SPECIAL PUBLICATION SJ2004-SP31

# SEDIMENT TRANSPORT AND DETRITAL TRANSFER CHANGES RELATED TO MINIMUM FLOWS AND LEVELS

UFL/COEL-2004/003

# SEDIMENT TRANSPORT AND DETRITAL TRANSFER CHANGES RELATED TO MINIMUM FLOWS AND LEVELS

**Final Report**

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June 2004

# SUMMARY

Methods to assess changes in detrital transfer and sediment transport (and load) as part of the protocol for Minimum Flows and Levels (MFLs) in rivers and lakes are addressed in this report.

Detrital transfer in the present context refers to the movement of organic-rich sedimentary material from the banks into the water column when high water levels occur. This “external” contribution to sediment load is easily addressed as a simple linear equation relating the rate of transfer to the inundated bank area. The load-scaling coefficient of this equation is wholly site-specific.

A noteworthy issue to be considered is that total suspended solids (TSS) concentrations correlate in Florida not only with water discharge but also with seasonal conditions. We also note that the lack of a direct correlation between measured TSS concentration and peak discharge in the St. Johns River suggests that part of the suspended sediment transport in this river may be wash load. Since wash load is not present in the bed, it has no bearing on river morphology.

Long-term (years to decades) impacts of a change in the sediment transport regime can be assessed by use of numerical models that are reasonably robust, but a hindrance to their application is the lack of data on TSS concentrations, as well as on bottom sediment composition and accumulation rate. In addition, standard present-day modeling approaches are useful for short-term predictions but cannot accurately predict long-term changes to the system relevant to MFLs. This is so because morphodynamic relationships between river geometry and flow are neither well known nor adapted to existing models. As a result, sediment transport models can predict depth changes but not width, making their application untenable for long-term assessments. At any rate, it appears that in Florida’s sediment-starved environment the time-scale

for a river system to reach a new equilibrium following any significant perturbation may range from years to several decades. This being the case, the relationship between the design life of the project and physical changes to the system due to, for example, sea level rise must also be taken into consideration.

We have developed a simple morphodynamic model to illustrate the role of water off-take on river morphology. It must be pointed out, however, that this modeling technique is in its infancy and has not been applied to Florida's interior waters. In order to make an assessment of the state-of-the-art in morphodynamic modeling techniques for rivers, a comprehensive review of such models in use in the United States, The Netherlands, Italy and other countries should be carried out. A detailed assessment should be made of the immediate applicability of models to Florida, model and data limitations, and further improvements required.

A practical difficulty in any impact analysis is the fact that throughout the St. Johns River Water Management District as elsewhere in Florida, herbicidal spraying is carried out on a wide scale in lakes and rivers. This operation adds organic-rich black mud or muck in a way that is not easily quantified, and possibly overwhelms other sediment sources such as detrital transfer by natural means.

A detailed MFLs analysis including spatial variability in loading and the effect of off-take must rely on a comprehensive approach, e.g., the use of the Environmental Fluid Dynamics Code (EFDC). Even then, it may be difficult to accurately predict the resulting change in the bottom sediment redistribution, especially in a small lake, i.e., one in which the effect of off-take is significant throughout.

Significant sediment loads (including those due to detrital transfer) generally occur only under episodic flow peaks in Florida. Our initial assessment in regard to critical thresholds for detrital transfer and sediment transport is that relatively small off-takes, e.g., less than ~10% of annual water volume discharged, do not alter peak flows in any significant way, and may not show measurable changes in the frequency and rate (in relation to data noise and natural variability) over time scales of years to decades. We must also add that time periods on the order of decades cannot be considered in MFLs assessments without recognizing that natural changes in discharge and water level forcing, especially sea level rise, may entirely subsume any effects of off-take.

As a starting point for the establishment of critical thresholds we recommend the following task for possible implementation:

> We recommend that prismatic channel modeling, the basis of which we have demonstrated, be expanded to examine the off-take induced response (in peak magnitude and frequency of occurrence) of TSS and sediment load to long-term (i.e., decade or more) hydrologic time-series. With further developments, this analysis could be used to develop a general methodology for establishing MFLs thresholds. Transport of sand (as bedload and suspended load) as well as fine sediment (as suspended load) should be included. This effort, using presently available data, will determine if additional information on sediment transport is required, or whether in Florida's environment it is sufficient to develop a "generic template" for analysis applicable to most MFLs project assessments.

The modeling exercise will also assist in identifying data needs. If it is concluded that sediment related information is required for model methodology improvements, the following additional studies should be considered.

1) Given that MFLs assessments must rely on long-term hydrology, a test site where USGS stage-discharge gaging is carried out might be selected for a one-year monitoring

of flows and suspended solids. For *suspended load* the latest Acoustic Doppler Current Profiling (ADCP) technology may be used. Testing would include running cross-sectional transects by using an ADCP mounted on a vessel, along with deployment of a bottom-mounted ADCP. We also recommend that a trench (e.g., 2 m deep below existing bottom, 2-3 m wide in the flow direction and 5-10 m perpendicular to the flow) be excavated at the site to estimate *bedload* transport of sand.

This pilot study will serve as the basis to establish long-term sites at other cross-sections for the development of TSS-discharge rating relationships required for sediment transport modeling. Multiple sites may be required as the rating relationship is unique to each site. At present almost no synoptic information of this type is available.

2) Over a selected reach of a river or a lake, the Multi-Detector System for Underwater Sediment Activity (MEDUSA), a recently developed method for rapid surveying of bottom sediment composition (sand and fine sediment), may be tested for use throughout the St. Johns River Water Management District. This approach must include bottom coring for “ground-truthing” within the selected reaches. If successful, it can lead to the development of a database for bottom sediment accumulation and its transport under natural and anthropogenic influences. Information of this type is available for some lakes but the rivers generally have not been surveyed.

3) In order to make an assessment of the state-of-the-art in morphodynamic modeling techniques for rivers, a comprehensive review of such models in use in the United States, The Netherlands, Italy and other countries should be carried out. The immediate applicability of models to Florida, model limitations and further improvements required should be determined. Modeling using this approach is required for assessing long-term

(spatial and temporal) changes associated with MFLs. Presently such modeling is not carried out in Florida.

4) In conjunction with the above tasks we recommend that a database containing flow cross-section surveys be developed. This will require that a certain number of sites be accurately surveyed on a periodic basis. Bottom core data should be collected at these sites to assess and dated to determine deposition rates. This technique is a “non-modeling” albeit *post facto* means to assess MFLs related changes.

The above four follow-up tasks will set the basis for model improvement and possible data needs. Additional effort will be required to identify critical threshold criteria for off-take. These criteria depend on the relationship between key environmental parameters and the statistics of long-term hydrologic time series related to discharge, water level, sediment load and detrital transfer, and also on changes in the morphology of the water body.

**ACKNOWLEDGMENT**

The authors thank Bob Epting, Sonny Hall, Cliff Neubauer, Price Robison and Chris Ware of the St. Johns River Water Management District, Florida for their technical assistance and comments. Funds for the study were provided by the St. Johns River Water Management District.

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## 1. INTRODUCTION

Currently an effort is underway in Florida to establish Minimum Flows and Levels (MFLs) at rivers, lakes, wetlands and springs within the state. According to Section 62-40.473 of FAC, all MFLs projects must consider the protection of non-consumptive uses of water such as for navigation, recreation, fish and wildlife habitats, and other natural resources values. They define the maximum extent to which natural flows can be depleted or otherwise altered by human activity (Neubauer et al. 2004).

As a case in point, the St Johns River Water Management District (SJRWMD) is considering off-take of surface water from a presently unspecified location within the reach of the St. Johns River south of the town of Astor (Fig. 1.1). Off-take on the order of 7-10% of mean discharge might be available for consumptive use from the river or possibly from Lakes Dexter or Woodruff. Such off-take is considered to be an alternative to ground water usage.

In general, a total of ten conditional elements are specified for consideration in MFLs assessment:

1. Recreation in and on water
2. Fish and wildlife habitats and passage of fish
3. Estuarine resources
4. *Transfer of detrital material* (italics added)
5. Maintenance of freshwater storage and supply
6. Aesthetic and scenic attributes
7. Filtration and absorption of nutrients and other pollutants
8. *Sediment loads* (italics added)
9. Water quality
10. Navigation

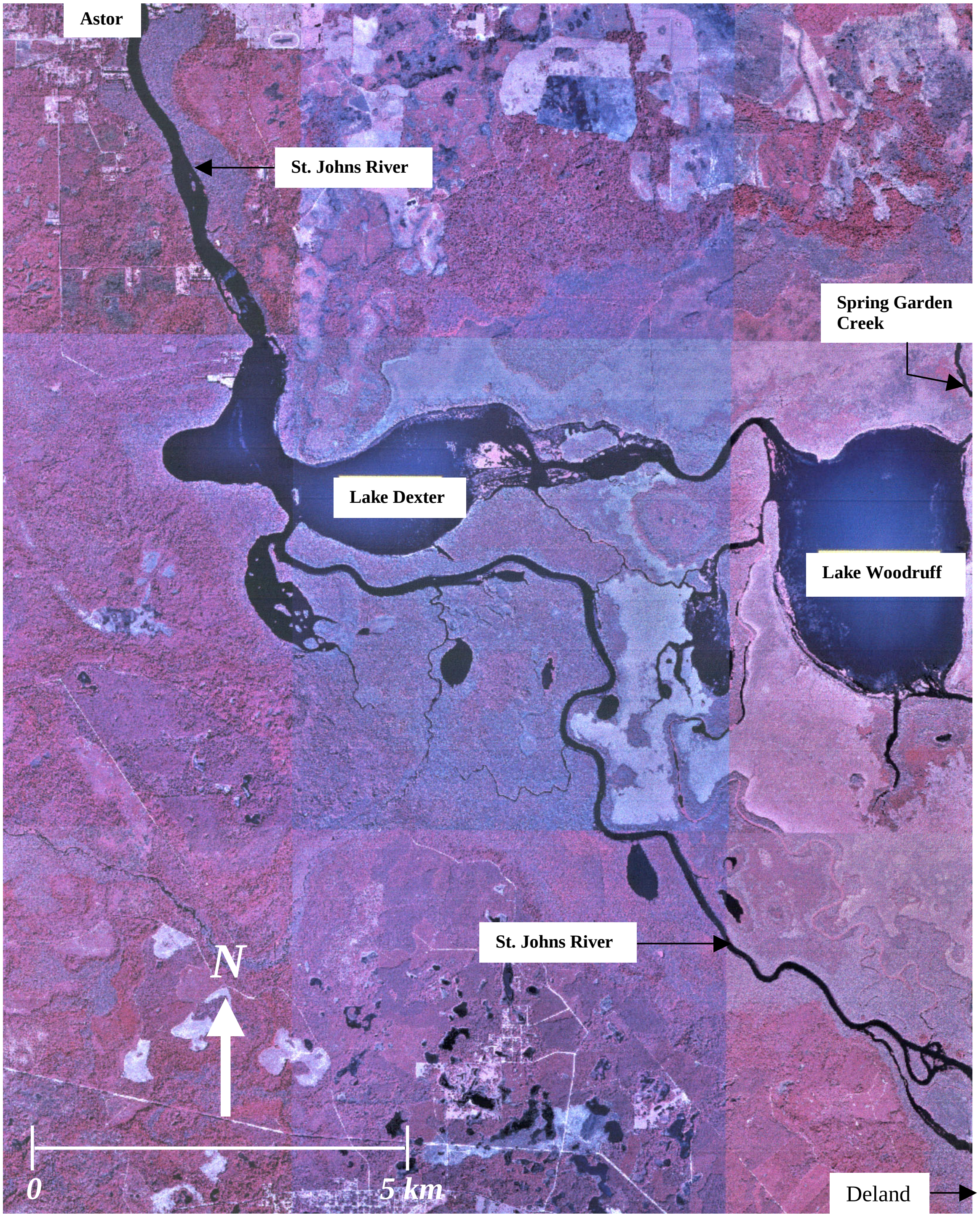


Figure 1.1 St. Johns River south of Astor and Lakes Dexter and Woodruff

Of these ten elements, this report will identify and document methods for assessing changes in *transfer of detrital material and sediment loads*, both related to hydrologic conditions limited by MFLs. In what follows, a summary of the relevant sedimentary setting in Florida will provide guidance in regard to the basis for these methods.

## 2. SEDIMENTARY ENVIRONMENT

### 2.1 Preamble

Methods for assessing sediment transport and detrital transfer are inherently dependent on the sedimentary setting. Consequently, a summary of this setting is provided, focusing on the St. Johns River.

### 2.2 St. Johns River and Tributaries

As mentioned, consideration is being given to installing an off-take for fresh water somewhere in the reach of the St. Johns River south of the town of Astor. This reach, indeed much of the river, is poorly surveyed for bed sediment and suspended sediment regimes. Many of the lakes as well are not well studied. Nevertheless, three reports provide relevant information: ECT (2002), Durell et al. (1998) and Kroening (2004).

### 2.3 Bed Sediment Regime

It can be anticipated that the bed sediment regime of the St. Johns River will not be as dynamic as that of the larger lakes, e.g., Dexter and Woodruff, which are susceptible to frequent wind stirring. Durell et al. (1998) provide sediment data for the bed samples analyzed as shown in Table 2.1.

Table 2.1 Bed sediment data (Durell et al. 1998)

| Sample | Moisture (%) | TOC[a] (%) | TS[a] (wet wt) (%) | TVS[a] (dry wt) (%) | Sand (%) | Silt (%) | Clay (%) | Mud (silt & clay) (%) |
|---|---|---|---|---|---|---|---|---|
| 02236000 | 62 | 2.35 | 38.4 | 7.2 | 64.5 | 32.9 | 2.6 | 35.5 |
| 20010002 | 27 | 0.27 | 72.9 | 0.6 | 93.1 | 5.8 | 1.1 | 6.9 |
| 20010003 | 60 | 2.03 | 40.4 | 4.3 | 62.7 | 34.5 | 2.7 | 37.2 |
| BLUSPA | 28 | 0.17 | 72.4 | 0.5 | 97.4 | 2.1 | 0.5 | 2.6 |

[a] TOC = Total Organic Content; TS = Total Solids; TVS = Total Volatile Solids.

In the above data, sample 02236000 is from the St Johns River near Deland, south of Lake Dexter and west of the town. Sample 2001002 is from a bridge at Route 40 near Astor, between Lakes George and Dexter, and 20010003 is from a bridge crossing Routes 17 and 92. BLUSPA is from Blue Springs near Orange City. In two cases the silt fractions are low and the sediments are correctly termed sands. The silt/sand mixtures near Deland (02236000) and at 20010003 will behave as cohesive sediment. Neither seems to be a site of progressive and large-scale black organic mud accumulation. Naturally, we know nothing about the mobility or otherwise of the sand and silty sand deposits. Further, some reaches of the St. Johns River near Deland have been dredged in the past. In these reaches mud may have accumulated since.

A matter of some relevance is the chemistry of bed sediments in the fairway of the river (Table 2.2). This is not a component of the present investigation but we do need to be aware of it. In particular, we observe that metal and organic contaminant levels are generally low. There are very few samples, but we particularly note the Phthalate concentrations in 20010002 in the river-bed at Route 40 near Astor. The site is located between Lakes Dexter and George. The value of 1580 μg/kg/%TOC appears to be high, especially in view of the fact that this is a sandy deposit (analysis is only of the fine organic sediment component). Phthalates are used in plastics manufacture to render plastics flexible.

Table 2.2 Chemical species in river-bed sediment (Durell et al. 1998)

| Chemical species | 2236000 St Johns R near Deland South of L Dexter | 20010002 St Johns R at Highway 40 between L George & L Dexter | 20010003 St Johns R at US Highway 17 & 92 in outlet to L Monroe | BLUSPA Blue Springs near Orange City |
|---|---|---|---|---|
| Organic Components. Normalized to %TOC µg/kg/% | | | | |
| Total PAH | 75.1 | 151 | 338 | 21.7 |
| Low PAH | 10.3 | 26.6 | 107 | 17 |
| High PAH | 64.8 | 124 | 231 | 4.68 |
| Total Phthalate | 19.6 | 1,580.00 | 13 | 366 |
| Organic Contaminants. PCB Pesticides & other Chlorinated Compounds Normalized to %TOC µg/kg/%TOC | | | | |
| Σ DDTs | 0.508 | ND | 1.66 | ND |
| Σ PCB | 2.59 | 14.8 | 3.11 | 13.9 |
| Σ Chloros | 1.71 | 4.82 | 0.886 | ND |
| Σ BHCs | ND | ND | ND | ND |
| Σ Chlordane | ND | ND | 0.186 | ND |
| DDE | 0.155 | ND | 0.598 | ND |
| DDD | 0.157 | ND | 0.201 | ND |
| DDT | 0.196 | ND | 0.856 | ND |
| Concentration Ranges for Major & Trace Metals. Normalized to %TOC µg/kg/%TOC | | | | |
| Al | 3420 | 2800 | 5070 | 14600 |
| As | 0.421 | ND | 0.429 | 2.59 |
| Cd | 0.0426 | ND | 0.067 | 2.11 |
| Cr | 5.66 | 9.67 | 7.93 | 48.1 |
| Cu | 1.54 | 3.54 | 2.24 | 8.07 |
| Fe | 1830 | 2650 | 3240 | 11700 |
| Pb | 3.4 | 4.83 | 4.25 | 16.5 |
| Li | 2.28 | 7.05 | 5.02 | 10.4 |
| Mn | 43 | 76 | 38.3 | 464 |
| Hg | 0.015 | 0.021 | 0.02 | 0.166 |
| Ni | 0.906 | 1.84 | 2.48 | 9.82 |
| Se | 0.268 | ND | 0.27 | 1.2 |
| Ag | 0.009 | 0.059 | 0.025 | ND |
| Sn | 0.3 | 0.827 | 0.313 | 1.72 |
| Zn | 8.3 | 9.3 | 6.21 | 35.7 |

## 2.4 Suspended Sediment Regime

We note that the report ECT (2002) uses the term “detrital” in reference to much of the sediment load of the St. Johns River and applies it to dead organic material. This is in keeping with the use of this term in MFLs assessment, but is not the sense generally used by sedimentologists. The sedimentologic definition is:

*Detrital* (same as clastic or *allogenic*) - Of sediments made up of fragments produced by breaking-up of earlier (igneous, sedimentary or metamorphic) rocks (e.g., Watt 1982).

*Detritus*: Material produced by the disintegration and weathering of rocks that have been moved from their site of origin (hence use of the word allogenic-allochthonous above means moved in from a site outside its present location, as opposed to authigenic - created in situ).

This is an important distinction, as it is of interest to know if sediments suspended in the St. Johns River are authigenic or allochthonous (detrital). In fact, evidence shows that little or none of the sediment in question may be detrital in the sedimentological sense. In this case recent data are instructive. Kroening (2004) has published velocity and suspended solids/chlorophyll-*a* data sets for the St. Johns River near Deland, Florida for the period October 1, 1999 to October 1, 2002. There are data sets of similar duration for the St. Johns River near Christmas and for Cocoa. (Christmas is upstream of Deland situated between Orlando and the JFK Space Center; Cocoa is on the Atlantic Coast.)

The data sets show a signal that can be interpreted as follows. At Christmas near Deland and at Cocoa there are autumnal flow peaks (Table 2.3). Importantly, the suspended sediment concentration in these data sets also shows a strong seasonality. In each year there are spring/early summer season peaks (Table 2.4).

Table 2.3 Flow peaks (Kroening 2004)

| Year | Christmas (month) | Deland (month) |
|---|---|---|
| 1999 | (?) Oct, Nov, Dec | (?) Oct, Nov, Dec |
| 2000 | Oct, Nov, Dec | Weak peaks |
| 2001 | Aug, Sep, Oct, Nov | Oct, Nov, Dec |
| 2002 | Jul, Aug, Sep (?) | Jul, Aug, Sep (?) |

(?) Means no data prior to or after.

Table 2.4 Suspended sediment peaks (Kroening, 2004)

| Year | Christmas (month) | Deland (month) |
|---|---|---|
| 1999 | No summer data | No summer data |
| 2000 | Mar, Apr, May, Jun, Jul, Aug, Dec | June |
| 2001 | Feb, Mar, Apr, May, Jun, Jul | Feb, Mar, Apr, May, Jun, Jul |
| 2002 | Feb, Mar, Apr, May, Jun | May, Jun, Jul |

The plots for Christmas and Deland shown in Figs. 2.1 and 2.2, respectively, indicate that suspended solids are high in spring and early summer, whereas velocities peak in autumn/winter. Indeed, the graphs are "reciprocals" one of the other (even though this trend is not found with regard to sediment load; Fig. 2.1). Conventionally, detrital, inorganic suspended sediment concentrations will peak in response to raised hydrodynamic stress (e.g., Graf 1971). In the present case, the higher autumn/winter velocities do not entrain sediment in significant quantities. If so, then it is a matter of importance to obtain long-term data on discharge and TSS in order to develop appropriate sediment rating relations applicable to rivers within the St. Johns River Water Management District.

The reason for the lack of synchronicity between flow and solids concentration is found in Kroening (2004), which reports a three-year data set for the same station at Deland and another near Cocoa. The data, Figs. 2.3 and 2.4, indicate that primary production, as determined by Chlorophyll-*a* production, peaks in spring/early-summer and bears an inverse relationship to river discharge.

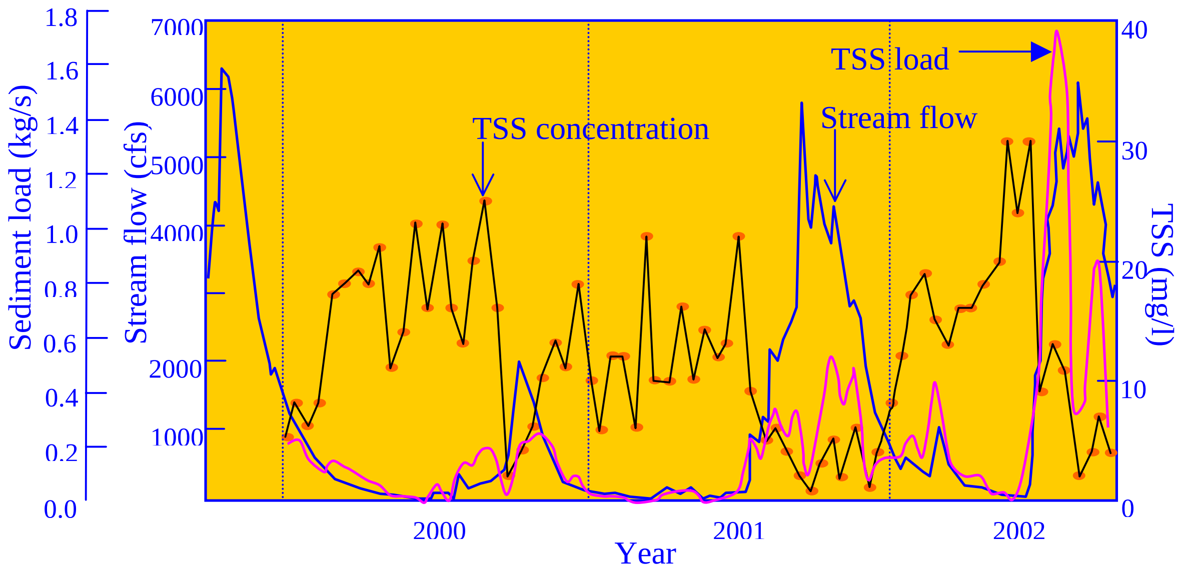


Figure 2.1 Stream flow discharge, total suspended solids concentration and sediment load data in St. Johns River near Christmas (Kroening 2004)

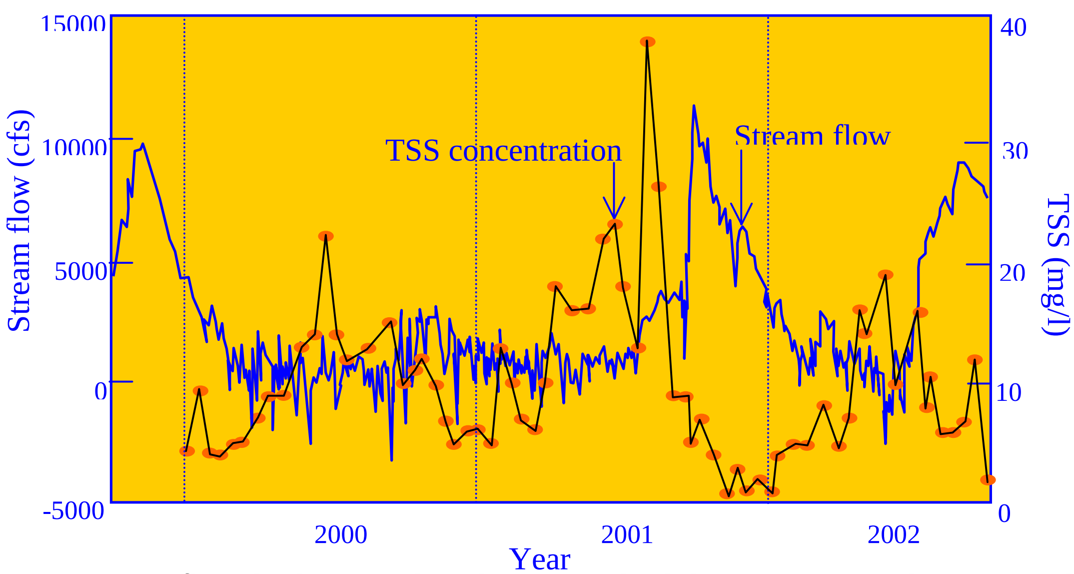


Figure 2.2 Stream flow discharge and total suspended solids concentration data in St. Johns River near Deland (Kroening 2004)

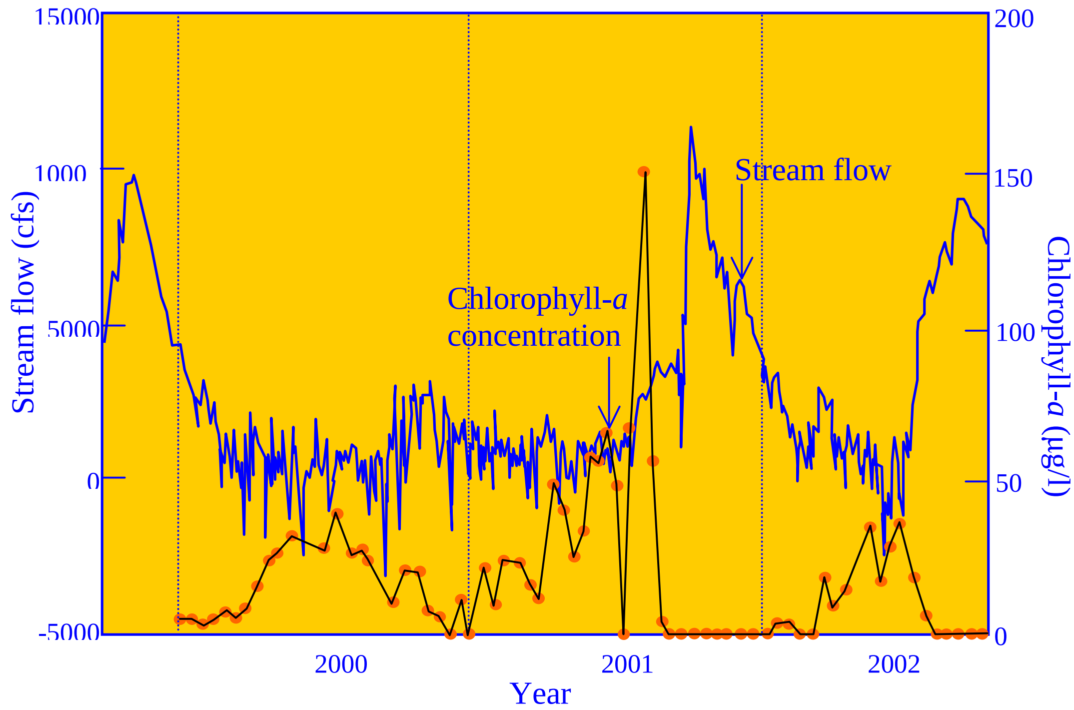


Figure 2.3 Stream flow discharge and chlorophyll-*a* concentration data in St. Johns River near Deland (Kroening 2004)

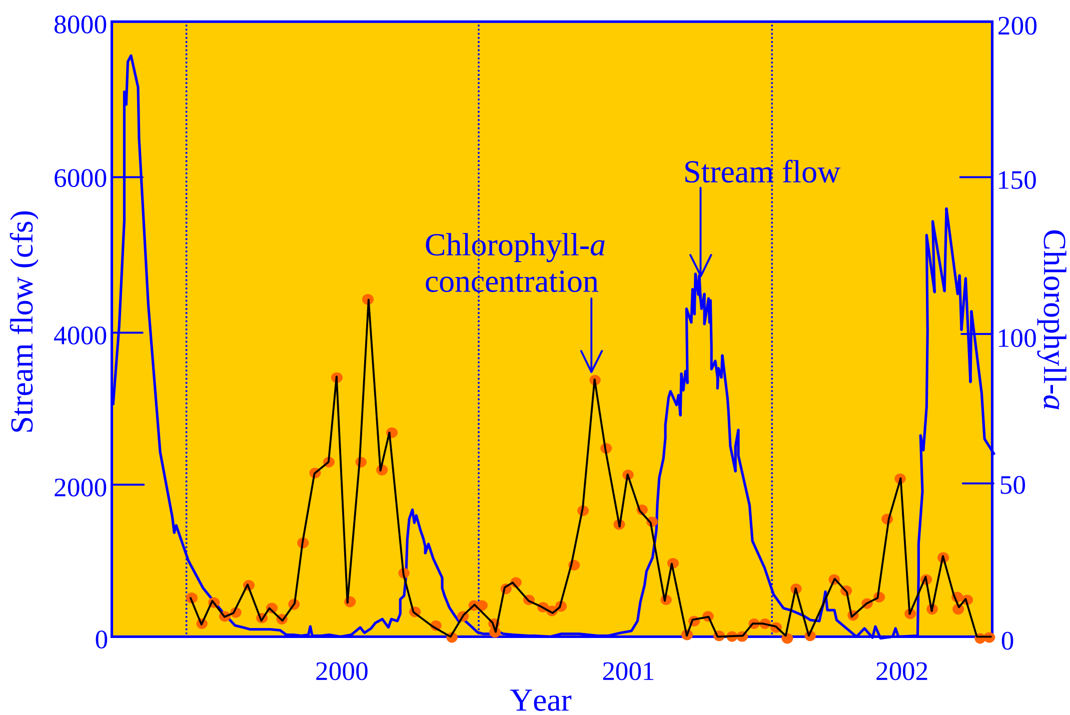


Figure 2.4 Stream flow discharge and chlorophyll-*a* concentration data in St. Johns River near Cocoa (Kroening 2004)

Table 2.5 Flow and chlorophyll peaks (Kroening 2004)

| Year | Deland | | Cocoa | |
|---|---|---|---|---|
| | Flow (month) | Chlorophyll-*a* (month) | Flow (month) | Chlorophyll-*a* (month) |
| 1999 | Oct, Nov, Dec | No data | (?)Oct, Nov | No data |
| 2000 | No peak | May, Jun, Jul, Aug | Oct | May, Jun, Jul, Aug, Sep |
| 2001 | Oct, Nov, Dec | Aug, May, Jun, Jul, Aug | Aug, Sep, Oct | Apr, May, Jun, Jul |
| 2002 | Jul, Aug, Sep(?) | Apr, May, Jun | Jul, Aug, Sep | May, Jun |

(?) Means no data prior to or after.

The evidence from the three-year data sets indicates that suspended solids concentrations in the St. Johns River peak in spring/early summer when flows are minimal and are a manifestation of primary production measured as Chlorophyll-*a*. In other words, i.e., the algal sediment appears to be authigenic and not detrital (in the way this term is defined in sedimentology).

In contrast to the above, we observe that in tidal reaches of the St. Johns River system, in the Cedar and Ortega estuaries, there is a detectable hydrodynamic component to the daily suspended sediment concentration. These data were obtained with a vessel-mounted Acoustic Doppler Current Profiler (ADCP) in conjunction with Sediview software (Mehta et al. 2004). In Fig. 2.5 we see the sediment flux at a cross-section. Sediment is entrained on the flood and ebb to some degree and settles to the bed at slack water. It is not known whether the sediment in the Cedar and Ortega is any different from that at Deland, Christmas and Cocoa. It might, for example, contain a significantly higher fraction of detrital, allochthonous, suspended sediment than is present in the unidirectional flow reaches of the river system. Nevertheless, total solids concentration was extremely low in these downstream reaches, only 2-14 mg/l. This cycle of the suspended sediment fraction in the lower reaches of the system may, alternatively, reflect no more than the effect of 6-hourly tidal slack water periods, which are absent from the unidirectional fluvial reaches.

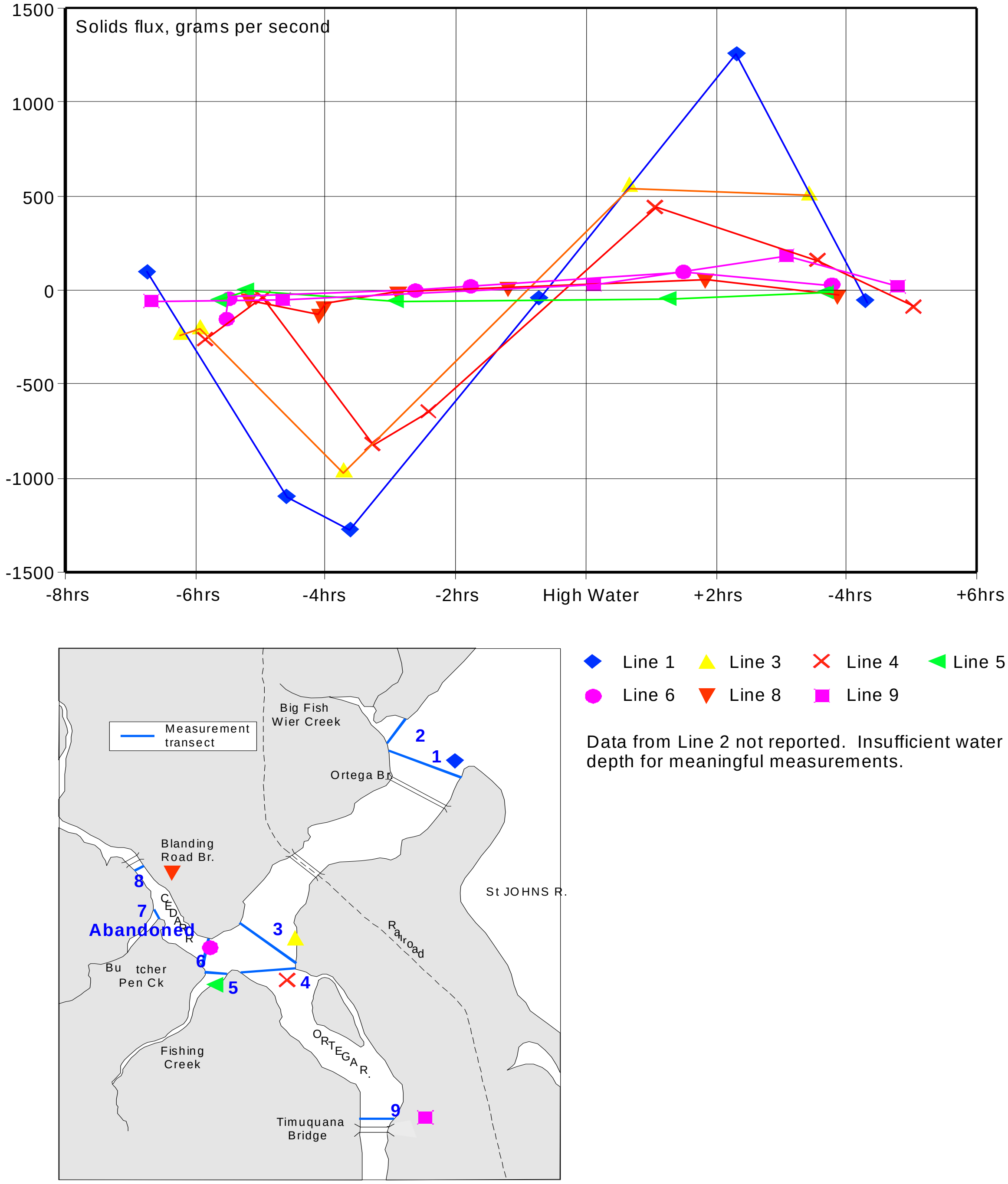


Figure 2.5 Sediment flux data measured with an Acoustic Doppler Current Profiler (ADCP) in Cedar and Ortega Rivers (Mehta et al. 2004)

With regard to the type of sediment, we must note that the three-year data sets of Kroening (2004) demonstrate flood discharge variations over this period but, presumably, do not include an extreme flood event (of a return period possibly equal to or greater than 10 years) that would be found in long-term hydrographs. We need to anticipate to the best extent possible, that an extreme flood event may well input *detrital* clays and sands swept out of the watershed and input to the river. Since these may have contrasting implications for MFLs, long-term records are useful in MFLs assessment.

## 2.5 Monitoring of Rivers

We are progressively developing an optimum ADCP installation to be mounted on an existing light beacon or a pile. It will be water surface mounted to follow variations in water elevation and will measure current, suspended sediment concentration and water level. The level data are only required to evaluate any loss of water surface elevation and its areal extent. It is unclear for now whether it would be best to use RDI Inc.'s Workhorse Sentinel or the Rio Grande system. This choice is merely a matter of power supplies, etc.

## 2.6 Lakes

### *2.6.1 Sediment Regime of Lakes*

The St Johns River watershed basin has a greater number of lakes of all sizes than occur in other water management districts in the state. The reason for the preponderance of lakes in the St. Johns River Water Management District region is related to the underlying geological control and the rock types in the basin.

The amount of black organic mud (or muck) in lakes in Florida is known from surveys to be variable but to follow three main patterns - evenly distributed, distributed around the margins and focused in hollows in the lake bed. Some lakes are devoid of muck. Danek et al. (1991)

studied a group of seven lakes only about 45 km west of Deland (Table 2.6). The mud volume was found to vary by a large amount. Figure 2.6 indicates that Beauclair, Dora, Weir and Yale all contain modest (volumetric) amounts of mud. Beauclair (organic sediment 75%) and Weir (73%) have the smallest proportion of their lake beds covered by significant depths of mud.

Figure 2.7 shows extensive areas with a thin-mud cover, or with no mud, in Lake Beauclair. There are similarly extensive areas in Lake Weir (Fig. 2.8), and more limited areas in Lake Dora (Fig. 2.9). By comparison, Lakes Griffin (organic sediment 95%), Harris (97%) and Eustis (94%) have extensive coverage.

