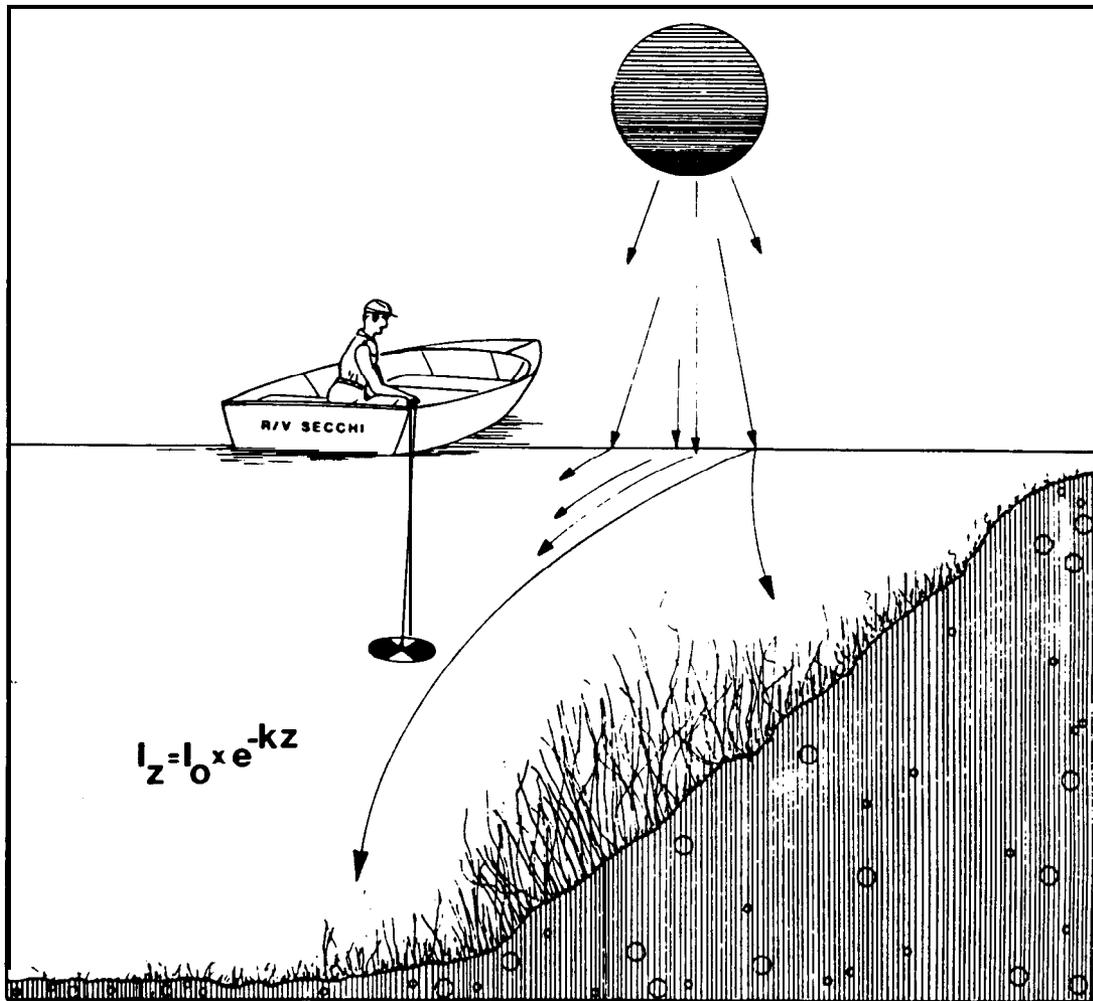


# THE LIGHT REQUIREMENTS OF SEAGRASSES



## RESULTS AND RECOMMENDATIONS OF A WORKSHOP



NOAA TECHNICAL MEMORANDUM NMFS-SEFC-287



## THE LIGHT REQUIREMENTS OF SEAGRASSES

Proceedings of a workshop to examine the capability of water quality criteria, standards and monitoring programs to protect seagrasses

Edited by W. Judson Kenworthy and Daniel E. Haunert

June 1991

Sponsored by

National Oceanic and Atmospheric Administration  
Coastal Ocean Program  
Estuarine Habitat Studies

South Florida Water Management District  
West Palm Beach, Florida

**Beaufort** Laboratory  
National Marine Fisheries Service, SEFC  
Beaufort, North Carolina

NOAA Technical Memorandum NMFS-SEFC-287

U.S. Department of Commerce  
Robert A. Moshbacher, Secretary  
National Oceanic and Atmospheric Administration  
John Knauss, Undersecretary  
National Marine Fisheries Service  
William W. Fox, Jr., Assistant Administrator for Fisheries

---

---

The Technical Memorandum series is used for documentation and timely communication of preliminary results, interim reports, or special-purpose information. Although the memoranda are not subject to complete formal review, they are expected to reflect sound professional work.

Copies of this report can be obtained from:

U.S. National Marine Fisheries Service  
**Beaufort** Laboratory  
Beaufort, NC 28516-9722

Correct citation of this report is:

Kenworthy, W. Judson and Daniel E. Haurert (eds.). 1991. The light requirements of seagrasses: proceedings of a workshop to examine the capability of water quality criteria, standards and monitoring programs to protect seagrasses. NOAA Technical Memorandum **NMFS-SEFC-287**.

The cover sketch was prepared by Curtis Lewis. We gratefully acknowledge the assistance of Stephen **DiPiero**, Mark Fonseca, staff of the South Florida Water Management District and the scientists and resource managers who contributed to the planning, preparation and implementation of the workshop.

Any reference to trade names within this document does not in any way represent an endorsement by NOAA, National Marine Fisheries Service.

## Table of Contents

Background .....	1
Workshop Agenda .....	2
Scientific Presentations .....	3
Resource Manager Presentations .....	11
Recommendations .....	16
Bibliography .....	18
List of Invited Speakers and Panels .....	19
Turbidity-Seagrass Workshop Agenda .....	22
Turbidity-Seagrass Workshop Abstracts .....	26
Session 1; Light Requirements of Temperate Seagrasses .....	26
Session 2; Light Requirements of Tropical Seagrasses .....	59
Session 3; Integrating Models with Laboratory and Field Data to Determine Sources of Light Attenuation and its Effects on the Distribution and Abundance of Seagrasses .....	114
Session 4; Development and Implementation of Federal Criteria and State Standards .....	145
Session 5; The Logistics and Scope of State and Local Water Quality Monitoring .....	152

## BACKGROUND

This workshop was convened in response to the recognition that extremely valuable economic, wildlife and aesthetic resources have been lost during the last several decades as a direct result of **seagrass** declines throughout our Nation's coastlines. The central role of seagrasses in maintaining the physical, chemical and biological integrity of many coastal ecosystems has been well documented (McRoy and Helfferich, 1977; Phillips and McRoy, 1980; Zieman, 1982; Phillips, 1984; Thayer et. al., 1984; Zieman and Zieman, 1989). **Seagrass** habitats provide nursery and feeding grounds for fish, shellfish and wildlife, including several endangered and numerous economically valuable species.

Ecological benefits resulting from the conservation and protection of fish and wildlife and the economic benefits derived from the enjoyment and harvest of **seagrass** community resources depend directly on the health and well being of **seagrass** habitats. These ecological functions and benefits, or from an economic standpoint, these services, are provided at no cost to the Nation's economy. All these functions are made possible by the ability of seagrasses to filter suspended material from the water column, inhibit resuspension of interred material, stabilize the bottom and perpetuate their existence through growth and reproduction. However, these services can be easily compromised by degradation of water quality, specifically water transparency (clarity). External perturbations to water transparency in **seagrass** habitats through dredging, nutrient loading, stormwater

runoff, agricultural drainage and boating activities upset the natural balance of water column filtration by seagrasses and, therefore, the stability and function of established and developing **seagrass** meadows. When the envelope of environmentally suitable light conditions is restricted, so are the growth and coverage of seagrasses, eventually leading to the loss of the valuable functions which fish, wildlife and man benefit from at no cost. Moreover, as these functions are lost, they are not easily replaced because: 1) many North American **seagrass** habitats are located in areas of naturally high environmental and anthropogenic stress, 2) the **seagrass** species pool available for recolonization is relatively small and, 3) recolonization rates are slow for many species (Thayer et al., 1984; Fonseca et al., 1987). Once they are lost, no other plant community can replace their unique functional attributes, nor the important feedback controls seagrasses have on their environment that serve to enhance and maintain their habitats. With that loss comes decreased bottom and shoreline stability, decreased macroepifaunal and infaunal abundance, increased sediment suspension, increased turbidity and decreased primary production: a situation that typically cannot be rectified by planting new **seagrass** plants because of the reduced light levels and enormous financial cost of restoration.

## WORKSHOP AGENDA

On November 7th and 8th 1990, twenty five scientists and resource managers from local, state and Federal agencies and academic institutions throughout the United States were invited to a workshop at the headquarters of the South Florida Water Management District in West Palm Beach Florida (see list at end).

The planned objectives of the workshop were to:

- 1) summarize the scientific knowledge regarding the light requirements of seagrasses,
- 2) examine the effects of modifying water transparency on the survival, distribution, abundance and growth of seagrasses, and
- 3) examine the capability of federal criteria and state or local water quality standards and water quality management programs to protect seagrasses from deteriorating water quality.

The workshop was sponsored cooperatively by Florida's Surface Water Improvement and Management Act (SWIM) Program, the South Florida Water Management District, the National Marine Fisheries Service, **Beaufort** Laboratory and the NOAA Coastal Ocean Program.

In addition to the expert speakers and panels, between 50 and 60 additional scientists, resource managers and planners attended and participated in discussions with the invited speakers. During the two days scientists and resource managers interacted through panel discussions which incorporated audience questions and answers. The dialogue between scientists and managers presented a unique opportunity to share information and exchange ideas between contemporary research programs and water quality management.

## DAY #1: SCIENTIFIC PRESENTATIONS

The scientific presentations covered a broad scope of laboratory physiological experiments, mesocosm research, field studies, and growth models for nearly all the **seagrass** species in the United States. Factors which influence the attenuation of light in the water column and control the growth of epiphytes on **seagrass** leaves also were discussed. These sessions were reviewed and synthesized in an open discussion period during the first evening.

Despite a wide diversity of experimental approaches, the scientists were able to demonstrate and verify by field, mesocosm, and modelling studies that the light requirements of temperate and tropical seagrasses are very similar, and are at least three to ten times greater than the traditional definitions used for the euphotic depth. The light level at which aquatic plants achieve net photosynthesis, the euphotic or compensation depth, has been defined as the depth in the water column where 1 to 5 % of the incident light remains (Ryther, 1956; Steemann-Nielsen, 1975). Unlike plankton, but similar to many submerged freshwater aquatic plants (Chambers and Kalff, 1985), seagrasses require at least 15 to 25% of the incident light just for maintenance. This is due to the large metabolic demand of their non-photosynthetic root and rhizome tissue which grow in anaerobic sediments and consume oxygen derived almost exclusively from leaf photosynthesis. In order to grow, reproduce and perpetuate their existence, seagrasses must produce more oxygen than needed for maintenance

respiration, therefore, they require more light than provided at the compensation point.

When developing guidance criteria, water quality regulations or management policy, it was emphasized that resource agencies must consider that seagrasses have higher light requirements than most other marine aquatic plants. Unlike phytoplankton, seagrasses are rooted on **the** bottom and are not usually transported upward into the **photic** zone as is frequently the case with plankton. This fixed position makes seagrasses particularly vulnerable to declining or fluctuating water transparency. For these reasons, special attention must be given to maintain and improve the level of water transparency in order to sustain and enhance existing **seagrass** populations.

Declines in **seagrass** abundance have been most pronounced at the deeper edges of **grassbed** distributions, strongly suggesting that these declines were related to decreasing water transparency. Many of these declines have been attributed to excessive nutrient loading in water bodies, and are correlated with increased light attenuation associated with extremely high levels of chlorophyll, dissolved inorganic nitrogen, dissolved inorganic phosphorus, leaf epiphytes and suspended sediments. Likewise, where nutrient discharges have been controlled or reduced, seagrasses have returned, demonstrating the potential for successfully restoring seagrasses by managing for improved water quality with specific long term goals for habitat protection and enhancement.

The scientific presentations demonstrated that **seagrass**

photosynthesis and growth are extremely sensitive to light levels. Field studies demonstrated that the maximum depth of **seagrass** distribution was correlated with average water column light attenuation, leading to the prediction that the overall abundance of seagrasses is a direct function of bathymetry and water column transparency. Experimental mesocosm research clearly demonstrated that exposure to varying degrees of increased light levels from 11% of the incident light to as much as 80%, linearly increased **seagrass** productivity. This relationship provides two critical management paradigms:

- 1) Incremental improvements in water clarity will yield corresponding higher **seagrass** productivity, deeper depth penetration, thus broader distribution.
- 2) Incremental degradation in water clarity will yield corresponding lower **seagrass** productivity, restricted depth penetration and thus, decreased distribution.

Based on conclusive scientific findings, it can now be unequivocally stated that the capacity of the coastal environment to withstand deterioration in water transparency is finite. Furthermore, once the buffering capacity is exceeded, additional declines in water transparency will continue to precipitate linear losses of **seagrass** habitat. Some seagrasses may demonstrate a temporary resiliency in their response to degradation of water transparency by drawing on stored reserves but, unless water transparency is significantly improved, there will be a predictable net loss in productivity and areal coverage. Scientists attending the workshop concluded that any enhancement of **seagrass** productivity through improved water clarity will lead to improved

growth, successful reproduction and an increase in the overall coverage and distribution of seagrasses. In turn this will enhance the fish, shellfish and wildlife resources dependent on **seagrass** habitat for food and shelter and improve shoreline and benthic stability, leading to direct aesthetic and economic benefits for man.

To abate further **seagrass** losses, and impacts to associated economic and aesthetic benefits, workshop participants concluded that immediate actions be taken by local, state and Federal authorities. These actions must be designed to prevent any further deterioration in water quality which would exacerbate the attenuation of light in the water column or increase the growth of epiphytic algae on the surfaces of the **seagrass** blades beyond that which is normally tolerated by the plants. Even though there may be uncertainties in specifying the quantitative aspects of the relationship between water transparency and **seagrass** survival, this uncertainty shouldn't prevent immediate actions designed to solve water quality problems.

Light penetration is the most important factor affecting **seagrass** growth and survival and is reduced either directly or indirectly by three major sources of light attenuation:

- 1) chlorophyll and microalgal or macroalgal blooms due to nutrient enrichment,
- 2) suspended sediments and,
- 3) color due to dissolved organic material.

Since these three sources are derived from both point and non-point discharges, water quality compliance criteria and

standards alone cannot be expected to control or abate these pollution problems. Water quality criteria and standards are based on minimum requirements for the survival of aquatic vascular plants. Federal water color criteria and the Florida transparency standard utilize the compensation depth for photosynthetic activity as the parameter to delineate the minimum allowable light level. The standard and criteria stipulate that the depth of the compensation point not be reduced by more than 10 % (substantially) compared to natural background. Because the history of significant human impacts to many coastal ecosystems is longer than the time frame over which water quality monitoring has established natural background values, the standards can only be used to maintain the status quo. A more comprehensive approach to water quality management must be adopted in order to increase light availability in environments which will support **seagrass** habitat.

Regional and waterbody specific management plans must be implemented which identify and control these major sources of light attenuation. It was demonstrated at the workshop that an existing light attenuation model and readily available water quality monitoring technology can be used to mathematically and physically decompose the general light attenuation measurement to indicate the relative importance of the three sources of light attenuation listed above. Use of this model in conjunction with an appropriately designed monitoring program would enable managers to focus attention on potential human impacts that are influencing water transparency. This approach would be an extremely valuable

management tool for sustaining long term improvements in water quality and habitat protection.

The habitat requirements for seagrasses and the conditions under which seagrasses will survive and grow can be documented and used to establish geographically specific water quality goals and objectives. For example, in the polyhaline region of the Chesapeake Bay, scientists and resource managers have utilized historical data bases, field surveys, laboratory experiments and numerical models to identify six frequently measured water quality parameters correlated with the growth and survival of the **seagrass Zostera marina** (Batiuk et al., 1990). These parameters are; 1) total suspended solids, 2) chlorophyll a, 3) dissolved inorganic nitrogen, 4) dissolved inorganic phosphorus, 5) Secchi depth, and 6) the light attenuation coefficient. Two of these parameters, total suspended solids and chlorophyll a, are directly responsible for water column light attenuation, while dissolved inorganic nitrogen and phosphorus act indirectly on light attenuation by stimulating pelagic, epiphytic and macroalgal growth. Secchi depth and the light attenuation coefficient are quantitative measures of the effect the other four parameters have on water transparency. The light attenuation coefficient should be obtained with sensors that measure photosynthetically active radiation (PAR); wavelengths which encompass the light utilized by seagrasses. Collectively, these six parameters plus an additional factor, water color, provide most of the quantitative information necessary to identify the potential sources of light

attenuation and their impact on **seagrass** growth.

Because of the predictable manner in which these parameters will effect water transparency and the similarity of light requirements for both temperate and tropical seagrasses, these scientifically established parameters also can be applied to the six subtropical **seagrass** species growing in the southeastern United States, Gulf of Mexico. and Caribbean basin. The numerical criteria assigned to these parameters may be significantly different than those proposed for the Chesapeake **Bay**, especially in oligotrophic waters with generally lower baseline values and a greater sensitivity to nutrient loading.

An example of the potential benefits of water quality management has been demonstrated in southwest Florida. Successful regrowth of seagrasses in Hillsborough Bay, a sub-estuary of Tampa Bay, has been associated with a reduction in nitrogen loading and a delayed (time **lag**) but steady decline in chlorophyll and planktonic filamentous blue green algae (Johansson and Lewis, 1990). Closely associated with these changes in water quality were improvements in water transparency as indicated by an increase in Secchi depths between 1981 and 1987. The reduction in nitrogen loading was accomplished by converting a single large wastewater treatment plant from primary to advanced wastewater treatment in 1980. Prior to these improvements in water quality, virtually all seagrasses in Hillsborough Bay were lost. However, between 1986 and 1989, **seagrass** coverage at study sites in Hillsborough and middle Tampa Bay doubled. Despite these improvements, relatively

high concentrations of chlorophyll a still persist, suggesting that other sources of nitrogen resulting from the local fertilizer manufacturing and shipping industry need to be controlled. Three important lessons are revealed by this apparent success story; 1) the persistent long-term degradation of water quality will lead to significant loss of **seagrass** resources, 2) improvements to water quality that increase transparency will lead to recovery of seagrasses, and 3) the realization of benefits from improvements in water quality will have a significant lag time in their response even if a broad scope of management practices is applied.

#### DAY #2: RESOURCE MANAGER PRESENTATIONS

Federal criteria and state water quality standards pertaining to the protection of seagrasses were addressed on the second day. Local and state water quality monitoring programs in Dade County, Florida, St. Johns Water Management District, Florida's 305b Program, the Texas Water Commission and the Chesapeake Bay Program were also presented.

The extent to which water quality monitoring is directly coupled to the protection of seagrasses and living marine resources varies widely on a state by state basis. In Texas, for example, nearly 40% of the routine water quality monitoring occurs in estuaries, yet there is no direct mechanism for coupling the measured parameters and **seagrass** habitat. Although the National Clean Water Act requires that each state report the quality of **its'** surface waters to the EPA every two years, the 305b program

as it is known, has severe shortcomings. In Florida, for example, there has been a ten year trend of declining monitoring and in the absence of an estuarine specific framework for interpretation, the qualitative and quantitative methods may be too simplistic to assess the status of estuarine water quality. Currently, measurements of PAR attenuation are not even included in the assessment formulas used to generate water quality indices.

Responsibilities for water quality monitoring, assessment and management is becoming more and more localized in Florida and elsewhere. Local government capabilities will vary enormously but, as demonstrated by the Dade County Department of Environmental Resources Management, a comprehensive knowledge of the water quality in Biscayne Bay and its' relationship to the living marine resources can be attained at the local level.

Success of any monitoring program depends on the cooperation among institutions and agencies within regions to obtain comprehensive coverage of water bodies and consistent protocols for data collection and analysis. Efforts to accomplish a regional assessment of water quality parameters directly related to the health and well being of seagrasses is beginning to be implemented in the Indian River Lagoon and Biscayne Bay under SWIM, Water Management Districts, and local sources of funding. These programs need scientific advice and practical assistance in order to improve their capabilities for protecting seagrasses. A national program designed to couple local and regional water quality monitoring programs with **seagrass** protection is needed.

The Chesapeake Bay regional water quality monitoring program, which includes state agencies with Bay jurisdictions, private and state academic institutions, and Federal agencies, is by far the most comprehensive and focused effort to establish scientifically defensible connections between water quality parameters and the distribution and growth of seagrasses. This program should serve as a model to encourage other coastal states to implement water quality monitoring and management plans that address coastal **seagrass** resources.

It was concluded that the continuing **seagrass** declines point out a fundamental flaw in water quality criteria, standards and management practices as currently implemented. The criteria and standards assume the current environment is healthy when, in fact, **seagrass** resources have been declining and continue to decline in virtually every estuary subjected to human impacts. The fact that these declines can be related to water column light attenuation argues strongly that the buffering capacity of the coastal environment has been exceeded and that further deterioration in water transparency will result in a proportional loss of **seagrass** habitat.

The potential for negative impacts by small, permitted reductions in water transparency on **seagrass** distributions were illustrated dramatically in the southern Indian River Florida. The maximum depth distribution of two important species, Halodule wrightii and Syrinsodium filiforme were strongly correlated with the annual average light attenuation coefficient, a relationship

which has been demonstrated for many other **seagrass** species as well. Given the bathymetry of the Indian River, which is a gently sloping bottom typical of many of our nation's coastal lagoons and coastal plain estuaries, the 10% reduction in transparency permitted in the Florida transparency standard would, in a very short time (< 6 months), result in an 8.3% reduction in **seagrass** coverage. Based on population growth data and assuming the 10% light reduction were corrected, the time scale for recovery from these losses is predicted to be on the order of several years to two decades. This prediction points out the necessity and urgency for implementing management plans that avoid long term loss of seagrasses.

Criteria and standards are written to allow for a certain amount of tolerance to degradation. However, there is little baseline data for transparency and PAR attenuation, or factors influencing these parameters, from which to measure the degradation. Therefore, declines are only addressable on a relative basis and resource managers have no target datum for which to strive. With our coastal water quality already at its life supporting limits in many areas, management practices must not only avoid further deterioration, but most importantly, they should promote significant improvements so that **seagrass** habitats can be restored. Otherwise we will continue to loose the living marine resources dependent on seagrasses.

The workshop participants voiced several recommendations concerning the protection and restoration of water quality in

coastal systems that could help maintain healthy and productive **seagrass** communities. These recommendations are compiled on the following page.

## RECOMMENDATIONS

1. There is an immediate need to implement comprehensive water quality protection and improvement plans. The buffering capacity in the receiving waters of most coastal systems in the U.S. has been exceeded and further deterioration in water transparency will lead to a direct loss of seagrasses. The urgency is partly due to the high cost and uncertainty of **seagrass** restoration by either natural or artificial means, even after water quality is improved.

2. Awareness of the degradation of water transparency and its effect on seagrasses must be incorporated into comprehensive water quality management programs that address a broad scope of non-point sources of pollution including; 1) discharge of colored water, 2) elevated suspended sediment loads, and 3) excessive nutrient loads which can lead to elevated water column light attenuation and blooms of epiphytic algae on the leaves of seagrasses. Water quality management programs should not rely solely on federal guidance criteria or state standards to protect seagrasses. Point sources of pollution, for which criteria and standards are targeted constitute only a portion of the source of deterioration in water transparency. Most importantly, standards and criteria currently pertaining to transparency define a compensation depth appropriate to plankton and not seagrasses.

3. Long term goals and objectives of water quality management programs should be firmly established to develop light attenuation standards and **seagrass** coverage. Baseline data on water transparency should originate from either clearly defined pristine environments or by hindcasting previous transparency conditions from historical information on **seagrass** distribution. Since the historical distribution of seagrasses were almost universally greater than now, current conditions may not represent a true baseline and should be used cautiously.

4. In order to better protect seagrasses, water quality monitoring programs should be designed with appropriate space and time scales to couple the measurement of water transparency to factors which are responsible for light attenuation so that sources of water quality problems are identified directly. PAP measurements should be made at multiple depths in the water column as the primary method of assessing transparency. At a minimum, attenuating factors should include color, chlorophyll suspended solids, and dissolved inorganic nitrogen and inorganic phosphorus. As needed, suspended solids may be partitioned into organic and inorganic fractions.

5. Applied research, using management practices as experimental treatments, should be implemented to test the direct effects of reducing specific sources of pollution on water transparency and **seagrass** growth and abundance. These management practices could include; 1) retrofitting and improving stormwater treatment systems, 2) replacing septic tanks and primary and secondary wastewater discharges with advanced wastewater treatment facilities, 4) improved watershed soil conservation practices, 5) motor vessel traffic control, 6) biological shoreline stabilization, and 7) biological filtration of discharge waters.

6. A basic research effort designed to identify the quantitative parameters in item 4 **should** be maintained in the subtropical-tropical **seagrass** system of the southeastern U.S., Gulf of Mexico, Puerto Rico and the U.S. Virgin Islands. Work now underway in the Chesapeake Bay Program which has identified the "habitat requirements" of **seagrass** in the **meso-** and polyhaline regions of the Bay should serve as a model to be applied to the subtropical and tropical seagrasses.

7. A national working group, consisting of scientists and resource managers working cooperatively, should be formed to direct and monitor the development of capabilities to protect seagrasses from water quality degradation.

## BIBLIOGRAPHY

- Batiuk, R. P. Heasley, R. Orth, K. Moore, J. Capelli, C. Stevenson, W. Dennison, L. Staver, V. Carter, N. Rybicki, R. Hickman, S. Kollar, and S. Bieber. 1990 (Draft). Chesapeake Bay submerged aquatic vegetation habitat requirements and restoration goals technical synthesis. U.S.E.P.A., Chesapeake Bay Program.
- Chambers, P.A., and J. Kalff. 1985. Depth distribution of biomass of submerged aquatic macrophyte communities in relation to Secchi depth. *Can. J. Fish. Aquat. Sci.* **42:701-709**.
- Fonseca, M.S., G.W. Thayer, and W.J. Kenworthy. 1987. The use of ecological data in the implementation and management of **seagrass** restorations. Pages 175-187 in M.J. Durako, R.C. Phillips and R.R. Lewis, eds., *Proceedings of the Symposium on Subtropical-Tropical Seagrasses of the Southeastern United States*. Florida Marine Research Publication No. 42.
- Johansson, J.O.R., and R.R. Lewis. 1990. Recent improvements of water quality and biological indicators in Hillsborough Bay, a highly impacted subdivision of Tampa Bay, Florida, U.S.A. *The International Conference on Marine Coastal Eutrophication*, Bologna, Italy, March 21-24, 1990.
- McRoy**, C.P., and C. Helfferich. 1977. **Seagrass** ecosystems, a scientific perspective. M. Dekker, New York. 314 p.
- Phillips, R.C. 1984. The ecology of **eelgrass** meadows in the Pacific Northwest: a community profile. U.S. Fish Wildl. Serv., **FWS/OBS-84/24**. 85 p.
- Phillips, R.C., and C.P. **McRoy**. 1980. Handbook of **seagrass** biology, an ecosystem perspective. Garland STPM Press, New York. 353 p.
- Ryther, J.H. 1956. Photosynthesis in the ocean as a function of light intensity. *Limnol. Oceanogr.* **1:61-71**.
- Steeman-Nielsen, E. 1975. Marine photosynthesis with special emphasis on the ecological aspects. Elsevier, New York. 141 p.
- Thayer, G.W., W.J. Kenworthy, and M.S. Fonseca. 1984. The ecology of **eelgrass** meadows of the Atlantic Coast: a community profile. U.S. Fish Wildl. Serv., **FWS/OBS-84/02**. 147 p.
- Zieman, J.C. 1982. The ecology of the seagrasses of south **Florida: a community profile**. U.S. Fish Wildl. Serv., **FWS/OBS-82/25**. 158 p.

Zieman, J.C. and R.T. Zieman. 1989. The ecology of the **seagrass** meadows of the west coast of Florida: a community profile. U.S. Fish Wildl. Serv., Biol. Rep. **85(7.25)**. 155 p.

LIST OF INVITED SPEARERS AND PANELS

Dr. Randy Alberte  
Oceanic Biology Program  
Code 1141 **MB/ONR**  
800 N. **Quincy** St.  
Arlington, Va. 22217

Rich Alleman  
Metro-Dade County  
Suite 1310  
111 N.W. First St.  
Miami, **Fl.** 33128

Dr. Robert April  
Criteria and Standards Div.  
**USEPA**  
Mail Code WH-585  
Washington, D.C. 20460

Rich **Batiuk\***  
**USEPA** Chesapeake Liaison Office  
410 Severn Ave. **#109**  
Annapolis, Md. 21403

Lou Burney  
Bureau of Surface Water Mgt.  
Florida Dept. of Environmental Regulation  
Twin Towers Building  
Tallahassee, Florida 32399

Marjorie Coombs  
Bureau of Surface Water Mgt.  
Florida Dept. of Environmental Regulation  
Twin Towers Building  
Tallahassee, Florida 32399

Dr. Bill **Dennison**  
Horn Point Environmental Lab  
Cambridge, Md. 21613

Dr. Ken **Dunton**  
University of Texas at Austin  
Marine Science Institute  
Port Aransas, Texas 78373

Dr. Mike Durako  
Florida Marine Research Institute  
100 Eighth Ave. S.E.  
St. Petersburg, Florida 33701

Mark Fonseca  
**Beaufort** Laboratory  
NMFS, NOAA  
Beaufort, N.C. 28516

Jim Fourgurean  
Dept. of Environmental Science  
University of Virginia  
Charlottesville, Virginia 22903

Dr. Charles **Gallegos**  
Smithsonian Environmental Research Center  
P.O. Box 28  
Edgewater, Md. 21037

Dr. Margaret O. Hall  
Florida Marine Research Institute  
100 Eighth Ave. S.E.  
St Petersburg, Florida 33701

Mike **Haire**\*  
Maryland Dept. of the Environment  
Chesapeake Bay Project  
2500 Broening Highway  
Baltimore, Md. 21224

John Higman  
St. Johns River Water Management District  
Palatka, **Fl** 32077

W. Judson Kenworthy  
**Beaufort** Laboratory  
NMFS, NOAA  
Beaufort, N.C. 28516

Jeff Kirkpatrick  
Texas Water Commission  
P.O. Box 13087 Capital Station  
Austin, Texas 78711

Ken Moore  
College of William and Mary  
Virginia Institute of Marine Sciences  
Gloucester Point, Virginia 23062

Dr. Hillary **Neckles**  
U.S. Fish and Wildlife Service  
LSU Wetlands Resources  
Baton Rouge, La. 70803-7500

Dr. Chris **Onuf**\*  
National Wetlands Research  
Campus Box 339  
Corpus **Christi** University  
Corpus **Christi**, Texas 78412

Dr. Robert Orth  
College of William and Mary  
Virginia Institute of Marine Science  
Gloucester Point, Virginia 23062

Dr. Frederick Short  
Jackson Estuarine Lab  
RFD 2 Adams Pt.  
Durham, NH. 03824

Dr. David Tomasko  
Florida Keys Land and Sea  
P.O. Box 536  
Marathon, Florida. 33050

Dr. Richard Wetzel  
College of William and Mary  
Virginia Institute of Marine Science  
Gloucester Point, Virginia 23062

Dr. Richard Zimmerman  
Hopkins Marine Station  
Dept. of Biology  
Stanford University  
Pacific Grove, Ca. 93950

Dr. Joseph Zieman  
Dept. of Environmental Sciences  
Univ. of Virginia  
Charlottesville, Virginia 22903

\* = submitted abstract but did not make a formal presentation  
or sit on a panel.

TURBIDITY-SEAGRASS WORKSHOP AGENDA

**Sponsors:**

The Surface Water Improvement and Management Act (SWIM) of Florida

South Florida Water Management District  
National Oceanographic and Atmospheric Administration Coastal Ocean Program.

**Location:**

South Florida Water Management District Headquarters, 3301 Gun Club Road, West Palm Beach, Florida.

**Workshop Schedule:** November 7th and 8th, 1990.

A. **Day 1, November 7th**

**8:00-Overall** Introduction (W. Judson Kenworthy, NOAA, NMFS, Beaufort, North Carolina and Dan Haunert, South Florida Water Management District)

1. Session 1; Light requirements of temperate seagrasses

**8:10-Introduction** to 1st session (William C. Dennison, Univ. of Maryland and Horn Point Environmental Labs)

**8:15-Prediction** of light requirements for **eelgrass** (Zostera marina L.) from numerical models (Richard C. Zimmerman and Randall S. Alberte, Univ. of Chicago and Hopkins Marine Station)

**8:40-Light** limitation on **seagrass** growth (Frederick T. Short, Jackson Estuarine Lab, Univ. of New Hampshire)

**9:05-Field** studies of the effects of variable water quality on temperate **seagrass** growth and survival (Ken Moore, College of William and Mary and VIMS)

**9:30-Break**

**9:45-Panel** discussion with moderator (William C. Dennison)

2. Session 2; Light requirements of tropical seagrasses

**10:20-Introduction** to second session' (Mike Durako, Florida Marine Research Institute)

**10:25-Photosynthesis**, respiration, and whole plant carbon balance of Halodule wrightii, Thalassia testudinum and Syringodium filiforme (Jim Fourqurean and Joseph C. Zieman, Univ. of Virginia)

**10:50-Seasonal** variations in the photosynthetic performance of Halodule wrightii measured in situ in Laguna Madre, Texas (Kenneth H. Dunton and David A. Tomasko Univ. of Texas, Marine Science Institute and Florida Keys Land and Sea Trust)

**11:15-Growth** and production of Halodule wrightii in relation to continuous measurements of underwater light levels in south Texas (David A. Tomasko and Kenneth H. Dunton, Florida Keys Land and Sea Trust and Univ. of Texas, Marine Science Institute)

**11:40-Lunch** Break

**1:15-Responses** of Thalassia Testudinum to in situ light reduction (Margaret O. Hall, Florida Marine Research Institute)

**1:40-Defining** the ecological compensation point of seagrasses Halodule wrightii, Syringodium filiforme from long-term submarine light regime monitoring in the southern Indian River (W. Judson Kenworthy, Mark S. Fonseca and Stephen J. DiPiero, Beaufort Lab, NMFS, NOAA). Results from a study in the Laguna Madre of Texas by Chris Onuf, USFWS, National Wetlands Research Center, Corpus Christi State University, Corpus Christi, Texas entitled Light requirements of Halodule wrightii, Syringodium filiforme, and Halophila enselmanni in a heterogeneous and variable environment inferred from long-term monitoring were also presented in this talk.

**2:05-Panel** discussion with moderator (Mike Durako)

**2:35-Break**

3. Session 3; Integrating models with laboratory and field data to determine sources of light attenuation and its effects on the distribution and abundance of seagrasses.

**2:50-Introduction** to third session ( Dick Wetzel, College of William and Mary and VIMS)

**3:00-Modelling** spectral light available to submerged aquatic vegetation (Charles Gallegos, David L. Correll and Jack W. Pierce, Smithsonian Environmental Research

Center and National Museum of Natural History)

**3:25-Complex** interactions among light-reducing variables in **seagrass** systems: simulation model predictions for long-term community stability (Hillary **Neckles**, U.S. Fish and Wildlife Service)

**3:50-Modelling** light availability for **seagrass** growth (Frederick T. Short, Jackson Estuarine Lab, Univ. of New Hampshire)

**4:15-Photosynthetic** and growth responses of tropical and temperate seagrasses in relation to secchi depth, light attenuation and daily light period (William C. Dennison, Univ. of Maryland and Horn Point Environmental Labs)

**4:40-Panel** discussion with moderator (Dick Wetzel)

**5:05-Dinner** Break

**7:30-Evening** session for discussion and question and answer period (Moderator Joseph C. Zieman, Univ. of Virginia)

B. **Day 2**, November 8th

**8:00-Opening** remarks (Mark S. Fonseca, **Beaufort** Lab, NMFS, NOAA)

1. Session 4; Development and implementation of federal criteria and state Standards.

**8:10-Introduction** to session 4 (Richard Batiuk, **USEPA**, Chesapeake Bay Liaison Office, Region III). Due to birth of his child Rich was unable to make this presentation and Robert Orth and Bill **Dennison** substituted for him

**8:15-Federal** water quality criteria program (Robert April, **USEPA** Water Quality Criteria and Standards Division)

**8:40-Florida's** Water Quality Standards Program (Marjorie **Coombs**, Florida Dept. of Environmental Regulation)

**9:05-Coordinating** the synthesis of two decades of Chesapeake Bay SAV research (Richard Batiuk, **USEPA** Chesapeake Bay Liaison Office).

**9:30-Break**

- 9:50-Panel** discussion with moderator (Bill Dennison)
3. Session 5; The logistics and scope of state and local water quality monitoring.
- 10:20-Introduction** to session (Lou Burney, Florida Dept. of Environmental Regulation)
- 10:25-A synopsis of water quality and monitoring in Biscayne Bay Florida (Rick Alleman, Dade County Environmental Resources Management)
- 10:50-Estuarine Water Quality** Monitoring for the Indian River Lagoon under the management of the St. Johns River Water Management District (John Higman, St. Johns River Water Management District)
- 11:15-Estuarine** water quality monitoring in Texas (Jeff Kirkpatrick, Texas Water Commission)
- 11:40-Lunch** Break
- 12:45-Maryland's** Chesapeake Bay water quality monitoring program and its relevance to SAV communities. (Mike **Haire**, Robert E. Magnien and Steven E. Bieber, Maryland Dept. of the Environment). Due to complications Mike **Haire** was unable to attend and Bill **Dennison** substituted for him.
- 1:10-State of Florida 305b Program for assessment of the status and trend of estuarine water quality (Lou Burney, Florida Dept. of Environmental Regulation)
- 1:35-Panel** discussion with moderator (Lou Burney)
4. Session 6; Discussion of alternative parameters for assessing water transparency based on the attenuation of Photosynthetically Active Radiation (PAR) or Secchi disk depth
- 2:15-Spatial** and temporal variation of PAR attenuation: examples from field studies in the southern Indian River ( W. Judson Kenworthy, **Beaufort** Lab, NMFS, NOAA)
- 2:40-Wrap** up discussion: Can we develop and implement a transparency parameter to protect seagrasses?

## SESSION 1; LIGHT REQUIREMENTS OF TEMPERATE SEAGRASSES

Zimmerman, Richard C. and Randall S. Alberte, Dept. Molecular Genetics and Cell Biology, The University of Chicago, Chicago, IL and Hopkins Marine Station, Pacific Grove, CA

### PREDICTION OF THE LIGHT REQUIREMENTS FOR **EELGRASS** (ZOSTERA MARINA L.) GROWTH FROM NUMERICAL MODELS

Seagrasses form the basis of critical, yet extremely fragile ecosystems in shallow coastal embayments and estuaries throughout the world. Although highly productive, these systems are particularly vulnerable to increases in water column turbidity that result from eutrophication, chronic upstream erosion, and periodic dredging of coastal environments. Light availability is recognized generally as the most important environmental factor regulating the depth distribution, density, and productivity of many species of submerged aquatic vegetation, including the temperate **seagrass** Zostera marina L. (eelgrass) (**Backman** and Barilotti, 1976; Orth and Moore, 1983; 1988; **Dennison** and Alberte, 1982; 1985; 1986). In that regard, the state of Florida and the Environmental Protection Agency (EPA) of the Federal Government have established transparency standards (based on traditional theories of light requirements for primary production) that are designed to limit increases in turbidity above historical background levels. Although enforcement of the turbidity standards can prevent further deterioration of the submarine light environment, the standards cannot be used as a mandate to increase light availability in environments where the history of human disturbance is significantly longer than the baseline of environmental data.

For example, human-induced turbidity has affected the San Francisco Bay estuary in California since the mid-19th century. Studies of water column transparency, euphotic depth and primary production, however, have been conducted in a systematic way only in the last 20 years. Turbidity is extremely high in the bay and coefficients of attenuation often exceed  $3.0 \text{ m}^{-1}$  (Secchi depths  $\leq 0.75 \text{ m}$ ). Thus, the euphotic zone (depth of the 1% light level) is less than 2 m. With mixed layer depths frequently in excess of **5m**, this results in very low rates of phytoplankton productivity in San Francisco Bay (Alpine and Cloern, 1988). High turbidity also limits the depth distribution of important aquatic macrophytes, such as Zostera marina, to extremely shallow fringes and shoals (Zimmerman et al., in press). Thus, water quality standards that merely serve to maintain the status quo of light transparency will not permit environmentally-sound management of this resource or expansion of primary production in San Francisco Bay or any other estuarine environment heavily affected by a history of anthropogenic turbidity.

The success of any **seagrass** management program is strictly dependent upon the maintenance of a physical environment that will ensure initial establishment and support long-term growth. Although simple models (e.g. Dennison, 1987; Zimmerman et al., 1990) can be useful management tools in specific habitats after extensive **field-**calibration, the models are not general enough to be applied with confidence to management-oriented problems in other regions without extensive testing. Furthermore, the data base of field observations

from most habitats required to test even simple models is currently inadequate, particularly with respect to temporal variations in light availability and the physiological response of different **eelgrass** populations.

Genetic variability among **eelgrass** populations can also have significant impacts on growth and physiology, as isolated populations may have evolved specific adaptations to unique features of their individual habitats. This is known as ecotypic differentiation. Zostera marina, a true marine "**weed**", has a cosmopolitan distribution in temperate oceans of the northern hemisphere. Thus, this species is expected to show genetic diversity at the population level that may have important regional implications for the restoration and management of this resource. **Eelgrass** may be excluded from the Chesapeake Bay in areas where the mean diffuse attenuation coefficient (**k**) of the water column exceeded  $1.7 \text{ m}^{-1}$  (Wetzel and **Penhale, 1983**), while it is capable of growing in areas of San Francisco Bay where mean diffuse attenuation coefficients exceed  $3.0 \text{ m}^{-1}$  (Alpine and Cloern, 1988; Wyllie-Echeverria and **Rutten, 1989**; Zimmerman et al., in press). The extent to which such differences reflect environmental influences and/or genetically-based differentiation in **eelgrass** performance is unclear.

