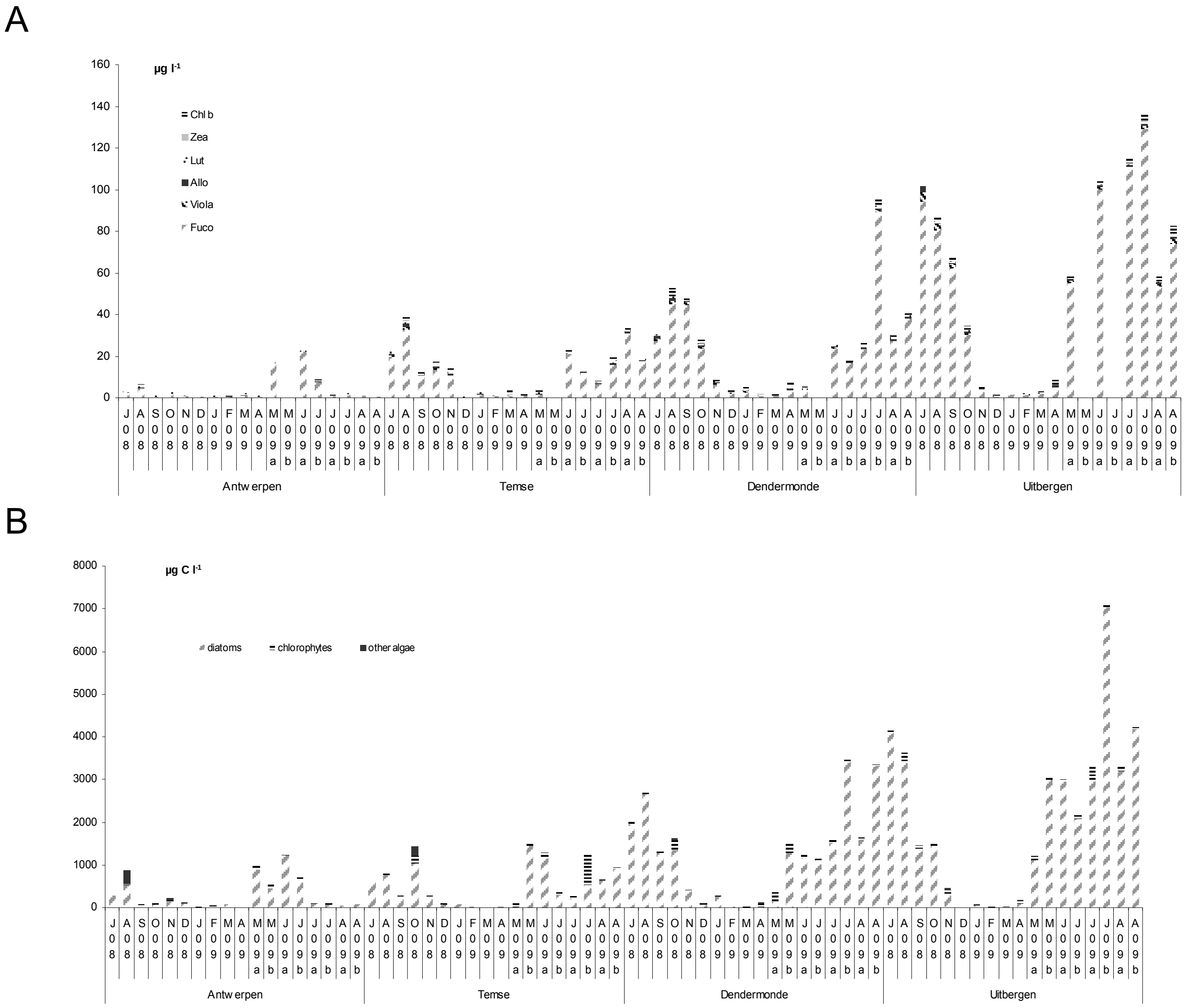

Figure 1.6: Phytoplankton pigment composition in µg l-1 (A) and biomass in µg C l-1 divided in diatoms, chlorophytes and other phytoplankton groups (euglenophytes, cryptophytes, cyanobacteria and dinophytes) on the base of cell counts (B) in the different zones of the Scheldt estuary in 2008-2009.

Fucoxanthin was in the tributaries the most important pigment as well, especially in the Rupel, Dijle, Nete and the Durme, which indicates the dominance of diatoms. Chlorophyll b became more important in the river Dender and Bovenschelde, which is an indication of the presence of chlorophytes. Other pigments were found only in minor concentrations (figure 1.7 to figure 1.10).

The **Rupel** river is dominated by diatoms throughout the year. Winter is dominated by *A. normanii*, while *S. hantzschii* became the most important species in spring. The summer is dominated by *Cyclotella* spp. The river never became dominated by chlorophytes, although they gained importance during summer.

In the **Nete** river, S. hantzschii was the most important species during winter. In spring, chlorophytes became most important, while summer and autumn were dominated by *Cyclotella* spp. *S. hantzschii* dominated the phytoplankton community in spring in the **Dijle** river. During summer there was a shared dominance by *A. normanii* and *Cyclotella* spp. The river **Bovenschelde** is dominated by chlorophytes, especially *Desmodesmus* spp. In spring, *S. hantzschii* became the most important species. In the **Dender** *S. hantzschii* was always dominant except in July and September 2008, when *A. normanii* became most important and in October 2008 when chlorophytes dominated the phytoplankton, more specific *Desmodesmus* spp. The **Durme** was dominated by *A. normanii* throughout the year. No other species became as important as that diatom. In August 2008 a bloom of *A. normanii* occurred, with exceptional high biomass of 6648 mg C l^{-1} reported. Phytoplankton communities in the docks at **Zandvliet** showed a co-dominance of *A. normanii* and *Thallasiosira* spp. The latter became more important during spring. In April 2009 the dinophyte *Gymnodinium* spp. was also important.

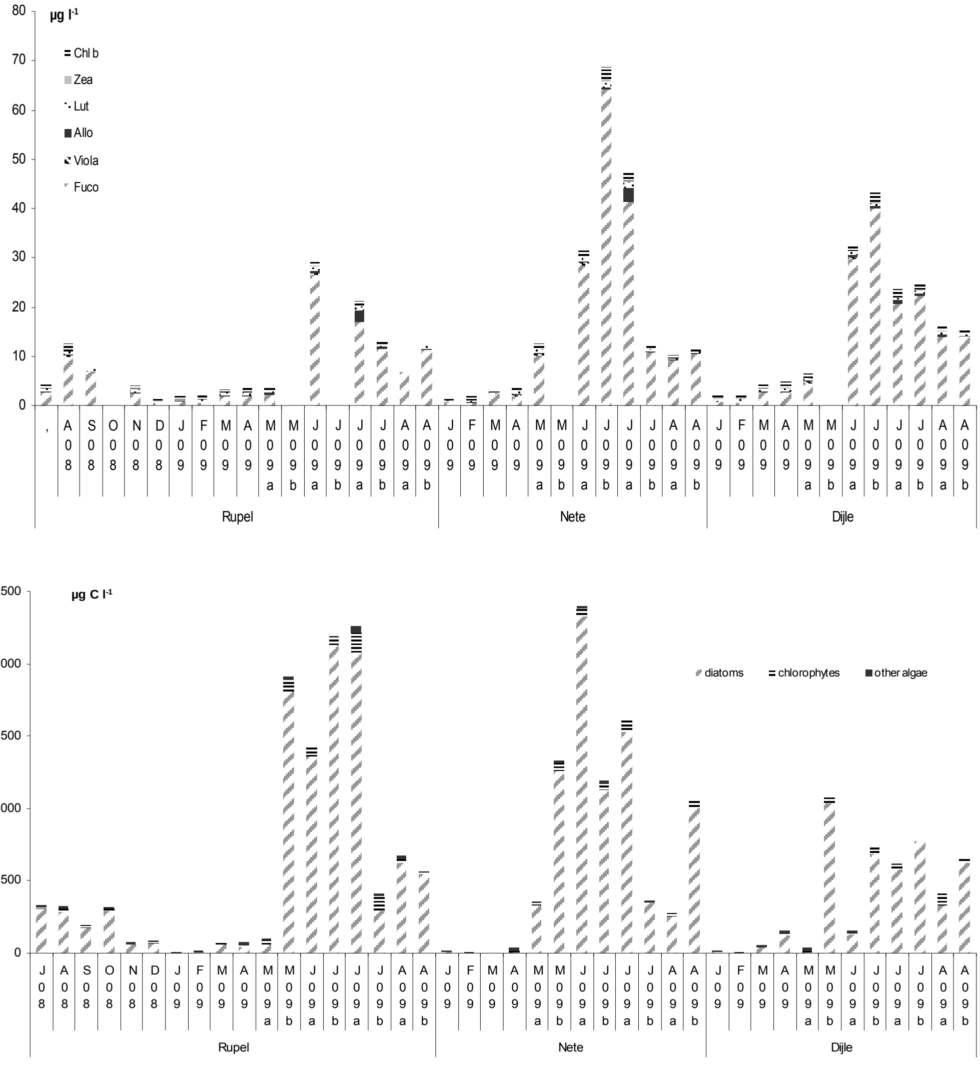

Figure 1.7: Phytoplankton pigment composition in µg l^{-1} and biomass in µg C l^{-1} divided in diatoms, chlorophytes and other algae groups (euglenophytes, cryptophytes, cyanobacteria and dinophytes) on the base of cell counts in the Rupel, Nete and Dijle in 2008-2009.

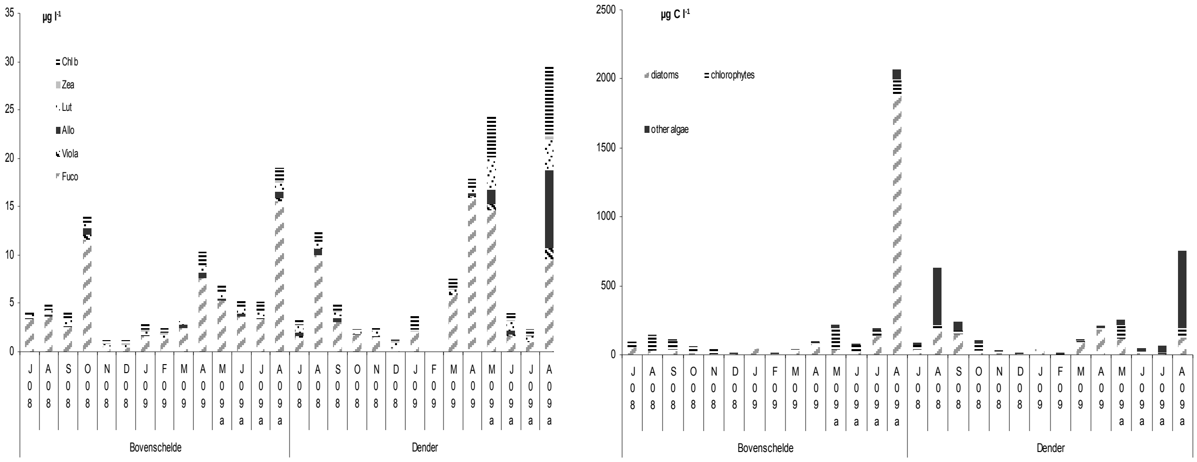

Figure 1.8: Phytoplankton pigment composition in µg l-1 and biomass in µg C l-1 divided in diatoms, chlorophytes and other algae groups (euglenophytes, cryptophytes, cyanobacteria and dinophytes) on the base of cell counts in the Bovenschelde and the Dender in 2008-2009.

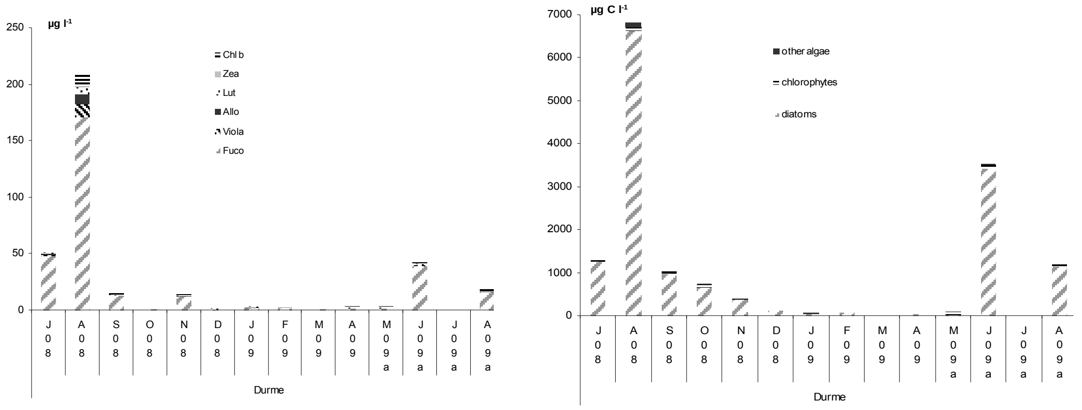

Figure 1.9: Phytoplankton pigment composition in µg l-1 and biomass in µg C l-1 divided in diatoms, chlorophytes and other algae groups (euglenophytes, cryptophytes, cyanobacteria and dinophytes) on the base of cell counts in the Durme in 2008-2009.

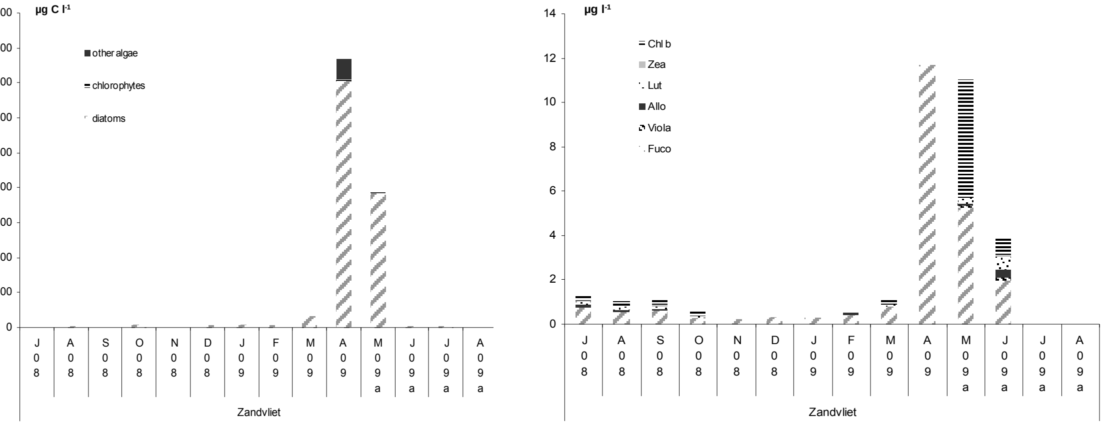

Figure 1.10: Phytoplankton biomass in µg C l-1 divided in diatoms, chlorophytes and other algae groups (euglenophytes, cryptophytes, cyanobacteria and dinophytes) on the base of cell counts and pigment composition in µg l-1 in Zandvliet in 2008-2009.

5.1.6. Dynamics in chlorophyll a concentration during the tidal cycle

In September 2008 three 13h measurements were carried out at different stations in the Scheldt estuary to assess the impact of the tide on the phytoplankton populations. These measurements were carried out at Schellebelle (freshwater zone), at Kruibeke (brackish zone) and the river Rupel, always between two high tides. Chlorophyll a was measured with HPLC as described before (figure 1.11).

During low tide, a decrease in chlorophyll a concentration was observed at the Rupel station. This is probably due to the fact that during high tide, phytoplankton is transported into the river from the Scheldt estuary. During low tide, phytoplankton poor water from the rivers upstream the Rupel, enters the Rupel. Same pattern was observed during 13h measurements in the Rupel in 2007.

Phytoplankton biomass at Schellebelle shows an alternating pattern during low tide. At the beginning of low tide a small decrease is observed, immediately followed by an increase. Then phytoplankton biomass decreased, again followed by an increase. This pattern is different from the one seen in 2007, when a decrease is observed during low tide. At Kruibeke, an increase of chlorophyll a concentration was reported during low tide, just like in 2007. The difference in the timing of the chlorophyll a maximum suggests that the maximal chlorophyll a concentration in the estuary was situated upstream Kruibeke and downstream Schellebelle.

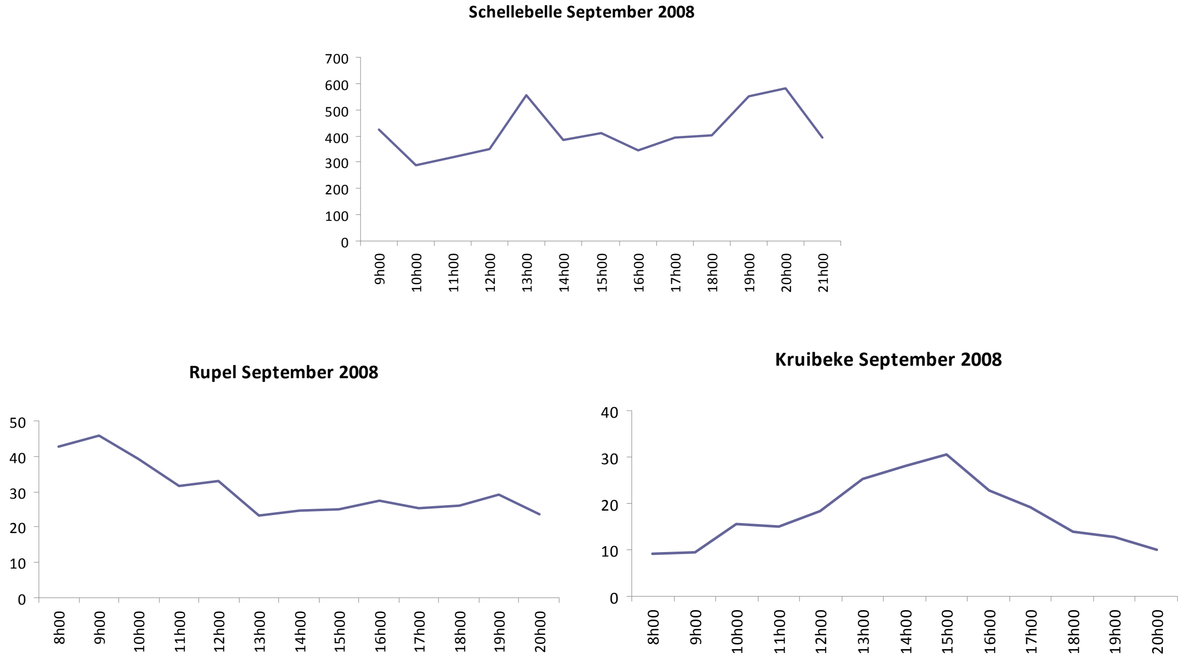

Figure 1.11: Dynamics in phytoplankton biomass (µg chl a l^{-1}) during the tidal cycle measured during 13h campaigns in summer 2008 in the Scheldt estuary and the Rupel.

5.2. Monitoring phytobenthos biomass and community composition on intertidal flats

5.2.1. Methodology

From July 2008 onwards, samples were collected monthly at 5 selected intertidal flats in the Schelde estuary between Gent and the Dutch-Belgian border (figure 9): the silty and the sandy flat at Groot Buitenschoor at the Belgian-Dutch border, Boerenschans, Ballooi and Appels. At each intertidal flat, between 3 and 5 stations situated along a transect between the high and low water mark were sampled. At each station, 3 replicate samples were mixed to average out small-scale patchiness. The upper 2 mm of sediment was frozen *in situ* with liquid nitrogen and sampled using a contact core. The frozen samples were kept in liquid nitrogen during the fieldwork and stored at -80 °C in the laboratory until analysis. The sediment samples were freeze-dried prior to analysis to remove water, and pigments were extracted in acetone (90 %) using sonnication. Pigments were analyzed using HPLC as described above for phytoplankton samples.

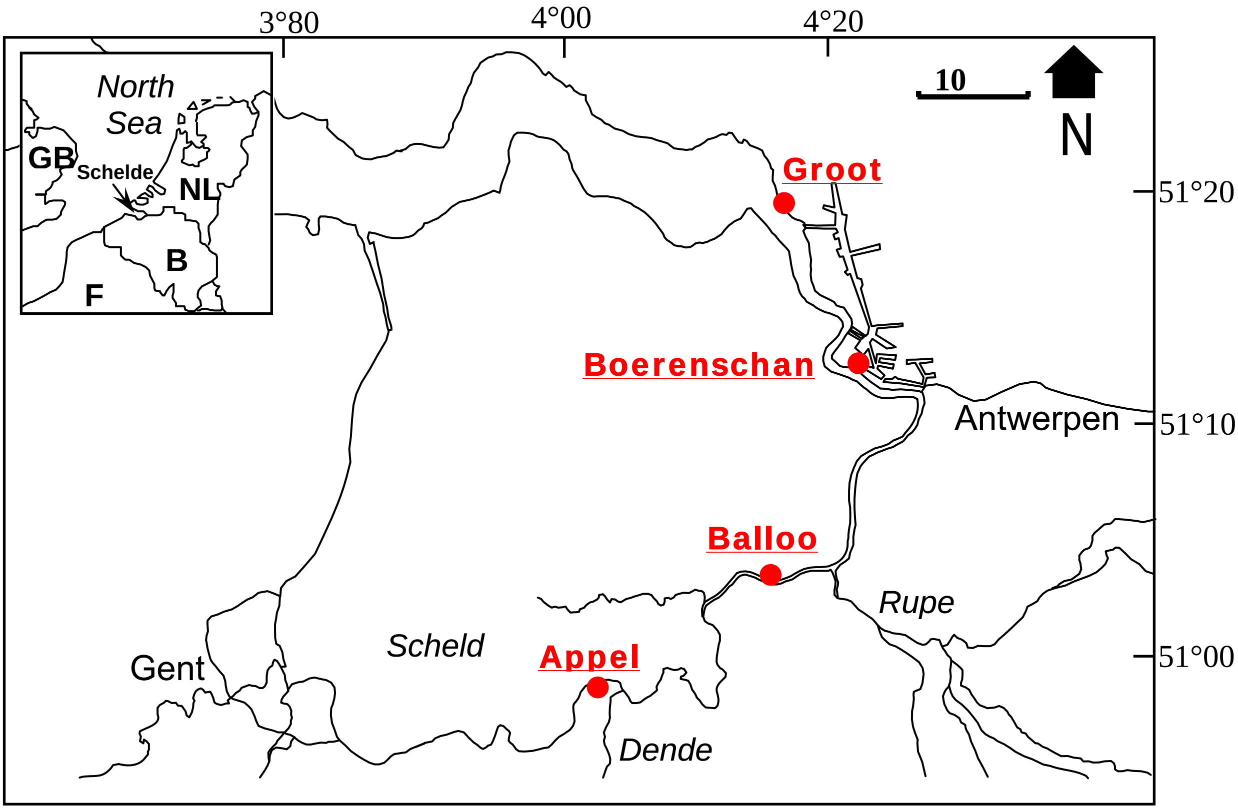

Figure 1.12: The Scheldt estuary with location of the intertidal flats that were sampled for the study of the phytobenthos development in 2008-2009.

5.2.2. Phytobenthos biomass

Maximal chlorophyll a concentrations were observed in summer 2008 at the intertidal flat in Appels. In July average concentrations of 122 mg chl a m^{-2} were reported, in September average concentrations of 116 mg chl a m^{-2}. These values are similar to the ones observed in summer 2007, when maximal concentrations of 103 mg chl a m^{-2} in July and 124 mg chl a m^{-2} in August were reached, also in Appels (figure 1.13).

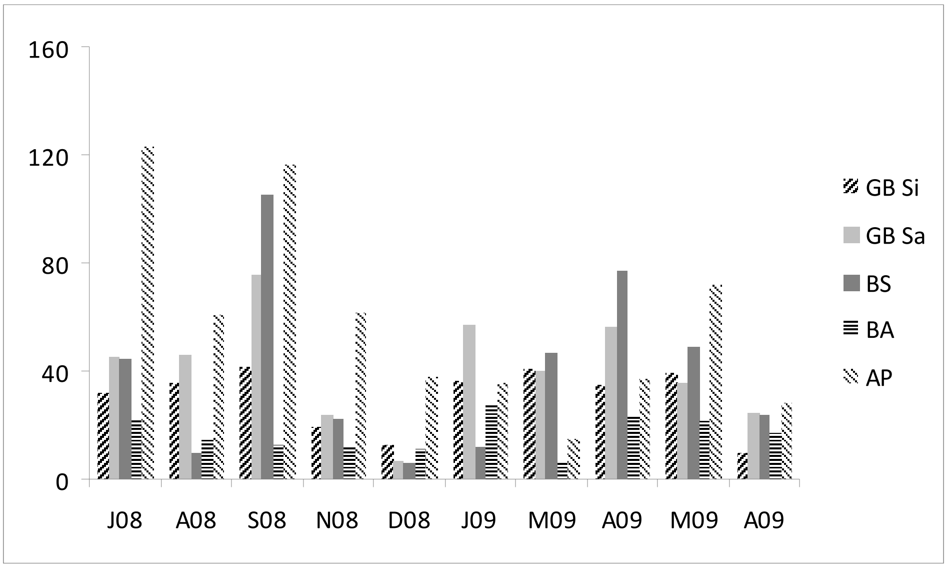

Figure 1.13: Average phytobenthos biomass in mg chl a m^{-2} for each tidal flat sampled along the Scheldt estuary in 2008-2009. Each column is the averaged biomass from 3-4 locations along a transect from the high water mark to the low water at each intertidal flat.

At all intertidal flats, chlorophyll a concentration decreased in general from the high water line to the low water line (figure 1.14). This can probably be ascribed to the fact that the sediment near the low water line is only exposed to the light during a short period at low tide while during high tide, the sediment is covered by turbid water so that phytobenthos receives no light. Also, sediments near the low water line are exposed to strong tidal currents which may result in losses of phytobenthos populations due to re-suspension.

5.2.3. Phytobenthos species composition

Pigment analysis revealed always a dominance of fucoxanthin, the characteristic pigment of diatoms (figure 1.15). Especially the intertidal flats at Groot Buitenschoor, rarely contain any other pigments in significant quantities. Towards the freshwater zone, also other pigments became more important. At Appels, high levels of chlorophyll b concentration indicate the presence of green algae. In August 2008 chlorophyll b was the most important pigment at the high water line at Appels. Also neoxanthin, indicative for euglenophytes and zeaxanthin, indicative for cyanobacteria were present at certain times. Euglenophytes were especially present around the high water line at the intertidal flats of Ballooi and Appels, which was clearly visible in the field as green mats. Comparison between phytoplankton and phytobenthos showed a strong similarity in species composition, which possibly suggest that part of the phytobenthos community is composed by settled phytoplankton.

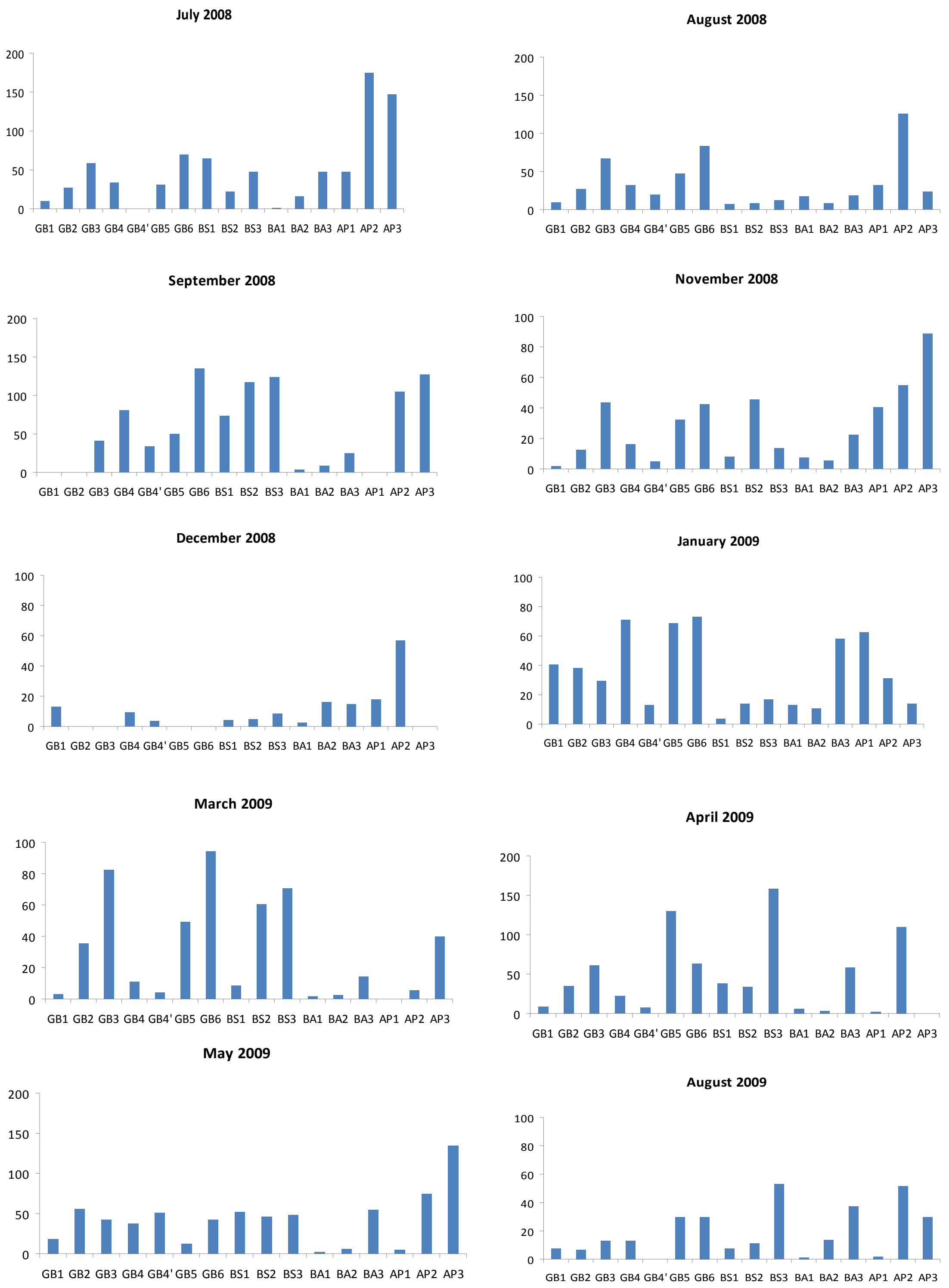

Figure 1.14: Spatial and temporal variability in phytobenthos biomass in mg chl a m^{-2} along intertidal flats from the Scheldt estuary, sampled from July 2008 till August 2009. (GB: Groot Buitenschoor, BS: Boerenschans, BA: Ballooi, AP: Appels).

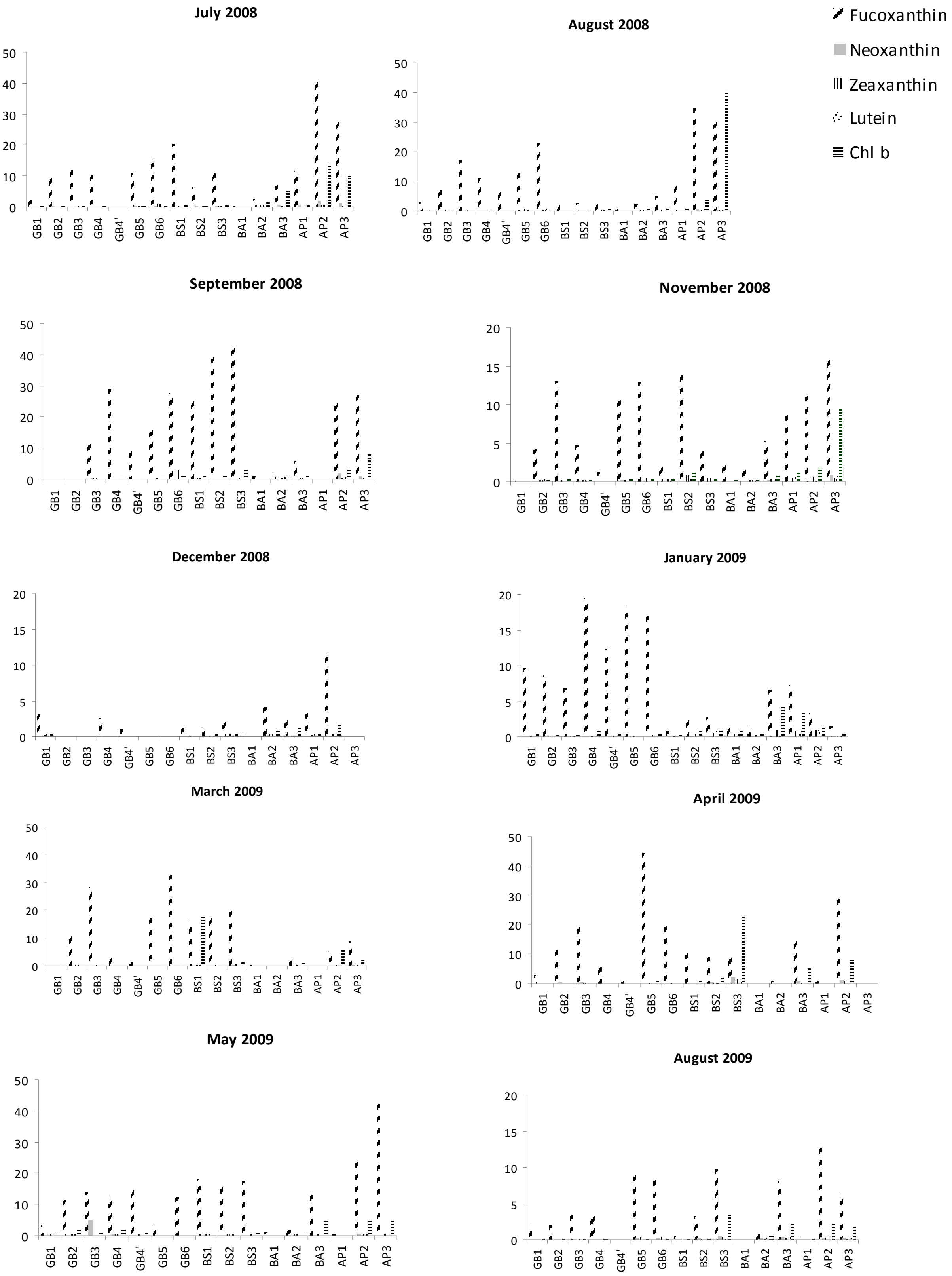

Figure 1.15: Spatial and temporal variability in phytobenthos pigment composition in µg m^{-2} along intertidal flats from the Scheldt estuary, sampled from July 2008 till August 2009. (GB: Groot Buitenschoor, BS: Boerenschans, BA: Ballooi, AP: Appels).

5.3. Future studies

- Continued monitoring of species composition by microscopical counts at selected stations in the estuary and in the tributaries
- Continued monitoring of phytoplankton pigments by HPLC analyses
- Continued monitoring of the Schelde tributaries (Bovenschelde, Dender, Durme, Rupel and Zandvliet)
- Continued monitoring chl a during 13h measurements at Lippenbroek and Kruibeke and the river Rupel
- Continued monitoring of phytobenthos from the intertidal flats of the Schelde estuary

5.4. References

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Wright, S.W. & Jeffrey, S.W. 1997 High-resolution HPLC system for chlorophylls and carotenoids of marine phytoplankton. In S.W. Jeffrey, R.F.C. Mantoura and S.W. Wright (ed.) Phytoplankton pigments in oceanography. UNESCO Publishing, Paris: 327-341.

Hoofdstuk 6. Studie naar de primaire productie

Jean-Pierre Vanderborght

Eindverslag voor deelstudie 1 (perceel 1), periode juli 2008 - juni 2009

Université Libre de Bruxelles
Laboratoire d'Océanographie Chimique et de Géochimie des Eaux

Dir. Prof. Dr. Lei CHOU

6.1. Introduction

The present report synthesizes the results obtained between July 2008 and June 2009 in the framework of the OMES monitoring program ("Deel 5: Studie naar de primaire productie"). During this period, seven monthly cruises and three tidal cycles ("13-hour cycles") in the estuary (Kruibeke – Schellebelle) and in the Rupel (Boom) have been performed. In 2009, the monthly survey has been extended to 4 complementary stations in the Rupel (Niel – Rumst), the Dijle and the Nete rivers. As before, the on-board activity comprised the acquisition of several water quality parameters and the determination of the *in situ* solar light penetration. The measurement of phytoplankton primary productivity was performed in the laboratory the same day as the water sampling.

Water quality parameters (temperature, conductivity/salinity, dissolved oxygen, pH), turbidity and chlorophyll fluorescence have been routinely measured using a multi-parameter sensor YSI Model 6600EDS. Light attenuation coefficient (k_d) was measured using a pair of horizontal (downward) irradiance sensors immerged at two depths separated by a fixed vertical distance (40 cm). At each station, data acquisition was performed during 3 to 8 minutes at a rate of one measurement per second. From these time series (to be found in Annex B : Light climate), the amplitude of the short term fluctuation of the coefficient k_d may be characterised by computing the value of the standard deviation. Suspended particulate matter (SPM) concentrations was determined by filtration (GF/F) for comparison purpose with turbidity and light attenuation coefficients.

The determination of the photosynthetic parameters (photosynthetic efficiency α and maximum specific rate of photosynthesis P^B_{max}) at nine stations along the estuarine gradient (+ three complementary stations in the Rupel (Boom), Dijle and Nete rivers in 2009) has been carried out as previously using ^{14}C incorporation in water samples submitted to a gradient of light intensity in a temperature-controlled incubator. Alkalinity, chlorophyll a and phaeopigments concentrations have been measured in the laboratory on the samples used for ^{14}C incubation. Dissolved inorganic carbon (DIC) concentration has been obtained from *in situ* pH data and laboratory determination of alkalinity, taking into account the influence of temperature and salinity on the carbon speciation.

6.2. Synthesis of the results

6.2.1. Light attenuation coefficient and euphotic depth

Between April and October 2008, the light attenuation coefficient for vertical irradiance k_d varied from 3.24 m^{-1} at the station "Grens" (July 2008) to 17.8 m^{-1} at the station Lippenbroek (August 2008). The observed maximum and minimum euphotic depths (defined as the depth where irradiance is equal to 1% of the value at the water surface) thus varied between 1.42 m and 0.26 m. The distribution of k_d values (based on all results from monthly profiles) is given in Figure 1. Only 65 % of the values are comprised between 3 and 8 m^{-1}, to be compared to values of 71 % in 2007, 74% in 2006, 65% in 2005 and 69% in 2004. This is the lower value observed since 2005, after a couple of years of narrower k_d distribution and a tendency towards higher water transparency. The median value of the overall distribution (including the results from both monthly and 13h monitoring) is equal to 7.50 m^{-1}, which corresponds to a euphotic depth of 0.61 m, the lowest value observed since 2002.

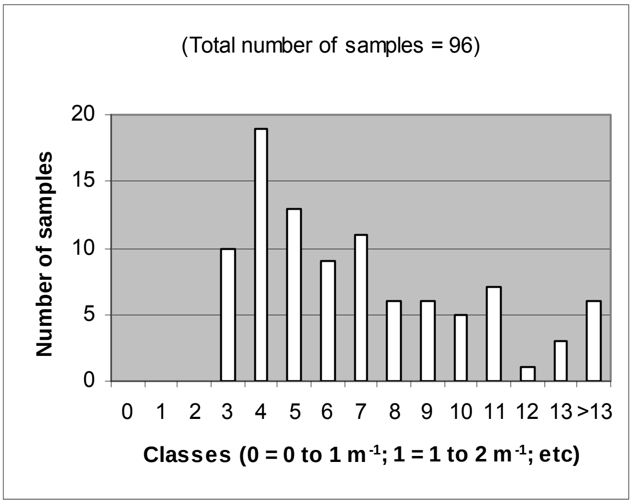

Figure 1: Distribution of measured k_d values (2008)

In Table 1, a synthesis of all results obtained since 2002 is presented, together with an up-dated characterization for the whole period. For 2009, which includes only 3 monitoring campaigns (April – June), preliminary results are also presented as an indication.

Results shown in Table 1 confirm the on going reduction of the average water transparency during the last three years (2006 – 2008) after a maximum observed in 2005. This evolution is not only noticeable for the median values, but also for the minimum and maximum values. Moreover, Figure 2 shows that the decrease of the median euphotic depth (i.e. the increase of the light attenuation coefficient) observed in 2008 can be entirely attributed to a major change in the optical properties of the estuary between the border (km58) and the Rupel (Kruibeke, km 85) (see bold line, open circles) . In contrast, the freshwater part of the estuary (upstream of the Rupel mouth) do not reveal in 2008 any marked difference with respect to the previous years: an average annual euphotic depth fluctuating between 0.81 and 0.45 m, with a mean value equal to 0.68 m (0.67 m in 2007), is observed in the upper, freshwater part of the estuary (between km 151 and km 94). In the downstream part however, it reaches only 1.1 m at the Belgian-Dutch border (against 1.8 m in 2007), with an average value of 0.85 m between km 58 and km 85. This value is about one half of the average value observed in 2005 for this part of the estuary (1.59 m).

Year	Number of samples	k_d (m^{-1})			Euphotic depth (m)		
		min	max	median	min	max	median
2002	203	1.77	16.5	5.11	0.28	2.60	0.90
2003	189	1.71	17.5	5.33	0.26	2.69	0.86
2004	199	2.01	17.2	5.76	0.27	2.29	0.80
2005	152	1.03	13.1	4.60	0.35	4.47	1.00
2006	124	1.84	12.4	5.33	0.37	2.50	0.85
2007	147	1.95	17.7	6.20	0.26	2.37	0.74
2008	132	3.24	17.8	7.50	0.26	1.42	0.61
2002 - 2008	**1205**			**5.49**			**0.84**
2009 (part)	**59**	**2.53**	**10.8**	**4.66**	**0.43**	**1.82**	**0.99**

Table 1: Statistical parameters of the distribution of k_d values

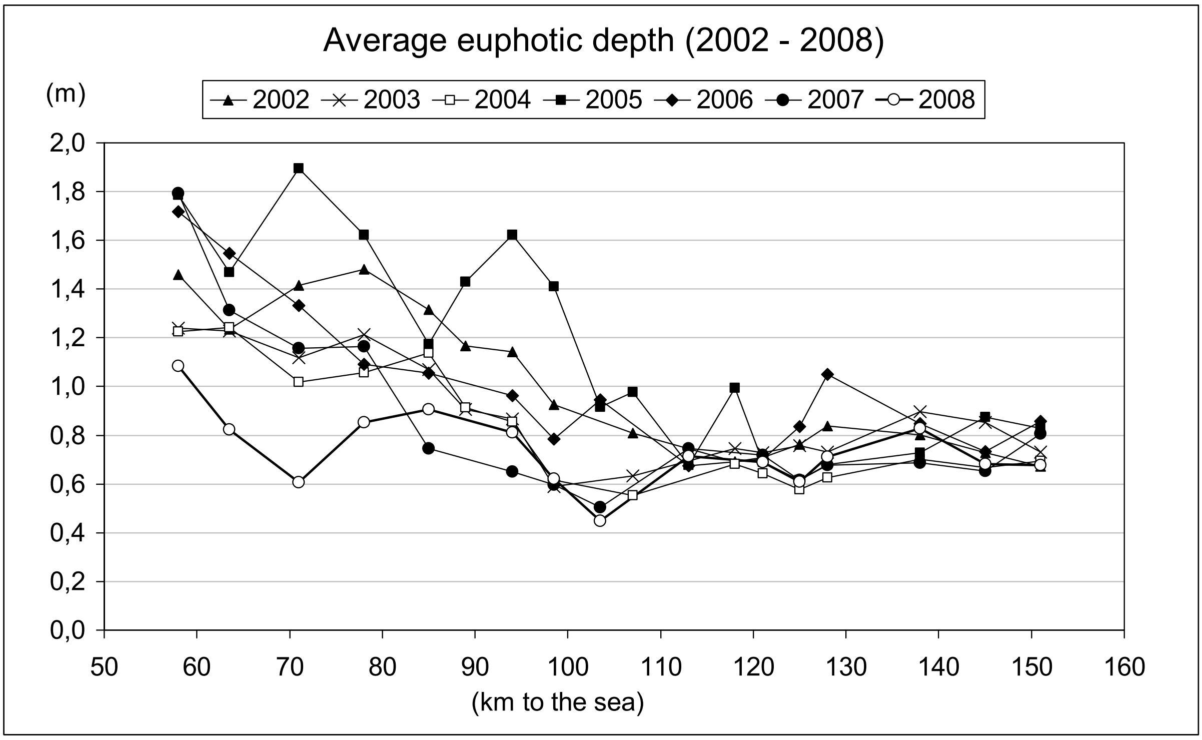

Figure 2

The year 2008 has now replaced the year 2007 as the "darkest" year since the start of our survey, an evolution that can largely be attributed to the very high SPM concentrations and turbidity values that have been observed in July, August and October 2008 in the downstream part of the estuary.

6.2.2. k_d – SPM relationship

The relationship between the attenuation coefficient k_d and the SPM concentration extracted from the 2008 dataset is shown in Figure 3.

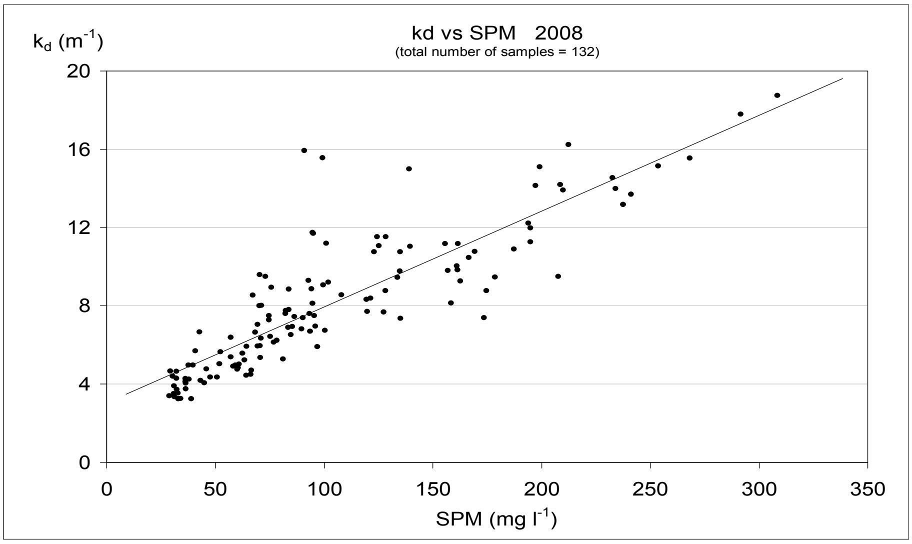

Figure 3: k_d – SPM correlation for 2008

Based on 132 pairs of k_d and SPM values, this relation may be expressed as follows:

$k_d = 0.049$ SPM $+ 3.049$ ($r^2 = 0.75$)

with k_d in m^{-1} and SPM in mg l^{-1}. For 2009, based on 59 couples of data only, the relation is :

$k_d = 0.053$ SPM $+ 2.087$ ($r^2 = 0.69$)

Figure 4 illustrates the complete dataset for the period 2002 – 2009 and the regression line based on the 1205 couples k_d – SPM available for the whole 7 ½ year period.

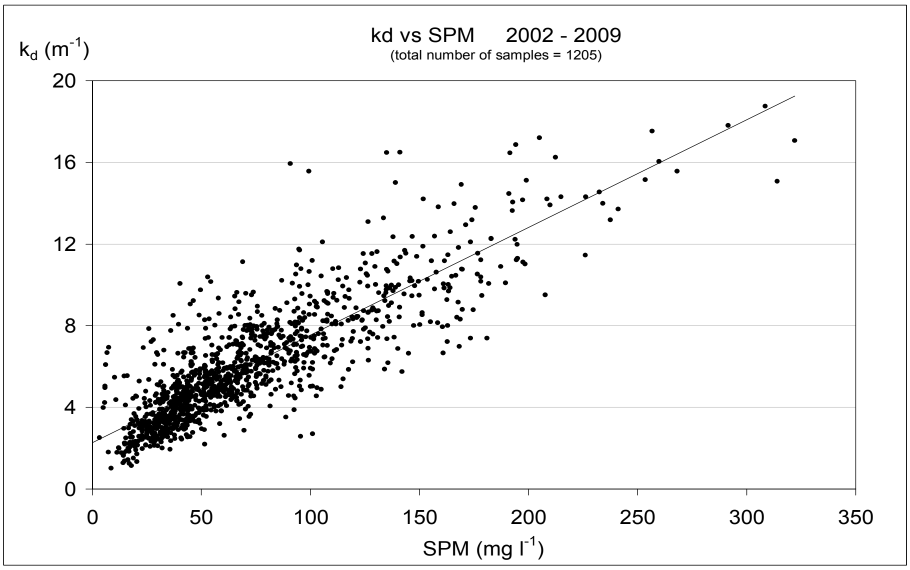

Figure 4: k_d – SPM correlation for the period 2002 – 2009

Table 2 synthesizes the relationships that have been observed between k_d and SPM since 2002, together with the linear regression obtained for the complete dataset. The slope and intersect of the regression line determined for the 2008 data set were almost identical to the 2006 and 2007 values. The hypothesis of a decreasing trend for the slope and of an increasing trend for the intercept put forward in the last two annual reports is therefore not confirmed.

Year	Slope	Intercept
2002	0.048	2.21
2003	0.055	2.06
2004	0.055	2.14
2005	0.059	1.81
2006	0.048	2.76
2007	0.046	2.98
2008	0.049	3.05
2009	0.053	2.09
2002 - 2009	$k_d = 0.053$ SPM $+ 2.27$ ($r^2 = 0.73$)	

Table 2: Linear regression between k_d and SPM

6.2.3. Light climate dynamics at the tidal timescale

The variations of k_d, turbidity and SPM during the 13-hour cycles are shown below for the 3 campaigns performed in 2008: Schellebelle (Figures 5 - 6); Rupel at Boom (Figures 7 - 8); Kruibeke (Figures 9 - 10). For each experimental site, the first graph illustrates the temporal variation of the irradiance attenuation coefficient k_d (line) and turbidity (line + dots) over one tide. In the second graph, the k_d time series is compared with the SPM concentrations measured on discrete water samples. As already observed in 2007, the temporal evolution of the irradiance attenuation coefficient k_d over one tide has always shown a remarkable correlation with the turbidity data at all 3 stations. Turbidity and SPM results have shown a good correlation in Boom and Kruibeke ($r^2 = 0.96$ at Kruibeke in September 2008, see Figure 11). This was however not the case in Schellebelle, where the SPM variation did not match the peak in both k_d and turbidity values observed between 14:00 and 17:00.

