Delta Outflows and Related Stressors Workshop

February 10 and 11, 2014

PWA Supporting Documentation

ECOSYSTEM CHANGES

NMFS & FWS Biological Opinions

Reasonable and Prudent Alternatives

DRAFT

CRITERIA	OCT	NOV	DEC	JAN	FEB	MAR	APR	MAY	JUN	JUL	AUG	SEP
National Marine Fisheries Service												
Action IV. 1.2 DCC Gate Operation	Oct 1 - Nov 30 Gates Are Closed if Fish Are Present		Dec 1 - 14 Dec 15 - Jan 31 Gates Are Closed		Feb 1 - May 15 Gates Are Closed per D1641			May 16 - Jun 15 up to 14 Days Closed per D1641				
Action IV. 2.1 San Joaquin River Inflow/Export Ratio			Gates Are Closed except for Experiments/WQ				Apr 1 - May 31 Maintain Vernalis I/E Ratio		Interim - Based on IOP Water Supply Long Term - Based on WY Type			
Action IV. 2.2 Six-Year Acoustic Tag Experiment						Mar 1 - Jun 15						
Action IV. 2.3 Old & Middle River Flow Management				Jan 1 - Jun 15 OMR (-5000 to -2500 cfs) until after Jun 1 Water Temperature @ Mossdale ≥ 72˚F for 7 Days								
Action IV. 3 Reduce Likelihood of Entrainment or Salvage		Nov 1 - Dec 31										
Fish & Wildlife Service												
Action 1 Adult Migration & Entrainment (1st Flush)			Dec 1 - 20 After Dec 20 Triggers: Turbidity or Salvage Off Ramps: Water Temperature or Biological									
Action 2 Adult Migration & Entrainment			Begins Imediately After Action 1. Suspension of Action: Flow Off Ramps: Water Temperature or Biological									
Action 3 Entrainment Protection of Larval Smelt					Triggers: Water Temperature or Biological Off Ramps: Water Temperature or June 30							
Action 4 Estuarine Habitat During Fall (X2)	Oct 1 - Nov 30											Sep 1 - 30
Action 5 Temporary Spring HORB & the TBP							Apr 1 - May 15					

Delta Compliance and Modeling Section

Revised 11/8/2013

Preliminary: Subject to Revision

NMFS BO http://swr.nmfs.noaa.gov/ocap/NMFS_Biological_and_Conference_Opinion_on_the_Long-Term_Operations_of_the_CVP_and_SWP.pdf

ACTION IV. 1.2 - OPERATION OF DCC TO ENHANCE PROTECTION OF EMIGRATING SALMONIDS/GREEN STURGEON

Page 635

Timing: October 1 – November 30
Triggers:

- Water quality criteria per D-1641 are met and either Knights Landing Catch Index (KLCI) or the Sacramento Catch Index (SCI) are greater than 3 fish per day but less than or equal to 5 fish per day. Within 24 hours of trigger, DCC gates are close. Gates will remain closed for 3 days
- Water quality criteria per D-1641 are met and either KLCI or SCI is greater than 5 fish per day. Within 24 hours, close the DCC gates and keep closed until the catch index is less than 3 fish per day at both the Knights Landing and Sacramento monitoring sites.
- KLCI or SCI triggers are met, but water quality criteria are not met per D-1641 criteria. DOSS reviews monitoring data and makes recommendation to NMFS and WOMT per procedures in Action IV. 5 (page 658).

Timing: December 1 – December 14
Triggers:

- Water quality criteria per D-1641 are met. DCC gates are closed. If Chinook salmon migration experiments are conducted during this time period, the DCC gates may be opened according to the experimental design, with NMFS' prior approval of the study.
- Water quality criteria are not met but both KLCI or SCI are less than 3 fish per day. DCC gates may be opened until water quality criteria are met. Once water quality criteria are met, the DCC gates will be closed within 24 hours of compliance.
- Water quality criteria are not met, but either KLCI or SCI is greater than 3 fish per day. DOSS reviews monitoring data and makes recommendation to NMFS and WOMT per procedures in Action IV. 5 (page 658).

Timing: December 15 – January 31
Triggers:

- DCC gates are closed December 15 – January 31.
- NMFS-approved experiments may be conducted. Agency sponsoring the experiment may request gate opening for up to 5 days; NMFS will determine whether opening is consistent with ESA obligations.
- One-time event between December 15 to January 5, when necessary to maintain Delta water quality in response to the astronomical high tide, coupled with low inflow conditions. Upon concurrence of NMFS, DCC gates may be opened one hour after sunrise to one hour before sunset, for up to 3 days, then return to full closure. USBR and DWR will also reduce Delta exports down to a health and safety level during the period of this action.

Timing: February 1 – June 15
Triggers:

- DCC gates are close February 1 – May 15 as per D1641 and WQCP.
- May 16 – June 15, DCC gates may be closed for up to 14 days as per D-1641 and 2006 WQCP, if NMFS determines it is necessary.

ACTION IV. 2.1 - MAINTAIN SAN JOAQUIN RIVER INFLOW/EXPORT RATIO

Page 641

Timing: April 1 – May 31 (*Phase II : Interim Operations* Beginning 2012)

1. Reclamation shall continue to implement the Goodwin flow schedule for the Stanislaus River prescribed in Action III.1.3 and Appendix 2-E.

2. Reclamation and DWR shall implement the Vernalis flow-to-combined export ratios in the following table, based on a 14-day running average

San Joaquin Valley Classification	Vernalis flow (cfs) CVP/SWP combined export ratio[11]
Critically Dry	1:1[12]
Dry	2:1
Below Normal	3:1
Above Normal	4:1
Wet	4:1
Vernalis flow equal to or greater than 21,750 cfs.	Unrestricted exports until flood receeds below 21,750 cfs.

Exception procedure for multiple dry years: If the previous 2 years plus current year of the San Joaquin Valley "60-20-20" Water year Hydrologic Classification and Indicators as defined in D-1641 and provided in the following table, is 6 or less, AND the New Melones Index is less than 1 MAF, exports shall be limited to a 1:1 ratio with San Joaquin River inflow, as measured at Vernalis.

San Joaquin Valley Classification	Indicator
Critically Dry	1
Dry	2
Below Normal	3
Above Normal	4
Wet	5

Exception procedure for Health and Saftey: If, by February 28 of a given year, Reclamation and DWR predict that they will not be able to achieve these ratios and make deliveries required for human health and safety, even after pursuing all options to augment inflow while preserving the ability to meet fish flow needs in all seasons, agencies may submit a plan to NMFS to maximize anadromous fish benefits while meeting health and safety needs.

[11] Exception to the ratio is provided for floods, where exports are not restricted until the flood recedes. See footnote 2 above.
[12] Minimum combined CVP and SWP exports is for health and safety. 2009 RPA with 2011 amendments

ACTION IV. 2.2 - ACOUSTIC TAG EXPERIMENT

Page 645

Timing: March 1 – March 31

- Exports will be operated in accordance with the requirements dictated by Action IV.2.3.

Timing: April 1 – May 30

- Exports will be dictated by the requirements of Action IV.2.1.

Timing: June 1 – June 15

- Reclamation shall operate to a minimum 1:1 inflow to export ratio, allowing exports to vary in relation to inflows from the San Joaquin to test varying flow to export ratios during this period. If daily water temperatures at Mossdale exceed 72°F for seven consecutive days during this period, then the inflow to export ratio may be relaxed.

ACTION IV. 2.3 - REDUCED EXPORTS TO LIMIT NEGATIVE FLOWS IN OMR DEPENDING ON PRESENCE OF SALMNOIDS

Page 648

Timing: January 1 – June 15

Action:

- Exports are managed to a level that produces a 14-day running average of the tidally filtered flow of -5,000 cfs in Old and Middle River (OMR). A five-day running average flow shall be calculated from the daily tidally filtered values and be no more than 25 percent more negative than the targeted requirement flow for the 14-day average flow.

Timing: January 1 – June 15
First Stage Trigger (increasing level of concern):

- Daily SWP/CVP older juvenile loss density (fish per taf) 1) is greater than incidental take limit divided by 2000 (2 percent WR JPE ÷ 2000), with a minimum value of 2.5 fish per taf, or 2) daily loss is greater than daily measured fish density divided by 12 taf (daily measured fish density ÷ 12 taf) or 3) CNFH CWT LFR or LSNFH CWT WR cumulative loss greater than 0.5%, or 4) daily loss of wild steelhead (intact adipose fin) is greater than the daily measured fish density divided by 12 taf (daily measured fish density ÷ 12 taf).

Action:

- Reduce exports to achieve an average net OMR flow of (minus) -3,500 cfs for a minimum of 5 consecutive days. The five day running average OMR flows shall be no more than 25 percent more negative than the targeted flow level at any time during the 5-day running average period (*e.g.*, -4,375 cfs average over five days). Resumption of (minus) -5,000 cfs flows is allowed when average daily fish density is less than trigger density for 3 consecutive days following the 5 consecutive days of export reduction. Reductions are required when any one criterion is met.

Timing: January 1 – June 15
Second Stage Trigger (analogous to high concern level):

- Daily SWP/CVP older juvenile loss density (fish per taf) is 1) greater than incidental take limit (2 percent of WR JPE) divided by 1000 (2 percent of WR JPE ÷ 1000), with a minimum value of 2.5 fish per taf, or 2) daily loss is greater than daily fish density divided by 8 taf (daily fish density ÷8 taf), or 3) CNFH CWT LFR or LSNFH CWT WR cumulative loss greater than 0.5%, or 4) daily loss of wild steelhead (intact adipose fin) is greater than the daily measured fish density divided by 8 taf (daily measured fish density ÷ 8 taf).

Action:

- Reduce exports to achieve an average net OMR flow of (minus) -2,500 cfs for a minimum 5 consecutive days. Resumption of (minus) -5,000 cfs flows is allowed when average daily fish density is less than trigger density for 3 consecutive days following the 5 consecutive days of export reduction. Reductions are required when any one criterion is met.

Timing: January 1 – June 15
End of Triggers:

- Continue action until June 15 or until average daily water temperature at Mossdale is greater than 72°F (22°C) for 7 consecutive days, whichever is earlier.

ACTION IV. 3 - REDUCE LIKELIHOOD OF ENTRAINMENT OR SALVAGE AT EXPORT FACILITIES

Page 652

Timing: November 1 – December 31
Triggers:

- Daily SWP/CVP older juvenile loss density greater than 8 fish/taf, or daily loss is greater than 95 fish/day, or Coleman National Fish Hatchery coded wire tagged late fall-run Chinook salmon (CNFH CWT LFR) or Livingston Stone National Fish Hatchery coded wire tagged winter-run (LSNFH CWT WR) cumulative loss greater than 0.5%. Reduce exports to a combined 6,000 cfs for 3 days or until CVP/SWP daily density is less than 8 fish/taf. Export reductions are required when any one of the four criteria is met.

- Daily SWP/CVP older juvenile loss density greater than 15 fish/taf, or daily loss is greater than 120 fish/day, or CNFH CWT LFR or LSNFH CWT WR cumulative loss greater than 0.5%. Reduce exports to a combined 4,000 cfs for 3 days or until CVP/SWP daily density is less than 8 fish/taf. Export reductions are required when any one of the four criteria is met.

Text for the Biological Opinion Actions have been condensed to fit within this limited space. Page numbers in the upper right hand corner of each Action block have been provided for referencing the actual document.

FWS BO http://www.fws.gov/sacramento/es/documents/SWP-CVP_OPs_BO_12-15_final_OCR.pdf

ACTION 1: ADULT MIGRATION AND ENTRAINMENT (FIRST FLUSH)

Page 329

Action: Limit exports so that the average daily OMR flow is no more negative than -2,000 cfs for a total duration of 14 days, with a 5-day running average no more negative than 2,500 cfs (within 25 percent).

Timing:
Part A: December 1 to December 20 – Based upon examination of turbidity data from Prisoner's Point, Holland Cut, and Victoria Canal and salvage data from CVP/SWP (see Triggers below), and other parameters important to the protection of delta smelt including, but not limited to, preceding conditions of X2, FMWT, and river flows; the SWG may recommend a start date to USFWS. USFWS will make the final determination.
Part B: After December 20 – The action will begin if the 3 day average turbidity at Prisoner's Point, Holland Court, and Victoria Canal exceeds 12 NTU. However, the SWG can recommend a delayed start or interruption based on other conditions such as Delta inflow that may affect vulnerability to entrainment

Triggers (Part B): Turbidity: 3-day average of 12 NTU or greater at all three stations (Prisoner's Point, Holland Cut, Victoria Canal) or Salvage: Three days of delta smelt salvage after December 20 at either facility or cumulative daily salvage count that is above a risk threshold based upon the "daily salvage index" approach reflected in a daily salvage index value >0.5 (daily delta smelt salvage >one half prior year FMWT index value). The window for triggering Action 1 concludes when either off-ramp condition described below is met. These off-ramp conditions may occur without Action 1 ever being triggered. If this occurs, then Action 3 is triggered, unless USFWS concludes on the basis of the totality of available information that Action 2 should be implemented instead.

Off-ramps:
Temperature: Water temperature reaches 12°C based on a three station daily mean at Mossdale, Antioch and Rio Vista. *or*
Biological: Onset of spawning (presence of spent females in SKT or at Banks or Jones).

ACTION 2: ADULT MIGRATION AND ENTRAINMENT

Page 352

Action: The range of net daily OMR flows will be no more negative than -1,250 to -5,000 cfs. Depending on extant conditions (and the general guidelines below), specific OMR flows within this range are recommended by the SWG from the onset of Action 2 through its termination. The SWG will provide weekly recommendations based upon review of the sampling data, from real-time salvage data at the CVP and SWP, and utilizing most up-to-date technological expertise and knowledge relating populations status and predicted distribution to monitored physical variable of flow and turbidity. USFWS will make the final determination.

Timing: Beginning immediately after Action 1. Before this date (in time for operators to I implement the flow requirement) the SWG will recommend specific required OMR flows based on salvage and on physical and biological data on an ongoing basis. If Action 1 is not implemented, the SWG may recommend a start date for the implementation of Action 2 to protect adult delta smelt.

Suspension of Action: Flow: OMR flow requirements do not apply whenever a three day flow average is greater than or equal to 90,000 cfs in Sacramento River at Rio Vista and 10,000 cfs in San Joaquin River at Vernalis. Once such flows have abated, the OMR flow requirements of the Action are again in place.

Off-ramps:
Temperature: Water temperature reaches 12°C based on a three station daily average (Rio Vista, Antioch, Mossdale). *or*
Biological: Onset of spawning (presence of spent females in SKT or at either facility).

ACTION 3: ENTRAINMENT PROTECTION OF LARVAL SMELT

Page 367

Action: Net daily OMR flow will be no more negative than -1,250 to -5,000 cfs based on a 14-day running average with a simultaneous 5-day running average within 25 percent of the applicable requirement for OMR. Depending on extant conditions (and the general guidelines below) specific OMR flows within this range are recommended by the SWG from the onset of Action 3 through its termination. The SWG will provide these recommendations based upon a weekly review of sampling data, from real-time salvage data at the CVP/SWP, and expertise and knowledge relating population status and predicted distribution to monitored physical variables of flow and turbidity. USFWS will make the final determination.

Timing: Initiate the action after reaching the triggers below, which are indicative of spawning activity and the probable presence of larval delta smelt in the South and Central Delta. Based upon daily salvage data, the SWG may recommend an earlier start to Action 3. USFWS will make the final determination.

Triggers:
Temperature: When temperature reaches 12°C based on three-station average at Mossdale, Antioch, and Rio Vista. *or*
Biological: Onset of spawning (presence of spent females in SKT or at either facility).

Off-ramps:
Temporal: June 30 *or*
Temperature: Water temperature reaches a daily average of 25°C for three consecutive days at Clifton Court Forebay.

ACTION 4: ESTUARINE HABITAT DURING FALL

Page 369

Action: Subject to adaptive management as described below, provide sufficient Delta outflow to maintain average X2 for September and October no greater (no further eastward) than 74 km in the Fall following Wet years and 81 km in the Fall following Above Normal years. The monthly average X2 must be maintained at or seaward of these values for each individual month and not averaged over the two month period. In November, the inflow to CVP/SWP reservoirs in the Sacramento Basin will be added to reservoir releases to provide an added increment of Delta inflow and to augment Delta outflow up to the fall target. The action will be evaluated and may be modified or terminated as determined by USFWS.

Timing: September 1 to November 30

Triggers: Wet and above normal WY type classification from the 1995 Water Quality Control Plan that is used to implement D-1641.

ACTION 5: TEMPORARY SPRING HEAD OF OLD RIVER BARRIER (HORB) AND THE TEMPORARY BARRIER PROJECT (TBP)

Page 377

Action: Do not install HORB if delta smelt entrainment is a concern. If installation of the HORB is not allowed, the agricultural barriers would be installed as described in the Project Description. If installation of the HORB is allowed, the agricultural barriers could be installed, but the flap gates would be tied in the open position until May 15.

Timing: The timing of the action would vary depending on the conditions. The normal installation of the spring temporary HORB and the TBP is in April.

Triggers: For Delta smelt, installation of the HORB will only occur when PTM results show that entrainment levels of Delta smelt will not increase beyond 1 percent at Station 815 as a result of installing the HORB.

Bay-Delta Standards

Contained in D-1641

DRAFT

<table>
<tr><th>CRITERIA</th><th>JAN</th><th>FEB</th><th>MAR</th><th>APR</th><th>MAY</th><th>JUN</th><th>JUL</th><th>AUG</th><th>SEP</th><th>OCT</th><th>NOV</th><th>DEC</th></tr>
<tr><td colspan="13"><i>FLOW/OPERATIONAL</i></td></tr>
<tr><td colspan="13">• Fish and Wildlife</td></tr>
<tr><td>SWP/CVP Export Limits</td><td></td><td></td><td></td><td colspan="2">1,500cfs [1]</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Export/Inflow Ratio [2]</td><td>65%</td><td colspan="5">35% of Delta Inflow [3]</td><td colspan="6">65% of Delta Inflow</td></tr>
<tr><td>Minimum Delta Outflow</td><td>[4]</td><td></td><td></td><td></td><td></td><td></td><td colspan="6">3,000 - 8,000 cfs [4]</td></tr>
<tr><td>Habitat Protection Outflow</td><td></td><td colspan="5">7,100 - 29,200 cfs [5]</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Salinity Starting Condition [6]</td><td></td><td>[6]</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td colspan="13">River Flows:</td></tr>
<tr><td>@ Rio Vista</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td colspan="4">3,000 - 4,500 cfs [7]</td></tr>
<tr><td>@ Vernalis - Base</td><td></td><td colspan="3">710 - 3,420 cfs [8]</td><td colspan="2">[8]</td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>- Pulse</td><td></td><td></td><td></td><td colspan="2">[9]</td><td></td><td></td><td></td><td></td><td>+28TAF</td><td></td><td></td></tr>
<tr><td>Delta Cross Channel Gates</td><td>[10]</td><td colspan="4">Closed</td><td colspan="2">[11]</td><td></td><td></td><td></td><td colspan="2">Conditional [10]</td></tr>
<tr><td colspan="13"><i>WATER QUALITY STANDARDS</i></td></tr>
<tr><td colspan="13">• Municipal and Industrial</td></tr>
<tr><td>All Export Locations</td><td colspan="12">≤ 250 mg/l Cl</td></tr>
<tr><td>Contra Costa Canal</td><td colspan="12">150 mg/l Cl for the required number of days [12]</td></tr>
<tr><td colspan="13">• Agriculture</td></tr>
<tr><td>Western/Interior Delta</td><td></td><td></td><td></td><td colspan="5">Max.14-day average EC mmhos/cm [13]</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Southern Delta [14]</td><td colspan="3">1.0 mS</td><td colspan="5">30 day running avg EC 0.7 mS</td><td colspan="4">1.0 mS</td></tr>
<tr><td colspan="13">• Fish and Wildlife</td></tr>
<tr><td>San Joaquin River Salinity [15]</td><td></td><td></td><td></td><td colspan="2">14-day avg; 0.44 EC</td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
<tr><td>Suisun Marsh Salinity [16]</td><td>12.5 EC</td><td colspan="2">8.0 EC</td><td colspan="2">11.0 EC</td><td></td><td></td><td></td><td></td><td>19.0 EC</td><td>[17]</td><td>15.5 EC</td></tr>
</table>

[#] **See Footnotes**

Operations Compliance and Studies Section

Revised 9/29/00

Preliminary: Subject to Revision

Footnotes

[1] Maximum 3-day running average of combined export rate (cfs) which includes Tracy Pumping Plant and Clifton Court Forebay Inflow less Byron-Bethany pumping.

Year Type	All
Apr15 - May15*	The greater of 1,500 or 100% of 3-day avg. Vernalis flow

* This time period may need to be adjusted to coincide with fish migration. Maximum export rate may be varied by CalFed Op's group.

[2] The maximum percentage of average Delta inflow (use 3-day average for balanced conditions with storage withdrawal, otherwise use 14-day average) diverted at Clifton Court Forebay (excluding Byron-Bethany pumping) and Tracy Pumping Plant using a 3-day average. (These percentages may be adjusted upward or downward depending on biological conditions, providing there is no net water cost.)

[3] The maximum percent Delta inflow diverted for Feb may vary depending on the January 8RI.

Jan 8RI	Feb exp. limit
≤ 1.0 MAF	45%
between 1.0 & 1.5 MAF	35%-45%
> 1.5 MAF	35%

[4] Minimum monthly average Delta outflow (cfs). If monthly standard ≤ 5,000 cfs, then the 7-day average must be within 1,000 cfs of standard; if monthly standard > 5,000 cfs, then the 7-day average must be ≥ 80% of standard.

Year Type	All	W	AN	BN	D	C
Jan	4,500*					
Jul		8,000	8,000	6,500	5,000	4,000
Aug		4,000	4,000	4,000	3,500	3,000
Sep	3,000					
Oct		4,000	4,000	4,000	4,000	3,000
Nov-Dec		4,500	4,500	4,500	4,500	3,500

* Increase to 6,000 if the Dec 8RI is greater than 800 TAF

[5] Minimum 3-day running average of daily Delta outflow of 7,100 cfs OR: either the daily average or 14-day running average EC at Collinsville is less than 2.64 mmhos/cm (This standard for March may be relaxed if the Feb 8RI is less than 500 TAF. The standard does not apply in May and June if the May estimate of the SRI IS < 8.1 MAF at the 90% exceedence level in which case a minimum 14-day running average flow of 4,000 cfs is required.) For additional Delta outflow objectives, see **TABLE A**.

[6] February starting salinity: If Jan 8RI > 900 TAF, then the daily or 14-day running average EC @ Collinsville must be ≤ 2.64 mmhos/cm for at least one day between Feb 1-14. If Jan 8RI is between 650 TAF and 900 TAF, then the CalFed Op's group will determine if this requirement must be met.

[7] Rio Vista minimum monthly average flow rate in cfs (the 7-day running average shall not be less than 1,000 below the monthly objective).

Year Type	All	W	AN	BN	D	C
Sep	3,000					
Oct		4,000	4,000	4,000	4,000	3,000
Nov-Dec		4,500	4,500	4,500	4,500	3,500

[8] BASE Vernalis minimum monthly average flow rate in cfs (the 7-day running average shall not be less than 20% below the objective). Take the higher objective if X2 is required to be west of Chipps Island.

Year Type	All	W	AN	BN	D	C
Feb-Apr14 and May16-Jun		2,130 or 3,420	2,130 or 3,420	1,420 or 2,280	1,420 or 2,280	710 or 1,140

[9] PULSE Vernalis minimum monthly average flow rate in cfs. Take the higher objective if X2 is required to be at or west of Chipps Island.

Year Type	All	W	AN	BN	D	C
Apr15 - May15		7,330 or 8,620	5,730 or 7,020	4,620 or 5,480	4,020 or 4,880	3,110 or 3,540
Oct	1,000*					

* Up to an additional 28 TAF pulse/attraction flow to bring flows up to a monthly average of 2,000 cfs except for a critical year following a critical year. Time period based on real-time monitoring and determined by CalFed Op's group.

[10] For the Nov-Jan period, Delta Cross Channel gates may be closed for up to a total of 45 days.

[11] For the May 21-June 15 period, close Delta Cross Channel gates for a total of 14 days per CALFED Op's group. During the period the Delta cross channel gates may close 4 consecutive days each week, excluding weekends.

[12] Minimum # of days that the mean daily chlorides ≤ 150 mg/l must be provided in intervals of not less than 2 weeks duration. Standard applies at Contra Costa Canal Intake or Antioch Water Works Intake.

Year Type	W	AN	BN	D	C
# Days	240	190	175	165	155

[13] The maximum14-day running average of mean daily EC (mmhos/cm) depends on water year type.

	WESTERN DELTA				INTERIOR DELTA			
	Sac River @ Emmaton		SJR @ Jersey Point		Mokelumne R @ Terminous		SJR @ San Andreas	
Year Type	0.45 EC from April 1 to date shown	EC value from date shown to Aug15 *	0.45 EC from April 1 to date shown	EC value from date shown to Aug15 *	0.45 EC from April 1 to date shown	EC value from date shown to Aug15 *	0.45 EC from April 1 to date shown	EC value from date shown to Aug15 *
W	Aug 15		Aug 15		Aug 15		Aug 15	
AN	Jul 1	0.63	Aug 15		Aug 15		Aug 15	
BN	Jun 20	1.14	Jun 20	0.74	Aug 15		Aug 15	
D	Jun 15	1.67	Jun 15	1.35	Aug 15		Jun 25	0.58
C		2.78		2.20		0.54		0.87

* When no date is shown, EC limit continues from April 1.

[14] As per D-1641, for San Joaquin River at Vernalis: however, the April through August maximum 30- day running average EC for San Joaquin River at Brandt Bridge,Old River near Middle River, and Old River at Tracy Road Bridge shall be 1.0 EC until April 1, 2005 when the value will be 0.7 EC.

[15] Compliance will be determined between Jersey Point & Prisoners Point.
Does not apply in critical years or in May when the May 90% forecast of SRI ≤ 8.1 MAF.

[16] During deficiency period, the maximum monthly average mhtEC at Western Suisun Marsh stations as per SMPA is:

Month	mhtEC
Oct	19.0
Nov	16.5
Dec-Mar	15.6
Apr	14.0
May	12.5

[17] In November, maximum monthly average mhtEC = 16.5 for Western Marsh stations and maximum monthly average mhtEC = 15.5 for Eastern Marsh stations in all periods types.

TABLE A

Number of Days When Max. Daily Average Electrical Conductivity of 2.64 mmhos/cm Must Be Maintained at Chipps Island and Port Chicago. (This can also be met with a maximum 14-day running average EC of 2.64 mmhos/cm, or 3-day running average Delta outflows of 11,400 cfs and 29,200 cfs, respectively.) Port Chicago Standard is triggered only when the 14-day average EC for the last day of the previous month is 2.64 mmhos/cm or less. PMI is previous month's 8RI. If salinity/flow objectives are met for a greater number of days than required for any month, the excess days shall be applied towards the following month's requirement. The number of day's for values of the PMI between those specified below shall be determined by linear interpolation.

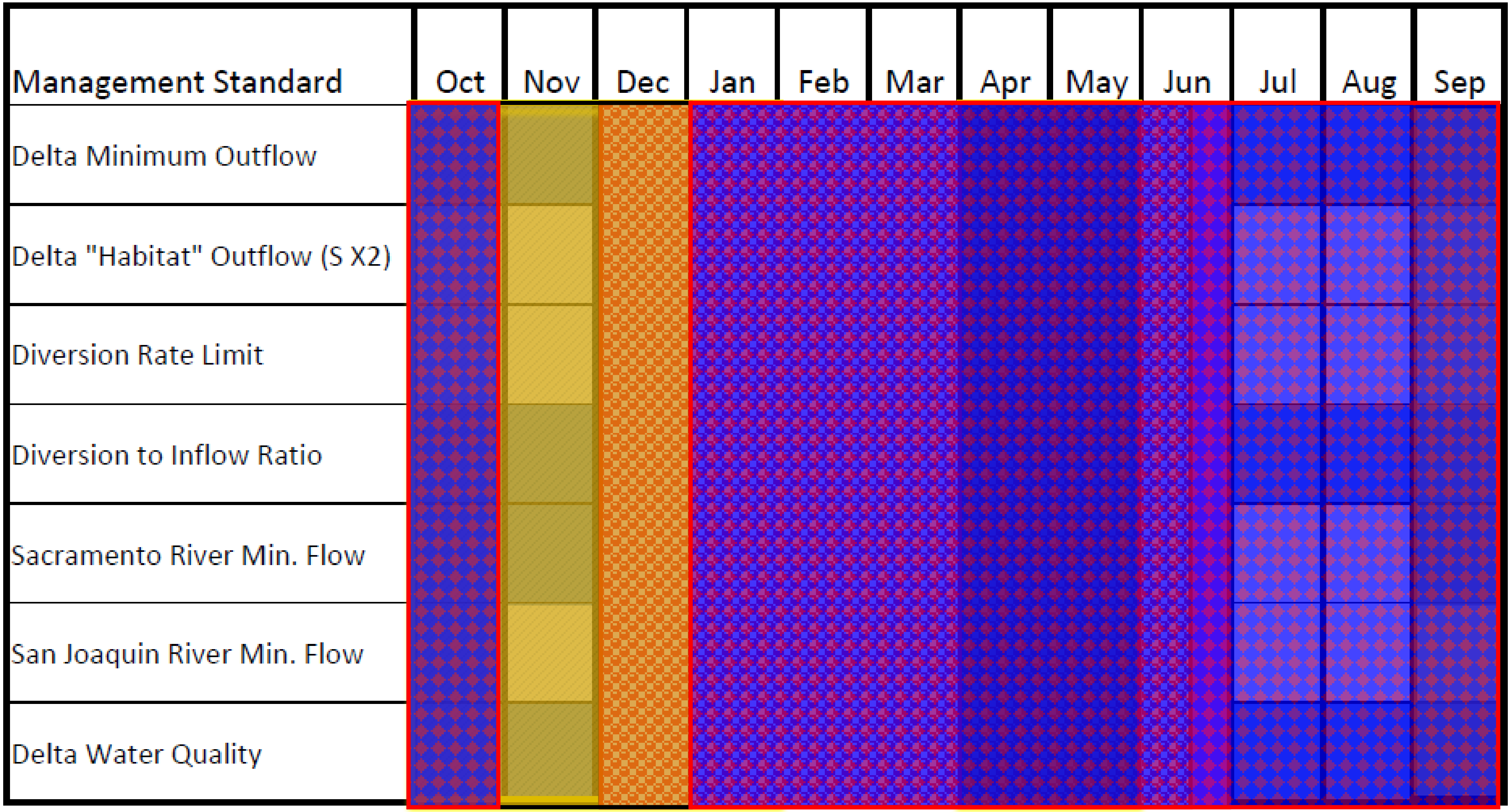
PMI	Chipps Island (Chipps Island Station D10)				
(TAF)	FEB	MAR	APR	MAY	JUN
≤ 500	0	0	0	0	0
750	0	0	0	0	0
1000	28*	12	2	0	0
1250	28	31	6	0	0
1500	28	31	13	0	0
1750	28	31	20	0	0
2000	28	31	25	1	0
2250	28	31	27	3	0
2500	28	31	29	11	1
2750	28	31	29	20	2
3000	28	31	30	27	4
3250	28	31	30	29	8
3500	28	31	30	30	13
3750	28	31	30	31	18
4000	28	31	30	31	23
4250	28	31	30	31	25
4500	28	31	30	31	27
4750	28	31	30	31	28
5000	28	31	30	31	29
5250	28	31	30	31	29
≥ 5500	28	31	30	31	30

*When 800 TAF < PMI < 1000 TAF, the number of days is determined by linear interpolation between 0 and 28 days.

PMI	Port Chicago (continuous recorder at Port Chicago)				
(TAF)	FEB	MAR	APR	MAY	JUN
0	0	0	0	0	0
250	1	0	0	0	0
500	4	1	0	0	0
750	8	2	0	0	0
1000	12	4	0	0	0
1250	15	6	1	0	0
1500	18	9	1	0	0
1750	20	12	2	0	0
2000	21	15	4	0	0
2250	22	17	5	1	0
2500	23	19	8	1	0
2750	24	21	10	2	0
3000	25	23	12	4	0
3250	25	24	14	6	0
3500	25	25	16	9	0
3750	26	26	18	12	0
4000	26	27	20	15	0
4250	26	27	21	18	1
4500	26	28	23	21	2
4750	27	28	24	23	3
5000	27	28	25	25	4
5250	27	29	25	26	6
5500	27	29	26	28	9
5750	27	29	27	28	13
6000	27	29	27	29	16
6250	27	30	27	29	19
6500	27	30	28	30	22
6750	27	30	28	30	24
7000	27	30	28	30	26
7250	27	30	28	30	27
7500	27	30	29	30	28
7750	27	30	29	31	28
8000	27	30	29	31	29
8250	28	30	29	31	29
8500	28	30	29	31	29
8750	28	30	29	31	30
9000	28	30	29	31	30
9250	28	30	29	31	30
9500	28	31	29	31	30
9750	28	31	29	31	30
10000	28	31	30	31	30
> 10000	28	31	30	31	30

When Regulations Affect the CVP & SWP

plus NMFS Temperature and Flow Standards



Management Standard	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep
Delta Minimum Outflow												
Delta "Habitat" Outflow (S X2)												
Diversion Rate Limit												
Diversion to Inflow Ratio												
Sacramento River Min. Flow												
San Joaquin River Min. Flow												
Delta Water Quality												

Ecosystem Changes to the Bay-Delta Estuary: A Technical Assessment of Available Scientific Information

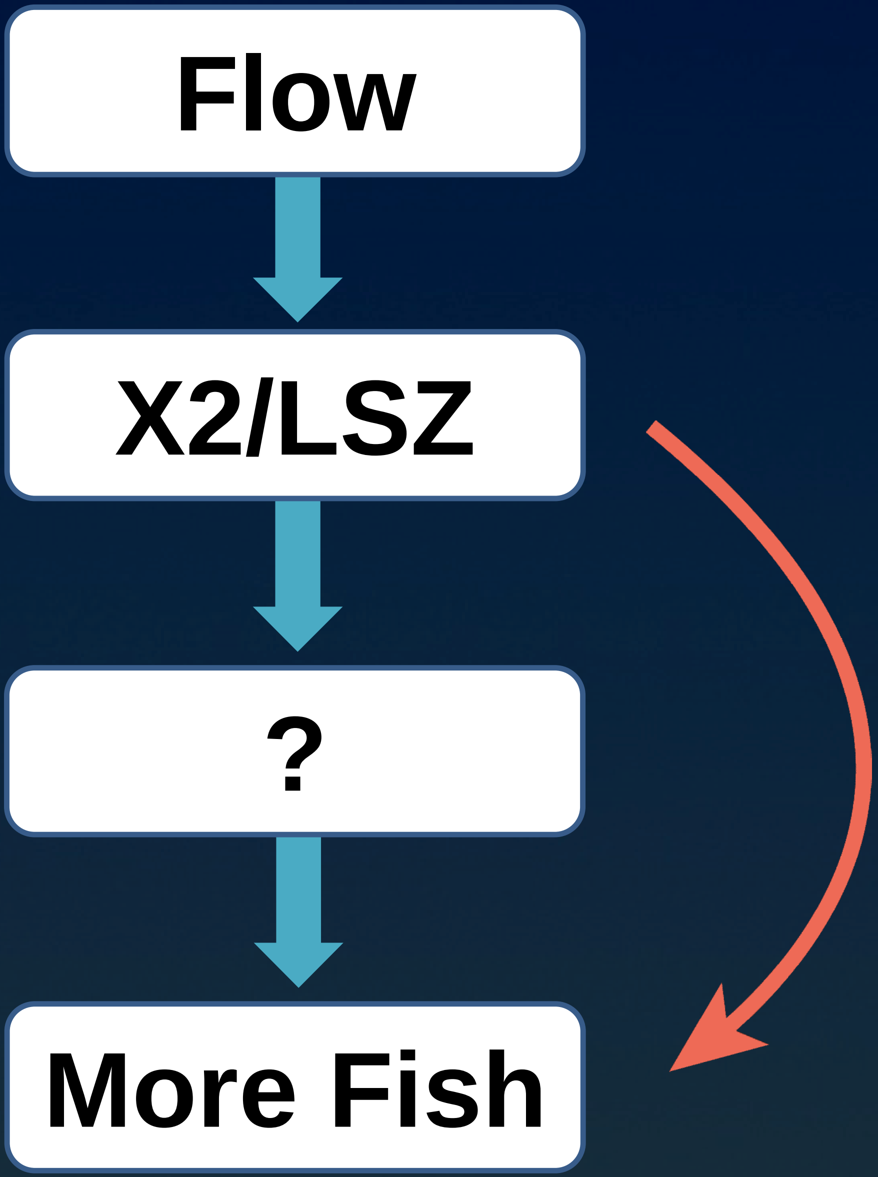
State Water Resources Control Board
Phase II Comprehensive Review of the Bay-Delta Plan

Workshop 1: Ecosystem Changes and Low-Salinity Zone

September 6, 2012

Submitted by: State Water Contractors, Inc.
San Luis & Delta-Mendota Water Authority

Bay-Delta Model?