We have examined the effects of genetically-based differences in physiological performance on growth and productivity of 3 geographically isolated **eelgrass** populations from California growing in experimental common gardens in **Elkhorn** Slough near

Monterey Bay, California. While differences in physiological performance (photosynthesis and respiration) measured in the laboratory were consistent with genetic differences indicated by leaf width morphology and defined by restriction fragment length polymorphism (**RFLPs**) of genomic DNA, the performance differences have not yielded a significant effect on the calculation of daily light requirements, measurements of growth or mean carbohydrate content of photosynthetic shoots in the field (Britting et al., in review).

Model calculations of carbon-balance based on assessment of daily metabolic activity indicate that Z. marina requires somewhere between 3 and 10 h of irradiance-saturated photosynthesis each day (termed  $H_{sat}$ ; **Dennison** and Alberte, 1982) to meet the demands of respiration and growth (Dennison and Alberte, 1985; 1986; Marsh et al., 1986 ; Zimmerman et al., 1989). The uncertainty in this estimate reflects the combined effects of environmental influences (temperature and light) on metabolic activity and the distribution of biomass between roots and shoots. Thus, it is difficult to define a single "**critical**" value of  $H_{sat}$  that predicts or ensures long-term growth and survival of Zostera marina in all environments.

A better understanding of the short term (daily) scales of variation in the physical environment may improve significantly our estimates of **eelgrass** light requirements. Within San Francisco Bay, Z. marina is limited to different depths at 5 different sites (**Fig.1**, from Zimmerman et al., in press). Rather than being limited

by a single "**critical**" value of light availability, however, depth limits of **eelgrass** at the different sites cross lines of mean constant  $H_{\text{sat}}$  as the mean attenuation coefficient ( $k$ ) increases. Thus, minimum daily light requirements for **eelgrass** survival appear to increase with mean turbidity perhaps because the variance in light attenuation also increases with the mean value. As a result, transient periods of extreme turbidity may be critical in determining the depth limits of **eelgrass** in some locations in San Francisco Bay. These transient events occur on time scales that are generally undersampled by traditional weekly or monthly monitoring programs (Zimmerman and Kremer, 1984). Undersampling can result in a serious underestimation of the real turbidity and a significant overestimate of  $H_{\text{sat}}$ . Consequently, detailed information regarding relevant time scales of variation in critical environmental parameters, such as light availability, are absolutely essential for reliable application of **any** model calculations to the management of submerged aquatic macrophytes.

Recent experiments have expanded our appreciation of the critical role that roots play in the question of **eelgrass** survival. **Eelgrass** roots exist within the anoxic sediment layer and are dependent on the transport of photosynthetically-produced oxygen from the shoots to maintain aerobic respiration during the light period (Smith et al., 1984). Roots must be able to tolerate prolonged periods of anoxia each night and even through some days of extremely low light availability. While anoxic, most metabolic processes (protein synthesis, carbohydrate transport and growth)

are inhibited or greatly reduced (Smith, 1989). We have found that extending the anoxic period to 18 h or more (simulating  $k \geq 3.0$  or secchi depths  $\geq 0.75$  m) may disrupt the transport of carbohydrates to root tissue (Zimmerman and Alberte, in prep.). Prolonged exposure (up to 30 d) to short daily periods of photosynthesis (< 6h) leads to carbon limitation in roots well before significant carbon limitation is observed in shoots (Fig. 2, from Zimmerman and Alberte, in prep.). Although there is evidence for some metabolic adaptation by Z. marina to increase rates of carbohydrate transport to roots under shortened daily light periods, this adaptation is not enough to prevent carbohydrate depletion of the roots. Thus, light availability may regulate the depth distribution of **eelgrass** by controlling rates of carbon transport to roots independently of whole-plant carbon balance. Studies that will define this relationship are currently in progress.

As a result of these studies, our modeling efforts have gone beyond simple questions of shoot and/or whole plant carbon balance to more detailed examinations of carbon partitioning between shoots and roots, and how these dynamics are affected by the length of the daily light (or aerobic) period. The model, as currently conceived, simulates the transport of carbohydrate from leaves to roots as regulated by the daily light period through the activities of the enzymes sucrose phosphate synthase (SPS) and sucrose synthase (SS) (Fig. 3). These enzymes are reliable indicators of the rate of source-to-sink sucrose transport in a variety of higher plants (Huber et al., 1985; Lowell et al., 1989). Rates of photosynthesis,

growth, sucrose synthesis, transport and catabolism are driven by light availability and coupled to each other by a series of partial differential equations based conceptually on the cell quota model proposed by Droop (1973). Laboratory experiments are currently under way that will provide physiological data to parameterize the model. We have just initiated a field program to collect the necessary data on in situ light availability, growth, and carbon partitioning with both **subtidal** and intertidal populations of **eelgrass** that can be used to test the model.

Although individual criteria of water clarity can be developed to protect specific **seagrass** populations in specific habitats, the ability to transfer these criteria to other populations or environments cannot be considered universal. Thus, in the short term it will be necessary to continue with detailed studies of specific populations and habitats threatened by environmental change while simultaneously working toward the long-term goal of developing a general model (or set of criteria) based on a full mechanistic understanding of the role light availability plays in regulating the productivity and distribution of seagrasses.

#### REFERENCES

- Alpine, A.E. and Cloern, J.E. 1988. PHYTOPLANKTON GROWTH RATES IN A LIGHT-LIMITED ENVIRONMENT, SAN FRANCISCO BAY. Mar. Ecol. Prog. Ser., **44:167-173**.
- Backman**, T.W. and Barilotti, D.C. 1976. IRRADIANCE REDUCTION: EFFECTS ON STANDING CROPS OF THE **EELGRASS** ZOSTERA MARINA IN A COASTAL LAGOON. Mar. Biol., **34:33-40**.
- Britting, S.A., Zimmerman, R.C., and Alberte, R.S. In review. CHARACTERIZATION OF DISJUNCT POPULATIONS OF ZOSTERA MARINA L.

(EELGRASS) FROM CALIFORNIA. I. GROWTH AND PHYSIOLOGICAL PERFORMANCE. Mar. Biol.

Dennison, W.C. 1987. EFFECTS OF LIGHT ON **SEAGRASS** PHOTOSYNTHESIS, GROWTH AND DEPTH DISTRIBUTION. Aquat. Bot., **27:15-26**.

Dennison, W.C. and Alberte, R.S. 1982. PHOTOSYNTHETIC RESPONSE OF ZOSTERA MARINA L. (EELGRASS) TO IN SITU MANIPULATIONS OF LIGHT INTENSITY, Oecologia, **55:137-144**.

Dennison, W.C. and Alberte, R.S. 1985. ROLE OF DAILY LIGHT PERIOD IN THE DEPTH DISTRIBUTION OF ZOSTERA MARINA (EELGRASS). Mar. Ecol. Prog. Ser., **25:51-61**.

Dennison, W.C. and Alberte, R.S. 1986. PHOTOADAPTATION AND GROWTH OF ZOSTERA MARINA L. (EELGRASS) TRANSPLANTS ALONG A DEPTH GRADIENT. J. Exp. Mar. Biol. Ecol., **98:265-282**.

Droop, M.R. 1973. SOME THOUGHTS ON NUTRIENT LIMITATION IN ALGAE. J. Phycol. **9:264-272**.

Huber, S.C., Kerr, P.S., Kalt-Torres, W. 1985. REGULATION OF SUCROSE FORMATION AND MOVEMENT. pp. 199-215. in R.L. Heath and J. Priess (eds), REGULATION OF CARBON PARTITIONING IN PHOTOSYNTHESIS TISSUES. Am. Soc. Plant Phys., Rockville, Md.

Lowell, C.A., Tomlinson, P.T. and Koch, K.E. 1989. **SUCROSE-METABOLIZING ENZYMES** IN TRANSPORT TISSUE AND ADJACENT SINK STRUCTURES IN DEVELOPING CITRUS FRUIT. Plant Physiol. **90:1394-1402**.

Marsh, J.A., Dennison, W.C. and Alberte, R.S. 1986. EFFECTS OF TEMPERATURE ON PHOTOSYNTHESIS AND RESPIRATION IN **EELGRASS** (ZOSTERA MARINA L.). J. Exp. Mar. Biol. Ecol., **101:257-267**.

Orth, R.J. and Moore, K.A. 1983. CHESAPEAKE BAY: AN UNPRECEDENTED DECLINE IN SUBMERGED AQUATIC VEGETATION. Science, **222:51-53**.

Orth, R.J. and Moore, K.A. 1988. DISTRIBUTION OF ZOSTERA MARINA L. AND RUPPIA MARITIMA L. SENSU LATO ALONG DEPTH GRADIENTS IN THE LOWER CHESAPEAKE BAY, U.S.A. Aquat. Bot., **32:291-305**.

Smith, R.D. 1989. ANAEROBIC METABOLISM IN THE ROOTS OF ZOSTERA MARINA L. Ph.D. Dissertation, The University of Chicago, Chicago, 241 pp.

Smith, R.D., Dennison, W.C. and Alberte, R.S. 1984. THE ROLE OF **SEAGRASS** PHOTOSYNTHESIS IN AEROBIC ROOT PROCESSES. Plant Physiol. **74:1055-1058**.

Wetzel, R.L. and **Penhale**, P.A. 1983. PRODUCTION ECOLOGY OF **SEAGRASS** COMMUNITIES IN THE LOWER CHESAPEAKE BAY. Mar. Tech. Soc. J., **17:22-31**.

Wyllie-Echeverria, S. and **Rutten**, P.J. 1989. INVENTORY OF **EELGRASS** (ZOSTERA MARINA L.) IN SAN FRANCISCO/SAN PABLO BAY. Administrative Report, Southwest Regional NOAA, National Marine Fisheries Service, Terminal Island, California, **18pp.**

Zimmerman, R.C. and Kremer, J.N. 1984 EPISODIC NUTRIENT SUPPLY TO A KELP FOREST ECOSYSTEM IN SOUTHERN CALIFORNIA. J. Mar. Res. **42:591-604.**

Zimmerman, R.C., Smith, R.D. and Alberte, R.S. 1989. THERMAL ACCLIMATION AND WHOLE PLANT CARBON BALANCE IN ZOSTERA MARINA L. (EELGRASS). J. Exp. Mar. Biol. Ecol., **130:93-109.**

Zimmerman, R.C., Smith, R.D. and Alberte, R.S. 1990. **SEAGRASS** REVEGETATION: DEVELOPING A PREDICTIVE MODEL OF LIGHT REQUIREMENTS FOR ZOSTERA MARINA. pp. 6-12. In K. Merkel, R. Hoffman and J. Stuckrath (eds), Proceedings of the California **Eelgrass** Symposium. Sweetwater River Press, National City, California.

Zimmerman, R.C., Reguzzoni, J.L., Wyllie-Echeverria, S., Josselyn, M. and Alberte, R.S. In press. ASSESSMENT OF ENVIRONMENTAL SUITABILITY FOR GROWTH OF ZOSTERA MARINA L. IN SAN FRANCISCO BAY. Aquatic. Bot.

Zimmerman, R.C. and Alberte, R.S. In prep. EFFECTS OF DAILY LIGHT PERIOD ON SUCROSE METABOLISM AND DISTRIBUTION IN ZOSTERA MARINA L. (EELGRASS). Plant Physiol.

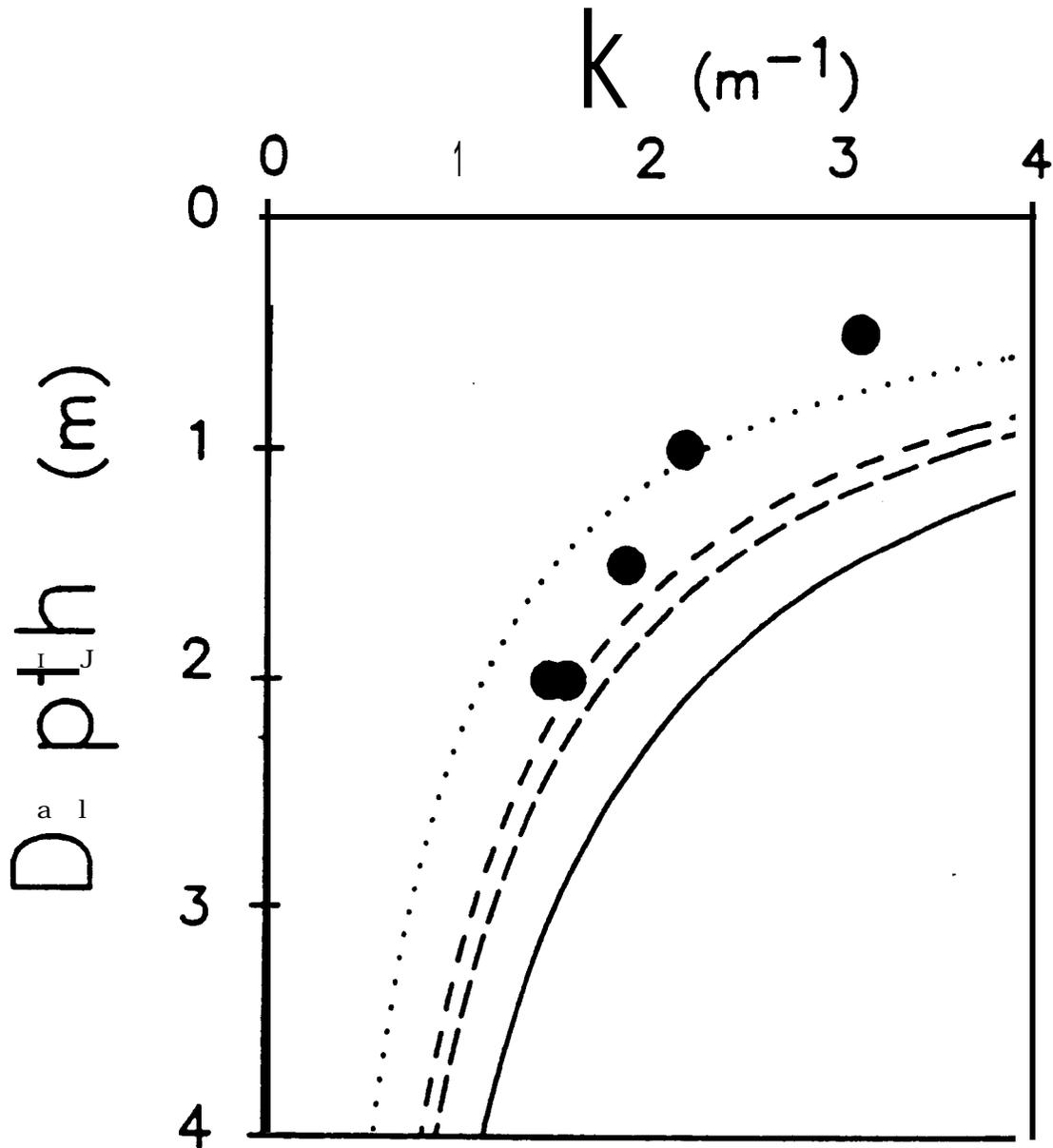


Figure 1. Depth limits of Zostera marina plotted as a function of mean diffuse attenuation ( $k$ ) for 5 sites in San Francisco Bay (from Zimmerman et al. in press). Depth profiles of constant  $H_{sat}$  periods illustrate that eelgrass distributions are not limited by the same value of  $H_{sat}$  at all sites, but require longer  $H_{sat}$  periods as mean  $k$  increases. Legend: (—) depth of the euphotic zone ( $1\% I_0$ ); (— — —)  $H_{sat} = 2$  h; (- - -)  $H_{sat} = 6$  h; (· · ·)  $H_{sat} = 10$  h.

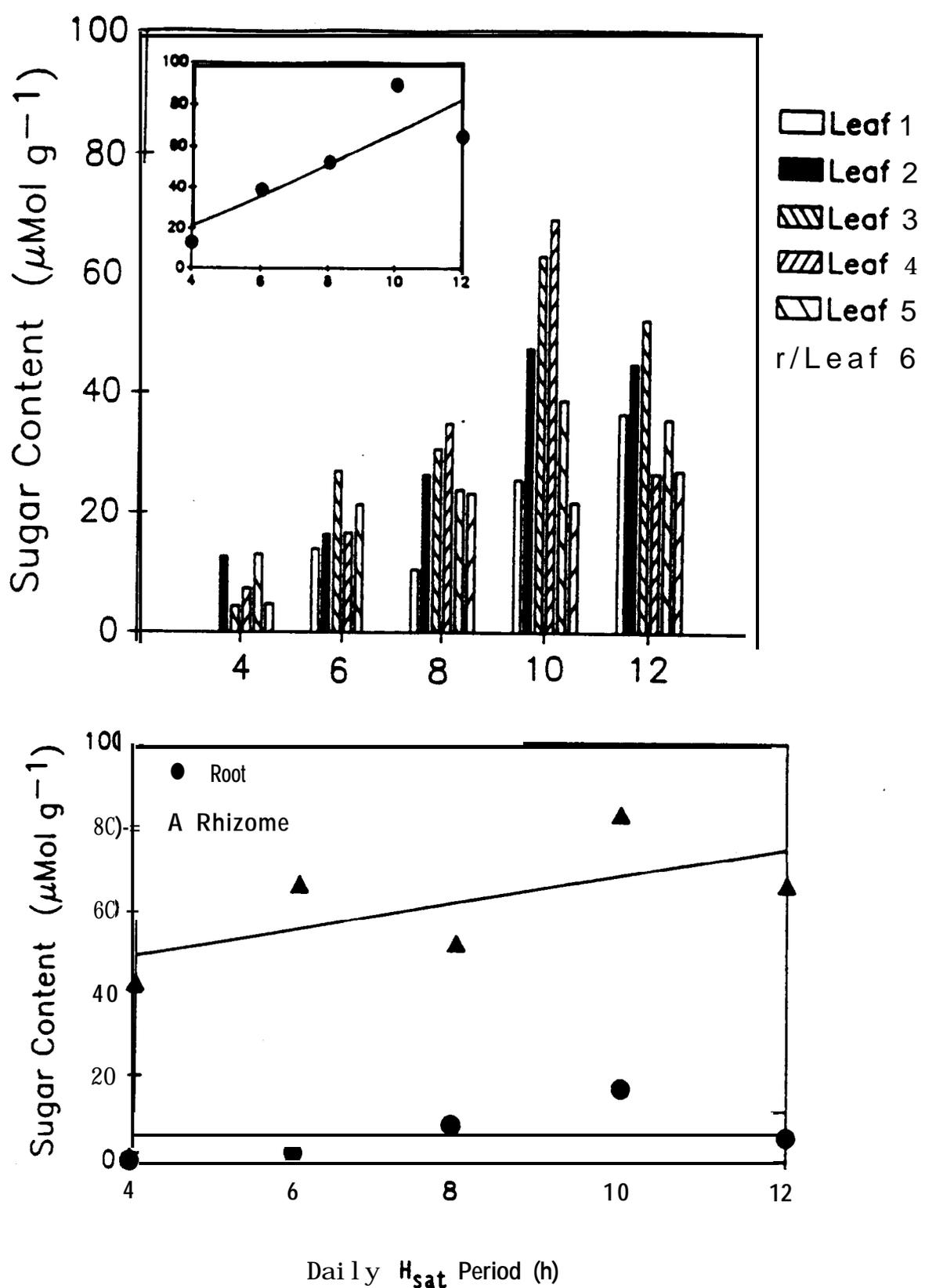


Figure 2. Sugar content of shoots (top) distributed among individual leaves as well as the average sugar content of integrated shoots (inset). Sugar content of rhizomes and roots (bottom). There was no measurable sugar in root tissue in the plant growing under 4 h  $H_{\text{sat}}$  even though measurable amounts of sugar were present in shoots and especially rhizomes. This suggests that short  $H_{\text{sat}}$  periods may limit **eelgrass** survival because sugar present in shoots and rhizomes cannot be transported to the roots.

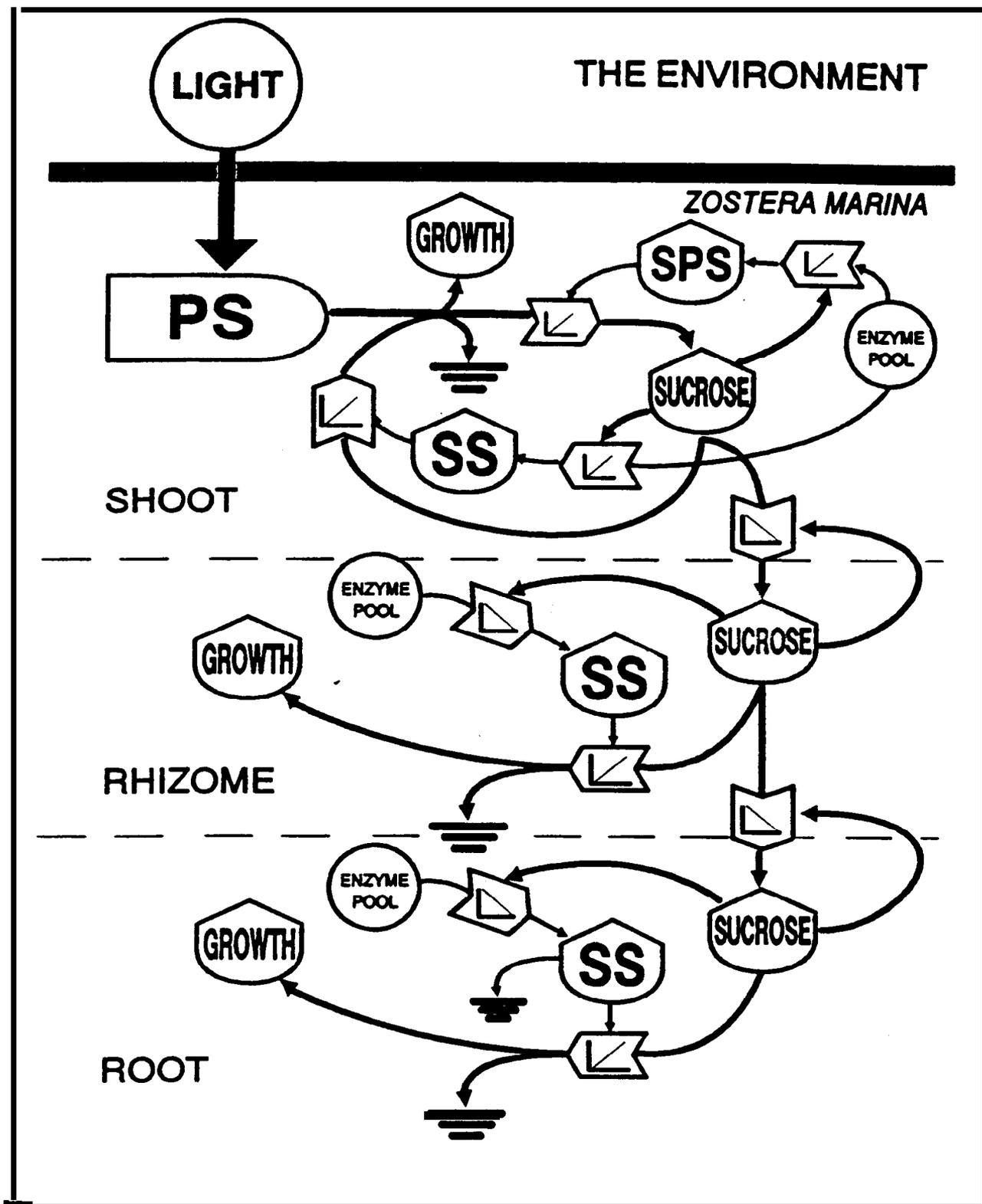


Figure 3 . Energy circuit diagram of the physiological dynamics regulating sucrose partitioning in the eelgrass *Zostera marina*. Symbols are after Odum (1983). Arrows represent the flow of energy and/or resources from one compartment to another. Circles represent resources under control of processes outside the formulated model, tank symbols represent accumulation of resources into specific pools that are affected by interactions with other terms in the model. Large block arrows indicate interaction terms. The nature of the interaction is indicated by graph symbols inside the block arrows. Positive slopes indicate that the interaction enhances the flow from one compartment to another, while negative slopes indicate the opposite.

Short, Frederick T., Jackson Estuarine Laboratory, University of New Hampshire, Durham, NH.

#### LIGHT LIMITATION ON **SEAGRASS** GROWTH

A series of mesocosm experiments were designed to examine the effects of reduced light intensity on the production, growth, and potential biomass of eelgrass, Zostera marina. Six outdoor tanks (1.5-2m) with a gradient of light intensity from 11% of surface light to full sunlight (94% 1 cm below surface) were planted with **eelgrass** seedlings in early June and the plants were allowed to grow to maturity. Reduced light levels were achieved by covering the tanks above the water level with neutral density screen which reduced incident light irradiance to **61%, 41%, 20%**, and 11% of the surface light conditions. It should be noted that the shading of these plants had no effect on the photoperiod; thus, only the effects of reduced light intensity reaching the **eelgrass** leaves was examined. Reduction in light intensity by shading is analogous to decreasing water clarity but not necessarily analogous to increased depth, which also alters the photoperiod of light reaching the plants. Throughout the mesocosm experiment, leaf growth, morphology and density were measured. At the conclusion of the experiment, total biomass was also assessed.

A marked difference in shoot density was observed between the variously shaded tanks as the season progressed (Fig. 1, top panel). Density at the lowest light level increased slightly then dropped to the initial planting density of 200 plants per square meter. Shoot density increased logarithmically with increased light

to a maximum density of  $>400$  shoots  $m^{-2}$ , achieved at the end of the experiment under 'full' light conditions (94% light level). Leaf growth measured as leaf elongation on a per shoot basis showed a significant linear increase with increased light intensity. However, specific growth rate, that is growth in cm of new leaf per cm of shoot per day showed little variation under the different light treatments. That is to say, the plants appear to have adapted to grow at a maximum specific production rate based on light availability. Utilizing density measurements to convert per shoot growth rates to per square meter growth rates, which combined the effects of increased growth, this measure showed a strong linear increase in production with light. These results indicate no evidence of saturation below ambient light levels when the plants were allowed to adapt to these different light conditions.

A significant change in leaf size also developed between treatments, with the plants in the most shaded tanks at the lowest light levels growing significantly longer than the plants at the higher light levels. In all cases, leaf length exceeded the water depth and the plants grew with a portion of the leaves horizontal on the tank surface. The increase in leaf size appears to be a morphological adaptation of the plants to reduced light intensity. The leaf surface area of individual plants increased, and the density of plants decreased concurrently in the tank with the lowest light level.

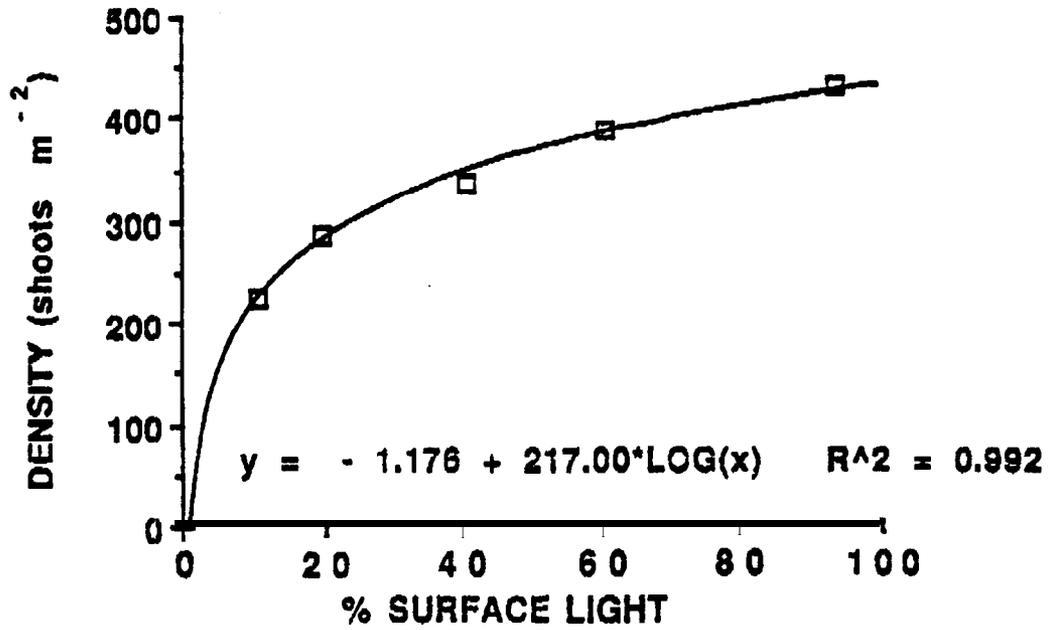
Standing leaf biomass was significantly higher at high light levels than at low light levels, reflecting the combined

differences in plant size and density. The increased shoot density at high light conditions overwhelmed the effects of larger plants at lower light levels. Standing leaf biomass exhibited a logarithmic increase with light levels.

It is clear that decreasing only the light intensity, which is analogous to decreasing water clarity (but not necessarily analogous to increased depth, which also alters the photoperiod of light reaching the plants), has a major effect on **eelgrass** production, biomass and morphology in experimental mesocosms. The plants respond to decreased light levels by lower growth ratio and biomass production. The plants respond morphologically by increasing leaf size and decreasing density at reduced light levels.

An unexpected result of this study is evidence of plant adaptation to maximize specific growth rate at all light conditions by adjusting morphology and shoot density. The specific growth rate varied the least of all parameters measured. However, shoot growth differences varied substantially among the light treatments, primarily in response to the plant morphometric configurations. These studies show that under quasi-natural conditions where the plants are allowed to adapt to different light levels, **eelgrass** does not demonstrate light saturation conditions in response to varying light and in fact overall plant production increases linearly in response to increased light intensity.

MESOCOSM EXPERIMENT -- July 1988



MESOCOSM EXPERIMENT -- September 1988

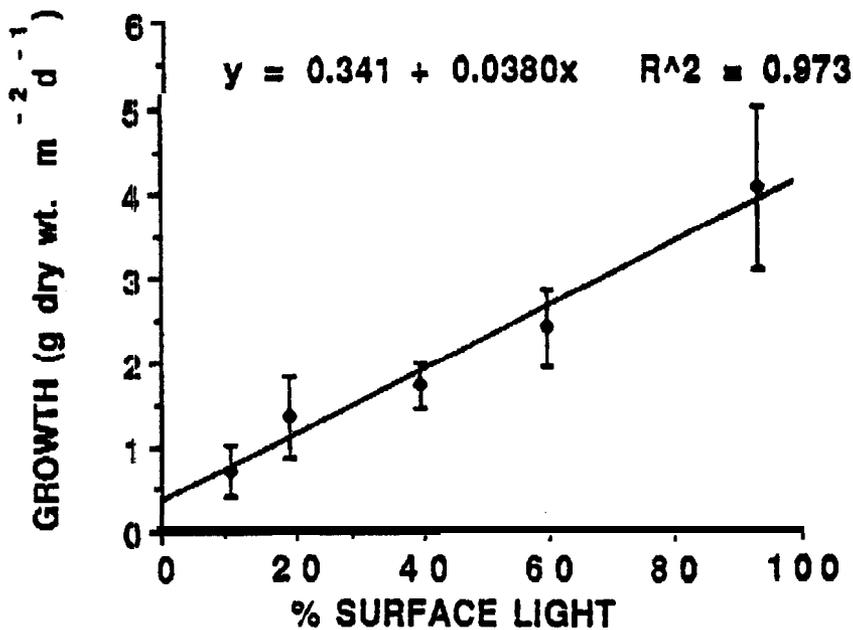


Figure 1: Results of mesocosm experiments with Zostera marina. Top panel are the results of density vs. % surface light. Bottom panel are the results of net growth vs. % surface light.

Moore, Kenneth A., **VIMS/** College of William and Mary, Gloucester Pt., VA.

FIELD STUDIES OF THE EFFECTS OF VARIABLE WATER QUALITY ON TEMPERATE **SEAGRASS** GROWTH AND SURVIVAL

INTRODUCTION: A **dieback** of **seagrass** beds dominated by Zostera marina L. was observed in the Chesapeake Bay in the late **1960's** and early **1970's**. The pattern of the **dieback**, which was greatest in the western tributaries and more pronounced in the deeper portions of the remaining beds, suggested that the losses may have been associated with enrichment due to the influence of river discharge in these areas (Orth and Moore 1983).

A series of field studies were therefore undertaken to determine the potential for environmental conditions to limit **seagrass** distribution in one southwestern tributary of the Bay. The goal of the study was to develop correlative relationships between factors in the water column including suspended sediment loads, light availability, inorganic nutrient levels, and **seagrass** growth and survival. Specific objectives were to monitor the water quality and light environment at a series of **dieback** and surviving sites, to relate any site differences to differences in potential plant growth, and, using these two sets of information to determine the levels of these factors which characterize **seagrass** communities in this region.

In the summary of field results presented here the data are integrated by season and potentially critical periods are identified. The results are then compared to several transparency standards which have been used to protect marine aquatic vascular

plants from deteriorations in water quality.

METHODS: The lower York River ( $37^{\circ}15'$ ,  $76^{\circ}30'W$ ) was chosen as a main study area, since it had a **seagrass** decline characteristic of the entire lower bay region and within a relatively short area included sites that experienced complete **dieback**, partial **dieback**, or were only minimally affected by loss. Stations selected for monitoring were in broad, **subtidal** flats ranging from the mouth of the York to the historical limits of **seagrass** growth, 27 km upriver (**Fig.1**).

Transplants of Zostera marina, following techniques of Fonseca et al. (1982, 1985), were used to determine the current capacity of the various sites to support vegetation. After transplanting in the fall of each year survivorship was monitored at monthly to bimonthly intervals until either no plants remained at a site or the planting units had coalesced. During certain years plants were also sampled for biomass, density and epiphyte loading. Macrophyte growth was studied in situ, from April 1985 to July 1986, using transplanted turfs of Z. marina, including sediments, which were placed in polyethylene boxes and submerged at the vegetated and **dieback** sites, Gloucester Point and **Claybank** respectively. Growth was determined using a modified leaf marking technique after Sand-Jensen (1975).

Triplicate subsurface water column samples were taken at biweekly intervals starting in August, 1984. Parameters measured included nitrite, nitrate, ammonium, inorganic phosphorus, chlorophyll a, total, inorganic and organic suspended matter,

salinity, and temperature. Concurrently, diffuse downwelling attenuation of photosynthetically active radiation (PAR) was determined through water column profiles of photosynthetic photon flux density (PPFD) measured with a LI-COR, LI-192 underwater cosine corrected sensor. Additionally, underwater PPFD was measured continuously from August, 1986, to September, 1987, at the Gloucester Point and **Claybank** sites using arrays of two underwater cosine corrected sensors placed vertically a fixed distance apart. The sensors were cleaned frequently and the measured PPFD corrected for fouling by assuming a linear rate of light reduction due to fouling between cleanings.

RESULTS AND DISCUSSION: Since 1979 there have been no successful long-term transplants of Z. marina at **dieback** sites at or upriver of the **Mumfort** Island area. In contrast, beginning at Gloucester Point, the most upriver location where natural stands remain, the transplants have always been successful. This suggests that lack of regrowth in the **dieback** areas has not simply been due to lack of propagules but rather to environmental conditions at the sites. Plants transplanted at all the sites did well after some initial losses due to wave scouring or burrowing activities of fish and crustaceans. However, beginning in the spring of each year, planting units at the upriver sites died out with no survival past mid to late summer. Long-term survival of planting units at the downriver, naturally vegetated sites was approximately 60 percent.

A bimodal seasonal pattern of aboveground growth was observed in Z. marina with highest growth rates in the spring and a second

period of increased growth in the fall (Fig. 2). Significant differences in growth rates between the Gloucester Point and **Claybank** sites were observed only during these spring and fall periods ( $p < .05$ ). Transplant rhizome growth followed a similar pattern and was comparable at both sites from November to March ( $p < .05$ ), with rates at both sites quite low. Beginning in March, rates were greater at Gloucester Point ( $p < .05$ ), with maximum growth observed between March and May.

The lack of significant differences in growth of transplants at the **dieback** and vegetated sites during the summer and winter may have likely been due to extremes in the water temperature which limited production during these periods. This suggests that although the summer is stressful to the seagrasses and they, in fact, die out at upriver sites during this period as leaf loss exceeds leaf production, adequate growth during the spring or fall may be critical to their summertime survival. Therefore any major environmental differences between sites which results in differing potential to support vegetation, may likely occur during critical spring or fall periods.

Because of a general lack of information regarding temporal relationships between levels of environmental factors and macrophyte growth and survival, the environmental data was aggregated to seasonal averages. Seasonal characterization of Z. marina was determined by relating aboveground plant growth to water temperature. This permitted relationships to be developed between plant response and environmental conditions that were based upon

plant-derived seasonal growth patterns. To accomplish this, the 0°C to 30°C and 30°C to 0°C periods in the annual temperature cycle were treated independently. For each temperature period, unique regressions were fit to both the increasing and decreasing portions of the growth curve using log rate vs. inverse temperature transformations. The two resultant equations for each temperature period were solved for the **maximum** growth rate and inflection temperature, and the temperature cutoffs at which growth equals 50% of this maximum rate were determined. The annual temperature cycle was therefore divided into four distinct, biologically determined seasons that reflect the bimodal growth pattern of Z. marina, characteristic of the polyhaline region of the bay. These temperature-derived seasons (spring 9-23°C; summer 23-30-25°C; winter 13-0-9°C) were then be used to compare water quality parameters for the individual stations.

Enriched levels of inorganic nutrients can promote increased epiphytic loading on **seagrass** leaves, reduction of the light and carbon at leaf surfaces, and resultant decreases in production. A two-way **ANOVA** with main effects of site and season was used to identify differences in yearly seasonal averages of inorganic nutrients from different sites. Essentially, nutrient levels rose earlier and maintained higher levels longer each fall and winter at upriver sites when compared to downriver areas. For example, this resulted in significantly higher ( $p < .05$ ) levels at the **dieback** site at **Claybank** during these seasons of most years when compared to the surviving sites at Gloucester Point or Guinea Marsh. Overall

seasonal means for the 1984-1989 period (Fig.3) illustrate the pattern for inorganic nitrogen. Inorganic phosphorus was generally higher upriver during all seasons at the **dieback** sites (Fig.4). However, N:P ratios suggest that nitrogen should be limiting for epiphyte growth during much of the year except during the late fall and winter when inorganic nitrogen levels increase to highest levels.

In the transplant experiments a relationship between increased epiphyte loading and in situ nutrient levels was not apparent. During the fall and winter, epiphyte levels were usually higher downriver than upriver. There was a general increase in epiphyte levels with distance upriver during the late spring, which may have influenced the differential patterns of **seagrass** production and survival. **Wetzel** and Neckles (1986) suggested that epiphyte accumulation had little effect on **seagrass** survival under average light levels, however they predicted that under conditions of high water column light attenuation, relatively small changes in epiphyte loading would have dramatic effects. Epiphytic loading is controlled by many physical, chemical and biological processes in addition to nutrient supply, not the least of which is epiphytic grazing rates, that can vary both spatially and temporally. Therefore, that differences were not observed in the field is not unusual. Further work to be presented by Neckles in this workshop suggests from controlled microcosm experiments and model simulations, that nutrient enrichment can, in fact, increase the **seagrass** community's response to increased water column light

attenuation, resulting in decreased long-term stability.

Given an adequate invertebrate epiphyte grazer community, model studies have suggested that light availability through the water column is the principal factor limiting **seagrass** growth in this region (Wetzel and **Neckles** 1986). Turbidity in the York River generally increases with distance upriver, paralleling that observed for total suspended solids with higher levels most pronounced upriver during the spring. **Stepwise** multiple regression of total inorganic matter (TIM), total organic matter (TOM) and chlorophyll on Kd demonstrates that TOM and chlorophyll on Kd demonstrates that TOM and chlorophyll a add only a small increment to the variation in Kd explained by TIM. Therefore phytoplankton or phytoplankton derived material in the water column likely play a much smaller role in blocking sunlight from the **seagrass** than do the suspended silts and clays. An estimation of percent total attenuated light due to phytoplankton, as calculated from chlorophyll a concentrations (after Bannister, 1974), indicate that, in general, the values are less than 20 percent. This suggest that river-born sediments are the principle component of turbidity in this region.

Determination of light requirements for long-term **seagrass** growth and survival from field studies in this region may be approached in several ways. If one assumes nominal epiphyte populations in all areas , then light levels at the deepest limits of **seagrass** growth in natural areas **should represent** a long-term minimum. Studies of **seagrass** growth in the Guinea marsh region at

the mouth of the York by Orth and Moore (1988) reveal this depth to be approximately **1.5m** below mean sea level (MSL) for Z. marina. Seasonal PPFD intensities of PAR at this depth as determined from biweekly monitoring from 1984 to 1989 are presented in Fig.5. These data indicate seasonal mean light levels during the summer and fall of approximately 20 percent, with increasing transparency in the winter and spring to 25 and 30 percent, respectively.

The differential growth and survival of Z. marina transplants also provide insight as to light requirements of this temperate species. Fig.6 presents the five year, seasonal means of PAR at the transplant depth of 1.0 m MSL for three York River sites. At this depth transplants survive at Guinea Marsh and Gloucester Point but die by the end of summer at Claybank, with reduced growth during the spring and fall. Turbidity levels are greatest in the summer at all sites, although light penetration is still **quite** high at the mouth of the York at Guinea Marsh. Average light at Gloucester Point is approximately 20 percent of surface with **Claybank** approximately 14 percent. However, this difference is not consistent between these two sites from year to year and during some summers light levels are equal. Transparency at all sites increases during the fall and winter, although the **Claybank** site continues to be relatively more turbid than the surviving downriver sites. Lower growth during the fall at **Claybank** relative to Gloucester Point correlates well with reduced light levels. Light levels at **Claybank** transplant depth, although 22 percent of the surface, are still well below the over 30 percent observed at the

surviving areas. During the critical spring period, transparency at the downriver, vegetated sites remains high, while the precipitous increase in turbidity at **Claybank** results in significantly lower light levels **equal** to only 10 percent of surface. Mean light levels of the Gloucester Point transplants are **equal** to or greater than levels at the compensation depth (1.5m MSL) of **seagrass** growth at Guinea Marsh during all four seasons, while levels at **Claybank** are much below these during the spring and summer. Continuous light records at Gloucester Point and **Claybank** during 1986-87 generally paralleled the patterns observed in the biweekly sampling, with significantly lower ( $P > .05$ ) mean seasonal light levels at **Claybank** during both the spring and fall with no significant differences during the summer or winter.