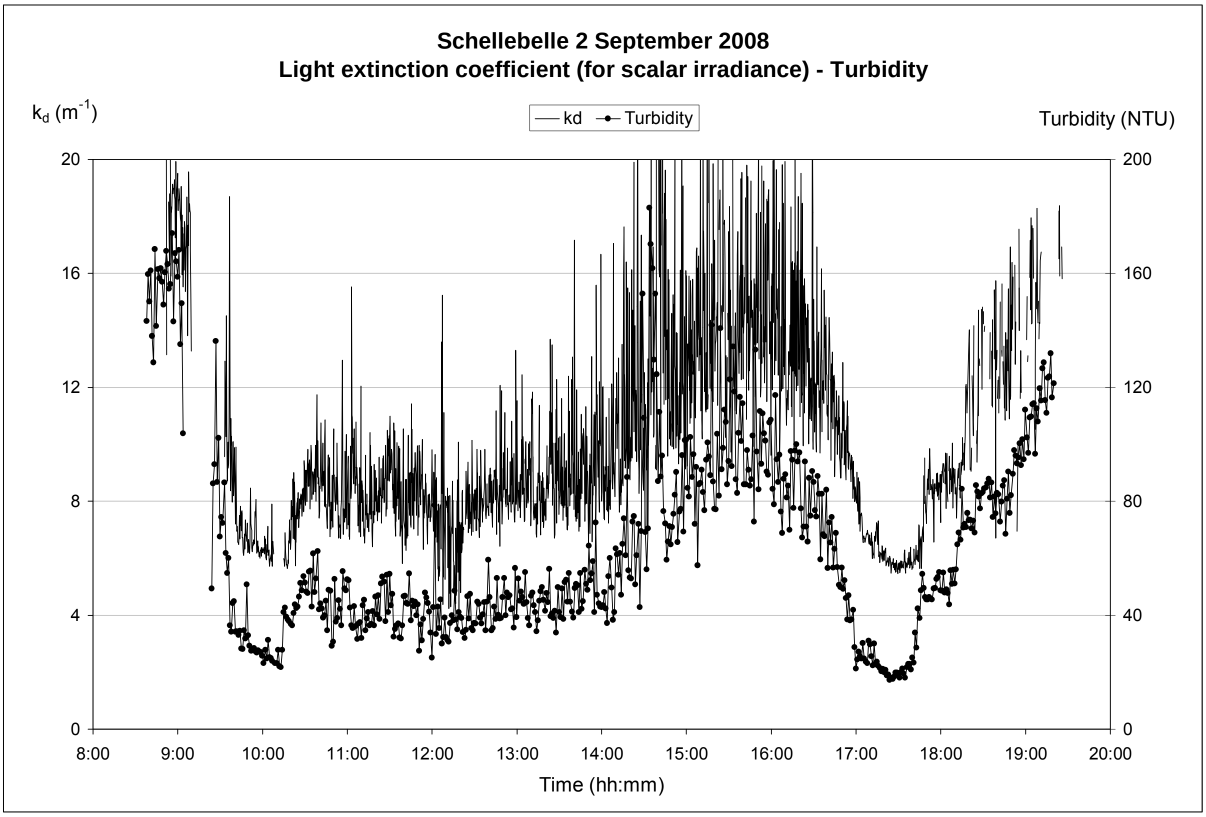

Figure 5

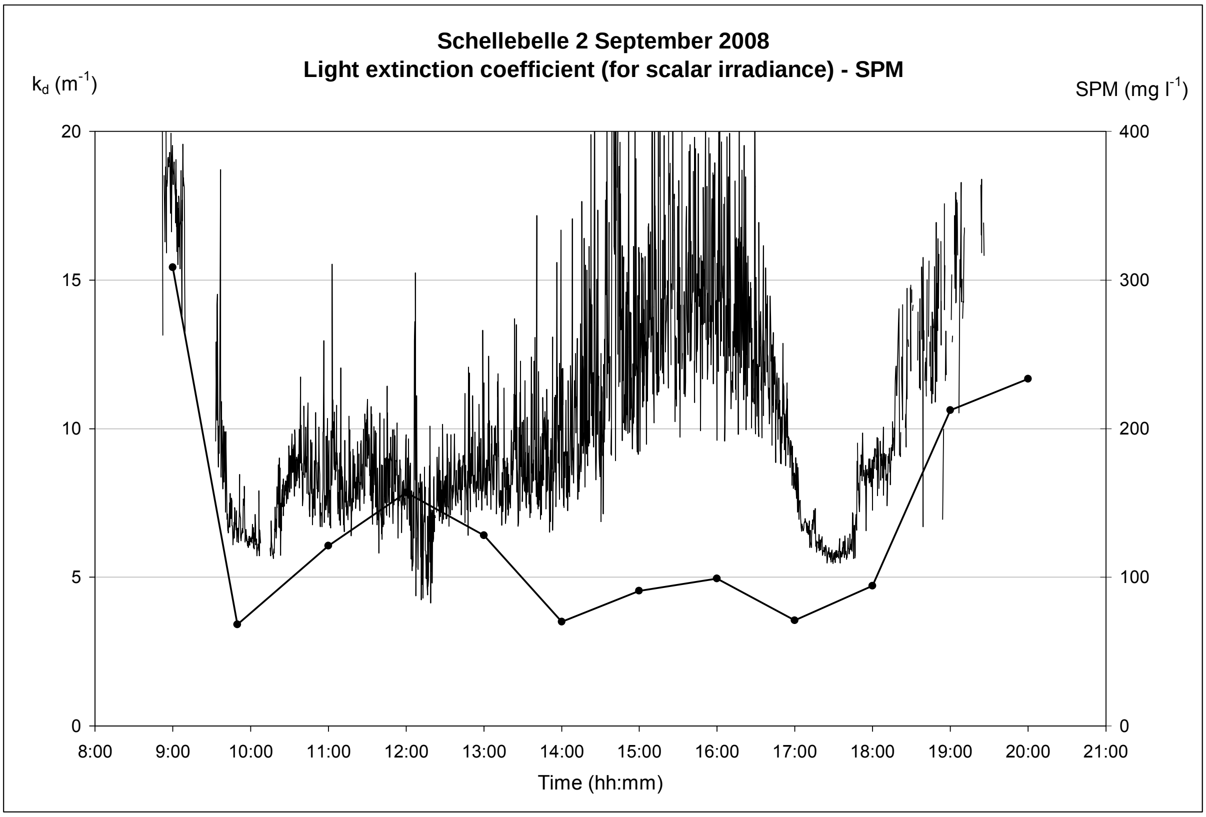

Figure 6

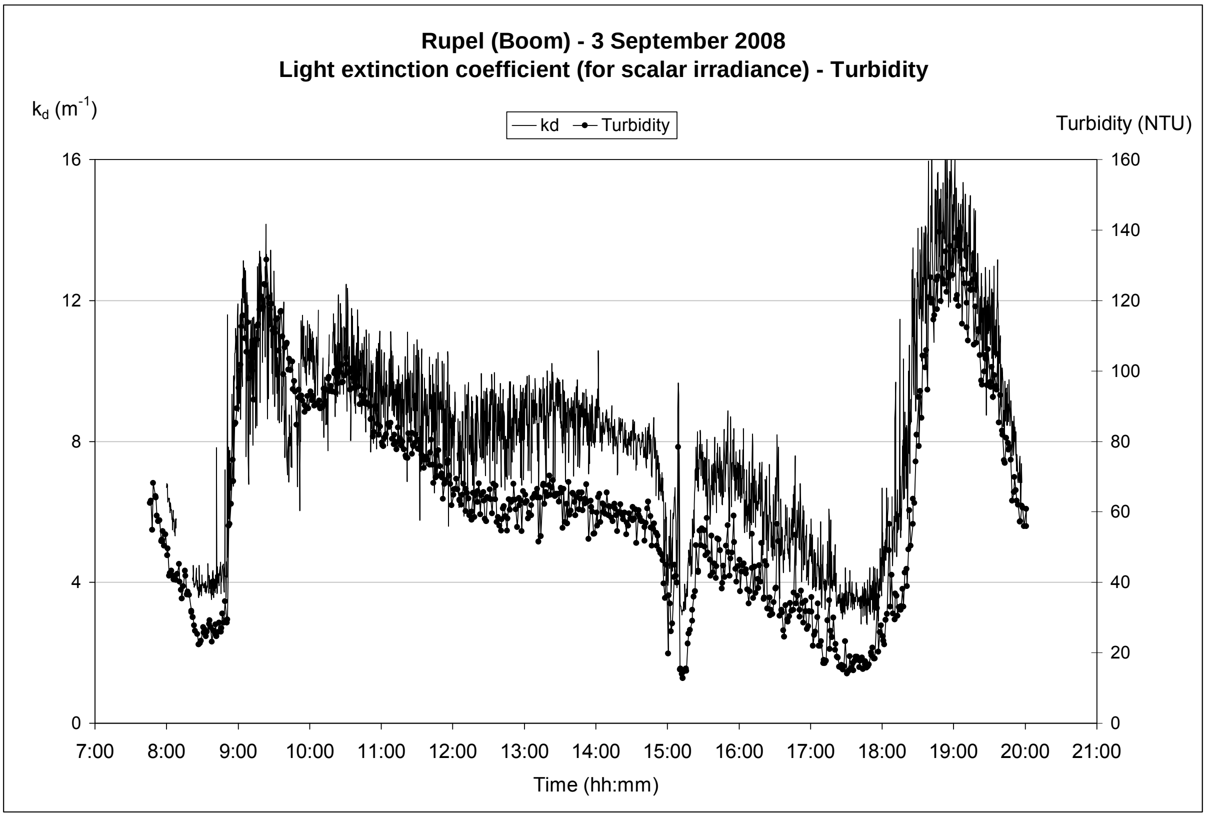

Figure 7

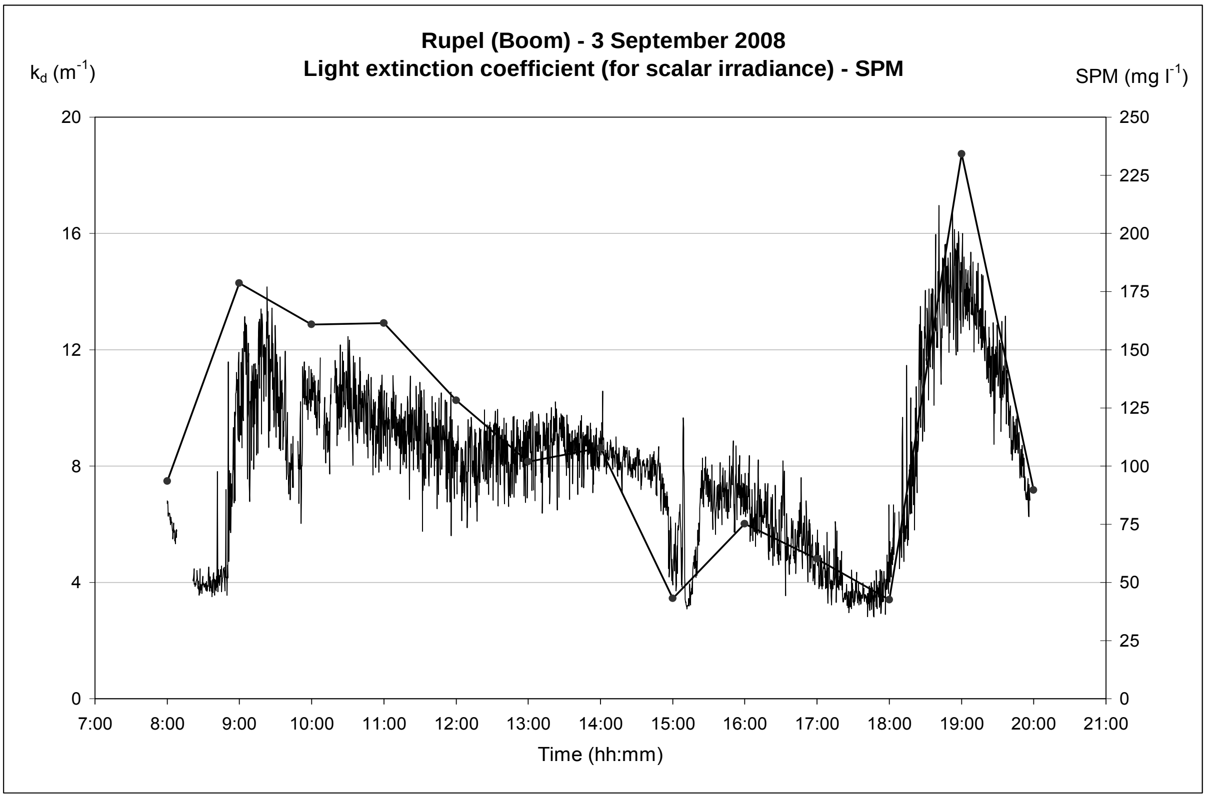

Figure 8

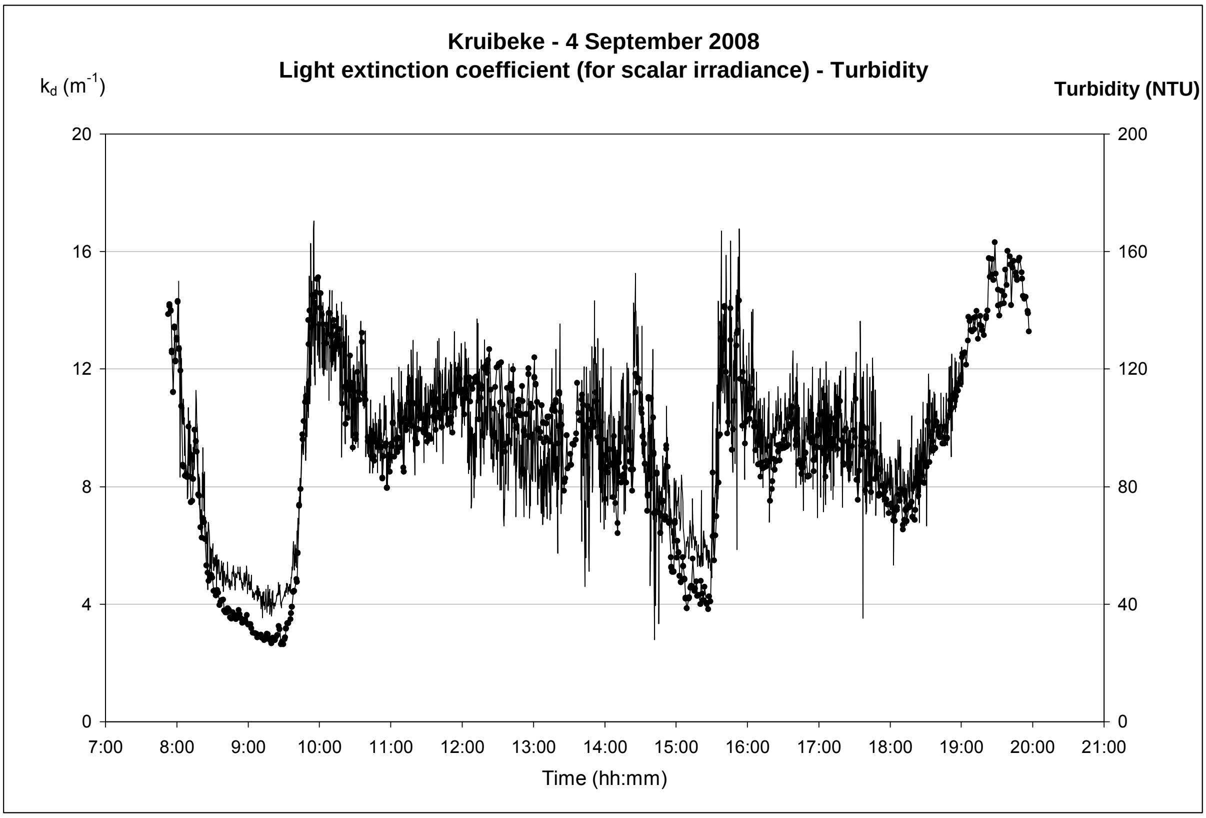

Figure 9

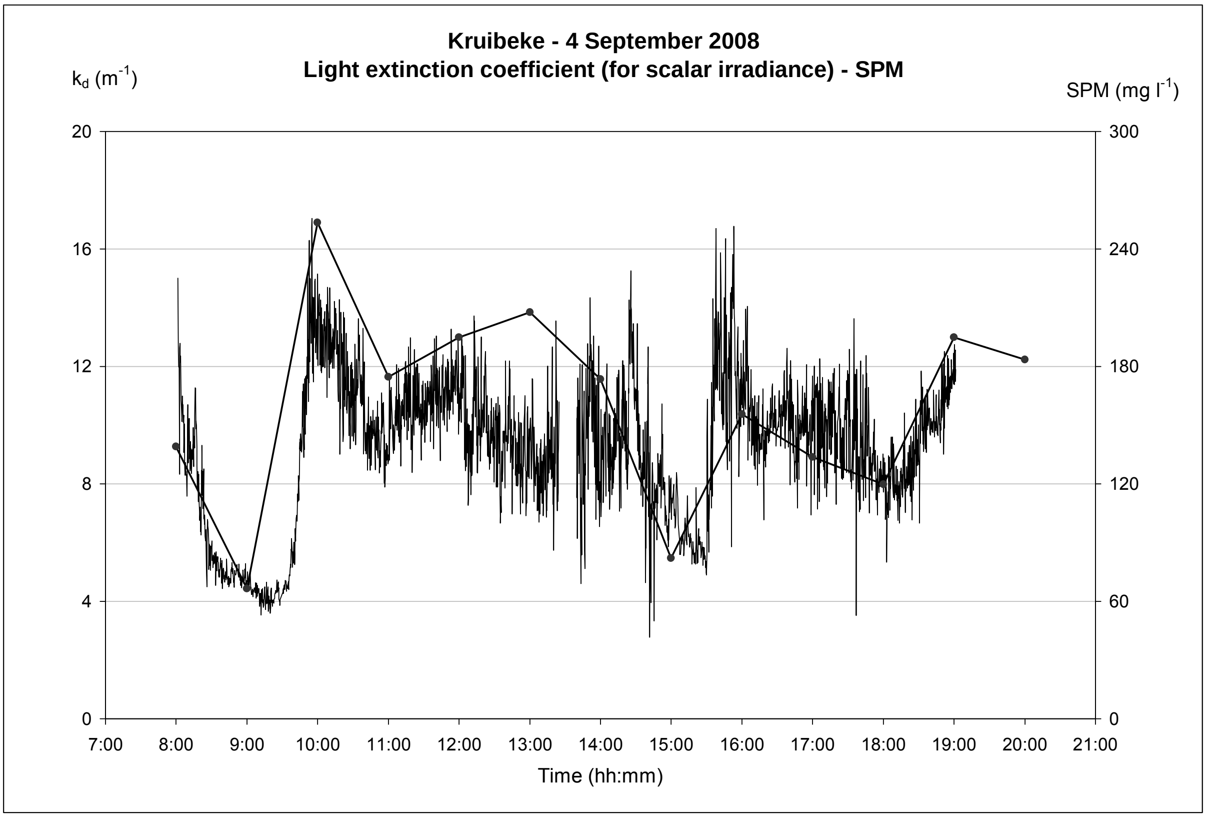

Figure 10

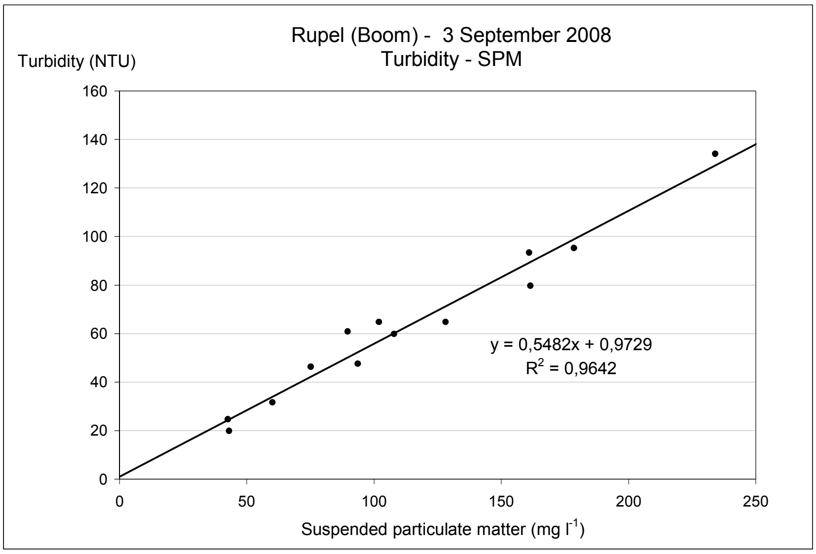

Figure 11: Turbidity – SPM correlation, 13-h cycle in Boom

6.2.4. Longitudinal chlorophyll a distribution

In July, August and September 2008, chlorophyll *a* concentrations larger than 100 µg l^{-1} have been observed at all stations situated upstream of km 120 (Dendermonde). In particular, extremely high values (up to 650 µg l^{-1} in Melle) were measured in July 2008, associated with a strong oxygen oversaturation (> 130 % in Melle). During these three summer months, a chlorophyll *a* maximum was observed at (or near) Melle (km 51), followed by a decrease in chlorophyll *a* concentration in the seaward direction. Phytoplankton blooms were thus able to develop close to the upward limit of the tidal river, due to the low freshwater input from the upper Scheldt and to the associated increase of the water residence time during this period (July, 2nd decade: 18.2, with a minimum at 0.8 on the 21st ; August, 2nd decade: 20.5; September, 1st decade: 18.1; all flows at Melle in m^3 s^{-1}). On the contrary, in spring (May- June) and early autumn (October), an increasing chlorophyll *a* concentration was observed downstream of Melle with a maximum concentration below Dendermonde, indicating that phytoplankton growth was active along the estuarine gradient. It thus reveals that no flushing of the phytoplankton population occurred at the intermediate values of freshwater discharge typically observed during these two periods, and that nutrient (silica) limitation was not reached in the upper tidal river.

Chlorophyll *a* concentrations measured in spring 2009 indicate a strong phytoplankton growth in the upstream reach of the estuary (between Melle and Uitbergen), already starting in May but particularly intense in June, with a maximum concentration above 400 µg l^{-1}. This has been accompanied by high oxygen oversaturation (120 % at Uitbergen in June 2009). On the other hand, a very intense phytoplankton bloom was also observed in May 2009, for the first time since the beginning of our survey, between km 71 and km 58: chlorophyll *a* concentrations exceeding 100 µg l^{-1} and oxygen oversaturation close to 130 % were recorded. This is a clear indication that a shift from heterotrophy towards autotrophy is presently taking place within the lower estuary between Antwerpen and the Belgian-Dutch border.

6.2.5. Photosynthetic parameters

The photosynthetic efficiency α and, whenever possible, the maximum specific rate of photosynthesis P^B_{max} have been determined between July 2008 and June 2008 at 9 stations along the estuarine gradient and 3 stations in the Rupel system. The detailed dataset, including all experimental PE curves, is available digitally available at VLIZ.

In 2008, α values varied between 0.003 and 0.053 (expressed in µgC.m^2.s/µgChl.h.µE), with about 72 % of the values comprised between 0.04 and 0.016. The observed distribution is almost identical to the distribution observed in 2007. It is shown in Figure 12, and the complete dataset for 2008 (April to October) is plotted in Figure 13. In the freshwater part of the estuary (km 90 to 151), α values show limited longitudinal and seasonal variations, the higher values (between 0.015 and 0.020) being reached in August (and September near Temse). For all other months, α values remained lower than 0.015. Values higher than 0.020 were observed during summer (July, August and September) in the brackish part of the estuary (km 58 to 90), but remained lower than 0.015 in April, May and October over the whole estuary.

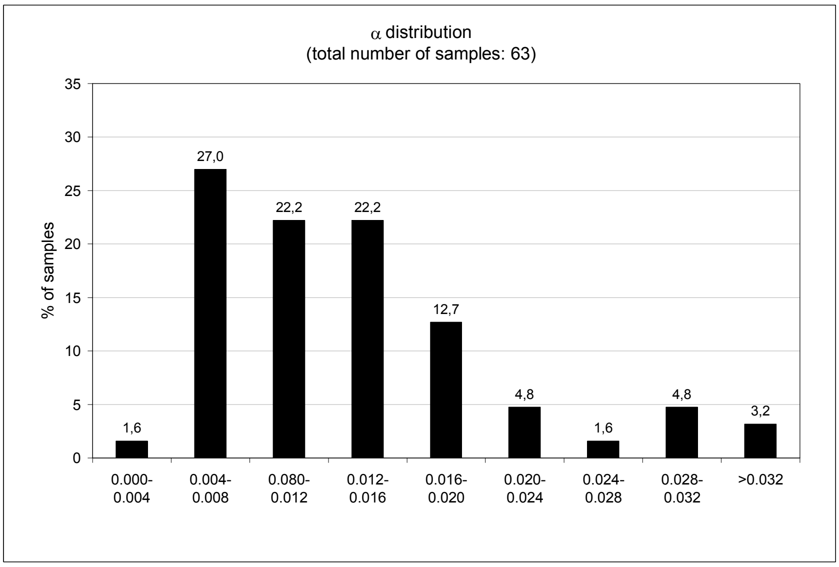

Figure 12: Distribution of α values in 2008 (α expressed in µgC.m².s/µgChl.h.µE)

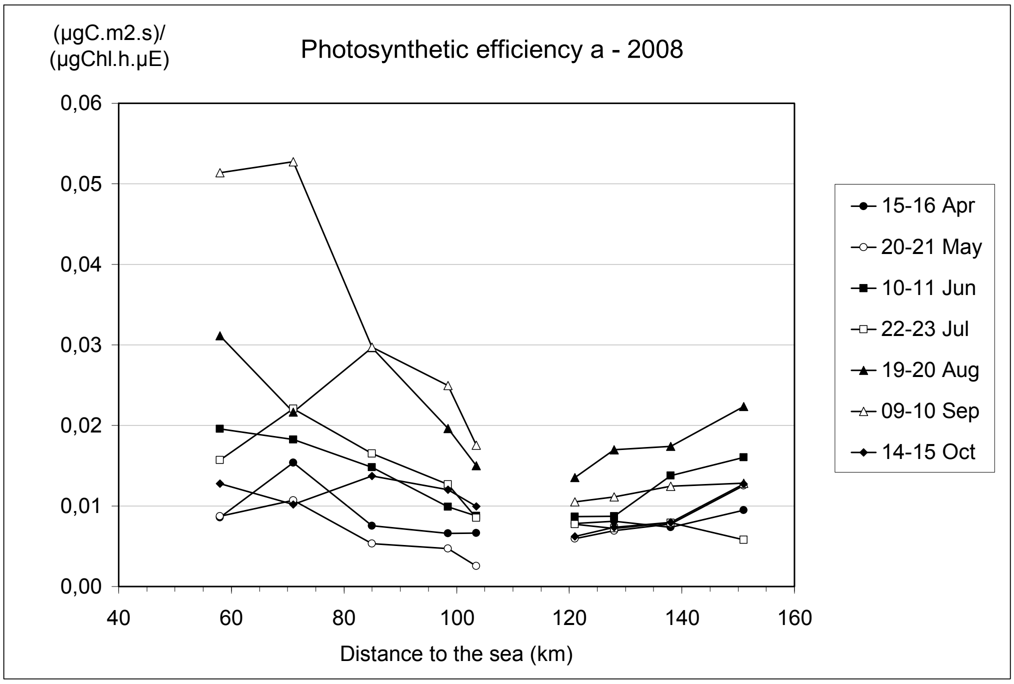

Figure 13: Seasonal evolution of α values for 2008

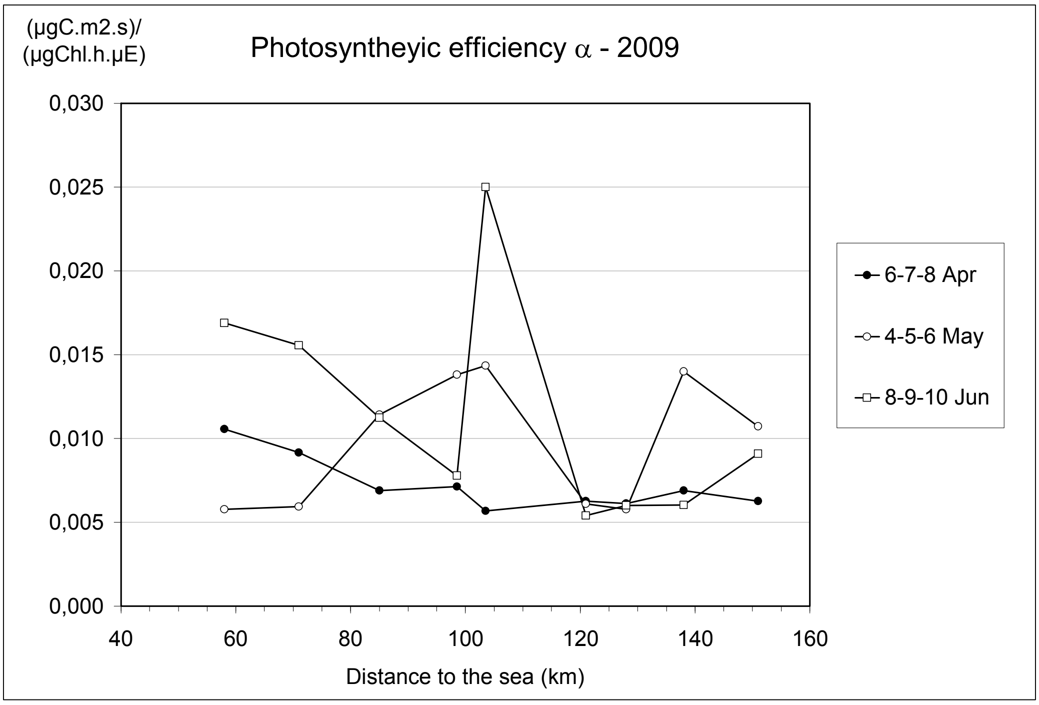

Figure 14: Seasonal evolution of α values for 2009 (part)

The situation is comparable for spring 2009 (April – June), with values under 0.010 in April, up to 0.015 in May and 0.025 in June. The general pattern may therefore be synthesized as follows: the lower values are observed in spring (April – June) and autumn (October) over the whole estuary; higher values are measured in summer (June to September), the highest values (> 0.025) being observed in August and September in the downstream stations, particularly near the border. This is in opposition with the observations for 2008, which showed the occurrence of the highest α values in summer at the most upstream locations.

Figure 15 shows the distribution of the maximum specific rate of photosynthesis in 30 samples taken in 2008. As pointed out previously, a number of experimental P-E curves display a quasi-linear behaviour over the range of irradiance used during laboratory incubations, leading to I_k (saturation irradiance) values unrealistically larger than 2000 µE m^{-2} s^{-1}. In 2008, 33 PE curves (out of 63) displayed a linear behaviour, essentially in April, May, June and October. Light saturation could therefore be achieved during ^{14}C incubation mainly on summer samples (July, Augustus and September).

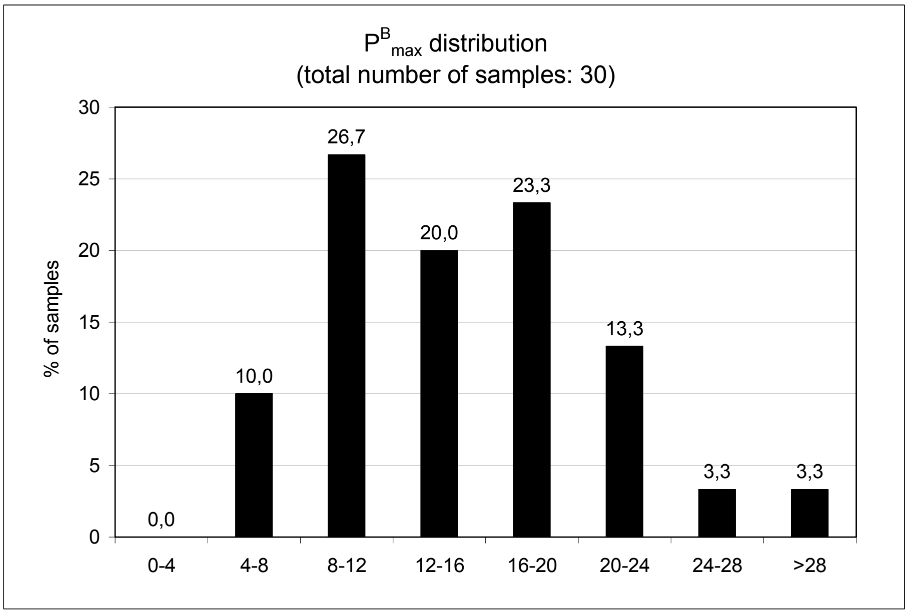

Figure 15: Distribution of P^B_{max} values in 2008 (P^B_{max} expressed in µgC/µgChl a)

Figure 16 shows that a majority of maximum rates are comprised between 10 and 20 µg of carbon per µg chlorophyll a and per hour, with only two samples above 25. No longitudinal or seasonal trend can be inferred from these data.

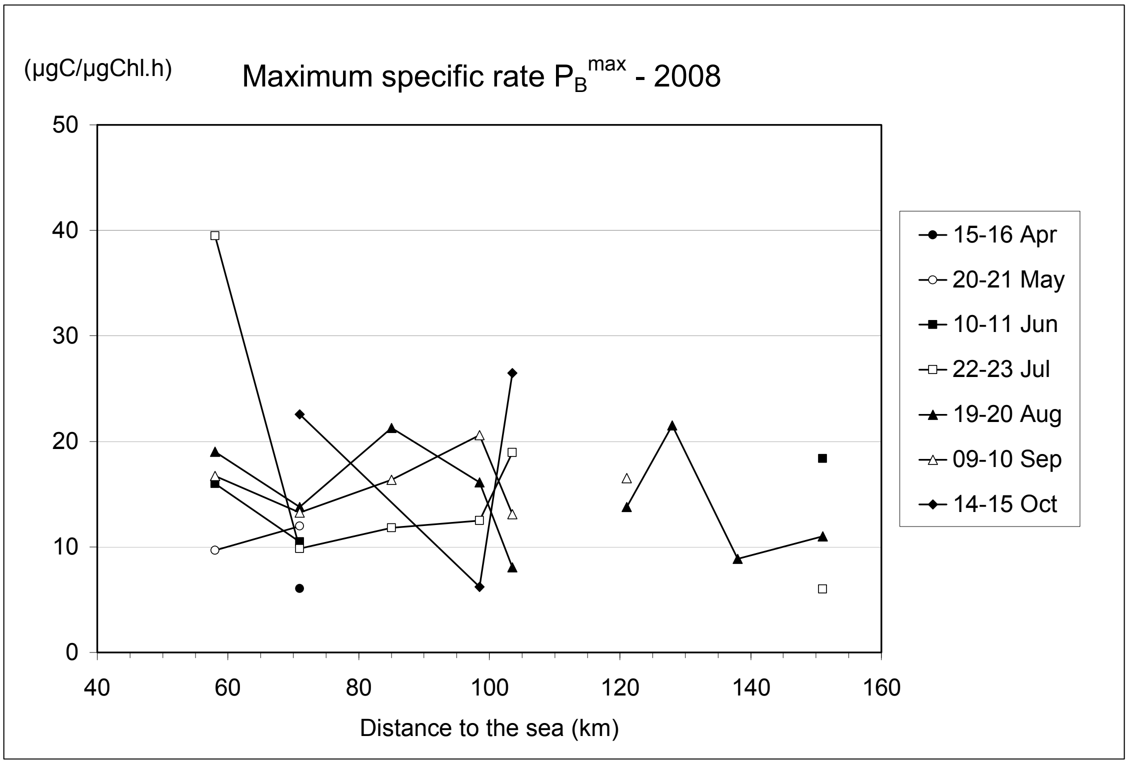

Figure 16: Seasonal evolution of P^B_{max} values for 2008

6.3. Annexes

The following data files (xls files) have been supplied to the coordinator and to VLIZ, the data manager of the Omes project:

Annex A: Parameters
- A.1: Monthly profiles
  - (Contain temperature, conductivity, salinity, oxygen, pH, alkalinity and DIC data)
    - Monthly profiles 2008 – Parameters.xls
    - Monthly profiles 2009 (April-June) – Parameters.xls
- B.1: 13 hour cycles
  - (Contain temperature, conductivity, salinity, oxygen and pH data)
    - 1 - Schellebelle 2008 – Parameters.xls
    - 2 - Rupel 2008 – Parameters.xls
    - 3 - Kruibeke 2008 – Parameters.xls

Annex B: Light climate
- B1: Monthly profiles
  - (Contain detailed incident + underwater light measurements and k_d data)
    - 08-04 (April 2008) – Light.xls
    - 08-05 (May 2008) – Light.xls
    - 08-06 (June 2008) – Light.xls
    - 08-07 (July 2008) – Light.xls
    - 08-08 (Augustus 2008) – Light.xls
    - 08-09 (September 2008) – Light.xls
    - 08-10 (October 2008) – Light.xls
    - 09-04(April 2009) – Light.xls
    - 09-05 (May 2009) – Light.xls
    - 09-06 (June 2009) – Light.xls
  - (Contain a synthesis of k_d euphotic depth data)
    - Monthly profiles 2008 – Light climate.xls
    - Monthly profiles 2009 (April-June) – Light climate.xls
  - (Contain kd, turbidity and SPM data)
    - Monthly profiles 2008 – SPM Turbidity.xls
    - Monthly profiles 2009 (April-June) – SPM turbidity.xls
- B.2: 13 hour cycles
  - (Contain detailed incident + underwater light measurement, k_d, SPM and turbidity data)
    - 1 - Schellebelle 2008 – Kd.xls
    - 2 - Rupel 2008 – Kd.xls
    - 3 - Kruibeke 2008 – Kd.xls
- B.3: All data
  - (Contain a synthesis of k_d and SPM data)
    - kd – SPM – all data – 2008.xls
    - kd – SPM – all data – 2009.xls
    - kd – SPM – all data – 2002-2009.xls

Annex C: Primary productivity

- C.1: Monthly profiles
  - (Contain detailed PE curves, α, P_b^{max} and I_k data)
    - 08-04 (April 2008) – PP.xls
    - 08-05 (May 2008) – PP.xls
    - 08-06 (June 2008) – PP.xls
    - 08-07 (July 2008) – PP.xls
    - 08-08 (Augustus 2008) – PP.xls
    - 08-09 (September 2008) – PP.xls
    - 08-10 (October 2008) – PP.xls
    - 09-04 (April 2009) – PP.xls
    - 09-05 (May 2009) – PP.xls
    - 09-06 (June 2009) – PP.xls
  - (contain a synthesis of α and P_b^{max} data)
    - Monthly profiles 2008 – Photosynthetic parameters.xls
    - Monthly profiles 2009 (April-June) – Photosynthetic parameters.xls
  - (contains chlorophyll a and phaeopigment data)
    - Monthly profiles 2008 – Pigments.xls
    - Monthly profiles 2009 (April-June) – Pigments.xls
- C.2: 13 hour cycles
  - (contain chlorophyll a and phaeopigment data)
    - 1 - Schellebelle 2008 – Pigments.xls
    - 2 - Rupel 2008 – Pigments.xls
    - 3 - Kruibeke 2008 – Pigments.xls

Hoofdstuk 7. Micro- en mesozoöplankton

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Eindverslag voor deelstudie 6 (perceel 6), periode juli 2008 - augustus 2009

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Samenvatting

De maandelijkse monitoring van micro – en mesozooplankton werd in 2008 - 2009 voortgezet. Vanaf januari 2009 werd de staalname uitgebreid tot 15 stations, in overeenkomst met de MONEOS doelstellingen. In het zomerhalfjaar verhoogde de staalnamefrequentie van maandelijks naar tweewekelijks.

De opvolging van de gemiddelde abundanties voor copepoden en cladoceren in het voorjaar toont aan dat deze in het zoetwatertransect van de Zeeschelde consistent hoger blijven dan voor de OMES periode. De abundantie van cladoceren in het brakwatertraject neemt licht toe, deze van copepoden daarentegen neemt de laatste jaren licht af. Voor de typisch estuariene copeode *Eurytemora affinis* hebben we kunnen aantonen welke omgevingsfactoren zijn al dan niet voorkomen in het stroomopwaarts traject van de Zeeschelde verklaren. Aan de hand van OPLS analyse is gebleken dat niet alleen de zuurstofconcentratie stroomopwaarts een drempelwaarde (3 mg l^{-1}) moet overschrijden, maar dat ook de minimum zuurstofconcentratie in het middengebied tenminste 1.3 mg l^{-1} met bedragen opdat de stroomafwaartse populatie stroomopwaarts zou kunnen migreren. Gezien deze zuurstofbarrière mogelijk ook voor andere organismen (mysidacëen, vissen) geldt, lijkt het aangewezen de nodige aandacht te besteden aan maatregelen om de zuurstofconcentratie in het middengebied te bevorderen.

Summary

During 2008, monthly sampling for micro – and mesozooplankton was continued. From January 2009 onwards, sampling campaigns were carried at 15 stations, in concordance to the MONEOS stipulations. In summer, the sampling frequency was increased from once to twice a month.

Using the 1996-2007 database on the spatio-temporal distribution of *Eurytemora affinis*, a typical estuarine copepod, we have analysed which environmental conditions explain its recent occurrence in the upstream part of the Schelde. Using PLS analysis, it is shown by that its presence upstream is conditioned by a sufficient oxygen concentration (> 4 mg l^{-1}) that is associated with temperature. However, not only the environmental conditions in the upstream zone are important: the frequent presence of an O2 minimum zone in the mid estuary (O2 min < 1.3 mg l^{-1}) seems to block the transition of the downstream *E. affinis* population in upstream direction. Occasionally, the bulk of the population is however situated upstream. During these periods, high E. affinis abundance is also observed in the Durme tributary. Our findings suggest

the possibility to use *E. affinis* as an "indicator" species of water quality, but also lead us to stress the necessity to consider conditions over the entire estuary, not exclusively in the zone of interest. So the existence of a low oxygen concentration in the middle part of the Zeeschelde and the possibilities to improve oxygen concentration in this area should be a prior concern for the Zeeschelde research and management.

Aknowledgements
With thank Tom Maris and Dimitri Van Pelt for the sampling and logistic support. P Meire and S. Van Damme put the OMES environmental data-set at our disposal.

7.1. Introduction

The research carried out on micro- and mesozooplankton in the frame of the OMES project aims at following the evolution of the zooplankton composition and its spatio – temporal distribution in the Schelde estuary. Because of its central position, as a link between primary production and higher food levels, zooplankton is an essential component in the functioning of any aquatic ecosystem. Its drifting with the currents makes it also a good indicator of spatio – temporal variation in hydrological and concurrent water quality conditions.

At present disposing of a 11 year database on zooplankton in the brackishwater-freshwater stretch of the Schelde, we are now able to analyse the response of the zooplankton community to the restoration of the Schelde estuary. This report considers on the evolution of the abundance of the spring populations of the main groups of mesozooplankton (calanoid and cyclopoid copepods and cladocerans). It pays special attention to the recent more frequent occurrence of *Eurytemora affinis*, a typical estuarine copepod, in the freshwater upstream part of the estuary.

7.2. Material and Methods

7.2.1. Sampling microzooplankton

Samples are taken by the ECOBE laboratory of the University of Antwerp (UA). Transport to the 'Laboratoire d'Ecologie Fonctionelle (EcoLab) of the University of Toulouse III is done by courier or by EcoLab personnel.

10 OMES–monitoring stations, 6 in de Schelde (Boei 87, Antwerpen, Temse, Dendermonde, Uitbergen, Melle) and the 4 tributaries (Bovenschelde, Durme, Dender and Rupel) are sampled. From January 2009 onwards, sampling campaigns were performed at 15 stations. (Boei 87, km57), Antwerpen, (km 78,5), Kruibeke (km 88), Temse (km 97,5) Lippenbroek (km 11), Dendermonde (km 121,5), Uitbergen (km 140), Melle (km 155), 3 stations in the Rupel basin: Rupel (Boom), Beneden Nete en Dijle (estuarien) and 4 boundaries Bovenschelde, Dender, Durme en Haven (Zandvlietsluis).

For microzooplankton, a 2 litres sub-sample of the < 50 µm filtrate available from sampling for mesozooplankton (see further) is used. The 2 litres sample is filtered through 20 µm, and the retained microzooplankton anaesthetized with a carbonated solution and finally fixed in 4% formalin.

7.2.2. Analysis microzooplankton

Determination is carried out at species level as far as possible. The analysis of the microzooplankton fraction is limited to Rotifera. The main references used are: Ruttner-Kolisko 1972, Pontin 1978, Pourriot et al. 1986, Segers 1995. Abundance is counted under binocular (90x), and converted to number of organisms m^{-3}. Details of the procedure are given in Tackx et al. 2004b.

7.2.3. Sampling and analysis mesozooplankton

Sampling stations for mesozooplankton are the same as those for microzooplankton. 50 litres water is taken with a bucket at surface and filtered through a 50 µm net. The collected mesozooplankton is processed in the same way as described above for the microzooplankton samples.

Determination is carried out at species level when possible. For copepods this is possible from Copepodits V onwards. Most important references used are: Dussart 1967, 1969; Kiefer 1978; Amoros 1984; Margaritora 1985, Einsle 1996, Karaytug 1999, Ueda 2003.

7.2.4. Analysis of the long – term zooplankton data-series in the Zeeschelde

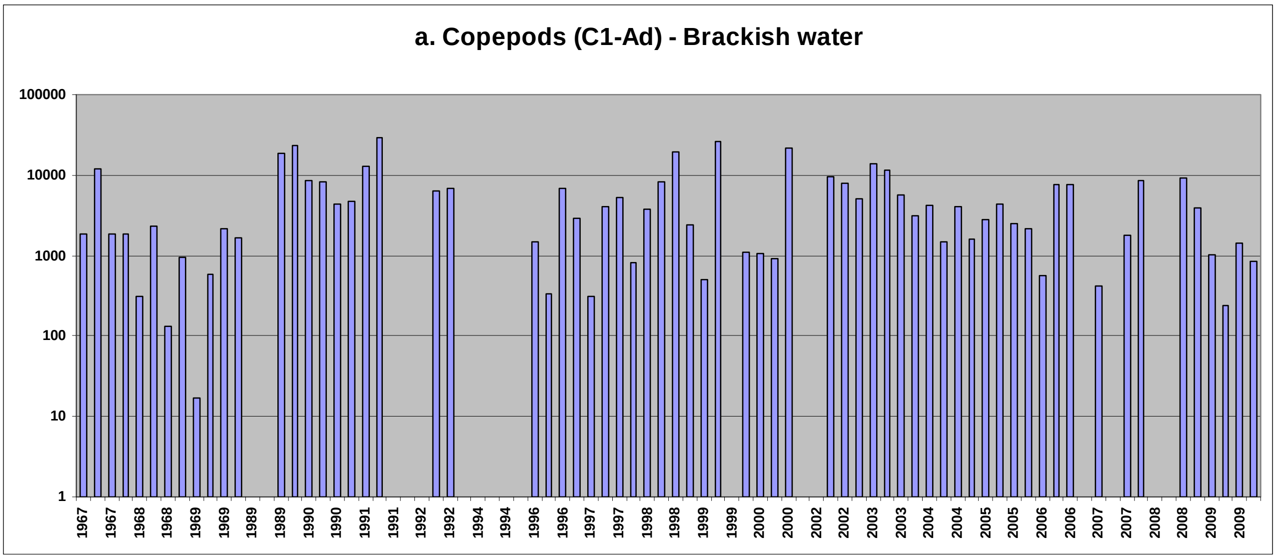
The data-set used covers the zooplankton data for the years 1996 – 2007, with the exception of 2001, when no sampling took place. For the years 1996, '97, '98, all 16 OMES stations have been analysed. For the other years, abundance adapt are only analysed for 6 stations: km 58, 78, 98, 121, 138, 151. The 1996-2009 data were also combined with earlier data on zooplankton (De Pauw, 1973 ; Soetaert & van Rijswijk, 1993) in order to look at longer time trends. Because of the limited availability of earlier data, this could only be done for the spring period (February- may). In this analysis, the brackish water (km 57,5 - km 71,5) and the freshwater (km 78,5 - 155) were considered separately. This reports considers this spring series, an analysis of the total annual series is presently carried out in the frame of a master's research project.

7.3. Results and discussion

The evolution of the spring (February - May) mesozooplankton community for the period 1989-2009 in the brackish (Antwerp – Belgian/Dutch boarder) and the freshwater (Antwerp – Gent) zone is represented in Figs. 1a,b; 2a,b for copepods and cladocerans respectively.

Calanoid abundance in the brackish zone remains fairly stable between 1000 and 10 000 ind. m^{-3} over the entire period (Fig. 1a). We note that abundance in 2009 has rarely reached values > 10 000 ind. m^{-3}.

The increase over an order of magnitude (from 100 to 1000 ind m^{-3}) of the copepod abundance in the freshwater area (Fig. 1b) since 1996 as compared to the 1967-1992 period has been reported previously (Tackx et al., 2005a,b; 2006;2008). The OMES – monitoring presently permits to observe that copepod abundance values of > 1000 ind. m^{-3} are regularly observed in the freshwater reach since 1996. Calanoid abundance frequently reaches values above 1000 individuals m^{-3} since 2002, whereas this was more seldom the case in previous years.



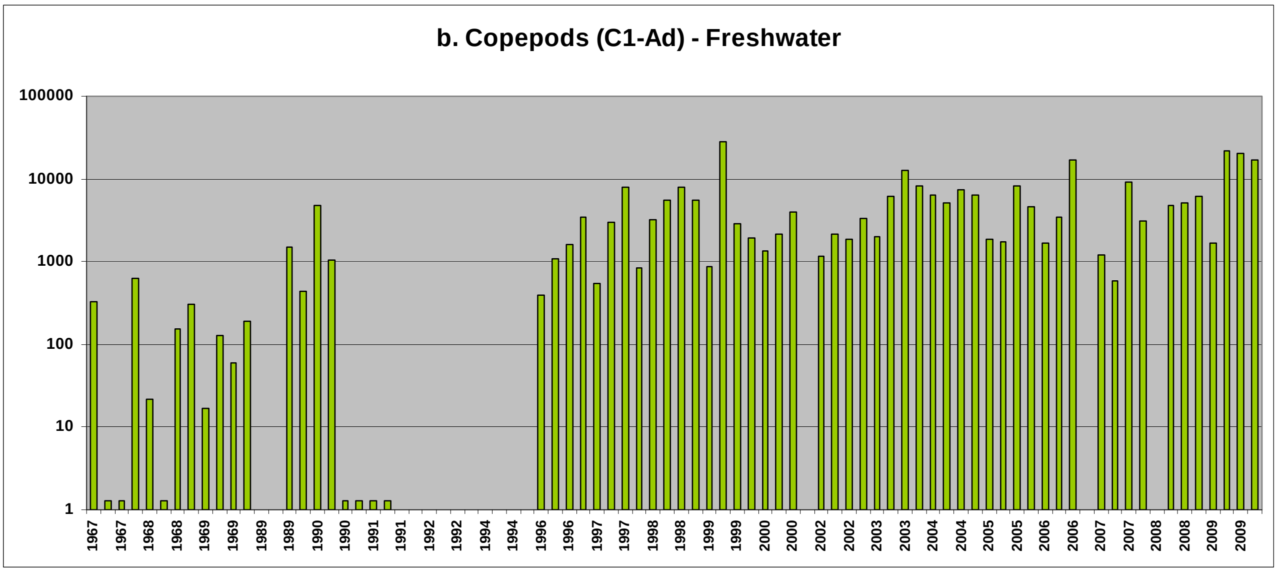


Fig 1. Monthly mean abundance of copepods (CV-adults) in the a) brackish and b) freshwater reach of the Schelde estuary. Only spring months (February- May) are considered.

Cladocerans are little abundant (10-100 ind. m^{-3}) in the brackish water zone (Fig. 2a), which is logic as they are mainly freshwater type organisms. Nevertheless, in recent years abundance has several time exceeded 1000 ind. m^{-3}.

In the freshwater zone, cladocerans (Fig. 2b) have increased by an order of magnitude since 1996 in comparison to previous observations during 1967-1989 (Tackx et al., 2005). Since 1998, they reach abundance values between 100 and 1000 ind. m^{-3} on a regular basis, occasionally exceeding > 10 000 ind. m^{-3}.

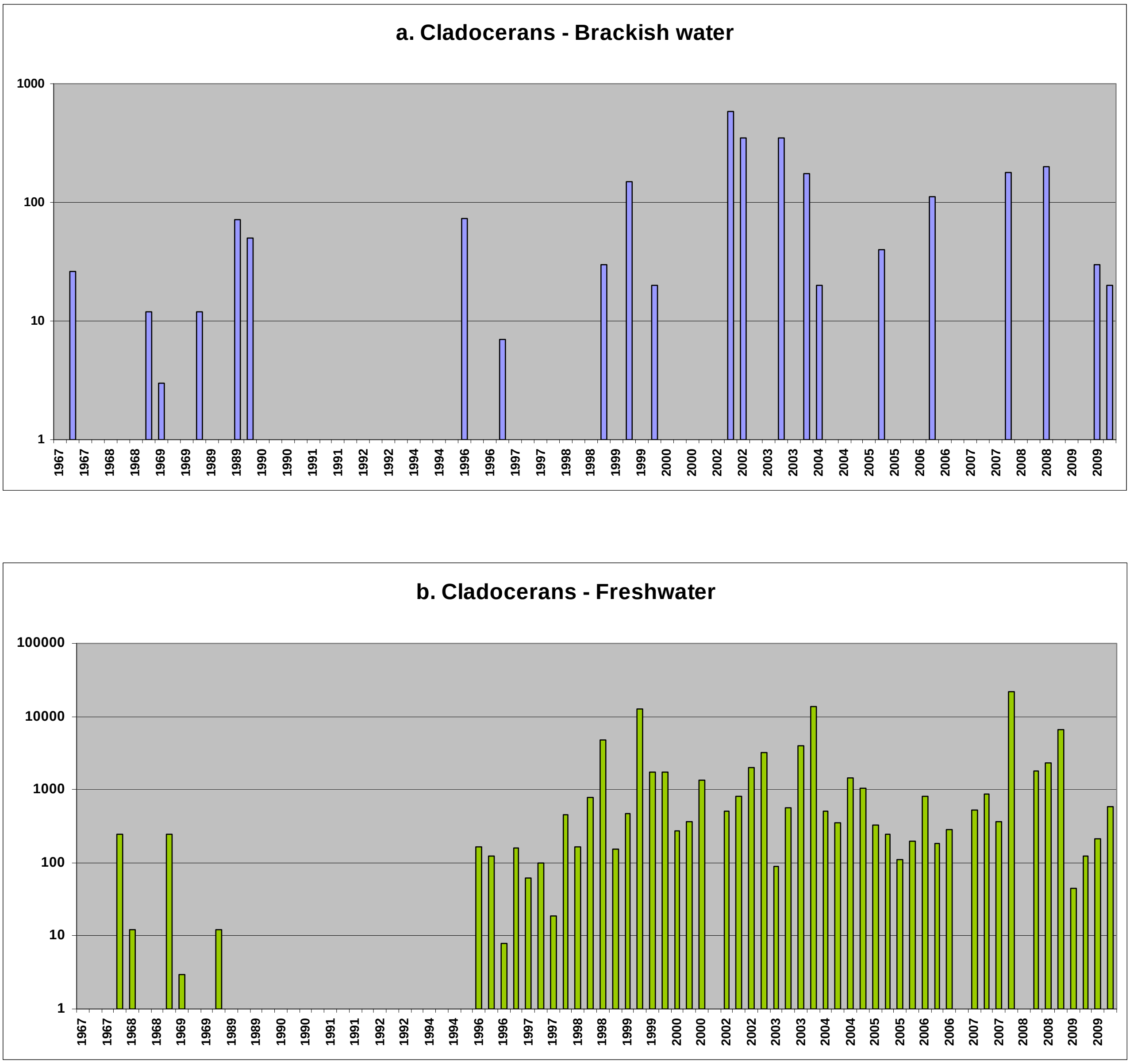
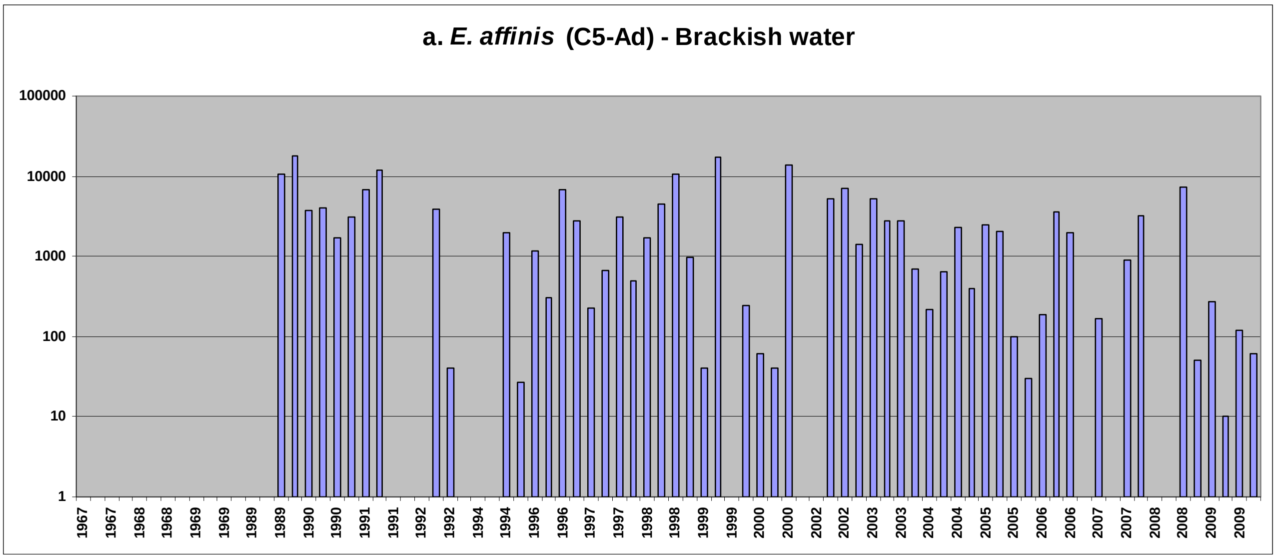


Fig 2. Monthly mean abundance of cladocerans in the a) brackish and b) freshwater reach of the Schelde estuary. Only spring months (February- May) are considered.

The abundance of the estuarine copepod *E. affinis* in the brackish zone shows a tendency for decline in recent years (since around 2005). In the freshwater zone, on the contrary, it has occurred in higher abundance (between 100 and 1000 ind. m^{-3}) since 1996 than before (Appeltans et al., 2003). It is observed at abundances > 10 000 ind. m^{-3} in 2009.



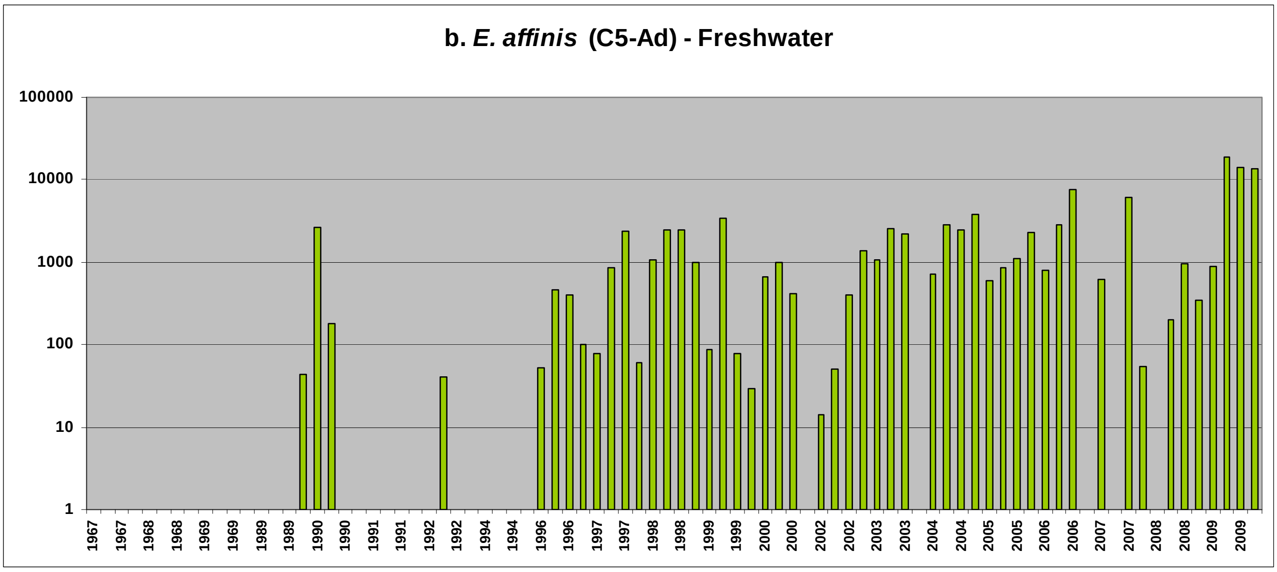


Fig 3. Monthly mean abundance of Eurytemora affinis in the a) brackish and b) freshwater reach of the Schelde estuary. Only spring months (February- May) are considered.

The increase of *E. affinis* in the freshwater area has been analysed in detail and is subject a paper submitted recently to Estuarine and coastal Shelf Science (Mialet et al., in sub.). The occurrence of *E. affinis* in the freshwater zone is irregular in abundance and in distance upstream, such as demonstrated in Fig. 4.

During some months, such as March 1998, *E. affinis* is present and abundant as far upstream as km 134 (Fig. 4a) whereas during other months, such as April 1997, *E. affinis* remains downstream km 106 and is quasi absent upstream from this station (Fig. 4b). Inversely, it sometimes happens that the bulk of the population is located in the freshwater reaches, upstream km 103, such as for example during February and March 2004 (Fig. 4c). This variable distribution could be a response to (local) environmental conditions in the upstream Schelde.

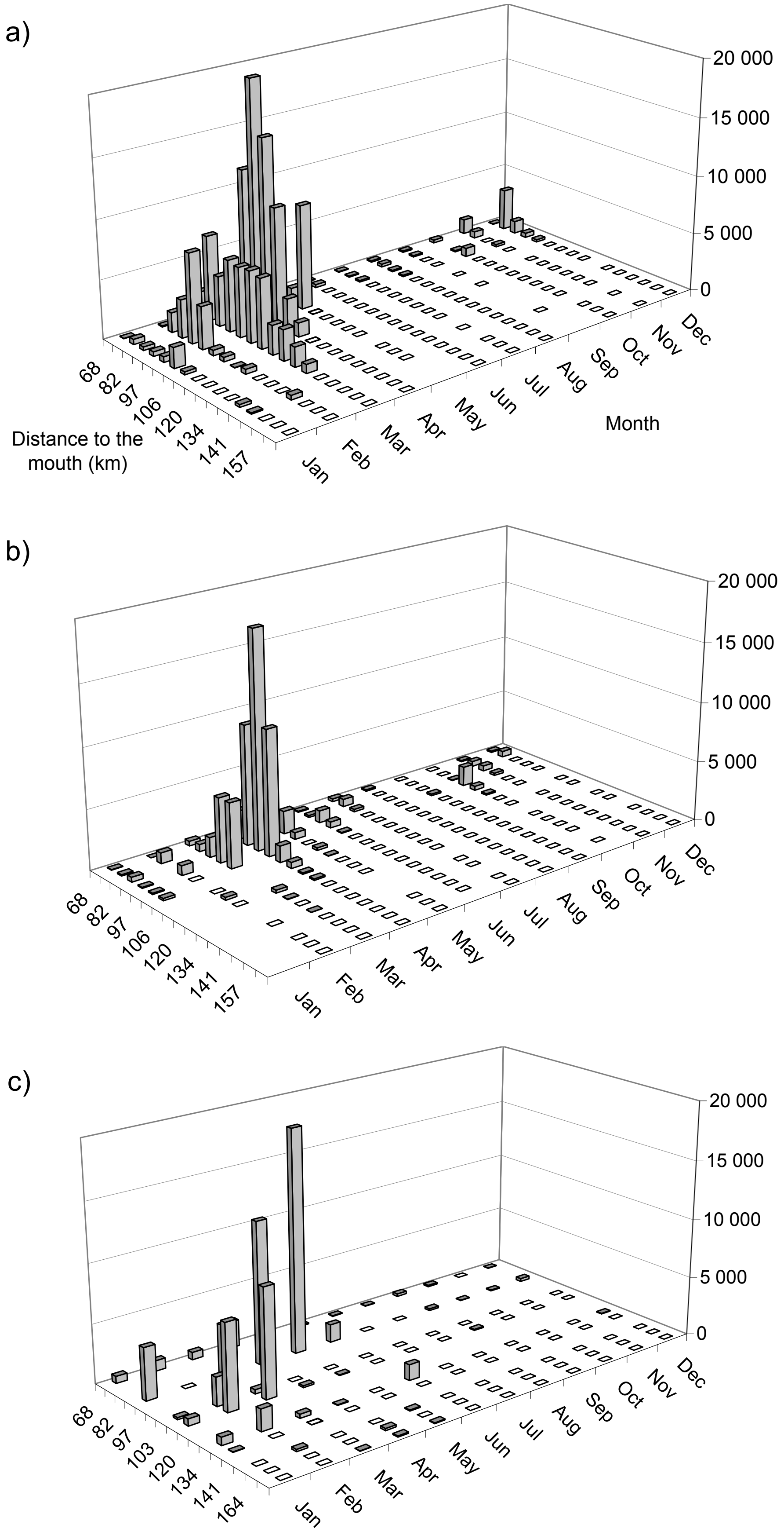


Fig. 4: Examples of various spatial distributions of Eurytemora affinis adults in the Scheldt estuary, during 1998 (a), 1997 (b) and 2004 (c). White squares mean null values.

We have therefore tried to identify the environmental variables that best explain the upstream distribution of *E. affinis*, based on predictors importance and regressions coefficients (Hoskuldsson, 1988).
An important element in understanding the spatial distribution is the observation that there is a zone in which oxygen concentration is always low (Fig 5). This zone fluctuates in extension and positioning,but is always positioned somewhere in between situated in between km 82 and km 110.

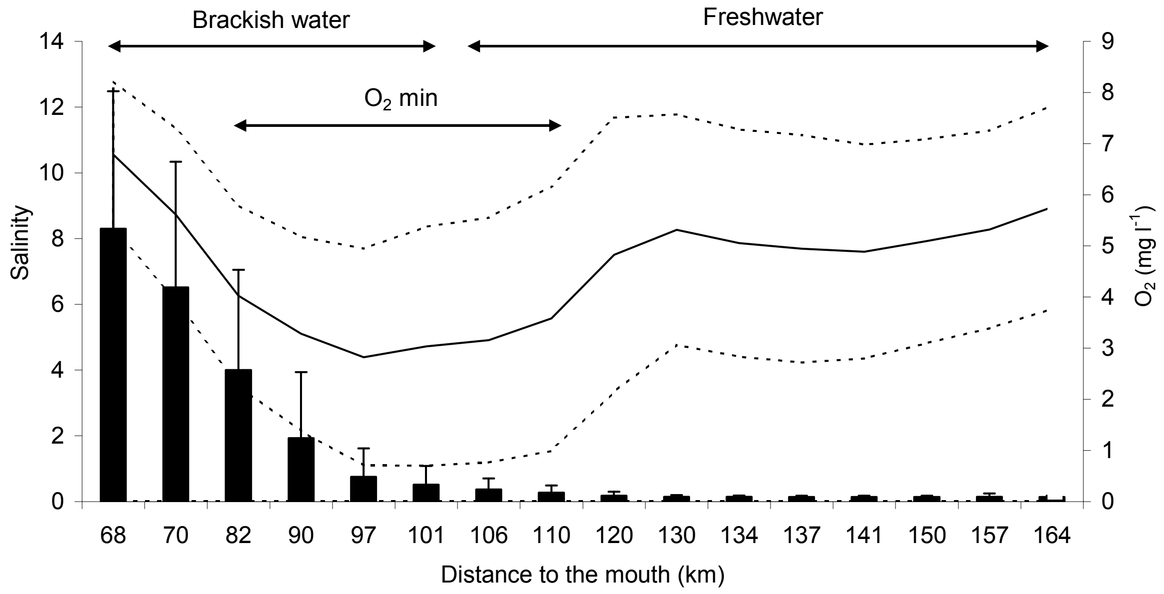


Fig. 5. Spring spatial distribution of mean salinity (black bars) and mean oxygen concentration (black line, dotted lines for standard deviation) in the Scheldt estuary over the period 1996-2007. The locations of the minimal oxygen concentrations observed for all samplings are situated in the "O2 min" range (data ECOBE).

In order to study to understand the occurrence of *E. affinis* 'upstream', we have used the station at which oxygen concentration is minimum (O2 min) to separate 'upstream and downstream'. So this limit is not a fixed station, but varies in positioning with the oxygen concentration pattern in this zone between km 82 and km 110.

We considered two ways to characterise the distribution of *E. affinis* in the estuary. Firstly we quantified it simply by its upstream (cfr below) abundance. Secondly, we characterized its relative abundance in the upstream part using an « Upstream/Downstream Homogeneity index » for every sampling date, using the following formula:

$$UDH = 1 - (|D - U|) / (D + U)$$

D: Downstream mean *Eurytemora affinis* abundance
U: Upstream mean *Eurytemora affinis* abundance

The effect of various environmental factors on UDH was tested by partial least square (PLS) analysis. The results are shown in Table 1.

Table 1. Partial least squares regression results, using the station where the lowest dissolved oxygen concentration was measured as upstream/downstream fringe. Significant results are figured in bold and marked with an asterisk. O2 min: lowest dissolved oxygen concentration measured in the estuary, O2: upstream mean dissolved oxygen concentration, Kj-N: upstream Kj-N mean concentration, tot P: upstream tot P mean concentration, CHL a: upstream mean Chl a concentration, SPM: upstream mean SPM concentration, T: upstream mean temperature, Q: run off at km 68. See text for explanation.

Dependant variables		*E. affinis* Upstr.	UDH	O_2	Kj-N
Predictors importances	O_2 min	**1.37 ***	**1.41 ***	-	1.12
	O_2	**1.49 ***	**1.37 ***	-	1.06
	Kj-N	**0.97 ***	0.48	**1.08 ***	-
	tot P	0.94	0.42	**1.36 ***	**2.09 ***
	CHL *a*	0.08	0.01	0.12	0.37
	SPM	1.28	**1.30 ***	0.83	0.05
	T	0.62	**1.38 ***	**1.38 ***	0.27
	Q	0.14	0.37	0.64	0.20
Predictors coefficients	O_2 min	**+ 0.18 ***	**+ 0.18 ***	-	- 0.19
	O_2	**+ 0.19 ***	**+ 0.18 ***	-	- 0.18
	Kj-N	**- 0.13 ***	- 0.06	**- 0.23 ***	-
	tot P	- 0.12	- 0.06	**- 0.30 ***	**+ 0.36 ***
	CHL *a*	- 0.01	0	- 0.03	- 0.06
	SPM	+ 0.17	**+ 0.17 ***	+ 0.18	- 0.01
	T	- 0.08	**- 0.18 ***	**- 0.30 ***	- 0.05
	Q	- 0.02	- 0.05	+ 0.14	- 0.03
R^2Y		0.38	0.39	0.41	0.47
Q^2		0.33	0.32	0.24	0.33

R^2Y indexes are rather good in all analyses. O2 min and O2 are the most important and significant factors explaining upstream abundance and UDH, with a positive influence. SPM is significant in explaining UDH, with a positive influence. Kj-N also negatively influences upstream mean abundance and T negatively influences UDH. Contrarily to upstream mean abundance, UDH is better explained by O2 min than by O2. O2 is negatively influenced by Kj-N, tot P and T (Table 1). Kj-N is strongly related to tot P but not to T, which therefore has an independent impact on O2 from Kj-N. Given that the Kj-N importance coefficient is superior to that for the tot P in explaining upstream abundance of *E. affinis* and UDH, we can consider Kj-N concentration to represent both the Kj-N and tot P effect on *E. affinis* upstream – downstream distribution. O2 or O2 min importance are higher than that of Kj-N, tot P and T in explaining UDH and upstream abundance. Therefore, oxygen has its own independent effect on the distribution of *E. affinis*.