Flow

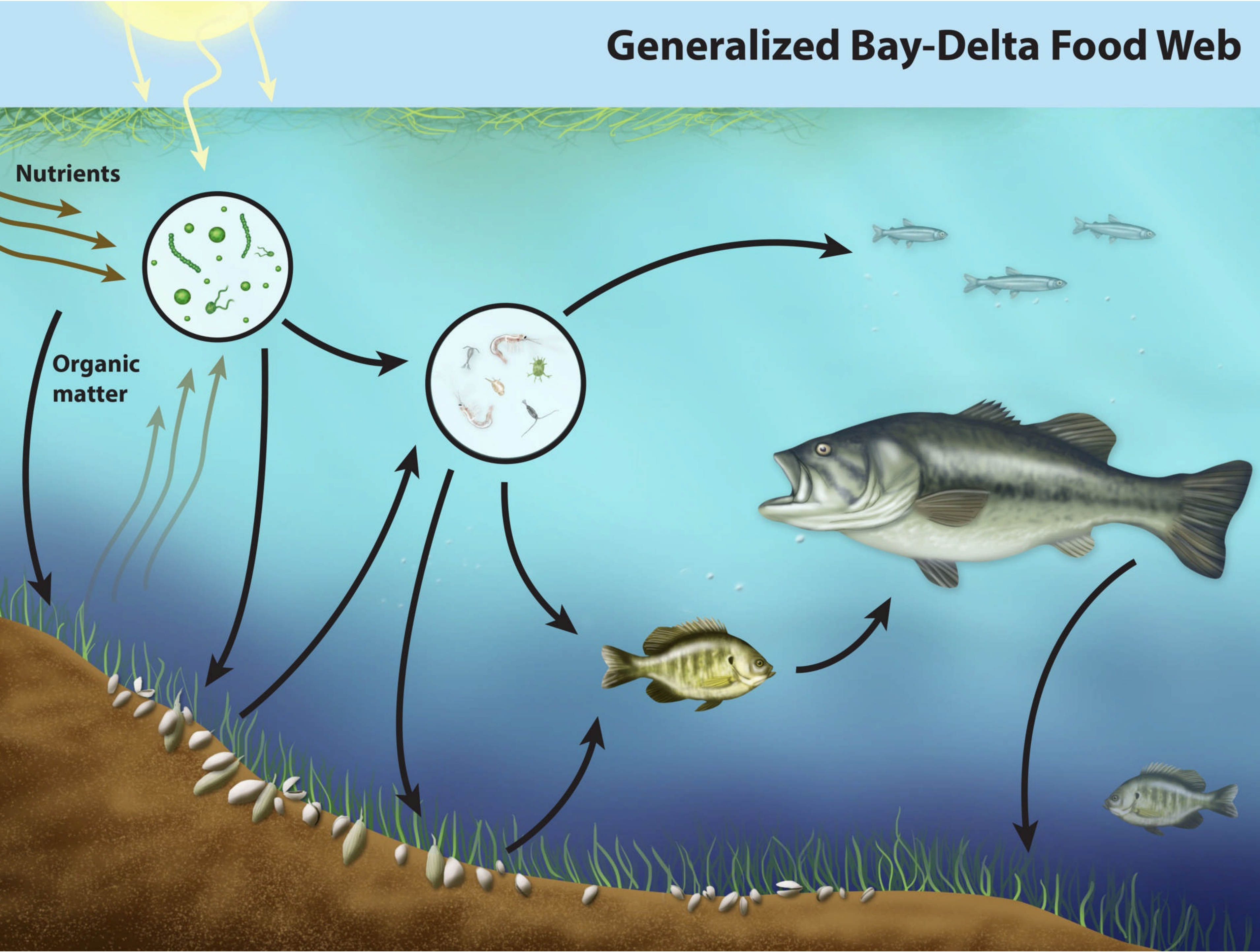
X2/LSZ

?

More Fish

“Although increases in quantity of habitat may contribute, the mechanism chiefly responsible for the X2 relationship for longfin smelt remains unknown.”

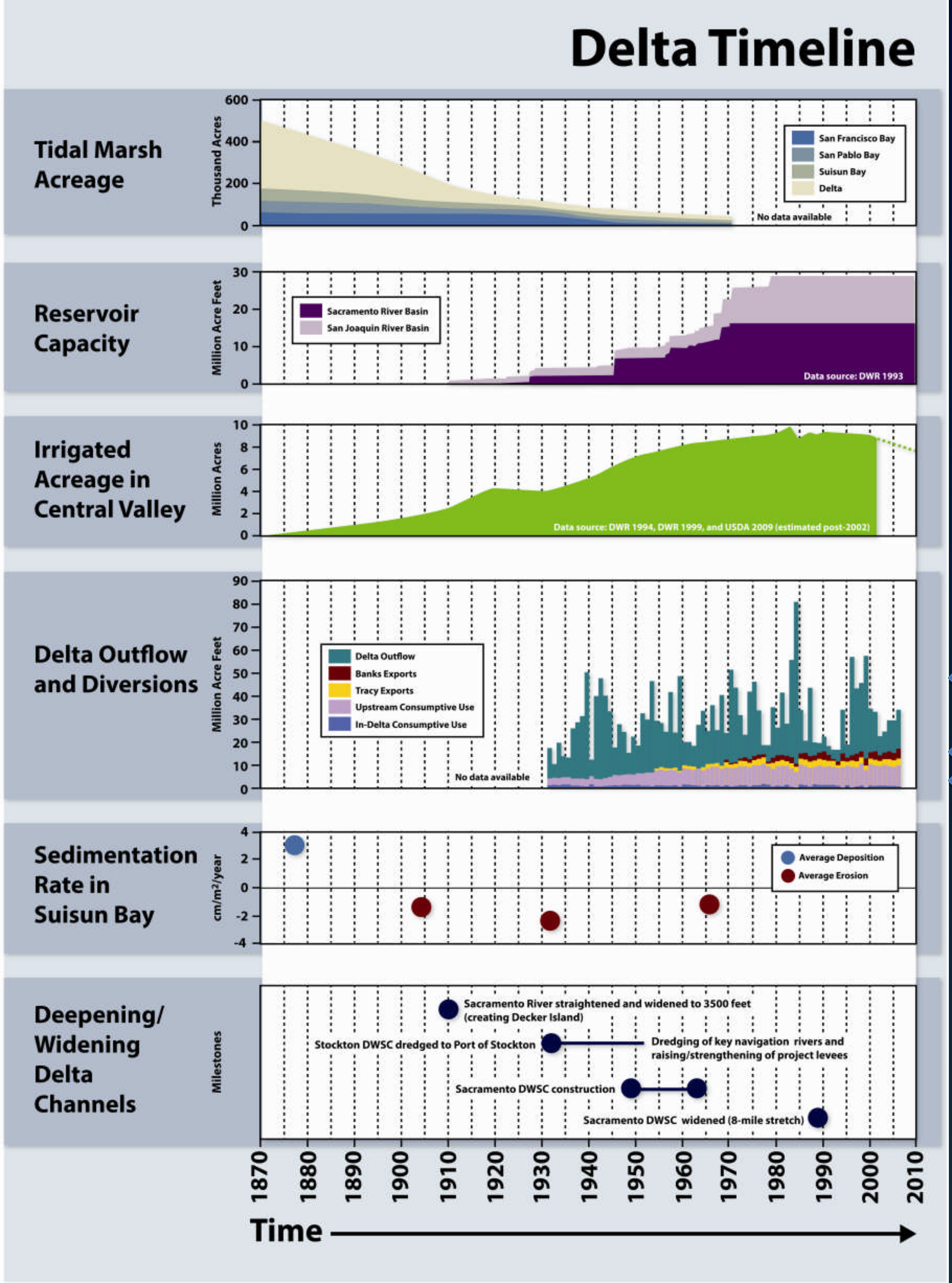
Kimmerer (2009)



Generalized Bay-Delta Food Web

Investigating Ecosystem Changes

- Physical
- Biological
- Chemical

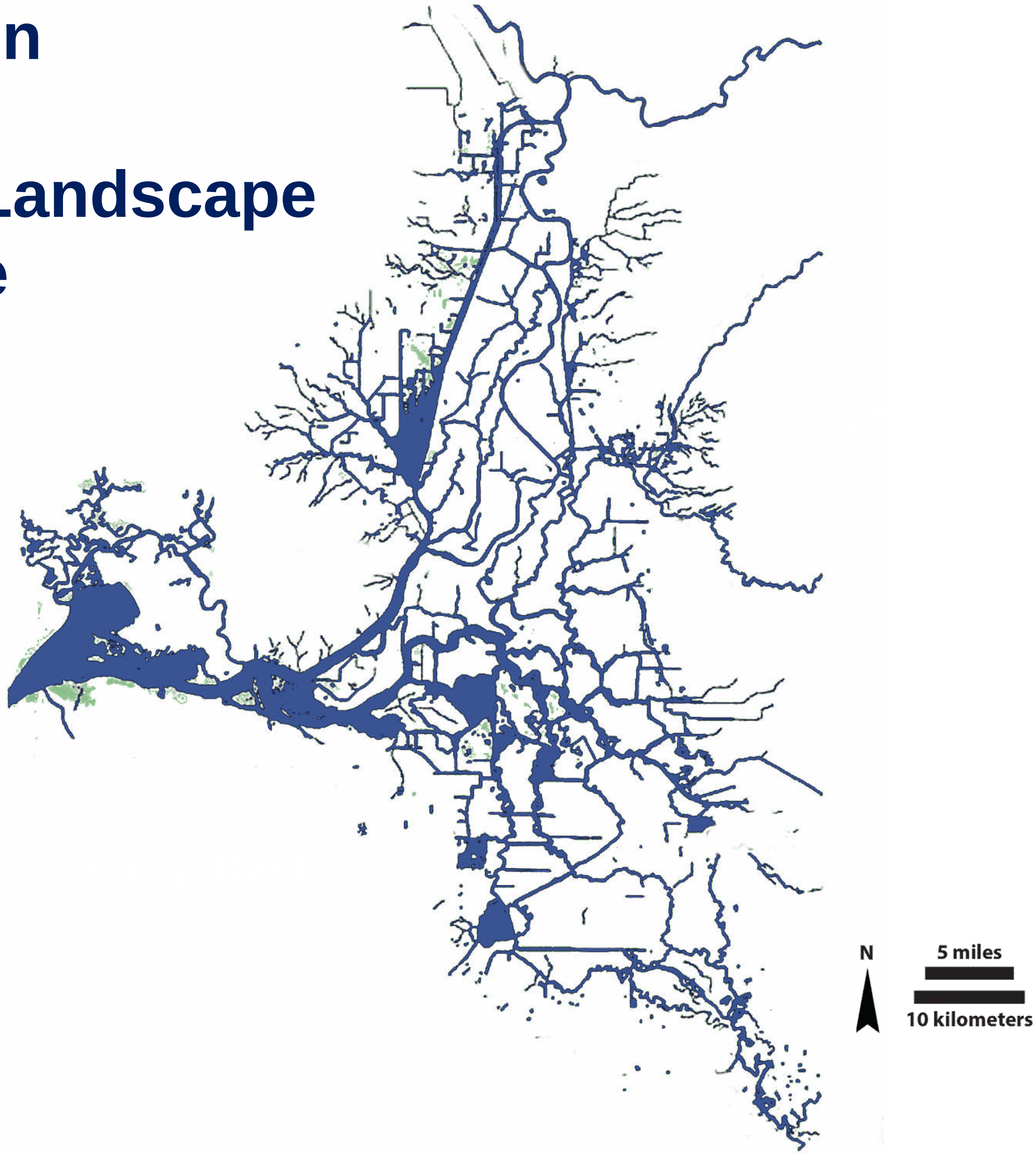


Delta outflow

Exports

Consumptive use

Changes in Bay-Delta Physical Landscape Over Time



🟦 Waterways

🟩 Tidal Wetlands

Bay-Delta Ecosystem Changes

1980 - 2011

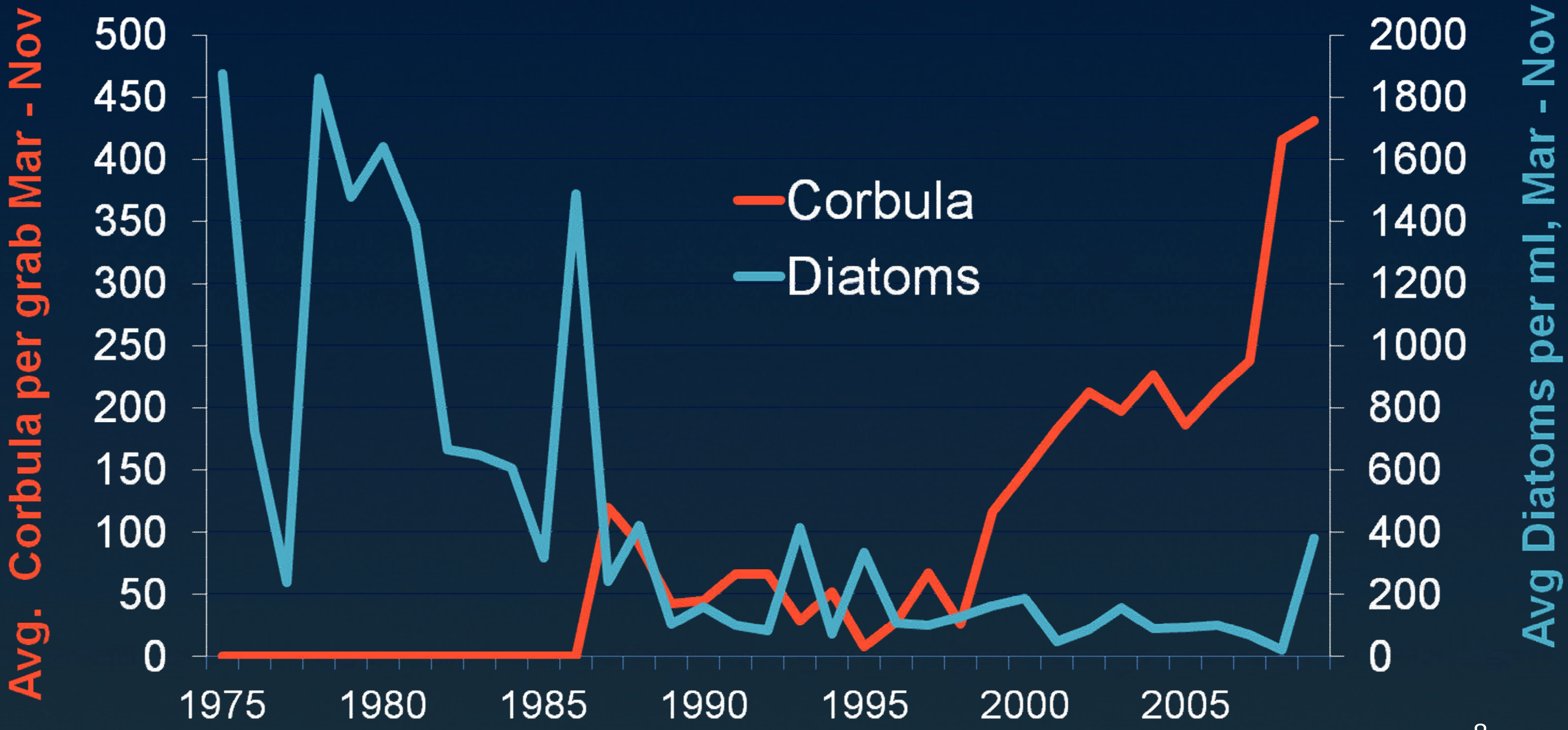
Eurytemora

Limnoithona

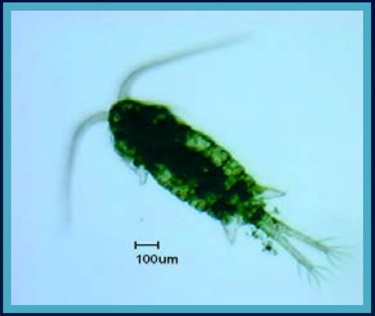
- ↑ Nutrients
- ↑ Clams
- ↓ Diatoms and chlorophyll
- ↓ Summer/fall turbidity
- ↓ *Eurytemora*, high value food
- ↑ *Limnoithona*, low value food
- ↑ Toxic Algae
- ↑ Submerged plants (SAV)
- ↑ Predator Fish (Bass, etc)
- ↓ Pelagic fish

Corbula amurensis
(Overbite clam)

Corbula and Diatoms Suisun Bay 1975 - 2011

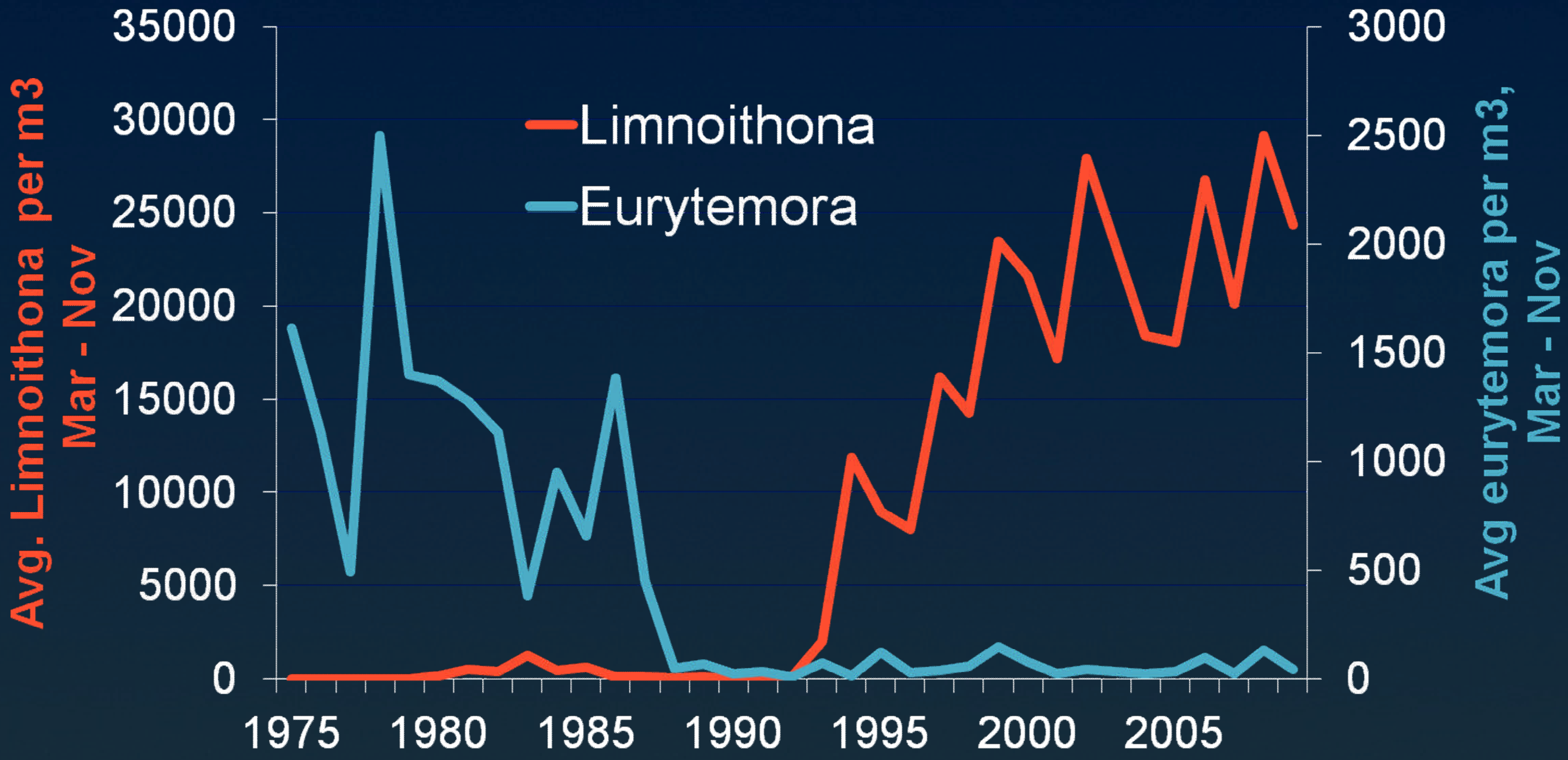


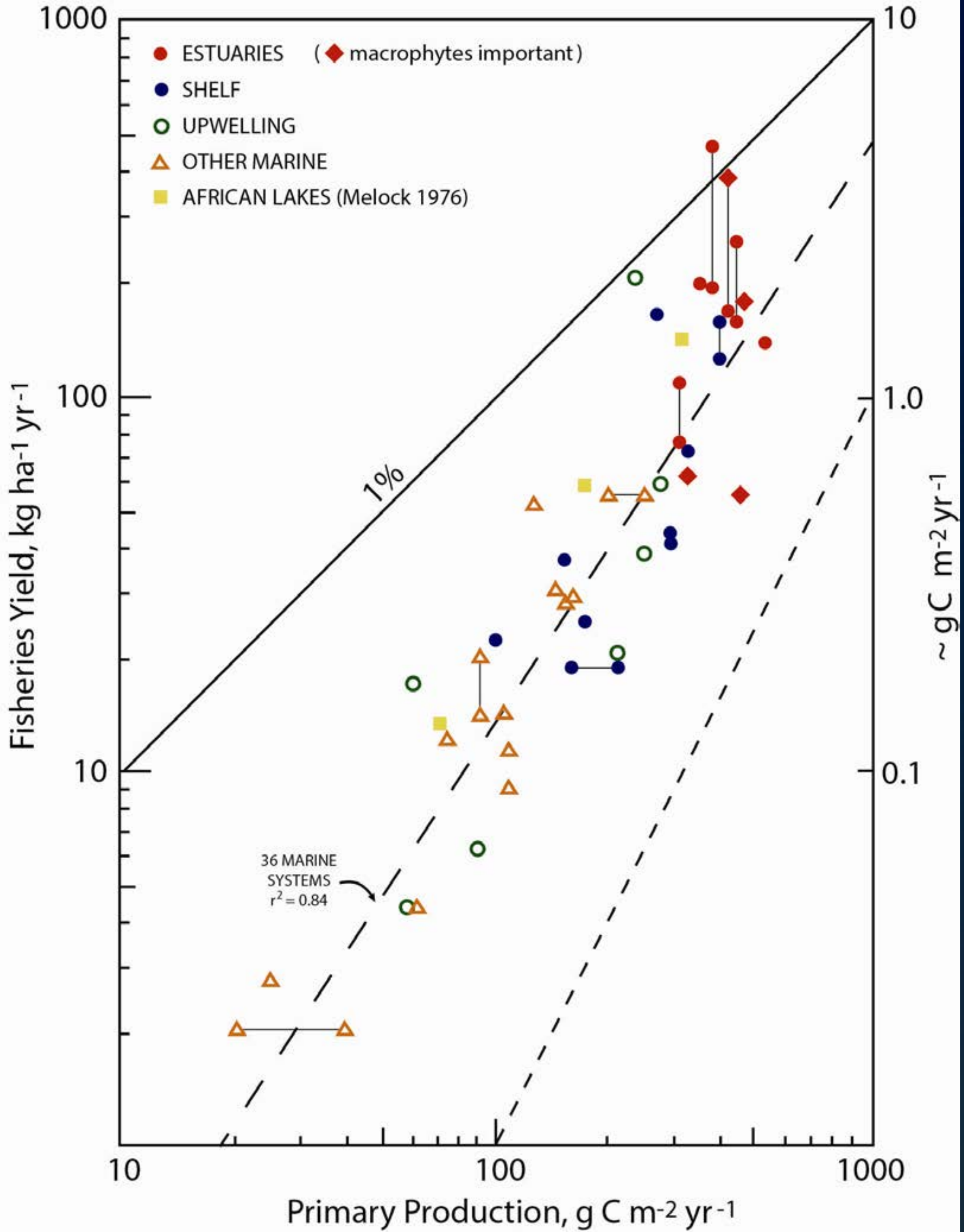
Limnoithona and Eurytemora
Suisun Bay **1975 - 2011**



Eurytemora

Limnoithona

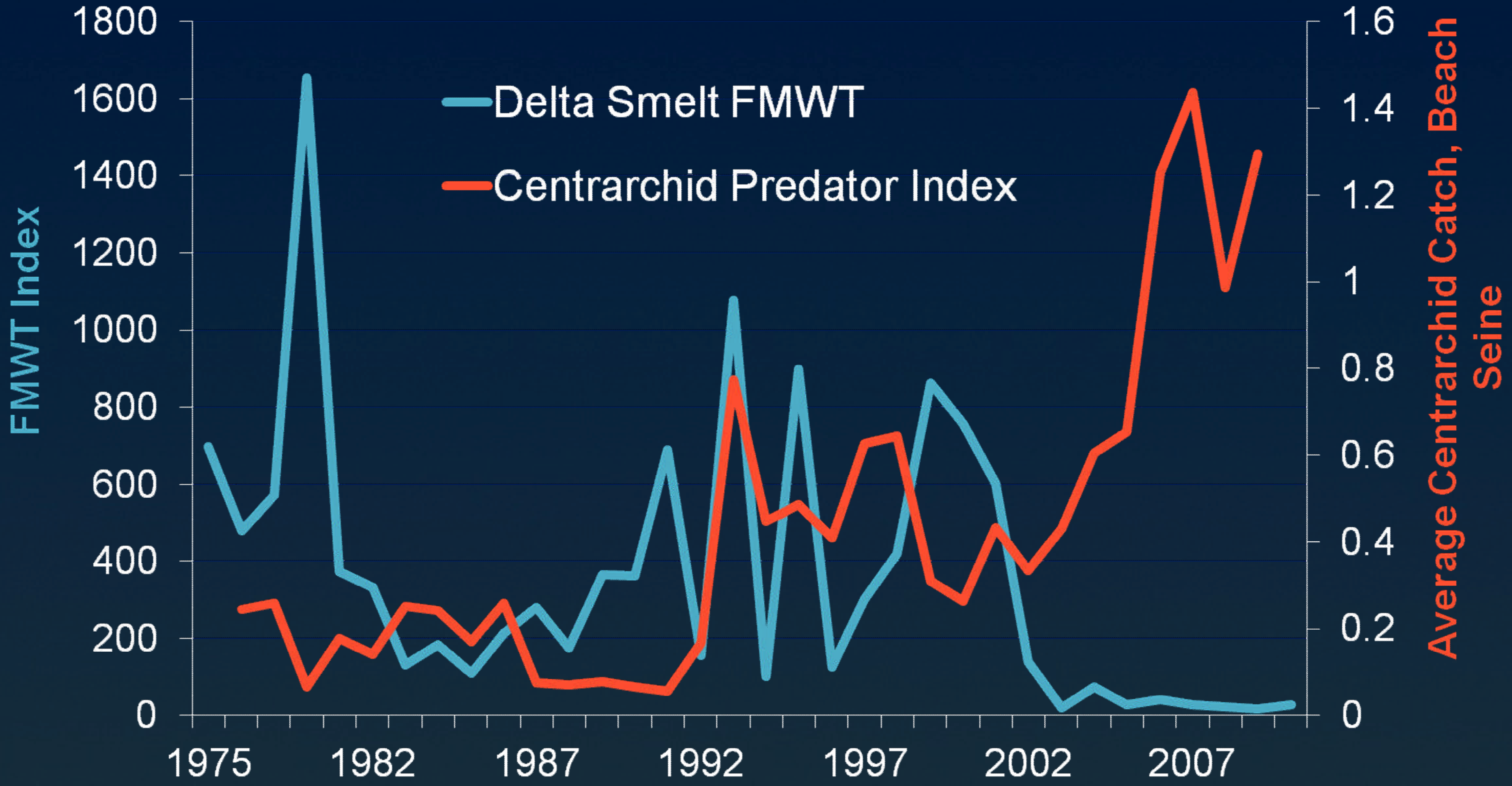




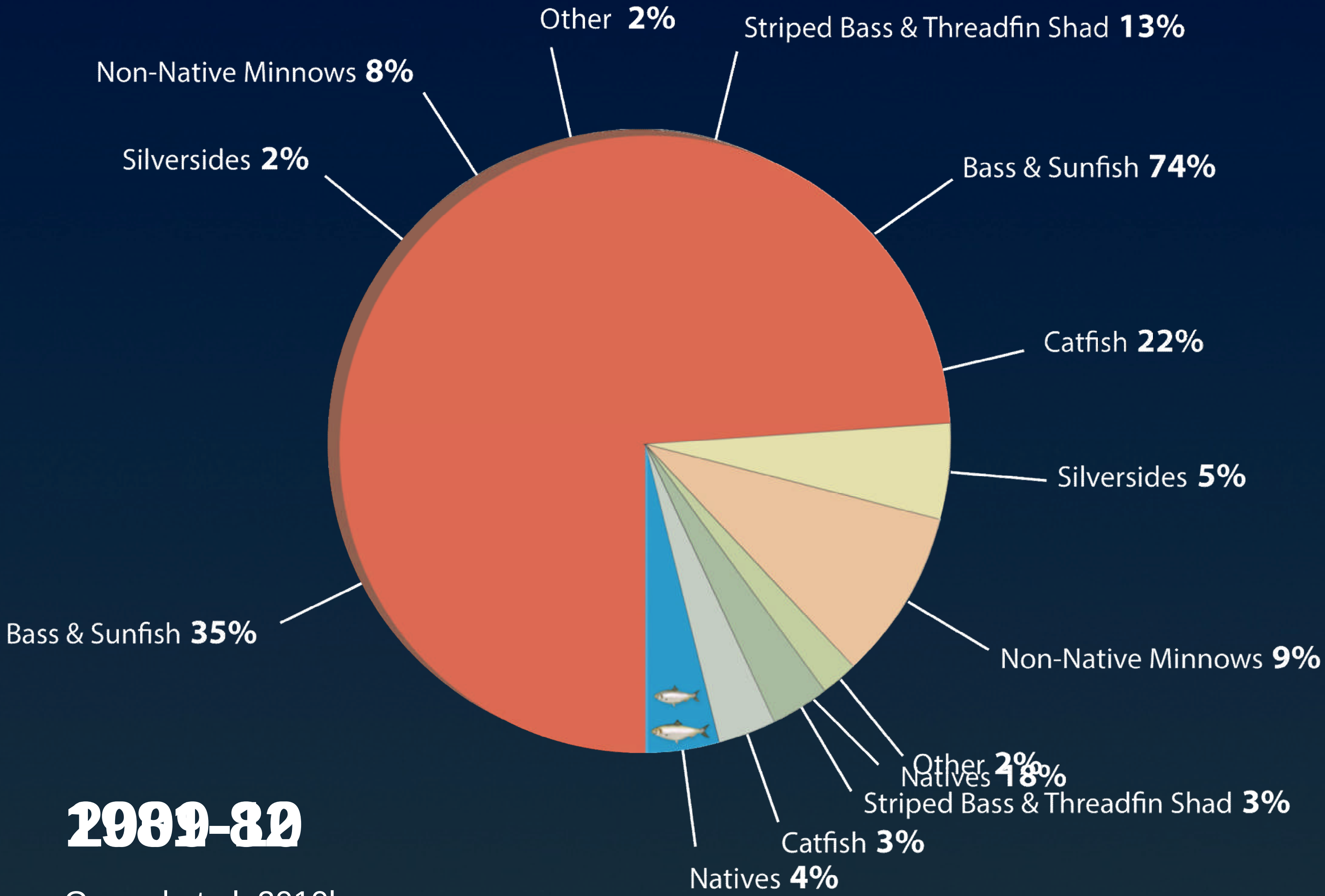
Source: Nixon, 1988

Centrarchid Predator Index and Delta Smelt Fall Abundance

1975 - 2011



Rise of the Centrarchids



Source: Conrad et al. 2010b

Possible Ecosystem Stressors

- Food web
- Physical landscape
- Water Temperature
- Turbidity
- Flows and diversions

Speakers and Topics

David Fullerton	- Overview
Dr. Paul Hutton	- Historical Flows
Sheila Greene	- Natural Flow Functions
Dr. Chuck Hanson	- Habitat & Invasives
Dr. Pat Glibert	- Nutrients & the Food Web

Flow & Salinity Time Trends

in Perspective

- Unimpaired flow ≠ Natural flow
- Climate must be accounted for when evaluating time trends
- CVP-SWP operation is NOT the primary driver of change between the two most recent decades