The minimum seasonal light penetration observed at Gloucester Point during any season (spring or fall) when significant differences in transplant growth were observed between it and Claybank, may also be used to characterize the light requirements of Z. marina in this region. Along the upriver gradient of increasing turbidity observed in the York during these periods, Gloucester Point is the most upstream site of successful transplant survival. Therefore the maximum seasonal turbidity observed here may represent the extreme condition for one year under which the **seagrass** may survive. Although, certainly, high turbidity during the summer would also likely result in **seagrass dieoff**, given comparable levels of turbidity among the sites during the summer (a condition which was observed during a number of years), spring

or fall might be considered the critical periods determining **long-term** successful survival. This maximum seasonal turbidity was observed to be **Kd=2.0**, or 13.5 percent of insolation at the Gloucester Point transplant depth of **1.0m** MSL.

Each of these field derived, seasonal estimates of minimum light requirements of the temperate **seagrass** Z. marina (ranging from 13.5 to 20 percent) .are well above both the Florida transparency standard of 1 percent and the NTAC color criteria of 5 percent that have been used to determine the compensation depth of plant growth. These Chesapeake Bay data suggest that the standards are too low and the criteria would not be appropriate to predict long-term temperate **seagrass** survival. The duration and timing of variable water quality conditions on **seagrass** survival are as yet not well understood. Nutrient enrichment, which in many cases co-occurs with conditions of increasing turbidity, presents a confounding influence that has the potential to increase the relative effect of increasing particle loads on **seagrass** growth. As illustrated for the bimodal pattern of **seagrass** growth observed in the Chesapeake Bay region, the period critical to **seagrass** survival may not be the summer with its stressful high temperatures alone, but rather other periods when relative decreases in light availability may be greatest.

#### REFERENCES

Bannister, T.T. 1974. PRODUCTION EQUATIONS IN TERMS OF CHLOROPHYLL CONCENTRATION, QUANTUM YIELD, AND UPPER LIMITS TO PRODUCTION. *Limnol. Oceanogr.* 19 (1):1-12.

Fonseca, M.S., Kenworthy, W.J. and Thayer, G.W. 1982. A LOW-COST PLANTING TECHNIQUE FOR **EELGRASS** (Zostera marina L.). Coastal Engineering Tech. Aid No. 82-6. U.S. Army, Corps of Engineers, Coastal Engineering Research Center Ft. Belvoir, Va. **15pp.**

Fonseca, M.S., Kenworthy, W.J., Thayer, G.W., Heller, D.Y. and Cheap, K.M. 1985. TRANSPLANTING OF SEAGRASSES ZOSTERA MARINA AND HALODULE WRIGHTII FOR SEDIMENT STABILIZATION AND HABITAT DEVELOPMENT ON THE EAST COAST OF THE UNITED STATES. Technical Report El-85-9. U.S. Army engineer Waterways Experiment Station, Vicksburg, Miss. **64pp.**

Orth, R.J. and Moore, K.A. 1983. CHESAPEAKE BAY: AN UNPRECEDENTED DECLINE IN SUBMERGED AQUATIC VEGETATION. *Science* **222:51-53.**

Orth, R.J. and Moore, K.A. 1988. DISTRIBUTION OF ZOSTERA MARINA L. AND RUPPIA MARITIMA L. SENSU LATO, ALONG DEPTH GRADIENTS IN THE LOWER CHESAPEAKE BAY, U.S.A. *Aquat. Bot.*, 32: 291-305.

Sand-Jensen, K. 1975. BIOMASS, NET PRODUCTION AND GROWTH DYNAMICS IN A **EELGRASS** (ZOSTERA MARINA L.) POPULATION IN VELLERUP VIG, DENMARK. *Ophelia* **14:185-201.**

Wetzel, R.L. and Neckles, H.A. 1986. A MODEL OF ZOSTERA MARINA L. PHOTOSYNTHESIS AND GROWTH: SIMULATED EFFECTS OF SELECTED **PHYSICAL-CHEMICAL** VARIABLES AND BIOLOGICAL INTERACTIONS. *Aquat. Bot.* 26: 307-323.

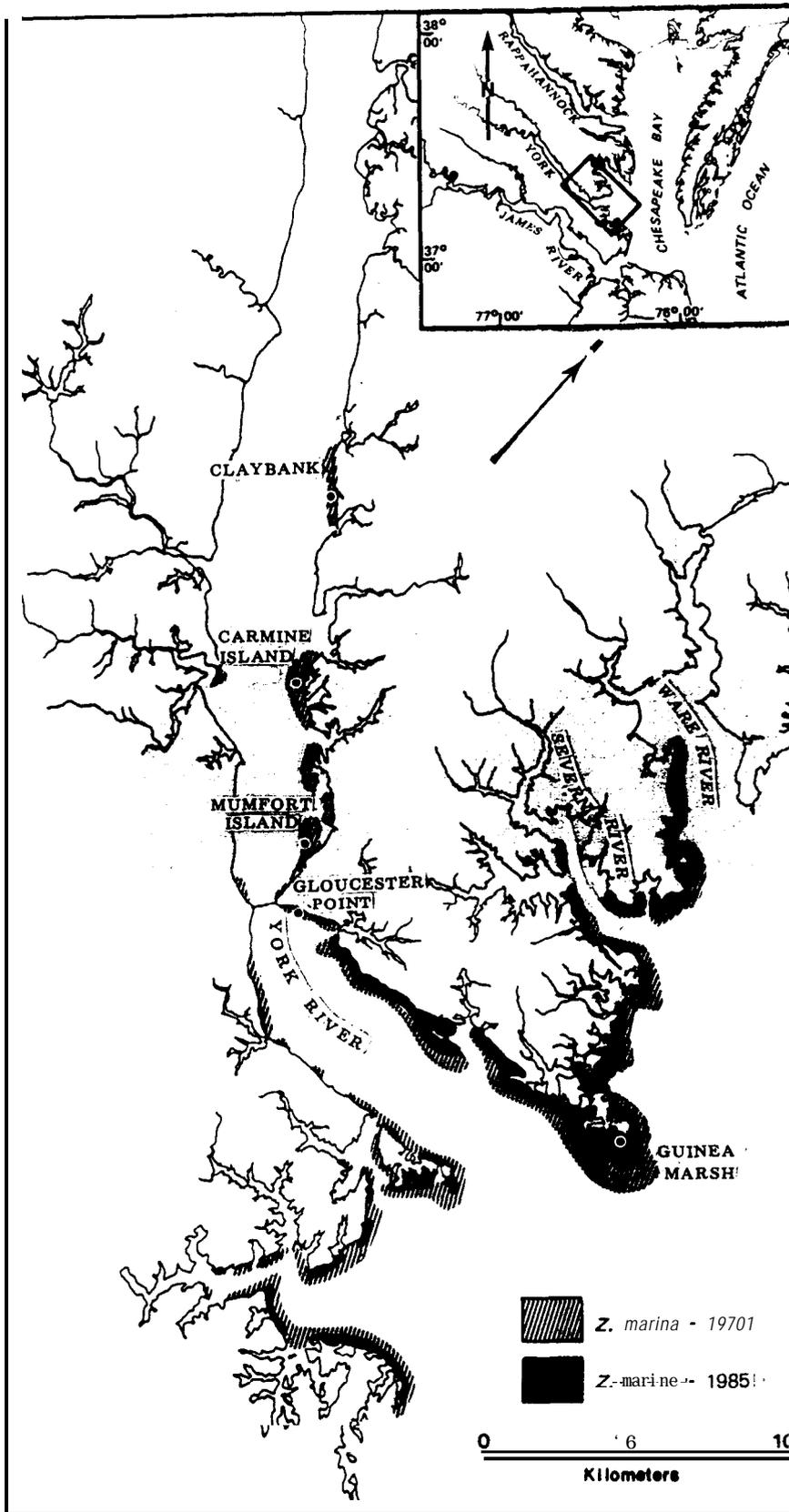
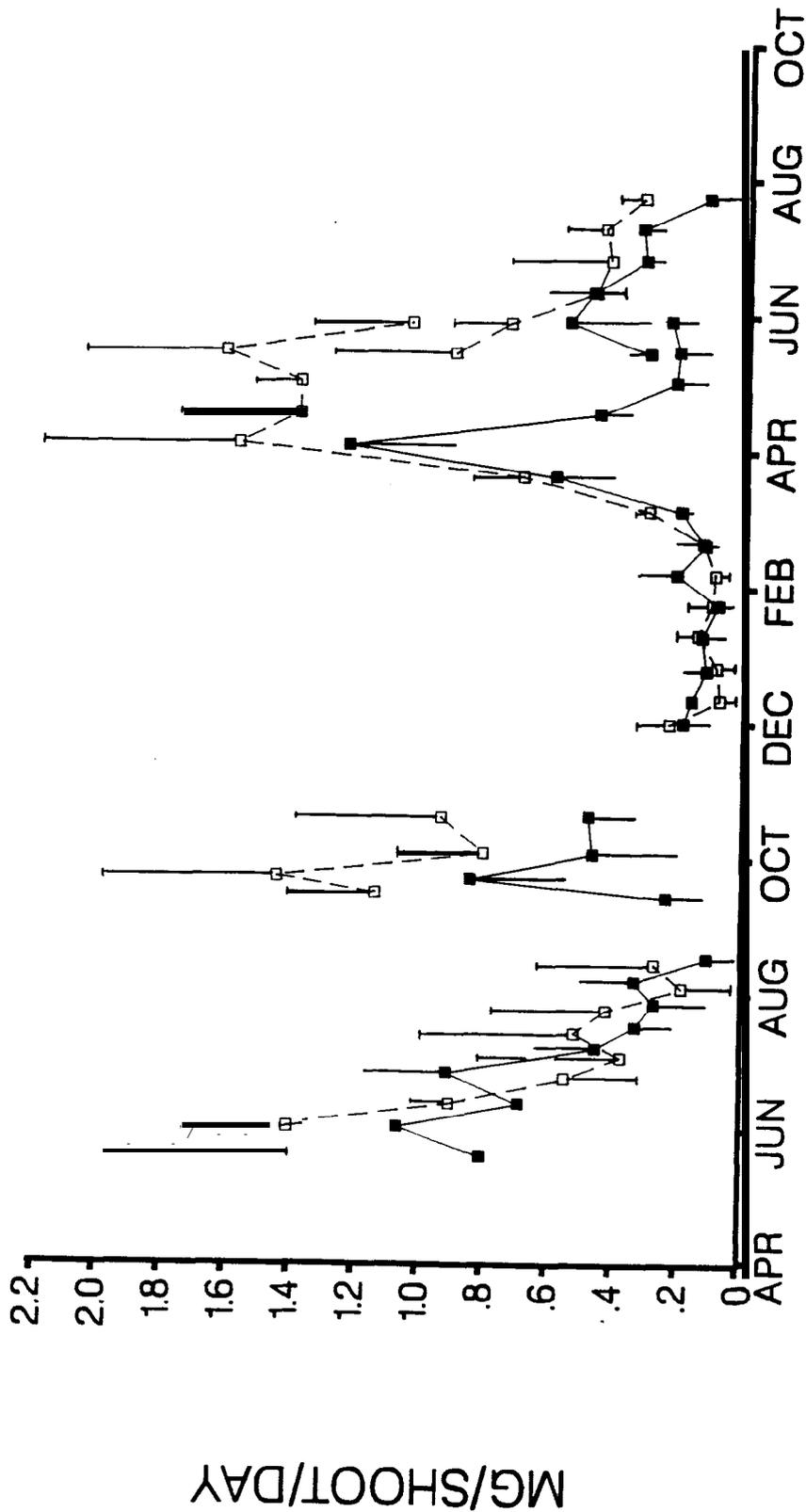


Figure 1: Map of study sites



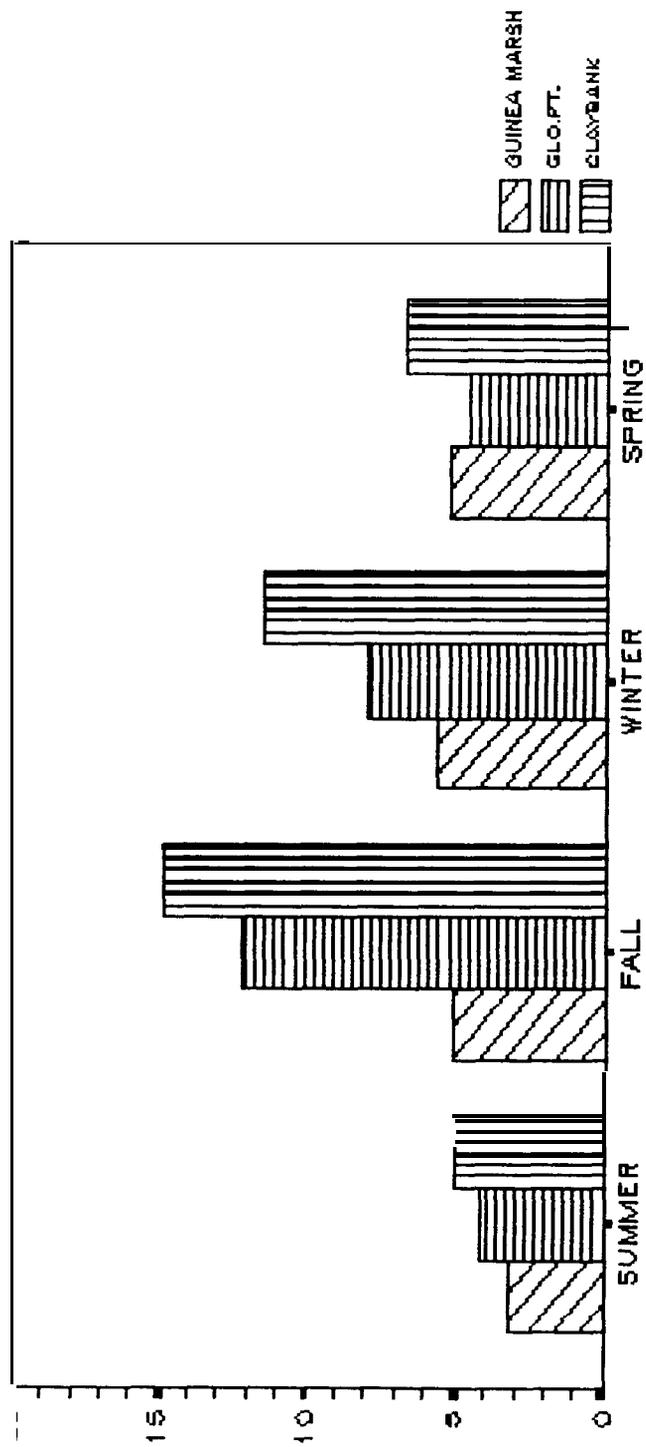
DATE (1985-1986)

Figure 2. Growth of *Zostera marina* transplants at Gloucester Point (□) and Claybank (■) as a function of time of year.

Figure 3: Seasonal averages of dissolved inorganic nitrogen at three sites.

### YORK RIVER - 1984-1989

#### DISSOLVED INORGANIC NITROGEN



UG-ATL

Figure 4: Seasonal averages of dissolved inorganic phosphorus at three sites.

### YORK RIVER - 1984-1989 DISSOLVED INORGANIC PHOSPHORUS

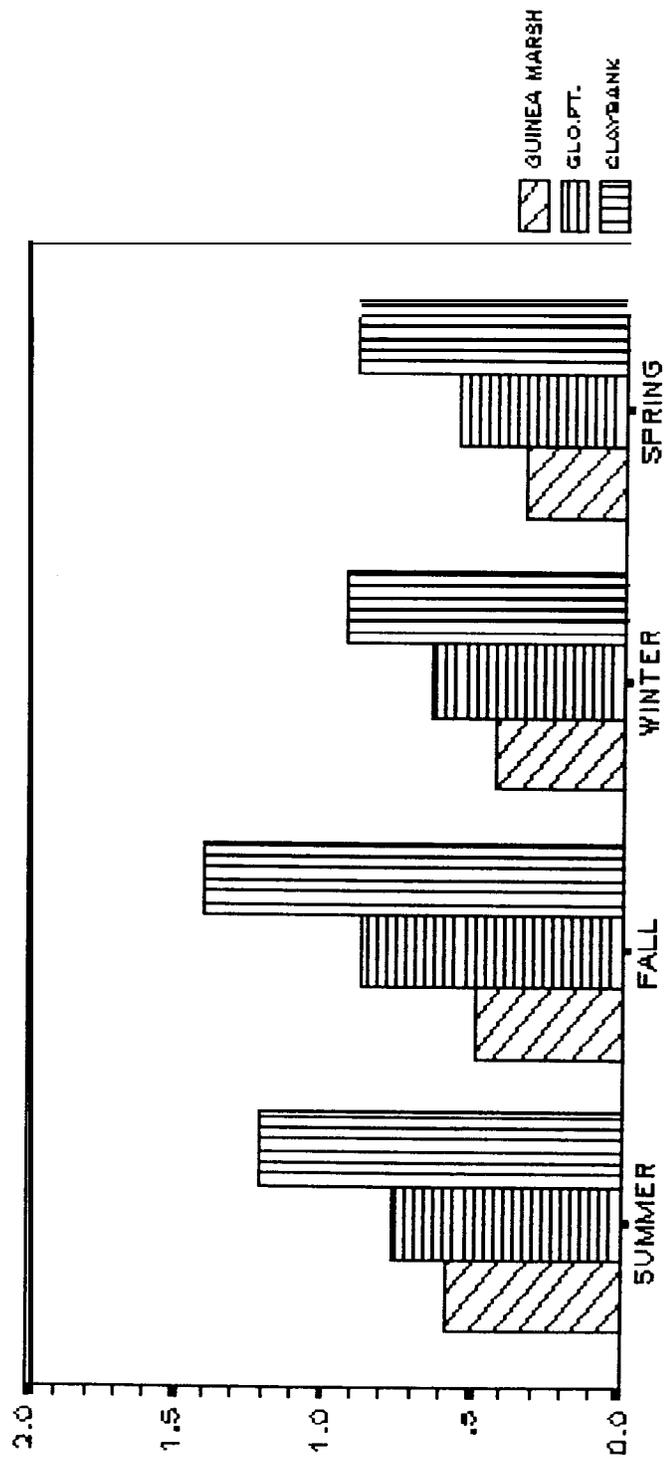


Figure 5: Percent of surface photosynthetically active radiation (PAR) reaching the lower depth limit of Zostera marina at Guinea marsh during four periods of the year.

GUINEA MARSH - 1984-1989  
LIGHT AT EELGRASS COMPENSATION DEPTH

(1.5M MSL)

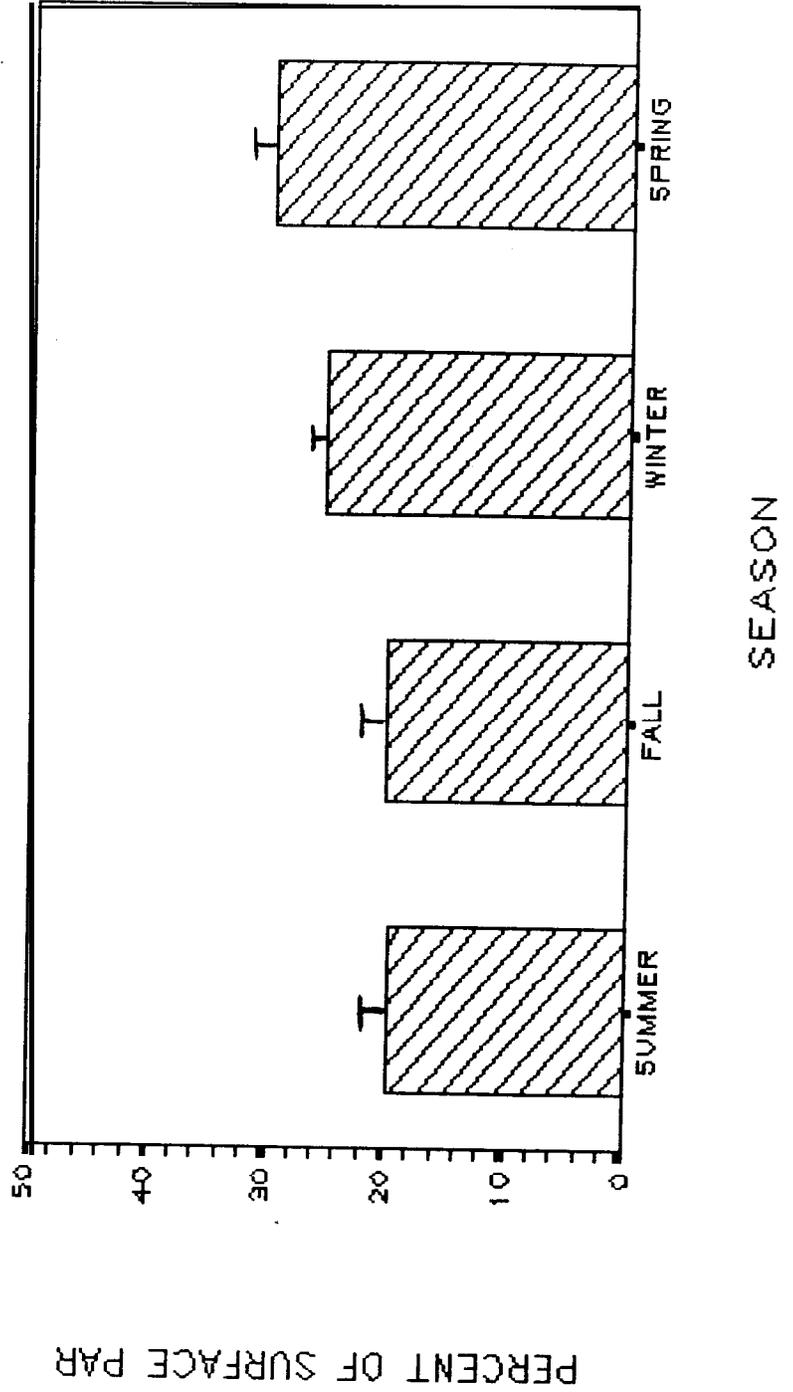
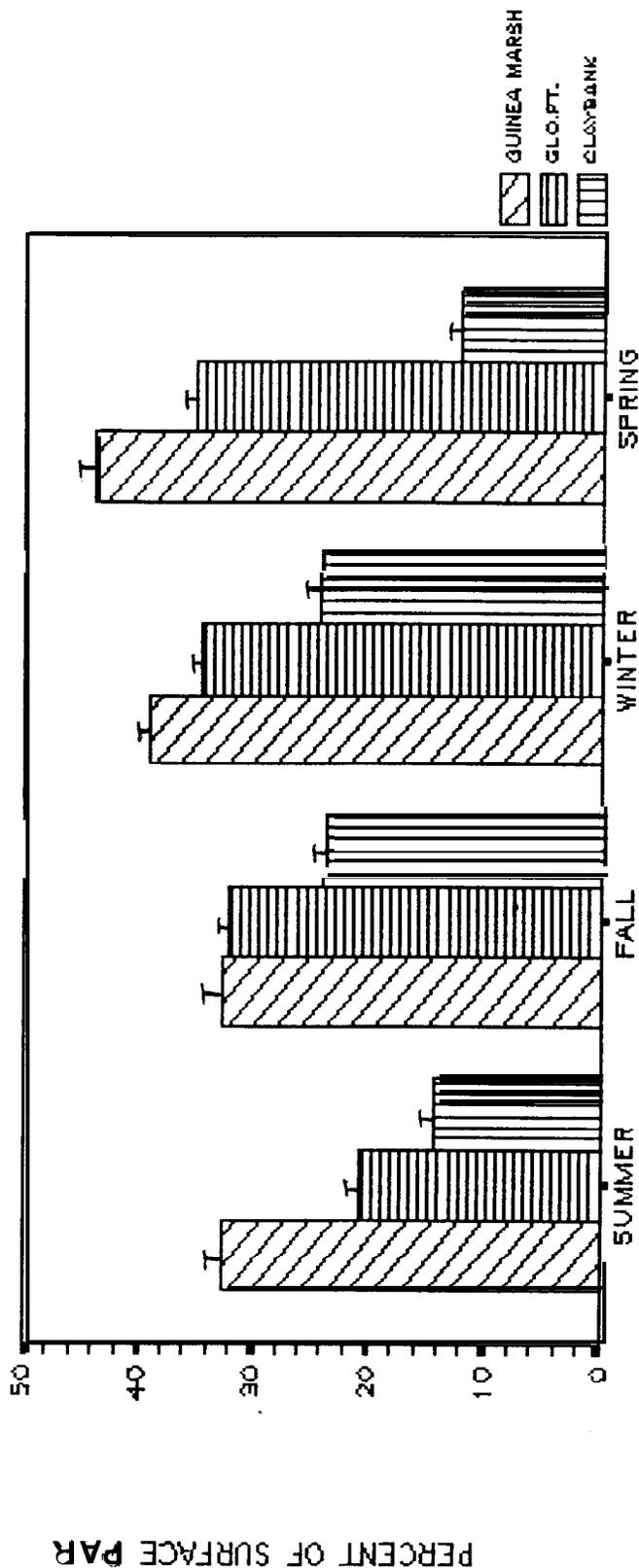


Figure 6: Percent of surface photosynthetically active radiation (PAR) reaching the 1.0 m depth at three sites during four periods of the year.

YORK RIVER - 1984-1989  
SEASONAL LIGHT PENETRATION (1.0M MSL)



## **SESSION 2; LIGHT REQUIREMENTS OF TROPICAL SEAGRASSES**

Fourqurean, James W. and Joseph C. Zieman, Department of Environmental Sciences, University of Virginia, Charlottesville, VA 22903.

### PHOTOSYNTHESIS, RESPIRATION AND WHOLE PLANT CARBON BUDGETS OF THALASSIA TESTUDINUM, HALODULE WRIGHTII AND SYRINGODIUM FILIFORME.

In the last 50 years, there has been a precipitous decline in the areal extent of **seagrass** beds in many parts of the world. Often these declines have been linked to reduction in water quality and increased turbidity, which decrease the amount of light available to benthic plants (Zieman 1975, Orth and Moore 1983, Cambridge and **McCoomb** 1984, Giesen et al. 1990, Larkum and West 1990). Reduced light levels may restrict seagrasses to shallower areas, since the depth limits of some seagrasses are considered to be controlled by the penetration of light in the water column (see **Dennison** 1987 for review). Because of the well documented importance of **seagrass** beds in the nearshore waters of Florida (Zieman 1982, Zieman and Zieman **1989**), the response of seagrasses to man-induced increases in turbidity and light attenuation must be understood to allow for effective management of the turbidity of the coastal zone.

The euphotic zone in oceanography is defined as the depth zone where adequate light is available to fuel photosynthetic rates sufficient to balance respiratory requirements. For **phytoplankton**-dominated systems, the euphotic zone penetrates from the surface to a depth at which roughly 1% of surface irradiance remains. Much

more light is required by seagrasses than phytoplankton, since the carbon balance of seagrasses is more complex than that of phytoplankton due to the increased structural complexity of seagrasses (Fourqurean and Zieman 1991). A large portion of the biomass of seagrasses is allocated to non-photosynthetic tissue that must be supported by the photosynthesis of the leaves. The apportionment of plant resources into leaves and non-photosynthetic tissues can vary not only between species, as is evident by the different growth forms of **seagrass** species, but also within a species.

In this study, we compare the whole plant carbon budget of the three major **seagrass** species in south Florida, Thalassia testudinum, Halodule wrightii and Syringodium filiforme using a technique developed for T. testudinum (Fourqurean and Zieman 1991). Laboratory measures of photosynthesis vs irradiance curves were generated using the oxygen evolution of intact shoots in sealed chambers at 25-30 °C. Apportionment of biomass into photosynthetic and non-photosynthetic tissues was measured in the field, and the respiration rates of the individual tissues were measured. The P/I curves and apportionment and respiration data were used to build carbon budgets for each species.

All three species showed typical light dependent, photosynthesis vs irradiance behavior, with no apparent photoinhibition at the light levels used in this study (0 - 1000  $\mu\text{E m}^{-2}\text{s}^{-1}$ ). Rates of photosynthesis varied between species, with H. wrightii exhibiting much higher photosynthetic rates at all light

intensities than T. testudinum and S. filiforme (Figure 1). For all species, variation in the measured rate increased as light intensity increased. For modelling purposes, a simple hyperbolic P/I model ( $P = P_{\max} \alpha I / (P_{\max} + \alpha I)$ , Burk and Lineweaver 1935) was fit to the data.

Core samples were collected in **seagrass** beds from various locations around the upper Florida Keys, and the living **seagrass** biomass was separated into four tissue types: green leaves, upright short shoots, rhizomes and roots. There were substantially different patterns of apportionment of biomass between the different species (Table 1). Photosynthetic leaves made up 15.0% +/- 1.0% of the total biomass of T. testudinum, 20.4% +/- 1.7% of H. wrishtii and 19.0% +/- 1.8% of S. filiforme. There was substantial variation within each species, with leaves comprising 5.6% to 29.9% of the total biomass of T. testudinum, 9.8% to 38% of H. wrishtii and 11.4% to 29.8% of S. filiforme. **Non-**photosynthetic structures account for 80% to 85% of total plant biomass, but their contribution to the respiratory demand of seagrasses has been largely ignored.

The respiration rate of each tissue type of each species was assessed by incubating tissue sections at 25-30 °C in stirred BOD bottles and measuring oxygen uptake rates. Within a species, there were significant differences in respiration rates between tissue types (Figure 2). In all three species, green leaves had the highest respiration rates, followed in descending order by roots, short shoots and rhizomes. There were differences between species

as well. For each tissue type, H. wrishtii always had the highest respiration rate, followed by S. filiforme. T. testudinum had the lowest respiration rates for each tissue. The relative contribution of each plant part to the total respiratory demand of the plant was computed for each species by multiplying the proportion of each tissue by the respiration rate of that tissue. Leaves accounted for 42.6% of total respiration for T. testudinum, 52.8% for H. wriahtii and 41.2% for S. filiforme.

A simple type of budget including photosynthesis and respiration may be made at the level of the leaves. The light level at which photosynthesis just balances leaf respiration is called the leaf compensation irradiance, or  $I_{\text{leaf}}$ . Compensation values for isolated leaf tissue have long been estimated for seagrasses, and range between 9 and 45  $\mu\text{E m}^{-2}\text{s}^{-1}$  (Dennison 1987, Dawes and Tomasko 1988). We found compensation irradiance values for leaves of Thalassia testudinum, Halodule wrishtii and Syringodium filiforme to be 15, 33 and 14  $\mu\text{E m}^{-2}\text{s}^{-1}$ , respectively. If surface irradiance at noon is roughly 2000  $\mu\text{E m}^{-2}\text{s}^{-1}$ , then these values indicate that **seagrass** leaves can maintain a positive carbon balance with light levels of about 0.5% to 1.5% of the maximum daily irradiance. The compensation irradiance for leaf tissue is of limited utility in predicting survival of seagrasses, however, since for most of the day surface irradiance is well below the maximum. Also, photosynthesis and respiration budgets for whole plants must take into account the respiratory demand of the **non-**photosynthetic parts of the plants. Predicting photosynthetic

requirements, based on the behavior of the leaves only, will severely underestimate the amount of light required to supply all of the energy needs of the entire plant.

The amount of light required for the photosynthesis of the leaves to balance the respiratory needs of the whole plant is the plant compensation irradiance, or  $I_{c \text{ plant}}$ . In calculating  $I_{c \text{ plant}}$ , oxygen uptake during respiration was converted to units of carbon using a respiratory quotient of 1 (RQ = moles C oxidized / moles  $O_2$  consumed) and oxygen released during photosynthesis was converted to units of carbon assuming a photosynthetic quotient of 1.2 (PQ = moles  $O_2$  evolved / moles C fixed). Using the mean apportionment of Florida Bay seagrasses into tissue types (Table 1) and the mean respiratory rates of each tissue (Figure 2), the mean  $I_{c \text{ plant}}$  values for Florida Bay seagrasses were 40, 65 and 35  $\mu E \text{ m}^{-2} \text{ s}^{-1}$  for Thalassia testudinum, Halodule wrightii and Syringodium filiforme, respectively. Note that these values are at least twice as great as the values of  $I_{c \text{ leaf}}$  given above.

The relative apportionment of **seagrass** biomass is not fixed at the mean values shown in Table 1, however. Variations in the relative importance of tissues may be caused by such environmental variables as nutrient availability or sediment type (Zieman, 1982). The effect of variation in the relative importance of the leaves to  $I_{c \text{ plant}}$  can be substantial (Figure 3). At the maximum leaf importance measured for Florida Bay Thalassia testudinum,  $I_{c \text{ plant}}$  was as low as 25  $\mu E \text{ m}^{-2} \text{ s}^{-1}$ , and at the minimum importance of leaves as high as 85  $\mu E \text{ m}^{-2} \text{ s}^{-1}$ . Similarly broad ranges apply to the  $I_{c \text{ plant}}$  of

Halodule wrightii and Syrinaodium filiforme.

The amount of light required to sustain seagrasses in an area will depend on many factors, including the relative apportionment of biomass into leaves and non-photosynthetic structures. This relative apportionment may be controlled by factors associated with sediment nutrient supply and stability (Zieman 1982). In order to properly apply this type of model to prediction of **seagrass** depth distribution, seasonal estimates of relative biomass apportionment must be made as well as measures of light availability. It is clear, however, that seagrasses require much more light than the 1% of surface irradiance required by planktonic primary producers. The values of  $I_{\text{plant}}$  calculated here indicate approximately 10 - 20 % of daily average surface irradiance must reach the bottom in order for seagrasses to maintain a net positive carbon balance.

REFERENCES

- Burk, D. and Lineweaver, H. 1935. THE KINETIC MECHANISMS OF PHOTOSYNTHESIS. Cold Spring Harbor Symp. quant. Biol. 3:165-183.
- Cambridge, M.L. and **McCoomb**, A.J. 1984. THE LOSS OF SEAGRASSES IN **COCKBURN** SOUND, WESTERN AUSTRALIA. I. THE TIME COURSE AND MAGNITUDE OF **SEAGRASS** DECLINE IN RELATION TO INDUSTRIAL DEVELOPMENT. Aquatic Botany **20:229-243**.
- Dawes, C.J. and D.A. Tomasko. 1988. DEPTH DISTRIBUTION OF THALASSIA TESTUDINUM IN TWO MEADOWS ON THE WEST COAST OF FLORIDA: A DIFFERENCE IN EFFECT OF LIGHT AVAILABILITY. P.S.Z.N.I.: Marine Ecology. **9(2):123-130**.
- Dennison, W.C. 1987. EFFECTS OF LIGHT ON **SEAGRASS** PHOTOSYNTHESIS, GROWTH AND DEPTH DISTRIBUTION. Aquatic Botany **27:15-26**.
- Fourqurean, J.W. and Zieman, J.C. 1991. PHOTOSYNTHESIS,

RESPIRATION AND THE WHOLE PLANT CARBON BUDGET OF THALASSIA TESTUDINUM. Marine Ecology - Prog. Ser. **69:161-170.**

Giesen, W.B.J.T., van Katwijk, M.M. and den Hartog, C. 1990. **EELGRASS** CONDITION AND TURBIDITY IN THE DUTCH **WADDEN** SEA. Aquatic Botany **37:71-85.**

Larkum, A.W.D. and West, R.J. 1990. LONG-TERM CHANGES OF **SEAGRASS** MEADOWS OF BOTANY BAY, AUSTRALIA. Aquatic Botany **37:55-70.**

Orth, R.J. and Moore, K.J. 1983. CHESAPEAKE BAY: AN UNPRECEDENTED DECLINE IN SUBMERGED AQUATIC VEGETATION. Science **222:51-53.**

Zieman, J.C. 1975. TROPICAL **SEAGRASS** ECOSYSTEMS AND POLLUTION. pp 63-74 in: Wood, E.J.F. and Johannes, R.E. (eds.) TROPICAL MARINE POLLUTION. Elsevier oceanography series 12. Elsevier, NY.

Zieman, J.C. 1982. THE ECOLOGY OF SEAGRASSES OF SOUTH FLORIDA: A COMMUNITY PROFILE. U.S. Fish and Wildl. Ser. **FWS/OBS-82/25.** 158 pp.

Zieman, J.C. and Zieman, R.T. 1989. THE ECOLOGY OF THE **SEAGRASS** MEADOWS OF THE WEST COAST OF FLORIDA: A COMMUNITY PROFILE. U.S. Fish and Wildl. Ser. Biol. Rep. **85(7.25).** 155 pp.

Table 1. Apportionment of total plant biomass into tissue types for Florida Bay seagrasses. Values are fraction of total living plant biomass.

	Leaf	Short Shoot	Rhizome	Root
<u>Thalassia testudinum</u>	0.15	0.40	0.35	0.10
<u>Halodule wrightii</u>	0.20	0.18	0.45	0.16
<u>Syringodium filiforme</u>	0.19	0.18	0.48	0.15

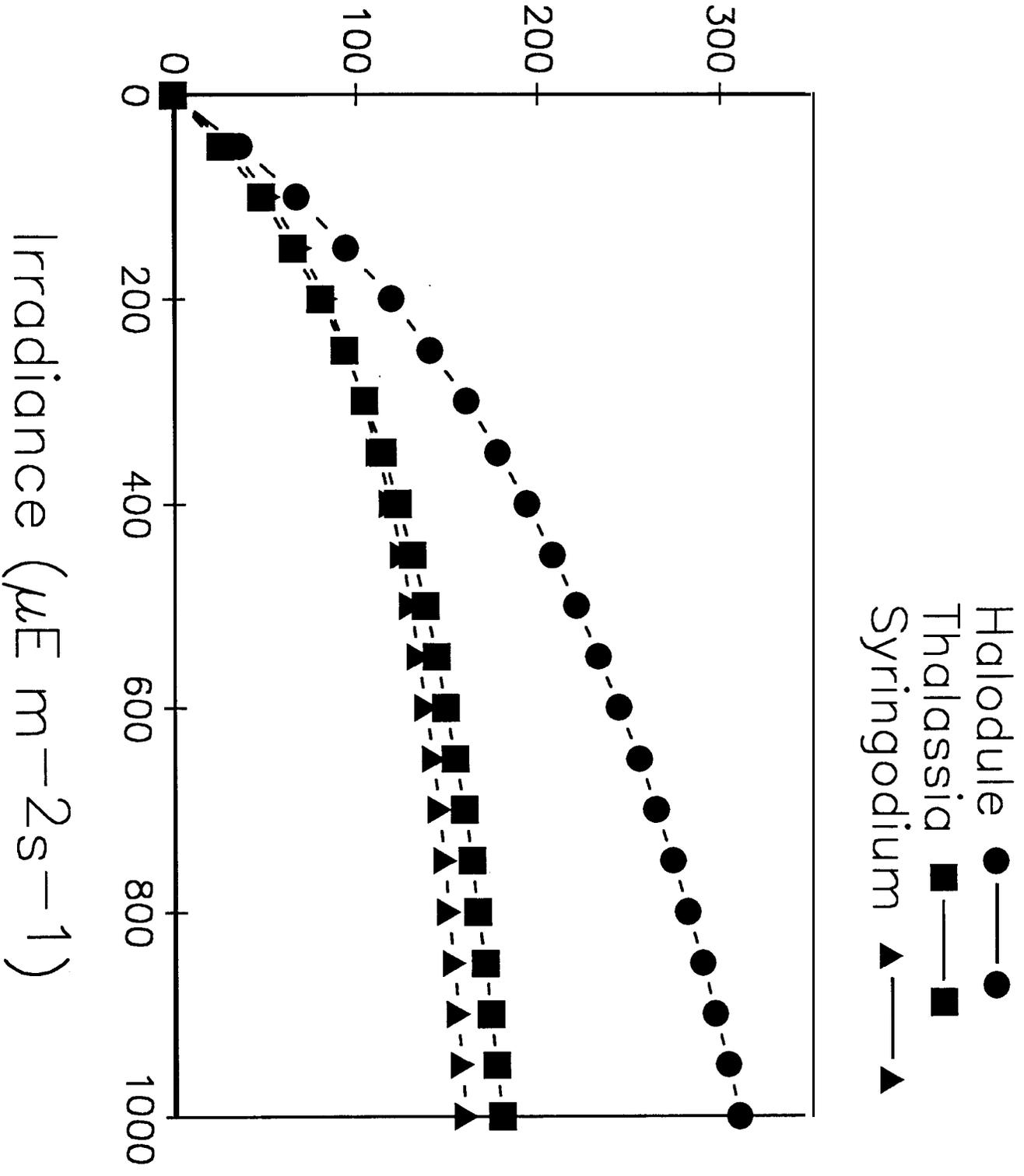
## FIGURE CAPTIONS

Figure 1. Best-fit hyperbolic curves for gross photosynthesis as a function of irradiance for intact shoots of Thalassia testudinum, Halodule wrightii and Syringodium filiforme.