In conclusion, O2 min, O2, Kj-N, SPM and T seem to be the most likely factors governing the upstream-downstream distribution of *E. affinis* in the Scheldt estuary. To visualise the combined effect of upstream O2 concentration and the potential O_2 min barrier, we plotted UDH values in a O2 min and O2 biplot (Fig. 6). We divided UDH values in 2 groups with a k-mean analysis to separate higher and lower values.

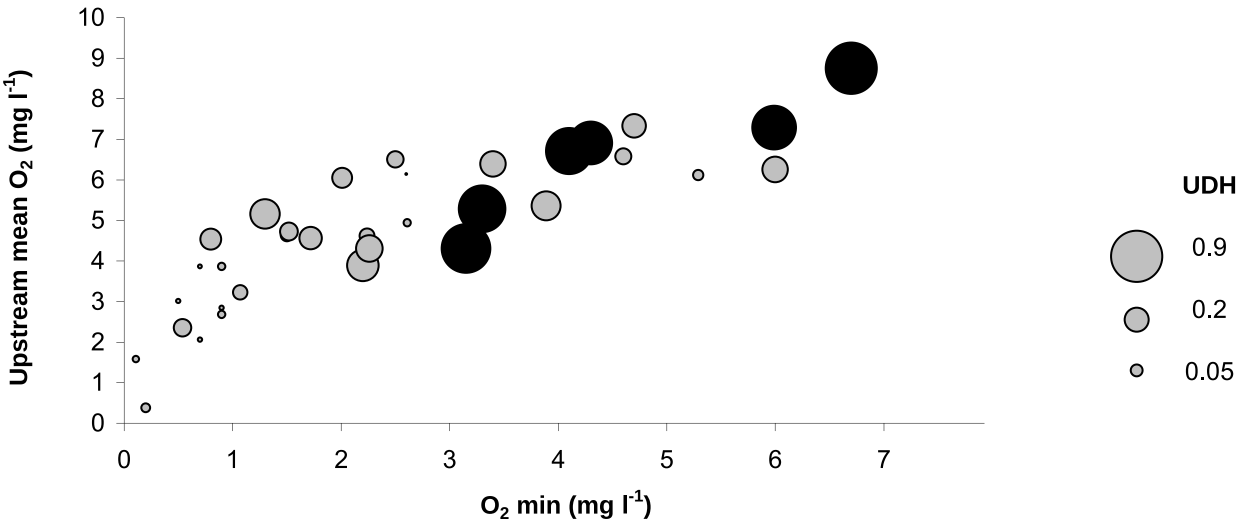


Fig. 6. Relation between UDH, O2 min and O2. Two UDH groups (high values in black, lower values in grey) were divided by a k-mean analysis.

A set of very high UDH values (Fig. 5) are observed when O2 min is superior to 3 mg l^{-1}. If we consider upstream O2, we find these high UDH values above 4 mg l^{-1}. Nevertheless, at these oxygen concentrations, several low homogeneity values are observed as well, mainly (8 cases out of 14) in the area corresponding to O2 min < 3 mg l^{-1}.In the zone corresponding to upstream O2 < 4 mg l^{-1} and O2 min <1.3 mg l^{-1}, only very low UDH values are observed. This figure also illustrates that there is a very clear relation ($p < 10^{-13}$) between upstream mean O2 and O2 min.

7.4. Conclusion and perspectives

Spring mean abundance of copepods and cladocerans continues to show higher values in the freshwater part of the Schelde, as compared to pre-OMES data. The research in progress will allow to consider mean population abundance over the entire year, considering the 1996-2009 period. As done in this report for *E. affinis*, we will also be able to analyse which environmental variables are most strongly explaining temporal and/or spatial variability within the main groups of zooplankton.

The case study of *E. affinis* has permitted us to show the importance of oxygen concentration in assuring living conditions for this organism. An important result is the fact that our findings show that not only the oxygen concentration in the zone considered (so in our case the upstream Zeeschelde) is important, but that the oxygen concentration downstream can act as a block to the upstream migration of *E. affinis*. It is likely that similar features define the living conditions for other organisms (mysids, fish) in the upstream Zeeschelde.

So the existence of the low oxygen concentration in the middle part of the Zeeschelde and the possibilities to improve oxygen concentration in this area should be a prior concern for the Zeeschelde research and management.

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Hoofdstuk 8. Opwaardering van de Omes databank

Annelies Goffin
Klaas Deneudt

Eindverslag voor deelstudie 7 (perceel 7), periode 2008-2009

Flanders Marine Institute/ Vlaams Instituut voor de Zee VLIZ
Marine & Coastal Research & Management in Flanders

Dir. Jan Mees

8.1. Inleiding

Na de grondige opwaardering van de OMES databank is de belangrijkste taak van het VLIZ in het OMES project het beheer en actueel houden van de databank. Daarnaast werd extra aandacht besteed aan het screenen van de Lippenbroek datasets in het kader van verder datamanagement binnen dit experiment.

De databank wordt actueel gehouden door het toevoegen, kwaliteitscontroleren en integreren van de nieuw gegenereerde gegevens uit de lopende monitoring van 2008-2009. Zoals voorzien in de planning werd op vraag van de UA een werkversie van de OMES databank gevormd. Deze databank is een transporteerbare extractie uit de VLIZ IMERS database en zal op regelmatige basis op geautomatiseerde wijze geüpdated worden. De db is voorzien van een zoekform zodat data snel en op een gebruiksvriendelijke manier uit de databank geëxtraheerd kan worden.

Aan de volgende activiteiten werd gedurende 2009 verder aandacht besteed:

- Kritische evaluatie databank en mogelijke structurele en technische aanpassingen
- Kwaliteitscontrole
- Data beleid
- Communicatie
- Databank
    - Toevoegen van gegevens
    - Extractie db uit IMERS
    - Lippenbroek experiment

8.2. Kwaliteitscontrole

De kwaliteitscontroles bestaan uit een eerste check op vlak van duplicate data, afwijkende eenheden, lege velden .... Een groot deel van die controle gebeurt reeds tijdens het integreren in de IMERS databank. Deze kwaliteitscontrole werd hoofdzakelijk in 2007 uitgevoerd. In 2008 werd vooral gekeken naar inhoudelijke controle op vlak van 'onmogelijke waarden'. Door de dimensionaliteit van de staalnames (tijd, plaats, matrix,..) vergt deze controle een grondige achtergrondkennis en werd deze dan ook uitgevoerd in samenwerking met Tom Maris van de Onderzoeksgroep Ecosysteembeheer (UA-Ecobe). In 2008 werden bijna alle waterkwaliteitsgegevens gescreend. In 2009 werd de kwaliteitscontrole van de andere parameters verder gezet.

Werkwijze: zie rapport 2007-2008
Aan de hand van grafieken werden potentiële outliers visueel geïdentificeerd. Aangezien het seizoen (temperatuursafhankelijheid) en de afstand tot de monding van belang zijn bij het interpreteren van de resultaten van verschillende parameters, werden de gegevens uitgezet per maand en per afstand t.o.v. Vlissingen.
De ruwe gegevens werden samengevat in een tabel. De grafieken en ruwe data werden ter nazicht doorgestuurd naar Ecobe, UA, die de 'echte outliers' en 'aanvaardbare waarden' identificeerden. De nodige kwaliteitslabels werden door VLIZ direct toegekend in de OMES databank.

8.3. Databeleid

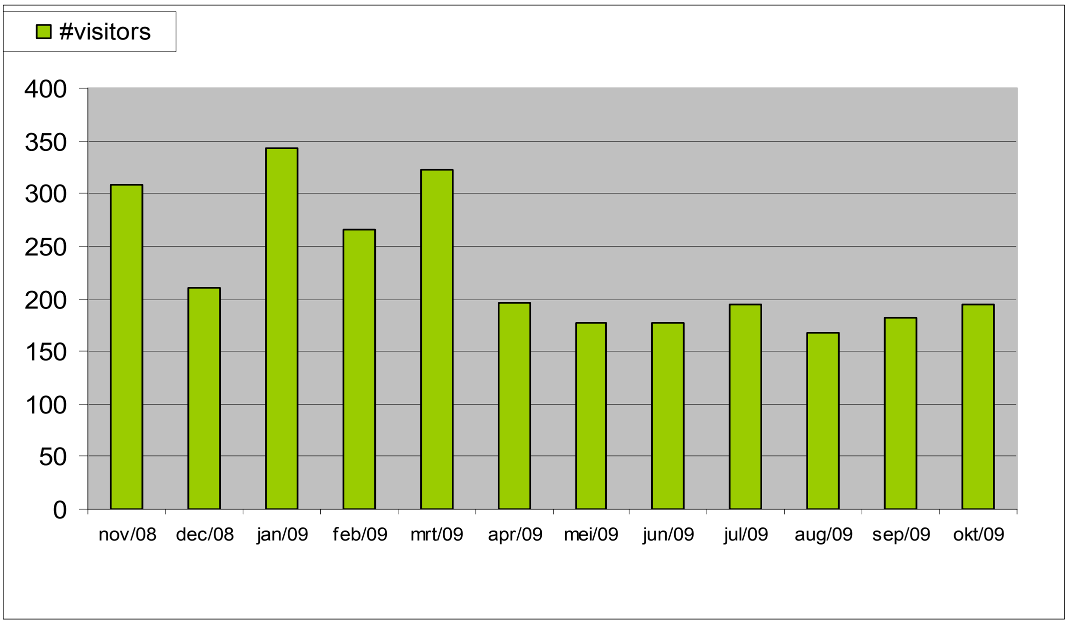
Herverdeling van de OMES gegevens gebeurde conform het databeleid dat in 2008 werd vastgelegd. Specifieke data aanvragen werden gecommuniceerd met de coördinator van het OMES project en werden technisch verder opgevolgd door het VLIZ. In het kader van MONEOS zal het bestaande databeleid verder moeten worden aangepast.

8.4. Communicatie

8.4.1. Website

Het VLIZ host de website op ***http://www.vliz.be/projects/omes*** .

De website wordt up to date gehouden, maar geen grote veranderingen werden toegepast ten opzichte van 2008. Het aantal bezoekers van de website werd tijdens de volledige duur van het project vastgelegd en we zien een stabilisatie van het aantal bezoekers na de pieken die gepaard gingen met eerste maanden van publicatie van de OMES website.



8.5. Databank

8.5.1. Toevoegen van gegevens aan de databank

De OMES databank wordt verder aangevuld met nieuwe monitoringsgegevens. De integratie van de nieuwe datasets 2008-2009 worden eind dit jaar gefinaliseerd. In 2007-2008 werden gegevens opgespoord die relevant kunnen zijn voor het OMES project: het gaat hier om historische gegevens (data-archeologie) of meer recente data (vb hydrometrische metingen AWZ,..) die een nuttige aanvulling vormen op de bestaande meetreeksen in de databank. De opname van deze datasets in de OMES database werd uitgesteld aangezien het VLIZ werkt aan de ontwikkeling van een dataportaal voor de Schelde.

Het VLIZ startte in 2008, op vraag van de Afdeling Maritieme Toegang met de ontwikkeling van een dataportaal binnen het project Scheldemonitor. Onderzoeksgegevens verzameld binnen het Schelde-estuarium of data relevant voor Schelde onderzoek zullen beschikbaar worden gemaakt via het Schelde dataportaal. Tal van Nederlandse en Vlaamse instituten dragen hiertoe bij.. Veel van de data relevant voor het OMES project zullen dus na verloop van tijd beschikbaar komen via het dataportaal. Voor deze datasets wordt dus opname in de OMES database minder relevant. Indien de specifieke vraag zou komen, kunnen wel extra datasets in de OMES database opgenomen worden.

8.5.2. Archivering OMES Datafiles

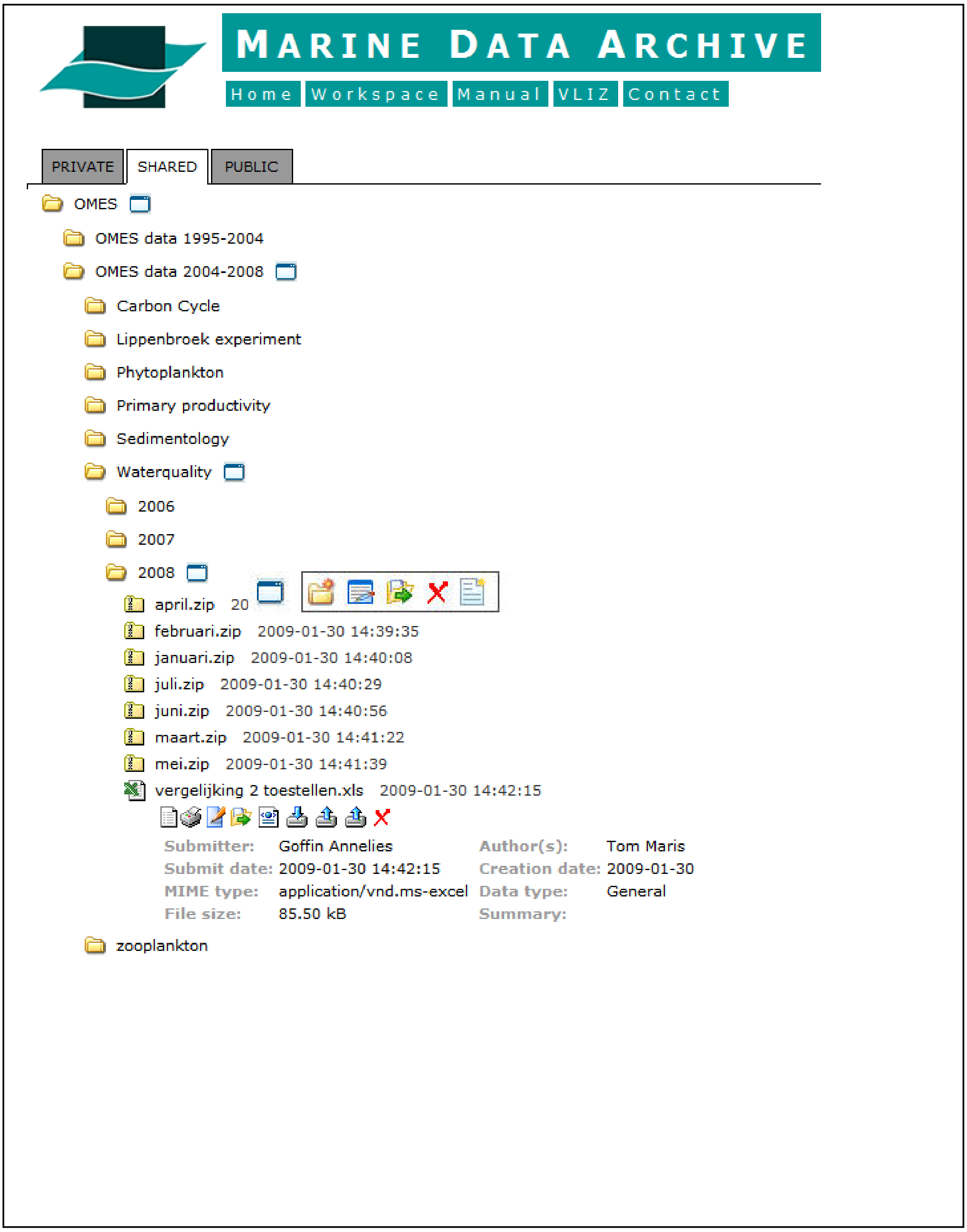
Binnen het marien en estuarien onderzoek is er een nood aan de bescherming van de fysische integriteit van datasets. Het archiveren, beschrijven en bewaren van datasets in een centraal systeem is van uiterst belang.

Het Marien Data Archief (MDA) van het VLIZ is een online systeem speciaal ontwikkeld voor mariene wetenschappers, onderzoeksinstituten en projectgroepen (***http://mda.vliz.be***).
Het stelt hen in staat om datafiles op een gedocumenteerde manier te archiveren op een externe locatie. Elke file is voorzien van metadata die de inhoud van de file beschrijft. De MDA is een beveiligd archief waartoe toegang kan worden verleend aan de hand van 'reading' en/of 'writing rights'.

Binnen projecten kan dit archief dienen als eerste stap naar datamanagement nl. het centraliseren van projectdata. Deze datafiles kunnen op de MDA gedeeld en geconsulteerd worden door de selecte groep van projectpartners.

Het VLIZ heeft alle OMES projectdata verzameld op het Marien Data Archief. Hiervoor werd een structuur van subfolders opgesteld zodat datafiles op een duidelijke en logische manier kunnen teruggevonden worden.

Momenteel hebben enkel de projectcoördinator en enkele Omes partners toegang tot de OMES_MDA. Toegang kan verleend worden aan de andere partners op aanvraag.



OMES
OMES data 1995-2004
OMES data 2004-2008
Carbon Cycle
Lippenbroek experiment
Phytoplankton
Primary productivity
Sedimentology
Waterquality
2006
2007
2008
april.zip 20
februari.zip 2009-01-30 14:39:35
januari.zip 2009-01-30 14:40:08
juli.zip 2009-01-30 14:40:29
juni.zip 2009-01-30 14:40:56
maart.zip 2009-01-30 14:41:22
mei.zip 2009-01-30 14:41:39
vergelijking 2 toestellen.xls 2009-01-30 14:42:15

Submitter:	Goffin Annelies	Author(s):	Tom Maris
Submit date:	2009-01-30 14:42:15	Creation date:	2009-01-30
MIME type:	application/vnd.ms-excel	Data type:	General
File size:	85.50 kB	Summary:	

zooplankton

Structuur van de projectmap OMES in het Marien Data Archief

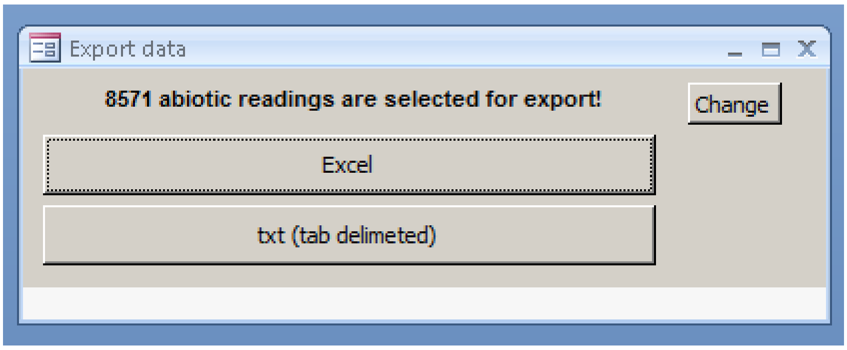
8.5.3. Extractie db uit IMERS

Door het VLIZ werd een access -databank ontwikkeld die de partners van het OMES-project toelaat om op een gebruiksvriendelijke manier de volledige OMES-dataset te bevragen. De inhoud van deze databank bestaat uit een extractie van de OMES -gegevens uit het IMERS-datasysteem van het VLIZ .
De databank kan echter niet gebruikt worden om nieuwe data in te voeren. Data invoer en beheer blijft één van de taken binnen het datamanagement van het OMES project en wordt dus door het VLIZ uitgevoerd. Op regelmatige basis zal dus een update van de OMES-databank uitgevoerd worden.

De databank heeft twee functies:

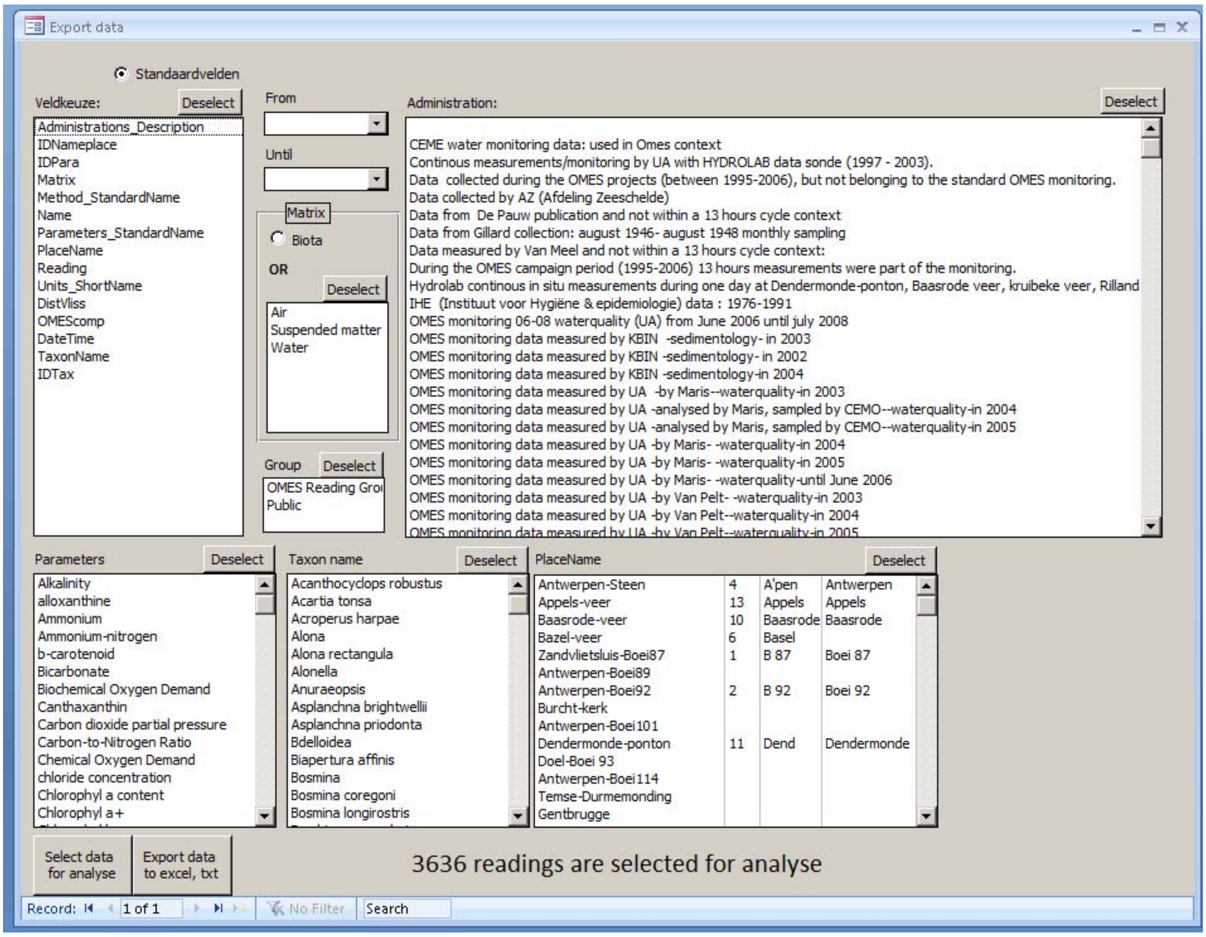
- Data kan geselecteerd worden om te exporteren naar de txt-format of excell. De output is één enkele tabel. Deze tabel kan verder dienen als invoerfile voor andere programma's. Het is mogelijk zelf queries te schrijven op deze tabel, maar een gebruiker kan ook via een formulier data selecteren zonder dat hij specifieke kennis van Access nodig heeft. Tijdstip, type dataset, parameters en stations of plaatsnamen zijn de belangrijkste selectiecriteria

- In de databank werden enkele eenvoudige analysetools voorzien. Deze laten een eerste verkenning van de geselecteerde data toe. Eenvoudige beschrijvende statistieken (zie illustratie XX) en grafieken (zie illustratie XX) kunnen opgevraagd worden.



8571 abiotic readings are selected for export!

Exportformulier van de OMES-databank



● Standaardvelden

Veldkeuze:
Administrations_Description
IDNameplace
IDPara
Matrix
Method_StandardName
Name
Parameters_StandardName
PlaceName
Reading
Units_ShortName
DistVliss
OMEScomp
DateTime
TaxonName
IDTax

Matrix: ○ Biota OR
Air
Suspended matter
Water

Group:
OMES Reading Groi
Public

Administration:
CEME water monitoring data: used in Omes context
Continous measurements/monitoring by UA with HYDROLAB data sonde (1997 - 2003).
Data collected during the OMES projects (between 1995-2006), but not belonging to the standard OMES monitoring.
Data collected by AZ (Afdeling Zeeschelde)
Data from De Pauw publication and not within a 13 hours cycle context
Data from Gillard collection: august 1946- august 1948 monthly sampling
Data measured by Van Meel and not within a 13 hours cycle context:
During the OMES campaign period (1995-2006) 13 hours measurements were part of the monitoring.
Hydrolab continous in situ measurements during one day at Dendermonde-ponton, Baasrode veer, kruibeke veer, Rilland
IHE (Instituut voor Hygiëne & epidemiologie) data : 1976-1991
OMES monitoring 06-08 waterquality (UA) from June 2006 until july 2008
OMES monitoring data measured by KBIN -sedimentology- in 2003
OMES monitoring data measured by KBIN -sedimentology- in 2002
OMES monitoring data measured by KBIN -sedimentology-in 2004
OMES monitoring data measured by UA -by Maris--waterquality-in 2003
OMES monitoring data measured by UA -analysed by Maris, sampled by CEMO--waterquality-in 2004
OMES monitoring data measured by UA -analysed by Maris, sampled by CEMO--waterquality-in 2005
OMES monitoring data measured by UA -by Maris- -waterquality-in 2004
OMES monitoring data measured by UA -by Maris- -waterquality-in 2005
OMES monitoring data measured by UA -by Maris- -waterquality-until June 2006
OMES monitoring data measured by UA -by Van Pelt- -waterquality-in 2003
OMES monitoring data measured by UA -by Van Pelt--waterquality-in 2004
OMES monitoring data measured by UA -by Van Pelt--waterquality-in 2005

Parameters:
Alkalinity
alloxanthine
Ammonium
Ammonium-nitrogen
b-carotenoid
Bicarbonate
Biochemical Oxygen Demand
Canthaxanthin
Carbon dioxide partial pressure
Carbon-to-Nitrogen Ratio
Chemical Oxygen Demand
chloride concentration
Chlorophyl a content
Chlorophyl a+

Taxon name:
Acanthocyclops robustus
Acartia tonsa
Acroperus harpae
Alona
Alona rectangula
Alonella
Anuraeopsis
Asplanchna brightwellii
Asplanchna priodonta
Bdelloidea
Biapertura affinis
Bosmina
Bosmina coregoni
Bosmina longirostris

PlaceName:

Antwerpen-Steen	4	A'pen	Antwerpen
Appels-veer	13	Appels	Appels
Baasrode-veer	10	Baasrode	Baasrode
Bazel-veer	6	Basel	
Zandvlietsluis-Boei87	1	B 87	Boei 87
Antwerpen-Boei89			
Antwerpen-Boei92	2	B 92	Boei 92
Burcht-kerk			
Antwerpen-Boei101			
Dendermonde-ponton	11	Dend	Dendermonde
Doel-Boei 93			
Antwerpen-Boei114			
Temse-Durmemonding			
Gentbrugge			

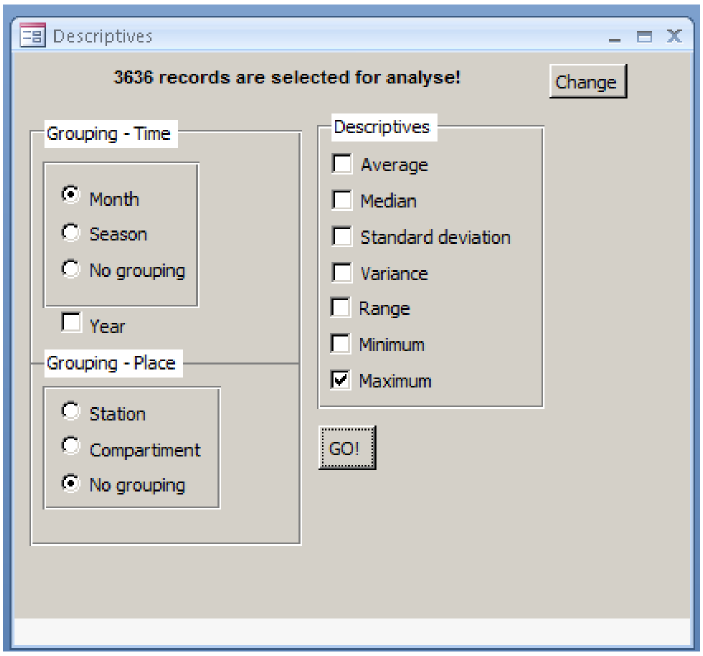
3636 readings are selected for analyse

Selectieformulier van de OMES databank

Output: Beschrijvende statistieken en grafieken

De beschrijvende statistieken gemiddelde, maximum, minimum, range, variantie, standaarddeviatie kunnen opgevraagd worden van geselecteerde data.
Om de grafieken aan te maken wordt gebruik gemaakt van het open source softwarepakket R. Op basis van de geselecteerde data en de gekozen grafiek maakt de Access-databank een datafile en R-scriptfile aan. Op de achtergrond wordt R aangeroepen die deze bestanden gebruikt om een grafiek aan te maken. De R -files zijn toegankelijk voor de gebruiker en kunnen rechtstreeks in R verder aangepast worden. Drie types van grafieken zijn voorzien:

- een boxplot waarmee een parameter kan samengevat worden ten opzichte van de afstand tot Vlissingen of OMES-compartiment
- een scatterplot waarmee twee parameters ten opzichte van mekaar kunnen uitgezet worden. In de scatterplot kan een groepering aangeduid worden op basis van maand, seizoen of jaar.
- Een 3D plot waarmee een parameter in Scheldebekken kan voorgesteld worden.



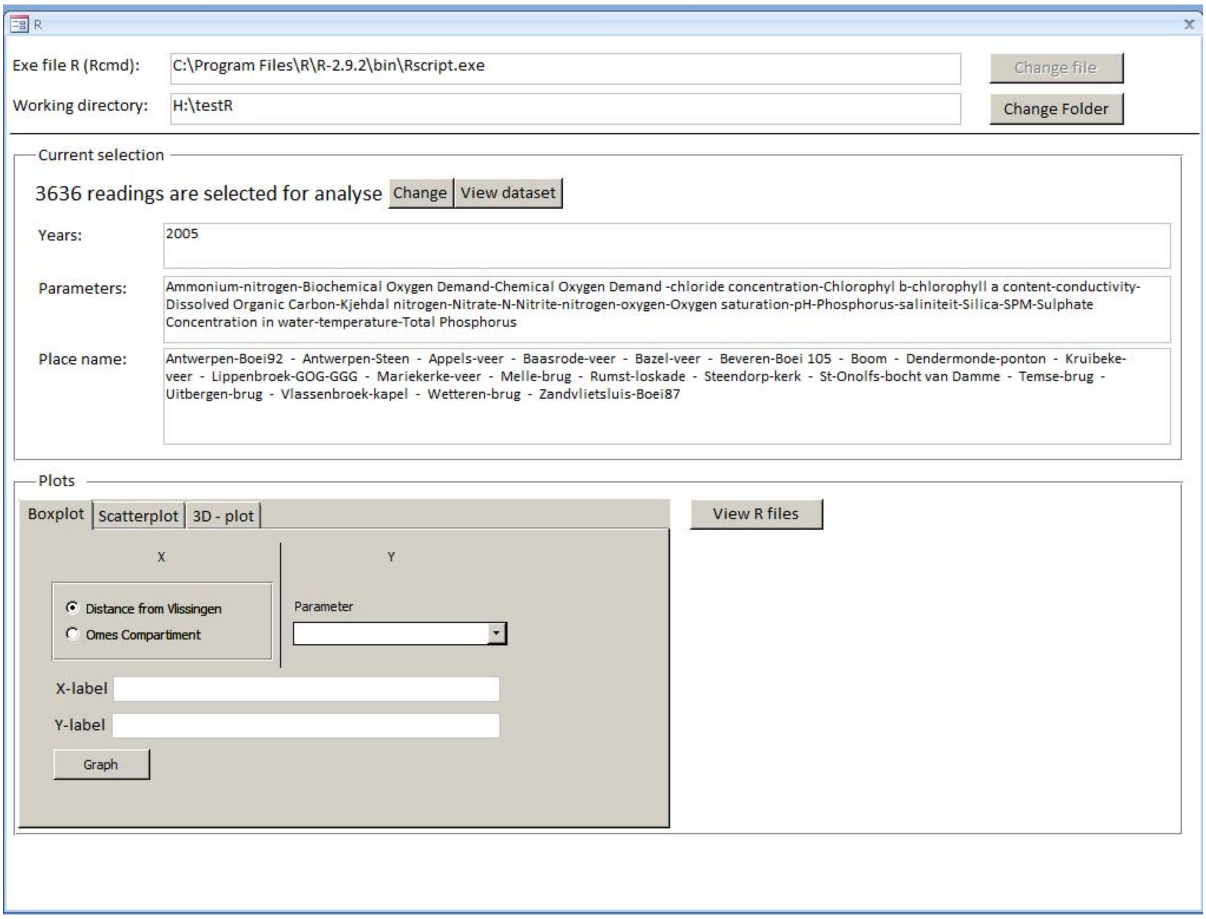
3636 records are selected for analyse!

Grouping - Time
● Month
○ Season
○ No grouping
☐ Year

Grouping - Place
○ Station
○ Compartiment
● No grouping

Descriptives
☐ Average
☐ Median
☐ Standard deviation
☐ Variance
☐ Range
☐ Minimum
☑ Maximum

Formulier voor statistische verwerking



Exe file R (Rcmd): C:\Program Files\R\R-2.9.2\bin\Rscript.exe

Working directory: H:\testR

Current selection

3636 readings are selected for analyse

Years: 2005

Parameters: Ammonium-nitrogen-Biochemical Oxygen Demand-Chemical Oxygen Demand -chloride concentration-Chlorophyl b-chlorophyll a content-conductivity-Dissolved Organic Carbon-Kjehdal nitrogen-Nitrate-N-Nitrite-nitrogen-oxygen-Oxygen saturation-pH-Phosphorus-saliniteit-Silica-SPM-Sulphate Concentration in water-temperature-Total Phosphorus

Place name: Antwerpen-Boei92 - Antwerpen-Steen - Appels-veer - Baasrode-veer - Bazel-veer - Beveren-Boei 105 - Boom - Dendermonde-ponton - Kruibeke-veer - Lippenbroek-GOG-GGG - Mariekerke-veer - Melle-brug - Rumst-loskade - Steendorp-kerk - St-Onolfs-bocht van Damme - Temse-brug - Uitbergen-brug - Vlassenbroek-kapel - Wetteren-brug - Zandvlietsluis-Boei87

Plots

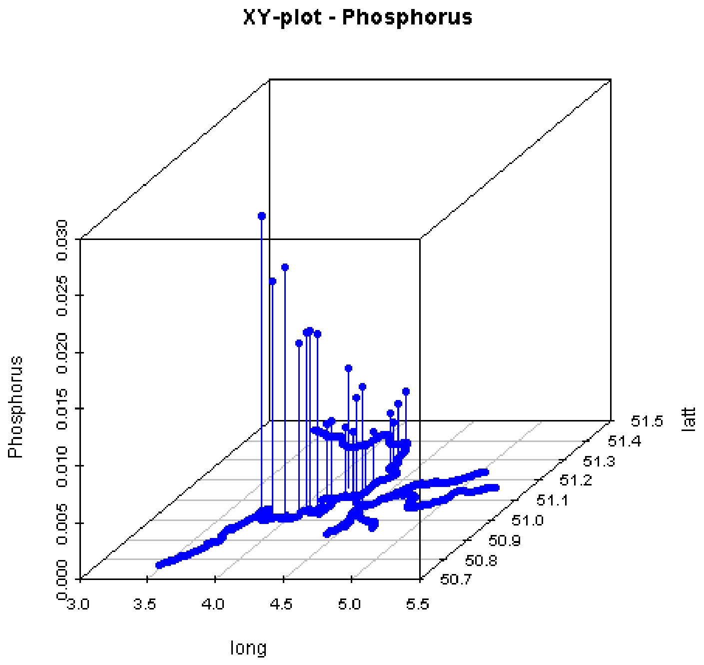
X
● Distance from Vlissingen
○ Omes Compartiment

Y
Parameter

X-label

Y-label

Opvragen van grafieken



Voorbeeld van een geografisch 3D-grafiek waarbij gegevens werden geplot op de Scheldecontouren.

8.5.4. Het Lippenbroek experiment

Lippenbroek is een multidisciplinair experiment. (http://www.vliz.be/projects/omes/news.php) Binnen dit experiment worden zowel de evolutie van de waterkwaliteit, biota, vegetatie, de sedimentologie, geulvorming,... opgevolgd. Dit levert een grote hoeveelheid data.
Een eerste stap in het databeheer van Lippenbroek is het opslaan en centraliseren van alle datafiles verzameld binnen het experiment. Hiervoor werd het Marien Data Archief van het VLIZ ter beschikking gesteld (uitleg MDA zie 1.3.2).
Tot nu toe werden enkel voorbeeldfiles van de verschillende types staalnamen in Lippenbroek opgeladen. In november zullen een groot deel van de datafiles beschikbaar zijn op de MDA onder de map Lippenbroek_experiment.

Een volgende stap in het databeheer is de ontwikkeling van een access database voor het Lippenbroek-experiment. Aan de hand van de voorbeeldfiles die reeds werden opgeladen in de MDA, werd de data eerst onderzocht op vlak van type, parameters, mogelijke problemen/onvolledigheden, tijdsreeksen. Door de multidisciplinariteit van de data moest onderzocht worden in welke mate integratie van de data mogelijk is. Voor het einde van dit jaar zal de databank in samenwerking met de UA Lippenbroek partners vorm krijgen. Alle data zal opgeslagen worden in de databank, maar integratie zal slechts doorgevoerd worden in mate van het mogelijke.

Hoofdstuk 9. Effecten van waterkwaliteit en getij op overstromingsgebieden

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S. Jacobs
J. Teuchies
T. Cox
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Campus Drie Eiken, Universiteitsplein 1, 2160 Wilrijk.

Eindverslag voor de deelstudie 8 (perceel 8), periode 2007-2008, met inbegrip van onderzoek verricht te Lippenbroek van:

Perceel 1: Maris T., T. Cox & P. Meire (UA-ECOBE)
Perceel 3: Chen M., S. Wartel, F. de Smedt, E. Van den Storme (VUB-HYDR)
Vandenbruwaene W. & Temmerman S. (UA-PLG)
Perceel 4: Van Burm E., R. Dasseville, J. Van Wichelen & W. Vyverman (UGent)
Perceel 5: Vanderborght J.P. (ULB)
Perceel 6: Tackx M., B. Mialet & F. Azemar (Université Paul Sabatier Toulouse)

9.1. Inleiding

Onderzoek naar de effecten van waterkwaliteit en getij op overstromingsgebieden, en in het bijzonder op gecontroleerde overstromingsgebieden met een gecontroleerd gereduceerd getij, vormen de kerntaak van perceel 8. Omdat een multidisciplaire aanpak noodzakelijk is om de vele invloeden van water en getij op een overstromingsgebied zo goede mogelijk in beeld te brengen, is dit onderzoekshoofdstuk niet het resultaat van 1 perceel, maar een samenwerking van verschillende percelen uit het Omes bestek. Met name het onderzoek in het gecontroleerd overstromingsgebied Lippenbroek, is het resultaat van een samenwerking van verschillende Omes percelen, bijgestaan door externe onderzoeksprogramma`s, onder coördinatie van perceel 8. In dit hoofdstuk worden de resultaten van perceel 8 en de bijdragen van de enkele andere percelen samengebracht. Wanneer in dit hoofdstuk resultaten worden gerapporteerd afkomstig van een ander perceel dan perceel 8, wordt dit uitdrukkelijk vermeld.

De voorbije jaren was het onderzoek van perceel 8 beperkt tot de mesocosmosopstellingen te Wilrijk en Kruibeke. Sinds 2006 is de klemtoon van het onderzoek van perceel 8 verschoven naar het nieuwe pilootproject Lippenbroek. Het onderzoek in de mesocosmosopstelling te Wilrijk werd afgebouwd, en is sinds 2009 geen onderdeel meer van Omes.

Dit hoofdstuk herneemt grotendeels de inleidende paragrafen uit het vorige omes rapport. Op deze wijze kunnen zij die niet vertrouwd zijn met het opzet van Lippenbroek, kennis maken met dit boeiende pilootproject.

Het Lippenbroek onderzoek loopt nu reeds 4 jaar. Verschillende van de onderzoeksresultaten werden intussen voorgelegd ter publicatie in internationale vakliteratuur, of zijn intussen gepubliceerd. In dit hoofdstuk worden daarom reeds verschillende resultaten van de voorbije 4 jaar geïntegreerd en gebundeld, indien mogelijk in de vorm van een onderzoeksartikel.

9.1.1. Waarom onderzoek naar gecontroleerde overstromingsgebieden met een gecontroleerd gereduceerd getij?

Zowel in oppervlakte als in kwaliteit gaan estuaria wereldwijd achteruit. De Zeeschelde kent een zeer grote input van organische belasting en nutriënten. Deze grote input, gepaard met het verlies aan functies, bezorgden de Schelde haar kwalijke reputatie van sterk vervuilde rivier. Ondanks de vele inspanningen op vlak van waterzuivering en de verbetering die wordt waargenomen in vele delen van het estuarium, blijft de waterkwaliteit in de Zeeschelde een hinderpaal voor de ontwikkeling van een duurzaam ecosysteem (Van Damme et al., 2005).

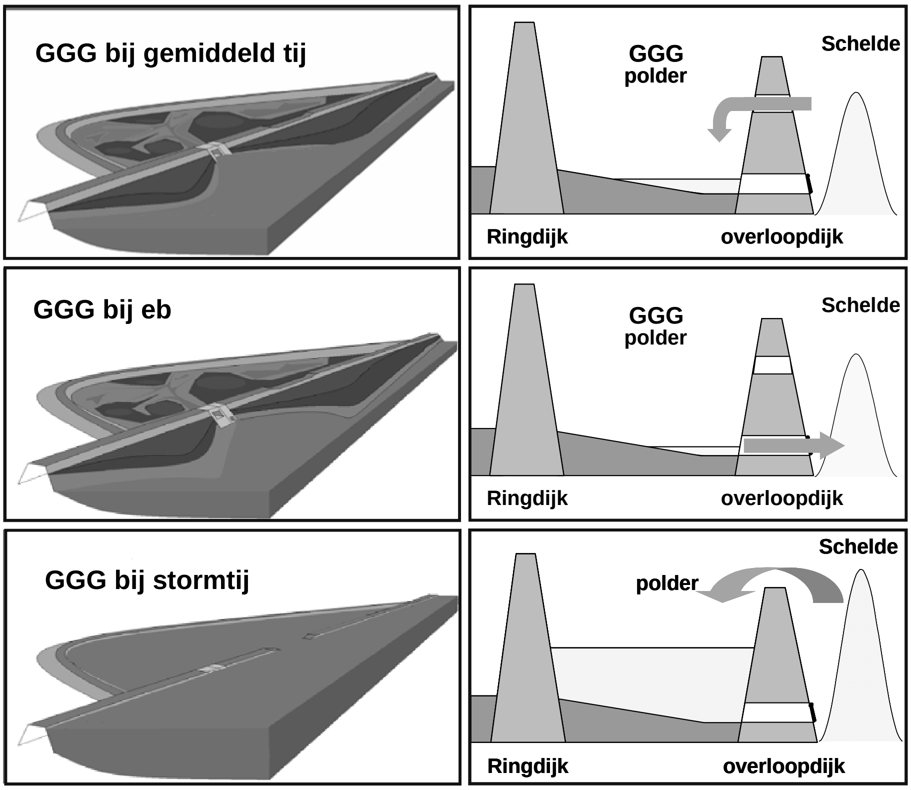
In de meeste Europese estuaria is het areaal slikken en schorren de voorbije eeuw drastisch ingekrompen. Het Schelde-estuarium vormt hierop geen uitzondering: tussen 1900 en 1990 kromp de Zeeschelde met 781 ha in tot 4923 ha (Meire et al., 2005). Deze verliezen, in hoofdzaak door inpolderingen en dijkwerken, troffen vooral de slikken en schorren. Het aandeel van deze intergetijdengebieden in de Zeeschelde daalde dan ook van 38% naar 28%. Dit betekent een belangrijk verlies aan habitat en essentiële ecosysteemfuncties.

Herstel van ecosysteemfuncties, wat in hoofdzaak neerkomt op meer ruimte creëren voor slikken en schorren, zal de ecologische kwaliteit van de Zeeschelde sterk verbeteren. Ruimte is echter schaars en dus kostbaar in het dichtbevolkte Vlaanderen. Het vernieuwde Sigmaplan, met focus op veiligheid en natuurlijkheid, biedt nieuwe perspectieven voor slik- en schorherstel. Het Sigmaplan, het Vlaamse plan dat het Zeescheldebekken moet behoeden voor overstromingen, voorziet o.a. in de aanleg van verschillende gecontroleerde overstromingsgebieden. Deze gebieden, met in de eerste plaats een veiligheidsfunctie, kunnen mits een aangepast sluizensysteem bijdragen tot een herstel van belangrijke estuariene functies: we spreken van een GOG-GGG, een gecontroleerd overstromingsgebied met gecontroleerd gereduceerd getij (Meire et al., 2005). Door het creëren van de juiste randvoorwaarden (vooral juiste tijregime), wordt gestreefd naar een zo dynamisch mogelijk systeem waarbij de natuur zelf de belangrijkste sturende factor wordt. Na het geven van de juiste aanzet, doet de natuur zelf het werk. Er moet getracht worden het beheer en de artificiële ingrepen tot een minimum te beperken zodat beheersmatig bijsturen enkel moet indien een gewenste evolutie niet of niet snel genoeg kan starten.

9.1.2. Werking van een GOG-GGG

De gecontroleerde overstromingsgebieden (GOG) zijn laaggelegen polders die van het estuarium gescheiden worden door een verlaagde overloopdijk. Tijdens een stormtij, wanneer in het estuarium gevaarlijk hoge waterstanden dreigen, wordt het peil van de overloopdijk overschreden en vult de polder zich met vloedwater (Figuur 9.1). Door het grote waterbergende vermogen van de GOG-polders kunnen zij de waterstand in het estuarium significant verlagen en zo andere gebieden vrijwaren van overstromingen.

Om een slik- en schorecosysteem te introduceren in een GOG-polder, die gemiddeld slechts 1 à 2 maal per jaar onderloopt bij stormvloed, is een sluizensysteem vereist dat dagelijkse uitwisseling van Scheldewater mogelijk maakt. Slikken zullen zich ontwikkelen in die delen van de polder die dagelijks overspoeld worden met Scheldewater. Schorren daarentegen overspoelen enkel rond springtij. Voor de ontwikkeling van een slik-schor ecosysteem is het dus van essentieel belang dat het getij in de polder een grote springtij-doodtij variatie kent, waarbij het doodtij slechts een beperkt deel van de polder overspoelt; bij springtij staat quasi de ganse polder blank. Tussen het spring- en doodtijniveau ligt een hele waaier aan overstromingsfrequenties en dus mogelijkheden voor habitatdifferentiatie.



Figuur 9.1 Werking van het GOG-GGG bij gemiddeld getij (boven), bij eb (midden) en bij stormtij (onder). Wanneer de Schelde bij gemiddeld getij het drempelpeil van de sluis bereikt, vult de polder zich gedeeltelijk. Bij eb stroomt het water terug weg via de lager gelegen uitwatering. Occasioneel, bij stormtij, wordt het peil van de overloopdijk overschreden en vult de polder zich met enkele meters.

Het introduceren van de volledige tijslag van de Schelde in de GOG-polder is uitgesloten. Als voor natuurontwikkeling grote volumes getijdenwater de polder vullen, komt het waterbergend vermogen en dus de veiligheidsfunctie van een GOG in het gedrang. Ook vanuit ecologisch standpunt is de volledige tijslag niet wenselijk. Om bij doodtij slechts een beperkte oppervlakte te overstromen in de zeer laag gelegen GOG-polder, moet het hoogwaterpeil sterk gereduceerd worden. Bij springtij zijn ook niet de hoge waterstanden van de Schelde vereist: net alles overstromen kan volstaan voor het ontwikkelen van vele ecosysteemfuncties. Voor nutriëntcyclering is bijvoorbeeld niet zozeer de grote massa bulkwater van belang, wel het nasijpelingswater.

Met een eenvoudige duiker doorheen de dijk kan de tijslag sterk gereduceerd worden, maar wordt ook de essentiële springtij-doodtij variatie grotendeels teniet gedaan. Een systeem met hoge inlaatsluizen en lage uitlaat kan het getij reduceren mét behoud van de springtij-doodtij variatie. Dat is het basisprincipe van een GGG: een Gecontroleerd Gereduceerd Getij met behoud van springtij-doodtijvariatie zodat de overstromingsfrequentie op het polderoppervlak vergelijkbaar is met deze in de natuurlijke slik- en schorgebieden (Cox et al., 2006, Maris et al., 2008). De werking is eenvoudig (Figuur 9.1): stijgt het water in het estuarium boven het drempelpeil van de inlaat, dan stroom het water vrij in de polder. De instroom duurt net zo lang tot het waterpeil weer onder het drempelpeil zakt. Uitstroom gebeurt gravitair via een klassieke poldersluis, waarbij een klep opent bij eb en wordt dichtgeduwd bij vloed.

9.1.3. Onderzoeksvragen

Het concept GOG-GGG is uniek. Systemen met een gereduceerd, maar wel gedempt getij werden reeds beproefd (bv. Pelletier et al., 2004), maar deze vertonen niet dezelfde abiotiek (weinig tot geen springtij-doodtijvariatie) als de buitendijkse slikken en schorren. Een getij reguleren tot op polderniveau met behoud van springtij-doodtijvariatie is nieuw. Vooraleer dit principe op grote schaal toe te passen in het kader van Sigma is een grondige proefopzet een must. Er wordt een strategie in drie stappen gevolgd, met telkens toenemende complexiteit.

Stap 1: Mesocosmos experiment te Wilrijk

Onderzoek naar de groei van riet bij verschillende overstrominsfrequenties en verschillende bodemtextuur

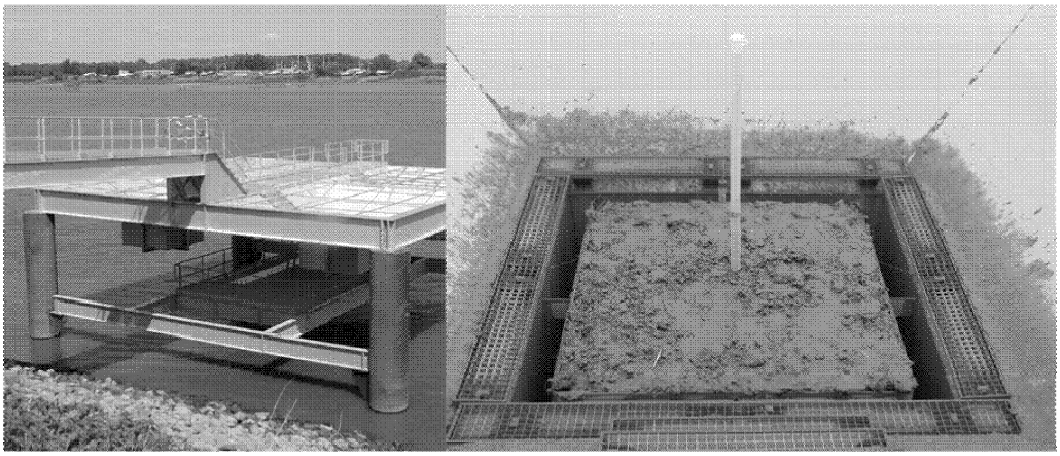
Riet *(Phragmites australis)* is op de schorren langsheen de Zeeschelde de dominante soort. Ook in GGG's kan riet een belangrijke rol spelen. Daarom wordt in het mesocosmosexperiment te Wilrijk (Figuur 9.2) de ontwikkeling van riet opgevolgd bij afwijkende overstromingsregimes. De experimenten worden uitgevoerd op 2 verschillende bodemtypes, die courant voorkomen in het estuarium. Dit onderzoek wordt nu afgerond. Vanaf oktober 2009 is deze opstelling niet meer opgenomen in het Omes onderzoek van perceel 8.

Figuur 9.2 mesocosmosopstelling te Wilrijk

Stap 2: Mesocosmos experiment te Kruibeke

Onderzoek naar het gedrag van zware metalen in bodem en overstromingswater onder verschillende overstromingsregimes, en de interactie met riet, bij verschillende bodemtextuur

In deze mesocosmosopstelling (Figuur 9.3) wordt naast de effecten van getij en bodemtextuur, gekeken naar het gedrag van zware metalen. Vele polders langsheen de Zeeschelde kennen een historische vervuiling met zware metalen. Verandering van de redoxpotentiaal in de bodem ten gevolge van overstromingingen, kan het gedrag van deze metalen beïnvloeden. Ook het overspoelende water zelf kan nog zware metalen bevatten. In deze studie wordt de rol van riet in de biobeschikbaarheid van zware metalen bestudeerd, evenals in invloed van zware metalen op de ontwikkeling van riet. In hoofdstuk 10.2 worden de resultaten van perceel 8 toegelicht.

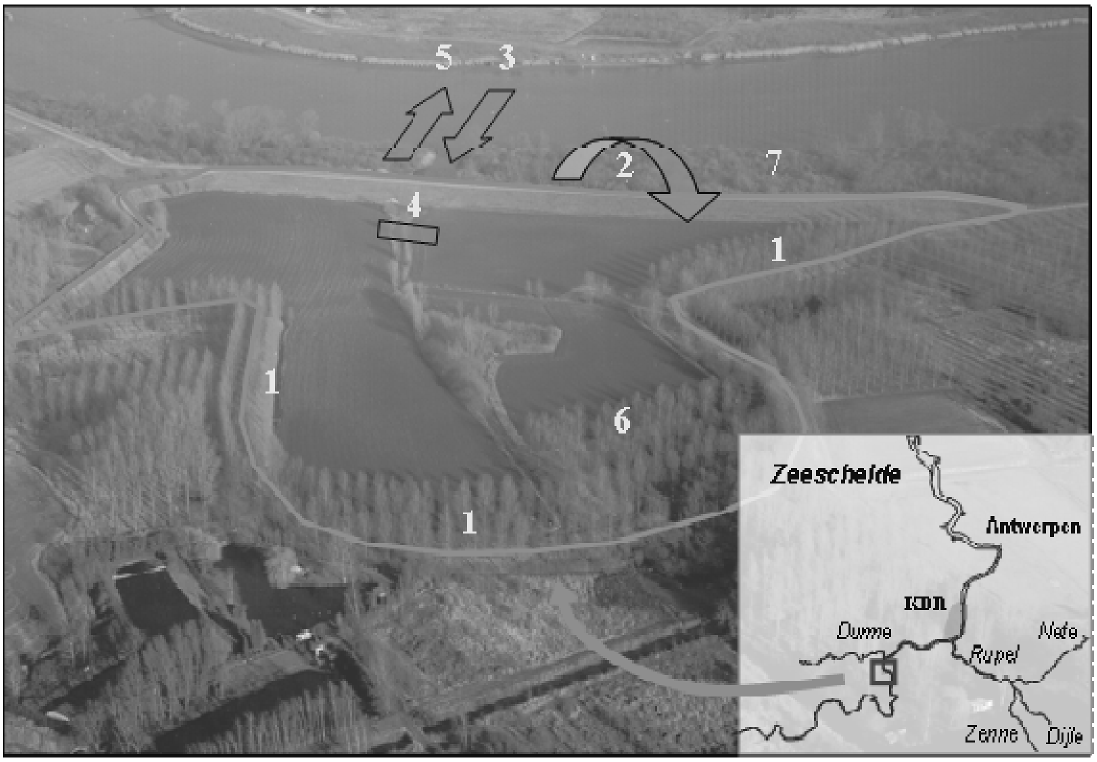


Figuur 9.3 mesocosmosopstelling te Kruibeke

Stap 3: Pilootproject voor een GOG-GGG te Lippenbroek

Onderzoek naar het ecosysteemfunctioneren van een GOG-GGG in de Zeeschelde op een pilootschaal van 10 hectare

In het pilootproject Lippenbroek te Hamme (Figuur 9.4) wordt het GOG-GGG-princiepe sinds maart 2006 uitvoerig opgevolgd. Dit pilootproject zal uitwijzen of in GOG-GGG's duurzame ecologische structuren en functies kunnen ontwikkelen, kwalitatief en kwantitatief gelijkwaardig aan deze van buitendijkse slikken en schorren. Bij dit project wordt perceel 8 ondersteund door de percelen 1, 3, 4, 5 en 6.



Figuur 9.4 Overzichtskaart van Lippenbroek en situering in het estuarium (inzet). De overzichtskaart toont Lippenbroek voor aanvang van de werken, waarop nog duidelijk de sporen van landbouw en populierenaanplant (6) te zien zijn. De werken omhelsden: bouw van ringdijk (1), overloopdijk (2), inlaatsluis (3) en woelkom met staalnamebrug (4). (5) is de bestaande uitwateringssluis, (7) het schor "De Plaat".

9.1.4. Monitoringsopzet Lippenbroek

Hydrodynamiek vormt een cruciale indicator, aangezien de hydrologie de voornaamste drijfkracht is achter fysische, biologische en chemische processen in intertidale gebieden (Zedler et al., 2000). Het hydrologisch regime in een GOG-GGG wijkt licht af van buitendijkse situaties, en kan een potentieel determinerende impact hebben op de structuren en ecosysteemfuncties in het onderzoeksgebied. Een tijregime met duidelijke springtij-doodtij wordt effectief waargenomen, maar wel met gewijzigde overstromingsduur en -hoogte. Intensieve monitoring gaat na wat de invloed hiervan is op het totale ecologisch functioneren, met aandacht voor o.a. vegetatie, plankton en benthos, vissen en vogels, water- en bodemkwaliteit. Dit heeft geleid tot een ambitieus spatio-temporeel monitoringsopzet (Tabel 9.1).

Tabel 9.1: Ruimtelijk-temporele opzet van het monitoringsprogramma

Spacio-temporal monitoring scheme

Monitoring at sites	# sites	frequency
sedimentation/erosion	10	6/year
soil compaction	10	4/year
granulometry	10	4/year
nutrients (soil)	10	4/year
heavy metals (soil)	4 (10)	4/year
benthic fauna	10	4/year
bioturbation	4	4 times
vegetation	10	4/year
tidal regime	10	continuous
grond water analysis	10	2/year

Monitoring at sluices	frequency
discharge in - out / water balance	continuous
tidal regime	continuous
light climate	4/year
suspended matter	continuous
Basic water quality (temp, conduct, O_2, turbidity, pH)	semi cont.
Water quality (e.g. nutrients, metals, pigments, BOD)	4/year

Monitoring entire CRT	frequency
creek morphology	2/year
vegetation mapping	4/year
birds	52/year
fish	2/year

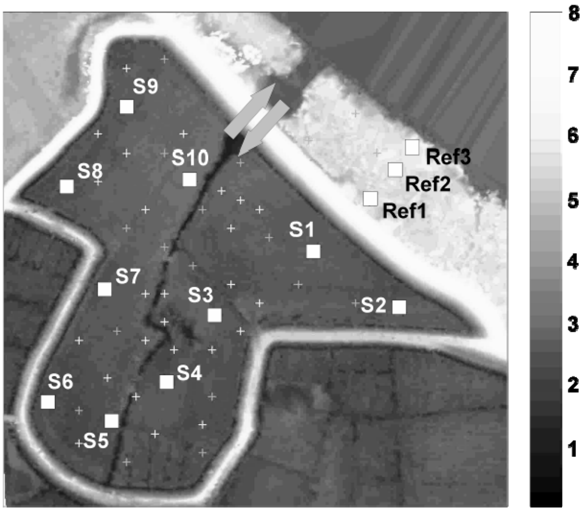
Sedimentatie krijgt hierbij bijzondere aandacht. Sedimentatie is niet enkel van belang vanuit ecologisch standpunt, het is ook cruciaal voor het veiligheidsaspect: de komberging van het GOG-GGG mag niet verloren gaan. Omdat sedimentatie in een GOG-GGG niet meer rechtstreeks afhankelijk is van de hoogteligging van de bodem, maar vooral afhangt van de massa water die via de sluizen het terrein binnenstroomt, gelden bestaande wetmatigheden niet meer. Het negatieve feedback mechanisme, waarbij hoge schorren minder frequent overspoelen en dus minder sterk ophogen, gaat niet volledig op in GGG's.