Unimpaired ≠ Natural

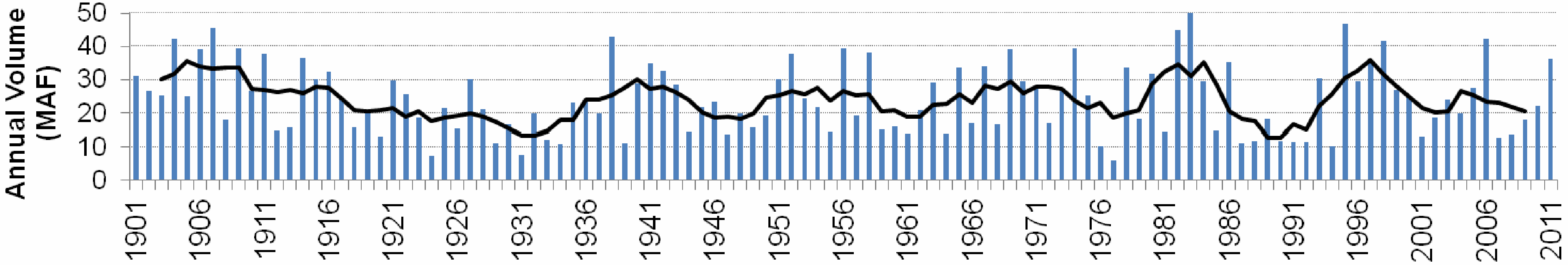
- Unimpaired flow is a calculation

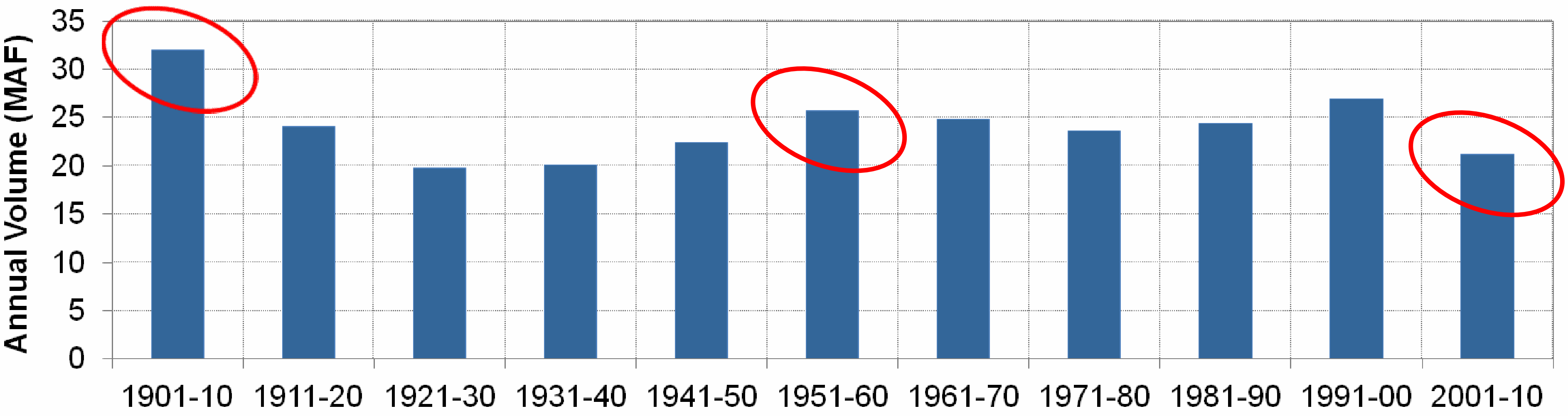
- Unimpaired flow calculations are NOT good approximations for natural conditions
  - Levees
  - Channelization and dredging

- Early 20th Century conditions ≠ natural conditions

Climate

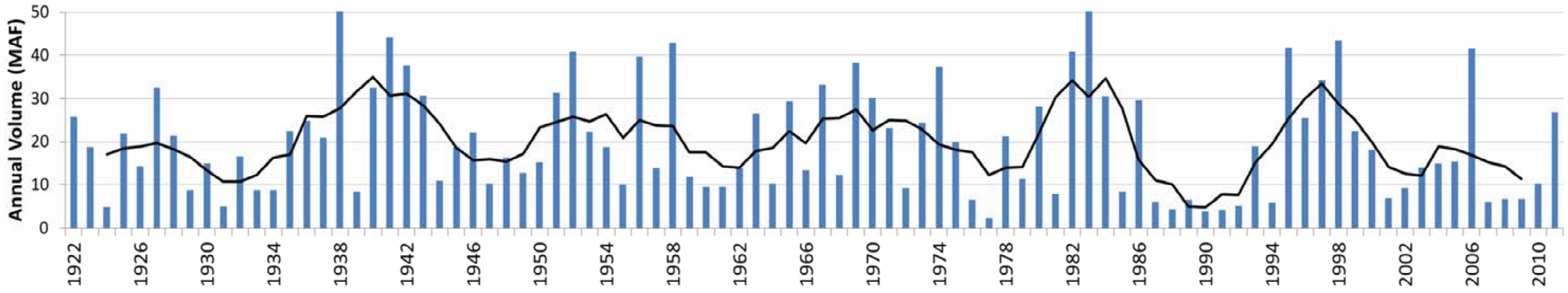
Measured by Eight River Index

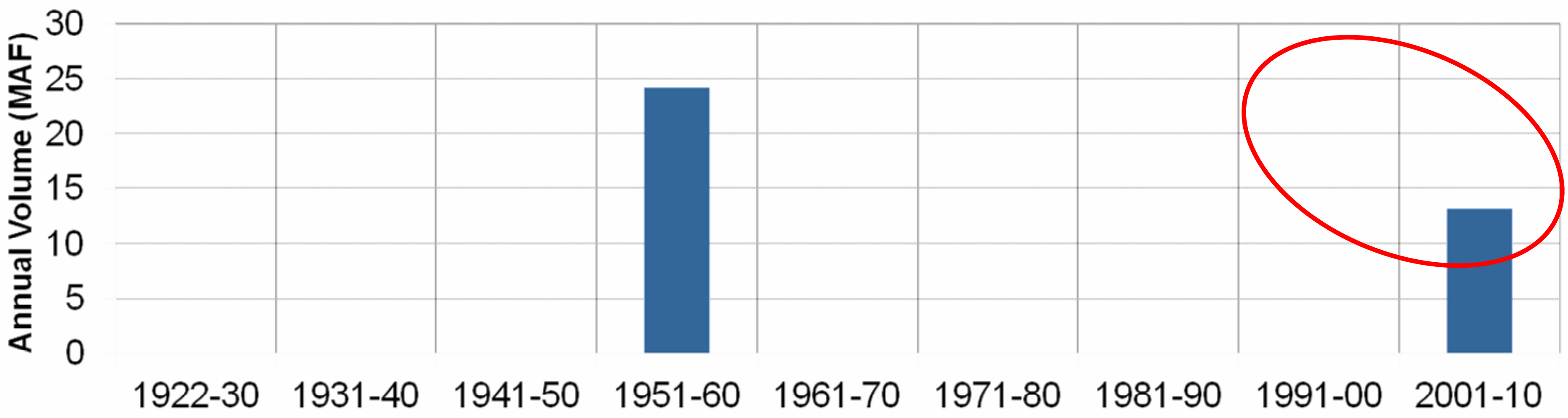




Data Sources: CDEC and Dept. of Public Works (1923) Flow in California Streams

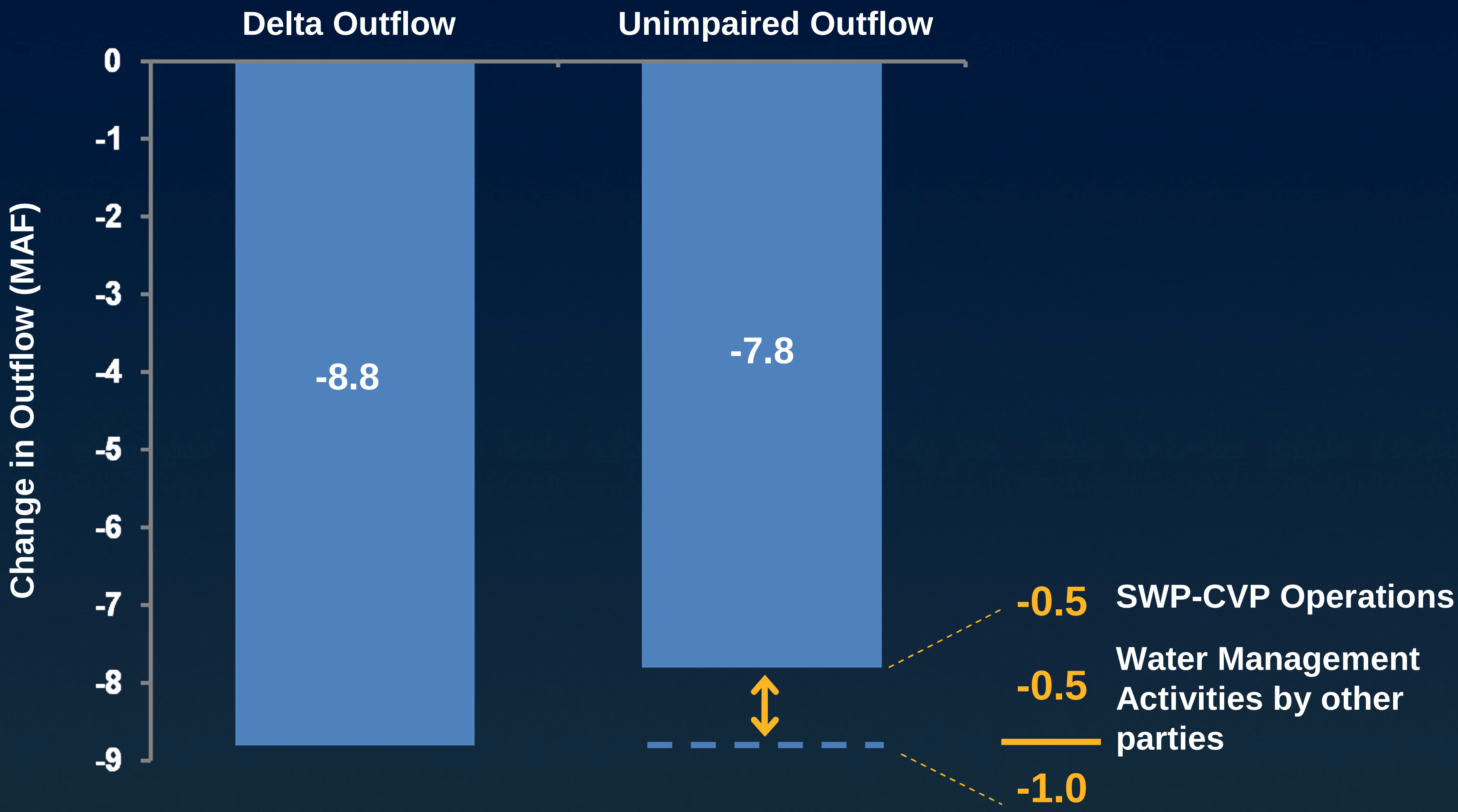
Annual Delta Outflow





Data Sources: DAYFLOW (Water Years 1930-2010) and DWR Bay-Delta Office (Water Years 1922-1929)

Annual differences between 1990s & 2000s



-0.5 **SWP-CVP Operations**

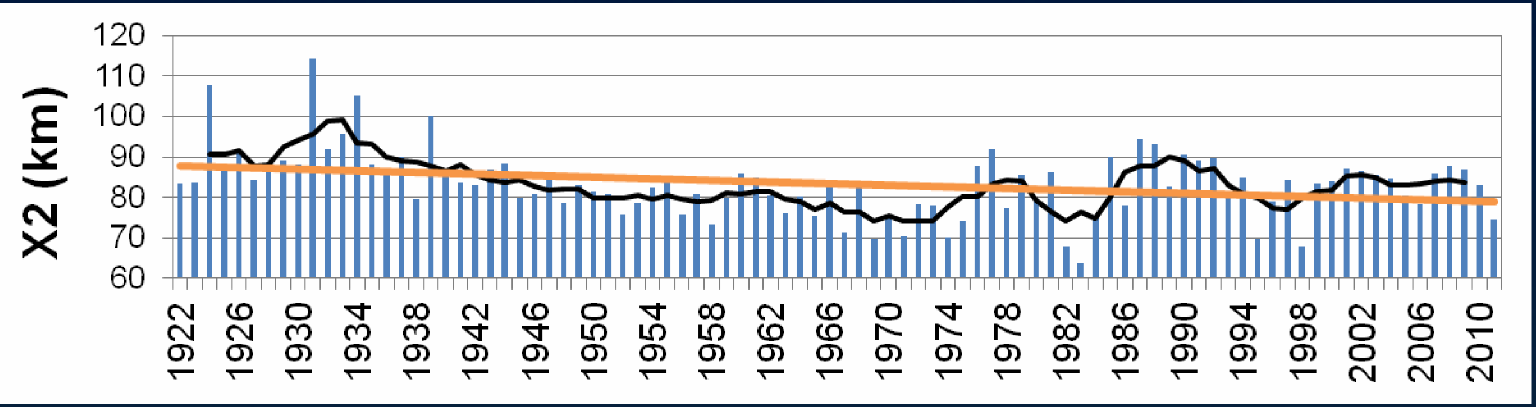
-0.5 **Water Management Activities by other parties**

-1.0

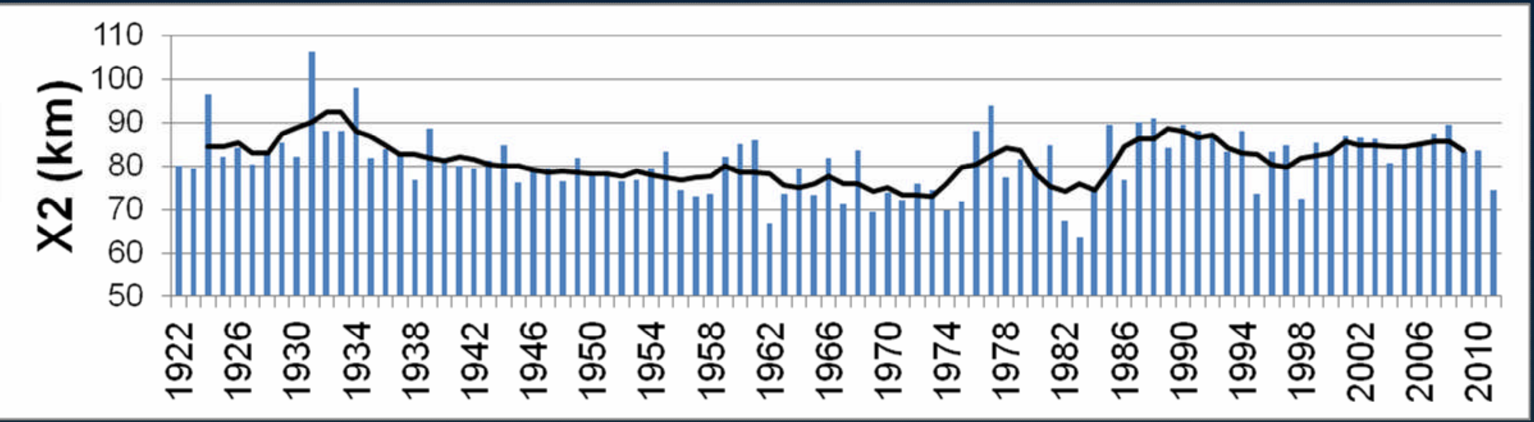
Data Sources: DAYFLOW, CDEC, DWR Bay-Delta Office

Fall X2 Position

September



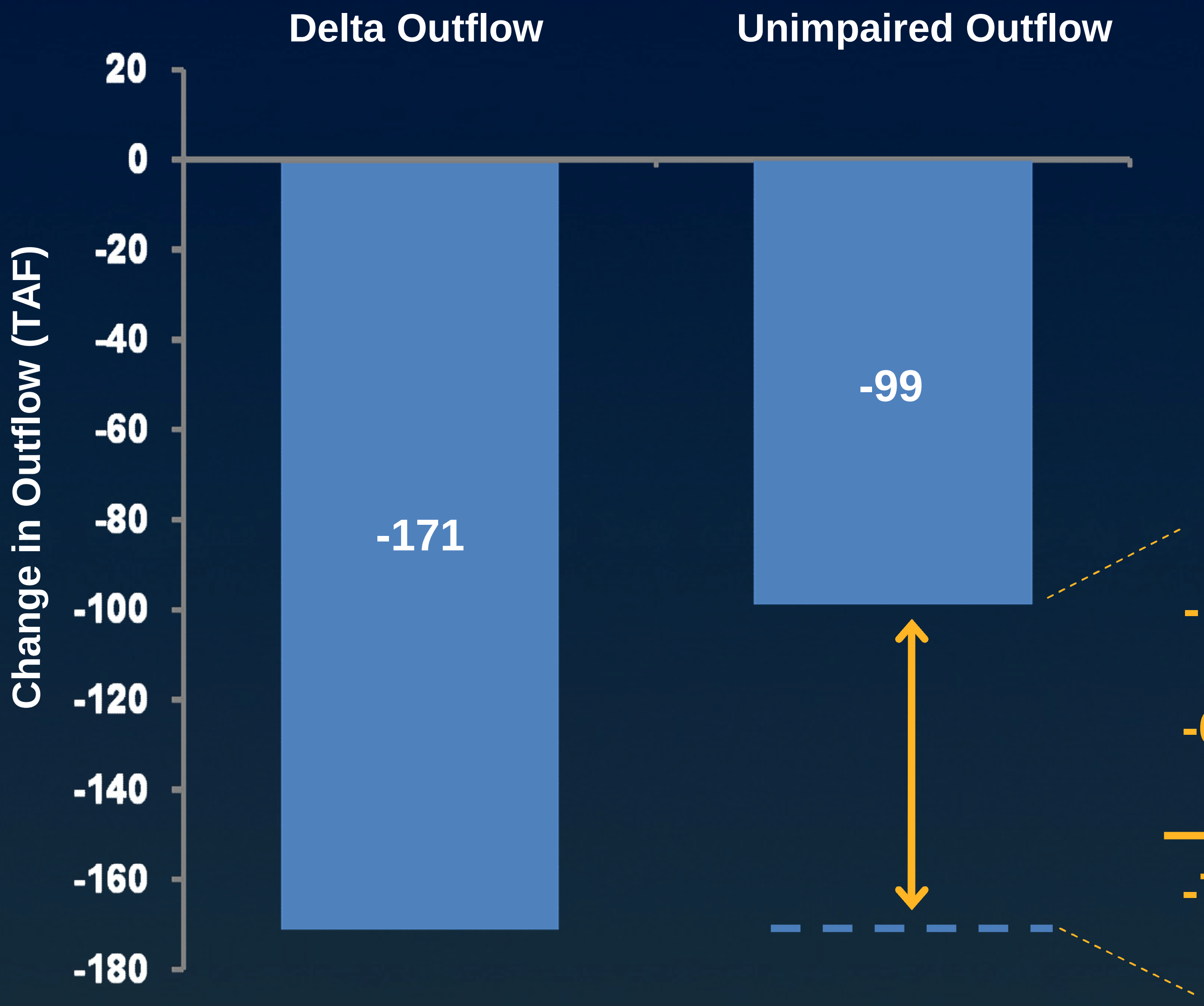
October



FRESHER

Data Source: X2 position calculated from monthly flow using K-M equation (Jassby et. al. 1995)

September differences between 1990s & 2000s



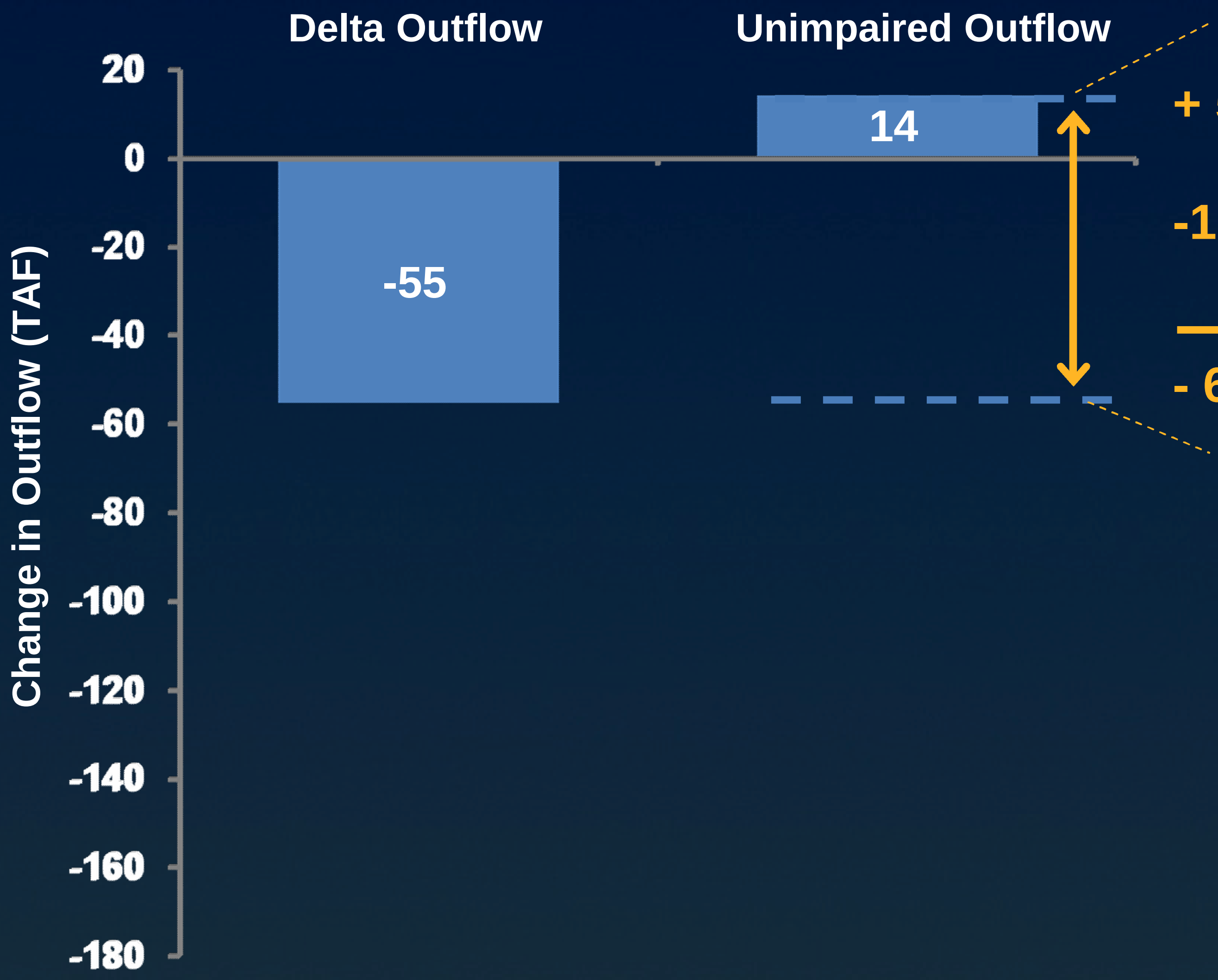
- 4 SWP-CVP Operations

-68 Water Management Activities by other parties

-72

Data Sources: DAYFLOW, CDEC, DWR Bay-Delta Office

October differences between 1990s & 2000s



+ 57 **SWP-CVP Operations**

-126 **Water Management Activities by other parties**

- 69

Data Sources: DAYFLOW, CDEC, DWR Bay-Delta Office

“Natural Flow” Approach is Uncertain In Highly Altered System

“Can reestablishing the natural flow regime serve as a useful management and restoration goal? We believe that it can, although to varying degrees, depending on the present extent of human intervention and flow alteration affecting a particular river.”

Poff et al. (1997)

"Natural Flow" Approach is Uncertain In Highly Altered System

The advice from aquatic ecologists on environmental flows might be regarded at this point in time *"as largely untested hypotheses about the flows that aquatic organisms need and how rivers function in relation to flow regime."*

Bunn and Arthington (2002)

Fluvial and Estuarine Systems Differ

Characteristic	Rivers	Estuaries
Body of pertinent literature	**Large**	**Small**
Understanding of flow effects	**Limited**	**Very limited**
Biota	**Limited diversity**	**More diverse**
Ecological interactions	**Less complicated**	**Much more complicated**
Water masses	Fresh only	Fresh and salt
Flow direction	Unidirectional	Reversing
Antecedent effects	Moderate	Potentially very important
Pollutant flushing	Rainfall runoff	Rainfall runoff and tidal flows
Water Quality changes	Downstream of source	Both upstream and downstream of source
Depth determined in	Flow	Primarily tides
Flow cross section determined by:	Sedimentary regime	Sedimentary regime, flocculation, littoral drift
Nutrient levels	Richer	Poorer

Source: Adapted from Pierson et al. 2002

Fluvial and Estuarine Systems Differ

Characteristic	Rivers	Estuaries
Body of pertinent literature	**Large**	**Small**
Understanding of flow effects	**Limited**	**Very limited**
Biota	**Limited diversity**	**More diverse**
Ecological interactions	**Less complicated**	**Much more complicated**
Water masses	**Fresh only**	**Fresh and salt**
Flow direction	**Unidirectional**	**Reversing**
Antecedent effects	**Moderate**	**Potentially very important**
Pollutant flushing	Rainfall runoff	Rainfall runoff and tidal flows
Water Quality changes	Downstream of source	Both upstream and downstream of source
Depth determined in	Flow	Primarily tides
Flow cross section determined by:	Sedimentary regime	Sedimentary regime, flocculation, littoral drift
Nutrient levels	Richer	Poorer

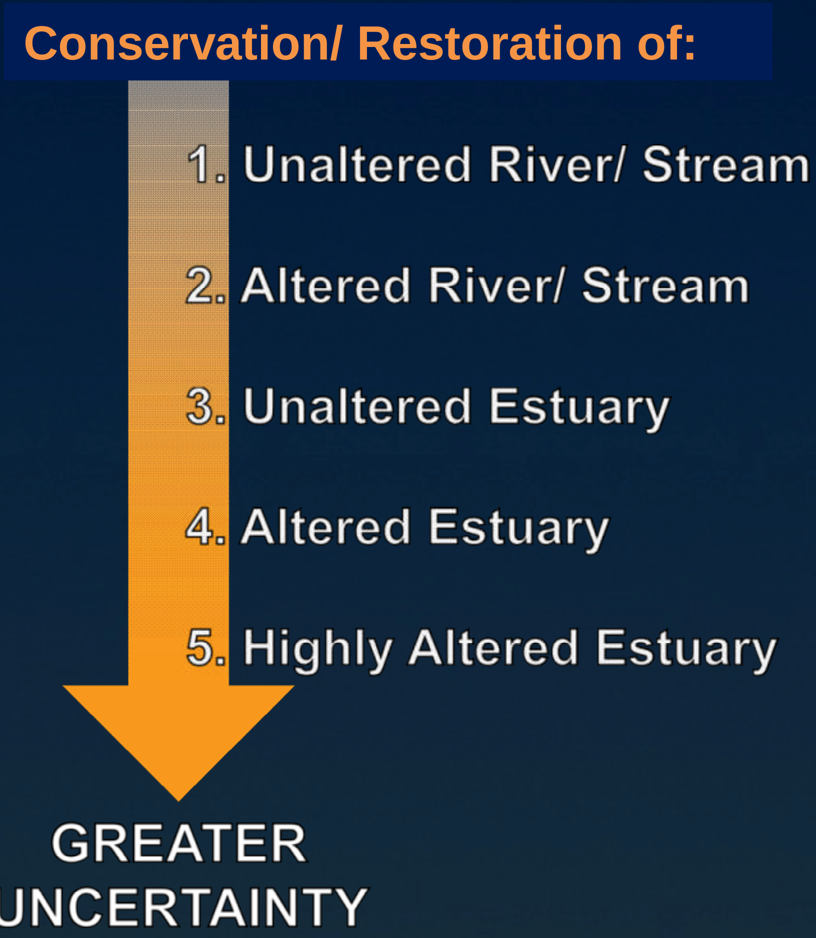
Source: Adapted from Pierson et al. 2002

Fluvial and Estuarine Systems Differ

Characteristic	Rivers	Estuaries
Body of pertinent literature	Large	Small
Understanding of flow effects	Limited	Very limited

Source: Adapted from Pierson et al. 2002

“Natural Flow” Approach is Uncertain In Highly Altered System



Conservation/ Restoration of:

1. Unaltered River/ Stream
2. Altered River/ Stream
3. Unaltered Estuary
4. Altered Estuary
5. Highly Altered Estuary

GREATER
UNCERTAINTY

Uncertainty in the Bay Delta System

Changes in reservoir releases cannot:

- Restore habitat complexity
- Supply depositional materials
- Restore widespread seasonal floodplain inundation
- Restore natural nutrient balance
- Decrease Delta water temperature

Proposed management of the LSZ at 60 – 74 km

Without citation, and unsupported / inconclusive within the scientific literature:

- No citation provided
- No correlation between flow and phytoplankton in Suisun Complex after the clam (Alpine/Jassby 1992, Kimmerer 2002)
- No correlation between X2 and Delta smelt abundance or summer distribution (Kimmerer 2002, Nobriga et al, 2008)
- Delta smelt distribution shifted to more northerly and fresher location; Cache Slough Complex (Sommer et al 2011)

Proposed management of the LSZ at 60 – 74 km

USEPA conceded:

"There are large scale declines over time in the abundance of species, especially pelagic species, but there is not good information, and a wider range of opinion, on the cause(s) / mechanisms leading to these declines. The role of the LSZ in these abundance declines is uncertain."

USEPA Workshop Summary: Technical Workshop on Estuarine Habitat in the Bay Delta Estuary (2012)

Proposed management of the LSZ at 60 – 74 km

Preliminary results in Brown et al., stated:

“Many of the predictions either could not be evaluated with the data available or the needed data are not being collected. Most of the predictions that could be addressed involved either the abiotic habitat components (i.e., the physical environment) or delta smelt responses. ***In general, the FLaSH investigation has been largely inconclusive as of the writing of this report.”***

Brown et al (2012)

Proposed management of the LSZ at 60 – 74 km

Asserted with no exploration of the biological mechanisms underlying correlations.

Jassby cautioned:

***"By ignoring variables other than X2 (or Q_{out}) we could therefore be in danger of imposing inappropriate standards**, either too stringent or too lenient. The mere fact of a correlation between some ecosystem property and an indicator such as X, is therefore not sufficient grounds for using the indicator as a policy variable."*

Jassby 1995

Proposed management of the LSZ at 60 – 74 km

Asserted with no exploration of the biological mechanisms underlying correlations.

Kimmerer recognised:

"These relationships to flow may be due to several potential mechanisms, each with its own locus and period of effectiveness, but no mechanism has been conclusively shown to underlie the flow relationship of any species."

kimmerer 2002

Disconnect between the LSZ and food and turbidity

Uncertain or inconclusive or sometimes contradicted in the scientific literature

- **No positive correlation between flow and phytoplankton in Suisun Bay, because of the invasion of the clam and nutrient imbalance (Alpine/Cloern 1992, Kimmerer 2002).**

- **The FLaSH studies reported lower phytoplankton in Suisun and higher outside Suisun, AND delta smelt growth was not related to salinity (FLaSH 2012).**

- **Potential food supply in Suisun Marsh, therefore recommended restoring marsh habitat (Muller et al 2002)**

Based on our review of the available science:

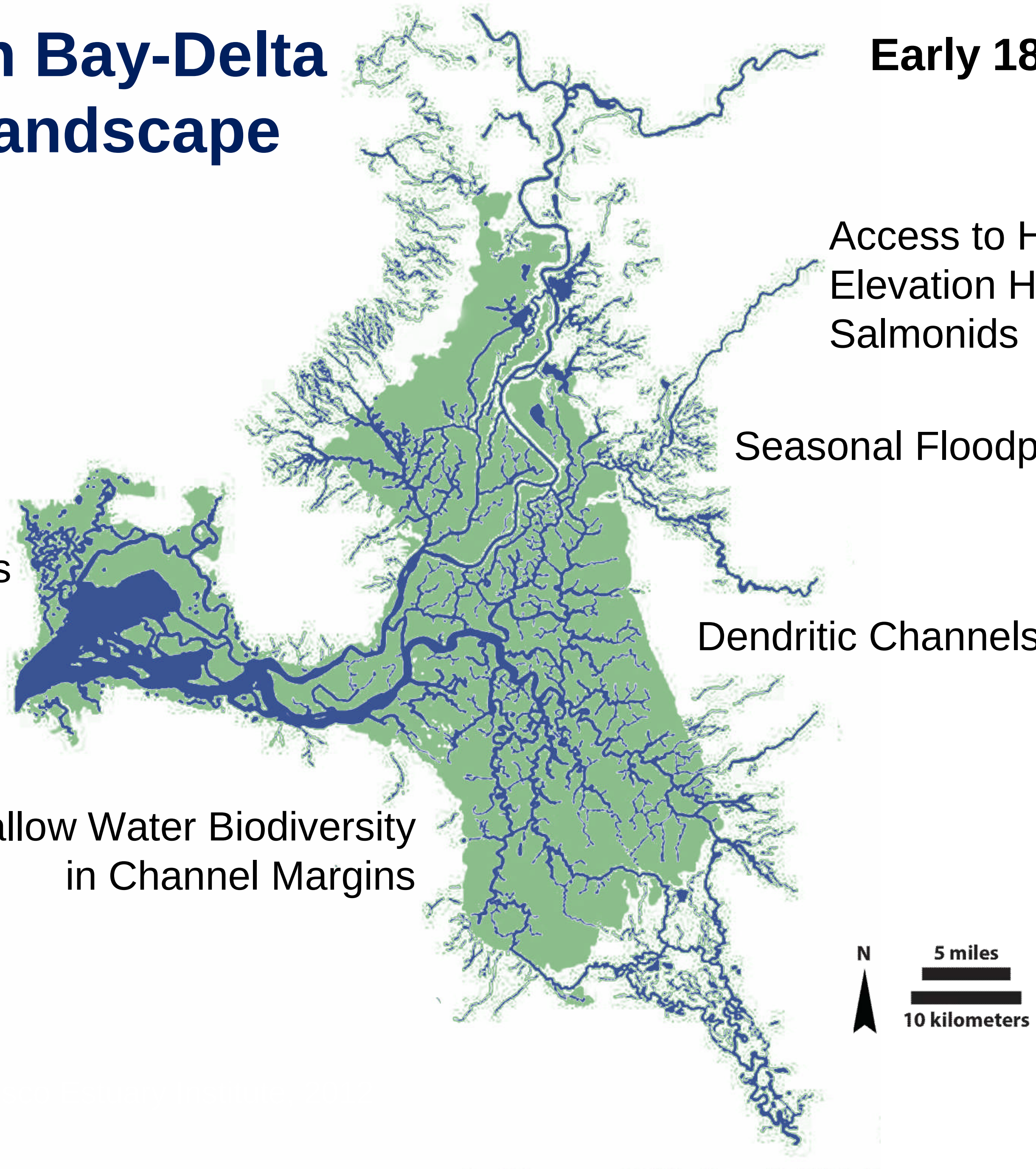
Given the highly altered state of the Bay-Delta estuary, it is highly uncertain that mimicking "natural" flows would restore biological functions.

Based on our review of the available science:

Large changes in flow made under scientific uncertainty could lead to large adverse impacts to beneficial uses.

