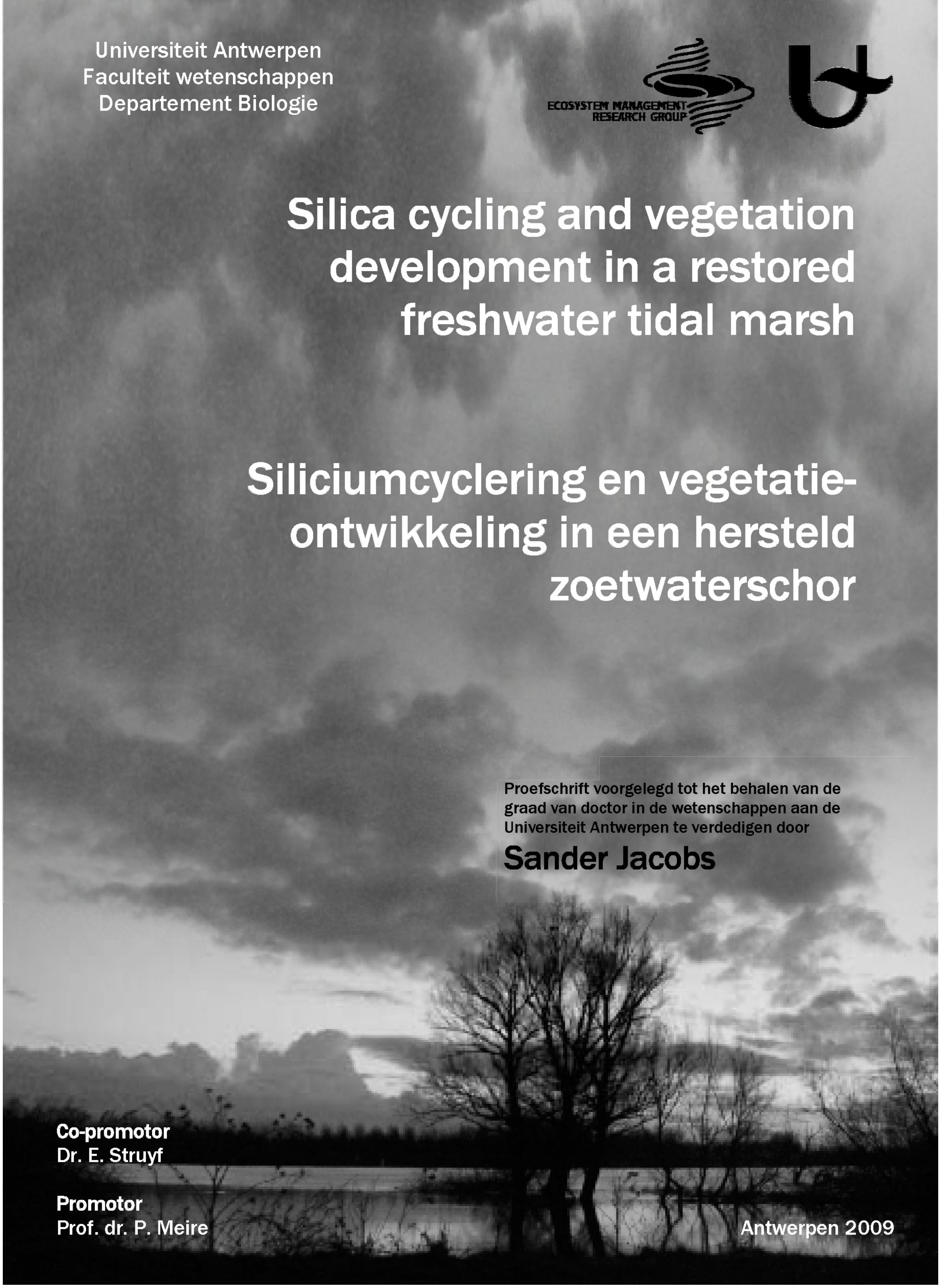

Universiteit Antwerpen
Faculteit wetenschappen
Departement Biologie

ECOSYSTEM MANAGEMENT
RESEARCH GROUP

Silica cycling and vegetation development in a restored freshwater tidal marsh

Siliciumcyclering en vegetatie-ontwikkeling in een hersteld zoetwaterschor

Proefschrift voorgelegd tot het behalen van de graad van doctor in de wetenschappen aan de Universiteit Antwerpen te verdedigen door

Sander Jacobs

Co-promotor
Dr. E. Struyf

Promotor
Prof. dr. P. Meire

Antwerpen 2009

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> Rivers know this: there is no hurry. We shall get there some day.
>
> *Pooh's Little Instruction Book*
> inspired by A. A. Milne

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Echt bedankt.
Gent, 29/09/2009

Siliciumcyclering en vegetatie-ontwikkeling in een hersteld zoetwaterschor

Ecosystemen produceren een brede waaier aan grondstoffen die broodnodig zijn voor de mens. Samen met de groei van de menselijke populatie groeit ook de vraag naar deze grondstoffen en diensten van ecosystemen en dus de impact van onze globale voetafdruk. Velen verkeerden lang in de verkeerde veronderstelling dat deze ecosysteemdiensten gratis, onkwetsbaar en onuitputtelijk zijn. Maar vandaag wordt de impact van menselijk gebruik en misbruik meer en meer duidelijk: lucht- en waterkwaliteit worden meer en meer bedreigd, oceanen zijn overbevist, plagen en ziektes breiden uit buiten hun historische grenzen, ontbossing bedreigt de natuurlijke bescherming tegen overstromingen rond bewoning, enz. Onderzoek wijst uit dat ongeveer 40-50% van het ijsvrije landoppervlak zwaar vervormd of beschadigd is door menselijke activiteiten, dat 66% van de visgronden overbevist of bijna uitgeput zijn, dat atmosferische CO_2 met meer dan 30% is toegenomen sinds de aanvang van de industriële revolutie, en dat bijna 25% van de vogelsoorten is uitgestorven in de voorbije 2000 jaar. Meer en meer realiseert men zich hierdoor dat ecosysteemdiensten niet alleen eindig zijn en bedreigd worden, maar dat de afweging tussen korte en lange termijn menselijke belangen zeer dringend moet gemaakt worden. Het beschermen en herstellen van ecosystemen is in deze optiek dus een kwestie van zelfbehoud.

Estuaria vormen de overgang tussen land en zee en zijn een vitale schakel in het functioneren van de biosfeer. Estuaria worden beschouwd als één van de productiefste ecosystemen en belangrijkste leveranciers van ecosysteemdiensten, zoals biogeochemische cyclering van nutriënten, zuivering van water, buffering tegen overstromingen, handhaven van biodiversiteit, productie van biomassa of kraamkamerfunctie voor voedselbronnen als vis en schaaldieren.
Tegelijk lijden net estuaria zwaar onder de menselijke impact doordat ze afvoerwater van grote gebieden concentreren: ze worden beïnvloed door het cumulatieve effect van vervuiling, erosie en andere problemen in het stroombekken. Een stijgende urbanisatie, het intensifiëren van de landbouw en de industrie hebben geleid tot een gestegen influx aan poluenten, stikstof (N) en fosfor (P). Op wereldschaal gaat het om een stijging met een factor twee, maar in vele estuaria is het probleem veel groter. Ook worden estuaria fysiek bedreigd door inpolderen, afdammen, baggeren en afleiden van de bovenstromen.
De stijgende zeespiegel en klimaatverandering doen de vraag naar de ecosysteemdiensten van estuaria acuut stijgen, terwijl tegelijk de veerkracht van deze ecosystemen net wordt ondermijnd. Het behouden en herstellen van robuuste en veerkrachtige estuariene ecosystemen is daarom cruciaal.

De Schelde is een perfect voorbeeld van een gedegradeerd estuarium. Het stroombekken van de Schelde wordt door meer dan 10 miljoen mensen bewoond. Inefficiënt lokaal waterbeheer en veranderingen in landgebruik leidden in het einde van de zeventiger jaren tot zuurstofloze periodes en sterfte van organismen. Velen herinneren zich de Schelde in deze periode als een zwarte, stinkende rivier. Tegelijk wordt hele estuarium intensief gebruikt voor de toegang van de Antwerpse haven naar de Noordzee en het hinterland, en wordt een deel van de bovenstroomse afvoer afgeleid voor de toegankelijkheid van de havens van Gent, Terneuzen en Zeebrugge. Bovendien kampte het estuarium met een ernstig veiligheidsprobleem, met als climax de rampzalige overstromingen van februari 1973.

De Europese Instandhoudingsdoelstellingen (IHD) zijn een gevolg van het groeiende belang van robuuste en gezonde ecosystemen. De IHD vereisen het opstellen en navolgen van maatregelen waarmee economische ontwikkeling wordt verbonden aan de juridisch afdwingbare voorwaarde van een robuust ecosysteem. De IHD-Schelde, die werden opgesteld in 2005, omvatten een soorten-, habitat- en ecosysteemluik.

In dit laatste wordt een gewenste oppervlakte getijdengebied berekend, grotendeels gebaseerd op recent onderzoek van de siliciumcyclus: door de historisch gestegen concentraties van N en P ten opzichte van Si, verschuift de samenstelling van de planktongemeenschap, die voor een groot deel bestaat uit diatomeeën welke silicium vereisen voor de opbouw van hun schaaltjes. Het relatieve belang van diatomeeën in de planktongemeenschap daalt, terwijl alle silicium wordt geconsumeerd (silicium-depletie). Zo'n verschuiving in de planktongemeenschap, die de basis vormt van het estuariene en oceanische voedselweb, kan rampzalige gevolgen hebben. Recent onderzoek wees echter uit dat schorren bij silicium-depletie opgelost silicium naleveren en dus de veerkracht van het ecosysteem vergroten door te fungeren als silicium-buffer. Het verder onderzoeken van de siliciumcyclus van getijdengebieden is één van de doelstellingen van dit doctoraat. Voor het Schelde-estuarium betekende het formuleren van de IHD het creëren van duizenden hectaren getijdengebied en wetlands langsheen het estuarium, met als deadline het jaar 2030.

Het Sigmaplan – dat na de overstromingen van 1973 ontstond om bescherming te bieden aan alle inwoners van het Schelde-estuarium – was anno 2000 aan herziening toe door de stijgende zeespiegel: de ambities werden opgevijzeld en het voor het behalen van een afdoende bescherming langs het hele estuarium werd gekozen voor het voorzien van waterberging tijdens stormtij.

In december 2004 besliste de Vlaamse regering dat het vernieuwde sigma-plan drie doelstellingen moest verwezenlijken: Toegankelijkheid – Veiligheid – Natuurlijkheid. De schaarste aan ruimte en de vele belanghebbende sectoren in Vlaanderen maakten dit tot een moeilijke oefening. Ook het combineren van overstromingsgebied voor natuurontwikkeling (IHD) en waterberging (Veiligheid) stelde technische problemen: in een overstromingsgebied kan niet zomaar een ecologisch functioneel getijdengebied worden gecreëerd, omdat de ingepolderde gebieden langs de Schelde zeer laag liggen ten opzichte van de huidige getij-amplitude, zodat een klassieke ontpoldering zou resulteren in het creëren van onderwater- of slikhabitat daar waar een slik-schorgradiënt is vereist.

Gecontroleerd Gereduceerd Getij (GGG) is een waterbouw-technische oplossing voor bovenstaand probleem: door het voorzien van een hoge en aanpasbare inwateringsdrempel, gecombineerd met een lage uitwateringssluis in één richting kan in laaggelegen polders een gereduceerd getij worden gerealiseerd dat zowel toelaat de hele getijdengradiënt te creëren als springtij-doodtij cycli volgt. Dit unieke systeem, dat uitblinkt in eenvoud, biedt mogelijkheden om waterberging bij stormtij te combineren met het creëren van getijdennatuur in ingepolderde gebieden. De GGG-techniek heeft echter tot gevolg dat het getij in de polder afwijkt van het getij op de estuariene schorren, bijvoorbeeld door een verlengde overstromingsduur en het vaker voorkomen van langere periodes van eb of vloed. Daarom werd een pilootproject gestart om de impact van GGG op het ontwikkelen van estuariene natuur na te gaan. Dit gebied, een voormalige landbouwpolder van ca. 8ha, is het belangrijkste studiegebied van dit doctoraat, dat focust op de silicium-buffering en vegetatieontwikkeling van deze site.

Dit doctoraat heeft enerzijds tot doel een bijdrage te leveren aan het fundamentele onderzoek van de siliciumcyclus, met een focus op de uitwisseling in getijdengebieden. Anderzijds beoogt het de kennis over herstel van estuaria te vergroten, meerbepaald door het evalueren van het potentieel van de GGG-techniek voor schorherstel . Daarom zijn de hoofdstukken veelal een combinatie van toegepaste en theoretische aspecten:

Hoofdstuk 1 leidt de algemene concepten in die het kader van dit onderzoek vormen.

Hoofdstuk 2 beschrijft de kenmerkende vegetatie van zoetwaterschorren in noordwest Europa. Het onderzoek vond plaats in de zoetwaterzone van het estuarium, en onderzoek over vegetatie en ecologie van zoetwatergetijdengebieden is zeldzaam. In dit hoofdstuk worden beschikbare data over vegetatie van zoetwaterschorren beschreven en de bedreigingen besproken. De resultaten dienen later als vergelijkingspunt voor de vegetatieontwikkeling in het onderzoeksgebied. Dit hoofdstuk werd gepubliceerd als hoofdstuk in een boek.

Hoofdstuk 3 vergelijkt bovenstaande referentiebeschrijving met de ontwikkeling van vegetatie in de pilootsite waar gecontroleerd gereduceerd getij werd gerealiseerd. Vegetatie-ontwikkeling was snel, met een bijna onmiddellijk verdwijnen van terrestrische pioniers en een gestage kolonisatie van estuariene soorten gedurende de eerste seizoenen. Er werd een duidelijke invloed van hoogteligging (overstromingsregime) geobserveerd, wat wijst op de ontwikkeling van de vereiste gradiënt, terwijl de aanwezigheid van terrestrische pioniers de vegetatieontwikeling vertraagt. Het belangrijkste resultaat is echter het verschijnen van plantengemeenschappen die zeldzaam zijn langsheen de Schelde en andere gedegradeerde estuaria, maar die wel worden beschreven voor historische zoetwatergetijdengebieden. Dit kan te wijten zijn aan de afwijkende tijkarakteristieken van de GGG-techniek, of aan de beschutte ligging die de site beschermt tegen grote stroomsnelheden in het estuarium. Dit hoofdstuk werd gepubliceerd in artikelvorm.

Hoofdstuk 4 maakt de link met de siliciumcyclus. Het focust op plantaardig silicium in de ontwikkelende getijdenzone van het GGG-gebied. De groottes van siliciumtransporten doorheen ecosystemen zijn niet goed gekend omdat er weinig gepubliceerde onderzoeken zijn die ook silicium in rekening brengen. Dit onderzoek maakt een inschatting van de totale siliciumstock in de schorvegetatie. De vegetatie omvat een significante siliciumstock, hoofdzakelijk geconcentreerd in enkele silicium-accumulerende soorten. Het belang van deze siliciumstock voor de nalevering van opgelost silicium op momenten van estuariene depletie is echter nog onzeker. Dit hoofdstuk maakt deel uit van een artikel in voorbereiding.

In het vijfde hoofdstuk wordt de siliciuminhoud van estuariene sedimenten vergeleken met die van de pilootsite. De verschillen tussen estuariene en GGG-sedimenten verdwijnen al na het eerste seizoen. De siliciuminhoud varieert echter sterk, zowel tussen habitats als tussen verschillende dieptelagen. De geobserveerde patronen worden gelinkt processen als sedimentatie, ecologie van benthische algen, grondwaterfluxen, oplossing en reprecipitatie. Deze hypotheses worden gesynthetiseerd in een conceptueel model voor silicium-biogeochmie van getijdengebieden. Dit hoofdstuk is ingediend in artikelvorm.

Het zesde hoofdstuk is een directe evaluatie van de silicium-buffering door de pilootsite. De procentuele nalevering gedurende volledige tijcycli onder verschillende omstandigheden worden vergeleken met eerder gemeten nalevering van estuariene schorren. De pilootsite vertoont dezelfde naleveringspatronen maar is nog niet op het niveau van de estuariene schorren. Om dit verder te verklaren worden verschillende habitats binnen in het studiegebied vergeleken en worden processen die betrokken zijn bij de silicium-buffering geëvalueerd. Zowel de nog geringe morfologische ontwikkeling als de afwijkende getijkarakteristieken kunnen aan de basis liggen van dit geobserveerde verschil. Dit hoofdstuk werd gepubliceerd in artikelvorm.

In hoofdstuk zeven worden de conclusies van de voorgaande hoofdstukken geplaatst in het kader van estuarien herstel. Gecombineerd met bijkomende observaties en preliminaire conclusies wordt een eerste evaluatie gemaakt van het potentieel van de GGG-techniek voor het creëren van functionele estuariene habitats. Ook worden aanbevelingen voor verder onderzoek gegeven. Delen van dit hoofdstuk zijn vervat in een ingediend artikel.

Chapter 1
Introduction

1.1 Estuaries

Estuaries - as the transitional zone between land-based ecosystems and the world ocean - have since long been intriguing areas to scientists of many disciplines. Estuaries are vital to the biosphere's functioning. This is expressed by their complex geology, hydrology and morphology, their prominent role in the historical and actual support of economies and ecosystems, their manifest dynamics that both sustain and put at risk all inhabiting organisms and their prominent role as biogeochemical filter for land-ocean exchanges.

Estuaries and coastal marine ecosystems are cited among the most productive biomes of the world, and serve important life-support systems also for human beings (Day et al. 1989, Costanza et al. 1993). Estuaries support many important ecosystem functions: biogeochemical cycling and movement of nutrients, purification of water, mitigation of floods, maintenance of biodiversity, biological production, nursery grounds for commercial fish and crustacean species, etc. (Daily et al. 1997, De Groot 1997, Meire et al. 1998).

Other highly productive systems, such as coral reefs and tropical rain forests, differ greatly in how their productivity is achieved. Reefs and tropical rain forests efficiently recycle the limited resources through a very diverse ecosystem. In contrast, the estuarine ecosystems achieve very high productivities -despite their low species richness- through the continuous arrival of new nutrient supplies. Estuarine and marine food webs are based essentially on diatoms (Irigoien et al. 2002, Kimmerer 2005), which make out the bulk of the phytoplankton community. Zooplankton feeds on this community and on detritus, and forms the main food source for zoobenthos, shellfish and young fish. Highest trophic levels like fish, birds and humans top the food web.

Another distinctive feature contrasting estuaries from other biomes is the nature and variability of the physicochemical forces that influence these ecosystems. Within small geographic regions, many estuaries experience widely varying conditions of temperature, salinity, concentrations of a wide variety of chemicals, and plant and animal densities, many of whose are mediated by water movement over relatively short timescales (Day et al. 1989, McLusky and Elliot 2004).

Estuarine biotas are therefore geomorphologically very dynamic and ephemeral systems, influenced by both sea and land changes, resulting in a complex mixture of many different habitat types.

Habitats do not exist in isolation, but rather have physical, chemical and biological links between them, for example in their hydrology, in sediment transport, in the transfer of nutrients and in the way mobile species move between them both seasonally and during single tidal cycles (Meire et al. 2005). Even small estuaries are typically composed of a mosaic between four and nine major habitat types (subtidal, intertidal mudflats, intertidal sand flats, marshes, shingles, rocky shores, lagoons, sand-dunes and grazing marshes/coastal grassland (Davidson et al. 1991)). Despite the many different habitat types, relatively large and unpredictable variations in salinity (physiological stress) and water movement (physical stress) tend to limit the number of organisms capable of adapting to these extreme conditions (Day et al. 1989, McLusky 1989), hence the before mentioned lower species richness compared to the freshwater reaches above the tidal limit or the truly marine habitat outside the estuary. Although estuaries contain relatively few species, the abundance and biomass of organisms is usually very high. For more than a century, estuaries and their fringing marshes have thus been classified as essential habitat for a wide scope of organisms, and up to 80% of marine recreational and commercial species are believed to have estuarine dependent life stages (Weinstein and Kreeger 2000). Accordingly, an estimate of the economical value of these ecosystem functions (goods and services) indicated that estuaries are among the most valuable ecosystems in the world (Costanza et al. 1997).

Freshwater tidal reaches of estuaries have received little attention with respect to anthropogenic impacts compared to the other estuarine parts. While large stretches of freshwater estuarine reaches were destroyed by weir constructions, the remaining areas were ignored by freshwater biologists because of tidal excursion and by estuarine scientists because they defined the end of estuaries at the upper limit of salt intrusion (Schuchard et al. 1993, Elliot and McLusky 2002). Moreover, the common assumption that water volumes are too small and residence times too short for freshwater processes to have any effect at the scale of the estuary is being replaced by the hypothesis that higher process rates mitigate the volume effect, increasing the importance of freshwater tidal reaches for nutrient turnover (Vanderborght et al. 2007, Van der Zee et al. 2007).

Tidal marshes are found along coasts and estuaries where tidal action is significant but wave action is rather limited. They are typically located high in the intertidal zone around high water level. Tidal marshes are characterised by a rather flat vegetated surface, called the marsh platform, which is dissected by branched networks of tidal creeks that narrow and shallow inland. As a consequence of their elevation they are only flooded by the highest high water levels occurring around spring tide and they are typically overgrown by specific marsh plants which are able to withstand tidal inundation. At their seaward edge, tidal marshes are mostly bordered by tidal flats, which have a lower elevation and which are flooded every tidal cycle. The lower limit of vegetation development is situated in this zone.

Tidal marshes are world-widely recognised as valuable habitats for plants and animals, such as unique salt marsh species and large populations of water and wading birds. Furthermore, they play an important role as sinks and/or sources for suspended sediments, nutrients and pollutants from and to the adjacent marine/estuarine ecosystem (e.g. Allen and Pye 1992). This recognition has led to the protection of tidal marshes by several international legislations. Connected to this ecological value, tidal marshes have an economical value as breeding grounds for fish, shellfish and crustaceans. Also, they protect inland and infrastructure against flooding, by storing water and damping waves and flows during high water levels associated with storm surges.

1.2 Estuarine degradation

Estuarine degradation comes in the direct physical form of habitat loss due to embankments, dredging activities, upstream discharge deviation in transport waterways and construction of harbour infrastructure and weirs. Recently, large losses of tidal marshes were reported globally (Wolters et al. 2008). These losses have been attributed to several factors associated with human development and climate change. (Adam 2002). In times of relative sea level rise, tidal marshes must increase in elevation in order to avoid being 'drowned'. This can either be achieved by sedimentation (Temmerman 2003) or by inland movement. In situations where land claims have resulted in consuming the inland buffer, tidal marshes are not able to migrate inland. This phenomenon is called 'coastal squeeze'. In the following section, estuarine degradation with focus on silica will be discussed.

Estuarine degradation is often apparent in the form of eutrophication, which is one of the most important anthropogenic problems estuarine systems are confronted with. Since estuaries concentrate run-off from large land surface in relatively small water bodies, they are affected by the cumulative effects of human activities and land-use (Cloern 2001). Increased urbanisation, agriculture and industrial activities have led to large increases in the delivery of N and P along the land-ocean continuum. On a global basis, the fluxes of these elements to the oceans have increased by a factor two. Increased N and P loadings increase primary production and, in many systems, diatom production (Ragueneau et al. 2006). Subsequent increase in sedimentation rate of diatoms can lead to increasing amounts of diatoms and BSi (Biogenic silica) being stored in the sediments, eventually leading to a reduction in DSi (Dissolved silica) supplies to the water column, especially in systems with long residence times (Conley et al. 1993). Also, several hydrological alterations have been observed to lead to Si-depletion (Ragueneau et al. 2006). Kilham's hypothesis (1971) that silica depletion by excessive inputs of N and P could shift algal communities, was strengthened by observations in various environments across the globe (e.g. Schelske and Stoermer 1971, Officer and Ryther 1980, Conley et al. 1993) and harmful effects of silica depletion (or N/P excess) on coastal food web structure and energy dissipation to higher trophic levels were put into evidence (e.g. Smayda 1990, Turner et al. 1998, see also Ragueneau et al. 2006). Eutrophication phenomena in estuaries are thus related to the balance between N, P and Si in river loading, and depend on interactions between human activities and natural processes in the watershed, which ultimately determine the riverine nutrient delivery into the marine environment (Officer & Ryther 1980, Cugier et al. 2005). Eutrophication can cause anoxia, extreme turbidity, toxic algal blooms, and reduction in the ability of coastal food webs to support higher trophic levels (Cugier et al. 2005, Tréguer et al. 1995, Kimmerer 2005) and the vital estuarine goods and services.

It has long been thought that at the same time as N and P fluxes increased, dissolved silica (Dsi) fluxes have remained constant because the major source of DSi to rivers is natural silicate rock weathering (Subramanian et al. 2006). However, it has recently been recognised that there are significant anthropogenic sources of DSi. Soluble silicates such as sodium silicates (water glass) are some of the largest-volume synthetic chemicals in the world and are used in both industrial and household products (van Dokkum et al. 2004). Additionally, various forms of soluble silicates are being used as commercial fertilisers (Datnoff et al. 2001). The quantities of the additional sources of DSi are poorly constrained at present. Little is known about changes in the production and weathering of Si with changes in land use (Likens et al. 1970). The long-term effects of soil disturbance with agriculture (e.g. Conley et al. 2008) and the more recent effects of acid rain are also unknown. General consensus remains that N and P are delivered in excess of Si in many temperate European and North American river systems (Billen et al. 2007). It is difficult to determine the extent to which anthropogenic inputs of DSi may compensate for observed reductions in DSi fluxes. However, recent estimates suggest that the additional load of anthropogenic DSi into rivers from industries and households contributes only 2 percent of the annual DSi in Western Europe (van Dokkum et al. 2004).

1.3 Tidal marsh restoration

The value of tidal marshes, their degradation and legislative protection have urged many governments to preserve and restore these habitats. Today, tens of thousands of hectares of degraded and/or isolated tidal marshes are therefore being restored worldwide. Yet, whether restored areas reach functional equivalency to 'natural' systems is subject of heated debate (Weinstein and Kreeger 2000).

To restore tidal marsh surface, managed realignment -breaching the dike- is a technique used in various countries, albeit under a different name and driven by different incentives (Wolters et al. 2008). The effectiveness of this technique is severely debated (Hughes and Paramor 2004, Morris et al. 2004), and a whole range of considerations has to be taken into account (reviewed in French 2006). Elevation is the key factor in the planning of managed realignment (French 2006, Cundy et al. 2002). Elevation is also the main criterion for suitable site selection as it relates to the frequency and duration of tidal inundation (French 2006). Most of the sites adjacent to estuaries have been lowered by years of dewatering and compaction. This typically leaves them below the levels of contemporary marshes within the same system, who have kept in pace with sea level rise (French 2006). The difference in elevation thus rules out many sites for potential realignment.

This problem can be 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). This strongly restricts the number of potential realignment sites. In the US, it is common to increase site elevation prior to breaching (French 2006). However, artificial site elevation has been the most common reason for site failure (Quammen 1986, Perry et al. 2001). A third option is to install a restricted tidal exchange (RTE). This has the advantage of tempering the tidal wave in the site to an acceptable level, but the technique cuts out the spring-neap variation, since always the same water volume is entering. The spring-neap variation provides the variation in flooding frequencies and duration needed for optimal habitat diversification. According to French (2006) one of the major discrepancies in the realignment technique is the obligation to only select suitable sites or increase elevations artificially.

The Controlled Reduced Tide (CRT) technique however permits to install a tidal regime in virtually every lowered site, and in the same time install an adaptive spring-neap variation. This increases the number of suitable sites and avoids problems with suboptimal tidal regimes and artificial elevation increase with dredged material. This new technique is discussed further on in this chapter.

Also, nutrient management strategies are needed for sustainable use of coastal zones (Conley 2002). In many countries, large reductions in N and P loads are obtained through purification of urban wastewater. However, non-point sources (precipitation, agricultural runoff) still supply excessive inputs of N and P, which are difficult to tackle. Estuaries, as the physical bottleneck of land-sea transport, provide a last opportunity to manage waters entering the coastal zone. Apart from the pioneering work of Struyf (2006), there are to our knowledge no publications focussing on the direct management of the harmful effects of Si-depletion in the estuary itself.

1.4 Silica in estuaries and tidal marshes

Silica - normal people associate this element with applications in plastic surgery or micro-electronics. Even less normal people like most ecologists seem to consider silica dull as ditchwater, since it is so abundantly present in ecosystems. Some geeks however obstinately continue the study of this element and its ecological cycles. The following section aims to improve appreciation for the ambitions and endeavours of silica scientists.

Silicon - on a global scale - is the second-most abundant element in the earth's crust with 28.8% (Wollast and McKenzie 1983, Wedepohl 1995). It occurs in more than 370 rock-forming minerals. Being an element of almost all parent materials, Si is one of the basic components in most soils. Soils show a huge span of Si contents ranging from <1 wt.-% up to 45 wt.-%. Some secondary Si-enriched horizons are almost entirely built up by Si (>95 wt.-%).

Silica plays a very important role in the global carbon cycle through weathering processes and Si fluxes to the oceans. Weathering reactions consume CO_2, resulting eventually in the storage of $H_2CO_3^-$ as carbonates. The processes of silica weathering take part in the regulation of atmospheric CO_2 (Berner et al. 1983). Moreover, the C-sequestrations in oceans are strongly coupled with the global cycle of Si (Treguer et al. 1995, Ragueneau et al. 2000, Yool and Tyrell 2003). This is because diatoms -which need Si for their skeleton-, comprise approx. 50% of the biomass in today's oceans (Treguer and Pondhaven 2000). At non-limiting N and P concentrations, larger Si-supply to oceans results in higher export flux of C to marine sediments and more C finally being removed from the atmospheric CO_2 pool (Sommer et al. 2006), as diatoms are buried in marine sediments.

In today's oceans approx. 5 Tmol Si/yr are supplied from the continents by rivers, which equals 80% of the annual fluxes into oceans (Tréguer et al. 1995, Conley 2002). The ultimate source of all continental Si fluxes to the oceans are indeed weathering processes in terrestrial biogeosystems, but this silica is transferred and recycled between various pools (mineral, organo-mineral, biogenic) before finally entering the ocean. In general, knowledge about size, transformations and properties of the biogenic silica (BSi) pools is very scarce (Sommer et al. 2006). In estuaries, which form the last stop in the land-sea transport, dissolved silica is taken up seasonally by benthic and pelagic diatom communities (Wollast and De Broeu 1971), and by estuarine marsh vegetation (Struyf et al. 2005).

The estuarine phytoplankton community is always dominated by diatoms when DSi concentrations are more than 2 µM (Egge and Aksnes 1992). A Si/C molar ratio of 0.79 ± 0.43 has been found for twelve freshwater species (Sicko-Goad et al. 1984), whereas a value of 0.13 ± 0.04 is reported for twenty-seven marine species (Brezinsky 1985). The freshwater diatom community thus potentially comprises an important amount of transported silica. Conley (1997) estimated that 16% of the gross riverine Si load is delivered to the world ocean as BSi. This estimate hides enormous spatial and temporal variability: during blooms, BSi may constitute one third of the gross silica delivery, as in the Danube (Ragueneau et al. 2002b) or it may even exceed DSi inputs, as in the Rhine (Admiraal et al. 1990).

Apart from the transport to the ocean and internal pelagic recycling, silica is also translocated to subtidal and intertidal sediments. Seasonal sedimentation and burial of phytoplankton production is a major source of nutrients for subtidal sediments in open coastal areas (Graf et al. 1982, Gili and Coma 1998). Tidal marshes also consist almost entirely of estuarine sediments which settle on the marsh surface and vegetation. Moreover, benthic microalgal assemblages, consisting primarily of diatoms (McIntyre et al. 1996, Wulff et al. 1997, Facca et al. 2002), are known to flourish on intertidal flats (Ragueneau et al. 2006) and can have major control on nutrient fluxes at the sediment-water interface (Sundbäck and Granéli 1988, Sundbäck et al. 1991, Rizzo 1990). Indeed, most ASi (amorphous silica, meaning biogenic and other easily soluble fractions of silica) in intertidal sediments is comprised of diatoms (>80%, Conley & Guo, unpubl.). Freshwater diatoms have relatively high silica content compared to more downstream species (Conley & Kilham 1989), and permanent deposition of Asi (Amorphous silica) in freshwater tidal marshes and sediments could be more efficient compared to more downstream sites (Conley and Kilham 1989). Struyf et al. (2006) found ASi-concentrations of approx.

597 g.m^{-2} on freshwater tidal marshes, and on a mesohaline tidal marsh, an estimated ASi-content of 82 g.m^{-2} was reported (Norris and Hackney 1999). These observations point to freshwater intertidal marshes as important silica sinks, with sedimentation and growth of diatoms as the major import processes.

The Si content of terrestrial plants is higher than the average value for soils, ranging from 0.1 wt. % to 10 wt. % (dry weight basis, Raven 1983; Epstein 1999), although some plants have higher Si-contents. The silica taken up by plants is either stored in biomass or returned to the soil through litter fall and subsequent decomposition. The magnitudes of the fluxes of Si in terrestrial ecosystems is still not well established because published studies on biomass and litter fall including Si data are scarce. An overview is given in Conley et al. (2006). Largest fluxes are found in ecosystems with highest productivity (e.g. Lucas et al. 1993), and exceptionally large values are observed in vegetations consisting of silica-accumulators like rice, sugar cane (Berthelsen et al. 2001) or -in estuaries- reed.

DSi taken up by marsh plants from the pore waters is stored as amorphous silicon dioxide in plant phytoliths. It is accumulated during the plant life cycle and only released to the environment after the plant decomposes. Struyf et al. (2005) studied silica accumulation in several estuarine vegetations on a tidal marsh, and found silica-stocks of characteristic *Phragmites australis* dominated vegetations comparable to those found in an Amazon rainforest, while yearly litter fall deposition of BSi in the reed marsh is much higher compared to forest ecosystems (Phragmites australis 68g BSi m^{-2}y^{-1}; forest ecosystems 0.5-4.1 g BSi m^{-2}y^{-1}; Struyf et al. 2007b). At peak biomass, freshwater tidal marsh vegetation contained about 227±61 kg.ha^{-1}, of which 96% in reed dominated vegetations. Moreover, Struyf et al. (2007) showed that over 98% of the silica in *Phragmites australis* litter dissolved within one year.

This dissolved silica is transported from the pore water to the estuarine water column by advective transport at high tides and seepage (gravitational drainage) at low tides. The physical properties of this transport are still to be disentangled.

Altogether, this leads to an estimated DSi-export potential of 213±57 kg.(ha.y)$^{-1}$ for freshwater tidal marshes, which largely exceeds maximal values found in literature (e.g. Alexandre et al. 1997: 54 ± 5 to 70 ± 7 kg.(ha.y)$^{-1}$ for rain forest on latisol in Congo). This does not take into account the potential export from the sediments.

However, despite this large potential silica export, tidal marshes in general remain an overall silica sink since all silica in this system originates from abovementioned imported amorphous silica deposits which mostly orginate from plankton communities, benthic as well as pelagic, of which large amounts are buried when marshes accrete.

1.5 The silica management hypothesis

A modelling study by Struyf et al. (2007) quantified the ability of a freshwater tidal marsh to recycle and retain Si through its development from initial marsh to climax vegetation stage. Significant differences in sediment ASi accumulation were described during this development, with more efficient retention in young, lowly elevated tidal marshes. Tidal marshes export dissolved silica (DSi) to the estuary as ASi is recycled (Struyf et al 2006). This export is highest at moments of estuarine silica depletion and could thus enhance the resilience of the estuary to silica limitation. Still, little information is available concerning the spatial and temporal variability of the ASi buffer in tidal marshes, especially in recently restored sites. As excessive N and P inputs from diffuse sources are difficult to tackle and might continue to cause nutrient ratio imbalance, restoration of tidal marshes might offer a last chance opportunity to manage Si:N:P ratio's before the water enters the ocean.