Table 2.6  Sediment in Deland Area Lakes (Danek et al. 1991)

| Lake | Black mud volume ($m^3$) | Surface area ($km^2$) | Depth (av.) (m) | Depth (max.) (m) | Sediment thickness (max.) (m) |
|---|---|---|---|---|---|
| Beauclair | $9.1x10^6$ | 4.4 | 2.0 | 4.7 | 5.2 |
| Dora | $5.3x10^7$ | 17.9 | 3.0 | 5.2 | 5.4 |
| Eustis | $1.1x10^8$ | 31.5 | 3.4 | 6.6 | 5.4 |
| Griffin | $2.8x10^8$ | 38.2 | 2.3 | 6.1 | 6.7 |
| Harris/Little Harris | $9.0x10^7$ | 75.9 | 3.7 | 9.7 | 8.8 |
| Weir | $1.3x10^8$ | 16.4 | 5.7 | 9.7 | 3.8 |
| Yale | $6.1x10^7$ | 22.9 | 3.7 | 7.9 | 5.6 |

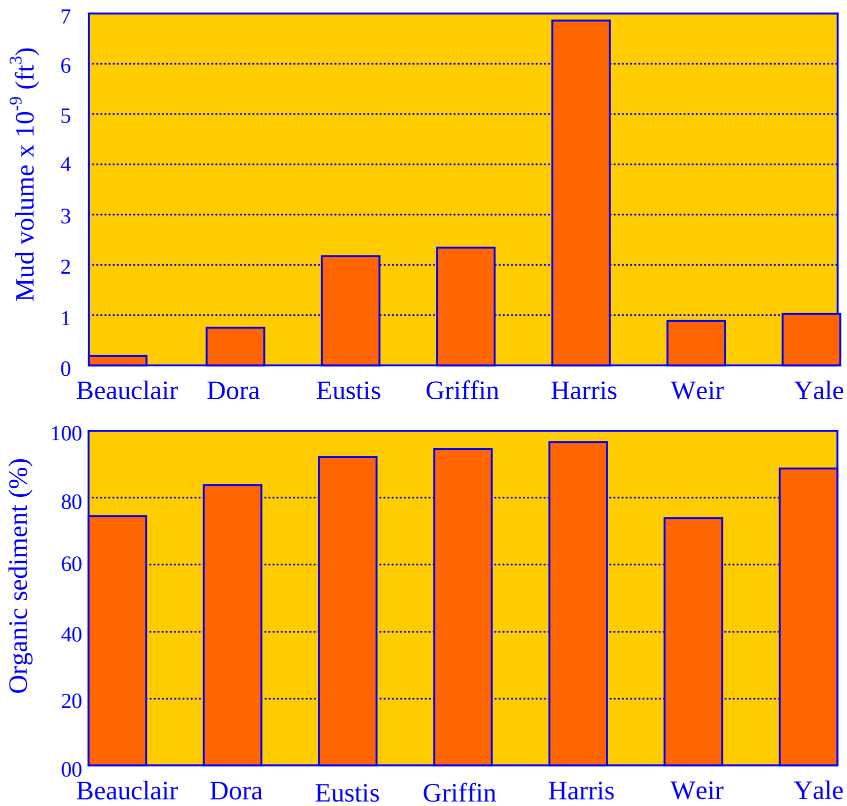


Figure 2.6  Mud volume and organic sediment content in seven lakes (Danek et al. 1991)

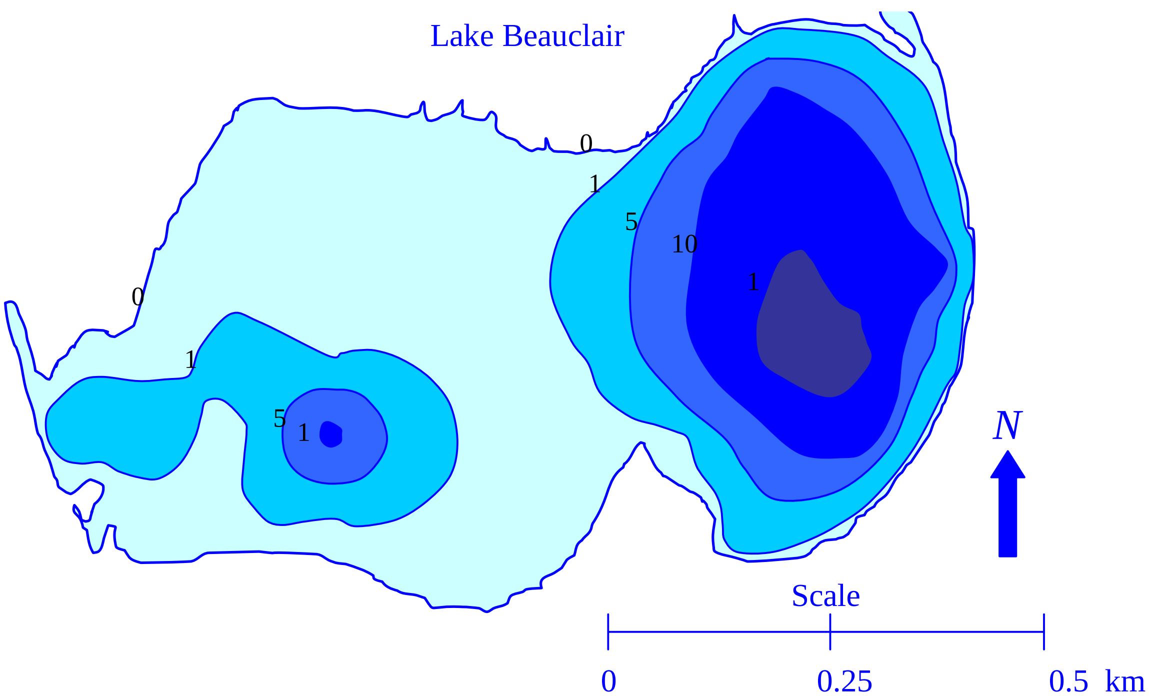


Figure 2.7  Bathymetry of Lake Beauclair (mud thickness is in feet) (Danek et al. 1991)

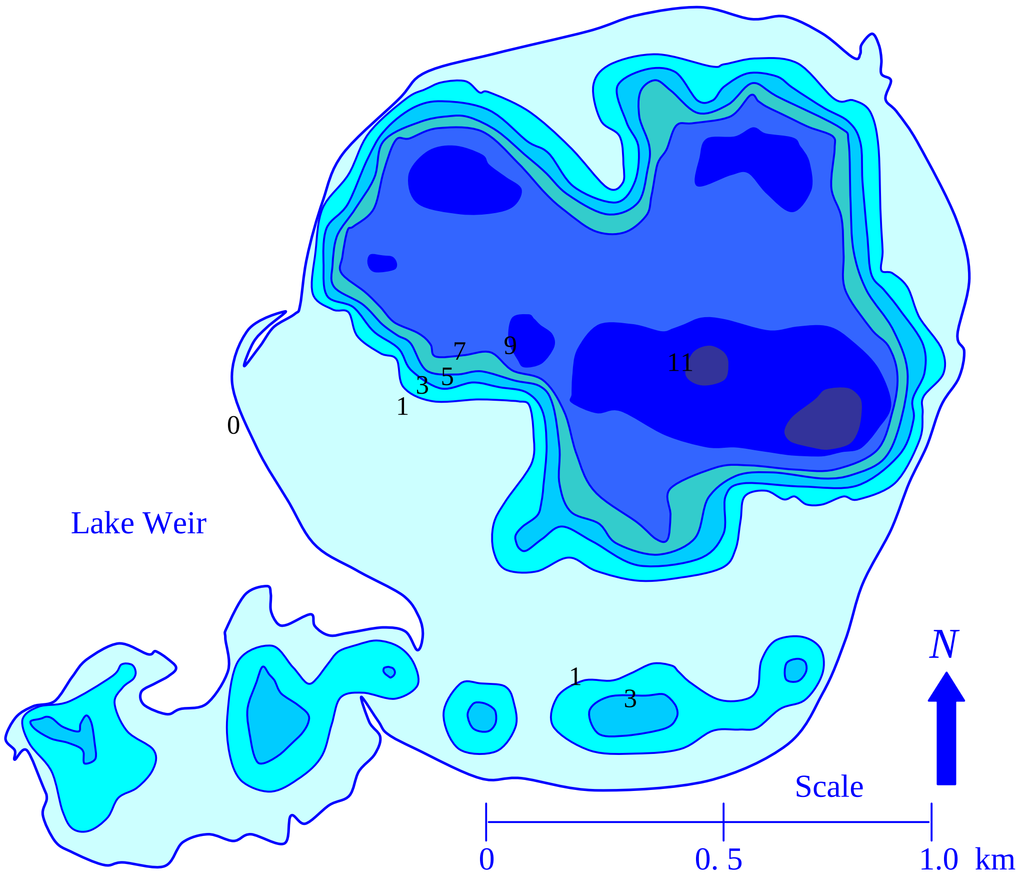


Figure 2.8 Bathymetry of Lake Weir (mud thickness is in feet) (Danek et al. 1991)

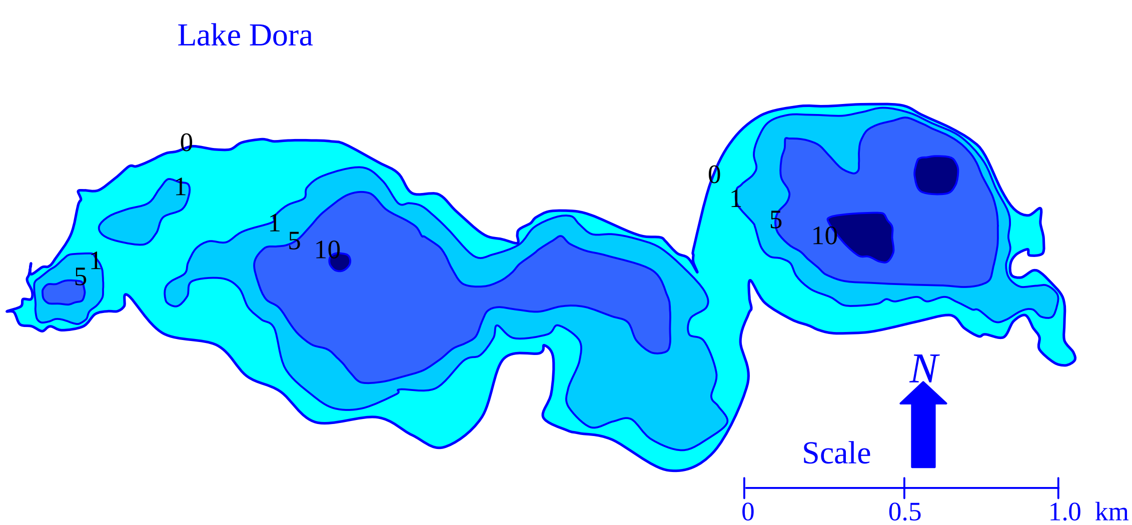


Figure 2.9 Bathymetry of Lake Dora (mud thickness is in feet) (Danek et al. 1991)

Danek et al. (1991) make no comment on the reasons for the variability of organic black mud cover, confining their attention to the consequences for management of either raising or lowering lake levels. It is notable that lake water depth, as with sediment depth, was also quite variable. Lake Weir has the second greatest volume but the fourth largest area, i.e., it is relatively deep (Table 2.6).

A corollary question is whether the lake sediments themselves or their input are contaminated. No chemical analysis for black organic mud in Lake Dexter, if present, has been published but measurements are available from Lake George, Lake Woodruff (a single sample) and Lake Monroe. A sample has also been analyzed from Lake Winnemissett near Deland, and another from Lake Winona near De Leon Springs and Lake Woodruff (Table 2.7).

The sampling and analyses were undertaken by Durell et al. (1998). There are particularly low concentrations of contaminants in Lakes Woodruff and Winnemissett. In Lake Monroe, concentrations of a range of contaminants are slightly elevated compared to these smaller lakes. It is noted that Monroe, in contrast to the two above, is near Deltona (to the north) and Sanford (to the south), which may lead to these slightly raised concentrations. For some reason Lake Winona has raised concentrations of some contaminants (e.g., total phthalate, DDT). This lake occurs in a semi-rural area east of Deland.

Table 2.7 Chemical species in lake-bed sediment (Durell et al. 1998)

| Chemical Species | L Woodruff | L Monroe | L Winnemissett | L Winona |
|---|---|---|---|---|
| Organic Components. Normalized to %TOC in μg/kg/%TOC | | | | |
| Total PAH | 13.4 | 126 | 20.1 | 157 |
| Low PAH | 4.13 | 17.3 | 4.57 | 36.9 |
| High PAH | 9.24 | 109 | 15.5 | 120 |
| Total Phthalate | 61.4 | 7.08 | 44.9 | 501.1 |
| Organic Contaminants, PCB Pesticides & other Chlorinated Compounds | | | | |
| Normalized to %TOC in μg/kg/% TOC | | | | |
| Σ DDTs | 0.409 | 1.87 | 0.556 | 3.23 |
| Σ PCB | 2.05 | 3.07 | 2.58 | 24.5 |
| Σ Chloros | 0.845 | 1.22 | 1.34 | ND |
| Σ BHCs | ND | 0.303 | ND | ND |
| Σ Chlordane | ND | 0.119 | 0.257 | ND |
| DDE | 0.124 | 1.17 | 0.323 | 0.928 |
| DDD | 0.286 | 0.496 | 0.233 | 1.76 |
| DDT | ND | 0.207 | ND | 0.545 |
| Concentration Ranges for Major & Trace Metals. Normalized to %TOC in μg/kg/%TOC | | | | |
| Al | 1310 | 2430 | 2590 | 7300 |
| As | 0.14 | 0.153 | 0.558 | 0.562 |
| Cd | 0.012 | 0.049 | 0.065 | ND |
| Cr | 1.82 | 3.98 | 2.51 | 27.6 |
| Cu | 0.465 | 1.65 | 1.51 | 7.58 |
| Fe | 793 | 1710 | 537 | 5680 |
| Pb | 1.19 | 2.31 | 6.34 | 19.7 |
| Li | 0.935 | 2.32 | 1.43 | 13.6 |
| Mn | 4.79 | 8.11 | 14.1 | 265 |
| Hg | 0.0078 | 0.224 | 0.045 | 0.48 |
| Ni | 0.58 | 1.22 | 0.965 | 3.57 |
| Se | 0.126 | 0.106 | 0.651 | ND |
| Ag | 0.0069 | 0.048 | 0.007 | ND |
| Sn | 0.056 | 0.178 | 0.22 | 2.07 |
| Zn | 1.33 | 4.7 | 9.83 | 58.7 |

Brenner et al. (1999) studied spatial and temporal patterns of sediment and nutrient storage in the Upper St Johns River watershed. They produce no map but do itemize black organic mud depth variations in lakes of these reaches (Table 2.8).

Table 2.8 Black organic mud thicknesses (Brenner et al., 1999)

| Lake | Sediment depth (cm) |
|---|---|
| Hell n' Blazes | 20-102 |
| Sawgrass | 13-41 |
| Washington | 0-64 |
| Winder | 0-18 |
| Poinsett | 0-48 |

Brenner et al. (1999) indicated that lakes Hell n' Blazes and Sawgrass have soft organic deposits distributed uniformly throughout. These have dense and extensive stands of submerged epiphytes and associated macrophytes, which reduce flow velocity and intercept sediment particles. This, combined with a short fetch, prevents resuspension and downstream transport of sediments. In contrast, Lake Washington has large expanses of open water. Much of the lake bed is characterized by a sandy bottom and black organic mud is largely restricted to the northern end of the lake. Brenner et al. suggest that Washington is probably wind mixed and resuspended black organic muds are redistributed to downstream sites. There is negligible discussion of Lakes Winder and Poinsett.

Another lake in the St. Johns River headwaters, Blue Cypress Lake, has been studied by Brenner et al. (2001). From the description given it is not clear whether the lake is vegetation-covered or whether the bed is covered by black organic mud.

### *2.6.2 Monitoring of Lakes*

The monitoring regime to evaluate MFLs changes for a lake contrasts with that of a unidirectional flow river. There may be no evident "upstream" or "downstream" locations relative to an off-take to place recording/monitoring devices. It may be the case that no permanent long-term monitoring is required, but in any case monitoring would involve at least one tower fitted with a water level gauge, a flow meter and suspended solids monitors. Some limited bed sampling before and after the installation would be used to establish any changes in the bed sediment regime.

# 3. TASKS AND RESPONSES

The report is structured on the basis of responses to five assigned tasks. Details are included in Appendices A through F.

## 3.1 Task 1(a) Definitions

*Task: Discuss differences and similarities between lakes and river systems with respect to sediment transport and detrital transfer.*

*Response:* At the outset it is essential to define: 1) sediment transport, 2) sediment load and 3) detrital transfer, in the MFLs context.

### *3.1.1 Sediment Transport*

According to common (but possibly not universal) definition (excluding aeolian transport), sediment transport includes *subaqueous* movement of all particulate matter. In general, for a given flow, transport depends on sediment sources and composition. Composition is most commonly described by particle size, and classified as coarse or fine (Table 3.1). The word "clay" is also used to describe fine-grained soil having plasticity. One can avoid confusion by employing "clay size" rather than merely "clay" to denote a particle smaller than 2µm (2 microns; 1 µm = $10^{-6}$ m), because clay is a specific type of inorganic (crystalline) mineral. Table 3.1 separates particles at the 0.063 mm boundary such that particles less than this size are classified as fine.

Table 3.1 Classification of particles by size (Graf 1971)

| Name | Size (mm) | Designation |
|---|---|---|
| Boulder | > 305 | Coarse |
| Cobble | 52 to 305 | |
| Gravel | 2 to 52 | |
| Sand | 0.063 to 2 | |
| Silt | 0.002 (20 µm) to 0.063 (63 µm) | Fine |
| Clay | < 0.002 (2 µm) | |

### *3.1.2 Sediment Load*

Sediment *load* is (total) sediment mass or volume per unit time, e.g., kilograms or cubic meters per second, in transport. When the motion of particles is one of rolling, sliding or jumping (also called saltating), and when this particle motion takes place close to the bed, the associated sediment transport of bed material is commonly referred to as *bedload* transport (Fig. 3.1).

When the entire motion of the solid particles is such that they are surrounded by fluid, the particles are said to move in suspension, i.e., they are transported as *suspended load*. Due to the weight of the particles there is a tendency for settling, which, however, is counter-balanced by the irregular motion of the fluid particles and associated turbulent diffusion. Thus the flow condition determines if and when a given size fraction will be in suspension. Furthermore, sediment particles that are part of suspended load at one time may, at another time, be part of bedload, and vice versa. There exists an active, continuous, interchange between the suspended load and bedload, and also between the bedload and the bed itself.

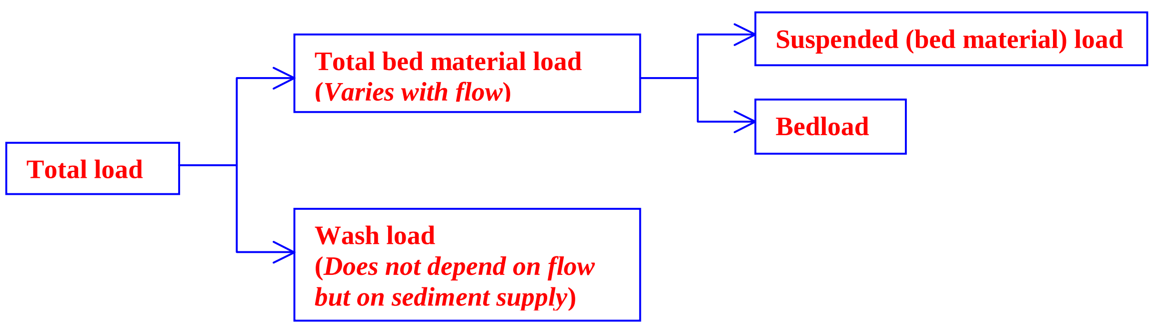


Figure 3.1 Sediment load definitions. Note that total load equals total bed material load minus wash load.

*Wash load* consists of very fine particles that tend to remain suspended under given flow conditions. These particles are brought into the stream by overland flow - dust and light-weight organic matter (e.g., algae) - and are usually not present in quantities in the bed material. Therefore, wash load is not bed material load. There can be flow conditions under which wash load at one site may become bed material load at another. For example, ebb flow through sandy tidal inlets may contain fine sediment that does not deposit there and is not contained in the bed. However, the same material deposits offshore once the flow velocity decreases to ~15 cm/s or less.

The sum of bedload and suspended load is called total load. Depending on the definition used in a particular study, total load may include only bed material load, but if wash load is significant and is included then the total load will be equal to total bed material load plus wash load.

At this point we must refer to the transport of sand, which is coarse sediment, versus clay, which is fine. Sand is transported as bed material load. In rivers both bedload and suspended load of sand occur, although in energetic rivers such as the Columbia in Oregon/Washington, suspended load tends to dominate. This is also the case in energetic estuaries. In contrast to sand, fine sediment is transported as suspended load. In fact, it is difficult to define a bedload for such sediment (see Appendix F).

Inasmuch as clay particles cohere (due to electrochemical surface forces) in the presence of even very small quantities of salts in water, they are usually found as aggregates or flocs consisting of thousands of individual particles. The settling velocities of these aggregates tend to be orders of magnitude greater than those of individual particles. Due to their ability settle (and erode), fine sediment aggregates are transported as bed material load. In contrast, unaggregated fine particles (where they are present) as well as light-weight organic particles often "flow through", and if so behave as wash load.

As we have noted in Section 2.4, the observed lack of a direct correlation between TSS and peak discharge suggests that part of the suspended sediment transport in the St. Johns River may be wash load. Since wash load is not present in bed, it has no bearing on river morphology. Conversely, withdrawal of wash load by water off-take cannot be detected by changes in river geometry. Direct measurement of TSS concentration is the only way by which changes in wash load can be assessed.

### *3.1.3 Detrital Transfer*

Detrital transfer in the sense relevant to MFLs (ECT, 2002) refers to the supply of sediment from the river bank (floodplain) or in the estuary the intertidal flat, into the stream. Accordingly, it is dependent on the hydroperiod and supply from vegetative sources. In lakes, water level and wind-induced surge are important determinants besides detrital supply. While the hydroperiod is predictable, supply is a wholly site-specific factor. As schematized in Fig. 3.2, detrital transfer loss (i.e., decrease due to water level reduction) along a bank of length $L$ due to a decrease in the (maximum) water level will in general be proportional to some power ($m$) of the loss of area ($A_o - A_N$), where $A_o$ is the "old" bank area, $A_N$ the "new" area and $K$ is a (load-scaling) proportionality constant.

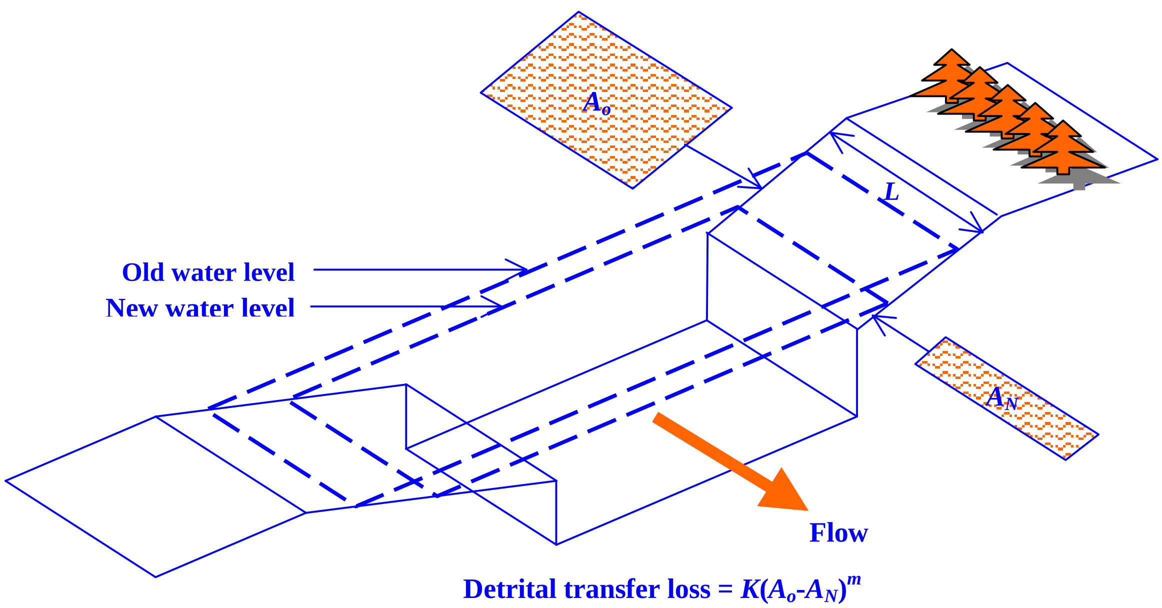


Figure 3.2 Detrital transfer loss due to water level (hydroperiod) decrease.

### *3.1.4 Lakes versus Rivers*

The above definitions apply to both lakes and rivers (including estuaries), but the following differences should be noted in reference to Florida.

1) In lakes, hydrodynamic forcing is mainly by wind-generated waves, except during storms when wind-induced surging has a role. Wind-induced forced oscillations (seiching) are usually less important. In rivers, sediment is forced by short-term-steady flows, and in such circumstances much of the sediment transport occurs under high freshwater flow conditions. In bays and river mouths tidal flows dominate except when river discharge is high. Wave action, occasional storm surge, and salinity-induced circulation provide additional forcing.

2) In most cases, sediment transport (due to wave action) in lakes is confined to fine-grained (<63 μm) material. Since fine sediment is (mainly) transported as suspended load, the total load is the sum of suspended load and wash load, with negligible contribution from bed load. Florida's rivers and estuaries in general transport both suspended load and bed load, the latter consisting (mainly) of coarse-grains including sand and (some) shell.

3) At open flow boundaries of a water body (lake, river or estuary), the sources are specified in terms of sediment load, but there can be sources interior to the body including local production of detritus and detrital transfer from the banks. Naturally, the bottom can also be a source depending on the strength of flow, sediment composition and compaction (density) and patterns of accumulation.

   From the perspective of MFLs, changes in the generation, transport and deposition of muck may be a significant issue, provided the amounts of bed material load (as opposed to wash load) in transport from this source are large. Muck is loosely defined as black or dark gray, fine-grained sediment with high water content, and containing decomposed organic matter admixed with silt and clay materials (Chapter 2, and Gleason and Stone 1994).

Sources and nature of muck in Florida's lakes, streams and the marine environment are discussed in Appendix A, where it is noted that the rate of production of muck has accelerated considerably over the past century, coinciding with human habitation.

**3.2 Task 1(b) Assessment of Sediment Transport and Detrital Transfer**

<u>Task</u>: *Discuss differences and similarities in the requirements for assessing sediment transport and detrital transfer.*

***3.2.1 Sediment Transport***

<u>*Response*</u>*:* For subaqueous sediment transport, the assessment procedures involve measurement of water level, current velocity and, focusing on fine sediment, the total suspended solids (TSS) concentration. These three together determine the sediment load. Instrumentation and methods used to calculate the load are well known, and in the present context we have identified two methods by examples, one involving the use of optical backscatter sensors (OBS), and the other the use of the acoustic Doppler current profiler (ADCP).

In Appendix B, OBS and ADCP measurements carried out near the C-18 Canal in southern Florida are presented as an example of the utility of such measurements in understanding sediment dynamics in Florida's (characteristically) episodically driven flow environment. Measurements of sediment flux with an ADCP in the Cedar/Ortega Rivers were described in Section 2.4. Subsequent to the Cedar/Ortega Rivers study (Mehta et al. 2004), improvements have been made to the ADCP technology to increase the accuracy of discharge measurements in shallow waters.

In Florida’s typically low-energy environment, OBS serve well because TSS concentrations are usually very low (< 30 mg/L). When strong flows occur they may rise by an order of magnitude, but often remain well below the 2,000-3,000 mg/L saturation limit of the sensors. If fine sediment is transported as fluid mud, the concentration limit is exceeded, since in that case concentrations may reach or exceed 5,000-10,000 mg/L. The transport of fines by this mode is believed to occur occasionally, with layers on the order of 5-10 cm in thickness at most. Long-term deployment of sensors, e.g., optical (transmissance) gages that can measure concentrations on the order of 20,000 mg/L, is logistically inconvenient for long-term data collection because of weekly or at least bi-weekly cleaning required against biofouling. When at rest, the thickness and TSS concentration of fluid mud are best measured by coring.

In Appendix C the Environmental Fluid Dynamics Code (EFDC) is used to illustrate the application of numerical codes to determine the effect of water (and sediment) off-take on the suspended sediment transport regime downstream. Two models are considered – one with the bathymetry and tidal conditions prevalent in the Cedar River, and the other an idealized prismatic channel subject to steady or quasi-steady flow. The change in the sediment transport regime is easily predicted for bed material load transport (as opposed to wash load). Such models however cannot predict the impact of off-take on river morphology with any degree of accuracy, because morphodynamic relationships governing flow and geometry are not included in the fullest sense, i.e., the models can predict depth change but not width, which limits their application to assessment of relatively short-term effects.

Also included in Appendix C is a simple graphical method for the calculation of deposition (shoaling) of suspended sand or fine sediment due to a velocity change. This is a special case of the prismatic channel model when the flow is steady and there is no resuspension of the deposit.

Another model application is given in Appendix D to estimate wind-induced resuspension of muck in a lake (Newnans Lake). In this example it is shown that simple, 1D-vertical modeling can be used to derive information on the relationship between the wind speed and the resuspended sediment load. A key feature of this analysis is the development of muck erosion and settling relationships that vary with the organic fraction in the sediment.

Detailed MFLs related analysis including spatial variability in sediment loading and the effect of off-take must rely on a more comprehensive approach, e.g., the use of EFDC. Even then, it may be difficult to accurately predict the resulting change in the bottom muck redistribution, especially in a small lake, i.e., one in which the effect of off-take is significant throughout.

### *3.2.2 Detrital Transfer*

Response: Detrital transfer as defined in this study tends to be highly episodic because most detrital transport, i.e., the movement of detritus transferred to the stream, occurs during periods of water level peaks. Presumably because the process is site-specific, data on detrital transfer are not available (ECT, 2002). Referring to Fig. 3.2, we define $g_{sL}$ as the lateral mass load (i.e., sediment mass per unit length of river per unit time) of detrital material entering the stream. Then, since detrital transfer *loss* must equal the corresponding *gain* in the stream, we have:

$$g_{sL} = \frac{K}{L}(A_o - A_N)^m \tag{3.1}$$

The common practice appears to be to set $m = 1$ (ECT, 2002) and calibrate for $K/L$[1].

[1] In hydraulics of sediment transport the term *detritus* is not used commonly, as one is concerned with *particles*.

## 3.3 Task 2(a) Review of Computer Models

*Task: Provide a review of computer models that might be used to assess changes in sediment transport and detrital transfer caused by altered hydrologic regimes.*

### *3.3.1 Sediment Transport*

Response: Models fall in three categories - those for estuaries, those for rivers, and those for lakes. An appropriate way to describe sediment transport modeling is by illustration. For estuarine application, Appendix C summarizes the use of EFDC to calculate the effect of water (and sediment) off-take on the change in sediment load downstream in an estuary modeled after the bathymetry and tide found in the Cedar River (but with a higher than normal river discharge). A second application is illustrated in the same appendix showing the effect of off-take on downstream sediment load in a 60 km long prismatic channel representing an idealized river. Finally, a graphical approach for deposition under (non-eroding) steady flows is also presented. This method is based on a simplified equation for sediment mass balance in the channel.

As noted in Section 3.2.1, Appendix D provides a model application for wind-induced sediment resuspension in a small lake. This is a 1D-vertical model, and it needs to be emphasized that while such a model can estimate the sediment load as a function of wind speed, the spatial effect of off-take for MFLs assessment must be modeled using a 2D or preferably a fully 3D numerical model code such as EFDC.

### *3.3.2 Detrital Transfer*

Response: Detrital *transfer* is expressed by Eq. (3.1), while detrital *transport* is part of stream particle transport and therefore must be modeled as such. As noted, the observation that at least part of the fine sediment load in the St. Johns River is uncorrelated to flow may indicate that part of the sediment load in the river derived from detrital transfer may act as wash load, i.e., one that does not deposit in the main channel but flows through. If so, and the fact that detrital transfer is partly supply controlled (so it cannot be correlated uniquely to water level or water velocity) suggests that it may be difficult to model detrital transfer and transport, and that the best option for MFLs assessment would be to institute a program of long-term measurements of TSS at selected points within the flow system. Such a protocol would develop a long-term database which can be used to identify natural variability in TSS in relation to the effect of off-take.

### 3.4 Task 2(b) Data Requirements for Modeling

Task: *Summarize minimum data requirements and typical costs for each model.*

*Response:* This question is best answered in a general sense. The key parameter governing the transport of particles of given diameter and density is the bed shear stress, which requires knowledge of the flow field including water level, velocity and bottom roughness. Sediment transport models characteristically require flow field generated by a hydrodynamic code. The effects of Coriolis acceleration, wind and saltwater-induced stratification may be significant enough to require inclusion of these factors, although in Florida's waterways they are often ignored. This is so because the low-latitude location of the state and because stratification is not always very strong except during freshets. Also ignored in Florida applications are the effects of very high TSS concentrations on the flow field. Where this effect is important, it becomes essential to dynamically couple the hydrodynamic code with the sediment transport code.

As for detrital transfer, based on Eq. (3.1) calibrated for $K/L$ and $m$, particle loading is specified as a lateral boundary condition in sediment transport modeling.

To address role of sediment transport in MFLs assessments in Florida, the appropriate strategy would be to begin with modeling, identifing threshold conditions on a generic basis and then plan for minimal (long-term) data collection provided this is found to be essential. Since the rates of sediment transport in Florida are very low (on the global scale), sediment transport modeling should be used to establish, as far as possible, a "generic template" for assessment of change in the sediment transport regime relative to MFLs. Data needs must then be prescribed to dealing with site-specific "deviations", if any, from the template. A study of this nature is proposed in Section 3.7.1.

### 3.5 Task 3 Simplified Methods for Assessment

*<u>Task</u>: Discuss and make recommendations about the possibility of using simplified (i.e., non-model) methods to assess changes in hydrologic systems with respect to transport of sediment and detritus. These simplified methods would most likely be based on hydrologic model results.*

*<u>Response</u>:* In general there are three elements related to methods for assessment.

#### *3.5.1 Long-term Field Monitoring*

The need is for long-term (e.g., 10 years) data on TSS and bottom sediment description. This type of information lags data on flows and levels because concern for sediment impacts related to MFLs is recent. To achieve database parity in this sense will mean the establishment of permanent sites (at selected flow cross-sections) within the District in coordination with U.S. geological Survey (USGS) stage-discharge measurements. At each site we recommend the following (in conjunction with discharge and water level data):

1) Measurement of daily TSS concentrations at (at least) two levels (near surface and near bottom) in the water column. Two levels are (minimally) required because the suspended bed material load tends to be considerably higher near the bottom than near the surface. Correlation of TSS must be sought with discharge (not load, which itself depends on discharge) and/or with other parameters such as chlorophyll-*a* on a daily basis because of the characteristically high variability in discharge/level, sediment supply and biological activity throughout the year. Present-day approaches involving acoustic transducers for TSS measurement and bottom sediment mapping have been

summarized in Appendix E. The ADCP has the advantage over optical backscatter gages (Appendix B) in that biofouling in not a significant problem with ADCP deployment.

The continuous seismic profiling (CSP) technique for bottom mapping described in Appendix E is new and untested within the District. If it is found to be suitable the payoff in terms of rapidity of measurement would be substantial.

2) Mineral composition of TSS (every 5 years). We have chosen a 5-year basis for repetition because of the slow changes in sediment composition (on an annual mean basis) that are typically observed in Florida.

3) Cross-section survey (every 5 years). Low sediment transport in Florida waters also cause slow changes in morphology. Consequently, more frequent surveying may not be justified.

4) Bottom coring to determine the vertical structure of density and mineral composition of sediment (every 5 years). This is quite important as this type of examination will allow the estimation of the thickness of the muck layer that is transported, or is vulnerable to transport. In the shallow waterways of Florida, vessel traffic also contributes to bottom sediment resuspension. The relationship between navigation depth, bottom mud density structure and mud erodibility is shown in Fig. 3.3 for the Cedar and Ortega River confluence area, where considerable deposition of muck has taken place over the decades (Mehta et al. 2004).

5) CSP has a high potential for extensive application for bottom composition mapping, if it is successful in Florida.

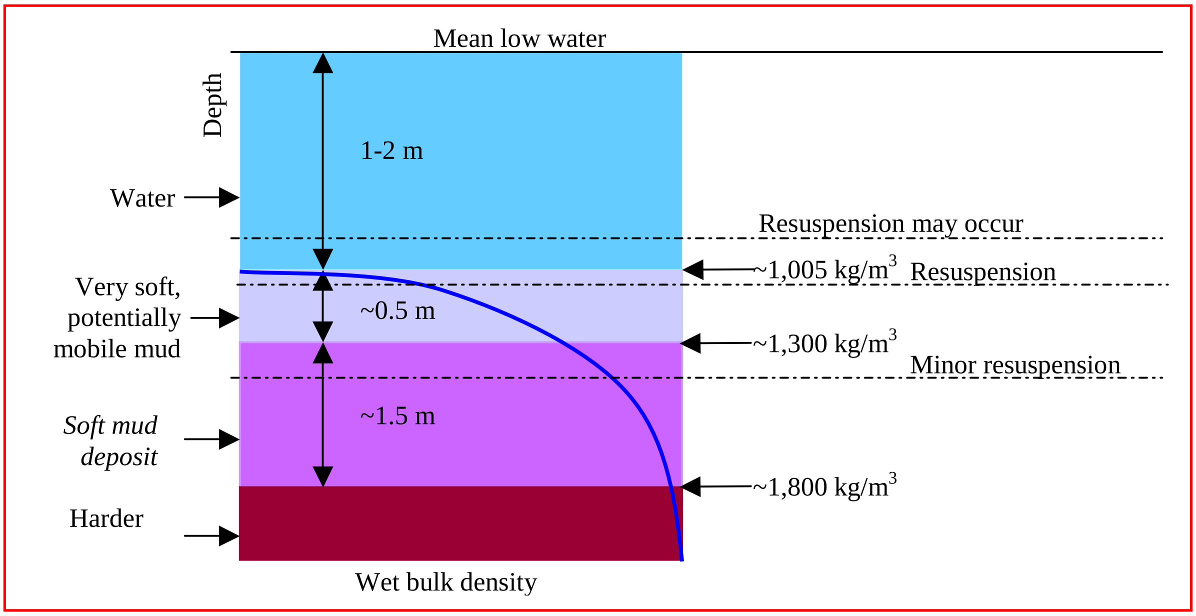


Figure 3.3 Navigable depth, density structure and erodibility of bottom sediment at the confluence of Cedar and Ortega Rivers (from Mehta et al. 2004)

### *3.5.2 Numerical Modeling*

The potential for application of numerical models of different levels of complexity has been described through Appendices C and D. Standard present-day approaches are useful for short-term predictions but cannot accurately predict long-term changes to the system relevant to MFLs. This is the case because morphodynamic relationships between river geometry and flow are neither well known nor adapted to models. As a result, models can predict depth changes but not width, which makes their application untenable for long-term assessments.

It appears that in Florida's sediment-starved environment the time-scale for a river system to reach a new equilibrium following any major perturbation may range from years to several decades (Appendix F). This being the case, the relationship between the design life of the project and physical changes to the system due to, for example, sea level rise must be taken into consideration.

### *3.5.3 Morphodynamic Modeling*

The fact that most, presently used, numerical sediment transport models for river and estuarine flows do not correctly predict long-term evolution of these water bodies has led to interest in the development of morphodynamic modeling to deal with (temporal and spatial) meso-scale processes (e.g., Dronkers 2004). An essential basis of this modeling is the inclusion of macro- or lumped-parameter relationships between the morphology of the water body and flow within it. When such a body is perturbed from its present state these relationships guide the model to define the new shape of the body, depending on the morphodynamic stability of the system.

A case in point is the regime concept relating discharge to cross-sectional area of the river. In Appendix F we have presented a simple method to consider the long-term impact of off-take on river morphology, hence of MFLs. Starting with the Einstein-Brown equation for sediment transport, we show that the regime equation relating river discharge $Q$ to cross-sectional area $A$ is compatible with sediment transport under live-bed (i.e., in the presence of sediment movement) equilibrium. As a result, a physical meaning can be provided to the (empirical) regime coefficients. Thus we find that for a given river the $Q$-$A$ relationship is directly dependent on sediment load. It follows that changing this load, e.g., due to off-take, will change the $Q$-$A$ relationship. This is discussed in relation to a hypothetical withdrawal scenario in the St. Johns River.

A limitation of the above approach is that it gives no indication of the time required to reach new equilibrium. Additional work is required to develop a more comprehensive, time-dependent model.

### 3.6 Task 4 Critical Thresholds

*Task: Discuss possible critical thresholds with respect to sediment and detrital transfer. These might include, for example, sensitive transition points in the hydrology/sediment transport relationships. Recommend possible study designs to understand and assess these thresholds.*

*Response:* There are two ways to establish critical thresholds for sediment load relative to MFLs inclusive of detrital transfer (and transport). The first is based on the short-term effect of off-take on the flow and sedimentary regime of water body, and the second is based on long-term morphodynamic changes. Study designs for the two approaches will be different.

#### *3.6.1 Short-term Effects*

High (e.g., > ~10-15%) off-take may have an immediate (weeks to months) effect on sediment load downstream, which can be predicted in a straightforward way by the application of a numerical model such as EFDC, as illustrated in Appendix C. These are of low importance to MFLs assessment, which is dependent on long-term (years to decades) changes. In any case, since MFLs are driven primarily by considerations of ecological effects (Neubauer et al. 2004), the effect of load change on key ecological parameters becomes as much a management question as science. This is so because there is a need to select as few such parameters as essential to keep the problem (of decision-making based on impact analysis) tractable.

A practical difficulty in any cause-effect analysis is the fact that, as noted, throughout Florida herbicide spraying is carried out on a wide scale in lakes and rivers. This operation adds organic-rich black mud or muck in a way that is not easily quantified, and possibly overwhelms other sediment sources such as detrital transfer by natural means. Given the usually very low TSS concentration in waters, this source of sediment will most likely continue to subsume any effects of off-take. By being a major ongoing perturbation to the sedimentary regime of Florida's waters, the effects of herbicide spraying may in fact subsume the objective of assessing effects of sediment load and detrital transfer in MFLs analyses.

It needs to be pointed out that the establishment of short-term critical thresholds is contingent upon the availability of *long-term* database, which is presently unavailable. Numerical modeling methods on the other hand are sufficiently robust for analyzing short-term effects, given the necessary input data. Accordingly, in Section 3.7 we have identified appropriate studies.

### *3.6.2 Long-Term Thresholds*

These deal with processes that change the morphology of the water body subject to off-take. The time scale will be on the order of years to decades, and is of relevance to MFLs assessment. A prediction of the new morphology and its magnitude relative to naturally induced perturbations are required to assess threshold requirements. For instance, quite simply, if the change in the river cross-section due to off-take is "small" compared to naturally occurring changes in the cross-section, then the effect of off-take would not be measurable. Here too a database upon which such a judgment can be made is lacking. Also, the required morphodynamic modeling is in its infancy.

Our initial assessment in regard to critical thresholds is that relatively small off-takes, e.g., less than ~10%, that do not alter these peaks in any significant way may not show measurable changes (in relation to data noise and natural variability) over time scales of years to decades. We must also add that time periods on the order of decades cannot be considered in MFLs assessments without recognizing that natural changes in forcing, especially sea level rise, may entirely subsume and effects of off-take.

Keeping the above points in mind, we have addressed the appropriate study design in the next section.

## 3.7 Task 5 Study Design to Assess MFLs Impacts

*Task: Recommend further analyses, data acquisition, or study design needed to assist DISTRICT in assessing changes in sediment transport and detrital transfer in the context of MFLs.*

*Response:* Such a study must be sequential, involving steps related mainly to long-term impact assessments. The following studies are recommended.