Changes in Bay-Delta Physical Landscape Over Time

Early 1800’s



Access to High Elevation Habitat for Salmonids

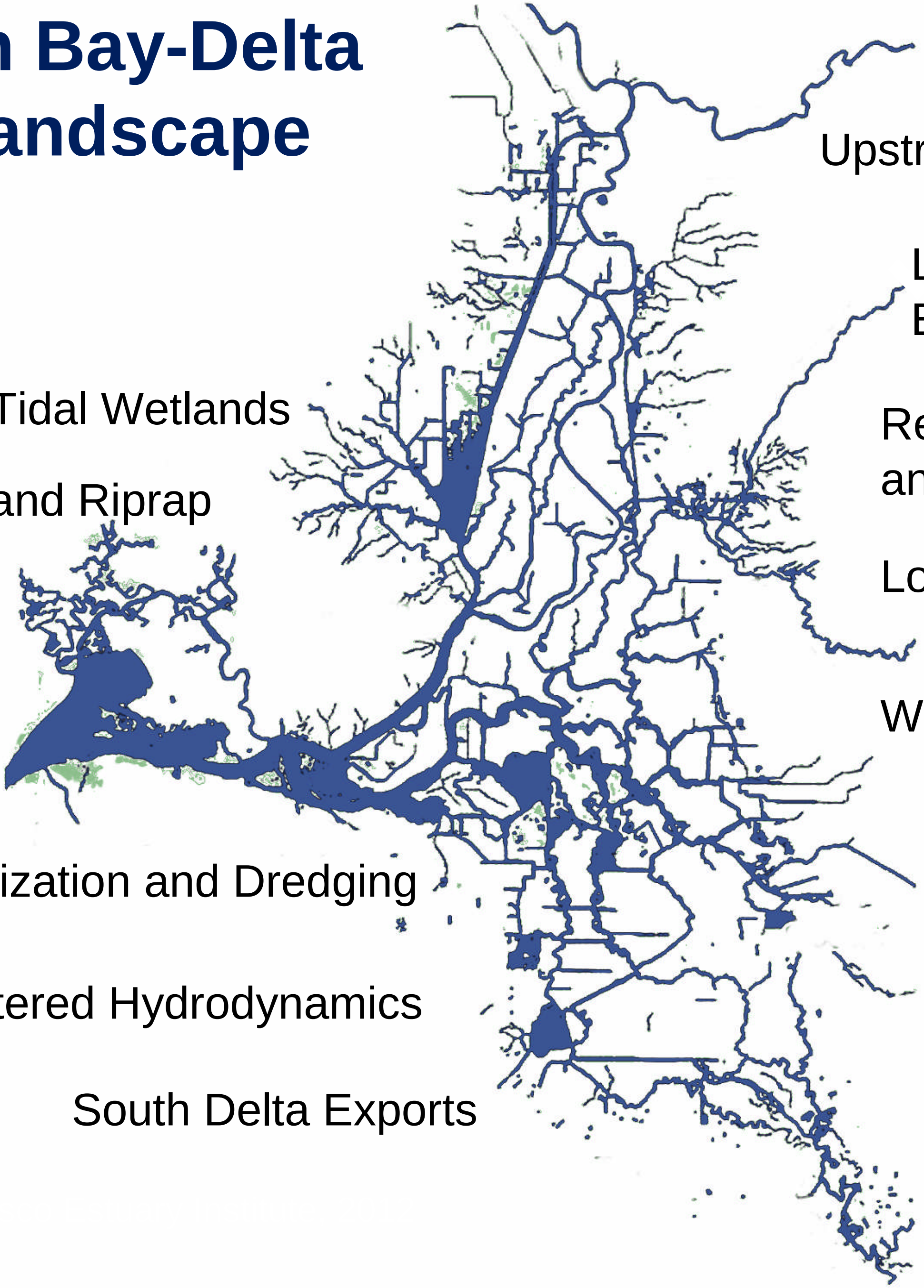
Seasonal Floodplains

Tidal Wetlands

Dendritic Channels

Shallow Water Biodiversity in Channel Margins

Changes in Bay-Delta Physical Landscape Over Time



Late 1990's

Upstream Water Diversions

Lost Access to High Elevation Habitat

Reduced Sediment and Gravel Input

Lost Floodplain Access

Within Delta Diversions

Loss of Tidal Wetlands

Levees and Riprap

Channelization and Dredging

Altered Hydrodynamics

South Delta Exports

Resultant Changes to Ecosystem Functions

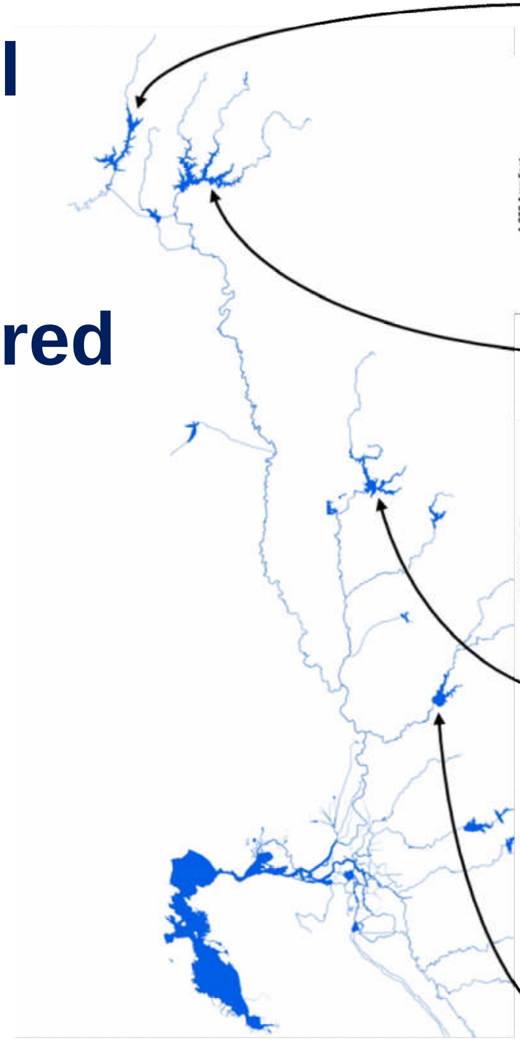
- Loss of wetland habitat
- Loss of access to floodplains
- Loss of shallow-water channel margin habitat
- Reduced production of organic matter and food

Potential Impacts of Flow Changes

In a highly altered system…

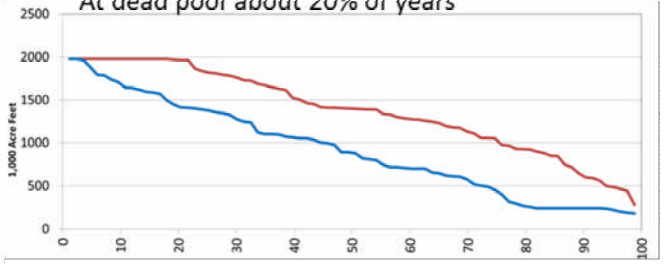
- Unimpaired flow could result in adverse impacts
- Increased winter-spring flow may provide uncertain benefits for some species and adversely impact others, such as Salmonids

Potential Impact of 50% Unimpaired Flow



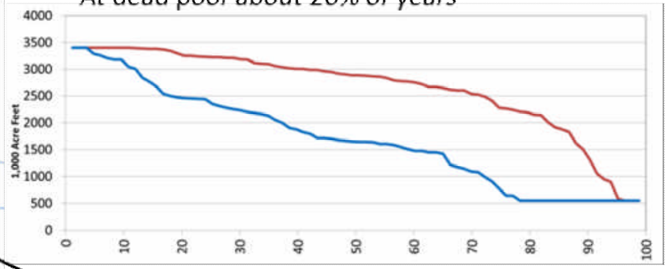
Trinity Reservoir

Average change in carryover = -460 TAF
At dead pool about 20% of years



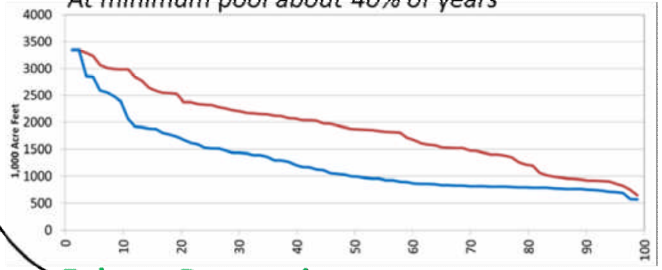
Shasta Reservoir

Average change in carryover = -960 TAF
At dead pool about 20% of years



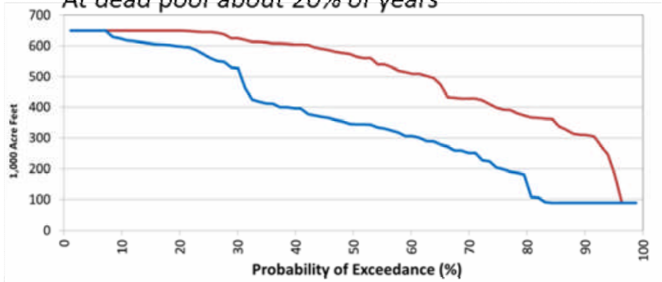
Oroville Reservoir

Average change in carryover = -620 TAF
At minimum pool about 40% of years



Folsom Reservoir

Average change in carryover = -150 TAF
At dead pool about 20% of years



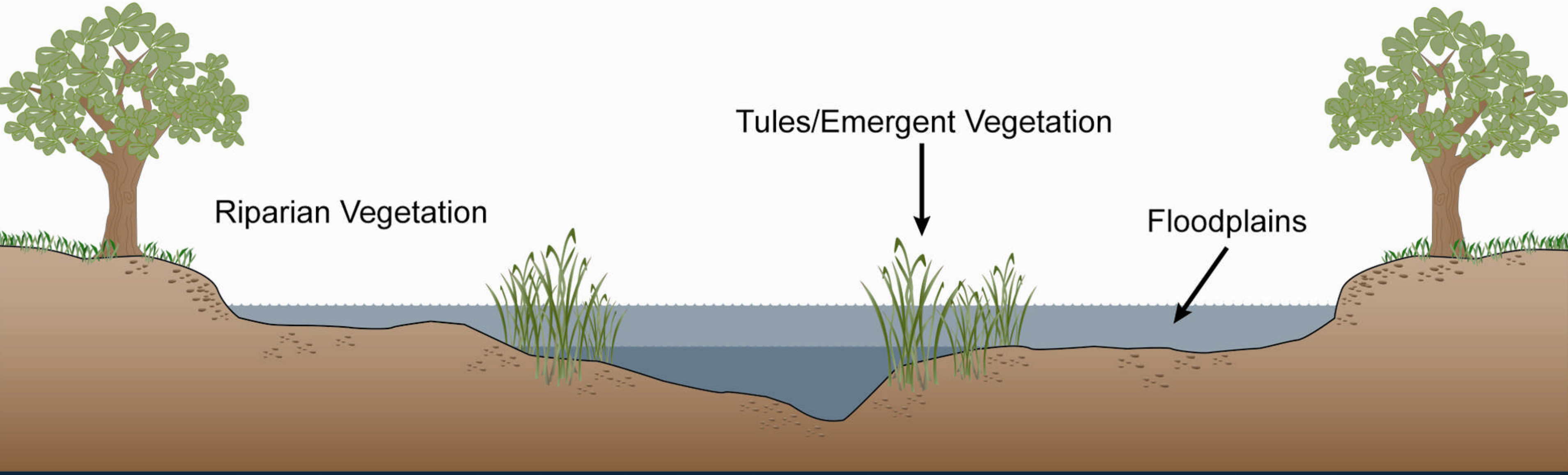
Source: MBK 2012

Effect of Rim Dams on Salmonids

- Dams block access to higher elevation habitat
  - Spawning and rearing occurs in downstream reaches
  - Exposure of eggs to high temperatures results in mortality
- Requires balance between flow and coldwater pool management
- Maintaining suitable temperature is challenging in dry years
- Increased winter-spring flows may reduce coldwater pool storage in the summer
  - Adverse impacts winter-run Chinook salmon

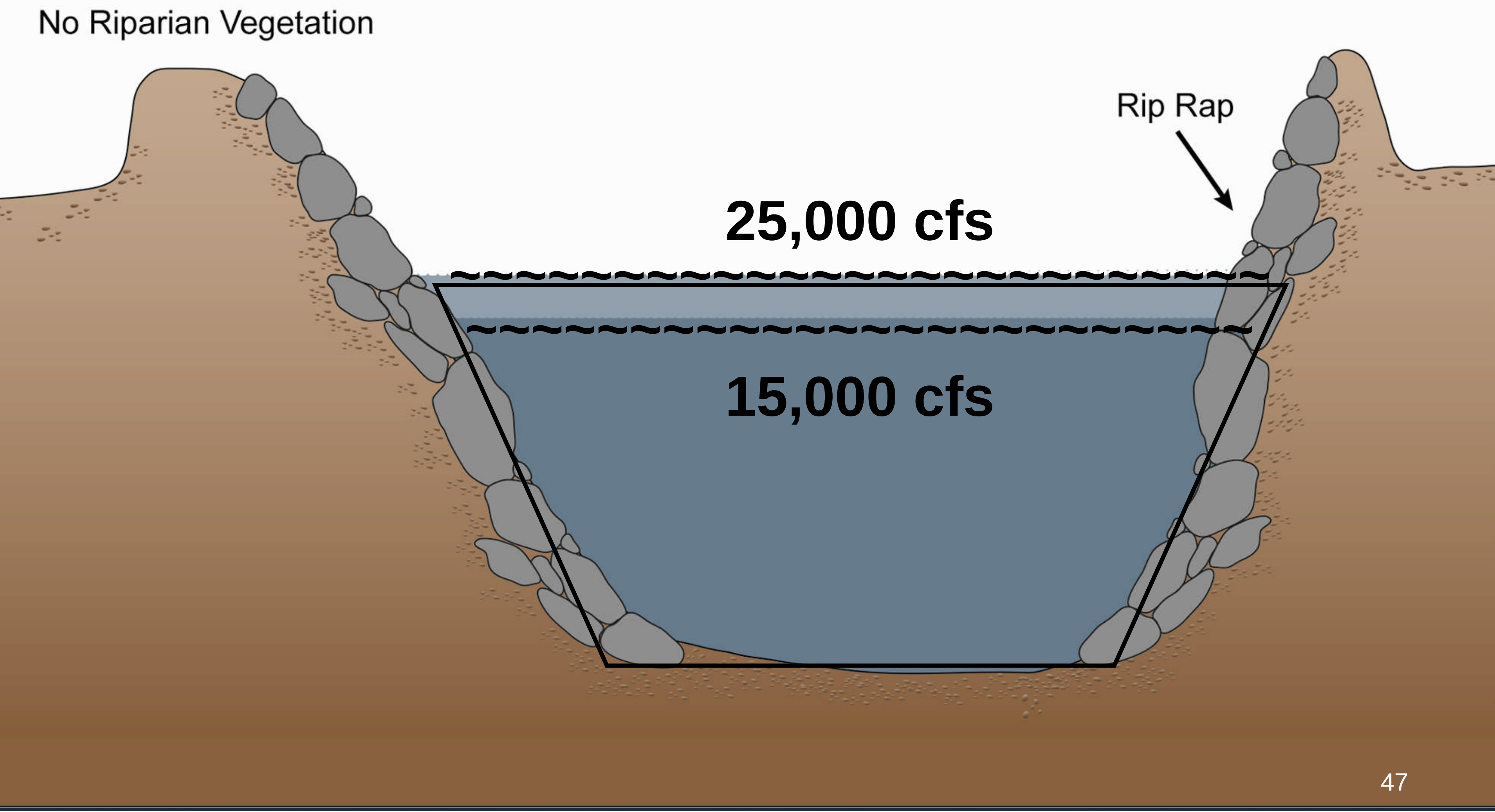
Sacramento River Cross-Section

HISTORIC



Sacramento River Cross-Section

CURRENT



Improving Aquatic Species Functions

- Tidal wetlands
  - Cover, rearing, food production
- Seasonal floodplains
  - Spawning, rearing and connectivity
- Shallow water low velocity channel margin
  - Food production

Dealing with Uncertainty in Restoration

- Based on current research (at Liberty Island & Northwest), habitat design should be:
  - Based on suitability and natural functions/processes
  - Compatible with tidal and river hydrodynamics, water quality, and natural processes such as sediment resuspension (turbidity)
  - Promote complexity such as depth, tidal currents, emergent vegetation
  - Dispersed to support various species and functions
  - Facilitate adaptive management
- Requires multidisciplinary collaborative monitoring

Outflow and Invasive Species

- Recent study suggest without support that water exports have produced “waves of invertebrate invasions”
- Exotic species have changed the ecological community
- Winder et al. (2011) cite prolonged drought and increased salinity intrusion as dominant factors for non-native invertebrate colonization
- Water operations also maintain Delta outflow and control salinity
- In dry years, there are dynamic interactions between salinity intrusion and water project operations
- The potential effect of water project operation on colonization by invasive species <u>has not been analyzed</u> and is an untested hypothesis

Key Points

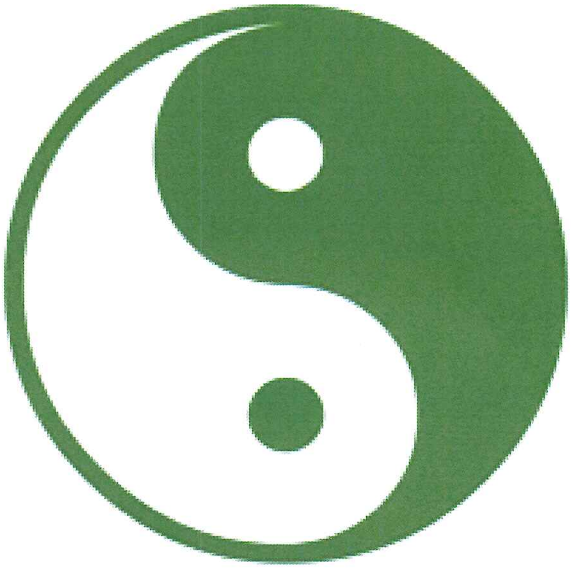
- The SWB should seek to understand the physical, chemical and biological changes that have occurred in the Bay Delta Estuary
- The SWB should endeavor to understand the underlying mechanisms stressing or the functions that flow serves in the Bay Delta Estuary, before considering whether to dedicate more water for environmental purposes
- Scientific literature shows habitat restoration and nutrient regulation could produce meaningful, positive changes to the Bay-Delta Estuary

Nutrients and the food web

Patricia Glibert
University of Maryland Center
for Environmental Science

“Ammonium is a paradoxical nutrient”

Ammonium is a preferred form of nitrogen for phytoplankton under some conditions



Ammonium can be inhibitory or even toxic under some conditions and to some species

“Sensitivity to NH_4^+ may be a universal biological phenomenon”

Britto and Kronzucker 2002

There is no controversy or uncertainty regarding the effects of ammonium on plants (including algae) and animals (including humans)

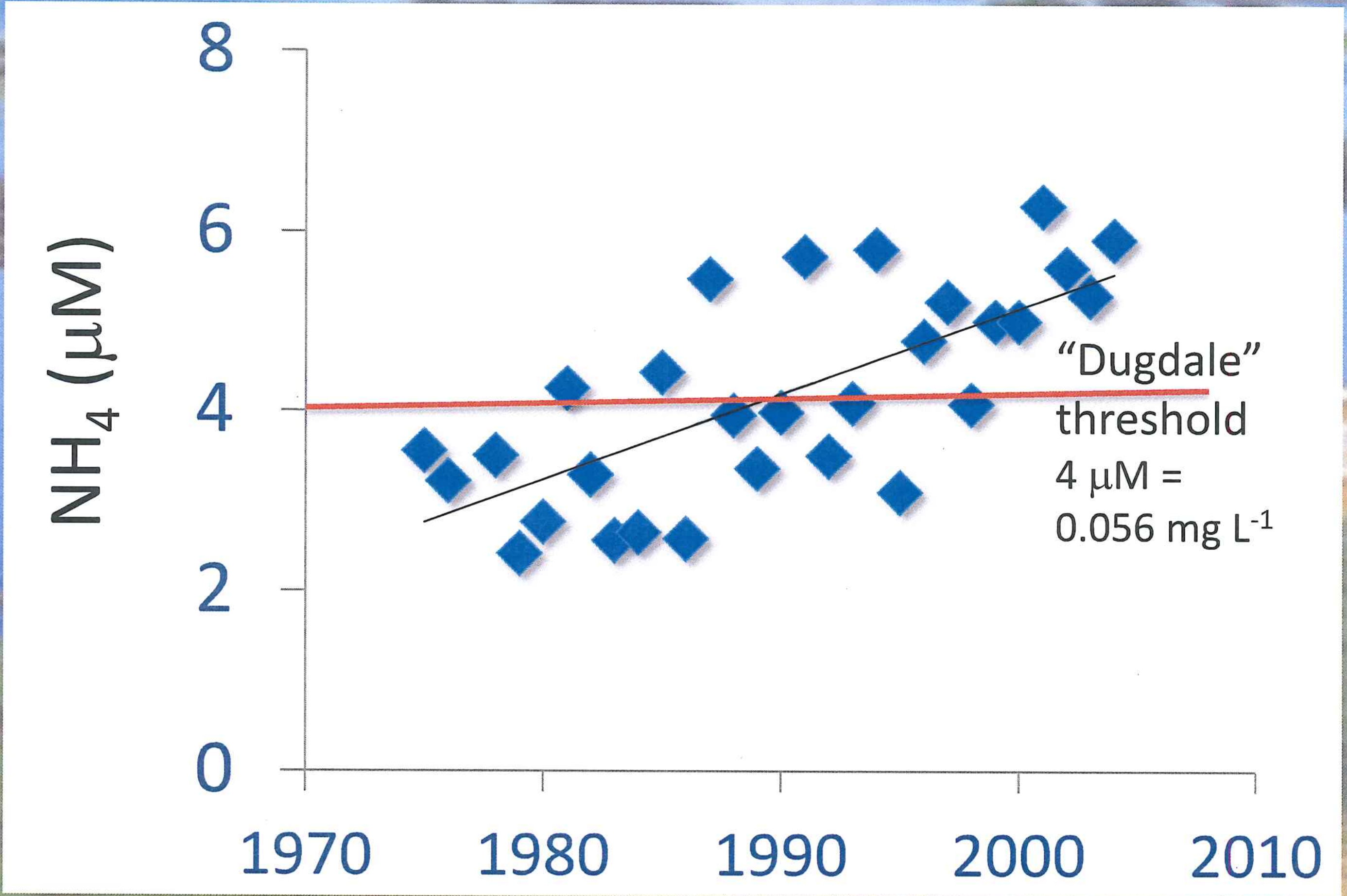
> "... but the threshold at which symptoms of toxicity become manifested differs..."

Concentrations at which ammonium has been shown to inhibit the uptake of nitrate by phytoplankton range from 1 μM or less to >20 μm

> "the extent and threshold concentrations involved depend on the species under study, its physiological status, and the environmental conditions to which this particular species or the natural assemblage has been exposed"

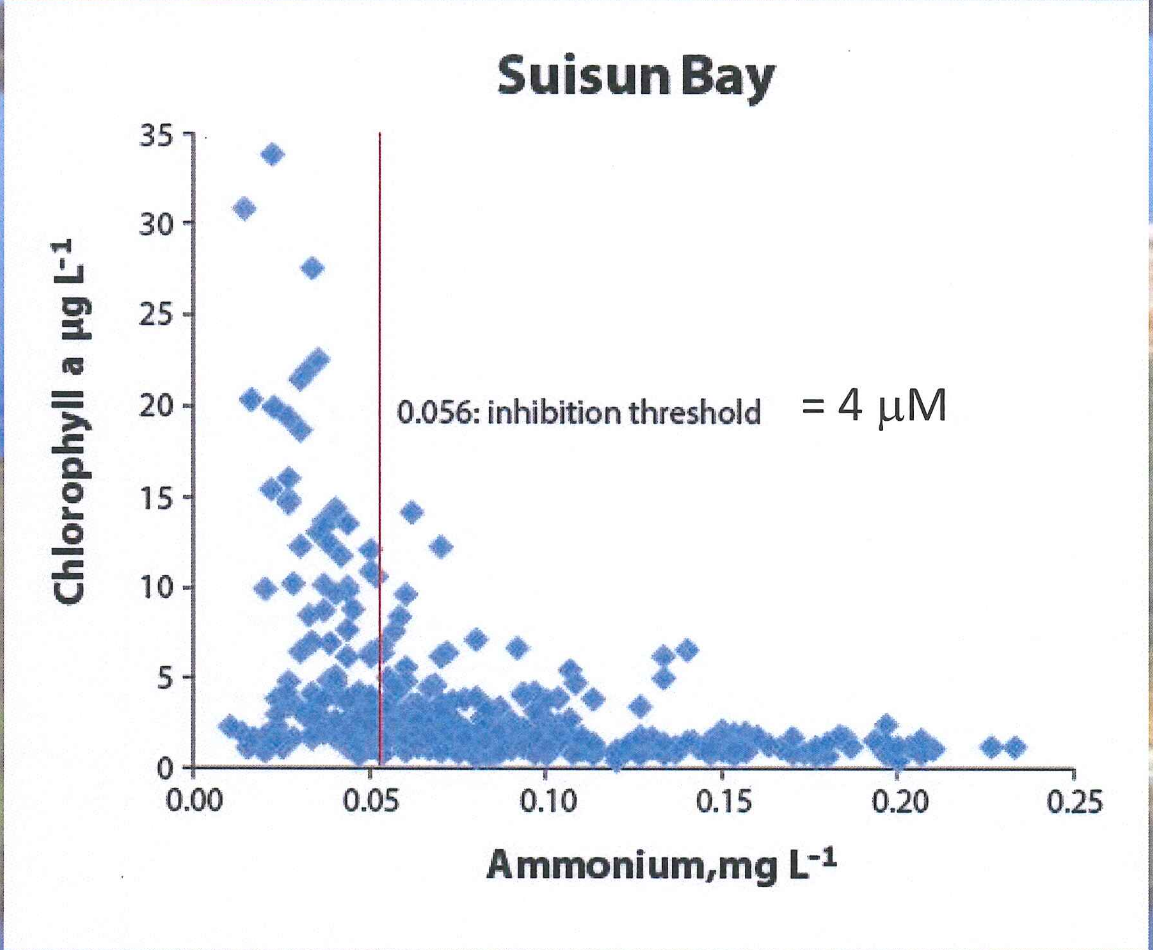
(Varela and Harrison 1999)

Ammonium in San Francisco Bay Delta: increasing over time

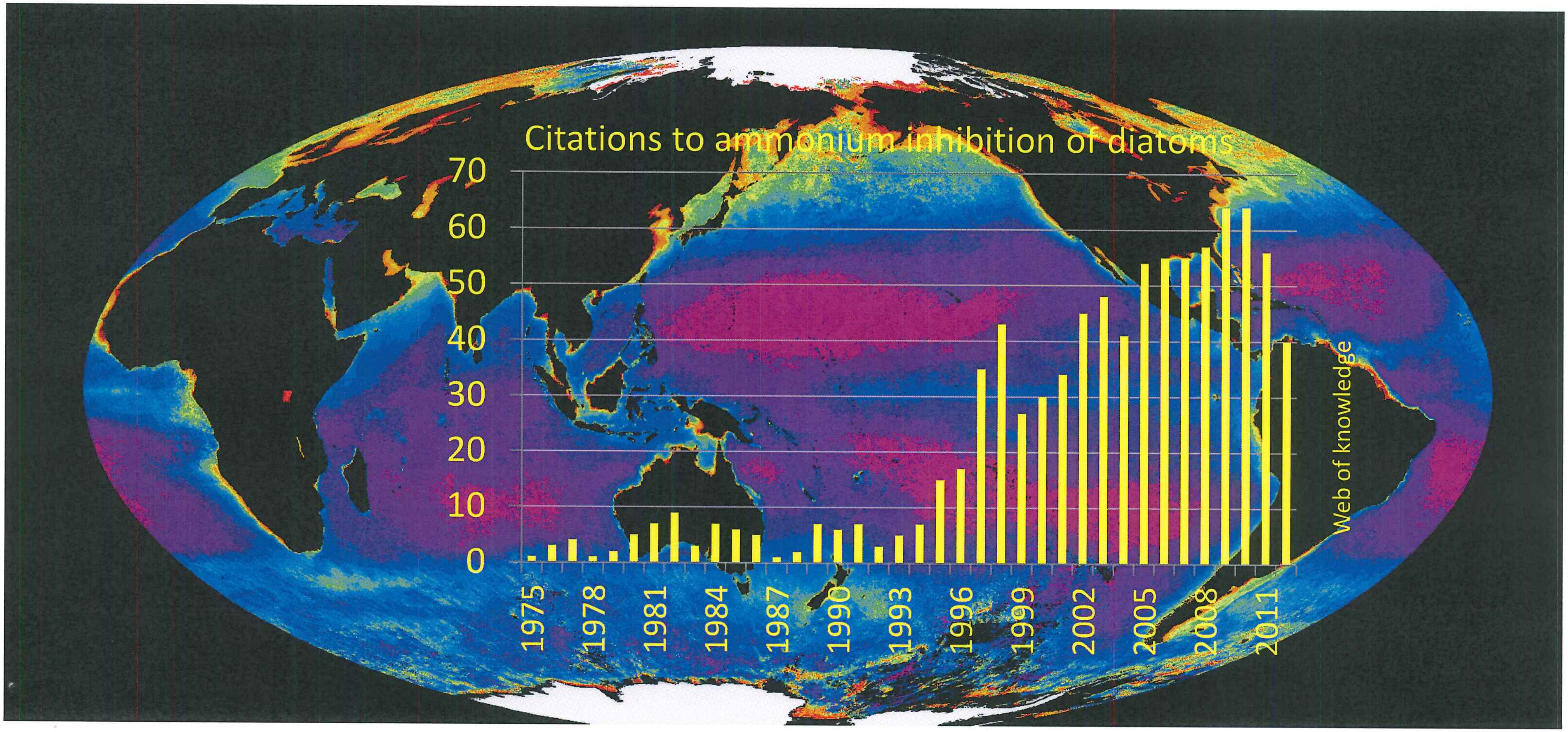


Glibert et al., 2011

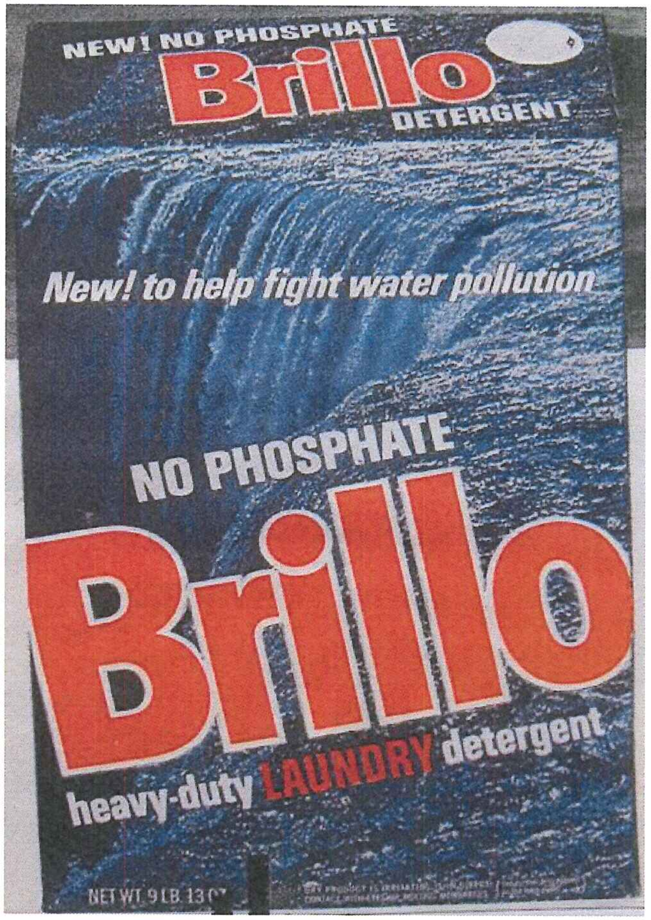
Overall, long-term ammonium data show strong negative relationship with chlorophyll in the water column



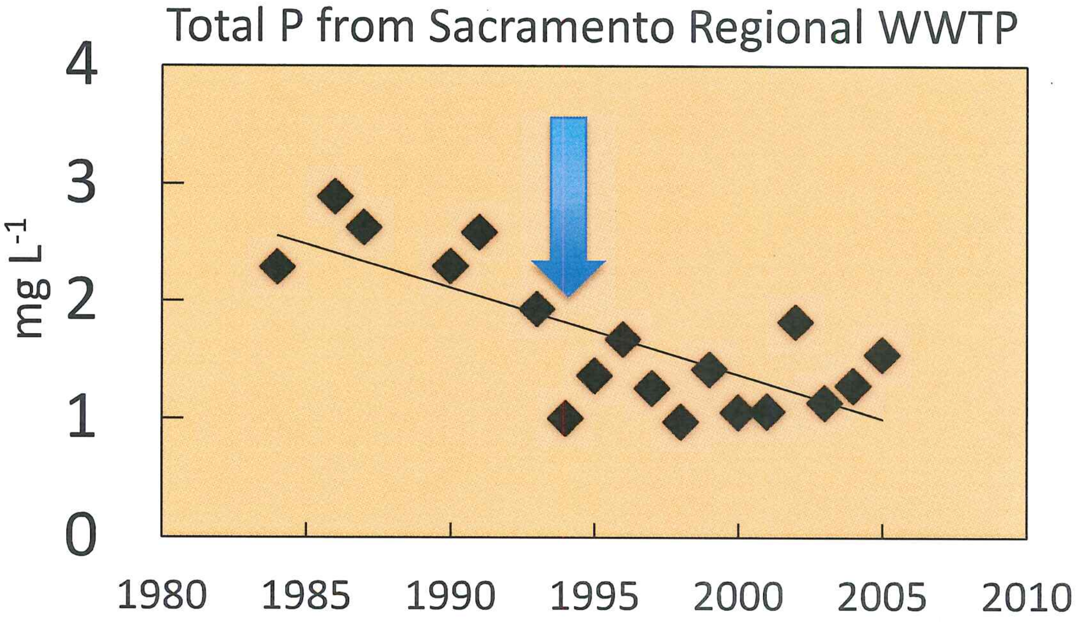
Inhibition of diatoms by ammonium is being increasingly recognized – and included in ecosystem models of ocean productivity



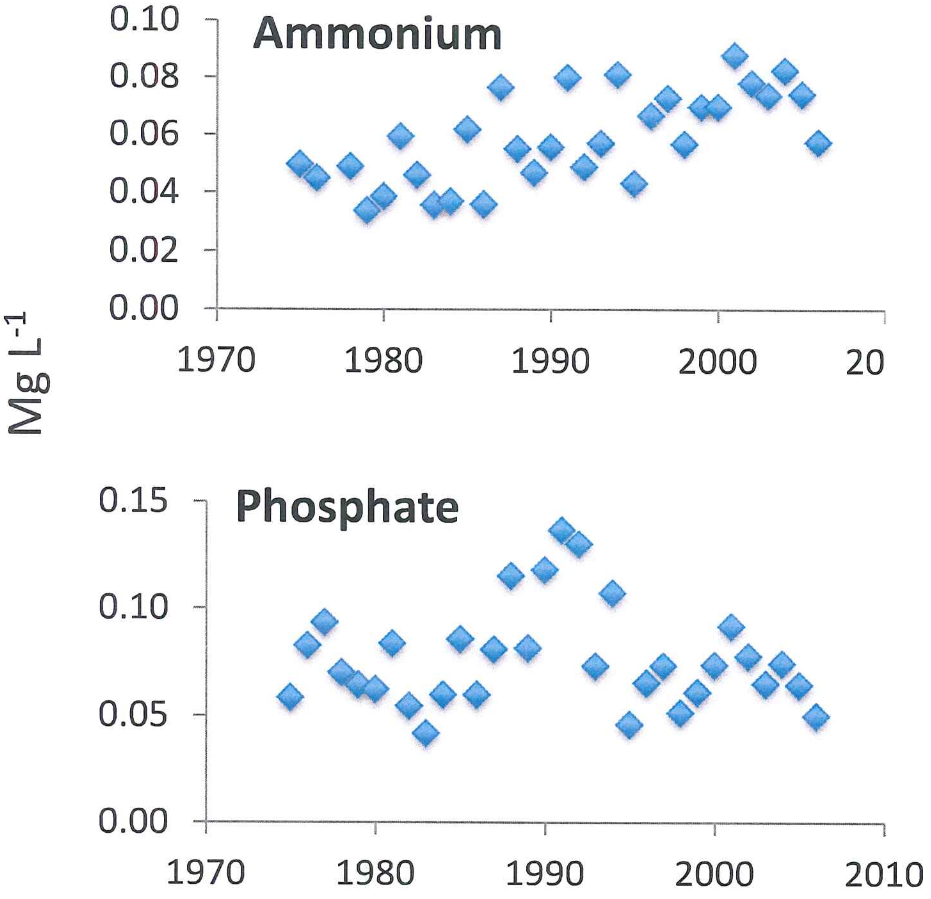
Ammonium is not the only nutrient that has changed