Struyf et al. (2005b) estimated the amount of tidal marsh that, based on their data, would be necessary to reduce Si-limitation in the Schelde estuary. They concluded that an additional 1500ha of functional tidal marshes would suffice to buffer problematic Si-depletion events as observed between 1996 and 1999 in the freshwater part of the estuary, and an additional 7300 to solve problems in the whole estuary. In a natural system, uninfluenced by anthropogenic pressures, Si marsh recycling will probably not be necessary to balance the Si:N:P ratios because of the low input of N and P. Depending on the reductions in N and P due to installation of wastewater treatment plants, anthropogenic impacted rivers like the Schelde could need these tidal marshes to cope with imbalanced nutrient inputs.

1.6 Restoration in the Schelde estuary

1.6.1 The Schelde estuary

The river Schelde, with a catchment covering 21.863km^2 is situated in the northwest of France (31%), the west of Belgium (Flanders, 61%) and the southwest of the Netherlands (8%). Most of the river basin is urban; the total population of the catchment numbers more than 10 million people, with densities varying from 100 to more than 2000 inhabitants per km^2 (averaging 400 inh.km^{-2}). The estuary of the Schelde is 170 km long and comprises both a freshwater part (80 km long) and a brackish-marine part (90 km long). The total length of tidally influenced rivers is about 240 km. Mean depth varies from about 14 m at the mouth to 7 m near Gent, where tidal influence is blocked by sluices. The tidal regime is semidiurnal, with mean neap and spring amplitudes at the mouth of 2.97 and 4.64 m respectively (Claessens and Meyvis 1994). As the tidal waves enters the estuary, these amplitudes increase to 4.49 and 5.93 m at km 90 and decrease further inland to 1.84 and 2.24 m (Temmerman et al. 2003). Strong northwest winds can create large storm surges resulting in water levels up to 3 m higher than mean high water level (Claessens and Meyvis 1994). Upstream discharge is mainly deviated in the canal Gent-Terneuzen and the canal Gent-Oostende. It varies between 50 m^3.s^{-1} during dry summer months and 300m^3/s during wet winter months (Taverniers et al. 2000). The averaged yearly discharge is 50±25 m^3.s^{-1} in the freshwater part (Struyf et al. 2004). Residence time is relatively large as a result of this small river discharge and the large tidal influence. It was estimated at 2-3 months for the brackish-marine part (Soetaert & Herman 1995b, Regnier et al. 1997). The water column is generally well mixed but the upper estuary may be slightly stratified during high peak discharges (Soetaert et al. 2006). Suspended sediment concentration varies from 30-60 mg.l^{-1} in the saline part (Van Eck et al. 1991) up to 100-200 mg.l^{-1} in the brackish part (Van Damme et al. 2001), and decreases again to 50-100 mg.l^{-1} in the upstream part of the estuary (Temmerman et al. 2003). As a result of tidal asymmetry, sediments accumulate in the upstream part, as more sediment is transported upstream during high tide than is transported downstream during low tide ('tidal pumping'). The cross-sectional area increases smoothly from the river to the mouth. Apart from abovementioned channelling of upstream parts, the estuary undergoes serious modifications by dredging activities to maintain and enlarge port facilities near Antwerp and transportation possibilities throughout the estuary.

The Schelde estuary is one of the most nutrient-rich systems in the world and the only true estuary in the Dutch Delta region (Soetaert et al. 2006). Ineffective local water management and changes in land use led to serious degradation of surface waters in the second half of the 20th century. By the end of the 1970s, symptoms such as temporal anoxia and organism death were common features in the upstream part of the Schelde. Nutrient concentrations and organic loads were among the highest recorded in Europe (Colijn et al. 2002, Philippart et al. 2007, Cox 2008). The implementation and gradual upgrade of sewage systems, combined with more stringent restrictions for industrial wastewater disposal, resulted in a gradual improvement of the surface water quality in general (Maeckelberge 2002), the improvement of water quality (Soetaert and Herman 1995a), suspended matter and sediments (Baeyens 1998) and the reappearance of higher organisms in the low-salinity region of the Schelde estuary (Appeltans et al. 2003).

Harmful algal blooms regularly occurred along the Belgian coastal zone during the last 50 years (Lancelot et al. 2007). However, Cox (2008) presents evidence for structural changes in the Schelde food web during the last decade: in this short period the freshwater tidal reaches changed from a nearly anoxic system dominated by heterotrophic processes to a system with recurring over-saturated day-time oxygen concentrations, frequently dominated by autotrophic processes. The recently observed diatom dominated algal blooms are accompanied by severe DSi depletion, reducing the flux of DSi to the downstream reaches (Cox et al. 2008).

1.6.2 The Sigma-plan

The Sigma-plan originated after the disastrous floods in Flanders on February, 3rd 1976. It was originally conceptualized as an overall security plan, providing flood protection for all people living in the urbanized Schelde floodplain. In December 2004, the Flemish government decided that for the new Sigma-plan, an optimal combination of three main goals - accessibility, safety and ecology – had to be obtained. Based on the optimized original safety plan on the one hand and a list with potentially valuable restoration projects on the other, 3 alternative integrated versions of the Sigma-plan were proposed. Each of these three alternatives combined the three goals, but differed in location of the sites, expected quality of the projects and the matrix of land use forms. These alternatives were checked independently on safety, ecology, and agricultural impact, including scientists and experts as well as stakeholder organizations. One final alternative was compiled (Couderé et al. 2005). On July, 22nd 2005, the Flemish government implemented this updated Sigma-plan. Main objectives are now the realization of safety and in the same time meet the European conservation goals for the Schelde estuary (Adriaensen et al. 2005).

Along the Schelde, flood controll areas are constructed with a lowered dike stretch at storm tide level. During storm surges, these areas stock the water mass. The whole plan, when completed, should protect all inhabitants against storms which theoretically recur every 10.000 years.

Concerning (re)creation of tidal marshes, the European conservation goals for the Schelde are based on a functional rather than a habitat- or species approach. Silica cycling, nitrogen removal and aeration are key processes, while sedimentation, fish migration, vegetation development and heavy metal behavior are crucial points to be taken into account. In total, some 2250ha of tidal marsh has to be created by 2030 (Jacobs et al. 2008). About 70% of this surface will be created with managed realignment; the other 30% will be realized through implementation of a controlled reduced tide (CRT) on lowered sites.

1.6.3 Controlled reduced tide

CRT – controlled reduced tide is a novel technique to restore tidal marshes. The conceptual idea for a CRT first appeared in the eighties. Several decades later, the decision was taken to develop a pilot area along the tidal Schelde, which is functioning since March 2006 and being closely monitored since then. Hypotheses concerning CRT functioning in Cox et al. (2006) and Maris et al. (2007) are being verified and guidelines for further implementation are constructed.

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 (Photo 1.1 and 1.2). Consequently, water flows out to a lower exit gravitational culvert when water levels in the river have again decreased (Fig. 1.3). 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.

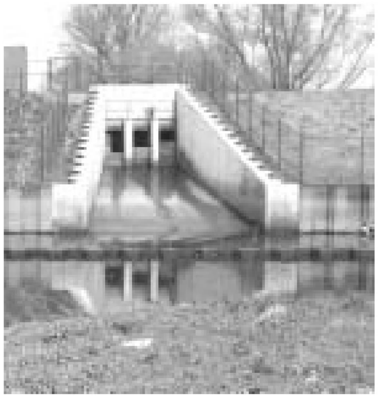

Photo 1.1 (left): Inlet culvert construction (as seen from within the pilot restoration site), consisting of three inlet culverts, each 1 m wide, with adjustable thresholds at different heights. Photo 1.2 (right): Close-up of the inlet culverts at spring tide flooding (as seen from within the pilot restoration site).

However, some critical conditions and potential disadvantages to the CRT technique must be noted:

Within-site topography sets the conditions for culvert dimensions and configurations.

If there is only a small elevation gradient present within the site, the flooding regime can be implemented with a limited number of culverts, since the volume of water needed to flood the entire surface is relatively small (Fig. 1.1A). Culvert thresholds at mean high water neap level could provide the desired gradient: at neap high water, a minimal amount of water flows in, while at spring high water, water levels exceed the threshold for a much longer period of time, so much more water will enter. It is then the width of the culverts which is decisive for the flooded surface.

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. 1.1B). 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. 1.2). Therefore, calculation of surface/volume relationships of the site and discharge estimates of the culverts is needed.

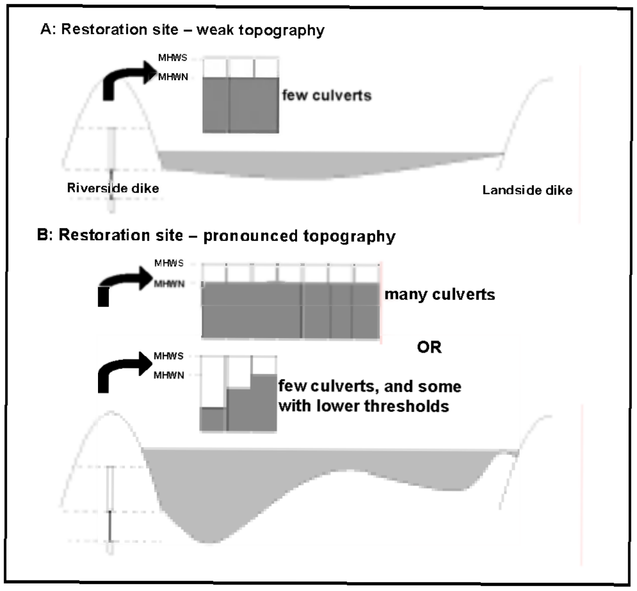

Fig. 1.1: Within site topography is decisive for the width and the threshold configuration of the inlet culverts. Sites with weak topography need less inlet culverts, while pronounced topography requires more volume to enter and thus more culverts or lower thresholds. (MHWS mean high water spring; MHWN mean high water neap)

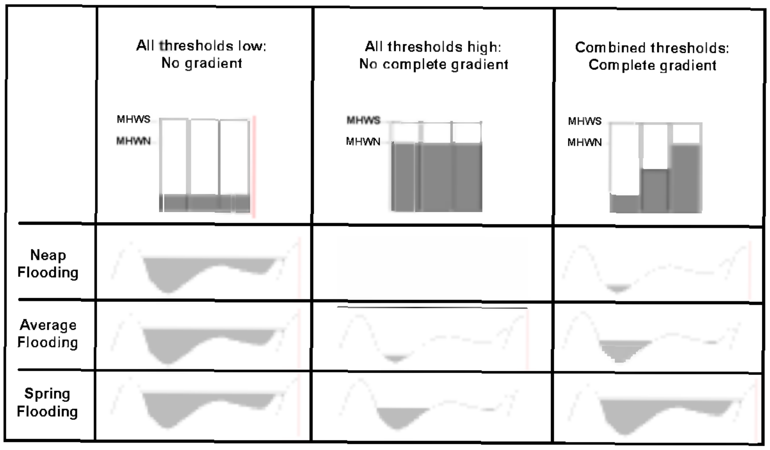

Fig. 1.2: Threshold configuration has to be fine tuned for optimal tidal flooding gradient.

The CRT's tidal envelope differs from the estuarine marshes.

After the water has entered through high inlet culverts, a stagnant high water phase develops, before it evacuates through the low outlet culverts (Fig.1.3). The length of this phase is dependent on the elevation difference of the site with the mean high water level. During 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 estuary is lower then the water level within the site (Fig. 1.3).

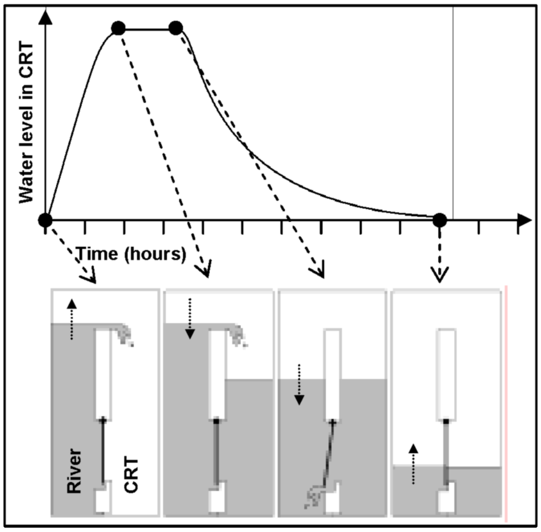

Fig. 1.3: typical CRT tidal envelope. Inflow phase when estuarine levels exceed inlet thresholds, stagnant phase inside CRT until water levels equal, and outstream as long as water pressure opens the outlet culvert.

Prolonged inundation times could have consequences on sediment biogeochemistry, plant and benthos colonization and nutrient exchanges, while hampered seepage could lower the exchange capacity with the estuary.

The prolonged inundation time remains, but the seepage can be enhanced by providing outlet culverts which are big enough to evacuate all the water within the given time frame.

The CRT's sedimentation process differs from the estuarine marshes.

When estuarine 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 estuarine 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 estuary. Fast accretion might be problematic if sedimentation rates during this catching up are too high for structures and processes to develop.

This could be countered 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 (typical 'merlon-crenel' topography, Fig. 1.4), which tidal marshes naturally develop when they accrete. This is structure is expected to develop in the CRT.

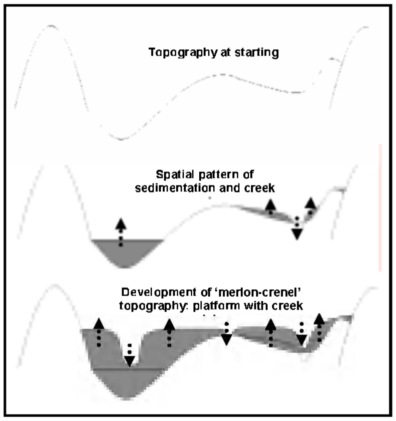

Fig. 1.4: Spatial sedimentation patterns as expected in the CRT.

Water currents inside CRT differ from the estuarine marshes.

On estuarine 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 are observed, this could again promote too fast sedimentation rates for typical marsh structures and processes to develop (see above). The lack of currents could also prevent development of natural drainage structures, which is of utmost importance for development of reference marsh topography as well as for seepage water exchange (see Chapter 5). 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 (see section Sigma plan), 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.

CRT excludes other societal functions of the site.

Implementation of controlled reduced tide severely impacts the given sites. Most former functions (inhabitation, agriculture, tree plantings, fishing ponds, camping, etc.) can no longer be exerted in any way.

However, other functions can be combined with CRT. All planned CRT's along the Schelde are designated as priority storm flood areas, where storm flood water volume is preferentially dissipated (see section Sigma plan). If the whole storage volume has to be preserved in the long run, accretion to mean spring flooding level must be prevented as described above.

The recreational value can be increased by providing access with boardwalks and by providing viewpoints for the instream culverts and bird watching (Fig. 1.5). 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 the total coastal defense scheme. This increases the public acceptability and political willingness to implement whole coastal defense schemes (French 2006, Weinstein 2007).

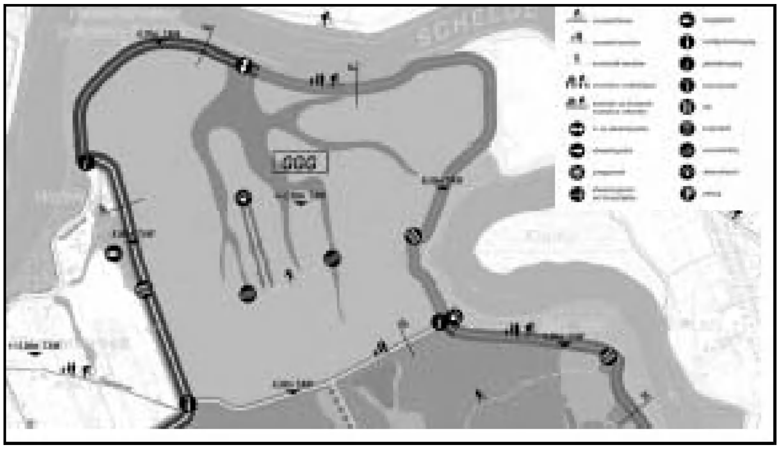

Fig. 1.5: recreational functions to be developed in a planned CRT project along the Schelde (source W&Z – ANB)

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 (Photo 1.3).

Photo 1.3: Aerial view of 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.

1.7 This thesis

This thesis focuses on vegetation development and silica cycling in a former agricultural area where a controlled reduced tide was implemented to restore freshwater tidal marsh habitat. It aims to contribute to the restoration practice of tidal habitats and to evaluate the potential of the CRT technique, with an emphasis on silica processing. This is why the presented manuscripts are mostly combinations of both applied and theoretical aspects. The results are presented in six chapters:

The first chapter just introduced you to this thesis.

The second chapter describes the vegetation of tidal freshwater marshes in northwest Europe. Available research on the vegetation and ecology of European tidal freshwater habitats is scarce. This chapter gives a descriptive review of available data on freshwater tidal marsh vegetation, its threats and their consequences.

These data are used in the third chapter to evaluate the habitat development in the pilot area where controlled reduced tide (CRT) was implemented at the beginning of this research. This is the first research which compares vegetation development in the pilot site to reference habitats

The fourth chapter makes the connection to the silica cycle. It focuses on the role of vegetation in silica cycling in the developing CRT. The magnitudes of the fluxes of Si in vegetations are still not well established because published studies on biomass including Si data are scarce. We aimed to make an estimate of this silica pool and discuss its relevance for estuarine silica cycling.

The fifth chapter compares the silica contents in estuarine sediment - which is considered the main pool silica cycling in tidal marshes- between estuarine sites and the CRT pilot marsh.

The sixth chapter presents a direct evaluation of functioning of the CRT pilot marsh with regard to silica buffering of Si-depletion events in the estuary, and again compares these results to earlier results in estuarine marshes.

The seventh chapter sketches restoration goals for structures and processes of freshwater tidal marshes, based on chapter 1. The findings of previous chapters in this restoration framework are summarized.
This chapter however clearly points out the high potential of the CRT technique for successful restoration of structures and processes of reference freshwater tidal marshes.
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 succes.

Enjoy…

Adam, P. Saltmarshes in a time of change, *Environmental Conservation* (29), 2002, pp. 39-61.

Admiraal, W., Breugem, P., Jacobs, D. and De Ruyter van Steveninck, E. Fixation of dissolved silicate in the river Rhine during diatom blooms, *Biogeochemistry* (9), 1990, pp. 175-185.

Adriaensen, F., Van Damme, S., Van den Bergh, E., Brys, R., Cox, T., Jacobs, S., Konings, P., Maes, J., Maris, T., Mertens, W., Nachtergale, L., Struyf, E., Van Braeckel, A., Van Hove, D. and Meire, P. Instandhoudingsdoelstellingen Schelde-estuarium(Rapport ECOBE 05R.82), Technical report, Universiteit Antwerpen, Antwerpen, 2005.

Alexandre, A., Meunier, J., Colin, F. and Koud, J. Plant impact on the biogeochemical cycle of silicon and reversed weathering processes, *Geochimica and Cosmochimica Acta* (61), 1997, pp. 677-682.

Allen, J. and Pye, K., (eds.) *Salt-marshes: morphodynamics, conservation and engineering significance*, Cambridge: Cambridge University Press, 1992.

Appeltans, W., Hannouti, A., Van Damme, S., Soetaert, K., Vanthomee, R., Tackx, M. and Meire, P. Zooplankton in the Schelde estuary (Belgium/The Netherlands). The distribution of Eurytemora affinis: effect of oxygen?, *Journal of Plankton Research* (25), 2003, pp. 1441-1445.

Baeyens, W. Evolution of trace metal concentrations in the Scheldt estuary, *Hydrobiologia* (366), 1998, pp. 157-167.

Berner, R., Lasaga, A. and Garrels, R. The carbonate-silicate cycle and its effect on atmospheric carbon dioxide over the past 100 million years, *American Journal of Science* (283), 1983, pp. 641-683.

Berthelsen, S., Noble, A. and Garside, A. Silicon research down under: Past, present and future, *in* Datnoff, L., G.H., S. and G.H., K., ed.,Silicon in agriculture, Amsterdam: Elsevier, 2001, pp. 241-256.

Billen, G. and Garnier, J. River basin nutrient delivery to the coastal sea: Assessing its potential to sustain new production of non-siliceous algae, *Marine Chemistry* (106), 2007, pp. 148–160.

Brzezinski, M. The Si:C:N ratio of marine diatoms: interspecific variability and the effect of some environmental variables, *Journal of phycology* (21), 1985, pp. 347-357.

Claessens, J. and Meyvis, L. Overview of the tidal observations in the Zeeschelde basin during the decade 1981-1990 [Overzicht van de tijwaarnemingen in het Zeescheldebekken gedurende het decennium 1981-1990], Technical report, Antwerpse Zeehavendienst: Antwerpen, Belgium, 1994.

Cloern, J. Our evolving conceptual model of the coastal eutrophication problem, *Marine Ecology Progress Series* (210), 2001, pp. 223-253.

Colijn, F., Hesse, K., Ladwig, N. and Tillman, U. Effects of the large-scale uncontrolled fertilisation process alongthe coastal North Sea, *Hydrobiologia* (484), 2002, pp. 133-148.

Conley, D. and Kilham, S. Differences in silica content between marine and freshwater diatoms, *Limnology and Oceanography* (34), 1989, pp. 205-213.

Conley, D. Riverine contribution of biogenic silica to the oceanic silica budget, *Limnology and Oceanography* (42), 1997, pp. 774-777.

Conley, D. Terrestrial ecosystems and the global biogeochemical silica cycle, *Global Biogeoch. Cycl* (16), 2002, pp. 1121-1129.

Conley, D., Likens, G., Buso, D., Saccone, L., Bailey, S. and Johnson, C. Deforestation causes increased dissolved silicate losses in the Hubbard Brook Experimental Forest, *Global Change Biology* (14), 2008, pp. 2548–2554.

Conley, D., Schelske, C. and Stoermer, E. modification of silica biogeochemistry with eurtrophication in aquatic systems, *Marine Ecology Progress Series* (101), 1993, pp. 179-192.

Conley, D., Sommer, M., Meunier, J., Kaczorek, D. and Saccone, L. Silicon in the terrestrial biogeosphere, *in* Ittekot, V., Unger, D., Humborg, C. and Tac An, N., ed.,The Silicon Cycle. Scope series 66, Islandpress Washington, 2006, pp. 13-28.

Costanza, R., dArge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., ONeill, R., Paruelo, J., Raskin, R., Sutton, P. and vandenBelt, M. The value of the worlds ecosystem services and natural capital, *Nature* (387), 1997, pp. 253-260.

Costanza, R., Kemp, W. and Boynton, W. Predictability, scale and diversity in coastal and estuarine ecosystems: implications for management:, *Ambio* (22), 1993, pp. 88-96.

Couderé, K.and Vincke, J., Nachtergaele, L., Van Den Bergh, E., Dauwe, W., Bulckaen, D. and Gauderis, J. Geactualiseerd Sigmaplan voor veiligheid en natuurlijkheid in het bekken van de Zeeschelde. Documentnummer 5145-507-042., Technical report, NV Waterwegen en Zeekanaal, 2005.

Cox, T. Oxygen and planktonic production in the freshwater tidal reaches of the Schelde estuary, 2008.

Cox, T., Maris, T., Vleeschauwer, P. D., Mulder, T. D., Soetaert, K. and Meire, P. Flood control areas as an opportunity to restore estuarine habitat, *Ecological Engineering* (28:1), 2006, pp. 55-63.

Cugier, P., Billen, G., Guillau, J., Garnier, J. and Ménesguen, A. Modelling the eutrophication of the Seine Bight (France) under historical, present and future riverine nutrient loading, *Journal of Hydrology* (304), 2005, pp. 381–396.

Cundy, A., Long, A., Hill, C., Spencer, C. and Croudace, I. Sedimentary response of Pagham Harbour, southern England to barrier breaching in AD 1910, *Geomorphology* (1156), 2002.

Daily, G., Alexander, S., Ehrlich, P., Goulder, L., Lubcheno, J., Matson, P., Mooney, H., Postel, S., Schneider, S., Tilman, D. and Woodwell, G. Ecosystem servives: benefits supplied to human society by natural ecosystems, *Ecological Society of America, Issues in Ecology* (2), 1997.

Datnoff, L., Snyder, G. and G.H., K., (eds.) *Silicon in agriculture. Studies in Plant Science 8*, Amsterdam: Elsevier, 2001.

Davidson, N., dA Laffoley, D., Doody, J., Way, L., Gordon, J., Key, R., Drake, M., Pienkowski, M., Mitchell, R. and Duff, K. Nature

Conservation and Estuaries of Great Brittain, Technical report, Nature Conservancy Council, Peterborough, 1991.

Day, J., Hall, A., Kemp, W. and Yanez-Arancibia, A. *Estuarine Ecology*, Wiley New York, 1989.

De Groot, R. Valuing Natural Ecosystems: from local services to global capital, *Position Paper for the European Symposium of Environmental Valuation, Vaux de Cernay, France* (), 1997.

Egge, J. and Asknes, D. Silicate as a regulating nutrient in phytoplankton competition, *Marine Ecology Progress Series* (38), 1992, pp. 281-290.

Elliot, M. and McLusky, D. The need for definitions in understandig estuaries, *Estuarine, Coastal and Shelf Science* (55), 2002, pp. 815-827.

Epstein, E. Silicon, *Annual Review of Plant Physiology and Plant Molecular Biology* (50), 1999, pp. 641-664.

Facca, C., Sfriso, A. and Socal, G. Temporal and spatial distribution of diatoms in the surface sediments of the Venice Lagoon, *Botanica Marina* (452), 2002, pp. 170-183.

French, P. W. Managed realignment - The developing story of a comparatively new approach to soft engineering, *Estuarine, Coastal and Shelf Science* (67:3), 2006, pp. 409-423.

Gili, J. and Coma, R. Benthic suspension feeders: their paramount role in littoral marine food webs, *Trends in Ecology and Evolution* (13), 1998, pp. 316-321.

Graf, G., Bengtsson, W., Diesner, U., Schulz, R. and Theede, H. Benthic response to sedimentation of a spring phytoplankton bloom: Process and budget. , *Marine Biology* (67), 1982, pp. 201-208.

Hughes, R. and Paramor, O. On the causes of the decline of saltmarshes in SE England and appropriate methods for reducing erosion and creating new marshes, *Journal of Applied Ecology* (41), 2004, pp. 440–448.

Irigoien, X., Harris, R., Verheye, H., Joly, P., Runge, J., Starr, M., Pond, D., Campbell, R., Shreeve, R., Ward, P., Smith, A., Dam, H., Peterson, W., Tirelli, V., Koski, M., Smith, T., Harbour, D. and Davidson, R. Copepod hatching success in marine ecosystems with high diatom concentrations, *Nature* (419), 2002, pp. 387-389.

Jacobs, S., Maris, T., Teuchies, J., Vandenbruwaene, W., Temmerman, S. and Meire, P. Eindrapport betreffende de functionele aspecten voor inrichting van GOG-GGG en Ontpoldering in het kader van het SIGMAPLAN, Zone 1 & 2., Technical report, University of Antwerp, Ecosystem Management Research Group, ECOBE 08-R115, 2008.

Kilham, P. A hypothesis concerning silica and the freshwater planktonic diatoms, *Limnology and Oceanography* (16), 1971, pp. 10-18.

Kimmerer, W. Long-term changes in apparent uptake of silica in the San Francisco estuary, *Limnology and Oceanography* (50), 2005, pp. 793–798.

Lancelot, C., Gypens, N., Billen, G., Garnier, J. and Roubeix, V. Testing an integrated river-ocean mathematical tool for linking marine-eutrophication to land use: the Phaeocystis-dominated Belgian coastal zone (southern North Sea) over the past 50 years, *Journal of Marine Systems* (64), 2007, pp. 216-228.

Likens, G., Bormann, F., Johnson, N., Fisher, D. and Pierce, R. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem, *Ecological Monographs* (40), 1970, pp. 23-47.

Lucas, Y., Luizao, F., Chauvel, A., Rouiller, J. and Nahon, D. The relation between biological activity of rainforests and mineral composition of soils, *Science* (260), 1993, pp. 521-523.

Maeckelberghe, H. De kwaliteit van het Vlaamse oppervlakteweter en afval-waterlozingen in 2002. Een toelichting bij de resultaten van de meetnetten van de Vlaamse Milieumaatschappij, Technical report, VMM, 2002.

Maris, T., Cox, T., Temmerman, S., De Vleeschauwer, P., Van Damme, S., De Mulder, T., Van den Bergh, E. and Meire, P. Tuning the tide: creating ecological conditions for tidal marsh development in a flood control area, *Hydrobiologia* (588), 2007, pp. 31-34.

McIntyre, D., Geider, R. and Miller, D. Microphytobenthos: the ecological role of the secret garden of unvegetated, shallow-water marine habitats, 1. Distribution, abundance and primary production, *Estuaries* (19), 1996, pp. 186-201.

McLusky, D. and Elliot, M. *The Estuarine Ecosystem: Ecology, Threats and Management*, Oxford University Press, Oxford, 2004.

McLusky, D. *The Estuarine Ecosystem*, Blackie & son Glasgow and London; Chapman & Hall New York, 1989.

Meire, P., Herman, P. and Santbergen, L. Ecologische structuren binnen het Schelde-stroomgebied: een essentiële voorwaarde voor ecologisch herstel en de veerkracht van het systeem, *Water* (102), 1998, pp. 315-322.

Meire, P., Ysebaert, T., Van Damme, S., Van den Bergh, E., Maris, T. and Struyf, E. The Scheldt estuary: a description of a changing ecosystem, *Hydrobiologia* (540), 2005, pp. 1–11.

Morris, R., Reach, I., Duffy, M., Collins, T. and Leafe, R. On the loss of salt marshes in south-east England and the relationship with Nereis diversicolor, *Journal of Applied Ecology* (41), 2004, pp. 787–791.

Norris, A. and Hackney, C. Silica content of a mesohaline tidal marsh in North Carolina, *Estuarine, Coastal and Shelf Science* (49), 1999, pp. 597-605.

Officer, C. and Ryther, J. The possible importance of silicon in marine eutrophication, *Marine Ecology Progress Series* (3), 1980, pp. 83-91.

Perry, J., Barnard, T., Bradshaw, J., Friedrichs, C., Havens, K., Mason, P., Priest, W. and Silberhorn, G. Creating tidal salt marshes in the Chesapeake Bay, *Journal of Coastal Research* (27), 2001, pp. 170–191.

Philippart, C., Beukema, J., Cadee, G., Dekker, R., Goedhart, P., Van Iperen, J., Leopold, M. and Herman, P. Impacts of nutrient reduction on coastal communities, *Ecosystems* (10), 2007, pp. 95-118.

Quammen, M. Measuring the success of wetland mitigation, *National Wetlands Newsletter* (), Sept–Oct 1986, pp. 6–8.

Ragueneau, O., Conley, D., Leynaert, A., Ni Longphuirts, S. and Slomp, C. Role of diatoms in Silicon Cycling and Coastal Marine Foodwebs, *in* Ittekot, V., Unger, D., Humborg, C. and Tac Ann, N., ed.,the silicon cycle. Scope series 66, Islandpress Washington, 2006.

Ragueneau, O., Lancelot, C., Egorov, V., Vervlimmeren, J., Cociasu, A., Déliat, G., Krastev, A., Daoud, N., Rousseau, V., Popovitchev, V., Brion, N., Popa, L. and Cauwet, G. Biogeochemical transformations of inorganic nutrients in the mixing zone between the Danube River and the northwestern Black Sea, *Estuarine, Coastal and Shelf Science* (54), 2002b, pp. 321-336.

Ragueneau, O., Tréguer, A., Leynaert, A., Anderson, R., Brzezinski, M., DeMaster, D., Rugdale, R., Dymond, J., Fisher, G., François, R., Heinze, C., Maier-Reimer, E., Martin-Jézéquel, J., Nelson, D. and Quéguiner, B. A review of the Si cycle in the modern ocean: Recent progress and missing gaps in the application of biogenic opal as a paleoproductivity proxy, *Global and Planetary Change* (26), 2002, pp. 315-366.

Raven, J. The transport and function of silicon in plants, *Biological Review* (58), 1983, pp. 179-207.

Regnier, P., Wollast, R. and Steefel, C. Long-term flux of reactive species in macrotidal estuaries: Estimates from a fully transient, multicomponent reaction-transport model, *Marine Chemistry* (58), 1997, pp. 127-145.

Rizzo, W. Nutrient exchanges between the water column and a subtidal benthic microalgal community, *Estuaries* (13), 1990, pp. 219-226.

Schelske, C. L. and Stoermer, E. Eutrophication, silica depletion and predicted changes in algal quality in Lake Michigan, *Science* (173), 1971, pp. 423-424.

Schuchard, B., Haesloop, U. and Schirmer, M. The tidal freshwater reach of the Weser estuary: riverine or estuarine?, *Aquatic Ecology* (27), 1993, pp. 215-226.

Sicko-Goad, L., Schelske, C. and Stoermer, E. Estimation of intracellular carbon and silica content of diatoms from natural assemblages using morphometric techniques, *Limnology and Oceanography* (29), 1984, pp. 1170-1178.

Smayda, T. Novel and nuisance of phytoplankton blooms in the sea: Evidence for a global epidemia, *in* Graneli, E., ed.,Toxic Marine Phytoplankton, New York: Elsevier, 1990, pp. 29-40.

Smith, V., Joye, S. and Howarth, R. Eutrophication of freshwater and marine ecosystems, *Limnology and Oceanography* (51), 2006, pp. 351-355.

Soetaert, K. and Herman, P. Carbon flows in the Westerschelde estuary (The Netherlands) evaluated by means of an ecosystem model (MOSES), *Hydrobiologia* (311), 1995a, pp. 215-224.

Soetaert, K. and Herman, P. Estimating estuarine residence times in the Westerschelde (The Netherlands) using a box model with fixes dispersion coefficients, *Hydrobiologia* (311), 1995b, pp. 225-246.

Soetaert, K., Middelburg, J., Heip, C., Meire, P., Van Damme, S. and Maris, T. Long-term change in dissolved inorganic nutrients in the heterotrophic Scheldt estuary (Belgium, The Netherlands), *Limnology and Oceanography* (51), 2006, pp. 409-423.

Sommer, M., Kaczorek, D., Kuzyakov, Y. and Breuer, J. Silicon pools and fluxes in soils and landscapes: A review, *Journal of Plant Nutrition and Soil Science* (169), 2006, pp. 294-314.

Struyf, E. The role of freshwater marshes in estuarine silica cycling (Scheldt estuary), 2005b.

Struyf, E., Damme, S. V., Gribsholt, B., Middelburg, J. and Meire, P. Biogenic silica in tidal freshwater marsh sediments and vegetation (Schelde estuary, Belgium), *Marine Ecology Progress Series* (303), 2005, pp. 51–60.

Struyf, E., Dausse, A., Van Damme, S., Bal, K., Gribsholt, B., Boschker, H., Middelburg, J. and Meire, P. Tidal marshes and biogenic silica recycling at the land-sea interface, *Limnology and Oceanography* (51), 2006, pp. 838–846.

Struyf, E., Temmerman, S. and Meire, P. Dynamics of biogenic Si in freshwater tidal marshes, Si regeneration and retention in marsh sediments (Scheldt estuary), *Biogeochemistry* (82), 2007, pp. 41–53.

Struyf, E., Van Damme, S., Gribsholt, B., Bal, K., Beauchard, O., Middelburg, J. and Meire, P. Phragmites australis and silica cycling in tidal wetlands, *Aquatic Botany* (87), 2007b, pp. 134-140.

Subramanian, V., Ittekot, V., Unger, D. and Madhavan, N. Silicate Weathering in South Asian Tropical River Basins, *in* Ittekot, V., Unger, D., Humborg, C. and Tac An, N., ed.,The Silicon Cycle. Scope Series 66, Islandpress Washington, 2006.

Sundback, K. and Graneli, W. Influence of microphytobenthos on the nutrient flux between sediment and water: a laboratory study., *Mar Ecol Prog Ser* (43), 1988, pp. 63-69.

Sundbäck, K., Enoksson, V., Granéli, W. and Pettersson, K. fluence of sublittoral microphytobenthos on the oxygen and nutrient flux between sediment and water: a laboratory continuous-flow study, *Marine Ecology Progress Series* (74), 1991, pp. 263–279.