Binnen dit onderzoeksprogramma neemt perceel 8 de organisatie en coördinatie voor haar rekening. Perceel 8 staat in voor de afbakening van onderzoeksplots, het opvolgen van basisgegevens zoals getij en waterstanden. Specifieke ondertaken spitsen zich onder andere toe op de grondwaterchemie, het gedrag van zware metalen, vegetatie en het gedrag van silicium in een GOG-GGG.

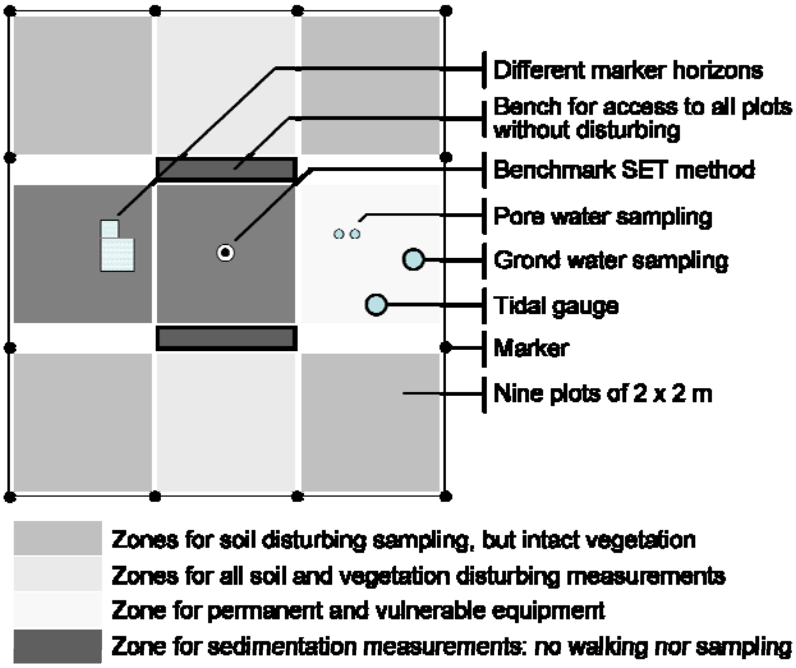
Andere percelen verrichtten onderzoekstaken die kaderen binnen de estuariene monitoringstaken van deze partner:
Perceel 1: Monitoring van de basiswaterkwaliteit
Perceel 3: partim VUB: Sediment en zwevende stof
partim UA-PLG: Sedimentatie en erosie; morfologische ontwikkeling
Perceel 4: Monitoring van fytoplankton en fytobenthos
Perceel 5: Primaire productie en lichtklimaat
Perceel 6: Zooplankton en zoobenthos

Aanvullend op de monitoring binnen het bestek van Omes, wordt bijkomend onderzoek verricht naar benthos, vis en vogels. Het onderzoek naar benthos wordt uitgevoerd door UA-ECOBE, de visbemonstering door INBO en de vogelwaarnemingen door UA-ECOBE. Resultaten van dit onderzoek zullen kort worden toegelicht in het samenvattende hoofdstuk.

Om de staalnames van alle partners ruimtelijk goed op elkaar af te stemmen, werd door perceel 8 een netwerk van staalnamepunten opgezet. Het betreft in hoofdzaak 10 random gekozen vaste sites voor intensieve bemonstering (Figuur 9.5). Om alle resultaten optimaal te linken, is staalname en monitoring voor diverse parameters op dezelfde lokatie een must. Om echter geen beïnvloeding te hebben van de ene monitoringstechniek op de andere (bijvoorbeeld bodemstalen nemen in een zone waar sedimentatie wordt gemeten), is elke site opgedeeld in verschillende zones, voorbestemd voor de verschillende monitoringstechnieken (Figuur 9.6). Op de 10 sites voor intensieve monitoring wordt de waterstand elke 5 minuten geregistreerd met een tijmeter. Aanvullend wordt ter hoogte van de in- en uitlaatconstructie aan de monitoringsbrug het waterpeil van de gracht en woelkom permanent opgevolgd.



Figuur 9.5: Kaart met hoogteligging, voor aanvang van de dijkwerken en sluisconstructie, met aanduiding van de 10 sites voor intensieve monitoring (S1 tot S10) en de monitoringsbrug ter hoogte van de sluizen.



Figuur 9.6: Schema van de intensieve monitoringssite (S1 tot S10), met aanduiding van de tijmeter (Tidal gauge). Elk van de 9 kwadranten binnen een site meet 2 bij 2 meter

9.1.5. Lippenbroek: Gebiedsbeschrijving

Lippenbroek is gelegen langs de linker Scheldeoever nabij Driegoten (Hamme, Oost-Vlaanderen (Figuur 9.4)), met een oppervlakte van ca 10 hectare en een hoogteligging van 2.50 à 3.00m TAW (buitendijkse schorren voor Lippenbroek liggen op 5.50 à 6.00 m TAW). Lippenbroek bevindt zich in de het zoete gedeelte van het Schelde estuarium, ruim 100 km van de monding en zo'n 10 km stroomopwaarts de Rupelmonding (Figuur 9.4). Hierdoor ontvangt Lippenbroek bij vloed een deel van de vuilvracht van de Zenne. Het instromende water kenmerkt zich dus door hoge nutriëntconcentraties en lage zuurstofverzadiging. Dit was echter geen argument om de start van Lippenbroek uit te stellen: GGG's moeten immers een cruciale rol spelen in het functionele herstel van het estuarium.

Tot voor enkele jaren was het gebied nog landbouwgrond met in hoofdzaak maïs en aardappelen en een kleine populierenaanplant in de natste zone. In 2004 startten de werken om het gebied om te vormen tot GOG-GGG: bouw van de nieuwe ringdijk op Sigmaniveau, verlagen van een overloopdijk aan Scheldezijde en de constructie van een nieuwe inlaatsluis voor GGG-werking. Ter hoogte van de sluizen werd een kom gegraven die aansluit op de bestaande poldersloot. Verdere graafwerken werden niet nodig geacht, omdat de bestaande sloot een goede aanzet kan vormen voor de ontwikkeling van een divers krekenstelsel in de polder. Wel werd de ringgracht tegen de dijk gedempt om dijkerosie en kortsluiting van het krekenstelsel te voorkomen.

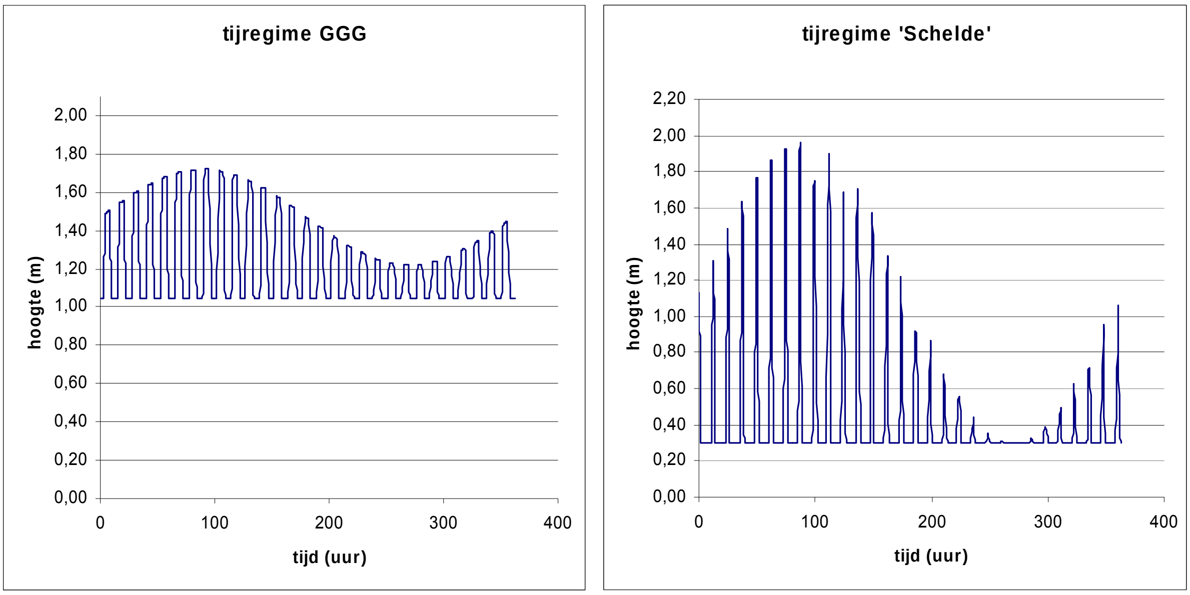
De nieuwe inlaatsluizen te Lippenbroek werden voorzien van schotbalken, waardoor een fijnstelling van het sluisdrempelpeil mogelijk werd. Verschillende "proefinwateringen" waarbij telkens voor enkele getijen water werd binnengelaten bij een welbepaald sluisdrempelpeil, lieten toe de meest geschikte sluisconfiguarie te vinden, dwz die sluisconfiguratie waarbij het getij in de polder het beste het getij op de schorren benadert. In maart 2006 konden de sluizen permanent geopend worden en werd Lippenbroek onderdeel van het estuarium.

9.2. Mesocosmosopstelling te Wilrijk

9.2.1. Inleiding

In de mesocosmos opstelling te Wilrijk (Figuur 9.7) wordt de invloed van overstromingsregime op de groei van *Phragmites australis*, riet, onderzocht. Er wordt met onder andere gekeken of de groie van riet zal afwijken, wannneer de plant onderhevig wordt aan het gewijzigd getijdenregime zoals dat voorkomt in een GGG. De planten groeien op 6 verschillende niveaus ten opzichte van het overstromingswater met een verschil in overstromingshoogte en duur tot gevolg. Naast deze 6 niveaus wordt de helft van de opstelling (4 replica's) blootgesteld aan een normaal getij en het andere deel aan een gecontroleerd, gereduceerd getij (ggg) (Figuur 9.8). Elk niveau wordt verder opgedeeld in een deel met zandgrond en een deel met leemgrond.

Figuur 9.7: rietgroei in de mesocosmos opstelling te Wilrijk.

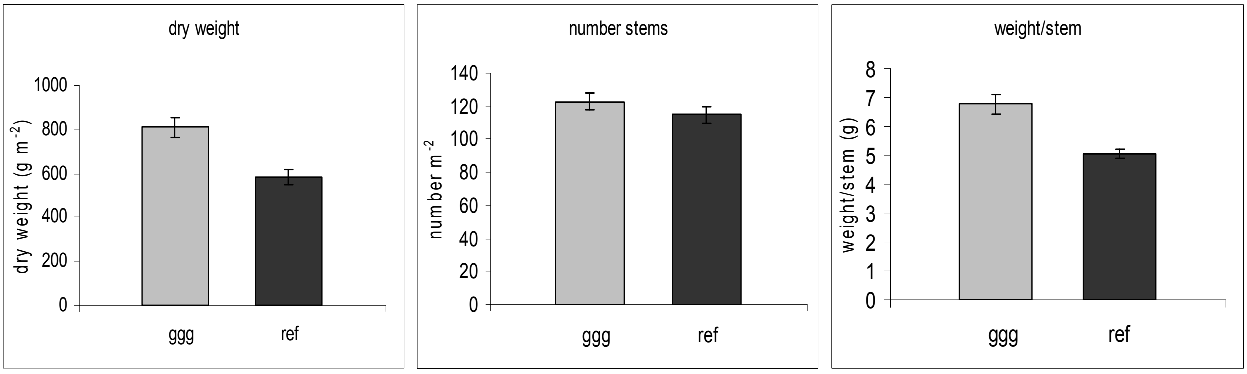


Figuur 9.8: GGG getijdenregime in de mesocosmos te Wilrijk

De planten groeien op 6 verschillende niveaus ten opzichte van het overstromingswater met een verschil in overstromingshoogte en duur tot gevolg. Naast deze 6 niveaus wordt de helft van de opstelling (4 replica's) blootgesteld aan een normaal getij en het andere deel aan een gecontroleerd, gereduceerd getij (GGG). Elk niveau wordt verder opgedeeld in een deel met zandgrond en een deel met leemgrond.

9.2.2. Resultaten

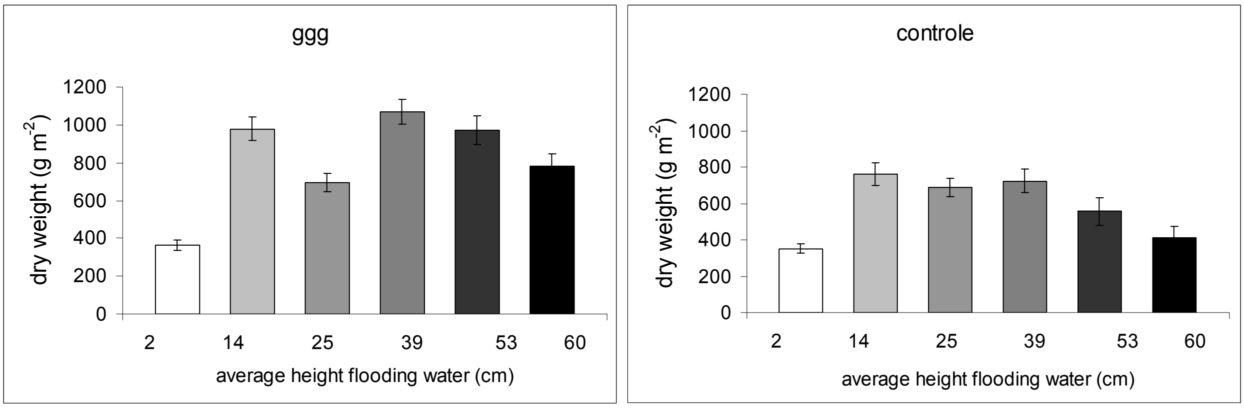
Het riet groei beter bij het ggg in vergelijking met de controle (Figuur 9.9). Hierbij zien we dat dit vooral het gevolg is van gewicht per stengel. Stengels onder het ggg regime zijn gemiddeld 1.74 g zwaarder ten opzichte van de controle. Het verschil in aantal stengels is klein. De totale biomassa per m² neemt met bijna 40% toe bij verandering naar GGG getij



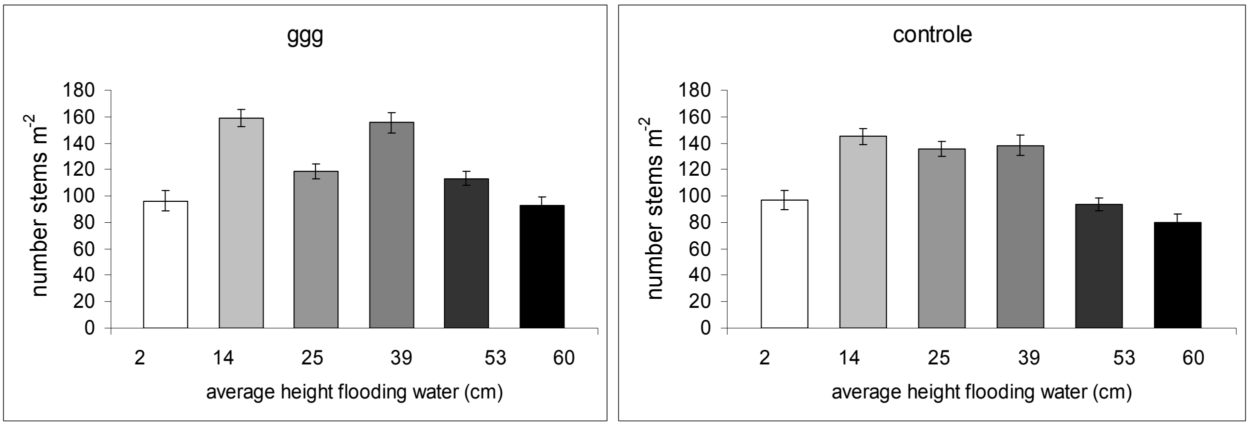
Figuur 9.9: Het gemiddelde over alle niveaus en bodemtypen (n=48) voor de verschillende getij types (controle – ggg). Drooggewicht en aantal stengels per m² en gemiddelde gewicht per stengel.

Het *Phragmites australis* drooggewicht verschilt tussen de overstromingsniveaus (Figuur 9.10). De trend tussen beide overstromingstypes (GGG – controle) is gelijk. Hierbij zien we een lage biomassa bij de laagste overstromingshoogte. Deze droogtestress neemt af met frequentere overstroming. De biomassa neemt daarbij toe tot een optimum om daarna weer af te nemen ten gevolge van stress door te hoge waterstanden en langere overstromingsperioden. Het is opmerkelijk dat de biomassa lager is bij een gemiddelde overstromingshoogte van 25 cm in vergelijking met de overstromingshoogte van 14 en 39 cm. Het verschil is sterker bij het GGG regime. Een mogelijke verklaring zou een verschil in groeistrategie kunnen zijn bij droogte of waterstress waarbij er bij verschillende overstromingskarakteristieken een optimale groei wordt bekomen. Hierbij kan deze verschillende strategie meer uitgesproken zijn onder het GGG regime waarbij de overstromingduur verlengd is. Andere mogelijke verklaringen zijn verschillen die worden veroorzaakt door het ontwerp van het experiment waarbij het niveau “25 cm overstromingswater” aan de tegenovergestelde zijde ligt van de naastliggende niveaus. Hierdoor is eventueel meer stress door wind of een lagere toestroom van nutriënten mogelijk en daardoor een lagere biomassa productie.

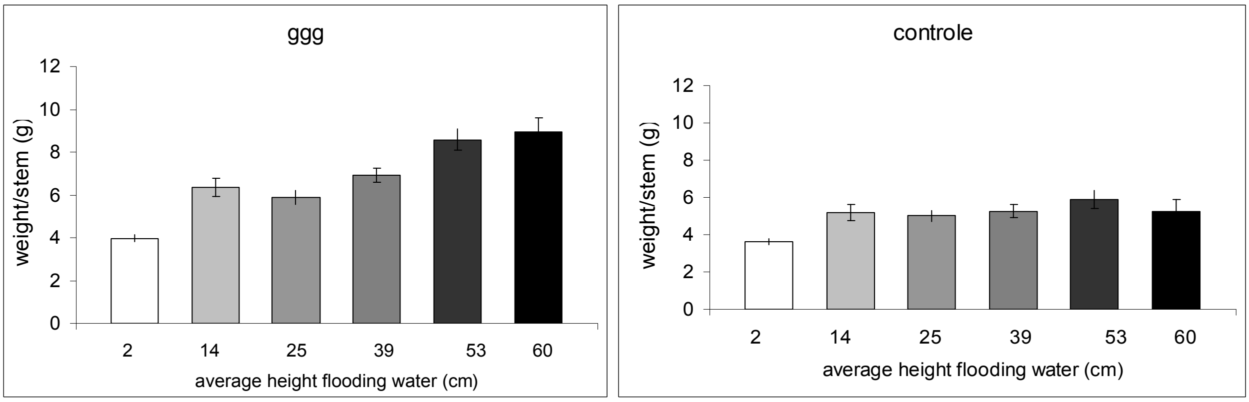
Het gewicht per stengel neemt toe met een grotere overstromingsduur en lengte (Figuur 9.11). Deze trend is meer uitgesproken onder het GGG regime. Vooral bij een grotere overstromingsduur en lengte is het gemiddelde gewicht per stengel veel groter bij ggg (8.9 g) in vergelijking met het controle getij (5.2 g). Het verschil in aantal stengels tussen het ggg en het controle regime is klein (Figuur 9.12). Het grotere totale drooggewicht onder het ggg regime is vooral te wijten aan een groter drooggewicht van individuele stengels onder dit getij in vergelijking met de controle.



Figuur 9.10: Het gemiddelde (n=8) drooggewicht per m² voor de verschillende overstromingsniveaus en voor de verschillende overstromingstypes (GGG – controle).



Figuur 9.11: Het gemiddelde (n=8) aantal stengels per m² voor de verschillende overstromingsniveaus en voor de verschillende overstromingstypes (ggg – controle).



Figuur 9.12: Het gemiddelde (n=8) gewicht per stengel voor de verschillende overstromingsniveaus en voor de verschillende overstromingstypes (ggg – controle).

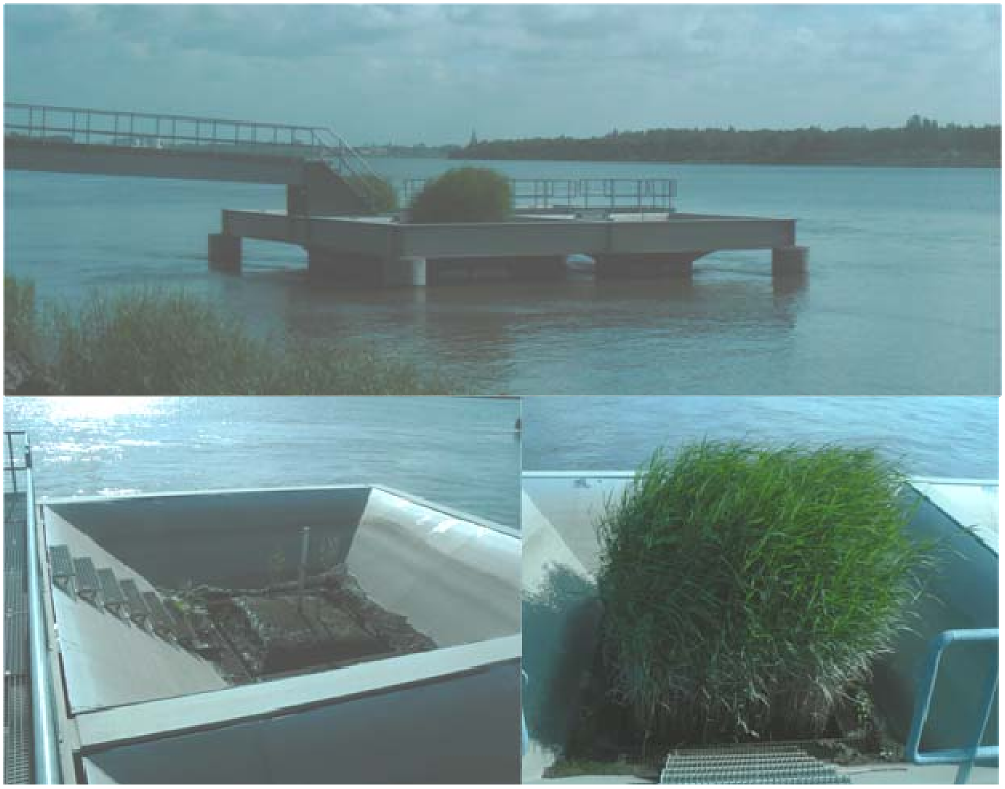
9.2.3. Besluit

De totale biomassa *Phragmites australis* is groter onder de omstandigheden van verlengde overstromingsduur van het gecontroleerde gereduceerde getij. Ook het overstromingsregime heeft een invloed op de biomassa productie. Hierbij zijn er twee optima met biomassa afname bij hogere en lagere overstromingsduur en lengte. Ook dit jaar werkte de regeling van het getij niet over het volledige groeiseizoen zoals geprogrammeerd. Dit kan eventueel een invloed hebben op de resultaten hier weergegeven. Wel zijn dit dezelfde trends die we ook groeiseizoen 2009 terug vinden, bij andere mankementen aan de regeling van het getij.

9.3. Mesocosmos opstelling te Kruibeke

9.3.1. Inleiding

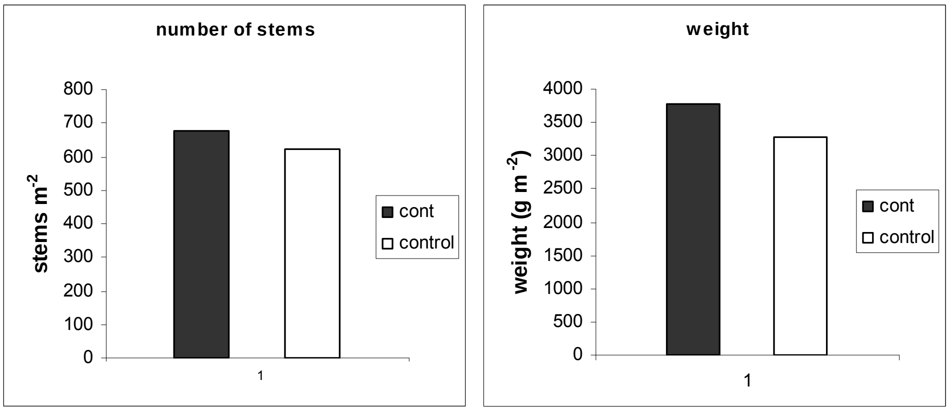
In de mesocosmos te Kruibeke (Figuur 9.13) wordt de invloed van rietgroei op bodemkarakteristieken in een gecontamineerde en niet gecontamineerde bodem onderzocht. In de opstelling in Kruibeke werden 4 containers in een platvorm in de Schelde gevuld met sediment. Twee containers werden gevuld met metaalgecontamineerde bodem, twee met een controle bodem met een lage concentratie aan metalen. Per bodemtype werden er in telkens 1 container rhizomen van *Phragmites australis* aangebracht. De containers overstromen met Scheldewater met een regime vergelijkbaar met schorren in dit deel van de rivier. De groei en opname van metalen wordt nu al enkele jaren opgevolgd. We moeten hierbij rekening houden dat sedimentatie in de opstelling erg snel gaat. Dit heeft tot gevolg dat de bovenste bodemlaag in elke container bestaat uit een laag van 30 cm Schelde sediment met dezelfde samenstelling en gehalten aan metalen.



Figuur 9.13: Foto’s van de mesocosmos opstelling in de Schelde ter hoogte van Kruibeke

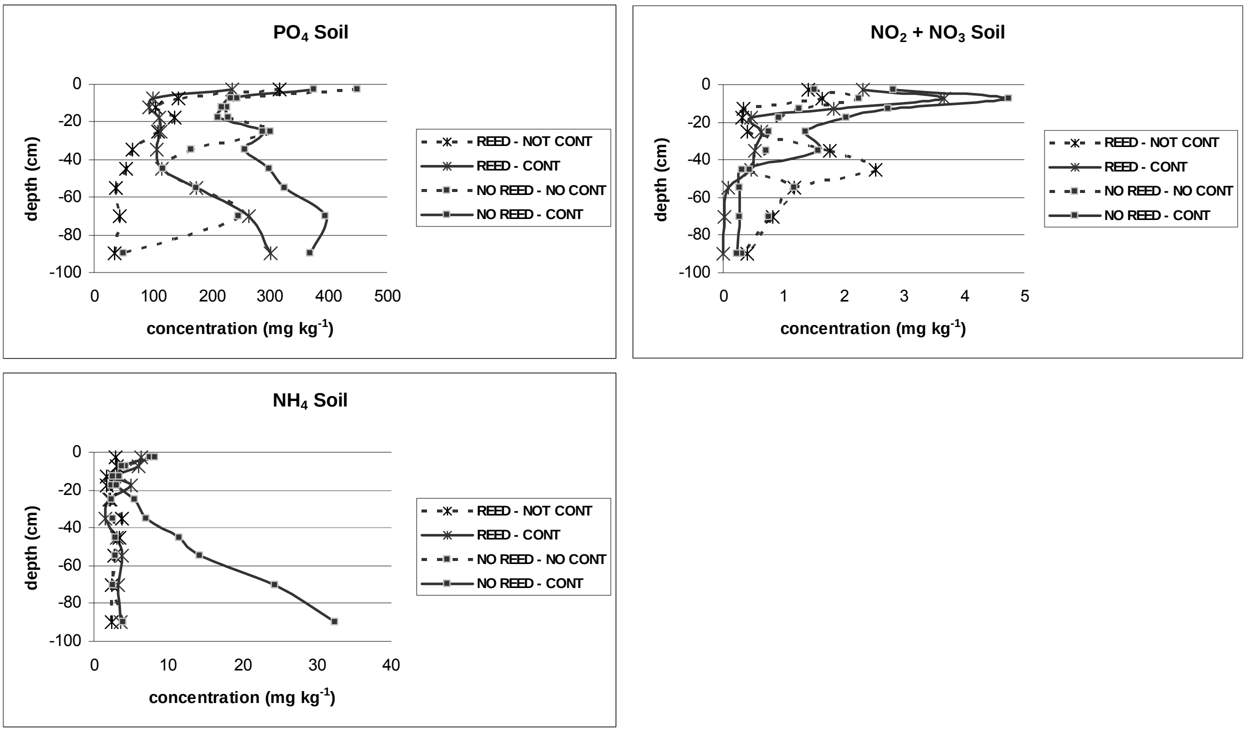
9.3.2. Resultaten

Vergelijkbaar met de resultaten van vorige jaren is ook dit groeiseizoen het gewicht en aantal stengels *Phragmites australis* per m² hoger in de metaal gecontamineerde bodem (Figuur 9.14). In de gecontamineerde bodem is het aantal stengels 624 m^{-2} en het drooggewicht 3767 g m^{-2} ten opzichte van 675 m^{-2} en 3275 g m^{-2} in de controle bodem.



Figuur 9.14: aantal stengels en het gewicht per m² voor de gecontamineerde en controle bodem.

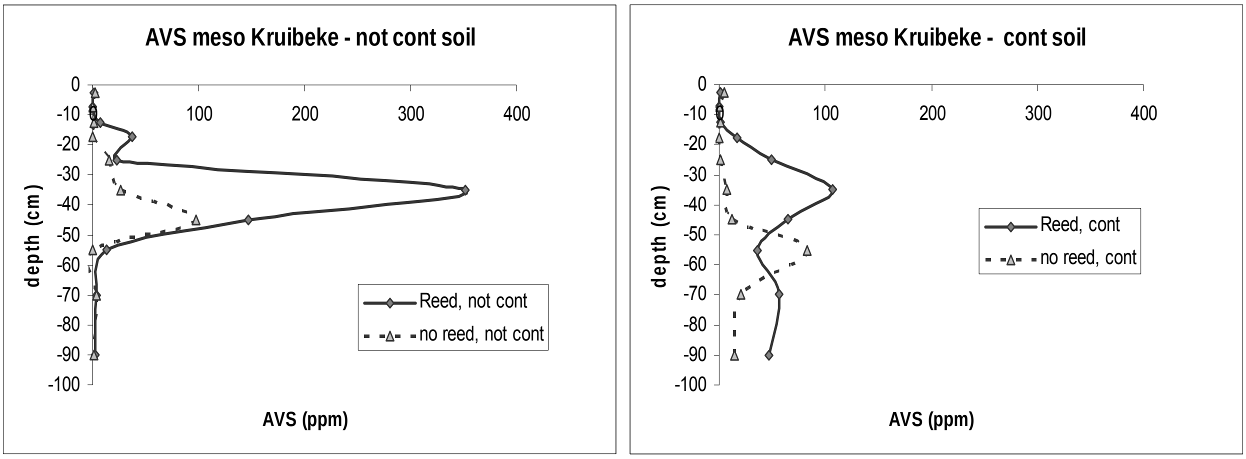
De nutriëntenconcentraties in de bodem variëren in de diepte (Figuur 9.15). De concentraties aan nitraat en nitriet nemen af met toenemende diepte. Deze geoxideerde vormen nemen waarschijnlijk af in de diepere anoxische bodemlagen. De concentraties aan nitraat en nitriet zijn voor alle diepten en alle containers lager dan de concentratie aan ammonium. Concentraties aan ammonium zijn vrij constant behalve voor de container zonder rietgroei met gecontamineerde bodem. Hier neemt de concentratie sterk toe met de diepte. De concentraties aan fosfaat in de bodem vertonen geen duidelijke trend.



Figuur 9.15: De gemiddelde (n=4) PO_4, $NO_2 + NO_3$ en NH_4 concentraties (mg kg^{-1}) in de bodem voor verschillende diepten (cm). Onderbroken lijnen voor de controle bodem, volle lijnen voor de gecontamineerde bodem. Containers met riet (= x) en zonder riet (= ▪)

In de bodem van de mesocosmosopstelling werd de concentratie aan acid volatile sulphides (AVS) bepaald. Dit is het gehalte aan sulfiden dat vrijkomt bij een extractie met 1N HCl. Sulfiden worden gevormd in de bodem bij afbraak van organisch materiaal onder anoxische omstandigheden waarbij sulfaten worden gereduceerd tot sulfiden. Deze sulfiden zijn beschreven als een belangrijke factor in het verlagen van de biobeschikbaarheid van enkele toxische zware metalen onder zuurstof arme omstandigheden. Metalen (vooral Fe en Mn) zullen neerslaan met deze sulfiden en andere metalen (Cd, Cu, Ni, Pb en Zn) zullen worden geïncorporeerd in deze structuur en minder beschikbaar worden voor opname door biota. Tijdens deze extractie komen ook metalen in oplossing (simultaniously extracted metals, SEM). De verhouding van AVS t.o.v. SEM kan een inschatting geven van de metaalbeschikbaarheid. De resultaten van SEM zijn momenteel nog niet beschikbaar.

De concentratie aan AVS in de bodem van de mesocosmosopstelling neemt toe met de diepte tot op een maximum om daarna weer af te nemen (Figuur 9.16). De maximum AVS concentratie ligt tussen de 30 en de 60 cm diepte. De AVS concentratie in de container met riet en niet gecontamineerde bodem stijgt sterker dan in de andere containers en bereikt een maximum waarde van 351.73 (± 276.22) ten opzichte van gemiddelde maximale concentraties onder de 100 ppm AVS in de andere containers. Deze verschillen kunnen te wijten zijn aan verschillen in redoxpotentiaal, verschillen in sulfaat - sulfiet gehalte van de bodem en verschillen in organisch materiaal. We zien grote variatie tussen de verschillende replica’s. Ruimtelijke variatie door verschillen in redoxpotentiaal op korte afstand zijn mogelijk.

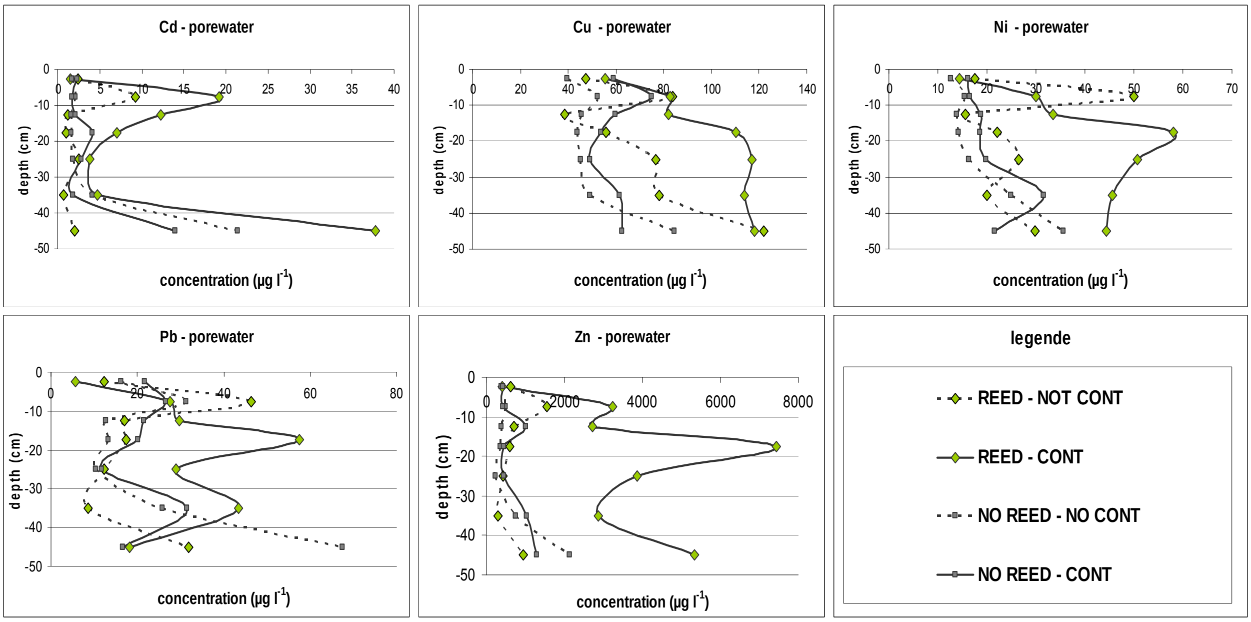


Figuur 9.16: Gemiddelde (n=4) concentratie Acid Volatile Sulphides (AVS) in het sediment op verschillende diepten in de verschillende containers. Onderbroken lijnen voor containers zonder riet, volle lijnen voor containers met riet.

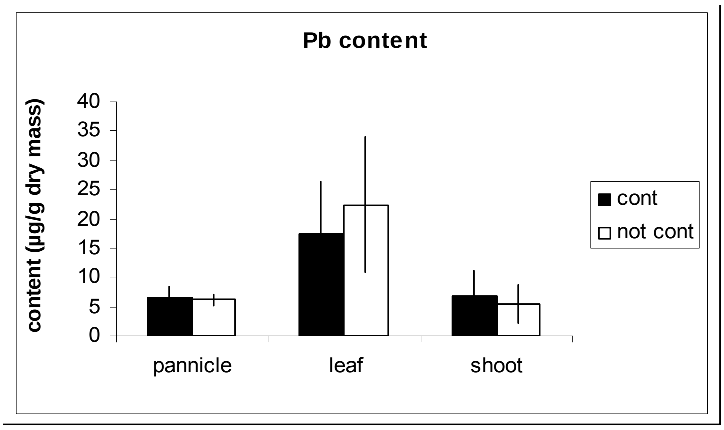
Het poriewater in de bodem (Figuur 9.17) kan worden beschouwd als de opneembare fractie voor planten en weerspiegelt de biobeschikbaarheid van metalen in de bodem. De gehalten aan metalen in de containers zonder riet (controle en gecontamineerde bodems) zijn vrij gelijkend. Dit wil zeggen dat de totale gehalte aan metalen in de bodem onder de omstandigheden van de mesocosmosopstelling niet bepalend is voor de biobeschikbaarheid. De metaalbeschikbaarheid lijkt kleiner in de containers zonder riet. Het is mogelijk dat het verlies aan zuurstof door wortels van de rietplanten de redoxpotentiaal verhoogt. Dit kan oxidatie van sulfiden en vrijkomen van geprecipiteerde metaalionen in het poriewater tot gevolg hebben. Het is wel opmerkelijk dat de beschikbaarheid ook in de bovenste lagen, waar de omgeving waarschijnlijk te oxisch is om sulfiden te vormen, hoger is in de containers met riet. Verder zijn de gehalten aan AVS algemeen ook hoger zijn in de containers met beplanting. Een mogelijke verklaring hiervoor is dat de bodemstalen om praktische redenen werden genomen voor het groeiseizoen. Het is mogelijk dat verlies van zuurstof uit de wortels kleiner is wanneer het riet nog niet groeit. Poriewater stalen werden wel genomen tijdens het groeiseizoen.

Algemeen zijn de gehalten aan metalen hoger in het poriënwater van de gecontamineerde container beplant met riet t.o.v. de andere containers. Ook in de bovenste 30 cm, die is ontstaan

door sedimentatie na het in werking stellen en dus een identieke samenstelling heeft. Dit zou erop kunnen wijzen dat rietgroei een invloed heeft op de allocatie van metalen in de bodem waarbij metalen uit diepere lagen naar het oppervlak worden getransporteerd.
We zien bij cadmium (Cd), lood (Pb) en zink (Zn) in het poriewater een trend waarbij de concentraties laag zijn tussen 20 en 40 cm diepte. Dit kan erop wijzen dat de beschikbaarheid van metalen afneemt door precipitatie met sulfiden op deze diepten (Figuur 9.18).



Figuur 9.17: Gemiddelde (n=3) concentratie metalen (cadmium, koper, nikkel, lood en zink) in het poriewater in de verschillende containers voor verschillende diepten (cm). Onderbroken lijnen voor de controle bodem, volle lijnen voor de gecontamineerde bodem. Containers met riet (= ◊) en zonder riet (= ▪).



Figuur 9.18: concentratie lood in de pluim, blad en stengel van *Phragmites australis* (data oogst 2004) in de gecontamineerde bodem (zwart) en de controle bodem (wit).

De concentratie aan lood, en ook aan andere zware metalen, verschilt niet tussen planten gegroeid in de gecontamineerde en de controle bodem. Riet staat bekend als een excluder en zal zeer weinig metalen transporteren naar bovengrondse plantendelen.

9.3.3. Besluit

In de containers zonder riet lijkt de totale concentratie aan metalen in de bodem geen invloed te hebben op de concentratie aan metalen in het poriewater en dus op metaalbeschikbaarheid. Rietgroei daarentegen lijkt de beschikbaarheid van metalen wel te beïnvloeden. We vinden hogere concentraties aan metalen in het poriewater in de containers met rietgroei. Aan de andere kant vinden we ook hogere gehalten aan sulfiden in de begroeide containers, waarbij we verwachten dat dit beschikbaarheid zou verlagen. De concentraties aan metalen in het poriënwater in de begroeide containers zijn hoger in de gecontamineerde bodem. We verwachten hier dan ook een grotere biobeschikbaarheid. Dit heeft echter geen negatieve invloed op de groei en de opname van metalen in de bovengrondse delen van de rietplanten.

9.4. Sedimentatie en erosie te Lippenbroek

Onderzoek uitgevoerd door perceel 3, partim UA: W. Vandenbruwaene & S. Temmerman.
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9.4.1. Methodiek

In het Lippenbroek wordt de sedimentatie-erosie opgevolgd gebruik makende van zowel lange termijn als korte termijn methodes. De lange termijn methode is een meting over meerdere springtij-doodtijcycli, de korte termijn methode is een meting over 1 eb-vloedcyclus.

Marker horizons, surface elevation tables en opnames van de geulenmorfologie zijn lange termijn metingen. De marker horizons bepalen de dikte van het sediment afgezet op een duidelijk herkenbare laag en dit voor 25 meetlocaties. Door de beperkte nauwkeurigheid van deze methode worden metingen slechts één maal per jaar uitgevoerd. De surface elevation tables meten elke twee maand heel nauwkeurig de hoogteveranderingen van het sedimentoppervlak voor 10 locaties. Naast sedimentatie kan deze methode dus ook erosie meten, iets wat niet mogelijk is met de marker horizons. Tenslotte wordt ook de ontwikkeling van de geulen opgevolgd gebruik makende van een total station. Voor de hoofd- en zijgeul worden zowel de thalweg als verschillende dwarstransecten opgemeten. Vanaf november 2006 werd ook de thalweg van enkele nieuwgevormde, duidelijk ontwikkelde geulen opgemeten waarbij de geuldiepte als meetcriterium werd gebruikt. Vanaf december 2008 werden bijkomend een aantal dwarstransecten opgemeten langsheen de nieuwe geulen. Geulmetingen worden jaarlijks uitgevoerd.
Sediment traps en siphon samplers worden gebruikt voor metingen gedurende één eb-vloedcyclus. De sediment traps worden uitgelegd op 50 verschillende meetlocaties. Ze meten per locatie de hoeveelheid afgezet sediment. De siphon samplers worden geplaatst in de hoofdgeul en bepalen de gesuspendeerde sedimentconcentratie in de waterkolom bij instroom. Deze twee meetmethodes worden 4-jaarlijks uitgevoerd, tijdens de 13-uursmetingen.

Tabel 9.2 geeft een overzicht van de data waarop de verschillende metingen zijn uitgevoerd. Voor een meer gedetailleerde beschrijving van de meetmethodes wordt verwezen naar het OMES eindrappport van 2007 (Vandenbruwaene en Temmerman, 2007).

Tabel 9.2: Overzicht van de meetdata voor de verschillende methodes. Elk puntje is een meetcampagne. SET = surface elevation table, MH = marker horizon, TS = total station (opmeten geulmorfologie), ST = sediment trap en SS = siphon sampler

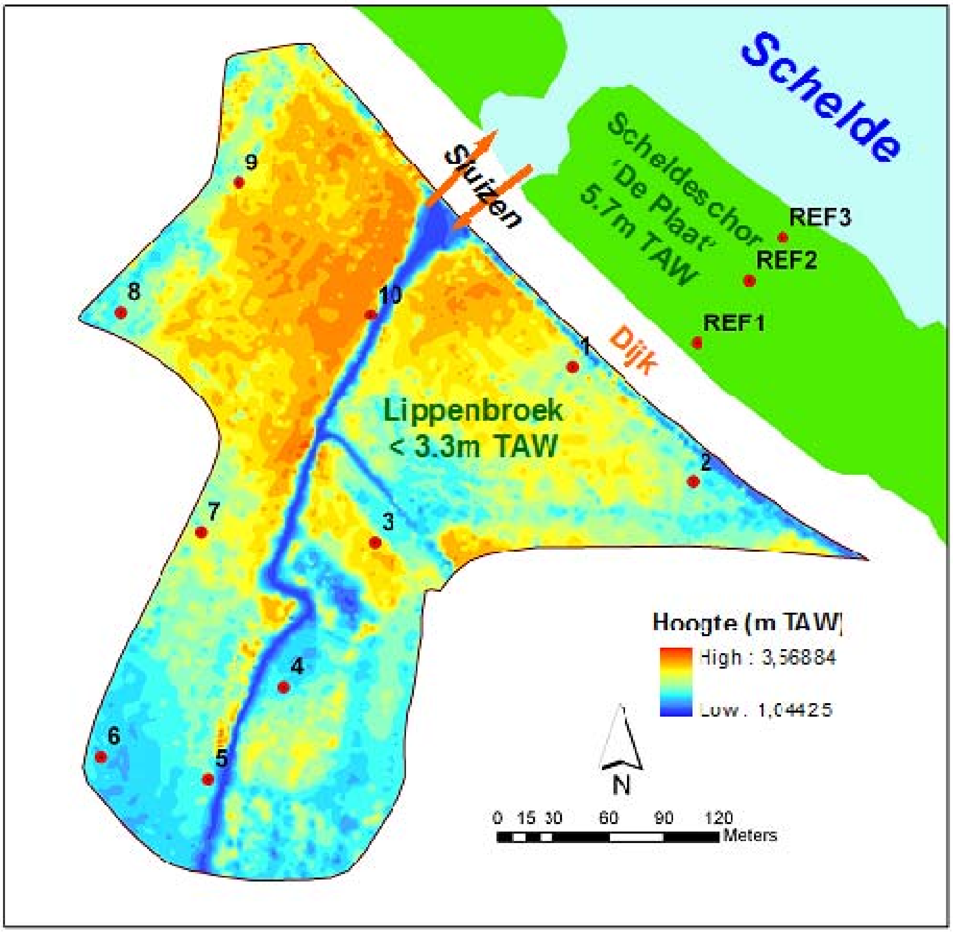
Methode	SET	MH	TS	ST	SS
Data	• mrt06 • 1, 2 jul06 • 21, 22 sep06 • 17, 19, 22 jan07 • 5, 7 mrt07 • 7, 8 mei07 • 16, 17 jul07 • 18, 19 sep07 • 30, 31 ok07; 1nov07 • 22, 23 jan08 • 20, 21 mrt08 • 26, 27 mei08 • 15, 16 jul08 • 1, 2, 3 sep08 • 17, 18 nov08 • 3, 4 feb09 • 4, 6 mrt09 • 14, 15 mei09 • 13, 14 jul09 • 17, 21, 22sep09	• 1, 2 feb 07 • 18, 19, 21 jul07 • 27feb09; 2, 3, 4, 6 mrt09	• 17 feb06; 2mrt06 • 9, 10 nov06; 11, 20, 21 dec06 • 12 tot 16 nov07 • 1 tot 3dec08	• 30aug06 • 11sep06 (13-uurs) • 25 ok06 (13-uurs) • 18mrt07 • 20mrt07 (13-uurs) • 5jun07 (13-uurs) • 1okt07 (13-uurs) • 8mei08 (13-uurs) • 3jun08 (13-uurs) • 6okt08 (13-uurs) • 4dec08 (13-uurs)	• 30aug06 • 11sep06 (13-uurs) • 18mrt07 • 5jun07 (13-uurs) • 1okt07 (13-uurs) • 8mei08 (13-uurs) • 3juni08 (13-uurs) • 4dec08 (13-uurs)

9.4.2. Sedimentatie in het Lippenbroek

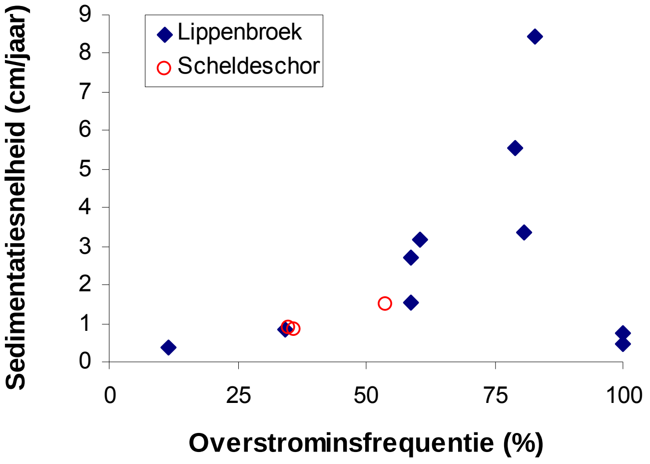
In totaal zijn er in het Lippenbroek 10 meetlocaties waar de lange termijn hoogteveranderingen van het sedimentoppervlak continu worden opgevolgd aan de hand van zogenaamde “Surface Elevation Tables” (SETs). Ter vergelijking zijn er 3 SET meetlocaties in het aangrenzende, buitendijkse schor ‘De Plaat’ (Figuur 9.19).

De sedimentatiesnelheid in het Lippenbroek vertoont grote ruimtelijke variaties die zijn gerelateerd aan ruimtelijke variaties in overstromingsfrequentie (Figuur 9.20). Laag gelegen delen van het Lippenbroek overstromen veel vaker en hogen daarom het snelste op (tot 8,4 cm/jaar tussen maart 2006 en september 2009, figuur 3, site 4), terwijl sommige hoog gelegen delen van het Lippenbroek zeer weinig overstroomd worden en daarom nauwelijks of zeer traag ophogen (< 0.5 cm/jaar, figuur 3, site 10). We benadrukken dat deze relatie tussen sedimentatiesnelheid en overstromingsfrequentie dezelfde is voor het Lippenbroek en voor het schor (Figuur 9.20). Met andere woorden de sedimentatiesnelheid in het Lippenbroek en op het schor is vergelijkbaar voor locaties met dezelfde overstromingsfrequentie. Uit vorige sedimentatiestudies (Temmerman et al. 2003; 2004; 2007) weten we dat sedimentatiesnelheden van 1,5 tot 10 cm/jaar op de schorren in de Zeeschelde normaal zijn, en gerelateerd zijn aan variaties in hoogteligging en overstromingsfrequentie. Het Lippenbroek gedraagt zich dus op een gelijkaardige manier als een natuurlijk schor.

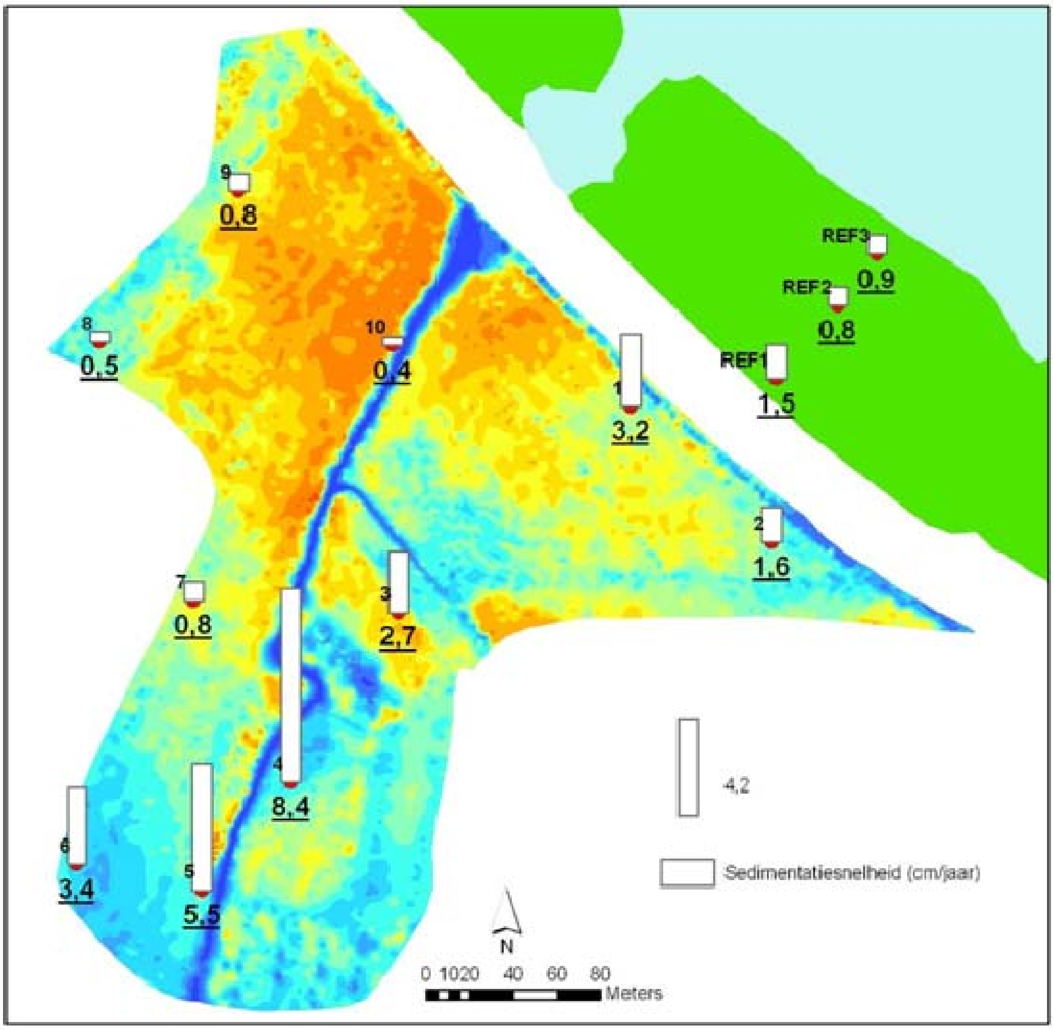
Twee meetlocaties (8 en 9) in het Lippenbroek staan continu onder water en daardoor wordt hier geen verband gevonden tussen sedimentatiesnelheid en overstromingsfrequentie (Figuur 9.20, locaties met 100% overstromingsfrequentie). Beide meetlocaties liggen in een depressie die slecht wordt gedraineerd door een verhoging in de topografie. Enkel bij hoge springtijen is er input van nieuw water en sediment, en als gevolg zijn de sedimentatiesnelheden er laag.



Figuur 9.19: Het Lippenbroek en het aangrenzend Scheldeschor 'De Plaat', met aanduiding van de SET locaties waar sedimentatie en erosie continu wordt opgevolgd. De morfologie van het Lippenbroek is weergegeven op basis van het Digitaal Hoogtemodel Vlaanderen voor de voormalige polder (dus vóór de introductie van de gereduceerde getijwerking in maart 2006)

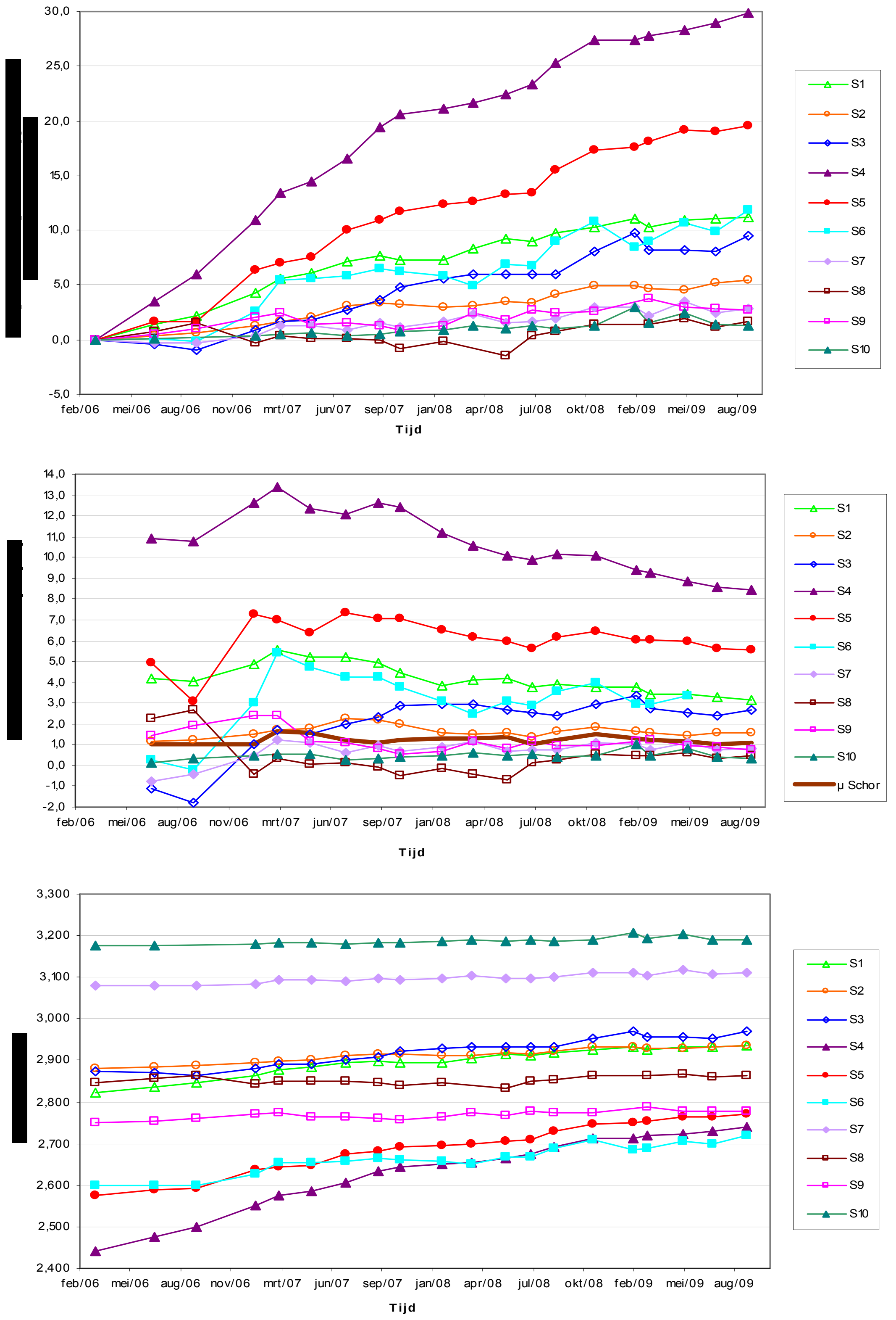


Figuur 9.20: Relatie tussen de sedimentatiesnelheid (cm/jaar) en de overstromingsfrequntie (% van hoogwaters), zowel voor de meetlocaties in het Lippenbroek als het schor "De Plaat". De sedimentatiesnelheden zijn berekend voor de periode maart 2006 t/m september 2009. Overstromingfrequenties naar Maris et al. (2008)



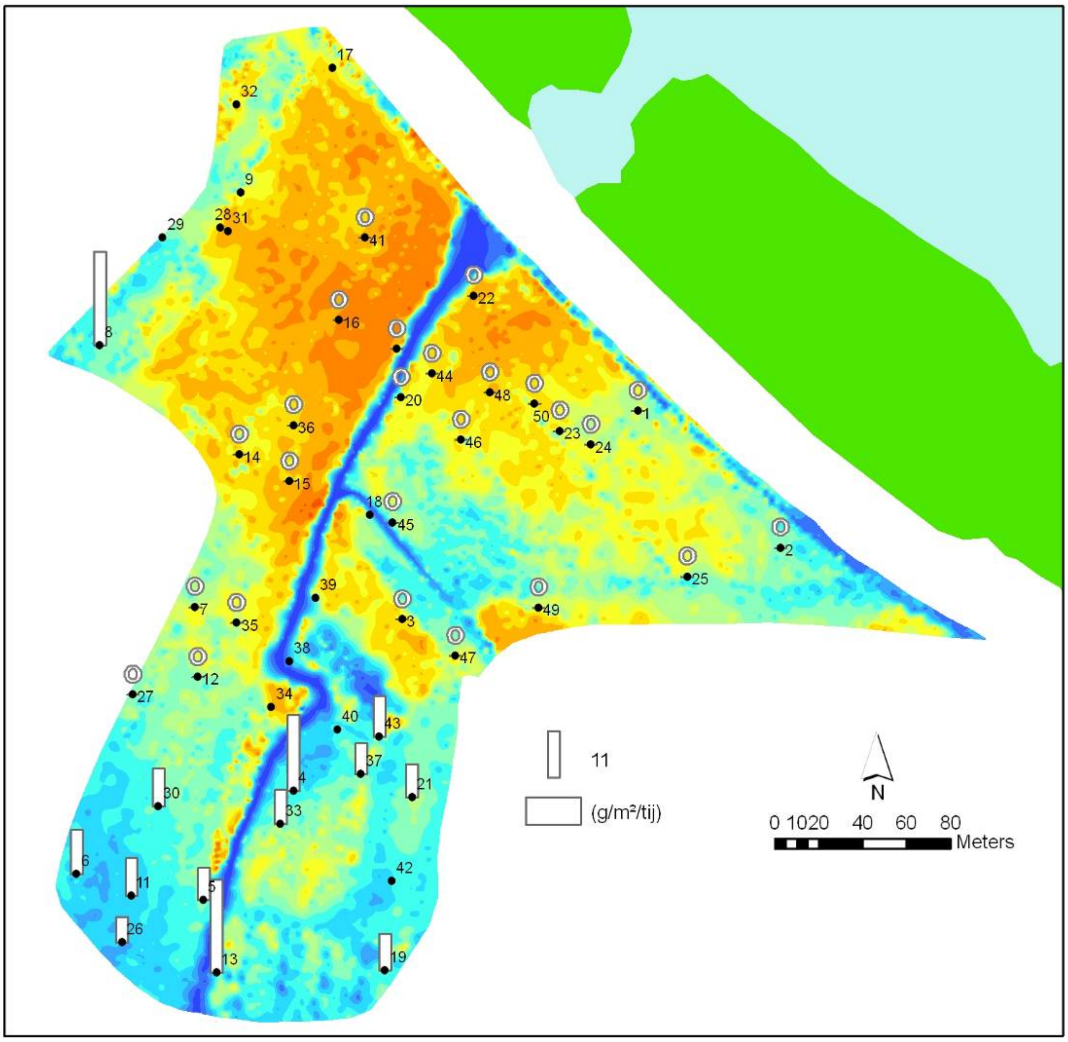
Figuur 9.21: Ruimtelijke variatie in sedimentatiesnelheid (cm/jaar). Onderlijnde cijfers zijn de sedimentatiesnelheden, kleine cijfers de meetlocaties. Sedimentatiesnelheden zijn verrekend over de periode tussen maart 2006 en september 2009

Het frequenter overstromen van laag gelegen sites zorgt ervoor dat er meer sediment wordt afgezet op deze sites (Figuur 9.22a). Dit resulteert in een afvlakking van de topografie (Figuur 9.22c). Het heeft echter ook tot gevolg dat er na verloop van tijd een duidelijke vermindering is in overstromingsfrequentie, en bijgevolg een vermindering van sedimentatiesnelheid (e.g. site 4 en 5, Figuur 9.22b). Voor de hoogst gelegen sites is de ophoging klein, waardoor de sedimentatiesnelheid vrij constant is (e.g. site 7 en 10, Figuur 9.22b).