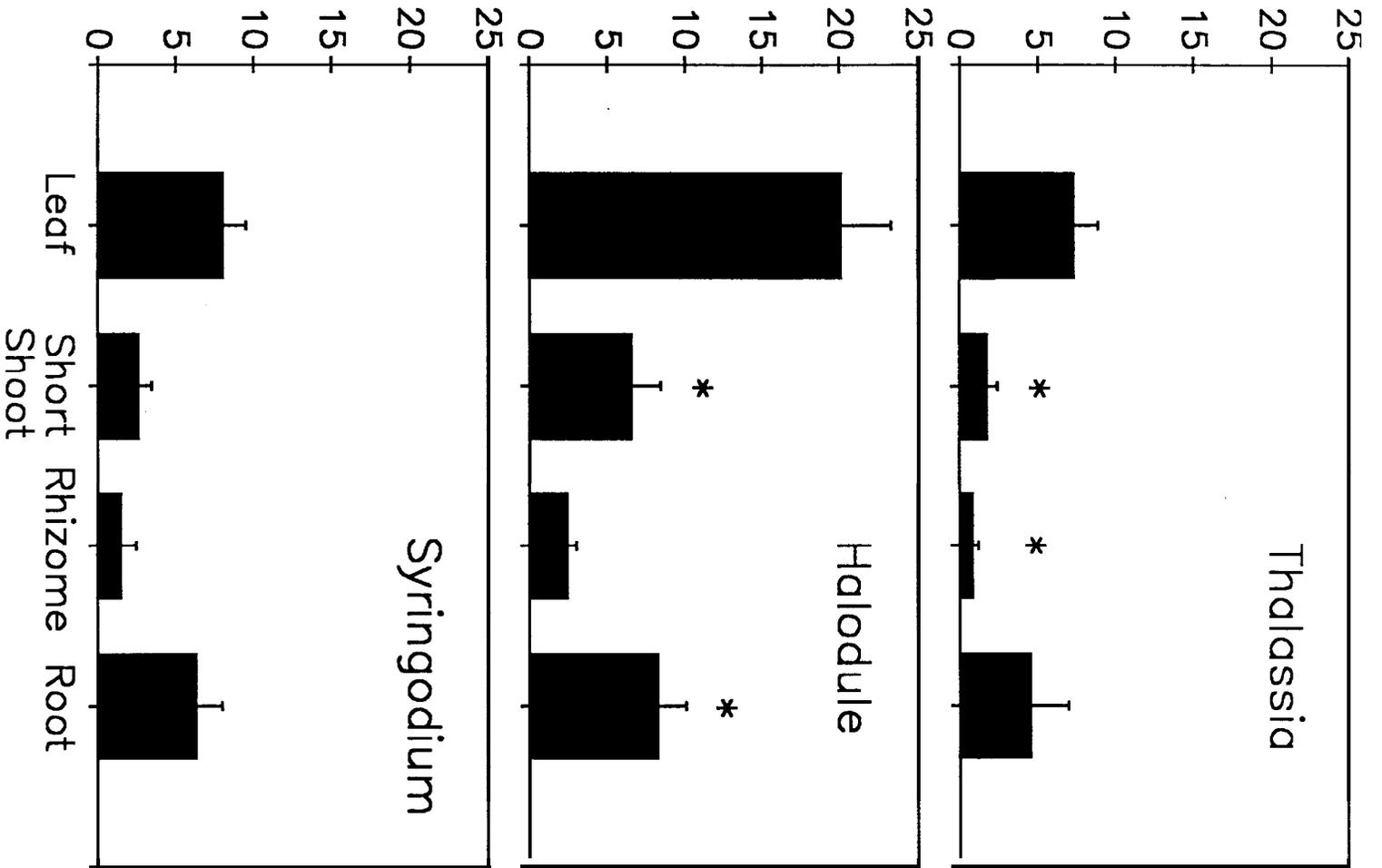
Figure 2. Respiration rates of tissues of the seagrasses Thalassia testudinum, Halodule wrightii, and Syringodium filiforme. Asterisks designate tissues within species not significantly different at  $p \leq 0.05$  (ANOVA).

Figure 3. Whole plant compensation irradiance as a function of the relative importance of leaf tissue to the total plant biomass.

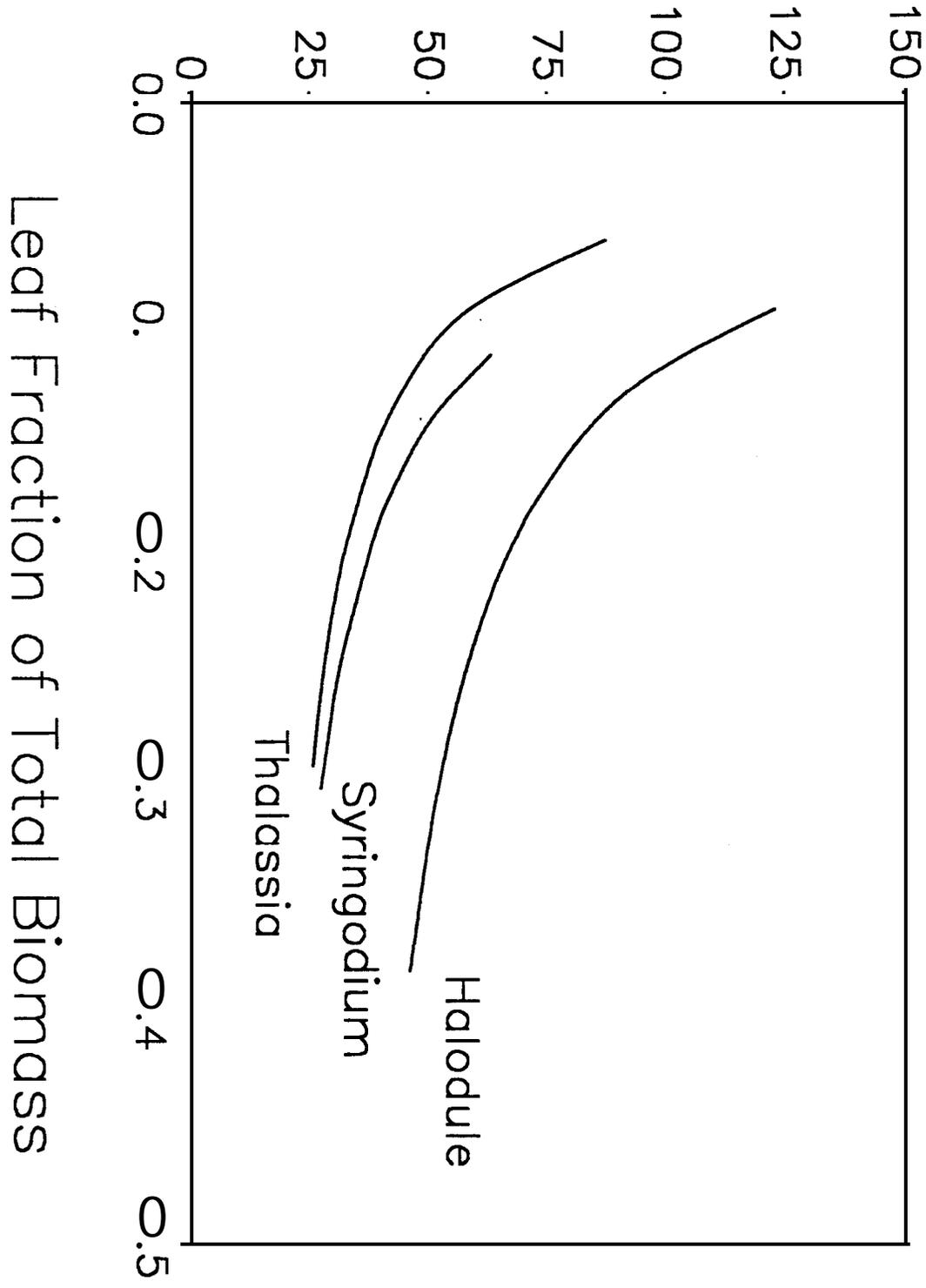
# Gross Photosynthesis ( $\mu\text{g O}_2 \text{gdw}^{-1} \text{min}^{-1}$ )



Respiration ( $\mu\text{g O}_2 \text{gdw}^{-1} \text{min}^{-1}$ )



$I_c$  plant ( $\mu E m^{-2} s^{-1}$ )



Dunton<sup>1</sup>, Kenneth H. and David A. Tomasko<sup>2</sup>, <sup>1</sup>Marine Science Institute, University of Texas at Austin, Port Aransas, Texas.  
<sup>2</sup>Florida Keys Land and Sea Trust, Marathon, Florida

SEASONAL VARIATIONS IN THE PHOTOSYNTHETIC PERFORMANCE OF HALODULE WRIGHTII MEASURED IN SITU IN LAGUNA MADRE, TEXAS.

Seasonal variations in the photosynthetic performance of the **seagrass** Halodule wrightii were determined from whole plants incubated in situ within a subtropical Texas **seagrass** community. We calculated photosynthesis-irradiance (P-I) curves from rates of oxygen evolution within four 5.0 L chambers placed on the seabed by divers. Oxygen measurements were collected continuously at 15-minute intervals using an ENDECO 1125 pulsed dissolved oxygen sensor controller. Underwater photosynthetically active radiation (PAR) was measured at one-minute intervals and integrated every 5 minutes on a continuous basis using a LI-193SA spherical quantum sensor which provided input to a LI-1000 datalogger. The dry weight of the **seagrass** within each chamber was used in the rate calculations of photosynthesis and respiration, expressed as  $\mu\text{mol O}_2$  evolved or consumed  $\text{mg}^{-1} \text{hr}^{-1}$ .

The respiratory demand of belowground tissue of Halodule wrightii (roots and rhizomes) and aboveground photosynthetic tissue (blades) was determined through dark bottle incubations. Initial and final dissolved oxygen levels were ascertained through Winkler titrations. The total respiratory demand of the plant, expressed as  $\mu\text{moles O}_2$   $\text{gdw (blade tissue)}^{-1} \text{hr}^{-1}$ , was determined from tissue respiration measurements and root-shoot ratios (RSR) obtained during each visit to the study site. Photosynthetic oxygen

evolution within each chamber was corrected for animal, bacterial, and chemical oxygen demand via dark night-time incubations conducted the previous evening. Photosynthetic contributions from other plant sources were not significant. Algal epiphytes were largely absent and were minimized through removal of the top few cm of blade tissue 2-3 weeks prior to the experimental work. Macroalgae were also thoroughly raked from the experimental area and chambers were always checked to insure the absence of macroalgae before the start of each incubation. No corrections for phytoplankton production within the chambers was necessary as their overall contribution within each chamber seldom exceeded  $\pm 5 \mu\text{mol O}_2 \text{ hr}^{-1}$ .

P-I parameters, including photosynthetic capacity ( $P_{\text{max}}$ ), photosynthetic quantum efficiency ( $\alpha$ , the moles of inorganic carbon fixed [or oxygen evolved] per mole of PAR absorbed) was estimated simultaneously using a **derivative-free** algorithm as evaluated by the hyperbolic tangent model of Jassby and **Platt** (1976). Saturating irradiance ( $I_k$ ) was calculated as  $P_{\text{max}}/\alpha$ ; whole plant compensation irradiance ( $I_{\text{cp}}$ ) was determined from estimates of plant (blade and corresponding root-rhizome) respiration using the hyperbolic tangent function.

The measurement of oxygen evolution in measuring photosynthetic production in seagrasses has sometimes been criticized because of lag problems involving oxygen storage in the **lacunae**. However, continuous daytime measurements of dissolved inorganic carbon (DIC) and dissolved oxygen within a chamber

(Fig.1) did not reveal any lag effect with respect to oxygen. Moreover, the total net molar change in both oxygen and carbon was nearly equivalent, indicating a P:Q of approximating unity. Based on these data, we believe that the oxygen technique used here is a reliable and accurate reflection of photosynthesis production.

Results of seasonal measurements in root/rhizome and blade respiration are shown in Figure 2. No clear pattern in these variations could be detected, but blade respiration usually ranged from 40 to 80  $\mu\text{mol O}_2 \text{gdw}^{-1} \text{hr}^{-1}$ , compared to only 10 to 40  $\mu\text{mol O}_2 \text{gdw}^{-1} \text{hr}^{-1}$  for roots and rhizomes. However, because root/rhizomes biomass was 2 to 3 times greater than that of blades, its contribution to the plants total respiratory demand was usually higher (Fig. 3).

Seasonal changes in the P-I parameters of Halodule wrightii, based on our in-situ measurements, are shown in table 1. Average  $I_k$  was 308.2  $\mu\text{mol m}^{-2} \text{s}^{-1}$ , compared to  $I_{cp}$ , which was 72.7  $\mu\text{mol m}^{-2} \text{s}^{-1}$ . Seasonal variation in  $I_k$  and  $I_{cp}$  are shown in figure 4 in relation to both temperature and porewater  $\text{NH}_4^+$ . No obvious relationship is apparent, although the variation in  $I_k$  may be following an annual cycle, reflective of some type of endogenous rhythm which we may be able to confirm over the next several months. In contrast, values for  $I_{cp}$  were remarkably consistent over the past year, ranging between 28 and 109  $\mu\text{mol m}^{-2} \text{s}^{-1}$ . With the exception of July 1989,  $\alpha$  values ranged between 0.6 and 1.6, reflecting the importance of light levels that are well above  $I_{cp}$  for maintaining at least moderate levels of plant production.

Table 1. Seasonal variation in the P-I parameters of *Halodule wrightii*, based on in situ measurements of oxygen evolution.

DATE	$P_{\max}^1$	$\alpha^2$	$I_{cp}^3$	$I_k^3$
May 1989	533.41	1.52	108.70	350.90
July 1989	707.31	5.56	27.66	127.21
Sept. 1989	426.85	1.58	59.05	270.16
Nov. 1989	385.91	1.12	87.73	344.56
Jan. 1990	1104.65	2.44	55.17	452.72
Mar. 1990	--	0.65	235.14	--
May 1990	222.72	0.61	75.97	365.10
July 1990	313.44	1.27	94.39	246.80

<sup>1</sup>Expressed  $\mu\text{mol oxygen gdw}^{-1} \text{hr}^{-1}$ .

<sup>2</sup>Expressed as  $\mu\text{mol oxygen gdw}^{-1} \text{hr}^{-1} / \mu\text{mol photons m}^{-2} \text{s}^{-1}$ .

<sup>3</sup>Expressed as  $\mu\text{mol photons m}^{-2} \text{sec}^{-1}$ .

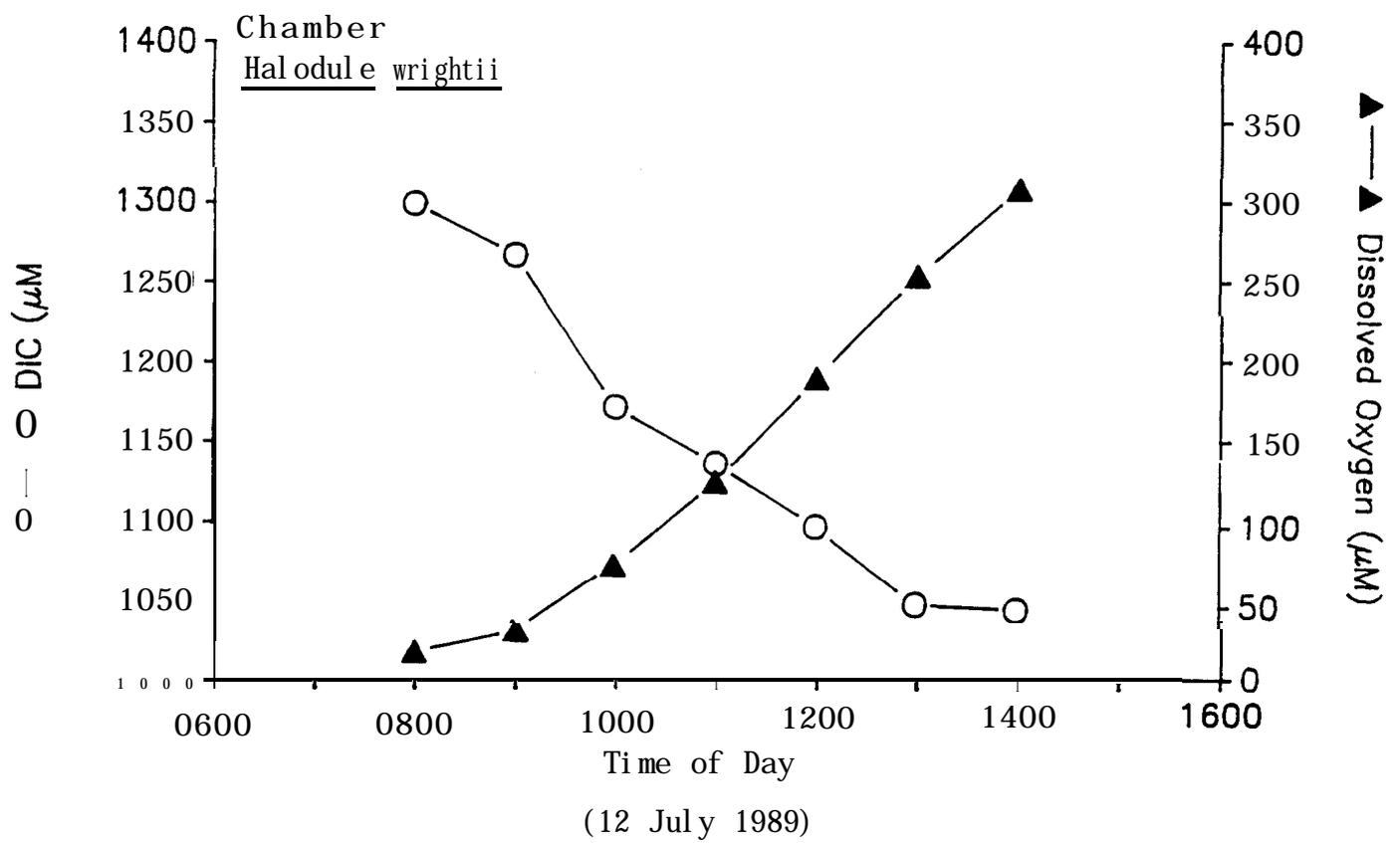


Figure 1. Daytime variation in dissolved inorganic carbon (DIC) and oxygen levels within a chamber containing *Halodule wrightii*.

Halodule wrightii

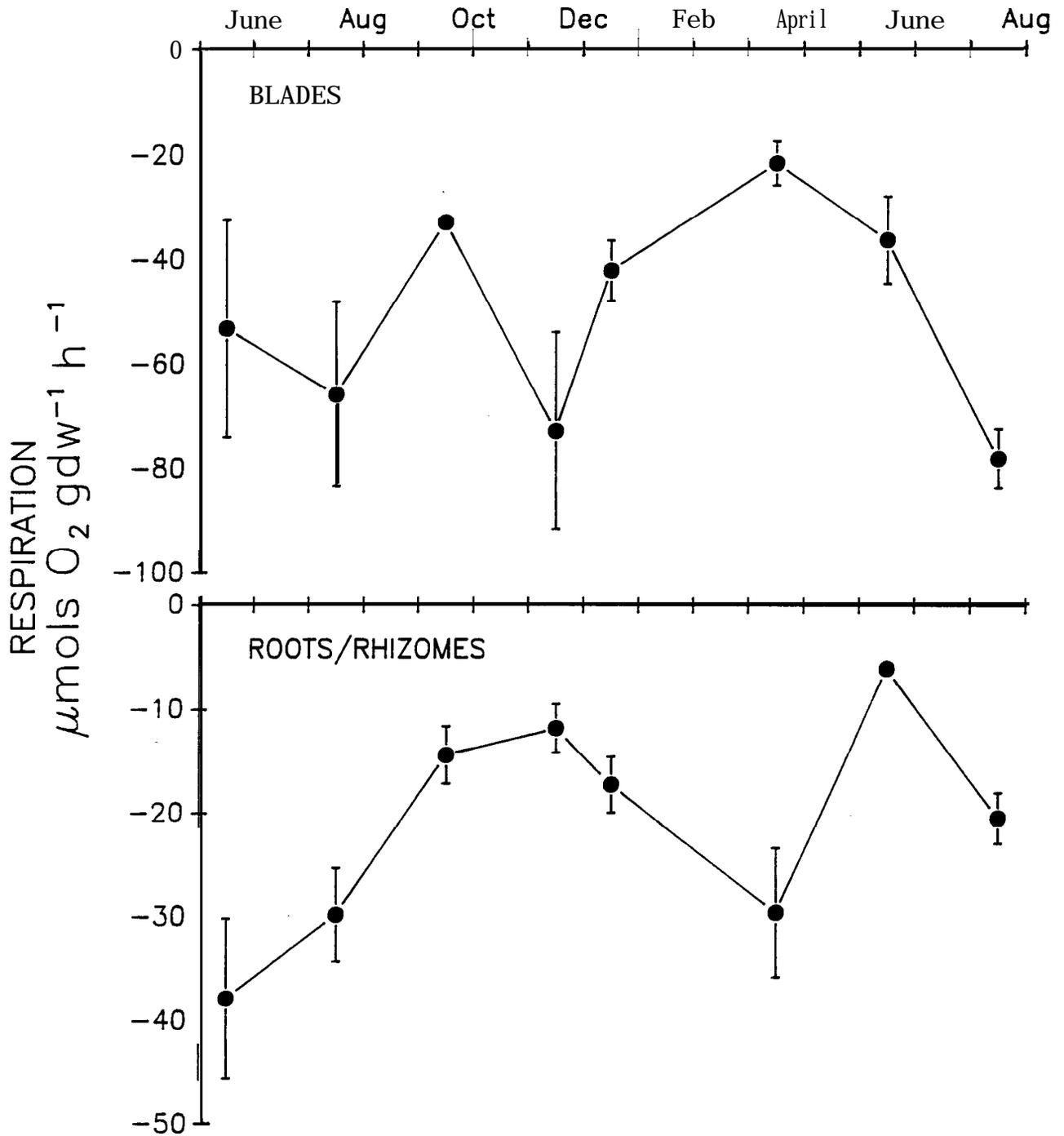


Figure 2. Seasonal variations in root/rhizome and blade respiration in *Halodule wrightii*.

Values  $\bar{x} \pm SE$  (n = 4).

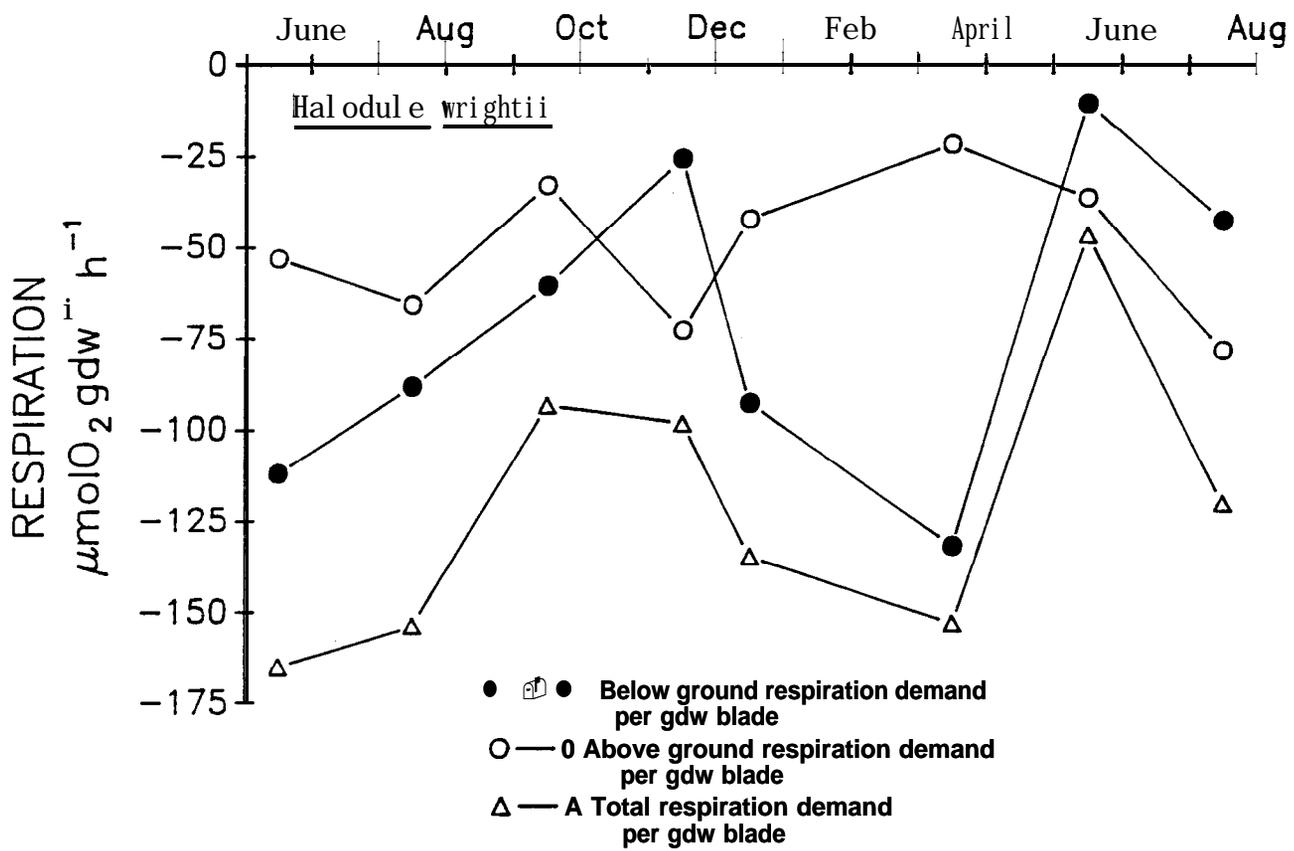


Figure 3. Seasonal variations in root/rhizome, blade and total respiration in *Halodule wrightii* corrected to mg dry wt of blade tissue. The variation in the contribution of root/rhizome oxygen consumption to the total plant respiratory demand is related to seasonal changes in the RSR ratio.

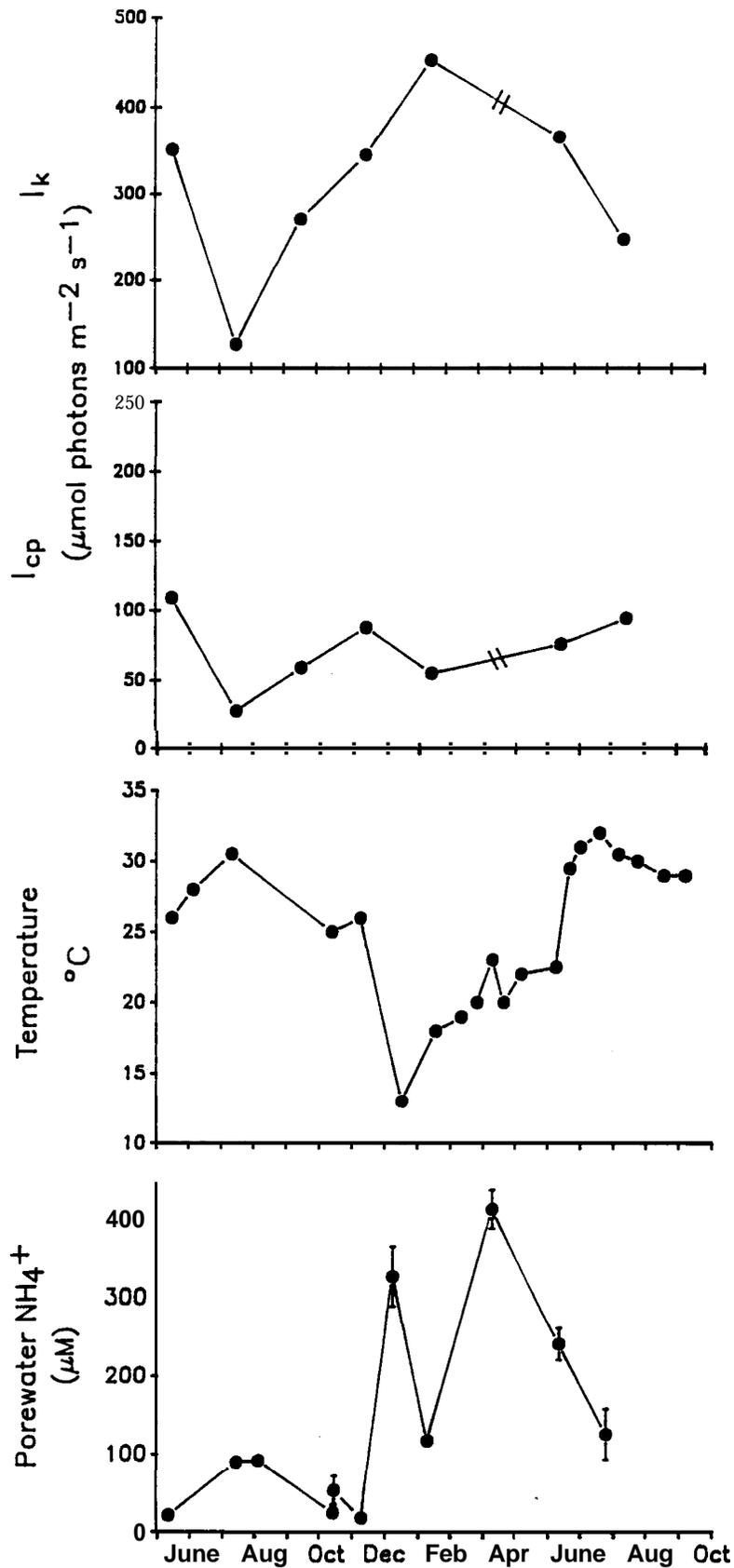


Figure 4. Variations in  $I_k$  and  $I_{cp}$  in relation to temperature and porewater  $NH_4^+$  in *Halodule wrightii* based on in situ measurements of blade photosynthesis and total plant respiration.

Tomasko', David A. and **Dunton**<sup>2</sup>, Kenneth H., 'Florida Keys Land and Sea Trust, Marathon, FL. <sup>2</sup>**Marine** Science Institute, The University of Texas at Austin, Port Aransas, TX.

GROWTH AND PRODUCTION OF HALODULE WRIGHTII IN RELATION TO CONTINUOUS MEASUREMENTS OF UNDERWATER LIGHT LEVELS IN SOUTH TEXAS

Despite the significance of the light environment to **seagrass** biomass and productivity, little research has been done that directly examines the relationship between irradiance and biomass or irradiance and productivity. Much of this shortcoming can be explained by the difficulty involved with measuring underwater light. Even if the light environment could be adequately quantified, physiological responses of seagrasses to changes in their light environment vary sufficiently with time and space to minimize the inference space within which experimental conclusions can be expected to hold true.

The first obstacle faced when modelling **seagrass** responses to light is that of measuring underwater light. Secchi disk depths are commonly used to indirectly measure light penetration, but the relationship between light attenuation and secchi disk depth degenerates when "**color**" is a significant component of light attenuation. Chambers and **Prepas** (1988) found that for lakes of equal **trophic** state, light penetrates to greater depths when color is low, despite similar secchi disk depths. The inconstant relationship between secchi disk depth and light attenuation has spurred some researchers into modelling underwater irradiance based on combining regularly-measured light attenuation coefficients with continuous measurement of irradiance using land-based quantum

meters.

Problems with this second method of modelling underwater irradiance involve events which affect underwater light, but not light measured above the surface. Clear sunny days following the passage of a cold front may result in high irradiance measurements on land, but increased wave energy due to windy conditions would reflect more light from the air-water interface, suspend sediments in the water column, and thus reduce the available light underwater. Tidal state can dramatically alter light attenuation coefficients, but it is difficult to build tidal influences into underwater light models. Also, due to the effects of solar elevation on the balance between direct light and diffuse sky light, light "**tails off**" more rapidly underwater in the morning and evening than it does on land (Holmes and Klein, 1987).

These problems seem to point out the obvious: if one is interested in quantifying underwater light, one should then measure it. If direct measurements of underwater light can be made in arctic Alaska (**Dunton** and Jodwalis, 1988) and Laguna Madre, Texas, there seems to be no reason why it cannot be done in Florida waters. However, this problem solved still leaves the question of how to model **seagrass** responses to light. Since the early work of Drew (**1978**), several researchers have used various methods to determine **seagrass** photosynthetic rates as a function of irradiance (P-I curves). Even when techniques have remained similar, **seagrass** P-I curves have shown variability as a function of water depth (Drew, 1978; Dawes and Tomasko, 1988) and season (Drew, 1978;

Bulthuis, 1983; Libes, 1986). Thus, **seagrass** responses to seasonal and/or depth-related differences in the light environment would themselves vary with season and water depth.

This presentation involves a comparison of continuous measurements of underwater light with information on productivity and growth of Halodule wrightii in Laguna Madre, Texas. Problems and promises of this particular approach are discussed.

The relationship between underwater light and blade growth was complex, and varied with season. Maximum daily underwater irradiance (in  $\mu\text{moles photons m}^{-2} \text{s}^{-1}$ ) varied between less than 100 to over 1300. During the time period when data on underwater light and productivity were both collected, we have divided the data for site 153 into four time blocks; May through July of 1989, January through February of 1990, March through May of 1990, and June through July of 1990 (**Fig.1**).

A casual examination of the data would suggest that the difference in blade growth rates between May - July 1989 and January - February 1990 is due to the large decline in the percent of days when irradiance exceeded 1000  $\mu\text{moles photons m}^{-2} \text{s}^{-1}$ . However, the last two time blocks, March - May 1990 and June - July 1990, both have smaller percentages of days with high maximum daily irradiance than May - July 1989, yet blade growth rates are similar. Clearly, variation in the number of days when irradiance exceeds 1000  $\mu\text{moles photons m}^{-2} \text{s}^{-1}$  does not seem to dramatically affect blade growth rates. Low water temperatures are responsible for the low blade growth rates in January - February 1989, despite

higher light levels versus March - May 1990 and June - July 1990. If days of very high light ( $2\ 1000\ \mu\text{moles photons m}^{-2}\ \text{s}^{-1}$ ) do not seem to be important for productivity in Halodule wrightii, the question remains what level(s) of light are important?

Using data from the in situ P-I curves from **Dunton** and Tomasko (this volume), it is apparent that for Halodule wrightii, critical irradiances are lower than levels commonly measured in the field, but are higher than critical values calculated for other species of seagrasses. Values for  $I_c$  ( $P_c$ , divided by the initial slope of the P-I curve) ranged from a low of 130 to a high of 452  $\mu\text{moles photons m}^{-2}\ \text{s}^{-1}$ , with a yearly average of 308. This compares with the literature values of 70 to 166 (see Dennison, 1987 for review). The reasons for the higher values here versus those in the literature are unclear, but probably involve the uniqueness of the experimental approach of **Dunton** and Tomasko (this volume).

Compensation irradiance values ( $I_c$ ) ranged from 30 to 100  $\mu\text{moles photons m}^{-2}\ \text{s}^{-1}$ , with a yearly average of 73. Literature values range from 9 to 26 (Dennison, 1987). The higher values calculated in **Dunton** and Tomasko (this volume) are undoubtedly due to the inclusion of respiratory demands of non-photosynthetic material in the calculation of compensation irradiances.

In summation, critical values of irradiance for Halodule wrightii in Laguna Madre, Texas are intermediate between high values often encountered in the field, and low values given for the other species of seagrasses. Using an average surface irradiance value of 2000  $\mu\text{moles photons m}^{-2}\ \text{s}^{-1}$ , it would seem that Halodule

wrishtii requires at least 2 to 5 percent of the surface irradiance just to maintain a positive energy balance during daylight hours. Underwater irradiance must reach 7 to 23 percent irradiance for I, to be equaled or exceeded. Compensation irradiance values are the barest minimum light levels necessary for **seagrass** maintenance metabolism; they cannot sustain continued somatic growth. The fact that even these levels exceed state mandated values indicates the inadequacy of the present state standards for protecting seagrasses in Florida waters.

#### REFERENCES

Bulthuis, D.A. (1983). EFFECTS OF TEMPERATURE ON THE PHOTOSYNTHESIS-IRRADIANCE CURVE OF THE AUSTRALIAN SEAGRASS, HETEROZOSTERA TASMANICA Mar. Biol. Lett. 4: 47-57.

Chambers, P.A., **Prepas**, E.E. (1988). UNDERWATER SPECTRAL ATTENUATION AND ITS EFFECTS ON THE MAXIMUM DEPTH OF ANGIOSPERM COLONIZATION. Can. J. Fish. Aquat. Sci. 45: 1011-1017.

Dawes, C.J., Tomasko, D.A. (1988). DEPTH DISTRIBUTION OF THALASSIA TESTUDINUM IN TWO MEADOWS ON THE WEST COAST OF FLORIDA; A DIFFERENCE IN EFFECT OF LIGHT AVAILABILITY. P.S.Z.N.I. Mar. Ecol. 9: 123-130.

Dennison, W.C. (1987). EFFECTS OF LIGHT ON **SEAGRASS** PHOTOSYNTHESIS, GROWTH AND DEPTH DISTRIBUTION. Aquat. Bot. 27: 15-26.

Drew, E.A. (1978). FACTORS AFFECTING PHOTOSYNTHESIS AND ITS SEASONAL VARIATION IN THE **SEAGRASS** CYMODOCEA NODOSA (UCRIA) ASCHERS, AND POSIDONIA OCEANICA (L) DELILE IN THE MEDITERRANEAN. J Exp. Mar. Biol. Ecol. 31: 173-194.

**Dunton**, K.H., Jodwalis, C.M. (1988). PHOTOSYNTHESIS PERFORMANCE OF LAMINARIA SOLIDUNGULA MEASURED IN SITU IN THE ALASKAN HIGH ARCTIC. Mar. Biol. 98: 277-285.

Holmes, M.G., Klein, W.H. (1987). THE LIGHT AND TEMPERATURE ENVIRONMENTS. pp. 3-22. In: **R.M.M.** Crawford (Ed.) Plant Life in Aquatic and Amphibious Habitats. Blackwell Scientific Publications, Oxford.

Libes, M. (1986). PRODUCTIVITY-IRRADIANCE RELATIONSHIP OF POSIDONIA OCEANICA AND ITS EPIPHYTES. Aquat. Bot. 26: 285-306.

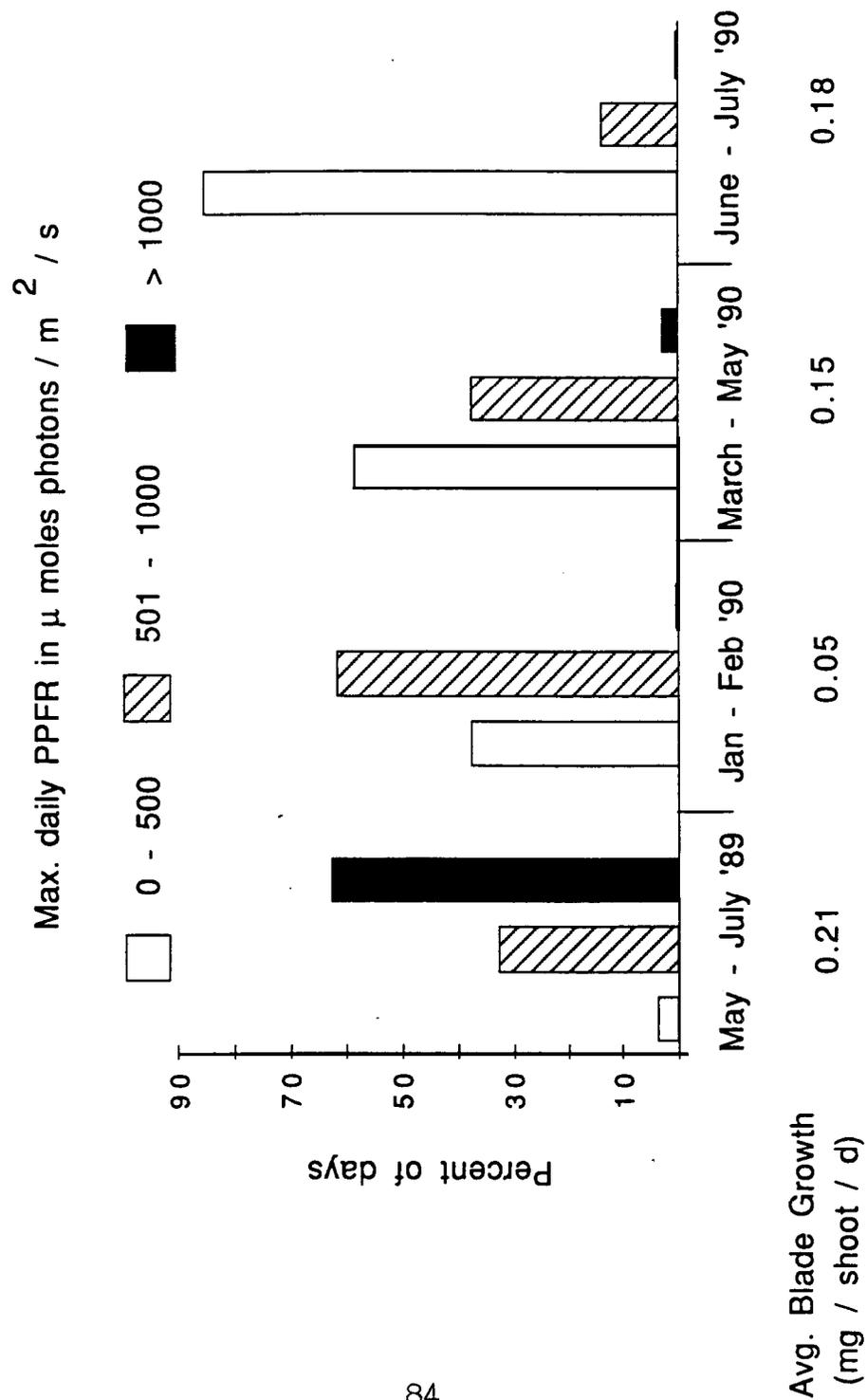


FIGURE 1

Hall, Margaret O., David A. Tomasko', and Francis X. Courtney, Florida Marine Research Institute, St. Petersburg, Florida. 'Florida Keys Land and Sea Trust, Marathon, Florida.

RESPONSES OF THALASSIA TESTUDINUM TO IN SITU LIGHT REDUCTION.

Historical records reveal that Florida, whose coastlines support some of the **world's** most extensive **seagrass** beds, has lost approximately one-third of the **seagrass** meadows that were present in the **1940's**. Reduction in the amount of light reaching **seagrass** blades is widely considered the major reason for **seagrass** decline in coastal waters. To test this idea, a study was designed to examine the effects of decreased light availability on the abundance, morphology, growth, and production of Thalassia testudinum (turtlegrass), the dominant **seagrass** in the Caribbean and the Gulf of Mexico. The study was conducted in a Thalassia meadow located near the mouth of Tampa Bay, Florida. Shoot density at this site decreases from the shallow to the deep margin, suggesting the lower edge of the meadow may be light limited.