Taverniers, E. Zeescheldebekken: afvoer van de Schelde in 1999., Technical report, Ministerie van de Vlaamse Gemeenschap AWZ Afdeling Maritieme Schelde Antwerpen, 2000.

Temmerman, S., Govers, G., Meire, P. and Wartel, S. Modelling long-term tidal marsh growth under changing tidal conditions and suspended sediment concentrations, Scheldt estuary, Belgium, *Marine Geology* (193), 2003, pp. 151-169.

Tréguer, P., Nelson, D. M., van Bennekorn, A. J., DeMaster, D. J., Leynaert, A. and Quéguiner, B. The silica balance in the world ocean: a reestimate., *Science* (268), 1995, pp. 375-379.

Turner, R., Qureshi, N., Rabalais, N., Dortch, Q., Justic, D., Shaw, R. and Cope, J. Fluctuating silicate:nitrate ratios and the coastal plankton food webs, *Proceedings of the National Academy of Science, USA* (95), 1998, pp. 13048-13051.

Van Damme, S., Struyf, E., Maris, T., Ysebaert, T., Dehairs, F., Tackx, M., Heip, C. and Meire, P. Spatial and temporal patterns of water quality along the estuarine salinity gradient of the Scheldt estuary (Belgium and The Netherlands): results of an integrated monitoring approach, *Hydrobiology* (540), 2005, pp. 29–45.

van der Zee, C., Roevros, N. and Chou, L. Phosphorus speciation, transformation and retention in the Scheldt estuary (Belgium/The Netherlands) from the freshwater tidal limits to the North Sea, *Marine Chemistry* (106), 2007, pp. 76-91.

Van Dokkum, H., Hulskotte, J., Kramer, K. and Wilmot, J. Emission, fate and effects of soluble silicates (waterglass) in the aquatic environment, *Environmental Science and Technology* (38), 2004, pp. 515-521.

Van Eck, G., De Pauw, N., Van Langenbergh, M. and Verreet, G. Emissies, gehalten, gedrag en effecten van (micro)verontreinigingen in het stroomgebied van de Schelde en het Schelde-estuarium, *Water* (60), 1991, pp. 84-99.

Vanderborght, J.-P., Folmer, I. M., Aguilera, D. R., Uhrenholdt, T. and Regnier, P. Reactive-transport modelling of C, N, and O2 in a river-estuarine-coastal zone system: Application to the Scheldt estuary,, *Marine Chemistry* (106), 2007, pp. 92-110.

Wedepohl, K. The composition of the continental crust, *Geochimica et Cosmochimica Acta* (59), 1995, pp. 1217-1232.

Weinstein, M. and Kreeger, D., (eds.) *Concepts and Controversies in Tidal Marsh Ecology*, New York: Kluwer Press, 2000.

Weinstein, M. Ecological restoration and estuarine management: plaving people in a coastal landscape, *Journal of Applied Ecology* (4:2), 2007, pp. 1-9.

Wollast, R. and De Broeu, F. Study of the behaviour od dissolved silica in the estuary of the Scheldt, *Geochimica and Cosmochimica Acta* (35), 1971, pp. 613-620.

Wollast, R. and Mackenzie, F. The global cycle of silica, *in* Aston, S., ed.,Silicon geochemistry and biogeochemistry, London Academic Press, 1983, pp. 39-76.

Wolters, M., Garbutt, A., Bekker, R. M., Bakker, J. P. and Carey, P. D. Restoration of salt-marsh vegetation in relation to site suitability, species pool and dispersal traits, *Journal of Applied Ecology* (45), 2008, pp. 904-912.

Wulff, A. S. K., Nilsson, C., Carlson, L. and Johnsson, B. Effect of sediment load on the microphytobenthic community of a shallow-water sandy sediment, *Estuaries* (203), 1997, pp. 547-558.

Yool, A. and Tyrrell, T. Role of diatoms in regulating the oceans silicon cyclev, *Global Biogeochemical Cycles* (17), 2003, pp. 1103.

Chapter 2
Plant communities in European tidal freshwater wetlands

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2.1 Abstract

Tidal freshwater marshes have become scarce in Europe because of drastic human alterations of estuarine geomorphology. The main tidal freshwater marshes in Europe are found nowadays along the Schelde, Elbe and Rhine-Meuse estuaries. The Schelde and Elbe are characterized by high tidal amplitudes, while the Rhine-Meuse has little tidal amplitude due to hydrologic alterations over recent decades. The major vegetation types in tidal freshwater marshes in these three estuarine systems are separately described and compared. Vegetation zonation is highly dependent on flooding frequency. *Phragmites australis*, *Salix* spp., *Bolboschoenus maritimus* and *Schoenoplectus* spp. are the most frequent species of European tidal freshwater marshes. Two endemic species have been described from the Elbe, and one characteristic subspecies for the Schelde, Rhine-Meuse and Elbe estuaries. *Impatiens glandulifera* is the most widespread invasive species. In the past, vegetation in tidal freshwater marshes was commercially managed and exploited but today these practices have almost completely disappeared.

2.2 Introduction

Tidal freshwater marshes have decreased greatly in abundance and extent in Europe, primarily due to human alteration of estuarine habitats. Construction of embankments and dikes, harbour expansion, and conversion of tidal freshwater marsh habitats into agricultural uses and urban development are the primary factors that have claimed a substantial portion of the original tidal freshwater marshes. For example, between 1900 and 1990, the total area of marshes and mudflats along the Zeeschelde (the Belgian part of the Schelde estuary that includes fresh water and brackish habitats) was reduced from 2192 ha to 1411 ha (-36%) (Meire et al. 2005). Still, substantial areas with tidal freshwater marshes occur in the Schelde (Belgium), the Weser and the Elbe (Germany, 5,000 ha) estuaries. The portion of the Rhine-Meuse estuary in The Netherlands that contained tidal freshwater marshes (Biesbosch, 8,000 ha) began to change following the almost complete elimination of tidal amplitude after the construction of huge storm surge barriers. Today only about 600 ha of tidal freshwater marshes with tidal amplitude has persisted along the river Oude Maas and 200 ha in scattered locations in the northern half of the Rhine-Meuse estuary. Tidal freshwater marshes persist in other western European estuaries (e.g., Gironde, Loire, Seine and across the UK and Ireland) but we could not locate any descriptions of the vegetation from those areas.

Completely natural tidal freshwater marshes no longer occur in Europe due to a large number of factors that result in alteration of hydrologic conditions as well as other factors that have negative impacts on wetlands. Conversion of wetland habitats into other uses, alteration of wetland hydrology and elimination of tidal regimes have resulted from the construction of embankments and levee fortifications (Preisinger et al. 1991). Waves from ships and other anthropogenic influences also impact wetland habitats in tidal fresh water portions of Western European estuaries. Table 2.1 summarizes wetland information for the Elbe, Weser, Biesbosch, Oude Maas, and Schelde estuaries.

Table 2.1: Area (in ha) of the main types of vegetation in freshwater marshes along Elbe, Weser, Oude Maas, Biesbosch and Schelde. Biesbosch marshes are relict with small tidal amplitude; only herb and willow-dominated marshes are distinguished for Biesbosch, Elbe and Weser. The area for the Weser might be too high as brackish marshes and tidal freshwater marshes have not been differentiated. Sources: Schröder (2007), Steege (2007), Strucker (1996), Saris (1987), Van Allemeersch et al. (1999).

Estuary	Tidal amplitude	*Schoeno-plectus* or *Bolbo-schoenus* dominant	Only *Phragmites* dominant	*Phragmites* -tall herbs co-domination	Tall herbs dominant	*Salix* shrubs	*Salix* woods	*Salix* coppice / plantations
Elbe	2.4 – 3.6	1222				448		
Weser	3.8 – 4.1	1088				214		
Oude Maas	1.1 m	18	28	112		19	99	134
Biesbosch	0.2 – 0.6 m	-	1920			1530		
Schelde	2-6 m	?	23	49	218	191	75	0

In this chapter we focus on the main plant communities occurring in tidal freshwater marshes along the Schelde estuary (Belgium), along the Rhine-Meuse estuary (The Netherlands) and along the Elbe estuary (Germany). In all of the tidal freshwater marshes described, helophytes, such as reed and bulrush, are the dominant plant life-form (Ellenberg 1963, Coops et al. 1999). Several factors influence the characteristics of vegetation in tidal freshwater marshes. The most important physical factors are depth and duration of inundation, wave exposure and sediment type. We also discuss temporal changes in tidal freshwater marshes as a function of the amount of sedimentation and the variations in surface elevation of the wetland. Elevation is important as it governs flooding frequency, depth and duration, which are in turn tightly linked to soil development (Zonneveld 1960).

Compared to North America, the available research on the vegetation and ecology of European tidal freshwater marshes is scarce. For this reason, a large part of this overview is based on literature that is not readily available to an international audience. We have compiled information from national reports, PhD theses, and national books that have focused specifically on tidal freshwater marshes or have considered them as part of a larger study topic.

2.3 Vegetation pattern of the Schelde, Rhine-Meuse, and Elbe

A general vegetation pattern is consistent in European tidal freshwater marshes and the pattern is almost completely governed by plant responses to differences in elevation and flooding characteristics (Zonneveld 1960). At the same time, accretion of sediment creates conditions that result in vegetation change over time that can be referred to as succession. On the lowest tidal flats, which are flooded twice daily, vegetation is limited to non-vascular benthic algae such as species of *Vaucheria*. In low-lying tidal flats areas where the elevation is slightly higher as a result of sediment accumulation, macrophytes such as *Schoenoplectus* spp. (or *Bolboschoenus maritimus*) colonize. Macrophyte growth promotes further sediment accumulation and increased surface elevation. Elevated creek banks are dominated by dense stands of herbaceous vegetation while on the lower marsh macrophytes such as *Typha angustifolia* and *Phragmites australis* are typical dominants. Dead shoots of the two species can remain standing for two years or more. Once shoots collapse (wind, ice or water flow), they add to the abundantly rich and dynamic litter layer, often observed in tidal freshwater marshes. The litter layer plays an important role in the gradual increase in elevation of the tidal wetlands, by capturing sediment imported with flood-water (Rooth et al. 2003).

In the past, many habitats dominated by *Phragmites australis* were commercially harvested. The yearly mowing and removal of reed litter reduced precluded tall herb and Salix settlement, resulting in largely monospecific *P. australis* stands. This practice is still locally applied as a nature conservation measure.

Increasing sedimentation eventually results in a reduction in the frequency of flooding and subsequent replacement of reed-dominated vegetation by other herbs species., e.g., *Heracleum sphondylium* and *Epilobium hirsutum*. In this phase the first *Salix* trees can establish, and the increased accretion and input of litter will stimulate the last stage of tidal freshwater marsh vegetation development (e.g., the last phase of succession), which are forests dominated by species of *Salix* occur (Fig. 2.1).

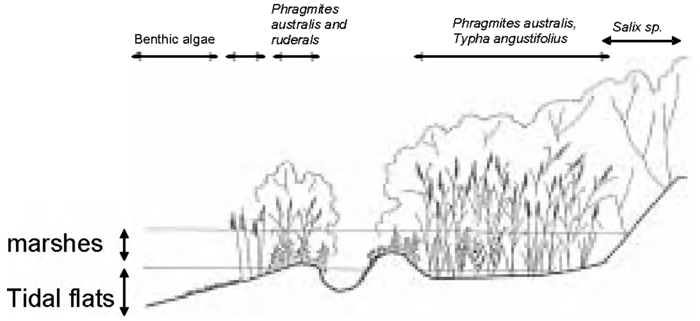

Fig. 2.1: General vegetation zonation in Belgian tidal freshwater marshes. Lower line: ca. 80% inundation, upper line less than 5% inundation frequency. Source: Van Allemeersch et al. (1999).

Invasive species

European tidal freshwater marshes have in recent decades witnessed the large-scale invasion of *Impatiens glandulifera*, or the ornamental jewelweed, which is native to the Southern Himalaya. It has not only invaded tidal freshwater marshes but other moist habitats (Beerling & Perrins 1993) all over Western Europe. A recent study concluded that I. glandulifera exerts only negligible influence on characteristics and species richness of invaded riparian vegetation, and presents no threat to native plant diversity (Hejda & Pysek 2006).

Another invasive species from the same genus is *Impatiens capensis*, originating from eastern North America where it occurs in a wide range of habitats from upland forests to non-tidal freshwater wetlands and tidal freshwater marshes (Chapter 5). This species has been reported from UK, France, Belgium and The Netherlands in estuaries and lower stretches of rivers. In the middle of the 1990s *I. capensis* was observed just east of Rotterdam and during the last ten years, it has become widely distributed in the tidal freshwater marshes in The Netherlands, having been observed in at least ten separate locations (E Weeda pers. comm.). The ecological niche of *I. capensis* is comparable with *I. glandulifera* occuring in reedbeds, dense herb vegetation, and open willow forests.

Other invasive plant species in tidal freshwater marshes were described by Zonneveld (1999) from the Biesbosch region. Zonneveld postulated that after the reduction in tidal amplitude in 1970, the dominant species in tidal freshwater marshes began to change and the species that invaded the sites differed temporally. In the seventies it was *Angelica archangelica*, in the eighties it was *Solidago gigantea*, and in the nineties *I. glandulifera* became dominant. Currently *Heracleum mantegazzianum* is spreading in tidal freshwater marshes in the Biesbosch.

In the next sections, tidal freshwater marsh vegetation characteristics are describe for the main estuaries of northwestern Europe; the Schelde, Rhine-Meuse (including Oude Maas), and Elbe.

2.3.1 Schelde (Belgium)

The Schelde estuary is highly eutrophic (Meire et al. 2005) even though in recent years a reduction of nutrient levels has occurred (Soetaert et al. 2006). Nitrogen levels are especially high and originate primarily from non-point sources, which are most difficult to control (Struyf et al. 2004). In fresh water portions of the estuary that are not dominated by emergent wetlands, productivity is limited by light availability, as turbidity in the fresh water portion of the Schelde estuary is high (Meire et al. 2005). This description of the tidal freshwater marsh vegetation that follows is based on Meire et al. (1992), Criel et al. (1999) and Van Allemeersch et al. (1999).

In the tidal freshwater marshes of the Schelde the low marshes are infrequent. *Schoenoplectus* spp.- and *Bolboschoenus maritimus*-dominated vegetation is scarce in tidal freshwater marshes along the Schelde, probably as a result of channel reconstructions, which removed habitats that both species occur in and introduced pollution that prevents vegetation growth. On older pictures of the fresh water section of the Schelde estuary (Massart 1908), extensive *Schoenoplectus* spp. and *B. maritimus* belts were present. In recent years, bulrush is starting to recover, but its abundance is still far from abundance in the beginning of the 20th century (P Meire pers. obs.). The tidal freshwater marsh habitats along the Schelde that is most widespread are the high marshes, which have surface elevations that are situated near mean high water level. Tidal amplitude along the fresh water portions of the Schelde can be as high as 6 m and tidal amplitude varies monthly and seasonally in response to lunar events and river hydrology. As a result, during neap tides most of the tidal freshwater marsh habitats are not flooded while they are almost all inundated during spring tides. Due to high rates of production and high rates of sediment accumulation, the high marsh habitats have been keeping pace with sea-level rise (Temmerman et al. 2004).

The tidal freshwater marshes along the Schelde are reasonably uniform along the longitudinal gradient of the fresh water section of the estuary. Pioneer vegetation has an open nature and, in contrast to the other vegetation types in Schelde tidal freshwater marshes, is short (< 0.5 m) and uniform (flooding frequency 80-100%). These tidal freshwater marshes are mainly dominated by *Persicaria hydropiper* with *Veronica anagallis-aquatica, Callitriche stagnalis, Apium nodiflorum* and *Rorippa amphibia* as frequently-occurring co-dominants. At the interface between low and high marsh habitats, *Phalaris arundinacea* and *Lythrum salicaria* thrive. The occurrence of species like *P. australis*, *I. glandulifera, Epilobium hirsutum* and *Urtica dioica* often indicates the succession towards *Phragmites* or tall-herb dominated marsh sites.

At elevations situated directly above the pioneer vegetation (flooding frequency averages 60-80%), mainly *P. australis*-dominated vegetation is found. However, *P. australis* individuals are found at a very wide range of flooding frequencies – from 7 to 92%. In the Schelde estuary, as in the other tidal freshwater marshes, *P. australis* can reach exceptional heights of 4 m or more, due to the abundant availability of nutrients. An elaborate network of underground roots and rhizomes is associated with wetlands dominated by reed. Although *P. australis* is the most common dominant species in this type of wetland, macrophytes are often found scattered through the nearly monospecific reed patches. *Caltha palustris* ssp. *araneosa* is restricted in distribution to the Schelde, Rhine-Meuse and Elbe estuaries and abundant at the beginning of the growing season (March – April). *C. palustris* flowers early, avoiding the tough competition with the fully developed reed shoots. *C. palustris* ssp. *araneosa* forms specific spider-like structures, essentially roots emerging from thick nodes in the shoot (Van Steenis 1971). These roots are transported with flooding water after the collapse of the shoot (vegetative reproduction). Other species, which are found growing in the *Phragmites australis* stands but are never co-dominant include *Solanum dulcamara, Valeriana officinalis, Typha latifolia, Iris pseudacorus, Phalaris arundinacea, Lythrum salicaria, Stachys palustris, Eupatorium cannabinum* and *Glyceria maxima.*

In the past, reed-dominated wetlands were frequently commercially exploited. This agricultural management conserved large monospecific stands of reed. In contrast to Dutch practices, no evidence was found for the preferential culture of high, broad-leaved individuals. With the disappearance of this local economical activity, many of the largely monospecific reed stands have disappeared. Over time, the monospecific reed stands have gradually increased in elevation, allowing colonization by other species, mostly the native *U. dioica* and *E. hirsutum* and non-native *I. glandulifera*. These three species are abundant in almost all of the tidal freshwater marshes along the Schelde estuary. After colonization, the wetland gradually evolves towards co-domination of reed and tall herb vegetation. Large stands of tall herb-*P. australis* co-dominated vegetation (flooding frequency ca. 40-60%) currently occurs along the fresh water portions of the Schelde. Common species associated with reed are *Calystegia sepium, Galium aparine, Rumex obtusifolius, Persicaria hydropiper* and *Cirsium arvense*. In sites with < 40% flooding frequency, the tall herbs become dominant.

The settlement of various willow species in elevated *P. australis* or tall herb-dominated wetland is the first stage in the development of *Salix*-dominated wetland forests, especially in habitats where the average flooding frequency is between 0 and 40%. The main trees are *Salix fragilis* and *S. alba*. The main shrubs are *S. cinerea, S. caprea, S. triandra, S. fragilis, S. dasyclados* and *S. viminalis*. Non-woody species are: *Solanum dulcamara, Cardamine amara, Rumex obtusifolius, I. glandulifera, C. palustris* ssp. *araneosa, E. hirsutum* and *P. australis*.

A special type of historically exploited wetland forest in tidal freshwater marshes is the coppiced osier-thickets that were dominated by *Salix* spp., and with *Alnus glutinosa* often as a co-dominant. Twigs from osier-thickets that are harvested every year are used in basket braiding while twigs of plants harvested or every 3-4 years are used in dike fortification. Most osier-thickets have been abandoned and have evolved towards *Salix* forests with *Alnus glutinosa* now occurring mainly on or near dikes.

In 2004, Struyf (2005) performed an intensive mapping of the most important tidal freshwater marsh plant species in the Tielrode marsh along the Schelde (Table 2.2). This site was considered to be representative of the majority of tidal freshwater marshes along the river. Struyf mapped pure *P. australis* vegetation, a mixed ruderal-reed vegetation (still dominated by *P. australis*), a *Salix* vegetation and a tall-herb dominated vegetation. Twelve plots were mapped in each vegetation type. The dominant species were *E. hirsutum, I. glandulifera, U. dioica* and *P. australis*, which comprised, together with *Salix*, the bulk of the biomass. At the height of the growing season (September) biomass was estimated by random sampling of 10 individuals of all species in each of the four types of vegetation, and multiplying average occurrence frequency with average individual biomass. *Salix* biomass was not estimated, because analyzing would result in the destruction of the vegetation as such. The total average biomass of the herbs in the four types of vegetation ranges from 1.4 to 2.25 kg/m^2, whereas the number of individuals of the incorporated species ranges from 26 to 103.

Table 2.2: Frequency and biomass of the major herb species dominating the tidal freshwater marshes along the Schelde, as observed in one representative site (Tielrode marsh, 2004). SE = standard error. Source: Struyf (2005).

Vegetation	Species	Individuals		Biomass	
		#/m^2	SE	kg/m^2	SE
Phragmites australis vegetation	*Phragmites australis*	103	7	2,25	0,31
Phragmites australis dominated with tall herb intrusion	*Phragmites australis*	53	5	1,17	0,18
	Impatiens glandulifera	1	1	0,19	0,07
	Urtica dioica	6	2	0,01	0,01
	Epilobium hirsutum	1	1	0,02	0,02
Salix spp. vegetation	*Phragmites australis*	5	3	0,39	0,2
	Impatiens glandulifera	8	2	0,72	0,22
	Urtica dioica	10	3	0,36	0,1
	Epilobium hirsutum	3	2	0,39	0,28
Tall-herb dominated vegetation	*Phragmites australis*	19	6	0,44	0,16
	Impatiens glandulifera	23	8	0,68	0,25
	Urtica dioica	6	2	0,07	0,02
	Epilobium hirsutum	7	3	0,36	0,15

2.3.2 Biesbosch and Oude Maas (The Netherlands)

The ecology of most of the tidal freshwater marsh habitats in the Rhine-Meuse Delta (especially the area 'Biesbosch') has been changed dramatically since the closing of the Haringvliet sluices in 1970, which created a barrier between the rivers and the North Sea. The Haringvliet dam (including sluices) was aimed at protection of the hinterland from flooding and at reduction of the salt intrusion to improve fresh water supply for agriculture and human consumption (drinking water). The ecological effects of closing in the estuary were enormous. Large inter-tidal areas disappeared or changed dramatically (Ferguson & Wolff 1983, Kuijpers 1995, Smit et al. 1997). The average tidal amplitude decreased from 2 m in 1960 (Zonneveld 1960) to 0.2-0.3 m after the closure of the Haringvliet (De Boois 1982). Essentially, the construction of the dam has changed most of the former estuary into a micro-tidal inland fresh water basin (Smit et al. 1997). Only in the most northern parts of the Biesbosch does a 70-80 cm tidal amplitude remain (Weeda et al. 2003). Shortly after the closure, intertidal flat habitats, which bordered the emergent wetland areas, began to change because they were not adapted to the decreased vertical tidal amplitude and lower current velocities. Erosion by wind induced waves gradually changed the gentle slopes of the tidal flats into steep slopes bordering the intertidal wetlands.

Before the closure of the Haringvliet and the construction of other important dikes to prevent storm flooding, the 'Biesbosch' area consisted of approximately 8,000 ha of tidal freshwater marshes, including willow forests, flats and channels (Zonneveld 1960, Verhey et al. 1961). After closure, only the open sea connection of the Rotterdam harbour and some river branches along the Rhine and the Meuse still had tides of 1 m (Barendregt 2005). The most important tidal freshwater marshes of the Netherlands are nowadays found along the river branch of the Oude Maas ('Old Meuse') with a 1 m tidal wave, and in the Sliedrechtse Biesbosch, the most northern and seaward part of the Biesbosch (Weeda et al. 2003, Barendregt 2005). Bulrush species, reed, and willows dominate the vegetation. The following review of the vegetation is derived mostly from Zonneveld (1960, 1999), Van Wirdum (1972), De Boois (1982) and Weeda et al. (2003).

Emergent wetlands

In contrast to tidal freshwater marshes along the Schelde, *Bolboschoenus maritimus, Schoenoplectus triqueter* and *Schoenoplectus lacustris* were the dominant species of intertidal habitats in the Biesbosch, hence the areas name (Bies = Dutch word for bulrush; Bosch = bush). Bulrush dominated tidal freshwater marshes were situated at lowest elevations, and represented the earliest phase in the succession; the name of the vegetation association is the *Alismato-Scirpetum maritimi* (Schaminee et al. 1995). One of the most characteristic species of the tidal freshwater marshes, *S. triqueter*, was present at lowest locations with distinct wave dynamics and high flooding depths, particularly where waves added to erosion. *S. triqueter* appeared to have a broader tolerance of inundation than *B. maritimus* (Deegan et al. 2005). Increased navigation on most rivers induced increased erosion. To counteract erosion, at many locations small stone defences were added after 1970, which together with reduced tidal amplitude, resulted in the almost complete elimination of *S. triqueter*. Today, *S. triqueter* is still present but mostly in newly levelled areas, where increased flooding and high water depth stimulates its growth.

Coops & Smit (1988) investigated the ecology of the three bulrush species. *S. lacustris* occurred in low elevation habitats with muddy soil; *S. triqueter* occured on sandy-clay soils at medium elevations; *B. maritimus* occurred on sandy soils at higher elevation. Water level fluctuation is the most important factor controlling seed germination and seedling establishment of the different bulrush species (Clevering 1995, Clevering et al. 1996).

In sheltered parts of the tidal creeks, which almost never dry completely, *Nuphar lutea, Potamageton pusillus* and *P. perfoliatus* dominated locally. *Potamogeton pectinatus*, *Zannichellia palustris palustris*, *Sagittaria sagittifolia, Veronica anagallis-aquatica, V. catenata* and *Sparganium emersum* sometimes also occurred in or near the gullies.

Special (semi)aquatic species that occur in tidal freshwater marshes are species of *Elatine*. Although extreme rare and not completely associated with tidal freshwater marshes, *Elatine hydropiper* and *E. triandra* both occurred in this system in The Netherlands (Weeda et al., 1987). Even more striking is the frequent presence of these species in the tidal freshwater marshes in sub-fossil finds from 6000-600 years ago (Brinkkemper et al. 2007), again from the lower parts of rivers and the tidal freshwater marshes. In this publication the (ancient and present) connection of the species with fresh water dynamics and periods with higher temperatures is indicated.

Phragmites australis and *Typha angustifolia* can colonize bulrush-dominated tidal freshwater marshes in the next stage of succession and co-domination by both bulrush and reed occurs as an intermediate stage in succession. *T. angustifolia* can grow in areas where the tidal depth of the water is as great as 1 m (Coops et al. 1999); consequently belts of this species often occur on the open-water side of *P. australis* stands. *T.*

angustifolia mainly occurred at flooding frequencies higher than 60%. *Callitriche stagnalis*, *Vaucheria* spp., *Apium nodiflorum* and *Alisma plantago-aquatica* often occurred at similar flooding frequencies as *T. angustifolia.* This association is described as *Typho-Phragmitetum* (Schaminee et al. 1995). A different pathway of vegetation development can occur when *Phalaris arundinacea* becomes the dominant species in unmanaged tidal freshwater marshes in the low-marsh habitat. The short-living species *Veronica anagallis-aquatica, Persicaria hydropiper* and *Callitriche stagnalis* had their optimum in pioneer communities on bare soil.

S. lacustris and *P. australis* were often harvested; if so, it was management that mainly determined which species was dominant. In most situations, however, managers preferred stands dominated by *Phragmites.* In *P. australis*-dominated habitats, high rates of sediment accretion occurred, inducing further succession to *U. dioica* and *E. hirsutum* dominated vegetation or to *Salix* dominated forests. Similar to the locations in the Schelde estuary, *C. palustris* ssp. *araneosa* evaded competition with *P. australis* by early flowering. The same strategy is used by *Cardamine amara, Ranunculus ficaria* and *Poa trivialis.*

In this stage of the succession, several 'inundation classes' could be distinguished: *Cardamine amara, Senecio paludosus, Persicaria hydropiper,* and *Lythrum salicaria* occurred over a very wide range of flooding frequencies (0-100%) and the plants were often inundated for long periods in over 70 cm of water. In contrast, species like *Stachys palustris* and *Ranunculus sceleratus* only occurred at flooding frequencies higher than 60%, but did not withstand long periods of deep inundation (maximally 50 cm). *E. hirsutum, Symphytum officinale* and *Lycopus europeaus* tolerated only flooding frequencies below 60%, and only tolerate shallow inundation. Natural levees bordering creeks provided habitat for several tall-herb species like *Valeriana officinalis, Anthriscus sylvestris, U. dioica* and *Heracleum sphondylium.* These plants generally did not grow at flooding frequencies higher than 50%. Poorly drained inner wetland basins were frequently dominated by *Sparganium erectum, T angustifolia, T. latifolia* and *Glyceria maxima.*

A characteristic species, flowering in spring (end April – beginning May) before *Phragmites australis* is full-grown, is *Leucojum aestivum.* This is a bulbous plant growing up to 1 m, bearing bell-shaped white flowers, and resembling a huge snowdrop (*Galanthus*) (Fig. 2.2). Seeds and bulbs are water dispersed. This species is only locally found and has its optimum in reed marshes along the Oude Maas. *Leucojum* has a very specific ecological habitat: it needs very wet conditions, with a mix of clay layers and organic matter. It is suggested that in the Biesbosch region, sediments were only recently deposited and dynamics do not resemble the specific habitat type enough, so that the species could not yet settle in that area. *Leucojum* is also distributed in reed-dominated wetlands in the centre of the country in areas that historically had been part of the coast (River Utrechtse Vecht). Some old records are available from the tidal freshwater marshes along the Schelde (Mennema 1967) and the Elbe (H-H Poppendieck pers. comm.).

Relevés with *Leucojum aestivum* from 1971 (archives Zuid-Hollands Landschap) are given in Table 2.3; they describe the herb rich association with the name *Valeriano-Senecionetum fluviatilis* (Stortelder et al. 1999). Recently the same location also supports a vegetation that includes *Symphytum officinale, Filipendula ulmaria* and *Thalictrum flavum* (A Barendregt pers. obs.). The modern vegetation persists best under conditions where annual mowing is applied. If mowing is absent, light limitation strongly hinders its development. This association is characteristic of The Netherlands and is present in two regions: the (present and former) tidal freshwater marsh areas in the Rhine-Meuse estuary and the delta of the river IJssel, another branch of river Rhine at the former coastline to the north (Weeda et al. 2005). Next to *L. aestivum*, the species *Senecio fluviatilis* indicates the tidal influences and the restricted distribution pattern, since the distribution and ecological niche in (former) tidal freshwater regions are fully comparable.

Fig. 2.2: ***Leucojum aestivum*** in a reed marsh along the Oude Maas (NL). Photo by A. Barendregt.

Table 2.3: Vegetation description of 8 plots (ca. 6 m2) along the Oude Maas in reed marsh with Leucojum aestivum in the year 1971; quantities of species according the Braun-Blanquet scale. Source: unpublished data, archive ZHL, Rotterdam

Relevé number	1	2	3	4	5	6	7	8
Leucojum aestivum	1	1	2a	1	2a	3	1	2b
Phragmites australis	1	1	2m	1	1	1	1	1
Cardamine amara	1	1	2a	2a	1	1	+	
Poa trivialis	1	1	2a	1	3	2a	2a	
Calystegia septum	1				1	1	+	1
Valeriana officinalis	1	1					+	2a
Caltha palustris	+		2a	+	+	+		
Galium aparine	1	1					+	+
Iris pseudacorus	+	1				1		1
Anthriscus sylvestris	4	3	+	r				
Ranunculus ficaria	2a	2a	2m					
Senecio fluviatilis	1	+						
Phalaris arundinacea			2a		1			
Epilobium hirsutum	+	1						1
Rumex obtusifolius				+	r		1	
Angelica archangelica				+			+	1
Solanum dulcamara	+							
Urtica dioica		2a						
Humulus lupulus		+						
Ranunculus auricomus			1					
Rumex crispus					r			
Myosotis scorpioides					r			
Lysimachia vulgaris					1			
Atriplex hastata						r		
Galeopsis spp.								1
Equisetum fluviatile								1
Carex acuta								+

Willow forests

The last succession stage was, as observed in other regions in Europe, Salix-dominated tidal forest (Wolf et al. 1997). In osier-thickets, Salix viminalis, S. alba, S. dasyclados, S. caprea, S. triandra and S. purpurea were dominant. Osier-thickets were harvested each 3, 4 or 5 years. Poorly drained, unmanaged thickets were invaded by *Lythrum salicaria*, *Senecio paludosus*, *Stachys palustris*, *Lycopus europaeus* and *Bidens frondosa.* Well-drained osier-thickets on the other hand were often the habitat for *Caltha palustris*, *Poa trivialis*, *Urtica dioica*, *Cardamine amara*, *Anthriscus sylvestris*, *Ranunculus auricomus* and *Rumex obtusifolius.*
In addition to the vascular plants, it should be stated that moss and lichen species are well represented in osier thickets. High air-humidity and relatively undisturbed areas with many dead trunks provide excellent growing conditions for several species, especially epiphytic ones (see: Van der Pluijm 1995). Even in the Oude Maas region, very close to the Rotterdam harbour and its oil refineries, very sensitive species for air pollution such as the lichen *Usnea* spp.. are present in the centre of the willow forests (A Barendregt pers. obs.). Moss genera such as *Marchantia*, *Lunularia*, *Fissidens*, *Mnium*, *Plagiomnium* and *Rhizomnium* are represented with several species. One of the most rare and characteristic species, in the Sliedrechtse Biesbosch, is *Timmia megapolitana.* Terrestrial moss and mushroom species are poorly represented due to the flooding in the tidal freshwater marshes.
Comparable with the zonation in the Biesbosch (Zonneveld 1960), the elevation results in different flooding regimes that still occur along the Oude Maas. Osier beds that are flooded twice daily are characterized by *Callitriche stagnalis* and *Vaucheria* spp., and, most notably, by the absence of many other species. In higher elevation habitats (i.e., close to mean high tide) *Cardamine amara* and *Myosotis scorpioides* reach their optimum. Osier beds at the highest elevations are characterized by *Filipendula ulmaria, Circaea lutetiana, Heracleum sphondylium* and *Humulus lupulus.* Willow vegetation belongs to the association of *Cardamino amarae-Salicetum albae* of the alliance *Salicion albae* (Stortelder et al. 1995), although the highest zonation inclines towards the *Alno-Padion* alliance.

The production and cycling of biomass in osier coppices is very high. To illustrate this, the biomass in a coppice at Oude Maas that was cut in February, was measured in September 2005 (A Barendregt unpubl. data). The herbs were 200 g/m^2 dry weight, willow leaf biomass was 500 g/m^2 and one-year old branches were 2000 g/m^2. This illustrates something about the potential biomass production, as already clear from the Schelde with its 4 m high reed. A willow coppice produced in dry weight as a whole 27 Mtons/ha/year. However, the following spring almost all recognizable remnants of herbs and leaves would be absent; the breakdown of the tissues is very rapid.

The structure of the osier stands changes if they lie fallow for many years. Willows grow to 10-15 m and become susceptible to storm damage because the shallow root system can not support the large aboveground biomass. Fallen willow trunks however will continue to grow in this ecosystem. The final result is an Atlantic humid 'mangrove', almost impossible to penetrate through all the fallen trunks and new grown vertical branches, but extremely rich in niches, especially by the holes created by the roots of the fallen trees. Species of the alliance of *Aldo-Padion*) invade the forests. Species such as *Carex remota, Scrophularia nodosa, Festuca gigantea, Impatiens noli-tangere, Cornus sanguinea, Ribes nigrum* and *Ribes rubrum,* are characteristic for these conditions. Recently *Chrysosplenium alternifolium* and *Carex strigosa*, two characteristic species for tempest-struck, unmanaged willow forests were observed in Biesbosch (Weeda 2006).

2.3.3 Elbe (Germany)

The tidal Elbe stretches around 142 km from the mouth at Cuxhaven to the created barrier at Geesthacht. The fresh water tidal part of the Elbe is about 80 km long. The estuary is characterised by a macro-tidal regime (amplitude about 3.6 m) and highly variable water depth. The harbour of Hamburg is situated at the upstream fresh water reaches and the river is characterized as having intense ship traffic in association with a highly industrialized landscape. The river is highly eutrophic (Riedel-Lorjé & Gaument 1982) and O_2 depletion occurs in the fresh water portion of the estuary during early summer. Compared to the Schelde estuary, however, the Elbe estuary is generally better oxygenized and has lower partial CO_2-pressures, indicating a less heterotrophic character (Goosen et al. 1999).