Figuur 9.22: Vergelijking van de sedimentatie in de tijd thv de 10 sites. Sedimentatie weergegeven als relatieve hoogteverandering van het sedimentoppervlak (a), sedimentatiesnelheid (b) en absolute hoogteverandering (c)

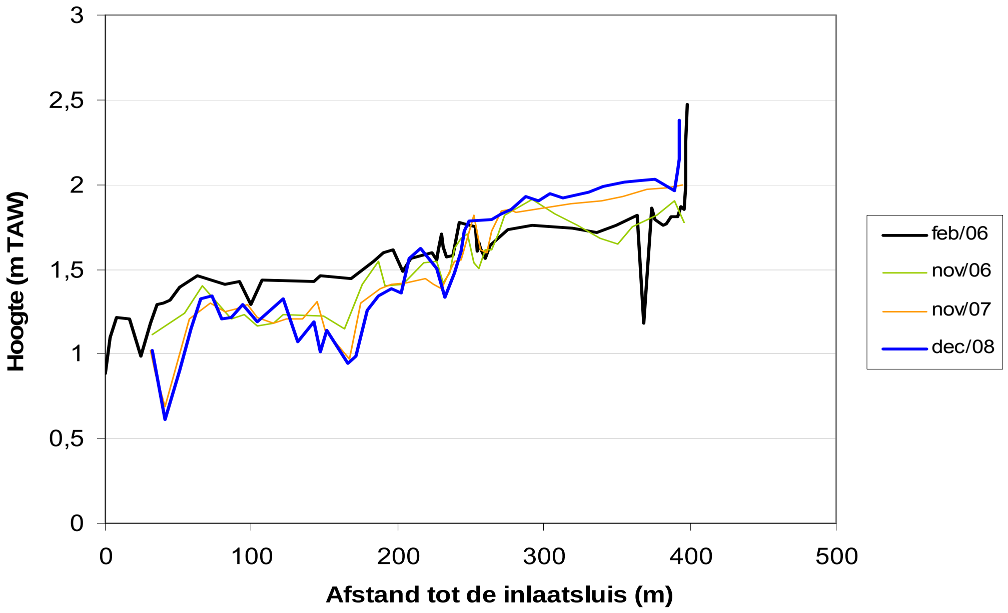
Naast de opvolging van de lange termijn sedimentatie worden ook metingen gedaan over 1 getijcyclus (op 50 locaties). Op die manier kan worden nagegaan wat het effect is van hoogwaterpeil, inkomende sedimentconcentraties, hoogteligging, afstand tot het geulnetwerk en afstand tot de inlaatsluis op de hoeveelheden afgezet sediment. In Figuur 9.23 wordt het ruimtelijk sedimentatiepatroon weergegeven voor de laatst uitgevoerde 13-uursmeting. Het is duidelijk dat dit een meting is tijdens doodtij waarbij enkel de laagst gelegen delen achteraan het Lippenbroek overstroomd worden. De hoeveelheden afgezet sediment zijn laag. Voor een volledig overzicht van ruimtelijke sedimentatiepatronen over 1 getij wordt verwezen naar de vorige OMES rapporten (Vandenbruwaene en Temmerman 2007; 2008).



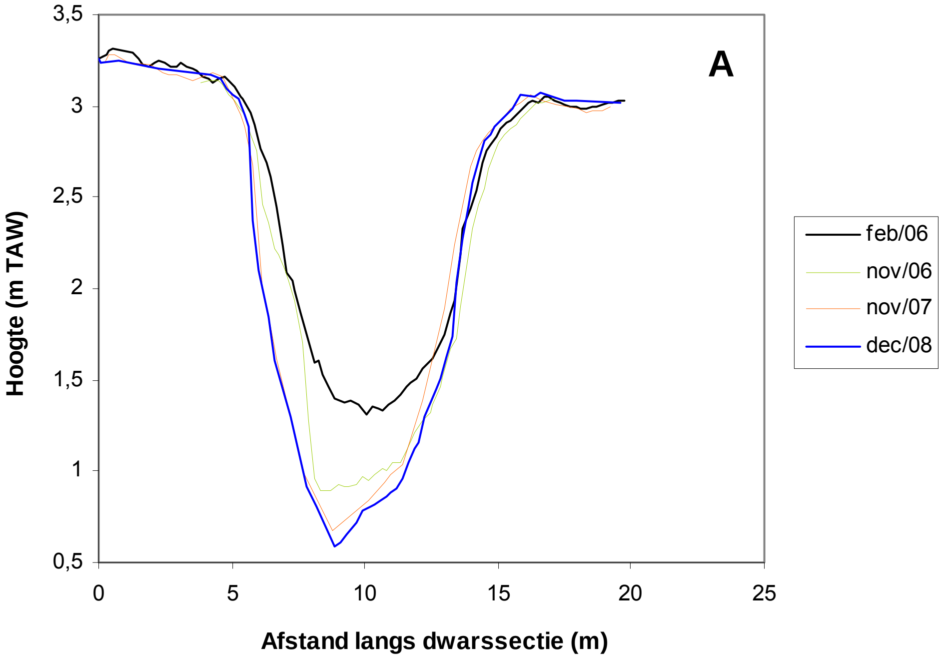
Figuur 9.23: Ruimtelijk sedimentatiepatroon voor het getij van 4 december 2008 (g/m²/tij). Meetlocaties met het 'O' symbool zijn niet overstroomd gedurende dit getij

9.4.3. Erosie in het Lippenbroek

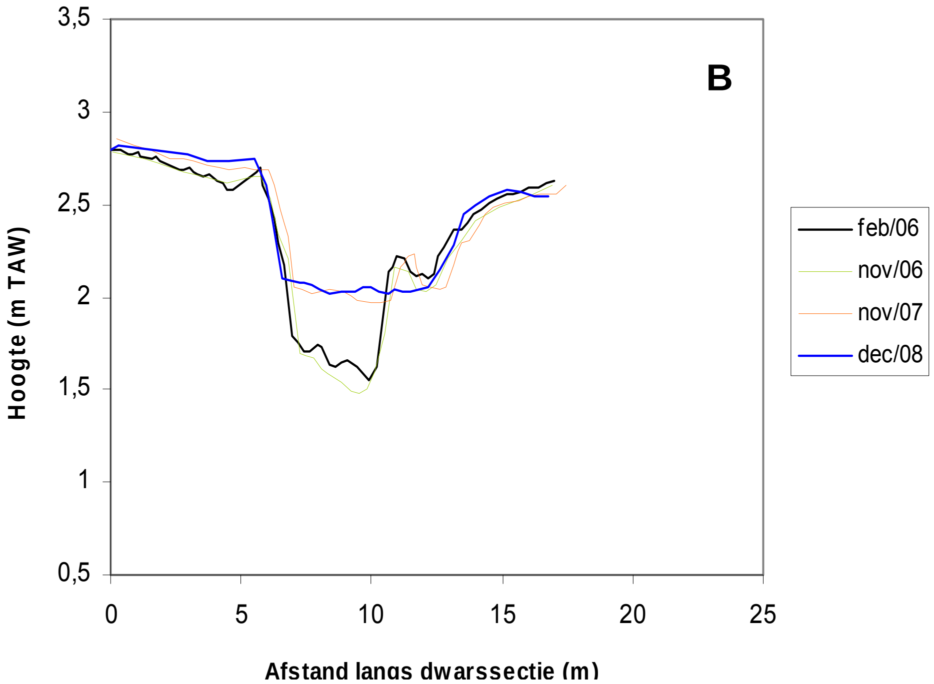
In de voormalige Lippenbroekpolder komen 2 grachten voor, een min of meer NZ lopende hoofdgracht en een kleinere daarop aansluitende zijgracht (Figuur 9.19). Figuur 9.24 geeft de morfologische ontwikkeling van het lengteprofiel van de hoofdgeul weer. Het is duidelijk dat er erosie optreedt in het deel van de gracht dat dicht tegen de inlaatsluis is gelegen, en dat er sedimentatie optreedt in het deel van de gracht achteraan in de polder op een grotere afstand van de sluis. De erosie dicht tegen de inlaatsluis komt eveneens tot uiting in de dwarssecties (Figuur 9.25a). Dicht tegen de sluis is de thalweg op een kleine 3 jaar ongeveer 0.8 m dieper geworden door erosie. Het grootste deel van deze uitdieping vond plaats in het eerste jaar na de inwerkingtreding van het GGG. Anderzijds is er tot 0.5 m sediment afgezet in de gracht achteraan in het Lippenbroek, ook hier is de verandering het grootst geweest in het eerste jaar (Figuur 9.25b). Deze veranderingen in de dimensies van de grachten is duidelijk gerelateerd aan de getijdenwerking: het drainagegebied van de gracht dicht bij de sluis is groter dan van de gracht achteraan in de polder, zodat de vloed- en ebdebieten groter zijn dichter bij de sluis. Bijgevolg is de gracht dichtbij de sluis verder uitgeschuurd en achteraan in de polder gedeeltelijk opgevuld. Vergelijking van de geuldimensies in het Lippenbroek met geuldimensies op natuurlijke schorren in de Zeeschelde geeft duidelijk aan dat de geulen in het Lippenbroek nog in volle ontwikkeling zijn en nog geen evenwichtsituatie hebben bereikt.



Figuur 9.24: Lengteprofielontwikkeling van de hoofdgeul in het Lippenbroek, voor de periode maart 2006 t/m december 2008.



Figuur 9.25: Dwarsprofielontwikkeling van de hoofdgeul in het Lippenbroek, voor de periode maart 2006 t/m december 2008. (9.25a) dwarstransect 1 dicht tegen de inlaatsluis, waar duidelijk erosie van de geul is opgetreden.



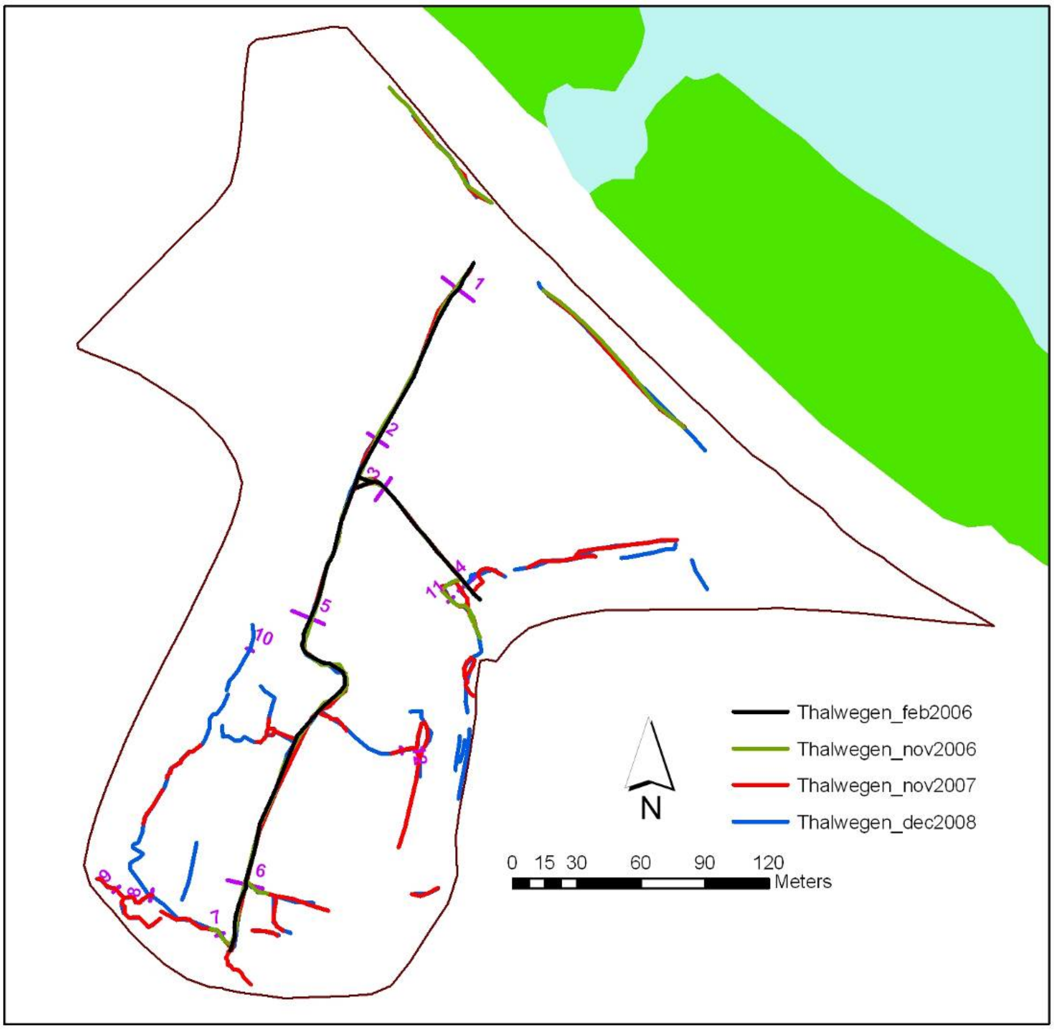
Figuur 9.25: Dwarsprofielontwikkeling van de hoofdgeul in het Lippenbroek, voor de periode maart 2006 t/m december 2008. (9.25b) dwarstransect 6 verder van de inlaatsluis achteraan in het Lippenbroek, waar duidelijk opvulling van de geul is opgetreden. De locaties van de dwarstransecten zijn aangeduid in Figuur 9.26

Naast de morfodynamiek in de hoofd- en zijgeul ontstonden er ook nieuwe geultjes. De ruimtelijke uitbreiding van het geulnetwerk in de tijd is voorgesteld in Figuur 9.26. In december 2008 was er ruim 1 km aan nieuw gevormde geultjes waarvan het grootste deel zich ontwikkelde in de lager gelegen delen van het Lippenbroek. Deze nieuw gevormde geulen zijn typisch vrij breed (ca. 1 m) en relatief ondiep (0.1 tot 0.35 m diep). Vergelijking van geuldichtheden (aantal geulen per oppervlakte) tussen het Lippenbroek en natuurlijke schorren geeft ook hier aan dat het geulnetwerk nog geen evenwicht heeft bereikt en zich hoogstwaarschijnlijk nog verder zal uitbreiden de komende jaren.

9.4.4. Bespreking en conclusies

Het is duidelijk dat sedimentatie belangrijk is op het Lippenbroekplatform. De sedimentatie is variabel in de ruimte, laag gelegen sites hebben hogere overstromingsfrequenties en dus hogere sedimentatiesnelheden, maar is ook variabel in de tijd, de sterke ophoging van laag gelegen sites zorgt voor een afname van de overstromingsfrequentie en sedimentatiesnelheid. Belangrijk is dat Lippenbroeksites met overstromingsfequenties vergelijkbaar met het aangrenzende schor ook vergelijkbare sedimentatiesnelheden kennen. Het berekenen van een gemiddelde sedimentatiesnelheid voor het volledige Lippenbroek is vrij complex. Hiervoor wordt verwezen naar een synthesenota uitgevoerd door de Universiteit Antwerpen, in opdracht van Waterwegen & Zeekanaal N.V. Afdeling Zeeschelde (Temmerman, 2009).

Ondanks de sedimentatie op het Lippenbroekplatform is er echter ook belangrijke erosie in het gebied. Er is uitdieping in de bestaande hoofd- en zijgeul en er heeft zich een netwerk van nieuwe geultjes ontwikkeld. De verhouding van sedimentatie en erosie in het Lippenbroek (bepalend voor het waterbergend vermogen), krijgt de volgende maanden verdere aandacht.



Figuur 9.26: Planimetrische evolutie van het geulnetwerk in het Lippenbroek voor de periode maart 2006 t/m december 2008. Paarse symbolen duiden de locaties van de dwarssecties aan (12 in totaal).

9.4.5. Referenties

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9.5. Sediment en zwevende stof te lippenbroek

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9.5.1. Introduction

Parcel 3 deals with the sedimentological study of the Flemish part of the Schelde-estuary. In the Lippenbroek area, attention is paid to the in- and out flowing sediment and its relation to the sediment behavior in the river. Therefore full tide measurements and sample analyses were performed in the mouth of the main creek (bridge 1) crossing the Lippenbroek area from northeast to southwest.

9.5.2. Field measurements of suspended matter

The Lippenbroek area is flooded twice a day. Inflow starts approximately 1 hour before high water in the river (at a tidal height of 4.75 m) and lasts for more than 2 hours. The outflow, through an outlet gate situated beneath the inlet gate, starts when the tidal height is below 3.5 m in the river and lasts for more than 4 hours. During this process suspended matter is transported from the river into the Lippenbroek area. Part of it settles on the marsh surface resulting in a decreasing storage capacity of the area. In order to understand this sedimentation process knowledge is required from the sediment movement in the river in front of the inlet gate, through the in- and outlet gates, and in the Lippenbroek area itself. This part of the parceel 3 deals with the movement of sediments both in the Schelde in front of the Lippenbroek area and through the main creek that crosses the Lippenbroek area from northeast to southwest. The sampling site was located on a bridge (bridge 1), and the suspended matter concentration (SMC) was monitored.

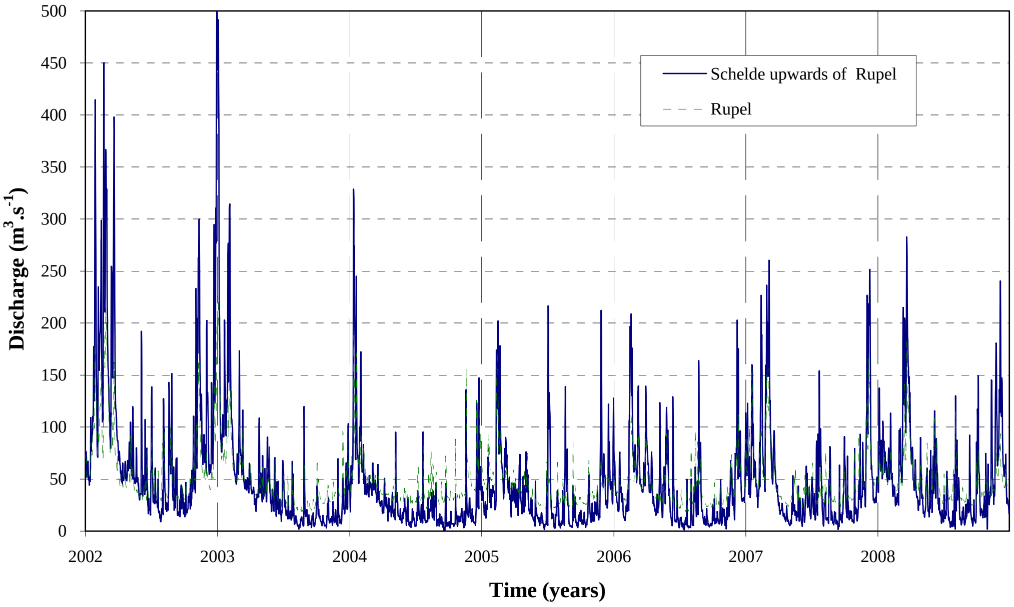
Tabel 9.3: Overview of full tide measurements in the Lippenbroek area with indication of tidal range (at Driegoten or Tielrode) and discharges of the upper Schelde at Melle for the days of measuring. Data of tide and river discharge by courtesy of E. Taverniers, Flanders Hydraulics Research, Antwerp.

date	bridge		Schelde		creek	tidal range	river discharge (Melle) $m^3.s^{-1}$	
	1	2	L	R		m	1 day	7 days
16-05-2006	✓	✓				5.92	9.3	31.5
03-07-2006	✓		✓			5.35	9.5	6.6
11-09-2006	✓					6.21	4.7	6.5
25-10-2006	✓		✓		✓	5.89	19.3	13.5
21-03-2007	✓					6.63	81.8	49.8
05-06-2007	✓		✓	✓		5.56	5.0	14.7
01-10-2007	✓		✓			5.90	47.5	22.4
08-05-2008	✓		✓			6.22	13.6	40.3
06-10-2008	✓	✓			✓	5.15	111.3	63.2
04-12-2008	✓					5.39	91.2	110.2

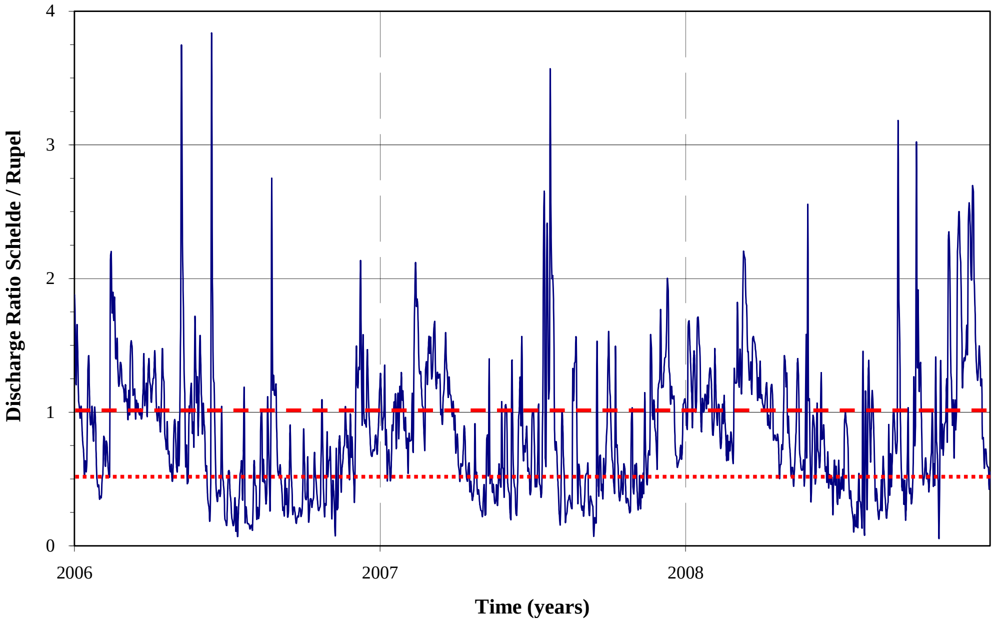
Tabel 9.3 lists the measuring campaigns in the Lippenbroek area. In 2008, three full tide measurements were performed. The river discharge of the Schelde is presented in the Figuur 9.27. The river discharges at Melle on the days of the measurements were appreciably higher than the ones in the previous years. It can be noticed that during the period 2006-2008 the discharge of the Schelde was often below the discharge of the Rupel (Figuur 9.27), and even sometimes less than 50% of the discharge of the Rupel. The river discharge ratio Schelde to Rupel is shown in the Figuur 9.28 and Figuur 9.29 and Tabel 9.4.

Tabel 9.4: Annual averages of river discharge of the upper Schelde, the Rupel, the total discharge of Schelde and Rupel at Schelle and the ratio Schelde/Rupel.

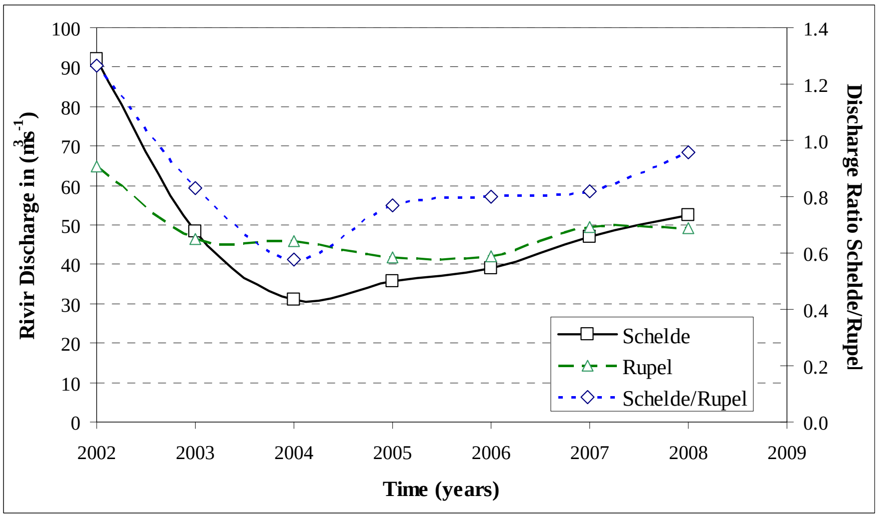
Average Annual Discharges				
Year	Schelde $m^3 s^{-1}$	Rupel $m^3 s^{-1}$	Schelle $m^3 s^{-1}$	Ratio Schelde/Rupel
2002	92	65	157	1.27
2003	48	46	95	0.83
2004	31	46	77	0.58
2005	36	42	77	0.77
2006	39	42	81	0.80
2007	47	49	96	0.82
2008	52	49	102	0.96



Figuur 9.27: River discharge of the upper Schelde and the Rupel (by courtesy of E. Taverniers, Flanders Hydraulics, Antwerpen).



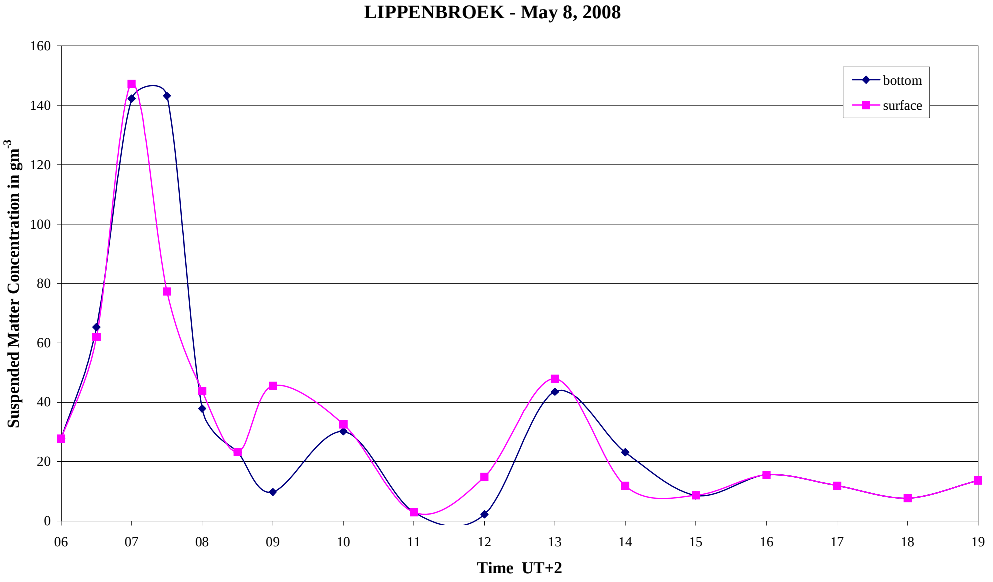
Figuur 9.28: The river discharge ratio Schelde at Schelle to Rupel at Boom (by courtesy of E. Taverniers, Flanders Hydraulics, Antwerpen).



Figuur 9.29: River discharges of the upper Schelde, the Rupel, and the Schelde to Rupel discharge ratio between 2002 and 2008.

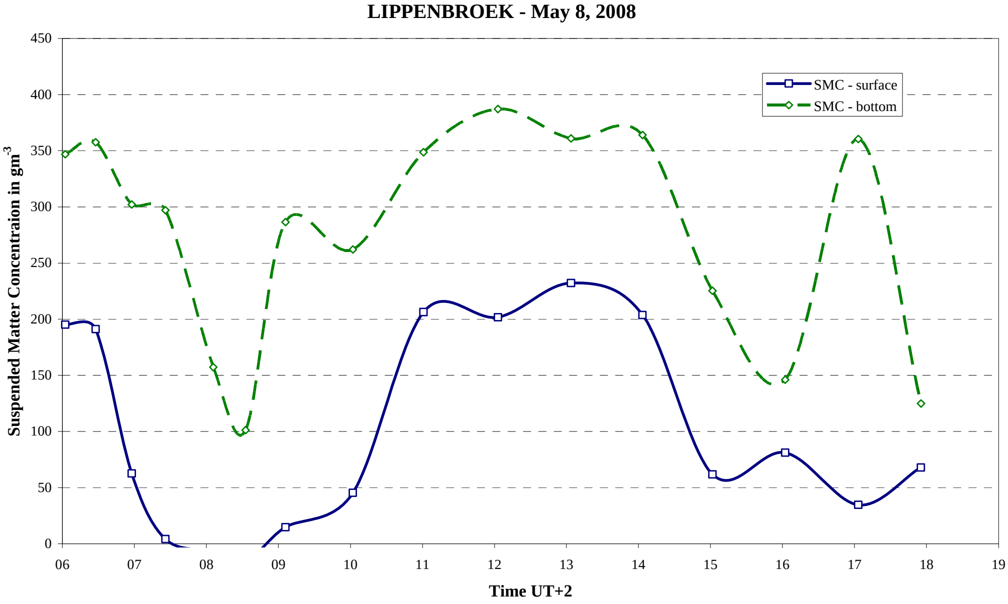
9.5.3. Measurements on May 8, 2008

The measurement campaign of May 8, 2008, started at 6:00 and ended at 19:00 (Figuur 9.30). The tidal range was 6.22 m and the river discharge at Melle was extremely low (13.6 $m^3 s^{-1}$). When the inflow started, the SMC was very low, below 10 g m^{-3}. It increased rapidly to a maximum of 150 g m^{-3} around 7:00 for a short time. It then decreased progressively to approximately 20 g m^{-3} remaining low till the outflow at around 12:00. During the outflow the SMC increased to approximately 40 g m^{-3} at 13:00 and then decreased to below 10 g m^{-3} and remained stable till the next inflow at around 19:00 (Figuur 9.30).



Figuur 9.30: Suspended matter concentration near the surface and near the bottom in the Lippenbroek

During this tidal cycle measurement, the SMC was also measured in the river Schelde in front of the tidal creek leading to the inlet gate (Figuur 9.31). During the inflow period (06:00 till 8:30) the SMC near the surface dropped from 300 g m^{-3} to very low values. It can be noticed that the SMC in the creek increased very fast during the inflow and reached more or less the same value as in the Schelde around 7:00. After that the SMC in the creek decreased although less rapidly than in the river Schelde.

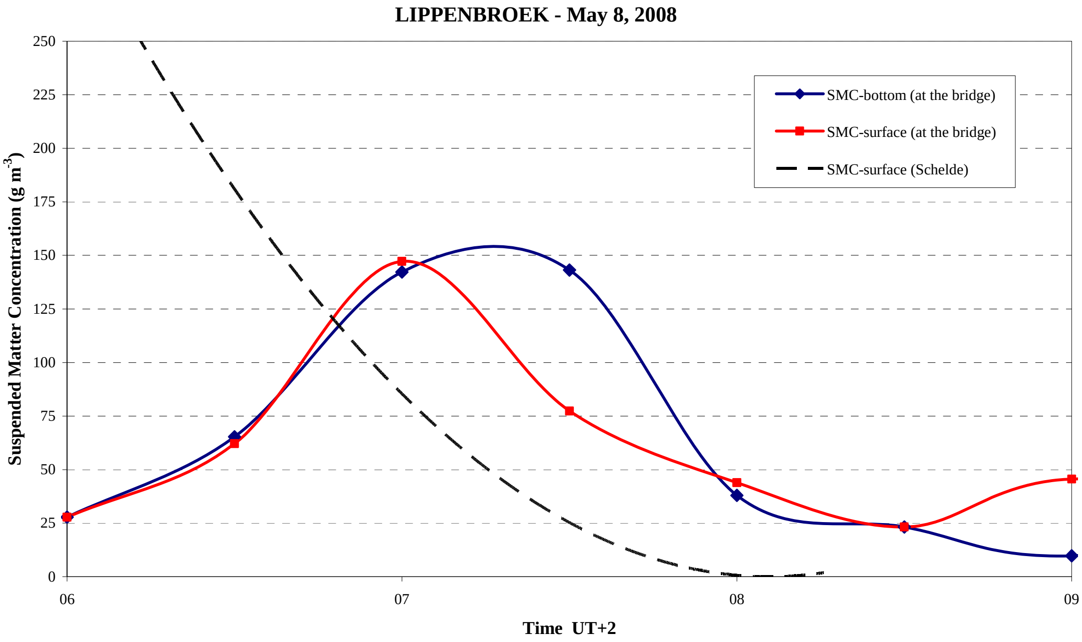


Figuur 9.31: Near surface and near-bottom suspended matter concentrations in the river Schelde in front of the inlet gate to the Lippenbroek area

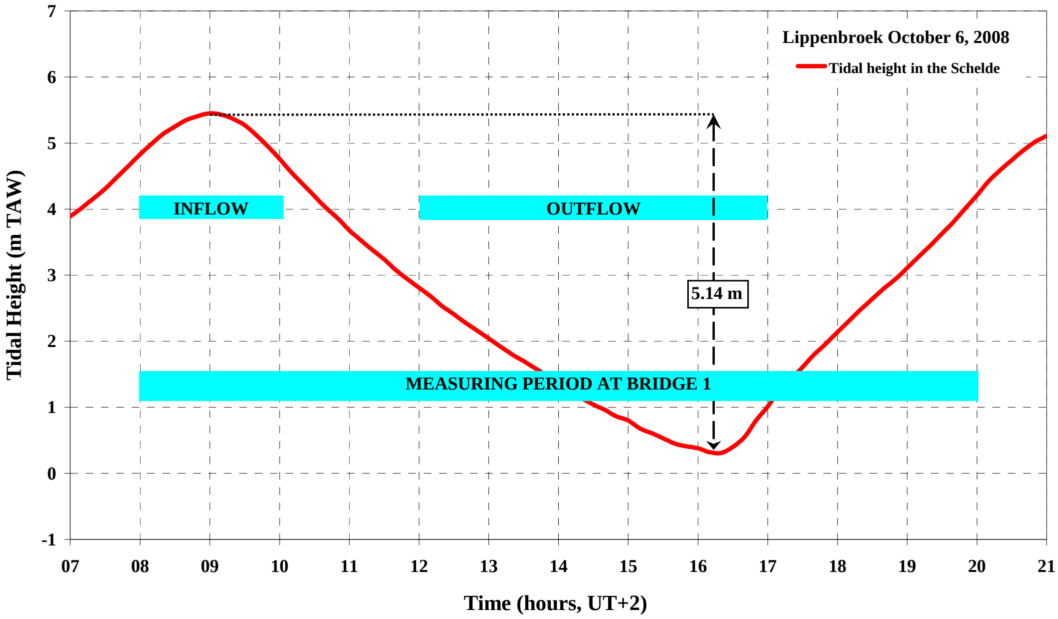
It is observed that the SMC near the bridge increased with the inflowing water reaching a maximum around 7:00 and then decreased rapidly (Figuur 9.32). Near the bottom the high SMC values remained for a longer period than near the surface. It can also be observed that the SMC near the bridge was initially much lower than the SMC near the surface in the river Schelde but came to a very close value as in the river Schelde when it was at its maximum (around 7:00). The same process was observed in previous measurements and reported in our early OMES report.

9.5.4. Measurements on October 6, 2008

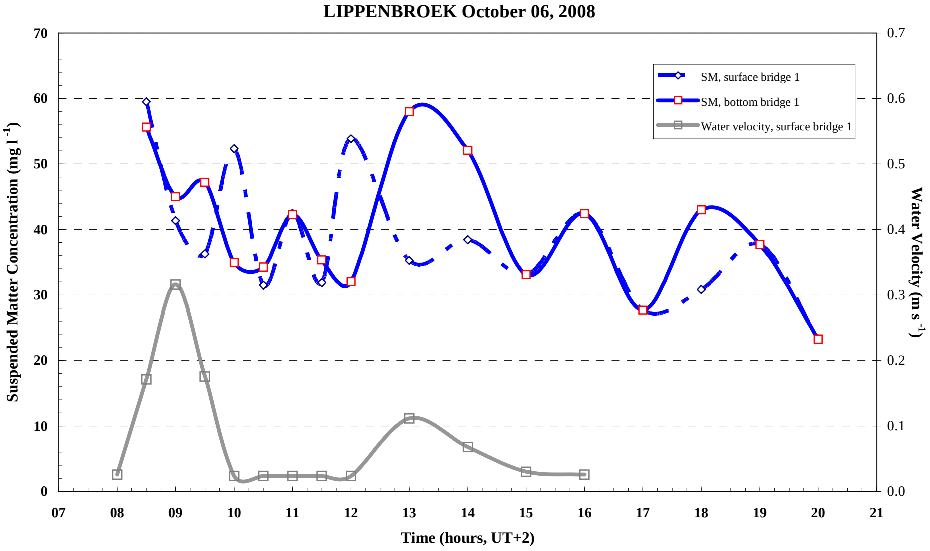
The measurement campaign of October 6, 2008, started at 8:00 and ended at 20:00 (Figuur 9.33). The tidal range was 5.15 m and the river discharge at Melle was quite high reaching 111.3 m^3 s^{-1}. When the inflow started, the SMC first increased rapidly from 20 to 60 mg l^{-1}, then progressively decreased to approximately 30 mg l^{-1} and stayed low till the outflow at around 12:00 (Figuur 9.34). When the outflow started, the SMC increased to approximately 60 mg l^{-1} in an hour, then fluctuated between 30 and 40 mg l^{-1}, and dropped to about 20 mg l^{-1} at the end of this tidal cycle. The SMC near bottom evolved in the similar trend and magnitude as the SMC close to the surface (Figuur 9.34). The SMC near the bridge 2, which is at the upper part of the creek, showed quite close values to the ones acquired at the bridge 1 (Figuur 9.35). Two hours after the inflow or about one hour after the maximum flow velocity was recorded, around 10:00, a high value of SMC was observed in the upper part of the creek. This high value might be resulted from local erosion.



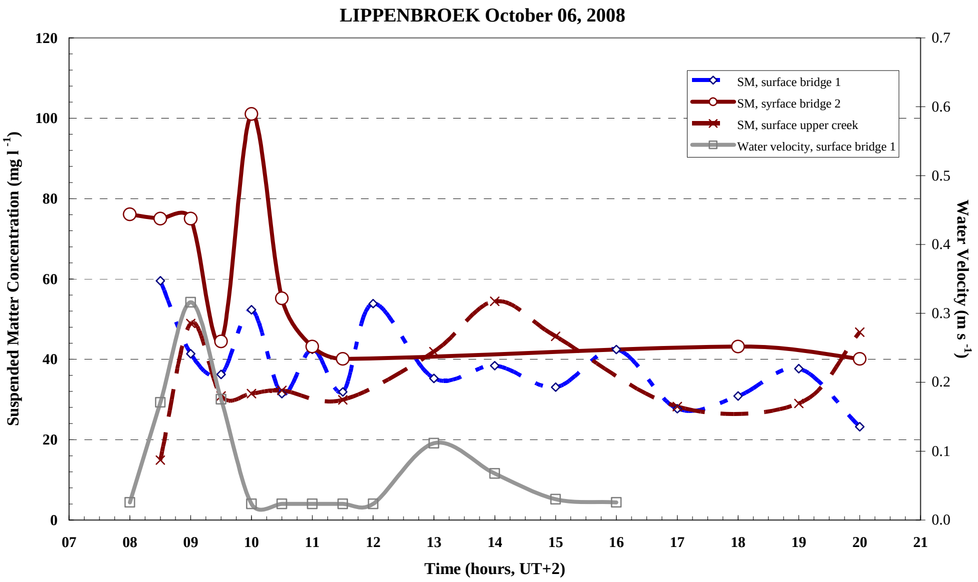
Figuur 9.32: Comparison of SMC in the river Schelde and in the Lippenbroek



Figuur 9.33: Tidal data at the Schelde in front of the Lippenbroek area on October 6, 2008



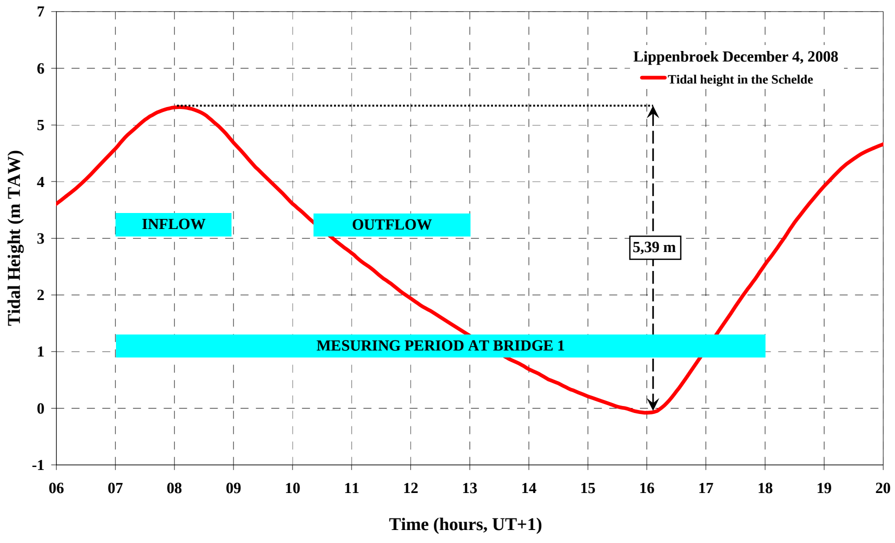
Figuur 9.34: Water velocity measured at the bridge 1; suspended matter concentration near the surface and near the bottom at bridge the 1



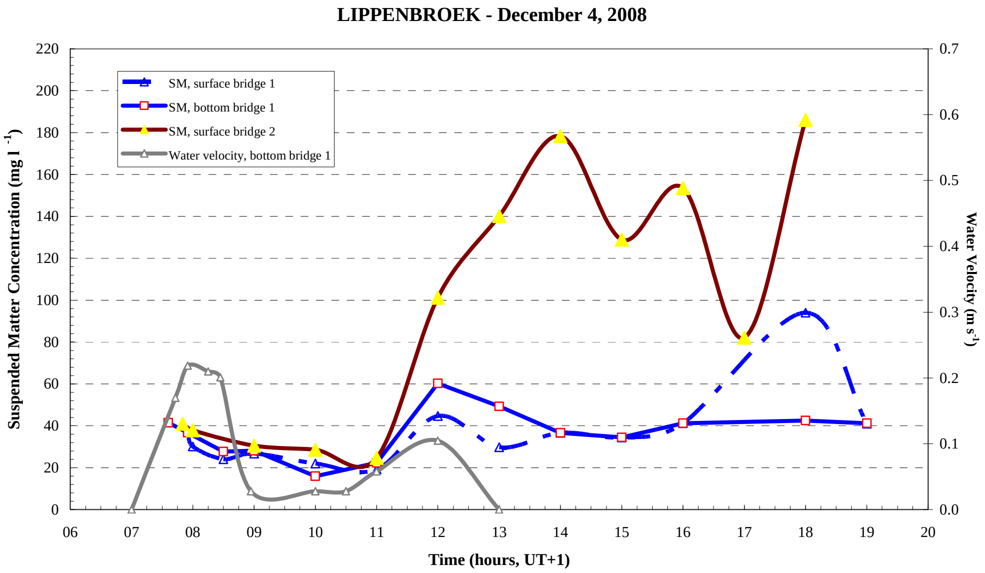
Figuur 9.35: Water velocity measured at the bridge 1; suspended matter concentration near the surface at the bridge 1 and bridge 2, in the upper part of the creek.

9.5.5. Measurements on December 4, 2008

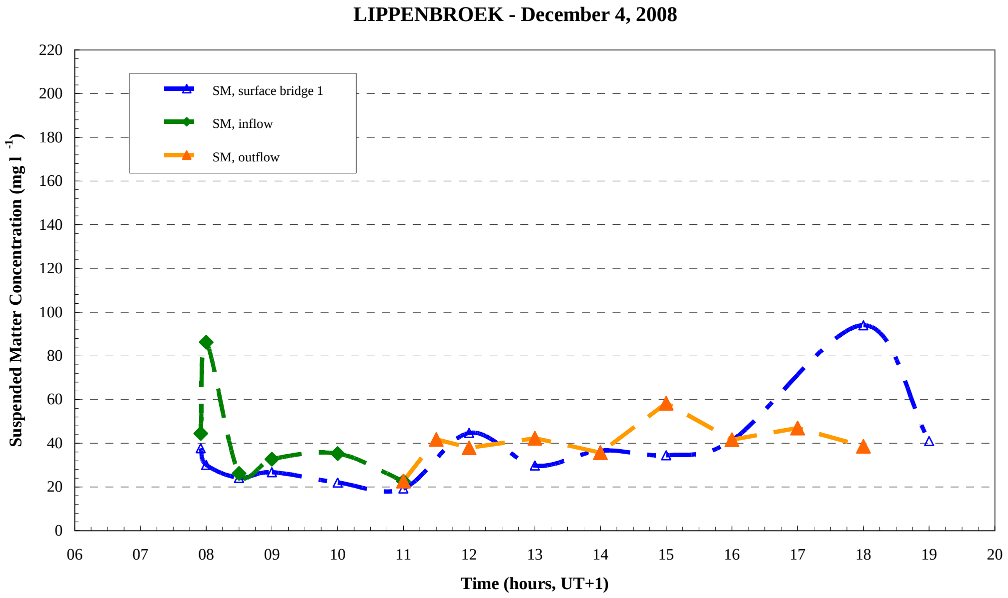
The measurement campaign of December 4, 2008, started at 7:00 and ended at 19:00 (Figuur 9.36). The tidal range was 5. 39 m and the river discharge at Melle was 91.2 m^3 s^{-1}. When the inflow started, the SMC remained below 40 mg l^{-1} and decrease slowly down to 20 mg l^{-1} in about 4 hours (Figuur 9.37). When the outflow started, it increased progressively to approximately 60 mg l^{-1} and stayed low till 16:00. Near the end of the tidal cycle a sudden increase of the SMC up to 80 mg l^{-1} was observed. The SMC near bottom showed a similar trend and magnitude as the SMC close to the surface (Figuur 9.37). The SMC near the bridge 2, which is at the upper part of the creek, showed quite close values to the ones acquired at the bridge 1 during the inflow period (Figuur 9.37). However, when the outflow started around 11:00, a fast increment of the SMC up to 100 was recorded, and fluctuated around 140 mg l^{-1} was observed. The SMC at the in- and outlet gates was presented in the Figuur 9.38.



Figuur 9.36: data in the Schelde in front of the Lippenbroek area on December 4, 2008.



Figuur 9.37: Suspended matter concentrations at bridge 1 (surface and bottom), bridge 2 (surface) and the water velocity at bridge 1 on December 4, 2008.



Figuur 9.38: Suspended matter concentration near the in- and outlet gates as observed during the measurements on December 4, 2008.

9.5.6. Correlation between tidal range and suspended matter concentration

An effort was made in order to evaluate the relationships between tidal range and suspended matter concentration resulted from all the measurements since 2006. The relationship between tidal range and the observed SMC at the bridge 1 during the inflow has a R^2 value of 0.6. And the relationship between the SMC in the Schelde and at the bridge 1 has a R^2 value of 0.5. This correlation indicates that a non negligible part of the suspended matter from the Schelde somehow did not reach the bridge 1 after being channeled into the Lippenbroek. The sediment transport dynamics in relation to the morphology in between the Schelde and the bridge 1 in the Lippenbroek needs to be addressed and studied in future.

9.6. Vegetatieontwikkeling in Lippenbroek

9.6.1. Inleiding

Voor de opvolging van de vegetatie successie, werden op 10 random gekozen locaties in het Lippenbroek en drie langs de slik-schor gradiënt op een referentieschor, telkens 7 permanente quadraten uitgezet. De 10 random locaties zijn de sites voor intensieve monitoring (S1 tot S10; Figuur 9.5).

Deze 70 PQ's werden 4 maal per jaar bemonsterd. Naast staalnames voor fysicochemiche bodemeigenschappen werden volgende variabelen genoteerd: Vegetatie, Strooisel, Algen, Mos, Diatomeeen, Vrije bodem, Water, Gemiddelde hoogte, vegetatie (cm), Helling (%), Microrelief (cm), Uitschieters (species + hoogte), Species (species+ bedekking). Bedekkingen werden allen geschat als percentage, voor percentages lager dan 3 werd ook telkens het aantal individuen genoteerd.

De gegevens van lente 2006 tot lente 2008 werden gepubliceerd in Estuarine, coastal and shelf science. Hieronder wordt de inhoud van het artikel weergegeven. De monitoringsresultaten na deze datum werden op dezelfde wijze genalyseerd, en toegevoegd achteraan dit artikel:

"Restoration of tidal freshwater vegetation using controlled reduced tide (CRT) along the Schelde Estuary (Belgium)."

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Key words: Estuaries, Managed Realignment, Restoration, Vegetation, Tide Gauges, Trajectory, Belgium, Schelde estuary, controlled reduced tide (CRT) 51°05'10"N, 4°10'20"E

9.6.2. Abstract

Throughout the world, tidal marshes are being restored to obtain natural protection against recurring storm surges and sea level rise and to preserve the resources these habitats provide. Managed realignment, or breaching of the seaward dikes, is a restoration technique increasingly used; yet unsuitable site elevation has been a major constraint to habitat development. Controlled reduced tide (CRT), a new technique to implement tidal regime on low elevated sites, could offer solutions for several problems associated with reconstruction or mitigation of tidal marshes.

We followed vegetation development at 3 month intervals during two years after implementation of the first pilot CRT treatment, which aimed to restore tidal freshwater marsh habitat in an agricultural site where elevation relative to the estuarine marshes has lowered. The development of the plant community and its driving forces (flooding, remnant vegetation) were studied. Arrival and establishment of typical tidal freshwater marsh vegetation was rapid, and a clear elevation gradient developed by eradication of terrestrial species and colonisation by estuarine species. Remnant vegetation substantially reduced the number of new established species. General development trajectory as well as the changing effect of driving forces and their interaction point to a swift ecosystem shift during the first two surveys, and a gradual species saturation during the seven following surveys. Vegetation communities seem to deviate from the nearby tidal freshwater marshes, but show similarities with tidal freshwater marshes described for the beginning of the 20th century. The CRT-technique provides strong potential for durable, adaptive restoration of tidal marshes on sites with low elevation. Wider implementation of the CRT technique could increase the total surface of tidal freshwater marshes with fully developed vegetation gradients.

9.6.3. Introduction

The worldwide extent and abundance of intertidal habitat has greatly decreased, primarily due to human alteration of estuarine habitats. Tidal freshwater marshes are ecosystems with specific environmental conditions which provide essential habitat for characteristic vegetation and waterfowl (Barne *et al.* 1998, Struyf et al. 2009). Their role in estuarine biogeochemistry has been studied and reviewed intensively in recent decades (e.g. Dame *et al.* 1986, Howarth *et al.* 1996), emphasising their role in C, N and P transformations (e.g. Spurrier & Kjerfve, 1988; Childers & Day, 1988; Gribsholt *et al.* 2005) and in the buffering of estuarine silica concentrations (Struyf *et al.* 2006, 2007, Struyf & Conley 2008, Jacobs *et al.* 2008). They also dissipate and control tidal currents and waves (Möller & Spencer 2002), protecting landward sea defences from scour and erosion.

Impacts include construction of embankments and dikes, harbor expansion and conversion of tidal marsh for agricultural uses and urban development. Nowadays, managed realignment is a technique which is increasingly used to restore intertidal habitat by the removal or breaching of dikes to restore tidal influence (reviewed by French, 2006). Elevation is a key factor in planning of managed realignment and for suitable site selection (e.g. Brooke 1991; Cundy, 2002) as it relates directly to frequency, height and duration of tidal inundation, which are the main factors in vegetation development. Agricultural sites adjacent to estuaries are often lower in elevation as a result of compaction. Typically they are low the levels of contemporary marshes within the same system, which have often increased in height with sedimentation (Temmerman *et al.* 2003). The lower elevation rules out many sites for potential realignment, since this would result in entire flooding of the site every tidal cycle and would thus result in development of completely non-vegetated intertidal mudflats. It is not certain if and how fast such mudflats could evolve to a vegetated marsh system. This problem has been countered in different ways. In the UK, suitable sites which approximate the desired elevation level are selected and natural processes are allowed to proceed (French 2006). In the US, it is common to increase site elevation prior to dike breaching by adding sediment (Brooke 1992). However, artificial site elevation has been a common reason for project failure (Quammen 1986, Perry *et al.* 2001). A third option is to install a restricted tidal exchange (e.g. Lamberth & Haycock 2002, Hirons *et al.* 2002). This has the advantage of lowering the tidal range in the site to an acceptable level, but the technique cuts out spring-neap tide variations needed for habitat development.

In 2006, a novel technique was presented (Cox *et al.* 2006, Maris *et al*, 2007) to restore tidal marshes on sites with low elevation. It provides a site with a controlled reduced tide (CRT). This technique allows implementation of a restricted tidal regime with neap and spring tides, by the use of high inlet culverts and low outlet valves. Restoring intertidal habitat with CRT is based on the very simple idea of detaching the site's elevation and the tidal regime it receives: a high inlet culvert in the dike allows only a limited volume to enter the site which depends on the height of high tide. The ebb phase only starts after the water level in the estuary has lowered to the inundation level of the site, when a lower gravitational culvert opens. Fine-tuning of inlet culvert level permits installation of a tidal regime with a pronounced spring-neap variation. At spring tide, the whole site is flooded, while at neap tide, only the central creek fills. However, despite the variation in inundation frequencies, there are some physical constraints to the system. A prolonged stagnant phase of several hours at high tide is observed inside the CRT (absent in reference tidal marshes), and tidal amplitude is less. The lower amplitude potentially lowers accretion rates, but this might be compensated by the slackened currents and the stagnant phase which promote complete settling of sediments. Adaptive management by fine-tuning of the culverts towards specific management goals is straightforward, and no complex mechanical or electronic systems are used. Detailed technical explications can be found in Maris *et al.* (2007) and Cox *et al.* (2006). We hypothesise that the CRT-technique could vastly increase the number of suitable sites for habitat restoration.

It can take several years before the vegetation of a restored site resembles that of a historical or nearby reference situation (Garbutt *et al.* 2006). This slow development may be due to dispersal constraints of the target species (Bisschoff 2002; Bissels *et al.* 2004; Wolters *et al.* 2008) or to environmental conditions that are unsuitable for their establishment (Morris *et al.* 2004). Despite these uncertainties, the Flemish government decided in 2006 to implement the CRT scheme in ca. 1500ha of tidal restoration projects, so that eventually (by 2030) new CRT habitat will comprise 36% of tidal freshwater marshes along the Schelde, or about 16% of the European tidal freshwater marshes (Struyf et al. 2009). Also, international application of the CRT technique for both fresh- and saltwater tidal marsh restoration might follow, while the capacity of the CRT technique to restore reference habitats and functions has still to be assessed. The objective of this study was to evaluate the potential of the CRT technique for restoration of intertidal habitat. Three questions were addressed by intensively studying two years of vegetation development in a pilot CRT pilot project:

(1) Which species did establish compared to European tidal freshwater marsh reference habitats along the Schelde, Elbe, and Rhine-Meuse estuaries? This question was addressed by qualitatively comparing the established species community to literature descriptions of European tidal freshwater marsh reference habitats (Schelde, Elbe, and Rhine-Meuse) in the discussion. The available research on the vegetation and ecology of European tidal freshwater marshes is scarce. To our knowledge, no statistically comparable dataset of reference freshwater tidal marshes is available. Only Struyf et al (2009) have compiled information from national books and reports and PhD theses from the last 50 years that have focused specifically on tidal freshwater marshes or that were part of a larger study topic. Moreover, present European freshwater tidal marshes are severely impacted, and cannot be considered as true reference habitat.

(2) What is the relative importance of driving forces (flooding gradient vs. remnant vegetation) in influencing the initial establishment of plant communities in the CRT site?

(3) What is the relative importance of flooding gradient vs. remnant vegetation in influencing the plant community composition through time? Here, the temporal tendencies of the driving forces' effects and system trajectory were analysed, and future expectations discussed.

Our results emphasise the capacity of CRT to rapidly develop a specific intertidal community with vegetation gradients characteristic for reference tidal freshwater marshes.

9.6.4. Methods

Site description and design scheme

The Schelde estuary is a highly eutrophic, macrotidal estuary, that has been extensively described (Wollast 1988, Meire *et al.* 2005, Soetaert *et al.* 2006). Its 21.863 km^2 drainage basin, situated in France, Belgium and the Netherlands, is inhabited by over 10 million individuals. The study area is a pilot CRT, the "Lippenbroek" (surface approximately 80.000 m^2), situated at Hamme, Belgium (51°05'10"N; 4°10'20"E). Mean tidal amplitude at springtide in the Schelde at this location is about 6 m. The site is situated in the freshwater tidal zone (diurnal tides) of the estuary, with an averaged specific conductivity ranging from 1000 in winter to 1500 $\mu S*cm^{-1}$ in summer (salinity approx 0.5 to 0.75 PSU). The study site is 2.5 to 3 m lower in elevation than the contemporary tidal marshes and managed realignment would result in total flooding of the whole area every tidal cycle. The study area was mostly used as cropland, in an intensive rotation system with *Zea mays (*Maize*), Daucus carota* (Carrot) and *Solanum tuberosum* (Potatoe); lower parts were planted with *Populus sp.* trees (Poplar) or over-grown with naturally established *Salix sp.* trees (Willow) (Fig. 1A). The rotation system was abandoned in 2003. During the two-year construction phase (2003-2005), crops were replaced by a dense pioneer vegetation (mainly *Epilobium hirsutum* (Great Willowherb) and *Urtica dioica* (Stinging Nettle)), and part of the site was devegetated because of the culvert construction activities (Fig. 1B). Tidal inundation was restored by means of CRT in March 2006. Sedimentation and erosion processes are transforming the site's topography towards a typical tidal marsh platform incised by creeks (Jacobs 2009).

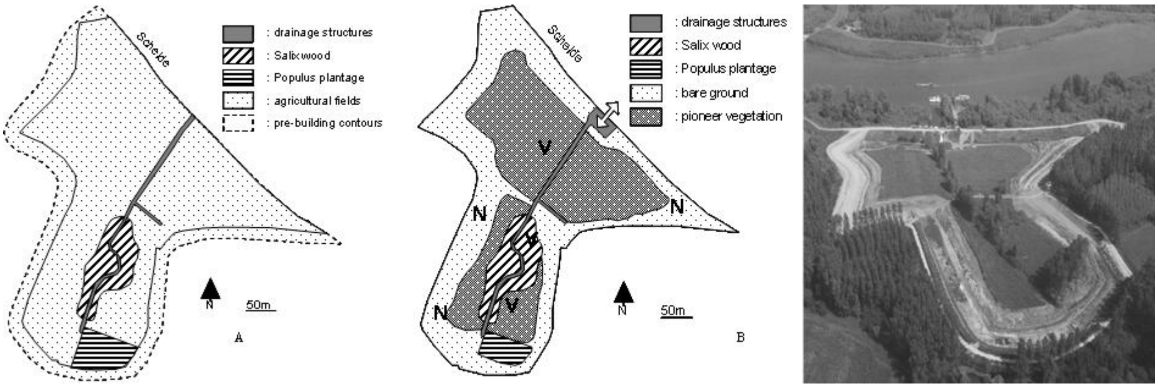

Figure 1. Schematic overview of the study site before (A) and after construction (B). Sampling locations with ("V") and without ("nV") initial vegetation are shown in B. The central creek is shown in gray, the white arrow indicates the culverts' location. C is an aerial view of the site just after construction.