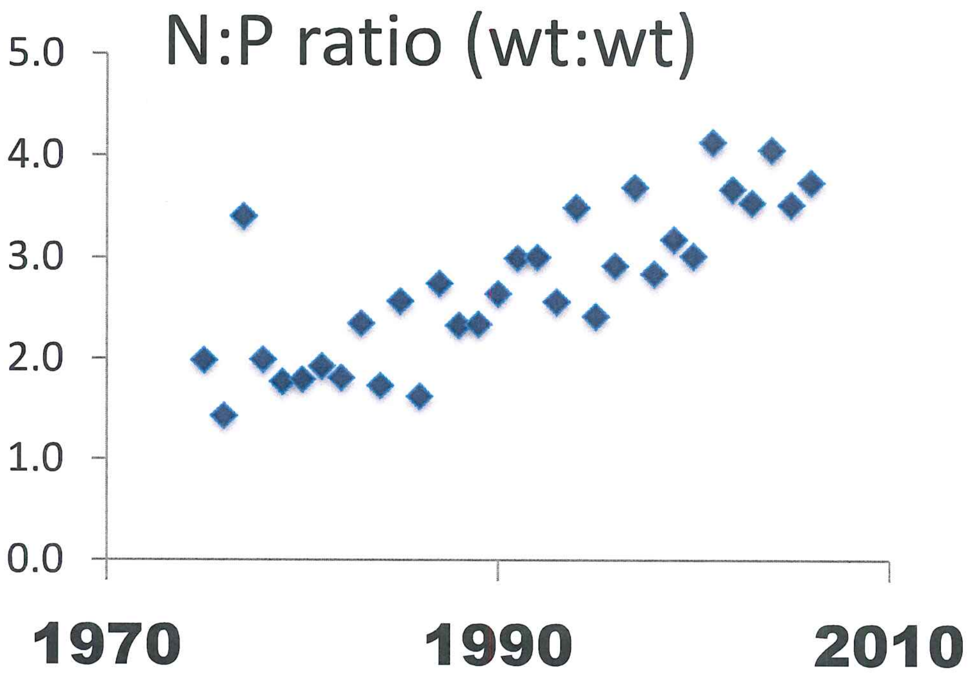


By the mid 1990s the switch to non-P detergents had occurred throughout the US and Europe. The manufacturers made the switch regardless of state (or country) laws to avoid region-by-region formulations



Nitrogen: Phosphorus ratios have increased

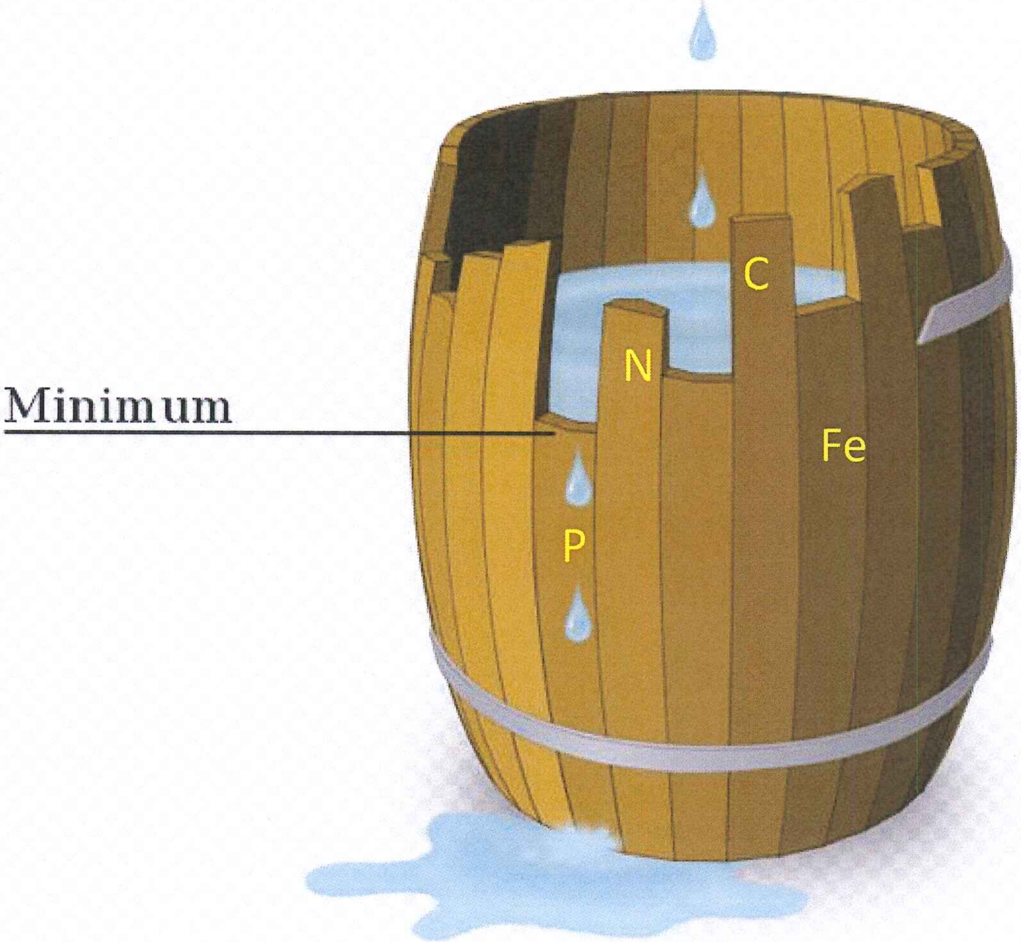




Liebig' Law of the minimum

Growth is limited by the nutrient that is in shortest supply relative to the needs of the organism.

But- in this system the nutrients, including P, rarely are low enough to be considered "limiting" using simple, classic metrics (half-saturations constants)



Classic Dogma:

> "There should be no selective effect ... that might distinguish between the potential performance of any pair of planktonic algae, so long as the resource concentrations are able to saturate the growth demand..." Reynolds 1999

i.e., if nutrients are 'sufficient" they should not regulate the species composition of the algae

Contemporary Perspective:

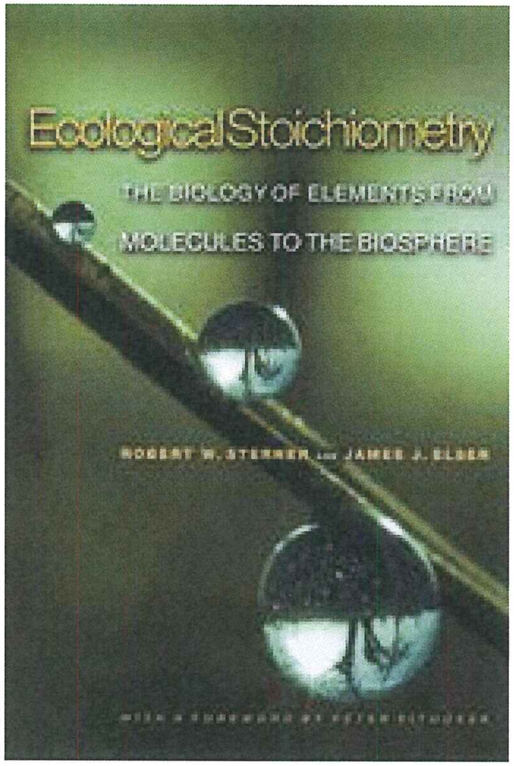
While total nutrient load sets the total amount of productivity/biomass of an ecosystem,

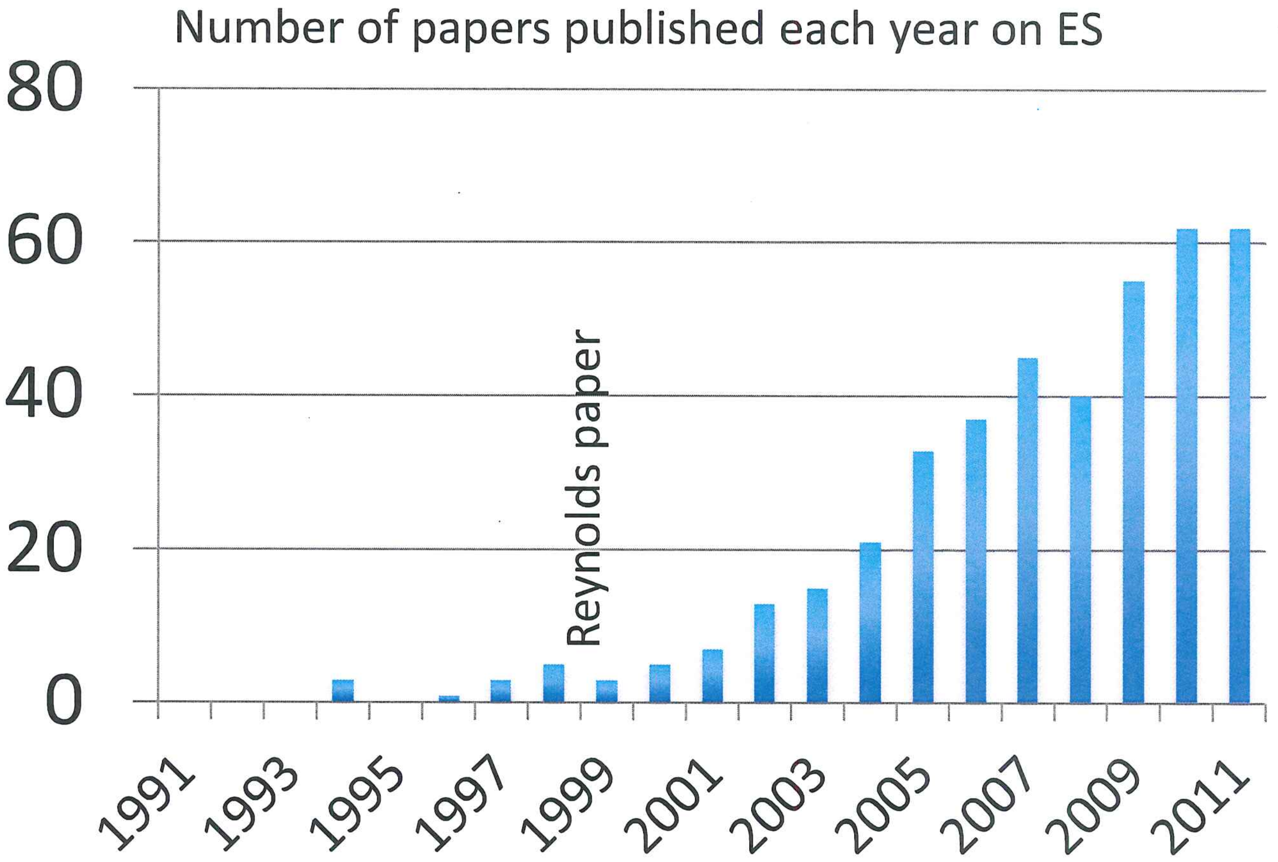
The relative proportions of nutrients sets the QUALITY

(who is there and how they are doing)

- The balance of chemical resources and their interactions;
- Match/mis-match between organismal requirement for nutrients and their availability

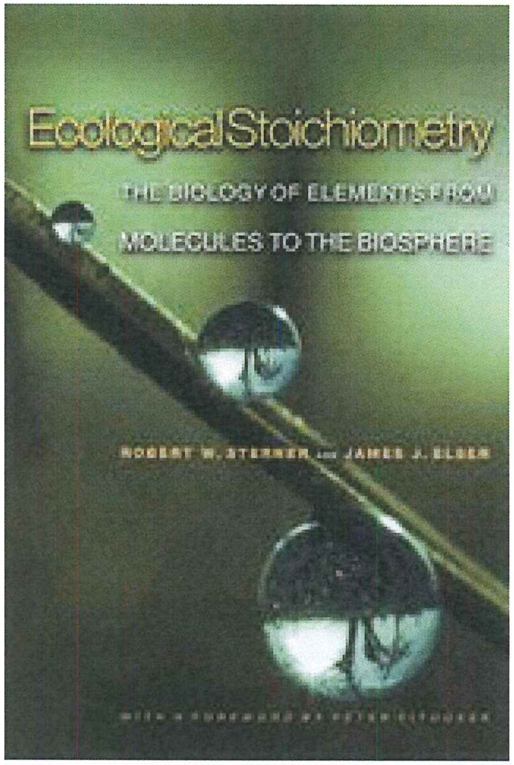
This concept formalized as Ecological Stoichiometry





Web of knowledge

This concept formalized as Ecological Stoichiometry

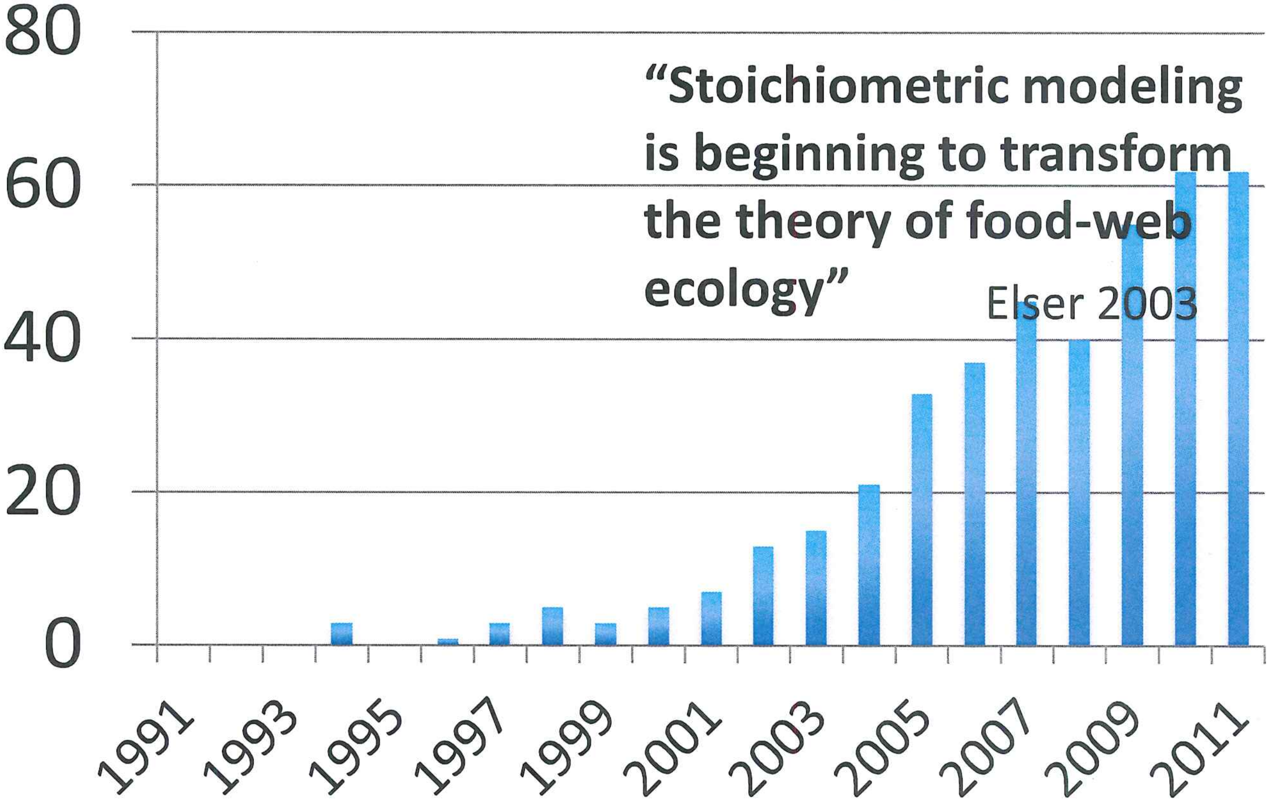


Ecological Stoichiometry

THE BIOLOGY OF ELEMENTS FROM MOLECULES TO THE BIOSPHERE

ROBERT W. STERNER and JAMES J. ELSER

Number of papers published each year on ES



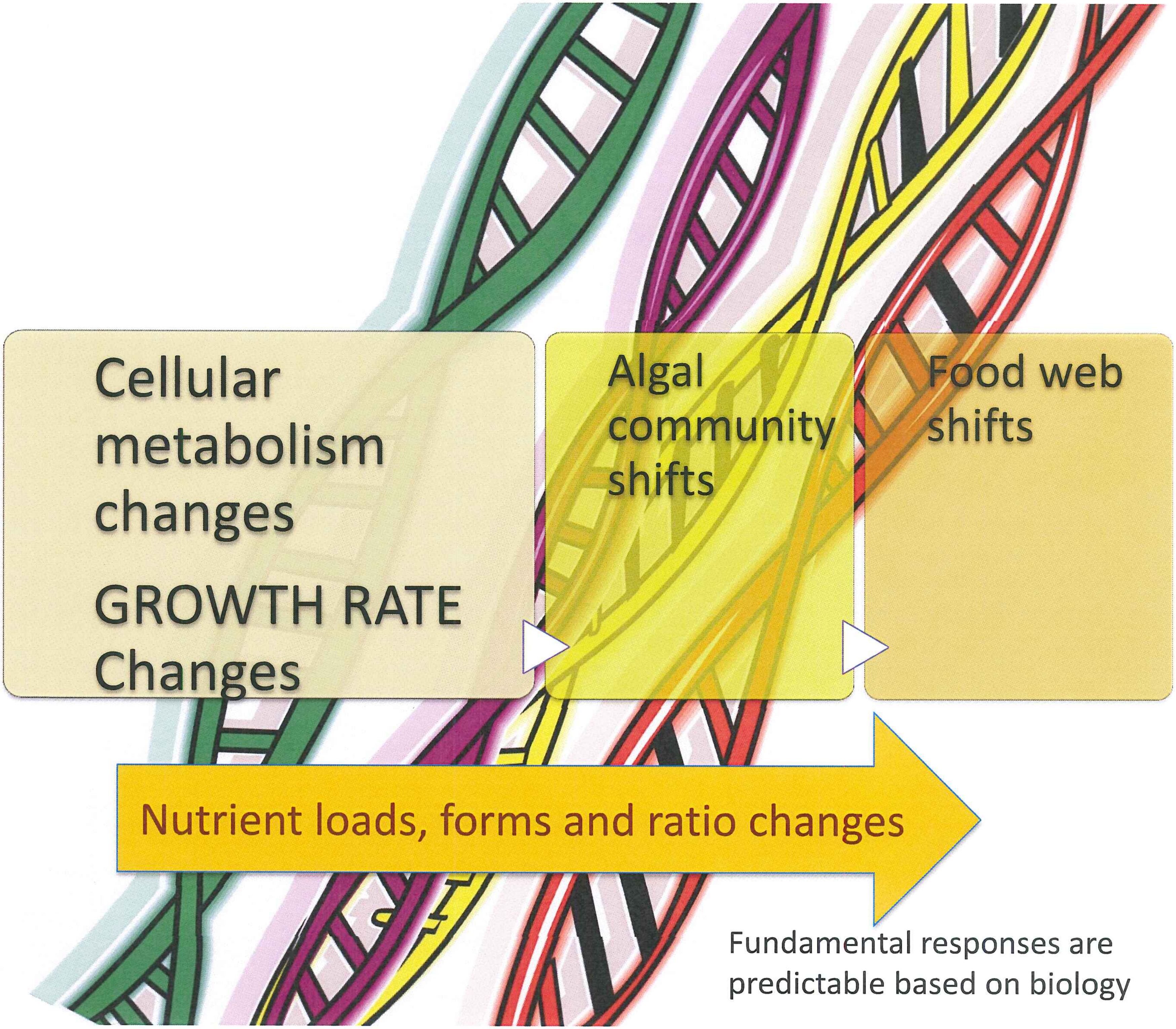
"Stoichiometric modeling is beginning to transform the theory of food-web ecology"

Elser 2003

Web of knowledge

Nutrient composition regulates organism metabolism at concentrations well above those that are limiting

- *It's a balancing act.*
- Organisms sequester what they need and dissipate the rest.
- Too much presents metabolic costs as does too little
- Organisms have different needs for, and different strategies to regulate their nutrients



Cellular metabolism changes

GROWTH RATE Changes

Algal community shifts

Food web shifts

Nutrient loads, forms and ratio changes

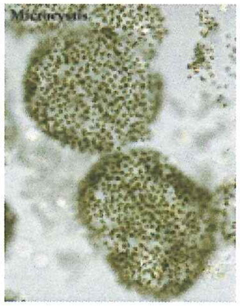
Fundamental responses are predictable based on biology

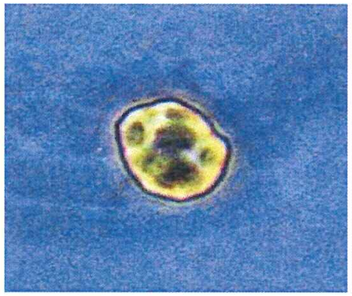
Ecological stoichiometry at the base of the Food Web

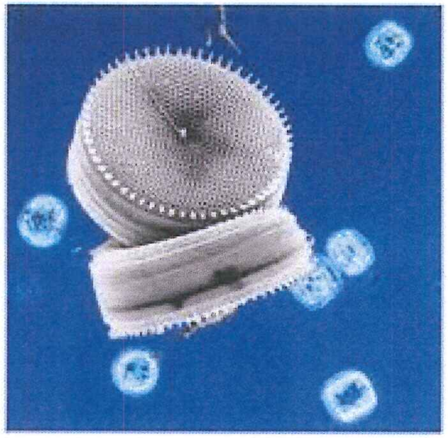
- Algae generally follow a

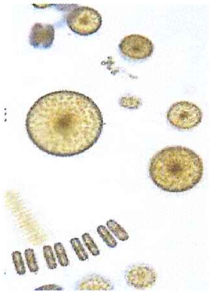
“*you are what you eat*” strategy

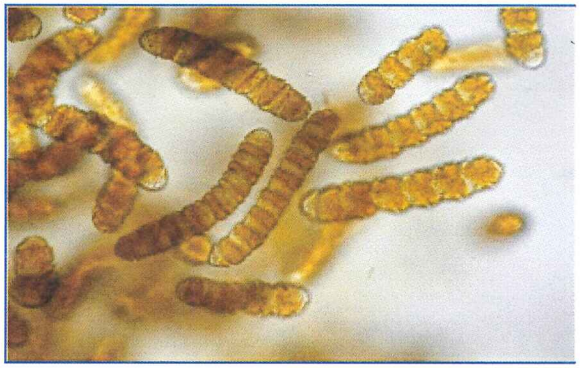
- They have limited capability to regulate their nutrient content



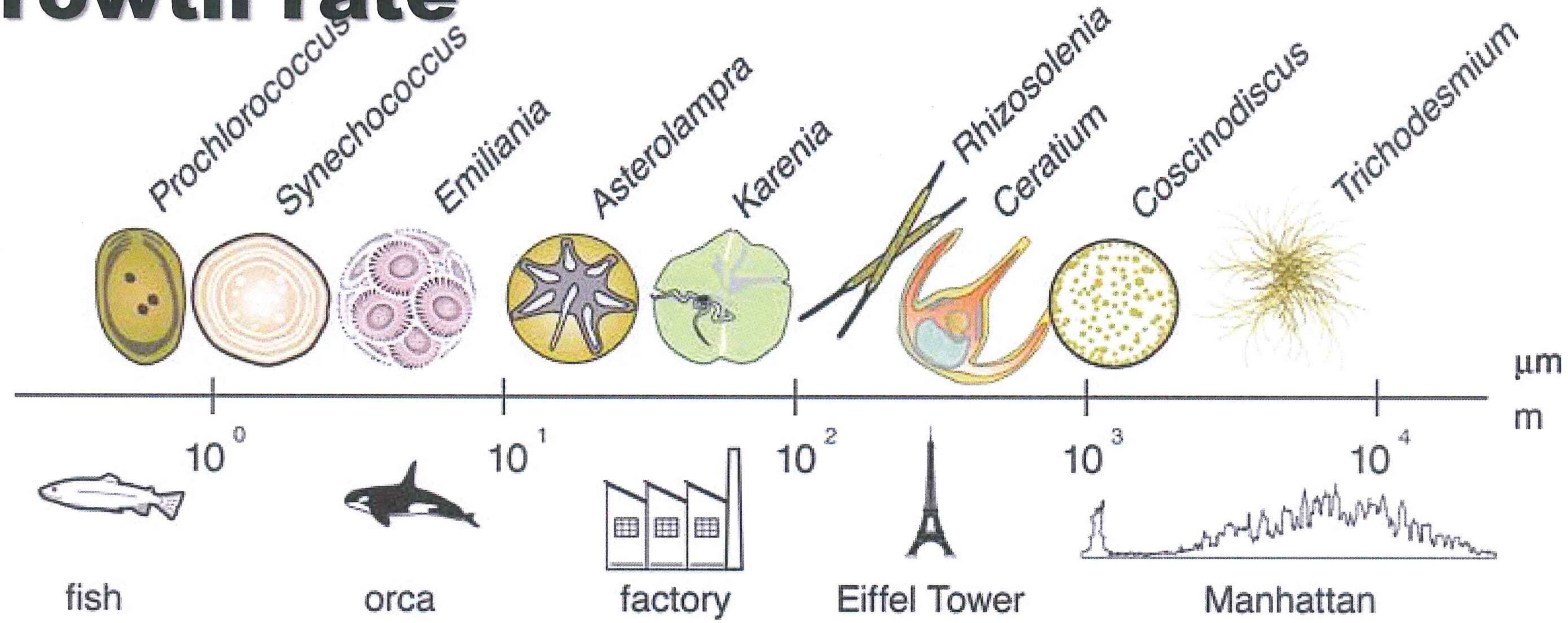






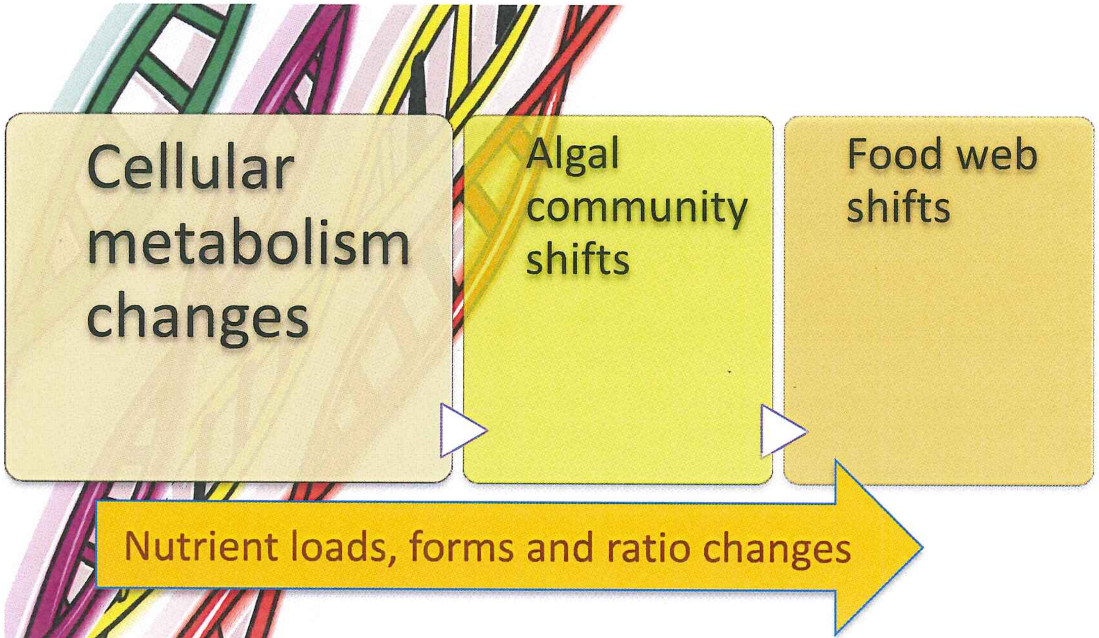


Across species there is a great diversity in size, composition and growth rate



Organism	Cell Size	C:P
Synechococcus	10^0 µm^3	~100
Diatoms	10^2- 10^3 µm^3	~50
Cryptophytes	10^1-10^4 µm^3	~60

Data and figure from Finkel et al. 2010



Cellular physiology changes across the entire spectrum of substrate availability

Cell size

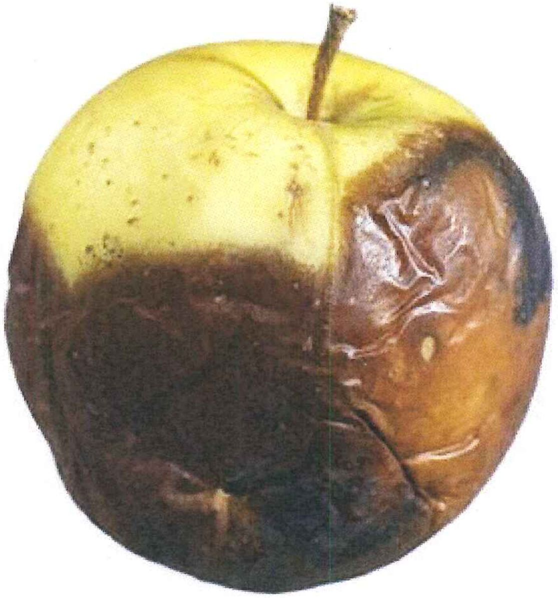
Cellular pigmentation

N:C, C:P, N:P

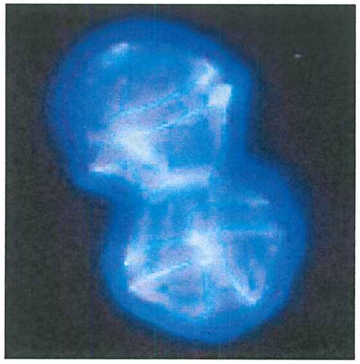
Enzyme activities

Growth rate

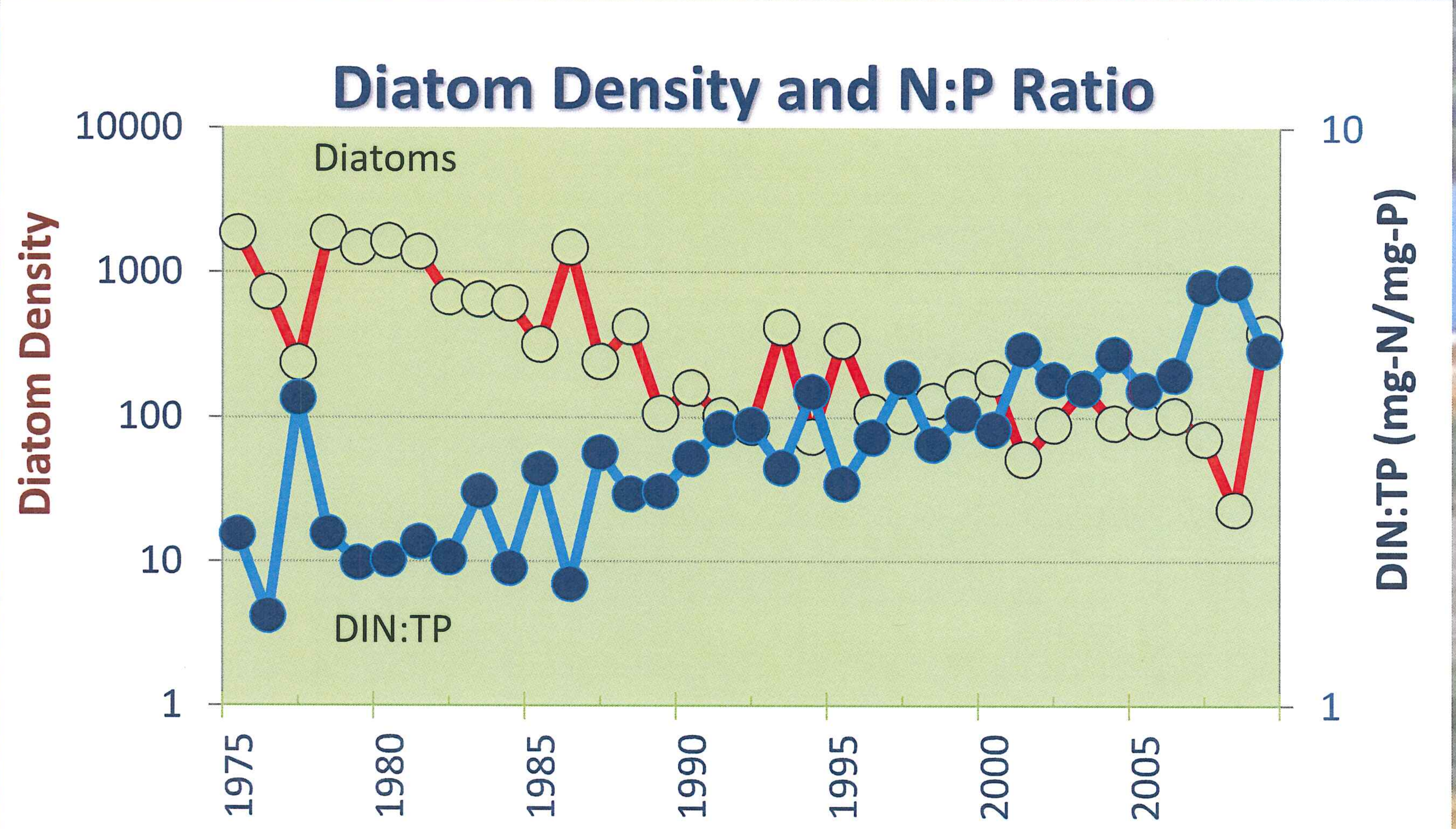
Toxin production



Even good food can go bad



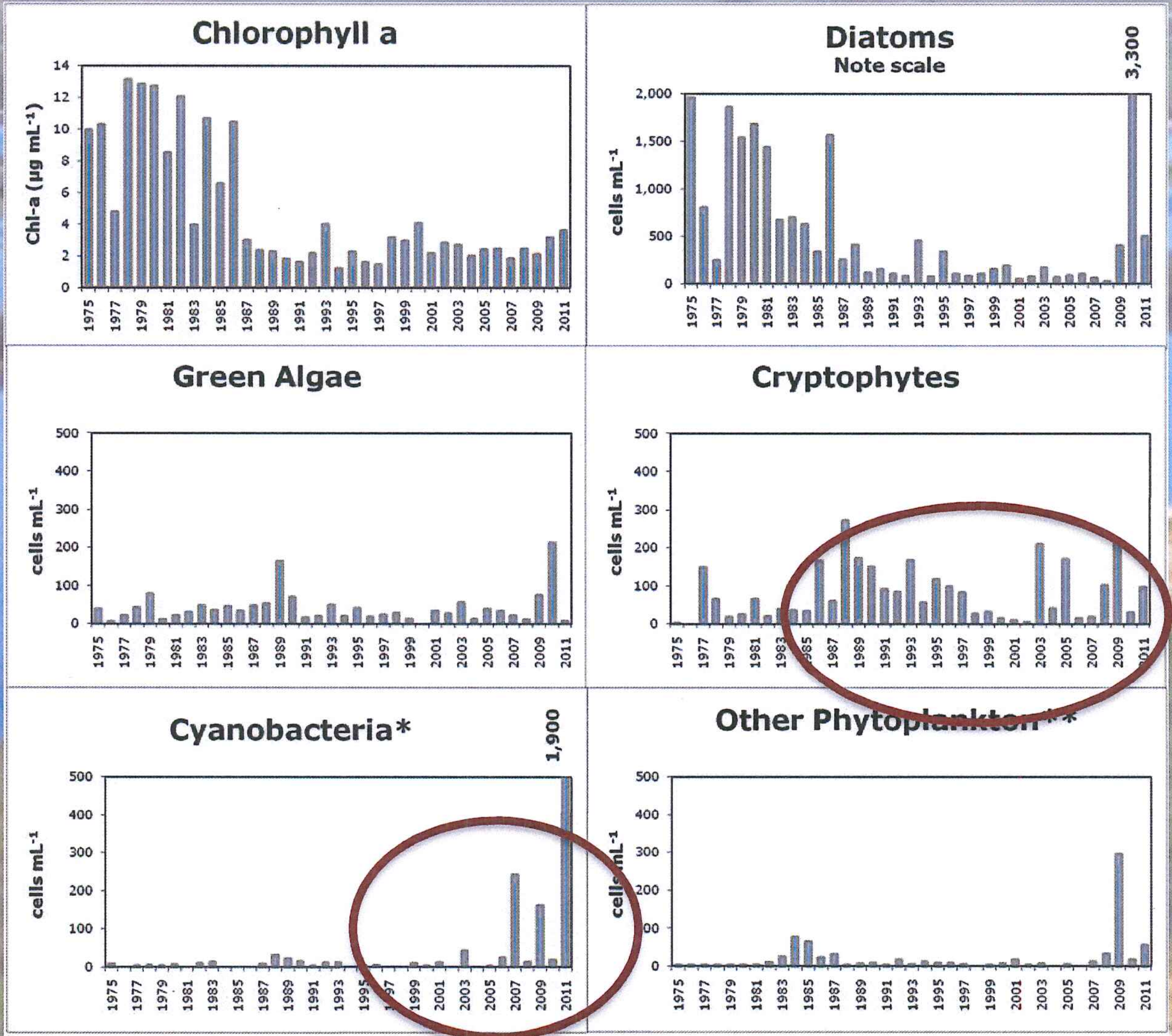
Algal community composition changes



San Francisco Bay Delta Annual averages

Glibert et al., 2011

Algal community composition changes



Chlorophyll a

Diatoms

Note scale

Green Algae

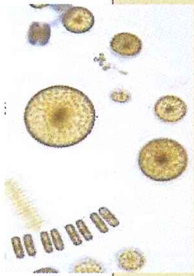
Cryptophytes

Cyanobacteria*

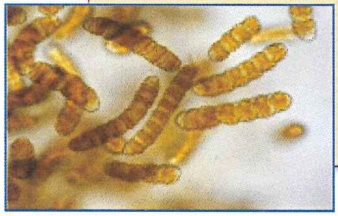
Other Phytoplankton**

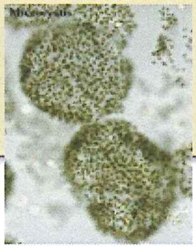
The QUALITY of algae has changed, not just the QUANTITY