### *3.7.1 Prismatic Channel Modeling*

We recommend that the prismatic channel modeling (of Appendix C) be expanded to examine the off-take induced response (in peak magnitude and frequency of occurrence) of TSS and sediment load to long-term (i.e., decade or more) hydrologic time-series of water level, discharge and sediment. With further developments, this analysis could be used to develop a general methodology for establishing MFLs thresholds, keeping in mind the anticipated long-term morphodynamic changes. Transport of sand (as bedload and suspended load) as well as fine sediment (as suspended load) should be included.

This effort, which could be on the order of a year in length, must establish if additional information on sediment transport is required, or whether in Florida's environment it is sufficient to develop a "generic template" for analysis applicable to most MFLs project assessments.

The modeling exercise will also assist in identifying data needs, and if it is concluded that sediment related information is required for model methodology improvements, the following additional studies should be considered.

### *3.7.2 Prototype Test for TSS-Discharge Rating Relationship*

A test site where USGS stage-discharge gaging is carried out might be selected for a one-year monitoring of flows and suspended solids. For *suspended load* the latest ADCP technology (e.g., RDI-Sediview) may be used. Testing would include running cross-sectional transects with an ADCP mounted on a vessel, and also deployment of a bottom-mounted ADCP (for a year). We also recommend that a trench (e.g., 2 m deep below existing bottom, 2-3 m wide in the flow direction and 5-10 m perpendicular to the flow) be excavated at the site to estimate *bedload* transport of sand.

This test will serve as the basis to establish long-term sites at other cross-sections for the development of TSS-discharge type relationships required for sediment transport modeling.

***3.7.3 Prototype Test for Rapid Bottom Sediment Composition Survey***

Over a selected reach of a river and or a lake, the Multi-Detector System for Underwater Sediment Activity (MEDUSA), a recently developed CSP technique for rapid surveying of bottom sediment composition (sand and mud) should be tested for use within the St. Johns River Water Management District. This approach must include bottom coring for "ground-truthing" within the selected reaches (e.g., Jaeger and Hart, 2001). Bottom core data should be dated to determine deposition rates (Jaeger, 2004). This technique is a "non-modeling" albeit *post facto* means to assess MFLs related changes.

***3.7.4 Review of Morphodynamic Modeling***

In order to make an assessment of the state-of-the-art in morphodynamic modeling techniques for rivers, a comprehensive review of such models in use in the United States, The Netherlands, Italy and other countries should be carried out. A detailed assessment should be made of the immediate applicability of models to Florida, model limitations and further improvements required.

***3.7.5 Cross-section Database***

In conjunction with the tasks under Sections 3.7.2, 3.7.3 and 3.7.4, we recommend that a database containing flow cross-section surveys be developed. This will require that a certain number of sites be accurately surveyed on a periodic basis. Bottom core data should be collected at these sites to determine sediment composition in the vertical and lateral directions.

The above four follow-up tasks (per Sections 3.7.2, 3.7.3, 3.7.4 and 3.7.5) will set the basis for model improvement and possible data needs. Additional effort will be required to identify critical threshold criteria for off-take. These criteria depend on the relationship between key environmental parameters and the statistics of long-term hydrologic time series related to discharge, water level, sediment load and detrital transfer, and also on changes in the morphology of the water body.

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# APPENDIX A: SOURCES AND NATURE OF ORGANIC-RICH FINE SEDIMENTS IN FLORIDA[1]

## A.1 SUMMARY

The Floridian environment gives rise to prolific and highly varied organic deposits. This overview considers sources of carbon and the nature of the deposits this element forms. The environmental conditions leading to "carbonate-forming" and to "peat-forming" deposits are described. Carbonate depositional systems occur mainly in the south of the state and at the coast. Peat type deposits include mangrove peats at the marine coast and the vegetation-type-dependent, fresh water Everglades Okeelanta, Gandy and Loxahatchee peats of the interior. None of these give rise to major management problems. The "peat-forming" sedimentary regime also includes the problematical finely divided organic black muds locally called "mucks".

The marine coast, together with the terrestrial environment, waterways and lakes of Florida, are large scale sources of carbon production, the fate of which is not always clear. What is apparent is that poorly consolidated black muds are presenting severe problems in a large number of lakes, estuaries and marine coastal zones in Florida. The rate of production of black muds has accelerated a great deal in the last century and in the last few decades the degree to which loosely bound industrial contaminants have become associated with these has created a major challenge to society.

Examples of black mud production in the various typical regimes, based on co-operative research programs in the University of Florida over more than a decade and focused mainly in lakes and estuaries, are presented. An overview of black organic muds around the marine coast is provided. Original research on black muds in

[1] Paper presented by R. Kirby at the 2003 International Conference on Cohesive Sediment Transport, Virginia Institute of Marine Science, Gloucester Point, VA.

the Cedar/Ortega estuaries, a tributary of the major St Johns River system, is described. Similarly, literature review reveals environmental conditions determining the production rate, together with external and internal forcing factors determining black mud distribution in small lakes. This includes broad even distribution, focusing in bathymetric depressions or alternatively deposition around the littoral margin. Further original research on the issue of black mud distribution in large lakes, this time Lake Okeechobee, indicates an additional source of carbon production, namely the repeated wind-induced nutrient cycling leading to plankton blooms. None of the known forcing functions, alone or in combination, seem wholly to explain the distribution of black mud in the lake. Options for remediation and some trends for the future are suggested. Not only is black mud management a challenge, but is also a reminder that aqueous microbial systems are still imperfectly understood in terms of carbon sources.

## A.2 INTRODUCTION

A wide range of materials, derived from both the plant and animal kingdoms, fall under the general category of organic sediments. A broad perspective on these was provided by Twenhofel (1926 revised 1950). Twenhofel pointed out that almost every type of organic matter may be segregated and form a deposit solely of its own kind, although more commonly several are intermingled and, more often still, they include sediments other than those of organic origin. All of these are abundant in Florida. Organic sediments occur widely in rivers, lakes, estuaries and nearshore waters. Although these latter two are compared and contrasted, it is fresh water organic sediments, and especially remobilizable components of these, which receive greatest emphasis. In fresh waters, environmental conditions largely determine whether carbon is taken up in carbonate or in a range of other organic sediments. Both are described with emphasis being concentrated on the latter. Within this latter group materials may be allochthonous (coming in from outside the system) or autochthonous (generated in situ) in origin, and will differ in a major way depending upon whether they are input from terrestrial sources, littoral sources in lakes, or open waters in larger lakes. Again, it is the often very soft and refractory black organic mud which is the focus of interest. What emerges from a literature review is that the nature of such sediment remains rather poorly investigated in comparison with its emerging applied importance.

The black organic-rich deposits have received a number of different names, a fact that possibly heightens the confusion. Locally these materials are called “muck”, but in the literature similar deposits are termed “gyttja”, a Swedish word, “sapropel” (sapros = rotten, pel = mud), “ooze”, “black mud” or, in recognition of its loose and fluid nature, “unconsolidated flocculent” (UCF) material which may grade a deeper levels beneath the sediment surface to “consolidated flocculent” (CF) material.

Earlier this century, such material was of interest as a result of the quest by pioneers for productive land. Drainage and containment of lakes formerly having wide and highly variable shallow margins gave rise, where such material was exposed, to productive land. Now a less desirable situation has emerged. Stabilization of lake levels, together with raised nutrient inputs, has led to a large increase in the production and trapping of such materials, which, moreover, provide a host for poorly-bound nutrients. These materials are very readily re-entrained and the nutrients remobilized such at 80-90% of nutrients contributed to a system can be those derived from the sediments (Olila & Reddy, 1993). Moreover, they also provide a host for poorly-bound anthropogenic contaminants.

Reid & Wood (1976) note "we are very much lacking in our knowledge of the formation and role of many organic substances". Similarly, according to Darnell (1967), "literature dealing with the organic detritus problem is diffuse, widely scattered and written in a variety of languages". "In addition, the actual work on the problem appears to be very small" and "the significance of organic detritus is inferred from several lines of circumstantial evidence". Much of what was written in these 1960s and 1970s accounts remains applicable today.

### A.3 CARBON SOURCES

An overview of the various ways in which organic sediments originate will provide a perspective for more detailed consideration. Such materials may be distinguished in a number of ways and these are drawn out below.

Carbon is manifest in the Floridian environment in two principle regimes, either as a "carbonate-forming" or, alternatively, as a "peat forming" depositional system. Carbonate sediments formed by solitary or colonial (reef-building) organisms, or by precipitation or secretion from plants and animals are widely found in Florida, especially at the coast (Cohen et al, 1994; Brown & Cohen, 1995) but are not considered in detail here. Instead, this evaluation focuses on the so-called "peat-forming depositional system" although, in reality, derivation, chiefly from plant activities, is itself subject to further and distinct subdivision.

Aside from this initial subdivision, consideration of where carbon in the environment comes from reveals three separate sources. An initial and possibly unexpected point is that the generation of carbon in respect of one of these three subdivisions is not well established. Each of the three subdivisions applies to fresh, brackish and marine systems.

With reference to the first, fresh water systems, these three input sources may be distinguished as: 1) terrestrial inputs via rivers, aquatic plants in rivers; 2) small lakes or the margins of bigger lakes, i.e., the fresh littoral zone; and 3) open water systems in larger lakes with little or no link to terrestrial inputs or the indigenous littoral zone.

Terrestrial systems tend to be abundant in carbon. For example, woodland and grassland such as that found in Florida can be expected to have a biomass somewhere in the region of 50g C $m^{-2}$ of area (Webster et al, 1995). Much of this material, when it dies, is degraded in terrestrial soil systems, which are very efficient at breaking down leafy and woody debris. However, although large quantities of such material also reach fresh water environments, a contrasted situation is manifest here. Firstly, terrestrial vegetation contains a very high percentage of the skeletal materials lignin and cellulose, which are highly recalcitrant or refractory in fresh water. Secondly, fresh water "soils" of rivers and lakes are not equivalent to terrestrial ones, being far less efficient at degrading organic materials. One may reasonably surmise, therefore, that although terrestrial biomass may be high, neither its intrinsic nature nor the fresh water system itself is able to process it rapidly and carbonaceous material breaks down very slowly.

The second, varied and readily evident, source is macroscopic water plants of the rivers, canals, shallow and littoral zones of lakes themselves. Patten (ed.) (1990) has composed a flow diagram of the nutrient dynamics and carbon production of a wetland system (see also Mehta et al, 1997). Mulholland (1980) in Patten (ed.) (1990) shows the organic carbon flux of a swamp ecosystem in coastal North Carolina. These swamps include a wide range of floral species including reeds, sedges, grasses, macrophytes, pond weeds, etc. Almost all such plants have a pronounced seasonal cycle of growth and die-back, resulting in a yearly biomass not dissimilar to that of terrestrial woodland and grassland. Also, almost all die-back material falls into the stream, lake or marsh it is growing in, thus maximizing organic capture by the system. On account of the similar need for these materials to stand erect in the air, they also produce highly refractory organic debris. Such material is less resistant to breakdown than much terrestrial debris, but still contributes in a more minor way to productivity than simple consideration of its biomass might indicate. Many shallow water plant species have less of a need for a skeleton than terrestrial plants, being supported during their life by the carrier fluid they are in. These plant communities are covered in part by the terrestrial classification (one, above), but more generally by this marginal/littoral marsh/ shallow coastal classification (two). These materials break down into a range of organic sediments which are specified later. A map of the Everglades, such as that produced in Davis & Ogden (eds) (1994) (Figure A.1), shows the breadth in the range of organic sediments derived from these.

The third source of organic carbon is microbial, phytoplankton carbon and its derivations or dependants. Such material constitutes the bulk of the organics produced in the open waters of large water bodies. Westlake (1980), (and in Mehta et al (1997)), shows the flow of energy and organic material in a planktonic ecosystem, whereas Saunders (1971), (and in Mehta) quantifies the carbon flow in the microbial system of a lake. In contrast to the two former sources, the biomass of such a system is always much smaller, typically in the region 1.5g C $m^{-2}$ of water surface. This much lower biomass is partly compensated by the lack of any “structural” carbon, as required by emergent organisms and plants, which render these latter materials not readily utilizable. Furthermore, whereas emergent forms tend to “crop” only once a year or less, microbial systems routinely produce 5-6 crops $yr^{-1}$. As a result, the

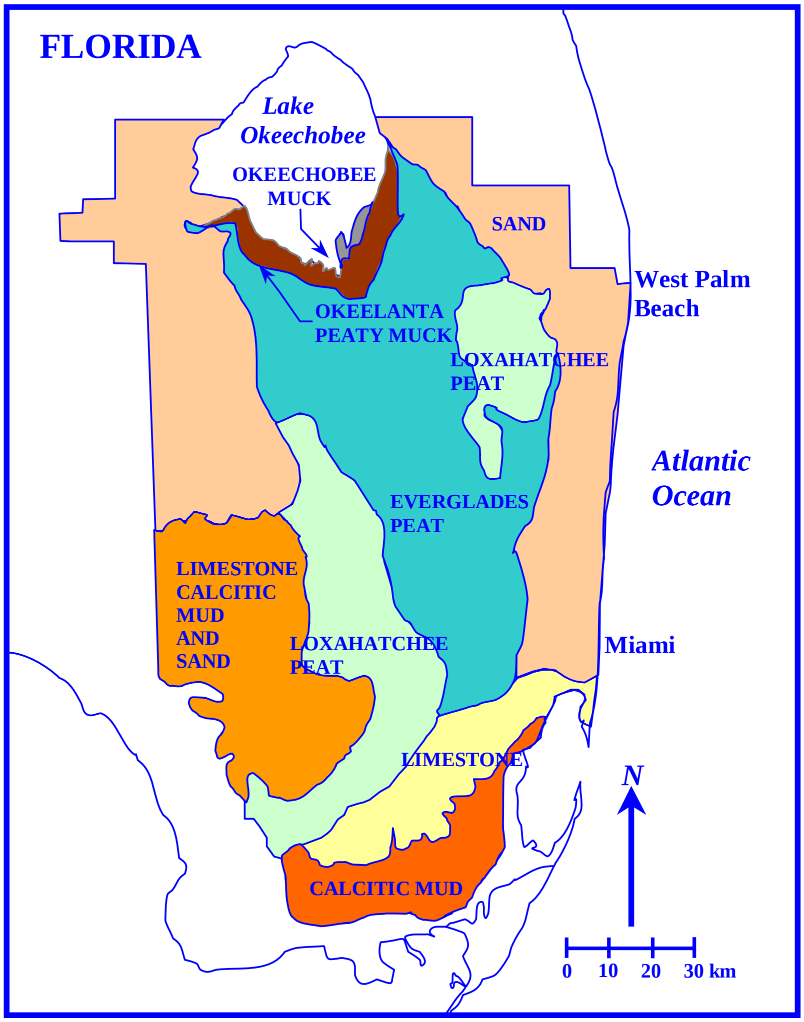


Figure A.1 Surface sediments covering southern Florida (adapted from Jones et al, 1948 in Davis & Ogden eds, 1994).

relative poverty of "maintained" carbon in open water bodies is largely offset by its replaceability.

Taking the issue of source a further step forward, whereas the derivation of carbon in the two first mentioned sources is readily obvious, namely the surrounding plants and to a lesser degree, animals themselves, the source for the third is not obvious. Indeed, relatively little is known about the source of carbon in open bodies of water. These open water environments represent an important area of practical concern to water management agencies. With this in mind, some of the conundrums still facing freshwater biologists are set out in the discussion.

Finally, in considering where in the coastal system carbon arises, the readily recognized sources are fluvial input from the continents and autochthonous production in the coastal zone itself. On the basis of such a hypothesis, one might reasonably anticipate that continents and their coastlines were sources and the ocean a sink. Walsh (1991), however, has provided an independent estimate of the annual supply of onwelling nitrate from the deep sea to the continental shelves and finds that it may balance the offshore flux of carbon, suggesting that the continental margins and deep sea are equally important in carbon and nitrogen biogeochemical cycles.

Having set the scene, above, for sources of organic material in sediments in Florida, the manner in which these may present themselves is laid out below.

## A.4 MARINE AND BRACKISH WATER ORGANIC SEDIMENTS

### A.4.1 *Coastal Black Organic Mud*

In the Florida peninsula we can ascertain that fluvial sources do give rise to organic sediments at a number of sites and that this has several undesirable characteristics. Trefry et al (1992) report that coastal waterways in Florida are being stressed by inputs of fine-grained organic-rich sediment from their river systems. They carried out an investigation of one extreme example of this in Manatee Pocket in the St Lucie Estuary on the SE coast of Florida. Here black muds, in many ways comparable to those accumulating in lakes in the state, incorporating more than 4% organic carbon, have accumulated to a depth of about 1m (range 19-204cm, mean 122cm) and occupy a volume of 340,000$m^{-3}$. The deposits are believed to have formed as soil and detrital organic debris, and to have been trapped in the system during the last 100 years of regional development. These materials give rise to direct and indirect problems. The natural sandy substrate of the pocket is blanketed by the black mud, altering the benthic community. Secondly, the materials are highly susceptible to resuspension, which increases turbidity, inhibits light penetration and causes a decline, along with the substrate changes, in the growth of sea grasses.

Brooks et al (1991) describe how many Gulf of Mexico estuaries have undergone alterations in sediment distribution patterns accompanied by increased sediment accumulation rates. This is directly attributed to urban development. This group studied Hillsborough Bay, the northeast lobe of Tampa Bay, which is the largest estuary on Florida's west coast and is surrounded by heavy urban development. The black mud deposits here are concentrated in low-energy bathymetric depressions. Deposition rates in the last several thousand years average 31-49cm per1000yr, whereas rates for the last 100 years have accelerated to 0.13-0.42cm $yr^{-1}$, a hundred fold increase.

Similar organic-rich sediments and parallel environmental concerns relate to areas of the Indian River Lagoon (Gu et al, 1987). McPherson et al (1990) describe a comparable situation in Charlotte Harbor in southwestern Florida, where carbon and nutrients are input to the headwaters from terrestrial discharges.

Chemical and sedimentological studies have been made of the sediments of the main stem of the St Johns River and its tributaries, for example, the Cedar/Ortega river system near Jacksonville. Much of the beds of these systems consist of weakly consolidated black muds. Cooper & Donaghue (1999) describe these sediments as "overwhelmingly fine-grained, averaging 80% fines, unusually high in moisture content, averaging 79%, and in organic material, averaging 29% by weight", all leading to extremely low dry bulk densities, averaging 0.24g $cm^{-3}$. This is the dark colloidal, highly organic-rich mud, locally called "muck". Various organic geochemical analyses on this material, Mulholland & Olson (1972) and Tissot & Welte (1984), suggest for the entrance to the Cedar/Ortega Rivers a marine-derived portion of the black muds in the region 20-30%. C:N ratios of many core samples were found to lie in the range 10-14:1 and did not vary significantly with depth. This and other contributory evidence is consistent with the organic components in terrestrial soils and in the surface sediments of lakes. The values are consistent with derivation of a large proportion of the black muds from degraded higher plant (terrestrial) sources. The various contributing analyses suggest that in the reaches of the Cedar/Ortega River entrance perhaps 10-20% of the black muddy sediment is derived from autochthonous, planktonic algal, aquatic sources.

The sedimentary regime of the bed of the Cedar/Ortega River estuaries is known from 172 cores taken by Morgan & Eklund (1995) and 51 grab samples obtained by Battelle Ocean Sciences (1998). The sediments of the Ortega River exhibit the highest organic content (Figure A.2) and the smallest anthropogenic alteration of any in the system. The Ortega watershed remains heavily forested and perhaps the most significant change in the last 100 years has been an increase in the sedimentation rate? Deforestation and residential development in the creek sediments entering the Cedar/Ortega River system is manifest perhaps by the suggested raised sedimentation rate, but also by multiple layers of wood chips and occasional yard grass (grass cuttings) in the sedimentary sequence. The sediments of the Cedar River show the greatest anthropogenic effects. The upper black mud reaches contain sand and clay deposits washed off deforested and urbanized areas. There are no wood chip zones to indicate recent deforestation but the sediments are significantly contaminated by fuel oil and a wide range of industrial pollutants. The hypothesized accelerated deposition rates over the last 100 years are exacerbated further along the north coast

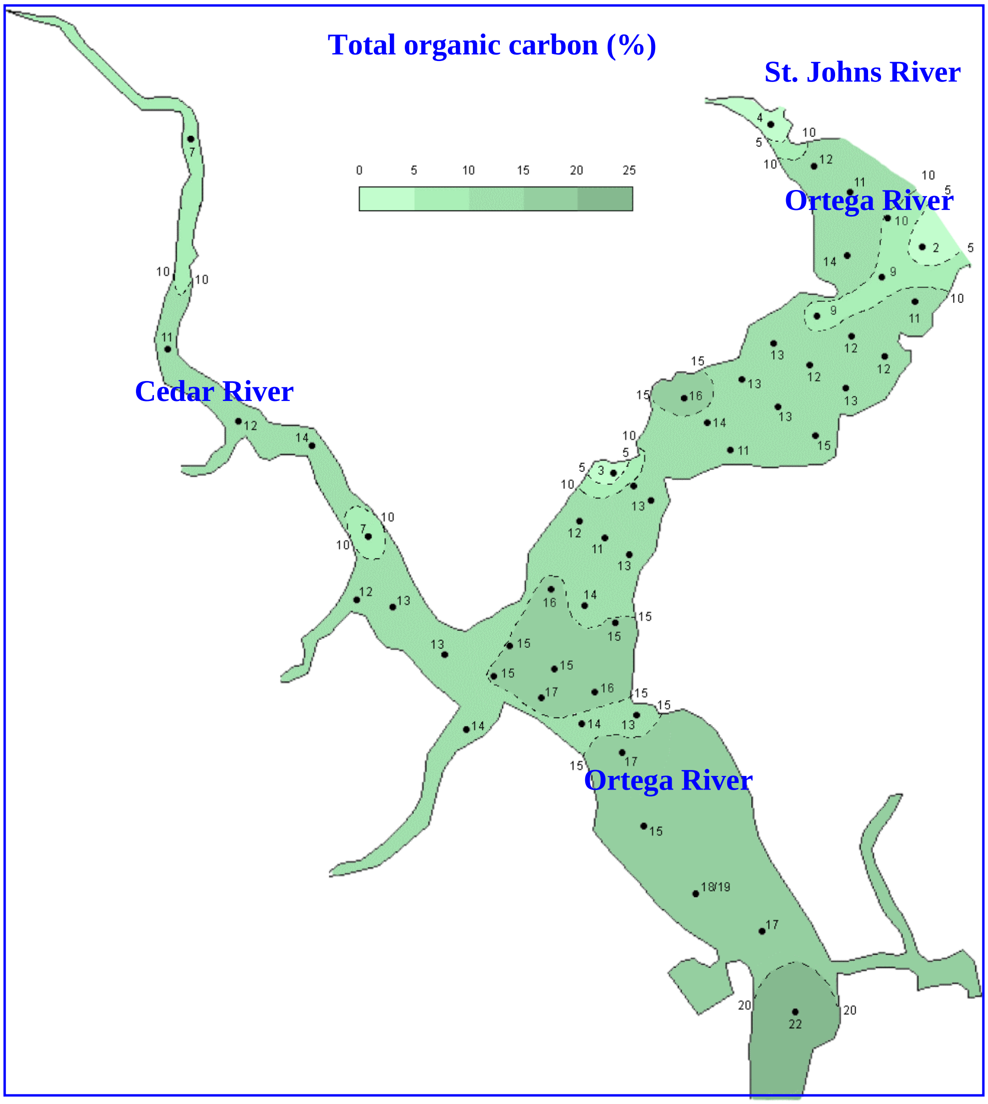


Figure A.2 Variation in percent Total Organic Carbon (TOC) in black organic mud samples in the lower reaches of the Cedar and Ortega Rivers. The high TOC of Ortega River samples, spreading into the inner confluent region, together with the low TOC at the confluence with the St Johns River is evident (from Mehta et al, 2004).

of the confluent reaches of the Cedar/Ortega Rivers by the blocking effect caused by a number of large marinas. The sedimentation rate reaches 20mm $yr^{-1}$ in this zone, but more generally occurs in the range of 4-8mm $yr^{-1}$. Measured values for points in the main stem of the St Johns River range between 6 and 39mm $yr^{-1}$, Mehta et al (2004).

### A.4.2 *Mangroves, Seagrass and Spartina*

The Floridian coast is especially productive regarding organic sediments and includes extensive mangrove swamps, *Spartina* marshes and seagrass beds. Each can contribute a large carbon flux to the system, which may maintain primary production or become incorporated into organic sediments. Where abundant and prolific, mangroves are especially productive, leading to the formation of mangrove peat - a very pure form of organic sediment.

#### A.4.1.1 Mangroves and Seagrass

Mangroves are extensive in some coastal areas around the Florida coast, as shown for the extreme southern tip of the peninsula (Figure A.3). Fleming et al (1990) specify seagrasses and mangroves as the major carbon producers in southern Florida fringe coastal communities. *Thalassia testudinum* dominates the seagrasses and *Rhizophora mangle* the mangroves. The detrital material from leaf decomposition is suggested by these authors to be the basis of trophic food chains in coastal ecosystems in southern Florida.

Mangrove forests are net exporters of organic carbon and mangrove carbon also supports food chains within riverine and basin communities along the southwest coast of Florida. Thus, it is obvious that these two groups of plants are extremely important in ecosystems, although, other than in the production of mangrove peat, no other organic sediment appears to have been specifically associated with them. On a similar theme, Twilley (1983) measured litter dynamics in basin mangrove forests and the productivity of adjacent coastal waters. The tides were found to control the monthly rates of organic carbon export. Surface export by rain and tidal exchange was

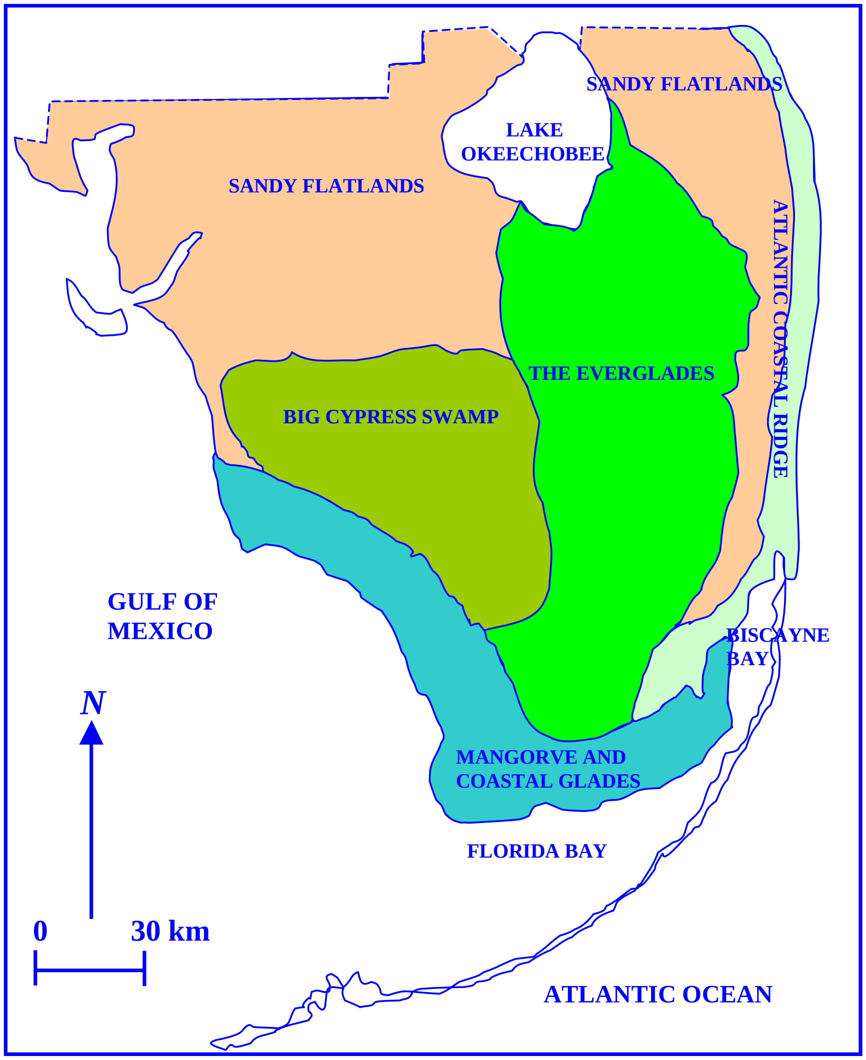


Figure A.3 Physiographic divisions in southern Florida including the coastal mangrove belt (adapted from Kohourt & Kalpiviski, 1967).

calculated to be 50.9g C $m^{-2}$ $yr^{-1}$, contrasting with export via seepage, which was estimated at 12.8g C $m^{-2}$ $yr^{-1}$. 75% was in the form of dissolved organic carbon, leaf export being minor by comparison. It was pointed out that this situation did not necessarily hold true for hurricanes.

#### A.4.1.2 *Spartina* Marshes

Wetland ecosystem productivity is generally high compared to other ecosystems. This is evident from an examination by Whittaker & Likens (1973) of ecosystem characteristics for 20 different environments ranging from barren rock and ice fields at one extreme to oceans and tropical forests. Their estimate of average annual production for wetlands is 1,125g C $m^{-2}$ $yr^{-1}$. They estimated that the average annual production rate for animals is 2.5g C $m^{-2}$ $yr^{-1}$ for all continents and about 9.0g C $m^{-2}$ $yr^{-1}$ for marshes and swamps. Wetlands consequently contribute a high proportion of carbon to surrounding systems, as well as within their own system, and it would not be surprising if there was a large output, in one form or another, of organic-rich sediment. The fate of carbon from these highly productive systems has, however, been a controversial issue.

Studies on saltmarsh functioning and exchanges between saltmarshes and coastal marine waters point out a great variability in terms of primary production, decomposition, import or export of organic matter and nutrients. In the coastal zone the energy balance of saltmarshes has been one of the most hotly debated issues amongst ecologists over the last three decades. In the 1960s, Odum (1969) concluded that 45% of production is removed by tides before marsh consumers have a chance to use it, and in so doing permits estuaries to support a wealth of animals. This new concept was supported by particulate organic detritus studies and was the origin of the "outwelling hypothesis", which proposes that saltmarshes produce more material than can be degraded or stored within the system, and that the excess is exported to coastal waters, where it supports ocean productivity. However, when whole systems from land to sea were later considered, it became clear that *Spartina* marshes can be sinks for nutrients. This and other work led to many studies in a wide variety of systems and to conflicting evidence about whether saltmarshes were exporting or importing.

Childers (1994) and Dame (1994) carried out very large scale studies of this topic at sites along the eastern seaboard of the US. These showed that where tidal ranges are low, less than 1m, as is typical in Florida, marsh/open water exchanges of nutrient and organic matter are primarily intertidal and fluxes measured on these marshes are export-dominated. Further to the north, at sites having higher tidal ranges, horizontal sub-surface flow and subtidal benthic advection become important, as marshes tend to take up nutrients and organic matter from the inundating water column, while exporting these constituents to adjacent tidal creeks. These latter marshes thus tend to be importers of nutrients and organic matter.

## A.5 TERRESTRIAL AND FRESHWATER INCLUDING LAKE ORGANIC SEDIMENT

This group of environments may be subdivided into four categories. These are: carbonates, wetland peat deposits, streams and rivers, and lakes. The sediments derived from these four categories are presented. In the absence of detailed descriptions, organic sediments in streams and rivers are described from a statewide and general perspective. In contrast, far more work has been published on lakes and a broad statewide overview is presented with detail adequate to comprehending management options and consequences. Fresh terrestrial wetland deposits, mainly peats, together with carbonates, are presented with an emphasis on the Everglades and southern Florida, although this is merely the largest and best described of many comparable systems in the state.

The distribution of surface sediments of south Florida is shown in Figure A.1. The total system embraces the headwaters and drainage basin of the Kissimmee River and several smaller streams and rivers, the largest lake (Okeechobee), the southerly and part westerly directed outflow from this lake forming the Everglades Wetland and its broad outlet to the sea in the extreme south, the Shark River Slough. A number of obvious regional features emerge from studying Figure A.1. The first is that organic sediments are mainly carbonate muds in the south and peats in the north. Secondly, a number of the expansive peat provinces have been given names. These are the Everglades peat, the Okeelanta peat and two separate large expanses of Loxahatchee peat. The justification for giving and maintaining these names is that the peats derive from different plant groups and consequently show contrasting features. A third aspect is the application of the word “muck” to certain organic deposits. Again, there is more than one type of muck, as explained below.

### *A.5.1 Carbonate Sediment*

The environmental significance of calcitic mud is that it is an organically-derived sediment formed in environments subject to an annual flooded and dry period; whereas, in contrast, at the same sites, if flooding was longer or continuous and the regime anaerobic, not calcitic mud, but peat would form. Thus, the environment of deposition of calcitic mud is sparsely vegetated marsh, where the water surface is well lit for the photosynthesizing algae. Peat-forming marshes then tend to be more densely vegetated and more continuously flooded to prevent oxidation by subaerial exposure. Development of calcitic mud presently indicates relatively shallow water on average and considerable oxidation of organic material in the sediment throughout the year, but especially in the dry season.

The link between calcitic mud and shallower, shorter flood periods, and marsh peats with deeper, longer flooded sites is clearly evident in Taylor Slough. Calcitic muds are thus organically-produced sediments and therefore relevant to this review. However, in view of their environment of formation and the oxidation of organic debris during their formation, they have a much lower free carbon content (no analyses have come to light to quantify this). They are also less reactive than black muds, less readily erodible and very much prone to penecontemporaneous cementation, forming marl beds or cementstone type materials. As a result of these features, it is judged that they do not give rise to the practical problems associated with black muds and do not merit more detailed consideration here.

**A.5.2 *Wetland Peat Deposits***

Peats and mucks are formed either by wetland plant communities, in the case of peat, or, in the case of muck, in a nearshore, permanently-inundated environment in waterways or lakes and, especially in earlier times, e.g., adjacent to or in Lake Okeechobee. Peats originate from the preservation in situ of roots and rhizomes and, to a lesser extent, the stems and leaves of Everglades plant communities. Muck consists of a mixture of transported, possibly only negligibly transported, fine, disseminated organic matter and fine sedimentary or detrital mineral material. It often occurs as a very loosely consolidated black mud.

The main areal distribution of marsh peat types is closely related to bedrock topography, which in turn controls plant community development. Distribution of tree island or hammock peats is also influenced by bedrock topography. The shape of many Everglades tree islands and their associated peat ridges is controlled by predrainage flow through the Everglades, as evidenced by their narrow streamlined shapes.

The geographical distribution of the major peat types in the Everglades is shown on Figure A.1, They are distinctive one from another on account of the dominant vegetation from which they developed, although it must be pointed out that vegetational succession has resulted in contrasted plant communities occupying them at some locations. The links between plant communities and freshwater organic sediment types are tabulated below (Table A.1, from Gleason & Stone, 1994).

Table A.1 Linkage between organic sediment type and plant community

| Organic Sediment Type | Plant Community |
|---|---|
| Everglades peat | Sawgrass marsh |
| Loxahatchee peat | Water lily marsh or slough |
| Gandy peat | Tree island |
| Okeelanta peaty muck | Elderberry-willow swamp[a] |
| Okeechobee muck | Custard apple swamp[a] |
| Custard apple muck | Custard apple swamp |
| Torry muck (modern terminology synonymous with Okeechobee muck) | Custard apple swamp |

[a] Not necessarily the vegetation initially forming the organic sediment.

Everglades peat refers to sediments formed chiefly from sawgrass. This peat type is characteristically brown to black in color, fibrous to granular and fairly low in mineral content (about 10% dry weight). Such peats covered approximately 4,400km$^{-2}$of the Everglades marsh and are the most abundant of the organic sediments. It is an intrinsically infertile material.

Loxahatchee peat refers to peats derived primarily from the roots, rootlets and rhizomes of the white water lily *Nymphaea*. It is a light-colored, fibrous and spongy, reddish-brown peat. It covers 3,000km$^{-2}$ within the Everglades basin and is second in abundance to sawgrass peat. Loxahatchee peat is related to water lily slough communities or water lily co-dominated wet prairie vegetation. These peats form some of the thickest deposits, but also occur in shallow sediment profiles. Such peat also underlies Everglades peat in parts of the sawgrass Everglades.

Gandy peat is a rooty and sometimes leafy peat formed by forest vegetation on elongated Everglades tree islands. It often overlies Everglades and Loxahatchee peat on existing tree islands, but is rarely interbedded with sawgrass peat. This peat, in contrast with marsh peat types, forms in the exposed oxidizing zone above surface water. It covers only about 80km$^{-2}$ in total among thousands of widely scattered occurrences. This tree peat, though an areal associate of Loxahatchee peat, is concentrated particularly in two large areas of deeper peat, lower bedrock elevation and where there is the greatest indication of broadly focused predrainage-era surface water flow within the Everglades.

The Okeechobee or custard apple muck, now termed Torry muck, refers in a narrow sense to a sediment covering 130km$^{-2}$ around the southern and eastern shore of Lake Okeechobee (Gleason & Stone, 1994). Early in the century it was colonized by custard apple swamp forest. It was suggested that custard apple forest possibly had established itself on a mud flat at the southern end of the lake following drainage and embanking works, because no woody peat was found below the trees and underlying sediments were of a "fine transported nature". Torry mucks are, then, the onshore expression of black muds of open waters of Lake Okeechobee and are analogous to black muds now recognized in many other lakes in Florida which, in the main, have not been drained, embanked and lowered to the same extent, such that their black mud deposits have never emerged ashore.

The Okeelanta peaty muck is characterized by a unique stratigraphy and in the past by an unusual willow and elderberry vegetation. It formerly presented a three layer succession in which the middle zone is described as a plastic layer of blackish sedimentary muck overlaid with a surface layer of brown fibrous peat, mainly the roots and rhizomes of sawgrass.

The several types of peat described are capable of containing nutrients in their interstitial waters to a degree comparable to the black mud deposits. However, even peats still submerged in lakes, streams and rivers cannot give rise to the severe nutrient cycling problems which the black muds suffer from. They are massive, fibrous and resistant deposits never susceptible to reworking in the same way as the unconsolidated flocculent muds. Consequently, they are not a focus of practical management concern in this regard and there is no necessity to examine them further in this review.

### A.5.3 *Streams and Rivers*

For some reason these are especially poorly studied. Recourse is made in part to either very broad and general or highly site specific information. In a review of organic processes in streams of the eastern US, Webster et al (1995) provide considerable relevant information, although naturally this is focused on water. These investigators point out that many streams and rivers are highly colored by dissolved organic material. The natural vegetation, which provides one important source of organic allocthonous input, is the Southern Mixed Hardwood vegetation association dominated by oaks, bald cypress and tupelos. Large clearances of this to provide agricultural land have led to vegetation change in many parts of Florida. Precipitation and stream flow is concentrated in summer months in Florida, although where streams drain large limestone aquifers the drainage may be dominated by springs. Webster et al (1995) have estimated a wide number of attributes of gross organic inputs to Florida streams (see also Mehta et al, 1997), ie. annual litter fall, annual wood input, net primary periphyton production in streams, net primary production of macrophytes, benthic respiration rates, breakdown rates of tree leaves, particulate organic matter concentrations in Florida streams, dissolved organic carbon concentrations, etc.

### A.5.4 *Lakes*

#### A.5.4.1Background

There are in excess of 8,000 lakes in Florida. Brenner et al (1995) note that only about 10% of these have been studied limnologically and, indeed, it is only in the past decade that the pace of detailed studies and extensive reviews has become intensive. A review in the late 1990s indicated that close to 400 lakes have been studied (Mehta et al, 1997). Information on organic sediment in lakes is consequently somewhat limited. Such knowledge as is available is reviewed below.

Where information on organic sediment is available, the two prime questions to be posed are “how is it distributed in lakes?” and “what does it consist of?” With regard to the latter, peat deposits, calcitic mud and black organic-rich muds are involved. Peat and calcitic muds are not readily destabilized and provide a poor host for nutrient contaminants. In contrast, a number of practical problems arise from the black muds, which are a focus of interest here. Turning to the distribution of black muds, in addition to the very limited nature of survey and composition data, it is apparent from the literature that distribution is not simple and straight-forward, but controlled by a differing balance of processes in different sites. One can but hope that the large proportion of these controlling factors has been recognized. Those which have are presented below.

**A.5.4.2 Composition**

Biogeochemical analyses have teased out the proportionate contributors to black muds in the Cedar/Ortega Rivers (see above). The organic black mud in lakes has a very high percentage of amorphous plant debris. Its potential origins are terrigenous sources in the watershed, the littoral zone and, for lakes with large open waters, the lake itself. A characteristic feature of the material is that it is very largely highly refractory material, but highly reactive with respect to nutrients. The fact that a very large proportion of black mud derives from plant sources is evident from published figures for carbon content, Table A.2.