Monitoring

Vegetation was studied at six locations from April 2006 to April 2008, beginning just after tidal regime was reinstalled (March 1st, 2006). The locations were situated throughout the CRT pilot site, and were selected to have different elevations as well as non-vegetated (nV) and vegetated (V) initial conditions (stratified random, Fig. 1). At each of the locations, 7 permanent plots (2×2 m) were set out next to each other. As such, the 42 plots fully represented the main gradients found in the pilot area. Plots were monitored at low tide 9 times (April-July-October-January 2006 and 2007 and April 2008) to closely monitor short-term changes. During each survey, cover per species was noted for each species per plot. Cover was estimated as % relative to the plot's surface area; resulting in a discrete dataset rather than in traditional cover classes, which are difficult to analyse. EUNIS database (EUNIS 2008) was used as a taxonomic checklist.

A database of 378 separate vegetation inventories was used for the analysis (9 dates × 6 locations × 7 plots). Plots were assigned to three elevation-classes: Low (L), Medium (M) and high (H). Water levels were calculated from water level dataloggers. Location elevation, as expected, governed hydrological variables (Fig. 2). Although total variance of the variables between the replicate plots within locations is large, its effect was highly significant on flooding frequency, duration and water height (multiple *t*-test; $p < 0.0001$ for all comparisons between groups H, M and L). The considered elevation factor was thus used as proxy for flooding characteristics. For each elevation class, two locations are considered: initially vegetated (V) and initially bare zones (nV).

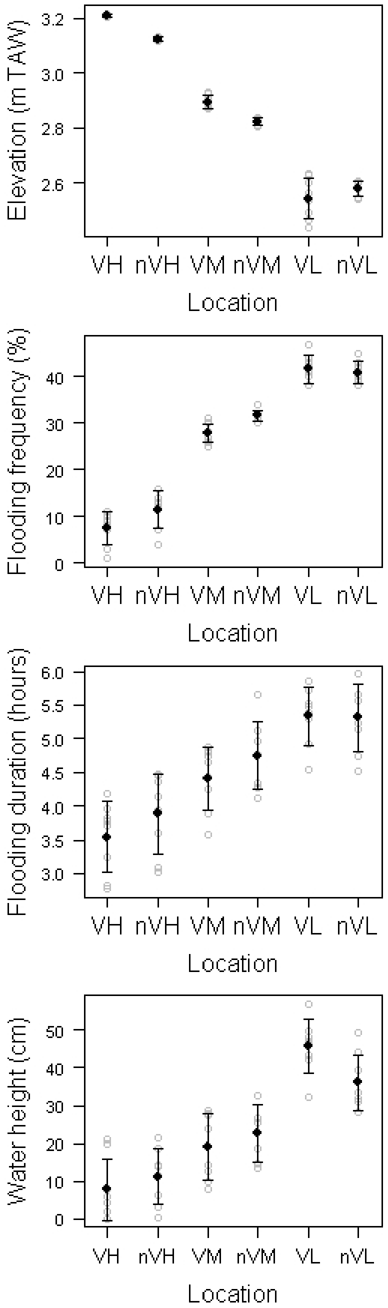

Figure 2. Hydrological variables and elevation gradient for the study locations during the study period. For initially vegetated sites (V) as well as for initially non-vegetated sites (nV), three elevation classes were considered: low (L), medium (M) and high (H). For elevation, grey points represent the seasonal measurements (SET, Boumans and Day, 1993; and Cahoon et al. 1995).

For other variables, grey points are seasonal averaged values. Hydrological variables are defined according to Cox et al. (2006) and calculated from continuous time series using the seasonal elevation. Black points are yearly mean values; bars extend to yearly mean ± standard deviation of these seasonal means. Note that variance of seasonal values is not shown. mTAW is the general Belgian elevation reference level.

Data analysis

First, spatial patterns of the species community were explored by means of Principal Component Analysis on Instrumental Variable (PCAIV; Sabatier *et al.* 1989, Lebreton *et al.* 1991) after log (x + 1) transformation and centering of the plots × species table. Statistical significance of effects was tested using a permutation test (9999 iterations, Manly, 1991).
Second, for each species, total yearly integrated cover was summed over the first 4 surveys and last 4 surveys separately (thus leaving out the 2008 surveys). This equally accounts for species with different seasonal growth maxima. Between-year variation was calculated as the relative difference in cover between the last 4 and first 4 surveys. This provided a general score for each species at each location. Individual species scores were compared with patterns emerging from the PCAIV.
Third, effects of elevation, remnant initial vegetation and their interaction were quantified and tested by means of PCAIV for each seasonal subset separately to explore temporal developments of the between location difference in species composition.
Fourth, to separate a general system trajectory from seasonal patterns based on the observed vegetation development, Multiple Co-inertia Analysis was used (MCOA; Chessel & Hanafi 1996). This technique emphasises the common changes ('synchrony') in species assemblages despite spatial or temporal fluctuations of species distributions in the system (Thioulouse *et al.* 2004; Bady *et al.* 2004). Therefore, the dataset was ordered in six tables (one per location, dates × species, and the summed species covers of the 7 replicate plots in the table). Consequently, the resulting ordination engenders purely temporal gradients. For an extensive application and method review, see Bady *et al.* (2004).
All computations and associated graphical representations were implemented using the *ade4* package (Chessel *et al.* 2004) available in *R* freeware (R 2008). An extensive description of these techniques can be found in the additional file provided online.

9.6.5. Results

Effect of remnant vegetation and elevation

PCAIV of conjoined effect of elevation and remnant vegetation highlights two main axes (Fig. 3A). This joined spatial effect was highly significant ($p < 0.0001$) and explained 39 % of the total variance. The first factorial plane (Fig. 3A) shows a community ordination responding significantly to elevation effect in both V and nV gradients (explained inertia 13.9 %; $p < 0.0001$). Although vegetation effect was also highly significant (explained inertia 13.8 %; $p < 0.0001$), a convergence in low-elevation plots was observed. A significant interaction between elevation and vegetation (explained inertia 10.9 %; $p < 0.0001$) confirmed a floristic similarity at low elevations vs. a floristic divergence at high elevations (Fig. 3A). Although a clear temporal element is apparent when considering the first (Fig. 3B) and last survey (Fig. 3C) separately, initially vegetated M- and H-plots have remained distinct from their non-vegetated equivalents even after two years (Fig. 3B and 4C).

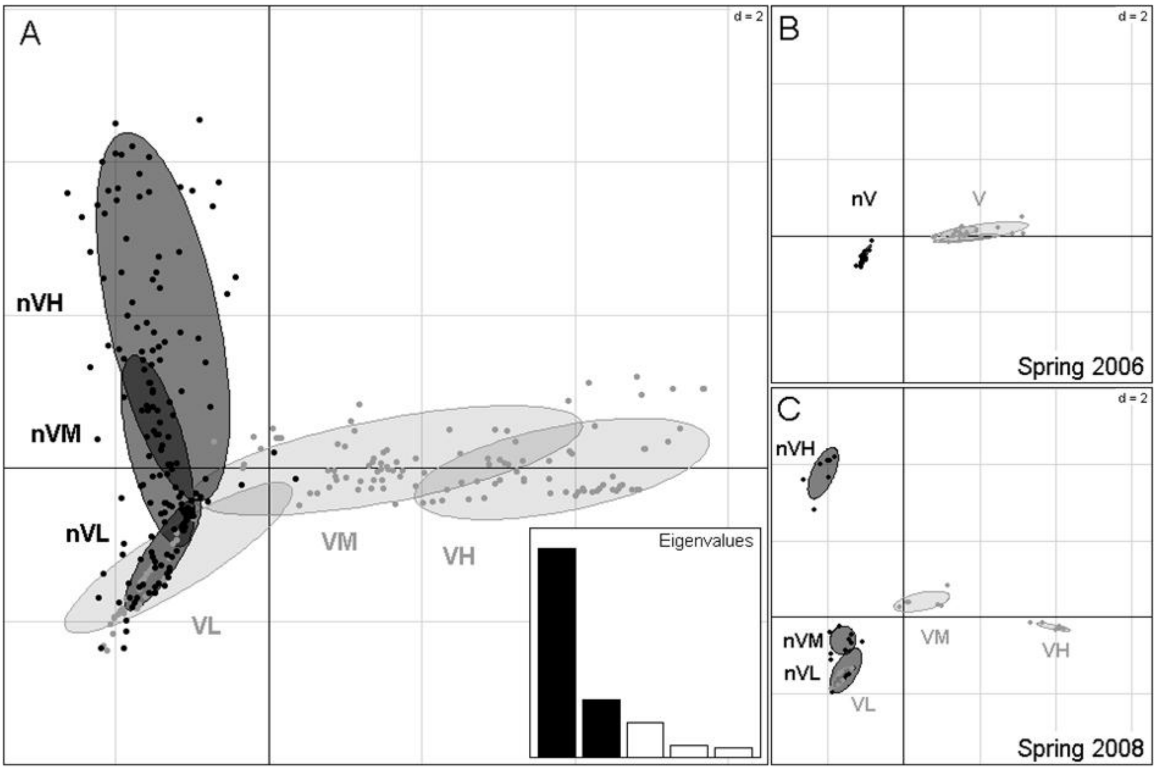

Figure 3. PCAIV of location effect. A) First factorial plane and eigenvalue diagram. F1, 65 %; F2, 18%. B) Within-date structure considering only the first spring survey. H, M and L are not marked because of complete overlap. C) Within-date structure considering only the last spring survey.

The analysis thus gives evidence for a structuring effect of both driving forces: elevation as well as remnant vegetation impact vegetation development. Moreover, it reveals a convergence of initially non- and vegetated plots in low elevated locations, pointing to a strong interaction between the two factors, i.e. a lower impact of remnant initial vegetation at lower elevations. This pattern is reflected in the cover developments for individual species (Fig. 4A&B).

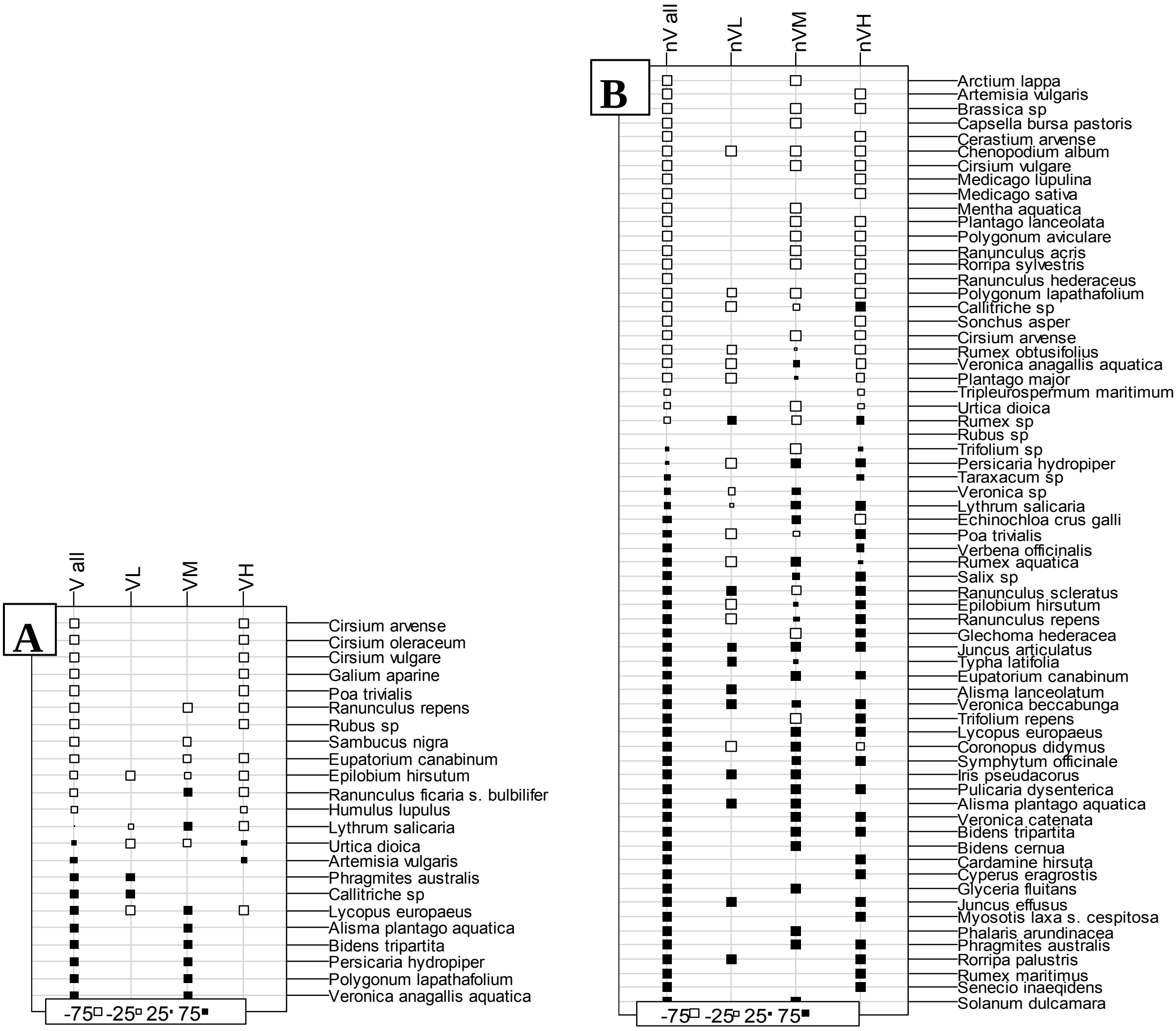

Figure 4. Percent difference in cumulative vegetation cover between the two complete seasonal cycles studied (cumulative cover over the year masks seasonal differences that might appear between years). Open quadrats represents negative, closed ones positive percent.) for the initially vegetated (A) and non-vegetated plots (B). Columns represent all (all), High (H), moderate (M) and low-elevation (L) locations. The ninth survey (spring 2008) was not incorporated in this analysis. Species are ordered following the 'Vall' column.

Species composition

Plots in all V zones (VL, VM, VH) had fewer species (Fig. 4A, Tab. 1). A dense vegetation cover with co-dominating ruderal species (*Epilobium hirsutum* and *Urtica dioica*) established on the abandoned and fertile agricultural land during the two year building period, and did this in a fairly homogenous way over the elevation gradient.

VL-plots changed rapidly after installation of the tidal regime: ruderal dominants were eliminated and replaced by a less dense hydrophyte and wetland pioneer vegetation. *Epilobium* partly disappeared and *Urtica* completely died during the following surveys. *Phragmites australis* (Common reed) is now quickly invading the plots (Fig. 4A). This first development coincides with the disappearance of the VL-nVL convergence on the first PCAIV axis.

VM-plots did not change suddenly, but the ruderal dominants were replaced by new colonisers. These are also found in the nVM-plots, but *Phragmites australis*, *Salix* sp. and other helophytes found in nVM-plots are missing from the VM-plots(Fig. 4B).

Finally, although VH-plots lost some terrestrial species, *Urtica dioica* strengthened its already dominant position (Fig. 4A).

The initially non-vegetated plots generally accommodate more species. The nVL-plots (Fig. 4B) developed a mainly helophyte community, appearing during the first year and developing further during the second year. Also, hydrophytes emerged, while marsh and terrestrial species disappeared in the second year (Fig. 4B).

The nVM-plots hosted many more species than the nVL-plots. Apart from the helophytes present on the nVL-plots, other helophytes gained cover, associated with many typical tidal freshwater marsh species and even some terrestrial species. However, some of the terrestrial species decreased in cover again during the second year. Also, the first *Salix* sp. individuals emerged (Fig. 4B). The nVH-plots accommodated the widest range of species, from hydrophytes to typical marsh species. In fact, most of the species appearing in the nVM-plots appeared also in the H zones, explaining the overlap of these categories in Fig. 3A. However, none of these became dominant. In contrast to the nVM-plots, typical tidal freshwater marsh-helophytes did not strengthen their position in these highest zones (Fig. 4B). An extensive description of species composition can be found in the additional file provided online.

The total number of new species appearing after the first survey in all plots differed between nV and V and throughout the elevation gradient, as did the number of species losing or gaining cover when comparing the two years (Tab.1).

Table 1. Number of species first appearing after the first survey, decreasing and increasing in total cover between the two studied seasonal cycles for all plots of the different locations. Percentages are given to the total number of species observed at low (L), mid (M) and hiugh (H) locations for initially vegetated (V) and nonvegetated (nV) plots.

	nVL	nVM	nVH	VL	VM	VH
New species	24 (100%)	48 (100%)	55 (100%)	4 (67%)	9 (69%)	3 (2%)
Decreasing species	14 (58%)	21 (44%)	22 (40%)	4 (67%)	5 (38%)	13 (86%)
Increasing species	10 (41%)	27 (56%)	32 (58%)	2 (33%)	8 (62%)	2 (13%)

Temporal development and system trajectory

PCAIV provided effects of the single factors and their interaction per survey (Fig. 5). According to the permutation tests, all three effects were significant at all surveys ($p < 0.001$) except the factor elevation and its interaction with remnant vegetation in the first survey ($p = 0.371$ and $p = 0.338$ respectively), underlining robustly different experimental starting conditions.

Elevation effect had already reached its maximum by the first survey and only varied slightly after that (Fig. 5). Vegetation effect suddenly dropped during the first survey, and showed an overall decrease after that. The interactive effect, driven by the difference in elevation impact on the two communities (V and nV), grew stronger. This analysis indicates that elevation and remnant vegetation are the main driving forces and furthermore demonstrates the importance of frequent monitoring during the first surveys. In addition, it shows remnant vegetation can have a long lasting effect and interact with elevation (difference in elevation effect on V and nV communities).

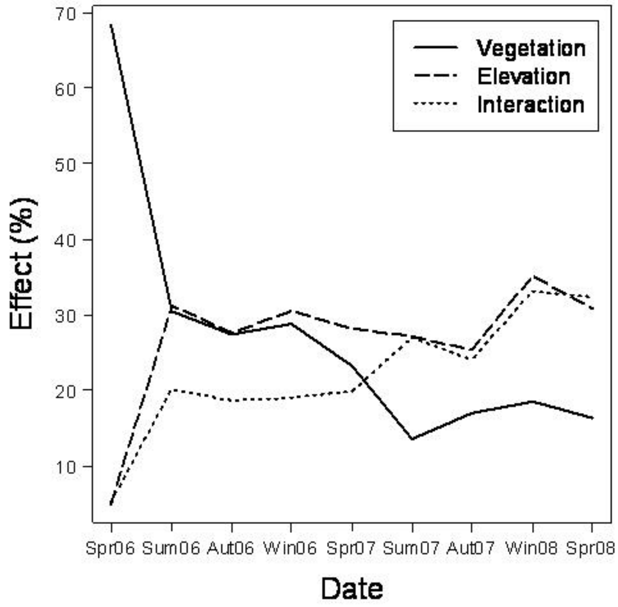

Figure 5. Temporal changes in elevation, remnant vegetation and interaction effects on community structure from spring 2006 (Spr06) to spring 2008 (Spr08) surveys. Effects are calculated by means of PCAIV for each season.

Correlations between each of the six location tables and the average MCOA axes' scores are high (0.88, 0.95, 0.92, 0.95, 0.83, 0.98 for VH, nVH, VM, nVM, VL, nVL respectively) and were all highly significant (*p*-value < 0.001). This confirms a strong common temporal pattern between the six locations. MCOA identified two main axes (Fig. 6A). The first axis scores follow the seasonal cycles (Fig. 6B) while the second axis scores increase irreversibly (Fig. 6C). These temporal gradients are thus composed of reversible and irreversible processes respectively, and the second axis, as the irreversible part of the system's temporal pattern, can be interpreted as a system trajectory.

Between the two first dates after initial inundation, second axis scores drop, matching observations of quick elimination of terrestrial species (Fig. 4) as well as switching factor effects (Fig. 5). During the second survey this score rises steeply, coinciding with the emergence of many wetland colonising species on the site. Then, the trajectory slowly approaches saturation, with some minor drops from winter to spring in 2006 and 2007, but without return to initial scores.

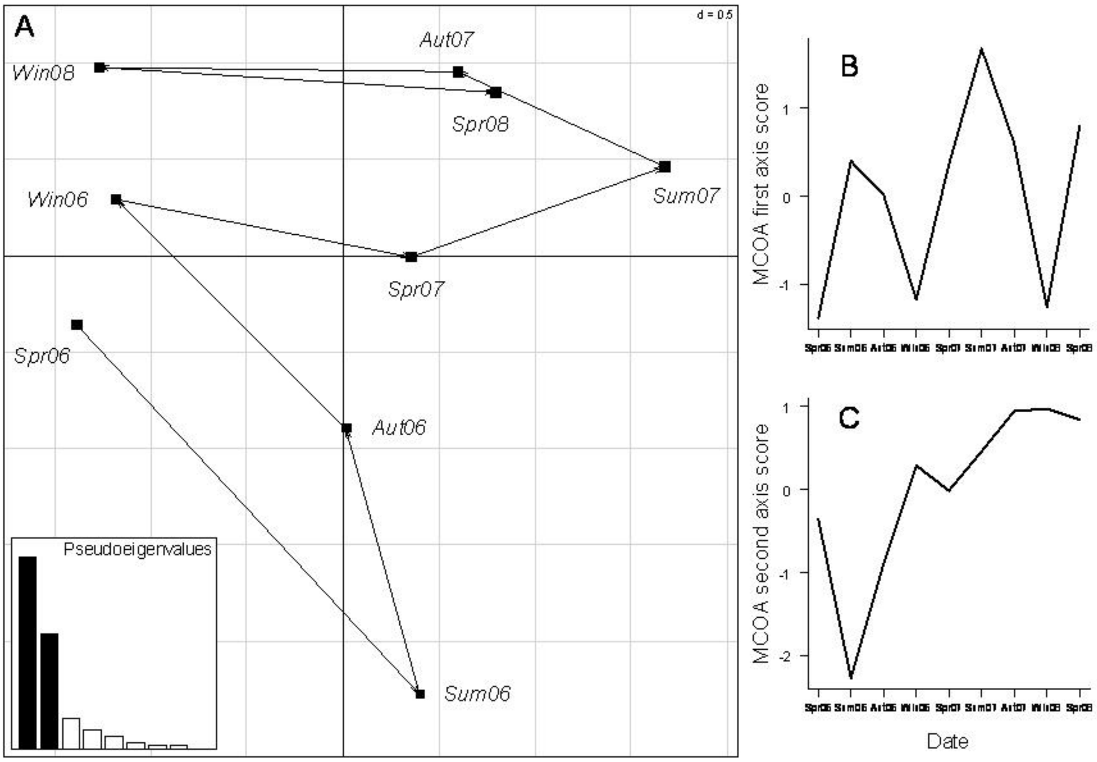

Figure 6. Multiple co-inertia analysis (MCOA). A) first factorial plane and pseudo-eigenvalue diagram; F1, 49 %; F2, 26 %. B) and C): MCOA axes scores from A vs. seasonal succession from spring 2006 (Spr06) to spring 2008 (Spr08).

9.6.6. Discussion

Which species established compared to European tidal freshwater marsh reference habitats (Schelde, Elbe, and Rhine-Meuse Estuaries)

The well-developed community on the nV-plots covers about half of the pilot CRT site. Some of the lowest zones are, like the reference tidal freshwater marsh, covered only with *Vaucheria*, (Xanthophyceae) or even have no cover at all (tidal pools and creeks). The hydrophyte community strongly resembles reference tidal freshwater marsh sites. However, some species like *Alisma plantago-aquatica* (American waterplantain), *Alisma lanceolata* (Lanceolate Water-plantain) and *Veronica beccabunga* (Brooklime) appear in higher densities and at relatively higher elevated plots in the CRT plots compared to the Schelde tidal freshwater marshes. Moreover, the helophyte community, which is the dominant plant life-form for all known tidal freshwater marshes (Ellenberg 1963, Struyf et al. 2009) completely deviates from the present marsh vegetations in the Schelde. *Glyceria maxima* (Reed Sweetgrass), *Phalaris arundinacea*, (Reed Canary Grass) *Typha latifolia* (Common Bulrush) and *Bolboschoenus maritimus* (Sea Club-Rush) are (co)dominant in large patches whilst the only helophyte establishing dominantly in the Schelde marshes is *Phragmites australis*. Along the Schelde, no reliable data of tidal freshwater marsh vegetation before increased anthropogenic pollution and erosion are available, yet photographs of historic tidal freshwater marsh show a typical *Bolboschoenus* community in the lower marsh zones (Massart 1908). For the Biesbosch (the Netherlands), this historical reference situation is well described. In the pre-1970 Biesbosch, abovementioned helophytes all dominated large zones. Later on, vegetation in the Biesbosch changed due to the closure of the Haringvliet dam in 1970 (Zonneveld 1960; Struyf et al. 2009) and the enhanced shipping and wind wave erosion (De

Boois 1982, Struyf et al. 2009). Along the Elbe (Germany), similar helophyte reference communities are described for the lower tidal freshwater marsh elevations.

Whether this deviation of the pioneer and helophyte communities from the nearby Schelde marshes could be explained by water quantity or quality effects remains to be studied. The prolonged inundation time at high tide (stagnant phase of approx. 2 hours) could influence vegetation composition in the long run, although after two years, it does not seem to hamper vegetation development in general. Also, vegetation establishment could be influenced by the increased dissolved oxygen concentrations of the water entering the CRT, both by culvert and surface aeration, lowering the potentially harmful effect of hypoxia on certain species.

We hypothesise, however, that the lower current velocities in the CRT provided sediment settling conditions similar to the historical reference situation, while in the estuary, the high current velocities hamper colonisation of mudflats by reference vegetation communities and erode the marsh edges.

What is the relative importance of flooding gradient vs. remnant vegetation in influencing the initial establishment of plant communities in the CRT study area?

Controlled reduced tide (CRT), implemented on an agricultural site with low elevation, caused the immediate development of a typical intertidal vegetation gradient, from unvegetated mudflats, over helophyte pioneer zones to *Salix* colonisation zones. This gradient is linked to site elevation and within-site elevation gradient (French 2006). To restore the whole range of tidal freshwater marsh habitats, it is therefore crucial to provide the whole range of flooding characteristics. It appears that CRT offers this possibility by implementing the complete tidal regime with spring-neap variations onto a given site, even if this site has a low elevation compared to the estuarine marshes, or smaller within-site elevation gradients (Cox *et al.* 2006, Maris *et al.* 2007).

This is the first study evaluating vegetation development in a CRT site. We conclude that after two years, typical species from the whole intertidal gradient arrived and established in our pilot CRT site: from terrestrial pioneering species (e.g. *Epilobium hirsutum*) to dominant helophyte species (e.g. *Phragmites australis, Typha latifolia*) and *Salix* seedlings. All species found in the CRT site are present in the estuary, but many of them never occur as dominants (e.g. *Typha latifolia, Glyceria maxima*) and some pioneers are even scarce because the estuary lacks the processes which reset succession. However, connectivity between remaining populations is potentially high as many seeds and propagules are transported by the tide. The immediate colonisation indeed proves that adequate adjacent sources were still available in the estuary.

On the nV-plots, arrival of new species was not a constraining factor. Arrival and establishment (as the number of species appearing and gaining cover respectively) increased with elevation, as expected from previous studies at salt marshes (Wolters *et al.* 2005a and 2005b, Cornu & Sadro 2002, Williams & Orr 2002).

On the V-plots, development of a typical tidal freshwater marsh community was less obvious. On the highest elevated VH-plots the decreasing remnant species were not at all replaced by a tidal freshwater marsh community. This lack of tidal freshwater marsh species establishing at VH-plots could be explained by the persistence of the dominant species *Urtica dioica,* which gained cover and could have hampered establishment of other species by competition for light (Ungar 1998) or nutrients (Levine, et al. 1998). Also, seed dispersal might be hampered by dense *Urtica* stands. The persistence of the *Urtica* stands is also reflected in the strong interaction between elevation and vegetation effects.

On intermediate elevations more tidal freshwater marsh species established, whilst on the lower VL-plots, the dense remnant vegetation community disappeared immediately during the first survey, but surprisingly the number of tidal freshwater marsh species establishing was still six fold less compared to the non-vegetated plots at similar elevations. The hampering impact of remnant vegetation can indeed persist for a long time by formation of anoxic and sodium enriched mud by litter decomposition in the soil (MacLeod *et al.* 1999), which was also shown by the long lasting effect of remnant vegetation in our PCAIV analysis.

What is the relative importance of flooding gradient vs. remnant vegetation in influencing the plant community composition through time?

CRT regime caused a sudden system change during the first 5 months. Eradication of terrestrial species from the lowest zone took place during the first few months, as reported earlier for salt marsh restoration (Wolters *et al.* 2008), and was followed by quick colonisation of the initially non-vegetated zones. The CRT's development then slowed down and appears to stabilise after about one year. Blackwell *et al.* (2004) show important short-term changes in environmental variables after managed realignment of an abandoned agricultural field in Devon, UK. Other studies also show that vegetation change can take place rapidly (Atkinson *et al.* 2001, Alpin & Posey 2000, Edwards & Proffit 2003) and then slows after the first growing season after restoration of tidal regime (Roman *et al.* 2002). However, the apparent stable state reached in the trajectory from the second summer on might be misleading. Firstly, interspecific processes (e.g. competition, facilitation) determining the climax state of the marsh (Ungar 1998, Levine et al. 1998) have yet to start gaining influence, and the magnitude of their effects is as yet unknown. Secondly, interspecific processes are known to provoke development over longer terms, which might not appear in this short-term trajectory. Probably, effects of the two factors studied here will lose influence in favour of these processes.

The climax succession stage, *Salix*-dominated tidal forests, did not develop fully as yet, but *Salix* individuals have arrived throughout the medium and high elevated zones of the CRT, even in young *Phragmites* stands. It is expected that *Salix* forests will outcompete the present vegetation for light in these zones, as they do in the marshes along the estuary. Future vegetation pattern will depend on the flooding-imposed abiotic constraints of the lower zones (i.e. Snow & Vince 1984, Cooper 1982, Wiehe 1935, Armstrong *et al.* 1985) and on facilitation or competition for resources at the upper zones (Ungar 1998, Levine, Brewer & Bertness 1998). This succession could be slowed down by traditional management (regular mowing and cutting of the *Salix* woods), or by adaptive management of the culverts: the level of the inlet culvert can be adjusted to obtain different ratios of mudflat and low and high intertidal marsh. Also, an equivalent of natural dynamic reversal of succession could be considered by temporary flushing the CRT site with completely opened culverts.

Although this research clearly shows development of intertidal gradients and reference vegetation communities under CRT-regime, only incorporation of abiotic and biotic variables in the evaluation of the marsh habitat development can provide answers to the questions concerning *how* they develop.

9.6.7. Conclusions

- After two years of controlled reduced tide, typical tidal freshwater vegetation has been restored in a former agricultural site. Establishment of an intertidal plant community in the CRT site was quick, with fast eradication of terrestrial species and colonisation by tidal freshwater marsh species already during the first few months
- However, presence of initial terrestrial vegetation slowed down establishment at all locations. Removal of remnant vegetation prior to culvert opening could decrease this impact.

- The specific features of the studied CRT seem to provide shelter for communities which have disappeared along the degraded estuary, but are described for several European references.
- The CRT technique potentially provides a simple and efficient way to develop intertidal gradients on sites with low elevations, which are not suitable for traditional managed realignment, that would result in development of non-vegetated intertidal mudflats of which it is not certain if and how fast they could evolve to a vegetated marsh system resembling reference marsh sites.
- The CRT-approach could vastly increase the number of suitable sites for intertidal habitat restoration. Of the 3000 ha of estuarine sites designated to be restored in Flanders the following decennia, about half is to be created on using CRT. This technique enhances opportunities to combine different societal functions (safety, agriculture, ecology, inhabitation…) in coastal defence schemes and will increase the public acceptability and political willingness to implement coastal defence schemes (French 2006, Weinstein 2007).

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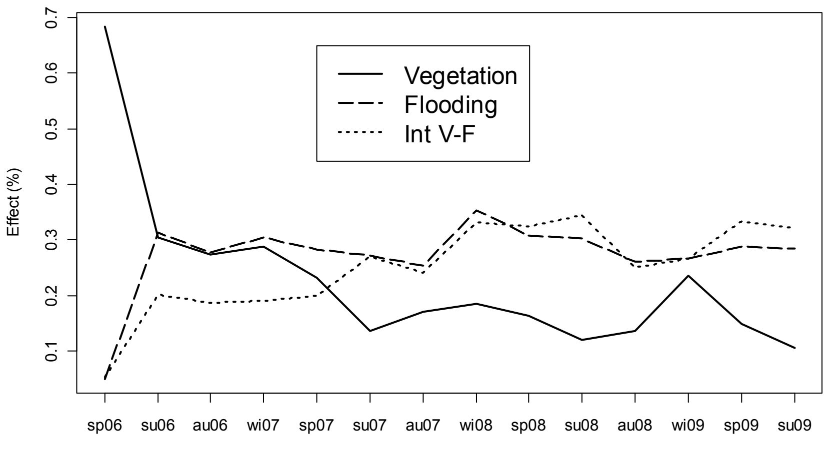
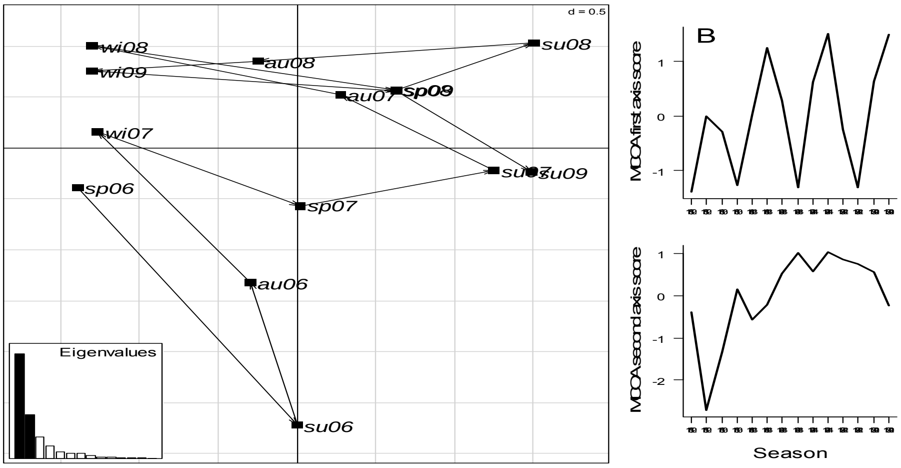
9.6.10. Aanvullingen 2008-2009

De meest recente data zijn nog niet opgenomen in bovenstaande publicatie. Daarom is hieronder een update gemaakt van enkele figuren met de meest recente gegevens uit de Omes campagnes.

PCA-IV analyse:



MCOA-analyse:



9.7. Zware metalen te Lippenbroek

Nutriënten en zware metalen werden onderzocht in het grondwater op de 10 sites te Lippenbroek (Tabel 9.5 en

Tabel 9.6). De metaal concentraties in het grondwater verschillen sterk tussen de sites. Hierbij zijn er geen duidelijke verschillen tussen sites met een hoge overstromingsfrequentie (4, 5, 6) een intermediaire overstromingsfrequentie (1, 2, 3) een lage overstromingsfrequentie (7, 10) of permanent overstroomd (8, 9).

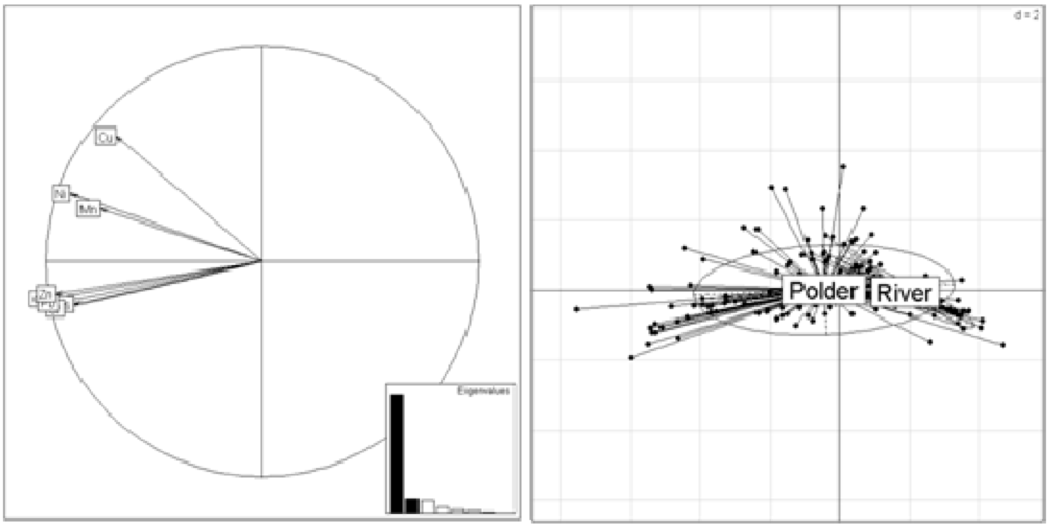
Tabel 9.5: Concentraties aan nutriënten, anionen, kationen, pH en conductiviteit in het grondwater van de 10 sites in het Lippenbroek.

site	pH	Cond	$NO_2^-+NO_3^-$	NH_4^+	PO_4^{3-}	HCO_3^-	SO_4^{2-}	Cl^-	Na	K	Ca	Mg	Fe
		µS/cm	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l
1	7.05	1420	0.29	0.32	0.30	562	89	112	82.6	13.3	233	21.2	0.12
2	7.30	1363	0.12	<0,08	0.18	427	158	150	93.4	9.1	198	18.2	0.01
3	7.35	1198	0.29	<0,08	0.28	371	127	136	82.4	10.6	170	15.9	<0,01
4	7.30	1340	0.16	2.8	0.06	634	71	106	73.1	14.0	218	20.0	0.07
5	7.60	1304	0.34	2.1	0.41	612	49	103	72.8	14.7	204	19.5	0.02
6	7.65	1204	0.21	2.9	0.11	587	25	106	68.9	16.8	176	18.3	0.08
7	7.65	1222	1.3	1.7	0.06	454	93	129	70.3	14.1	180	16.0	0.06
8	7.85	1160	2.2	0.23	0.08	302	91	165	98.9	17.3	129	16.6	0.01
9	8.20	1111	1.5	<0,08	0.09	258	92	169	98.7	17.2	114	16.0	<0,01
10	8.05	1097	1.5	<0,08	0.09	245	94	171	104	16.2	113	15.4	<0,01

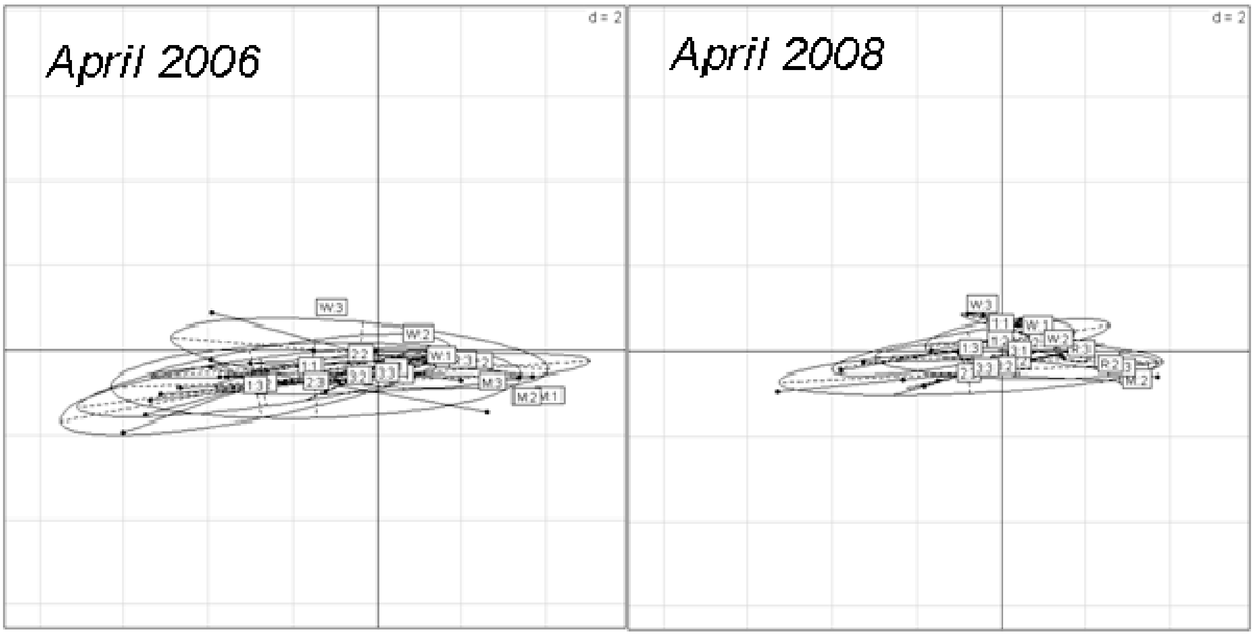
Tabel 9.6: Concentraties aan zware metalen in het grondwater van 9 sites in het Lippenbroek. Site 3 ontbreekt door een te klein staal.

site	Ag	As	Cd	Co	Cu	Cr	Ni	Pb	Zn	Mn
	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l
1	0.228	168.577	24.294	29.610	250.724	112.145	54.721	359.963	2375.970	5780.586
2	0.133	37.296	7.136	8.253	86.212	26.527	18.509	91.810	609.110	968.158
4	0.230	282.164	6.259	13.473	105.907	40.277	23.188	88.289	635.945	3427.054
5	0.154	106.516	2.294	6.462	60.042	15.967	14.070	36.971	301.156	1646.222
6	0.454	55.995	4.089	9.122	60.573	28.067	16.812	61.693	370.198	2943.414
7	0.315	117.446	25.720	37.211	300.019	134.996	72.417	443.120	3164.264	4592.504
8	0.355	44.879	2.622	5.477	49.897	18.815	13.702	62.680	258.065	1318.871
9	0.173	18.244	0.519	1.691	19.430	4.993	6.132	13.944	69.776	496.258
10	0.137	13.731	0.587	1.690	82.374	4.041	5.783	14.442	129.016	635.155

Uit de gegevens van de PCA blijkt dat bij de in werking treden van het gebied de gehalten aan metalen in het sediment van de bovenste 15 cm in het Lippenbroek hoger zijn dan in de aangrenzende slikken en schorren (Figuur 9.39). De concentraties aan metalen nemen echter af in de volgende jaren (Figuur 9.40). De historische metaal contaminatie bevond zich in het Lippenbroek nabij het oppervlak. In de buitendijkse gebieden werden de vervuilde lagen door continue sedimentatie begraven onder minder vervuild slib de laatste decennia. Door het in werking treden van het GGG zorgde de snelle sedimentatie ook in de voormalige polder voor een afname van de concentraties aan metalen in de bovenste sediment lagen.



Figuur 9.39: a) PCA van de dataset met concentraties aan metalen in sedimenten van de bovenste 15 cm. Variabelen op de twee hoofdassen en eigenwaarden-diagram. b) datapunten van het Lippenbroek (polder) en van de referentieschorren en slikken (river) bij het in werking treden van het GGG in 2006.



Figuur 9.40: PCA met eigenwaarden en hoofdassen volgens Figuur 9.39. Hierbij zijn W, R en M de referentie slikken en schorren en 1, 2 en 3 zijn sites in het Lippenbroek.

9.8. Silicium: spatio-temporele aspecten

9.8.1. Inleiding

Silicium neemt een bijzondere plaats in in het nutriëntenonderzoek te Lippenbroek. Daarom wordt dit element afzonderlijk besproken. Silicium is immers een essentieel voedingelement voor diatomeeën, welke de basis vormen van de voedselketen. Stikstof en fosfor zijn ook essentiële elementen, maar worden in de Zeeschelde verondersteld in voldoende hoge concentraties aanwezig te zijn. Silicium daarentegen is soms het limiterende element, en kan hierdoor verschuivingen in het voedselweb veroorzaken.

Omwille van deze belangrijke rol van silicium, is hier in de Instandhoudingsdoelstellingen voor het Schelde-estuarium bijzonderde aandacht besteed. De nieuw aan te leggen gebieden in het kdaer van het Sigmaplan hebben expliciete doelstellingen gekregen wat betreft Silicium export naar de Schelde. Daarom wordt binnen Omes ook bijzondere aandacht besteed aan de opvolging van dit element.

Op 10 random gekozen sites in het Lippenbroek en 6 referentielocaties werd Silicium bemonsterd in poriewater en sediment. Bijkomend werden massabalansen voor silicium opgesteld voor de 13-uurs en 26-uursmetingen. De resultaten hiervan werden gepubliceerd in verschillende artikels, hieronder weergegeven:

- **Spatiotemporal aspects of silica buffering in restored tidal marshes**

- **Amorphous and dissolved silica stocks in restored and reference freshwater tidal marshes**

De inhoud van het eerste artikel wordt hieronder gepresenteerd, de inhoud van het tweede staat weergegeven in 9.9.

“Spatiotemporal aspects of silica buffering in restored tidal marshes”

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Keywords: Wetlands, Restoration, Tidal flats, Nutrient cycles, Eutrophication, Silica, Schelde estuary, Belgium, 51°03’53”N; 4°08’55”E

9.8.2. Abstract

Losses of pelagic diatom production, resulting from silica limitation have not only been blamed for toxic algal blooms, but for the reduction in ability of coastal food webs to support higher trophic levels. Recent research has shown the importance of advective seepage water fluxes of dissolved silica (DSi) from freshwater marshes to pelagic waters during moments of riverine Si-limitation. In this study, we investigated the potential impact of recently installed new tidal areas along the Schelde estuary, located in former polder areas and characterised by so-called controlled reduced tidal regimes (CRT). Nine mass-balance studies were conducted in a newly constructed CRT in the freshwater Schelde estuary. During complete tidal cycles both DSi and amorphous silica (ASi) concentrations were monitored at the entrance culverts and in different habitats in the marsh. A swift DSi-delivery capacity was observed despite the shifted spatiotemporal frame of exchange processes compared to reference marshes. As silica-accumulating vegetation is not yet present, and difference with reference marshes’ deliveries is surprisingly small, we indicate diatomaceous debris and phytoliths to be the main silica source. Although further research is necessary on the driving forces of the different processes involved, restoration of former agricultural areas under CRT-regime provide the potential to buffer silica in the estuary.

9.8.3. Introduction

Estuaries are biogeochemical hot-spots and are amongst the most productive ecosystems of the world (Costanza et al. 1993). As the interface between terrestrial and coastal waters, they support processes that are central to the planet’s functioning (Costanza et al. 1997). Estuaries are characterized by steep chemical gradients and complex dynamics, resulting in major transformations in the amount, the chemical nature and the timing of material fluxes.

Eutrophication is one of the most important problems which confronts these systems. Eutrophication phenomena in estuaries are related to the balance between N, P and Si in river loading, and are thus dependant on the interactions between human activity and natural processes in the watershed, which ultimately determine the riverine nutrient delivery into the marine environment (Officer & Ryther 1980, Billen & Garnier 1997, Lancelot et al. 1997, Cugier et al. 2005). Eutrophication can cause anoxia, extreme turbidity and even toxicity in coastal areas and lakes, mostly provoked by shifts in plankton community following excessive inputs of N and P compared to Si. Decreases in the availability of silica relative to N and P in estuaries may result in a shift in the phytoplanktonic community from a dominance of diatoms to other phytoplankton forms as cyanobacteria or toxic dinoflagellate, affecting zooplankton and fisheries (see also Chicharo et al. 2006, Wolanski et al. 2006). Losses of diatom production, resulting from silica limitation, have not only been blamed for these toxic algal blooms, but for the reduction in ability of coastal food webs to support higher trophic levels (Cugier et al. 2005, Tréguer et al. 1995, Kimmerer 2005). Estuarine and marine foodwebs are based essentially on diatoms (Irigoien et al. 2002, Kimmerer 2005). Dissolved silica concentrations have since long been known to control diatom populations (Wang & Evans 1969), diatom blooms (Tessenow 1966, Schelske & Stoermer 1971, Davis et al. 1978), and seasonal succession in plankton communities (Kilham 1977). In fact, the availability of dissolved silica (DSi) has been shown to control diatom silica production rates, at least seasonally, in every natural system examined to date (Nelson & Brzezinsky 1990, Nelson & Tréguer 1992, Brzezinski and Nelson 1996, Nelson & Dortch 1996, Brzezinsky et al. 1998, Bidle & Azam 2001).

Within the estuarine ecosystem, fringing tidal marshes act as a biogeochemical filter, removing inorganic and organic substances from the floodwaters and changing substance speciation (e.g. Gribsholt et al. 2005). The interaction between tidal marshes and estuaries or coastal zones received much attention through numerous exchange studies (e.g. Valiela et al. 1978; Spurrier & Kjerfve, 1988; Whiting et al. 1989; Childers et al. 1993), with emphasis on C, P and N. Dominant questions were whether marshes were importing or exporting N, P, C or particulate matter, often testing the 'outwelling' hypothesis (e.g. Dame et al. 1986), which states that a large part of the organic matter produced in the intertidal marshes is not used in internal trophic chains but is transported into the adjacent sea areas and increases their productivity. Only a limited number of mass balance studies have targeted freshwater tidal marshes (e.g. Childers & Day, 1988; Gribsholt et al. 2005; Struyf et al. 2006). The freshwater systems are characterized by botanical properties resembling inland freshwater wetlands and by more direct contact with human-impacted river water. These characteristics make freshwater tidal marshes potentially important process interfaces. Struyf et al. (2006) have shown the importance of advective seepage water fluxes of dissolved silica (DSi) from freshwater marshes to pelagic waters during moments of riverine Si-limitation. Tidal freshwater marshes contain large amorphous silica stocks in marsh soils, built up through sedimentation of diatom shells and incorporation of silica in marsh vegetation (Struyf et al. 2005). Export is the result of consequent dissolution of this amorphous silica (ASi) in marsh pore water from litter and sediments, and advective export of marsh pore- and puddle water between tidal flooding events (Struyf et al. 2007a,b). Silica limitation of diatoms (Conley et al. 1993, Smayda 1997) and the consequent negative effects on food web structure may be avoided. However, data is only available from few tidal freshwater and conclusions are presently only applicable on a local scale. Furthermore, a recent review stresses the need for more research on silica cycling in wetlands, as it rivals their impact on other biogeochemical cycles and, to date, this topic has not received sufficient attention (Struyf & Conley 2008).

In this study, we investigated the potential impact of recently installed new tidal areas along the Schelde estuary, located in former polder areas and characterised by so-called controlled reduced tidal regimes (CRT's) (Maris et al. 2007, Cox et al. 2006). Along the Schelde estuary, more than 50 % of marsh area will eventually be located in such areas, and may result in international application. This article focuses on the silica biogeochemistry within these new systems and aims to explore spatiotemporal patterns of deposition and dissolution in recently flooded formerly agricultural polder areas. In the first implemented CRT, an intensive spatiotemporal sampling scheme was carried out during the first 16 months of development. This research expands the growing awareness that ecosystems and associated biogenically fixed amorphous Si rather than geological weathering control silica availability in the aquatic environment on a shorter, biological timescale (Conley 2002, Derry et al. 2005, Humborg et al. 2004).

9.8.4. Materials & Methods

Nine mass-balance studies were conducted in a newly constructed CRT in the freshwater Schelde estuary: on May 16th; July 3rd; September 10th and 11th and October 10th 2006, and on March 20th and 21st and June 4th and 5th 2007. During 9 complete tidal cycles both DSi and ASi concentrations were monitored at the entrance culverts as well as in different habitats in the marsh.

Study area

The Schelde estuary is one of the last European estuaries with a complete fresh- to saltwater tidal gradient, located in the Netherlands and Belgium. Maps and extensive descriptions of hydrology and ecology can be found in several recent papers (Temmerman et al. 2003, Meire et al. 2005; Van Damme et al. 2005; Soetaert et al. 2006). The studied CRT area is a newly constructed inundation area, the "Lippenbroek" (surface approximately 80.000 m^2), situated at Moerzeke (51°03'53"N; 4°08'55"E). Maximal tidal amplitude in the Schelde at this point is approximately 6m. The area was mostly used as cropland (rotation system with *Zea mays* and *Solanum tuberosum*; the lower parts were planted with *Populus sp.* trees or over-grown with *Salix sp.* trees (Fig. 1A). The rotation system was abandoned in 2003.

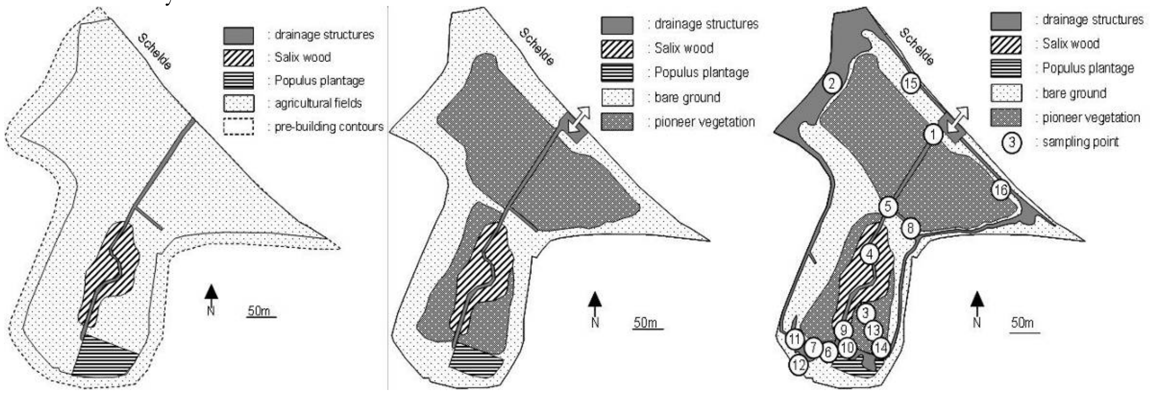


Figure 1: schematic overview of study site before (A, left) and after building works (B, middle). C (right): Sampling locations in the CRT. Sampling intensity at locations is given in Tables 1 & 2.

During the two-year construction phase (2003-2005), crops were replaced by pioneer vegetation (mainly *Epilobium hirsutum* and *Urtica dioica*) (Fig. 1B). Part of the polder was devegetated due to building construction work (Fig. 1B). Tidal inundation was initiated in March 2006. Since the first inundation, vegetation has been progressively replaced by flood-tolerant species (mainly *Lythrum salicaria, Lycopus europaeus* and *Phragmites australis*) (Fig. 3). Because site elevation is several meters under mean high water level, reconstruction of spring-neap tide flooding variation required the construction of separate inlet culvert and outlet culvert (Maris et al. 2007). At the riverside, an inlet culvert permits flooding from 4.80m TAW and higher, whilst a valved outlet culvert guarantees one-way emptying from 1.5m TAW and lower (TAW is the Belgian Ordnance Level, which is approx. 2.3m below mean sea level at the Belgian coast). Consequently, only the top of the tidal cycle is permitted to flood the polder surface. This results in a controlled reduced tidal area (CRT) with unique tidal features, such as a pronounced spring-neap variation and a prolonged stagnant phase (Fig. 2, for details see Maris et al. 2007, Cox et al. 2006). The marsh is surrounded by a dike at 8m TAW. Because of the deep artificial dike bases and thick riverine clay deposit in the CRT, ground water fluxes were assumed to be small compared to observed tidal surface water fluxes.

Figure 2: Vegetation development in devegetated zones. A and B (upper left and right) show overview; C and D (lower l & r) detail. A and C are taken in spring 2006 (one month after first inundations), B in summer 2006, and D in summer 2007.

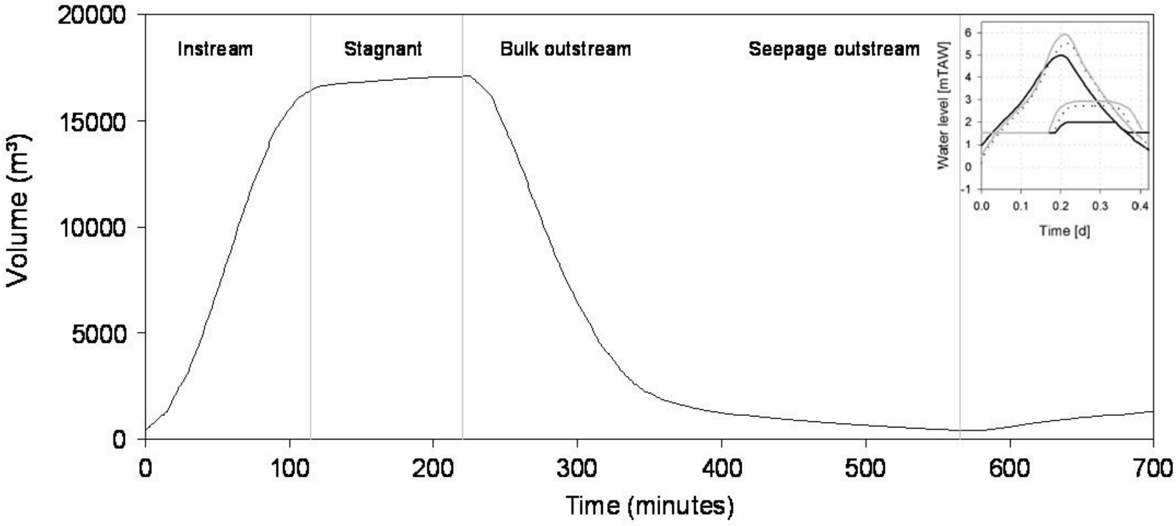


Figure 3: Water mass balance during a typical tide in Lippenbroek. Inset illustrates tidal curves at typical neap (–), mean (· · ·) and spring (–) tide outside and inside Lippenbroek CRT (from Cox et al. 2006). Grey lines indicate (0) start instream, (110) stop instream =start stagnant phase, (220)

start outstream = stop stagnant phase and (580) stop outstream. Outstream consists of a bulk outstream (overmarsh tidal frame) and a seepage phase (here at approx. 340min.Phase lines are indicated in relevant figures throughout the MS.

Sampling

A total of 796 data points were obtained during the 9 mass balance studies. Surface water samples were collected at the entrance and outlet culvert (1 in Fig. 1C) and in selected habitats throughout the marsh (2-16 in Fig. 1C). Sampling covered the full 13 hours of the tidal cycle for May, July and October 2006 campaigns, and double cycles of one night (“a” in text) plus day (”b” in text) of 26 hours for September 2006 and March and June 2007 campaigns. Sampling intensity was highest during the first campaign (Table 1 & 2). This intensity was necessary to explore the spatial patterns in the marsh; however this exhaustive scheme was not entirely repeated during all campaigns. The selection of habitats during subsequent campaigns was based on maximal cover of different habitat features. A selection of samples was analysed for ASi (table 2), also covering different habitat features.

DSi-samples										
Location	**May 06**	**Jul 06**	**Sep 06a**	**Sep 06b**	**Oct 06**	**Mar 07a**	**Mar 07b**	**Jun 07a**	**Jun 07b**	**Total**
1	53	8	15	17	24	16	11	15	15	174
2	16	23	57	11	12			5	5	129
8	7		12	12	12					43
6	10	10			12			5	5	42
5	10	8			12					30
3	8	8			12					28
17		21								21
4	10	9								19
15	16									16
16	14									14
12	12									12
9	8									8
11	7									7
10	5									5
14		5								5
13		3								3
7	3									3
Totaal	179	95	84	40	84	16	11	25	25	559

Table 1: Sampling intensity at different locations (see Fig. 1C) and during different campaigns

ASi-Samples										
Location	**May 06**	**Jul 06**	**Sep 06a**	**Sep 06b**	**Oct 06**	**Mar 07a**	**Mar 07b**	**Jun 07a**	**Jun 07b**	**Total**
1	12	16	4	5	12	8	6	15	15	93
2		9	7	7	12			5	5	45
6	6	5			12			5	5	33
8			4	6	12					22
3		3			12					15
17		12								12
5	6									6
4	6									6
11	5									5
Total	35	45	15	18	60	8	6	25	25	237

Table 2: Selection of samples analysed for ASi at different locations (see Fig. 1C) and during different campaigns

Samples were taken approximately 10 cm below the surface, and stored in dark incubators at 5°C for a maximum of 24h. Dissolved silica (DSi) was analyzed on a Thermo IRIS ICP (Inductively Coupled Plasmaspectrophotometer) (Iris®). For each of the samples analysed for ASi (table 2), 3 sub samples of 25ml each were filtered over 0.45 µm filters, from a well-mixed total sample of 150 ml. After drying at 20°C, ASi was extracted from the filters in a 0.1M Na_2CO_3 solution at 80°C in a shaker bath. Sub samples were taken at 60, 120 and 180 min. Blank extractions revealed insignificant DSi release from filters, recipients or chemicals. [ASi] (mg l^{-1}) was then calculated by extrapolating the linear line through the three extraction points in a time-extracted silica plot (De Master 1991). This approach corrects for the additional release of Si from mineral silicates. The ASi wet-alkaline extraction is prone to additional release of DSi from amorphous mineral silicates. Despite its flaws, ASi wet-alkaline extraction is for the moment still the most representative method to analyse for ASi (Saccone et al. 2007).