Three shade and three control plots were established at both the shallow (0.75 m below MLW) and the deep (2 m below MLW) edges of the Thalassia bed in late February 1989. The amount of light reaching the **seagrass** was reduced in the field using shading screens constructed of neutral density nursery cloth attached to PVC frames that were 1.5 m on a side. Frames were held 0.5 m above the canopy with galvanized pipes. Ambient light reduction by shading screens averaged 60% near the shallow edge, and 65% at the deep margin. Fouled shading screens were replaced with clean ones

every one to two weeks, depending on how rapidly fouling occurred. Light was measured with a LICOR LI-1000 quantum meter and a 4 pi sensor under both clean and dirty screens, and in control plots whenever screens were changed.

Only the 1 m<sup>2</sup> areas in the centers of the 2.25 m<sup>2</sup> experimental plots were sampled to avoid possible edge effects. Experimental durations were 1, 3, 6, 9, and 13 months after the initiation of shading. A variety of parameters that might respond to shading were examined: shoot density, mean maximum blade length, leaf width, number of leaves per shoot, leaf area index, leaf relative growth rate, shoot turnover time, plastochrone interval, leaf production per shoot, leaf production per m<sup>2</sup>, and leaf biomass per m<sup>2</sup>. Data from each experimental duration were analyzed by one-way ANOVA, with Duncan's multiple range tests to isolate differences.

Shoot densities (Figure 1a) were generally higher in the shallow than the deep areas throughout the study. The response to shading was not rapid, and a clear trend of lower densities in shaded vs. control plots did not appear until 9 months **post**-shading. However, after 13 months of light reduction, density in the shallow shade treatment was 25% lower, and in the deep shade treatment over 60% lower than corresponding controls. Slower declines in Thalassia density in response to shading relative to seagrasses such as Zostera marina could be due to the larger amounts of below-ground reserves stored by turtlegrass.

One of our objectives was to determine if characteristics of the shallow edge of the **seagrass** bed would come to resemble those

of the deep margin if light was reduced. After 13 months, shoot density in the shallow shade treatment approached that of the deep control. Light measurement made during screen changes suggest that the shallow shade and deep control treatments received similar amounts of light.

Leaf area per shoot (Figure 1b) integrates changes in leaf length, width, and blade number that may occur in response to reduced light. Blade area of Thalassia might be expected to increase with decreases in light quantity. After 1 month of shading, leaf areas were slightly larger in the shaded vs. control shoots, reflecting the somewhat longer and wider blades recorded in shaded plots. The increased leafiness in shaded treatments did not persist, however, and by 9 months post-shading, leaf area per shoot was higher in controls than in shaded plots at both the shallow and the deep stations.

Leaf area index (LAI) (Figure 1c) synthesizes changes in both leaf area and shoot density which may occur with light reduction. **LAI** was generally higher at the shallow than at the deep station, reflecting higher shoot densities recorded there. Trends in **LAI** were similar to those observed for leaf area per shoot. **LAI** was higher in shaded plots than in controls in the deep, but especially in the shallow shade plots 1 month post-shading. However, after 3 months of light reduction and throughout the remainder of the study, **LAI** was usually higher in control than shaded plots at both the shallow and the deep stations. **LAI's** were similar in the shallow shade and deep control plots at 13 months post-shading,

suggesting that differences in density and morphology observed between the shallow and the deep margins of the Thalassia meadow may be attributable to differences in light availability.

Leaf relative growth rates (LRGR) (Figure 2a) might be expected to decline with decreasing light. LRGR did not appear to vary with depth. Growth rates were generally higher in control plots than in shaded plots at both the shallow and deep stations at 3 months post-shading through the remainder of the study. The effects of reduced light were much more apparent after 3 and 6 months of shading than after 13 months. The 13 month sampling period occurred in April, when Thalassia generally exhibits high growth rates. It appears that the effects of reduced light can be obscured by the endogenous seasonal growth patterns of Thalassia. Growth in all plots was slow in December 1989, at 9 months **post**-shading.

Shoot turnover time (Figure 2b) is inversely related to LRGR, and should reflect changes in growth rate which occur in response to shading. Turnover time was generally higher in shaded plots than in control plots at 3, 6, and 9 months postshading. Response in turnover time to shading was much greater in the deep than the shallow site. Notably, after 13 months of shading, differences in turnover time between treatments and controls decreased. Trends in plastochrone interval were similar to those of shoot turnover time. Perhaps below-ground reserves enabled seasonal growth patterns in Thalassia to outweigh the effects of reduced light.

Leaf production per  $\text{m}^2$  (Figure 3a) integrates leaf production

rates per shoot and shoot density per  $\text{m}^2$ . After 3 months of light reduction and throughout the rest of the study, shaded treatments exhibited lower leaf production on an areal basis than corresponding controls. Response to shading was initially greater at the deep vs. the shallow edge of the **seagrass** bed, however, by the end of the study, production in the shallow shade treatment was only 40% of the control. Production levels were similar in shallow shade and deep control plots after 13 months, suggesting that differences in areal production often observed between the shallow and deep stations may be related to differences in light availability.

Reduction in leaf biomass (Figure 3b) has been commonly observed in response to decreased light availability. Lower biomass per  $\text{m}^2$  was recorded in shaded vs. control plots at 3 months **post-**shading at both shallow and deep stations. Differences between treatments became more dramatic with time. After 13 months of shading, leaf biomass had dropped by 50% in the shallow, and 70% in the deep shade plots in relation to controls. Above-ground biomass was generally higher at the shallow than at the deep station, largely due to higher shoot densities at the shallow edge of the turtlegrass meadow. However, biomass in the shallow shade and deep control plots were quite similar by the end of the study, suggesting that differences in biomass observed with depth could be the result of corresponding decreases in ambient light.

In summary:

1. This study provides experimental evidence that reduction

in the amount of light reaching the subtropical **seagrass** Thalassia testudinum may elicit changes in density, morphology, growth, and production characteristics.

2. Plants living near the deep edge of the Thalassia meadow were more responsive to light reduction than plants at the shallow edge. These results should not be surprising, if turtlegrass at the deep site is living nearer to **it's** minimum light requirement.

3. Experimental light reduction appeared to elicit both a short-term and a long-term response in leaf area of Thalassia testudinum. After 1 month of shading, leaf areas were slightly greater in shaded treatments than in corresponding controls. With continued shading, leaf area of turtlegrass shoots declined in shaded plots at both the shallow and the deep stations. These findings are similar to those for Posidonia, which exhibited increased blade lengths during the first few months of shading, however, blade lengths decreased with continued shading.

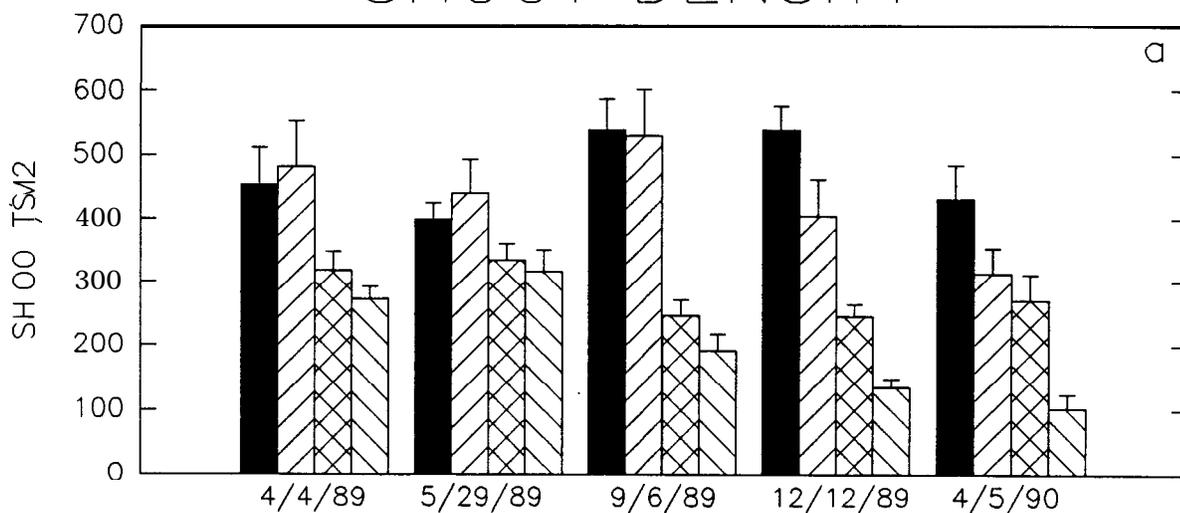
4. Response times of seagrasses to shading appear to be species specific. For example, declines in shoot density have been shown to occur much more rapidly in Zostera, Heterozostera, and Ruppia than were observed for Thalassia or Posidonia. These differences may be related to the more extensive below-ground reserves present in Thalassia and Posidonia as opposed to some other seagrasses.

5. **LRGR**, shoot turnover time, and plastochrone interval appeared to rebound from the effects of shading during the 13th month of our study, which coincided with the beginning of the

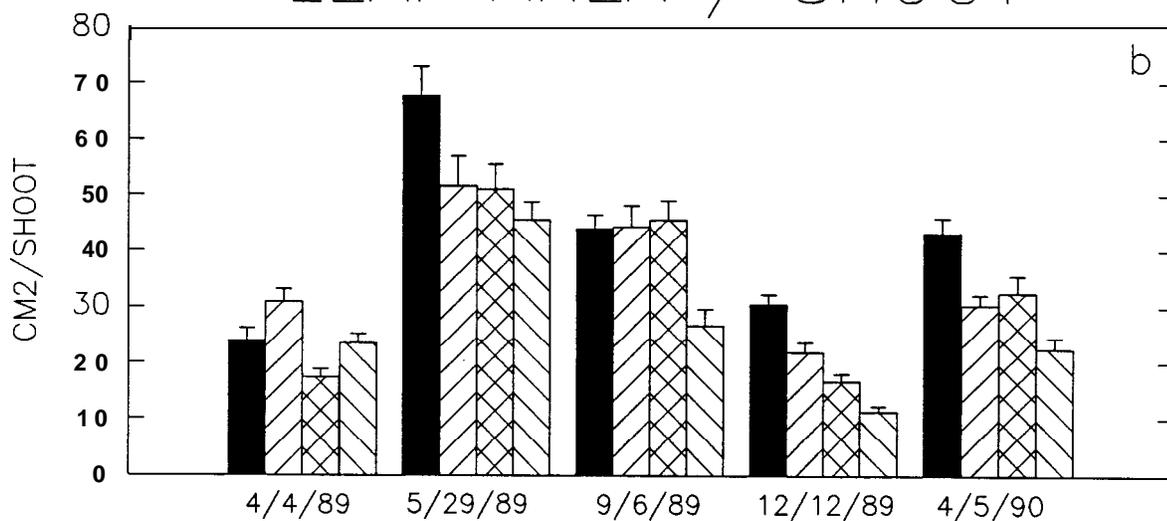
spring growing season. Although there were fewer, and somewhat smaller shoots present in the shaded vs. control plots 13 months post-shading, shoots in shaded plots were growing about as rapidly as shoots in control plots. Perhaps the typical seasonal pattern in growth and production of Thalassia was possible because **below-ground** reserves were still available to ameliorate the effects of reduced light.

6. Our results suggest that many of the characteristics of Thalassia that vary with depth may be attributable to a corresponding decrease in ambient light availability. During the course of this investigation, shallow shaded plots became similar to deep control plots in characteristics such as shoot density, leaf biomass, areal leaf production, and LAI. The fact that plants growing in the shallow region began to take on characteristics similar to those of the deep region when shaded illustrates the importance of light in controlling the growth and abundance of turtlegrass. Some of the characteristics which have been shown to respond to experimental light reduction may become useful management tools, as they may provide some early warning signals indicative of light stress.

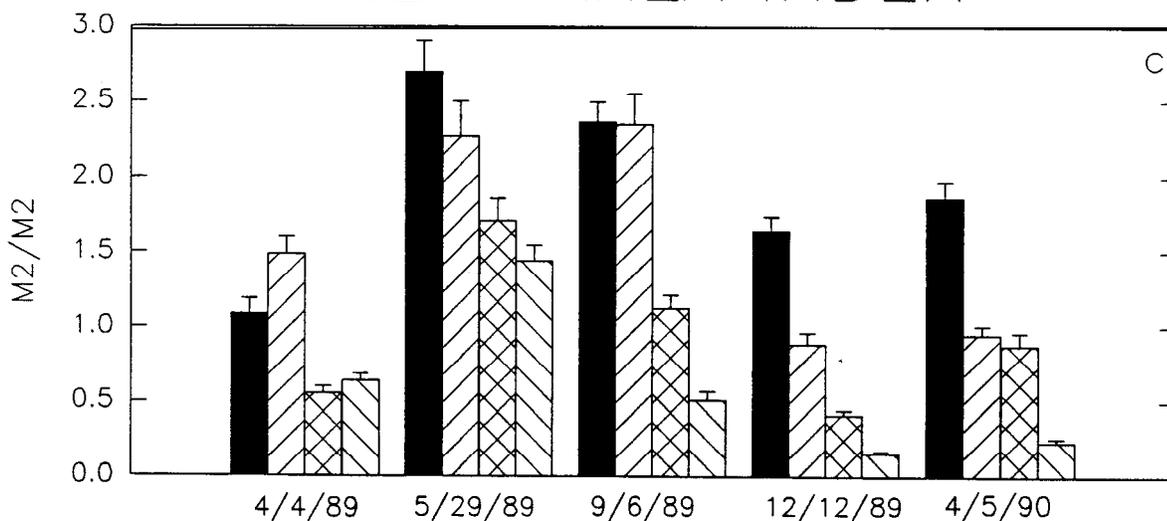
# SHOOT DENSITY



# LEAF AREA / SHOOT



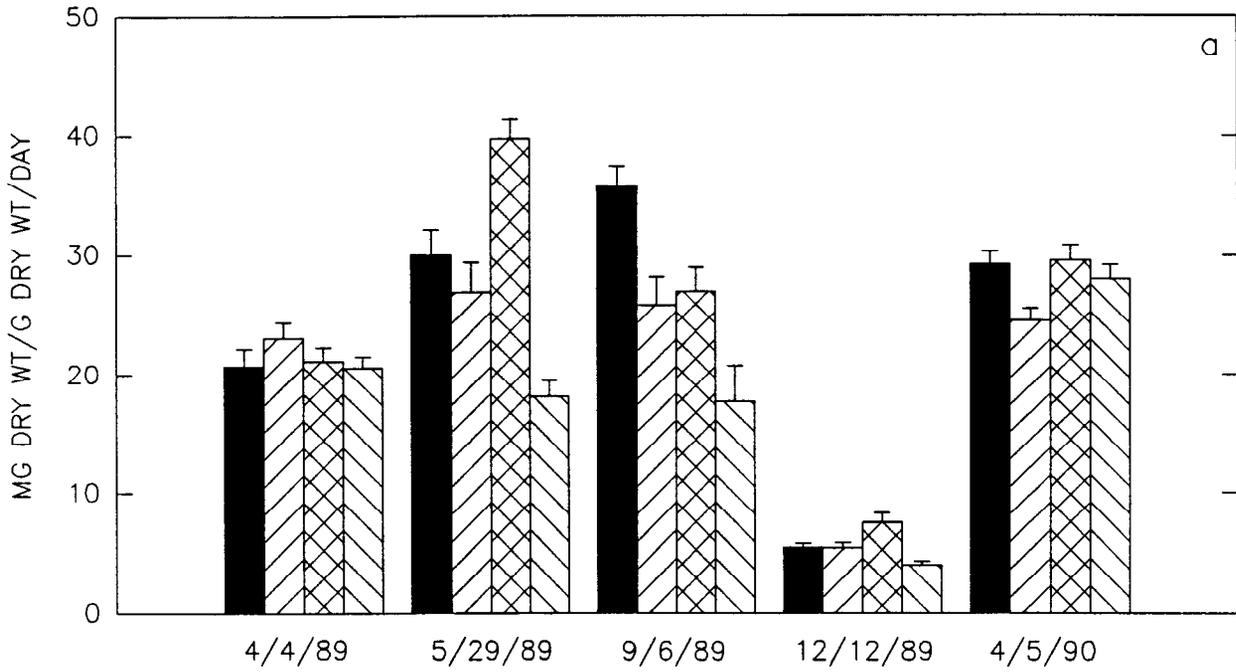
# LEAF AREA INDEX



Shallow Control
  Shallow Shade
  Deep Control
  Deep Shade

Figure 1

# LEAF RELATIVE GROWTH RATE



# TURNOVER TIME

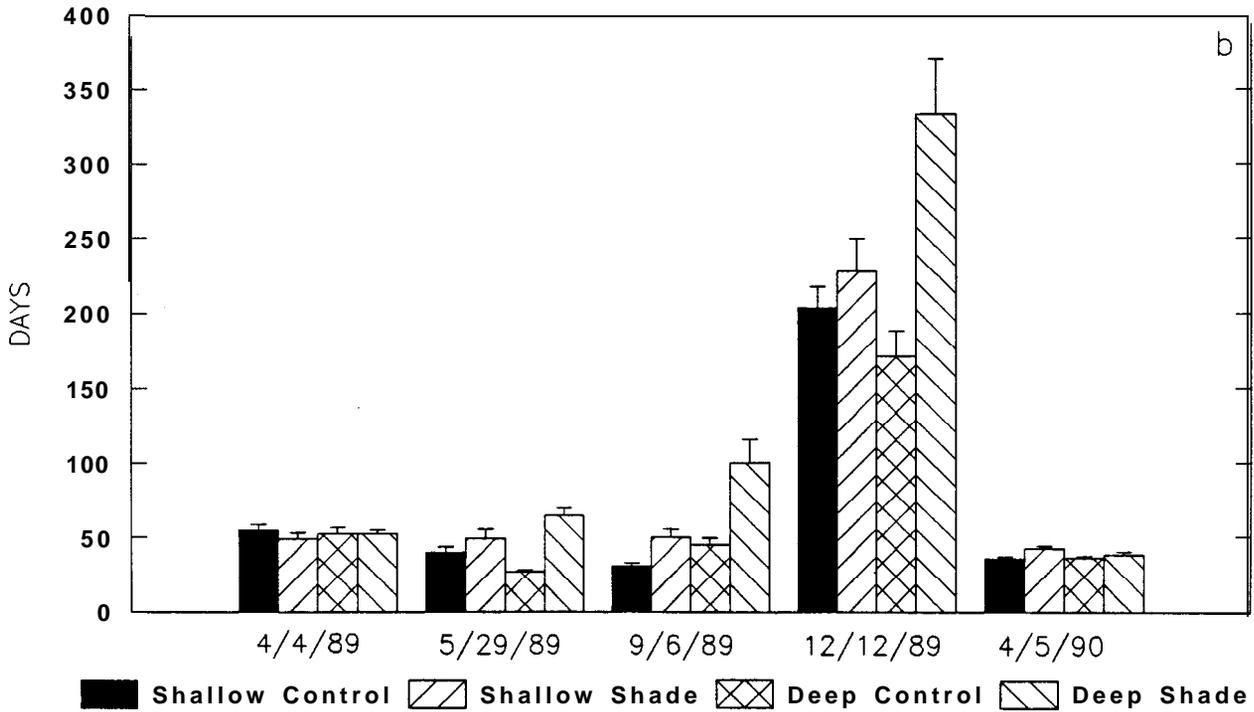
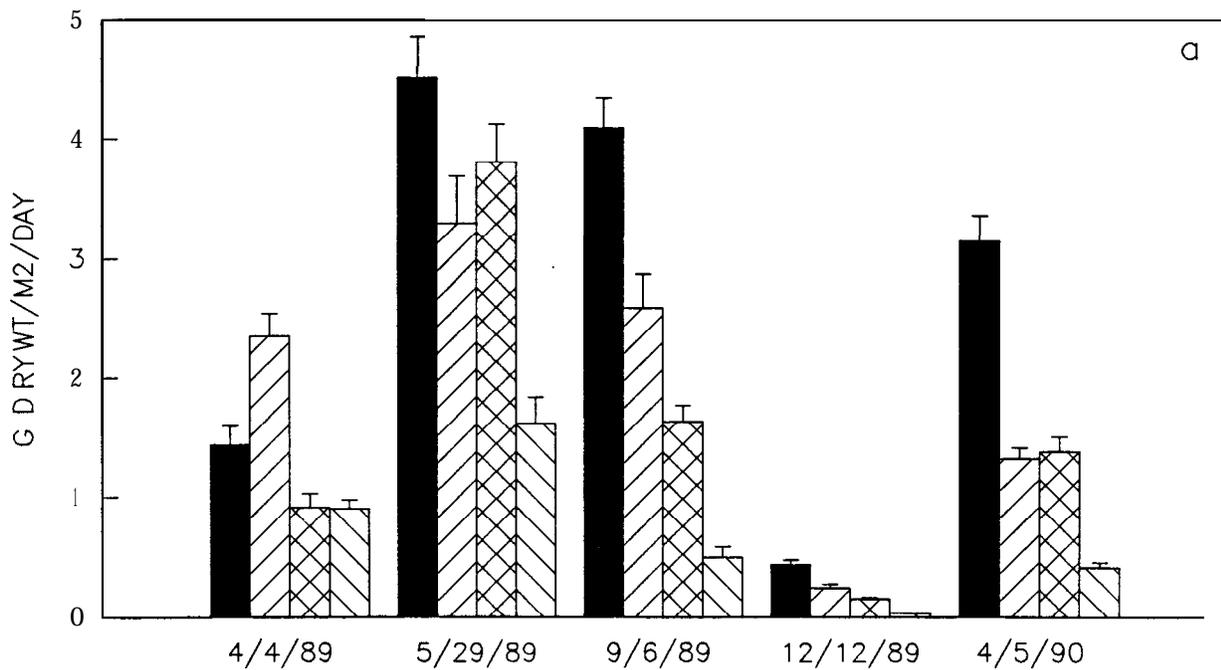


Figure 2

# LEAF PRODUCTION / SQUARE METER



# LEAF BIOMASS

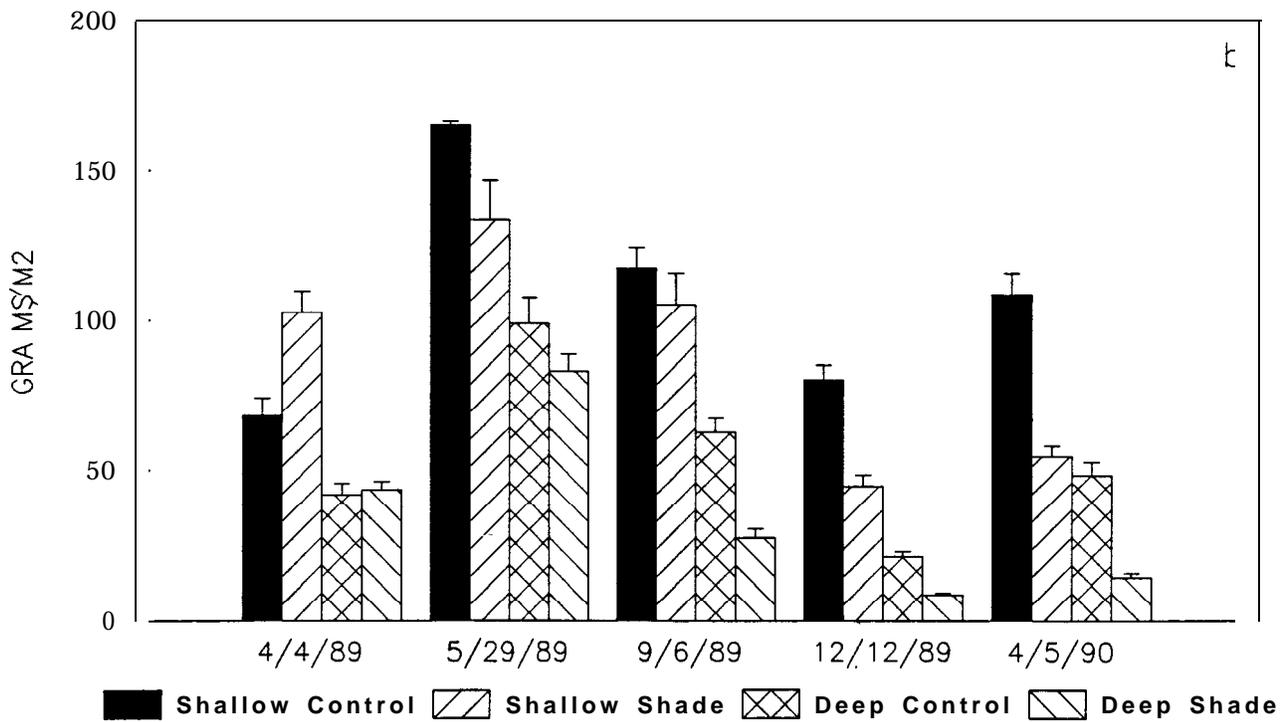


Figure 3

Onuf, Christopher P., U.S. Fish and Wildlife Service, National Wetlands Research Center, Corpus **Christi**, TX.

LIGHT REQUIREMENTS OF HALODULE WRIGHTII, SYRINGODIUM FILIFORME, AND HALOPHILA ENGELMANNI IN A HETEROGENEOUS AND VARIABLE ENVIRONMENT INFERRED FROM LONG-TERM MONITORING.

Comparisons among **seagrass** distributional surveys in the lower Laguna Madre of Texas conducted in 1967, 1974, and 1988 reveal progressive, major losses of **seagrass** cover. Losses were restricted to deeper parts of the laguna. The time-course and location of the loss correspond closely with differences in the intensity of maintenance dredging activity between locations and over time. These observations lead obviously to the hypothesis that light limitation sets the outer boundary of **seagrass** meadows in this embayment and that a reduction in water clarity resulting from increased turbidity caused by maintenance dredging is responsible for meadow retreat.

Imminent notice of maintenance dredging, together with the strong circumstantial evidence implicating dredging in the loss of large expanses of **seagrass** meadow, prompted me to begin monitoring the underwater light regime of the laguna near Port Mansfield, Texas, in 1988 (**Fig.1**). Here, the time-block contrasts used to test for dredging effects are ignored, and the data are aggregated to allow the most powerful discrimination of spatial patterns.

The data set used in this analysis consists of approximately 2400 determinations of extinction coefficients calculated from light vs depth profiles obtained on 56 days in 20 months between January 1988 and September 1989. Measurements of Photosynthetically

Active Radiation (PAR) were measured with a **LiCor** 1000 datalogger equipped with a terrestrial quantum sensor mounted above the boat and a spherical underwater quantum sensor mounted on a lowering frame. Depth was recorded for each profile, as was bottom cover (determined from what came up on the anchor). The observations were made along east-west transects and were arbitrarily grouped into 0.2' intervals of longitude .for computing means and mapping. On average, each interval on each transect was sampled on 13 dates. The east-west transects originated in the interior of continuous meadows growing on the sand flats fringing Padre Island, the coastal barrier that separates Laguna Madre from the Gulf of Mexico, and ended either in the middle of the embayment, beyond the deep edge of the meadow, or at the western shore.

The percentage of surface PAR reaching the bottom was computed for each depth profile as  $100 \cdot e^{-kz}$ , where  $k$  is the mean of Beer's law extinction coefficient over all intervals of a depth profile, and  $z$  is bottom depth (m) at the time of light measurement. This indirect approach had to be used because the **"surface"** reading is from 5-10 cm below the surface and the deepest possible reading for the apparatus is 20-25 cm above the bottom. Because the underwater light measurements were made sequentially rather than simultaneously, light measurements at different depths can be confounded by changes in atmospheric conditions between measurements. To factor out these atmospheric effects, ratios between the simultaneous readings for underwater and terrestrial sensors were used in computing the  $k$  values for each depth stratum,

rather than the raw underwater readings. A 0.2 (0.3 km) interval of longitude was considered to be within a **seagrass** meadow if >50% of anchor samples collected in the interval over the 20 months had seagrass. The sensitivity of boundaries to other thresholds for meadow designation can be evaluated from Fig. 2d.

Simple contour maps of depth, turbidity as measured by extinction coefficient, and % of surface PAR reaching the bottom are displayed in the same figure with a map % of samples with **seagrass** cover to give a rough idea of relations among relevant factors. Depths >2m occur only in the northern half of the study area (Fig. 2a). There is an abrupt **dropoff** to depths >1m along the mainland shore in the north, while shallow flats extend for kilometers beyond the limit of light sampling on the east side, toward Padre Island. Waters are most turbid in shallow to intermediate depths along the mainland shore and at intermediate depths on the east side (Fig. 2b). Waters are clearest in shallow areas on the east side. The aggregate effect of depth and turbidity is that light at the bottom is greatest on the Padre Island flats and least in deep waters of the north and in waters of intermediate depth in the south, extending even into shallow waters along the mainland shore (**Fig.2c**). The outer boundary of **seagrass** meadow as defined by >50% frequency of anchor samples with **seagrass** generally lies in the 5-20% light-at-bottom zone (**Fig.2d**).

Because of the heterogeneity of the system with respect to turbidity and basin morphology, the study area was subdivided into more homogeneous quadrants for a more critical analysis of **seagrass**

distribution in relation to light availability. The quadrants were defined by the intersecting navigation channels that traversed the basin (NE = north of Port Mansfield Channel and east of the Gulf intracoastal Waterway, etc.) "**Edge**" in Fig 3-5 is the outermost **0.3km** interval on a transect meeting the criterion of meadow (>50% of anchor samples with seagrass). Presumably, light is sufficient to allow development of close to continuous **seagrass** cover at that point. The change from meadow to bare bottom commences at an unspecified distance out from this nominal edge. This artifact of data aggregation and meadow definition accounts for the wide divergence in bottom light at the nominal edge in Fig. 3. The closest that we can resolve the level at which light becomes limiting to meadow development is 0.3 km out from the nominal edge. There, the long-term mean values of % of surface PAR reaching the bottom converge around 15, ranging from 12 in the southwest quadrant of the study and to 21 in the southeast.

Two strong suspicions arise about the validity and utility of this crude assessment of light in relation to the outer edge of **seagrass** meadows. 1) The average of 13 observations per sampling location over 20 months may not be adequate to characterize the long-term light regime in a shallow embayment subject to strong and variable winds. 2) The mean may not be a valid summary statistic of the light regime (For instance, if the frequency distribution is bimodal).

I checked these possibilities by simulating light at the bottom along a transect from inside the meadow in the northeast

quadrant out into the middle of the laguna and comparing these to the aggregate of the actual observations classified according to the same intervals referred to the edge of the meadow. The simulation was constructed from hourly predicted extinction coefficient values and hourly water depths at each station from 0600 to 2000 hours, three days each month for a year, using Weather Service hourly wind speed and direction records and NOS hourly water level records. Multiple regression relations were used to predict the extinction coefficients from a set of wind speed and direction, water level, and location variables.

The overall mean percentages of light at the bottom are in close agreement between observed and simulated records outside the meadow but diverge considerably for the two stations farthest into the meadow (Fig.4). In all likelihood, neither the simulation nor the aggregate of observations should be believed for the latter stations. Sampling frequency was one quarter what it was elsewhere, because the stations were inaccessible when water level was low. The next sampling interval out from the nominal edge of the meadow is most crucial to the assessment of limiting levels of light. The similarity of predicted and observed values here is a validation of the earlier findings.

The frequency distribution of % light at the bottom at different distances from the edge of the meadow show the **>40%** light-at-the-bottom category as most frequent within the meadow and an abrupt drop in it to the first station outside the bed (**Fig.5a,b**). Conversely, the **1-5%** light-at-the-bottom category is

low in the meadow and rises abruptly at the first station outside. The light regime of the first sampling interval outside the meadow is bimodal in both the simulated and observed records. This may mean that the long-term average is not the best descriptor of limiting conditions. Some threshold frequency of higher or lower light might be critical. However, at least in the observed record (**fig.5a**), the larger mode is in the same % light-at-the-bottom category as the mean, so a major discrepancy is unlikely.

Differences in the outer limits of the three species that occurred in the study area are minor (Fig.6). Where Syringodium filiforme is absent, Halodule wrightii extends in continuous cover into areas of as low light as Syringodium. In fact, biomass vs depth comparisons indicate that Halodule achieves highest biomass precisely in the depth zone from which Syringodium displaces it when present (unpublished observations). Halophila enselmanni is restricted largely to the outer edge of **seagrass** meadows but in few instances actually occurs farther out than the other two species. Although the species differ appreciably in their "centers of **gravity**" along a light intensity axis, there are no demonstrable differences in their low light limits.

Despite the low spatial resolution of this analysis and limitations imposed by using anchor samples to define meadow location, the overall analysis is **quite** robust. Differences between any particular pair of stations are likely to be artifactual, because at that level, the extreme differences in conditions that could be represented by the subset of sampling dates for each

location easily could overwhelm any between-location difference. However, the analyses for the subdivisions of the study area are aggregated from at least 4 transects (NE quadrant - 10 transects, SE - 6, SW - 4, NW - 6), and sample sizes and number of dates sampled for each interval in or out from the meadow's edge are much larger than the mean of 13 for any single interval. Thus there is a blind brute force element to the analysis that strengthens conclusions. The congruence of estimated light limits among subdivisions of the study area with grossly different topographic and water clarity characteristics lend further support to the basic conclusion of this study. At least 15% transmission of PAR is required to sustain meadow development in this mixed assemblage, shallow subtropical system. These findings extend those for the NMFS program in **Hobe** Sound, Florida, to other key species and to a system with a highly variable light environment.

## FIGURE LEGENDS

Figure 1. Map of Laguna Madre showing the study area.

Figure 2. Contour maps of depth (a), turbidity as measured by extinction coefficient (b), % of surface irradiance reaching bottom (c), % of anchor samples in a sampling interval bringing up seagrasses and boundaries enclosing intervals with >50% frequency of **seagrass** occurrence (d).

Figure 3. % light at the bottom in relation to the nominal outer edge of **seagrass** meadow in the 4 quadrants of the study area.

Figure 4. Comparison for the northeast quadrant between predicted % light at bottom from multiple regression relations of turbidity on wind speed and direction, water exchange, and location variables and the mean of all observations made during the study.  $n = 480$  for each predicted value (16 hourly values per day approximately every 10th day in 1988) and  $n$  ranges from 15 to 140 for the observed values.

Figure 5. Frequency distributions of observed (a) and simulated (b) % light at the bottom at different distances from the outer edge of the **seagrass** meadow in the northeast quadrant.

Figure 6. Contour map of % of surface irradiance reaching bottom (a) and locations of samples with *Halodule wrightii* (b), *Syringodium filiforme* (c), and *Halophila engelmannii* dominant. **Seagrass** boundary is repeated from Fig. 2d.

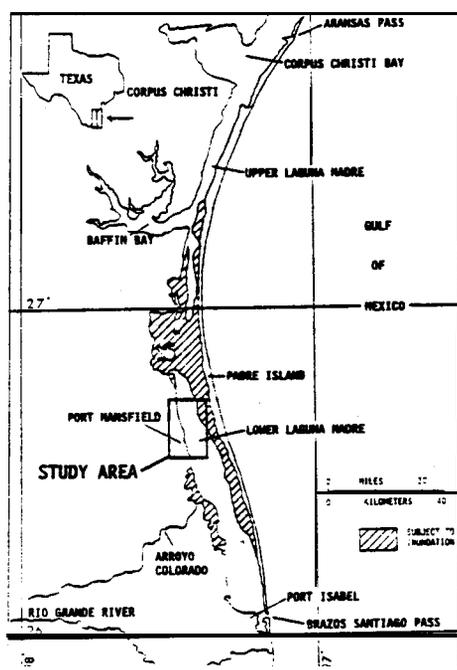


FIGURE 1.



LIGHT AT BOTTOM RELATED TO MEADOW EDGE  
DIFFERENT PARTS OF STUDY AREA

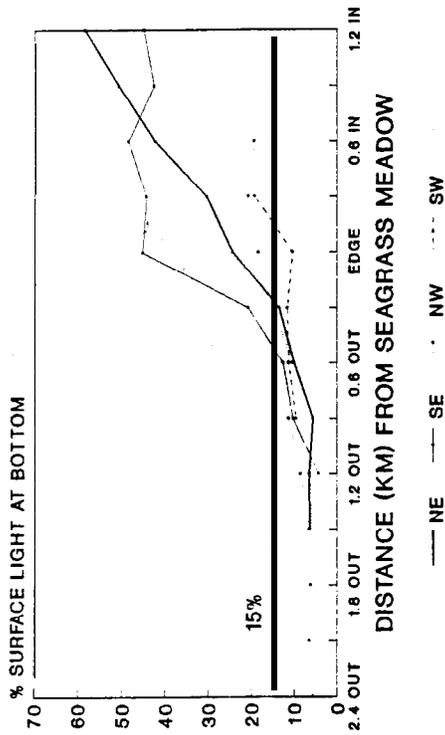


FIGURE 3.

MEAN ANNUAL % LIGHT AT BOTTOM  
PREDICTED FROM WIND DATA AND OBSERVED

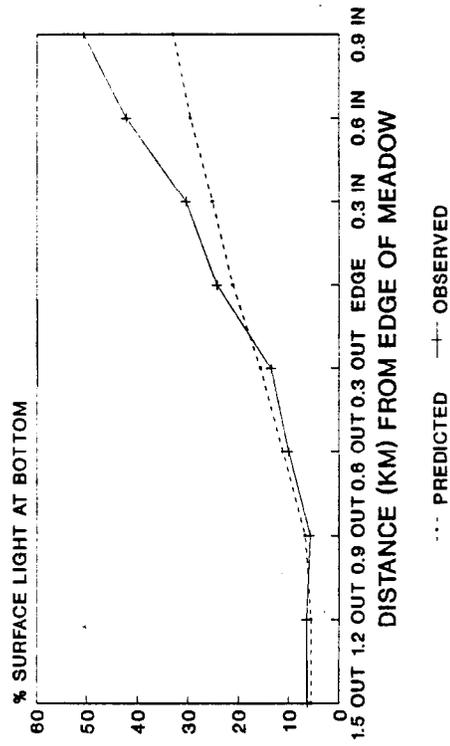
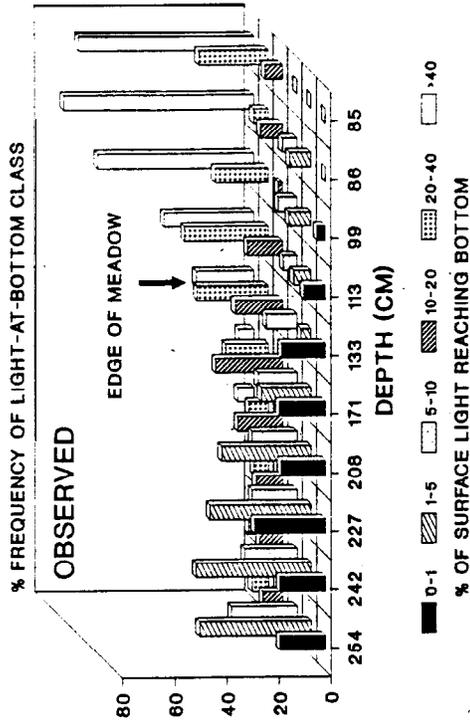


FIGURE 4.

a % LIGHT AT BOTTOM-FREQUENCY DIAGRAMS  
AT 0.3 KM INTERVALS FROM MEADOW'S EDGE



b % LIGHT AT BOTTOM-FREQUENCY DIAGRAMS  
AT 0.3 KM INTERVALS ALONG A TRANSECT

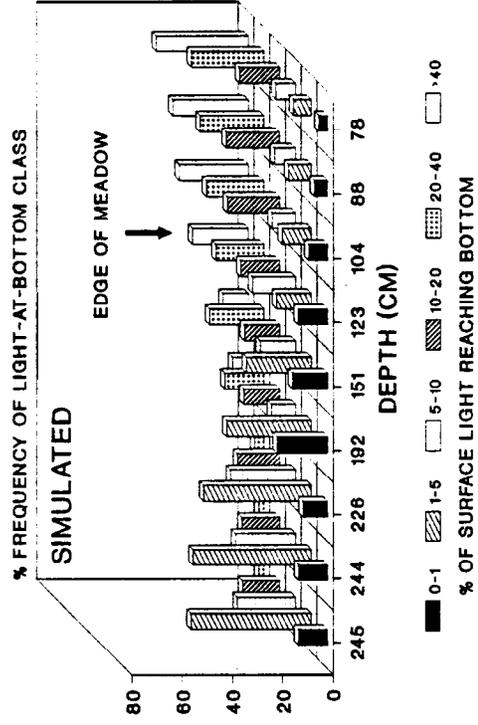
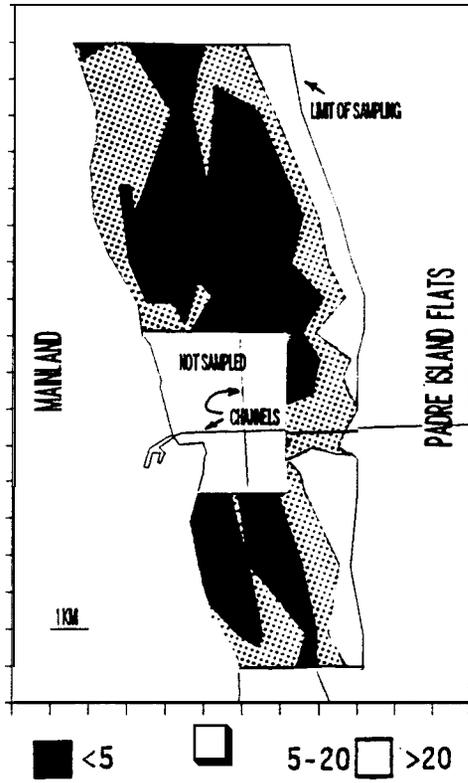
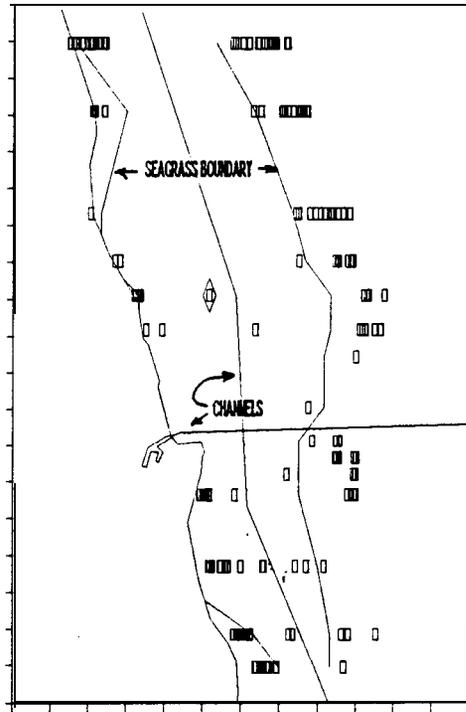


FIGURE 5.