Table A.2 Sediments in three lakes

| Lake | Sediment | Data | Source |
|---|---|---|---|
| Apopka | Mud | 325-395g $kg^{-1}$ organic carbon dry wt. | Olila & Reddy,1993 |
| Okeechobee | Mud | 174g $kg^{-1}$ organic carbon dry wt<br>Littoral Sediment: 39g $kg^{-1}$ organic carbon dry wt<br>Peat: 425g $kg^{-1}$ organic carbon dry wt | Olila & Reddy,1993 |
| Newnans | Mud | Mud Range: 4,700$gm^{-2}$ to 6,250$gm^{-2}$ at northern end | Gottgens & Crisman, 1991 |
| Newnans | Mud | Mud Range: NE SE W<br>Org.Fraction (g $cm^{-2}$) 20 95 10-15<br>Org. Matter (%) 10 35 5 | Gottgens, 1994 |

Bearing in mind these are dry weight fractions, it is easy to appreciate that in volume terms this amorphous organic material represents the very large proportion of sediment. Indeed, Binford & Brenner (1986), relying on their own unpublished data, state that virtually all the carbon in Florida lake sediments is organic, carbonate-rich lakes being uncommon. Total carbon, as sedimentary organic matter in 97 lakes examined by themselves, was reported at a near-constant 54%.

Other components noted by Olila & Reddy (1993) include total phosphorus (22-44m mol P $kg^{-1}$ in Lake Apopka), 1M HCl extractable (amorphous) Fe, Al and Mg, and the minerals sepiolite, calcite and dolomite. (Very possibly the carbonate minerals are produced autochthonously). Sand, shell, and siliceous microfossils, of which sponge spicules are frequently more abundant than diatoms, provide a minor component.

In open water eutrophic lakes the source of organic carbon is variously attributed to “recent algal deposits” and “allochthonous particulate organic matter”, (Reddy & Graetz, 1991). Lake Apopka has, (Olila & Reddy, 1993), “a high primary productivity and exhibits massive deposition of dead algal cells in the sediments" forming the unconsolidated flocculent layer. Because summer storms generate waves which entrain portions of the unconsolidated flocculent layer, leading to diatom peaks in the water column, this must also imply that diatomaceous debris is an abundant component of the black mud of many lakes.

In contrast, the tonnage of carbon in submersed vegetation around the shores of Lake Okeechobee is calculated by Zimba et al (1995) to be (Table A.3):

Table A.3 Carbon load in Lake Okeechobee

| Vegetation | Carbon (Metric tonnes) | |
|---|---|---|
| | 1990 | 1991 |
| Hydrilla | 7,269 | 8,531 |
| Potamogeton | 5,805 | 6,446 |
| Najas | 13 | 1,221 |
| Chara | 3,967 | 2,469 |
| Vallisneria | 7,991 | 9,569 |
| Total | 25,044 | 28,235 |

The fate of this material and its relation to open water production has not been established.

**A.5.4.3 Distribution of Black Mud**

*General Description:* Literature search indicates that relatively few studies of black mud distribution have been carried out, and those that have reveal wide, but explicable, contrasts in the spread of these materials. In recognition of this situation, one approach, thought at one time to hold promise in a general sense in order to address the 8,000 or so lakes, is to seek out a surrogate which might permit an authority to obtain an impression of what to expect. In pursuit of such an indicator, Beaver & Crisman (1991) studied the variation in trophic state of lakes in north, central and south Florida from published data on 68 lakes, supported by chlorophyll-*a* data from a further 138. Two trophic states were identified, clear and colored, and each subdivided into oligotrophic, mesotrophic, eutrophic and hypereutrophic subcategories. The results were set beside information which already existed for tropical and temperate lakes to the south and north of Florida, respectively, and fitted closely with what might have been predicted.

It was found that areal productivity variation is statistically greater in north Florida than in central, and that south Florida was less variable again. The degree of variation in algal biomass was comparable between zones, but the variation in productivity per unit plankton decreased from north to south Florida. Gross primary productivity also reduced from north to south. If this information alone determined carbon flux to black mud on the bed, it would be useful information.

Figure A.4 shows the relationship between trophic state (as TSI Chlorophyll-*a*) and accumulation rate of organic matter in dated cores. Organic sediment accumulation rate is calculated from the uppermost horizon (equivalent to 5-10 years) in the cores. More than 60% of the variance of TSI from lake to lake is explained by the net organic accumulation rate, and those lakes apparently forming outliers to the trend and reducing the correlation can be readily explained. In spite of these exhaustive studies by Beaver & Crisman (1991) and by Binford & Brenner (1986), attempting to apply trophic status index studies as a ready surrogate for predictions of organic black mud production rate in the majority of lakes where no significant surveys of mud status has been undertaken, it is now clear that too many other extraneous factors, ten were itemized, impinge on the issue for this to be a viable option. Thus, for the moment there seems no substitute for actual surveys of individual lakes, although some overall patterns can be drawn out.

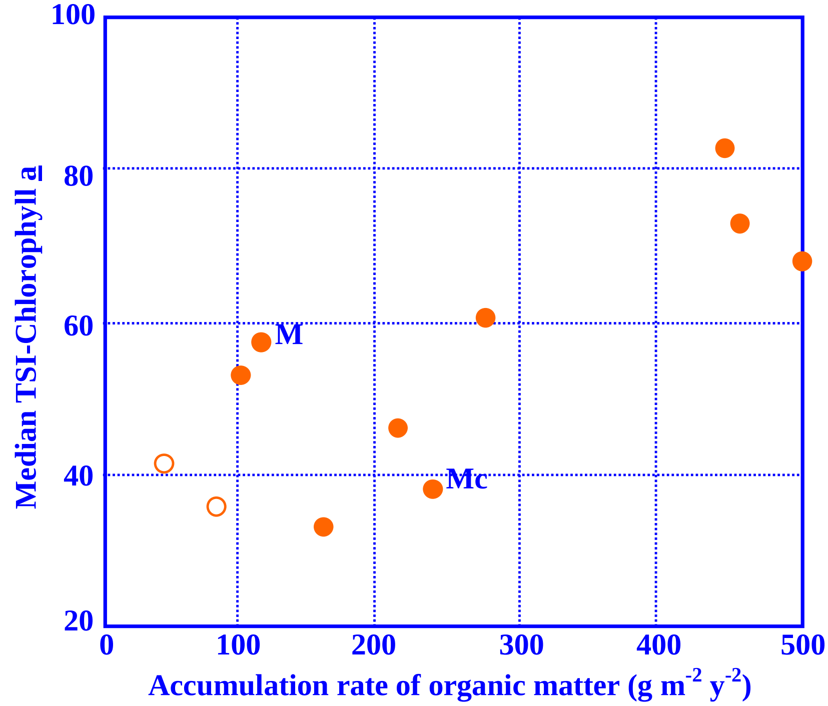


Figure A.4 Relationship between trophic state and accumulation rate of organic matter in lakes with $^{210}$Pb-dated cores. The accumulation rate (g $m^{-2}$ $yr^{-1}$) is the mean of the previous 10 years in each lake. The two open circles represent cores from Lakes Sheelar and Annie, reported and later re-evaluated from Thompson (1981) M-Lake Minnehaha, Mc-Lake McCloud (adapted from Binford & Brenner, 1986).

*Small Lakes:* Whitmore et al (1996) have reported sediment distribution patterns from seven Florida lakes and some distribution patterns from others are known. They studied Lakes Maggiore, Hollingsworth, Clear, Thonotosassa, Marianna, Parker and Seminole. These lakes, as with many others in Florida, are highly productive, and a superficial assumption might be that a high flux of organic breakdown products would give rise to a thick and even deposit of black muds on the beds of such lakes. Following surveys, probing and coring these lakes and from experience with others, they discovered some hypereutrophic lakes in which organic sediments were of minimal scale or completely lacking. This situation arose due to the shallow depth, frequent mixing, lack of stratification, and warm temperatures, all of which promote rapid breakdown of organic material.

In lakes containing a significant amount of black mud, including the seven above, an unexpected discovery was that sediment distributions were highly variable. Whitmore et al (1996) were able to categorize black mud distributions into three types: 1) uniform across the lake; 2) confined to deeper zones where these were present; and 3) restricted to the littoral zone and embayments, Figure A.5.

These distributions and aspects of their consequences are discussed in detail in Mehta et al (1997) but not repeated here. The necessarily limited and laboriously conducted investigation by Whitmore et al is, nevertheless, of great value, indicating, as it does, that several factors contribute to black mud distribution in small shallow lakes: these being the strength and direction of the wind and the amount of shelter, the presence and distribution of macrophytes, basin shape, and basin bathymetry. These appreciations are vital in the consideration of management options.

*Large Lakes:* The largest, and in many ways statewide most important water body, is Lake Okeechobee. With respect to distribution and nature of its black mud deposits, the existence of this lake introduces a further controlling factor. In addition to the meteorologic, hydrologic, topographic and geologic factors relevant to the suite of lakes described by Whitmore et al (1996), Okeechobee is of a size large enough to permit the establishment of large scale patchiness of its phytoplankton composition. This phenomenon is familiar to oceanographers and in the lake is another possible contributor to variability in black mud distribution. The nature and distribution of black mud has been investigated by several research groups.

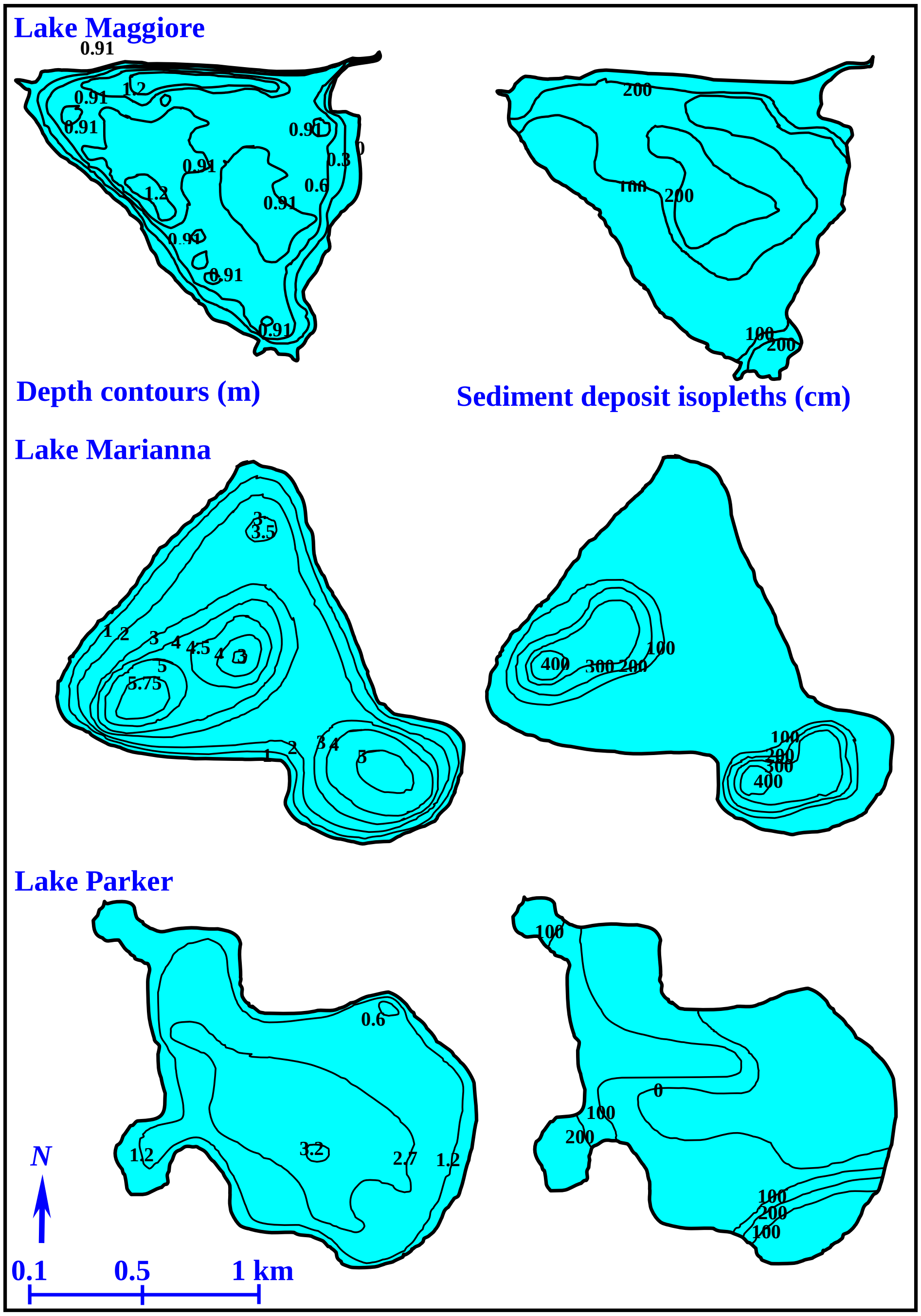


Figure A.5 Maps of organic black mud distribution in Lake Maggiore (relatively even distribution across whole lake), Lake Marianna (mud concentrated in bathymetric hollows), and Lake Parker (mud concentrated in peripheral areas and embayments) adapted from Whitmore et al, 1996.

The lake occupies an area of 1,700km$^{-2}$ (Phlips et al, 1997), or 1,810km$^{-2}$, (Stoermer et al, 1992). It is roughly circular in shape and close to 48km in diameter. It is a eutrophic lake with a maximum depth of about 4.5m$^{-1}$. Many attributes of the lake are poorly established, not least of which relate to the distribution and origin of its black mud.

Considering the plan distribution of black mud or its variation with depth, much is known but more questions remain outstanding. In plan the sediment is disposed in four major facies - 44% black mud, 28% marl and sand, 19% littoral and 9% peat (Olila & Reddy, 1993).

The bed sediment maps, Figure A.6 (Gleason et al,1978 and Gleason & Stone, 1994) and Figure A.7 (Phlips et al, 1997) do not include a large zone of littoral peat around the NE littoral zone mapped from coring surveys by Kirby et al (1994). Admittedly this is buried for much of its extent by black mud or black mud and sand, but is exposed at the lake bed in places. It is important because it complements a similar extensive littoral outcrop around the southern margin. Other than this, the main extent of the differing sediment types is now reasonably well recognized.

Maps of mud thickness contours (Figure A.8) are found in Kirby et al (1994). The black mud patch is asymmetrically disposed in the lake, occupying much of its center, with a northerly tongue reaching to Taylor Creek. The littoral zone, even on the eastern margin, is virtually devoid of black mud, as is the zone around the mouth of the Kissimmee River. The reason for the absence of black mud in the approaches to the Kissimmee River is not altogether clear, although almost the entire black mud body is contained within the -3m bathymetric contour. The total volume of the mud patch is approximately 193 x 10$^{-6}$ m$^{-3}$.

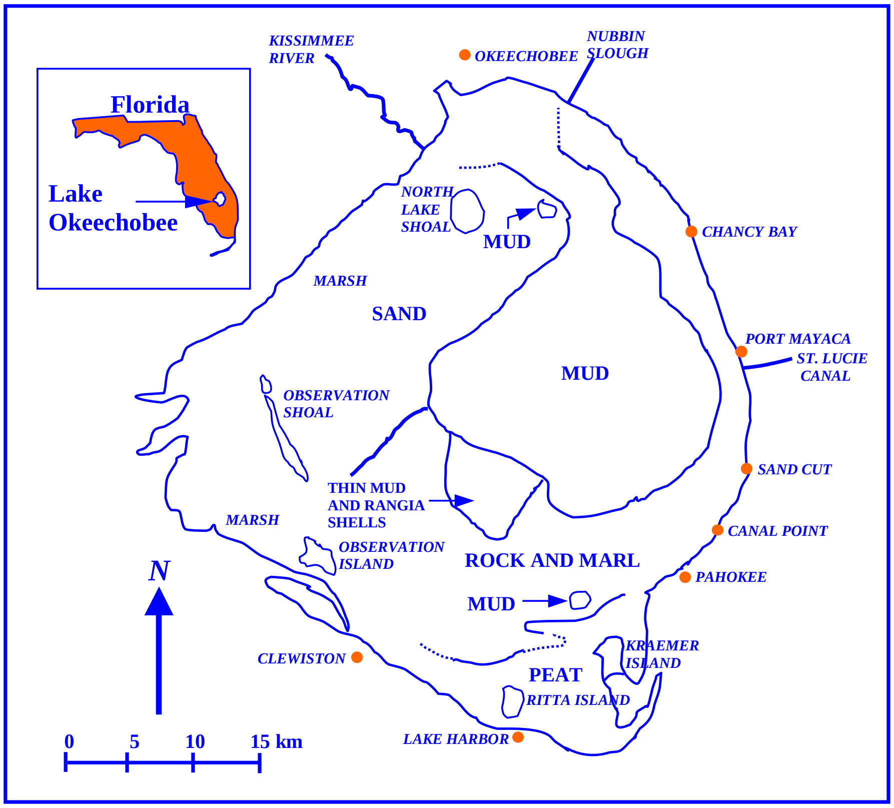


Figure A.6 Bottom sediment types in Lake Okeechobee (adapted from Gleason et al, 1978).

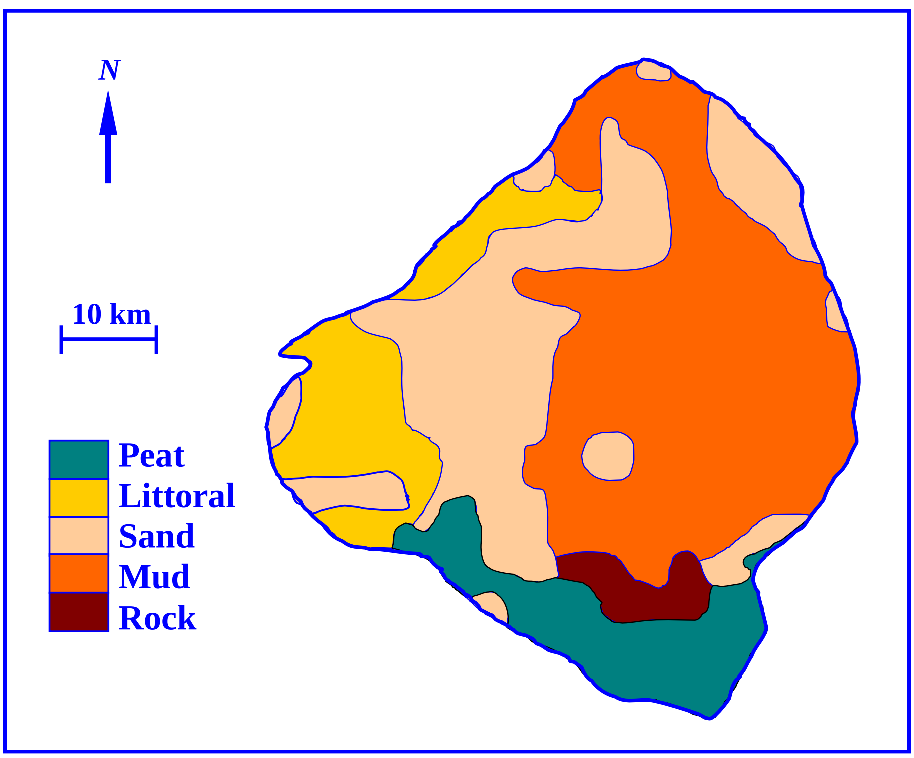


Figure A.7 Distribution of bottom sediment types in Lake Okeechobee (adapted from Phlips et al, 1997).

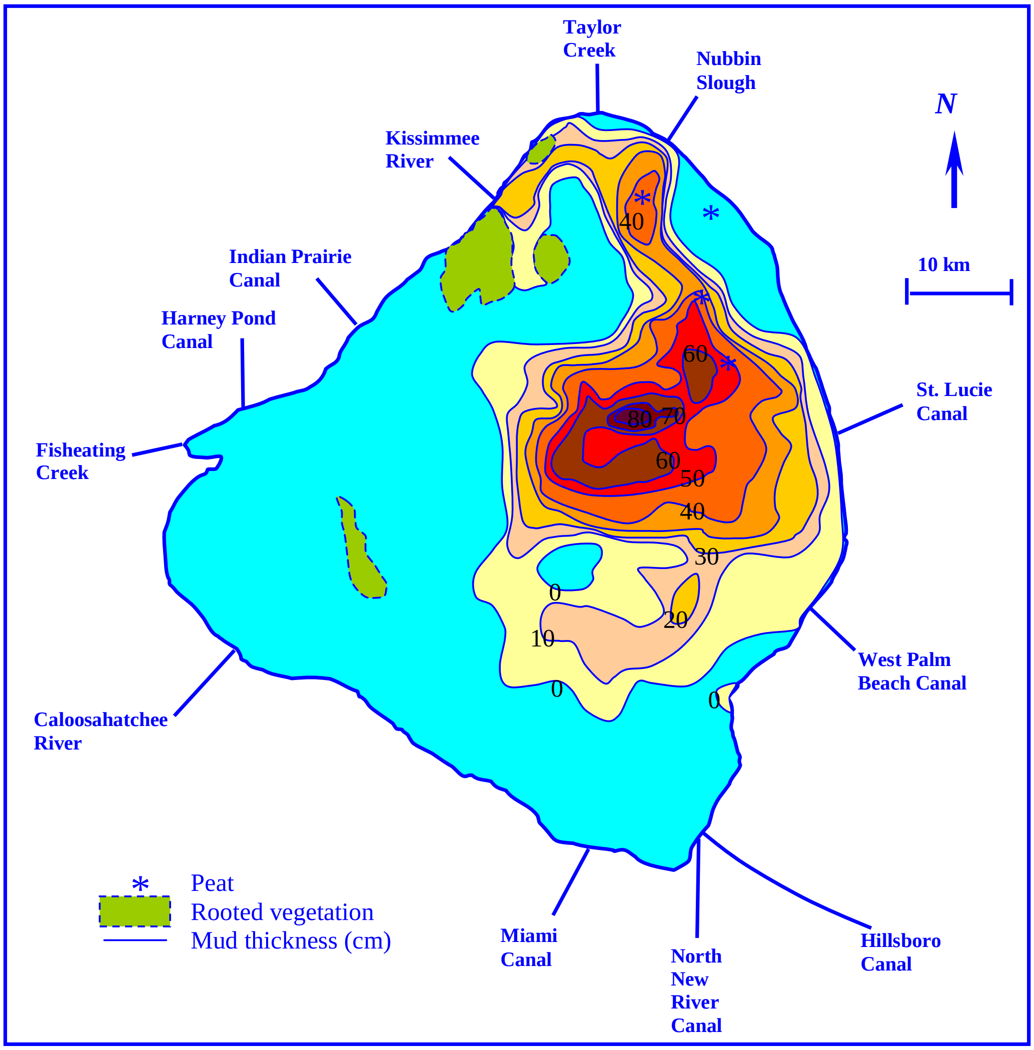


Figure A.8 Distribution and depth of unconsolidated sediment types in Lake Okeechobee. Away from the shore, organic sediment mainly occupies the northern two-thirds of the lake bed (after Kirby et al, 1994).

One important clue to the distribution of black mud in Lake Okeechobee may, therefore, be its association with the bathymetry. In this respect it may be comparable to Lakes Thonotassa, Marianna and Clear Lake in the descriptions of Whitmore et al (1996). However, there are very likely other contributing factors and issues. In addition to the possibly unexpected absence of black mud from the Kissimmee River approaches, the carbon production of submerged littoral vegetation is given by Zimba et al (1995) as 25,000 metric tonnes in 1990 and 28,000 tonnes in 1991. Evidently very little, if any, of this forms black mud and accumulates close to its site of production. The proportions exported in dissolved, compared to particulate form appear unknown and the fate, especially whether they escape the lake southwards or instead are reworked into the mud patch, are unknown.

Phlips et al (1997) has provided an important contribution to the issue of black mud distribution in Lake Okeechobee. These investigations have distinguished four ecologically distinct regions of the lake waters (Figure A.9). These are:

1. The large central region of muddy bottom sediments coincident with the deepest water and frequently overlain by a turbid water column.
2. A northern region of muddy sediments in intermediate water depths subject to the major influence of nutrient-rich inflows.
3. A western perimeter region with firm bed materials and subject to frequent plankton blooms.
4. A littoral fringe region along the western and southern boundaries of the lake enclosing a significant proportion of the submerged aquatic vegetation, combined with an extensive littoral community encompassing about 20% of the lake area.

The ecological distribution for the water column in Figure A.9 is similar to all the plots of bed sediment distribution (Figures A.6, A.7 & A.8). Studies reveal that the annual mean bio-volume of phytoplankton in Lake Okeechobee is about $3.2 \times 10^{-6}$ $\mu m^{-3}$ $ml^{-1}$. Mean annual phytoplankton volume was lowest in the central region due to elevated turbidity over the mud patch, 3 times higher in the north and 7-8 times higher in southerly and westerly regions. Similarly, taking diatoms alone, the annual mean

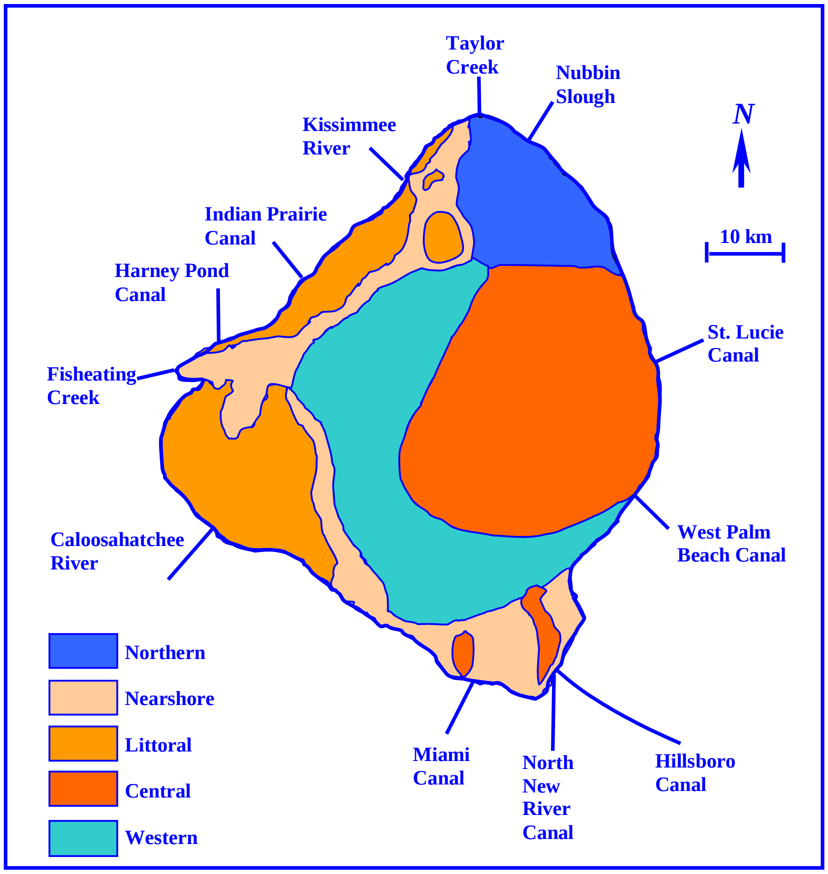


Figure A.9 Geographical boundaries of ecologically distinct zones in the water body of Lake Okeechobee (adapted from Phlips et al, 1997).

diatom biovolume was lowest in northern and central sites and had three to eight times the biovolume at southerly and western sites.

The region of transition between the predominantly light-limited center and north, and the nutrient-limited (by competition) littoral fringe is the site of the most frequent plankton blooms and a region of heightened productivity. It appears evident then, as with allochthanous inputs from rivers and autochthonous productivity of the macrophytes, particularly those of the broad western fringe, that a third source of organic carbon, the phytoplankton, is also not coincident with the zone of black mud.

The various transfer processes from these three to the mud patch have yet to be determined by researchers.

The history of these black mud deposits can be elucidated somewhat from X-radiographs of cores, (Kirby, 1994), and radiochemical, microfloral and microfaunal analyses, (Stoermer et al, 1992). X-radiographs of thick (0.5cm) vertical slices from the axial zones of black mud cores show fine and delicate dark and light zones no more than a few microns thick towards the bases of cores. Possibly these alternations represent the preserved history of individual plankton blooms in the lake? A dilemma posed by this fabric is how such individual layers could remain distinguishable if they had lain at shallow elevations in the black mud succession for many years and undergone frequent resuspension. Possibly they relate to periods of historically deeper water? The presence of a delicate primary fabric also implies that, certainly for the basal parts of the successions, at no time have the black muds acted as the host for significant populations of macroscopic burrowing organisms, otherwise a secondary biogenic fabric would dominate. This observation is consistent with the relative scarcity of macroscopic burrowing organisms in the modern surficial muds.

Stoermer et al (1992) established an age chronology for a single core based on $^{210}Pb$ analysis. This gives a mean deposition rate of 1.35mm $yr^{-1}$ and, assuming a steady sedimentation rate through time, a start date for deposition soon after 1610AD. The microfossil analyses appear to show four well-defined environmental changes. The relative abundance of planktonic to benthic diatoms is taken to indicate varying lake levels, latterly linked to man's attempts to regulate this. Diatoms also provide strong evidence for increases in the level of eutrophication during the period of Western settlement, which has also been accompanied by progressive salinization. In some respects the stratigraphy established leaves a number of outstanding questions.

Four coarse-grade sediment horizons are distinguishable and damaging hurricanes hit the Okeechobee region in 1890, 1926, 1928 and 1949. By analogy with less severe storm events, one might reasonably anticipate that they would cause large scale and deep winnowing of black muds. Increased river flows might input the coarser grade material or a lag deposit might be left on the lake bed following winnowing. However, only the uppermost sandy horizon appears coincident with the 1949 event and only the second, dated 1880-1900, appears coincident with a large change in lake water level. Consequently, whereas these scientific enquiries have established a number of aspects of lake history, many more remain unresolved.

## A.6 DISCUSSION

It is made clear above that Florida has a wealth and variety of organic sediments. For the most part the provenance for these is simple, well understood and needs no further elaboration. This is not the situation in large open water lakes where freshwater biology and especially microbiology dominates but is even today poorly understood. A few words concerning emerging issues and their significance for management may be appropriate.

Where the organic carbon comes from in a primary sense remains a vital question in aquatic biology. Whereas the 1970s work tended to treat systems in terms of productivity for phytoplankton, this is not the only issue, as somewhere in the region of 97% of daily carbon production is excreted back out into the water as glycolate, where it is available for re-use. Phytoplankton, therefore, recycle most of the carbon they acquire and the system is not best looked at in terms of productivity alone.

This issue of the role of phytoplankton and the magnitude of their input to carbon fluxes has another aspect, which bears in a major way on the ultimate source of the carbon. It has traditionally been thought that phytoplankton occupy the lowest rung of the food ladder by fixing carbon in the form of chlorophyll-*a* in the presence of sunlight. Phytoplankton are thought to be predated by zooplankton and in turn by the meiofauna, etc. In this traditional scenario, bacteria have the role of scavengers acting on biological secretions, dead cells etc. during the decomposition process.

A new and unproven suggestion is that bacteria may be the primary reservoir of carbon. The hypothesis is (Colin Reynolds, Freshwater Biological Association and Institute of Freshwater Ecology, UK, personal communication) that bacteria are what phytoplankton feed and depend on.

These recent ideas of reconsidering the 1970s global carbon productivity fluxes and reassessing the role of bacteria are the current challenges in aquatic microbiology but, as must be apparent in this context, have important engineering connotations where detrimental anthropogenic alterations to natural systems and the objectives of remediation are concerned.

Having considered sources of carbon in open lake water bodies, some modern thoughts concerning pathways through the system and indicators of alternative processing systems are in order. In regard to these, there are two end members. In the deep ocean all organic carbon is used up in the water column before it reaches the bed. On the other hand, in shallow water systems all the carbon may reach the bed before it can be handled in the water column. Naturally there is a spectrum between these extremes, the point of the examples being that it is important for engineering to appreciate why and where carbon is utilized.

It is suggested that, without recourse to detailed whole system appreciation, the zoo-plankton population of a water body are the key to how carbon is being cycled within the system. The zooplankton response is determined by two attributes of its potential food, firstly, that it is the right size and, secondly, that it is present in sufficient quantity.

In eutrophic systems, those abundantly supplied with nutrients, the carbon base is supplied by abundant bacteria and phytoplankton. These are grazed upon by the zooplankton species, *Daphnia,* or its equivalents, as an indicator that carbon is being cycled through the microphytoplankton. In these highly productive lakes the algal biomass will be high and of the order of 50% of the carbon cycled through zooplankton and higher organisms becomes nucleated in faecal pellets and other aggregates and a large fraction will reach the bed. This is the traditional concept of carbon processing. The remaining 50% of production will be excreted back to the water as $CO_2$. In contrast, when systems are superabundantly supplied with nutrients the algae produced are large colonial species far too large for a *Daphnia* to feed on and, instead, the indicator zooplankton will be rotifers and small crustacea dependant on the bacterial web. Productivity is high and again much will reach the bed.

There are two contrasting situations. In oligotrophic lakes, ie. those deficient in nutrients, resource limitation inhibits phytoplankton production and zooplankton feeding strategies dependent solely upon filtering, applicable to the prolific systems, are no longer viable. *Daphnia* and its relatives are replaced in such systems by calanoid copepods.

The alternative feeding strategies of these species present a large difference compared to the traditional food chain photosynthesis -phytoplankton - zooplankton - fish, etc.

Much more than 50% of the carbon fixed by copepods in oligotrophic lakes, via either of these feeding routes, is pushed out into the water as $CO_2$ and would be available for recycling. A relatively minor amount reaches the bed.

Thus, zooplankton species and feeding strategies are good indicators of how carbon is being cycled and exhibit three routes for carbon processing. Should remediation shift an open water system from a eutrophic to an oligotrophic status, one would anticipate the above mentioned changes in the cycling of carbon. Similarly, if an oligotrophic system was created, it may be important to anticipate whether a photosynthetic or ciliate feeding dependence on the zooplankton will be initiated. Much of the understanding needed in order to manage these systems is only now beginning to be fitted together after a rather prolonged investigative period. Suffice to say, this needs to be pursued if organic sediments and their management are to be placed on a firmer footing in future.

A further management implication of black mud biology, strongly evident from Cedar/ Ortega River and from Lake Okeechobee sediments and presumably the norm, is the scarcity and shallow burrowing depth of the macrobenthic infauna. On the one hand this severely restricts macrobenthic recycling of nutrients and pollutants, whilst on the other hand it serves to ensure that recent anthropogenic pollutants are confined to the shallow surficial layers of black mud deposits and not deeply interred by bioturbation.

## A.7 CONCLUSIONS

Organic-rich sediments in Florida are common, extensive, and typified by a broad range of types. In some of their natural states they are poorly described and understood. In particular, the source of this carbon is not always obvious. Carbon-rich sediment manifests itself as two contrasted types. In environments subject to a flooded and a dry period it occurs as a carbonate-forming sediment province whereas in permanently flooded and anaerobic regimes the “peat-forming” province arises.

This overview concentrates on the latter. These latter are chiefly derived from plant activities and readily subject to further subdivision.

Three of the fundamental sources of organics to fresh water systems are: 1) terrestrial plants; 2) aquatic plants in streams, small lakes or the margins of big lakes; and 3) big open lakes not linked to terrestrial or to littoral sources. The latter involve what is loosely termed microbial "plankton" carbon and involves the bulk of the organics produced in open waters of large lakes. Terrestrial soils produce a high biomass which, moreover, is readily decomposed in such environments. Organic materials are less readily broken down by aqueous "soils". By comparison microbial systems produce a much lower biomass but, in contrast, this is much more readily utilizable. Moreover, microbial systems frequently "crop" 5-6 times $yr^{-1}$.

In respect of inland organic sediments, terrestrial wetland peat deposits are also widely developed in Florida, most notably in the extensive Everglades area to the south. It is shown that bedrock topography has determined a number of diagnostic vegetation communities and these, in turn, have given rise to a number of distinctive and extensive peats, the Everglades, Okeelanta, Gandy and Loxahatchee peats. The peat deposits are refractory, highly resistant to reworking and generally infertile. They are not themselves linked to black mud production or to nutrient cycling. As a consequence, they don't give rise to major management challenges.

Organic sediment also occurs in the minor and major drainage channels, creeks, streams, rivers and estuaries, in small and large lakes and in marine inlets. It mainly takes the form of extremely finely divided, colloidal black mud which is poorly flocculated and consolidated, is highly susceptible to reworking and is in modern times a host for nutrients and a broad range of contaminants. Consequently, they are a focus of management concern.

The organic sediments of small water courses in Florida are only poorly described, although their natural carbon inputs from the terrestrial environment are documented. The three ultimate carbon sources to rivers and estuaries in Florida are set out above. These water courses are also the ultimate source of nutrient enrichment. Carbon sources to lakes are terrestrial, littoral lacustrine (aqueous) or “algal”. Florida has in excess of 8,000 lakes, only 10% of which have been studied limnologically. In only very few has the bed sediment been investigated. Lakes may be subdivided into clear and colored water and each further subdivided into oligotrophic, mesotrophic, eutrophic or hypereutrophic. Black mud distribution in these may be uniform, confined to deeper zones, restricted to the littoral zone etc. and is determined by factors such as the strength and direction of the wind, amount of shelter, presence and distribution of macrophytes, basin shape and bathymetry. In large open water lakes the carbon for the black mud is produced autochthonously from microbial, “algal”, sources quite separate from the land or littoral marginal production.

The acceleration in black mud production rate in the last century is suggested to be anthropogenically, nutrient-induced and to be focused not on terrestrial but on aquatic macrofloral and algal sources. Evidence for production rates enhanced by up to a hundred fold has been presented. For example, in specified coastal embayments 90% of total primary production is said to be caused by submerged macrophytes and benthic algae. The types of black mud problems in small and large lakes, in rivers and estuaries and in coastal areas are referred to in passing.

Other fundamental carbon sources in Florida are the surrounding oceans, of which, however, scant mention is made, and, fifth, the sea coasts. Major marine coastal carbon sources include seagrasses, saltmarshes and mangroves. There has been controversy in recent decades over the specific and generic role of saltmarshes in carbon cycling. It seems to be proven now that saltmarshes are highly productive, and, arising from the low tidal range in Florida, they are also large scale carbon exporters. This is not so elsewhere. There are no specific organic sediments diagnostic of *Spartina* marshes. Mangroves and seagrasses are also major carbon producers in southern Florida fringe coastal communities. Seagrasses and perhaps mangrove vegetation, too, are said to be the basis for trophic food chains in the region. Only mangroves are well documented to be a direct source of marine organic sediment in the form of mangrove peat. Whether carbon production rates from *Spartina* marshes, mangroves and seagrasses have risen or fallen in the last century and the extent to which free carbon liberated from decomposition and export from these sources might be contributing to the undoubted build up of black muds in coastal sediment sinks seems not to have been addressed.

Black mud remediation attempts have involved or considered use of drawdown, water level raising, capping, sealing, fetch reduction using wind breaks and breakwaters and dredging. Whereas the ubiquitous distribution of black mud presents severe management challenges, the attempts considered for the moment are conservative. The organic black muds of Florida do at least have a number of readily identifiable beneficial uses and emerging technologies render these significantly more readily attainable.

Finally, a cautionary perspective is provided of some of the ongoing challenges in freshwater microbiology concerning ultimate carbon sources and the way systems perturbed by further anthropogenic interference, such as in the MFLs context, might react.

# APPENDIX B: OPTICAL BACKSCATTER AND ACOUSTIC PROFILING FOR SEDIMENT TRANSPORT[1]

## B.1 SUMMARY

Time-series measurements of current velocity, TSS concentration, salinity and temperature from a station downstream of the C-18 Canal are presented. This canal acts as a tributary to the Loxahatchee River on the southeastern coast of Florida. The data were collected over an 18-day period during which the S-46 flow control structure was opened, allowing the ensuing effects sediment resuspension to be measured. This presentation serves as an example of the use of the optical backscatter sensor and the acoustic Doppler current profiler for measuring sediment loads in Florida's episodically driven flows.

## B.2 INTRODUCTION

As part of a study to examine options for sedimentation control in the Loxahatchee River estuary along southern Florida's Atlantic coast, data collection (water level, currents, TSS concentration, salinity and temperature) was carried out at three sites (UF1, UF2 and UF3) shown in Fig. B.1. Of present interest is UF3 in the Southwest Fork of the river, where it was desired to examine the effect of opening the S-46 sluice gate control in the C-18 Canal. The geographical coordinates of the site were: latitude 26$^{o}$ 56' 36.78" N and longitude 80$^{o}$ 07' 17.34" W.

[1]From: Patra, R. R., and Mehta, A. J. (2004). Sedimentation issues in low-energy estuaries: the Loxahatchee, Florida. *Report UFL/COEL-2004/002*, Coastal and Oceanographic Engineering Program, Department of Civil and Coastal Engineering, University of Florida, Gainesville.


Figure B.1 Measurement stations in the Loxahatchee River estuary.

The following instruments were deployed: an Aquadopp acoustic current profiler, three Sea Point optical backscatter turbidity sensors, a Vitel conductivity/temperature sensor for measurement of salinity and temperature, and a Transmetric water pressure recorder. Time series data were obtained over an 18-day period in 2002.

## B.3 RESULTS

Data presented in Figs. B.2-B.5 highlight the episodic nature of sediment advection down the Southwest Fork governed by the operation of the S-46 sluice gate. Except for the water temperature record, these data on current velocity, salinity and TSS concentration from site UF-3 are 12-hour means, so tidal oscillations have been largely filtered out from them.