Ecological stoichiometry at higher trophic levels



Changes in the QUALITY and quantity of the algae



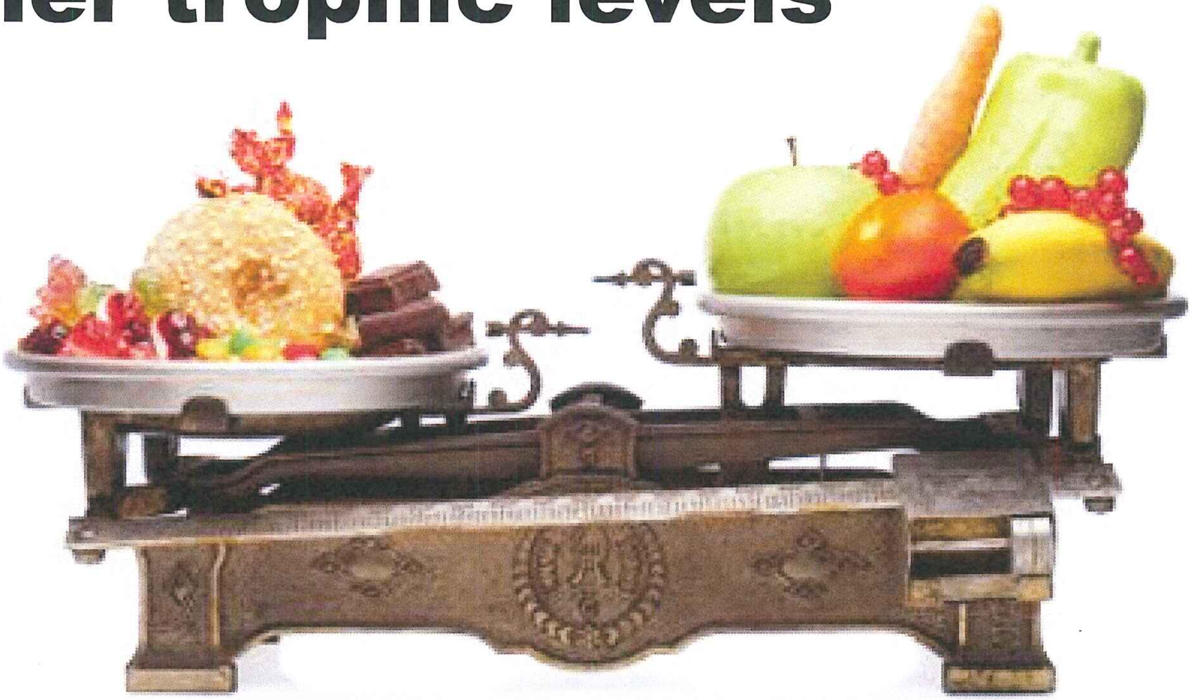


Grazers change in GROWTH, community shifts

Food web shifts

Nutrient loads, forms and ratio changes

Ecological stoichiometry at higher trophic levels



Changes begin at the metabolic scale
Changes in diet can affect reproduction, egg viability, and ultimately population success, ***GROWTH***

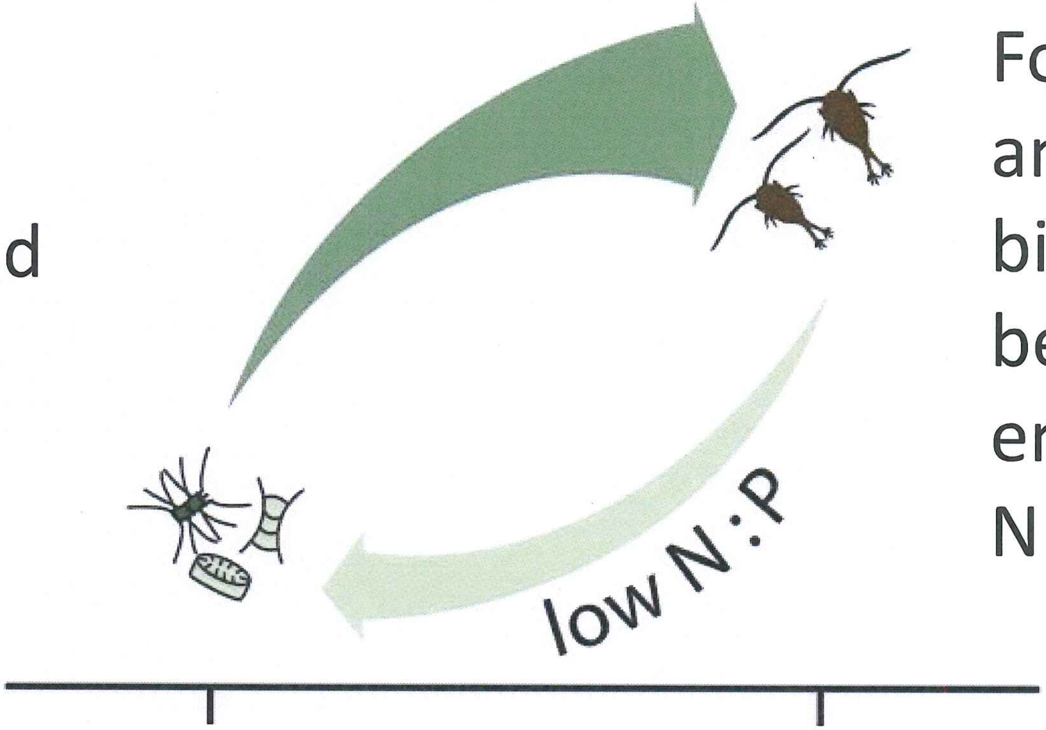
Ecological stoichiometry at higher trophic levels

“An animal’s elemental composition is linked to its evolved structure and life history…

It takes a different proportion of nutrients to make skeleton and bones than it does to make muscle; Different types of organisms thrive as nutrition changes

Ecological stoichiometry at higher trophic levels

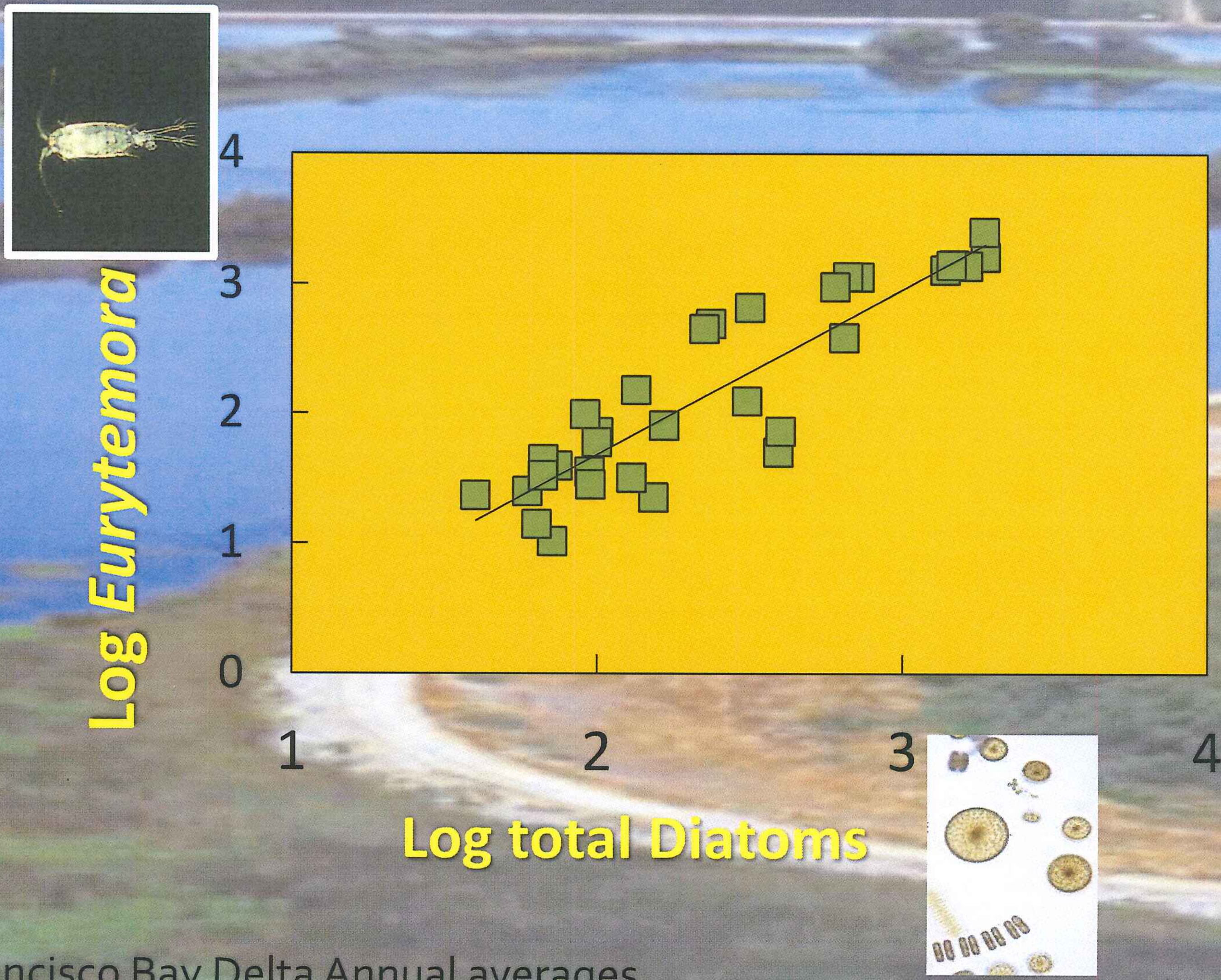
Stoichiometric needs of the secondary consumer and the prey are "fine tuned"



N:P Ratio

Food biomass and grazer biomass may be at opposite ends of the N:P spectrum

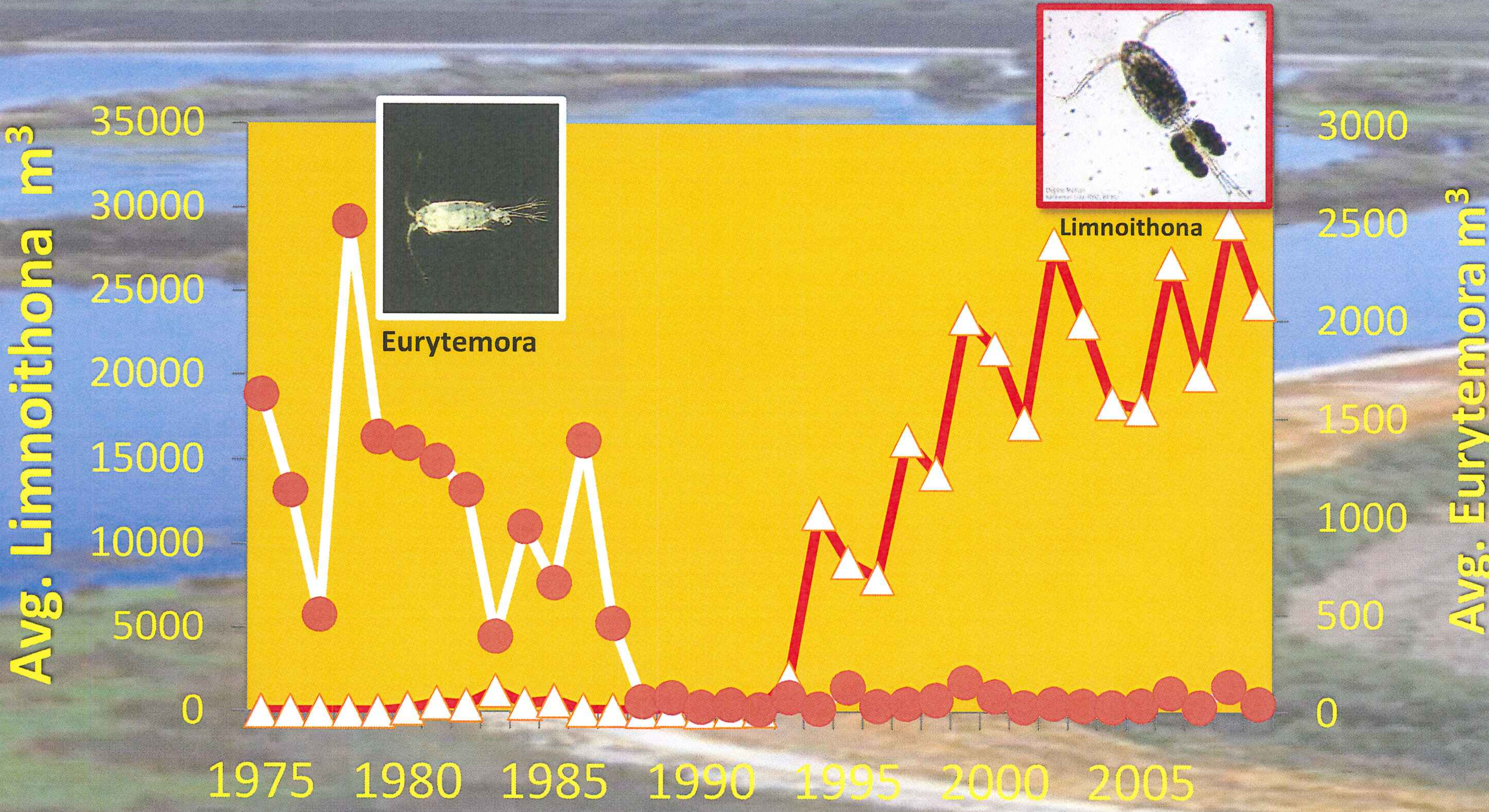
Changes at the bottom of the food web alter the community at the top



San Francisco Bay Delta Annual averages

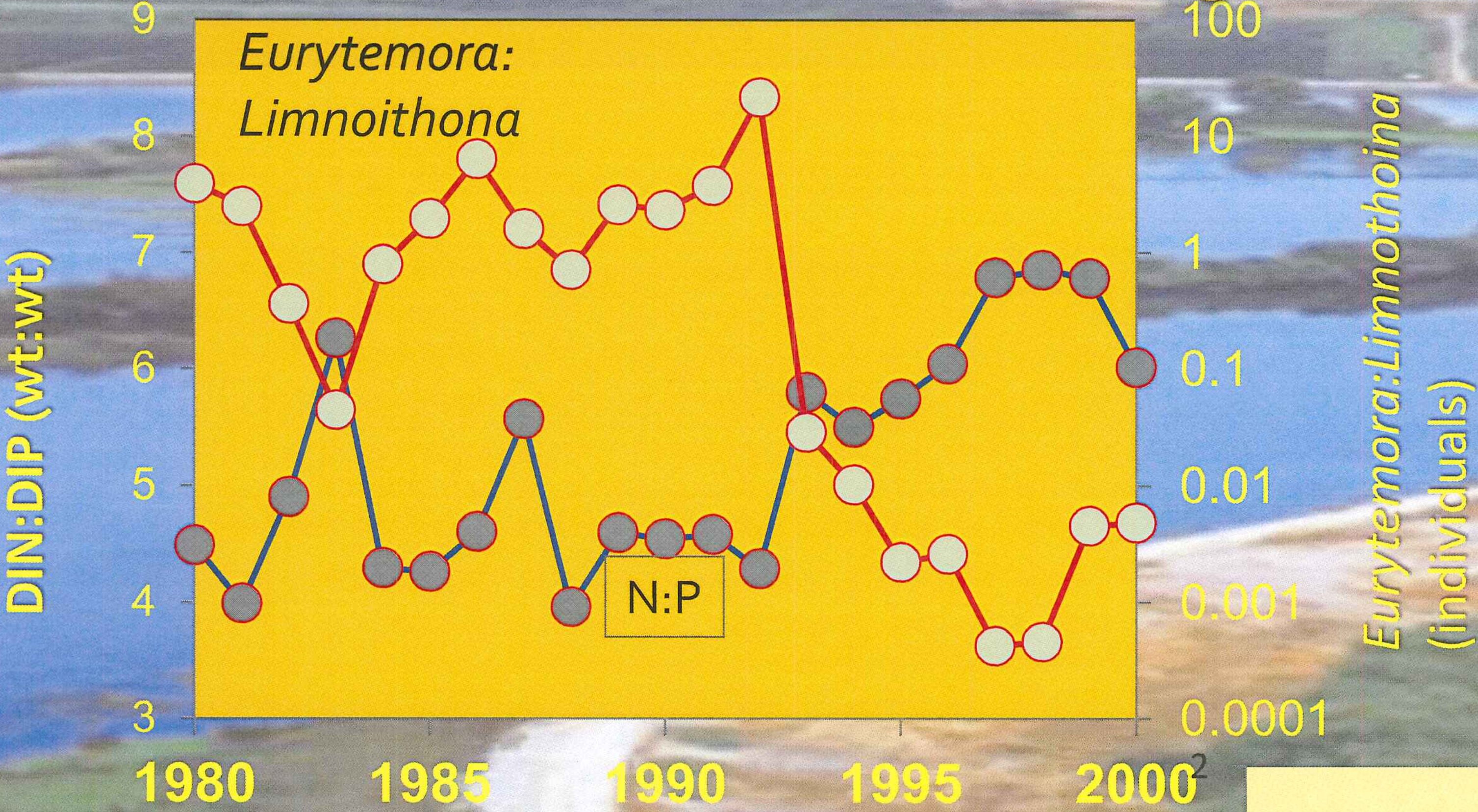
Glibert et al. 2011

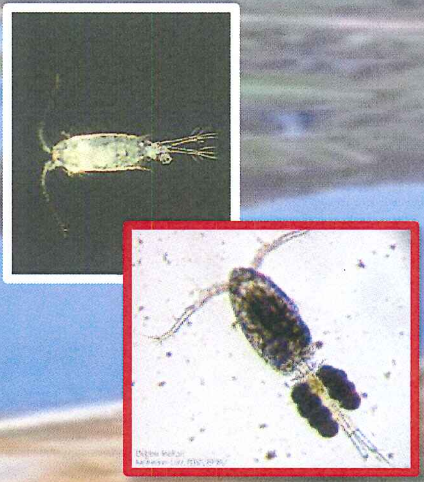
Changes at the bottom of the food web alter the community at the top

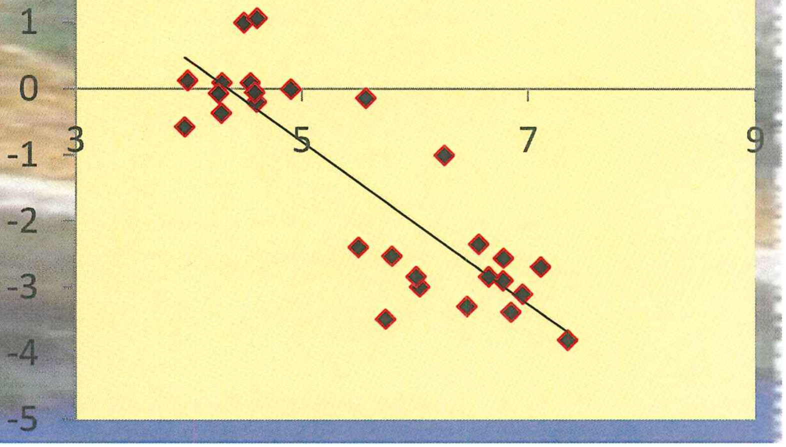


Glibert et al., 2011

Changes at the bottom of the food web alter the community at the top



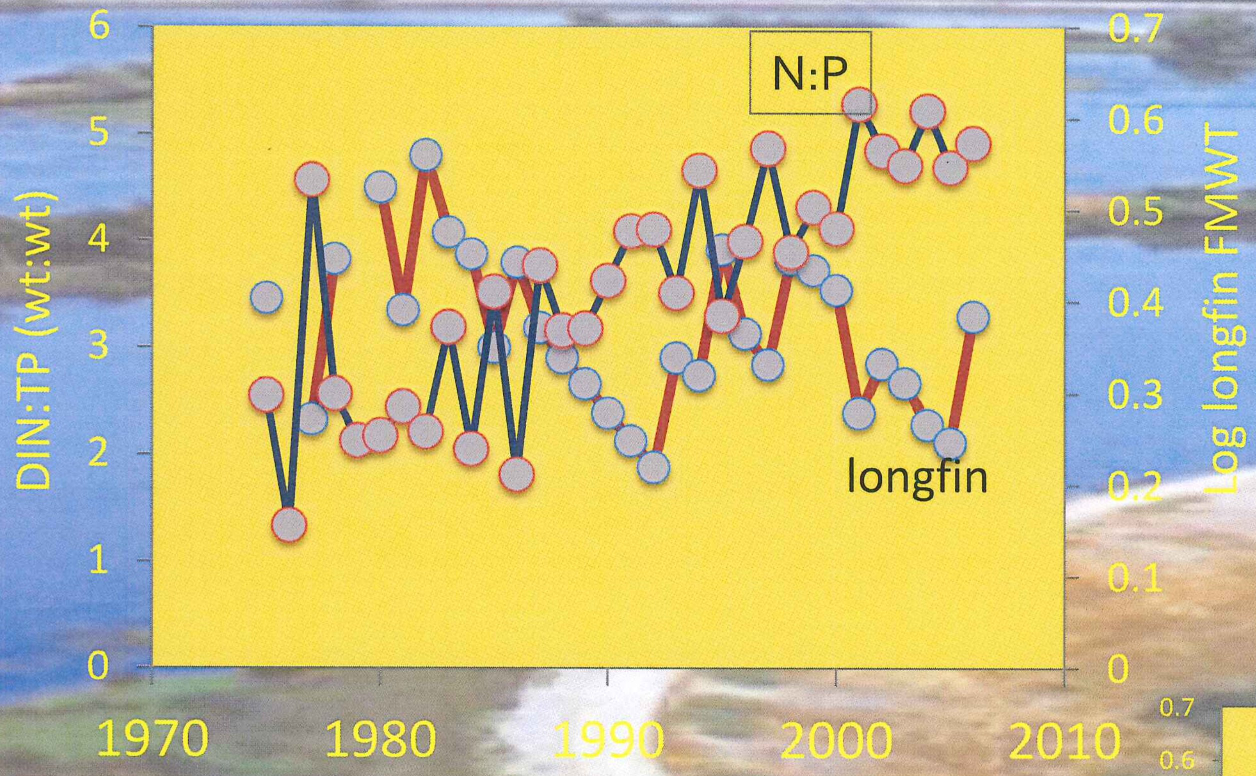


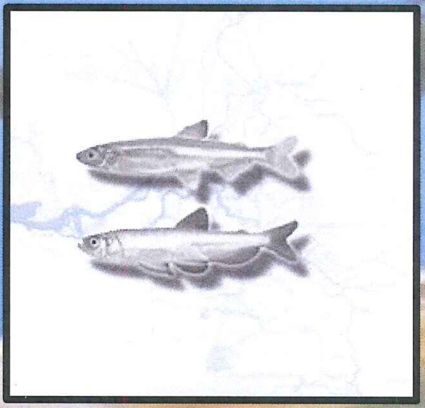


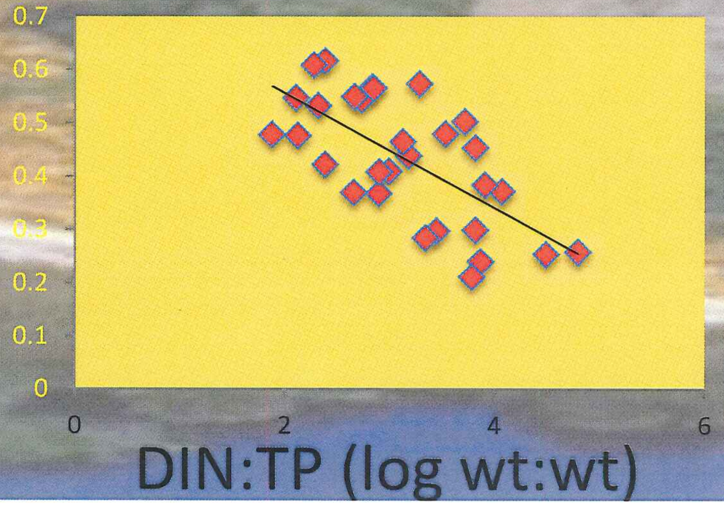
San Francisco Bay Delta Annual averages

Glibert et al. 2011

Changes at the bottom of the food web alter the community at the top



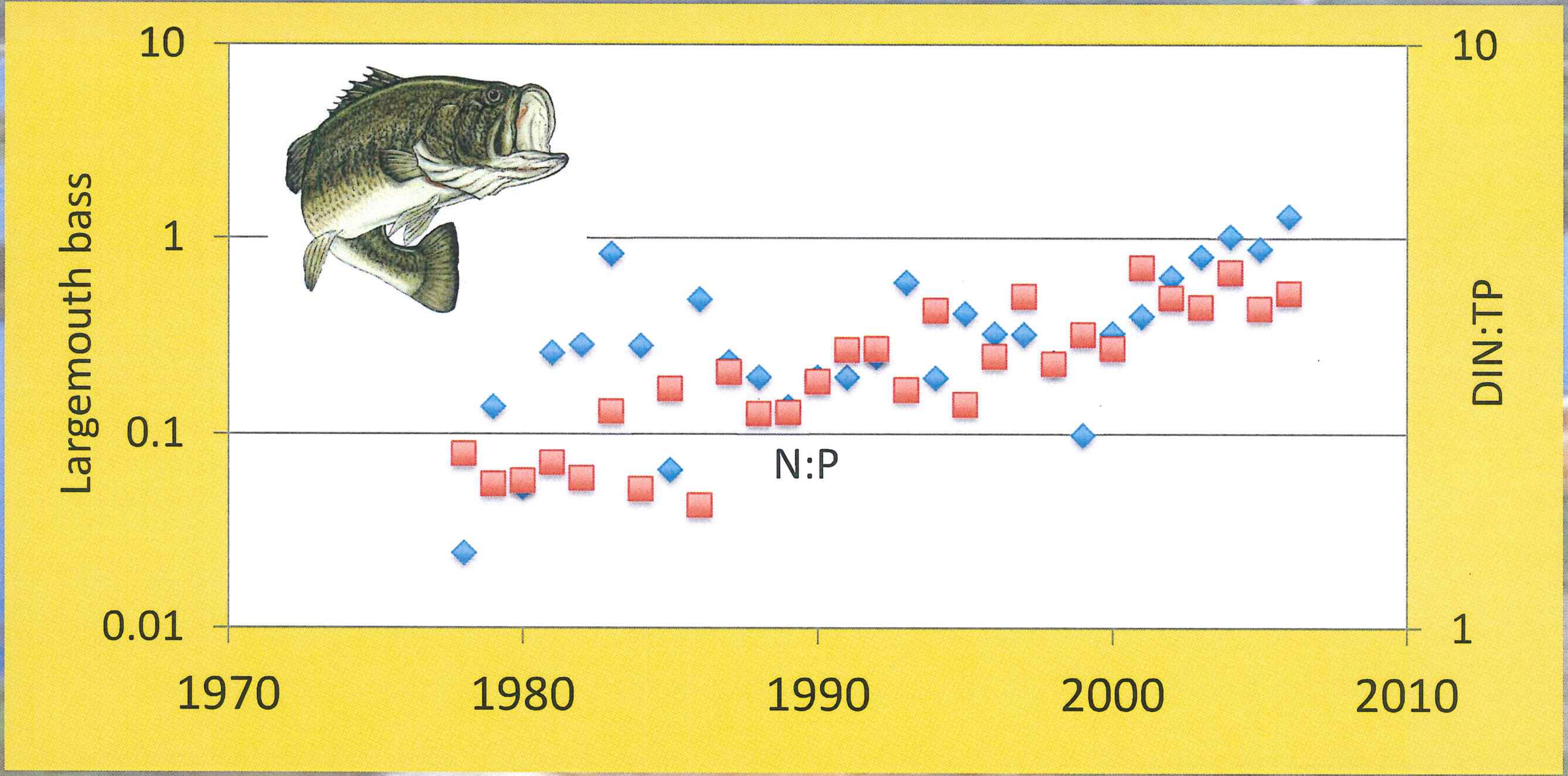




San Francisco Bay Delta Annual averages

Glibert et al., 2011

Changes at the bottom of the food web alter the community at the top

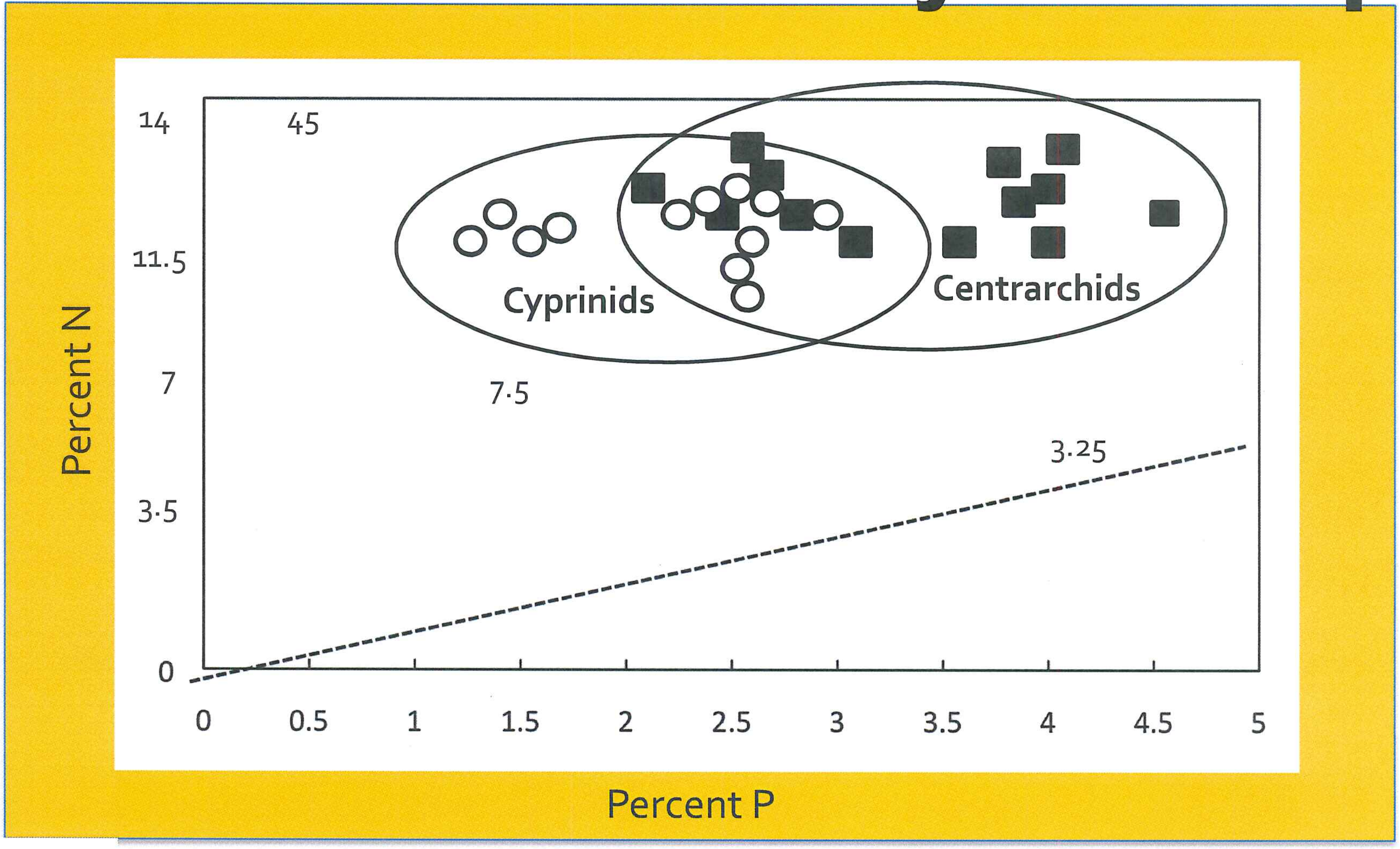


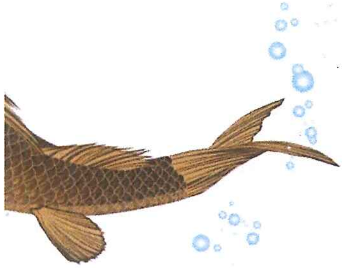
"As one ascends the pelagic food web…trophic groups grow increasingly nutrient and especially P rich…"