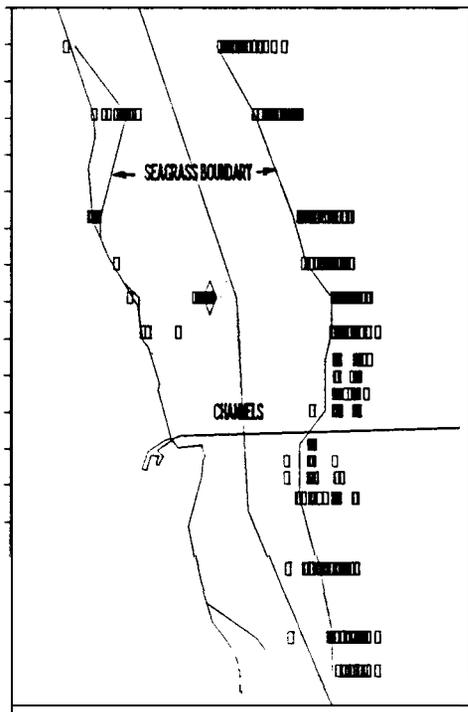
**a** % OF SURFACE LIGHT REACHING BOTTOM



**b** SAMPLES WITH HALODULE DOMINANT



**c** SAMPLES WITH SYRINGODIUM DOMINANT



**d** SAMPLES WITH HALOPHILA DOMINANT

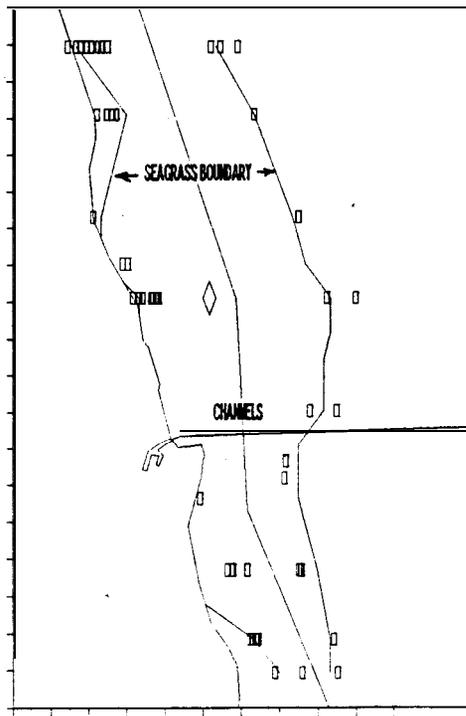


FIGURE 6.

Kenworthy, W. Judson, Mark S. Fonseca, and Stephen J. DiPiero, Beaufort Lab, NMFS, NOAA, Beaufort, N.C.

DEFINING THE ECOLOGICAL LIGHT COMPENSATION POINT FOR SEAGRASSES HALODULE WRIGHTII AND SYRINGODIUM FILIFORME FROM LONG-TERM SUBMARINE LIGHT REGIME MONITORING IN THE SOUTHERN INDIAN RIVER.

Between March 1987 and September 1990 we measured the attenuation of photosynthetically active radiation (PAR) in a shallow system of coastal lagoons at the southern end of the Indian River Lagoon in Florida. Attenuation coefficients (k) were determined from light profiles taken at between 16 and 24 stations which were visited weekly. The submarine light profiles were obtained with a spherical quantum sensor (Li-Cor) between the hours of ten AM and two PM. At least five light measurements were taken in each profile and were corrected for fluctuating incident light due to cloud cover with a reference sensor located on the deck of the boat. The maximum depth of the profiles were two meters. All data were recorded and stored on a data logger which also served as the calibrated response meter for the sensor. Values of light attenuation (k) were calculated with a SAS computer algorithm. From the k values we estimated the amount of incident light transmitted through the water column to the maximum depths occupied by the seagrasses.

The areal distribution of seagrasses was determined with 1/10,000 scale color aerial photography taken in April 1988 and May 1989 and ground verified throughout the study period. One hundred and thirty shore normal transects separated by 100m were surveyed at five meter intervals in the lagoon to determine the

species composition and depth penetration of seagrasses.

The aerial photography was able to discriminate the distribution of Halodule wrightii and Syringodium filiforme but not the genus Halophila. Areas of the lagoon occupied by H. wrightii and S. filiforme were easily distinguished but the signature for Halophila appeared as unvegetated bottom. H. wrightii and S. filiforme grew to maximum depths of approximately 1.75 to 2.0 m (Figures 1 and 2). Average depth for H. wrightii was 0.91 m and 1.1 m for S. filiforme. At depths exceeding 2.0 m Halophila decipiens, H. johnsonii and H. enselmanni grew. H. decipiens covered as much as 100% of the bottom in some of the deeper areas of the lagoon between April and October. H. johnsonii covered only about 3 to 5 % of the deeper area but was present all year long. H. enselmanni was only sparsely distributed in the most interior part of the lagoon.

There was a definite recurring seasonal cycle in the amount of light transmitted to 2.0 m; the lower depth limit of H. wrightii and S. filiforme (Figure 3). Maximum values of 40 to 50 % transmittance occurred in summer between May and August and 5 to 20 % between September and April. There were significant shorter term fluctuations within these larger time windows when either turbid or extremely clear water penetrated the lagoon. However, the majority of the values exceeded 10 to 15 % of the incident light (Figure 4). The lowest values, 5-7 % occurred after the passage of Hurricane Floyd in October of 1987 when attenuation coefficients were between 1.5 and 2.0. Following the

storm it took 70 to 80 days for the lagoon to return to what appeared to be normal winter attenuation values. Tidal excursion from Jupiter Inlet is an important hydrographic parameter that influences the transparency of the water in the interior portion of **Hobe** Sound.

Neither water currents nor sediment characteristics could explain the sharp threshold of depth distribution for H. wrightii and S. filiforme at 2.0 m. In the deeper waters of the lagoon vegetated by Halophila species and outside of the channel, maximum current velocities do not exceed 20 cm  $\text{sec}^{-1}$ . The near bottom shear velocities developed by these current speeds are well below values known to uproot seagrasses. In addition, the sediments are unconsolidated quartz sands with 10 to 15 % silt clay and are typical of sediments which support **seagrass** growth. Since these environmental parameters are well within the tolerable limits of seagrasses, we hypothesize that the lower depth distribution of H. wrightii and S. filiforme is due to light limitation. It appears as though the ecological light compensation point for these two species are in excess of 10 to 15 % of the incident light and are much greater than the previously reported values of 1 to 5 % for seagrasses and other aquatic plants. Criteria and standards for water quality which are based on lower values of compensation points probably cannot be used to protect seagrasses from deteriorating water transparency (turbidity).

#### FIGURE LEGENDS

FIGURE 1: Qualitative illustration of the depth distribution of seagrasses in **Hobe** Sound, Florida.

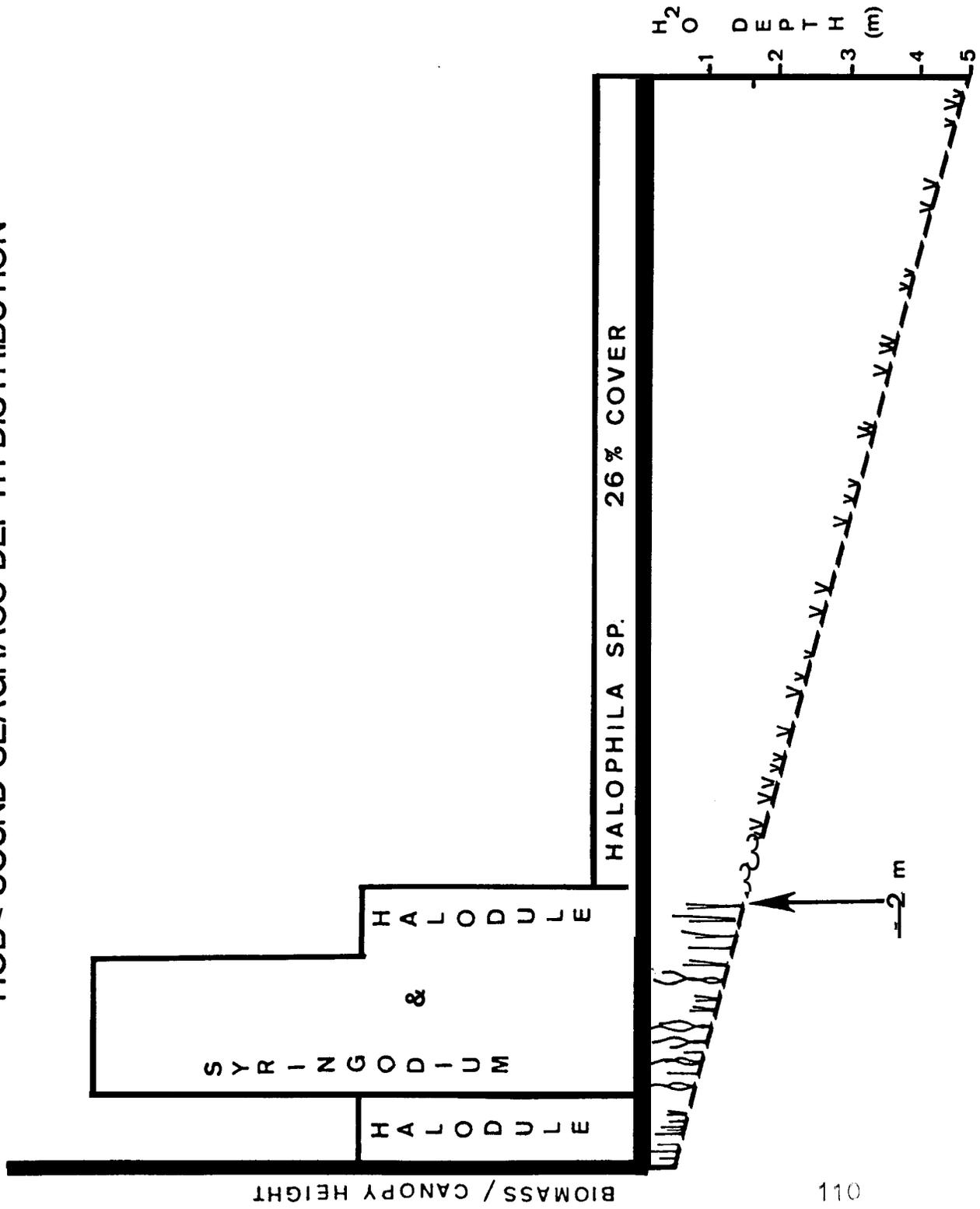
FIGURE 2: Frequency of occurrence of seagrasses H. wrightii and S. filiforme as a function of water depth in **Hobe** Sound, Florida.

FIGURE 3: Percent of incident photosynthetically active radiation reaching a depth of 2m in **Hobe** Sound, Florida.

FIGURE 4: Frequency distribution of percent incident photosynthetically active radiation reaching 2m depth in **Hobe** Sound, Florida.

FIGURE 1

HOBΞ SOUND SEAGRASS DEPTH DISTRIBUTION



SPECIES DEPTH DISTRIBUTION  
HOBE SOUND

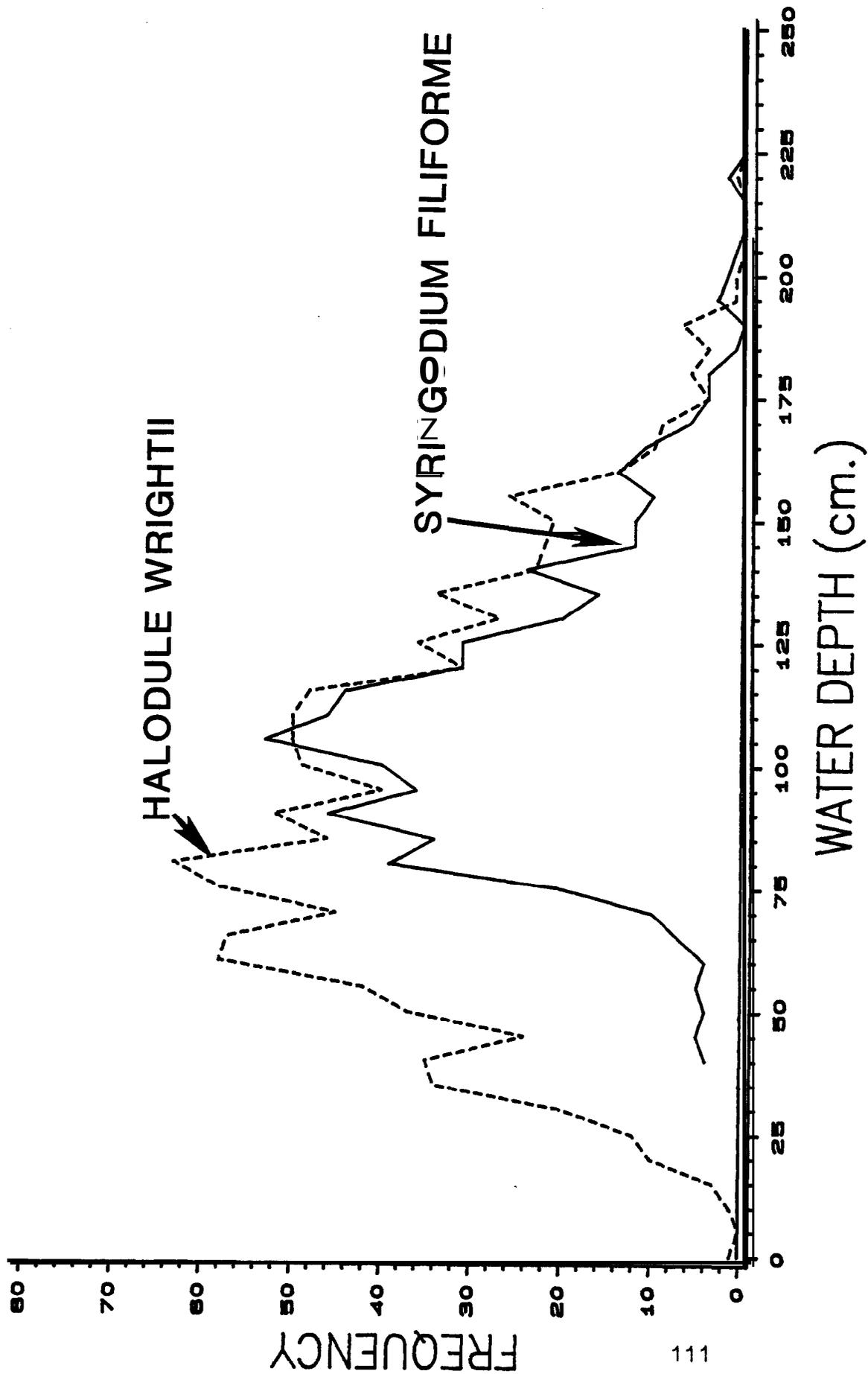


FIGURE 2

FIGURE 3

PERCENT LIGHT AT TWO METERS DEPTH  
HOBE SOUND

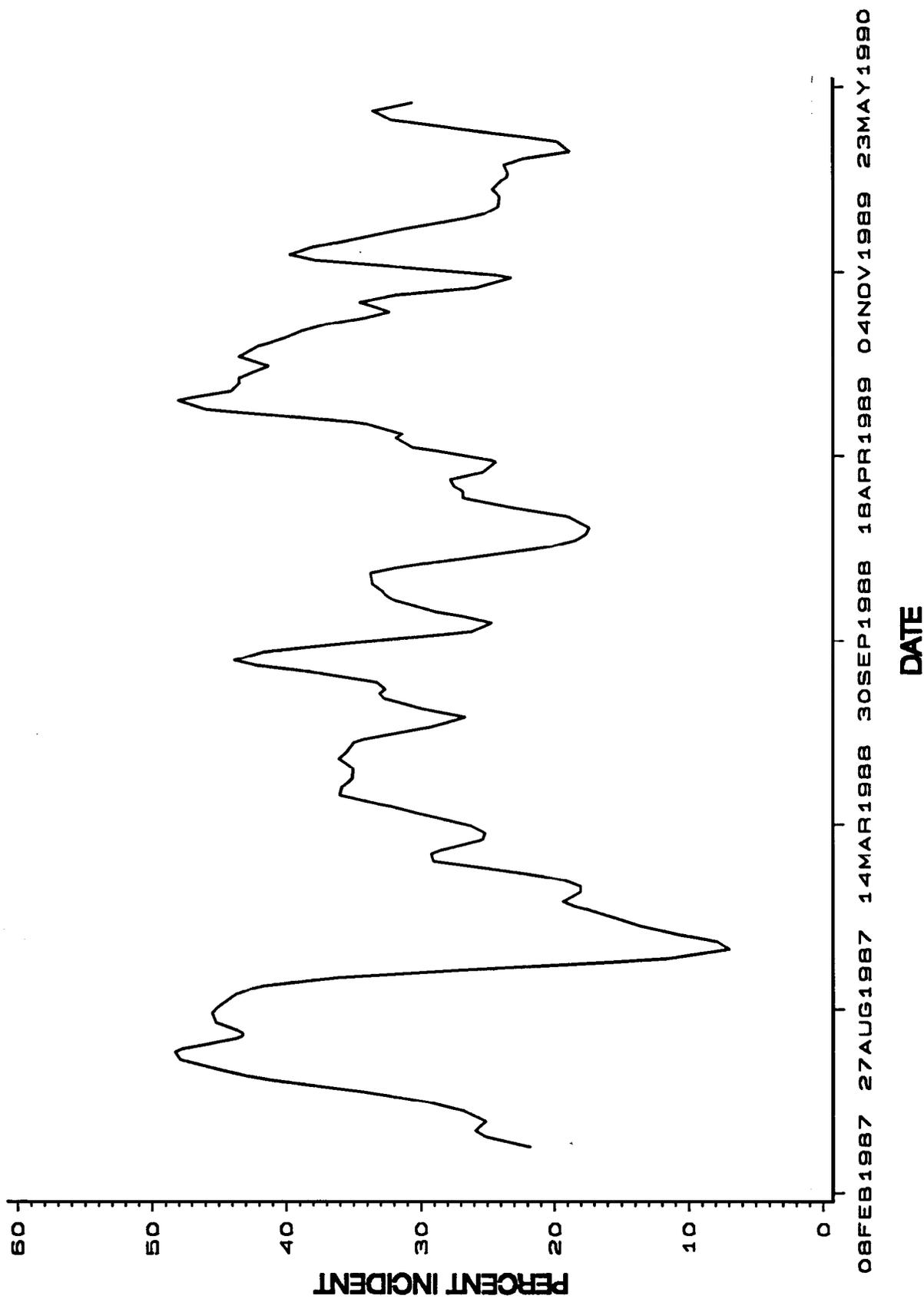
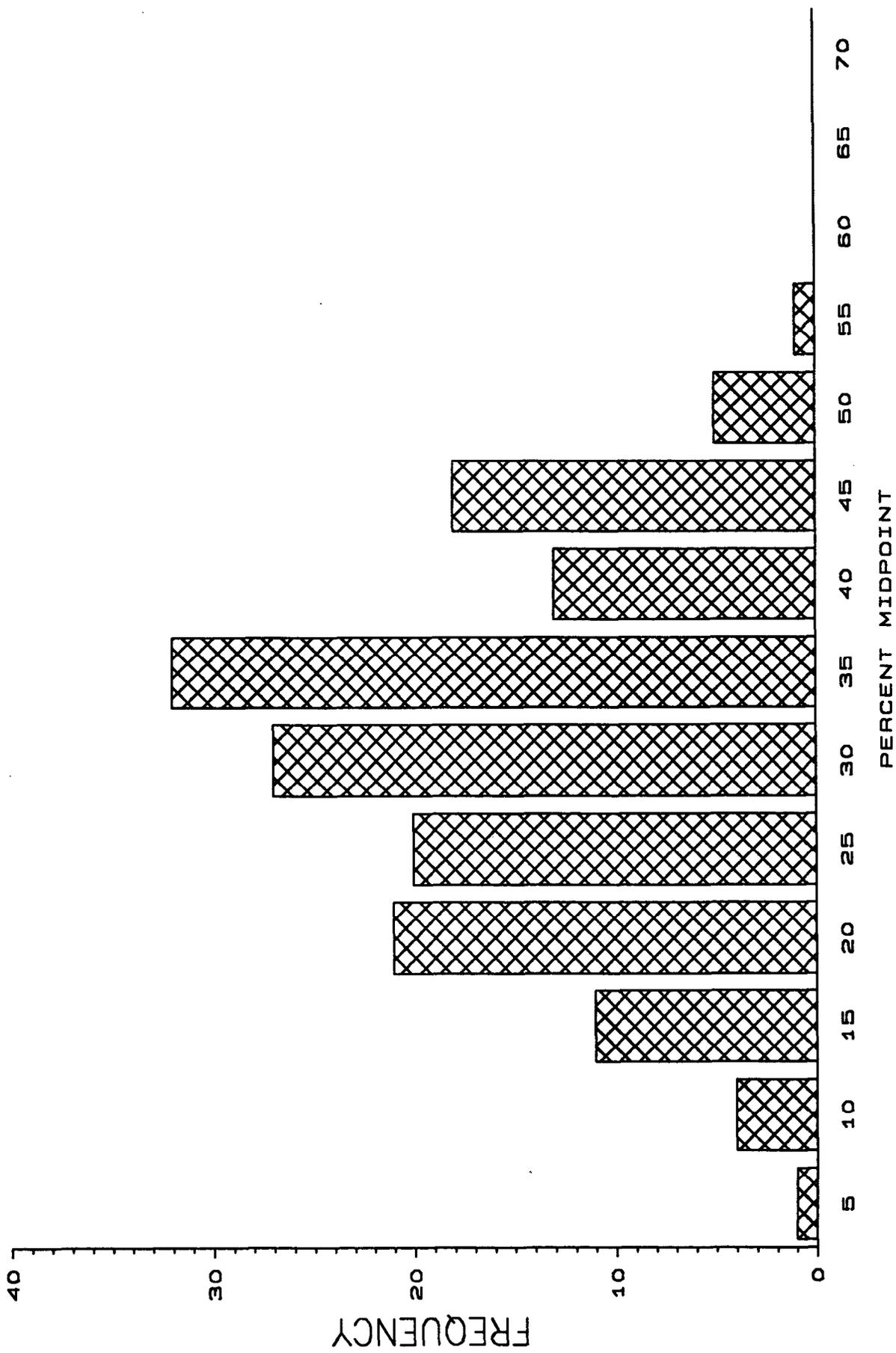


FIGURE 4

FREQUENCY OF PERCENT INCIDENT LIGHT  
TWO METERS IN HOBE SOUND



SESSION 3; INTEGRATING MODELS WITH LABORATORY AND FIELD DATA TO **DETERMINE** SOURCES OF LIGHT ATTENUATION AND ITS EFFECTS ON THE DISTRIBUTION AND ABUNDANCE OF SEAGRASSES.

Gallegos, Charles L., David L. Correll, and Jack Pierce, Smithsonian Environmental Research Center, Edgewater, MD.

MODELING SPECTRAL LIGHT AVAILABLE TO SUBMERGED AQUATIC VEGETATION

One of the major long-term biological changes in Chesapeake Bay over the last 20 years has been the severe decline in the 11 native species of submerged aquatic vegetation (SAV). Principal causes of the SAV decline are usually attributed to increases in the runoff of agricultural herbicides, toxic discharges, suspended sediments, or plant nutrients. The latter two factors, suspended sediment and nutrients, impact SAV indirectly through their effects on the penetration of light through the water column and its availability at the leaf surface. Increased nutrient loading reduces light availability by stimulating algal growth, including phytoplankton in the water column and epiphytes on the leaf surface.

Regardless of the proximate cause of SAV decline, it is clear that adequate availability of light at the plant leaf-surface represents a minimum requirement for SAV persistence or recovery. Building on recent studies of spectral light penetration in the Rhode River, we developed a model relating optical properties of the water column to the concentrations of light-absorbing and scattering materials dissolved and suspended in the water. The model partitions the contribution to total absorption and scattering coefficients amongst the various suspended and dissolved

materials. Here we use the model to predict **photic** depths as a function of water quality constituents, for a range of assumed compensation points, and examine the site-specificity of the model using data from the Indian River, **Fl.**

In this work we distinguished between inherent optical properties, which are determined solely by the contents of the water, and apparent optical **properties**, which are a function of both the content of the water and the ambient light field. Inherent optical properties have the advantage that the contributions of individual components to the overall property are strictly additive, which is not true of apparent optical properties. Thus we may partition the total absorption coefficient at a wavelength  $\lambda$ ,  $a_t(\lambda)$ , (an inherent optical property) as the sum of the absorption due to pure water  $a_w(\lambda)$ , dissolved organic matter  $a_d(\lambda)$ , and particulate material  $a_p(\lambda)$ . That is

$$a_t(\lambda) = a_w(\lambda) + a_d(\lambda) + a_p(\lambda) \quad (1)$$

Absorption by particulate material may be further divided into the contributions due to phytoplankton,  $a_{pl}(\lambda)$ , and that due to mineral and organic detritus,  $a_{det}(\lambda)$ .

Measurements of spectral diffuse attenuation coefficients were made on 15 sampling occasions in 1988; 2-3 sites were occupied on each trip at a total of 7 different locations, 6 in the Rhode River, and one in Chesapeake Bay near marker 73 in the main shipping channel. Diffuse attenuation coefficients were measured

using a spectral radiometer designed and built in house. The instrument uses interference filters to isolate narrow regions of the visible and near infrared spectrum (**Fig.1**). Bandwidths of the filters vary so that highest resolution (**10nm**) is obtained in the wavebands where the spectrum changes most rapidly, which are also the wavebands in which plant pigments absorb most strongly. Similar measurements were made on 15 **occasions** in the Indian River, FL.

We used Kirk's (1984) model to extract estimates of absorption and scattering coefficients, two inherent optical properties, from measurements of diffuse attenuation. Using a Monte Carlo model of the propagation of photons underwater, Kirk (1984) determined that the diffuse attenuation coefficient averaged over the **euphotic** zone,  $k_d(av)$ , could be described as a function of the absorption and scattering coefficients,  $a$ , and  $b$

$$k_d(av) = \frac{1}{\mu_0} [a_t^2 + (0.425\mu_0 - 0.190)a_t b]^{1/2} \quad (2)$$

where  $\mu_0$  is the cosine of the solar zenith angle refracted at the air water interface, which may be calculated from location and time of day. We calculated  $b$  by assuming that water itself is the only significant absorbing substance in the 715 nm waveband, using  $a_w(715)=1.002 \text{ m}^{-1}$ . We then calculated an estimated scattering coefficient,  $b(715)$ , by rearrangement of eq. 1 and used that value for all wavebands.

Estimated scattering coefficients in the Rhode River ranged from 1.73-55.3  $\text{m}^{-1}$  and were linearly related to the concentration

of mineral suspended solids, [MSS] (Fig. 2a). By least squares regression we estimated the relation (Gallegos et al 1990)

$$b = 2.41[\text{MSS}] + 0.32, (r^2=0.92, n=15).$$

After determining scattering coefficients we estimated  $a_t(\lambda)$  from eq. 1. Using laboratory measurements of absorption by filtered water and by particulate material collected on glassfiber filters, we were able to partition  $a_t(\lambda)$  into contributions due to dissolved organic matter, phytoplankton pigments, and particulate detritus (Fig. 3). Absorption by each of these constituents has a characteristic spectral shape (Fig. 3). Thus the problem of predicting  $a_t(\lambda)$  becomes that of scaling the specific curves in terms of measurable water quality parameters. Dissolved organic matter and particulate detritus have spectral shapes that decrease exponentially with wavelength; thus scaling requires determination of a spectral slope and absorption at a characteristic wavelength, chosen here to be 400 nm. Absorption by dissolved substances at 400 nm,  $a_d(400)$ , was linearly related to dissolved organic carbon, [DOC], by the relation (see Fig. 2b)

$$a_d(400) = 0.27[\text{DOC}] + 0.35 (r^2=0.81, n=13),$$

and mean spectral slope was  $0.0104 \text{ nm}^{-1}$ . Absorption by particulate detritus at 400 nm,  $a_x(400)$ , was linearly related to total suspended solids, [TSS], by the relation (see Fig. 2c)

$$a_x(400) = 0.15[\text{TSS}] - 0.13 \quad (r^2=0.88, n=14),$$

and the mean spectral slope was  $0.012 \text{ nm}^{-1}$ . Absorption by phytoplankton normalized to the concentration of chlorophyll,  $a_{\text{ph}}^*(A)$ , does not have a convenient mathematical representation, so mean values were tabulated at 5 nm intervals (for brevity, not shown here; see **Gallegos et al.** 1990).

These tabulated values and the regressions given above, together with measurements of phytoplankton chlorophyll, [TSS], [DOC], [MSS] and tabulated values of absorption due to pure water give a complete set of equations for predicting  $a_x(A)$  and  $b$  from water quality parameters in the Rhode River and nearby Chesapeake Bay. Using predicted  $a_x(A)$  and  $b$  with Eq. 1 we found that  $k_d(\lambda)$ , the empirical descriptor of light available at a depth, could be predicted within an error of about 21%.

When predicting light available for plant growth, it is important to distinguish between photosynthetically available radiation (PAR), and photosynthetically usable radiation (PUR). PAR is the photon flux density equally weighted between 400-700 nm. PUR is the photon flux density weighted by the absorption spectrum of the plants of interest. Because of the strong spectral dependence of  $k_d(\lambda)$  in these waters, the wavelengths most efficiently utilized by plants are rapidly attenuated.

Using the model developed for the **Rhode River**, we can predict  $z_p(\text{PUR})$  as a function of water quality parameters for any

percentage of surface incident irradiance of interest. Figure 4a gives contours of  $z_p(\text{PUR})$  based on the penetration of 1% of surface irradiance as a function of chlorophyll and [MSS]. For ease of presentation in 2 dimensions, we have assumed that [DOC] covaries with chlorophyll and [TSS] covaries with [MSS] according to statistical relationships determined for the Rhode River, although in actual use the model carries no such restrictions. Contours on the figure denote lines of equal  $Z_p(\text{PUR})$ . When using 5% or 20% of surface irradiance as the compensation level for SAV growth, the water quality requirements become much more stringent (**Fig. 4b-c**).

Although the particular regressions used to model inherent optical properties in the Rhode River may be site specific, the approach used to derive them is applicable in any estuarine region. Analysis of data from the Indian River, FL. is incomplete, but preliminary analysis suggests that suspended sediments in the Indian River area have a lower optical scattering cross section than those in the Rhodes River, MD (Fig.5). Consequently, water quality criteria required to achieve a given **photic** depth standard may be somewhat less stringent (Fig. 6).

At least 2 critical gaps prevent immediate application of the present model for decision making purposes. First, the model was developed from measurements made in very turbid water; on only one occasion did the measured 1%  $Z_p(\text{PUR})$  exceed 4 m; most measurements fell between 1-3m. Many of the areas in the Chesapeake Bay that traditionally supported SAV beds are in waters deeper than those examined here. Application of the equations to predict optical

coefficients in such regions would require extrapolation of regressions to conditions less turbid than those encountered during model development. Secondly, prediction of  $Z_p(\text{PUR})$  based on suspended and dissolved constituents in the water column represents minimal, i.e. necessary conditions for SAV survival. Attached epiphytic material and other deposited solids reduce light available for SAV even further. However, conclusions about the significance of light absorption by epiphytic material requires better knowledge of the spectral absorption properties of the attached material.

#### REFERENCES

- Gallegos, C.L., D.L. Correll, and J.W. Pierce. 1990. MODELING SPECTRAL DIFFUSE ATTENUATION, ABSORPTION, AND SCATTERING COEFFICIENTS IN A TURBID ESTUARY. *Limnol. Oceanogr.* in press.
- Kirk., J.T.O., 1984. DEPENDENCE OF RELATIONSHIP BETWEEN APPARENT AND INHERENT OPTICAL PROPERTIES OF WATER ON SOLAR ALTITUDE. *Limnol. Oceanogr.* 29: 350-356.

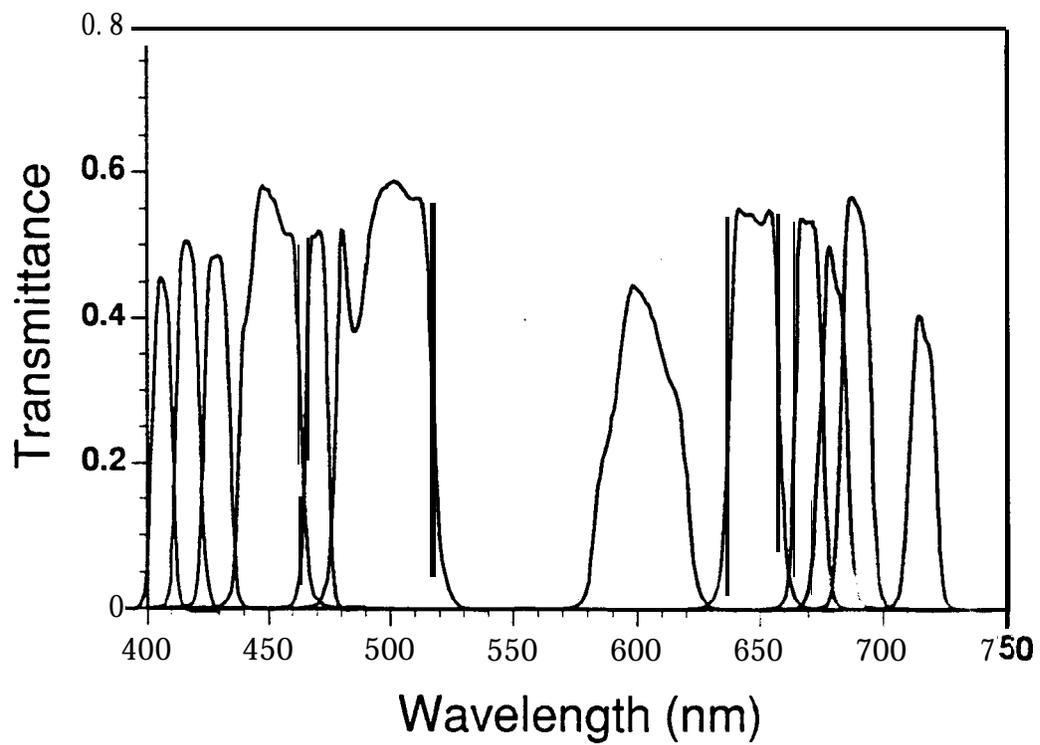


Figure 1. Transmittance spectra (dimensionless) of interference filters used in underwater spectral radiometer.

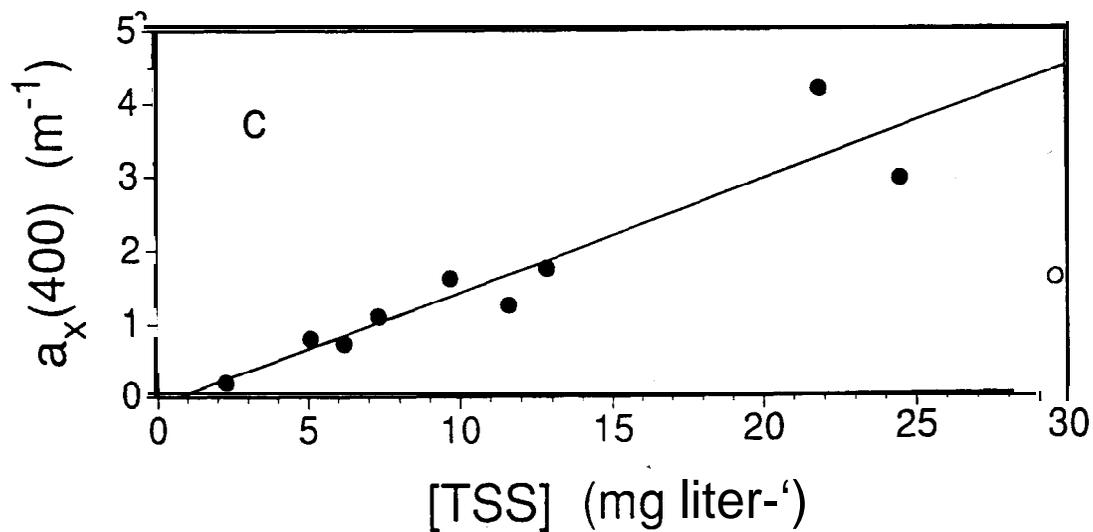
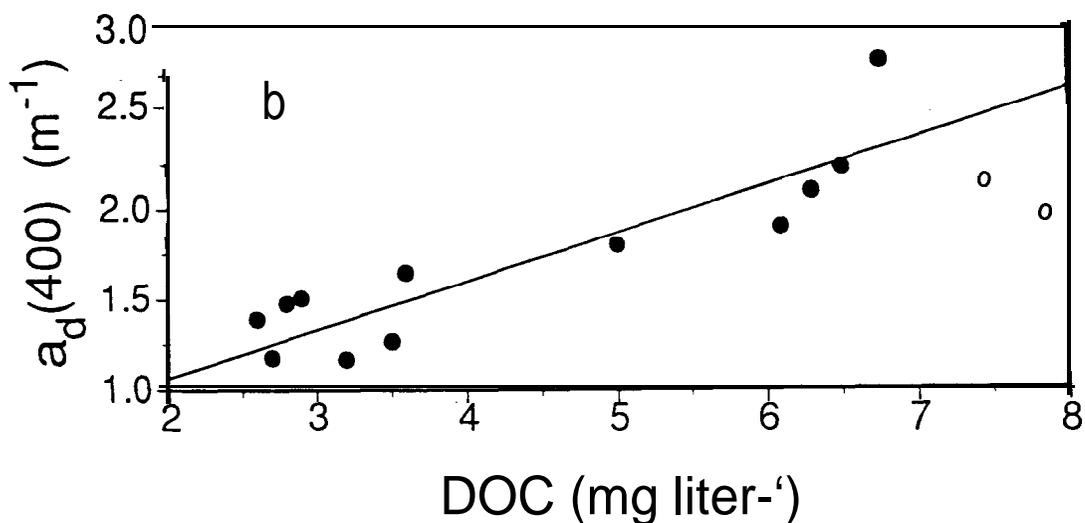
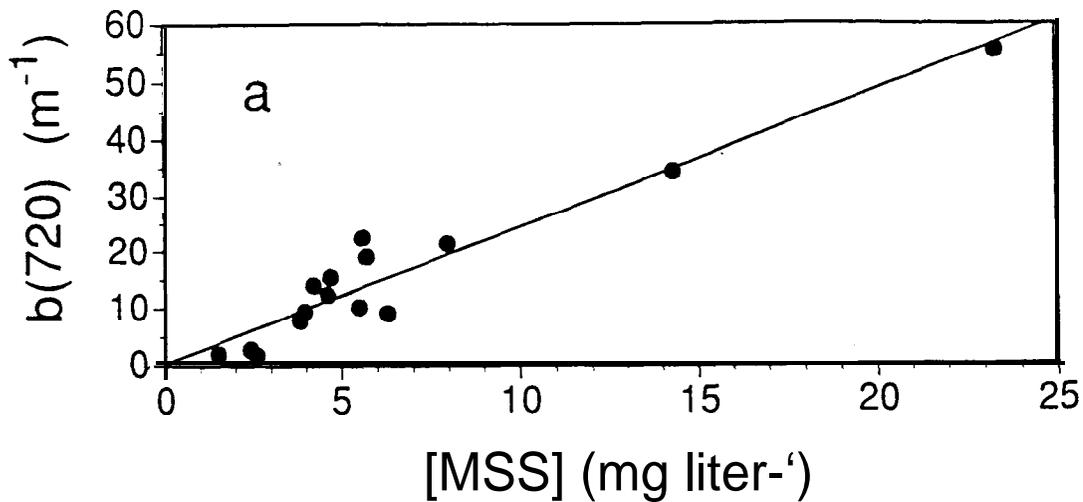


Figure 2. Regressions of optical coefficients against water quality parameters. (a) Scattering coefficient estimated from diffuse attenuation coefficient at 720 nm [ $b(720)$ ] against mineral suspended solids (MSS). (b) Absorption by dissolved matter at 400 nm [ $a_d(400)$ ] against dissolved organic carbon (DOC). (c) Absorption of depigmented particulate material at 400 nm [ $a_x(400)$ ] against total suspended solids (TSS); one outlier (open circle) was omitted from regression.