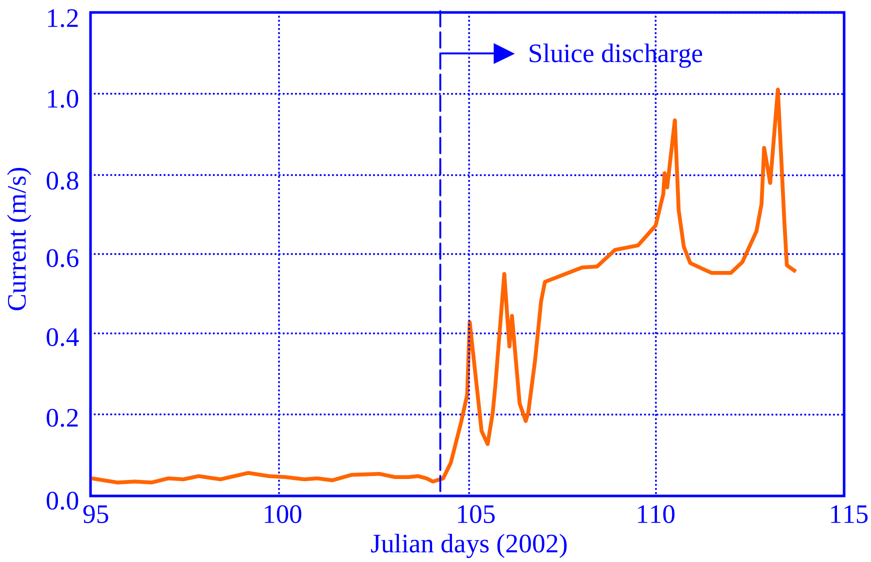


Figure B.2 Effect of S-46 operation (beginning April 14th, 2002) on the transport regime in the Southwest Fork: 12-hour-mean current speed derived from Aquadopp.

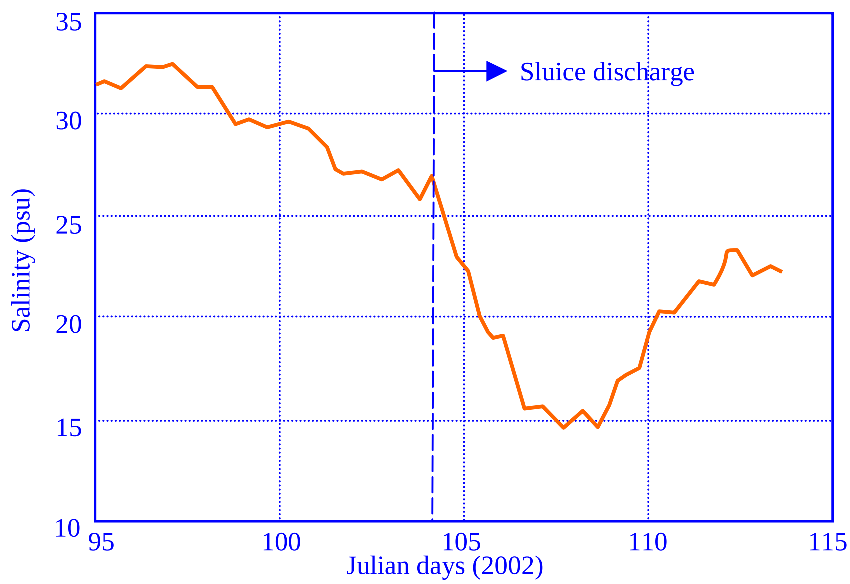


Figure B.3 Effect of S-46 operation on the transport regime in the Southwest Fork: 12-hour-mean salinity.

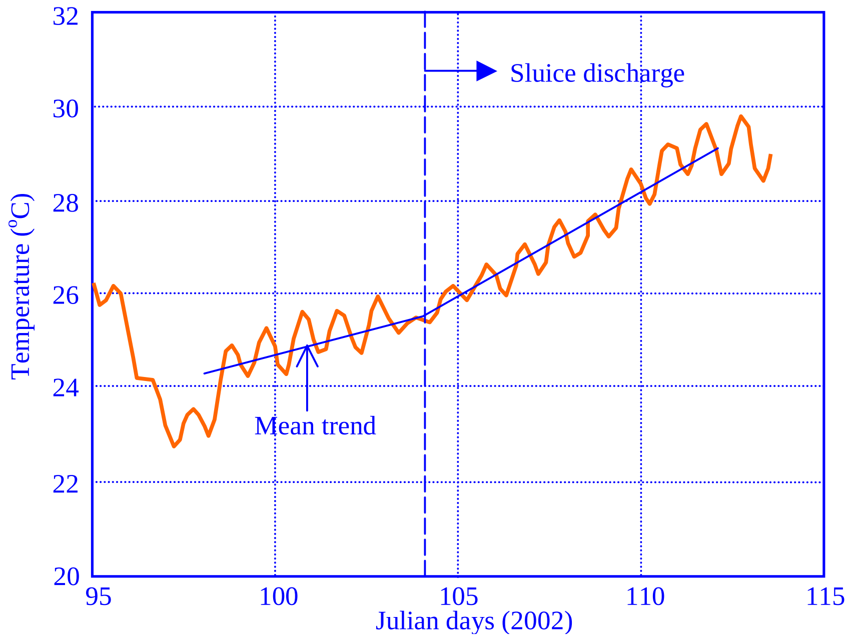


Figure B.4 Effect of S-46 operation on the transport regime in the Southwest Fork: water temperature.

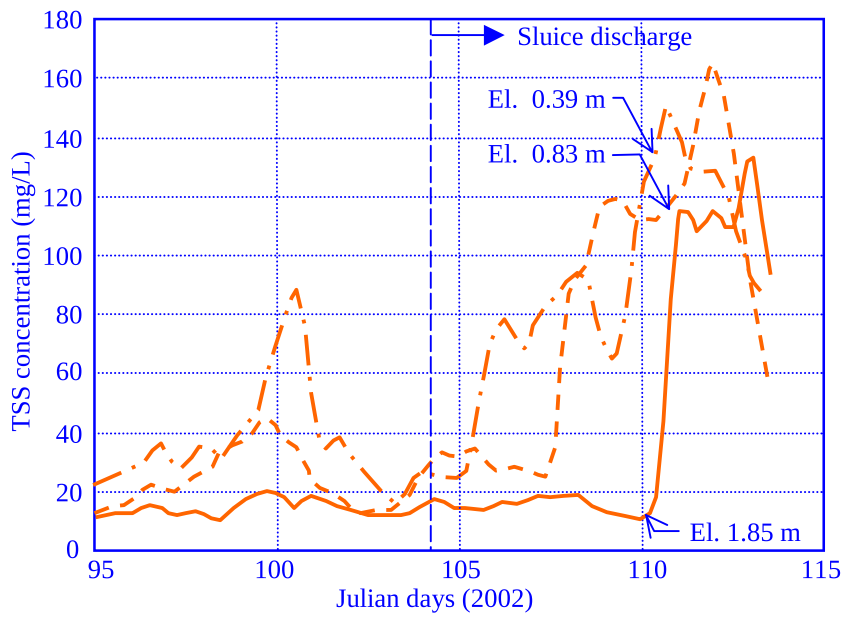


Figure B.5 Effect of S-46 operation on the transport regime in the Southwest Fork: 12-hour-mean TSS concentration data using OBS at three elevations from bottom.

On Julian day 104 (April 14$^{th}$), 2002 the sluice gate was opened. Until then the current meter recorded a small (almost negligible) flow velocity downstream. Yet, over the ensuing ten days the velocity rose to as high as 1 m/s (Fig. B.2) and the salinity dropped by as much as 12 psu due to freshwater flow (Fig. B.3).

Water temperature rose somewhat (Fig. B.4), presumably because the canal water was slightly warmer than the ocean, and TSS concentration increased dramatically from the usual 10-20 mg/L to 130-160 mg/L (Fig. B.5).

The TSS response lagged the current by several days because of the ~2.5 km distance between the sluice and the measuring station UF-3. This time lag reflects the retarding effect of tidal oscillations and associated particle settling and resuspension lags on sediment advection from the sluice to UF-3. Nevertheless it is noteworthy that despite this lag, TSS concentration rose by an order of magnitude over the ambient, implying that, ultimately, sediment transport down C-18 Canal is largely determined by the frequency and intensity of rainfall within the contiguous watershed.

## B.4 CONCLUDING COMMENTS

In Florida's typically low energy environment, OBS serve well because TSS concentrations are usually very low (< 30 mg/L). When strong flows occur they may rise by an order of magnitude, but often remain well below the 2,000-3,000 mg/L saturation limit of the sensors. If muck is transported as fluid mud, the concentration limit is exceeded, since in that case concentrations may reach or exceed 10,000 mg/L. This transport of muck by this mode is believed to occur occasionally, with layers that do exceed 5-15 cm in thickness. Long-term deployment of sensors, e.g., optical sensors that can measure concentrations on the order of 20,000 mg/L, is logistically inconvenient because of weekly or at least bi-weekly cleaning required against biofouling, the thickness and TSS concentration in muck is best measured where it occurs by coring.

## APPENDIX C: NUMERICAL MODELING TO ASSESS THE EFFECT OF WATER WITHDRAWAL ON SEDIMENT TRANSPORT

### C.1 SUMMARY

The Environmental Fluid Dynamics Code is used to illustrate the application of numerical codes to determine the effect of water (and sediment) off-take on the suspended sediment transport regime downstream. Two models are considered – one with the bathymetry and tidal conditions prevalent in the Cedar River in the Jacksonville area, and the other an idealized prismatic channel with steady flow. The change in the sediment transport regime is easily predicted for bed material load transport (as opposed to wash load). Such models however cannot predict the impact of off-take on river/lake morphology with any degree of accuracy, because morphodynamic relationships governing flow and geometry are not included in the fullest sense - the models can predict depth change but not width, which limits their application to short-term impacts.

### C.2 INTRODUCTION

Two models, using the Environmental Fluid Dynamics Code (EFDC) (Hamrick 1996), were developed and run to evaluate the effect of water withdrawal on suspended sediment and deposited bed sediment regimes. Model 1 was meant to show the utility of EFDC in dealing with off-takes in realistic situations. Model 2 was included as an idealized version of Model 1 which can be used to develop criteria for MFLs by coupling the response of water level with that of TSS at different rates of withdrawal. Brief descriptions of these models are as follows.

Model 1: An existing three-dimensional (3D) hydrodynamic, salinity and sediment transport model (described below) of the tidal Cedar River that was developed for the St. Johns River Water Management District (Mehta et al. 2004) was run to investigate the effect of withdrawing water on the sediment regime at two different sites along the modeling domain up to 100% of the flow in the upper Cedar River. The Cedar River is shown in Figure C.1.

Model 2: A 60 km reach of a 100 m wide prismatic channel with an assumed mean depth of 5 m, assumed discharge of 168.5 $m^3/s$, and assumed bed slope of $10^{-4}$ was modeled to evaluate the effect of withdrawing water during steady, uniform flows in this idealized waterway. Water was withdrawn 30 km downstream of the upstream boundary of this model. The following two versions of this model were developed:

Model 2a: Only deposition occurs in the 60 km uniform channel
Model 2b: Both deposition and erosion occur

Models 2a,b were run to investigate the effect of withdrawing water on the sediment regime at two different discharges at mid-reach. Model 2b was also used to investigate the effect of withdrawing water 15 km downstream of the upstream boundary during an unsteady flow hydrograph.

## C.3 MODEL DESCRIPTION

### C.3.1 The Environmental Fluid Dynamics Code

EFDC solves the three-dimensional (3D), vertically hydrostatic, free surface, turbulent-averaged equations of motions for a variable density fluid. The model uses a stretched (or sigma) vertical coordinate, and Cartesian or curvilinear, orthogonal horizontal coordinates. The physics incorporated in the EFDC model and many aspects of its computational scheme are equivalent to the Blumberg-Mellor model (Blumberg and Mellor 1987) and the U. S. Army Corps of Engineers Chesapeake Bay model (Johnson et al. 1993). Dynamically coupled transport equations for turbulent kinetic energy, turbulent length scale, salinity and temperature are also solved. The two turbulence parameter transport equations implement the Mellor-Yamada level 2.5 turbulence closure scheme (Mellor and Yamada 1982) as modified by Galperin et al. (1988). The EFDC model also simultaneously solves an arbitrary number of Eulerian transport-transformation equations

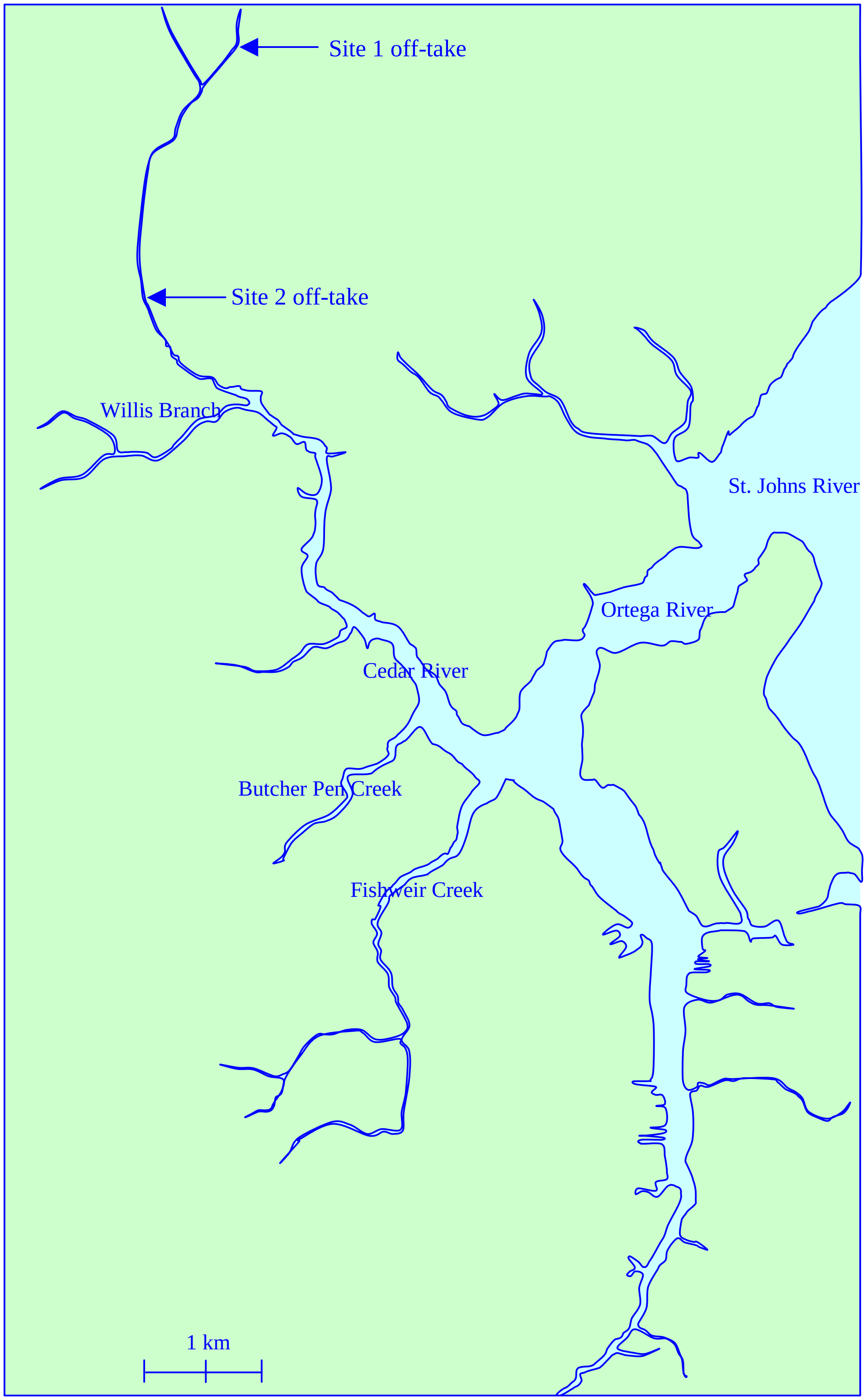


Figure C.1 Cedar River and its confluence with Ortega and St Johns Rivers (after Paramygin 2002).

for dissolved and suspended materials (e.g., cohesive and non-cohesive sediments, and contaminants), as well as bedload transport for non-cohesive sediment. The EFDC model also simulates drying and wetting in shallow areas (e.g., tidal marshes, mud flats, floodplains) using a mass conservation scheme. In addition, EFDC can simulate discharge control structures such as weirs, spillways and culverts, and it can read in the results from a wave transformation model to incorporate: a) radiation stresses due to wind-generated gravity waves; b) dissipation due to wave breaking and bottom friction; and c) wave-current boundary layer interactions.

EFDC was developed by John M. Hamrick while he was a faculty member at the Virginia Institute of Marine Science (Hamrick 1992). Dr. Hamrick now works for Tetra Tech, Inc. in Fairfax, VA. Continued development and support of EFDC has been provided by Tetra Tech, with this work mostly funded by the U.S. Environmental Protection Agency (EPA). The model has been applied to numerous rivers and estuaries in this country. It has been used to investigate a wide range of environmental problems including: a) sediment, contaminant and pathogenic organism transport and fate from both point and non-point sources; b) power plant cooling water discharges; c) oyster and crab larvae transport; d) dredging and dredge spoil disposal alternatives, and e) water quality problems in estuaries. A recent evaluation of contaminated sediment transport models reported that EFDC is one of the top-ranked public domain models for simulated the transport of sediments and contaminants in surface waters (Imhoff et al. 2003). Recently (Rouhani et al. 2004) EFDC was used for MFLs determination for Blue Springs located within the District.

The theoretical and computational basis for EFDC is described in Hamrick (1992), and a user manual for EFDC is given in Hamrick (1996). The description of the sediment transport algorithms incorporated in EFDC is given by Hamrick (2000).

### C.3.2 Model 1 – Tidal CR Model

The curvilinear-orthogonal grid for the Cedar River (CR) model consists of 845 computational cells. To simulate the partially stratified estuarine flow in the lower half of the modeled reach of the Cedar River, six vertical layers were used to represent the water column. The bathymetry of the modeling domain, the grid and locations of the six open water boundaries (BC1–BC6) are shown in Figure C.2. Enlargements of the downstream and upstream ends of the CR grid are shown in Figures C.3 and C.4, respectively. As seen in these figures, five cells were used to represent the lateral variability in flow and transported constituents, i.e., dissolved salt and sediment, over the entire length of the modeling domain. At BC1–BC5, the time series for discharge and suspended sediment concentration were obtained by the St. Johns River Water Management District (SJRWMD) using the SWMM model (Freeman 2001). Figures C.5 and C.6 show representative time series plots of the SWMM predicted discharges and suspended sediment concentrations at BC1–BC5. In addition, the SWMM predicted time series of direct runoff and non-point source sediment loads from subwatersheds along the Cedar River were added to the appropriate computational cells along the CR. The stage, salinity and suspended sediment concentration boundary conditions at the downstream boundary (BC6) were generated by the Cedar-Ortega-St. Johns River model developed for the SJRWMD (Mehta et al. 2004). The semi-diurnal tidal signal applied at the downstream boundary (BC6) is shown in Figure C.7.

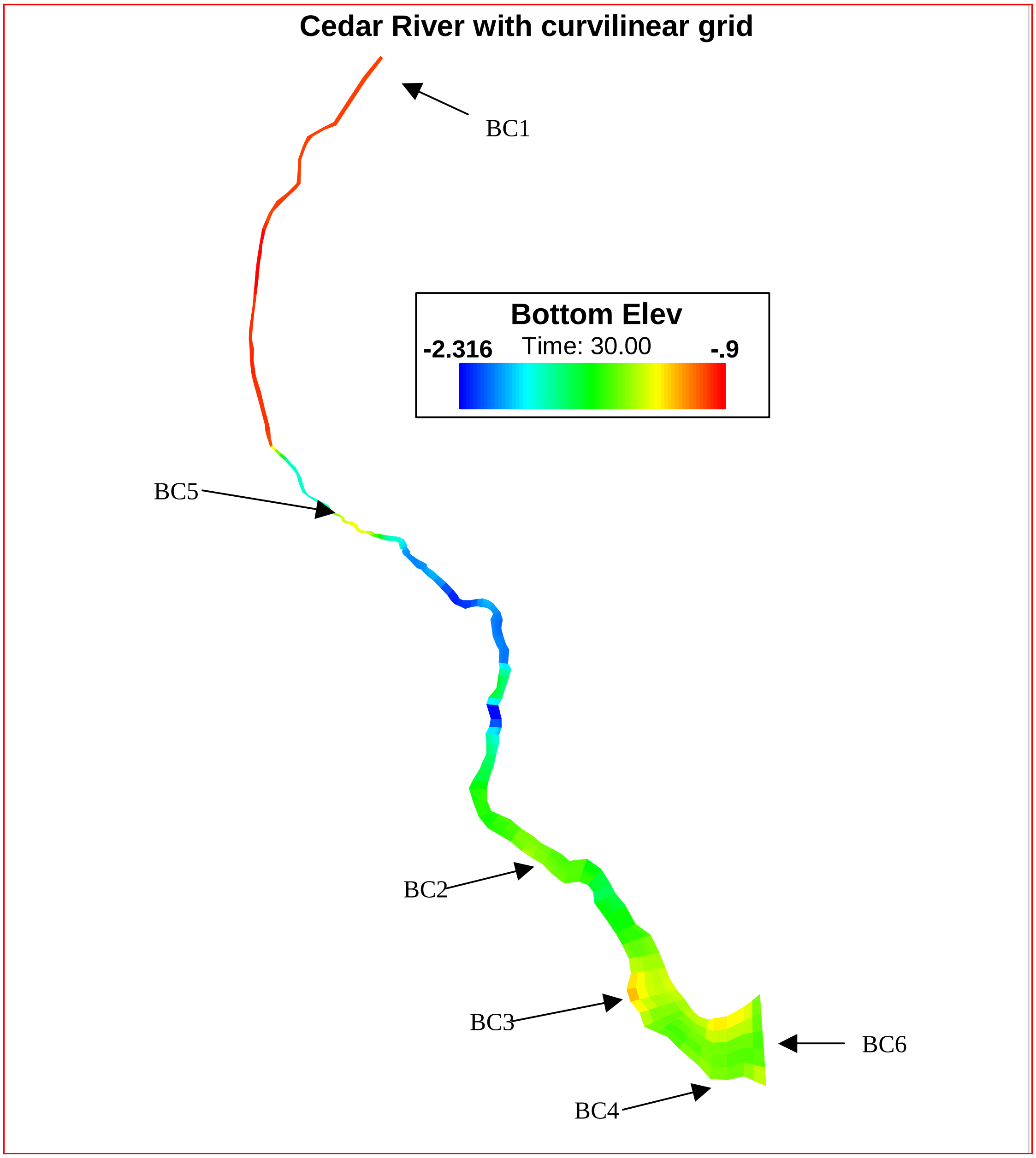


Figure C.2 Bathymetry of the Cedar River modeling domain. Bottom elevations are in meters with respect to NGVD. BC1 – Cedar River; BC2 – Williamson Creek; BC3 – Butcher Pen Creek; BC4 – Fishing Creek; BC5 – Willis Branch.

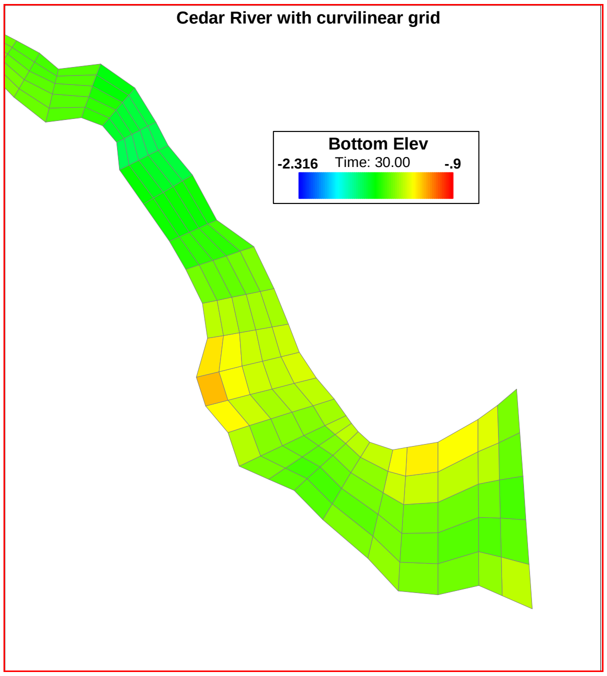


Figure C.3 Downstream End of the Cedar River grid.

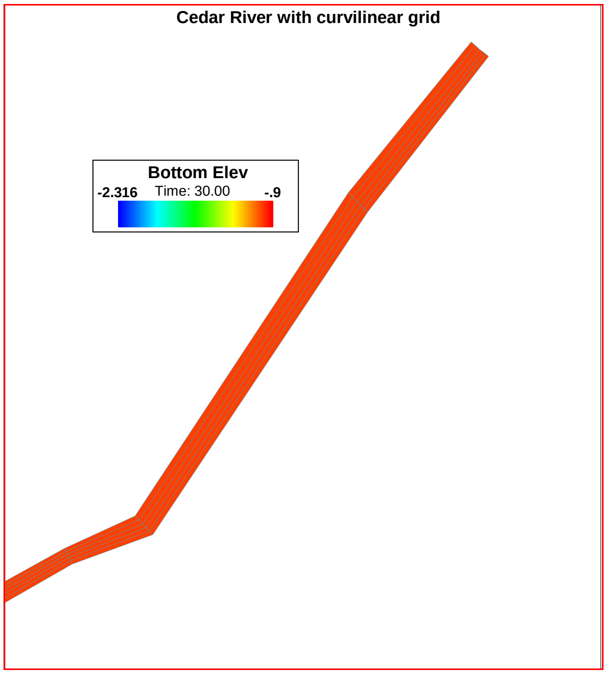


Figure C.4 Upstream end of the Cedar River grid.

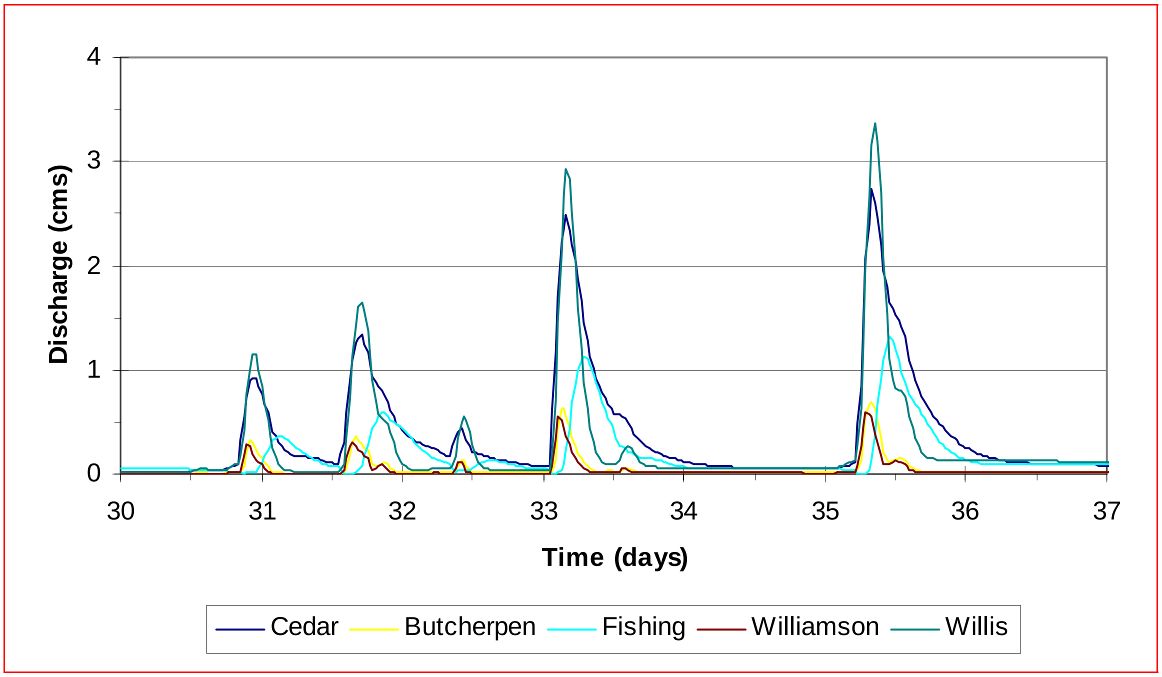


Figure C.5 Freshwater inflow time series for the Cedar River at BC1 – BC5.

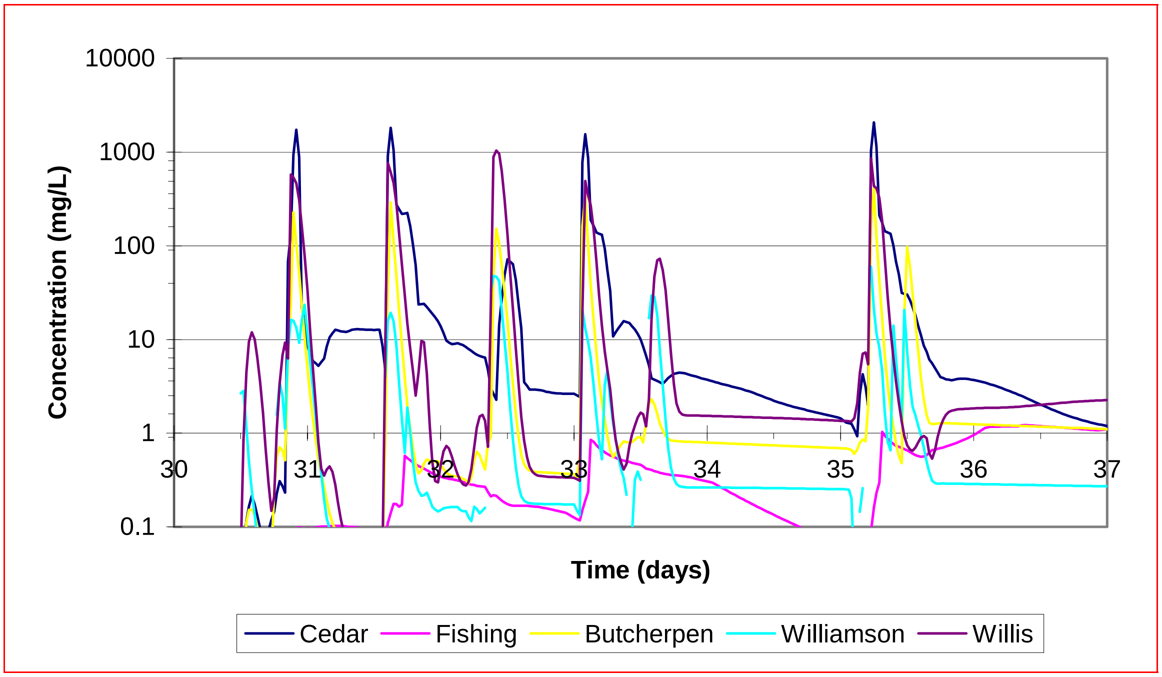


Figure C.6 Suspended sediment concentration time series for the Cedar River at BC1 – BC5.

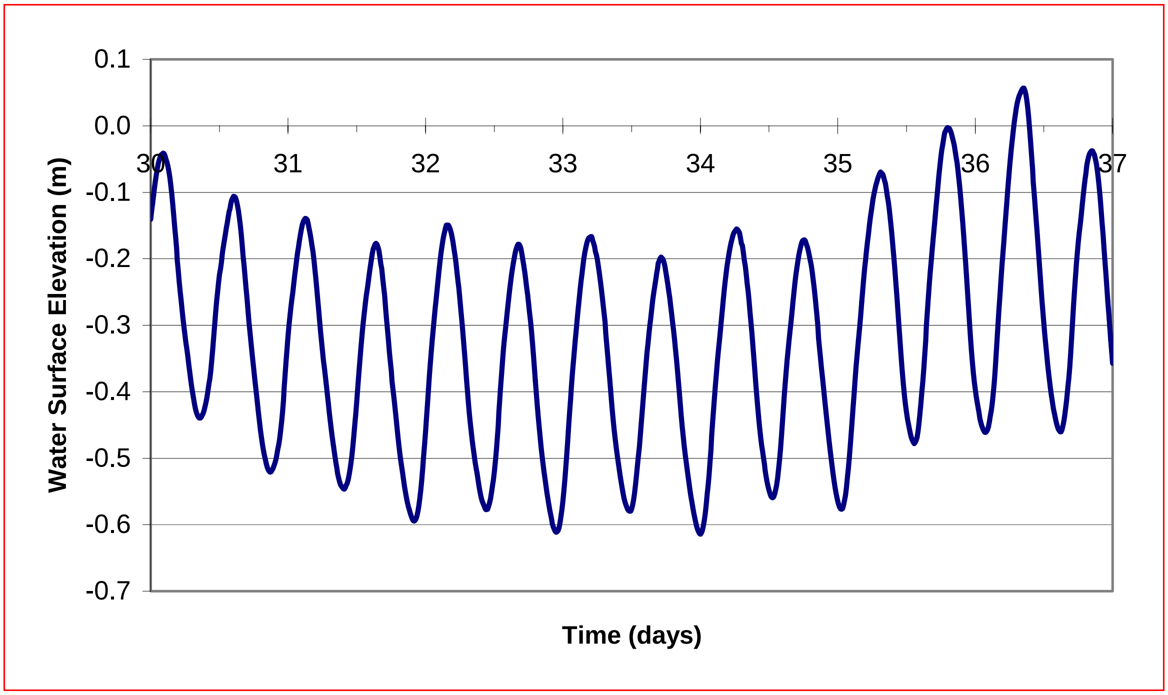


Figure C.7 Tidal signal used for the CR hydrodynamic boundary condition at BC6.

The vertically varying salinity time series applied at the middle cell at the downstream boundary are shown in Figure C.8. Similar time series are applied at the other four cells at BC6. Driven by the specified boundary conditions, the CR model was initially run (cold-started) for a 30-day spin-up period. A time-step of two seconds was used. This model runs at a speed of 148 simulated days per day on a 2.4 GHz Pentium 4 computer. The restart file generated by this run was used to hot-start the seven-day water withdrawal simulations.

### C.3.3 Model 2 – 60 km Prismatic Channel

Model 2 consisted on a 60 km reach of a 100 m wide prismatic channel with an assumed mean depth of 5 m, assumed bed slope of $10^{-4}$ and an assumed discharge of 168.5 $m^3/s$. The Cartesian grid for this model consists of 1,500 computational cells – 5 cells wide by 300 cells in the longitudinal direction. Six vertical layers were used to

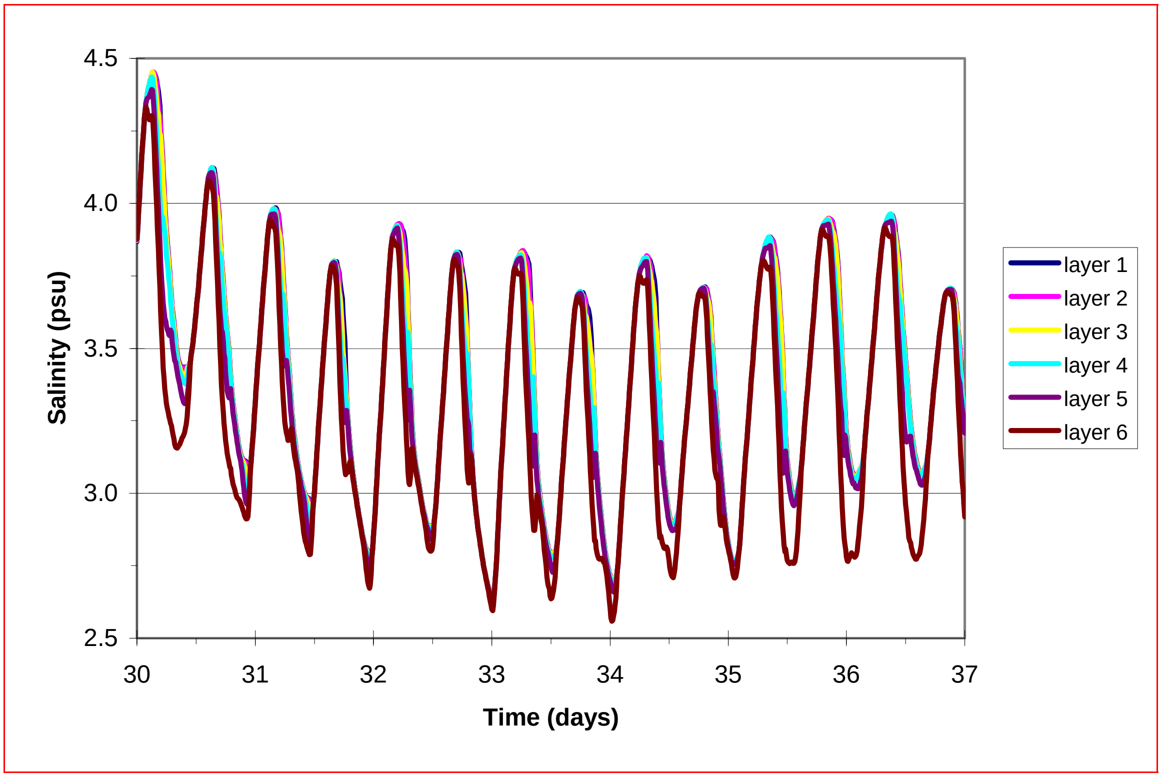


Figure C.8 Cedar River salinity time series at BC6, with layer 1 being the bottom layer.

represent the water column in each cell. To ensure that the flow was fully established at the withdrawal location, water was simulated to be withdrawn 30 km downstream of the upstream boundary of this model. The following two versions of this model were developed:

Model 2a: Only deposition occurs in the 60 km channel
Model 2b: Both deposition and erosion occur

These two models were run for seven-day simulations to investigate the effect of withdrawing water on the sediment regime at two different discharges at mid-reach. Model 2b was also used to investigate the effect of withdrawing water 15 km downstream of the upstream boundary during a 65-day unsteady flow hydrograph.

## C.4 MODEL SCENARIOS

The models described in the previous section were used to evaluate the (short-term) impact of water withdrawal on the sedimentary regimes in the modeled water bodies. The 15 withdrawal scenarios defined in Table C.1 were simulated. The first nine scenarios were run using the tidal Cedar River model. The location of Site 1, one of two withdrawal locations simulated, is shown in Figure C.1. The other withdrawal location, Site 2, is located approximately 200 m downstream of TG1 (see Figure C.1).

Table C.1 Water withdrawal scenarios

| Scenario No. | Model No. | Withdrawal | Site of Withdrawal |
|---|---|---|---|
| 1 | 1 | 0 % | - |
| 2 | 1 | 5 % | Site 1 |
| 3 | 1 | 10 % | Site 1 |
| 4 | 1 | 40 % | Site 1 |
| 5 | 1 | 100 % | Site 1 |
| 6 | 1 | 5 % | Site 2 |
| 7 | 1 | 10 % | Site 2 |
| 8 | 1 | 40 % | Site 2 |
| 9 | 1 | 100 % | Site 2 |
| 10 | 2a | 0 | mid-reach |
| 11 | 2a | 8.43 $m^3/s$ | mid-reach |
| 12 | 2a | 16.85 $m^3/s$ | mid-reach |
| 13 | 2b | 0 | mid-reach |
| 14 | 2b | 8.43 $m^3/s$ | mid-reach |
| 15 | 2b | 16.85 $m^3/s$ | mid-reach |
| 16 | 2b | 0% | 15-km |
| 17 | 2b | 10% | 15-km |
| 18 | 2b | 20% | 15-km |

Scenario 1 is the baseline case in which no water is withdrawn from any location in the Cedar River.

In Scenarios 2–5 varying percentages (from 5% to 100%) of the time-varying freshwater flow into the Cedar River upstream boundary were withdrawn at Site 1, whereas in Scenarios 6–9 the same percentages of water were withdrawn at Site 2.

Scenarios 10–15 were run using the prismatic channel model, with Scenarios 10–12 run using Model 2a, and Scenarios 13–15 using Model 2b. In these six scenarios, the given flow rate, varying from 0% to 10% of the discharge in the channel, is withdrawn from the middle (i.e., 30 km from the upstream end) of the 60 km long channel.

Scenarios 16-18 were run using Model 2b using an unsteady discharge and TSS hydrographs (see Figure C.9) at the upstream boundary. In Scenarios 17 and 18, water is simulated to be withdrawn at a distance of 15 km downstream from the upstream end. Figure C.10 show the discharge time series for these three scenarios at mid-channel (i.e., 30 km downstream of the upstream end). For these three scenarios, a 10-day cold start model run was used to spin up the model, followed by a 65-day hot start model run. The results discussed in the next section pertain to the 65-day hot start model runs.