Approximately 1,000 years ago humans started to alter the landscape of the Elbe estuary significantly by building dikes around their settlements on former wetland areas. In the following centuries, dikes were perpetually heightened and built closer to the shoreline of the Elbe. In consequence, the potential flooding area was increasingly restricted (Garniel & Mierwald 1996). In the region of Hamburg non-diked wetlands decreased in area from approximately 1500 ha in 1880 to less than 500 ha in 1980 (Preisinger 1991), mainly due to the development of the harbor of Hamburg. After a major flood in 1962 a large dike-construction program was implemented along the whole Elbe estuary leading to the loss of approximately 75% of the former tidal freshwater marsh area and also to the destruction of large parts of one of Europe's largest tidal freshwater marsh complex (Haseldorfer Marsch). In addition, building of floodgates at the confluences to the Elbe restricted the tidal influence of all tributaries. By repeated dredging during the last 150 years the main channel of the Elbe was deepened from a water depth of 3.5 m in the 19th century up to 14.5 m today. All these described changes are at least partly responsible for the observed increase of the tidal amplitude in Hamburg from 1.7 m at the end of the 19th century to more than 3.5 m today (see: Jensen 2007).

Vegetation description

Despite the long-lasting human influence on the Elbe Estuary, remnants of tidal freshwater marshes with their characteristic vegetation are still present. The following vegetation description derives from Kötter (1961), Wolf (1988), Preisinger (1991), Oertling (1992) and Below (2006), as well as from our own unpublished data. In general the tidal freshwater marsh vegetation of the Elbe Estuary can be characterized by a typical zonation reflecting the position of the sites in relation to mean high tide (e.g. Kötter 1961, Raabe 1986). Some differences regarding the zonation pattern of the tidal freshwater marshes of the upper and the lower tidal fresh water reaches (east and west from Hamburg, respectively) have been described (Seelig 1992, Oertling 1992). As a consequence of human alterations, both the spatial extent of the tidal freshwater marshes and their distribution pattern along the elevation gradient have changed during the last decades (Immeyer 1996). In the following paragraphs, the general zonation pattern and major temporal changes in tidal freshwater marsh composition and distribution will be described in some more detail, some information on the occurrence of submerged species in the Elbe estuary will be provided, and two endemic plant species will be introduced.

Within the tidal freshwater marshes dominated by herbaceous vegetation three characteristic zones (pioneer zone, low marsh, high marsh) can be distinguished along the elevation gradient. Standing crop (g/m^2) and species diversity data for these zones were analysed in summer 2006 and are presented in Fig. 2.3 (G Engels pers. comm.). Bulrushes dominate the wetlands that occur at the lowest end of the tidal zone. Most often *Bolboschoenus maritimus* dominates this so-called pioneer zone. The second most important species is *Schoenoplectus tabernemontani*. These species of the pioneer zone are found up to 1.5 to 1.75 m below mean high tide (MHT). Although the lowest habitats are protected against wave erosion because tidal flooding rapidly inundates them, *Eleocharis uniglumis* replaces the above mentioned bulrushes at heavily wave-exposed sites. In general, *B. maritimus* promotes sedimentation and thus facilitates succession towards reed-dominated marshes. *Schoenoplectus triqueter*, another bulrush species characteristic for the pioneer zone, has declined during the last decades and is today only found at a few locations along the Elbe Estuary. Further species, which regularly occur in the pioneer zone are *Nasturtium officinale*, *Callitriche* spec., and especially on sandy substrates *Deschampsia wibeliana*.

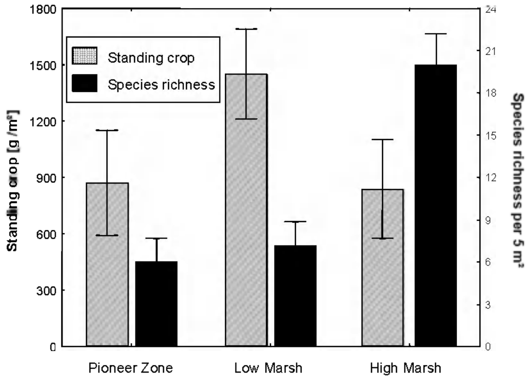

Fig. 2.3: Standing crop and species richness of the vegetation of the pioneer zone (dominated by *Bolboschoenus maritimus*), the low marsh community (dominated by *Phragmites australis*) and the high marsh community (co-dominated by tall herbs) of tidal freshwater marshes at the Elbe Estuary (G Engels pers. comm.).

Low marsh vegetation is mainly dominated by *Phragmites australis*. which forms large and almost mono-specific stands along the Elbe. Other species are *Typha angustifolia*, *T. latifolia*, *Phalaris arundinacea* and *Glyceria maxima* and any of these species can dominate some areas within this elevation zone. There seem to be no general differences in abiotic conditions between sites occupied by these different species and it has been hypothesized that initial establishment might be the main factor responsible for the spatial distribution pattern of the species (see: Wolf 1988). First *P. australis* tillers can be found up to approximately 1.2 m below MHT, but close reed stands most often establish at approximately 0.5 m below MHT. Beside these matrix species *Caltha palustris*, *Lythrum salicaria*, *Agrostis stolonifera* and again *Deschampsia wibeliana* are often occurring in these lower marshes as subordinate species.

The high marsh vegetation zone is most often also dominated by *Phragmites*, but in addition certain tall-growing herb species (e.g., *Calystegia sepium*, *Cirsium arvense*, *Epilobium hirsutum*, *Angelica archangelica*, *Urtica dioica*) are co-dominants. These high marsh species are restricted to sites above MHT. Usually high marsh vegetation is found only as a narrow belt between the lower-lying marsh communities and the non-marsh sites higher in elevation which are either dominated by trees and/or shrubs (under near to natural conditions) or by grasslands (on dikes, which are mostly grazed by sheep). In common to the Schelde estuary and the Biesbosch, the upper reaches of the Elbe estuary has in recent decades witnessed invasion of *Impatiens glandulifera*, however a change in the native plant species diversity is not reported.

Within the woody communities of the tidal freshwater marshes, a low marsh habitat referred to as softwood riparian forests and a high marsh habitats referred to as the hardwood forests are distinguished. The softwood forests are dominated by different *Salix* species (e.g., *Salix alba, S. fragilis*) in the tree layer. In the herb layer of the lowest *Salix* forests *Caltha palustris*, *Callitriche* spp., *Amblystegium riparium* and *Cardamine amara* occur, whereas *Rubus caesius*, *Persicaria hydropiper* and *Geum urbanum* take over at higher elevations. In the highest reaches of the Elbe estuary, hardwood species were once common. Today they occur only fragmentary, mainly because their former habitat is lying landward of dikes. Only scattered individuals of *Ulmus laevis*, *U. minor*, *U. glabra*, *Fraxinus excelsior*, *Alnus glutinosa*, *Euonymus europaea* and *Crataegus* spp. are present. Some of the sites are partly covered with planted tree species like *Populus* spp. today. *Alliaria petiolata, Anthriscus sylvestris, Galium aparine, Rubus* spp. and *Geum urbanum* are dominant herb species at high elevations, with flooding frequencies lower than 30%.

Changes in vegetation zonation

This general zonation pattern of the tidal wetland vegetation is of course only a simplification of the distribution of different species and communities in the Elbe Estuary. One major deviation from this zonation is the fact that the so-called pioneer zone is missing upstream from Hamburg. In upstream areas, where the tidal amplitude diminishes and where the wave energy becomes much smaller due to a lower fetch, *P. australis* dominates also the lowest marshes and *B. maritimus* and *S. tabernemontani* are almost lacking (Seelig 1992).

Beside this spatial variation in zonation pattern, temporal changes in the vegetation of tidal freshwater marshes could be suspected as a consequence of the above-mentioned human alterations. While the main zonation pattern of tidal freshwater marsh vegetation remained the same during the last 50 years, some changes occurred, which could be most likely attributed to changes of the hydro-regime. Immeyer (1996) found that at present the lower borders of marshes are mostly found at higher elevations than in the 1950s (according to Kötter 1961). She argues that the flooding duration has increased at the former lower boundaries of the pioneer zone due to changes of the hydro-regime (see also: Chapter 17). Further, she showed that flooding duration at elevations of today's lower boundaries of the vegetation almost exactly equals the flooding duration at elevations of the lower boundaries of the pioneer zone in the 1950s.

Changes between 1975 and 2002 in the spatial extent of tidal freshwater marshes in general, and in reed-dominated vegetation especially, have been analyzed by Schröder (2007). He found that reeds decreased by approximately 20 ha, mainly as a consequence of preceding succession to softwood willow-dominated forests in the upper reaches of the Elbe Estuary, upstream from Hamburg. Contrary to this finding the area with *P. australis* increased from 750 to 950 ha in the tidal freshwater marshes reaches downstream from Hamburg. Schröder (2007) hypothesizes that changes in land-use (abandonment of grassland management), the development of new habitats after the creation of man-made islands with material from channel dredging, as well as high sedimentation rates might contribute to this increase in tidal freshwater marsh area. However, the increase of tidal reeds between 1975 and 2002 should be carefully evaluated against the background of abovementioned destruction of 75% of the former flooding area after 1962.

Submersed vegetation

In the 19th century, several submersed species occurred in the fresh water part of the Elbe estuary (e.g., *Groenlandia densa*, *Potamogeton pectinatus*, *Zannichellia palustris*; Eschenburg & Schulz 1927). Kötter (1961) mentions that most of them have disappeared almost completely due to bank fortifications and increasing pollution, and that they could only be observed near the confluence with tributaries or near sluices in the 1950s. The most often observed aquatic species at that time were *Ceratophyllum demersum* and *Lemna* spp. (Kötter 1961). However, recently Stiller (2007) found small populations of *Chara vulgaris*, *Zannichellia palustris* and *Limosella aquatica* on a mudflat approximately 80 cm below MHT on the island of Nesssand. *Z. palustris* and *Chara vulgaris* were last found in the tidal fresh water zone of the Elbe in the 1920s and in 1890s, respectively (cf. Krieg & Kies 1989).

Endemic species

Two species are endemic for the Elbe estuary (Burkart 2001). The occurrence of the first, *Oenanthe conioides,* has dramatically declined over the last decades and it is now considered severely endangered. The species is listed in annexes II and IV of the FFH-directive of the European Union today. Its endangered status is mostly due to its endemic character and severe habitat loss in recent decades caused by harbour development, dike construction, and erosion as a consequence of channel deepening. The species is only found at elevations below MHT in a narrow belt between the pioneer zone and the reed dominated low marsh community with flooding frequencies of almost 100%. See Neubecker et al. (2005) for more information.

Another endemic species along the Elbe estuary (endemic to Elbe, Eider and Weser) is *Deschampsia wibeliana*. In contrast to *O. conioides* this species has considerably increased in abundance along the shores of the Elbe Estuary during the last century. Still it is considered as endangered because of its restricted distributional range. *D. wibeliana* occurs on sandy beaches as well as in the pioneer zone of tidal freshwater marshes and in gaps of the low marsh communities. In addition it is today one of the few plant species which regularly establishes itself on stone fortifications (Von Weihe & Reese 1968). As both endemic species are closely related to common and widespread taxa (*Oenanthe aquatica, Deschampsia cespitosa*), which occur in a wide range of wetlands in temperate Europe, but not under tidal conditions, the question arises whether the endemics of the Elbe estuary might have sympatrically evolved during the Holocene as a consequence of pronounced disruptive selection (cf. Kadereit & Kadereit 2005).

2.4 Conclusions

tidal freshwater marshes have almost became a rarely habitat-type in European estuaries, especially in undisturbed conditions, due mostly to historical and ongoing human management. The three main tidal fresh water areas in Europe we described are all characterized by a zonation in vegetation, along a hydrological gradient. Zonneveld (1960, 1999) linked the types of vegetation to the flooding frequency and at the same time to the soil development.

Few plant species are endemic to the European tidal fresh water zone, which is not surprising since isolation is absent due to connections with up- and downstream ecosystems. One characteristic subspecies for the tidal freshwater marshes has been described for the complete area of the Schelde, Rhine-Meuse and Elbe (*Caltha palustris* ssp. *araneosa*), while two endemics are present only along the Elbe freshwater estuary (*Deschampsia wibeliana, Oenanthe conioides*). Characteristic is the presence of *Leucojum aestivum, Schoenoplectus triqueter, Senecio fluviatilis* and *Senecio paludosus* in the tidal freshwater marshes. With their distribution and ecological relations those species indicate that the tidal freshwater marsh is an ecosystem with characteristic conditions and related species. In general, the vegetation from the three described tidal freshwater marsh regions is comparable, with low and high marsh and willow forests. On creek banks, tall herbs often persist as the climax vegetation. The tall-herb community differs between the estuaries: for example *Anthriscus sylvestris* is common in Elbe and Biesbosch, but is only rarely encountered in Schelde tidal freshwater marshes (and there not in the tidal system, but mostly on dikes).

European tidal freshwater marshes are at the moment fully incorporated in activities with human management or consequences of human activities. The quality of the river water itself is changing for the better in last decades (Van Dijk et al. 1994, Soetaert et al. 2006), improving the general conditions in the tidal freshwater marshes. Aquatic (floating and submerged) vegetation in the European tidal freshwater marshes was poorly represented in the past 50 years, mostly due to restricted water quality. At sheltered locations in tidal creeks, *Sagittaria saggitifolia*, *Nuphar lutea* and *Potamageton* spp. used to grow. During the last decade a spectacular recovery of the aquatic vegetation has been observed in the Biesbosch (e.g., with *Potamageton nodosus* and *P. perfoliatus*).

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Barendregt A (1980) Vergelijkend onderzoek naar de aktieve bodemfauna van de getijdegrienden langs Oude Maas (Z.H.). Zoological Museum, Utrecht University, NL.

Barendregt A (2005) The impact of flooding regime on ecosystems in a tidal freshwater area. Ecohydrology & Hydrobiology 5: 95-102.

Beerling DJ & Perrins JM (1993) Biological Flora of the British Isles. *Impatiens glandulifera* Royle (Impatiens roylei Walp.). Journal of Ecology 81: 367-382.

Below H (2006) Das Naturschutzgebiet Heuckenlock und seine Vegetation: Süsswasser-Gezeitenbereich der Elbe (Nachexkursion). In: Härdtle W, Horst K & Prüter J (eds) Jahrbuch des Naturwissenschaftlichen Vereins für das Fürstentum Lüneburg von 1851 e.v.: Flora und Vegetation im Nordöstlichen Niedersachsen. GER.

Brinkkemper O, Weeda EJ, Bohncke SJP & Kuijper WJ (2007) The past and present occurrence of *Elatine* and implications for palaeoenvironmental reconstructions. Vegetation History and Archaeobotany 17

Burkart M (2001) River corridor plants (stromtalpflanzen) in central European lowland: a review of a poorly understood plant distribution pattern. Global Ecology and Biogeography 10: 449-468.

Clevering OA (1995) Germination and seedling emergence of *Scirpus lacustris* L. and *Scirpus maritimus* L. with special reference to the restoration of wetlands. Aquatic Botany 50: 63-78.

Clevering OA, Blom CWPM & Van Vierssen W (1996) Growth and morphology of *Scirpus lacustris* and *S. maritimus* seedlings as affected by water level and light availability. Functional Ecology 10: 289-296.

Coops H & Smit H (1988) Biezen langs de Oude Maas. De Levende Natuur 89: 106-110. NL

Coops H, Geilen N & Van der Velde G (1999) Helophyte zonation in two regulated estuarine areas in the Netherlands: vegetation analysis and relationship with hydrological factors. Estuaries 22: 657-668.

Criel B, Muylaert W, Hoffmann M, De Loose L & Meire P (1999) Vegetatiemodellering van de buitendijkse gebieden langs de Zeeschelde. Rapport Onderzoek Milieu-effecten Sigmaplan (OMES), AMIS DS7.2, deelstudie 8. Gent / Brussel, BEL.

De Boois H (1982) Veranderingen in het milieu en de vegetatie in de Biesbosch door de afluiting van het Haringvliet. PhD Thesis, Wageningen Agricultural University, Wageningen, NL.

Deegan B, Harrington TJ & Dundon P (2005) Effects of salinity and inundation regime on growth and distribution of *Schoenoplectus triqueter*. Aquatic Botany 81: 199-211.

Ellenberg H (1963) Vegetation Mitteleuropas mit den Alpen. E. Ulmer Verlag, Stuttgart, GER.

Eschenburg H & Schulz A (1927) Gemarkungsflora von Holm und Plön. Aus den Schriften des Naturwissenschaft-lichen Vereins für Schleswig-Holstein 18: 62-161. GER.

Ferguson HA & Wolff WJ (1983) The Haringvliet-project: the development of the Rhine-Meuse estuary from tidal inlet to stagnant freshwater lake. Water Science and Technology 16: 11-26.

Garniel A & Mierwald U (1996) Changes in the morphology and vegetation along the human-altered shoreline of the Lower Elbe. In: Nordstrom KF & Roman CT (eds) Estuarine Shores: Evolution, Environments and Human Alterations, pp. 375-396. John Wiley & Sons, Chichester, UK.

Goosen NK, Kromkamp J, Peene J, Van Rijswijk P & Van Breugel P (1999) Bacterial and phytoplankton production in the maximum turbidity zone of three European estuaries: the Elbe, Westerschelde and Gironde. Journal of Marine Systems 22: 151-171.

Hejda M & Pysek P (2006) What is the impact of *Impatiens glandulifera* on species diversity of invaded riparian vegetation? Biological conservation 132: 143-152.

Immeyer J (1996) Untersuchungen zur Veränderung der ökologischen Bedingungen der Tideröhrichte an der Unterelbe zwischen den Kartierungen von Kötter und heute (1961-1995). Diplomarbeit am Institut für Angewandte Botanik, Universität Hamburg, GER.

Jensen K (2007) Röhrichte in Ästuaren: Verbreitung, Ökosystemfunktion und Gefährdung. BFG Veranstaltungen 2: 5-19. GER.

Kadereit G & Kadereit JW (2005) Phylogenetic relationships, evolutionary origin, taxonomic status and genetic structure of the endangered local Lower Elbe river (Germany) endemic *Oenanthe conioides* (Nolte ex Rchb. f.) Lange (Apiaceae): ITS and AFLP evidence. Flora 200: 15-29. GER.

Kötter F (1961) Die Pflanzengesellschaften der Unterelbe. Archiv für Hydrobiologie, Suppl. 26: 106-184.

Krieg H & Kies L (1989) Artenschutzprogramm Armleuchteralgen (Charophyta) und Süßwasser-Rotalgen (Rhodophyta) im Gebiet der Freien und Hansestadt Hamburg. Naturschutz und Landschaftspflege in Hamburg, Schriftenreihe der Umweltbehörde 30: 1-39. GER.

Kuijpers JWM (1995) Ecological restoration of the Rhine/Meuse estuary. Water Science and Technology 31(8): 187-195.

Massart J (1908) Essai du géographie botanique des districts littoraux et alluviaux de la Belgique. Recueil de l'Institut Botanique Léo Errera VII: 1g - 121e. Brussel, BEL.

Meire P, Rossaert G, De Regge N, Ysebaert T & Kuijken E (1992) Het Schelde-estuarium: ecologische beschrijving en een visie op de toekomst. Rapport RUG-WWE 28, IN nr. A. 92.57. Gent, Brussel, BEL.

Meire P, Ysebaert T, Van Damme S, Van den Bergh E, Maris T & Struyf E (2005) The Schelde estuary from past to future: a description of a changing ecosystem. Hydrobiologia 540: 1-11.

Mennema J (1967) *Leucojum aestivum* L. in België teruggevonden. Gorteria 3: 108. NL.

Neubecker J, Kohler S, Obst G & Jensen K (2005) Der Schierlings-Wasserfenchel – erfolgreiche Ansiedlung einer prioritären FFH-Art an der Elbe. Naturschutz and Landschaftsplanung 37: 248-255. GER

Oertling W (1992) Profil-Typen der Ufer-Vegetation der Unterelbe im Bereich und unterhalb der Mitteltidehochwasser-Linie. Institut für Angewandte Botanik der Universität Hamburg, Beiheft 3: 37-74. GER.

Preisinger H (1991) Strukturanalyse and Zeigerwert der Auen- and Ufervegetation im Hamburger Hafen- and Hafenrandgebiet. Dissertationes Botanicae Band 174. Cramer, Berlin, GER.

Raabe EW (1986) Die Gliederung der Ufervegetation der Elbe unterhalb Hamburg. Mitteilungen Natur- und Umweltschutz Hamburg 2: 117-141. GER.

Riedel-Lorjé JC & Gaument T (1982) 100 Jahre Elbe-Forschung – Hydrobiologische Situation und Fischbestand 1842-1943 unter dem Einfluss von Stromverbau und Sieleinleitungen. Archiv für Hydrobiologie, Suppl. 61: 317-376.

Rooth JE, Stevenson JC & Cornwell JC (2003) Increased sediment accretion rates following invasion by *Phragmites australis*: the role of litter. Estuaries 26: 475-483.

Saris F (1987) Patroon en proces in een zoetwater(getijden)delta. Rapport SCMO, Delft, NL.

Schaminee JHJ, Weeda EJ & Westhoff V (1995) De Vegetatie van Nederland – deel 2 – Plantengemeenschappen van wateren, moerassen en natte heiden. Opulus Press, Uppsala, Sweden.

Schröder U (2007) Aktuelle und historische Röhrichtentwicklung an Unter- und Außenelbe. BFG Veranstaltungen 2: 20-30. GER.

Seelig A (1992) Profil-Typen und Standorte der Elbufer-Vegetation zwischen Staustufe Geesthacht und Bunthäuser Spitze im Bereich der Mitteltidehochwasserlinie. Institut für Angewandte Botanik der Universität Hamburg, Beiheft 3: 5-36. GER.

Smit H, Van der Velde G., Smits R & Coops H (1997) Ecosystem responses in the Rhine-Meuse delta during two decades after enclosure and steps toward estuary restoration. Estuaries 20: 504-520.

Soetaert K, Middelburg JJ, Meire P, Van Damme S & Maris T (2006) Long-term change in dissolved organic nutrients in the heterotrophic Schelde estuary (Belgium, the Netherlands). Limnology and Oceanography 51: 409-423.

Steege V (2007) Die Entwicklung der Tide-Röhrichte an der Weser und ihr Leitbild. BFG Veranstaltungen 2: 31-45. GER.

Stiller G (2007) Vorgezogene überblicksweise Überwachung der Qualitätskomponenten Makrophyten und Angiospermen in der Tideelbe gemäß EG-Wasserrahmenrichtlinie. Gutachten im Auftrag von Sonderaufgabenbereich Tideelbe der ARGE ELBE, Wassergütestelle Elbe, Hamburg, GER.

Stortelder AHF, Schaminee JHJ & Hommel PWFM (1999) De Vegetatie van Nederland – Deel 5 – Planten-gemeenschappen van ruigten, struwelen en bossen. Opulus Press, Uppsala, Sweden.

Strucker RCW (1996) Vogels van de zoetwatergetijderivier de Oude Maas. Maasdam. Haveka, Alblasserdam, NL.

Struyf E, Van Damme S & Meire P (2004) Possible effects of climate change on estuarine nutrient fluxes: a case study in the highly nutrified Schelde estuary (Belgium, The Netherlands). Estuarine Coastal and Shelf Science 60: 649-661.

Struyf E (2005) The role of freshwater marshes in estuarine silica cycling (Schelde estuary). PhD-thesis, Antwerp, BEL.

Temmerman S, Govers G, Wartel S & Meire P (2004) Modelling estuarine variations in tidal marsh sedimentation: response to changing sea level and suspended sediment concentrations. Marine Geology 212: 1-19.

Van Allemeersch R, Hoffmann M & Meire P (1999) Ontwerpbeheersplan voor het Vlaams natuurreservaat slikken en schorren van Schelde en Durme. University of Gent / Institute for Nature Conservation, Brussel, BEL.

Van der Pluijm A (1995) De mos- en korstmosflora van de Biesbosch. Report Staatsbosbeheer regio Brabant-West, Biesbosch, Werkendam, NL.

Van Dijk GL, Van Liere L, Admiraal W, Bannink BA & Cappon JJ (1994) Present state of the water quality of European rivers and implications for management. The Science of the Total Environment 145: 187-195.

Van Steenis CGGJ (1971). De zoetwatergetijdedotter van de Biesbosch en de Oude Maas, *Caltha palustris* L. var. *araneosa*, var.nov. Gorteria 5: 213-219. NL.

Van Wirdum G (1972) Rapport over vegetatie-onderzoek langs de Oude Maas, 1969-1971. Archives Rijnmond, Rotterdam, NL.

Verhey CJ, Heyligers PC, Lebret T & Zonneveld IS (1961) De Biesbosch, land van het levende water. Thieme, Zutphen, NL.

Von Weihe K & Reese G (1968) *Deschampsia wibeliana* Sonder (Parl.) – Beiträge zur Monographie einer Art des Tidegebietes. Botanische Jahrbücher 88: 1-48. GER.

Weeda EJ (2006) Crisis in de boomlaag, feest in de kruidlaag: *Carex strigosa* Huds. terug langs de benedenrivieren. Stratiotes 32: 31- 50. NL.

Weeda EJ, Westra R, Westra C & Westra T (1987) Nederlandse Ecologische Flora. Wilde planten en hun relaties 2. IVN, Amsterdam, NL.

Weeda EJ, Van der Neut JJM, Boesveld AAM & Weel BAM (2003) Nationaal Park De Biesbosch: Schatkamer van de wilde flora. Een overzicht van zeldzame en bedreigde vaatplanten. Alterra / Staatsbosbeheer. Wageningen, NL.

Weeda EJ, Schaminee JHJ & Van Duuren L (2005) Atlas van Plantengemeenschappen in Nederland – deel 4 – Bossen, struwelen en ruigten. KNNV Uitgeverij, Utrecht, NL.

Wolf A (1988) Röhrichte und Rieder des holsteinischen Elbufers unterhalb Hamburgs. Schriften des Naturwissenschaftlichen Vereins für Schleswig-Holstein 58: 55-68. GER.

Wolf RJAM, Vrielink JG & De Waal RW (1997) Riverine woodlands in the Netherlands. Global Ecology and Biodiversity Letters 6: 287-295.

Zonneveld IS (1960) De Brabantse Biesbosch. Een studie van bodem en vegetatie in een zoetwatergetijdedelta. PhD thesis, Agricultural University, Wageningen, NL.

Zonneveld IS (1999) De Biesbosch een halve eeuw gevolgd. Staatsbosbeheer / Uitgeverij Uniepers, Abcoude, NL.

Chapter 3

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

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3.1 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.

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

3.2 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.

3.3 Methods

3.3.1 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.

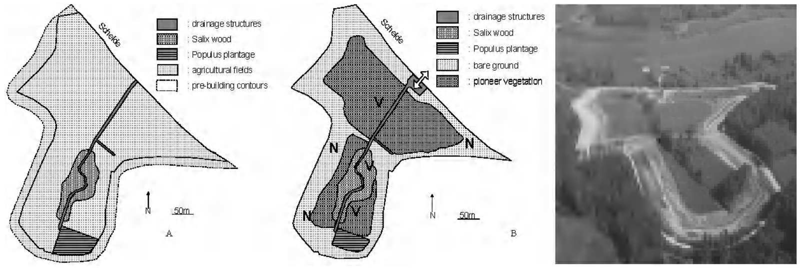

Figure 3.1. Schematic overview of the study site before (A) and after construction (B). Sampling locations with ("V") and without ("nV") initial vegetation are shown in B. The central creek is shown in gray, the white arrow indicates the culverts' location. C is an aerial view of the site just after construction.

3.3.2 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).

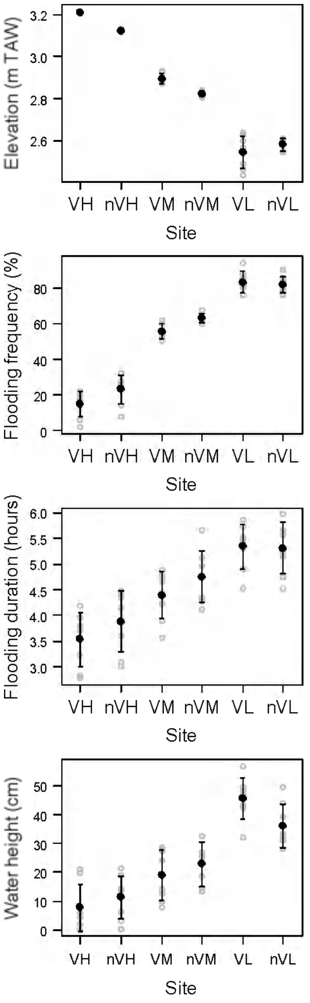

Figure 3.2. Hydrological variables and elevation gradient for the study locations during the study period. For initially vegetated sites (V) as well as for initially non-vegetated sites (nV), three elevation classes were considered: low (L), medium (M) and high (H). For elevation, grey points represent 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.

3.3.3 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.

3.4 Results

3.4.1 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).

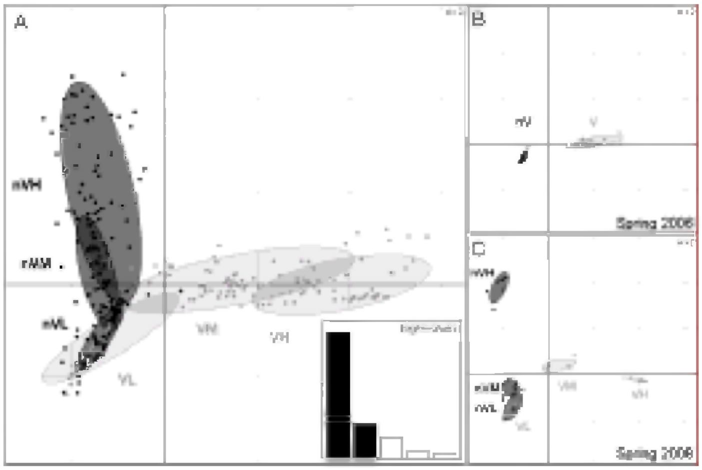

Figure 3.3. PCAIV of location effect. A) First factorial plane and eigenvalue diagram. F1, 65 %; F2, 18%. B) Within-date structure considering only the first spring survey. H, M and L are not marked because of complete overlap. C) Within-date structure considering only the last spring survey.

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).

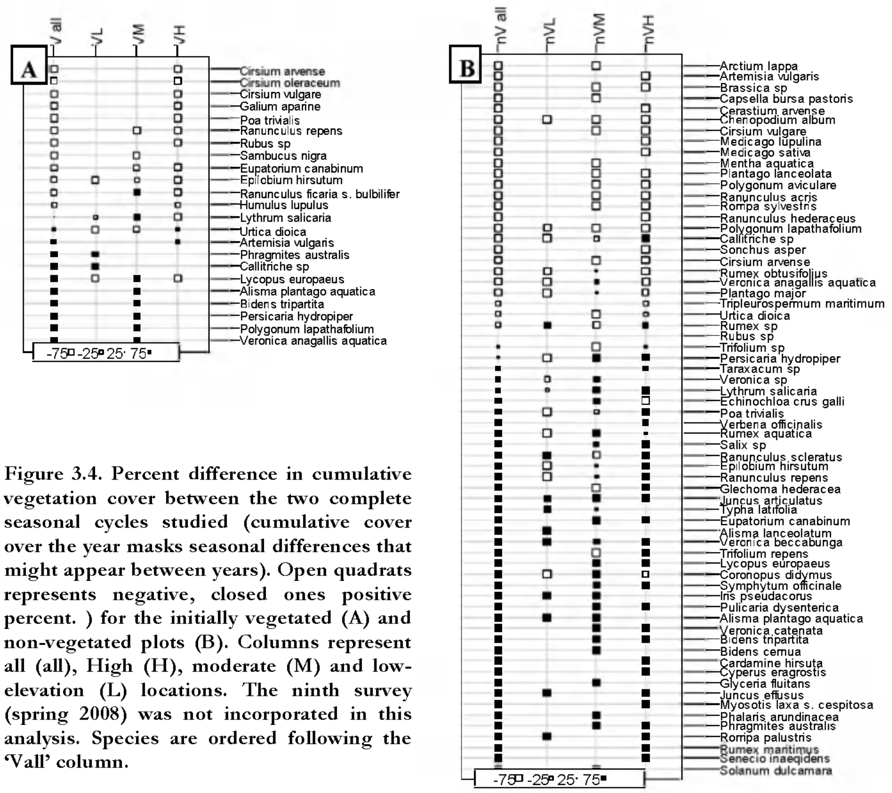

Figure 3.4. Percent difference in cumulative vegetation cover between the two complete seasonal cycles studied (cumulative cover over the year masks seasonal differences that might appear between years). Open quadrats represents negative, closed ones positive percent.) for the initially vegetated (A) 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.

3.4.2 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 3.1. 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%)

3.4.3 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).

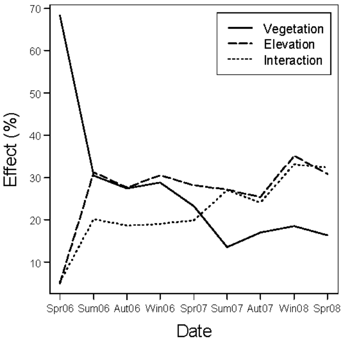

Figure 3.5. Temporal changes in elevation, remnant vegetation and interaction effects on community structure from spring 2006 (Spr06) to spring 2008 (Spr08) surveys. Effects are calculated by means of PCAIV for each season.

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.

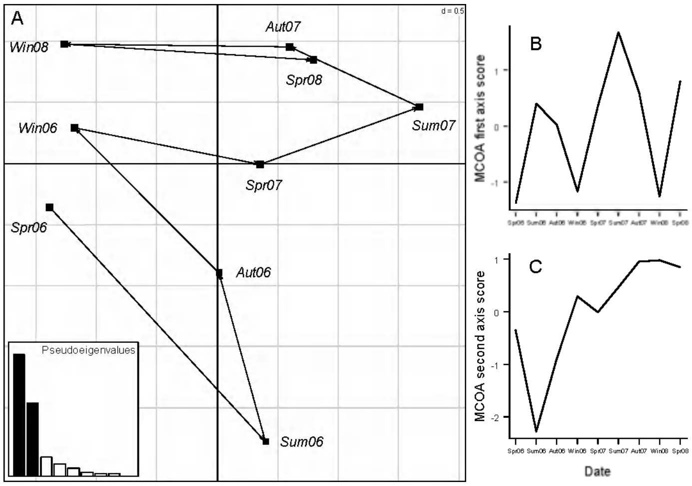

Figure 6. Multiple co-inertia analysis (MCOA). A) first factorial plane and pseudo-eigenvalue diagram; F1, 49 %; F2, 26 %. B) and C): MCOA axes scores from A vs. seasonal succession from spring 2006 (Spr06) to spring 2008 (Spr08).

3.5 Discussion

3.5.1 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.

3.5.2 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.

3.5.3 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.

3.6 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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Alphin, T. and Posey, M. 2000. Long-term trends in vegetation dominance and infaunal community composition in created marshes. *Wetlands Ecology and Management* **8**: 317–325.

Armstrong, W., Wright, E., Lythe, S. and Gaynard, T. 1985. Plant zonation and the effects of the spring–neap tidal cycle on soil aeration in a Humber salt marsh. *Journal of Ecology* **73**: 323–339.

Atkinson, P., Crooks, S., Grant, A. and Rehfisch, M. 2001. The success of creation and restoration schemes in producing intertidal habitat suitable for waterbirds. English Nature Research Report No. 425(425) Peterborough, UK.

Bady, P., Doledec, S., Dumont, B. and Fruget, J. 2004. Multiple co-inertia analysis: a tool for assessing synchrony in the temporal variability of aquatic communities. *Comptes Rendus Biologies* **327**:29-36.

Barne, J., Robson, C., Kaznowska, S., Doody, J. and Davidson, N. 1995. *Coasts and Seas of the United Kingdom, Region 6: Flamborough Head to Great Yarmouth.* Joint Nature Conservation Committee, Peterborough, UK.