Rhode River, Mouth  
27 Sep 1988

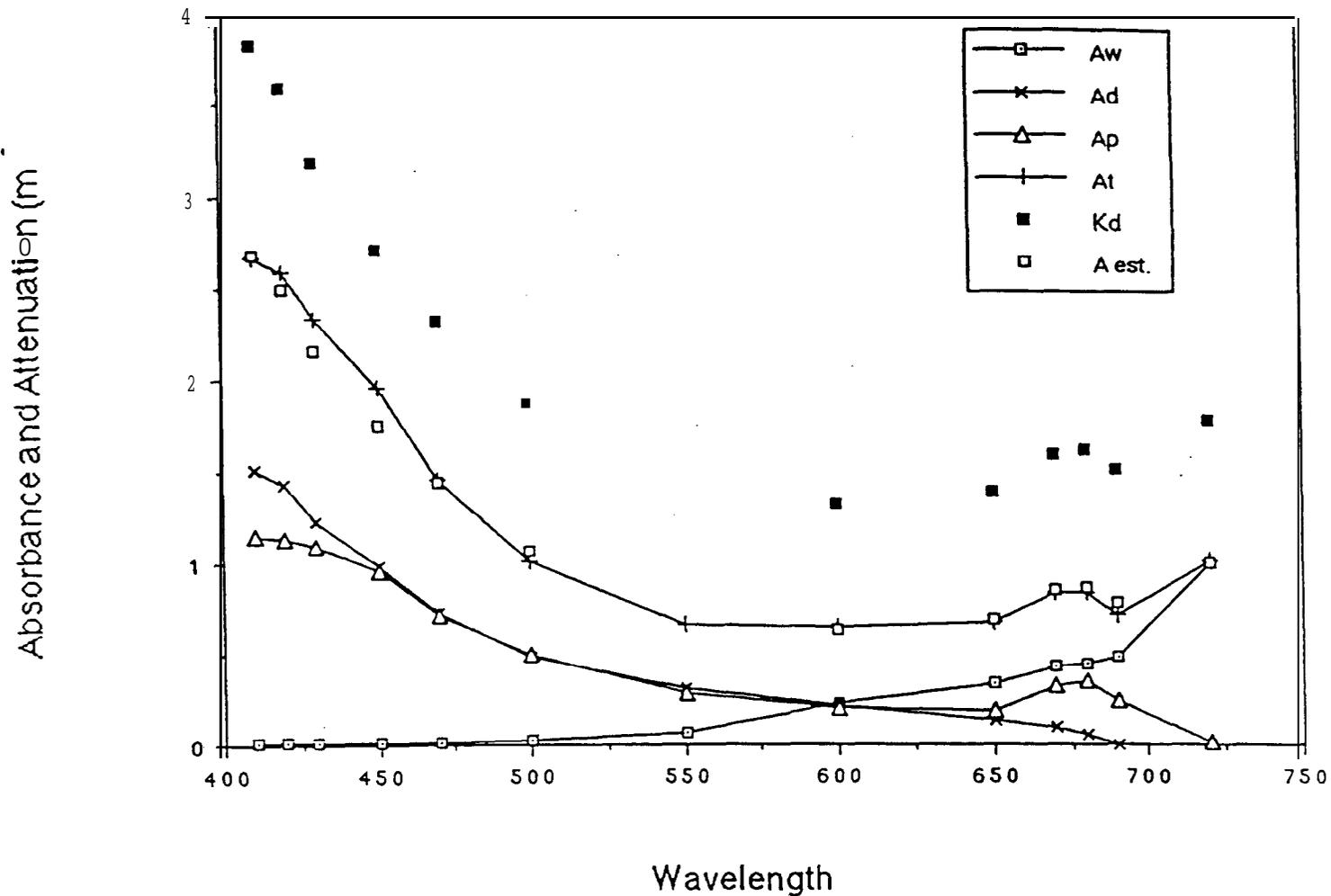


Fig. 3. Measured diffuse attenuation coefficients (filled squares), and extracted absorption coefficients (open squares) determined from field measurements at the Rhode River. Absorption by dissolved substances (Ad) and particulate material (Ap) are determined from laboratory measurements, and absorption due to pure water (Aw) is known from literature. Total absorption (At) is sum of Aw + Ap + Ad.

# Rhode River, MD

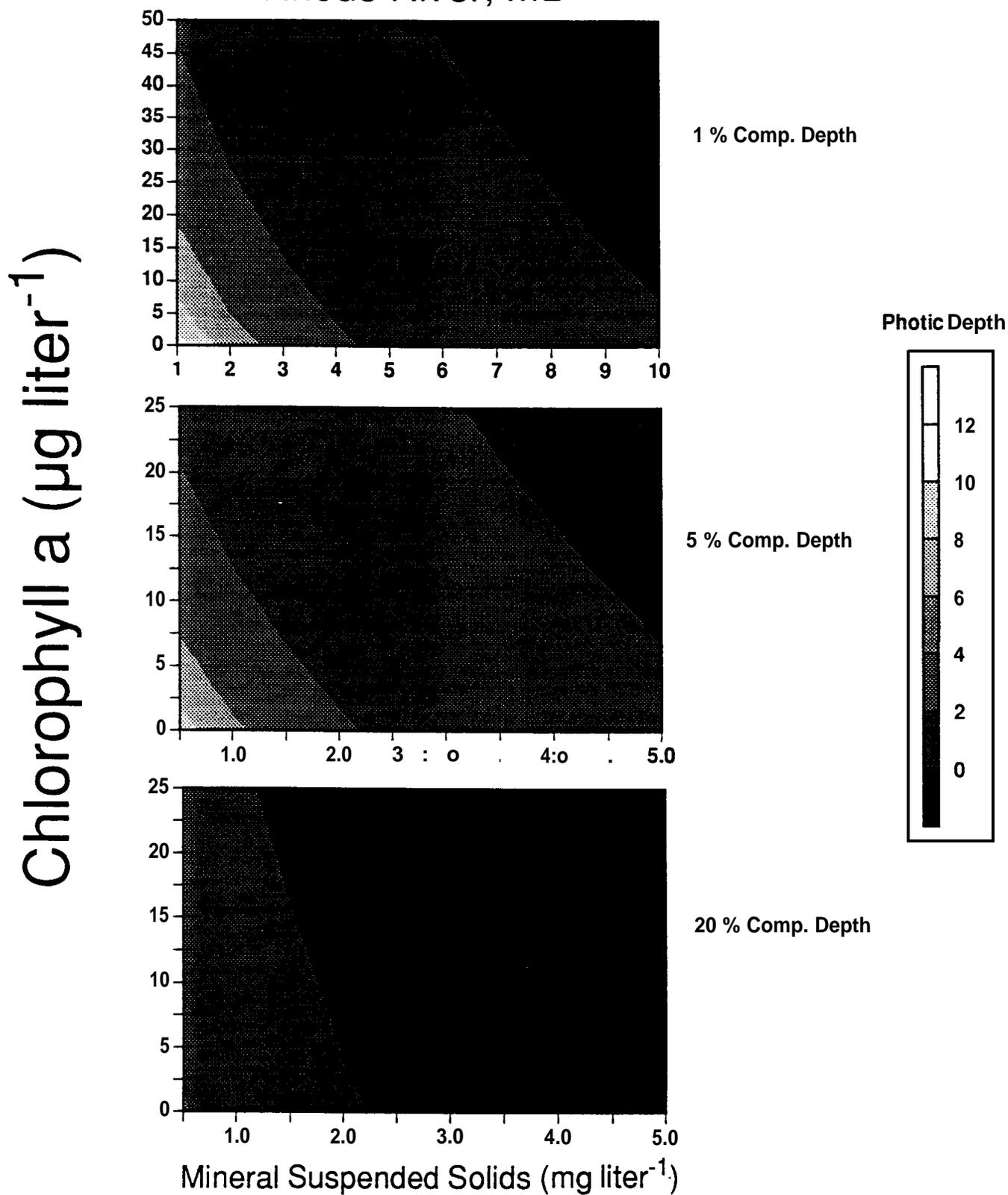


Figure 4. Contours of photic depths as a function of mineral suspended solids and chlorophyll a. Contours are based on depth of penetration of 1% (upper), 5% (middle) and 20% of surface incident irradiance, and are in terms of photosynthetically usable radiation (PUR).

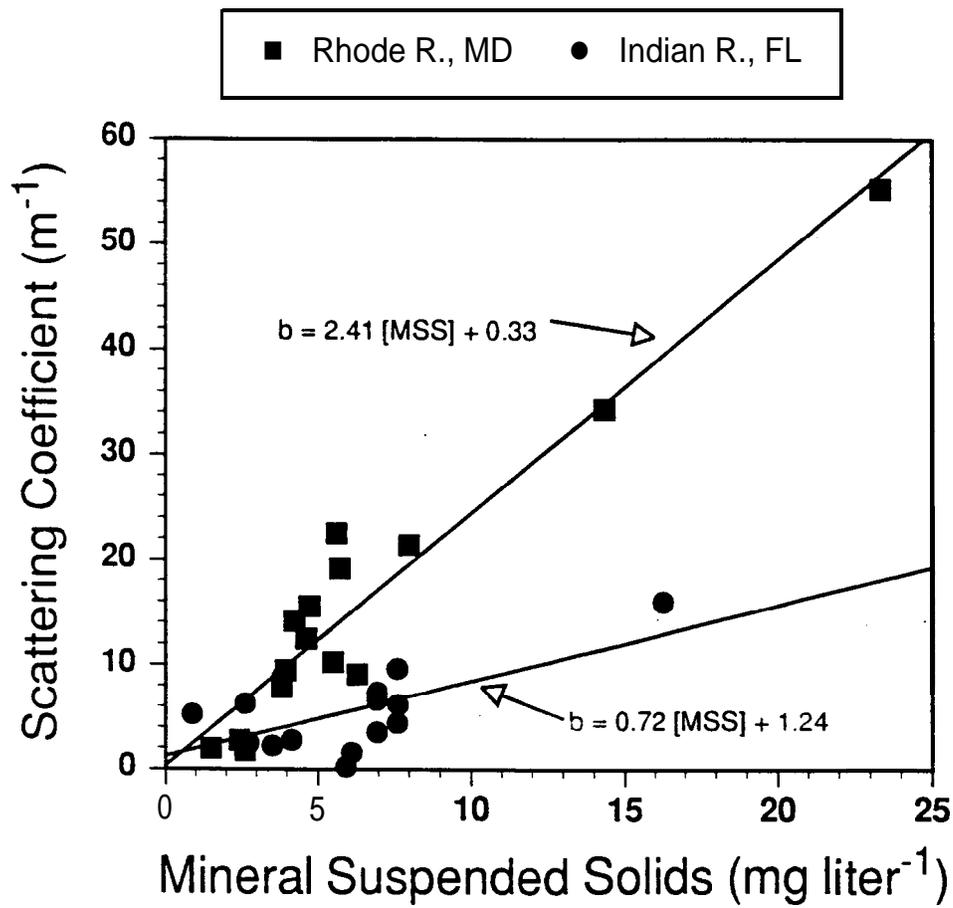


Figure 5. Comparison of scattering coefficients determined in the Indian River, FL, with those determined in the Rhode River, MD.

# Indian River, FL

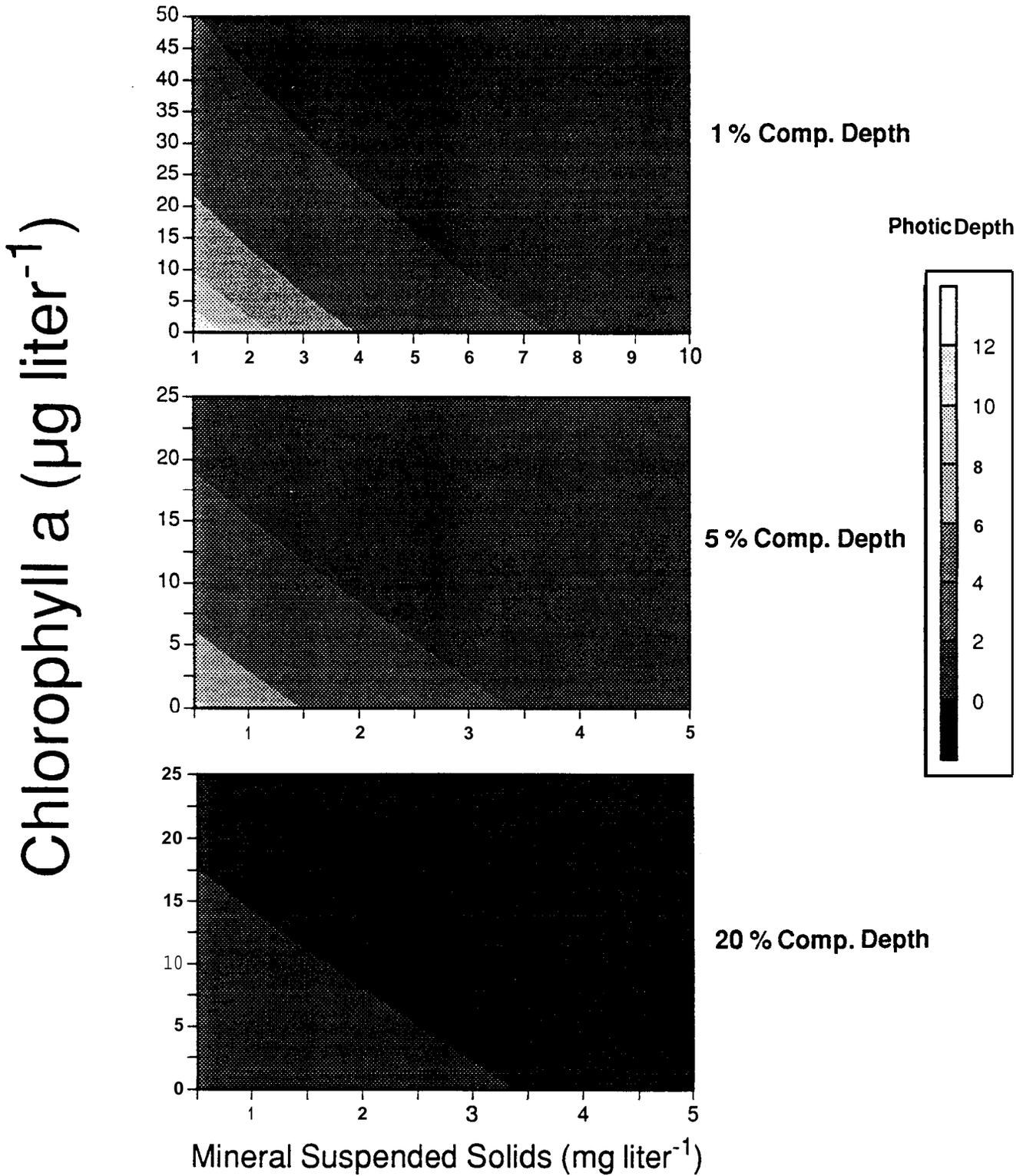


Figure 6. As Fig. 4, but with **photic** depths determined using relationship between scattering coefficients and MSS as determined for the Indian River, FL.

**Neckles**, Hilary A., U.S. Fish and Wildlife Service, LSU Center for Wetland Resources, Baton Rouge, LA

COMPLEX INTERACTIONS AMONG LIGHT-REDUCING VARIABLES IN **SEAGRASS** SYSTEMS: SIMULATION MODEL PREDICTIONS FOR LONG-TERM COMMUNITY STABILITY

Light reaching leaf surfaces of submerged macrophytes is a result of attenuation by the water column and by an epiphytic matrix of algae, microbes, and associated detritus. Whereas scattering and absorption of light by dissolved compounds and suspended particles in the water column is well documented, only recently has the potential for material attached to leaf surfaces to reduce light transmittance been addressed. Studies from temperate and tropical submerged macrophyte communities have measured light attenuation through epiphytic material as an exponential function of epiphytic density (e.g. Bulthuis and Woelkerling 1983, Sand-Jensen and Borum 1983, **Twilley** et al. 1985, Silberstein et al. 1986, Kemp et al. 1988). The spectral selectivity of such attenuation and its ultimate effect on the macrophyte light environment depend on the composition of the epiphytic matrix. **Eelgrass** epiphyton in Chesapeake Bay, for example, is a heterogeneous assemblage of diatoms, cyanobacteria, heterotrophic protists and bacteria, and inorganic and organic debris. Although this epiphytic material attenuates blue wavelengths most rapidly, there are no windows of high transmittance throughout the photosynthetically active spectral range (Table 1). This broad-band attenuation indicates the potential for epiphytic material to reduce the light available for

macrophyte photosynthesis. The relative effect of epiphytic light attenuation on macrophyte production is influenced further by incident irradiance and macrophyte photosynthetic efficiency. **Seagrass** community persistence thus depends on complex interactions among macrophyte photosynthetic characteristics and those environmental variables controlling both water clarity and the accrual of epiphytic biomass..

The environmental factor cited most frequently as causing increases in epiphytic fouling and consequent declines in abundance of submerged macrophytes is anthropogenic nutrient enrichment. Data from **eelgrass** microcosms on Chesapeake Bay suggest that the indirect effects of nutrient enrichment on macrophyte production vary seasonally with levels of other environmental variables and macrophyte light requirements. For example, at irradiances and densities of epiphytic grazers typical of stable **eelgrass** habitat, enrichment with nitrogen and phosphorus to concentrations correlated with regional **eelgrass** declines resulted in reduced macrophyte production during the early summer only. Grazer population densities during this period had not yet reached annual maxima, allowing moderate levels of epiphytic biomass to accumulate; high summer water temperatures may have caused low **eelgrass** photosynthesis:respiration ratios or reduced ability to translocate photosynthate, either of which could have enhanced the relative effect of epiphytic light attenuation on macrophyte production. At higher grazer population densities of late summer, only when grazers were removed did enrichment result in high levels

of epiphytic biomass correlated with reduced macrophyte production. There were no indirect effects of enrichment on macrophyte production during spring and fall, regardless of intermediate effects on epiphytic biomass; this may have been a result of the comparatively low light requirements of **eelgrass** at low temperatures. These experiments show that changes in environmental conditions which increase epiphytic accumulation, such as nutrient enrichment or reduced grazing intensity, can result in seasonal reductions in macrophyte production. Therefore, in areas where light reaching leaf surfaces is mediated by epiphytic attenuation, the distribution of seagrasses predicted from macrophyte light requirements and water column attenuation alone **may** not be realized. The complex interactions among multiple environmental variables, epiphytic biomass, and macrophyte production, however, preclude generalizations from short-term experiments to long-term macrophyte persistence.

Simulation modeling offers a means to relate these experimental results to predictions for community behavior. Incorporating nutrient enrichment as an environmental control into a model of **eelgrass** production in Chesapeake Bay (Wetzel and **Neckles** 1986) allowed the combined effects of various **light-reducing** factors on macrophyte community stability to be assessed. **Eelgrass** photosynthesis was modeled as a hyperbolic function of light at leaf surfaces, which in turn was limited by an epiphytic matrix. Epiphytic biomass in the model increased through the photosynthesis of microflora and was removed by grazers. Nutrient

enrichment was modeled implicitly as the increase in the maximum rate of epiphytic photosynthesis which yielded seasonal epiphytic densities in agreement with data from microcosm experiments.

Nominal environmental conditions in the model represented long-term averages for stable **eelgrass** communities in Chesapeake Bay. Simulated increases in turbidity from a water column light attenuation coefficient ( $k$ ) of  $1.0 \text{ m}^{-1}$  to  $1.5 \text{ m}^{-1}$  reduced **eelgrass** standing stocks but did not result in loss of the community (Table 2). However, a similar increase in turbidity coupled with additional epiphytic light attenuation due to nutrient enrichment or elimination of grazing caused macrophyte loss. These simulations suggest that long-term macrophyte responses to the submarine light regime are influenced by seasonal epiphytic accumulations, and that turbidity criteria reflecting macrophyte light requirements in unenriched waters would not ensure macrophyte persistence under eutrophic conditions. Water quality monitoring programs in **seagrass** habitats should include not only variables controlling water column light attenuation but also those influencing epiphytic accrual.

#### REFERENCES

- Bulthuis, D.A., and Woelkerling, W.J. 1983. BIOMASS ACCUMULATION AND SHADING EFFECTS OF EPIPHYTES ON LEAVES OF THE SEAGRASS, HETEROZOSTERA TASMANICA, IN VICTORIA, AUSTRALIA. *Aquat. Bot.* **16:137-148.**
- Kemp, W.M., Boynton, W.R., Murray, L., Madden, C.J., Wetzel, R.L., and Vera **Herrera**, F. 1988. LIGHT RELATIONS FOR THE **SEAGRASS** THALASSIA TESTUDINUM AND ITS EPIPHYTIC 'ALGAE IN A TROPICAL ESTUARINE ENVIRONMENT. pp. 193-206. **In:**A. Yanez-Arancibia and J.W. Day, Jr. (Eds.) *Ecology of Coastal Ecosystems in the Southern Gulf*

of Mexico: the Terminos Lagoon Region. Inst. Cienc. **del** Mar y Limnol. UNAM, Coast. Ecol. Inst. LSU, Editorial Universitaria, Mexico.

Sand-Jensen, K. and Borum, J. 1983. REGULATION OF GROWTH OF **EELGRASS** (ZOSTERA MARINA L.) IN DANISH COASTAL WATERS. Mar. Tech. Soc. J. 17:15-21.

Silberstein, K., Chiffings, S.W., and **McComb**, A.J. 1986. THE LOSS OF **SEAGRASS** IN **COCKBURN** SOUND, WESTERN AUSTRALIA. III. THE EFFECT OF EPIPHYTES ON PRODUCTIVITY OF POSIDONIA AUSTRALIS HOOK. F. Aquat. Bot. 24:355-371.

**Twilley**, R.R., Kemp, W.J., Staver, K.W., Stevenson, J.C., and Boynton, W.R. 1985. NUTRIENT ENRICHMENT OF ESTUARINE SUBMERSED VASCULAR PLANT COMMUNITIES. 1. ALGAL GROWTH AND EFFECTS ON PRODUCTION OF PLANT AND ASSOCIATED COMMUNITIES. Mar. Ecol. Prog. Ser. 23:179-191.

Wetzel, R.L., and **Neckles**, H.A. 1986. A MODEL OF ZOSTERA MARINA L. PHOTOSYNTHESIS AND GROWTH: SIMULATED EFFECTS OF SELECTED **PHYSICAL-CHEMICAL** VARIABLES AND BIOLOGICAL INTERACTIONS. Aquat. Bot. 26:307-323.

Table 1. Spectral epiphytic attenuation coefficients ( $\text{cm}^2 \text{mg}^{-1}$ ) from **eelgrass** microcosms on Chesapeake Bay (April-June 1988). Wavelengths represent midpoints of 10nm bands. Coefficients were calculated from the proportion of incident light transmitted through epiphytic suspensions ( $n=137$ ) following Beer's law.

Wavelength (nm)	Coefficient
410	.45
441	.41
488	.36
507	.35
570	.28
589	.28
625	.27
656	.27
694	.26

Table 2. Model predictions for annual maximum **eelgrass** leaf biomass ( $\text{gC m}^{-2}$ ) under varying environmental conditions in Chesapeake Bay. Asterisks indicate loss of the community over a 10-year simulation. Average conditions for stable **eelgrass** communities are represented by grazers present,  $k=1.0 \text{ m}^{-1}$ , and ambient nutrient concentrations. Enriched nutrient concentrations represent 3x ambient, based upon observed differences between presently and formerly vegetated sites in the region.

Grazers	<u>Light attenuation</u>	<u>Dissolved nutrient concentrations</u>	
	$k \text{ (m}^{-1}\text{)}$	Ambient	Enriched
Present	1.0	141	96
	1.25	136	104
	1.5	107	*
Absent	1.0	105	*
	1.25	53	*
	1.5	*	*

Dennison, William C., Horn Point Environmental Labs, University of Maryland, P.O. Box 775, Cambridge, MD 21613

PHOTOSYNTHETIC AND GROWTH RESPONSES OF TROPICAL AND TEMPERATE SEAGRASSES IN RELATION TO SECCHI DEPTH, LIGHT ATTENUATION AND DAILY LIGHT PERIOD

**Seagrass** Declines

Worldwide, populations of seagrasses have been affected by human activities. In particular, environmental perturbations resulting in reductions in the light available to seagrasses have been implicated in numerous **seagrass** declines (e.g., den Hartog and Polderman 1975, Peres and **Picard** 1975, Orth and Moore 1983, Kemp et al. 1983, Cambridge and McComb 1984; **Dennison** et al. 1989). Well-documented case studies from Europe (e.g., Giesen et al **1990**), North America (e.g., Costa 1988) and Australia (e.g., Cambridge and McComb 1984) have demonstrated the ubiquitous nature of the problems associated with nutrient enrichments in coastal waters. In Chesapeake Bay, **seagrass** and freshwater macrophyte declines have occurred in all reaches of the estuary, from freshwater to polyhaline regions (Orth and Moore 1983). Freshwater macrophyte resurgences have been recently observed in some areas of Chesapeake Bay (Carter and Rybicki 1986, Orth and Nowak **1990**), but **seagrass** and freshwater macrophyte abundance still remains near its lowest levels in recorded history. Agricultural development and urbanization of the Chesapeake Bay watershed have led to changes in sediment runoff and nutrient loadings, leading to changes in water quality that affect seagrass. Most of the nutrient and sediment inputs to Chesapeake Bay are derived from non-point sources and therefore it is rather

difficult to quantify historical patterns of water quality, yet the well-documented, large scale **seagrass** declines in Chesapeake Bay are evidence for changes in historical patterns of water quality. Experimental mesocosms were used to test macrophyte responses to increased nutrient loadings in Chesapeake Bay (Kemp et al. 1983), with decreases in macrophyte biomass correlated with increased nutrient loading rates, epiphyte and phytoplankton biomass.

The central role of light availability for **seagrass** has been demonstrated in numerous field, laboratory and modelling studies. The low light environments of coastal waters has led to various **seagrass** adaptations to ameliorate some of the suboptimal light conditions through pigment composition, biochemical and structural adaptations (Wiginton and **McMillan** 1979, **Dennison** and **Alberte**, 1986). In spite of these adaptations, experimental evidence demonstrating light limitation of **seagrass** growth has been obtained by in situ manipulations of light intensity (**Backman** and **Barilotti** 1976, **Bulthuis** 1983, **Dennison** and **Alberte** 1985, **Williams** and **Dennison** 1990). Changes in year-to-year variability in light availability leading to changes in **seagrass** abundance have been reported (**Wetzel** and **Penhale** 1983). In addition, a model that relates instantaneous photosynthetic responses of **seagrass** to light availability provides a means of relating changes in light attenuation in the water column to changes in **seagrass** productivity and depth penetration (**Dennison** 1987). This model ( $H_{\text{sat}}/H_{\text{comp}}$ ) provides a predicted relationship

between light attenuation coefficient ( $k$ ;  $\text{m}^{-1}$ ) and maximum depth limit of **seagrass** in which the depth limit (in m) is equal to  $1.6/k$ .

#### Secchi Depth/Light Attenuation Conversion

The use of a Secchi disc to estimate light attenuation of the water column is based on a convenient coincidence. Light that is visible to the human eye is remarkably similar in terms of wavelength to light available to plants for photosynthesis (Photosynthetically Active Radiation = 400 - 700 nm). The Secchi depth measurement has the advantage of being a simple field measurement and has been in use for over a century. More recently, photoelectric light meters have been commercially available and are used extensively to measure underwater light fields. These light meters ideally measure light as  $\mu\text{mol}$  of quanta between 400 - 700 nm wavelengths. The measurement of light quanta (= photons) is relevant, since photosynthesis is a quantum process. Discrepancies in light attenuation measured by the Secchi disc versus light attenuation measured by a photosynthetically active radiation light meter are addressed through the application of a conversion calculation.

Conversion factors between Secchi depth and light attenuation coefficient ( $k$ ) were originally developed for clear ocean waters and more recently formulated for various estuaries. Considerable discussion over the relative merits of making such conversions has occurred, both historically (e.g., Poole and

Atkins 1929) and recently (e.g., Preisendorfer 1986, Megard and Berman 1989). Developing a relevant conversion factor is particularly important when utilizing historical data sets containing Secchi data (e.g., Giesen et al. 1990). As simple a measurement that the Secchi depth appears, there are, however, many subjective influences on making such a measurement which have been codified into 10 "**laws of the Secchi disk**"

(Preisendorfer 1986). In spite of the subjective aspects to measuring Secchi depth, in open ocean situations they appear to be as accurate and precise as photoelectric sensors (Megard and Berman 1989).

The application of Secchi depth measurements in determining light attenuation in turbid, coastal waters has problems not encountered in open ocean situations. Organic detritus from decaying plant material (e.g., salt marsh plants, seagrasses and terrestrial plants) can attenuate light both as particulate matter and dissolved matter. Water in some estuaries is often tea-colored as a result of the decomposing plant matter that has leached **humic** substances. As a result of this colored material in the water column, discrepancies between what the human eye perceives and what the photoelectric light meter measures becomes acute. Secchi depths in these portions of the estuary may not be good estimates of light attenuation. Large adjustments in the conversion factor between Secchi depth and light attenuation coefficient are required in these regions.. Simultaneous measurements of Secchi depth and light attenuation need to be

performed for each water body in order to develop accurate conversion factors.

Use of a photoelectric light meter is obviously an easy way to avoid the problems of developing conversion factors (e.g., Preisendorfer 1986). Equally important in a turbid estuary is the precise measurement water depth that must accompany a Secchi or light reading. Since light extinction is an exponential decay function, relatively small changes in the measurement of water depth in turbid waters lead to large changes in the calculated light attenuation coefficient. Obviously sea state affects the accurate measurement of water depth and leads to an eleventh "**law of the Secchi disk**" for estuaries.

Conversion factors for various water bodies have been formulated by simultaneous Secchi depth and light attenuation measurements and is a matter of considerable dispute. Even the original conversion factor of  $k = 1.7/\text{Secchi depth}$  proposed by Poole and Atkins (1929) using measurements taken in the English Channel has been recalculated by Walker (1980) to be 1.45 and by Megard and Berman (1989) to be 1.6. Conversion factors formulated for oceanic waters are not directly applicable to coastal waters, however. Lower conversion factors than the Poole and Atkins value of 1.7 have been determined for turbid waters; Holmes (1970) = 1.44, Walker (1980) = 1.46. A recent study carried out to cover the Secchi depth range of 0.5 to 2.0 m incorporated measurements made by 8 independent researchers and determined an average conversion factor of  $k = 1.65/\text{Secchi depth}$

(Giesen et al. 1990). Differences in conversion factors lead to small changes in the determination of light attenuation coefficients in very turbid waters. For example, only a 5% discrepancy between k values occurs when comparing conversion factors of 1.4 vs. 1.7 in water columns with a Secchi depth of 0.5 m.

#### Minimum Light Requirements

Minimum light requirements for seagrasses can be determined where the maximum depth limit and light attenuation coefficient are simultaneously measured. Percent of incident light that corresponds to maximum depth penetration of seagrasses can be determined by using the exponential light attenuation relationship:

$$I_z = I_0 * e^{-kz} \quad (1)$$

where  $I_z$  is the light at depth  $z$ ,  $I_0$  is the light at the water surface,  $k$  is the light attenuation coefficient and  $z$  is the depth. Assuming that the minimum light requirement is the light level at the maximum depth penetration of the seagrass, the depth  $z$  in equation (1) can be determined rearranging equation (1) to:

$$I_z/I_0 = e^{-kz} \quad (2)$$

to yield the fraction of light remaining at depth  $z$ . Multiplying the fraction  $I_z/I_0$  by 100 yields a percent and gives the minimum light requirement as a time-integrated proportion of surface irradiance necessary to sustain **seagrass** at its deepest habitat. The conversion between Secchi depth to  $k$  that was used for

literature values is  $k = 1.65/\text{Secchi depth}$  (from Giesen et al., 1990).

The average minimum light requirement for freshwater macrophytes from 88 lakes in Canada was determined to be **21.4+2.4%** (Chambers and Kalff, 1985). The minimum light requirements for seagrasses range from 2.5 to **24.4%**, depending upon species (Table 1). The variation in minimum light requirements can be attributed to differences in physiological and morphological adaptations of the various species. **Seagrass** genera with low minimum light requirements such as Heterozostera and Halophila grow deeper than other **seagrass** species where they co-occur (Shepherd and Robertson 1989, Coles et al. 1989, respectively), also indicating that minimum light requirements vary between species. The predominant temperate water **seagrass** in the U.S., Zostera marina, has minimum light requirements that have been independently determined to be about 20% from three different locations. Z. marina in Chesapeake Bay has minimum light requirements of **23.9%**, integrated over the entire year (see Moore, this volume).

## References

- Backman**, T.W. and D.C. Barilotti. 1976. Irradiance reduction: effects on standing crops of the eelgrass, Zostera marina, in a coastal lagoon. Mar. Biol. 34: 33-40.
- Beer, S. and Y. Waisel. 1982. Effects of light and pressure on photosynthesis in two seagrasses. Aq. Bot. 13: 331-337.
- Borum, J. 1983. The quantitative role of macrophytes, epiphytes, and phytoplankton under different nutrient conditions in Roskilde Fjord, Denmark. **Proc.** Int. Symp. Aquat. Macrophytes, Nijmegen. pp. 35-40.
- Bulthuis, D.A. 1983. Effects of in situ light reduction on density and growth of the **seagrass** Heterozostera tasmanica (Martens ex **Aschers.**) den Hartog in Western Port, Victoria, Australia. J. Exp. Mar. Biol. Ecol. 67: 91-103.
- Cambridge, M.L. and A.J. McComb. 1984. The loss of seagrasses in **Cockburn** Sound, Western Australia. I. The time course and magnitude of **seagrass** decline in relation to industrial development. Aq. Bot. 20: 229-243.
- Carter, V. and N.B. Rybicki. 1986. Resurgence of submersed aquatic macrophytes in the tidal Potomac River. Estuaries 9: 368-375.
- Chambers, P.A. and J. Kalff. 1985. Depth distribution and biomass of submersed aquatic macrophyte communities in relation to Secchi depth. Can. J. Fish. Aquat. Sci. 42: 701-709.
- Coles, R.G., I.R. Poiner and H. **Kirkman**. Regional studies -- seagrasses on North-Eastern Australia. In: Biology of Seagrasses: A Treatise on the Biology of Seagrasses With Special Reference to the Australian Region. Eds. A.W.D. Larkum, A.J. McComb and S.A. Shepard. Elsevier, Amsterdam. pp. 261-278.
- Costa, J.E. 1988. Distribution, production and historical changes in abundance of **eelgrass** (Zostera marina) in Southeastern Massachusetts. **PhD** Thesis, Boston University, 354 pp.
- den Hartog, C. and P.J.G. Polderman. 1975. Changes in the **seagrass** populations of the Dutch Waddenzee. Aq. Bot. 1: 141-147.
- Dennison, W.C. and R.S. Alberte. 1985. Role of daily light period in the depth distribution of Zostera marina (eelgrass). Mar. Ecol. Prog. Ser. 25: 51-61.

- Dennison, W.C. and R.S. Alberte. 1986. Photoadaptation and growth of Zostera marina L. (eelgrass) transplants along a depth gradient. J. Exp. Mar. Biol. Ecol. 98: 265-282.
- Dennison, W.C. 1987. Effects of light on **seagrass** photosynthesis, growth and depth distribution. Aq. Bot. 27: 15-26.
- Drew, E.A. 1978. Factors affecting photosynthesis and its seasonal variation in the seagrasses Cymodocea nodosa (Ucria) **Aschers**, and Posidonia oceanica (L.) Delile in the Mediterranean. J. Exp. Mar. Biol. Ecol. 31: 173-194.
- Giesen, W.B.J.T., M.M. van Katwijk, and C. den Hartog. 1990. **Eelgrass** condition and turbidity in the Dutch **Wadden** Sea. Aq. Bot. 37: 71-85.
- Kemp, W.M., R.R. Twilley, J.C. Stevenson, W.R. Boynton, and J.C. Means. 1983. The decline of submerged vascular plants in Upper Chesapeake Bay: Summary of results concerning possible causes. Mar. Tech. **Soc. J.** 17: 78-89.
- Holmes, R.W. 1970. The Secchi disk in turbid coastal waters. Limnol. Oceanogr. 15: 688-694.
- Megard, R.O. and T. Berman. 1989. Effects of algae on the Secchi transparency of the southeastern Mediterranean Sea. Limnol. Oceanogr. 34: 1640-1655.
- Orth, R.J. and K.A. Moore. 1983. Chesapeake Bay: An **unprecedented** decline in submerged aquatic vegetation. Science. 222: 51-53.
- Orth, R.J. and J.F. Nowak. 1990. Distribution of submerged aquatic vegetation in the Chesapeake Bay and tributaries and Chincoteague Bay -- 1989. Final report to EPA, Ches. Bay Prog., Annapolis. 249 pp.
- Ostenfeld, C.H. 1908. On the ecology and distribution of the grass-wrack (Zostera marina) in Danish waters. Report of the Danish Biol. Station, Copenhagen. 62 pp.
- Peres, J.M. and J. **Picard**. 1975. Causes de la rarefaction et de la disparition des **herbiers** de Posidonia oceanica sur les **cotes Francaises** de la Mediterranee. Aq. Bot. 1: 133-139.
- Poole, H.H. and W.R. Atkins. 1929. Photo-electric measurements of submarine illumination throughout the year. J. Mar. Biol. Assoc. U.K. 16: 297-324.
- Preisendorfer, R.W. 1986. Secchi disk science: Visual optics of natural waters. Limnol. Oceanogr. 31: 909-926.

Shepard, S.A. and E.L. Robertson. 1989. Regional studies -- seagrasses of South Australia, Western Victoria and Bass Strait. In: Biology of Seagrasses: A Treatise on the Biology of Seagrasses With Special Reference to the Australian Region. Eds. A.W.D. Larkum, A.J. **McComb** and S.A. Shepard. Elsevier, Amsterdam. pp. 211-229.

Vicente, V.P. and J.A. **Rivera**. 1982. Depth limits of the **seagrass** Thalassia testudinum (Konig) in Jobos and Guatanilla Bays, Puerto Rico. Carib. J. Sci. 17: 73-79.

Walker, T.A. 1980. A correction to the Poole and Atkins Secchi disc/light attenuation formula. J. Mar. Biol. Assoc. U.K. 60: 769-771.

Wetzel, R.L. and P.A. **Penhale**. 1983. Production ecology of **seagrass** communities in the lower Chesapeake Bay. Mar. Tech. Soc. J. 17: 22-31.

Wiginton, J.R. and C. **McMillan**. 1979. Chlorophyll composition under control light conditions as related to the distribution of seagrasses in Texas and the U.S. Virgin Islands. Aq. Bot. 12: 321-344.

Williams, **S.L** and W.C. Dennison. 1990. Light availability and diurnal growth of a green macroalga (Caulerpa cuoressoides) and a **seagrass** (Haloohila decioiens). Mar. Biol. 106: 437-443.

Table 1. Maximum depth limit, light attenuation coefficient (k) and minimum light requirements of various species of seagrasses. Where Secchi depths were reported,  $k = 1.65/\text{Secchi depth}$  (Giesen et al., 1990). Minimum light requirement calculated as percent light at the maximum depth limit using  $100 \times I_z/I_0 = e^{-kz}$ . Range of maximum depth limit and k values, and means  $\pm$  S.E. of minimum light requirement given in locations with **multiple** data points.

Genus Species(Reference)	Location	Maximum Depth Limit (m)	k; Light Attenuation Coefficient ( $m^{-1}$ )	Minimum Light Requirement (%)
<u>Thalassia</u> <u>testudinum</u> (1)	South coast, Puerto Rico	1.0-5.0	0.35-1.50	<b>24.4<math>\pm</math>4.2</b>
<u>Zostera</u> <u>marina</u> (2)	Kattegat, Denmark	3.7-10.1	0.16-0.36	<b>20.1<math>\pm</math>2.1</b>
<u>Zostera</u> <u>marina</u> (3)	Roskilde Fjord, Denmark	2.0-5.0	<b>0.32-0.92</b>	<b>19.4<math>\pm</math>1.3</b>
<u>Zostera</u> <u>marina</u> (4)	Woods Hole, MA, U.S.A.	6.0	0.28	18.6
<u>Syringodium</u> <u>filiforme</u> (5)	Hobe Sound, FL, U.S.A.	1.9	0.93	17.2
<u>Halodule</u> <u>wrightii</u> (5)	Hobe Sound, FL, U.S.A.	1.9	0.93	17.2
<u>Posidonia</u> <u>oceanica</u> (6)	Malta, Mediterranean	35.0	0.07	9.2
<u>Cymodocea</u> <u>nodosa</u> (6)	Malta, Mediterranean	38.5	0.07	7.3
<u>Heterozostera</u> <u>tasmanica</u> (7)	Victoria, Australia	3.8-9.8	0.36-0.85	<b>5.0<math>\pm</math>0.6</b>
<u>Halophila</u> <u>decipiens</u> (8)	St. Croix, Caribbean	40.0	0.08	4.4
<u>Halophila</u> <u>decipiens</u> (5)	Hobe Sound, FL, U.S.A.	4.0	0.93	2.5
<u>Halophila</u> <u>stipulacea</u> (9)	Gulf of <b>Eilat</b> , Red Sea	50.0	0.07	3.0

References:

- (1) Vicente and **Rivera**, 1982
- (2) Ostenfeld, 1908
- (3) Borum, 1983
- (4) Dennison, 1987
- (5) Kenworthy, 1990
- (6) Drew, 1978
- (7) Bulhuis, 1983
- (8) Williams and Dennison, 1990
- (9) Beer and Waisel, 1982

SESSION 4; DEVELOPMENT AND IMPLEMENTATION OF FEDERAL CRITERIA AND STATE STANDARDS.

Batiuk, Richard., U.S. Environmental Protection Agency,  
Chesapeake Bay Liaison Office, Annapolis, MD.

COORDINATING THE SYNTHESIS OF TWO DECADES OF CHESAPEAKE BAY SAV RESEARCH

Based on a synthesis and interpretation of findings from SAV research and monitoring programs since the **1970's**, habitat requirements and restoration goals for SAV have been established for Chesapeake Bay and its tributaries.

Habitat requirements were derived through four study areas covering all salinity regimes of the Bay. Interpretation of transplant and monitoring data from the upper Chesapeake Bay region and a decade of data tracking the revegetation of the upper tidal Potomac River yielded the habitat requirements for tidal fresh and oligohaline species. A variety of research and monitoring projects conducted since the 1970s and focused on the **Choptank** River provided data necessary to establish habitat requirements for mesohaline regions of the bay. Transplant and monitoring data supported by long term research in the York River were used to quantify the habitat needs of polyhaline SAV species.

Through a synthesis and multi-investigator interpretation of findings from these study areas, habitat requirements for light attenuation coefficient, secchi depth, total suspended solids, chlorophyll a, dissolved inorganic nitrogen and dissolved inorganic phosphorus were developed. The applicability of these

habitat requirements outside of the four study areas was established through a comparative analysis of 1987 and 1989 water quality and SAV distribution data **baywide** and the corresponding habitat requirements. The relative importance of and interactions between each of these parameters was fully explored through a conceptual model which characterizes direct and indirect impacts on SAV growth. The impacts of herbicides on SAV were also reviewed and habitat requirements set for selected compounds.

**Baywide** and regional SAV distribution and species diversity restoration goals were developed through analysis of SAV distribution and abundance survey data collected since the early **1970s**, review of historical species distributions and geographic overlays of factors influencing SAV propagation.