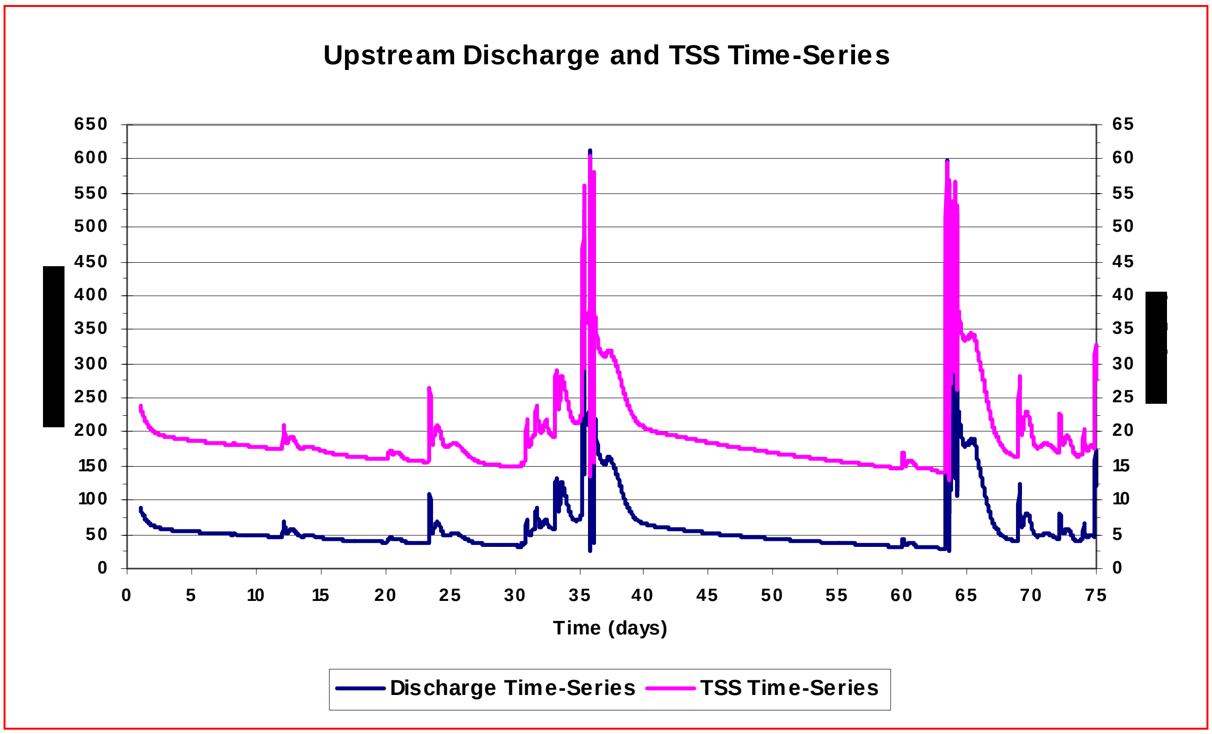


Figure C.9 Unsteady discharge and TSS hydrographs at upstream boundary used for scenarios 16-18.

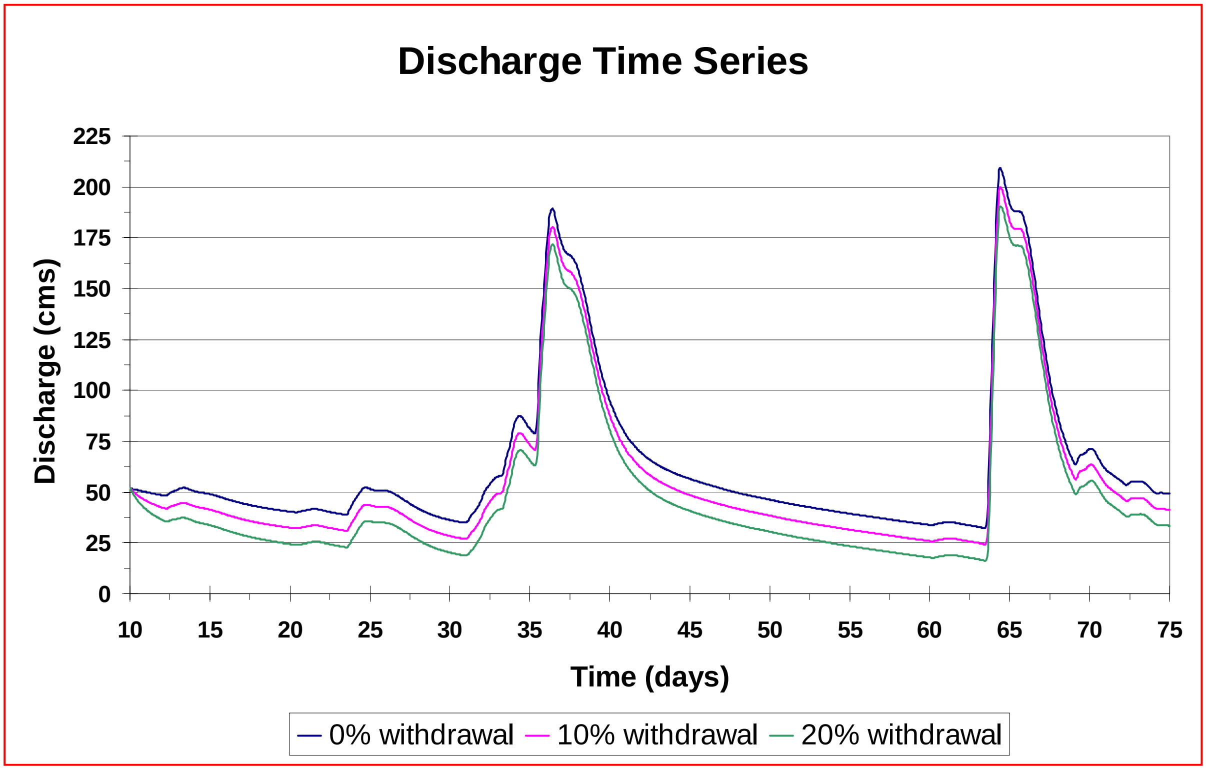


Figure C.10 Discharge time series at mid-channel for scenarios 16-18.

## C.5 MODEL RESULTS

### C.5.1 Model 1 – Tidal CR Model

Figure C.11 shows the location of two transects (marked as T1 and T2) along the Cedar River where the net sediment flux was calculated for 15 trapping scenarios (Mehta et al. 2004). Transect T1 is immediately downstream of Site 1 and Transect T2 is immediately downstream of Site 2. Figure C.12 shows the average suspended sediment concentrations at T1 and T2 over the seven-day simulation for Scenarios 2–5 as a function of the withdrawal percentage. As seen, the average sediment concentration at T1 first (insignificantly) increases up to a withdrawal percentage of 40% at Site 1, and then decreases sharply between the analyzed percentages of water withdrawals of 40% and

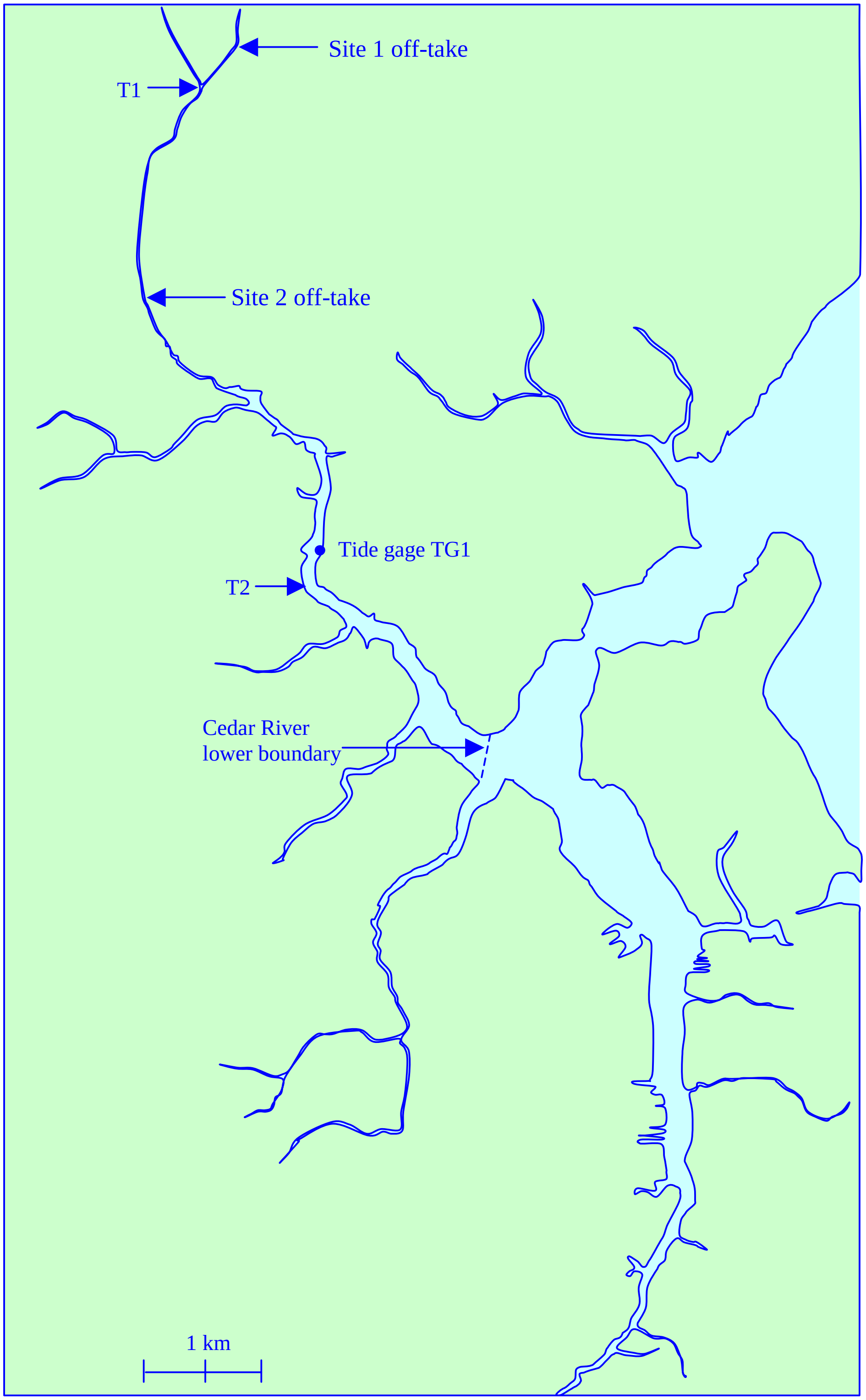


Figure C.11 Location of Transects T1 and T2 (from Mehta et al. 2004).

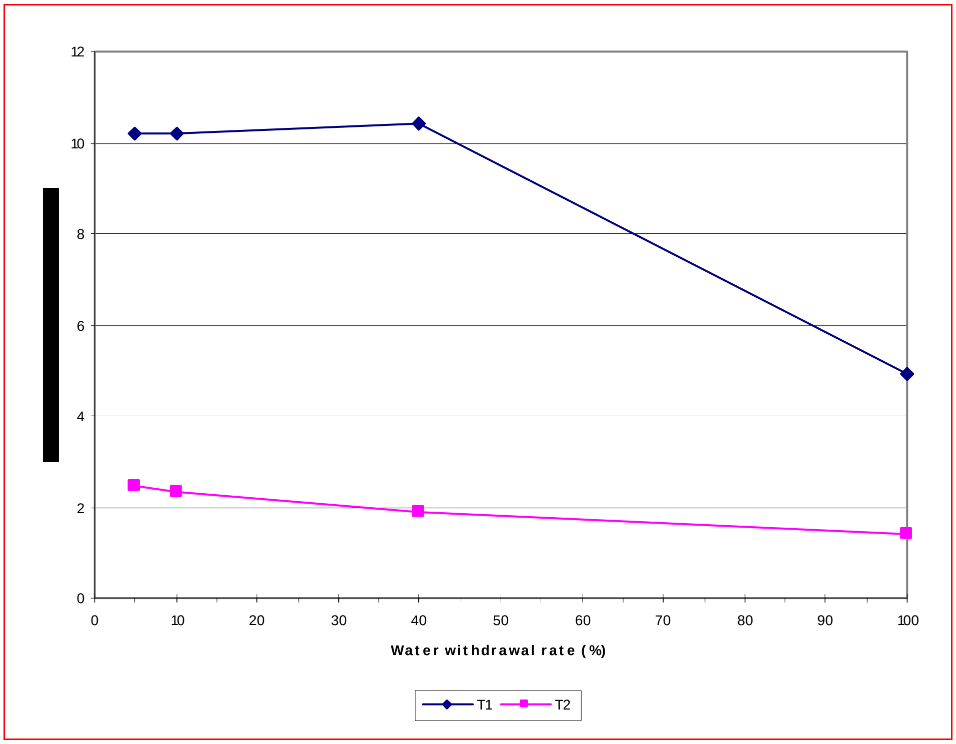


Figure C.12 Average suspended sediment concentration for Model 1 for scenarios 2 - 5 as a function of percentage of water withdrawal.

100%, while the average sediment concentration at T2 monotonically decreases with increasing water withdrawal at Site 1.

Figure C.13 shows the average suspended sediment concentrations at T2 over the seven-day simulation for Scenarios 6–9. This figure does not show the average sediment concentration at T1 since the water withdrawal location (Site 2) is well downstream of T1; thus, the withdrawal at Site 2 does not impact the average sediment concentration at T1. As seen, the average sediment concentrations at T2 decreases with increasing percentage of water withdrawal at Site 2.

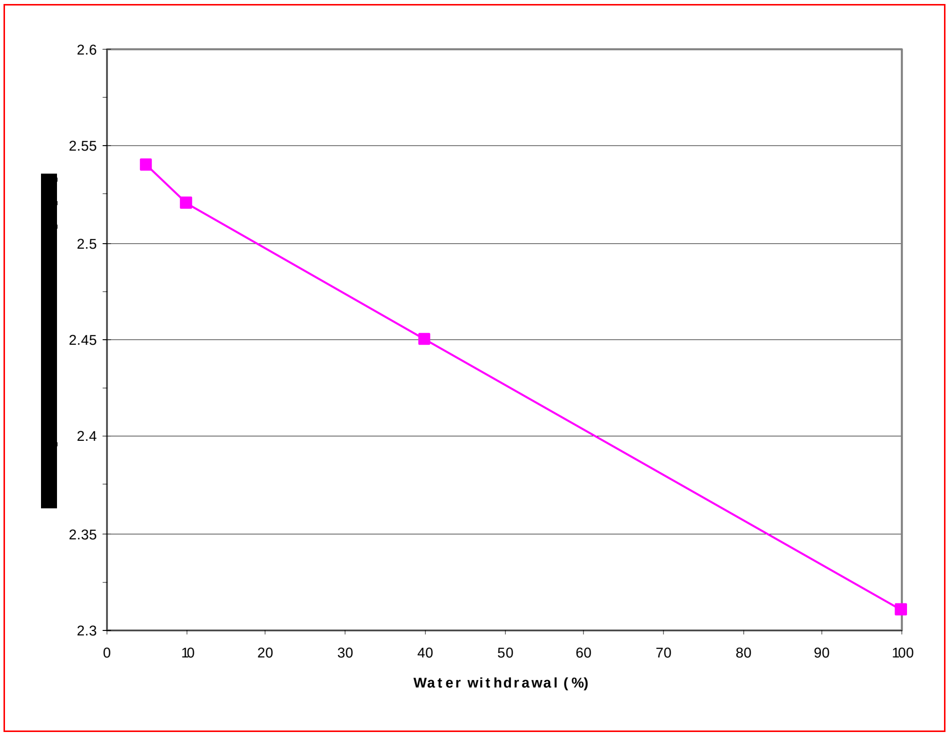


Figure C.13 Average suspended sediment concentration at T2 for Model 1 for scenarios 6 - 9 as a function of percentage of water withdrawal.

### C.5.3 Model 2a – 60 km Prismatic Channel

Figure C.14 shows the normalized net deposition for the 30 km reach downstream of the mid-reach water withdrawal site as a function of water withdrawal percentage. As seen, the net deposition decreases by up to 11.2% as the rate of water withdrawal increases up to 10% of the discharge in the channel. Figure C.15 shows the average suspended sediment concentration over the 7-day simulation at the end of the prismatic channel for Scenarios 10–12 (labeled Model 2a). As expected, the average concentration decreases, though only slightly, with increasing percentage of water withdrawal.

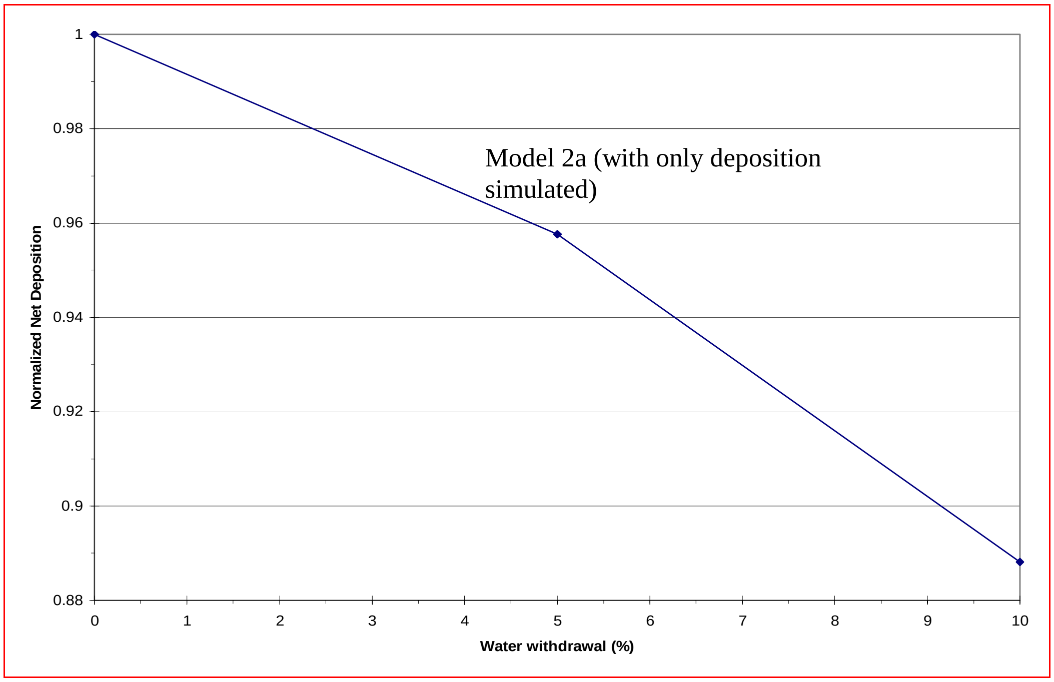


Figure C.14 Normalized net deposition downstream of the water withdrawal site as a function of percentage of water withdrawal for Model 2a.

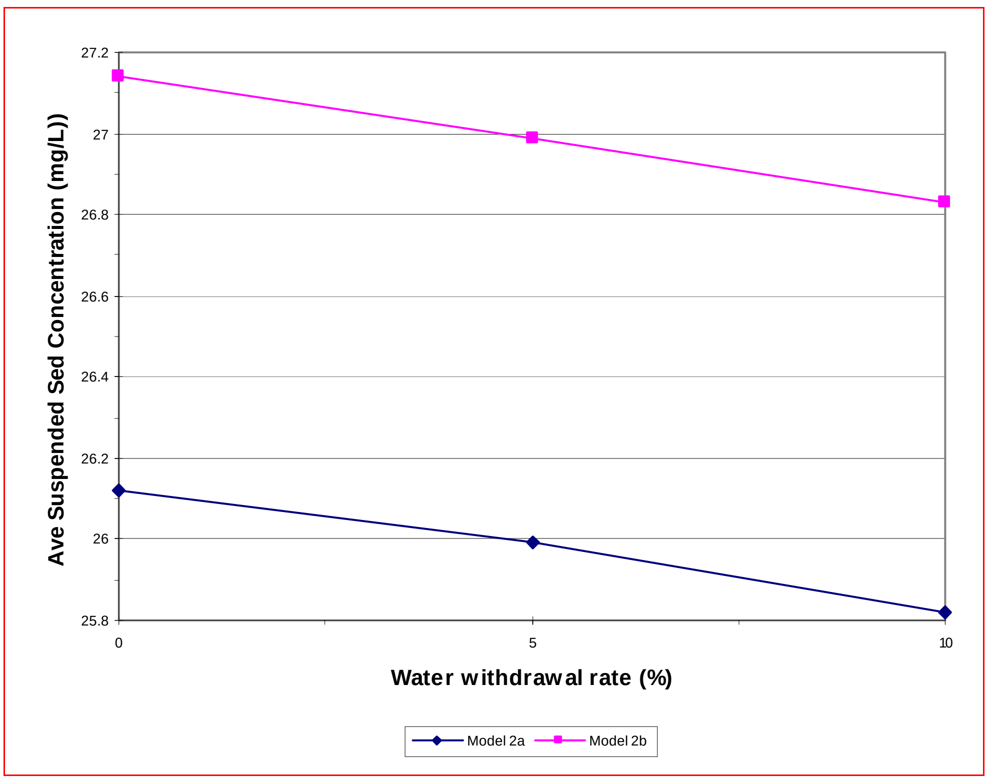


Figure C.15 Average suspended sediment concentration at downstream end of channel as a function of percentage of water withdrawal.

### C.5.4 Model 2b – 60 km Prismatic Channel

Figure C.16 shows the normalized net deposition for the 30 km reach downstream of the mid-reach water withdrawal site as a function of water withdrawal percentage. As seen, the net deposition decreases by up to 8.3% as the rate of water withdrawal increases up to 10% of the discharge in the channel. Figure C.15 shows the average suspended sediment concentration over the 7-day simulation at the end of the prismatic channel for Scenarios 13–15 (labeled Model 2b). As expected, the average concentration decreases, though only slightly, with increasing percentage of water withdrawal.

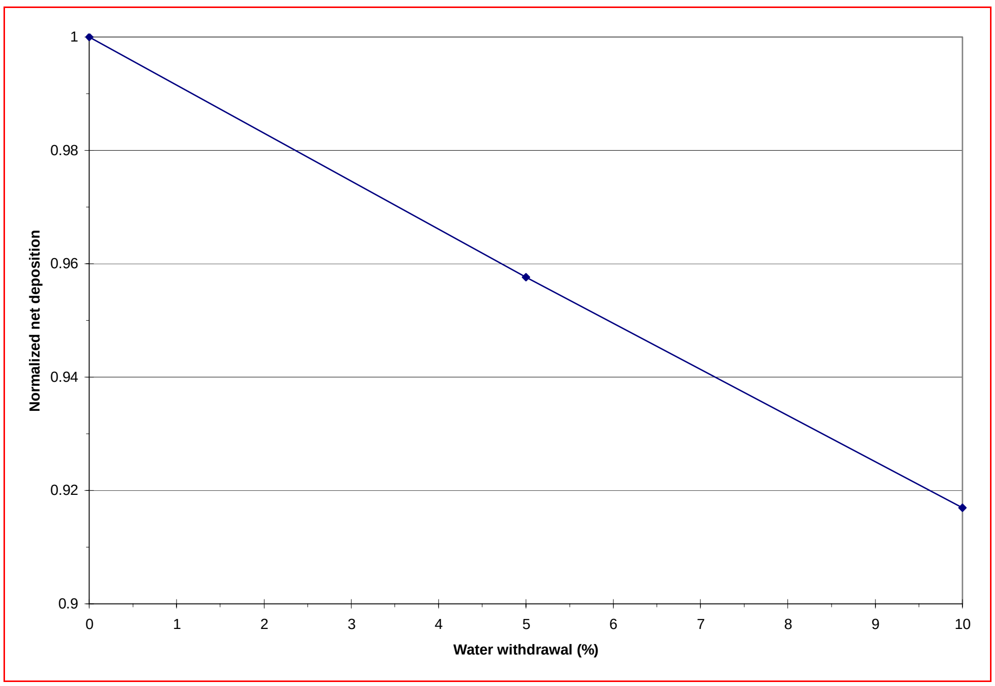


Figure C.16 Normalized net deposition downstream of the water withdrawal site as a function of percentage of water withdrawal for Model 2b.

### C.5.5 Model 2b with Unsteady Discharge and TSS Inflow Hydrographs

Figure C.17 shows the normalized net erosion that occurred for the 45 km reach downstream of the water withdrawal site as a function of water withdrawal percentage.

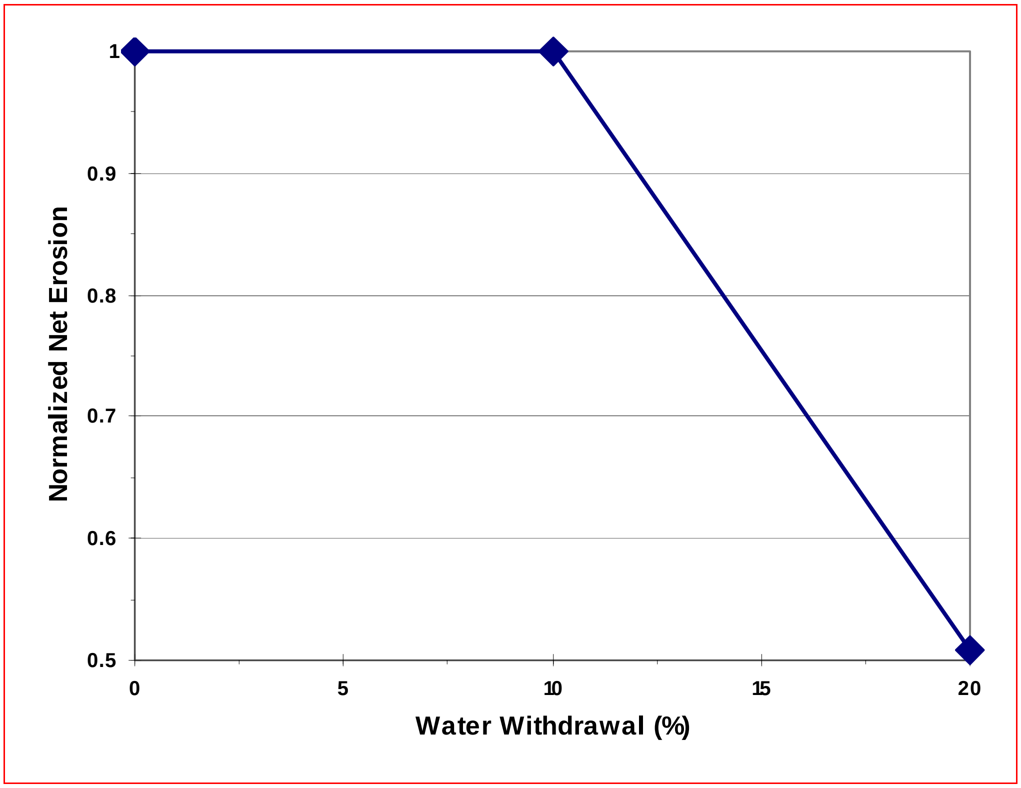


Figure C.17 Normalized net erosion downstream of the water withdrawal site as a function of percentage of water withdrawal for Model 2b with the unsteady hydrograph.

As seen, the net erosion was the same at 0% and 10% withdrawals, but decreased by approximately 50% at a 20% withdrawal. Figure C.18 shows the average suspended sediment concentration over the 65-day simulation at the end of the prismatic channel for Scenarios 16–18. As seen, the average concentration decreases slightly with increasing percentage of water withdrawal.

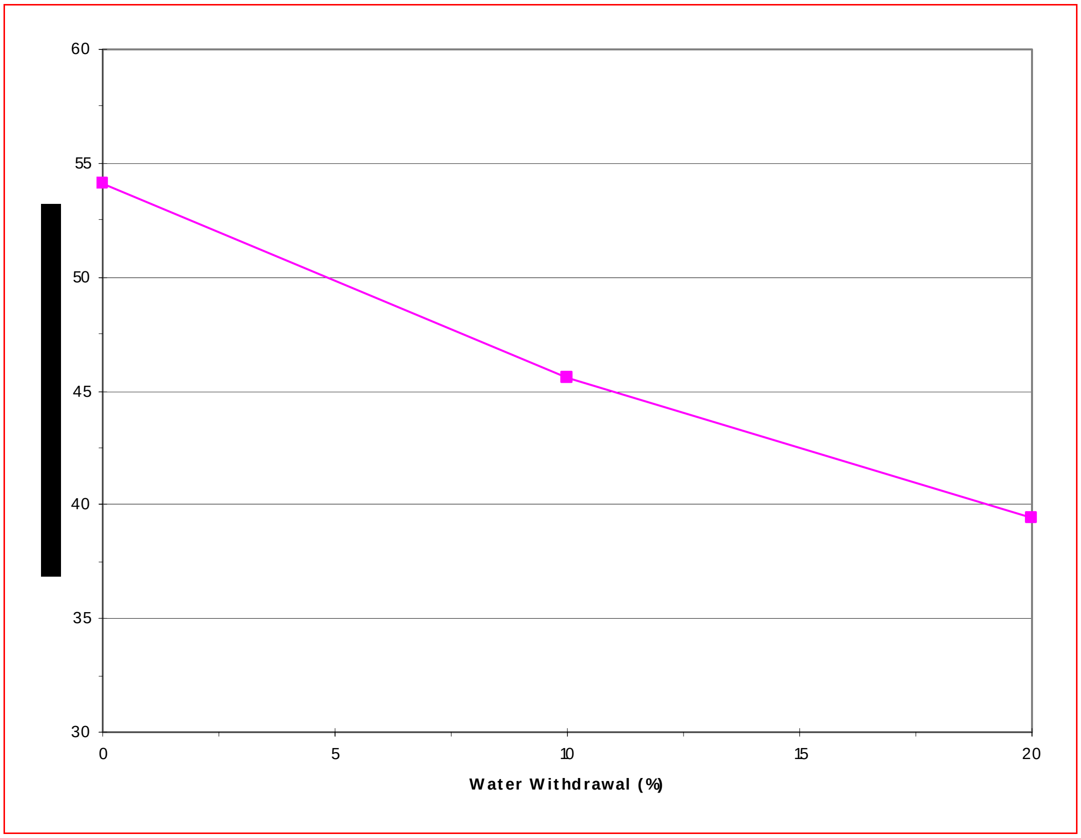


Figure C.18 Average suspended sediment concentration at downstream end of channel as a function of percentage of water withdrawal.

### C.5.6 Conclusions

The results seen in Figures C.14–16 for Scenarios 10–15 are the following:

1). As the rate of water withdrawal increases, the net deposition in the downstream reach decreases in an almost linear manner in this idealized prismatic channel.

2). The net deposition is the same in the two models for 0% and 5% water withdrawals. But for the 10% withdrawal simulations, the net deposition is less for Model 2b than it is for Model 2a.

3). As expected, the average suspended sediment concentrations at the downstream end of the channel are slightly higher for Model 2b than those for Model 2a.

These findings indicate that for a 10% water withdrawal rate, a small amount of erosion occurred since the net deposition for Model 2a (with only deposition) was more than that for Model 2b (with erosion and deposition). The result stated in the third bullet is not surprising since erosion was allowed in the simulations with Model 2b whereas it was not in the Model 2a simulations.

The results seen in Figures C.17–18 for Scenarios 16–18 are the following:

1). The net erosion in the downstream reach varied in a nonlinear manner with increasing water withdrawals.

2). The decrease in the average suspended sediment concentrations at the downstream end of the channel, coupled with the effect of withdrawals on the net erosion in the reach downstream of the water withdrawal site, indicate that water withdrawals greater than approximately 10% result in a decrease in the sediment load being transported out of the channel.

Three concluding thoughts are the following:

1). EFDC (and similar models) should not be used to predict the impact of withdrawal on river/lake morphology with any degree of accuracy, because morphodynamic relationships governing flow and river/lake geometry are not included in the code in the fullest sense - the models can predict depth change but not width, which limits its application to short-term impacts.

2). The presented analysis focuses on fine sediment transport behavior. For sand transport, for which only sparse data are available from the study area, the method of analysis will be analogous, and can be modeled using EFDC. This model enables calculation of both bed load and suspended (bed material) load.

3). MFLs are normally established by analyzing long-term (i.e., more than one year) hydrologic records. The purpose of the short-term (i.e., ~ two-months) modeling and analysis performed in this study is to demonstrate a modeling methodology that could be used to perform the long-term modeling and subsequent analysis required to establish MFLs.

## C.6 GRAPHICAL METHOD FOR STEADY, NON-EROSING FLOWS

Consider the case of the segment a waterway shown in Fig. C.19. The flow depth, h, is constant everywhere, B is the width and L is the length of the waterway. The coordinate x is measured along the thalweg of the waterway and z is the vertical coordinate. An advective change, $U\partial C/\partial x$, in the suspended sediment concentration C (where U = steady mean flow velocity) results from settling, represented by $W_s\partial C/\partial z$,

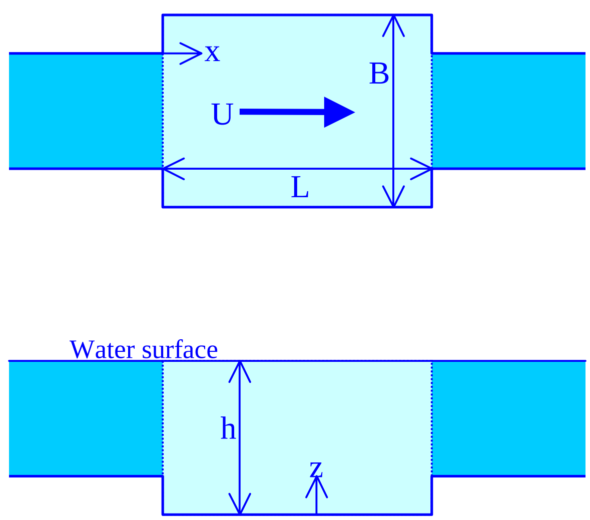


Figure C.19 Waterway segment as a deposition basin.

where $W_s$ is the particle settling velocity, and upward turbulent diffusion, represented by $\partial(\varepsilon\partial C/dz)/\partial z$, where ε = vertical diffusivity. Thus, mass balance gives

$$U\frac{\partial C}{\partial x} = \frac{\partial}{\partial z}\left(\varepsilon\frac{\partial C}{\partial z}\right) + W_s\frac{\partial C}{\partial z} \qquad \text{(C.1)}$$

In Eq. C.1, longitudinal (x-direction) dispersive transport is assumed to be small in comparison with the three indicated terms, and is therefore ignored. Considering a uniformly distributed concentration $C_o$ at the site of flow entry, the upstream boundary condition is

$$C = C_o \text{ at } x = 0 \qquad \text{(C.2)}$$

Further, considering the simple and common case of deposition without resuspension of the deposit, the bottom boundary condition is

$$\varepsilon\frac{\partial C}{\partial z} = 0 \text{ at } z = 0 \qquad \text{(C.3)}$$

and finally, there is no transport across the water surface so that

$$\varepsilon\frac{\partial C}{\partial z} + W_s C = 0 \text{ at } z = h \qquad \text{(C.4)}$$

Equation C.1 with the above conditions, plus assuming $\varepsilon$ to be independent of z and $W_s$ a constant, was solved numerically by, among others, Sarikaya (1977) by introducing the well-known parabolic distribution of $\varepsilon$:

$$\varepsilon = \kappa u_* z\left(1 - \frac{z}{h}\right) \qquad \text{(C.5)}$$

and the velocity was assumed to be logarithmically distributed:

$$\frac{u - U}{u_*} = \frac{1}{\kappa}\left(\ln\frac{z}{h} + 1\right) \qquad \text{(C.6)}$$

where $\kappa$ = Karman constant (= 0.4 for clear water), and $u_*$ = friction velocity. The solution is presented in graphical form in Fig. C.20, where the sediment removal ratio, r, is defined as

$$r = \frac{(q_s)_i - (q_s)_e}{(q_s)_i} \tag{C.7}$$

Here $(q_s)_i$ = amount of sediment in the influent and $(q_s)_e$ = amount of sediment in the effluent.

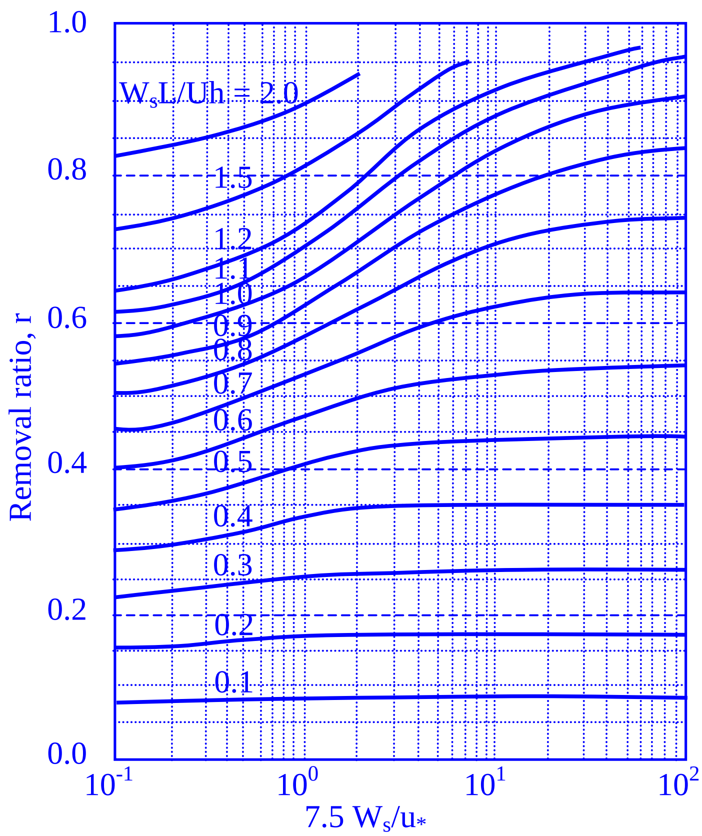


Figure C.20 Graph for calculating the removal ratio for a two-entrance basin.

As an example of the application of this method, consider the waterway shown in Fig. C.21. The segment in question is located between sections A-A and B-B. Although the flow is tidal and therefore oscillatory, sediment supply into the basin is known to be primarily through A-A (during flood flow). Thus, over flood, sediment is transported through A-A, a portion of which deposits in the segment and the remaining passes out of B-B. Resuspension of the deposit is negligible under the prevailing flow conditions. During ebb, negligible amount of sediment enters the segment through B-B and, once again, since resuspension is also negligible, the flow remains free of sediment. Relevant parameters are: average width of the segment, B = 300 m, the distance L between A-A and B-B = 3,000 m and the depth h = 10 m everywhere. Mean flood velocity U = 0.2 m/sec and the settling velocity $W_s = 2.1 \times 10^{-4}$ m/s. Selecting the Darcy-Weisbach friction factor f = 0.025 gives $u_* = \sqrt{f/8}\ U = 0.011$ m/s. Thus, with $W_s/u_* = 0.019$ and $W_sL/Uh = 0.32$, r = 0.24 from Fig. 8.8, which means that 24% of the incoming sediment settles out.

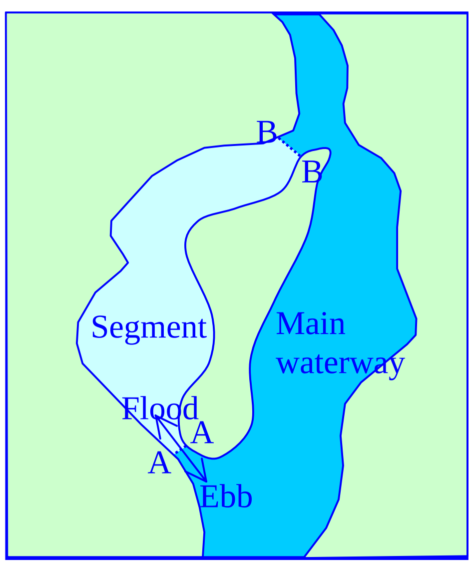


Figure C.21 Basin with two entrances and selective sediment transport.

Given the mean sediment flux through A-A during flood equal to $4 \times 10^{-3}$ kg/m$^2$ per second based on measurements, the rate of mass transport would be 12 kg/s, since the flow cross-section is 3,000 m$^2$. Over a semi-diurnal flood duration of 6.21 hours, $6.4 \times 10^4$ kg would be deposited. Given the measured dry bed density equal to 400 kg/m$^3$, the volume of deposit per cycle would be $1.61 \times 10^2$ m$^3$. Since the bed area is $9 \times 10^5$ m$^2$, the shoaling rate would be $1.79 \times 10^{-4}$ m per cycle. If this rate were on an annual average basis, the depth would decrease by about 12 cm in one year.

The above analysis can be applied to coarse or fine sediment. However, for fine sediment C, the TSS concentration, must be less than about 300 mg/L. Above this limit the settling velocity $W_s$ can not be assume to be constant, as it depends on C. In steady flows the application is more straightforward, and can be used calculate the change in sedimentation rate with change in velocity for MFLs analysis.

# APPENDIX D: RESUSPENSION OF ORGANIC-RICH FINE SEDIMENT IN A LAKE[1]

## D.1 SUMMARY

Recognizing the effect of organic matter on fine sediment aggregates, an effort has been made to empirically quantify the dependence of erosion and settling fluxes on the organic fraction using commonly employed functions for these fluxes. Based on data from studies in Florida's muck-laden waters, sediment density, erosion flux and settling velocity (hence flux) are shown to vary with organic content in a reasonably well-behaved manner. An examination of the effect of changing organic content on the potential for resuspension is considered for Newnans Lake, a shallow, hypereutrophic water body subject to episodic wind-wave action. Increasing organic content is shown to decrease the critical wind velocity for resuspension. A characteristic decrease in the organic content with depth within the lake bed implies that dredging the submerged bottom would increase the critical wind velocity due to this effect as well as the curtailed ability of waves to penetrate deeper water. Thus where resuspension is problematic, dredging enhances bed stability by two distinct, although not entirely uncorrelated, mechanisms.

## D.2 INTRODUCTION

The role of waterborne organic matter in modulating electrochemical bonds in clayey aggregates and consequently their resuspension was demonstrated decades ago (e.g., Kandiah, 1974). In response to the need to account for this effect for calculating the rates of scour and shoaling in waters laden with organic-rich sediments, data on the

[1] Paper presented by J. E. Gowland, A. J. Mehta, J. D. Stuck, C. V. John and T. M. Parchure at the 2003 International Conference on Cohesive Sediment Transport, Virginia Institute of Marine Science, Gloucester Point, VA.

erosion and settling fluxes have been collected in laboratory studies at the University of Florida on muck-like sediment derived from eight sites in peninsular Florida (Fig. D.1). In this report we have made an effort to show the utility of the resulting erosion and settling velocity functions by examining the potential for wind-wave induced resuspension in Newnans Lake in north-central Florida.