Baty, F.; Facompré, M.; Wiegand, J.; Schwager, J. and Brutsche, M. H. 2006. Analysis with respect to instrumental variables for the exploration of microarray data structures. *BMC Bioinformatics* **7**: 422.

Bertness, M. and Hacker, S. 1994. Physical stress and positive associations among marsh plants. *American Naturalist* **144**: 363-372.

Bischoff, A. 2002. Dispersal and establishment of floodplain grassland species as limiting factors in restoration, *Biological Conservation* **104**: 25–33.

Bissels, S., Hölzel, N., Donath, T. and Otte, A. 2004. Evaluation of restoration success in alluvial grasslands under contrasting flooding regimes. *Biological Conservation* **118**: 641–650.

Blackwell, M. S. A., Hogan, D. V. and Maltby, E. 2004. The short-term impact of managed realignment on soil environmental variables and hydrology. *Estuarine, Coastal and Shelf Science* **594**: 687-701.

Boumans, R.M.J. and Day Jr., J.W. 1993. High precision measurements of sediment elevation in shallow coastal areas using a sediment-erosion table. *Estuaries* **16**: 375–380

De Boois, H. 1982. Veranderingen in het milieu en de vegetatie in de Biesbosch door afsluiting van het Haringvliet , Masters thesis, Wageningen.

Dray S., Chessel D. and Thioulouse J. 2003. Co-inertia analysis and the linking of ecological data tables. Ecology, **84:** 3078-3089.

Brooke, J. 1992. Coastal defence, the retreat option. *Journal of the Institute of Water and Environmental Management* **6**: 151–157.

Brooke, J. 1991. Retreat, the best form of defence? *Heritage Coast* **6**: 4.

Cahoon, D.R., Reed, D.J. and Day Jr., J.W. 1995. Estimating shallow subsidence in microtidal salt marshes of the southeastern United States: Kaye and Barghoorn revisited. *Marine Geol.* **128**: 1–9

Chessel, D. and Hanafi, M. 1996. Analyses de la co-inertie de K nuages de points. *Rev. Stat. Appl* **44**: 35–60.

Chessel, D., Dufour, A.B. and Thioulouse, J. 2004. : The ade4 package-I- One-table methods. R News. **4**: 5-10.

Childers, D. and Day, J. 1988. A flow-through flume technique for quantifying nutrient and material fluxes in microtidal estuaries. *Estuarine, Coastal and Shelf Science* **27**: 483–494.

Cooper, A. 1982. he effects of salinity and waterlogging on the growth and cation uptake of salt marsh plants. *New Phytologist* **90**: 263–275.

Cornu, C. and Sadro, S. 2002. Physical and functional responses to experimental marsh surface elevation manipulation in Coos Bays south Slough. *Restoration Ecology* **10**: 474–486.

Cox, T., Maris, T., Vleeschauwer, P. D., Mulder, T. D., Soetaert, K. and Meire, P. 2006. Flood control areas as an opportunity to restore estuarine habitat. *Ecological Engineering* **28**: 55-63.

Criel, B., Muylaert, W., Hoffmann, M., De Loose, L. and Meire, P.

1999. Vegetatiemodellering van de buitendijkse gebieden langs de zeeschelde. Rapporten van het instituut voor natuurbehoud 1999(12), Instituut voor Natuurbehoud, Brussel : Belgium .

Cundy, A., Long, A., Hill, C., Spencer, C. and Croudace, I. 2002. Sedimentary response of Pagham Harbour, southern England to barrier breaching in AD 1910., *Geomorphology* **1156**.

Dame, R. T., Chzranowski, T., Bildstein, K., Kjervfer, B., McKellar, H., Nelson, D., Spurrier, J., Stancyk, S., Stevenson, H., Vernberg, J. .. and Zingmark, R. 1986. The outwelling hypothesis and North Inlet, South Carolina. *Marine Ecology Progress Series* **33**: 217-229.

Dolédec, S. and Chessel, D. 1994. Co-inertia analysis: an alternative method for studying species-enVironment relationships, *Freshwater Biology* **31**: 277–294.

Dolédec, S. and Chessel, D. 1987. Rythmes saisonniers et composantes stationnelles en milieu aquatique I- Description d'un plan d'observations complet par projection de variables. *Acta Œcologica. Œcologia Generalis* **8**: 403-426.

Dolédec, S.; Phillips, M.; Scarsbrook, R.; Riley, R. and Townsend, C. 2006. Comparison of structural and functional approaches to determining landuse effects on grassland stream invertebrate communities. *Journal of the North American Benthological Society.* **25**: 44–60.

Dray, S., Chessel, D. and Thioulouse, J. 2003. Co-inertia analysis and the linking of the ecological data tables. *Ecology* **84**: 3078–3089.

Edwards, K. and Proffitt, C. 2003. Comparison of wetland structural characteristics between created and natural salt marshes in southwest Louisiana, USA. *Wetlands* **23**: 344–356.

Ellenberg, H. 1963. *Vegetation Mitteleuropas mit den Alpen*, Ulmer, Stuttgart.

EUNIS 2008. European Nature Information System, http://eunis.eea.europa.eu/index.jsp

French, P. W. 2006. Managed realignment - The developing story of a comparatively new approach to soft engineering. *Estuarine, Coastal and Shelf Science* **67**: 409-423.

Garbutt, R., Reading, C., Wolters, M., Gray, A. and Rothery, P. () Monitoring the development of intertidal habitats on former agricultural land after the managed realignment of coastal defences at Tollesbury, Essex, UK. *Marine Pollution Bulletin* **53**.

Gerard, M., El Kahloun, M., Rymen, J., Beauchard, O. and Meire, P. 2008. Importance of mowing and flood frequency in promoting species richness in restored floodplains. *Journal of Applied Ecology,* **45**: 1780–1789

Gribsholt, B., Boschker, H., Struyf, E., Andersson, M., Tramper, A., Brabandere, L. D., Damme, S. V., Brion, N., Meire, P., Dehairs, F., Middelburg, J. and Heip, C. 2005. Nitrogen processing in a tidal freshwater marsh: a whole ecosystem 15N labeling study. *Limnology Oceanography* **50**: 1945–1959.

Hirons, G., Kew, J., Smith, K. and Pilcher, R. 2002. Regulated Tidal Exchange - a Saline Sollution? Report of a Study Tour to USA to Determine the Potential of Regulated Tidal Exchange for Coastal Habitat creation, restoration and Management in the UK.Technical report, Royal Society for the Protection of Birds.

Howarth, R., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., Downing, J., Elmgren, R., Caraco, N., Jordan, T., Berendse, F., Freney, J., Kudeyarov, V., Murdoch, P. and Zhao-Liang, Z. 1996. Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. In: *Nitrogen cycling in the North Atlantic Ocean and its watersheds.*, Kluwer Academic Publ., Dordrecht, the Netherlands, 141–180.

Jacobs, S., Struyf, E., Maris, T. and Meire, P. 2008. Spatiotemporal aspects of silica buffering in restored tidal marshes. *Estuarine, Coastal and Shelf Science* **80**:42-52.

Lamberth, C. and Haycock, N. 2002. Regulated Tidal Exchange: An Intertidal Habitat Creation Technique, Technical report, Report by Haycock Associates Limited.

Lebreton, J. D., Sabatier, R., Banco G. and Bacou A. M. 1991. Principal component and correspondence analyses with respect to instrumental variables : an overview of their role in studies of structure-activity and species- environment relationships. In J. Devillers and W. Karcher, editors. *Applied Multivariate Analysis in SAR and Environmental Studies*, Kluwer Academic Publishers, 85–114

Levine, J., Brewer, J. and Bertness, M. 1998. Nutrients, competition and plant zonation in a New England salt marsh. *Journal of Ecology* **86**: 285-292.

Macleod, C., Scrimshaw, M., Emmerson, R., Chang, Y. and Lester, J. 1999. Geochemical changes in metal and nutrient loading at Orplands Farm managed retreat site, Essex, UK (April 1995–1997). *Marine Pollution Bulletin* **38**: 1115–1125.

Manly, B. 1991. Randomization and Monte Carlo methods in Biology. Chapman and Hall, London.

Maris, T., Cox, T., Temmerman, S., De Vleeschauwer, P., Van Damme, S., De Mulder, T., Van den Bergh, E. and Meire, P. 2007. Tuning the tide: creating ecological conditions for tidal marsh development in a flood control area. *Hydrobiologia* **588**: 31-34.

Massart, J. 1908. Review of the botanical biogeography of the littoral and alluvial districts of Belgium, Rec. Inst. Bot. Léo Errera.

Meire, P., Ysebaert, T., Damme, S. V., den Bergh, E. V., Maris, T. and Struyf, E. 2005. The Scheldt estuary: a description of a changing ecosystem. *Hydrobiologia* **540**: 1–11.

Morris, R., Reach, I., Duffy, M., Collins, T. and Leafe, R. 2004. On the loss of salt marshes in south-east England and the relationship with Nereis diversicolor. *Journal of Applied Ecology* **41**: 787–791.

Möller, I. and Spencer, T. 2002. Wave dissipation over macro-tidal saltmarshes: effects of marsh edge typology and vegetation change. *Journal of Coastal Research* **36**: 506–521.

Paillex, A.; Castella, E. and Carron, G. 2007. Aquatic macroinvertebrate response along a gradient of lateralconnectivity in river floodplain channels. *Journal of the North American Benthological Society* **4**: 779–796.

Perry, J., Barnard, T., Bradshaw, J., Friedrichs, C., Havens, K., Mason, P., Priest, W. and Silberhorn, G. 2001. Creating tidal salt marshes in the Chesapeake Bay. *Journal of Coastal Research* **27**: 170–191.

Quammen, M. (Sept–Oct 1986) Measuring the success of wetland mitigation. *National Wetlands Newsletter*, 6–8.

R 2008. The R foundation for statistical computing, *http://www.R-project.org/* **[2.7.2]**.

Robert P. and Escoufier Y. 1976. A unifying tool for linear statistical methods: the RV-coefficient, *Appl. Stat.* **25**: 257-265.

Roman, C. T., Raposa, K. B., Adamowicz, S. C., James-Pirri, M. and Catena, J. G. 2002. Quantifying Vegetation and Nekton Response to Tidal Restoration of a New England Salt Marsh. *Restoration Ecology* **103**: 450-460.

Sabatier, R., Lebreton, J. and Chessel, D. 1989. *Multiway data analysis.*, Elsevier Science Publishers B.V., North-Holland, chapter Principal component analysis with instrumental variables as a tool for modelling composition data. 341–352.

Snow, A. A. and Vince, S. W. 1984. Plant zonation in an Alaskan salt marsh. II. An experimental study of the role of edaphic conditions. *Journal of Ecology* **72**: 669-684.

Soetaert, K.; Middelburg, J.; Heip, C.; Meire, P.; Van Damme, S. and Maris, T. 2006. , 'Long-term change in dissolved inorganic nutrients in the heterotrophic Scheldt estuary (Belgium, The Netherlands)', *Limnology and Oceanography* **51**, 409-423.

Spurrier, J. and Kjerfve, B. 1988. Estimating the net flux of nutrients between a salt marsh and a tidal creek. *Estuaries* **11**: 10–14.

Struyf Eric, Conley Daniel J. Silica: an essential nutrient in wetland biogeochemistry.- In: Frontiers in ecology and the environment, 7:2(2009), p. 88-94

Struyf, E., Damme, S. V., Gribsholt, B., Bal, K., Beauchard, O., ack J. Middelburg and Meire, P. 2007. Phragmites australis and silica cycling in tidal wetlands. *Aquatic Botany* **87**: 134-140.

Struyf, E., Dausse, A., Van Damme, S., Bal, K., Gribsholt, B., Boschker, H., Middelburg, J. and Meire, P. 2006. Tidal marshes and biogenic silica recycling at the land-sea interface. *Limnology and Oceanography* **51**: 838-846.

Struyf E., S. Jacobs, P. Meire, K. Jensen & A. Barendregt (2009) Plant communities of European tidal freshwater wetlands. In: Barendregt A., D.F. Whigham, A.H. Baldwin (Eds) Tidal Freshwater Wetlands, Backhuys Publishers, Leiden, The Netherlands. p 59-70.

Temmerman, S., Govers, G., Meire, P. and Wartel, S. 2003. Modelling long-term tidal marsh growth under changing tidal conditions and suspended sediment concentrations, Scheldt estuary, Belgium. *Marine Geology* **193**: 151-169

Thioulouse, J., Simier, M., Chessel, D. 2004. Simultanous analysis of a sequence of paired ecological tables. *Ecology* **85**: 272-283

Ungar, I. 1998. Are biotic factors significant in influencing the distribution of halophytes in saline habitats? *Botanical Review* **64**: 176-199.

Weinstein, M. 2007. Ecological restoration and estuarine management: placing people in a coastal landscape. *Journal of Applied Ecology* **4**: 1-9.

Wiehe, P. 1935. A quantitative study of the influence of tide upon the population of Salicornia europaea. *Journal of Ecology* **23**: 323–333.

Williams, P. and Orr, M. 2002. Physical development of restored breached levee salt marshes in the San Francisco Bay estuary. *Restoration Ecology* **10**: 527–542.

Wollast, R. 1988. ‚The Scheldt estuary, *in* W. Salomon; W.L. Bayne; E.K. Duursma and U. Forstner, ed.,'Pollution of the North-Sea: an assessment', Springer-Verlag, Berlin, pp. 183-193.

Wolters, M., Bakker, J. P., Bertness, M. D., Jefferies, R. L. and Möller, I.

2005. Saltmarsh erosion and restoration in south-east England: squeezing the evidence requires realignment, *Journal of Applied Ecology* **42**: 844-851.

Wolters, M., Garbutt, A. and Bakker, J. (2005a) Plant colonization after managed realignment: the relative importance of diaspore dispersal. *Journal of Applied Ecology* **42**: 770–777.

Wolters, M., Garbutt, A. and Bakker, J. P. (2005b) Salt-marsh restoration: evaluating the success of de-embankments in north-west Europe. *Biological Conservation* **123**: 249--268.

Wolters, M., Garbutt, A., Bekker, R. M., Bakker, J. P. and Carey, P. D. 2008. Restoration of salt-marsh vegetation in relation to site suitability, species pool and dispersal traits. *Journal of Applied Ecology* **45**: 904-912.

Zonneveld, I.S. 1960. De Brabantse Biesbosch : een studie van bodem en vegetatie van een zoetwatergetijdendelta.

Chapter 4

The silica pool in freshwater tidal marsh vegetations

Jacobs S., Teuchies J., Struyf E., Meire P.

4.1 Abstract

In estuaries, the last ecosystem in the land-sea transport of nutrients, dissolved silica (DSi) is taken up seasonally by benthic and pelagic diatom communities and by estuarine marsh vegetation. Delivery of DSi back to the estuary potentially enhances the resilience of the estuary against harmful effects of silica depletion events. However, the magnitude of the Si-fluxes in marsh ecosystems is still not well established because published studies on biomass including Si data are scarce. Freshwater tidal marsh vegetations, often mainly *Phragmites australis*, are thought to contribute to the silica buffering function of tidal marshes through enhanced DSi-export by dissolution of plant-bound BSi into the pore water and consequent seepage to the estuary.

In this study, a silica pool estimation was realised for an entire restored freshwater tidal marsh vegetation, using species-based cover-biomass relationships and silica analyses. Our estimate (7 to 45 kg BSi/ha) confirms that the silica pool in the vegetation of the studied marsh is growing, as observed colonisation by *Phragmites australis* will increase this pool.

The largest release of DSi from vegetation is expected to occur in early winter, while estuarine silica depletion events occur mostly during algal blooms in summer. Although the total amount of silica in freshwater tidal marsh vegetations largely exceeds amounts in other vegetations, this means there is a potential time gap between release of DSi and occurrence of silica depletion. Our results do however indicate that young freshwater marshes have a high potential to quickly build-up a large BSi stock with a few silica-accumulating species.

4.2 Introduction

The ultimate source of all continental Si fluxes to the oceans are weathering processes in terrestrial biogeosystems. Still, this silica is transferred between various pools (mineral, organo-mineral, biogenic) before finally entering the ocean. In general, knowledge about size, transformations and properties of the amorphous silica (ASi) pools is scarce (Sommer et al. 2006).

In estuaries, which form the last filter in the land-sea transport, dissolved silica is taken up seasonally by benthic and pelagic diatom communities (Wollast and De Broeu 1971), and by estuarine marsh vegetation (Struyf et al. 2005).

Tidal marshes form a silica sink, accreting amorphous silica (ASi) in sediments by deposition of diatoms and plant debris on the marsh floor (Struyf et al. 2007a, see Chapter 3). Part of this silica is dissolved again into the pore water, and available for uptake by plants and diatoms. Dissolved silica (DSi), taken up by marsh plants from pore waters, is stored as plant-bound amorphous silica or biogenic silica (BSi). It is accumulated during the plant life cycle and only released to the environment after the plant decomposes (Struyf et al. 2005).

DSi is also delivered directly to the estuary by seepage from marshes in between tidal floods (Struyf et al. 2006). As such, tidal marshes potentially buffer estuarine silica depletion events. These events are characterised by decreased silica concentrations in the river compared to the Redfield ratio (C:Si:N:P = 106:15:16:1). Depletion events can cause shifts in algal communities towards non-diatom dominance (Kilham 1971, Conley et al. 1993) and can have harmful effects on coastal food web structure and energy dissipation to higher trophic levels (Ragueneau et al. 2006). Silica delivery from tidal marshes during such events might enhance the resilience of the estuary to silica depletion. Knowledge about the size and mobility of silica pools in tidal marshes is thus essential.

The magnitude of the Si-fluxes in ecosystems is still not well established because published studies on biomass including Si data are scarce. An overview is given in Conley et al. (2006). Largest fluxes are found in ecosystems with highest productivity (e.g. Lucas et al. 1993), and exceptionally large values are observed in vegetations consisting of silica-accumulators like rice, sugar cane (Berthelsen et al. 2001) or reeds (Struyf et al. 2005). Freshwater tidal marsh vegetations, often mainly *Phragmites australis,* are thought to contribute to the silica buffering function of tidal marshes through enhanced DSi-export by dissolution of BSi in the pore water and consequent seepage to the estuary (Struyf et al. 2007b).

In this study, a whole-vegetation silica pool estimation was realised for an entire freshwater tidal marsh. Therefore, species-specific biomass estimates and silica content analyses were performed over two years, each time at the beginning as well as at the end of the growing season.

4.3 Methods

4.3.1 Site description

The Schelde estuary 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. The study area is an experimental site where controlled reduced tide (CRT) was implemented (Maris et al. 2007, Cox et al. 2006). Its surface is approximately 80.000 m^2, and it is situated at Hamme, Belgium (51°05'10"N; 4°10'20"E). Mean tidal amplitude at springtide in the Schelde at this point is about 6m. The site is situated in the freshwater tidal zone of the estuary, with an averaged specific conductivity ranging from 1000 (winter) to 1500 µS.cm-1 (summer). The elevation difference between the contemporary tidal marshes and the pilot site is 2.5 to 3 m. Tidal inundation was restored by means of CRT in March 2006. Restoring intertidal habitat with a controlled reduced tide is based on detaching the site's elevation and the tidal regime it receives: a high inlet culvert in the dike only allows the top of the tidal wave to enter the site. The ebb phase only starts after the water level in the river 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 polder is flooded, while at neap tide, only the central creek fills. Detailed technical explications can be found in Maris et al. 2007 and Cox et al. 2006. After two years of controlled reduced tide, typical tidal freshwater habitat has been restored in a former agricultural site (see Chapter 3). Establishment of an intertidal plant community in the CRT was quick, with fast eradication of terrestrial species and colonisation of reference species already during the first few months, and the specific features of the CRT provide shelter for communities which have disappeared along the degraded Schelde, but are described for several European references (see Chapter 3). Also, several silica delivery events were observed during estuarine silica limitation (Jacobs et al. 2008) during the first year of controlled reduced tidal (CRT) regime.

4.3.2 Sampling plan

Vegetation mapping was performed at the beginning of the growing season (April) and at the end (October) in 2007 and 2008. The Lippenbroek site was divided into polygons characterised by homogenous vegetations (Fig. 4.1). In each polygon, dominant species' cover (dominant species defined as species with at least 20% cover) was noted (**C**). Cover was defined as the total leaf area relative to the considered surface (in %). This avoids truncated relationships obtained with traditional cover estimates which level off at 100% (projected leaf area).

Aboveground biomass estimations were based on a concurrent sampling campaign. For all mapped species, cover was estimated in 2500 cm^2 plots and separately harvested and weighed. This was repeated until at least five replicates per campaign were obtained (approx. 60 to 80 harvested plots every campaign). These data allowed constructing cover-biomass relationships per species. Replicates were located throughout the marsh; cover estimates in the 2500 cm2 plots ranged within the mapped polygon values. Plant samples were dried for 48 hours at 70°C and weighed.

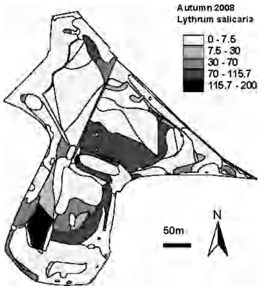

Fig. 4.1: Example map for the autumn 2008 inventory, with cover of *Lythrum salicaria* represented.

Silica content (**S**) of the plant species was measured in a third simultaneous campaign. Individual plants were harvested, dried for two days at 70°C, weighed, mixed and analysed. BSi content was analyzed on a Thermo IRIS ICP (Inductively Coupled Plasmaspectrophotometer) (Iris®) after sequential alkaline extraction for 4 hours in 1M Na_2CO_3 at 80°C. Of each mapped species, 5 replicate individuals were analysed every campaign. They were harvested throughout the marsh and covered a wide range of plant sizes.

Every campaign, 60 to 80 plots and individuals were harvested for biomass evaluation and silica analysis respectively. The last step of the analysis consisted of per species silica pool estimation (1).

$$\mathbf{E_{sp}=[\Sigma_{1\rightarrow n}\ (C.A)\ .f_{sp}.\ S_{sp}]} \qquad (1)$$

Where **C** is the estimated cover of a species in a certain polygon, **A** the surface area of the polygon, **f** the conversion factor to estimate biomass per percent cover, **n** the number of polygons where the species was noted, and **S** the silica content (mg/g) of the concerning species. The lower and upper range was calculated accordingly, using the 95% confidence intervals of the f-values multiplied by the standard deviations S-values (2):

$$\begin{gathered}\mathbf{[\Sigma_{1}\rightarrow_{n}(C.A_{)})\ .f_{sp(0.25)}.\ (S_{sp}\text{-}sd(S_{sp}))]} \\ \mathbf{< E_{sp} <} \\ \mathbf{[\Sigma_{1}\rightarrow_{n}(C.A)\ .f_{sp(0.95)}.\ (Ssp+sd(Ssp))]}\end{gathered} \qquad (2)$$

4.4 Results

4.4.1 Vegetation mapping

Only a few species were dominant in the polygons (Fig 4.2). Dominance shifted from more terrestrial species (e.g *Urtica dioica*. *Rumex aquatica*) to typical tidal marsh species (e.g *Lythrum salicaria*. *Phragmites australis*), although *Epilobium hirsutum* remains important (Fig. 4.2).

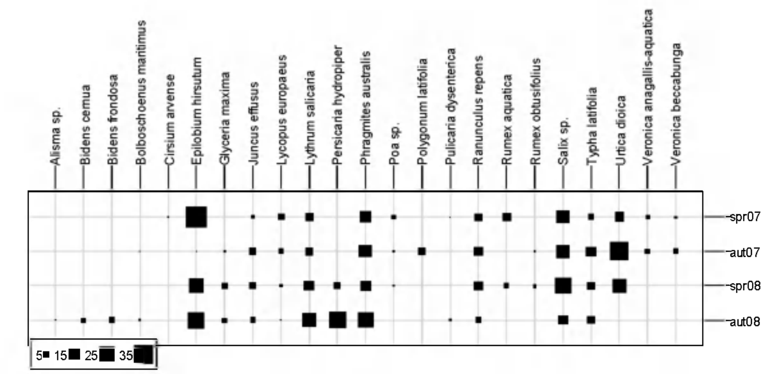

Fig. 4.2: Polygon vegetation mapping: dominant species per season. Abundance -as the sum of the polygon cover.polygon surface products (Σ(C.A)) for every species- was divided by the seasonal sum for all species and thus represents the percent contribution of the species to the vegetation of the season.

4.4.2 Cover-biomass relationships

A linear model for cover and biomass for every species was obtained with cover and dry weight of the 2500cm^2 plots. The model always includes the origin (0,0), to avoid illogic intercepts (e.g. zero cover with positive or negative mass). Per season, at least 5 replicates per species were available, but correlations improved by pooling all four seasons, increasing the number of replicates used in the cover-biomass relationship. Mean R-squared value of the models was 0.813 (range 0.357 - 0.996). 95% confidence intervals for the linear slopes were calculated to provide a realistic estimation range per species (Fig. 4.3 & 4.4).

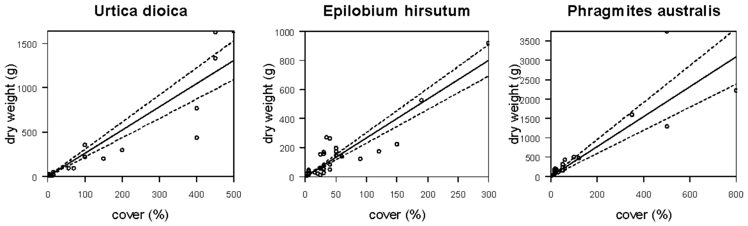

Fig. 4.3: Linear cover-biomass model (forced to intercept 0,0) for three species. R squared values for *Urtica dioica*, *Epilobium hirsutum* and *Phragmites australis* were 0.891, 0.886 and 0.809 respectively. 95% confidence intervals for the model slopes are represented in dashed lines, the model itself in solid lines.

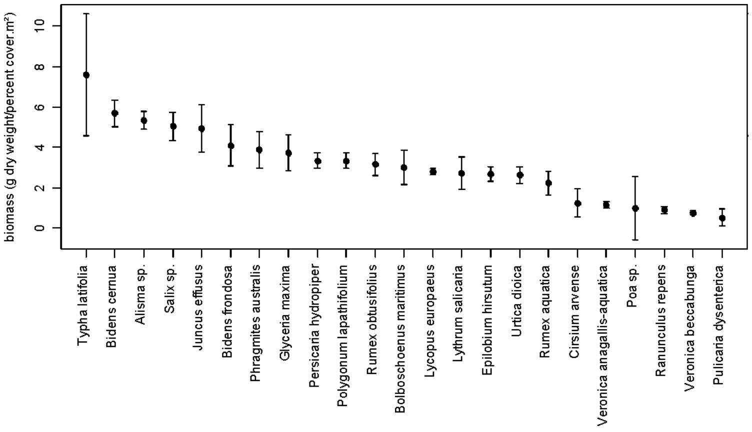

Fig. 4.4: Modelled cover-biomass slopes for all species (see Fig xxx). Error bars represent 95% confidence interval ranges for the modelled slope.

4.4.3 Silica content per species

Some species do not accumulate silica, and variation in silica content in these species was generally low (e.g. *Plantago major*, Fig. 4.5). *Phragmites australis* had the highest silica content, as well as the highest individual variation. Other accumulating species are mostly monocots, with some exceptional dicots like *Symphytum officinalis* and *Urtica dioica* (Fig. 4.5).
BSi contents in marsh plants were comparable to other studies (Struyf et al. 2005; Struyf & Conley 2009)

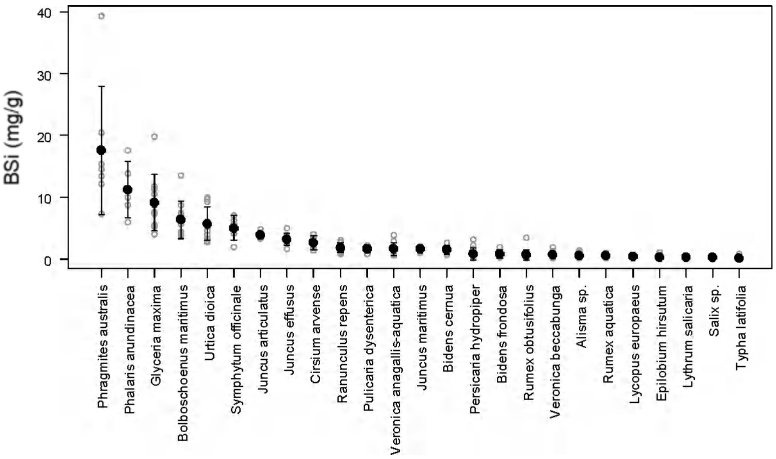

Fig. 4.5: BSi content (mg/g) of the sampled marsh species. Error bars represent standard deviation.

4.4.4 BSi pool in the entire marsh vegetation

Estimation of the total silica pool (summed biomass multiplied by BSi content, see methods), despite the large range, depicts a clear increase over the four campaigns (Fig. 4.6). *Phragmites australis* is clearly the biggest contributor to the total BSi pool (Fig. 4.7). During the first growing season (season 1-2 on Fig. 3.6) the BSi pool increased due to the bigger contribution of *Urtica* and *Juncus effusus* (Fig. 4.7). During the winter (season 2-3 on Fig. 4.6), the BSi pool remained the same, as the loss of *Urtica* and *Phragmites australis* was compensated by the increasing contribution of *Glyceria maxima* (Fig. 4.7). During the second growing season (season 3-4 on Fig. 4.6), the increase of the BSi pool can be attributed to the increase of *Phragmites australis* biomass contribution (Fig. 4.7).

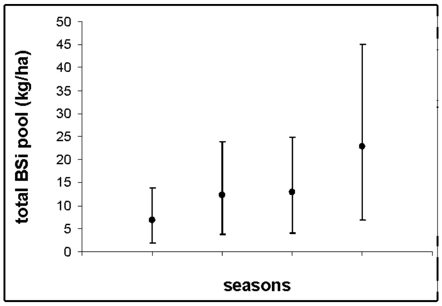

Fig. 4.6: Total biogenic silica pool in the marsh vegetation (kg/ha). Error bars represent minimal and maximal estimates, based on 95% confidence intervals of cover-biomass estimates and standard deviations of species' silica contents.

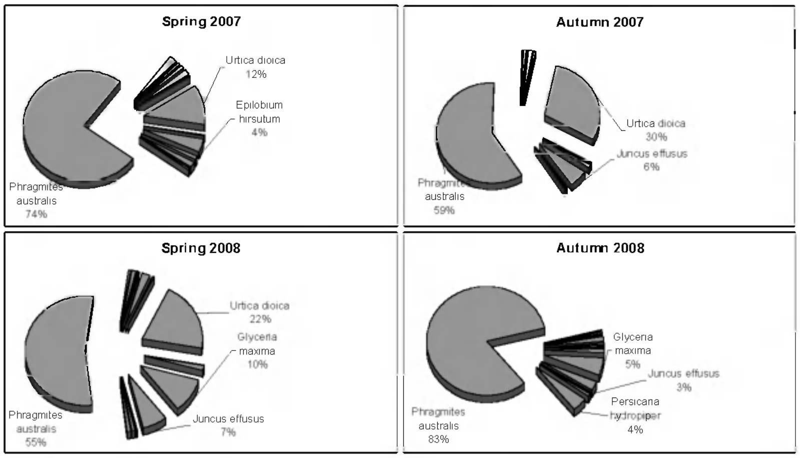

Fig. 4.7: Contributions of sampled species to total biogenic silica pool in the marsh vegetation for the four sampling campaigns. Species with contributions from 3% on are indicated.

4.5 Discussion

The cover-biomass method is superior to other techniques when a detailed species or life-form response is necessary (Flombaum and Sala 2007). In our case, an estimated biomass per species was obtained, and we assume it provided a direct and credible estimate for the entire vegetation. The high explained variability in our cover-biomass relationships indicate that our technique is suitable for the sampled high-productive ecosystem, although it required intensive in situ sampling. An estimation of cover as total leaf area instead of projected leaf area had to be conducted, and cross-checks for estimates by different observers had to be carried out. The choice of linearity and forcing through the origin of the models are supported by high explained variability.

The cover-biomass relationship is a technique seldom used. Mostly, allometric equations (which typically differ among life forms), scaled with density to obtain ecosystem-level biomass, are used (e.g. Adler et al. 2004; Johnson et al. 1988; Návar et al. 2004, Rittenhouse and Sneva 1977, Williamson et al. 1987). In the variable freshwater tidal habitat, different plant covers at similar densities for one same species are encountered. This rules out density as a stable biomass proxy for this habitat.

General floristic evolution was consistent with the findings in chapter 2. However, the four campaigns show some differences due to seasonal variations (e.g. the absence of *Epilobium hirsutum* in autumn 2007, when this species died off earlier than in 2008, while the opposite is true for *Urtica dioica*). Since no seasonal variations were found in biomass-cover relationships or silica content, one yearly campaign at peak biomass would suffice. To cope with between-year variations, a seasonal proxy could be used to time this campaign (e.g. the start of seed dispersion of *Lythrum salicaria*).

The silica content of the marsh species corroborates well with earlier results (Struyf et al. 2005; Struyf & Conley 2009). Next to *Phragmites australis*, some other monocots accumulate silica. The dicots which have higher silica contents seem to have in common that they have hardened hair structures on the leafs (e.g *Urtica dioica*, *Symphytum officinale*). The low silica content of *Typha latifolia* also corroborates with earlier findings. For characterisation of the total plant silica pool, monitoring of only a few silica accumulating species would suffice, since other species contain only a minimal part of the total silica found in the vegetation (Fig. 4.7).

In an earlier study, the freshwater tidal marsh vegetation contained about 227±61 kg BSi/ha at peak biomass, of which 96% in reed dominated vegetations (Struyf et al. 2005). Our estimate (7 to 45 kg BSi/ha) confirms that the silica pool in the vegetation is still far under its potential maximum, observed in reed dominated semi-natural tidal marshes. Indeed, the polygons where dense reed vegetation was found only covered part of the surface, although their abundance score was high due to the high productivity of these reed blocks. Reeds are quickly colonising the site, and it can be expected that this BSi-pool will increase (Fig. 4.8).

Fig. 4.8: Photograph of the same location in 2006 (4 months after restoring tidal influence) and 2008. Note that the reed patches have colonised a major part of the surface.

It must be noted that the development of a more diverse helophyte community, as observed in the CRT (see Chapter 3) could have an impact on the final total silica pooled in the vegetation. The competitive outcome of *Phragmites australis* versus *Typha latifolia* colonisation impacts the amounts of silica pooled in freshwater tidal marsh vegetations. The total amount of silica stocked in vegetations is thus potentially influenced by its composition and management. In many North American marshes for instance, colonization of *Typha*-dominated marshes with *Phragmites australis* could impact silica cycling in these wetlands. In Europe, management practices such as willow-cutting and mowing to promote reed growth could have an important effect on the amount of silica stocked in the marsh, while succession to *Salix* softwoods on the contrary could decrease the BSi pool. Also, recolonization of lower marshes by silica accumulators like *Phalaris arundinaceae*, *Glyceria maxima* and *Bolboschoenus maritimus* -whose populations along the Schelde declined during the last century- could have an impact on the total amount of BSi stocked in estuarine vegetation.

Struyf et al. (2007b) showed that over 98% of the silica in *Phragmites australis* litter dissolved within one year. The contribution of vegetation-BSi on DSi-export from marshes is thus potentially important: Struyf et al. (2007) estimated a DSi-export of 213±57 kg/ha/y for freshwater tidal marshes, which largely exceeds maximal values found in literature (e.g. Alexandre et al. 1997: 54 ± 5 to 70 ± 7 kg/ha/y for rain forest on latisol in Congo). A marsh with a reactive silica pool of 250 kg/ha BSi in its yearly vegetation potentially releases an extra 0.34 kg Si/ha/tidal cycle into the estuary. In the case of our study site, this would mean 2.72 kg Si/tide. This lies within the orders of magnitude already measured during the tidal exchange campaigns (Jacobs et al. 2008), meaning that further development of silica-accumulating vegetation could potentially increase DSi-delivery from the pilot site.