The habitat, distribution and species diversity restoration goals established through this synthesis process will prove valuable in planning, implementing and measuring the resultant success of the water quality and resource management actions necessary to ensure restoration of this critical component of the Chesapeake Bay ecosystem.

April, Robert W., and Kennard Potts, Criteria Branch, U.S. Environmental Protection Agency, Washington D.C.

#### LIGHT REQUIREMENTS OF SEAGRASSES: RELATIONSHIP TO THE FEDERAL WATER QUALITY PROGRAM

Under the Clean Water Act, the states have the primary role in water pollution control, with the Federal government serving in an advisory and supervisory capacity. Federal water quality criteria are guidance, not regulations. States may adopt Federal criteria, use other alternatives, or simply not issue a standard for a particular parameter, subject to EPA review.

Early in the Federal program (in the early **1970's**) two criteria were issued which are relevant to light requirements of seagrasses. These covered the parameters color, and suspended solids and turbidity. 45 of 57 States and Territories have adopted water quality standards based on these criteria. The present Federal program is largely oriented toward controlling impacts on aquatic animal life caused by toxic chemicals. A detailed methodology, based on a specified set of tests, is used to produce criteria. While there is a provision for setting criteria based on plant impacts, it has never been used, largely because plants are generally more resistant to toxic chemicals.

Problems due to diminished light are quite site-specific. Background levels of transparency and non-point source discharges are of major importance. Given these factors, it may be most appropriate to pursue regulatory solutions on a State by State basis.

Development of a new or revised Federal criteria would

require a demonstration of need for a Federal solution and a defensible methodology for setting a Federal criteria. It is not clear that any new criteria would be substantially more stringent than the existing ones, which basically call for no more than a 10% reduction in the compensation depth. It would still require work at the State level to implement standards. A new Federal program requiring States to 'adopt biological criteria (criteria based on the health of the ecology as a whole) is promising as an alternative regulatory tool for implementing light standards to protect seagrasses.

Coombs, Marge., Florida Dept. of Environmental Regulation,  
Tallahassee, FL.

#### FLORIDA'S WATER QUALITY STANDARDS PROGRAM

Florida's water quality standards system includes the following elements: a classification system for waterbodies of the state, including designated beneficial uses, an antidegradation policy, and criteria to protect the designated beneficial uses. Criteria are of 2 types: narrative criteria, which give a statement of a general goal (e.g., no chronic toxicity), and numeric, which are specific, defined levels for a parameter which should protect the designated beneficial uses. Various relief mechanisms are available for those instances where the state-wide criteria are inapplicable.

The rule adoption process (including adoption or revision of water quality criteria) involves the following: an economic impact analysis, public participation in workshops and possibly administrative hearings, and adoption by the Environmental Regulation Commission. Only the ERC may adopt state water quality standards or criteria.

Existing criteria which are related to the protection of seagrasses include the following: acute and chronic toxicity (none shall be present); nutrients (**can't** cause imbalance in aquatic flora or fauna); injurious substances, or substances which produce adverse impacts (none shall be present); turbidity (not to exceed 29 **NTUs** above natural background); and transparency (**depth** of the compensation point for photosynthetic activity shall not be reduced by more than 10% as compared to

natural background). "Natural **background**" water quality should be distinguished from "**background**".

Florida's antidegradation policy for surface waters states that discharges or other activities: a) cannot degrade the ambient water quality below the criteria established for the classification for the particular water body; b) can degrade the ambient water quality only if the facility of activity is shown to be clearly in public interest, taking into account: 1. public health, safety, and welfare, 2. conservation of fish and wildlife, endangered and threatened species (including their habitats), 3. fishing and recreational values, and marine productivity, and 4. whether the project is consistent with any approved SWIM plan for the water body. In addition to a) and b) it must be demonstrated that the following are not feasible: reuse, or alternatives such as land application or other discharge locations.

Formal implementation of water quality standards and criteria is most commonly done through the permitting programs of various governmental agencies. Design or performance criteria, which may be included in permits in order to implement the water quality criteria or other forms of regulation, should not be confused with the criteria themselves. Guidance, which may aid in the interpretation of narrative criteria or the public interest permitting criteria, may be used in addition to the numeric criteria. Such guidance may be useful where site-specific conditions do not allow for state-wide application. An example

of such guidance could be specific light requirements for Halodule in the **Hobe** Sound area of the Indian River, but which may not be the same for Halodule in the Big Bend area.

Future water quality standards efforts may include biocriteria. Additional data on cause and effect relationships between pollutants and seagrasses would aid in the development of biocriteria, in the implementation of existing standards, and in the development of broader policy and management decisions.

SESSION 5; THE LOGISTICS AND SCOPE **OF** STATE AND LOCAL WATER QUALITY MONITORING.

Alleman, Richard W., Metro-Dade County Dept. of Environmental Resources Management, Miami, FL.

A SYNOPSIS OF THE WATER QUALITY AND MONITORING PROGRAM IN BISCAYNE BAY, FLORIDA

In 1978, the local government and academic communities of Dade County, Florida acknowledged that a serious gap existed in the knowledge of the general water quality in Biscayne Bay on a comprehensive basis. A long-term, Bay-wide monitoring program was considered essential for a concerted planning and resource management effort. The Department of Environmental Resources Management proposed a relatively modest, but geographically comprehensive water quality monitoring program. The original goals of the monitoring program were to 1) establish a baseline of water quality data, 2) provide data for use either directly or indirectly in other key study efforts and 3) detect temporal trends in water quality. Funding sources have varied over the years, however, the original Program has been kept relatively intact. Improvements have been made by adding collection stations, parameters, and auxiliary components such as sediment and bivalve tissue analysis. The Program now benefits from improved quality assurance and comparability through various modifications ranging from sampling protocol to analytical techniques. Lessons have also been learned about reliable field equipment and redundancy of field gear. A computerized database has become essential to manage the volume of data, and presently

requires at least as much time and thought investment as the collection phase. The uses of the data must be considered, as well as, the end users. The Biscayne Bay Water Quality Monitoring Program has met the original stated goals and more. Despite variability in water quality as a result of natural processes and human activities, adequate data now exists to determine changes from **"typical"** conditions and to identify statistically significant temporal and geographic trends. The monitoring program increases in value with every month and continues to be an essential component of the effort to protect, restore and enhance Biscayne Bay.

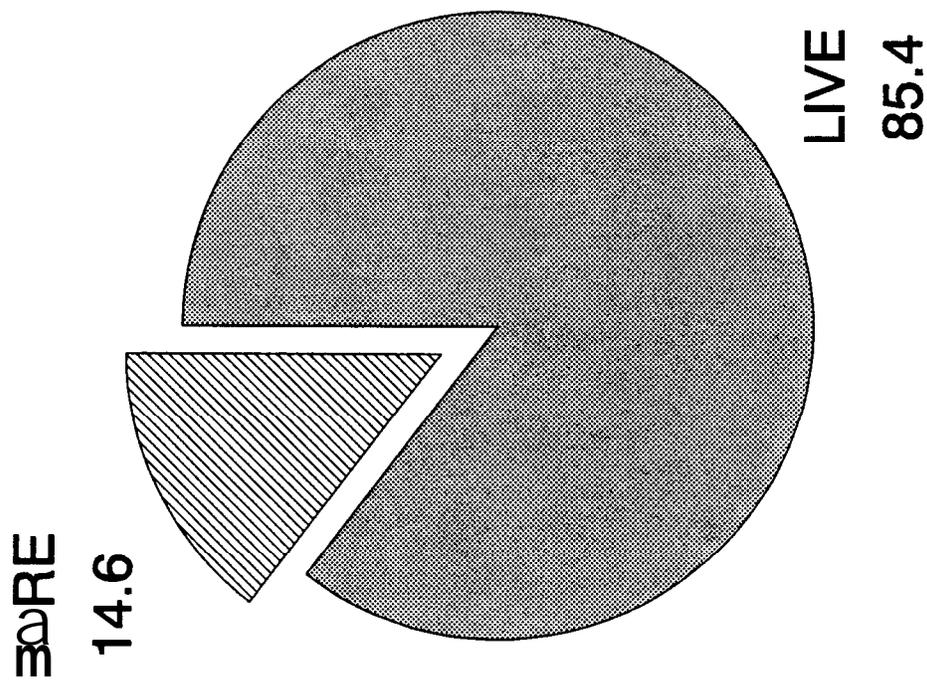
We have attempted to characterize water clarity with a variety of parameters. Samples are collected and analyzed for color, nephelometric turbidity and suspended solids. Water column photosynthetically active radiation (PAP) attenuation is measured in the field at 56 stations monthly. A Li-Cor, Inc. LI-1000 datalogger with two 5 pi sensors is used to record the PAP measurements in the water column. Average PAP attenuation coefficients (K) are then calculated and compiled in a database.

About 15 percent of the bottom of Biscayne Bay is barren (Fig. 1) excluding naturally barren bottom habitats such as hardground areas with thin veneers of sediment. Besides dredged channels, all of this bottom type lies within about three meters of the surface. Figure 2 suggests that a healthy bottom habitat in Biscayne Bay will not tolerate an average **K** greater than about 0.7 at three meters and, that even at a depth of one meter,

average **K** cannot exceed about 0.8. Barren bottom types in Biscayne Bay are characteristically associated with high average water column PAR attenuation values.

Comparisons of average turbidity, color and suspended solids values with average **K** values give variable results. Nephelometric turbidity does not appear to provide a highly reliable means of predicting light attenuation. Figure 3 shows a considerable degree of commingling of live and barren bottom types. Color, on the other hand, seems to be a better means of predicting water clarity, and Figure 4 shows that the live bottom types cluster toward the low end of the **K** scale with less overlap. Indeed, correlation coefficients are greater than 0.5 (**p<0.05**) between color and **K** (see table). Nephelometric turbidity values correlate poorly (**<0.45**) with **K** values. Suspended solids values should probably not be used as an indication of water clarity in Biscayne Bay based upon the poor correlations (0.08) given in the table.

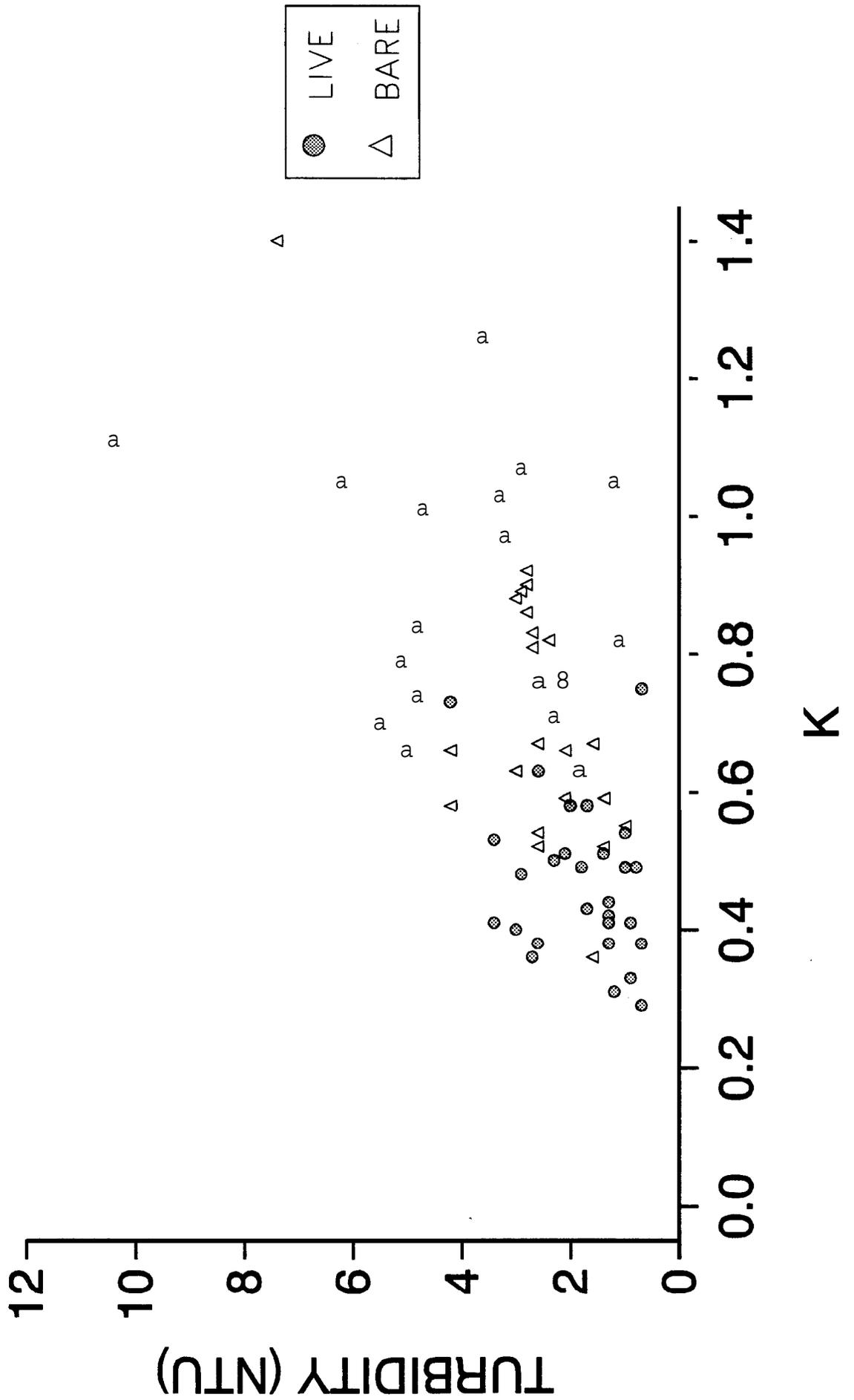
**Figure 1. Biscayne Bay percent live and barren bottom.  
Live bottom is either a seagrass or hardground  
community.**



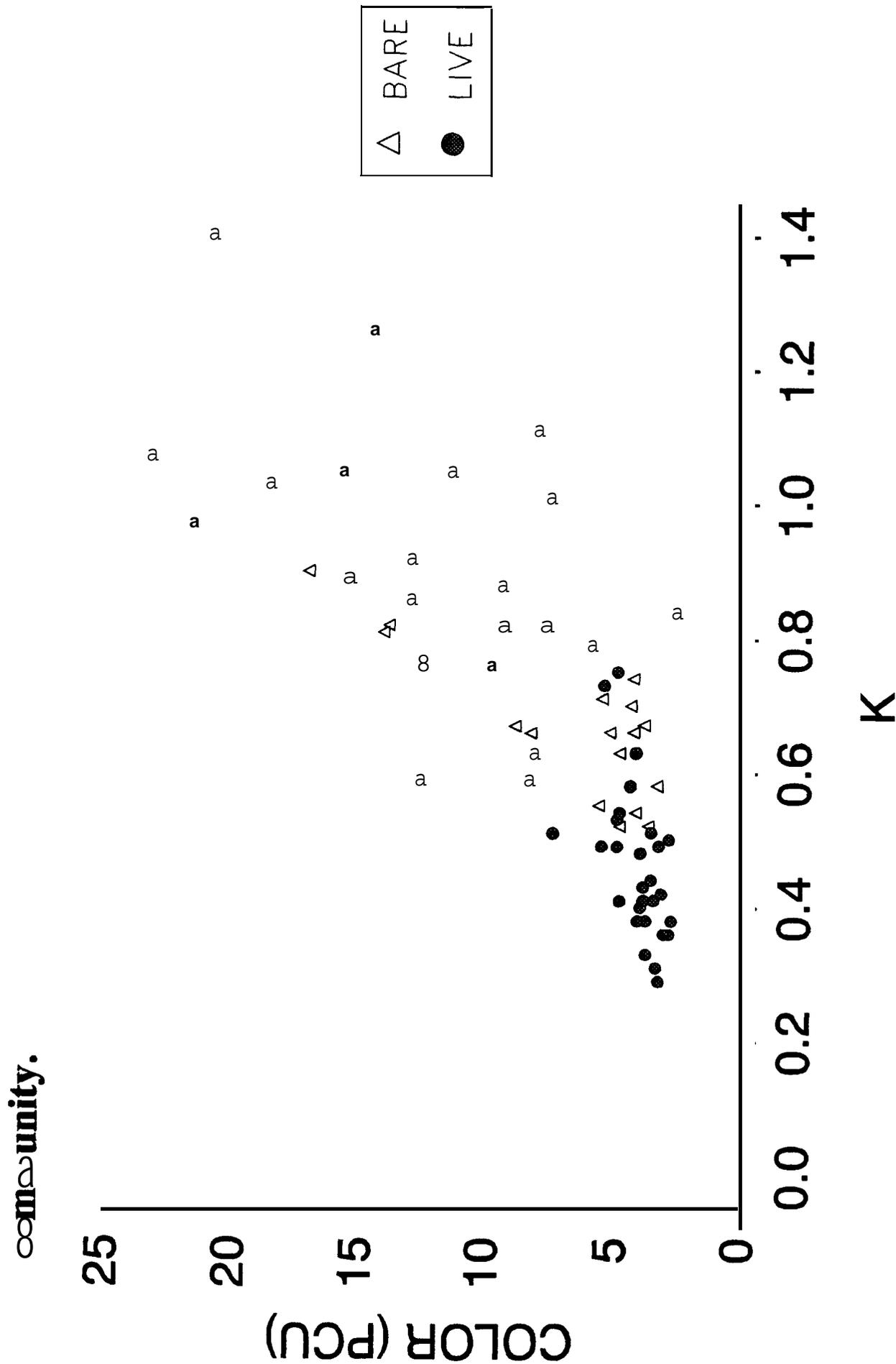
83.6 sq. km. or 20,600 acres are barren.



**Figure 3. Biscayne Bay PAR attenuation (K) vs. turbidity.**  
**Live bottom is either a seagrass or hard-ground community.**



**Figure 4. Biscayne Bay PAR attenuation (K) vs. color. Live bottom is either a seagrass or hardground community.**



# BISCAYNE BAY WATER QUALITY MONITORING PROGRAM

## P-4 RS<sup>2</sup> CORRELATION COEFFICIENTS:

	TURBIDITY	SUSP. SOLIDS	K
COLOR (PCU)	0.14	-0.15	0.54
TURBIDITY (NTU)	-	0.16	0.40
SUSP. SOLIDS (mg/L)	-	-	-0.08

## SPEARMAN CORRELATION COEFFICIENTS:

	TURBIDITY	SUSP. SOLIDS	K
COLOR (PCU)	0.23	-0.05	0.58
TURBIDITY (NTU)	-	0.08	0.44
SUSP. SOLIDS (mg/L)	-	-	-0.08

Higman, John., St. Johns River Water Management District,  
Palatka, **Fl.**

#### ESTUARINE WATER QUALITY MONITORING IN THE INDIAN RIVER LAGOON

The Indian River Lagoon (IRL) water quality monitoring network (WQMN) is a cooperative multi-agency program designed to give an accurate physical and chemical description of the water quality in the IRL. Sampling is done at 135 sites in the IRL at least four times each year. Information collected by this **long-term** water quality monitoring program, since 1989, is used to measure trends in lagoon water quality, evaluate responses to changing management practices, compare water quality among subbasins in the lagoon, and relate trends or changes to biological or ecosystem responses.

Water clarity is a indicator of water quality, and knowledge of water clarity is needed to appropriately manage the submerged aquatic vegetation (SAV) resource. Some of the physical and chemical parameters measured by the WQMN that indicate water clarity include total suspended solids, water color, turbidity, chlorophyll and secchi disk depth measurements.

Recently, participants in the WQMN have used irradiance meters to measure photosynthetically active radiation (PAR) and supplement other water clarity measurements. A vertical profile of PAR measurements is used to calculate light extinction coefficients. These extinction coefficients provide an estimate of the light available to SAV.

PAR values can tell us:

- a) if enough light reaches the bottom at a site to support SAV

growth (calculated compensation point is below the bottom), and  
b) the increased amount of lagoon bottom that could be colonized  
by SAV if light attenuation was decreased.

The use of extinction coefficients calculated from  
measurements made by irradiance meters can result in better  
management of the SAV resource because they are a direct  
indicator of the amount of light available to SAV. In addition,  
these coefficients can be compared within a waterbody and among  
waterbodies.

Kirkpatrick, Jeff., Texas Water Commission, Austin, TX

#### ESTUARINE WATER QUALITY MONITORING IN TEXAS.

Routine water quality monitoring of estuaries in Texas began in 1968 with quarterly monitoring of all distinct coastal segments. Parameters included field data and routine water chemistry at all sites. During the ensuing twenty-two years there have been refinements in delineating segments boundaries, new segments have been created, the scope of parameters monitored has expanded, and sampling frequencies have changed.

Eighty-two of 359 total classified segments in Texas (23%) are located in estuarine waters. These segments include tidal portions of major rivers **(10)**, tidal streams **(19)**, dredged canals and ship channels **(13)**, primary bays (18) and secondary bays **(21)**.

Presently, almost 40% of the total routine water quality monitoring sampling events in Texas are devoted to estuarine sites. Field data, consisting of temperature, dissolved oxygen, **pH**, conductivity, salinity, and secchi depth, are measured at all sites. Vertical profiles are made where possible. Routine water chemistry and fecal coliform analyses are made at almost all sites. Parameter groups monitored occasionally include sediment metals, sediment pesticides, metals in water, pesticides in water, metals and pesticides in tissue, benthic macroinvertebrates, nekton and plankton. About 36% of the total estuarine monitoring sampling events are from primary bays, followed by dredged canals **(26%)**, secondary bays **(14%)**, tidal

streams (12%), tidal rivers (9%), Gulf passes and boat basins comprise less than 3% of the estuarine sampling events.

Data from the monitoring program are used by many environmental groups, private companies and governmental agencies for various water quality assessment purposes. The Texas Water Commission uses the data to determine segment classification, whether established numerical criteria and uses are being met, to determine the status and trends in water quality, and to identify problem areas. The estuarine segments are periodically evaluated for the occurrence, extent, and severity of hypoxia, anoxia, and excessive algal production. Parameters utilized for these evaluations are dissolved oxygen, chlorophyll a, inorganic nitrogen, total phosphorus and orthophosphorus.

Marine macrophytes or parameters associated with their productivity, except for secchi depth, are not routinely monitored by the Texas Water Commission. Some protection of these ecologically valuable areas is afforded, however, by the antidegradation policy of the Texas Water Code. Figure 1 is a summary of the Statewide Monitoring System conducted by the Texas Water Commission for FY 91.

ESTUARINE MONITORING SAMPLING EVENTS PER YEAR

Parameter Group	TR	TS	DC	BB	PB	SB	GP	Total Estuarine	Total Inland	Total State	Estuarine Percent
Field Data	50	74	160	4	210	83	12	593	908	1501	39.5
Water Chemistry	50	74	136	4	210	83	12	569	908	1477	38.5
Fecal Coliform	50	74	136	4	210	83	12	569	904	1473	38.6
Sediment Metals	3	0	4	1	4	5	0	17	19	36	47.2
Sediment Pesticides	3	0	4	0	3	0	0	10	18	28	35.7
Water Metals	3	0	16	1	2	12	0	34	18	52	65.4
Water Pesticides	3	0	2	0	2	0	0	7	18	25	28.0
Tissue Metals and Pesticides	3	0	4	0	2	1	0	10	24	34	29.4
Benthic Microinvert.	0	4	0	0	29	0	0	33	87	120	27.5
Nekton	3	0	24	0	1	1	0	29	46	75	38.7
Plankton	0	4	8	0	2	0	0	14	5	19	73.7
<b>Total</b>	<b>168</b>	<b>230</b>	<b>494</b>	<b>14</b>	<b>675</b>	<b>268</b>	<b>36</b>	<b>1865</b>	<b>2955</b>	<b>4840</b>	<b>38.9</b>
<b>Estuarine Percent</b>	<b>8.9</b>	<b>12.2</b>	<b>26.2</b>	<b>0.7</b>	<b>35.8</b>	<b>14.2</b>	<b>1.9</b>	<b>100</b>			

TR - Tidal River  
 TS - Tidal Stream  
 DC - Dredged Canal  
 BB - Boat Basin  
 PB - Primary Bay  
 SB - Secondary Bay  
 GP - Gulf Pass

**Haire**, Michael S., Robert E. Magnien and Steven E. Bieber, Maryland Dept. of the Environment, Chesapeake Bay and Special Projects Program, Baltimore, MD.

MARYLAND'S CHESAPEAKE BAY WATER QUALITY MONITORING PROGRAM AND IT'S RELEVANCE TO SAV COMMUNITIES

The Chesapeake Bay suffers from a variety of water quality problems, the most pervasive of which is eutrophication. Having come to this conclusion in 1983 as a result of a lengthy EPA study, the bordering states and federal agencies agreed that strong action was necessary to restore the **Bay's** water quality and living resources which depend upon adequate water quality for their survival. In order to move forward with an aggressive restoration campaign it was agreed that a comprehensive, **long-term** water quality monitoring program was needed to guide and provide accountability for management actions. Since long-term Bay-wide water quality information was not available at the time of the EPA study, scientists trying to evaluate the Bay's condition were severely hampered in quantifying the extent and character of its problems. This piecemeal information was also of little value in defining the changes in water quality between the 1950's and **1970's** as pollution entering the Bay during that period increased. Given the large uncertainty, both then and now, in our understanding of the Bay's ecological relationships, a comprehensive long-term water quality monitoring program was seen as a way of providing a "bottom-line" answer to the questions about man's impact on the Bay and its response to pollution control actions.

Uncertainties surrounding the quantification of pollution

sources, their impact on water quality and, in turn, the impact of water quality on living resources limit effective remedial action. Fortunately, Maryland and other jurisdictions have not let this uncertainty paralyze their efforts to reduce pollution entering the Bay. Given some of the obvious pollution problems of the last several decades such as raw sewage, odors, fish kills, bacterial contamination, and massive algal blooms, it was clear that certain measures such as secondary treatment and disinfection at sewage treatment plants were necessary. In recent years, more sophisticated analyses using mathematical models have provided justification for more concerted action in heavily polluted tributaries. As many of these remedies have been implemented, some dramatic improvements have been documented.

Today, however, the effects of eutrophication are more subtle and widespread. Oxygen concentrations often drop to levels that will not support fish and shellfish. Reduced water clarity and nutrient enrichment have made it difficult for aquatic macrophyte communities, which provide important habitat, to flourish as they had in the past. The **Bay's** bottom sediments have been altered so that in many areas they no longer serve as suitable habitat for the once abundant oyster. The sediments can also magnify external nutrient inputs by recycling back into the water column a large part of the nutrients that are deposited to the bottom.

These continuing problems will clearly be more difficult and expensive to fix, especially in the face of increased population

pressures. Costly methods of advanced treatment at municipal and industrial plants are generally required now to make additional reductions in nutrient inputs from point sources. At the same time that nutrient removal is becoming more costly, the federal government is reducing support for these pollution controls.

**Nonpoint** sources of pollution are starting to be addressed by more aggressive agricultural and urban programs but their effectiveness in stemming nutrient inputs to the estuary is poorly understood.

The Chesapeake Bay Water Quality Monitoring Program that Maryland implemented in 1984 is specifically aimed at reducing current levels of uncertainty in pollution abatement strategies. Along with other evaluations, such as forecasting the outcome of alternative strategies using mathematical models, the monitoring program provides information sufficient to justify needed management actions. Concepts that went into the pollution design and the ways in which the information is being used as discussed below.

#### PROGRAM DESIGN

##### WATER QUALITY ISSUES AND MANAGEMENT INFORMATION NEEDS:

To ensure that the monitoring program would ultimately be useful for management, its design was founded upon an assessment

of information needs. This assessment commenced by defining the major water quality issues of concern. These were nutrient enrichment and its consequences: algal blooms, low dissolved oxygen and high turbidity. With observed declines in fish, shellfish, and SAV abundance, these water quality concerns were strongly related to impacts on these resources, either directly or through habitat deterioration. Toxicants were also a major concern, but there was much more uncertainty surrounding the effects of toxicants in the Bay and the information that would be useful for management purposes.

Starting with these broad management issues, more defined information needs were developed. This definition was facilitated by the listing of "management questions". These questions represented specific, practical pieces of information that managers would need from a monitoring program to effectively pursue the restoration of the Chesapeake Bay. These questions requested information such as:

- o What are the loading of phosphorus and nitrogen from point and **nonpoint** sources in a particular basin?
- o Where are **hypoxic** areas located and how severe are they?
- o Has water quality improved where major management actions have been implemented?
- o Is algal production limited by nitrogen, phosphorus or both?

From a long list of specific questions, or management information needs, three categories clearly emerged:

1. Questions about the present characteristics of water quality and pollution loadings.

2. Questions about trends or changes in water quality due to increasing or decreasing pollution.
3. Questions about basic processes affecting water quality and the impact on living resources.

These information categories become the three guiding objectives for the collection of monitoring information:

1. CHARACTERIZATION: Quantify the extent and nature of water quality problems.
2. DETECTION OF CHANGES OR TRENDS: Determine the response of key water quality variables to management action or inaction.
3. UNDERSTANDING OF PROCESSES: Develop and test hypotheses on how the Bay ecosystem functions, especially as it relates to anthropogenic stresses and management solutions.

Some additional themes also emerged from the long list of monitoring information needs. First, several types of water quality information would be required. Water quality is a general term and from a eutrophication standpoint it was clear that key physical, chemical and biological indicators were all needed to make informed management decisions. Second, the information needed to be collected Bay-wide in the **mainstem** and in each of the major Bay tributaries. This would provide data sufficient for decisions that reflect the Bay-wide nature of many problems as well as tributary-specific characteristics and solutions. Finally, the information needed to be collected consistently over a long period of time. Only with this long-term record could the monitoring program be effective in judging the success of management actions.

## PROGRAM COMPONENTS:

With specific information needs and objectives established, the design of the program could be formulated. From an assessment of the types of information that would be critical to management, six program components were formed. These were:

1. CHEMICAL/PHYSICAL PROPERTIES  
Variables: salinity, temperature, Secchi depth, dissolved **oxygen**, suspended solids, nutrient species (N,P,C and Si), phytoplankton pigments, heavy metal and organic compounds in surficial sediments. Stations/Yearly Sampling Frequency: **77/12-20** (sediments, 1)
2. PHYTOPLANKTON  
Variables: species counts, phytoplankton pigments by horizontal and vertical in vivo fluorescence, primary productivity, light penetration. Stations/Yearly Sampling Frequency: **14/18**
3. ZOOPLANKTON  
Variables: micro (**44 $\mu$ m-202 $\mu$ m**) and **meso (>202 $\mu$ m)** species counts, biomass. Stations/Yearly Sampling Frequency: **14/12**
4. BENTHIC ORGANISMS  
Variables: species counts, production, sediment characteristics, salinity, dissolved oxygen. Stations/yearly Sampling Frequency: **31/10**
5. ECOSYSTEM PROCESSES  
Variables: sediment-water column exchange rates of dissolved inorganic nutrients (N,P,Si) and oxygen; surficial sediment characteristics; deposition rates of particulate matter (total seston, N,P,C, phytoplankton pigments). Stations/Yearly Sampling Frequency: **10/4 (exchange); 1/20** (deposition).
6. RIVER INPUTS  
Variables: flow, suspended solids, nutrient species (N,P,C and Si), phytoplankton pigments. Stations/Yearly Sampling Frequency: **4/20-30** (flow-dependant).

Each program component was structured such that it would provide both sufficient information for that discipline and complement others. It was recognized that these program divisions

were somewhat artificial in the context of a complex, interacting ecosystem but that they were necessary to effectively design and manage such a large project. Knowing at the outset that the information would need to be interpreted across components, close linkages were created between most of the components by overlapping station locations and simultaneous sampling. This is particularly evident between the chemical/physical, phytoplankton and zooplankton components which are sampled concurrently from the same vessels. In this case, there are compelling scientific reasons for simultaneous sampling as well as the obvious resource efficiencies.

It was not feasible to implement all of the monitoring components in all of the Bay's tributaries. The chemical/physical component was conducted at the full complement of stations in the **mainstem** and tributaries. This component includes the most fundamental and interpretable water quality variables for management information needs. To provide a level of effort capable of yielding technically rigorous information, the other components were concentrated in the Bay's **mainstem** and Maryland's three largest tributaries. Baltimore Harbor was also included in most of the components due to its high pollutant impacts. By concentrating efforts in the largest systems of the Bay, the broader water quality responses to management actions and an improved understanding of processes could be firmly established for these systems. This fundamental understanding, along with direct measures of chemical/physical properties, could then be

utilized in the management of smaller systems. Variable selection methodology and spatial/temporal intensity were carefully evaluated for all components to provide a level of information that would be the minimum necessary to support confident management decisions making. Techniques such as statistical power analysis and pilot studies were used to evaluate study design where the literature did not provide sufficient guidance.

#### DATA ANALYSIS AND REPORTING:

The element of the monitoring program that finally brings the information into the management arena is data analysis and interpretation. Maryland's strategy includes both technical and non-technical summaries of the monitoring information and reflects the input of other sources of information into the decision making process. While competent technical analysis is the foundation for utilizing the monitoring information in management decisions, a parallel effort is being made to inform politicians and citizens. The politicians and citizens are viewed as being an important part of the water quality management decisions process. They supported the creation of the monitoring program and expect it to provide periodic reports on the "State of the **Bay**" as well as serving water quality managers in the regulatory agencies. Ultimately, the citizens of Maryland and their elected officials will decide the allocation of resources to restore the Bay.

## PROGRESS TO DATE

In the summer of 1984, following a year of planning, the monitoring program described above was implemented. With over 5 years of monitoring information to date, the program has largely achieved its first objective of establishing an initial Bay-wide characterization of water quality. In some tributary estuaries, such as the Potomac and Patuxent Rivers, clear changes in water quality have been established in response to nutrient loading reductions. In the case of the Potomac River, various monitoring programs had been in operation since the **1960's**. These data were combined with the more recent monitoring data to establish trends over an extended period of time. The Potomac River monitoring information clearly demonstrates that a monitoring program of the type now in place can provide definitive, quantitative answers on the response of an estuarine system to concerted management efforts. This capability fulfills the second major objective of the Maryland program.

Considerable progress has also been achieved on the third objective of the monitoring program, an improved understanding of processes related to water quality in Chesapeake Bay. A much clearer picture is emerging from analysis of the monitoring data on such key eutrophication issues as the sources and fates of nutrients, the spatial and temporal patterns of limiting nutrients and the utilization pathways for primary production.

This information is now being assembled into a "Technical Synthesis" that will include a rigorous analysis of the first 5 years of monitoring information. These results will be used to refine both mathematical and conceptual models which are being used to predict the response of the Bay to various management strategies.

Although the monitoring program has been quite successful in meeting its design objectives, the most important measure of its success is whether or not the monitoring information has been used in the management of Chesapeake Bay. Already, there are a number of examples where it has been used. The first two years of monitoring data were used in the development of a 2-dimensional mathematical model of the Bay. The nutrient reduction scenarios of this model led to the most dramatic commitment of the 1987 Bay Agreement - a 40% reduction of nitrogen and phosphorus by the year 2000. This agreement, signed by the Governors of Maryland, Virginia, and Pennsylvania, the mayor of Washington D.C., and top federal agency officials has resulted in specific nutrient control actions. Similarly, a thorough analysis of monitoring data and the use of monitoring data for the development of a mathematical model was the foundation of a decision by EPA to grant Maryland \$10 million to remove nitrogen from a major sewage treatment plant on the Patuxent River. The monitoring information has supported management in more subtle ways. By demonstrating positive responses to management actions in the Potomac and Patuxent Rivers, the monitoring information has greatly enhanced

the confidence among managers that nutrient loading reductions will produce desired results in the estuary.

#### FUTURE CHALLENGES AND DIRECTIONS

Many additional challenges confront the monitoring program as the Bay's future restoration plans are periodically reformulated and assessments are conducted on the results of previously implemented plans. The 1987 Bay Agreement commits to a **"re-evaluation"** of the goal to reduce nutrients by 40%. Interpretations of the monitoring information, both directly and through the use of improved mathematical models, will be the technical cornerstone for this re-evaluation. Because monitoring is now being conducted in most of the Bay's tributaries, the re-evaluation will provide an opportunity to improve upon strategies in many areas by resolving future plans on a basin-specific level. Analysis of the monitoring information to date has provided strong justification for the basin-specific approach for nutrient reduction since large differences exist between many of the tributary and **mainstem** regions in terms of the concentrations of nutrients in their waters and source and amounts of nutrient loading.

Another question that relates strongly to the monitoring programs is the concept of goal setting for water quality. In grossly polluted situations it is clear that desirable water quality conditions are not being met. As pollution reduction

programs eliminate many of these obvious problems, the question turns to how **"good"** does the water quality need to be. As logical as it seems to have a goal when considerable resources are allocated to mitigate a problem, the desired water quality, or **goal**, remains poorly defined. One definition of a water quality goal, that which protects human health, has received the most attention and at least some **guidelines** exist regarding bacterial contamination. Goals related to the eutrophication problem in estuaries have not received much attention except for dissolved oxygen. Levels below 4 or 5 **mg/l** are typically considered a stressful condition for fish and shellfish. In stratified estuaries such as the Chesapeake Bay, however, application of these guidelines is difficult since naturally occurring dissolved oxygen levels are often below these thresholds.

The current focus for setting goals is the relationship between water quality conditions and the Bay's living resources. There is a widely accepted perception that deteriorating water quality has contributed to many of the declines in Chesapeake **Bay's** living resources. Efforts are underway to identify the minimum water quality conditions that are necessary to support the successful reproduction and survival of key living resources including submerged aquatic vegetation (SAV).

The development of these water quality based habitat goals is most advanced for the Bay's submerged aquatic vegetation. Water transparency is viewed as the key water quality variable controlling SAV growth and survival, with total suspended solids,

chlorophyll and nutrients playing a more indirect role.

#### RELEVANCE OF MONITORING PROGRAM TO SAV COMMUNITIES

With the development of habitat goals for resources such as SAV, the water quality monitoring program has the potential to add a new dimension of information in support of Bay management actions. However, one of the possible limitations in using data from the monitoring program to assess SAV habitat is that stations are typically located in mid-channel areas while SAV is typically **located in** near-shore areas. While locating stations in mid-channel provides representative data for large regions, including deepwater hypoxia problems, there is the question of how representative these data may be in characterizing near-shore environments. Establishing the relationship between mid-channel and near-shore water quality for key variables thus becomes critical to determining whether this large body of information can be tapped to assess habitat conditions.

We are currently in the process of analyzing matched **near-**shore and mid-channel data for a diverse number of sites in the Chesapeake Bay. Variables for comparisons include light attenuation (direct or indirect from secchi depth) total suspended solids, chlorophyll, dissolved inorganic phosphorus and dissolved inorganic nitrogen. These variables were compared over the SAV growing season of April through October. Preliminary results indicate that for all of the investigated parameters

there is some degree of variation between mid-channel and **near-** shore locations, but few consistent biases. Among all of the variables, light attenuation was the most comparable.

Burney, Lou, Florida Dept. of Environmental Regulation,  
Tallahassee, **Fl.**

#### OVERVIEW FLORIDA 305(b) PROGRAM

\* The program was begun in 1978 in response to Sec. 305(b) of the National Clean Water Act, which requires each state to submit a report to EPA every two years, describing the quality of its surface and ground water (Illustrate 1990 report).

\* The 305(b) reports attempt to provide a general statewide assessment of water quality by water body type: stream, lake or estuary.

\* A **major** problem with estuaries has been the general lack of practical interpretive frameworks that consider the complex dynamics of estuaries (Historically, EPA has focused on lakes and streams).

\* In the absence of such frameworks, several approaches have been tried, with mixed results. Two basic methods are now utilized in Florida:

(1) Development of "typical values" observed in Florida estuaries (median values for 1,700 estuarine stations and percentile distribution of these values).

(2) Development of a **trophic** state index (**T.S.I**) Florida appears to be the only state that has employed this approach.

\* The TSI approach is based in part upon calculation of average TSI values for chlorophyll-a, Secchi depth, total nitrogen and total phosphorus, from which an overall TSI rating is derived for each estuarine segment. The parameters assessed do not have

numeric standards, but are considered measures of the overall health of the water body.

\* Without going into detail on the equations used to generate TSI values for individual parameters, the overall TSI rating for each estuary reach has a total possible score of 100 points. The score determines the overall water quality rating as follows:  
0 - 49 = good, 50 - 59 = fair, 60 - 100 = poor.

\* This part of the water quality assessment is computer generated, based upon actual data from STORET.

\* Where possible, the computer generated TSI is subjected to review by agency staff (DER, **WMD's**, local government, and the research community) having site-specific knowledge of biological conditions in the estuary, and through interviews, is then adjusted to take into account these personal understandings.

\* The 305(b) Technical Appendix provides: tables of data on mean values for each estuary reach; maps showing average water quality in terms of **"good"**, **"fair"**, or **"poor"**; as well as descriptive text on pertinent considerations such as anthropogenic sources, apparent trends, etc.

\* In short, the 305(b) Report is an admittedly simplistic attempt to reduce available water column data into a perspective of approximate water quality conditions around the state. It appears to hold up for this general purpose and is the only statewide, regularly published document to attempt this. It has been very useful in identifying problem areas needing attention under the SWIM program. Among the recognized shortcomings of the

program are:

1. Lack of consistent protocols for data collection (hence, the reliability of the data cannot be determined).
2. Inadequate capability for statewide data management (whereby data from state, regional, and local agencies can be integrated). The system is particularly weak in terms of trend assessment for estuaries.
3. Since the program focuses on the water column, the assessments do not reflect potential chemical problems manifested in the sediments.
4. In the absence of estuarine-specific interpretive frameworks, the program essentially employs the same methods used to assess eutrophication in lakes. Given the more complicated dynamics of estuaries, this may be a bit too simplistic to correctly assess the true health of estuaries.
5. There has been a ten year trend of declining monitoring of Florida's waters.

\* These problems apparently are not restricted to Florida, and are general difficulties experienced nationally.

\* The DER staff responsible for producing the 305(b) Reports are cognizant of the need for better interpretive frameworks for estuaries and welcome suggestions for practical improvements. The point of contact for this would be Mr. Joe Hand, in our Bureau of Surface Water Management (Ph. **904/488-6221**).

\* Florida is presently reevaluating its overall water quality monitoring needs. The present emphasis is on compliance monitoring until ambient monitoring needs become better defined.