Water and Silica mass Balances

All calculations and statistical analyses were performed in R (R 2006). Inlet and outlet culverts are the only exchange points with the river. Their dimensions are exactly known. Flow velocity was measured acoustically (Sontek 'Argonaut'). Water mass balances were calculated with an averaged discharge value throughout the water column for every two minutes, assuring accurate volume-weighing of concentration values during all tidal phases. Measurements, calibration and operation of the flow meters were perfomed by Flanders hydraulics research laboratory (WL) experts. Concentration profiles as well as nutrient discharges were calculated as linear interpolations (Becker et al. 1988). ASi-concentrations were measured on average 10 times (range 4 - 16) and DSi-concentrations 19 times (8-53) along each tidal cycle at the culverts. Interpolation provided 700 values/tidal cycle, for discharge (D) as well as concentrations. These values were used to calculate absolute mass balance by cumulative summing of (D (m^3) * [Si] (mg l^{-1})) along the instream and outstream phase separately. Stagnant phase and volumes entering through small leaks in the outstream culvert were not taken into consideration.

Total mass balances were first calculated as percentages ((∑(out) – ∑(in)) / ∑(in)) in order to compare between different tidal volumes, and then as absolute masses (∑(out)-∑(in)). In a conservative mass balance, it is assumed that there is no net import or export of water. However, due to inter-tide variations, stocking or surplus release of water volumes takes place. To compare between tides, this conservative correction was calculated as a percentage for each campaign, recalculated on the final mass balance and shown as a range. However, general patterns were not seriously influenced by this effect. (Fig. 5,6)

Additional tidal features were measured in order to compare between tides: average flooding height was calculated from total volume of each entering tide and total surface of the study area, water temperature was continuously monitored at culverts, and DSi- and ASi-concentrations were monitored in adjacent river water. General relationships between silica delivery and these tidal features were explored through PCA and ANOVA analysis (Chevenet et al. 1994).

9.8.5. Results

Concentration profiles

DSi concentration profiles show different seasonal patterns (Fig. 4). Instream phases (see Fig. 2) are marked by steep concentration changes, whilst the fluctuations during stagnant phase do not exceed 0.2 mg l^{-1}. Outstream concentration profiles are highly variable and show increases, decreases or both: at starting concentrations below 2 mg DSi L^{-1}, concentrations increase with 125.0% and 126.6% (June 2007 in Fig. 4) or even with a factor 17 (July 2006 in Fig. 4) at final concentration. For instream concentrations higher than 2 mg/l, profiles show slight increases (10.5% to 31.9%, May, September and October 2006 in Fig. 4). When instream concentrations become higher than 6 mg l^{-1}, profiles show a status quo or slight decrease (6%, March 2007 in Fig. 4) towards final concentrations.

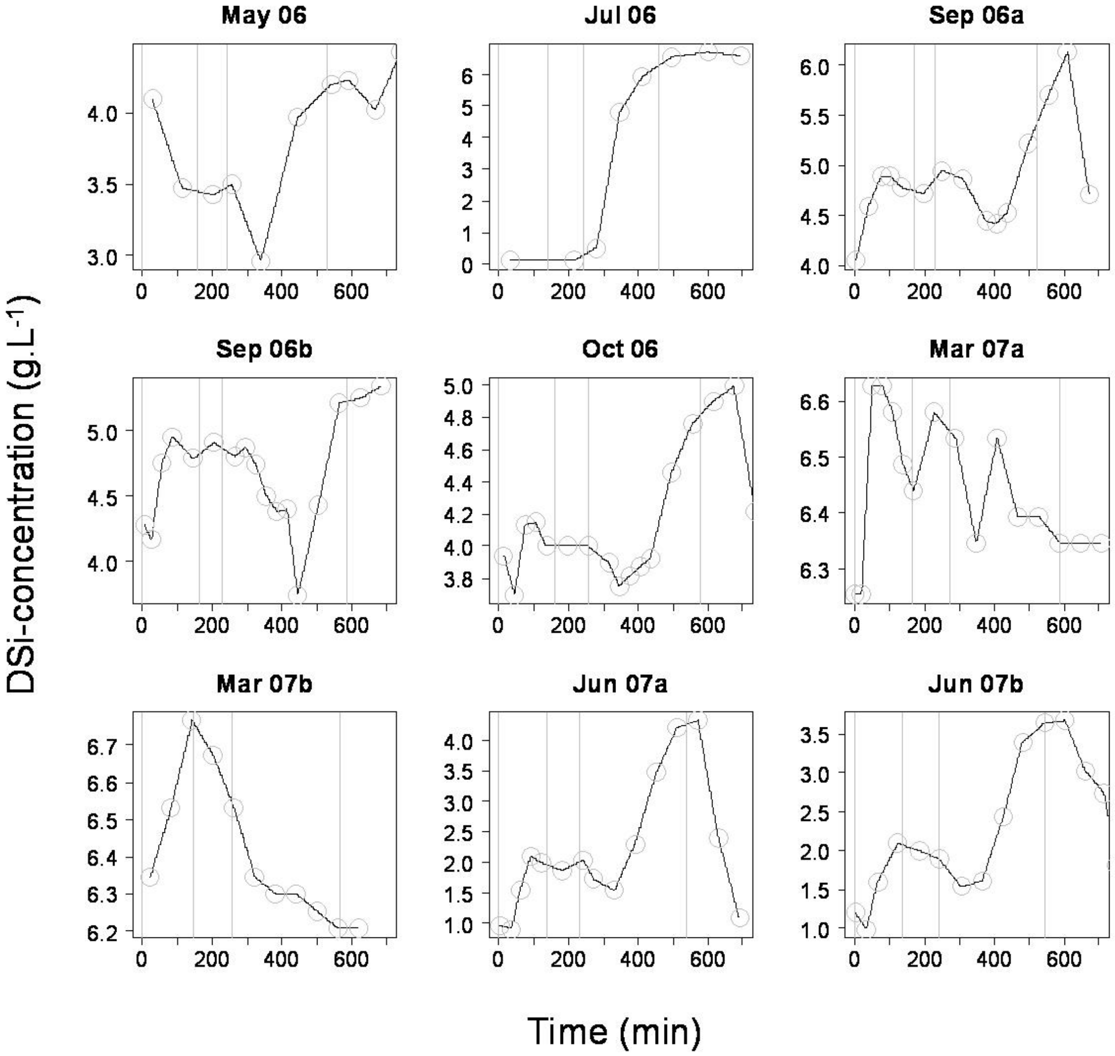


Figure 4: DSi-concentration profiles at in- and outstream location for all campaigns (tidal phases as in Fig. 3). Campaign month is indicated above each subpanel (dots are measured concentrations, lines represent linear extrapolation).

Concentration profiles of ASi present a more variable pattern over a smaller concentration range (not shown). Although for September 2006 and March 2007 only a limited number of samples was analysed for ASi, and differences are generally lower for ASi compared to DSi, there is a general evolution of increase or status quo (May, July, September 2006 and October 2006, not shown) towards strong ASi decrease in ASi- concentration profiles during later tidal cycle (March and June 2007b, not shown), with the exception of June 2007a.

Mass balances

Calculated DSi mass balances indicate enrichment of exported water in summer months (July 2006 and June 2007, Fig. 5 upper left), but also in late autumn (Oct. 2006) and during one spring campaign (March 2007). Although July 2006 shows spectacular enrichment in percentage, absolute numbers (Fig. 5 upper right) are lower due to small water mass at neap tide, while the opposite is true for the 2007 campaigns. ASi mass balances confirm the transition from slight ASi-delivery or status quo towards ASi-caption by the marsh (Fig. 5 lower graphs).

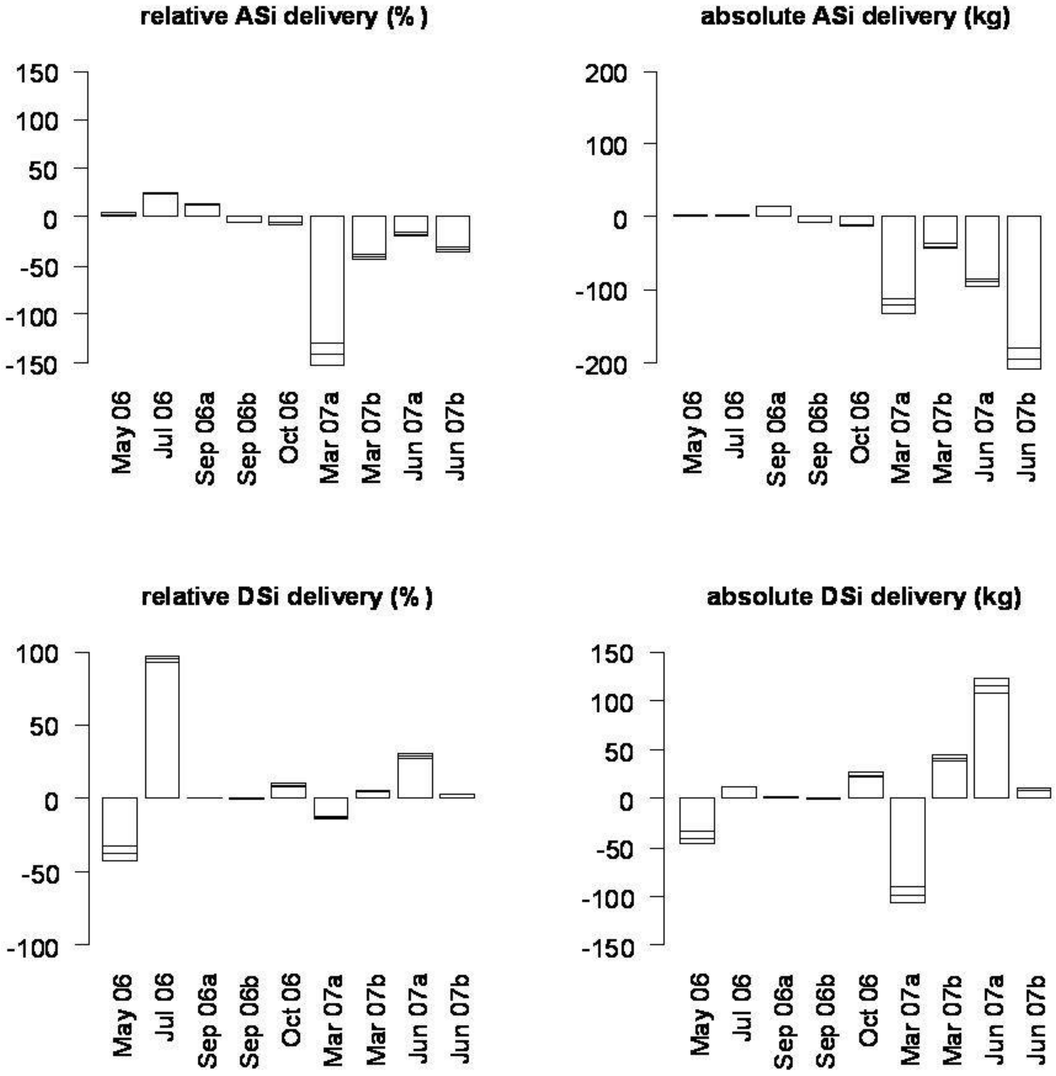


Figure 5: DSi (upper graphs) and ASi (lower graphs) mass balance of all campaigns. Balance is represented as percent, (out(g) –in(g)) / in(g))(left graphs); and in absolute numbers (kg delivered/retained)(right graphs). Error bars represent deviation from conservative mass balance, represented as percent in both directions.

Total silica mass balance (Fig. 6) is strongly influenced by ASi-retention, which adds up to Si retention in May and September 2006a and March 2007a (Fig. 5,6) or overrules DSi-delivery in June 2007b (Fig. 5,6). Yet, despite big ASi-retention in the March 2007b and June 2007a-campaigns , DSi-delivery can still be sufficient (Fig. 7 upper graphs) to provide the estuary with net silica (Fig. 6).

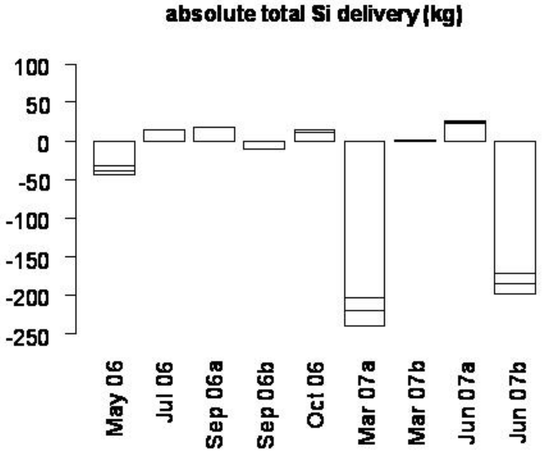


Figure 6: Absolute total silica mass balance (DSi+ASi). Error bars represent deviation from conservative mass balance, represented as percent in both directions.

Ephemeral and Diurnal aspects

Features of each tidal cycle are listed in Table 3. Non-numerical variable classes, being "day/night" show opposite differences in March vs. June 2007 campaigns. Variation of obtained numeric variables was maximised along two components (PCA correlation circle, see materials and methods, Fig. 7). Parallel vectors indicate high correlation along the two components, whilst squared vectors are not correlated. Opposite direction indicates a negative correlation. Explicit R-squared and p-values depend on the amount of variation explained by the components. This exploration of numerical variables shows potential effects of riverine DSi-concentration ("dsisch" in Fig. 7), average flooding depth ("depth" in Fig. 7) and temperature ("t" in Fig. 7) on ASi- and DSi- balance ("asi" and "dsi" in Fig. 7), while the balances are completely independent of riverine ASi-concentrations ("asisch" in Fig. 7).

Season	Asi (%)	Dsi (%)	$T_{schelde}$ (°C)	$[DS_{schelde}]$	$[ASi_{schelde}]$	mean water depth (cm)	Time instream	day/night	Tidal volume (m³)
May 06	3.6	-37.1	19.5	4.2	1.5	30.0	6:00	D	24000
July 06	24.7	94.8	24.0	0.2	1.5	5.6	9:30	D	4500
Sep 06a	13.2	1.0	18.0	4.0	1.5	42.5	18:20	N	34000
Sep 06b	-6.3	-0.2	22.5	4.2	2.0	43.8	6:35	D	35000
Oct 06	-7.1	9.3	14.4	8.5	3.5	33.8	5:45	D	27000
Mar 07a	-140.7	-12.2	7.9	6.2	1.5	96.3	16:25	N	77000
Mar 07b	-40.6	5.6	9.0	6.3	1.4	96.3	5:30	D	77000
Jun 07a	-17.3	29.6	19.8	1.0	3.5	17.5	18:46	N	14000
Jun 07b	-33.4	2.5	23.8	1.0	3.5	17.5	7:00	D	14000

Table 3: percent DSi and ASi delivery and general tidal features of different campaigns.

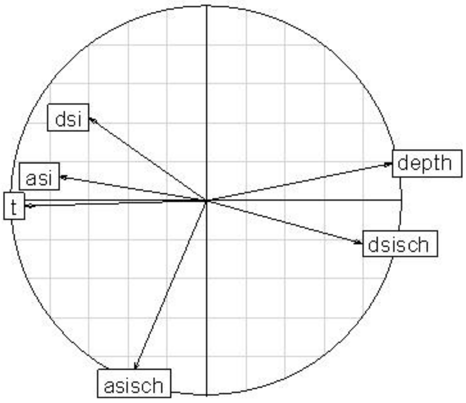


Figure 7: PCA correlation circle of table 3. Variation of numeric variables is maximised along two components. Mean water depth (“depth”), riverine ASi and DSi (“asisch” and “ dsisch”) and mean water temperature (“t”) are shown with ASi and DSi percent delivery (“asi”, “dsi”). Parallel vectors are highly correlated along the two components, while squared vectors are not correlated. Opposite direction indicates a negative correlation. Explicit R-squared and p-values depend on the amount of variation explained by the components.

Linear as well as logarithmic correlations were tested for numerical variables; highest significant R-squared values are shown. Percent DSi-delivery was negatively correlated with the logarithm of riverine DSi-concentration ($R^2 = 0.6397$, p-value 0.0096), and the logarithm of mean water depth ($R^2 = 0.52$, p-value 0.02), and not (significantly) correlated with temperature of outstream water, percent ASi delivery, or riverine ASi concentrations.
ASi-delivery was negatively correlated with the mean water depth ($R^2 = 0.5317$, p-value 0.0257) and positively with the logarithm of temperature ($R^2 = 0.5787$, p-value 0.0172), ASi delivery was independent of riverine ASi-concentrations (not shown).

Spatial Aspects

Concentration profiles observed at other locations were not used for local mass balances, because water volumes and tidal phases were not measured separately for these locations separately. However, interesting patterns are visible when comparing local concentration profiles: During the first campaign (May 2006, Table 1), intensive sampling of 15 locations throughout the marsh took place. A clear gradient in increasing vs. decreasing DSi-concentration profiles was observed (Fig. 8) despite the relatively small surface area, and the fact that the marsh was only functionally connected with the estuary less then two months before this campaign. Concentrations in the polder even out with the riverine concentration at bulk water entrance at the start of the instream phase. Concentrations appear unchanged during stagnant phase (minute 150 until 250 after instream in Fig. 8), but an hourglass pattern appears during outstream, when concentrations from different locations diverge strongly. (Minute 300; Fig. 8). Typical retention habitat during the May 2006 campaign was the tidal pool (site 2 on Fig. 8 and Fig. 1C), whilst opposing behaviour is observed at the lower mudflat (site 3 on Fig. 8 and Fig. 1C). Locations situated in-between these two average these two extremes, as do the outstream concentrations (site 1 on Fig. 8 and Fig. 1C) and. The hourglass structure returns, albeit less complete, in the other spatially sampled campaigns in the summer and autumn with net DSi-delivery (not shown).

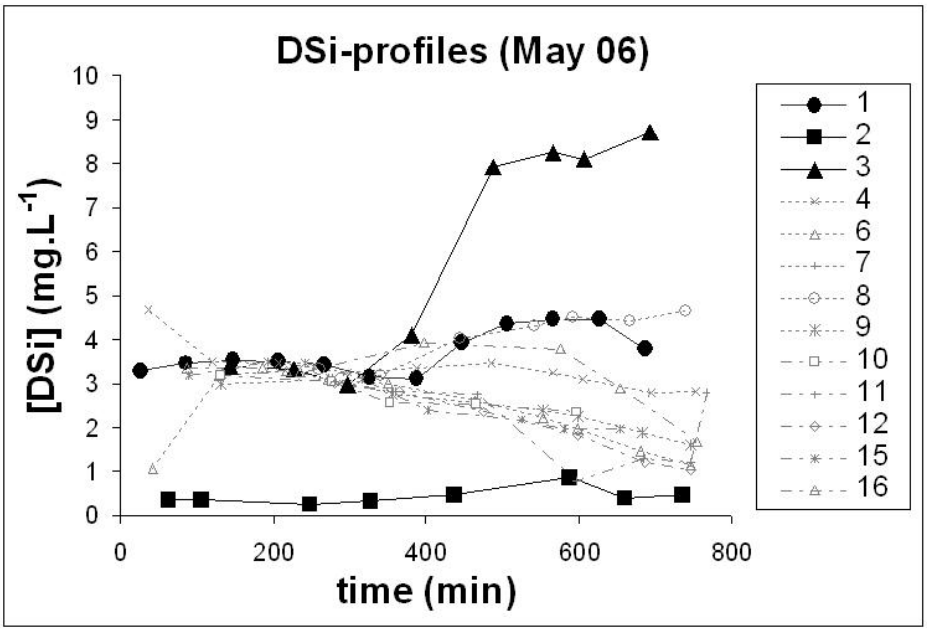
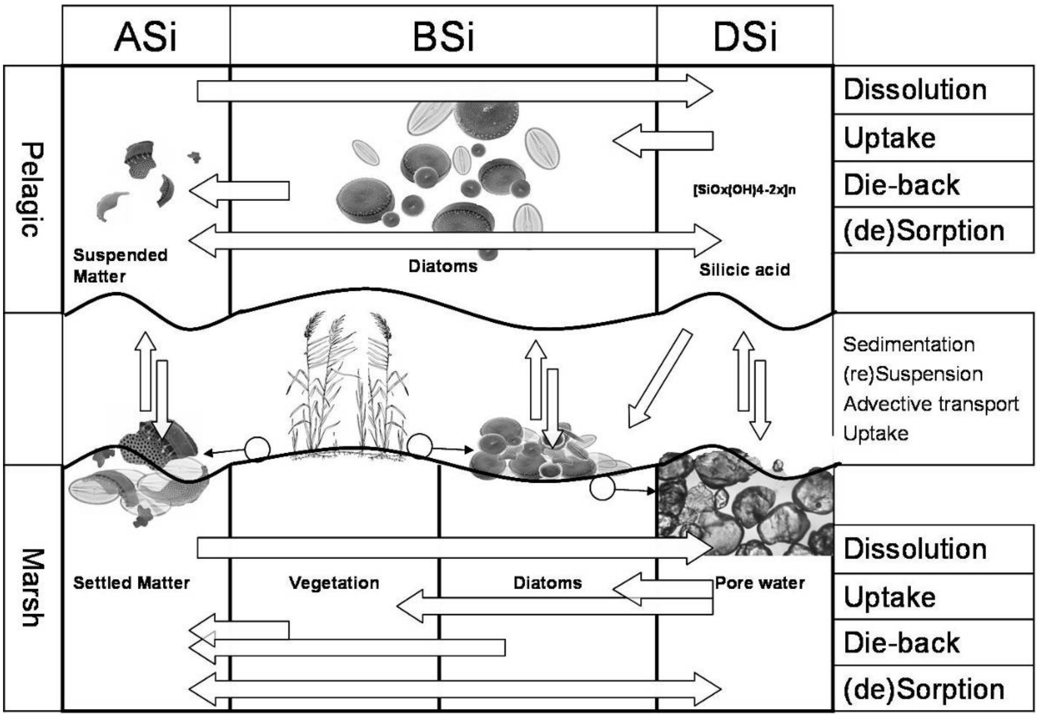


Figure 8: DSi- concentration profiles at all locations (see Fig. 3) sampled in the May 2006 campaign. Tidal pool (2) and lower mudflat (3), as well as outstream location (1) are shown in black, other locations in grey.

A similar spatial sampling took place for ASi (Table 2). Homogenisation of concentration at instream and spatial patterns in general are less clear. Still, the lower mudflat site delivers ASi during the May, July, October 2006 and June 2007 campaigns. Tidal pool habitat delivers ASi during the July 2007 campaign. Other locations were year-round ASi-sinks.

9.8.6. Discussion

Numerous processes are involved in the silica-exchange between tidal marsh and flooding water (Scheme 1). In the following overview, these processes are described and linked to the obtained data.



Scheme 1: schematic overview of processes involved in silica cycling in the intertidal ecosystem.

Diffusive and advective transport

In several systems, the importance of advective ground water fluxes to the estuarine nutrient balance has been emphasized (i.e. Herrera-Silveira 1998, Niencheski et al 2007, Hays & Ullman 2007). However, this mainly concerned sandy coasts, where ground water fluxes are more important than in the dense clay soils characterising the Schelde.

As previously explained, the key process in silica-buffering by tidal marshes is the swift replenishing of dissolved silica (DSi) in the flooding water during seepage (e.g. Struyf et al. 2006). Sediment-water silica fluxes are normally assumed to be straightforward, with release due to dissolution of siliceous material, controlled by temperature (e.g. Hammond et al, 1985, Mortimer et al. 1998) and consequent diffusion in the water. However, calculated diffusive silica fluxes deviate from measured fluxes (Callender & Hammond 1982, Mortimer et al. 1998), and the relatively low apparent rate constants obtained from pore water data cannot explain the observed fast dissolution kinetics (Nelson et al, 1995). Some diagenetic models invoke biogenic opal fractions (Wong & Grosch 1978, Dixit et al. 2002) and decreasing dissolution rate coefficient with advancing early diagenesis (McManus et al. 1995), or seek for solutions in Al-content and specific surface area of biosiliceous fragments (Dixit et al. 2001). Other authors, however, have suggested the importance of bioturbation (Berner 1980, Meile and Van Capellen 2005), resuspension (Mortimer et al. 1998), bioirrigation (Aller 1965, Mortimer et al. 1998), advection and subsurface circulation patterns (Vanderborght et al. 1977) in diagenetic equations.

Empirical data on tidal marsh exchange revealed the importance of advective transport and gravitational drainage for silica-delivery to the estuary (Struyf et al. 2006), suggesting a physical thrust of porewater refreshment by the flooding water, with tidal marshes acting as leaky dams (Hackney et al. 2000).

Our data confirm these conclusions: In all measured campaigns, DSi-concentrations did not increase during stagnant phase or during bulk outstream, even when water depths are low throughout the site. DSi-concentrations only increase when advective outflow or seepage starts. Seepage is characterized by the absence of continuous water bodies (overmarsh tide) in the marsh (except in tidal pools). Outflow is then generated by small streams within the litter and surface layer of the marsh, or by advective horizontal outflow at creek banks from within the sediment. The bulk-seepage switch coincides with a decrease in flow velocity, visible in the outstream curve. In fact, the end of overmarsh tide and start of seepage coincides with a sudden increase in DSi-concentrations in outstream water for almost all campaigns. This strengthens our hypothesis that DSi-efflux from marshes to the estuary is a purely physical process of gravity driven fluxes of Si-enriched pore and puddle water. This also explains why a recent modelling study, based only on diffusive fluxes directly into overmarsh tide, minimized the role of marshes in the estuarine silica cycle (Arndt et al. 2007). Empirical studies, including this one, have already indicated the importance of physical export processes (e.g. Vanderborght et al. 1977; Scudlark and Church 1989).

The often observed decline in concentrations at the very end of the outstream phase is probably due to riverine water slowly entering the exit culvert. This leaking volume, which is important for fish migration, probably lowered concentrations at the culvert sample location during the last samples of each tide, when outstream had already stopped.

Sedimentation and resuspension

Sedimentation and recycling of suspended particulate silica are key processes in the marine and lacustrine silica cycle (Tréguer et al. 1995, Bidle & Azam 2001). However, in estuaries, it has been shown that regeneration of silicic acid from particulate silica does not account for observed summer increases in silicic acid concentration (van Bennekom et al. 1974, Yamada & d'Elia 1984). In some estuaries, at least during summer, regenerated silicic acid inputs exceed allochtonous outputs to the water column (Wilke and Dayal 1982, Yamada & d'Elia 1984). These input result from regeneration of silicic acid from recently deposited biogenic silica in diatomaceous debris in the sediments (Van Bennekom et al. 1974), and from biogenic silica in vegetation litter in tidal marshes (Struyf et al. 2007b).

In our study site, only a small proportion of the vegetation consisted of *Phragmites australis* during the experiments, although the proportion has recently increased. The remaining biomass consisted of species which do not accumulate silica, with the possible exception of *Urtica dioica* (Struyf et al. 2005). DSi-delivery occurred equally from location 4 during the initial *Urtica*-dominated period, over the period of gradual decay of the *Urtica* vegetation, towards a rather bare mudflat stage. This may also indicate the dissolution of former buried phytoliths (plant silica bodies) from former vegetation (including crops like *Zea mays*). The observed, recent colonization by *Phragmites australis* and *Typha angustifolia* may enhance ASi available for dissolution and DSi-delivery to the pore water (and to the estuary) in the long run, since *poales* such as the latter species are well-known for their capacity to control silica fluxes (Blecker et al. 2006).

DSi-delivery appears, up until now, to take place independent of vegetation composition. Therefore, we suspect dissolution of deposited diatomaceous debris, next to phytoliths from former vegetation (*Urtica dioica*, *Zea mays*), to be the source of this DSi. Sedimentation rates are several times higher in the CRT then at reference marshes, which is typical for their status of young marshes (Temmerman et al. 2004). Also, sedimentation rates are highest at observed DSi-source locations (3 in Fig. 1C), and lowest at typical DSi-sink locations (2 in Fig. 1C) (Vandenbruwaene, unpubl.data).

When discussing patterns of ASi delivery, it is necessary to emphasize that it ASi concentrations are probably underestimated, as only surface samples were taken, and suspended solids concentrations are expected to be higher near the bottom. Also, patterns for ASi are less clear than DSi, although for July 06, October 06 and June 07 campaigns, sampling effort was increased. The ASi-concentrations drop between the moment of maximum instream discharge and the stagnant phase (not shown) probably points to important sedimentation, mostly during the stagnant phase. This is in accordance with our hypothesis that recently deposited materials contribute to the observed export of DSi, as settling sediments consist to a great extent of diatoms (De Schutter J., unpubl.data).

During outstream ASi-concentration patterns oppose DSi-profiles. The pattern of ASi concentration increase during bulk outflow, and decrease in the seepage phase, shows that ASi is exported with the bulk water, while DSi was mostly exported during seepage. During the first months, marsh surface was mostly unvegetated, and no creeks were present (as shown in pictures, Fig. 3). Throughout the first summer, small creeks developed, vegetation and algal layers began to consolidate marsh soil while sedimentation and erosion zones became apparent, as in reference marshes. Probably, this has increased deposition/resuspension ratio and promoted the high net ASi-retention observed in the later campaigns.

Diatom die-back and frustule dissolution

Although the effects of environmental factors (such as salinity, pH, temperature,…) on the *solubility* of silicic acid have been well studied (Greenberg & Price 1957, Hurd & Theyer 1975, Kamatani and Riley 1979), silicic acid is likely be found at concentrations considerably below saturation in the water column and surface sediments of most estuaries (Yamada & d'Elia 1984). According to these last authors, factors affecting *dissolution rate* of biogenic silica and the amount of substrate available are of greater importance in determining the rate of regeneration of silicic acid in estuaries (Yamada & d'Elia 1984). They point out that the deposition rate of biogenic material to the sediment surface, the salinity of the overlying water and the ambient temperature all have substantial effects on the silicic acid efflux from estuarine sediments. Bacteria have been shown to play a major role in dissolution of diatoms in oceanic (Bidle and Azam 2001) and lake systems (Patrick and Holding 1985). This bacterial mediation of potentially rapid and highly variable silica regeneration might also apply for intertidal marshes, especially in the lower tidal range, where decomposition of vegetation and consequent dissolution of phytoliths will play a less important role.

Our data confirm a positive temperature-dependency (Yamada and d'Elia 1984), since DSi-delivery increases with temperature (table 3, Fig. 9) albeit not significantly. Maximum relative deliveries in our study were observed in summer, when surface and water temperatures of 35°C and more (Beauchard, O. pers.comm) were observed on the mudflats and in small creeks during ebbs. Not only does increased temperature directly boost silica-efflux from sediments (Yamada & d'Elia 1984), but it also enhances efficiency of bacterial removal of the organic carbon matrix from diatoms, increasing the surface area of naked silica exposed to dissolution and fastening chemical dissolution rates (Bidle &Azam 2001). Moreover, the specific ebb conditions could affect bacterial activity on diatoms in a positive way, a mechanism that must be looked into.

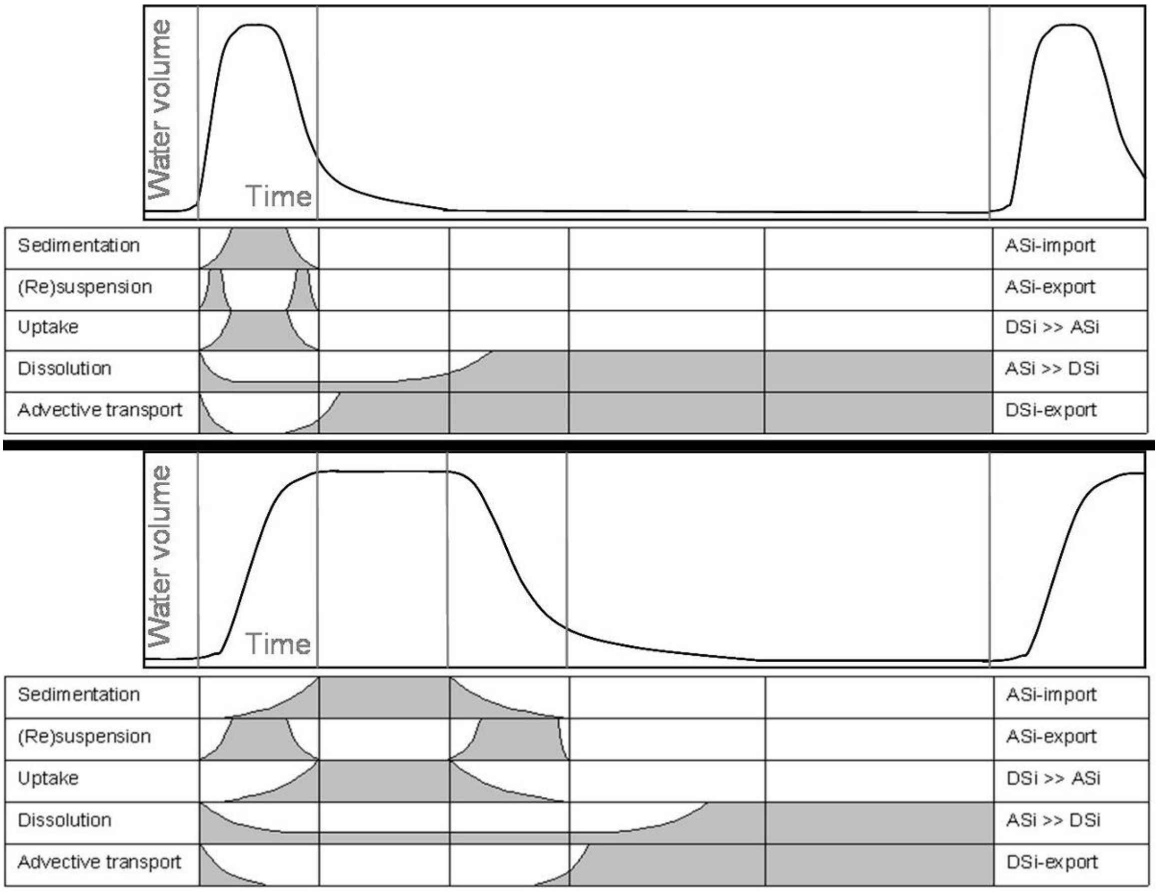
Next to the temperature, ambient DSi concentrations certainly play a role, as relative export will increase with lower incoming DSi concentrations, observed in summer. The exponential character of the relationships between DSi and concentration or volume of flooding water (Fig. 10A and B) seems logical: if the marsh has a certain delivery capacity, this standard amount will make up a bigger percent of incoming DSi when incoming DSi concentration or volume is lower. Struyf et al. (2006) suggest this enhanced export at lower ambient DSi might be due to increased ASi dissolution at undersaturated conditions (e.g. Greenwood et al. 2001, Rickert et al. 2002, Passow et al. 2003). Also, the relationship could be magnified by temporal concurrence of low riverine DSi concentrations with periods of high temperature in summer. The June 2007 mass balance showed DSi-delivery mainly taking place during the night. This suggests that day delivery might be lower, due to uptake of DSi by autochthonous diatoms. This diurnal aspect certainly deserves more attention.

Si-uptake by diatoms

The DSi-retention in the tidal marsh, until now not observed in important quantities (Struyf et al. 2006) may be the most striking difference of this study with earlier findings. The cause is almost certainly diatom uptake. In tidal pools, DSi-concentrations dropped immediately after instream, while oxygen oversaturation indicated high diatom primary production (Maris, pers observ). However, in later campaigns, benthic diatom populations were observed throughout the marsh (Jacobs, pers obs), as were decreasing DSi-profiles during stagnant and bulk phase (not shown). Settling of the sediments and very low depth compared to the pelagic optimizes light conditions and surface aeration. During the stagnant phase, which is a specific feature of the CRT (Maris et al. 2007; Cox et al. 2006), optimal conditions for diatom production are thus provided throughout the marsh. The fact that the July 2006 campaign does not show this decrease in DSi is a consequence of the concentrations being close to zero at that time in the estuary, resulting in big relative DSi-delivery during this campaign. Again, the day night variation observed in Si delivery seems to confirm DSi uptake by diatoms (see earlier); silica-incorporation by diatoms is halved in dark situation (Azam & Chisholm 1976), and surface temperatures at night are lower, resulting in higher DSi export during nights. The uptake process is dominant in spring, as low ambient DSi concentrations disadvantage production and promote relative export numbers in summer campaigns.

Interactions

In contrast to other systems where silica-fluxes are studied, processes influencing silica-cycling from tidal marshes are mostly separated in space and/or time. (Scheme 2). It is possible that specific physical features of tidal marches account for their recycling capacities. Specific tidal patterns in CRT's influence the observed processes and their temporal distribution, most likely in favour of import processes (Scheme 2).



Scheme 2: Schematic overview of timeframes of some key processes in silica cycling in tidal marshes and in Controlled Reduced Tidal areas

However, when DSi-deliveries are compared with earlier measured reference tidal marshes, the difference is surprisingly small (fig 9). As the absence of *Phragmites* vegetation in the CRT has not provoked large differences in DSi-delivery to the estuary up until now (Fig. 9; data from Struyf et al. 2006 and Van Damme et al., unpublished), we indicate the diatom population and dissolving frustules as the main biogenic silica-stock for estuaries, although the presence of phytoliths of former vegetation might also play a role.

The above-mentioned difference (Fig. 9) is mainly governed by two DSi-retention events, which are caused by DSi-uptake by diatoms, a process that occurs throughout the marsh at stagnant phase and in the tidal pool throughout the whole tidal cycle.

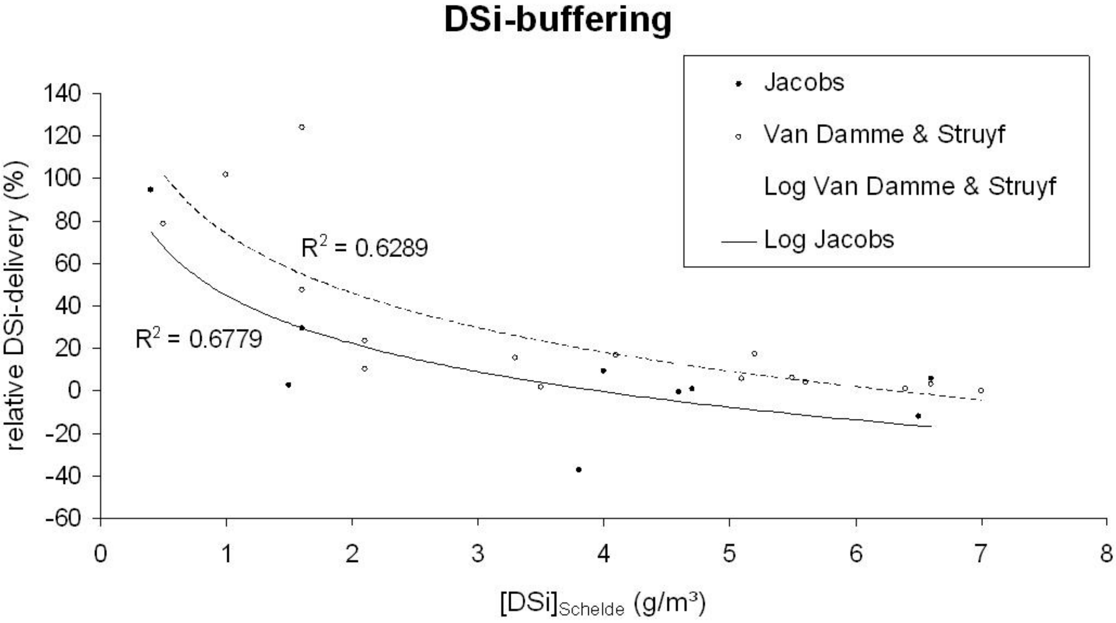


Figure 9: Comparison of DSi-buffering capacity of reference tidal marshes (open points, dashed curve) and CRT (filled points, full curve. Reference data from Struyf et al. 2006 and Van Damme et al. in prep.

9.8.7. Conclusions

The typical tidal features of CRT areas influence their silica cycling, both through increased potential for uptake of DSi and enhanced sedimentation of ASi during the stagnant phase and in tidal pools. As such, they differ from reference tidal marshes (Scheme 2). However, when DSi-deliveries are compared with earlier measured reference tidal marshes, the difference is surprisingly small. Export processes do not seem strongly limited by their decreased time budget. Moreover, at limiting conditions in the estuary, no DSi uptake in the stagnant phase took place, resulting in strong exports of DSi. This indicates that CRT areas are capable of fast build-up of Si-recycling capacity and swift DSi-delivery at limitation events. Despite the current absence of silica-accumulating wetland vegetations, CRT areas have the potential to fulfil their role as silica buffer in the estuary, both via diatom dissolution and dissolution of buried phytoliths (Fig. 16). Additional research concerning the diatomaceous versus vegetation silica stock in intertidal mudflats and marshes is necessary, as well as estimates of process rates to determine the limiting factors of silica buffering by marshes. Our main conclusion is that recently constructed CRTs along estuaries are capable of a silica buffering role comparable to older, reference tidal marshes. Since this restoration technique can be implemented on lowered sites, which are less suitable for managed realignment, the CRT technique has to be represented in the debate about intertidal function and habitat restoration today.

9.8.8. Acknowledgements

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9.9. Silicium: Amorfe en opgeloste silicium stocks

"Amorphous and dissolved silica stocks in restored and reference freshwater tidal marshes"

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9.9.1. Abstract

The global silica cycle is of significant importance to the earth's functioning, and strongly linked with the global carbon cycle and ocean primary production. Cycling in ecosystems plays a crucial role in the global transport of Si towards rivers, estuaries and eventually the ocean, but this biological buffer is only understood relatively poorly. Estuarine ecosystems are generally the last buffer between continental Si export and import of Si into coastal areas and the ocean. Recently, intertidal marshes were hypothesized to be crucial hot-spots in estuarine biological silica cycling. Tidal marshes form a sink for amorphous silica (ASi), and at the same time export dissolved silica

(DSi). It was hypothesized that this DSi export might enhance estuarine ecosystem resilience against silica depletion events associated with eutrophication. Here, we present the results of an extensive sampling campaign quantifying ASi and DSi stocks on several freshwater tidal marshes. Amorphous silica content ranged between 0 and 14 mg/g sediment; dissolved silica concentrations in pore water ranged between 180 and 640 μM. We have found strong indications that rapid shifts might occur, especially in the surface sediment layers, from a silica retention towards a silica delivery state. Our results shed new light on the role of marsh sediments in the estuarine silica cycling, and their potential role in buffering estuarine Si depletion events.

9.9.2. Introduction

Cycling of silica in ecosystems plays a crucial role in the global transport of Si towards rivers, estuaries and eventually the ocean (Conley 2002; Sommer et al. 2006). Although the ultimate source of all Si in the aquatic and terrestrial ecosystems is weathering of mineral silicates, a major part of the weathered silica enters a biological silica cycle by fixation in various organisms (vegetation, diatoms, sponges, testate amoebae) in the form of amorphous Si (ASi) (Conley 2002). This ASi is subsequently stored in ecosystem soils and sediments and it exerts an important control on the eventual fluxes of Si towards rivers and estuaries (Derry et al. 2005; Street-Perrott & Barker 2008). Although this biological buffer is important, it is only understood relatively poorly (Street-Perrott & Barker 2008; Struyf & Conley 2009). Yet quantifying its importance is essential: the efflux of Si from the continents to the ocean is a vital factor in the oceanic Si budget: 3% of yearly ocean diatom ASi production is lost to the deep ocean (Tréguer et al. 1995), and this is largely compensated for by riverine efflux of Si. The carbon burial associated with the diatom burial is one of the major sinks for atmospheric CO_2 (Rabosky & Sorhannus 2009). In addition, ratio of Si, N and P delivery to coastal zones plays an essential role in eutrophication of coastal zones (Cloern, 2001).

The biological silica cycle in estuarine ecosystems is generally the last buffer between continental Si export and import of Si into coastal areas and the ocean. Studies in the 20th century have highlighted an intense biological cycling in the water column and subtidal sediments (e.g. Yamada and d'Elia 1984; Conley and Malone 1992; Garnier et al. 1995; Muylaert et al. 2000): more recently intertidal marshes were hypothesized to be crucial hot-spots in estuarine biological silica cycling (Struyf & Conley 2009). Tidal marshes form a sink for ASi, which mainly occurs in silica rich vegetation (Struyf et al. 2005) and in sediments (Struyf et al. 2005). ASi in sediments originates from import of ASi with sedimentation (Struyf et al. 2007). At the same time, it has been observed that tidal marshes are net exporters of dissolved silica (DSi) to the estuary, as the large ASi pool is slowly recycled (Struyf et al. 2006). This export mainly occurs at times of silica depletion in the adjacent estuary (Struyf et al. 2006, Jacobs et al. 2008), and could enhance estuarine ecosystem resilience against these silica depletion events.

The Schelde estuary, as many other estuarine systems worldwide, suffers from such eutrophication related phenomena. Excessive anthropogenic inputs of N and P regularly cause silica depletion in both the freshwater and marine parts of the estuary (Van Damme et al. 2005), resulting in shifts in plankton community composition towards non-siliceous phytoplankton blooms, associated with anoxia, excessive algal blooms (Phaeocystis sp.) (Lancelot et al. 2005) and hampered silica export towards the ocean (Conley et al 1993).

The 'silica buffer hypothesis' states that restoration of intertidal marshes could enhance resilience of the ecosystem against these silica depletion events (Struyf & Conley 2009).

A modeling study (Struyf et al. 2007) quantified how a single freshwater tidal marsh site changed significantly in ASi build-up potential and rates of ASi recycling through its development from initial marsh to climax vegetation stage. This modeling study coupled sedimentation modeling with a simple empirical model for ASi dissolution, assuming gradually decreasing ASi recycling in marsh sediments with depth. Although the study was able to reconstruct the pattern of ASi burial in the sediments, it was clear that factors such as autochthonous development of diatoms and marsh surface erosion probably resulted in the large absolute deviations observed between modeled and observed sediment ASi profiles. More studies were clearly needed to quantify the storage of ASi in marsh ecosystems, characterized by different vegetations and in different stages of development, and to identify the processes (and their temporal and spatial variability) that control ASi recycling and storage in marsh ecosystems. In addition, the limited amount of marsh ecosystems studied for ASi storage (Norris & Hackney 1999; Hackney et al. 2000; Struyf & Conley 2009) prevent accurate quantification of spatial variability of ASi storage in these systems.

Here, we present the results of an extensive seasonal sampling campaign quantifying ASi and DSi stocks on a restored marsh site, as well as young and old reference freshwater tidal marshes, covering three depth layers and several typical marsh vegetations in the Schelde estuary. Although restored intertidal marshes quickly attain concentrations comparable to reference marshes, we have found strong indications that the ASi concentrations in marsh sediments can be highly variable on relatively short time scales, especially in the surface sediment layers. Our results indicate that rapid shifts might occur in marsh habitat, from a silica retention towards a silica delivery state. Our results shed new light on the role of marsh sediments in the estuarine silica cycling, and their potential role in buffering estuarine Si depletion events.

9.9.3. Materials and Methods

An extensive sampling campaign was carried out in the tidal fresh reaches of the marcrotidal, eutrophic Schelde estuary (Belgium) in 2006. Maps and detailed descriptions of hydrology and ecology of the Schelde can be found in several recent papers (Temmerman et al. 2003; Meire et al. 2005; Van Damme et al. 2005; Soetaert et al. 2006). Sediment cores (depth 25 cm, diameter 1 cm) were sampled at three tidal freshwater marshes (Fig.1): Appels (51°02’53.59”N; 04°04’12.23”E), a young tidal marsh (> 1935; Temmerman et al. 2004), De Plaat (51°05’ 10.64”N; 04°10’28.07”E), a fully developed climax vegetation tidal marsh, and the Lippenbroek (51°05’07.28”N; 04°10’17.52”E), a recently (2006) restored tidal marsh with a controlled reduced tide regime (Maris et al. 2007; Cox et al. 2006; Jacobs et al. 2008).

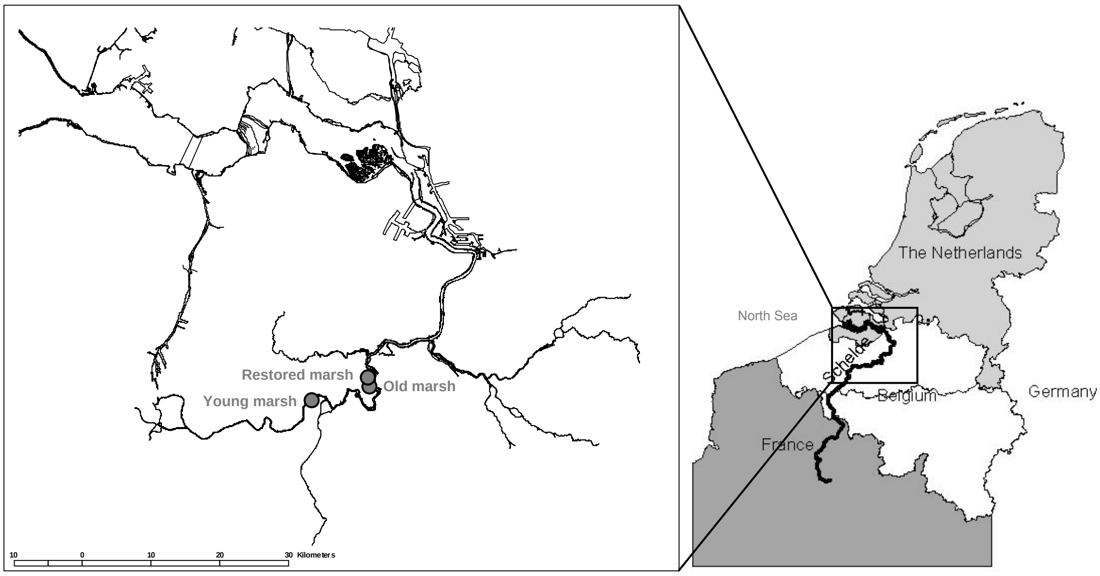

Figure 1: Schematic map of the Schelde estuary, with approximate position of the sampling sites.

The reference sampling locations consisted of homogenous habitats (reed beds, climax willow forests, annual forbs and non-vegetated mudflats) (Fig.2). In the restored site, climax vegetation did not yet develop: five locations along the elevation gradient were sampled (Fig.2).

At each sampling location, six cores were randomly taken every season (April, June, September, and December). The samples were pooled per season, sampling location and depth layer (0-5cm, 5-15cm, 15-25cm). From June on, the first cm was analyzed separately, adding a fourth layer. From June, location 9 was added to the restored site sampling plan to represent the then emerged tidal lake in the sampling plan, not yet developed in April 2006 (Tab.1). Statistical analysis considered the three depth layers – four seasons sampling schedule: the fourth layer was used for graphical discussion.

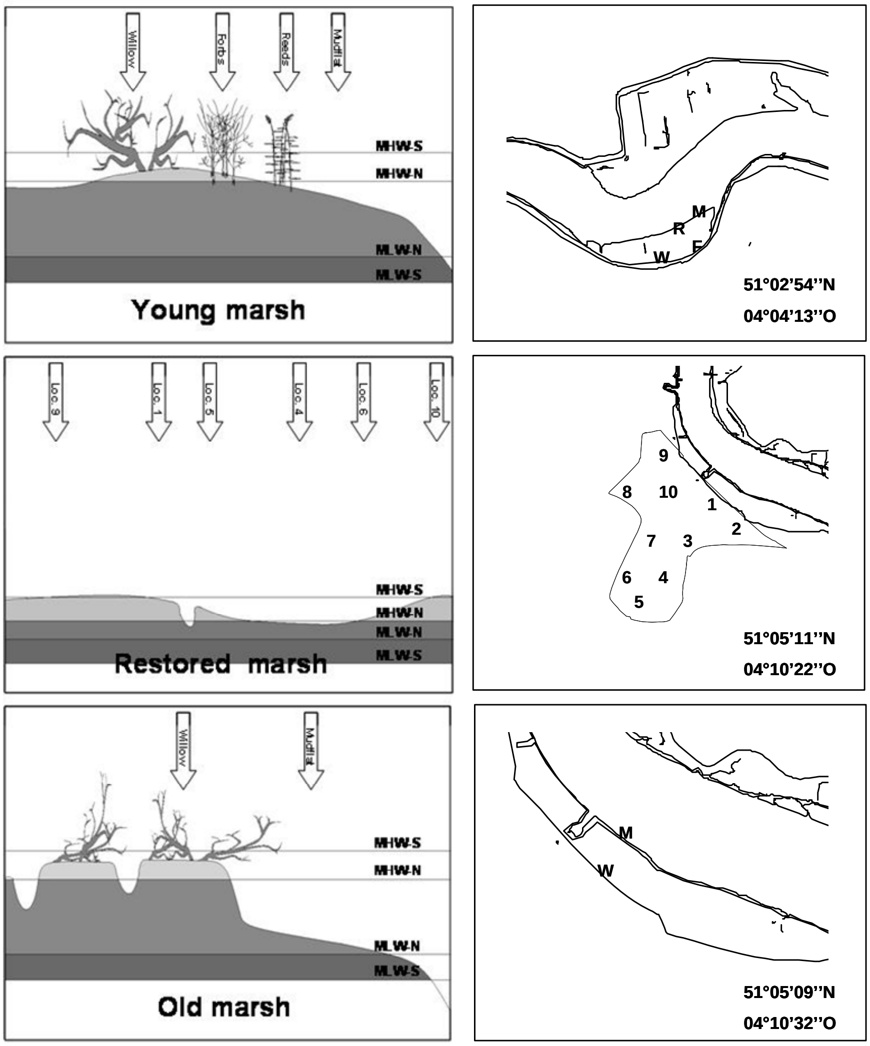

Figure 2: scheme of sampling locations and habitats per site, with elevation relative to tidal levels (MHW=Mean High Water, MLW=Mean Low Water, S=Spring tide, N= Neap tide)

Pore water was sampled every season with 15 cm porous rhizons (manufactured by Eijkelkamp, see also Struyf et al 2005) at all locations, except for location 10 and the young mudflat, which proved too dry for the used sampling procedure. Between two and four replicates were taken for the pore water. At the restored site, 5 extra locations were added to the pore water sampling.

Table 1: Sampling scheme. Numbers indicate the number of layers sampled. On each site, different habitats (M=mudflat; F=forbs; R=reeds; W=willow) (1, 10, 4, 5 and 6 are restored sampling locations) and three depth layers (0-5cm, 5-15cm, 15-25cm) were sampled at every season. From summer on, the surface layer (0-1cm) was collected separately, adding a fourth layer. Pore water was harvested every season at all locations, except for location 10 and Young site-Willow. Between two and four replicate pore water samples were taken. Extra pore water samples were taken throughout the restored site (loc. 2, 3, 7, 8) (not in table).

Site	**Young**		**Old**				**Restored**					
Habitat	**M**	**F**	**R**	**W**	**M**	**W**	**1**	**10**	**4**	**5**	**6**	**9**
spring	3	3	3	3	3	3	3	3	3	3	3	3
summer	4	4	4	4	4	4	4	4	4	4	4	4
autumn	4	4	4	4	4	4	4	4	4	4	4	4
winter	4	4	4	4	4	4	4	4	4	4	4	4
Totals	**15**	**15**	**15**	**15**	**15**	**15**	**15**	**15**	**15**	**15**	**15**	**15**

Sediment samples were dried at 75°C, homogenized with mortar and pestle and sieved over a 300µm sieve to dispose of the fine litter fraction. ASi content was analyzed on a Thermo IRIS ICP (Inductively Coupled Plasmaspectrophotometer) (Iris®) after sequential alkaline extraction for 3, 4 and 5 hours in 1M Na_2CO_3 at 80°C. ASi is extracted during the first hour, while lithogenic silica is released only slowly throughout the extraction. The corrected ASi value is calculated by extrapolating the regression line of the three extraction times to zero in a time-extracted silica plot (DeMaster 1981; Saccone et al. 2007). Pore water DSi was analyzed on a Thermo IRIS ICP.

Data analysis

All data points result from a mixture of pooled random samples and were therefore considered as representative for a certain habitat at a given time.
Data were further organized in a location-layer × season table on which a centered Principal Component Analysis (PCA; Pearson 1901) was conducted, ensuring a strictly spatial ordination of ASi contents. On the resulting factorial planes, differences between the sites (young – old – restored) and habitats (willow – reeds – forbs – mudflat – restored) were tested by means of between-group analyses (Dolédec & Chessel 1987; Dolédec & Chessel 1989). In this procedure, only the pattern of average positions of groups is considered and statistical significance of its inertia (i.e. multidimensional variance) is tested using a permutation test (Manly, 1991) based on the randomization of the lines of the table (999 iterations). All statistical analysis was performed in R (R 2006). ANOVA was used for DSi data exploration (Chambers 1992). The posthoc procedure used is referred to as Tukey Honest Significant Differences (TukeyHSD; Miller 1981; Yandell 1997).

9.9.4. Results

ASi in sediment

As a general pattern, ASi contents were more variable in the surface layer, which also contained maximal observed ASi contents (Fig.3). Individual depth profiles varied strongly between seasons and habitats, with no consequent depth patterns in the deeper layers (Fig. 3).

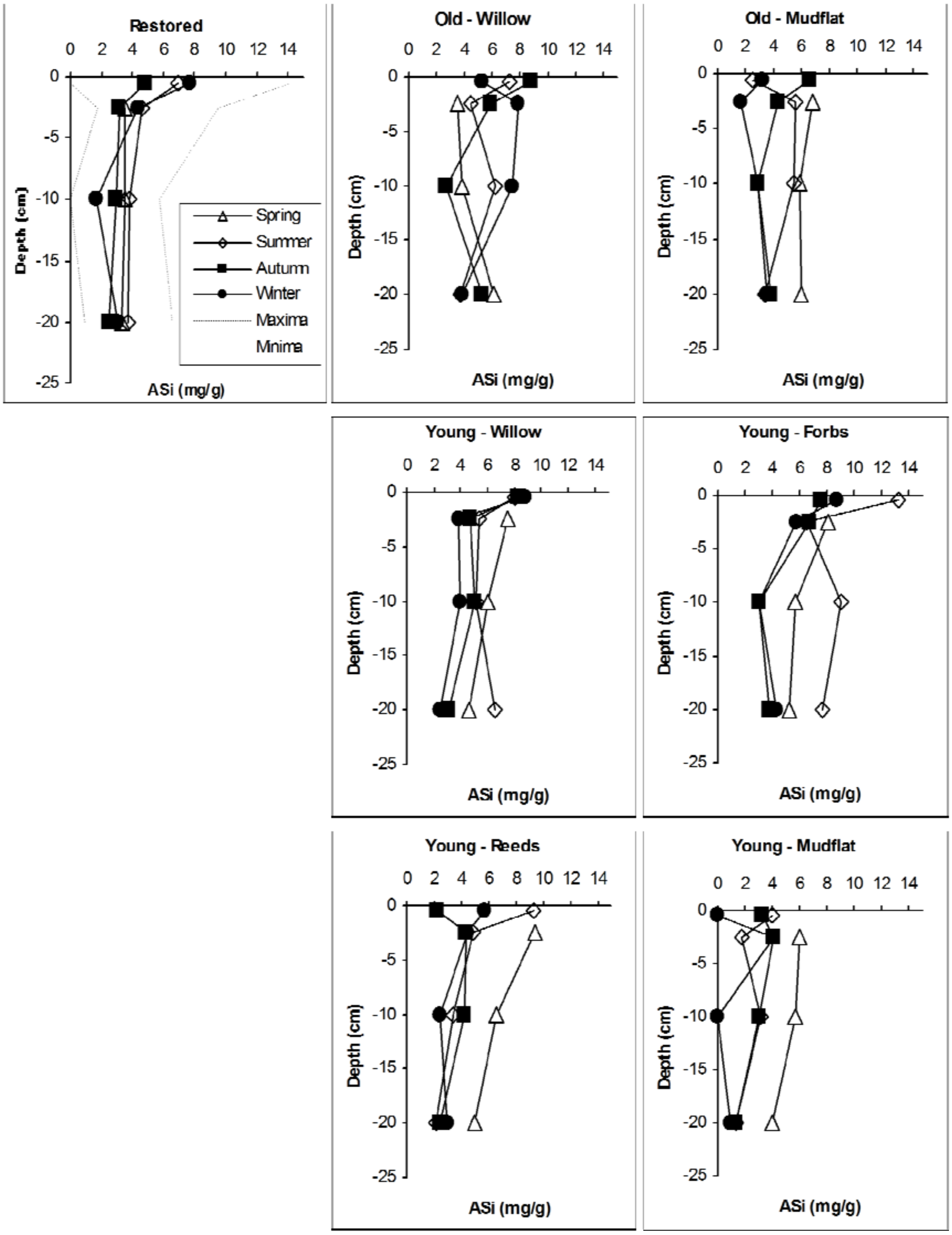

Figure 3: Depth profiles of ASi content in g/mg sediment. For restored locations (A), locations are considered as replicates, and averages are shown. Variance is represented in dashed lines as minimal and maximal concentrations found per layer. For the young and old reference site, different habitats are separately represented (B-G).