The question is *when* these DSi-release take place: most vegetation biomass decays during autumn and early winter (Fig. 4.9), while the dissolution process of plant BSi to DSi occurs fast, continuous and independent of the season (Struyf et al. 2007, 2007b).

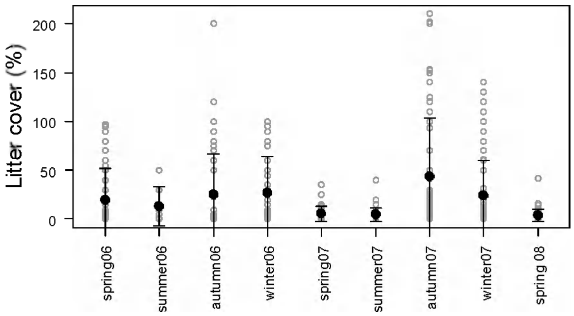

Fig. 4.9: Litter cover in 70 permanent plots throughout the study site, recorded every three months during the first two years. Cover was estimated as percent of litter surface compared to the plot surface. Relatively high spring values in 2006 are due to terrestrial vegetation die-back after introduction of the tidal influence (see Chapter 3).

This could mean that the release of litter DSi peaks in early winter. This peak release would not coincide with estuarine silica depletion events, which occur mostly during algal blooms in summer. Still, a year round litter layer remains throughout the year. Release of DSi from the litter could therefore still contribute to DSi export year-round. The study of year-round DSi export from the marsh would require a detailed and continuous study of Si release from marsh litter and total Si pool in marsh litter, concurrent with marsh-estuary exchange experiments.

Our research has determined the size of the reactive silica pool in the vegetation of a developing freshwater tidal marsh. Changes in dominance of single Si-acumulating species, whether they are caused by succesion or management practice, could severely influence the amount of silica stored in ecosystems. The silica in vegetation remains stocked during the growing season. This might mean that DSi flux from litter dissolution peaks at the end of the growing season, which would not coincide with summer silica-depletion events. This raises some doubts about the actual contribution of vegetation BSi pool to the buffering of these harmful events.

Adler, P., Milchunas, D., Lauenroth, W., Sala, O. and Burke, I. Functional traits of graminoids in semi-arid steppes: a test of grazing histories, *Journal of Applied Ecology* (41), 2004, pp. 653-663.

Alexandre, A., Meunier, J., Colin, F. and Koud, J. Plant impact on the biogeochemical cycle of silicon and reversed weathering processes, *Geochimica and Cosmochimica Acta* (61), 1997, pp. 677-682.

Berthelsen, S., Noble, A. and A.L., G. Silicon research down under: Past, present and future, *in* Datnoff, L., G.H., S. and G.H., K., ed.,Silicon in agriculture, Amsterdam: Elsevier, 2001, pp. 241-256.

Conley, D., Schelske, C. and Stoermer, E. modification of silica biogeochemistry with eurtrophication in aquatic systems, *Marine Ecology Progress Series* (101), 1993, pp. 179-192.

Conley, D., Sommer, M., Meunier, J., Kaczorek, D. and Saccone, L. Silicon in the terrestrial biogeosphere, *in* Ittekot, V., Unger, D., Humborg, C. and Tac An, N., ed.,The Silicon Cycle. Scope series 66, Islandpress Washington, 2006, pp. 13-28.

Cox, T., Maris, T., Vleeschauwer, P. D., Mulder, T. D., Soetaert, K. and Meire, P. Flood control areas as an opportunity to restore estuarine habitat, *Ecological Engineering* (28:1), 2006, pp. 55-63.

Flombaum, P. and Sala, O. A non-destructive and rapid method to estimate biomass and aboveground net primary production in arid environments, *Journal of Arid Environments* (69), 2007, pp. 352-358.

Jacobs, S., Struyf, E., Maris, T. and Meire, P. Spatiotemporal aspects of silica buffering in restored tidal marshes, *Estuarine, Coastal and Shelf Science* (80), 2008, pp. 42-52.

Johnson, P.S.and Johnson, C. and West, N. Estimation of phytomass for ungrazed crested wheatgrass plants using allometric equations, *Journal of Range Management* (41), 1988, pp. 421-425.

Kilham, P. A hypothesis concerning silica and the freshwater planktonic diatoms, *Limnology and Oceanography* (16), 1971, pp. 10-18.

Lucas, Y., Luizao, F., Chauvel, A., Rouiller, J. and Nahon, D. The relation between biological activity of rainforests and mineral composition of soils, *Science* (260), 1993, pp. 521-523.

Maris, T., Cox, T., Temmerman, S., De Vleeschauwer, P., Van Damme, S., De Mulder, T., Van den Bergh, E. and Meire, P. Tuning the tide: creating ecological conditions for tidal marsh development in a flood control area, *Hydrobiologia* (588), 2007, pp. 31-34.

Meire, P., Ysebaert, T., Damme, S. V., den Bergh, E. V., Maris, T. and Struyf, E. The Scheldt estuary: a description of a changing ecosystem, *Hydrobiologia* (540), 2005, pp. 1–11.

Návar, J., Méndez, E., Nájera, A., Graciano, J., Dale, V. and Parresol, B. Biomass equations for shrubs species of Tamaulipan thornscrub of North-eastern Mexico, *Journal of Arid Environments59* (59), 2004, pp. 657-674.

Ragueneau, O., Conley, D., Leynaert, A., Ni Longphuirts, S. and Slomp, C. Role of diatoms in Silicon Cycling and Coastal Marine Foodwebs, *in* Ittekot, V., Unger, D., Humborg, C. and Tac Ann, N., ed.,the silicon cycle. Scope series 66, Islandpress Washington, 2006.

Rittenhouse, L.-R.and Sneva, F.-A. A technique for estimating big sagebrush (Artemisia tridentata) production, *Journal of Range Management* (30), 1977, pp. 68-70.

Soetaert, K., Middelburg, J., Heip, C., Meire, P., Van Damme, S. and Maris, T. Long-term change in dissolved inorganic nutrients in the heterotrophic Scheldt estuary (Belgium, The Netherlands), *Limnology and Oceanography* (51), 2006, pp. 409-423.

Sommer, M., Kaczorek, D., Kuzyakov, Y. and Breuer, J. Silicon pools and fluxes in soils and landscapes: A review, *Journal of Plant Nutrition and Soil Science* (169), 2006, pp. 294-314.

Struyf, E. and Conley, D. Silica: an essential nutrient in wetland biogeochemistry., *Frontiers in Ecology and the Environment* (7), 2009, pp. 88-94.

Struyf, E., Damme, S. V., Gribsholt, B., Middelburg, J. and Meire, P. Biogenic silica in tidal freshwater marsh sediments and vegetation (Schelde estuary, Belgium), *Marine Ecology Progress Series* (303), 2005, pp. 51–60.

Struyf, E., Dausse, A., Damme, S. V., Bal, K., Gribsholt, B., Boschker, H., Middelburg, J. and Meire, P. Tidal marshes and biogenic silica recycling at the land-sea interface, *Limnology and Oceanography* (51), 2006, pp. 838–846.

Struyf, E., Temmerman, S. and Meire, P. Dynamics of biogenic Si in freshwater tidal marshes, Si regeneration and retention in marsh sediments (Scheldt estuary), *Biogeochemistry* (82), 2007a, pp. 41–53.

Struyf, E., Van Damme, S., Gribsholt, B., Bal, K., Beauchard, O., Middelburg, J. and Meire, P. Phragmites australis and silica cycling in tidal wetlands, *Aquatic Botany* (87), 2007b, pp. 134-140.

Williamson, S., Detling, J., Dodd, J. and Dyer, M. Nondestructive estimation of shortgrass aerial biomass, *Journal of Range Management* (40), 1987, pp. 254–256.

Wollast, R. The Scheldt estuary, *in* Salomon, W., Bayne, W., Duursma, E. and Forstner, U., ed.,Pollution of the North-Sea: an assessment, Berlin: Springer Verlag, 1988, pp. 183-193.

Wollast, R. and De Broeu, F. Study of the behaviour od dissolved silica in the estuary of the Scheldt, *Geochimica and Cosmochimica Acta* (35), 1971, pp. 613-620.

Chapter 5

Amorphous and dissolved silica patterns in freshwater tidal marshes

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5.1 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.

5.2 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.

5.3 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. 5.1): 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).

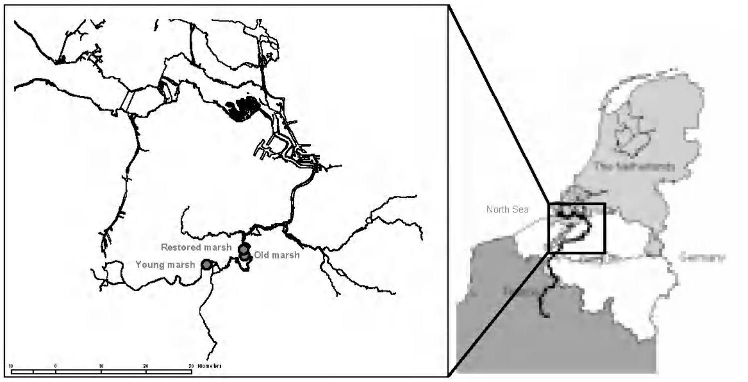

Fig. 5.1: Schematic map of the Schelde estuary, with approximate position of the sampling sites.

The reference sampling locations consisted of homogenous habitats (reed beds, climax willow forests, annual forbs and non-vegetated mudflats) (Fig. 5.2). In the restored site, climax vegetation did not yet develop: five locations along the elevation gradient were sampled (Fig. 5.2).

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 analysed separately, adding a fourth layer. From June, location 9 was added to the restored site sampling plan to represent the emerged tidal lake in the sampling plan, not yet developed in April 2006 (Tab. 5.1). Statistical analysis considered the three layers – four seasons sampling schedule: the fourth layer was used for graphical discussion.

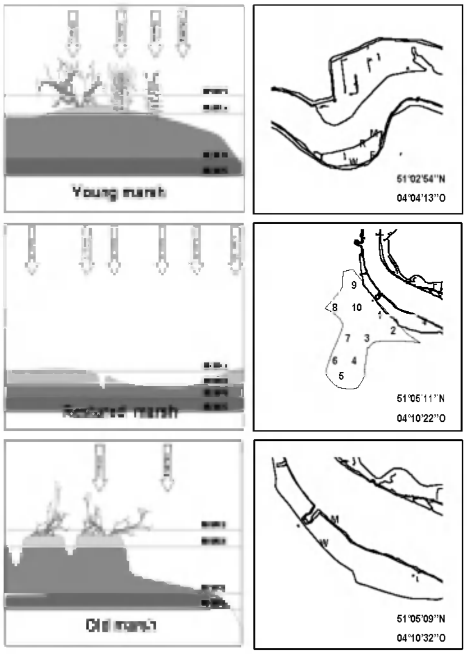

Fig. 5.2: scheme of sampling locations and habitats per site, with elevation relative to tidal levels (MHW Mean High Water, MLW Mean Low Water, S = Spring tide, N = Neap tide, M = mudflat, R = reeds, F = forbs, W = willow)

Pore water was sampled every season with 15 cm porous rhizons (Eykelkamp) 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 5.1: 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, homogenised 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 two hours, 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 analysed on a Thermo IRIS ICP.

5.3.1 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. multidimentional 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 (Chambers 1992) were used for DSi data exploration. The posthoc procedure used is referred to as Tukey Honest Significant Differences (TukeyHSD; Miller 1981; Yandell 1997).

5.4 Results

5.4.1 ASi contents

As a general pattern, ASi contents were more variable in the surface layer, which also contained maximal observed ASi contents (Fig. 5.3). A more scattered pattern was observed deeper down. Individual depth profiles varied strongly between seasons and habitats, with no consequent depth patterns in the deeper layers (Fig. 5.3).

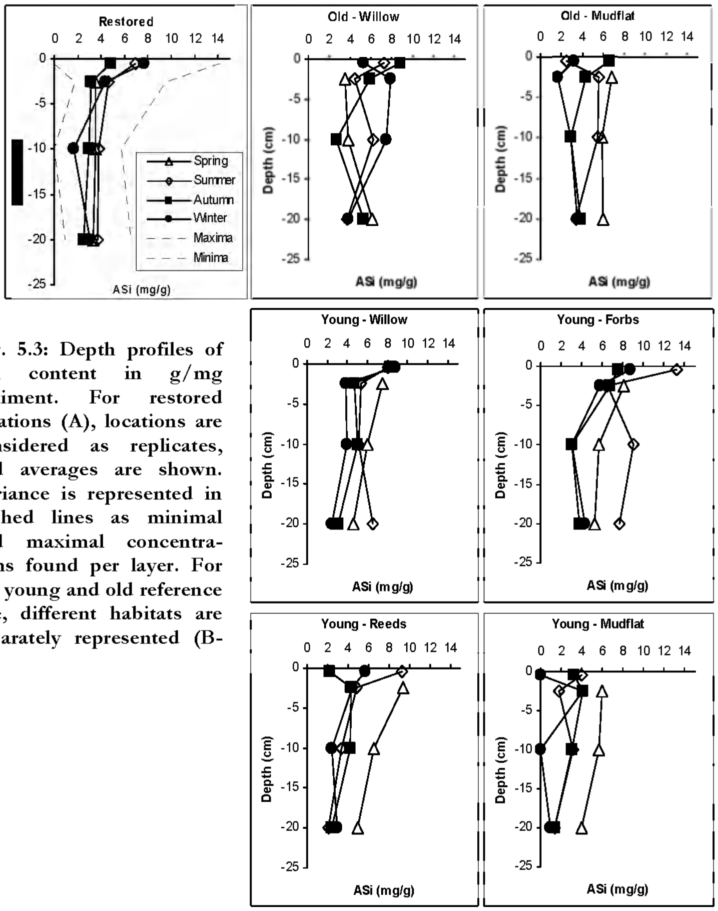

Fig. 5.3: Depth profiles of ASi content in g/mg sediment. For restored locations (A), locations are considered as replicates, and averages are shown. Variance is represented in dashed lines as minimal and maximal concentrations found per layer. For the young and old reference site, different habitats are separately represented (B-G).

PCA revealed three main axes (Fig. 5.4) 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 the first axis (x-axis). This results in a gradual increase in ASi content from mudflats to reeds, willow and forbs, with highest ASi concentrations observed in the vegetated reference sites (Fig. 5.4B). The restored locations are ranging over this whole gradient, but are bended towards the lower intertidal habitats. The second axis, opposing winter to spring (Fig 5.4A) 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. 5.4C, Fig. 5.3). The second axis opposes frequently flooded (mudflats and reeds) and high habitats (forbs and willow), with restored locations similar to the high habitats. The third axis (Fig. 5.4E-H) only separates the forbs habitat from all other habitats and the restored locations, due to its high summer ASi contents. Vertically, the depth profile with higher and more variable concentrations in the first layer is apparent on all three axes (Fig. 5.4D and H)

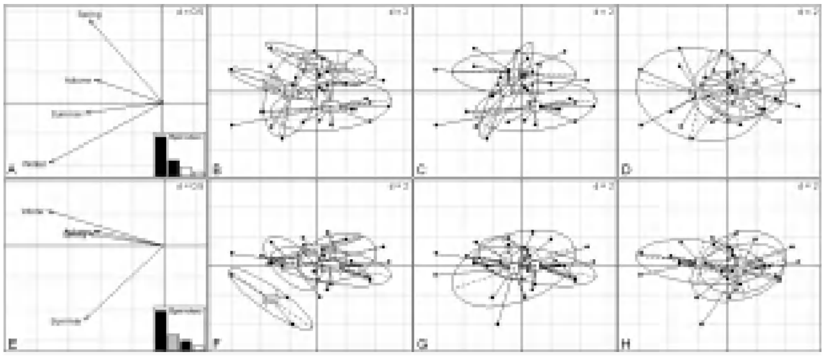

Fig. 5.4: PCA on ASi contents. A, B, C & D: axes 1 and 2. E, F, G & H: axes 1 and 3. A & E: Season's covariances and eigenvalues diagram. B, C, D, F, G, & H: sampling units grouped per site (Y = young; O = old; R = restored), habitat (Mud = mudflat, Rds = reeds, For = forbs, Wil = willow, Res = restored) and layer (1 = 0-5 cm; 2 = 5 – 15 cm; 3 = 15-25 cm).

5.4.2 DSi concentrations

Dissolved Silica concentrations in pore water ranged between 180 and 640 µM (Fig. 5.4). ANOVA revealed significant differences between seasons, sites, and habitats (Table 5.2, explained/total variability = 0.747).

Table 5.2: Summary of analysis of variance of the DSi (pore water) dataset. Interactions were removed one by one in decreasing order of significance, until only significant factors and interactions remained.

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 all other seasons (appendix 1, fig. 5.5). Differences between the three sites are caused by slightly lower values in the restored site compared to the old and the young site (appendix 1). Habitat differences do not clearly separate groups, but differences were generally situated in the restored site (appendix 1). Reference habitats did not differ significantly (appendix 1).

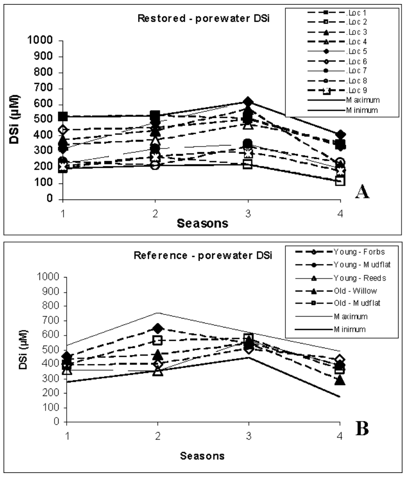

Fig. 5.5: seasonal differences in pore water DSi concentrations for the reference (A) and restored (B) locations. Minima and maxima observed are shown in dashed lines.

5.5 Discussion

5.5.1 Build-up of ASi-stock

ASi contents and DSi pore water concentrations of tidal marsh sediments in a freshwater macrotidal estuary (Schelde) were investigated in order to discuss the potential contribution of restoration sites to the estuarine silica sink. 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). 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 restored area is in origin a tidal marsh, which was embanked and used for agriculture for several centuries. However, it was deliberately flooded during winter seasons until the seventies, when dike heights along the Schelde were increased. This means that an import of ASi during winter could have been possible until then. Also, the crop rotation, which included the monocotyledon species *Zea mais* could have been a source of amorphous silica, as monocots accumulate silica in phytoliths. The vertical profile was probably homogenised by ploughing activities.

The ASi contents we observed initially in the restored site were slightly lower compared to the reference sites. The minimal difference was surprising, because the restored area was cut-off from regular tidal regime since its embankment several centuries ago.

Since the introduction of the tidal regime in March 2006; import of ASi clearly intensified and 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. This indicates that sedimentation gradually changes the restored site into a site similar to the reference sites.

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 amorphous silica content of the reeds sediments, where *Phragmites australis* thrives in high density, is not higher than other on locations. These observations suggest a minor importance of plant-ASi in the build-up of marsh silica stocks.

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.

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 arrived ASi.

5.5.2 ASi-recycling

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, where abundant growth of 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), departure from equilibrium (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 (0.0-10.1 cm) of fresh sediment which settled on the clay soil of the restored site.

Although these processes could explain variations of ASi content in the surface layer, neither sedimentation nor diatom growth can explain the apparent increases in ASi content in the deeper sediment layers. Struyf et al. (2005) showed there is a large spatial variation in the ASi content of deeper sediment layers in tidal marshes. These authors only found a significant depth gradient when averaging for all five sampling sites. Within their sampling sites however, similar large fluctuations were observed.

The existence of such large variability could nevertheless imply transformations in the deeper layers. The only apparent source of silica in these deeper layers is the dissolved silica in pore water, implying the presence of binding processes. 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). The observed variability might be partly due to occurrence of these processes.

5.5.3 Conclusions

Amorphous silica stocks of restored tidal marshes under CRT regime rapidly build up, and dissolution processes quickly initiate. The ongoing development of 1135ha of artificial areas along the Schelde, which will then make out 44% of the freshwater tidal marsh surface, will impact the estuarine silica cycle.

ASi build-up is dependent on a variety of processes, as in tidal marshes. This research provides a detailed empirical basis of silica cycling in intertidal surface sediments and restored marshes.

In the surface layer, concentrations are mainly controlled by biological processes. In situations where optimal conditions for benthic microalgal growth and/or sedimentation occur, we observed maximum ASi contents in the surface layer. On the other hand, longer periods of higher temperature, drought or lowered DSi concentrations in flooding water could enhance die-back of benthic microalgae or dissolution of ASi, which can explain the observed surface minima. In deeper layers, complex physicochemical processes could play a role in ASi-recycling.

The variability in our data indicate that field and lab measurements with high frequencies and spatial resolutions, as well as discrimination of different forms of ASi, are needed to estimate the total sink and buffer capacity of tidal marshes and their importance to the global silica cycle and the estuarine food web, as well to fine-tune restoration practices.

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Beckwith, R. S. & Reeve, R. 1963. Studies on soluble silica in soils. I. The sorption of silicic acid by soils and minerals. Australian Journal of Soil Research. 1:157-168.

Bidle, K. D. & Azam, F. 1999. Accelerated dissolution of diatom silica by marine bacterial assemblages. Nature. 397:508-512.

Boorman, L.; Garbutt, A. & Barrat, D. 1998. The role of vegetation in determining patterns of accretion of salt marsh. In Sedimentary Processes in the Intertidal Zone, eds. Black, K.; Paterson, D. & Cramp, A. 389-399

Chambers, J. M. 1992. Chapter 4 of Statistical Models in S. In Linear models, eds. Chambers, J. M. & Hastie, T. J. Wadsworth & Brooks/Cole..

Conley, D. & Kilham, S. 1989. Differences in silica content between marine and freshwater diatoms. Limnology and Oceanography 34:205-213.

Conley, D. & Malone, T. 1992. Annual cycle of dissolved silicate in Chesapeake Bay: implications for the production and fate of phytoplankton biomass. Marine Ecology Progress Series. 81:121–128.

Conley, D. 2002. Terrestrial ecosystems and the global biogeochemical silica cycle. Global Biogeochemical Cycles. 16:1121-1129.

Cugier, P., Billen, G., Guillau, J., Garnier, J. & Ménesguen, A. 2005. Modelling the eutrophication of the Seine Bight (France) under historical, present and future riverine nutrient loading. Journal of Hydrology. 304:381–396.

DeMaster, D. 1991. Measuring biogenic silica in marine sediments and suspended matter. Geophysical Monograph 63:363–367.

Derry, L. A., Kurtz, A. C., Ziegler, K. & Chadwick, O. A. 2005. Biological control of terrestrial silica cycling and export fluxes to watersheds. Nature. 433:728-731.

Dietzel, M. 2002. Interaction of polysilicic and monosilicic acid with mineral surfaces. In Water-rock interaction, eds. Stober, I.Bucher, K.

207-235 Kluwer, Netherlands.

Dolédec, S. & Chessel, D. 1987. Rythmes saisonniers et composantes stationnelles en milieu aquatique I- Description dun plan d'observations complet par projection de variables. Acta Œcologica, Œcologia Generalis. 8:403-426.

Dolédec, S. & Chessel, D. 1989. Rythmes saisonniers et composantes stationnelles en milieu aquatique. II. Prise en compte et élimination d'effets dans un tableau faunistique.

Drees, L. R.; Wilding, L. P.; Smeck, N. E. & Sankayi, A. L. 1989. Silica in soils: Quartz and disordered silica polymorphs. In Minerals in soil environments.SSSA Book series No.1, eds. Dixon, J. B.Weed, S. B. 913-974 Madison, WI.

Facca, C., Sfriso, A. & Socal, G. 2002. Phytoplankton changes and relationships with microphytobenthos and physico-chemical variables in the central part of the Venice lagoon. Estuarine Coastal and Shelf Science. 54:773-792.

Garnier, J., Billen, G. & Coste, M. 1995. Seasonal succession of diatoms and Chlorophyceae in the drainage network of the river Seine: Observations and modeling. Limnology and Oceanography. 40:750–765.

Harsh, J. B.; Chorover, J. & Nizeyimana, E. 2002. Allophane and imogolite. In Soil Mineralogy with environmental applications. Book Series SSSA No. 7, eds. Dixon, J. B.Schulze, D. G. 291-322 Madison.

Hurd, D. & Birdwhistell, S. 1983. On producing a more general model for biogenic silica dissolution. American Journal of Science. 283:1–28.

Hurd, D. 1973. Interactions of biogenic opal, sediment and seawater in the Central Equatorial Pacific. Geochimica and Cosmochimica Acta. 37:2257–2282.

Jacobs, S., Struyf, E., Maris, T. & Meire, P. 2008. Spatiotemporal aspects of silica buffering in restored tidal marshes. Estuarine Coastal and Shelf Science. 80:42-52.

Kamatani, A. 1982. Dissolution rates of silica from diatom decomposing at various temperature. Marine Biology. 68:91–96.

Kimmerer, W. 2005. Long-term changes in apparent uptake of silica in the San Francisco estuary. Limnology and Oceanography. 50:793–798.

Lawson, D., Hurd, D. & Pankratz, H. 1978. Silica decomposition rates of decomposing phytoplankton assemblages at various temperatures. American Journal of Science. 278:1373–1393.

Leonard, L. & Luther, M. 1995. Flow hydrodynamics in tidal marsh canopies. Limnology and Oceanography. 40:1474–1484.

Leonard, L.; Hine, A. & Luther, M. 1995. surficial sediment transport and deposition processes in a Juncus-Roemerianus marsh, west-central Florida. Journal of coastal research. 11: 322-336

Manly, B. 1991. Randomization and Monte Carlo methods in Biology. Chapman and Hall, London.

Maris, T., Cox, T., Temmerman, S., De Vleeschauwer, P., Van Damme, S., De Mulder, T., Van den Bergh, E. & Meire, P. 2007. Tuning the tide: creating ecological conditions for tidal marsh development in a flood control area. Hydrobiologia. 588:31-34.

Meire, P., Ysebaert, T., Damme, S. V., den Bergh, E. V., Maris, T. & Struyf, E. 2005. The Scheldt estuary: a description of a changing ecosystem. Hydrobiologia. 540:1–11.

Miller, R. G. 1981. Simultaneous Statistical Inference, Springer.

Muylaert, K., Sabbe, K. & Vyverman, W. 2000. Spatial and temporal dynamics of phytoplankton communities in a freshwater tidal estuary Schelde, (Belgium). Estuarine, Coastal and Shelf Science. 50:673–687.

Nelson, D., Tréguer, P., Brzezinski, M., Leynaert, A. & Quéguiner, B. 1995. Production and dissolution of biogenic silica in the ocean: revised global estimates, comparison with regional data and relationship to biogenic sedimentation. Global Biogeochemical Cycles. 9:359–372.

Patrick, S. & Holding, A. J. 1985. The effect of bacteria on the solubilization of silica in diatom frustules. Journal of Applied

Bacteriology. 59:7–16.

Pearson, K. 1901. On lines and planes of the closest fit to systems of points in space. Philosophical Magazine. 2:559-572.

Ragueneau, O.; Conley, D.; Leynaert, A.; Ni Longphuirts, S. & Slomp, C. 2006. Role of diatoms in Silicon Cycling and Coastal Marine Foodwebs. In the silicon cycle. Scope series 66, eds. Ittekot, V.; Unger, D.; Humborg, C. & Tac Ann, N. Islandpress Washington.

Saccone, L., Conley, D., Koning, E., Sauer, D., Sommer, M., Kaczorek, D., Blecker, S. & Kelly, E. 2007. Assessing the extraction and quantification of amorphous silica in soils of forest and grassland ecosystems. European Journal of Soil Science. 58:1446–1459.

Sarmiento, J. L. & Gruber, N. 2006. Ocean Biogeochemical Dynamics, Princeton, Woodstock: Princeton University Press.

Soetaert, K., Middelburg, J., Heip, C., Meire, P., Damme, S. V. & Maris, T. 2006. Long-term change in dissolved inorganic nutrients in the heterotrophic Scheldt estuary (Belgium, The Netherlands). Limnology and Oceanography. 51:409–423.

Sommer, M., Kaczorek, D., Kuzyakov, Y. & Breuer, J. 2006. Silicon pools and fluxes in soils and landscapes - a review. Journal of Plant Nutrition and Soil Science. 169. 310–329

Street-Perrott, A. & Barker, P. 2008. Biogenic silica, a neglected component of the coupled global biogeochemical cycles of carbon and silicon. Earth Surface Processes and Landforms. 33:1436-1457.

Struyf, E., Damme, S. V., Gribsholt, B., Middelburg, J. & Meire, P. 2005. Biogenic silica in tidal freshwater marsh sediments and vegetation (Schelde estuary, Belgium). Marine Ecology Progress Series. 303:51–60.

Struyf, E., Dausse, A., Damme, S. V., Bal, K., Gribsholt, B., Boschker, H., Middelburg, J. & Meire, P. 2006. Tidal marshes and biogenic silica recycling at the land-sea interface. Limnology and Oceanography. 51:838–846.

Struyf, E., Temmerman, S. & Meire, P. 2007. Dynamics of biogenic Si in freshwater tidal marshes, Si regeneration and retention in marsh sediments (Scheldt estuary). Biogeochemistry. 82:41–53.

Struyf, E., Van Damme, S., Gribsholt, B., Bal, K., Beauchard, O., Middelburg, J. & Meire, P. 2007b. Phragmites australis and silica cycling in tidal wetlands. Aquatic Botany. 87:134-140.

Temmerman, S., Govers, G., Meire, P. & Wartel, S. 2003. Modelling long-term tidal marsh growth under changing tidal conditions and suspended sediment concentrations, Scheldt estuary, Belgium. Marine Geology. 193:151-169.

Tréguer, P., Nelson, D. M., van Bennekorn, A. J., DeMaster, D. J., Leynaert, A. & Quéguiner, B. 1995. The silica balance in the world ocean: a reestimate. Science. 268:375-379.

van Bennekom, A., Jansen, J., van der Gaast, S., van Iperen, J. & Pieters, J. 1989. Aluminum-rich opal: an intermediate in the preservation of biogenic silica in the Zaire (Congo) deep-sea fan. Deep-Sea Research. 36:173–190.

Van Damme, S., Struyf, E., Maris, T., Ysebaert, T., Dehairs, F., Tackx, M., Heip, C. & Meire, P. 2005. Spatial and temporal patterns of water quality along the estuarine salinity gradient of the Scheldt estuary (Belgium and The Netherlands): results of an integrated monitoring approach. Hydrobiology. 540:29–45.

van der Zee, C., Roevros, N. & Chou, L. 2007. Phosphorus speciation, transformation and retention in the Scheldt estuary (Belgium/The Netherlands) from the freshwater tidal limits to the North Sea. Marine Chemistry. 106:76-91.

Vanderborght, J., Wollast, R. & Billen, G. 1977. Kinetic-models of diagenesis in disturbed sediments. 1. Mass-transfer properties and silica diagenesis. Limnology and Oceanography. 22:787–793.

Yamada, S. & D'Elia, C. 1984. Silicic acid regeneration from estuarine sediment cores, Marine Ecology Progress Series. 18:113–118.

Yandell, B. S. 1997. Practical Data Analysis for Designed Experiments, Chapman & Hall.

5.7 Appendix 1

DSi-posthoc: seasons	diff	lwr	Upr	P	
spring-autumn	-2.66	-3.96	-1.37	0.0000	***
summer-autumn	-0.95	-2.31	0.41	0.2682	
winter-autumn	-4.22	-5.57	-2.87	0.0000	***
summer-spring	1.71	0.53	2.90	0.0014	**
winter-spring	-1.56	-2.73	-0.39	0.0041	**
winter-summer	-3.27	-4.52	-2.03	0.0000	***

DSi-posthoc: sites	diff	lwr	Upr	p	
Restored-Old	-2.89	-3.94	-1.85	0.0000	***
Young-Old	-0.06	-1.22	1.09	0.9906	
Young-Restored	2.83	1.89	3.77	0.0000	***

DSi-posthoc: habitats	diff	lwr	Upr	p	
.Loc10-.Loc1	-4.34	-11.19	2.50	0.6466	
.Loc2-.Loc1	-8.21	-11.83	-4.58	0.0000	***
.Loc3-.Loc1	-3.14	-6.41	0.13	0.0736	.
.Loc4-.Loc1	-3.39	-6.81	0.04	0.0553	.
.Loc5-.Loc1	-0.85	-4.47	2.78	0.9999	
.Loc6-.Loc1	-1.80	-4.95	1.36	0.7926	
.Loc7-.Loc1	-6.25	-9.41	-3.10	0.0000	***
.Loc8-.Loc1	-6.78	-9.93	-3.62	0.0000	***
.Loc9-.Loc1	-7.38	-10.65	-4.10	0.0000	***
Forbs-.Loc1	-5.11	-8.09	-2.12	0.0000	***
Mudflat-.Loc1	-2.87	-5.41	-0.33	0.0126	*
Reeds-.Loc1	-5.35	-8.62	-2.08	0.0000	***
Willow-.Loc1	-4.60	-7.19	-2.01	0.0000	***

.Loc2-.Loc10	-3.86	-10.98	3.25	0.8409	
.Loc3-.Loc10	1.20	-5.74	8.15	1.0000	
.Loc4-.Loc10	0.96	-6.06	7.97	1.0000	
.Loc5-.Loc10	3.50	-3.62	10.61	0.9165	
.Loc6-.Loc10	2.55	-4.34	9.44	0.9916	
.Loc7-.Loc10	-1.91	-8.80	4.98	0.9995	
.Loc8-.Loc10	-2.43	-9.32	4.45	0.9945	
.Loc9-.Loc10	-3.03	-9.98	3.91	0.9651	
Forbs-.Loc10	-0.76	-7.58	6.05	1.0000	
Mudflat-.Loc10	1.48	-5.15	8.11	1.0000	
Reeds-.Loc10	-1.01	-7.95	5.94	1.0000	
Willow-.Loc10	-0.26	-6.91	6.39	1.0000	
.Loc3-.Loc2	5.07	1.26	8.87	0.0010	**
.Loc4-.Loc2	4.82	0.89	8.75	0.0040	**
.Loc5-.Loc2	7.36	3.25	11.47	0.0000	***
.Loc6-.Loc2	6.41	2.71	10.11	0.0000	***
.Loc7-.Loc2	1.95	-1.75	5.66	0.8664	
.Loc8-.Loc2	1.43	-2.27	5.13	0.9876	
.Loc9-.Loc2	0.83	-2.97	4.64	1.0000	
Forbs-.Loc2	3.10	-0.46	6.66	0.1583	
Mudflat-.Loc2	5.34	2.15	8.53	0.0000	***
Reeds-.Loc2	2.86	-0.94	6.66	0.3658	
Willow-.Loc2	3.60	0.37	6.84	0.0148	*
.Loc4-.Loc3	-0.25	-3.86	3.37	1.0000	
.Loc5-.Loc3	2.29	-1.51	6.10	0.7201	
.Loc6-.Loc3	1.35	-2.02	4.71	0.9831	
.Loc7-.Loc3	-3.11	-6.47	0.25	0.0999	
.Loc8-.Loc3	-3.64	-7.00	-0.28	0.0214	*
.Loc9-.Loc3	-4.23	-7.71	-0.76	0.0043	**
Forbs-.Loc3	-1.97	-5.17	1.23	0.6937	

Mudflat-.Loc3	0.27	-2.52	3.06	1.0000	
Reeds-.Loc3	-2.21	-5.68	1.26	0.6430	
Willow-.Loc3	-1.46	-4.30	1.37	0.8846	
.Loc5-.Loc4	2.54	-1.39	6.47	0.6187	
.Loc6-.Loc4	1.59	-1.92	5.10	0.9529	
.Loc7-.Loc4	-2.87	-6.37	0.64	0.2389	
.Loc8-.Loc4	-3.39	-6.90	0.12	0.0691	.
.Loc9-.Loc4	-3.99	-7.60	-0.37	0.0169	*
Forbs-.Loc4	-1.72	-5.07	1.63	0.8887	
Mudflat-.Loc4	0.52	-2.44	3.49	1.0000	
Reeds-.Loc4	-1.96	-5.57	1.65	0.8419	
Willow-.Loc4	-1.21	-4.22	1.79	0.9816	
.Loc6-.Loc5	-0.95	-4.65	2.75	0.9998	
.Loc7-.Loc5	-5.41	-9.11	-1.70	0.0002	**
.Loc8-.Loc5	-5.93	-9.63	-2.23	0.0000	***
.Loc9-.Loc5	-6.53	-10.33	-2.72	0.0000	***
Forbs-.Loc5	-4.26	-7.82	-0.70	0.0057	**
Mudflat-.Loc5	-2.02	-5.21	1.17	0.6513	
Reeds-.Loc5	-4.50	-8.30	-0.70	0.0067	**
Willow-.Loc5	-3.75	-6.99	-0.52	0.0087	**
.Loc7-.Loc6	-4.46	-7.71	-1.21	0.0006	***
.Loc8-.Loc6	-4.98	-8.23	-1.73	0.0001	***
.Loc9-.Loc6	-5.58	-8.94	-2.22	0.0000	***
Forbs-.Loc6	-3.31	-6.39	-0.23	0.0231	*
Mudflat-.Loc6	-1.07	-3.72	1.58	0.9816	
Reeds-.Loc6	-3.55	-6.91	-0.19	0.0280	*
Willow-.Loc6	-2.81	-5.51	-0.11	0.0333	*
.Loc8-.Loc7	-0.52	-3.77	2.72	1.0000	
.Loc9-.Loc7	-1.12	-4.48	2.24	0.9968	
Forbs-.Loc7	1.15	-1.94	4.23	0.9912	

Mudflat-.Loc7	3.39	0.73	6.04	0.0021	**
Reeds-.Loc7	0.90	-2.46	4.27	0.9997	
Willow-.Loc7	1.65	-1.05	4.35	0.7002	
.Loc9-.Loc8	-0.60	-3.96	2.76	1.0000	
Forbs-.Loc8	1.67	-1.41	4.75	0.8427	
Mudflat-.Loc8	3.91	1.26	6.56	0.0001	***
Reeds-.Loc8	1.42	-1.93	4.79	0.9719	
Willow-.Loc8	2.17	-0.52	4.87	0.2578	
Forbs-.Loc9	2.16	-0.93	5.47	0.4654	
Mudflat-.Loc9	4.50	1.72	7.30	0.0000	
Reeds-.Loc9	2.02	-1.44	5.50	0.7621	
Willow-.Loc9	2.77	-0.06	5.61	0.0618	.
Mudflat-Forbs	2.24	-0.20	4.69	0.1084	
Reeds-Forbs	-0.24	-3.44	2.96	1.0000	
Willow-Forbs	0.50	-1.99	3.00	1.0000	
Reeds-Mudflat	-2.48	-5.27	0.31	0.1366	
Willow-Mudflat	-1.73	-3.67	0.21	0.1311	
Willow-Reeds	0.74	-2.08	3.58	0.9997	

Signif. Codes: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Chapter 6

Spatiotemporal aspects of silica buffering in restored tidal marshes

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6.1 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 at 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 ASi concentrations were monitored at the entrance culverts as well as in different habitats in the marsh. Despite the shifted spatiotemporal frame in which exchange processes take place compared to reference marshes and the only recent restoration of tidal regime, a swift DSi-delivery capacity was observed. Since silica-accumulating vegetation is not yet present, and difference with reference marshes' deliveries is surprisingly small, we point towards diatomaceous debris and phytoliths as the main silica source. Although further research is needed on the driving forces of different processes involved, restoration of former agricultural areas under CRT-regime provide potential to buffer silica in the estuary.