San Francisco Bay Delta Annual aves 1975-2005

Glibert et al., 2011

Changes at the bottom of the food web alter the community at the top

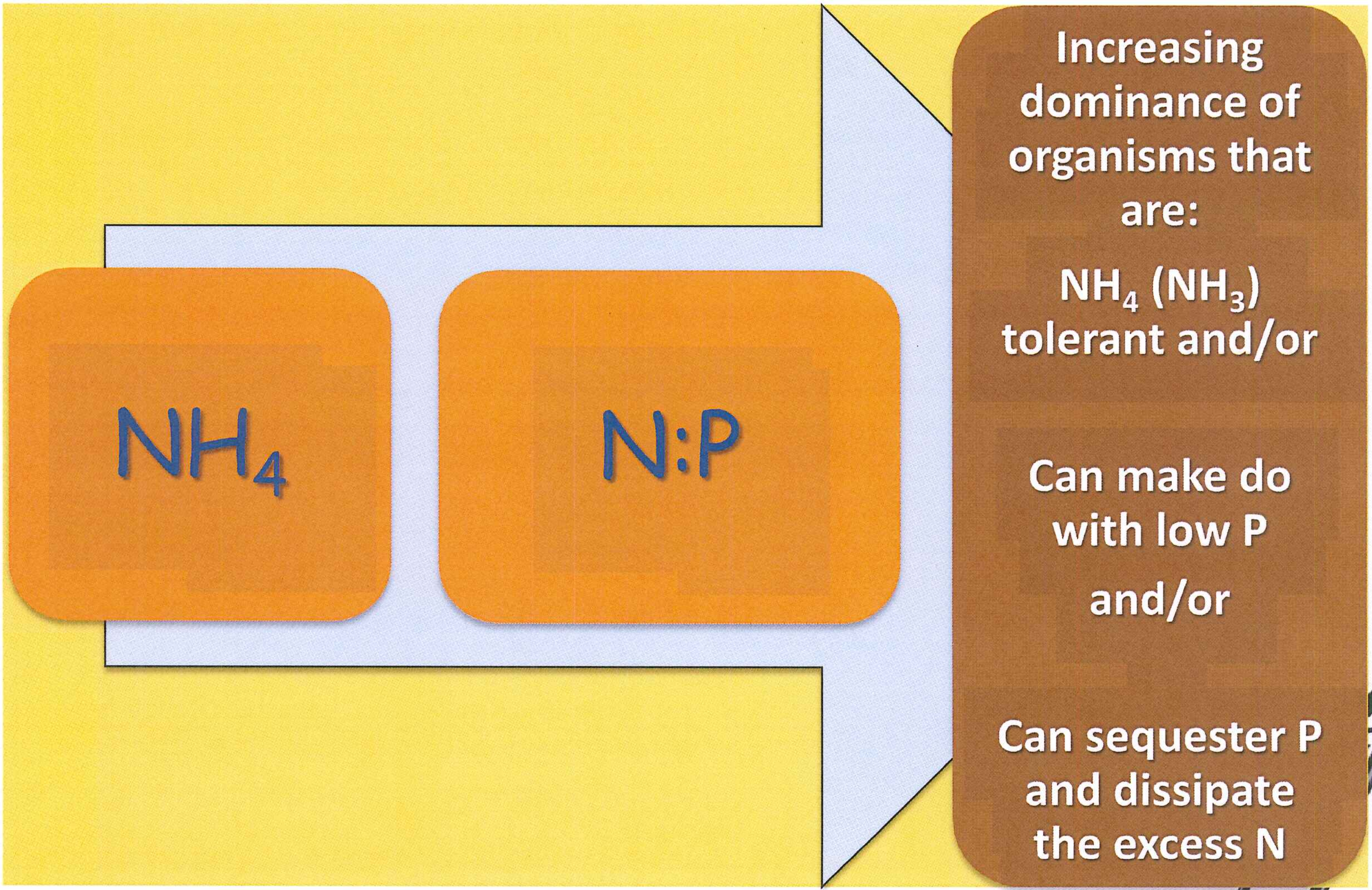




Highly evolved excretion mechanisms to help balance nutrient stoichiometry

Modified from Sterner and George 2000

Changes at the bottom of the food web alter the community at the top



NH_4

N:P

Increasing dominance of organisms that are:

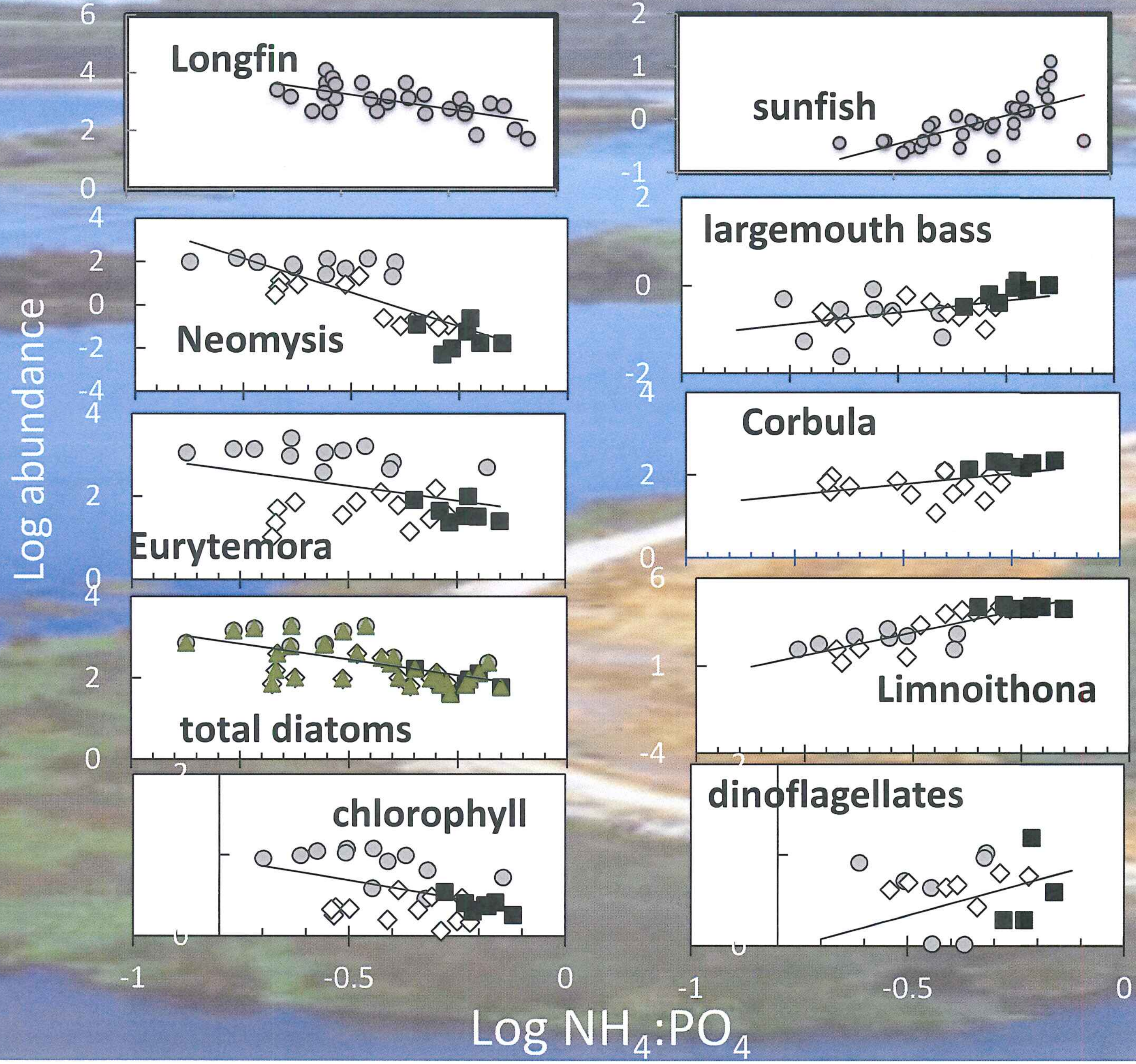
NH_4 (NH_3) tolerant and/or

Can make do with low P

and/or

Can sequester P and dissipate the excess N

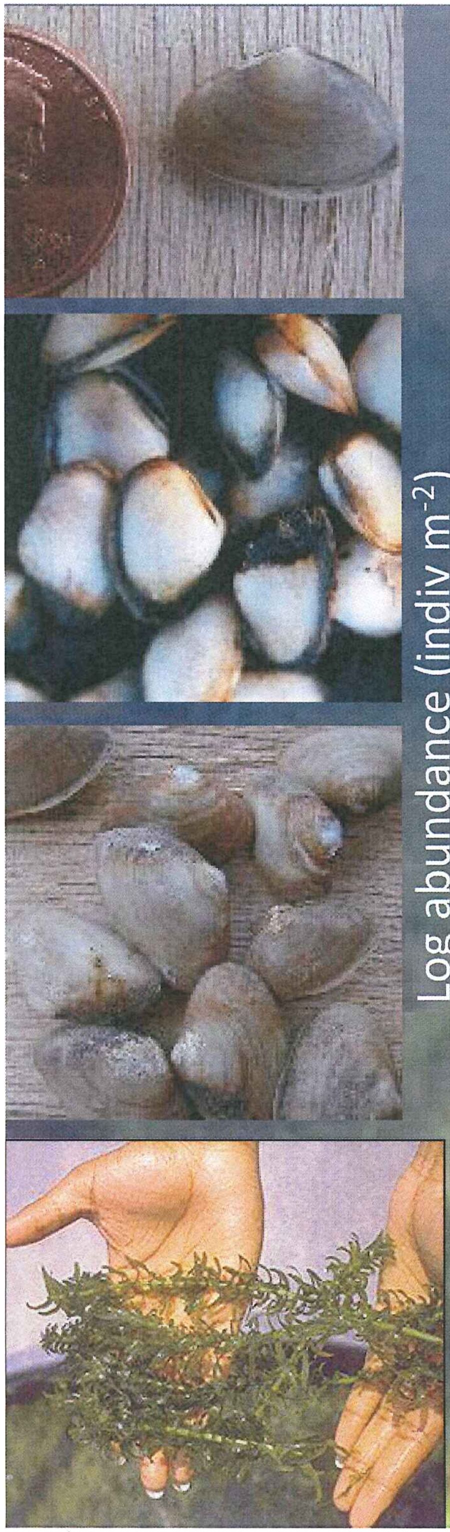
Changes at the bottom of the food web alter the community at the top



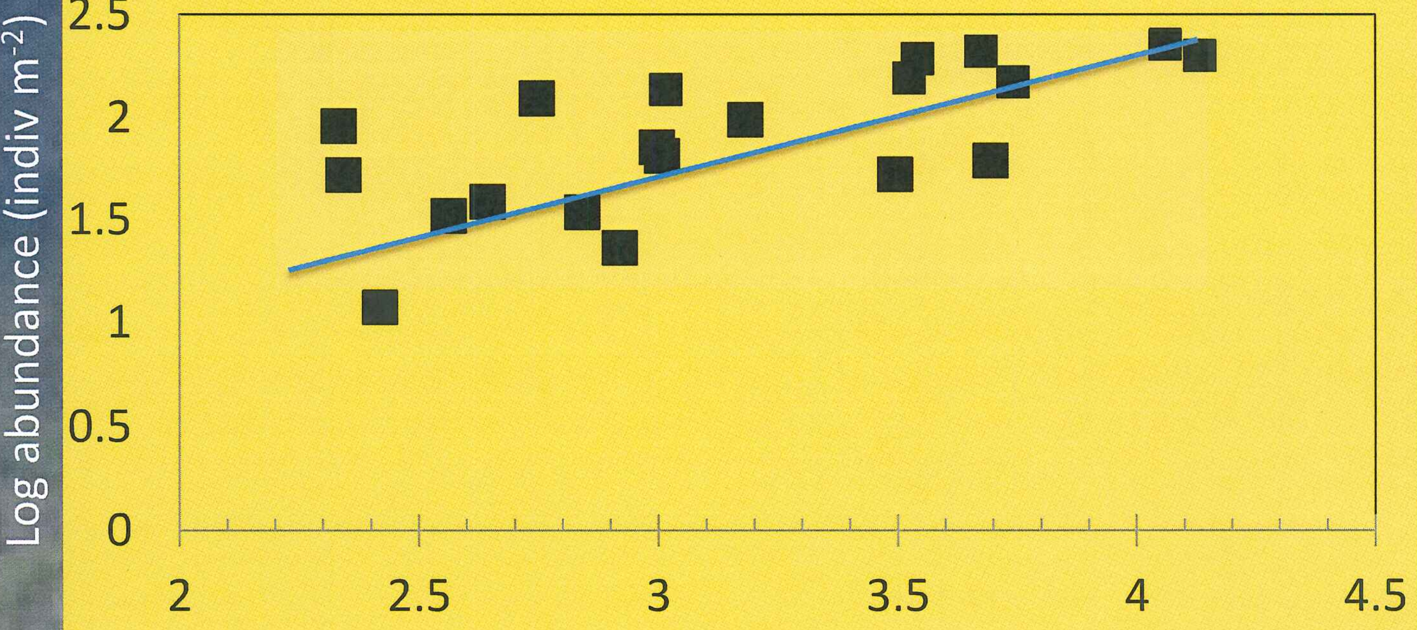
As nutrients change there are winners and losers at all levels of the food web

The winners and losers are predictable based on fundamental biological principles

Some of the winners are invasives

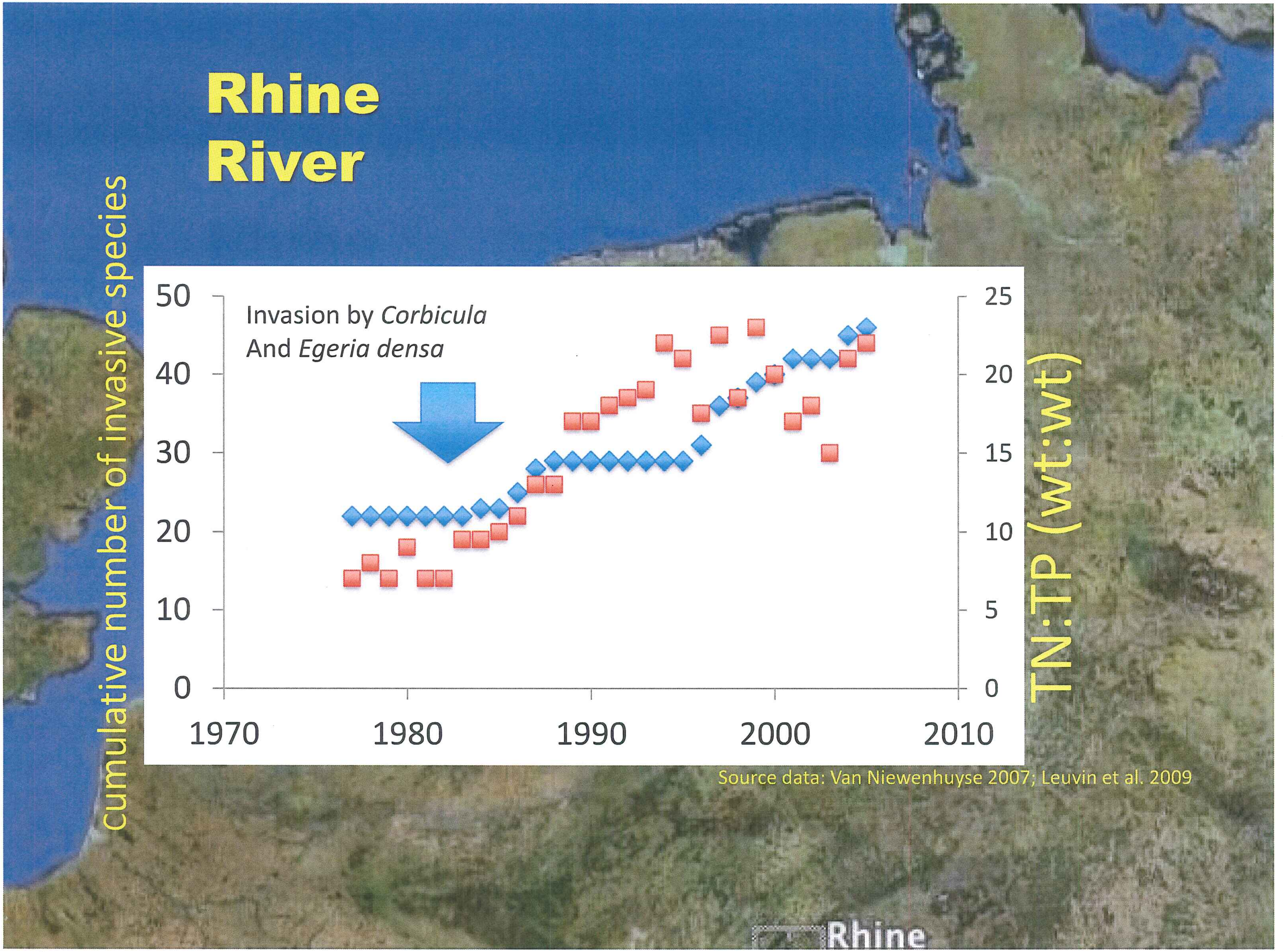


Potamocorbula



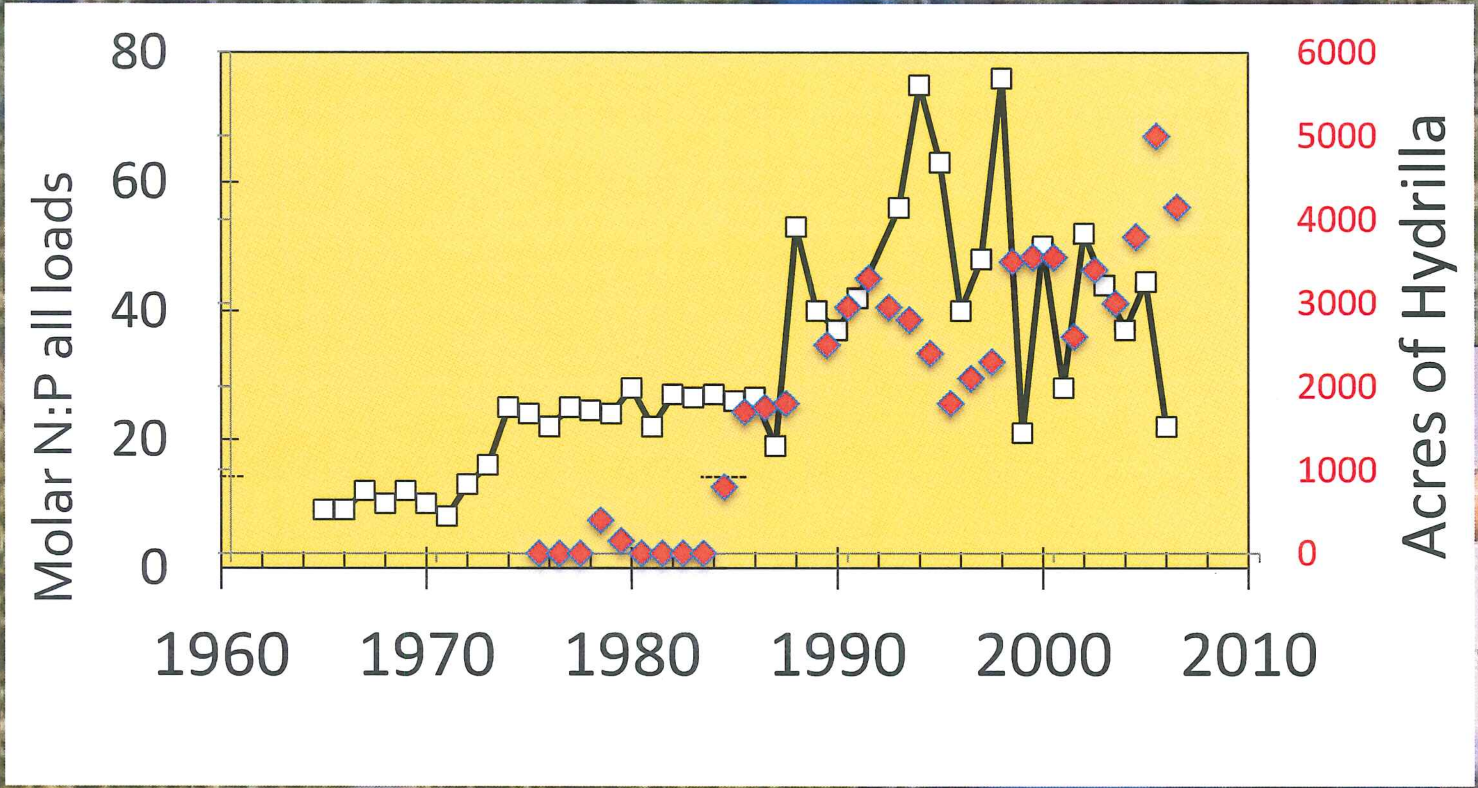
Invasive species may be as much a RESPONSE to nutrient and ecosystem change as they are a CAUSE of ecosystem change

Rhine River



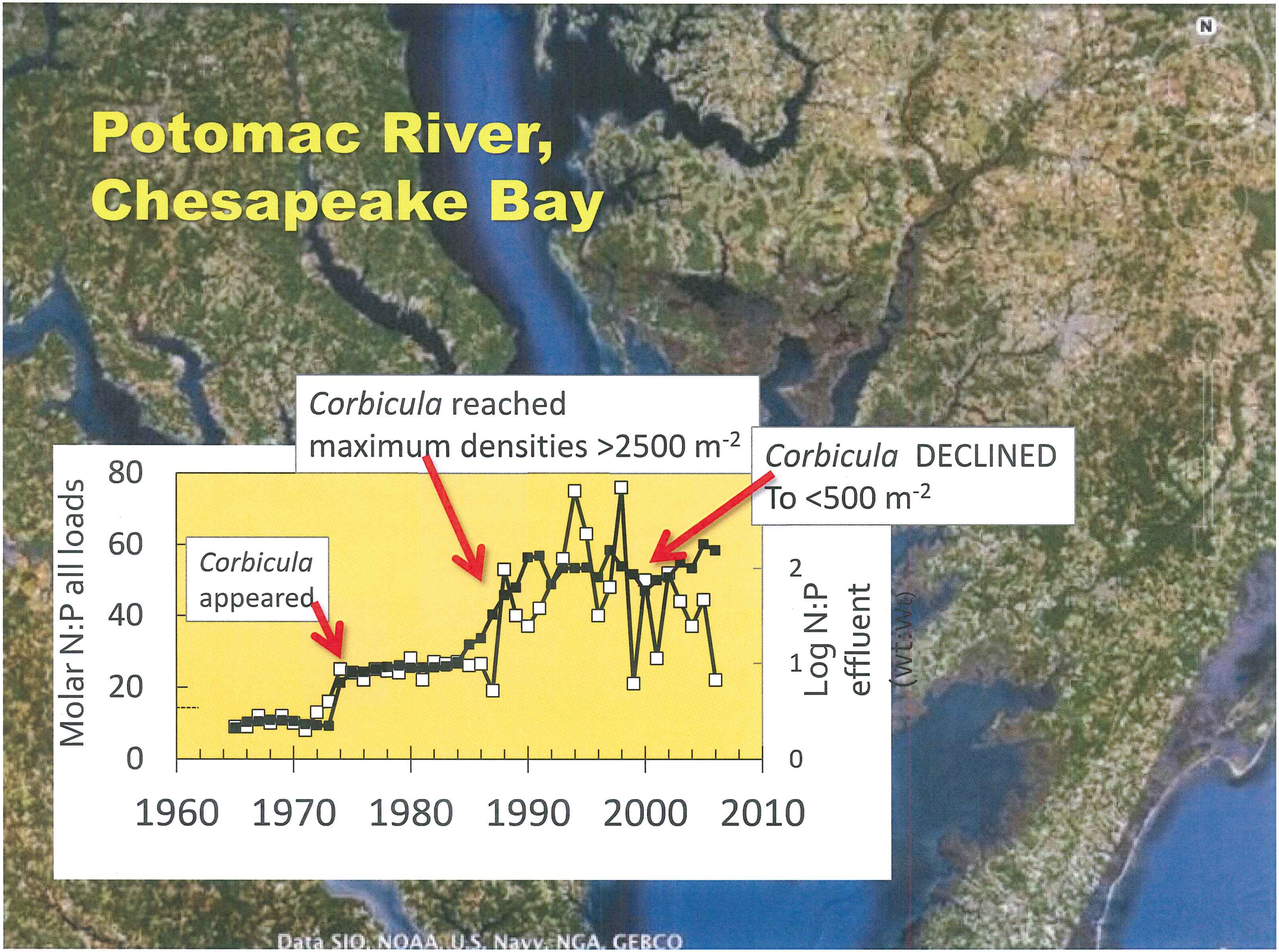
Source data: Van Niewenhuyse 2007; Leuvin et al. 2009

Potomac River, Chesapeake Bay



Data SIO, NOAA, U.S. Navy, NGA, GEBCO

Potomac River, Chesapeake Bay



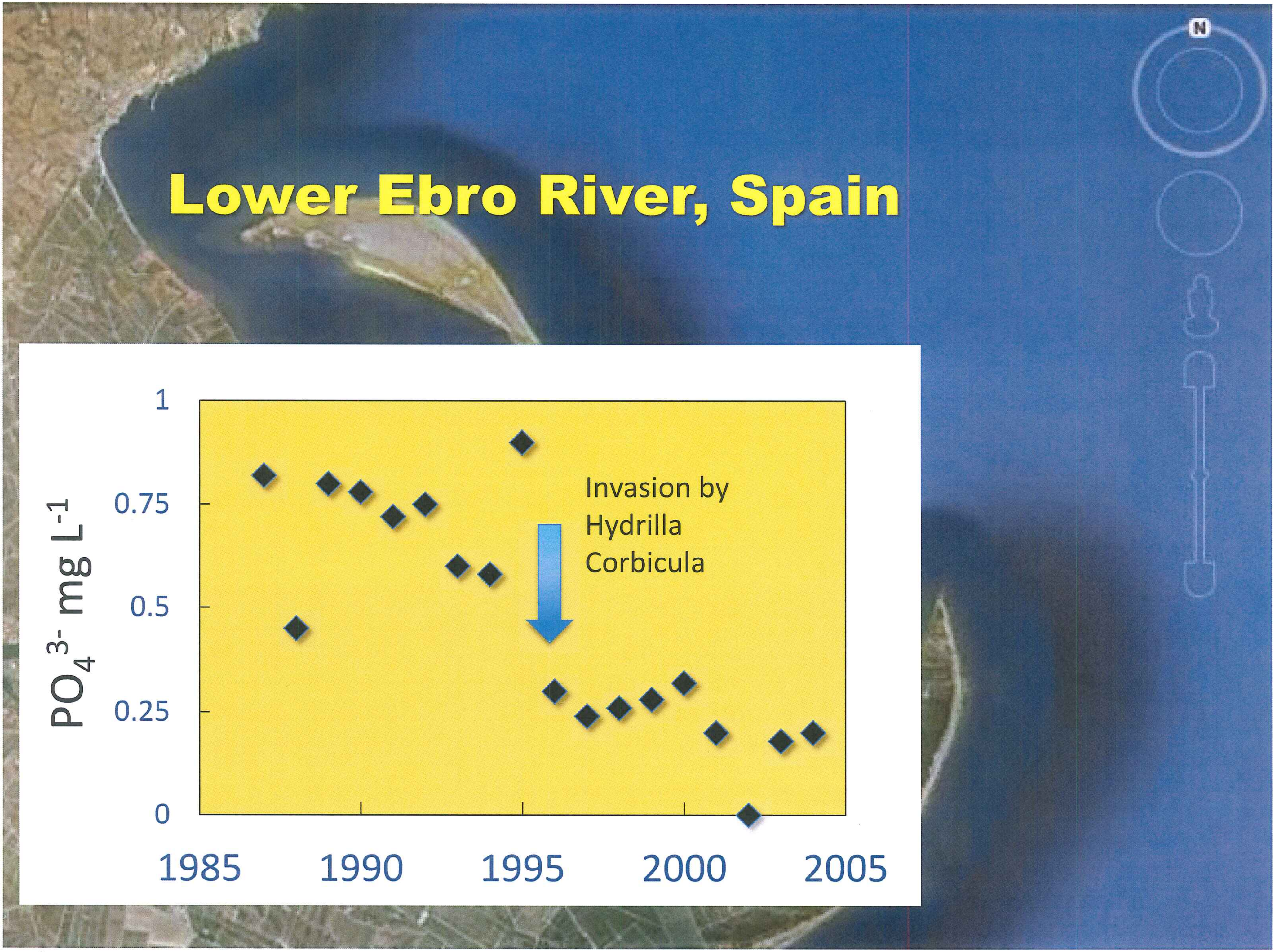
Corbicula appeared

Corbicula reached maximum densities >2500 m^{-2}

Corbicula DECLINED To <500 m^{-2}

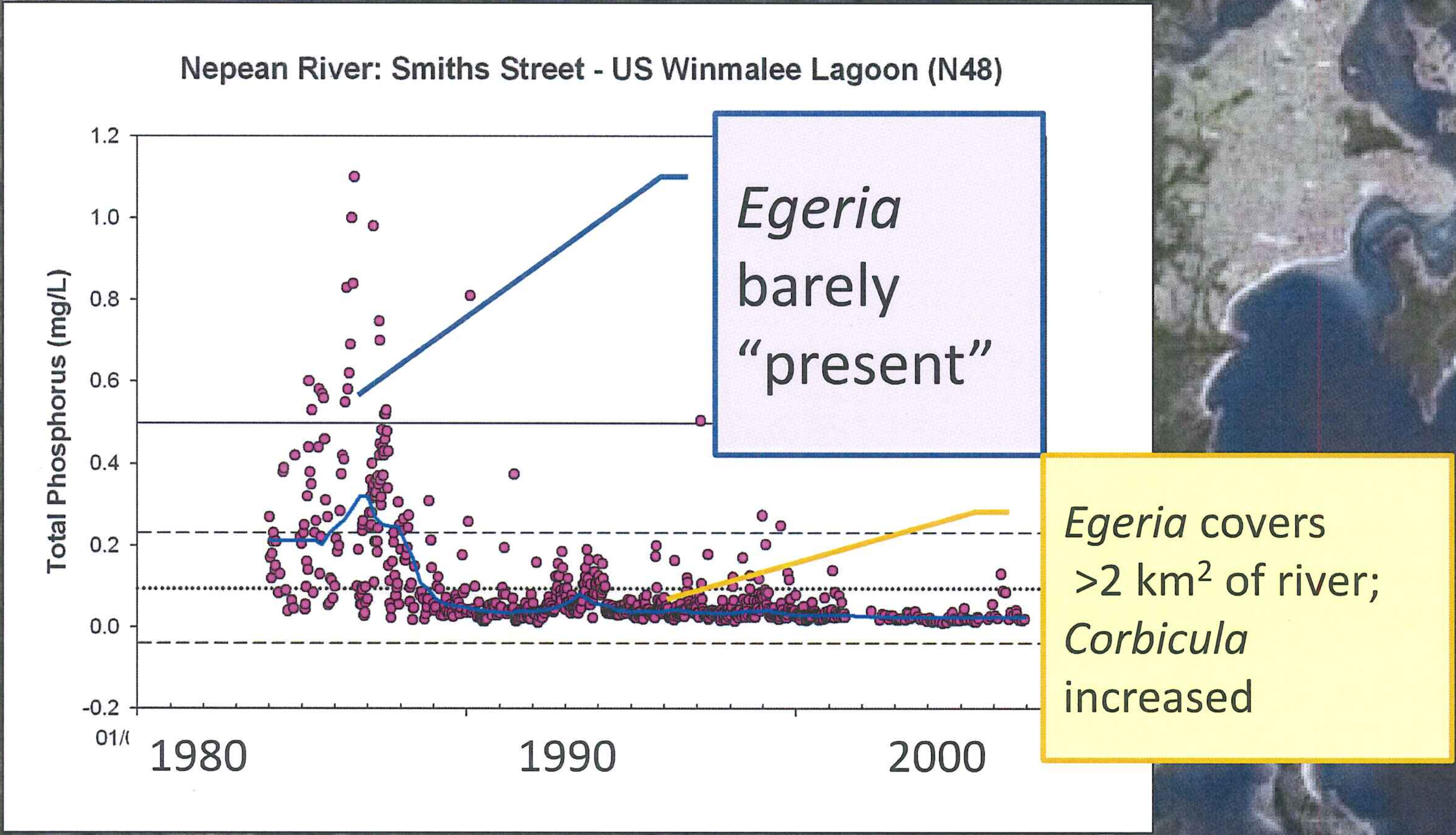
Data SIO, NOAA, U.S. Navy, NGA, GEBCO

Lower Ebro River, Spain



Invasion by
Hydrilla
Corbicula

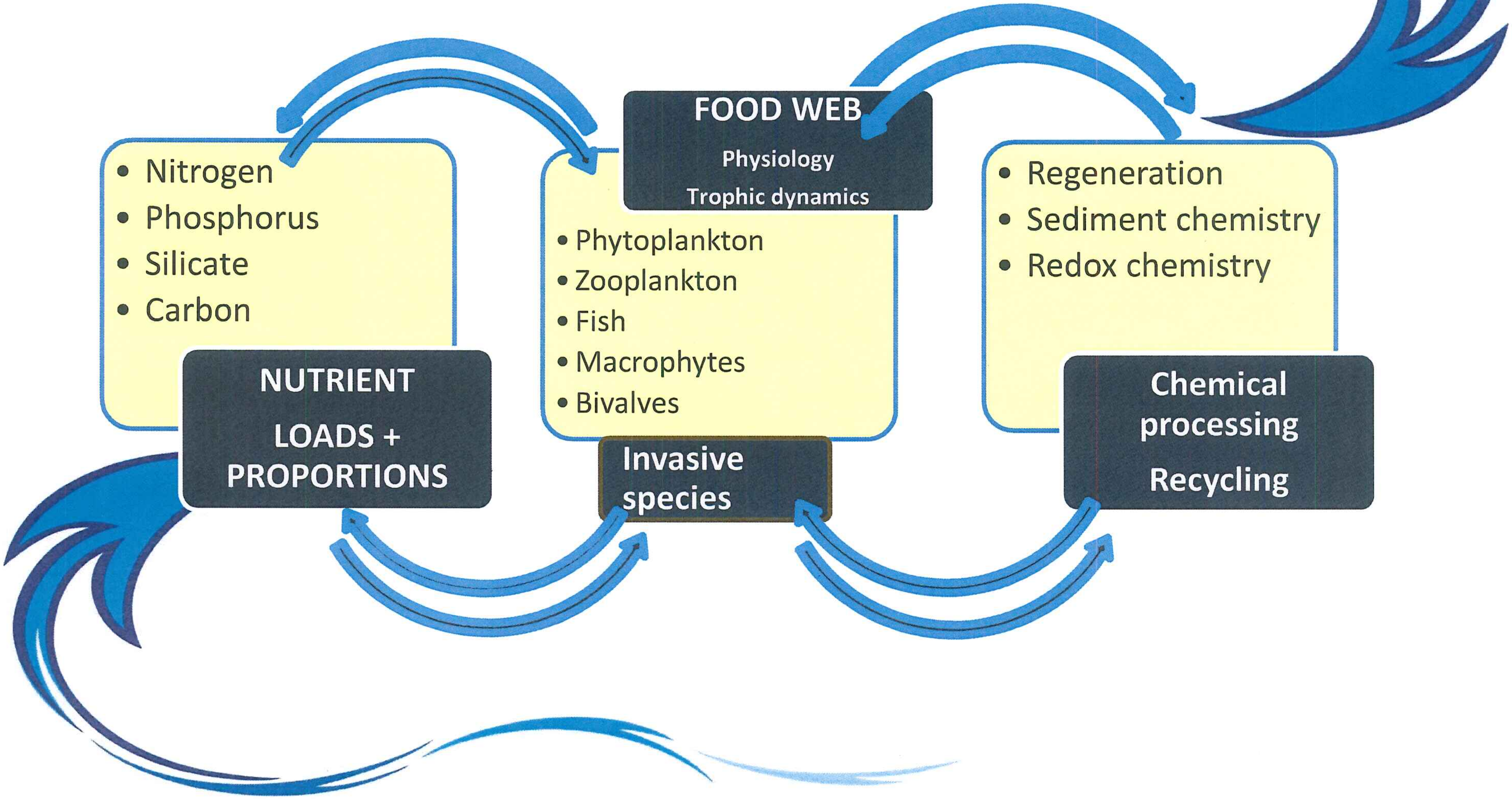
Lower Hawkesbury-Nepean River, Australia