Fig. D.1 Bottom muck sampling sites in Florida.

## D.3 EROSION AND SETTLING VELOCITY FUNCTIONS

### D.3.1 Aggregate Structure

Muck aggregates in water (e.g., Fig. D.2) tend to be heterogeneous in composition, containing inorganic particles (e.g., clay minerals) as well as organic material derived from a variety of sources such as diatoms, worm tubes, biopolymers and wood fragments (Decho, 2000). For the present purposes we will consider the organic fraction to be represented by the loss in mass upon ignition of the sample. We will further consider that all aggregates contain the same or similar inorganic and organic constituents,

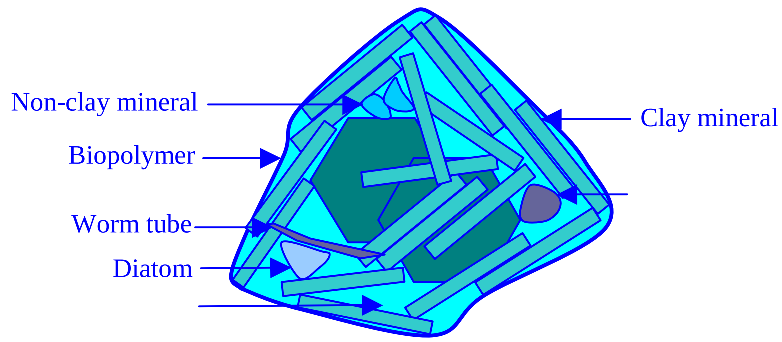


Fig. D.2 Fine sediment aggregate constituents.

but that the organic fraction may vary from sample to sample. This premise, which will allow us to link the variability in the erosion and settling properties of sediment directly with organic content, is rationalized by the historic occurrence of a unique source of inorganic sediment in peninsular Florida. There the clay minerals, mainly montmorillonite, kaolinite and chlorite, and non-clay calcareous, silicate and phosphatic sediments, are believed to have deposited during the Miocene (24M to 2M BP) as a result of their alluvial transport from southern Georgia to then submerged shallow Florida platform (Weaver and Beck, 1977). Also, the organic matter is largely derived from fresh and brackish water wetlands within a reasonably homogeneous subtropical environment.

Support for the above assertion concerning the uniformity of organic and inorganic mineral compositions is found in Figs. D.3 and D.4, in which the bottom sediment (wet) bulk density $\rho$ and the dry (bulk) density $\rho_D$ from six sites are plotted as functions of organic content ($OC$). These relationships should be considered to be applicable to reasonably consolidated beds, in correspondence with the state of the bottom samples from which the data were derived. Observe that despite data scatter arising from site to site variability in bottom samples collected by grab-samplers and

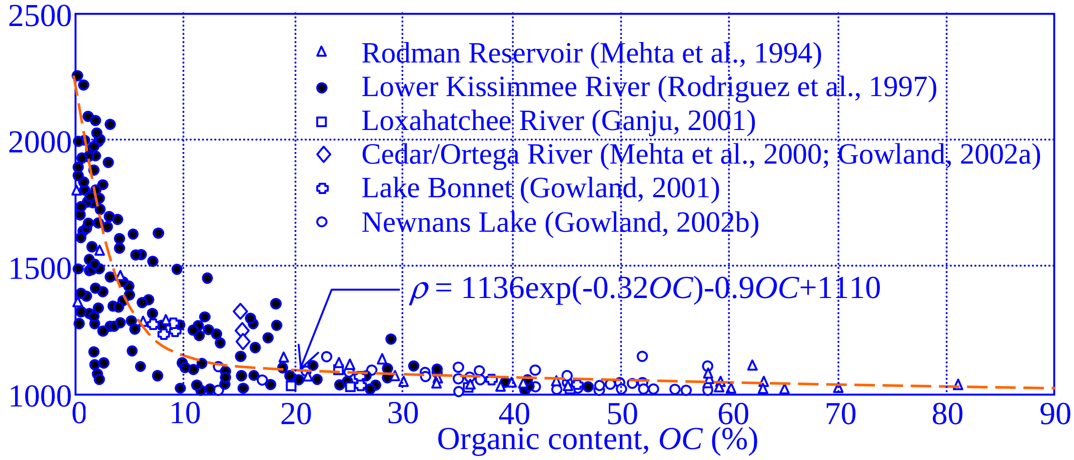


Fig. D.3 Variation of wet bulk density with organic content.

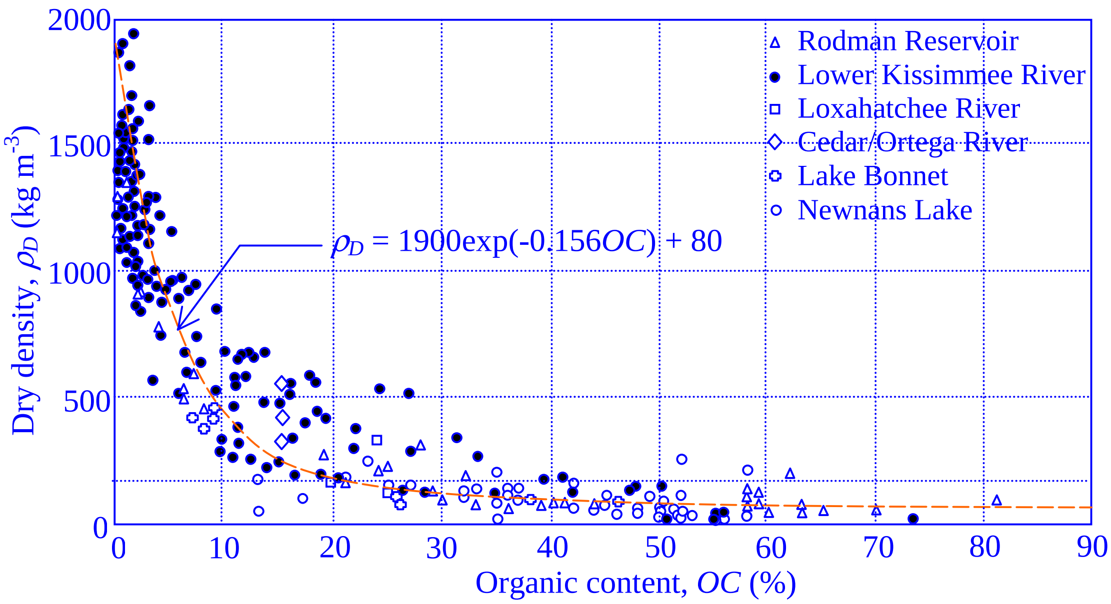


Fig. D.4 Variation of dry density with organic content; data from six sites.

corers and varying in thickness between ~0.1 m and ~1.5 m, mean trends can be identified and approximated as follows:

$$\rho = 1136\exp(-0.32OC) - 0.9OC + 1110 \tag{D.1}$$

$$\rho_D = 1900\exp(-0.156OC) + 80 \tag{D.2}$$

These relations are applicable in the range of $OC$ from 1 to 82%, and indicate a rapid decline in $\rho$ and $\rho_D$ as $OC$ increases from 1% to about 10%. This trend is the result of the aggregates acquiring an increasingly open structure, so that for a given total mass the volume increases.

### D.3.2 Erosion Function

For modeling bed erosion by waves we will select the simple linear equation

$$\varepsilon = \delta \frac{dc}{dt} = \varepsilon_N (\tau_b - \tau_s) \tag{D.3}$$

where $\varepsilon = \delta \partial c / \partial t$ is the erosion flux, $\delta$ is a characteristic bottom water layer height, $c$ is the total suspended solids concentration within this layer, $t$ is time, $\varepsilon_N$ is the erosion flux constant, $\tau_b$ is the bed shear stress (the periodic maximum value in case forcing is by waves) and $\tau_s$ is the bed shear strength against erosion (Dennett et al., 1998). For a bed of given density, $\varepsilon_N$ and $\tau_s$ were determined in laboratory apparatuses by applying a series of stresses $\tau_b$ of increasing magnitude over constant durations (typically ranging between 1 and 4 h) and measuring, for each $\tau_b$, the quantity $\varepsilon \approx h \Delta c / \Delta t$, where $h$ is the water depth in the apparatus, and $\Delta c$ is the increment in $c$ over a time duration $\Delta t$. From the best-fit line plot of $\varepsilon$ against $\tau_b$, $\varepsilon_N$ (line slope) and $\tau_s$ (intercept on the $\tau_b$ axis) were determined.

Several erosion tests were conducted in a Counter Rotating Annular Flume (CRAF), and others in a Particle Erosion Simulator (PES). In both devices beds of 3 to 5 cm thickness were allowed to consolidate between 24 and 96 h before the erosion test. The CRAF has been described in detail elsewhere (Parchure and Mehta, 1985; Stuck, 1996). Employing this flume, the results from a set of tests using muck from a farm drainage ditch in the Extended Agricultural Area (EAA) of south-central Florida (Fig. D.1) are shown in Fig. D.5.

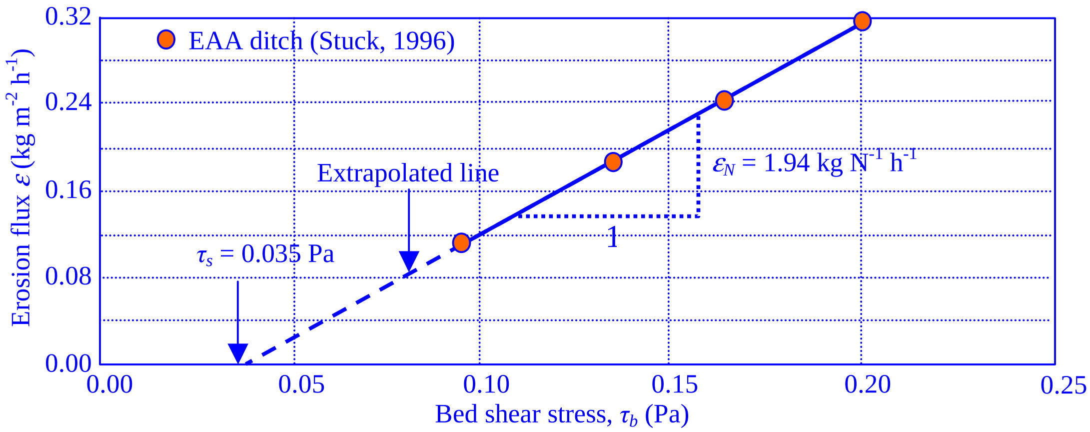


Fig. D.5 Erosion flux versus bed shear stress; data from an EAA ditch.

In the PES, described by Tsai and Lick (1986), the bed was prepared inside a 15 cm diameter perspex cylinder. A porous vertical grid submerged in water over the bed was then oscillated at different angular speeds (*rpm*), which caused the sediment to erode. The grid-associated shear stress was obtained from the calibration relationship $\tau_b$ $=5.91\text{x}10^{-4}$ x *rpm* derived by measuring and comparing the erosion of EAA muck concurrently in the PES and the CRAF (Stuck, 1996; Mehta et al., 1994).

Erosion test results using sediment from the sites in Fig. D.1 (with the exception of Lake Bonnet, for which such tests were not conducted) suggested that following the feasibility of representing the density measurements on a collective basis [Figs. D.3 and D.4; Eqs. (D.1) and (D.2)], the behaviors of $\varepsilon_N$ and $\tau_s$ may also be examined similarly. In agreement with the analysis of Mehta and Parchure (2000), in Fig. D.6 these two parameters, when plotted against each other using all available data, show a characteristic inverse dependence, i.e., as $\tau_s$ increases $\varepsilon_N$ decreases, rapidly at first and more slowly later as $\tau_s$ approaches unity. The initial value of $\varepsilon_N$ = 200 g N$^{-1}$ s$^{-1}$ is taken from that publication. Despite evident hiatuses in the data over the range of $\tau_s$ as well as data

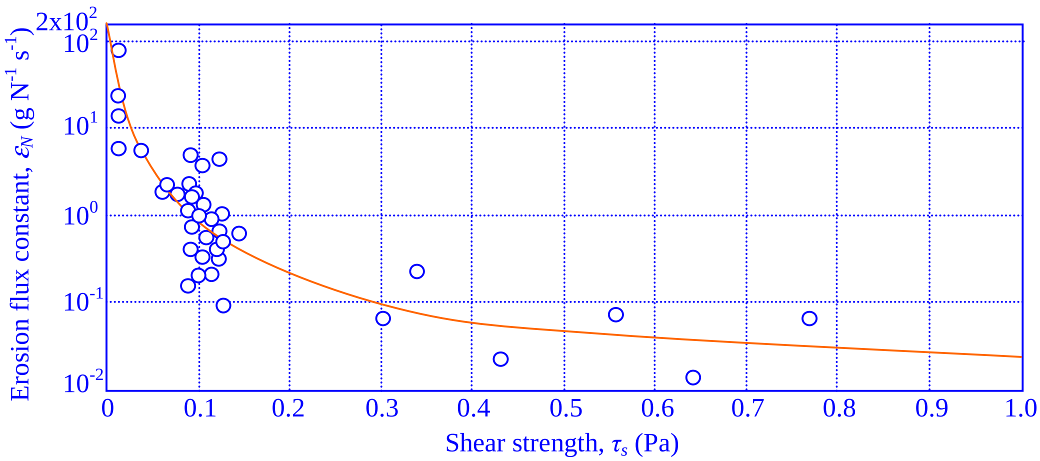


Fig. D.6 Variation of erosion flux constant with shear strength.

scatter, the plotted mean trend suggests that, in accordance with Eq. (D.3), increasing bed stability manifests itself as increasing strength and, hence, decreasing bed scour.

The shear strength itself shows a consistent variation with dry density ($\tau_s = 0.025\rho_D{}^{0.3}$; Fig. D.7). However, in comparison with what is found for clayey sediments, which carry exponents of $\rho_D$ between 1 and 2 (Mehta and Parchure, 2000), the dependence is weak (characterized by the exponent 0.3), and reflects the fact that organic-rich aggregates tend to effectively decrease (electrochemical) cohesion of the inorganic constituents. The outcome is that closer packing of particles with increasing density does not significantly increase aggregate strength.

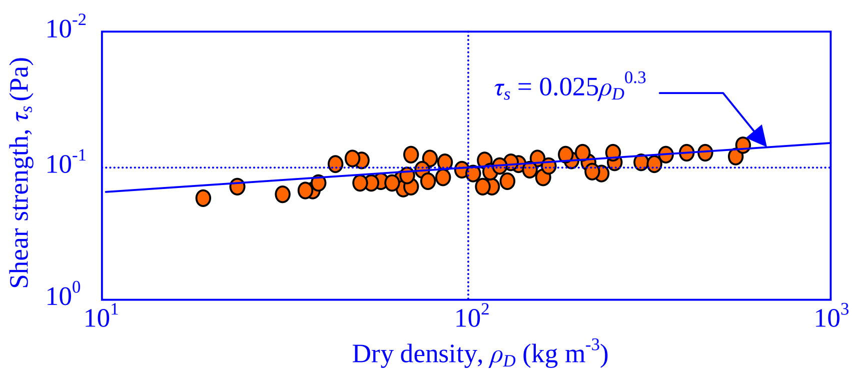


Fig. D.7 Shear strength variation with bed dry density.

### D.3.3 Settling Velocity Function

We will use the following function to describe the dependence of the settling velocity $w_s$ on suspended sediment concentration $c$ (Hwang, 1989):

$$w_s = w_0 \; , \quad c \le c_f \; ; \quad w_s = \frac{a\,c^n}{\left(b^2 + c^2\right)^m} \; , \quad c > c_f \tag{D.4}$$

in which $c_f$, typically ranging between 0.1 and 0.3 kg m$^{-3}$, is the concentration limit at and below which $w_s$ is practically free of the effect of inter-particle collisions and is assumed constant ($w_0$). The quantities $a$, $b$, $n$, and $m$ are sediment-specific coefficients that describe: 1) flocculation settling, in which $w_s$ increases with $c$ above $c_f$ due to aggregation brought about by collisions, and 2) hindered settling, in which $w_s$ decreases with increasing concentration at high values of $c$. Based on data from the Cedar/Ortega River system, the Loxahatchee River and Lake Okeechobee (Fig. D.1), Marván (2001) and Ganju (2001) noted that that the influence of organic content ($OC$) can be conveniently ascribed to the velocity-scaling coefficient $a$ while holding $b$, $n$, and $m$ constant. Accordingly, selecting $b$ = 6.4, $n$ = 1.8, and $m$ = 1.8, Table D.1 gives values of $a$, and the Stokes' diameter, $d_s$, calculated from the corresponding $w_0$. The latter is conveniently chosen as velocity at 0.25 kg m$^{-3}$ concentration.

Table D.1 Velocity-scaling coefficient $a$, free settling velocity $w_0$ and Stokes' diameter $d_s$

| Source in Florida | Mean organic content, $OC$ (%) | Scaling coefficient, $a$ | Free settling velocity, $w_0$ (m s$^{-1}$) | Stokes' diameter, $d_s$ (μm) |
|---|---|---|---|---|
| Low $OC$ mud | 2 | 0.20 | $2.60\times10^{-5}$ | 5 |
| Loxahatchee River | 15 | 0.19 | $1.80\times10^{-5}$ | 5 |
| Cedar/Ortega River | 28 | 0.16 | $1.65\times10^{-5}$ | 5 |
| Lake Okeechobee | 38 | 0.09 | $0.93\times10^{-5}$ | 4 |
| Lake Okeechobee | 40 | 0.08 | $0.78\times10^{-5}$ | 4 |
| Lake Okeechobee | 43 | 0.03 | $0.31\times10^{-5}$ | 3 |

The $a$-values in Table D.1 yield the following dependence on $OC$ (Fig. D.8):

$$a = a_0 + a_1 OC + a_2 OC^2 + a_3 \mathrm{OC}^3 + a_4 OC^4 \qquad \text{(D.5a)}$$

where $a_0 = 0.2$, $a_1 = 6.6\text{x}10^{-4}$, $a_2 = -1.7\text{x}10^{-4}$, $a_3 = 7.1\text{x}10^{-6}$ and $a_4 = -1.3\text{x}10^{-7}$ . Equation (D.4) along with these coefficient values is consistent with the trend of decreasing $d_s$ and $w_0$ with increasing $OC$, as the aggregates become lighter and also smaller because the effect of cohesion decreases. An example of Eq. (D.4) along with data from Table D.1 applied the Loxahatchee River is shown in Fig. D.9.

In Fig. D.10 free settling velocities $w_0$ from the sites in Table D.1 as well as Rodman Reservoir, Cedar/Ortega River system, Kissimmee River and Newnans Lake are plotted against $OC$. The data show the expected mean trend of decreasing $w_0$ with increasing $OC$ as the aggregates become lighter.

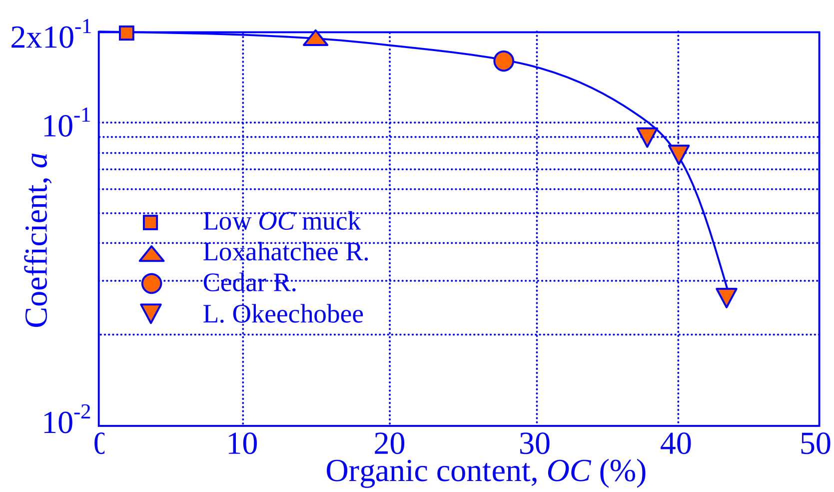


Fig. D.8 Variation of velocity coefficient $a$ with organic content.

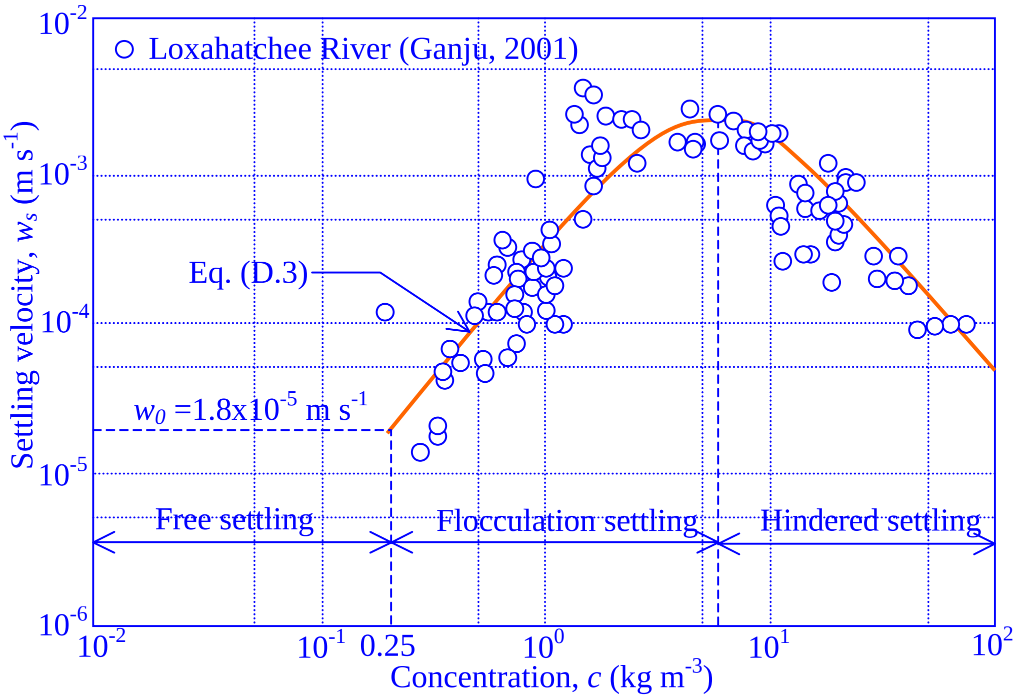


Fig. D.9 Settling velocity variation with concentration for Loxahatchee River sediment.

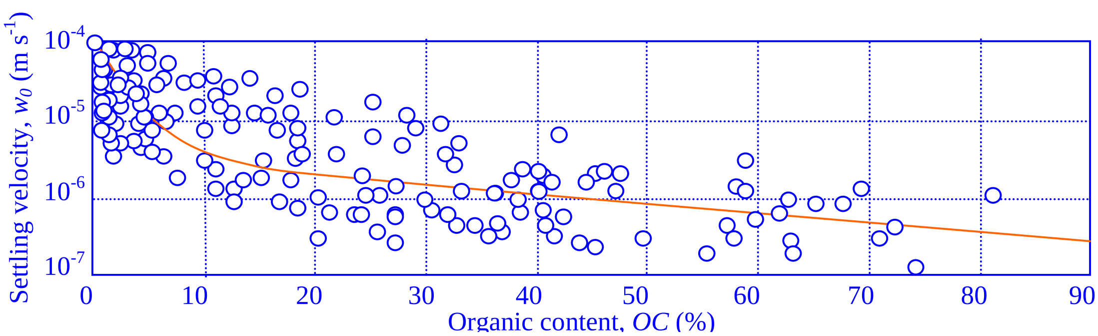


Fig. D.10 variation of free settling velocity with organic content.

## D.4 WIND-WAVE RESUSPENSION

### D.4.1 Bed Sediment Properties

For convenience of treatment we will divide Newnans Lake (Fig. D.11) into three sub-areas based on bathymetry – an inner open water zone, an outer open water zone and the exposed zone. In 2001, forty-five bottom muck samples (40 by push-cores in the open

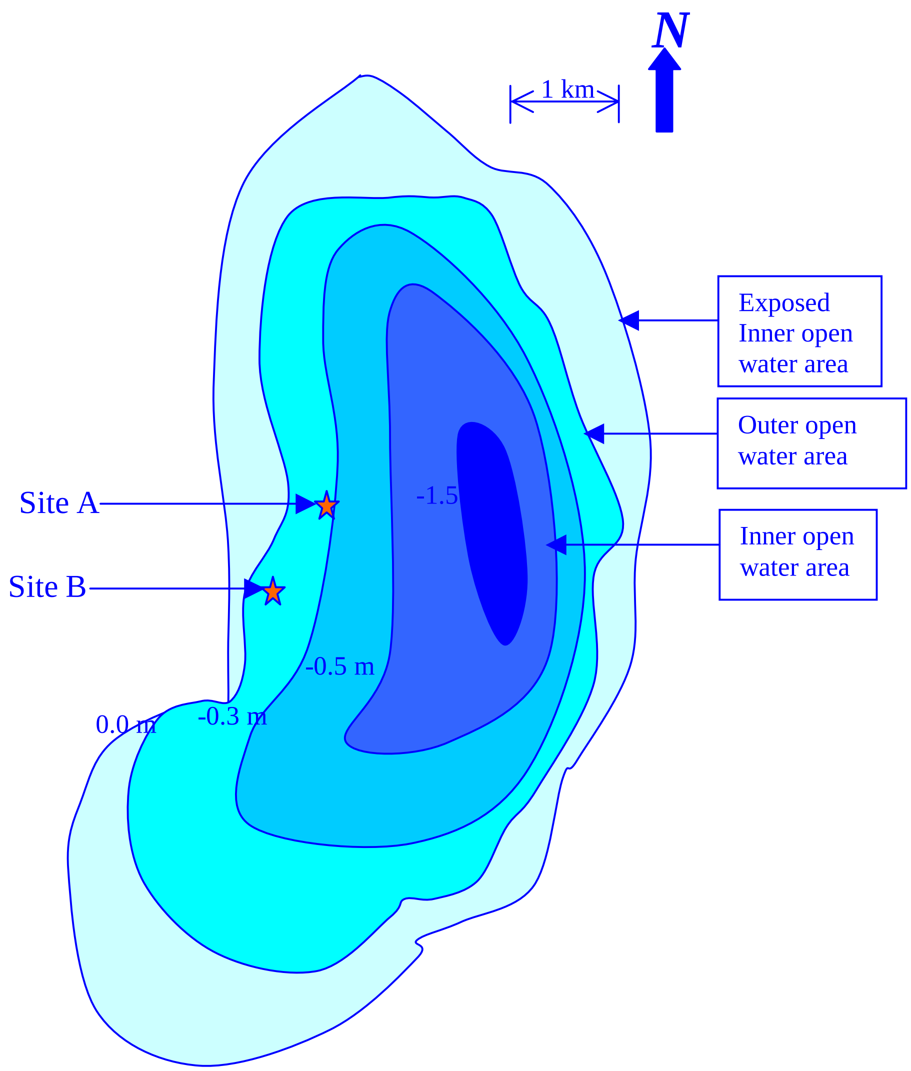


Fig. D.11 Newnans Lake zones based on bathymetry. Suspended sediment concentration was measured at Sites A and B.

water area and 5 by a shovel in the exposed area) were obtained, and short duration (~10 min) wind-wave and suspended sediment data were collected at Sites A and B in this shallow, hypereurophic water body. At that time the water level was considerably lower (by 1.1 m) compared to ~ 20 m for the 1995-1998 period, and almost 2 m below the El Niño high in February 1998 (Fig. D.12), resulting in the large exposed area seen in Fig. D.11.

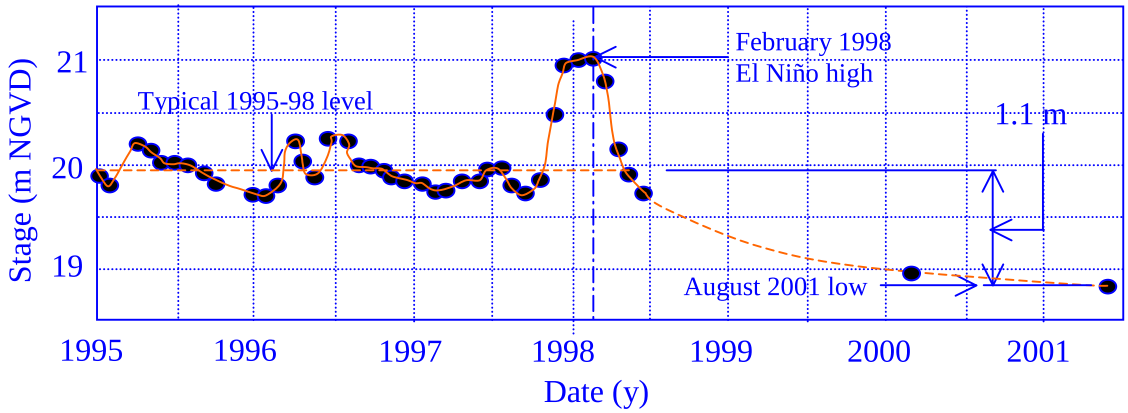


Fig. D.12 Stage variation in Newnans Lake, 1995-2001.

Total core lengths ranged from 1.05 to 1.74 m with a mean of 1.50 m. Accordingly, the submerged bed was divided into a weakly consolidated upper layer of 0.40 m mean thickness and a lower, more consolidated layer of 1.10 m (mean) thickness. The exposed area samples were from the top ~0.30 m. In Table D.2, the means and standard deviations of selected parameters (*OC*, $\rho$, dry bulk density $\rho_D$, $\rho_s$ and median particle diameter $d_{50}$) for these layers in the three zones are given. Note that the dry density is related to the other two densities by the mass balance, $\rho_D = [(\rho - \rho_w)/(\rho_s - \rho_w)]\rho_s$, where water density $\rho_w$ = 1,000 kg m$^{-3}$ is assumed for the lake.

Table D.2 Characteristic properties of upper and lower bed layers by zone

| Bed Layer | Zone | Organic content *OC* (%) | | Bulk density $\rho$ (kg m$^{-3}$) | | Dry density $\rho_D$ (kg m$^{-3}$) | | Granular density $\rho_s$ (kg m$^{-3}$) | | Median diameter $d_{50}$ (µm) | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Mean | S.D. | Mean | S.D. | Mean | S.D. | Mean | S.D. | Mean | S.D. |
| Upper | Exposed | 30 | 11 | 1156 | 63 | 267 | 105 | 2296 | 210 | 262 | 100 |
| Upper | Outer | 45 | 8 | 1050 | 30 | 98 | 56 | 2098 | 411 | 38 | 14 |
| Upper | Inner | 52 | 4 | 1023 | 11 | 51 | 27 | 1986 | 491 | 26 | 7 |
| Lower | Outer | 28 | 11 | 1090 | 28 | 161 | 43 | 2283 | 280 | 174 | 104 |
| Lower | Inner | 43 | 7 | 1047 | 29 | 94 | 36 | 2037 | 531 | 45 | 22 |

The parametric values in Table D.2 indicate that, overall, sediment in the inner open water zone had the highest organic content, the lowest bulk, dry and granular densities and the smallest particle size. At the opposite end was exposed bed, with the outer open water zone in-between. These trends correlate with decreasing *OC* from inner open water to the exposed zone, the latter having been subjected to desiccation and oxidation with a lowering of the lake level over the previous two years. In turn, the particles were large (260 μm) in this zone because of the loss of organic matter as well as the presence of sand. Sand also contributed to the large (190 μm) size in the lower layer of the outer open water zone. Elsewhere the material was fine-grained (< 63 μm).

The dependence of the erosion flux on organic content is evident from Fig. D.13, and conforms to the inverse variation of $\varepsilon_N$ with $\tau_s$ shown in Fig. D.6. Since $\rho_D$ decreases with increasing *OC* according to Eq. (D.2), the data in Fig. D.13 also imply an increase in $\tau_s$ with increasing $\rho_D$, which in turn is found to agree with the trend in Fig. D.7.

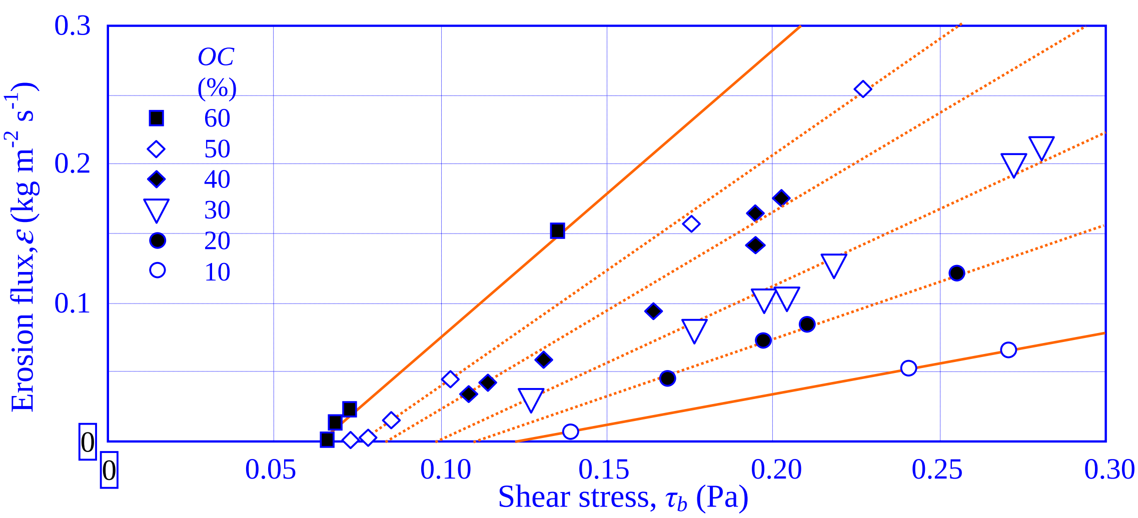


Fig. D.13 Erosion flux versus bed shear stress with lines of constant *OC*; data from Newnans Lake.

In Fig. D.14 settling velocity data obtained in a series of settling column tests are shown in accordance with Eq. (D.4). For plotting the curves it was found convenient to hold the coefficients $a$ (= 0.7), $n$ (= 1.4) and $m$ (= 2.5) constant and vary $b$, in contrast to the use of Eq. (D.5) based on an analysis (Ganju, 2002), in which $b$, $n$ and $m$ remain unchanged. The experimental points in Fig. D.14 are included after a careful reinterpretation of the original data (Gowland, 2002b). A consequence of data scatter in the plot is that the free settling velocity values given in Table D.3 exhibit large standard deviations compared to means.

In addition to the means and standard deviations of the free settling velocity characterizing the usually low concentration environment of the lake, erosion parameters are also given in Table D.3. In general, trends in the mean values imply that the organic-rich material in the inner open water area was potentially more erodible than in

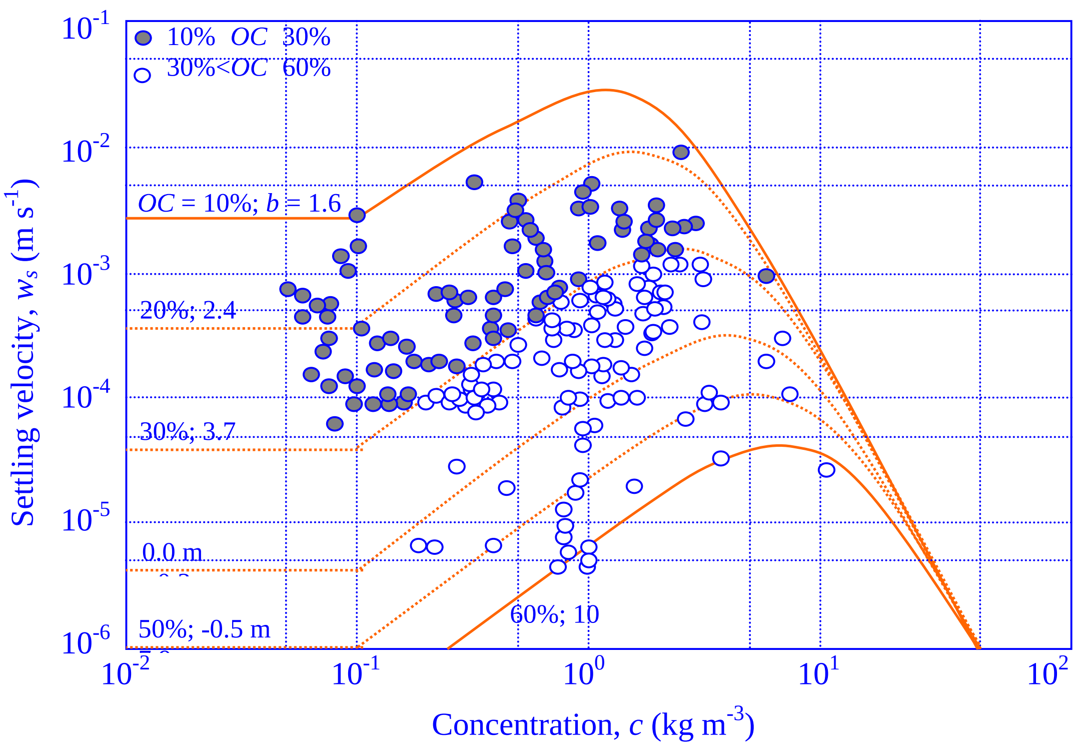


Fig. D.14 Settling velocity variation with concentration for Newnans Lake sediment.

Table D.3 Mean erosion parameters and median settling velocity for upper and lower bed layers by zone

| Bed layer | Zone | Organic content $OC$ (%) | | Erosion flux constant $\varepsilon_N$ (g $N^{-1}$ $s^{-1}$) | | Bed shear strength $\tau_s$ (Pa) | | Free settling velocity $w_0$ (m $s^{-1}$) | |
|---|---|---|---|---|---|---|---|---|---|
| | | Mean | S.D. | Mean | S.D. | Mean | S.D. | Mean | S.D. |
| Upper | Exposed | 30 | 11 | 0.87 | 0.35 | 0.08 | 0.013 | $2.3 \times 10^{-4}$ | $2.5 \times 10^{-4}$ |
| Upper | Outer | 45 | 8 | 1.61 | 0.25 | 0.08 | 0.009 | $2.4 \times 10^{-4}$ | $1.3 \times 10^{-4}$ |
| Upper | Inner | 52 | 4 | 1.82 | 0.13 | 0.07 | 0.005 | $1.3 \times 10^{-4}$ | $0.7 \times 10^{-4}$ |
| Lower | Outer | 28 | 11 | 0.99 | 0.38 | 0.10 | 0.014 | $5.8 \times 10^{-4}$ | $2.0 \times 10^{-4}$ |
| Lower | Inner | 43 | 7 | 1.53 | 0.23 | 0.08 | 0.008 | $2.9 \times 10^{-4}$ | $1.2 \times 10^{-4}$ |

outer open water. It must be noted however that in the deeper (than outer area) inner bottom the potential for erosion of the weaker material would be tempered by lower wave-induced bed shear stresses.

To a certain extent the settling velocities in Table D.3 show an inverse correlation with $OC$ due to the associated decrease in the density of the aggregates. For example, increasing $OC$ from 28 to 52% decreases $w_0$ from $5.8 \times 10^{-4}$ to $1.3 \times 10^{-4}$ m $s^{-1}$. Since the suspended material in the settling velocity tests did not contain sand particles, it did not correlate consistently with what was present in the bed. For example, in the exposed area, even though settling tests yielded a value of $w_0 = 2.7 \times 10^{-4}$ m $s^{-1}$ suggesting a high $OC$ (30%), grain size analysis of the bed sample yielded $d_{50} = 262$ μm.

### D.4.2 Concentration Profiles

We followed the technique presented by Li and Parchure (1998) to simulate the vertical profile of total suspended sediment solids concentration $c$ due to wind-wave action in the lake. For that purpose, the governing sediment mass balance is simply stated as

$$\frac{\partial c}{\partial t}-\frac{\partial}{\partial z}\left(K\frac{\partial c}{\partial z}+w_s c\right)=0 \qquad \text{(D.6)}$$

in which $t$ is time, $z$ is the vertical elevation coordinate with origin at the bed and $K$ is the sediment mass diffusion coefficient. According to Eq. (D.6) the temporal rate of change of concentration, $\partial c/\partial t$, is determined by the net vertical sediment flux, which is the algebraic sum of the diffusive flux, $K\partial c/\partial z$, and the settling flux, $w_s c$. The diffusion coefficient, which in general must accounting for the effect of the buoyancy of the sediment suspension, was obtained from

$$K=\frac{K_n}{(1+\alpha Ri)^{\beta}} \qquad \text{(D.7)}$$

where $K_n$ is the (neutral) diffusion coefficient for non-stratified flows, $\alpha$ (= 0.5) and $\beta$ (= 0.33) are sediment-specific coefficients and $Ri$ is the gradient Richardson number defined as (Li and Parchure, 1998)

$$Ri=-\left(\frac{g}{\rho_f}\right)\frac{\partial\rho_f/\partial z}{\left(\partial u/\partial z\right)^2} \qquad \text{(D.8)}$$

In Eq. D.8, $u$ is the horizontal (flow) velocity, $\rho_f=[1-(\rho_w/\rho_s)]+\rho_w$ is the fluid density and $g$ is gravitational acceleration. The neutral diffusivity was obtained from

$$K_n=\alpha_w H^2\sigma\frac{\sinh^2 k(z-h)}{2\sinh^2 kh} \qquad \text{(D.9)}$$

where $\alpha_w$ (=0.35) is a site-dependent coefficient, $H$ is the wave height, $\sigma$ = $2\pi/T$ is the wave angular frequency, $T$ is the wave period, $k$ = $2\pi/L$ is the wave number and $L$ is the wave length. For calculating $Ri$ the velocity $u$ as a function of $H$, $T$ and water depth $h$ (hence $L$) was calculated from a wave theory incorporating the bottom boundary layer (Li and Parchure, 1998). For calculating the erosion flux [Eq. (D.3)] the bed shear stress was estimated from: $\tau_b = 0.5 f_w \rho_w u_b^2$, where $f_w$ (= 0.026 for this lake; Gowland, 2002b) is the wave friction factor and $u_b$ is the wave velocity amplitude at the bottom. At the bed the net sediment flux was calculated from Eqs. (D.3) and (D.4).