6.2 Introduction

Estuaries are biogeochemical hot-spots and among 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.

One of the most important problems these systems are confronted with is eutrophication. Eutrophication phenomena in estuaries are related to the balance between N, P and Si in river loading, and thus depend on interactions between human activities 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. Losses of 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 (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, 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 few 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 at 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 thus be avoided. However, data from only a few tidal freshwater areas are available, and conclusions are as of yet 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, but has been far less studied (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 international application might follow. 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 on the growing conscience 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).

6.3 Materials and 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.

6.3.1 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 about 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. 6.1A). The rotation system was abandoned in 2003.

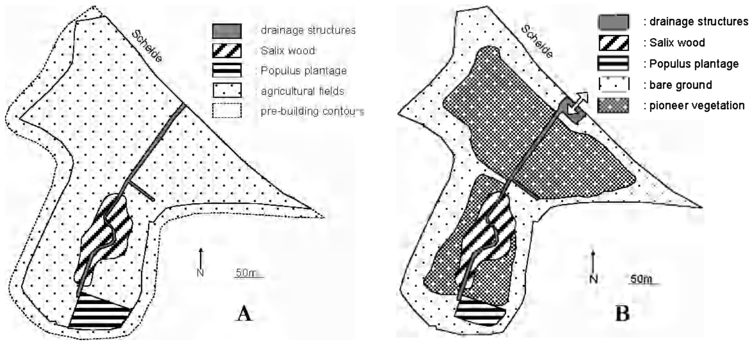

Fig. 6.1: schematic overview of study site before (A, left) and after building works (B, right).

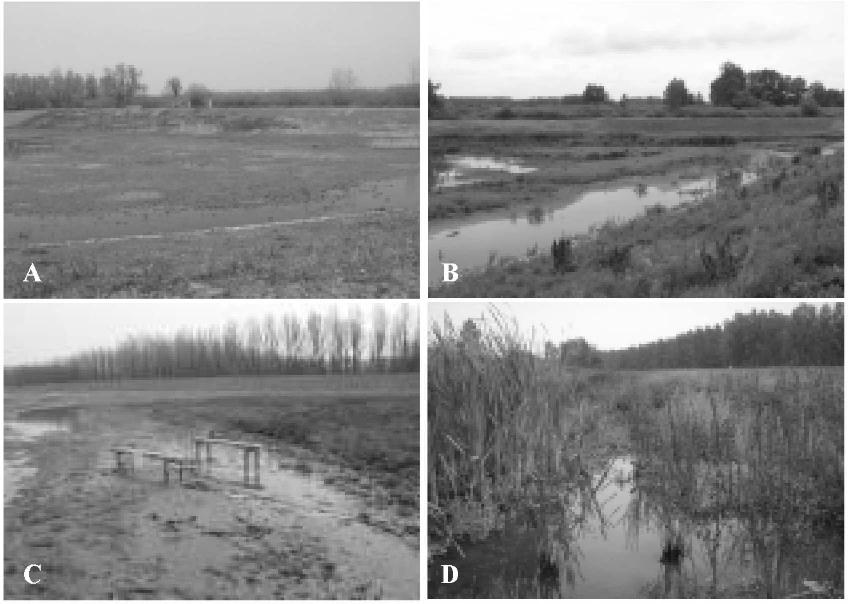

Fig. 6.2: Vegetation development in devegetated zones. A and B (upper left and right) show overview; C and D (lower l & r) detail. A and C are taken in spring 2006 (one month after first inundations), B in summer 2006, and D in summer 2007.

During the two-year construction phase (2003-2005), crops were replaced by pioneer vegetation (mainly *Epilobium hirsutum* and *Urtica dioica*) (Fig. 6.1B). Part of the polder was devegetated because of the building works (Fig. 6.1B). 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. 6.3). Because site elevation is several meters under mean high water level, reconstruction of spring-neap tide flooding variation required 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, while 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 wave 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. 6.3, for more details see Maris et al. 2007, Cox et al. 2006). The marsh is surrounded by a dike at 8m TAW.

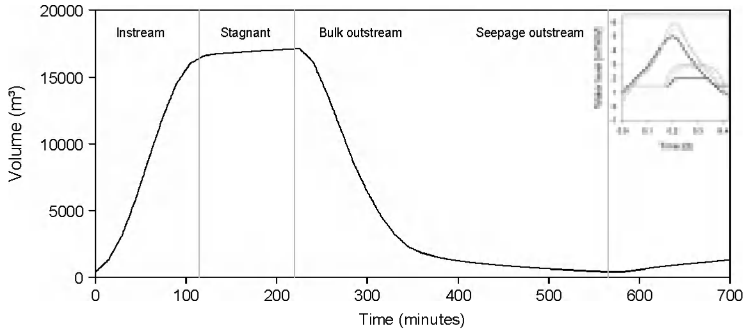

Fig. 6.3: Water mass balance during a typical tide in Lippenbroek. Inset illustrates tidal curves at typical neap (–), mean (· · ·) and spring (–) tide outside and inside Lippenbroek CRT (from Cox et al. 2006). Grey lines indicate (0) start instream, (110) stop instream =start stagnant phase, (220) start 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.

6.3.2 Sampling

A total of 796 data points were obtained during the 9 mass balance studies. Surface water samples were taken at the entrance and outlet culvert (1 in Fig. 6.4) and in selected habitats through the marsh (2-16 in Fig. 6.4). 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, Fig. 6.5-6.8 and Table 6.3)) plus day ("b" in text, Fig. 6.5-6.8 and Table 6.3) of 26 hours for September 2006 and March and June 2007 campaigns. Sampling intensity was highest during the first campaign (Table 6.1 & 6.2). This intensity was necessary to explore the spatial patterns in the marsh; however this exhaustive scheme was not entirely repeated during all campaigns. Selection of habitats during following campaigns was based on maximal cover of different habitat features. A selection of samples was analysed for ASi (table 6.2). This selection maximally covered different habitat features.

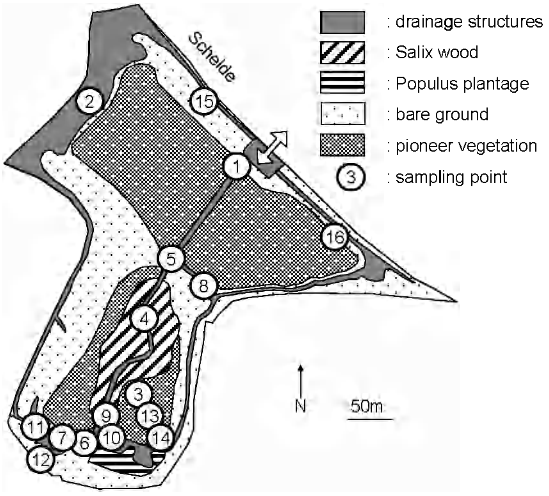

Fig. 6.4: Sampling locations in the CRT. Sampling intensity at locations is given in Tables 1 & 2.

Table 6.1: Sampling intensity at different locations (see Fig. 3) and during different campaigns

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 6.2: Selection of samples analysed for ASi at different locations (see Fig. 3) 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

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 6.2), 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 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).

6.3.3 Water and Silica mass Balances

All statistical analysis was 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. Measurements, calibration and operation of the flow meters were done 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.

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. For comparison between tides, this conservative correction was calculated as a percentage for each campaign, recalculated on the final mass balance and shown as a range. General patterns however were not gravely influenced by this effect. (Fig. 6.7)

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 trough PCA and ANOVA analysis (Chevenet et al. 1994).

6.4 Results

6.4.1 Concentration profiles

DSi concentration profiles show different seasonal patterns (Fig. 6.5). Instream phases (marked as in Fig. 6.3) are marked by steep concentration changes, while 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. 6.5) or even with a factor 17 (July 2006 in Fig. 6.5) at final concentration. At instream concentrations higher than 2 mg/l, profiles show slight increases (10.5% to 31.9%, May, September and October 2006 in Fig. 6.5). When instream concentrations become higher than 6 mg l^{-1}, profiles show a status quo or slight decrease (6%, March 2007 in Fig. 6.5) towards final concentrations.

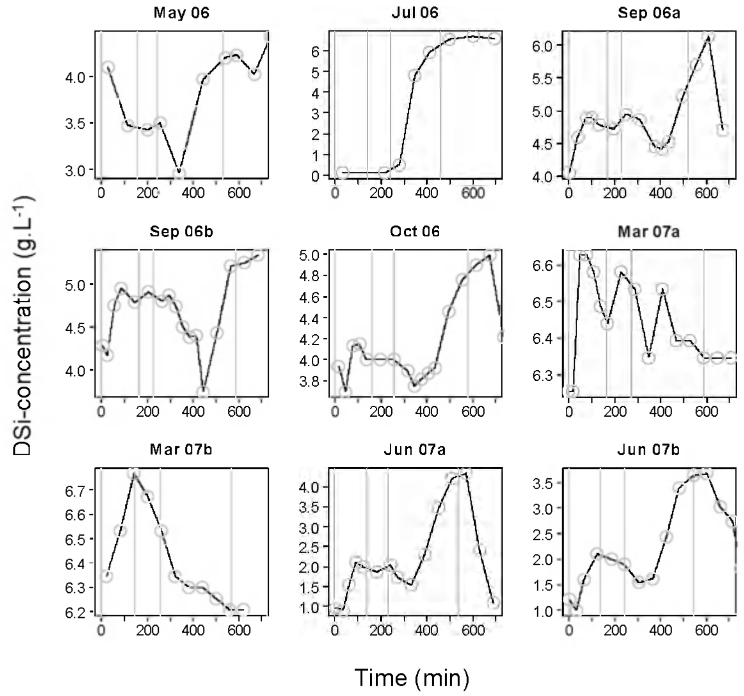

Fig. 6.5: DSi-concentration profiles at in- and outstream location for all campaigns (tidal phases as in Fig. 6.3). Campaign month is indicated above each subpanel (dots are measured concentrations, lines represent linear extrapolation).

Concentration profiles of ASi present a more variable pattern over a smaller concentration range (Fig. 6.6). 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, Fig. 6.6) towards strong ASi decrease in ASi-concentration profiles during later tidal cycle (March and June 2007b, Fig. 6.6), with the exception of June 2007a.

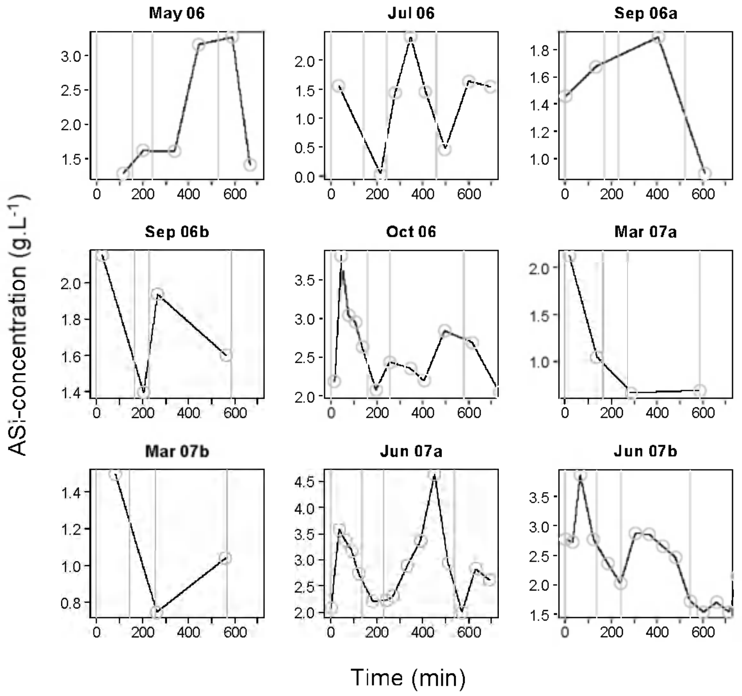

Fig. 6.6: ASi-concentration profiles at in- and outstream location (tidal phases as in Fig. 6.3) for all campaigns. Campaign month is indicated above each subpanel.

6.4.2 Mass balances

Calculated DSi mass balances indicate enrichment of exported water in summer months (July 2006 and June 2007, Fig. 7 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. 6.7 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. 6.7 lower graphs).

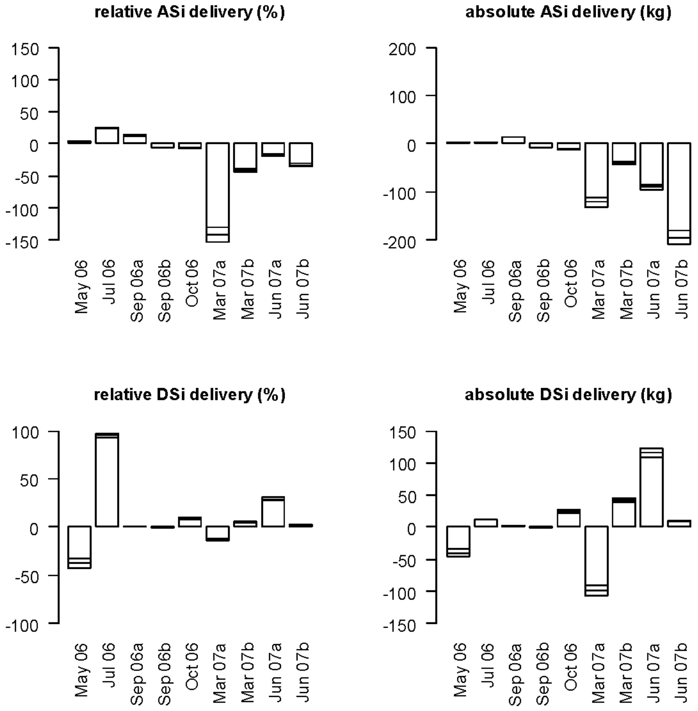

Fig 6.7: DSi (upper graphs) and ASi (lower graphs) mass balance of all campaigns. Balance is represented as percent, (out(g) –in(g)) / in(g))(left graphs); and in absolute numbers (kg delivered/retained)(right graphs). Error bars represent deviation from conservative mass balance, represented as percent in both directions.

Total silica mass balance (Fig. 6.8) is strongly influenced by ASi-retention, which adds up to Si retention in May and September 2006a and March 2007a (Fig. 6.7&6.8) or overrules DSi-delivery in June 2007b (Fig. 6.7&6.8). Yet, despite big ASi-retention in the March 2007b and June 2007a-campaigns , DSi-delivery can still be enough (Fig. 6.7 upper graphs) to provide the estuary with net silica (Fig. 6.8).

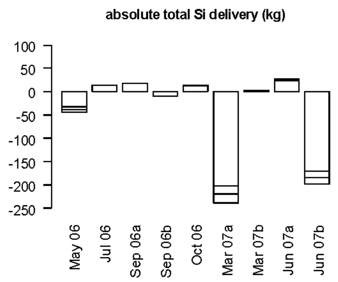

Fig. 6.8: Absolute total silica mass balance (DSi+ASi). Error bars represent deviation from conservative mass balance, represented as percent in both directions.

6.4.3 Ephemeral and Diurnal aspects

Features of each tidal cycle are listed in Table 6.3. 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. 9). Parallel vectors indicate high correlation 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. This exploration of numerical variables shows potential effects of riverine DSi-concentration ("dsisch" in Fig. 6.9), average flooding depth ("depth" in Fig. 6.9) and temperature ("t" in Fig. 6.9) on ASi- and DSi- balance ("asi and dsi in Fig. 9), while the balances are completely independent of riverine ASi-concentrations ("asisch" in Fig. 6.9)

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, Fig. 6.10 a), and the logarithm of mean water depth (R^2 = 0.52, p-value 0.02, Fig. 6.10b), and not (significantly) correlated with temperature of outstream water, percent ASi delivery, or riverine ASi concentrations (not shown).

Table 6.3: percent DSi and ASi delivery and general tidal features of different campaigns.

Season	Asi (%)	Dsi (%)	$T_{scheldt}$ (°C)	[DS$_{scheldt}$]	[ASi$_{scheldt}$]	mean water depth (cm)	Time instream	day/night	Tidal volume (m^3)
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

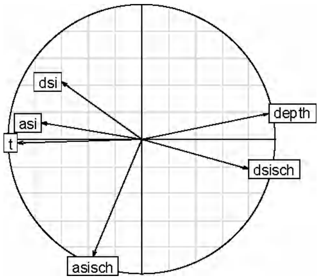

Figure 6.9: PCA correlation circle of table 3. Variation of numeric variables is maximised along two components. Mean water depth ("depth"), riverine ASi and DSi ("asisch" and " dsisch") and mean water temperature ("t") are shown with ASi and DSi percent delivery ("asi", "dsi"). Parallel vectors are 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.

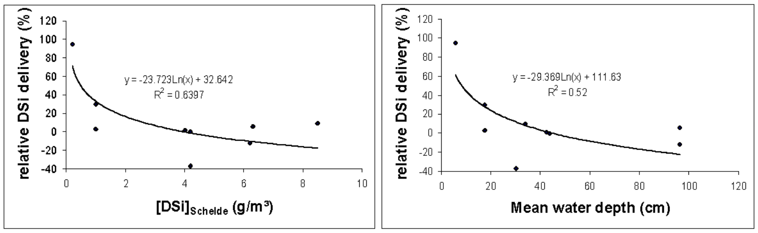

Fig. 6.10: Correlations between percent DSi-delivery and riverine DSi-concentration (a, left) and mean water depth of flooding (b, right).

ASi-delivery was negatively correlated with the mean water depth (R^2 = 0.5317, p-value 0.0257, Fig. 6.11a) and positively with the logarithm of temperature (R^2 = 0.5787, p-value 0.0172, Fig. 6.11b), ASi delivery was independent of riverine ASi-concentrations (not shown).

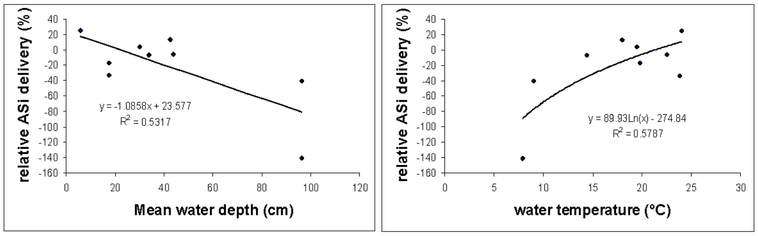

Fig. 6.11: Correlations between ASi-delivery and mean water depth of flooding(A,left) and mean water temperature during outstream (B,right).

6.4.4 Spatial Aspects

Concentration profiles observed at other locations were not used for local mass balances, because water volumes and tidal phases were not measured for these locations separately. Yet, interesting patterns are clear when comparing local concentration profiles: During the first campaign (May 2006, Table 6.1), intensive sampling of 15 locations throughout the marsh took place. Despite the relatively small surface area, and the marsh being functionally connected with the estuary less then two months before this campaign, a clear gradient in increasing vs. decreasing DSi-concentration profiles was observed (Fig. 6.12).

Concentrations in the polder even out with the riverine concentration at bulk water entrance. Concentrations appear unchanged during stagnant phase (minute 150 until 250 after instream in Fig. 6.12), but an hourglass pattern appears during outstream, when concentrations of different locations diverge strongly. (Minute 300; Fig. 6.12). Typical retention habitat in during the May 2006 campaign was the tidal pool (site 2 on Fig. 6.12 and Fig. 6.4), while opposite behaviour is observed at the lower mudflat (site 3 on Fig. 6.12 and Fig. 6.4). Locations in between nicely average these two extremes, as do the outstream concentrations (site 1 on Fig. 6.12 and Fig. 6.4) and. The hourglass structure returns, albeit less complete, in the other spatially sampled campaigns in summer and autumn with net DSi-delivery (not shown).

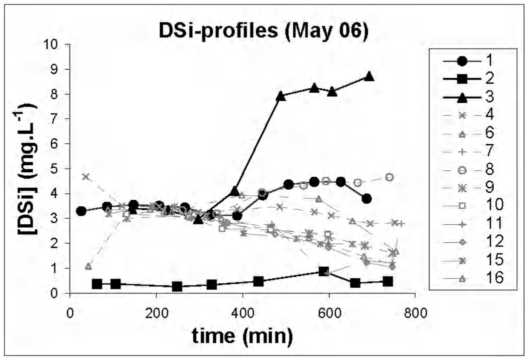

Fig. 6.12: DSi- concentration profiles at all locations (see Fig. 3) sampled in the May 2006 campaign. Tidal pool (2) and lower mudflat (3), as well as outstream location (1) are shown in black, other locations in grey.

A similar spatial sampling took place for ASi (Table 6.2). Homogenisation of concentration at instream and spatial patterns in general are less clear. Still, the lower mudflat site (site 3 in Fig. 6.13 and Fig. 6.4) delivers ASi during the May, July, October 2006 and June 2007 campaigns. Tidal pool habitat (site 2 in Fig. 6.13 and Fig. 6.4) delivers ASi during the July 2007 campaign (not shown). Other locations were year-round ASi-sinks.

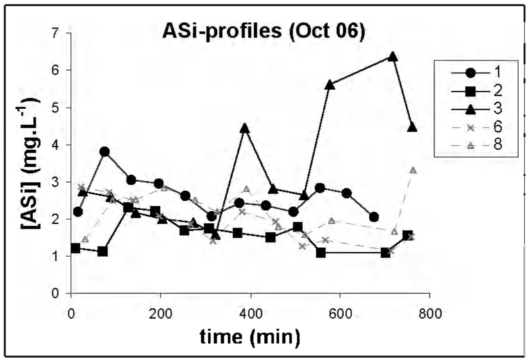

Fig. 6.13: ASi- concentration profiles at at all locations (see Fig. 3) sampled in the October 2006 campaign. Tidal pool (2) and lower mudflat (3), as well as outstream location (1) are shown in black, other locations in grey.

6.5 Discussion

Numerous processes are involved in the silica-exchange between tidal marsh and flooding water (Fig. 6.14). In the following overview, these processes are described and linked to the obtained data.

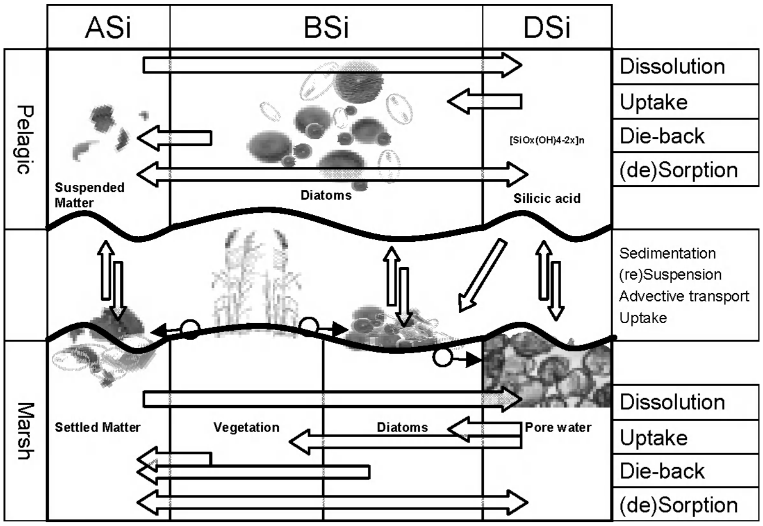

Fig. 6.14: schematic overview of processes involved in silica cycling in the intertidal ecosystem.

6.5.1 Diffusive and advective transport

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 can not explain the fast dissolution kinetics observed (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, and constitutes 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, have pointed earlier to 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.

6.5.2 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, regeneration of silicic acid from particulate silica has been shown by itself to be inadequate to 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 its proportion is recently expanding. 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 might 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* might 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 seems, 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. 6.4), and lowest at typical DSi-sink locations (2 in Fig. 6.4) (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 (Fig. 6.6) 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. 6.3). 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.

6.5.3 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 6.3, Fig. 6.9) 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.

6.5.4 Si-uptake by diatoms

Maybe the most striking difference of this study with earlier findings is the DSi-retention in the tidal marsh, until now not observed in important quantities (Struyf et al. 2006). The cause is almost certainly diatom uptake. In tidal pools, DSi-concentrations were observed to drop 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.

6.5.5 Interactions

In contrast of other systems where silica-fluxes are studied, processes influencing silica-cycling from tidal marshes are mostly separated in space and/or time. (Fig. 6.15). 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 (Fig. 6.15).

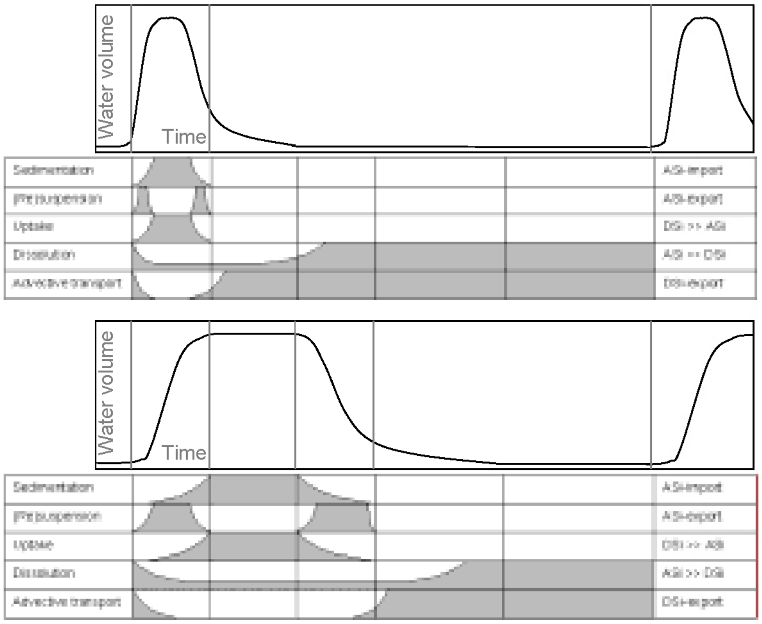

Figure 6.15: Schematic overview of timeframes of some key processes in silica cycling in tidal marshes and in Controlled Reduced Tidal areas

Yet, when DSi-deliveries are compared with earlier measured reference tidal marshes, the difference is surprisingly small (fig 6.16). Since the absence of *Phragmites* vegetation in the CRT does not provoke big differences in DSi-delivery to the estuary until now (Fig. 6.16; data from Struyf et al. 2006 and Van Damme et al., unpublished), we point to 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 abovementioned difference (Fig. 6.16) 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.

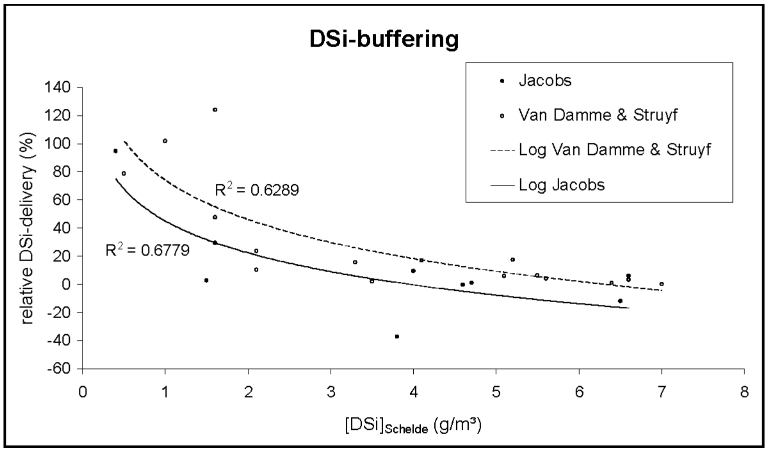

Figure 6.16: Comparison of DSi-buffering capacity of reference tidal marshes (open points, dashed curve) and CRT (filled points, full curve. Reference data from Struyf et al. 2006 and Van Damme et al. in prep.

6.6 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 (Fig. 6.15). Yet, 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 seems to show 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 through diatom dissolution and dissolution of buried phytoliths (Fig. 6.16). More research concerning the diatomaceous versus vegetation silica stock in intertidal mudflats and marshes is needed, as well as estimates of process rates to determine the limiting factors of silica buffering by marshes. Still, our main conclusion is that recently constructed CRTs along estuaries are capable of a silica buffering role comparable to older, reference tidal marshes.

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Aller,A.C. 1965., Quantifying solute distributions in the bioturbated zone of marine sediments by defining an average microenvironment. Geochimica et Cosmochimica Acta 44, 1955-1965.

Azam,F. and S.W.Chisholm., 1976. Silicic-Acid Uptake and Incorporation by Natural Marine-Phytoplankton Populations. Limnology and Oceanography 21, 427-435.

Becker,R.A., J.M.Chambers, and A.R.WILKS., 1988. The New S Language. Wadsworth & Brooks/Cole.

Berner,R.A., 1980. Early Diagenesis. Princeton University Press. Princeton and Oxford.

Bidle,K.D. and F.Azam., 2001. Bacterial control of silicon regeneration from diatom detritus, Significance of bacterial ectohydrolases and species identity. Limnology and Oceanography 46, 1606-1623.

Billen,G. and J.Garnier., 1997. The Phison River plume, coastal eutrophication in response to changes in land use and water management in the watershed. Aquatic Microbial Ecology 13, 3-17.

Blecker,S.W., R.L.McCulley, O.A.Chadwick, and E.F.Kelly., 2006. Biologic cycling of silica across a grassland bioclimosequence. Global Biogeochemical Cycles 20, GB3023.

Brzezinski,M.A. and D.M.Nelson., 1996. Chronic substrate limitation of silicic acid uptake rates in the western Sargasso Sea. Deep-Sea Research Part Ii-Topical Studies in Oceanography 43, 437-453.

Brzezinski,M.A., T.A.Villareal, and F.Lipschultz., 1998. Silica production and the contribution of diatoms to new and primary production in the central North Pacific. Marine Ecology-Progress Series 167, 89-104.

Callender,E. and D.E.Hammond., 1982. Nutrient Exchange Across the Sediment-Water Interface in the Potomac River Estuary. Estuarine Coastal and Shelf Science 15, 395-413.

Chevenet,F., S.Dolédec, and D.Chessel., 1994. A fuzzy coding approach for the analysis of long-term ecological data. Freshwater Biology 31, 295-309.

Childers,D.L. and J.W.Jr.Day., 1988. Direct quantification of nutrient and material fluwes between microtidal Gulf Coast wetlands and the estuarine column. Estuarine Coastal and Shelf Science 27, 486-494.

Childers,D.L., H.N.Jr.McKellar, R.Dame, F.Sklar, and E.Blood., 1993. A dynamic nutrient budget of subsystem interactions in a salt marsh estuary. Estuarine Coastal and Shelf Science 36, 105-131.

Conley,D.J., C.L.Schelske, and E.F.Stoermer., 1993. Modification of the biogeochemical cycle of silica with eutrophication. Marine Ecology-Progress Series 101, 179-192.

Conley,D.J., 2002. Terrestrial ecosystems and the global biogeochemical silica cycle. Global Biogeochemical Cycles 16.

Costanza,R., L.Wainger, C.Folke, and K.G.Maler., 1993. Modeling Complex Ecological Economic-Systems - Toward An Evolutionary, Dynamic Understanding of People and Nature. Bioscience 43, 545-555.

Costanza,R., R.dArge, R.deGroot, S.Farber, M.Grasso, B.Hannon, K.Limburg, S.Naeem, R.V.Oneill, J.Paruelo, R.G.Raskin, P.Sutton, and M.vandenBelt. 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253-260.

Cox,T., T.Maris, P.De Vleeschauwer, T.De Mulder, K.Soetaert, and P.Meire., 2006. Flood control areas as an opportunity to restore estuarine habitat. Ecological Engineering 28, 55-63.

Cugier,P., G.Billen, J.F.Guillau, J.Garnier, and A.Ménesguen., 2005. Modelling the eutrophication of the Seine Bight (France) under historical, present and future riverine nutrient loading. Journal of Hydrology 304, 381-396.

Dame,R.F., T.H.Chrzanowski, K.Bildstein, B.Kjeifve, H.McKellar, D.Nelson, J.D.Spurrier, S.Stancyk, H.Stevenson, F.Vernberg, and R.G.Zingmark., 1986. The outwelling hypothesis and North Inlet, South Carolina. Marine Ecology-Progress Series 33, 217-229.

Davis,C.O., N.F.Breitner, and P.J.Harrison., 1978. Continuous Culture of Marine Diatoms Under Silicon Limitation .3. Model of Si-Limited Diatom Growth. Limnology and Oceanography 23, 41-52.

DeMaster D.J., 1991. Measuring biogenic silica in marine sediments and suspended matter. Geophysical Monograph 63, 363-367.

Derry,L.A., A.C.Kurtz, K.Ziegler, and O.A.Chadwick., 2005. Biological control of terrestrial silica cycling and export fluxes to watersheds. Nature 433, 728-731.