PCA revealed three main axes (Fig. 4) structuring our ASi content dataset (F1, 58 %; F2, 23 %; F3, 14%). Both site (explained inertia = 15 %; $p = 0.031$) and habitat (explained inertia = 35 %; $p = 0.001$) were found to have a significant effect on the PCA structure. Fig. 4A indicates a concentration gradient characterized by positive season's covariances on axis 1 (x-axis). A gradual increase in ASi content is apparent from mudflats to reeds, willow and forbs, with highest ASi concentrations observed in the vegetated reference sites (Fig. 4B). The restored locations are ranging over this whole concentration gradient, but are bended towards the lower intertidal habitats. The second axis, opposing winter to spring (Fig 4A) originates almost completely from the lower spring ASi contents in the restored site compared to the young site and the forbs part of the older site (Fig. 4C, Fig. 3). The second axis also opposes frequently flooded (mudflats and reeds) and high habitats (forbs and willow), with restored locations similar to the high habitats. The third axis (Fig. 4E-H, vertical axis) only separates the forbs habitat from all other habitats and the restored locations, due to its high summer ASi contents. The depth profile with higher and more variable concentrations in the first layer is apparent on all three axes (Fig.4 D and H).

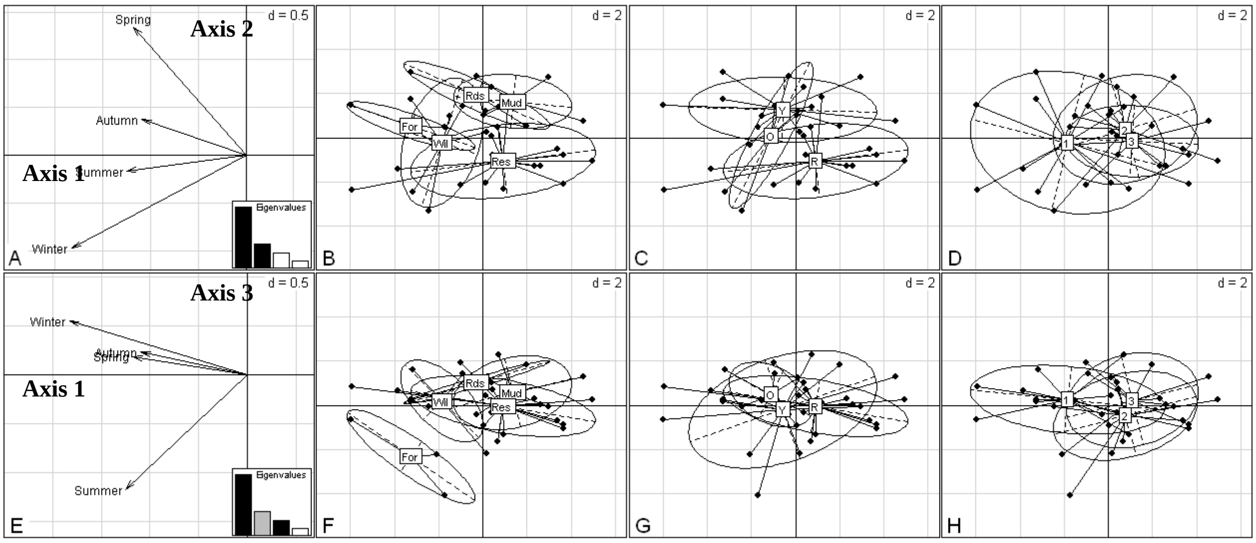

Figure 4: PCA on ASi contents. A, B, C & D: axes 1 and 2. E, F, G & H: axes 1 and 3. A & E: Season's covariances and eigenvalues diagram. B, C, D, F, G, & H: sampling units grouped per site (Y = young; O = old; R = restored), habitat (Mud = mudflat, Rds = reeds, For = forbs, Wil = willow, Res = restored) and layer (1 = 0-5 cm; 2 = 5 – 15 cm; 3 = 15-25 cm).

DSi concentrations

Dissolved Silica concentrations in pore water ranged between 180 and 640 μM (Fig.4). ANOVA revealed significant differences between seasons, sites, and habitats (Tab.2, explained/total variability = 0.747).

Table 2: Summary of three-way analysis of variance of the DSi (pore water) dataset.

Source	Sum Sq	Mean Sq	F value	Pr(>F)
season	298.83	99.61	27.80	0.0000
site	258.24	129.12	36.04	0.0000
habitat	583.12	48.59	13.56	0.0000
Residuals	386.91	3.58		

Seasonal differences separate a summer-autumn group from lower values in spring-winter (Fig.5). Differences between the three sites are caused by slightly lower values in the restored site compared to the old and the young site. Habitat differences do not clearly separate groups, but differences were generally situated in the restored site. Reference habitats did not differ significantly.

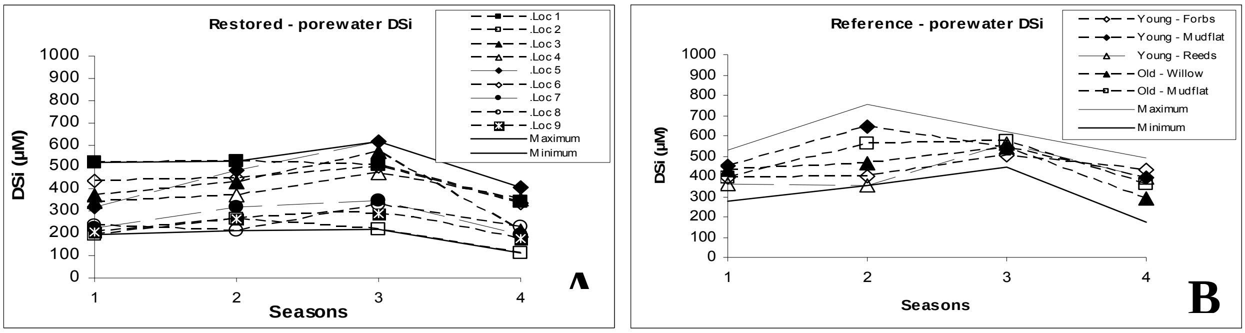

Figure 5: seasonal differences in pore water DSi concentrations for the reference (A) and restored (B) locations. Minima and maxima observed are shown in dashed lines.1,2,3 and 4 represent spring, summer, autumn and winter respectively.

9.9.5. Discussion

Build-up of the ASi-stock

ASi sediment contents and DSi pore water concentrations of several tidal marsh habitats in a freshwater macrotidal estuary (Schelde) were investigated in order to discuss the potential contribution of restoration sites to the estuarine silica cycle. In the reference marshes, big stocks of ASi are imported due to the settling of ASi-rich sediments (Struyf et al. 2007). Also benthic micro-algae, consisting primarily of diatoms, are known to flourish on intertidal flats (e.g. Facca et al. 2002; Ragueneau et al. 2006). Plant phytoliths may also constitute part of the ASi in intertidal marshes, but in the Schelde they have been shown to dissolve quickly (Struyf et al. 2007b), and most ASi in intertidal sediments is comprised of diatoms (>80%) (Conley & Guo, unpubl.).

The ASi contents initially observed in the restored site were low compared to the reference sites. The small difference however was surprising, because the restored area was cut-off from regular tidal regime since its embankment several centuries ago. However, it was deliberately flooded during winter seasons until the seventies, when dike heights along the Schelde were increased. ASi import during winter could explain the minimal difference with the average reference concentrations, while the vertical profile was probably homogenized by ploughing activities.

Since the introduction of the tidal regime in March 2006, import of ASi clearly intensified. More pronounced ASi-depth profiles developed in the restored site, with clear maxima at the surface. These maxima are similar to the ASi contents observed in reference tidal marshes. The build-up of the restored site's ASi contents to the reference levels took place without development of silica-accumulating vegetations, which were almost non-present at the time. Moreover, maximal reference values in the surface layer are found in vegetations which do not consist of dominant gramineous species, and ASi content of the reeds sediments, where *Phragmites australis* thrives in high density, is not higher than on other locations.

We conclude that restoration will gradually provide an ASi stock similar to the reference sites, but local plant-ASi is of minor importance in the build-up of this stock compared to sedimentation of ASi and/or growth of benthic diatoms.

Still, vegetation definitely contributes in an *indirect* way. Vegetation enhances sedimentation by trapping particles on leaves and stems and by lowering current velocity (Leonard et al. 1995; Leonard and Luther 1995; Boorman et al. 1998), which could also provide optimal conditions for benthic micro-algal populations to develop. The third PCA-axis, which originates from the maximum values in the forbs during summer, could reflect the spatial pattern of sediment input or benthic micro-algal growth. Although the proportion of gramineous species in the forbs habitat is small, and it thrives on the higher marsh reaches where sediment import should be relatively low, it consists of extremely dense annual herb vegetation, and sediments were observed to settle even on the vegetation itself. This could explain the observed ASi-maxima in summer.

ASi-recycling

The occasional ASi-minima observed in the surface layer, often lower than ASi contents in deeper layers and occurring at locations were maxima were observed earlier, could indicate an intense recycling of freshly deposited ASi.

The observed increase in DSi in pore water in autumn is late compared to reference tidal marshes, where already in summer DSi concentrations start to increase, as shown before by Struyf et al. (2005). This might indicate an intensification of dissolution processes after the first summer, when colonization by benthic micro-algal mats was observed (Jacobs, personal obs.).

Biological processes influencing diatom dissolution are degradation of organic frustule coatings by bacterial populations (Patrick and Holding 1985; Bidle and Azam 1999), embedding of diatom frustules in zooplankton fecal pellets or formation of aggregates (Ragueneau et al. 2006). Numerous physicochemical factors also affect the dissolution of ASi (reviewed by Nelson et al. 1995; Sarmiento and Gruber 2006). These include temperature (Lawson et al. 1978; Kamatani 1982), incorporation of trace elements such as Al in the chemical structure (Van Bennekom et al. 1989), equilibrium concentrations (Hurd 1973) or specific surface area (Hurd and Birdwhistel 1983).

The increased DSi concentrations coincide with the onset of DSi-export over this period, which was nevertheless 20% lower compared to reference sites (Jacobs et al. 2008). Indeed, DSi concentrations in the restored site remained slightly higher, which might indicate hampered seepage processes compared to the reference sites. This is consistent with the barely developed creeks and with the limited layer (0.-10 cm) of fresh sediment which settled on the clay soil of the restored site.

Although these processes might well explain variations of ASi content in the surface layer, neither sedimentation nor diatom population growth and decline can explain the apparent increases in ASi content in the deeper sediment layers. Struyf et al. (2005) also found a large spatial variation in the ASi content of deeper sediment layers in tidal marshes. The authors only obtained a significant depth gradient when averaging for all five sampling sites. The observed large fluctuations rule out a simple burial-dissolution model, and could imply transformations in the deeper layers. The only apparent source of silica in these deeper layers is the dissolved silica in pore water. Numerous forms of DSi-binding exist, such as formation of allophanes with soluble Al (Harsh et al. 2002), precipitations as almost pure, amorphous phases at mineral surfaces (Drees et al. 1989), chemical adsorption on surfaces of carbonates, aluminium hydroxides and iron hydroxides (Beckwith and Reeve 1963) or the building of polysilicic acid at Fe oxidate surfaces (Dietzel 2002). These processes must be considered when assessing silica transformation in tidal marshes.

Conclusions

ASi stocks in restored tidal marshes rapidly build up, and dissolution processes quickly initiate. The ongoing restoration of 1135ha of tidal marshes along the Schelde, which will then make out 44% of the river's freshwater tidal marsh surface, could impact Si budget in the Schelde and adjacent coastal waters.

Our data can suggest a more complex silica-cycling scheme than assumed by Struyf et al. (2006, 2007, 2007b). The silica stock in freshwater tidal marsh sediments is determined by input of diatoms and diatom debris through sedimentation and benthic growth, by the transformation processes of this ASi to dissolved silica, and by the amount of reactive sediment exchanging dissolved silica with the flooding water. Based on our observations, two conceptual models are proposed. The first model depicts a standard retention state (Fig.6). This situation occurs at conditions which:

- promote benthic algal growth
- promote DSi-uptake by diatoms or binding processes
- promote sedimentation of diatom debris
- hamper benthic diatom dieback
- hamper frustule dissolution

In this case, maximal ASi values are found in the upper centimeters of the sediment profile (‘observed ASi-profile’ in Fig.6). Benthic diatom population turnover and sedimentation provide ASi (‘sedimentation’ and ‘growth’ in Fig.6), which is partially dissolved (‘dieback’ and ‘dissolution’ in Fig.6) and partially buried by ongoing sedimentation. When this ASi is buried deeper, dissolution goes on, which explains decreasing ASi concentrations with depth. Dissolved silica in the pore water is drained (‘drainage’ in Fig.6), and could be translocated and bound as ASi (‘groundwater flux’ and ‘binding’ in Fig.6).

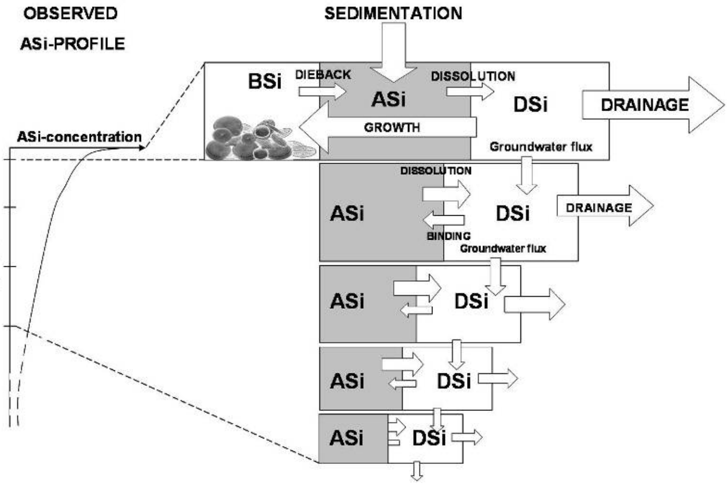

Figure 6: Conceptual model of silica cycling in freshwater tidal marsh sediments during silica retention (DSI: dissolved silica; BSi: biogenic silica; ASi: amorphous silica).

The opposite conditions provoke a ‘silica delivery state’, with a profile with minimal values observed in the upper centimeters (‘observed ASi-profile’ in Fig.7). Temperature and drought stress, estuarine silica depletion or predation could hamper diatom growth or cause dieback of the benthic diatom population. Silica depletion could cause a decline in diatom debris sedimentation. High temperatures, bacterial activity and enhanced concentration gradient could fasten dissolution of diatom frustules even in deeper layers. This provokes a shift in the ASi-DSi equilibrium, providing a surplus of DSi.

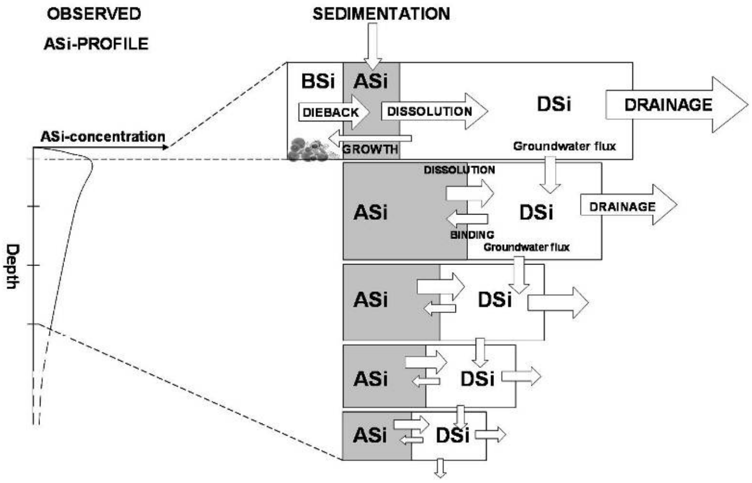

Figure 7: Conceptual model of silica cycling in freshwater tidal marsh sediments during silica delivery (DSI: dissolved silica; BSi: biogenic silica; ASi: amorphous silica).

Our concept opposes two extreme states, which are theoretical extremes of a continuum. It suggests that the sediment ASi-pool could potentially deliver DSi, mainly from the top centimeters. The 'delivery state' (Fig. 7) is probably more exceptional, since concentrations in the deeper (and older) layers remain high, which suggests that the 'retention state' (Fig. 6) is occurring most of the time. We hypothesize that conditions which promote the delivery state occur mainly during estuarine depletion events in summer. These hypotheses oppose the prevailing assumption that the entire ASi sediment stock has a slow reactivity. Upper layers, which consist of benthic algal populations and reactive frustules, might be capable to react quickly to changing conditions. A low-reactive sediment ASi-pool, indifferent to changes in environmental conditions, would result in a same delivered amount of silica at tides with the same volume and different conditions. During single-tide mass-balances however, different DSi-deliveries were observed at different conditions, and these earlier results indeed suggest a positive correlation with temperature and a negative one with DSi-concentrations in the flooding water (Jacobs et al. 2008).

Core incubation experiments and mathematical modeling are needed to disentangle the complex cycling of silica in intertidal sediments: simultaneous measurements of ASi and DSi stocks at higher resolutions and frequencies, flux estimates, processes which control diatom population dynamics, frustule dissolution and DSi binding reactions have to be incorporated. Ultimately, coupling with system-scale hydrological models is essential to quantify the impact on estuarine silica budgets and verify the silica buffer hypothesis

9.9.6. Acknowledgements

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9.10. Summary of the Lippenbroek pilot project

Restoration of a freshwater tidal marsh on a lowered rural site: A new method applied in the Schelde estuary (Belgium)

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keywords: estuarine restoration, managed realignment, regulated tidal exchange, freshwater tidal

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9.10.1. Abstract

Throughout the world, tidal marshes are restored to obtain natural protection against recurring storm surges and to preserve the goods and services these habitats provide. Managed realignment is a restoration technique increasingly used; yet unsuitable site elevation has been a major constraint to habitat development. Sites under controlled reduced tide (CRT) could offer solutions for several problems associated with reconstruction of tidal marshes. This paper presents the first results on tidal variation, nutrient processing, species colonization and habitat development in a pilot CRT. This qualitative overview aims to introduce this technique to scientists and restorationists worldwide, since after only three years of monitoring the pilot site, we conclude that the CRT-technique provides strong potential for durable, adaptive restoration of tidal marshes on lowered sites. Wider implementation of the CRT technique could increase the total surface of tidal freshwater marshes and ease implementation of restoration projects in coastal defence.

9.10.2. Introduction

The worldwide extent and abundance of intertidal habitat has greatly decreased, primarily due to human alteration of estuarine habitats. Impacts include construction of embankments or weirs, harbour expansion, deviation of freshwater inflow and conversion of tidal marsh for agricultural uses and urban and industrial development.

Tidal freshwater marshes are ecosystems with specific environmental conditions which provide essential habitat for characteristic vegetation and waterfowl (Barne *et al.* 1995, Struyf *et al.* 2008). Their role in estuarine biogeochemistry has been studied and reviewed intensively in recent decades (e.g. Dame *et al.* 1986, Howarth *et al.* 1996), emphasising their role in C, N and P transformations (e.g. Spurrier & Kjerfve, 1988; Childers & Day, 1988; Gribsholt *et al.* 2005) and in the buffering of estuarine silica concentrations (Struyf *et al.* 2006, 2007, Struyf & Conley 2008, Jacobs *et al.* 2008). They also dissipate and control tidal currents and waves (Möller & Spencer 2002), protecting landward sea defences from scour and erosion.

Nowadays, managed realignment is a technique which is increasingly used to restore intertidal habitat. This technique, which consists of the removal or breaching of dikes to restore tidal influence, is extensively reviewed in French (2006). Elevation is a key factor in planning of managed realignment and for suitable site selection (e.g. Brooke 1991; Cundy, 2002) as it relates

directly to frequency, height and duration of tidal inundation, which are the main factors in sedimentation patterns and vegetation development. Agricultural sites adjacent to estuaries have often lowered in elevation as a result of compaction. This typically leaves them below the levels of contemporary marshes within the same system, which have often increased in height with sedimentation (Temmerman *et al.* 2003). The difference in elevation rules out many sites for potential realignment, since this would result in entirely flooding of the site every tide and would thus result in development of completely non-vegetated intertidal mudflats, of which it is not certain if and how fast they could evolve to a vegetated marsh system.
This problem has been countered in different ways. In the UK, suitable sites which approximate the desired level are selected and natural processes are allowed to proceed (French 2006). In the US, it is common to increase site elevation prior to dike breaching (Brooke 1992). However, artificial site elevation has been a common reason for project failure (Quammen 1986, Perry *et al.* 2001). A third option is to install a restricted tidal exchange (e.g. Lamberth & Haycock 2002, Hirons *et al.* 2002). This technique, typically using one culvert for both in- and outflow, is often implemented in restoration projects in England and at the German North Sea Coast (Rupp-Armstrong & Nicholls, 2007). It has the advantage of lowering the tidal wave in the site to an acceptable level, but the technique cuts out spring-neap tide variation needed for optimal habitat diversification.

In 2006, a novel technique was presented (Cox *et al.* 2006, Maris *et al*, 2007) to restore tidal marshes on lowered sites. It provides a site with a controlled reduced tide (CRT). This technique allows implementation of a restricted tidal regime with neap and spring tides, by the use of high inlet culverts and low outlet valves. Adaptive management by fine-tuning of the culverts towards specific management goals is straightforward, and no complex mechanic or electronic systems are used. Cox et al. (2006) and Maris et al. (2007) hypothesised that the CRT-technique could successfully restore freshwater tidal marsh habitat on a lowered rural site.

The Schelde estuary (Fig. 1) is a highly eutrophic, macrotidal estuary, and has been extensively described (Wollast 1988, Meire *et al.* 2005, Soetaert *et al.* 2006). Its 21.863 km^2 drainage basin, situated in France, Belgium and the Netherlands, is inhabited by over 10 million individuals. In Europe, national and international legislations like the Water Frame Directive (WFD), Habitat and Bird Directive (HD/BD) dictate a 'no net loss' policy and the development of a 'sound ecosystem'. Local implementation of these legislations on the Schelde basin resulted in the current development of several thousands of ha of tidal restoration projects, of which ca. 1000ha with the CRT technique (Fig.2). Eventually (by 2030) new CRT habitat will comprise 36% of tidal freshwater marshes along the Schelde, or about 16% of tidal freshwater marsh described on a European scale (Struyf *et al.* 2008). However, no one has addressed the basal hypothesis as whether this technique could successfully restore freshwater tidal marsh habitat on a lowered rural site.

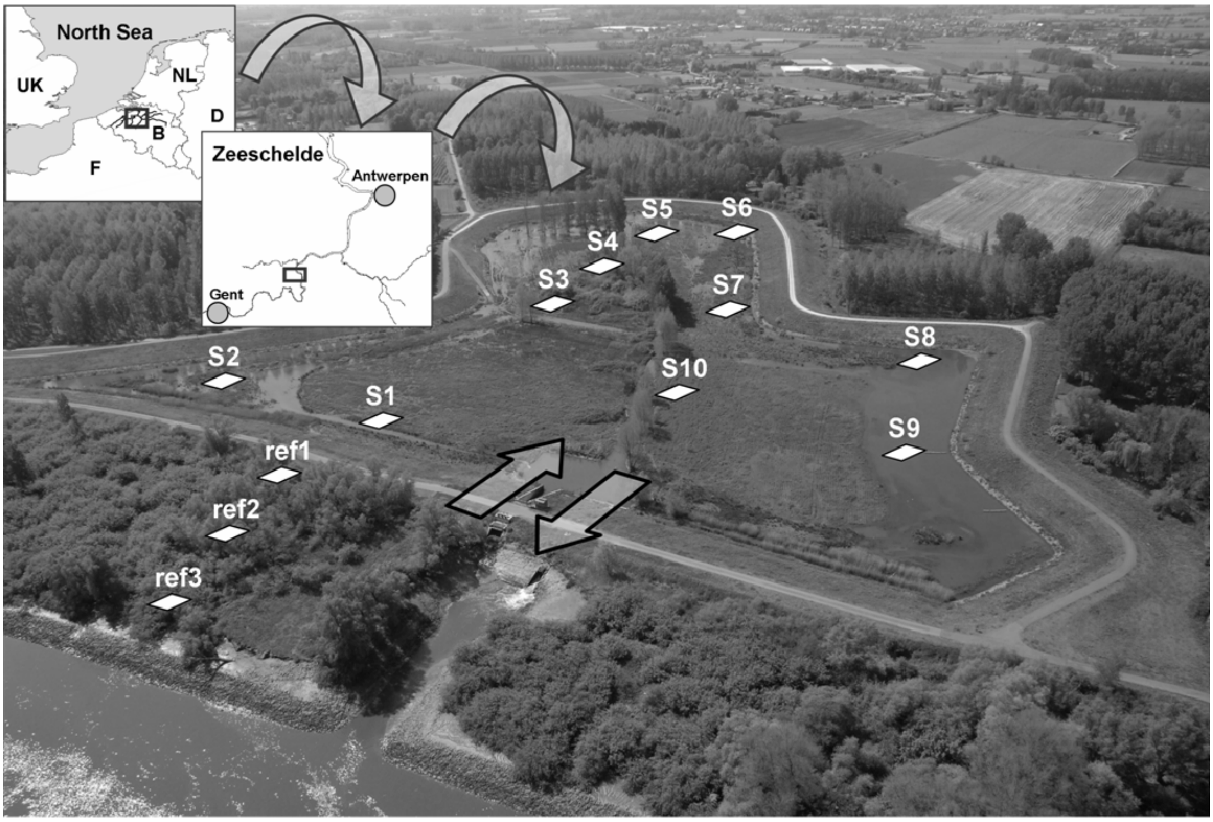

Figure 1: Situation and areal view of the CRT pilot site and Schelde estuary. Monitoring sites (S1 to S10) and reference sites on the adjacent estuarine marsh (Ref1 to Ref3) are marked with white squares. Arrows show the position of in- and outlet sluices.

Figure 2: Aerial view of current CRT construction along the Schelde. Photograph shows outlet (left) and inlet (right) construction which will allow implementation of CRT on several hundreds of hectares of lowered agricultural land. Riverside dike is situated above in the photograph.

Here, the first results on species colonization, habitat development, and nutrient processing in a pilot CRT are presented. The results were obtained from unpublished monitoring data, published papers and papers in preparation. European legislation demands certain functions and structures to be developed (table 1) in restored freshwater tidal marshes. Although splitting up the ecosystem in structures and functions is somewhat artificial and both are often linked (e.g. sedimentation process – topographic structure), we will use this list to structure the results.
This qualitative overview emphasises the capacity of CRT to rapidly develop a specific intertidal habitat with structures and functions characteristic for tidal freshwater marshes.

Table 1: processes (italic) and structures (bold) to be developed in restored freshwater tidal marshes in order to comply with European legislation.

processes and **structures**
Tidal cycling
Nutrient processing (Si, N, P, O)
Sedimentation
Tidal marsh topography
Tidal marsh vegetation
Tidal marsh fauna

9.10.3. Materials and Methods

The pilot site

The study area is a pilot CRT, the "Lippenbroek" (surface approximately 80.000 m^2), situated in the Schelde estuary, Belgium (51°05'10"N; 4°10'20"E, Fig.1). Mean tidal amplitude at springtide in the Schelde at this point is about 6 m. The site is situated in the freshwater tidal zone of the estuary, with an averaged salinity ranging from 0.5 (winter) to 0.8 PSU (summer). The difference in elevation between the contemporary tidal marshes and the pilot site is 2.5 to 3 m. Managed realignment would thus result in total flooding of the whole area every tidal cycle. The area was mostly used as intensive cropland, and abandoned in 2003. During the two-year construction phase (2003-2005), crops were replaced by a dense pioneer vegetation (mainly *Epilobium hirsutum* (Great Willowherb) and *Urtica dioica* (Stinging Nettle)), and part of the site was devegetated because of the building works. Tidal inundation was restored by means of CRT in March 2006.

CRT Functioning

The idea of restoring intertidal habitat with a controlled reduced tide is very simple: instead of breaching a dike to flood the polder area or installing a single restricting inlet culvert, an inlet culvert which only lets in the top of the tidal wave is created (Fig 3). Consequently, water flows gravitationally out to a lower exit culvert when water levels in the river have again decreased (Fig. 4). The site elevation and river tidal regime are thus detached. Fine-tuning of inlet culvert level permits installation of a tidal regime with a pronounced spring-neap variation. This aspect is of tremendous importance to restore the full range of intertidal habitats and has until now been lacking completely from other tidally restricted systems.

Figure 3 (left): Inlet culvert construction (as seen from within the pilot restoration site), consisting of three inlet culverts, each 1 m wide, with adjustable thresholds at different heights. Figure 3 (right): Close-up of the inlet culverts at spring tide flooding (as seen from within the pilot restoration site).

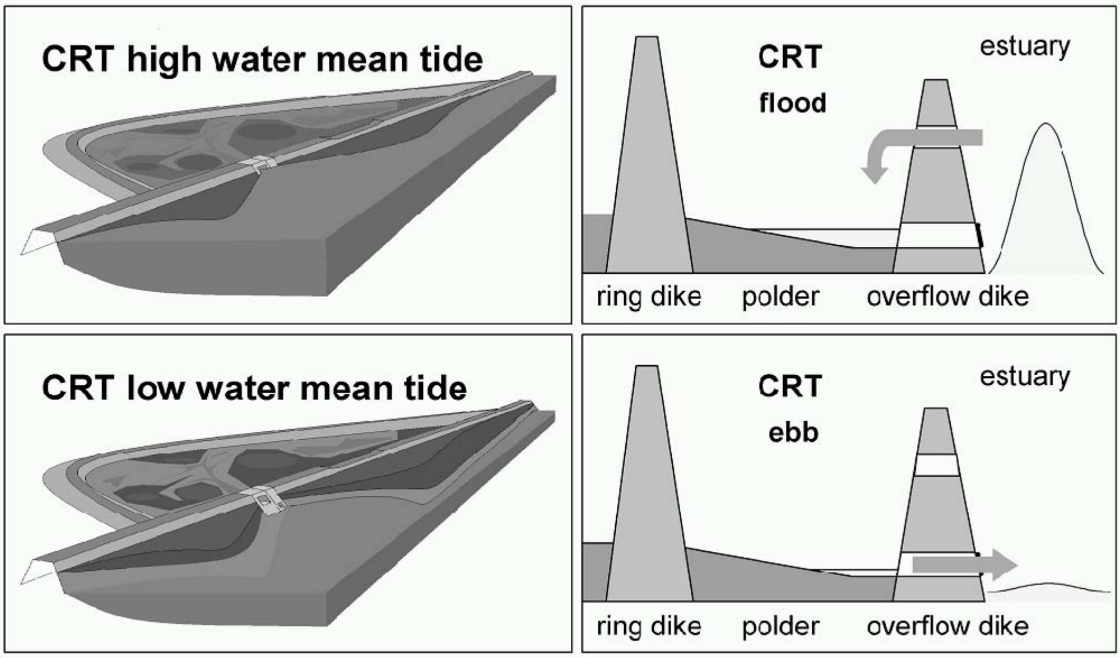

Figure 4: CRT Functioning. Inflow phase initiates when rising river level exceeds the inlet threshold, and continues throughout the rising and falling of the tide until the level drops back to this threshold. Outstream begins water pressure opens the outlet culvert, and continues until the river level rises again to close the valve.

Tuning the tide

When a site has a pronounced topography, or an extended ditch system, the water volumes needed to create a tidal gradient are much bigger. This would theoretically lead to more culverts at mean high water neap level, but the problem can also be solved by adding some culverts with lower thresholds (Fig. 5). Such a configuration permits bigger water volumes to enter every tide without increasing the number of culverts, which saves up on the building costs. Off course, in the latter case, precise fine tuning will be required according to the desired flooded surfaces at neap, average and spring high waters (Fig. 5). Therefore, calculation of surface/volume relationships of the site and discharge estimates of the culverts is needed. After a few test runs and fine-tuning of the inlet sluice sill level, a reduced tidal regime was introduced from March 2006 on. Sluice parameters are summarised in table 2.

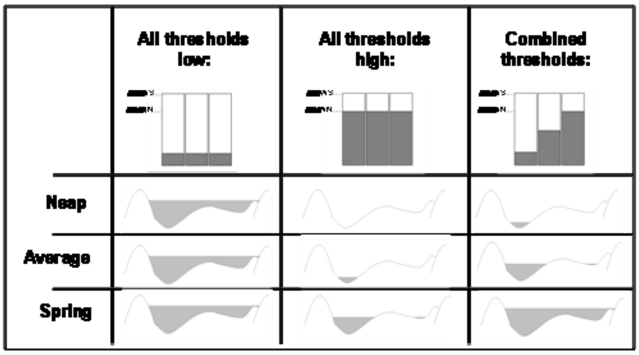

Figure 5: Threshold configuration has to be tuned for optimal tidal flooding gradient.

Table 2: Dimensions of the culverts of in and outlet sluices

	Inlet Sluices			**Outlet sluice**
	culvert 1	culvert 2	culvert 3	
length (m)	12,75	12,75	12,75	40,2
width (m)	0,75	0,75	0,75	1,5
sill level (m TAW)	4,7	5	5,3	1,5
height (m)	1,2	0,9	0,6	1,5

The Monitoring scheme

On ten randomly chosen monitoring sites, 9 plots (2 x 2m) for different sampling purposes are designated. This approach guarantees maximal comparability of different parameters, and minimises mutual interference of destructive and non-destructive measurement techniques. Vegetation, benthos, sedimentation and soil properties (e.g. compaction, granulometry, nutrients, and pollutants) are recorded in detail. At three reference sites on the adjacent marsh, the same parameters are being recorded.

Tidal characteristics are measured at each of the 10 sites and at the sluices. Diver dataloggers (Van Essen Instruments) monitor the water level by measuring the pressure of the water column with a pressure sensor. Correction for changes in air pressure is performed. From these water level records tidal characteristics are derived: absolute high (HW) and low water (LW) levels, inundation frequency (IF), inundation height and inundation time (for definitions see Cox et al. (2006)).

From May 2006 onward, water levels and flow velocities at the inlet and outlet sluices of the pilot CRT are monitored continuously on a 2 to 5-minute time interval. At distinguished moments (5 to 10 measurement~~s~~ campaigns for each in- and outlet sluice), discharge measurements were executed using portable measurement devices (e.g. Flo-Mate$^{\text{TM}}$ and Qliner). Comparing continuous time series with measured discharges made it possible to establish good relationships (linear regression with $R^2>0.8$) between the continuously measured data and resp. the actual water level and section-averaged flow velocity. Based on head-discharge measurements at the inlet sluices, it was found that the inflowing discharges could be obtained using the standard discharge equation for a sharp-crested weir, which provides field measurement accuracy in the order of 5% (e.g. Bos, 1989). Validated flow discharges through the outlet structure are obtained using validated water levels and flow velocities. At this time, the accuracy of the outflow discharges is estimated to be 5 to 10%. Discharges were used to calculate discharge weighted averaged nutrient concentrations. With these weighted averages, conservative mass balances were made.

Sedimentation and erosion is measured at the 10 sites using "Surface Elevation Tables" (SETs) with an accuracy of approximately 1 mm (Cahoon et al., 2002). In this way elevation change, which is the common effect of sedimentation, erosion and autocompaction, can be monitored. Creek morphology is monitored using a total station (Sokkia, with accuracy of 1 to 3 mm).

Water quality analysis and exchange studies (nutrients, sediments, metals, biota (fish, algae)) were performed near the sluices during several separate 13 hour campaigns: water quality was monitored every half hour during the flood phase, every hour during the ebb phase. Subsurface samples were taken near the sluice by means of bucket hauls, stored at 4°C and analysed within 24 hours using a SKALAR SA 5100 colorimeter. Dissolved silica was analysed spectrophotometrically on a Thermo IRIS ICP (Inductively Coupled Plasmaspectrophotometer).

Vegetation mapping, creek morphology and the monitoring of fish and birds is performed regularly, covering the entire surface of the pilot site. Additionally, spatial sedimentation patterns are monitored during short campaigns. Table 3 presents an overview of the spatiotemporal monitoring setup.

Table 3: spatio-temporal monitoring setup

processes and **structures**	**Monitoring**	**Spatial scale**	**Frequency**
Tidal cycling	Divers	10 sites	continuous
Nutrient processing (Si, N, P,O)	13h campaigns	Sluices	4/year
Sedimentation	SET	10 sites	6/year
Tidal marsh topography	mapping	Entire CRT	2/year
Tidal marsh vegetation	mapping	Entire CRT	2/year
Tidal marsh fauna	inventories	Entire CRT	weekly

9.10.4. Results: Processes

Spring-neap tidal cycling

The tidal amplitude is reduced, on average, from 5.5 m in the Schelde to 1.3 m in the pilot CRT and the HW level is about 3 m lower in the polder than in the Schelde (fig. 6). The spring- neap tidal cycle is however maintained, the difference between spring and neap tide highwater being approximately 1.0 m in the Schelde and in the pilot CRT. In contrast to estuarine marshes, in the CRT the low water levels are lowest during neap tides.

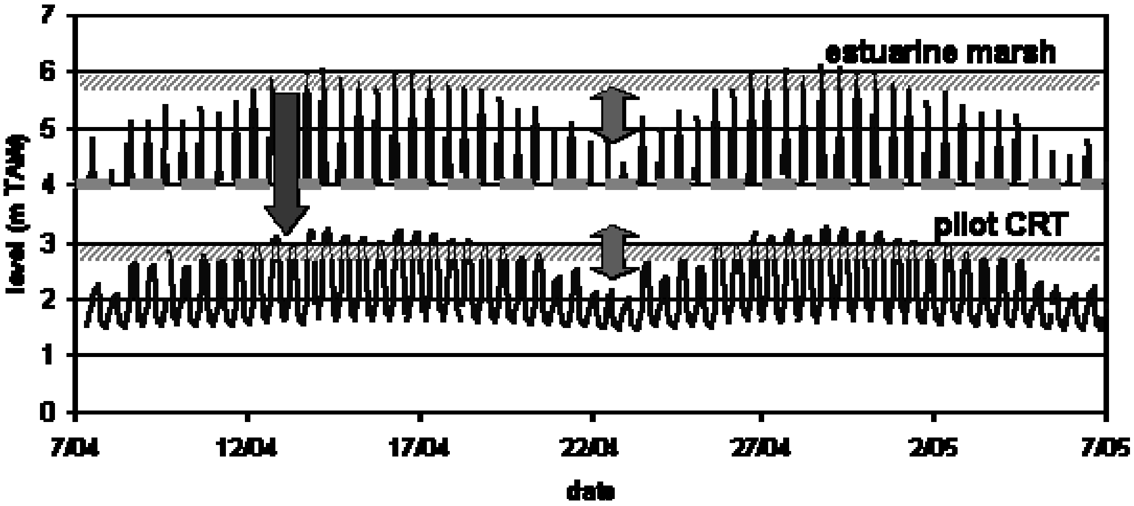

Figure 6: tidal waves in Schelde and The pilot CRT. Hatched areas indicate the elevation of the adjacent marsh and the mean surface elevation of the polder. The big arrow indicates the reduction of the high water levels by about 3 m. The smaller arrows show that the difference between HW spring tide and HW neap tide is preserved in the CRT. Note that for the tidal curve of the Schelde, only the upper parts (above 4 m TAW) are shown.

At the scale of one single tide, some important differences with the estuarine marshes occur: after the water has entered through high inlet culverts, a stagnant high water phase develops (1-2 hours), before it evacuates through the low outlet culverts, as predicted by Cox et al. (2006). The length of this phase is dependent on the elevation difference of the site with the mean high water level. The seepage phase, when water gravitationally drains from the marsh during low tides, the end of outstream can be blocked by the next flood. Both are consequences of the gravitational outlet sluice, which only opens when the water level in the river is lower then the water level within the site. This stagnant phase, together with the extreme maximal dry droughts and flooding durations, could well have consequences on sediment biogeochemistry, plant and benthos colonization and nutrient exchanges, while hampered seepage could lower the exchange capacity with the estuary.

Also, on a larger temporal scale, some important deviations from the river tides appear. During neap tides, the threshold of the inlet construction might not be reached, and no flooding at all occurs. This provokes drought events, which are longer (seasonal maxima 4 to 27 days) then in adjacent river marshes (seasonal maxima 12 to 14 days). Also, prolonged flooding events occur at storm tides, provoking extreme flooding durations much higher and more variable (seasonal maxima 12 to 35 hours) then in the river marshes (seasonal maxima 3 to 7 hours).

However, when analysing the characteristics of the inundations more in detail by calculating distribution of inundation frequencies (IF) for both the CRT and the adjacent estuarine marsh, very similar inundation frequencies (IF) are found. For comparing both sites with different elevation, elevation is expressed relative to the level corresponding with an IF of 50% (fig. 7). The fact that sites with high IF are rather scarce on the adjacent marsh while in the CRT surface with low IF is limited (fig. 8) is due to the site-specific topography of the pilot CRT and the typical topography of the estuarine marshes, which have a deficit in lower marsh surface because of marsh edge erosion.

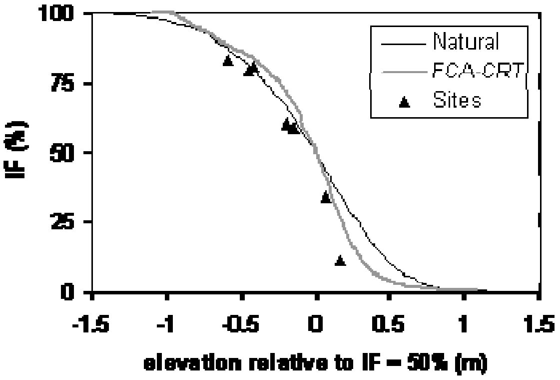

Figure 7: Inundation frequencies in the adjacent marsh and in the pilot CRT in function of elevation. Elevation is expressed relative to the level corresponding with an IF of 50%. Based on tidal data for the period March 2006 – March 2008, this IF corresponds to 5.63 m TAW on the reference marsh and 3.02 m TAW in the CRT. Black triangles represent the different research sites in the CRT.

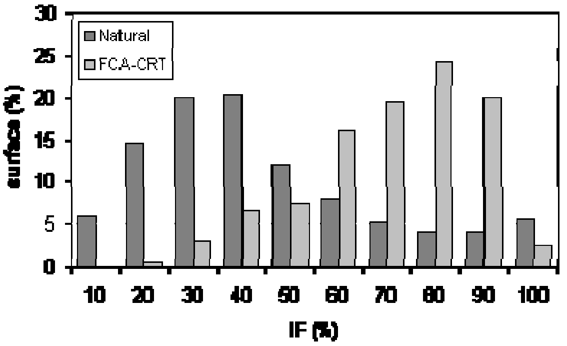

Figure 8: surface (% of total) of adjacent estuarine marsh and the pilot CRT for different inundation frequencies (10: IF between 0% and 10%, 20: IF between 11% and 20%, …).

Nutrient Processing

Silica buffering is one of the vital processes exerted by tidal marshes. By accumulating amorphous silica from diatom and plant debris, and releasing it as dissolved silica, which is available for uptake by diatoms. This release increases during events of silica depletion in the estuary, during which diatom communities decline and eutrophication phenomena appear. Jacobs et al. (in prep.) compared the silica processing of the pilot CRT with measurements in reference freshwater tidal marshes. Similar accumulations of amorphous silica, as well as a swift delivery of dissolved silica during depletion events were reported. These deliveries are about 20% lower then percentages observed in earlier research on river marshes. Their main conclusion is that recently constructed CRTs along estuaries are capable of a silica buffering role comparable to older, reference tidal marshes.

Retention of nitrogen and phosphorus is regarded as a potentially important process since it could partially compensate for excessive anthropogenic inputs of these nutrients. As detailed research on exchange mechanisms is ongoing, we can only conclude that monitoring results so far indicate no significant differences between the recorded CRT deliveries and earlier observations in river marshes made by Van Damme et al. (2009) (Fig.9)).

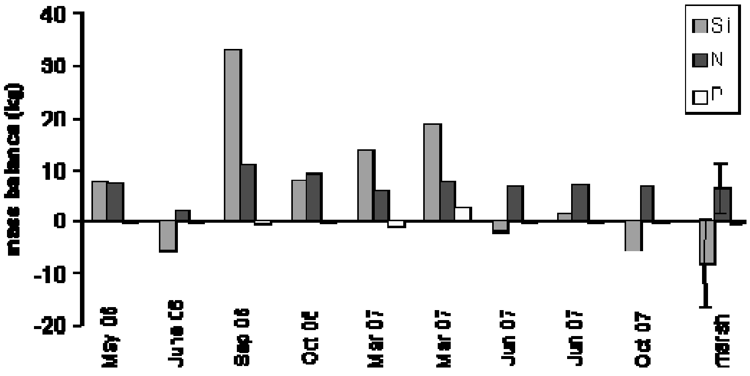

Figure 9: Mass balances for dissolved silica (Si), Total Dissolved Nitrogen (N) and Phoshate-P (P) for every tidal campaign in the CRT and for the Schelde estuarine marsh (marsh) (Van Damme et al. 2009). Positive values indicate retention, negative delivery.

Regardless weather or light conditions, the incoming, the often oxygen poor Schelde water is aerated by the instream wave, so that the pilot CRT never faced problems with anoxia. On the contrary, on sunny days, super saturation was recorded.

Sedimentation

At all sites (except for sites in a tidal pool), net accretion was observed, ranging from a few mm on the highest up to more than 10 cm/year initially on the lowest sites. Sedimentation clearly leads to a fast flattening of the area (fig 10). The highest accretion rates are however decreasing.

Elevation changes are related to inundation frequencies (fig 11). Sites with comparable inundation frequencies show similar accretion rates in the CRT as on the river marsh. The two locations in the CRT with an inundation frequency of 100% (Fig. 11) are situated in a tidal pool, to which few or no sediments are supplied during tidal inundation. Therefore elevation changes are limited on these locations.

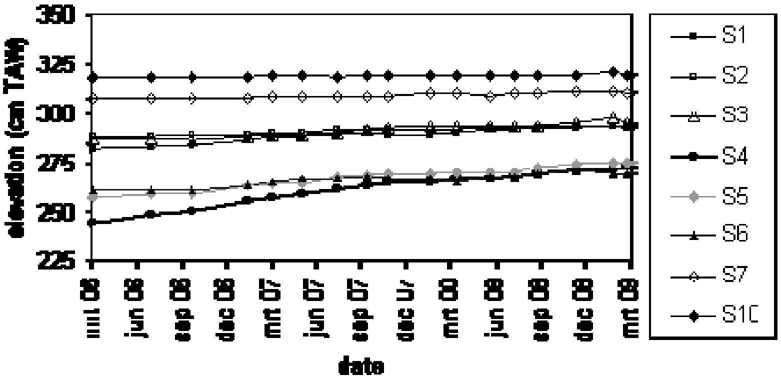

Figure 10: Elevation changes (cm) on different sites in The pilot CRT. Sites 8 and 9 (tidal pool) are not included.

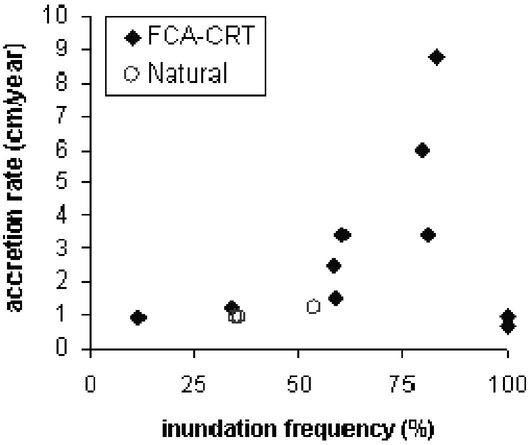

Figure 11: average accretion rate in function of IF at the three sites on the adjacent marsh and at all sites in the pilot CRT.

9.10.5. Results: Structures

Tidal MarshTopography

At the starting phase, topography was determined mainly by agricultural use, remaining drainage ditches and the building works. Sedimentation occurs throughout the site but elevation differences (and thus flooding differences) still persist. Already during the first months, small creeks developed more or less perpendicular to the main drainage ditches (fig. 12). 1029 meters of new creeks have formed three years after installing the tidal regime. In the pre-existing central channel, erosion was observed near the sluices and sedimentation near the head (fig 13). Also, meanders are developing, converting the former drainage ditch into a natural creek.

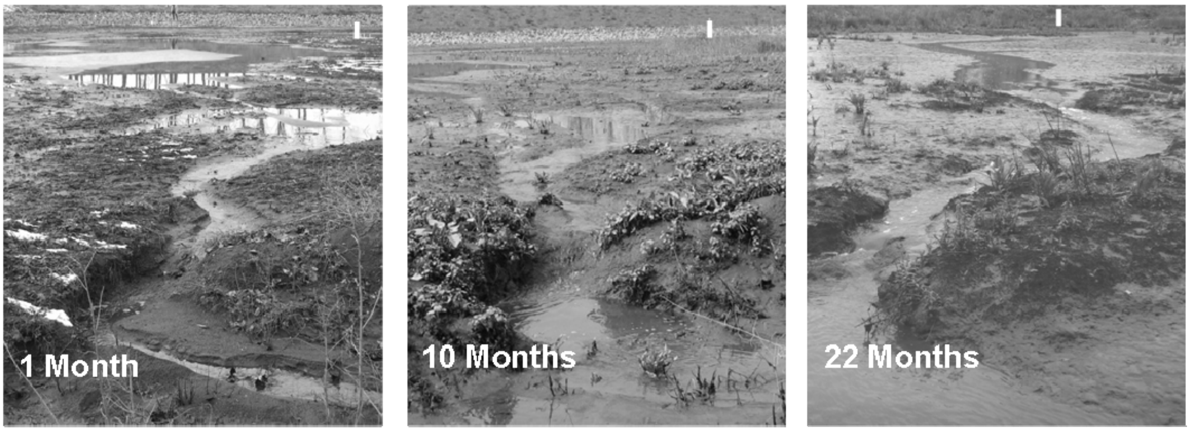

Figure 12: Development of a creek in the CRT. White square indicates position of the same stick in every photo. Note the deepening of the creek, decrease of stagnant water surface in the background, and emergence of mudflats and meandering structures.

New creeks incised the compacted agricultural soil, and many of them are still deepening (Fig 14). In the zones with high sedimentation rates, small creeks are more chaotic, sometimes disappearing and often changing shape. However, once a certain depth is established, they seem to consolidate and start eroding the accreted layers and underlying soils. Creek density and drainage capacity are not yet in equilibrium with the sites' surface area and exchanged water volumes, as indicated by the ongoing structural evolution. Further development will most likely result in a flat marsh platform (merlons), incised by creeks (krenels, Fig 14).

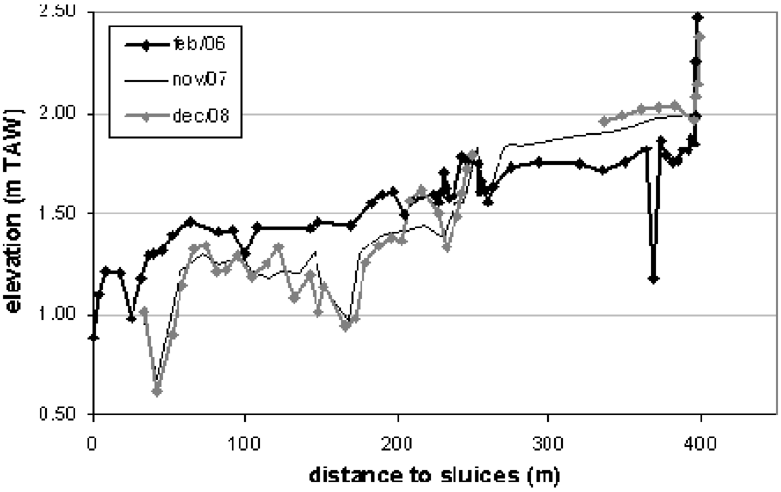

Figure 13: vertical evolution of the main creek in the pilot CRT

Figure 14: Topographic structures in the CRT. left: new developed creek. right: typical merlon-krenel topography.

Tidal Marsh Vegetation

Vegetation development is described in detail in Jacobs et al. (in prep). Already after two years of controlled reduced tide, an intertidal plant community has established in the pilot CRT, with fast eradication of terrestrial species and colonization of typical European freshwater tidal marsh species already during the first few months. Moreover, communities which have mostly disappeared along the degraded Schelde, but are described for several historic European references, are establishing in the pilot site. Presence of initial terrestrial vegetation slowed down establishment at higher as well as lower locations. Removal of remnant vegetation prior to culvert opening could decrease this impact. Young willow forest and *Phragmites* stands are colonizing the area.

Tidal Marsh Fauna

The first estuarine macroinvertebrates were observed after three months of tidal inundation. After one year taxonomic richness on the frequently flooded sites exceeded the richness observed in the river sites, and alike the vegetation development, taxa which are absent in the river are observed to establish in the CRT. Oligochaete densities six fold exceed the river site densities. After three years, macroinvertebrate colonization continues, albeit with lower pace, on the less frequently flooded sites.

During the bird monitoring, 98 species were inventoried. 30% of all observations consisted of aquatic bentivores, 54% concerned wetland generalists. Other observations concerned terrestrial species. Several rare and endangered species are regularly foraging in the CRT. Almost all fish species occurring in the river near the pilot site were recorded within the CRT (Simoens et al. 2007). Foraging behaviour in vegetated zones during high tide was regularly observed. Fish does not seem to enter the site through the high inlet sluices, but prefers to enter the site swimming against the current during outflow. Because the outgoing discharge rapidly increases, fish passage is assumed to be possible only during a limited period of time after outflow starts. Adaptations of the tidal gate are investigated to broaden the time frame for fish passage (Peeters et al., in prep.).

9.10.6. Discussion

Creating good tidal conditions is a key factor for tidal marsh restoration, because hydrodynamics are the main driving force behind several physical, chemical and biological processes in tidal areas (Zedler et al., 2000). The CRT technique, combining a high inlet and a low outlet sluice permits to introduce a tidal regime in low laying sites at the landward side of the dike, providing a clear spring-neap variation. Our results confirm the advantage of high inlet and low outlet sluices, as modelled by Cox et al (2006) and Maris et al. (2007). This technique increases the number of suitable sites and avoids problems with suboptimal tidal regimes and artificial elevation increase with dredged material. The pilot CRT faces a wide range of inundation frequencies, and thus potential for a diverse estuarine ecosystem. The site however faces altered inundation time and period, as predicted by Cox et al. (2006). Monitoring now shows that the development of an estuarine ecosystem is not hampered by prolonged inundation times when inundation height is limited.

Silica processing was shown to be comparable in the CRT as in river marshes (Jacobs et al. 2008; Jacobs et al, in prep), while preliminary monitoring data indicate that nitrogen, phosphorus and oxygen processing also occur in comparable ways to river marshes. CRTs act, just like estuarine marshes, as a sink for nitrogen. On average, one kg of TDIN per ha per tide was removed, meaning more than 5 ton/year for the whole of the pilot site. In the Blackwater estuary, nitrogen removal though burial and denitrification in MR sites was modelled to be 0.07 kg/ha.tide at low sedimentation rates (1.5 mm/year) and 0.23 kg/ha.tide at higher sedimentation rates (6 mm/year) (Shepherd et al., 2007). Using our fivefold sedimentation rate (approximately 30 mm/year on average), comparable removals are obtained (1.0 kg/ha.tide).

Oxygenation of incoming water in the CRT was obvious. Maris et al. (2006) illustrated the potential role of a high inlet sluice in aerating the incoming water. Regardless weather or light conditions, the incoming, the often oxygen poor Schelde water is aerated, so that the pilot CRT never faced problems with anoxia. On the contrary, on sunny days, super saturation was recorded. A shallow water depth and low dynamics create a more favourable light climate in the CRT than within the estuary, so primary production is assumed to be much higher.

A few years ago, aeration was considered to be a very important ecosystem service in the Schelde. With the general improvement of water quality and oxygen levels (Cox et al, in prep.), this service is becoming less important. Nevertheless, if hypoxia ever prevails in the estuary, a CRT will act as an oxygen rich refuge. Given the highly eutrofic status of the estuary (Van Damme et al, 2005) and increasing algal blooms (Cox et al, in prep), hypoxia is not unlikely to occur.

Sedimentation and erosion incite the development of a creek network and typical tidal marsh topography. Development of vegetation (Jacobs et al., in prep) and macroinvertebrate communities is spectacular and even surpasses the present river communities. Presence of bird and fish species indicates the installation of a fully developed intertidal ecosystem. In the CRT sites which are currently being built, biota are expected to develop even better as the habitat will be larger, less disturbed and more variable.

Our results confirm the restoration potential of the CRT technique, as hypothesised by Cox et al (2006) and Maris et al. (2007). However, some potential differences in between estuarine marsh and the CRT technique must be noted:

The CRT's sedimentation process differs from the river marshes.
When river marshes accrete, they become flooded less and less frequent with decreasing water volumes. At a certain point, sedimentation rates become slow, and marsh elevation equilibrates around the mean spring flooding level. The river tidal marshes thus keep in pace with increasing tidal influences. In the CRT however, the entering volume is governed only by the culvert configuration, independent from the marsh elevation. No equilibrium is expected to install itself. CRT marshes are bound to accrete until they reach the height of tidal marshes inside the river. Fast accretion might be problematic if sedimentation rates during this catching up are too high for structures and processes to develop. Fast sedimentation can be problematic for other functions within the CRT site. In the Schelde estuary, many CRT areas are located within flood control areas, thus combining safety and ecology measures. For safety however, preserving storage capacity in the embanked site is essential, so sedimentation should be minimized. This can be achieved by installing the equilibrium at will: the culvert configuration can be adapted so that the mean spring tide does not flood the marsh surface, but only the creeks. However, this requires a flattened topography with a certain creek volume, which tidal marshes naturally develop when they accrete. This is structure is observed to develop in the CRT. On the other hand, the system can also be manipulated to maximize accretion when fast changes in elevation are wanted, e.g. to allow a fast development of a tidal marsh morphology or to prepare a site for managed realignment.

Water currents inside CRT differ from the river marshes.
On river tidal marshes, flooding water mostly enters at a broad section, and during spring tides the river flows through the marsh. Inside the CRT, all water enters at a small culvert section, leaves through another small section, and during high tide (stagnant phase) no current is observed. Although lower dynamics than in the river bed could be advantageous for colonization of certain communities, this could again promote too fast sedimentation rates for typical marsh structures and processes to develop (see above). The lack of currents could also impede development of natural drainage structures, which is of utmost importance for development of reference marsh topography as well as for seepage water exchange. Another consequence of the sheltered position of CRT's is the protection from extreme storm flood currents which deprives them from the natural setback mechanism of storm erosion events. A lowered dike stretch at storm flood level however permits extreme flooding during storm tides, and extreme currents can be obtained by opening all culverts to the lowest level. Finally, if these measures do not obtain

natural setback and this is still desired, vegetation management or even topsoil removal could be considered. Up till now it is not clear if or at which term setback will eventually be necessary.
On the other hand, at the marsh edges in the estuary, increasing tidal amplitudes and current velocities cause strong erosion, provoking steeper slopes and reducing the habitat with high IF. The remaining habitat, situated on the steep slope, faces high current speeds, so that vegetation development is almost excluded. In the shelterd CRT, habitat with high IF but low dynamics was restored.

Abovementioned concerns are the subject of several research projects conducted in the pilot CRT. Many different aspects of the developing ecosystem are being investigated on this site by different project partners and PhD-students. In the end, this allows evaluation of the potential of the CRT technique and result in application guidelines for bigger CRTs that are being constructed.

CRT: a 'deus ex machina' solution for freshwater tidal marsh restoration?
Only one relatively small pilot CRT site has up till now been built. The tidal regime in the site has only been restored for three years. Research on desired structures and processes for restored tidal marshes is still premature, while quantitative requirements are still under discussion as science evolves. Any present conclusion on restoration success is thus qualitative (table 4).

Table 4 Processes (italic) and structures (bold) to be developed in restored freshwater tidal marshes in order to comply with European legislation; the present state compared to river marshes, and their expected future statecompared to present river marshes.

processes and **structures**	**Present CRT**	**Future CRT**
Tidal cycling	=	=
Nutrient processing (Si, N, P,O)	=/+	=/+
Sedimentation	=	=
Tidal marsh topography	-	=
Tidal marsh vegetation	+	+
Tidal marsh fauna	+	+

This paper however clearly points out the high potential of the CRT technique for successful restoration of structures and processes of reference freshwater tidal marshes. In fact, since the CRT-approach vastly increases the number of suitable sites for tidal habitat restoration, it even enhances opportunities to combine different societal functions in coastal defense schemes which include restoration projects. This increases the public acceptability and political willingness to implement whole coastal defense schemes (French 2006, Weinstein 2007). As more tidal marshes are currently being restored with CRT along the freshwater and brackish reaches of the Schelde, more case studies and longer time series will permit to verify this statement and further quantify restoration success.

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9.10.8. References

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