Inasmuch as Sites A and B (Fig. D.11) were close to each other, with the exception of water depth the same set of parametric values was selected for simulation purposes (Table D.4). Note that these site-specific values differ in some cases (e.g., free settling velocity) from the means given for those in the upper outer open water zone in Tables D.2 and D.3.

Table D.4 Input parameters for concentration profile simulation

| Parameter | Value |
|---|---|
| Water depth $h$ (m) | 0.25[a], 0.30[b] |
| Wave height $H$ (m) | 0.05 |
| Wave period $T$ (s) | 1.0 |
| Background $c$ (kg m$^{-3}$) | 0.04[c] |
| Bed granular density $\rho_s$ (kg m$^{-3}$) | 2059 |
| Bed dry density $\rho_D$ (kg m$^{-3}$) | 96 |
| Free settling velocity $w_0$ (m s$^{-1}$) | 0.1x10$^{-5}$ |
| Velocity coefficients $a$, $b$, $n$, $m$ | 0.7, 7.9, 2.5, 1.4 |
| $\varepsilon_N$ (g N$^{-1}$ s$^{-1}$) | 1.82 |
| $\tau_s$ (Pa) | 0.07 |

[a] At Site A; [b] At Site B; [c] Present as washload.

For the numerical solution of Eq. (D.6) the water column was discretized into 30 sublayers and the time step for simulation was 60 s. For Site A the resulting increase in concentration with continued wave action is shown in Fig. D.15. The initially uniform concentration of 0.04 kg m$^{-3}$ (not plotted) evolves into a bottom-weighted profile at 90 min, which seemingly agrees with measurements of concentration under a wave height of 0.05 m and a wave period of 1 s (Table D.4) at this site. Since the “initial time” for wave action in the lake was not measured (as a water level time-series), the comparison between

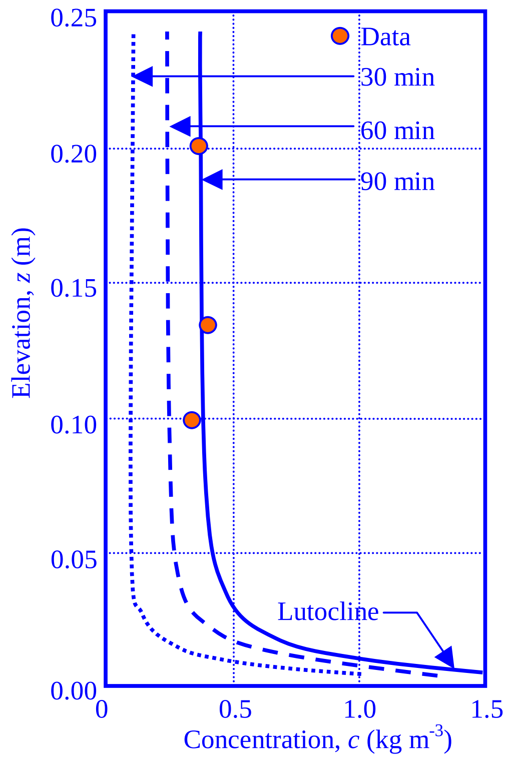


Fig. D.15 Concentration profiles at Sites A in Newnans Lake.

prediction and measurement is on a qualitative basis only. Nonetheless we note that the measured and simulated values of $c$ were both close to 0.4 kg m$^{-3}$, which is high in comparison with some other lakes (Hwang, 1989), and is an indicator of the easily erodible material at the bottom of this very shallow lake.

An interesting feature of the simulated profiles, not captured by the measurements because they were not made sufficiently close enough to the bed, is the development of a lutocline within the first few centimeters from the bottom. Concentrations in this layer well exceed 1 kg m$^{-3}$, and lead to the formation of fluid mud beneath the lutocline (Li and Parchure, 1998).

Without changing any of the constants in Table D.4 used in simulating the profiles of Fig. D.14, the profiles of Fig. D.15 were generating merely by increasing the water depth from 0.25 m at Site A to 0.30 m at Site B. In this case we find that under the same wave conditions resuspension was slower due to the greater water depth. Consequently, the concentration rose to only ~0.2 kg m$^{-3}$ in 120 min from the start of simulation. The measured values are commensurate with this value.

### D.4.3 Critical Wind Speed

As a corollary to the above simulation exercise we will now examine the correlation between wind speed and resuspension. Typical hourly-mean westerly surface winds at this lake are only on the order of 3 m s$^{-1}$. However, 2-min maximum values range between 10 and 20 m s$^{-1}$ over an effective fetch length of ~3,000 m during the December-August period. A cursory examination of the relationship between wind speed measured at a nearby site and wave height (measured by a graduated staff) in the lake showed that wave height prediction could be made using the empirical formulation of Young and Verhagen (1996) with only minor modifications (Gowland, 2000b).

In Fig. D.16 the mid-depth concentration $c_m$ is plotted against time for the inner open water zone (Tables D.2 and D.3) and assumed steady (or quasi-steady) wind speeds $U$ = 7, 8 and 9 m s$^{-1}$ (see Fig. D.17). At 7 m s$^{-1}$ no erosion took place. At 8 m s$^{-1}$ erosion did occur and as the time neared 400 min the rate of erosion began to decrease due to a corresponding increase in the rate of deposition of the suspended matter. However, an equilibrium condition marked by no-net increase in concentration was not attained in this simulation for 2 additional hours (Gowland, 2002b). An inference one may draw from this observation is that inasmuch as wind almost never remains steady for durations of such lengths (~10 h) in this region, episodically driven resuspension in the lake is characteristically in a perpetual state of disequilibrium, except when and if there is a prolonged period of calm.

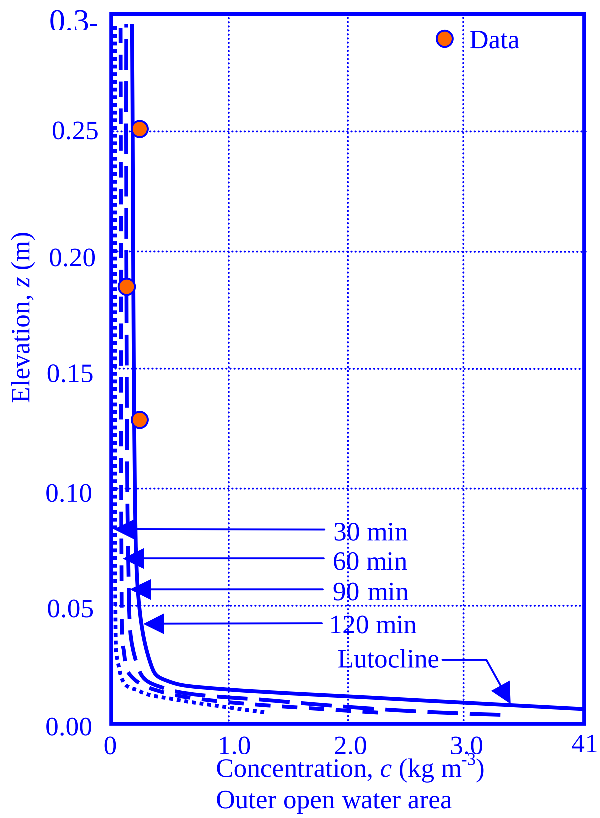


Fig. D.16 Concentration profiles at Sites Bin Newnans Lake.

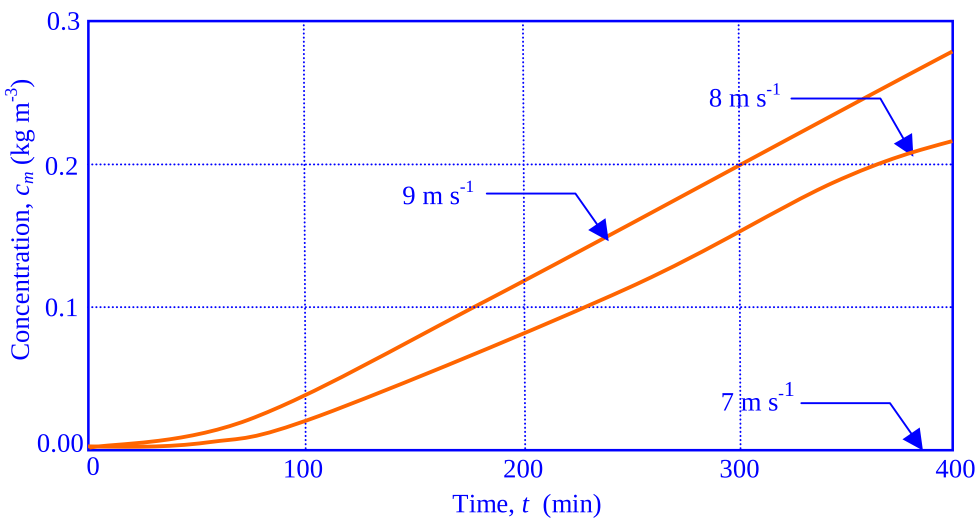


Fig. D.17 Wind-induced resuspension simulation in Newnans Lake.

In the above case the critical wind speed for initiation of resuspension, $U_c$, was found to be 7.2 m $s^{-1}$ in the outer zone and 7.5 m $s^{-1}$ in the inner zone. In that regard, it is interesting to examine the effect of changing the organic content of bed sediment on $U_c$. Let us consider a case wherein *OC* in the peripheral part of the outer open water zone is reduced from 45% to 30%, the same as in the exposed zone, during a severe but temporary drought. Using the same analytic approach and relationships dependent on *OC* we find that $U_c$ would increase from 7.2 to 7.8 m $s^{-1}$, implying greater bed stability.

Next let us look at a somewhat different case in which the bottom is dredged out from the inner open water zone. As a starting point it is necessary to examine the state of consolidation of the lake bed. Although the lake has been accumulating muck since its inception as a lake-like body about eight millennia ago (Holly, 1976), a substantial degree of additional accumulation occurred when in the 1970s a program of herbicide spraying was carried out to eliminate floating plants, and the dead material was allowed to settle out. For our purposes we will assume that this process was rapid enough (in relation to the time-scale of consolidation) for it to be treated as self-weight consolidation of “instantaneously formed” deposit at the bottom.

In Fig. D.18 we have shown results obtained from the self-weight consolidation equation of Been and Sills (1981) calibrated with sediment-specific parameters obtained from laboratory tests on the consolidation of the lake muck (Gowland, 2002b). The starting uniform density of 1,039 kg $m^{-3}$ represents a fresh deposit having an assumed thickness of 1.3 m. After 40 years of simulation the bed attained a density close to what was measured (~1,061 kg $m^{-3}$) by Brenner and Whitmore (1998). This period is in acceptable agreement with the over three-decade period between the initial accumulation

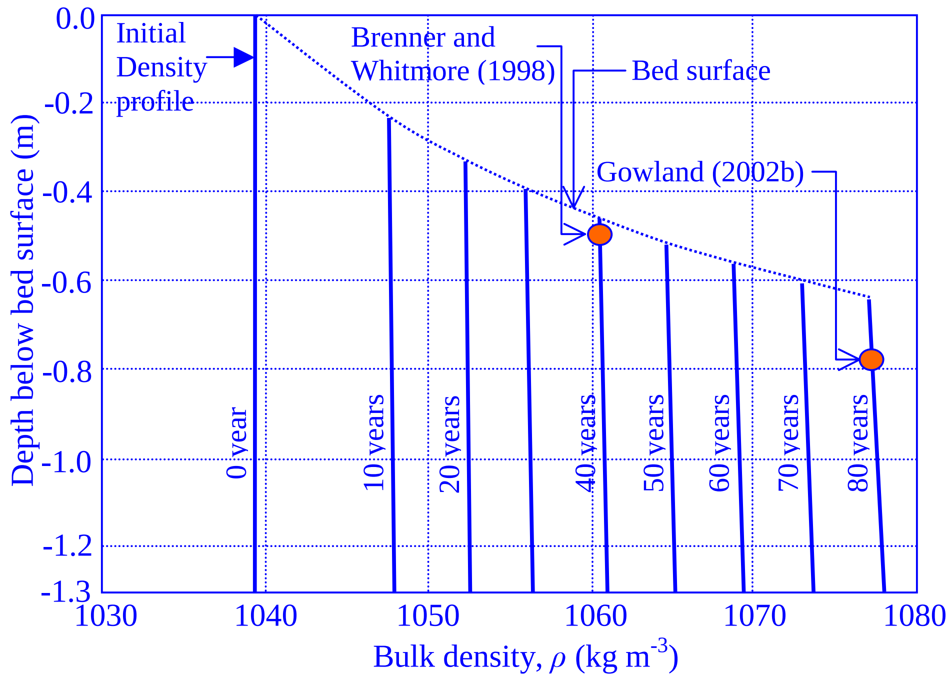


Fig. D.18 Bed density variation due to consolidation and data from Newnans Lake.

of material and density measurement. The density reported by Gowland (2002b), 1,078 kg $m^{-3}$ was higher, and according to the model would take 80 years of self-weight consolidation. This seeming discrepancy may be due to the limitation arising from the assumption that the bed everywhere in the inner open water area was subject to self-weight consolidation only. In actuality the effect of overburden in further densifying the bed may not be negligible.

Now we will consider the top 0.3 m of the bottom in the inner open water zone to be dredged out, and the properties of the newly exposed bed there to be described by the denser lower layer having a lower *OC* (Tables D.2 and D.3). It was found that in this case the critical wind speed for resuspension would increase from 7.5 to 10 m $s^{-1}$ (Table D.5). About one-half of this increase would be due to reduced wave penetration in deeper water, and the other due to a reduction in the organic content (from 52 to 43%) and concomitant increase in bed density.

Table D.5 Critical wind speeds for resuspension

| Condition (%) | Mean water Depth, $h$ (m) | Critical wind speed, $U_c$ ($m\ s^{-1}$) |
|---|---|---|
| Outer open water zone, $OC$ = 45% | 1.25 | 7.2 |
| Inner open water zone, $OC$ = 52% | 1.50 | 7.5 |
| Outer open water zone, $OC$ = 30% | 1.25 | 7.8 |
| Dredged inner zone, $OC$ = 43% | 1.80 | 10.0 |

## D.5 CONCLUDING COMMENTS

This examination of the effect of dredging, e.g., in the MFLs context, has evident relevance to areas where water quality degradation by frequent and significant resuspension of muck is a problem. The described approach provides a simple but useful tool for analyzing the basic physics linking wind to muck resuspension.

In the dredging problem we used measured values of the requisite parameters from Tables D.2 and D.3. In that regard we must note a constraint inherent in using the developed general relationships between bed density measures and organic content. This constraint arises because while density increase with depth is due to consolidation, reduction in organic content with depth is a biogenic process that is not uniquely correlated to densification by consolidation. We therefore recommend that the use of density-versus-organic content relationships be limited to naturally consolidated, top ~0.1-0.3 m thick muck layers that are not overly aged with respect to organic content.

# APPENDIX E: SEDIMENT MONITORING

## E.1 SUMMARY

For monitoring TSS, an ADCP along with software to process the backscatter signals for TSS concentration are recommended. As an alternative to the tedious method of bottom coring, it appears that rapid mapping of bottom muck can be carried out using continuous seismic profiling (CSP) techniques. These systems have not been tested in the study area however.

## E.2 MONITORING

A self-recording device for long, intermediate or short-term monitoring can be supplied. We suggest that in the initial period, perhaps as long as 10 or more years, “continuous” monitoring is required in order that the complete envelope of flow situations including prolonged droughts, high energy events and long intervening periods with slow change and steady flow are established. This is very poorly known at present.

## E.3 ACOUSTIC PROFILING

An RDI Acoustic Doppler Current Profiler (ADCP) along with the Sediview software (Land et al., 1998) mounted on a float, which is free to follow changes in water surface elevation and record level is recommended. It might be mounted on an existing light beacon or otherwise self-contained and battery powered. This type of flow monitoring device is able to measure the extremely low velocities typical of the vast proportion of the flow regime in Florida. It has been observed that such extremely low flows are characterized by highly variable directions, further emphasizing the need for an ADCP. At these low velocities one is more or less monitoring the large scale internal turbulence of the flow.

An ADCP can readily and completely measure the extremely low suspended sediment concentrations found for much of the time. Background concentrations in the lower Cedar/Ortega ranged between 1 and $15 mgl^{-1}$ and were confidently calibrated. Finding one part in a million parts is a most severe calibration challenge. The RDI ADCP can be used (with its attendant Sediview software) to determine solids concentration. During high energy events, possibly during plankton blooms etc., suspended solids concentrations will be much higher. The scale of difference in the upstream and downstream concentrations is expected to be minor/non-existent, but RDI ADCP (with Sediview) can determine this issue.

An ADCP has an advantage over conventional impeller current meters and over optical transmissance or backscatter meters that it is more or less immune from fouling effects from biofilms, epibenthos etc. Similarly, its results and operational integrity can be obtained remotely from telemetered data.

We recommend the following:

1). RDI ADCP with 20° beam angle - in fact 30° instruments are only made to special order these days.

2). RDI ADCP needs to be 1200 kHz in order to achieve the necessary accuracy and resolution in shallow water.

The application requires the RDI Inc.'s Workhorse Sentinel or Rio Grande ZedHed device specially developed for extremely shallow flows. This permits very narrow (in the vertical plane, i.e., 2 cm) "bins" of data to be obtained. It should be noted that RDI produces a 4 transducer Broadband ADCP array. It is protected by a patent.

Workhorse Sentinel or Rio Grande ZedHed ADCPs are becoming the industry standard. It is worth pointing out that the suspended sediment monitoring component has been developed by Dredging Research Ltd. (UK). Although it is widely and relatively uncritically used in the industry, DRL points out that a ZedHed works entirely in what is called the acoustic nearfield where initial spreading of the acoustic beam has not heretofore been well understood. A breakthrough has been achieved with a new algorithm to predict beam spreading in the nearfield, but this is still an area of active development.

It is likely that in this application the ADCP would be operated in what is called Mode 11 or 12. A detailed design will be worked up at a future stage.

**E.4 BOTTOM MUCK MAPPING**

A further requirement of any subsequent and practical stage is to undertake mapping surveys of distribution of black organic mud in lakes and critical reaches of the St. Johns River. Traditional point sampling surveys of these two types of system tend to be time consuming and require assumptions concerning continuity of bed material between adjacent sample localities that are often long distances apart. From time to time these assumptions of continuity break down. Any such breakdowns would be especially undesirable in circumstances where a significant engineering installation, such as an off-take station, is involved.

These days more rapid and all-embracing survey techniques are available and we recommend applying one or other of these. Modern high-resolution continuous seismic profiling (CSP) techniques using acoustic devices towed astern of a survey vessel near the water surface on a grid pattern can map the nature and depth of muddy deposits. We may resort to this type of survey approach, accompanied by a few grab or core samples used to quantify the signal returns. CSP techniques are, however, constrained by extremely shallow waters and by the presence of gas in the sediment, both of which are likely to occur in these areas. We have recent direct experience of these issues in the course of CSP surveys in Lake Okeechobee.

An alternative rapid method, which has a similarly high data capture rate, and also involves a device towed behind a boat on a grid pattern, in this case a scintillation counter. Such a transducer can be towed along in contact with the bottom. It is able to determine the atomic number of elements making up the chemical composition of sediment and so establish from spatial variations in the pattern of these how they are distributed on the bottom. It, like all methods, has certain limitations. In this case the intensity of the returns is determined by the geometry of the instrument in relation to the bed. This is unlikely to be a constant as the counter is dragged across a sandy surface but may sink completely and be surrounded by soft mud deposits. Furthermore, it only measures the materials it is directly in contact with and does not provide any indication of their depth. For this reason, again, ancillary and occasional coring or probing with a metal rod along the survey line will give an indication of the sediment depth. A modern version of this instrument is the MEDUSA (Van Wijngaarden, 2002).

# APPENDIX F: LONG-TERM IMPACT OF DISCHARGE REDUCTION ON RIVER MORPHOLOGY

## F.1 SUMMARY

Here we present an example of a simple morphodynamic method to determine the long-term impact of off-take on river morphology. Starting with the Einstein-Brown equation for sediment transport, we show that the well-known regime concept relating river discharge *Q* to cross-sectional area *A* is compatible with sediment transport under live-bed equilibrium. As a result, a physical meaning can be provided to the empirical regime coefficients. Thus we find that for a given river the *Q*-*A* relationship is directly dependent on the sediment load. It follows that changing this load, e.g., due to off-take, will change the *Q*-*A* relationship. This is discussed in relation to a hypothetical off-take scenario in the Middle St. Johns River.

## F.2 INTRODUCTION

Recent interest in establishing criteria for minimum flow and levels (MFLs) in the managed water districts of Florida have prompted the need to examine the potential for the long-term impact of off-take from a lake, river or an estuary (Neubauer et al., 2003). If off-take from a river or an estuarine channel also causes a measurable reduction in sediment load, its impact will be manifested as a reduction in flow velocity and increase in the rate of sediment deposition.

There are basically two ways by which the degree of impact can be assessed, one by full-fledged modeling of flow and sediment transport in the channel, and the second by a simpler approach taking advantage of our understanding of channel morphodynamics. The latter is a far more approximate approach than the former, but can yield order of magnitude values related to impact that can be used to decide whether a more detailed analysis, e.g., by full modeling, is needed. Here the morphodynamic approach, which relies on the relationship between flow discharge (hence velocity) and flow cross-sectional area under live-bed sedimentary equilibrium, is explored. Specifically, the following tasks are carried out: 1) The equilibrium relationship between discharge and flow area is reviewed for steady and periodic (tidal) flows in (open) channels, and 2) the impact of flow (coupled with sediment) reduction on channel morphology is examined via illustrative examples.

## F.3 LIVE-BED EQULIBRIUM

Under a constant flow velocity $u$, if the bottom shear stress $\tau$ is greater than the critical shear stress for erosion $\tau_c$, live-bed equilibrium (as opposed to clear water equilibrium below $\tau_c$ critical shear stress) the number of particles that erodes (by saltation) from the bed per unit bed area per unit time is equal to the number of particles that deposits from suspension per unit bed area per unit time [Fig. F.1(a)]. The sediment unit load $q_s$ (volume per unit time per unit channel width) is determined by $u$, hence $\tau$. If u decreases, e.g., due to water withdrawal, then will $q_s$ decrease [Fig. F.1(b)]. A well-known expression relating $q_s$ to $\tau$ is the Einstein-Brown approximation of the Einstein bed load equation

$$\varphi=40\left(\frac{1}{\psi}\right)^3 \tag{F.1}$$

In Eq. (F.1) $\varphi$, the bed load function, and $\psi$, the flow intensity, are defined as

$$\varphi=\frac{q_s}{w_s d} \tag{F.2}$$

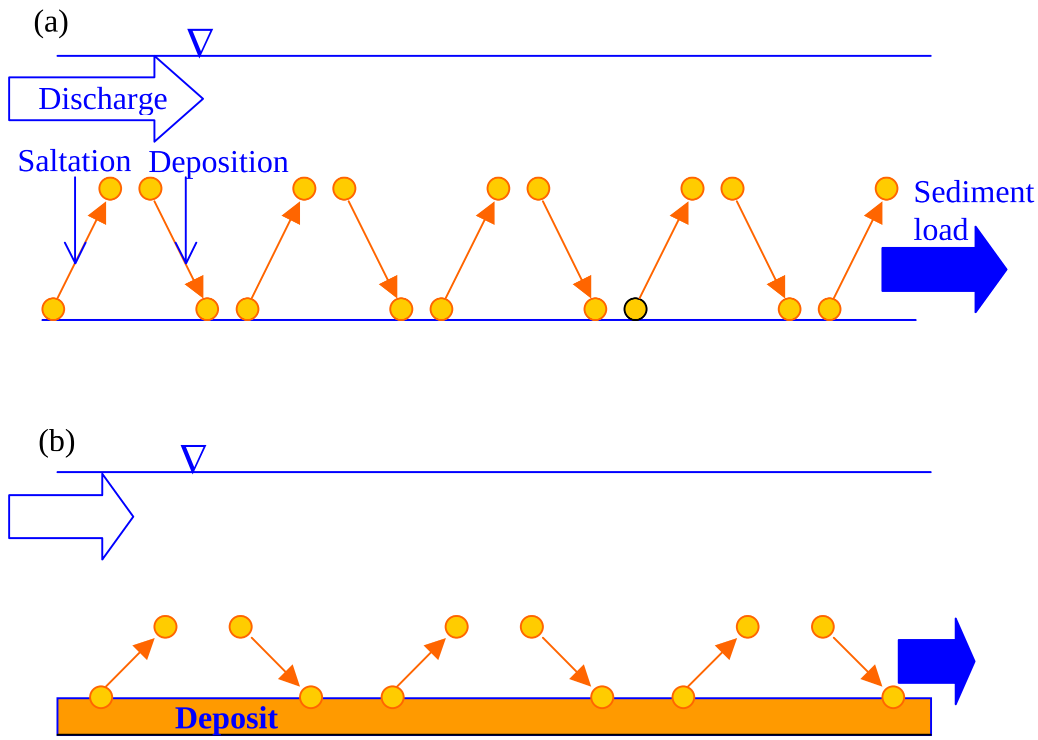


Figure F.1. Live-bed equilibrium in a channel: (a) Given discharge, and (b) deposition under reduced discharge.

$$\frac{1}{\psi} = \frac{\tau}{\gamma(s-1)d} = \frac{\rho u_*^2}{\gamma(s-1)d} \tag{F.3}$$

$$u_* = u\left(\frac{f}{8}\right)^{1/2} \tag{F.4}$$

where $w_s$ is the settling velocity of a particle of diameter $d$, $\gamma$ is the unit weight of water, $s$ is the grain specific weight, $u_*$ is the friction velocity and $f$ is the Darcy-Weisbach friction factor. We will conveniently consider $q_s$ to represent the total load, i.e., sum of be load and suspended load. Therefore,

$$\frac{q_s}{w_s} = \frac{0.078 f^3}{g^3 (s-1)^3 d^2} u^6 \tag{F.5}$$

where $g$ is the acceleration due to gravity. Since $u = Q/A$, where $Q$ is the discharge through flow area $A$,

$$q_s = \frac{0.078 f^3 w_s}{g^3 (s-1)^3 d^2} \left(\frac{Q}{A}\right)^6 = K_1 \left(\frac{Q}{A}\right)^6 \tag{F.6}$$

in which $K_1$ is treated as a (dimensional) constant, whose value depends on the relationship between $Q$ and $A$ specific to the river. In general, the empirical "regime" equation of Blench (1961) is

$$A = K_2 Q^m \tag{F.7}$$

in which $Q$ is now the (annual) average discharge, and $K_2$ and $m$ are river-specific coefficients. Data indicate the value of $m$ to be close to (but less than) unity (Blench, 1961). Combining Eqs. (F.6) and (F.7) we obtain the sediment rating relationship

$$q_s = K_3 Q^{6(1-m)} \tag{F.8}$$

where $K_3 = K_1 / K_2^6$ is another constant. Thus we note that $m$ cannot exceed the value 1, as it would mean a decrease in $q_s$ with an increase in $Q$. If $m = 1$, $q_s$ would become independent of $Q$, i.e., $q_s$ would be wash load rather than bed material load. Since however in general $1-m < 1$, $q_s$ is a gradually varying function of $Q$. Taking $m = 1$,

$$q_s = K_3 \tag{F.9}$$

which is statement of a river in live-bed equilibrium, in which the flow area $A$ bears a constant proportion ($K_2$) to mean discharge $Q$, and, under this morphodynamic condition, the sediment load $q_s$ is constant ($K_3$).

From Eq. (F.8) we obtain

$$\frac{dq_s}{q_s} = 6(1-m)\frac{dQ}{Q} \tag{F.10}$$

Or, in terms of finite differences

$$\frac{\Delta q_s}{q_s} = 6(1-m)\frac{\Delta Q}{Q} \tag{F.11}$$

which means that a reduction in discharge $Q$ by $\Delta Q$, will reduce $q_s$ by $\Delta q_s$. Three cases are of interest:

(1) Water is withdrawn at the rate $\Delta Q$ without any sediment along with it. In this case, sediment will deposit downstream of the withdrawal point at the rate $B\Delta q_s$, where $B$ is the local width of the river. Downstream discharge will be $Q-\Delta Q$, and suspended load (per unit river width) $q_s - \Delta q_s$.

(2) Water is withdrawn at the rate $\Delta Q$ along with sediment at the rate $\Delta q_s$ (per unit width of river). In this case no sediment deposition will occur downstream.

(3) Water is withdrawn at the rate $\Delta Q$ along with sediment at the rate $q_s$ (per unit width of river). In this case scour will occur downstream.

A limitation of Eq. (F.11) is that it provides no indication of after what length of the effect of withdrawal will be felt, since the change (decrease) in flow area $A$ with $Q$ per Eq. (F.7) is valid only for equilibrium states. In reality, the time required to reestablish morphodynamic equilibrium depends on sediment supply, and can vary from years in high sediment-load rivers to hundreds of years where sediment supply is low.

Let us consider regime data from the Middle St. John's River channels (Table F.1). Observe that in this case $m = 1$ is a reasonable assumption (Fig. F.2), which would imply no change in the sediment load, because with a decrease in discharge the flow area would contract in exact proportion to this change, so that $q_s$ would remain unaltered. Schematically, one can portray the change as shown in Fig. F.3. As the upstream area $A_u = h_u B_u$ decreases to $A_d = h_d B_d$, the load decreases from $Q_{su} = q_s B_u$ to $Q_{sd} = q_s B_d$.

Table F.1 Regime data from the Middle St. Johns River channels

| Station ID | $A$ (m$^2$) | $Q$ (m$^3$/s) |
|---|---|---|
| A – 2232400 | 170 | 29.9 |
| B – 2232500 | 201 | 36.4 |
| C – 2233500 | 35 | 9.2 |
| D – 2234000 | 505 | 53.0 |
| E – 2234500 | 467 | 62.9 |
| F – 2235000 | 66 | 8.5 |
| G – 2235200 | 12 | 2.4 |
| H – 2236000 | 585 | 89.8 |

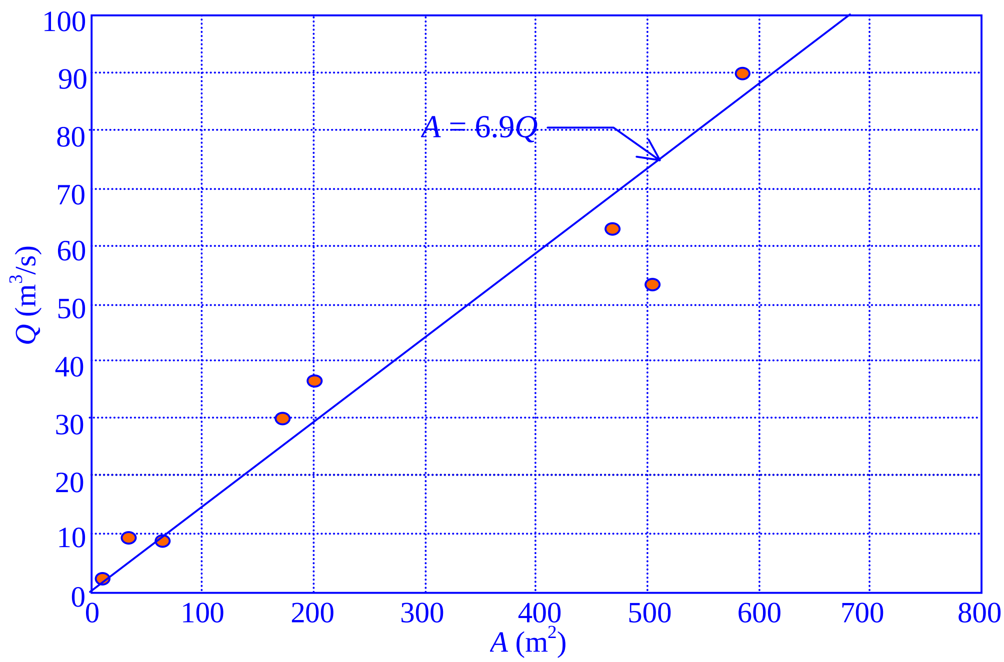


Figure F.2 Mean discharge versus flow area relationship for the Middle St. Johns River.

## F.4 LOAD CALCULATION

Let us estimate $q_s$ from Eq. (F.9). Let $d$ = 0.4 mm (= 4x10$^{-4}$ m) for which $w_s$ (quartz sand) = 0.05 m/s and $s$ = 2.65, and $f$ = 0.028 (a typical value). This yields [Eq. (F.6)] $K_1$ = 1.26x10$^{-4}$. Now from Fig. F.2 $K_2$ = 6.9. So $q_s$ = $K_3 = K_1 / K_2{}^6$ = 1.17x10$^{-9}$ m$^2$/s. Selecting a channel width of 75 m, the annual sediment transport will be 2.76 m$^3$/y.

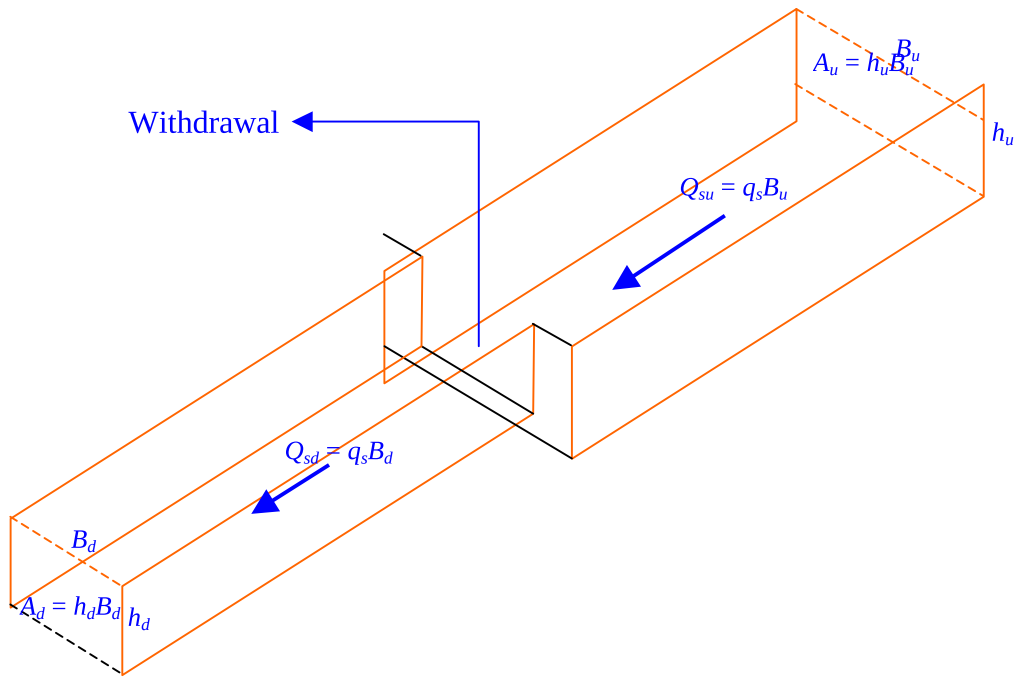


Figure F.3 Flow area and total load reduction due to water withdrawal.

## F.5 FINE SEDIMENT

The above analysis is valid for substrates dominated by coarse sediment, e.g., sand. Referring to Fig. F.4 we note that the occurrence of live-bed equilibrium for sand is determined by two conditions: one in which the instantaneous hydrodynamic lift $L$ (under turbulent flow) is (equal to or) greater than the buoyant weight of the particle, $W$, i.e., $L$  $W$, which causes entrainment of bed sediment, and the other given by $L<W$, which causes the suspended particle to deposit. In contrast, the entrainment and deposition of fine particles or flocs is somewhat different due to the property of cohesion. Let us represent cohesion simply by a representative force $F$, which binds the cohesive particle to the bed. Accordingly, the condition for incipient entrainment will now be $L/(W + F)$  1, where $W$ is the buoyant weight of the particle. On the other hand, the condition for deposition of a suspended particle will remain $L/W<1$.

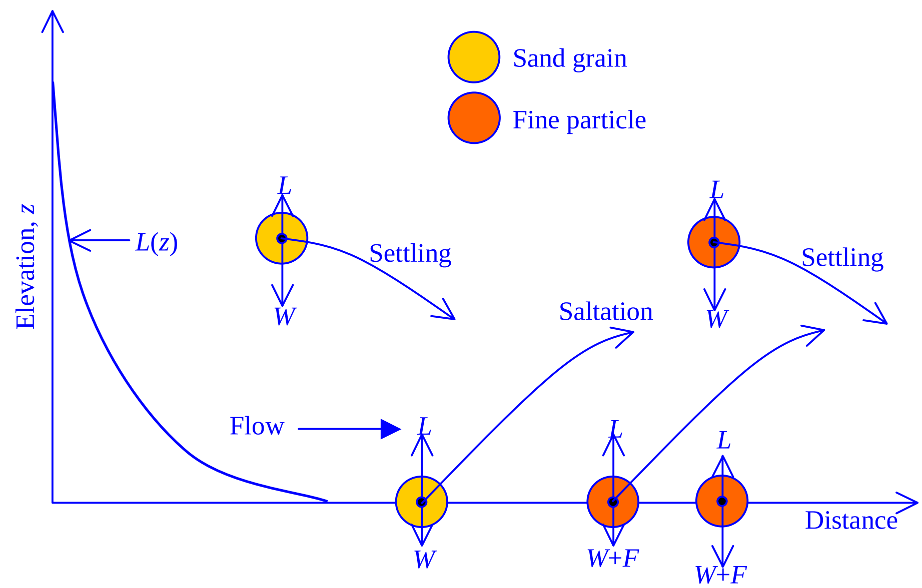


Figure F.4 Condition for incipient motion for sand grains and fine particles.

The above difference leads to the cases shown in Fig. F.5. If the instantaneous (turbulent) bed shear stress $\tau$ (which is proportional to lift $L$) is (equal to) or greater than the critical shear stress for erosion, $\tau_c$, a sand particle on the bed (surface) will erode, and if $\tau$ is less than $\tau_c$ the a suspended particle will deposit. In other words, statistically speaking erosion and deposition will occur simultaneously. In the case of fine cohesive sediment, due to cohesive bonding of bed particles the critical shear stress for erosion, $\tau_{ce}$ is higher than the critical shear stress for deposition $\tau_{cd}$. As a result, within the shear stress range $\tau_{cd}$ $\tau < \tau_{ce}$, there can be no erosion or deposition. Consequently, lived bed equilibrium is not fully established. Also, within this range of bed shear stress the sediment will behave as wash load.

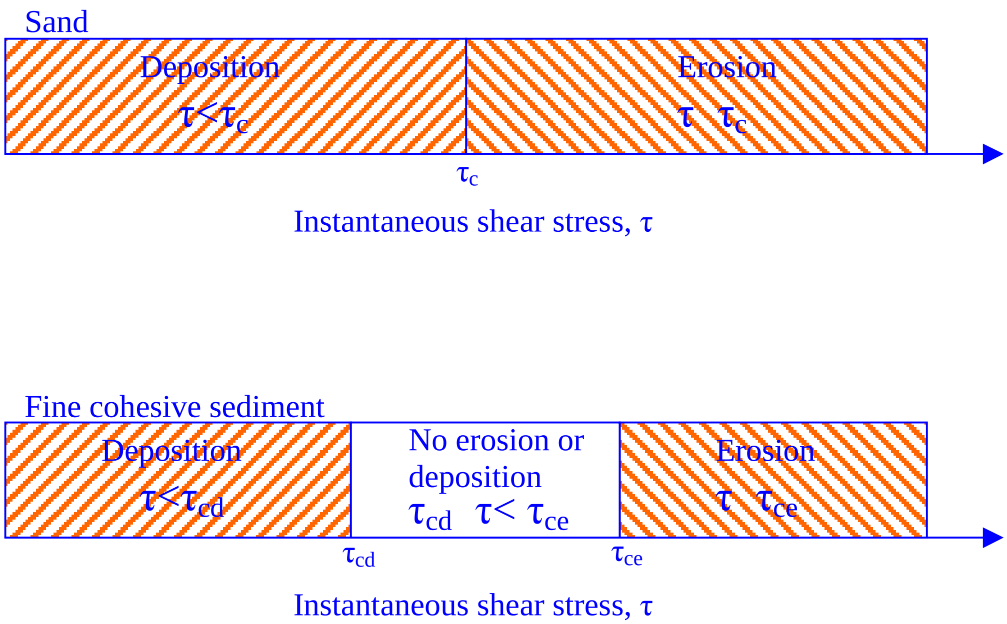


Figure F.5 Conditions for erosion and deposition of sand and fine cohesive sediment.