Dixit,S. and P.Van Capellen., 2002. Surface chemistry and reactivity of biogenic silica. Geochimica et Cosmochimica Acta 66, 2259-2269.

Greenberg,S.A. and E.W.Price., 1975. The solubility of silica in solutions of elektrolytes. Journal of Physical Chemistry 61, 1539-1541.

Greenwood,J.E., V.W.Truesdale, and A.R.Rendell., 2001. Biogenic silica dissolution in seawater - in vitro chemical kinetics. Progress in Oceanography 48, 1-23.

Gribsholt,B., H.T.S.Boschker, E.Struyf, M.Andersson, A.Tramper, L.De Brabandere, S.Van Damme, N.Brion, P.Meire, F.Deheirs, J.J.Middelburg, and C.H.R.Heip., 2005. Nitrogen processing in a tidal freshwater marsh, a whole ecosystem 15N labeling study. Limnology and Oceanography 50, 1945-1959.

Hackney,C.T., L.B.Cahoon, C.Prestos, and A.Norris. 2000. Silicon is the link between tidal marshes and estuarine fisheries, a new paradigm. In M.P.Weinstein and D.A.Kreeger, editors, Concepts and controversies in tidal marsh ecology. Kluwer Academic Publishers. London. 543-552.

Hammond,D.E., C.Fuller, D.Harmon, B.Hartman, M.Korosec, L.G.Miller, R.Rea, S.Warren, W.Berelson, and S.W.Hager., 1985. Benthic Fluxes in San-Francisco Bay. Hydrobiologia 129, 69-90.

Humborg,C., E.Smedberg, S.Blomqvist, C.Mörth, J.Brink, L.Rahm, A.Danielsson, and J.Sahlberg., 2004. Nutrient variations in boreal and subarctic Swedish Rivers, landscape controll of land-sea fluxes. Limnology and Oceanography 49, 1871-1883.

Hurd,D.C. and F.Theyer., 1975. Changes in the physical and chemical properties of biogenic silica from the Central Equatorial Pacific. I. Solubility, specific surface area, and solution rate constants of acid-cleaned samples. In T.R.P.J.Gibbs, editor, Analytical methods in oceanography 211-230.

Irigoien,X., R.P.Harris, H.M.Verheye, P.Joly, J.Runge, and M.Starr., 2002. Copepod hatching succes in marine ecosystems with high diatom concentrations. Nature 419, 387-389.

Kamatani,A. and J.P.Riley. 1979. Rate of Dissolution of Diatom Silica Walls in Seawater. Marine Biology 55, 29-35.

Kilham,P., 1971. A hypothesis concerning silica and the freshwater planktonic diatoms. Limnology and Oceanography 16, 10-18.

Kimmerer W., 2005. Long-term changes in apparent uptake of silica in the San Francisco estuary. Limnology and Oceanography 50, 793-798.

Lancelot,C., V.Rousseau, G.Billen, and D.Van Eeckhout. Coastal eutrophication of the Southern Bight of the North Sea, assesment and modelling. 2, 439-453. 1997. Berlin, Springer Verlag. Sensitivity of North Sea, Baltic Sea and Black Sea to anthropogenic and climatic changes. NATO-ASI Series 2, Environment Vol. 27.

Maris,T., T.Cox, S.Temmerman, P.De Vleeschauwer, T.De Mulder, E.Van den Bergh, and P.Meire., 2007. Tuning the tide, creating ecological conditions for tidal marsh development in a flood control area. Hydrobiologia 588, 31-43.

McManus,J., D.E.Hammond, W.M.Berelson, T.E.Kilgore, D.J.Demaster, O.Ragueneau, and R.W.Collier., 1995. Early diagenesis of biogenic opal, dissolution rates, kinetics, and paleoceanographic implications. Deep-Sea Research II 42, 871-903.

Meile,C., P.Berg, P.Van Capellen, and K.Tuncay., 2005. Solute-specific pore water irrigation, implications for chemical cycling in early diagenesis. Journal of Marine Research 63, 601-621.

Meire,P., T.Ysebaert, S.Van Damme, E.Van den Bergh, T.Maris, and E.Struyf., 2005. The Scheldt estuary, a description of a changing ecosystem. Hydrobiologia 540, 1-11.

Mortimer,R.J.G., M.D.Krom, P.G.Watson, P.Frickers, J.T.Davey, and R.J.Clifton., 1998. Sediment-water exchange of nutrients in the intertidal zone of the Humber estuary, UK. Marine Pollution Bulletin 37, 261-279.

Nelson,D.M. and M.A.Brzezinski., 1990. Kinetics of Silicic-Acid Uptake by Natural Diatom Assemblages in 2 Gulf-Stream Warm-Core Rings. Marine Ecology-Progress Series 62, 283-292.

Nelson,D.M. and P.Treguer., 1992. Role of Silicon As A Limiting Nutrient to Antarctic Diatoms - Evidence from Kinetic-Studies in the Ross Sea Ice-Edge Zone. Marine Ecology-Progress Series 80, 255-264.

Nelson,D.M., P.Tréguer, and B.Quéguiner., 1995. Production and dissolution of biogenic silica in the ocean, Revised global estimates, comparison with regional data and relationship to biogenic sedimentation. Global Biogeochemical Cycles 9, 359-372.

Nelson,D.M. and Q.Dortch., 1996. Silicic acid depletion and silicon limitation in the plume of the Mississippi River, Evidence from kinetic studies in spring and summer. Marine Ecology-Progress Series 136, 163-178.

Officer,C.B. and J.H.Ryther., 1980. The possible importance of silicon in the marine eutrophication. Marine Ecology-Progress Series 2, 93-91.

Passow,U., A.Engel, and H.Ploug., 2003. The role of aggregation for the dissolution of diatom frustules. FEMS Microbiology Ecology 46, 247-255.

Patrick,S. and A.J.Holding., 1985. The effect of bacteria on the solubilization of silica in diatom frustules. Journal of Applied Bacteriology 59, 7-16.

R 2006. [2.4.1]., 2006. The R Foundation for Statistical Computing.

Rickert,D., M.Schlüter, and K.Wallmann., 2002. Dissolution kinetics of biogenic silica from the water column to the sediments. Geochimica et Cosmochimica Acta 66, 439-455.

Saccone,L., D.J.Conley, E.Koning, D.Sauer, M.Sommer, D.Kaczorek, S.W.Blecker, and E.F.Kelly., 2007. Assessing the extraction and quantification of amorphous silica in soils of forest and grassland ecosystems. European Journal of Soil Science 58, 1446-1459.

Schelske,C.L. and E.F.Stoermer., 1971. Phosphorus, silica and and prdicted changes in algal quality in Lake Michigan. Science 173, 423-424.

Scudlark,J.R. and T.M.Church., 1989. The sedimentary fluw of nutrients at a Delaware salt marsh site, A geochemical perspective. Biogeochemistry 7, 55-75.

Smayda,T.J., 1997. Bloom dynamics, physiology, behavior, tropic effcts. Limnology and Oceanography 42, 1132-1136.

Soetaert,K., M.Hoffmann, P.Meire, M.Starink, D.Van Oevelen, S.Van Regenmortel, and T.Cox. 2004., Modelling growth and carbon allocation in two reed beds (Phragmites australis) in the Scheldt. Aquatic Botany 79, 211-234.

Spurrier,J.D. and B.Kjerfve., 1988. Estimating the net flux of nutrients between a salt marsh and a tidal creek. Estuaries. 11, 10-14.

Struyf,E., S.Van Damme, B.Gribsholt, J.J.Middelburg, and P.Meire., 2005. Biogenic silica in tidal freshwater marsh sediments and vegetation (Schelde estuary, Belgium). Marine Ecology-Progress Series 303, 51-60.

Struyf,E., A.Dausse, S.Van Damme, K.Bal, B.Gribsholt, H.T.S.Boschker, J.J.Middelburg, and P.Meire., 2006. Tidal marshes and biogenic silica recycling at the land-sea interface. Limnology and Oceanography 51, 838-846.

Struyf,E., S.Temmerman, and P.Meire., 2007a. Dynamics of biogenic Si in freshwater tidal marshes, Si regeneration and retention in marsh sediments (Scheldt estuary). Biogeochemistry 82, 41-53.

Struyf,E., S.Van Damme, B.Gribsholt, K. Bal, O. Beauchard, J.J. Middelburg and P.Meire., 2007b. Phragmites australis and Si-cycling in tidal wetlands. Aquatic botany 87 134-140.

Struyf,E. and D.J.Conley., 2008. Silica: an essential nutrient in wetland biogeochemistry. Frontiers in Ecology and environment, in press.

Temmerman,S., G.Govers, S.Wartel, and P.Meire. 2003., Spatial and temporal factors controlling short-term sedimentation in a salt and freshwater tidal marsh, Scheldt estuary, Belgium, SW Netherlands. Earth Surface Processes and Landforms 28, 739-755.

Temmerman,S., G.Govers, S.Wartel, and P.Meire. 2004., Modelling estuarine variations in tidal marsh sedimentation, response to a changing sea level and suspended sediment concentrations. Marine Geology 212, 1-19.

Tessenow,U. 1966., Untersuchungen über den Kieselsäurehaushalt der Binnengewasser. Archiv fuer Hydrobiologie.Supplementband 32, 1-136.

Treguer,P., D.M.Nelson, A.J.Vanbennekom, D.J.Demaster, A.Leynaert, and B.Queguiner., 1995. The Silica Balance in the World Ocean - A Reestimate. Science 268, 375-379.

Valiela,I., J.M.Teal, and W.G.Deuser., 2007. The nature of growth forms in the salt marsh grass Spartina alterniflora. American Naturalist 112, 461-470.

Van Bennekom,A.J., E.Krijgsman-van Hartingsveld, G.C.M.Van der Veer, and H.F.J.Van Voorst., 1974. The seasonal cycles of reactive silicate and suspended diatoms in the Dutch Wadden Sea. Netherlands Journal of Sea Research 8, 174-207.

Van Damme,S., E.Struyf, T.Maris, T.Ysebaert, F.Dehairs, M.Tackx, C.Heip, and P.Meire., 2005. Spatial and temporal patterns of water quality along the estuarine salinity gradient of the Scheldt estuary (Belgium and The Netherlands), results of an integrated monitoring approach. Hydrobiologia 540, 29-45.

Vanderborght,J.P., R.Wollast, and G.Billen., 1977. Kinetic-Models of Diagenesis in Disturbed Sediments .1. Mass-Transfer Properties and Silica Diagenesis. Limnology and Oceanography 22, 787-793.

Wang,W.C. and R.L.Evans., 1969. Variation of Silica and Diatoms in a Stream. Limnology and Oceanography 14, 941-944.

Whiting,G.J., H.N.McKellar, J.D.Spurrier, and T.G.Wolaver., 1989. Nitrogen exchange between a portion of vegetated salt marsh and the adjoining creek. Limnology and Oceanography 34, 463-473.

Wilke,R.J. and R.Dayal., 1982. The behaviour of iron, manganese and silicon in the Peconic Ricver estuary, New York. Estuarine Coastal and Shelf Science 15, 577-586.

Wong,G.T.F. and C.E.Grosch., 1978. A mathematical modal for the distribution of dissolved silicon in interstitial porewaters - and analytical approach. Journal of Marine Research 36, 735-750.

Yamada,S.S. and C.F.Delia., 1984. Silicic-Acid Regeneration from Estuarine Sediment Cores. Marine Ecology-Progress Series 18, 113-118.

Chapter 7
Conclusions

7.1 Freshwater tidal marsh restoration in the Schelde

7.1.1 Restoration goals

Reference state is a tricky concept. In many restoration projects, the reference state is a historical state of the ecosystem, of which it is supposed it had a higher (biodiversity or functional) value, often regardless of today's differing ecological and economical conditions. As for freshwater tidal marshes, particularly in highly urbanized estuaries, it is clear that a natural-historical reference is of little use to restoration ecologists.

Naturally, river mouths evolve irreversibly to estuaries and finally deltas. In many estuaries, anthropogenic interventions have severely impacted structures and processes since the Middle Ages. Historically, intertidal marshes were part of large dynamic floodplains with extended intertidal stretches buffering floods and storm events. In Belgium and the Netherlands for instance this delta comprised Rhine, Meuse, and Schelde and covered the north of Flanders up to the central Netherlands. Dissipation of the floods prevented deep longitudinal tidal intrusion. It is doubtful that there was significant and regular tidal influence in the freshwater parts of the Schelde estuary. Since a historical reference is of little use in the present socio-economical context, a 'reference state' is often based on desired structures and processes today, depending on goods and services considered important. Therefore, the term 'restoration goals' is more appropriate.

Our literature study shows that most of today's European tidal freshwater marshes occur in a suboptimal state compared to restoration goals (see Chapter 2). In the Schelde for instance, lateral erosion impacts the edges of the softwood marsh and flushes away young tidal marsh (Photo 7.1).

Photo 7.1: Eroded marsh edges in the Schelde estuary.

There is a lack of pioneer marshes and early succession stages. Similar problems are mentioned in the German Elbe. In the Biesbosch (the Netherlands) on the contrary, it is the lack of tidal dynamics which is blamed for impoverished vegetation composition. Still, tidal freshwater marshes throughout northwest Europe, although sparsely documented, have comparable structures: softwood floodplain forests (mostly willow, Photo 7.2) and helophyte stands (mostly reed) are the typical dominant structures.

Some important processes are repeatedly mentioned throughout the literature: tidal influence and its spatio-temporal variation create intertidal gradients, vegetation colonization and mudflat accretion form new marshes; occasional storm events naturally set back succession while waves (shipping or wind) and regular high current velocities exert continuous erosion and prevent development of young marsh.

Socio-economic goods and services were evaluated and incorporated into restoration goals in 2005, during the formulation of European restoration goals for the Schelde estuary (Adriaensen et al. 2005). Starting from the Schelde system services (Table 7.1), a priority list of desired processes for freshwater tidal marshes was put forward:

- Silica buffering
- Nitrogen retention
- Phosphorous retention
- Oxygenation
- Sedimentation

Photo 7.2: Typical freshwater tidal marsh platform: willow softwoods, accompanied mainly by *Poa* sp. and *Urtica dioica*.

Supporting services

- Primary productivity
- Nutrient cycling
- Water cycling
- Biodiversity
- Habitat for rare species or for global population
- Nursery function
- Migration route
- Soil formation

Regulating services

- Air quality regulation
- Climate regulation
- Water purification and waste treatment
- Regulation of transport of nutrients and contaminants
- Disease regulation
- Pest regulation
- Pollination
- Trophic-dynamic regulation
- Waterregulation (protection against flooding)
- Erosion regulation and sediment trap
- Maintaining habitat structure and features (eg. Tidal characteristics)
- Natural hazard regulation

Provisioning services

- Fresh water
- Clean air
- Food
- Fiber
- Fuel
- Genetic resources
- Biochemicals, natural medicines and pharmaceuticals
- Ornamental resources
- Fresh water
- cultural, scientific and educational values

Table 7.1: Ecosystem services of the estuary (after Adriaensen et al. 2005)

From literature, a detailed description -based on vegetations- for structural restoration goals was obtained. Incorporation of socio-economic goods and services completes the restoration goals for freshwater tidal marshes along the Schelde. These goals can be formulated as follows:

- Structurally, freshwater tidal marshes contain all vegetation stages, from mudflats with early pioneers, over intermediate succession stages to climax softwoods, with characteristic species as described for European tidal freshwater marshes nowadays (see Chapter 1, 2). This implies the presence of high marsh platforms, creek incisions as well as sheltered lower marsh stretches.
- Tidal influence and sedimentation allow continuous young marsh development and compensate for sporadic erosion losses. Waves and high current velocities should not erode entire habitats. Freshwater tidal marshes provide silica to the estuary at moments of depletion (Chapter 3, 4, 5), they retain nitrogen and phosphorous, oxygenate the flooding water, capture sediments and provide habitat for fish and birds.

7.1.2 Why Controlled Reduced Tide?

If freshwater tidal marshes have to be restored inside the Schelde river bed, some present problems in the Schelde will have to be tackled:
During the last century, regular anoxia events were observed, and some mudflats were observed to be black-colored year-round. In the 60's and 70's, severe industrial contamination with heavy metals and other pollutants took place, some of which are still being conserved in the marsh sediments. The only helophyte that withstood these conditions was *Phragmites australis*, which has been dominating large patches along the freshwater Schelde. Recently water quality has improved, black-colored mudflats have no longer been observed and other helophytes (e.g. *Bolboschoenus*) are again increasing in abundance along the Schelde. Since water quality is now quickly improving, biogeochemical equilibriums could be disturbed and unexpected events might occur, of

which we do not know if they will influence freshwater tidal marsh development or persistence. Stored metals might become available, algal blooms might cause anoxia again etc.

High current velocities in the Schelde prevent mudflats to accrete and be colonized (Photos 7.3 and 7.4). Although erosion events are considered necessary to set back succession, there is actually little shelter from erosion in the estuary, because all currents are confined between dikes, and tidal marshes are mostly slender stretches along them. As a result, the present marshes mostly have a steep eroding side, followed by a mudflat. High current velocities shift mudflat granulometry towards sand flats. It is impossible to hold on to the idea of restoring a natural equilibrium of marsh erosion and accretion within this hyperdynamic and confined estuary. There is an urgent need for a definition of a natural 'erosion event', or when and how many times these events are desired to take place.

Photo 7.3 (left): Typical riverside erosion of freshwater tidal marsh.

Photo 7.4 (right): Close-up of erosion impact on young developing marsh: about 15 cm of accreted soft sediments, along with pioneering plants and benthos, is washed away (note *Phragmites* rhizomes indicating former marsh height).

The problems with restoration within the river bed are often dodged by restoring areas outside the river dikes. This is the only option when space is limited or dynamics are disturbed within the river bed. Managed realignment, by removing or breaching the dikes, has become a traditional technique (French 2006). However, this technique can not be applied on most of the embanked areas along the Schelde: the tidal marshes have kept in pace with increased tidal influence by accretion and their elevation is now several meters higher than the elevation of historically embanked marshes, as is the case for many estuaries (see Chapter 3). Managed realignment would thus result in creation of subtidal and unvegetated intertidal mudflats, since the whole surface would flood every single tide. If conditions for sediment accretion and species colonization are met, tidal marshes will develop in due time, but it will take decennia or even centuries to catch up with the elevation difference.

In Chapter one, several advantages and deviating features of the controlled reduced tide (CRT) are described (see Chapter 1, CRT). The advantages of implementing a controlled reduced tide seem very convincing: tidal influence is detached from site elevation, flooding frequencies and flooded surface can be controlled along the full neap-spring flooding gradient, flooding regime can be adapted to changing conditions or objectives, the construction is simple and does not require continuous maintenance or complex electronics, tidal influence is restored with one-time intervention by building the culverts, etc. However, some specific features of the CRT (see Chapter 1, CRT) might prevent desired structures and processes to develop optimally: topography and culvert configuration might rule out the creation of an entire tidal gradient, the stagnant phase and differing seepage might change process rates and time frames, and the differing sedimentation pattern and water currents might influence marsh formation.

Therefore, evaluation of development of structures and processes in pilot CRT projects is crucial.

7.2 The pilot CRT: two years of development

7.2.1 Development of structures in the pilot CRT

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 freshwater tidal marsh species already during the first few months. Also, communities which have mostly disappeared along the degraded Schelde, but are described for several European references, are developing in the pilot site (see Chapter 2). 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 (Fig. 7.2C and 7.2D)

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. These creeks incised the compacted agricultural soil, and many of them are still deepening (Fig 7.1). 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 (Fig. 7.2A). 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 7.2B).

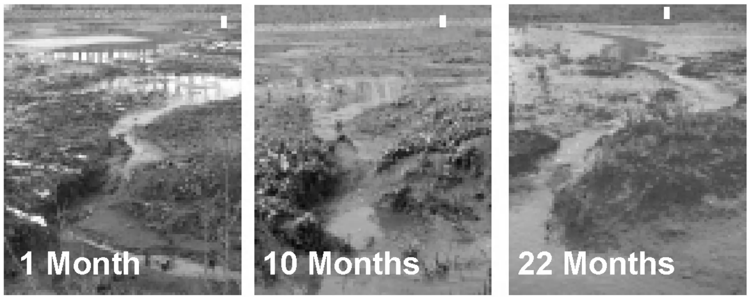

Fig. 7.1: Development of a creek in the CRT. White square indicates position of the same stick in every photo. Note the deepening of the creek, decrease of stagnant water surface in the background, and emergence of mudflats and meandering structures.

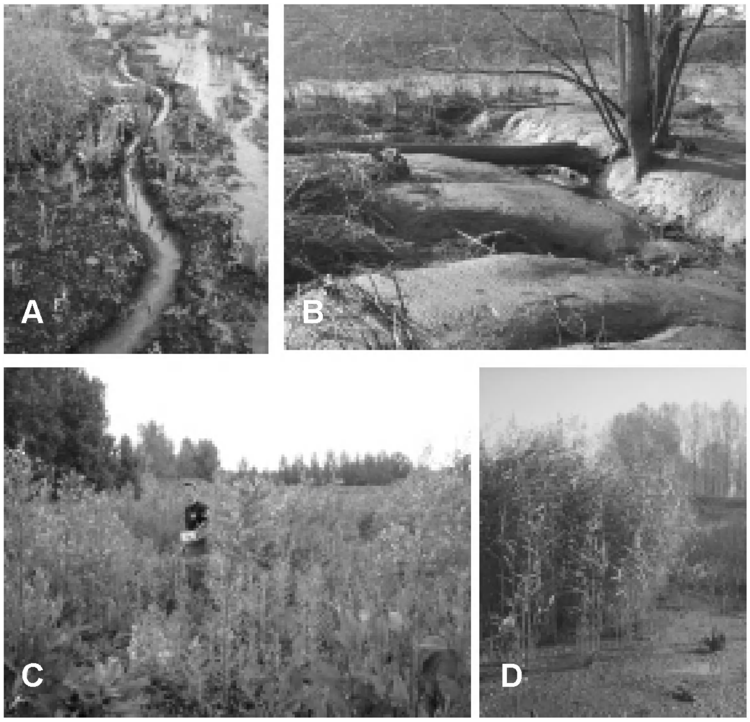

Fig. 7.2: Habitat development in the CRT. A: sheltered mudflats with pioneering species. B: typical merlon-krenel topography. C: young softwood colonization. D: *Phragmites* colonization.

Although morphological developments are still ongoing, many desired structures and typical vegetation communities are present in the pilot CRT, despite the minimal surface area and the short time span of the experiment. This proves that the CRT technique offers potential for restoration of a whole tidal gradient on one single lowered site. Up till now, there are no indications that the deviating tidal envelope, sedimentation and water currents cause problems for further development of desired structures in the pilot CRT. As structures further develop, research should focus on comparing habitat quantity (surface, density, volume, biomass, etc.) and quality (species richness, abiotic variables, etc.) with river habitats across Europe.

7.2.2 Development of processes in the pilot CRT

The deviating features of the CRT and the yet incomplete structures call for evaluation of the desired processes. In the frame of our research, it was not possible to evaluate all processes in depth. The following section focuses on the process of silica cycling inside the CRT. It also briefly presents some results on other processes, but without going into detail.

Silica buffering in the pilot CRT

Buffering of estuarine silica depletion events is one of the desired processes of tidal freshwater marshes. The delivery of silica during these events can enhance the resilience of the estuary against eutrophication phenomena (see Chapter 1).

To summarize the findings of Chapters 3, 4 and 5, conceptual models for both vegetation and sediment silica cycling are presented (Fig. 7.3,7.4 and 7.5). These models are used to highlight processes and structures involved in the CRT silica cycling. These processes and structures are then compared to our observations (see Chapter 3, 4 and 5) and future expectations are discussed.

Regarding silica cycling in tidal marsh vegetation, only a few Si-accumulating species are concerned. These are mainly *Poaceae*, of which *Phragmites* is by far the most important because of its high Si-concentrations and abundance (see Chapter 3).

During the growing season, plants take up dissolved silica from the pore water and translocate it to leaves and stems, where it is stored as biogenic silica. At the end of the growing season, these aboveground parts die and decompose, and most silica is again dissolved into the surrounding water, although a small part can be buried as amorphous silica in undissolved phytoliths.

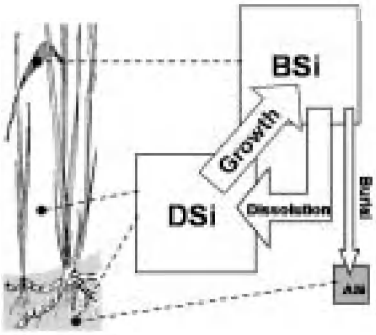

Fig. 7.3: Conceptual model of silica cycling in freshwater tidal marsh vegetation. Explanations in text.

The total amount of silica stored in the vegetation is small compared to the sediment silica stock, but since practically the whole vegetative stock is released every year, it could influence silica retention and delivery of freshwater tidal marshes (see chapter 3). The processes involved ('growth' and 'decomposition' in Fig. 7.3) are mainly linked to the seasonal cycle: helophytes such as *Phragmites* develop most of the aboveground biomass in spring and further accumulate silica during the growing season ('growth' in Fig. 7.3). In autumn, most shoots die, and a certain amount falls down and decomposes during autumn and winter. Biogenic silica dissolves quickly and continuously from decomposing litter ('dissolution' in Fig. 7.3). Although a year-round litter layer is present is some *Phragmites* stands, it remains to be seen wether this pool is currently substantial enough to contribute essentially to silica buffering functions of freshwater tidal marshes. Estuarine silica depletion events are mostly observed during the growing season, and the observed buffering requires a silica pool which reacts quickly to changing conditions associated with silica depletion events. The litter fraction is also mobile and flushed away to or imported from the estuary, which complicates the estimations of vegetative contribution to the silica mass balance.

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 and BSi to dissolved silica, and by the amount of reactive sediment exchanging dissolved silica with the flooding water. Based on our observations (see Chapter 4), two conceptual models are proposed: the first model depicts the standard retention state (Fig. 7.4). 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. 7.4; data see Chapter 4). Benthic diatom population turnover and sedimentation provide ASi ('sedimentation' and 'growth' in Fig. 7.4), which is partially dissolved ('dieback' and 'dissolution' in Fig. 7.4) 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. 7.4), and could be translocated and bound as ASi ('groundwater flux' and 'binding' in Fig. 7.4).

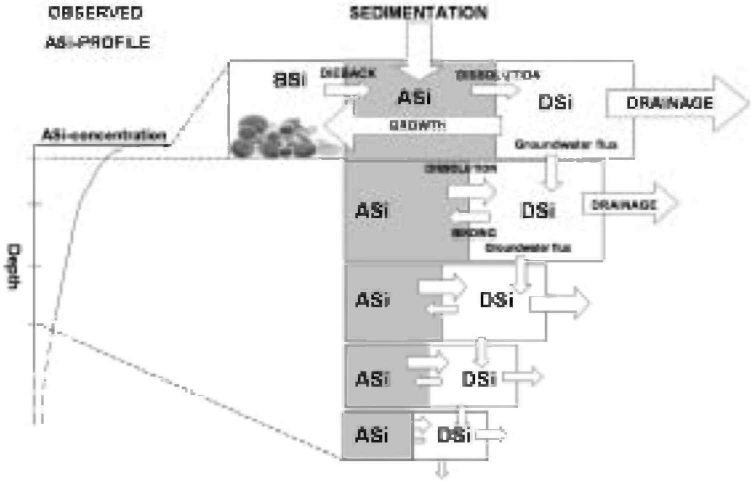

Fig. 7.4: Conceptual model of silica cycling in freshwater tidal marsh sediments during silica retention. Explanations in text.

The opposite conditions provoke a 'silica delivery state', with a profile with minimal values observed in the upper centimeters ('observed ASi-profile' in Fig. 7.5; data see Chapter 4). 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.

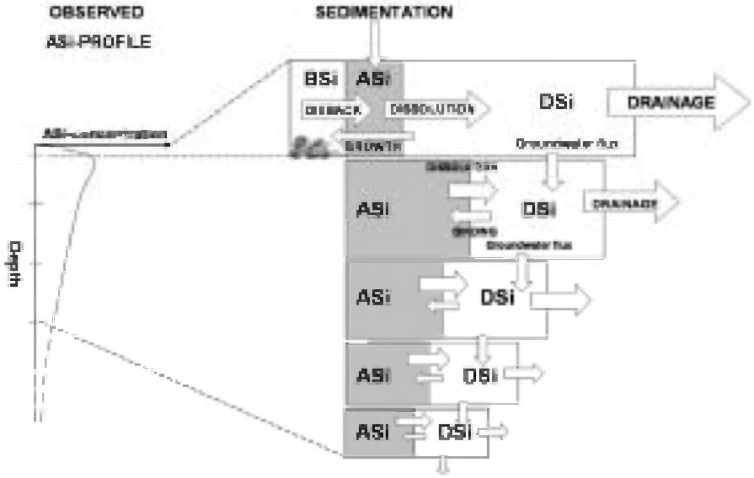

Fig. 7.5: Conceptual model of silica cycling in freshwater tidal marsh sediments during silica delivery. Explanations in text.

Our concept opposes two extreme states, which are theoretical extremes of a continuum. It shows that the sediment ASi-pool could potentially deliver DSi, mainly from the top centimeters. The 'delivery state' (Fig. 7.5) is probably more exceptional, since concentrations in the deeper (and older) layers remain high, which suggests that the 'retention state' (Fig. 7.4) is occurring most of the time. We hypothesize that conditions which promote the delivery state occur during estuarine depletion events in summer. The sediment silica stock could for this reason provide a swift buffering mechanism to estuarine silica depletion events. This hypothesis opposes the prevailing assumption that the entire ASi sediment stock has a slow reactivity. Upper layers, which consist of benthic algal populations and reactive frustules, are capable to react quickly. 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, different DSi-deliveries were observed at different conditions. Our results indeed suggest a positive correlation with temperature and a negative one with DSi-concentrations in the flooding water (see Chapter 5). This hypothesis could explain the observed DSi-delivery events.

Simultaneous measurements at higher frequencies and spatial resolutions of both ASi and DSi under different conditions are needed to check these hypotheses. As for now, it is not possible to chemically separate different forms of ASi and BSi.

As the CRT provides shelter for a more diverse helophyte community to develop, the competitive outcome of silica-accumulating species like *Phragmites* with species that do not accumulate silica but occupy the same ecological niche, like *Typha latifolia*, could influence the final vegetative silica stock. For the moment, *Phragmites'* total biomass is still far under the amounts found for river reed beds, but colonization is expected to continue and further increase the potential vegetative silica stock.

Specific conditions of the CRT could change process rates and timeframes and thus influence the sediment silica cycling compared to estuarine marshes (see Chapter 5). Enhanced sedimentation, prolonged inundation time and water aeration (see further in this chapter) could promote ASi input and benthic diatom growth, while longer dry periods could enhance diatom dieback and ASi dissolution in the pore water. A main structural setback could be expected from the minimal amount of reactive sediment and its poor exchange capacity with the still underdevelopped drainage system.

As for now, measured silica deliveries of the pilot CRT are about 80% of those measured in riverine freshwater tidal marshes (see Chapter 5). Probably, ongoing development of creeks and further sedimentation will increase the amount of reactive sediment and larger deliveries will probably be obtained. For instance, the presence of a badly drained tidal lake with high diatom productivity (and thus silica retention) has lowered the measured average silica deliveries balances (see Chapter 5). Creek development will eventually drain this lake and this will positively impact the measured silica mass balances.

Table 7.2: Processes (italic) and structures (bold) involved in silica cycling, their state in the pilot CRT compared to the estuarine freshwater tidal marshes, and their expected future state compared to the estuarine freshwater tidal marshes.

Structure/Process ~Si	CRT	Expected
Reed biomass	-	=
Reed growth	=	=
Litter dissolution	=/?	=/?
Reactive sediment	-	=
Drainage system	-	=
ASi sedimentation	+	+
Diatom Growth	+	+
Diatom Dieback	+	+
ASi dissolution	+/?	+
Groundwater flux	-	=
DSi Binding	-	=

The measured silica deliveries are high for a structurally underdeveloped marsh as this pilot site. ASi sedimentation and benthic diatom population sizes are expected to exceed these of estuarine marshes (Table 7.2). CRT areas could thus develop as potential hotspots for silica processing, which internally support the entire estuarine food web. Moreover, potentially negative impacts of missing structures (e.g. drainage system) and processes (e.g. groundwater flux) are expected to disappear (Table 7.2). These facts suggest that the CRT might even prove to be better in the buffering of estuarine silica depletion events than today's riverine marshes.

Other required processes in the pilot CRT

Other required processes to occur in restored tidal freshwater marshes are nitrogen and phosphorus retention, oxygenation, and sedimentation. Sedimentation processes are shown above (see section 2.1). In this section, we briefly present the first results of exchange studies in the CRT. Preliminary monitoring data indicate that nitrogen, phosphorus and oxygen processing also occur in comparable ways to river marshes.

. Although only a few exchange studies were performed in the Schelde estuarine marshes, on average there is a tendency for retention of total N, NO_2, NO_3 and NH_4 in the CRT. 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 Although more observations and further research is needed to solidify these findings, the expected nitrogen retention appears to occur in the CRT.

As for phosphorus, retention was observed only one time out of 9 in the CRT. Stocked phosphorus from fertilizers used during previous decades could explain the deliveries (as phosphorus is less mobile than nitrogen), and these might be expected to lower in the future. Although more observations and further research is needed, the CRT can retain PO4 under certain conditions.

Oxygenation of the flooding water might be better in the CRT compared to estuarine marshes. This has several possible reasons: firstly, the entering water is heavily stirred when it falls down to the CRT level. Secondly, during the stagnant phase, water is spread over the surface in a shallow layer, optimising surface aeration. At this point, 100% saturation is reached. Thirdly, sediment settles during the stagnant phase and light conditions are optimized for primary production and prevail for several hours often resulting in oxygen oversaturation. However, despite observations of oversaturation during field campaigns, the up till now available mass balance calculations do not fully affirm this hypothesis. More calculations and field campaigns have still to be performed.

7.3 CRT: a 'deus ex machina' solution for freshwater tidal marsh restoration?

Only one pilot CRT site of 8 ha has up till now been built. The tidal regime in the site has only been restored for two 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 conclusion on restoration success is thus preliminary.

This chapter however clearly points out the high potential of the CRT technique for successful restoration of structures and processes of reference freshwater tidal marshes (Table 7.3).

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 succes.

Table 7.3: Desired processes (italic) and structures (bold), their state in the pilot CRT compared to the estuarine freshwater tidal marshes, and their expected future state compared to the estuarine freshwater tidal marshes.

Desired **structures** and *processes*	CRT	Expected
Softwood forests	-	=
Helophyte vegetations	+	+
Pioneer vegetations	+	+
Merlon-krenel topography	-	=
Creek network	-	=
Si-buffering	80%	>100%
N-retention	(+)	?
P-retention	(+)	?
Oxygenation	(+)	+
Sedimentation	=	=

However, fundamental research is needed to justify freshwater tidal marsh restoration in the Schelde. The relative importance of tidal marsh processing compared to the estuarine loads is still unclear. The surface claim, based on previous research and backed by legislation, has to be detailed and updated continuously. To obtain enhanced and credible scientific backup for restoration policy and practice, societal goods and services of tidal marshes must be put central. These must be used to validate and, where necessary, adjust the restoration goals of tidal marshes.

A sustainable restoration science has to concentrate on gaps and uncertainties in the concepts supporting legislative claims and individual projects. All scientists have the responsibility to deliver policy support, albeit state-of the-art concepts or expert judgements. Restoration ecologists who are consulted by policy makers must stress the importance of adaptive policy and practice, both in the light of uncertainties and advances of science and of climate change impacts on ecosystems. Continuous, non-private funding is essential to guarantee sustainable qualitative science which serves the commonage.

Adriaensen Frank, Van Damme Stefan, Van den Bergh Erika, Van Hove Ditske, Brys Rein, Cox Tom, Jacobs Sander, Konings Peter, Maes Joachim, Maris Tom, Mertens Wim, Nachtergaele Lieven, Struyf Eric, Van Braeckel Alexander & Meire Patrick. 2005. Conservation goals for the Schelde estuary. Report Antwerp University, Ecosystem management research group (ECOBE).