Egeria barely "present"

Egeria covers >2 km^2 of river; *Corbicula* increased

Ultimately it's all connected ...and a grand mass balance



- Nitrogen
- Phosphorus
- Silicate
- Carbon

NUTRIENT

LOADS + PROPORTIONS

FOOD WEB

Physiology

Trophic dynamics

- Phytoplankton
- Zooplankton
- Fish
- Macrophytes
- Bivalves

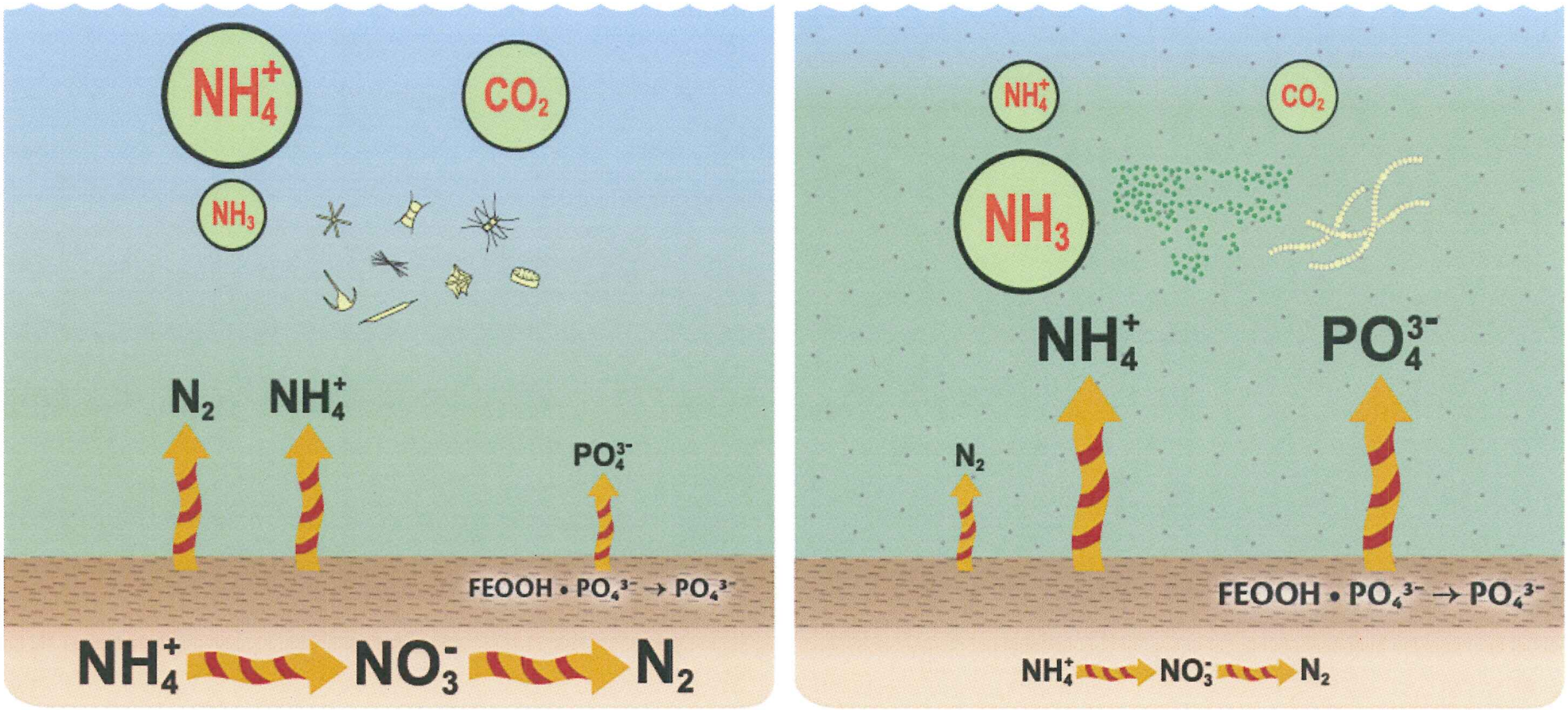
Invasive species

- Regeneration
- Sediment chemistry
- Redox chemistry

Chemical processing

Recycling

Nutrients and flow



Sediments are reservoirs of nutrients
Fluxes change with salinity, pH, temperature

Glibert et al. 2011

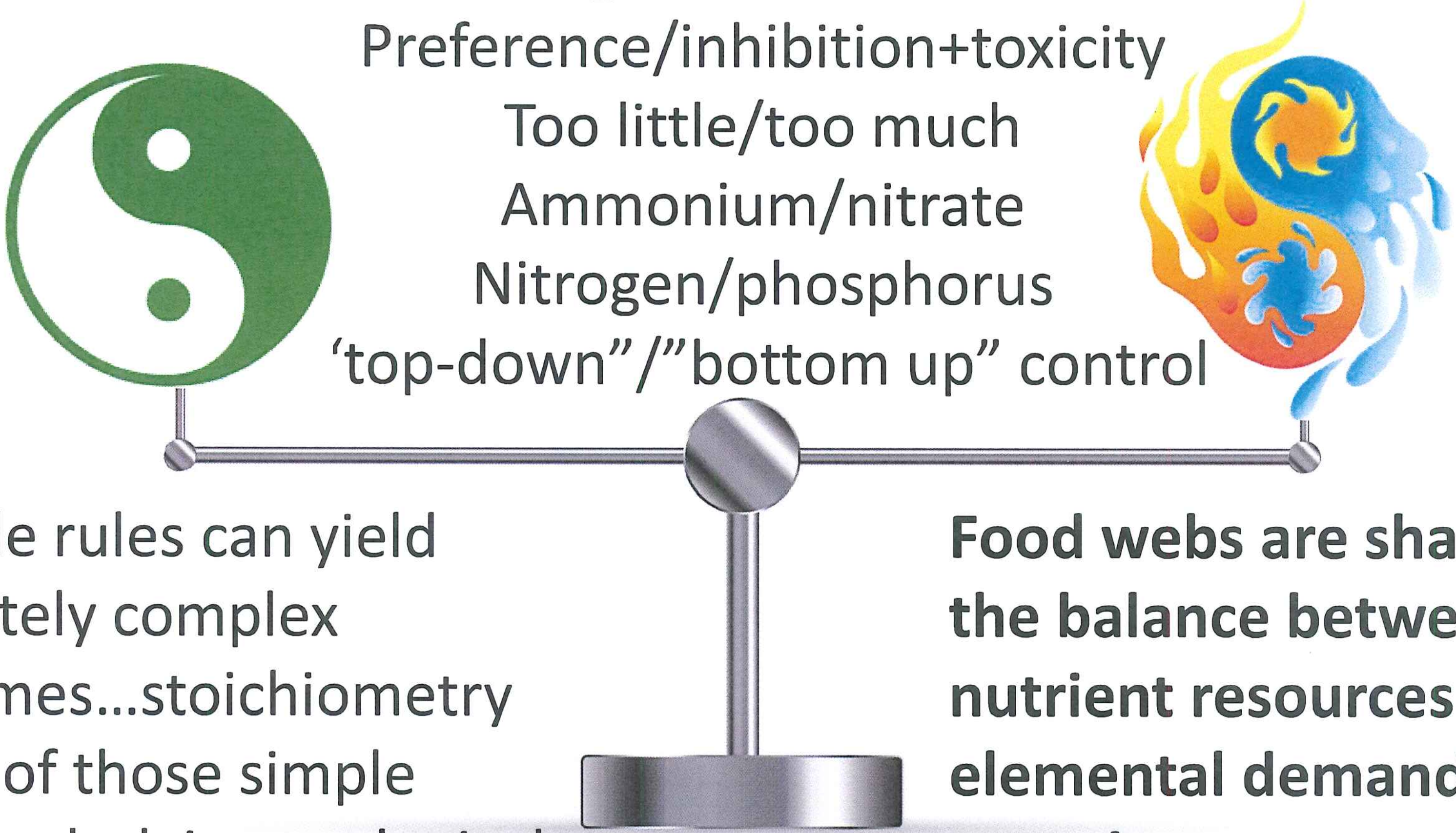
Nutrients and flow

Residence time (exposure time) to beneficial nutrients or inhibitory/toxic compounds

Flow

- Imports new nutrients
- Dilutes point source inputs
- Alters reactions at the sediment surface
- Exports nutrients downstream where they can form blooms displaced in time and space from the source.

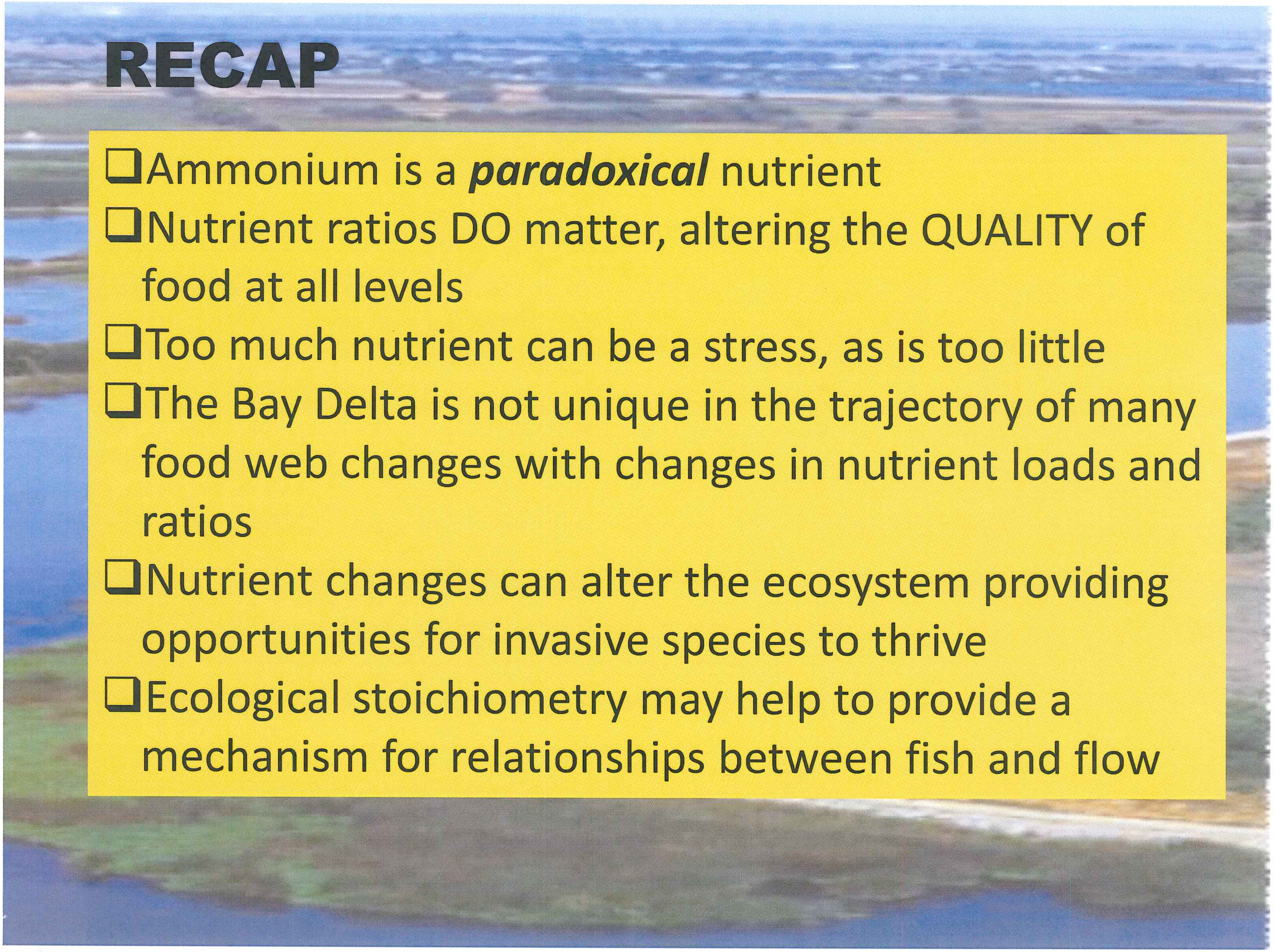
Ecosystem is in Dynamic Balance
Nutrients are part of that balance



Preference/inhibition+toxicity
Too little/too much
Ammonium/nitrate
Nitrogen/phosphorus
'top-down"/"bottom up" control

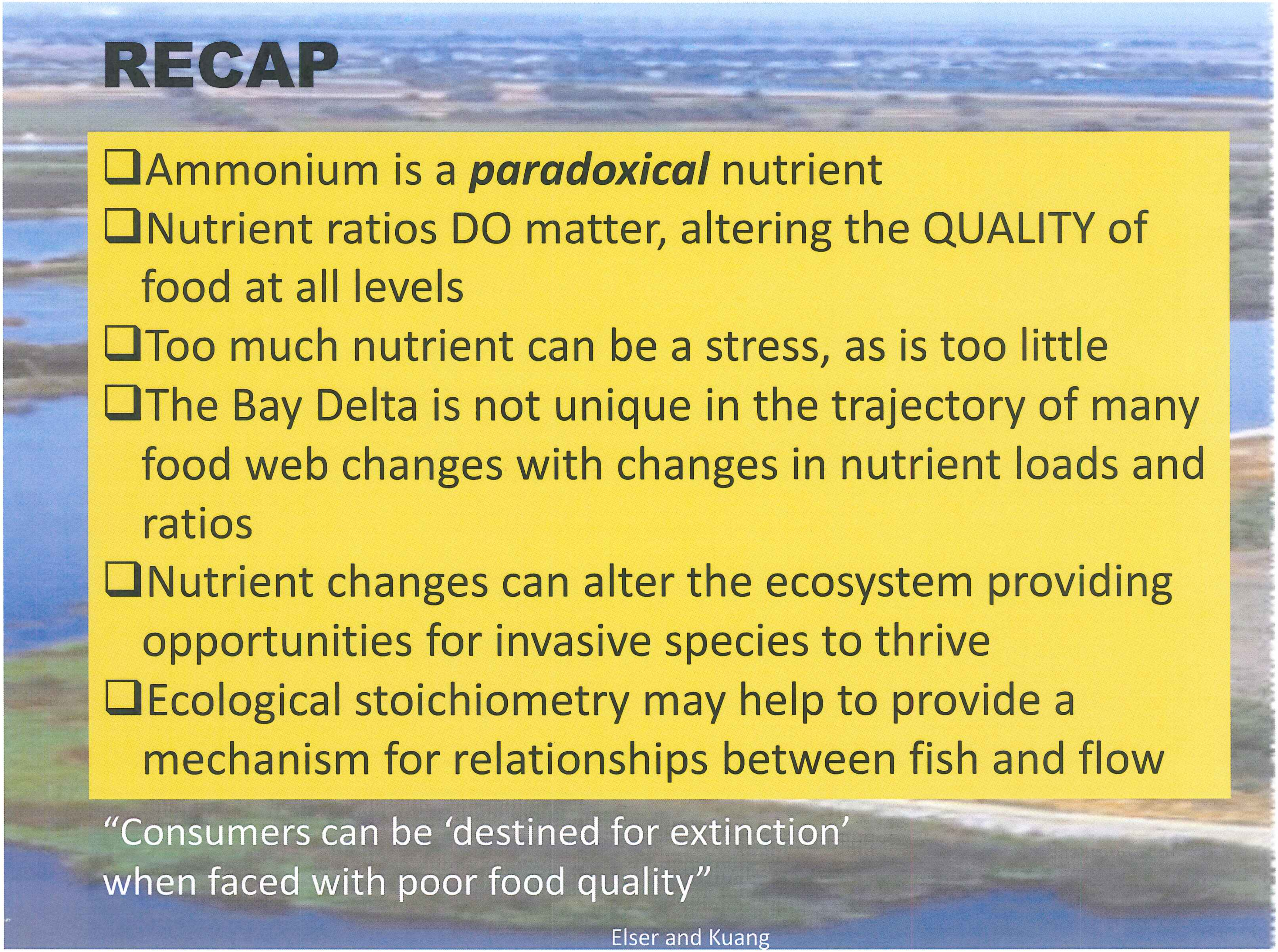
"Simple rules can yield intricately complex outcomes…stoichiometry is one of those simple rules underlying ecological and biological complexity"
Sterner and Elser 2002

Food webs are shaped by the balance between nutrient resources and the elemental demands of the organisms



RECAP

- ☐ Ammonium is a ***paradoxical*** nutrient
- ☐ Nutrient ratios DO matter, altering the QUALITY of food at all levels
- ☐ Too much nutrient can be a stress, as is too little
- ☐ The Bay Delta is not unique in the trajectory of many food web changes with changes in nutrient loads and ratios
- ☐ Nutrient changes can alter the ecosystem providing opportunities for invasive species to thrive
- ☐ Ecological stoichiometry may help to provide a mechanism for relationships between fish and flow



RECAP

- ☐ Ammonium is a ***paradoxical*** nutrient
- ☐ Nutrient ratios DO matter, altering the QUALITY of food at all levels
- ☐ Too much nutrient can be a stress, as is too little
- ☐ The Bay Delta is not unique in the trajectory of many food web changes with changes in nutrient loads and ratios
- ☐ Nutrient changes can alter the ecosystem providing opportunities for invasive species to thrive
- ☐ Ecological stoichiometry may help to provide a mechanism for relationships between fish and flow

"Consumers can be 'destined for extinction' when faced with poor food quality"

Elser and Kuang

Ecosystem Changes to the Bay-Delta Estuary: A Technical Assessment of Available Scientific Information

August 17, 2012

Submitted by: State Water Contractors, Inc.
San Luis & Delta-Mendota Water Authority

Table of Contents

Tables

Figures

Acronyms

AF	acre-foot/feet
BDCP	Bay Delta Conservation Plan
BiOp	biological opinion

C:P	ratio of carbon to phosphorus
cfs	cubic foot/feet per second
chl-*a*	chlorophyll-*a*
CVRWQCB	Central Valley Regional Water Quality Control Board
CVP	Central Valley Project
DEHP	bis(2-ethylhexyl)phthalate
DFG	California Department of Fish and Game
DIN:DIP	ratio of dissolved inorganic nitrogen to dissolved inorganic phosphorus
DIN:TP	ratio of dissolved inorganic nitrogen to total phosphorus
DWR	California Department of Water Resources
EC_{50}	median effect concentration; concentration that effects 50% of population
ETM	estuarine turbidity maximum
K-M	Kimmerer-Monismith (equation)
LC_{50}	lethal concentration 50, concentration that is lethal to 50% of population
MAF	million acre-feet
$\mu g\ g^{-1}$	microgram(s) per gram
$\mu g\ L^{-1}$	microgram(s) per liter
μmol	micromol(s)
$mg\ L^{-1}$	milligram(s) per liter
mm	millimeter(s)
MMHg	monomethylmercury
N:P	ratio of nitrogen to phosphorus
$ng\ L^{-1}$	nanogram(s) per liter
$NH_4:NO_3$	ratio of ammonium to nitrate
IEP	Interagency Ecological Program
PCB	polychlorinated biphenyls
POD	pelagic organism decline
SSC	suspended-solids concentration
SWB	State Water Board
SWRCB	State Water Resources Control Board
SWP	State Water Project
TAF	thousand acre-feet
USEPA	United States Environmental Protection Agency
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey

Executive Summary

ES1. Introduction

In the upcoming series of workshops, the State Water Resources Control Board (State Water Board) will receive information and conduct discussions regarding the scientific and technical basis for potential changes to the 2006 Water Quality Control Plan for the Bay-Delta. This presentation has been prepared to help inform the first of those workshops on *Ecosystem Changes and the Low Salinity Zone*.

The State Water Board should be applauded for recognizing the need to dedicate significant resources to ensure it can consider the available scientific information on key ecosystem attributes of the Bay-Delta, and other rivers and estuaries elsewhere. The State Water Contractors (SWC) and San Luis & Delta-Mendota Water Authority (SLDMWA) have attempted to compile and critically assess that scientific information below.

The State Water Board can draw on a two-decade history of attempts to enhance numbers of and reverse declining trends in native fish with flows through the Delta. A dominant operating conservation surmise has been that management of flow in the highly dynamic Bay-Delta estuary is the best direct route to enhancing the status of imperiled fishes. But, while a number of directed studies have found correlations between flows through the Delta and population changes in several fishes, no data are available that explain how flows affect the survival and persistence of the Delta's native species, and no studies support the assertion that additional flows will produce greater numbers of at-risk fishes, enhance the resources that support them, or compensate for the highly altered physical landscape that now accommodates both.

The review and analysis presented here indicates that unilaterally managing flows without addressing the actual environmental stressors that are directly and indirectly compromising at-risk fishes and their essential resources will not serve to reduce threats to those fishes' survival and recovery, and actually could contribute to further population declines. The best available science demonstrates that four changes in salient environmental attributes of the Delta have contributed to the current degraded state of the Bay-Delta ecosystem and appear to be proximate causes of declines in desired fishes:

1. changes in the composition of the food web that supports those fishes,
2. increases in water temperatures in portions of the estuary,
3. overall and localized reductions in turbidity, and
4. changes to the physical landscape.

The discussion of environmental changes that have disrupted the food web that supports the Delta's desired fishes presented here includes a review of available scientific information that supports or refutes six possible explanations for observed population declines – changes in 1) flows or flow patterns, 2) water appropriations, 3) contaminants, 4) invasive species, 5) nutrients, and 6) changes to the physical landscape. Some evidence exists that suggests that two or more of those sources of ecosystem change could operate in concert by positively and negatively reinforcing biological and biogeochemical feedbacks.These many changes to the Delta's ecosystems are considered immediately below, and explored in detail in the presentation that follows.

ES2. Changes to the Food Web

ES2.1 Observed Changes

The food web that supports native fishes in the Bay-Delta is different than that of decades ago. Primary productivity and phytoplankton biomass, as measured by chlorophyll-*a* (chl-*a*) concentrations, decreased significantly between 1975 and 1995 and still remain low. A paper that studies freshwater, marine and estuary systems around the world (Nixon, 1988) reports a strong relationship between production at the base of the food web (primary production) and production of fish (fishery yield). It provides an explanation for the low fishery production in the Bay-Delta estuary. Primary productivity data from the Bay-Delta estuary superimposed on the results of the study show very low and declining primary productivity in the Bay-Delta. Based on the results presented in the Nixon paper, that low primary productivity results in low fishery production.*The State Water Board should commission a study that directly applies the principles of the Nixon paper to the Bay-Delta estuary, one that compares total fishery yield to primary production within the Bay-Delta estuary.*

The link between changes (declines) in primary production and fish production in the Bay-Delta estuary is further supported by the shift in the dominant phytoplankton species from diatoms to less nutritious and sometimes toxic algal species. Secondary productivity has also changed, with larger, more accessible zooplankton species being replaced by smaller species. Because aquatic ecosystems are substantively determined by the structure and composition of their constituent food webs, efforts to increase the abundance of native fish in the Bay-Delta require identification of the most important factors affecting phytoplankton production. There is substantial scientific agreement that many of the observed changes in Delta primary productivity and food web composition are a result of introduction of non-native and invasive species, such as the Amur River clam, and that those changes have had significant impacts on the abundance and distribution of several desired species, such as the northern anchovy, mysid shrimp, and striped bass. Changes to the lower levels of the Delta food web also appear to have had detrimental impacts to other native species, including delta smelt and longfin smelt.

A number of explanations have been suggested to account for the observed changes to the composition of the Bay-Delta food web.

ES2.2 Possible Explanations of Changes to the Bay-Delta Food Web

Possible Explanation 1: Changes in flows and flow patterns that affect primary and secondary productivity

There is general agreement that freshwater flow is an important factor in primary and secondary productivity. In an estuary, freshwater flows deliver nutrients from upstream areas, re-suspend nutrients within the estuary, and enhance dispersion with salinity stratification. Soluble and particulate nutrient enrichment from freshwater inflow makes estuaries some of the most productive aquatic ecosystems.

Ecological processes in estuaries, however, are inherently complex due to their dynamic nature, mixing processes, and intricate ecological linkages. Because the Bay-Delta estuary is not a riverine system, but a highly altered tidal environment, the relationship between flow and productivity is neither simple nor linear. Recent research in the Bay-Delta on the relationship between through Delta flows and productivity have produced contradictory results, with some reporting variable results within different reaches of the same channel, some unable to detect a direct relationship, and some identifying an inverse relationship - such as in the Yolo flood bypass, where longer residence time appears to increase productivity. Several studies have concluded that numerous factors influence the relationship between residence time and phytoplankton biomass including growth and loss rates, nutrient balance, abundance of grazers, turbidity, temperature, and other factors.

The Bay-Delta estuary is a tidal environment dominated by instantaneous flows, which are largely beyond human control and exert a much greater influence than net flows, which are the basis for a number of regulatory requirements in the Bay-Delta. During the summer, tidal flows in a typical 25-hour tidal cycle can be greater than 300,000 cubic feet per second (cfs) compared to a net inflow of between 5,000-10,000 cfs. Because of these very large tidal flows, fish and other aquatic organisms have adapted strategies that allow them to maintain position or move around in the estuary, as evidenced by their continual presence in the face of large instantaneous flows. This is even the case for "weak swimmers," like delta smelt and their prey items.

The observed association between the abundance of some fish species and flow (as represented by the position of the X2 isohaline in the estuary) has led to the resource management proposition that more flows though the Delta will produce increases in fish abundance. However, a causal link between increased flow and fish abundance has not been established.

In many estuaries, freshwater flow regulates the location of the estuarine turbidity maximum (ETM) or entrapment zone where suspended particles, phytoplankton, and zooplankton accumulate to create a biologically rich aquatic environment. The establishment of the spring X2 outflow was largely based on the assertion that the location of the ETM and peaks in abundance of several desired species in the Bay-Delta occurs at 2 practical salinity units (psu). However, more recent research suggests that multiple ETMs exist in the Bay-Delta; they tend to be associated with channel bathymetry and bottom topography. Thus, unlike some other estuaries, locations of ETMs in the Bay-Delta may be largely decoupled from freshwater inflow.

Possible Explanation 2: Appropriation of water that serves to remove productivity from the system

Appropriation of water affects productivity in the Bay-Delta through three mechanisms. First, in-Delta water appropriations remove plankton biomass, both phytoplankton and zooplankton, from the Bay-Delta ecosystem. An analysis of phytoplankton mass balance found that net transport loss, including losses from outflow to Suisun Bay and with appropriated water, accounted for only 6 tons C day^{-1} compared to net production of 44-53 tons C day^{-1} and within Bay-Delta consumption of 38 to 47 tons C day^{-1}. While appropriations remove some of the carbon biomass, they appear to have little effect on the concentration of the remaining phytoplankton. Analysis of average March to June chl-*a* concentrations at plankton sampling stations in the central Delta show no apparent relationship between phytoplankton concentrations and rate of appropriations during this same time period.

Second, appropriation of water impacts residence time in the Bay-Delta by increasing flow rates through the Bay-Delta. Transport time can affect phytoplankton biomass; however, longer residence time does not necessarily translate to greater phytoplankton biomass. The spatial and temporal variations in growth and loss rates within the Bay-Delta are too great to predict with any accuracy the overall impact of changes in residence time.

Third, water appropriations might prevent plankton in the south Delta from reaching Suisun Bay. But, modeling results indicate that over 90% of the time, the San Joaquin River and other eastern tributaries combined contribute less than 10% of the total flow reaching Suisun Bay even with in-Delta appropriations by CVP and SWP completely shut off.

Possible Explanation 3: Increases in contaminants that have direct and indirect effects on the survival and health of aquatic organisms

The presence of a wide diversity of contaminants from both point and nonpoint sources has been documented over the last 25 years, primarily by the Central Valley Regional Water Quality Control Board and the Interagency Ecological Program (IEP). Several recent reviews conclude that contaminants have the potential to impact the ecosystem but must be considered in view of the complex interactions and uncertainties associated with potential exposures.

Contaminants enter the Bay-Delta estuary and its tributaries by runoff from urban and agricultural land uses, atmospheric deposition, municipal and industrial water treatment effluent, recreational and commercial boating activities, and from historic mining operations. Contaminant levels vary both spatially and temporally and many are highest following rain events (Kuivila and Hladik 2008). Several contaminants, including pesticides, metals, pharmaceuticals, personal care products and nutrients, have been detected in Bay-Delta estuary water and sediment and continue to be the focus of concern by regulatory agencies.

Possible Explanation 4: Increases in invasive species that have caused a significant decline in phytoplankton biomass with resultant impacts on higher trophic levels

There is universal agreement that the invasion by the Amur River clam has had a significant effect on chl-*a* levels in Suisun Bay, with its influence extending into the lower Sacramento and San Joaquin Rivers. The clams have been blamed for the density declines in diatoms, several zooplankton species, and various native fish species. Evidence exists that non-native species have contributed to changes to the Bay-Delta food web, a reduction in those species could increase productivity and populations of remaining, native species.

However, it appears that the arrival of invasive clam species does not fully explain the decrease in the Bay-Delta's productivity. Primary productivity, as measured by chlorophyll-*a* (chl-*a*) concentrations, began to decline prior to the arrival of the Amur River clam, has declined in seasons when clam biomass is low, and has continued to decline even where clam abundance has not. Phytoplankton levels have also declined in upstream parts of the estuary, in areas not affected by invasive clams, such as in the south Delta. The invasive aquatic plants, *Egeria densa*, may be one of the other factors suppressing productivity in the Delta. Egeria has been present in the Delta for about 50 years, but its coverage in channels and embayments began to expand significantly in the mid-1990s. The dense upper "canopy" formed by egeria blocks light that would normally be available to phytoplankton, thus contributing to reduced native fisheries production.

Efforts to control invasive species will need to be coordinated with efforts to address other factors suppressing productivity. While increasing flow has been proposed as a management tool to reduce the adverse impact of clams, the proposal is not supported by available science. Ecological responses in other ecosystems support the hypothesis that nutrient load reductions may reduce invasive clam abundance and improve ecosystem productivity.

Possible Explanation 5: Changes in nutrient concentrations, forms, and ratios cause changes in species composition and abundance at all trophic levels

Changes in nutrient loads are clearly impacting Bay-Delta ecosystem dynamics in complex ways that extend beyond eutrophication. In addition to increases in nutrients, changes in the form of available nutrients (chemical state, oxidized vs. reduced, organic vs. inorganic, dissolved vs. particulate) and the proportion of different nutrients produce adverse effects at both the scale of the primary producers and the entire ecosystem.

Total nutrient loads, including ammonium (mostly from a single source) have increased, and the relative proportion of ammonium (NH_4) to nitrate (NO_3) has changed. For decades, researchers have explored the relative use of – or relative preference for – different forms of nitrogen (N) by phytoplankton. Ammonium (NH_4) is generally considered to be the form of nitrogen preferred by phytoplankton because it requires less energy to assimilate than NO_3. However, it is also well documented that NH_4 can inhibit the uptake of NO_3, and thus can exhibit a strong negative control on phytoplankton productivity.

The effects of changes in the proportion of NH_4/NO_3 have been shown for the Bay-Delta in both field observations and lab experiments. One recent study identified a 60% decline in primary production in the Sacramento River below the Sacramento Regional Wastewater Treatment Plant, where NH_4 is discharged, compared to production levels above the Treatment Plant's outfall, and found evidence that high rates of carbon uptake are linked to phytoplankton NO_3 use. The increased proportion of NH_4 may help explain reduced primary production in Bay-Delta since the 1970s. The form of available nitrogen also affects phytoplankton species composition, as diatoms generally have a preference for NO_3, while dinoflagellates and cyanobacteria generally prefer more chemically reduced forms of nitrogen (NH_4, urea, organic nitrogen). Furthermore, the ratio of nitrogen to phosphorus (N:P) has doubled in the Bay-Delta over the last 35 years due to increased total nitrogen loads (from increased effluent discharge from wastewater treatment plants and other sources) and declining phosphorus loads (due to the regulation of detergents). A retrospective analysis of 30 years of data from the Bay-Delta found that variation in nutrient concentrations and ratios is highly correlated to variation in phytoplankton species composition and biomass at the base of the food web. Similar observations have been made in ecosystems elsewhere. The balance of N:P can affect other metabolic aspects of phytoplankton besides growth, including toxin production, cell membrane thickness, and other chemical constituents. Several ecosystems elsewhere have seen a resurgence of some native species and a decline of some invasive species, including invasive clams, following reductions in nutrient loads and a restoration of the N:P balance.

Possible Explanation 6: Changes in the land-water interface, including loss of floodplains and loss of tidal wetlands

The predevelopment Bay-Delta was an extensive, complex, and diverse environment, with narrow, meandering, and sinusoidal channels. The massive dredging of the Sacramento River for flood control in the 1920s deepened, widened and straightened the river, resulting in profound changes in the bathymetry and regional hydrodynamics. From 1860 to 1930, approximately 400,000 acres of tidal marsh were converted to Delta farm land, thereby cutting off the tidal prism. Miles of dendritic channels were eliminated and replaced with deep channels with lesser bathymetric diversity. These profound changes to the physical environment resulted in the hardening of the land-water interface, thereby isolating large geographic areas from natural tidal action and flood events.

In the last 150 years, approximately 95% of the tidal wetlands in the Delta have been lost. Shallow-water areas contiguous and adjacent to tidal wetlands and freshwater marshes support high phytoplankton growth rates and produce high quantity and quality productivity and can serve as relatively food-rich areas for desired fishes. Floodplains in the Bay-Delta estuary and in the watersheds of the Sacramento and San Joaquin rivers can produce high levels of phytoplankton and other algae, even after the short-duration flooding that occurs in the spring. Shallow water depth and long water residence time on floodplains facilitate settling of suspended solids, resulting in reduced turbidity and increased total solar radiation available for phytoplankton growth. Periodic small floods boost aquatic productivity of phytoplankton by delivering new pulses of nutrients, mixing waters, and exchanging organic materials with the river.

Reduction in primary productivity resulting from losses of wetlands across much of the Bay-Delta estuary is recognized by agency biologists as a key determinant of declines in zooplankton and the native fish that prey on them. While changes in the land-water interface from decades ago may not explain the most recent fish declines, they contribute to the overall low productivity of the Bay-Delta.

ES3. Changes to the Delta's Physical Landscape

Before European settlement, the Sacramento and San Joaquin Rivers flowed through approximately 400,000 acres of wetlands and other aquatic habitats in the Bay-Delta. The primary landscapes included flood basins in the north, tidal islands in the central Bay-Delta, and a complex network of channels formed by riverine processes in the south.

In the past 160 years, approximately 1,335 miles of levees were constructed and in-Delta channels were widened, straightened, deepened, connected, and in some instances gated, which have collectively altered the pattern and extent of diurnal tidal flows. Most upstream rivers and many of the contributing streams have been modified with dams, diversions, or other “improvements” that have separated channels from their floodplains, changing inflow patterns, and reducing sediment and nutrient inputs to the ecosystem.

The historical changes in the Bay-Delta landscape have affected more than just food web productivity. The complex assemblage of floodplains, freshwater and tidal wetlands, open water and upland habitats historically provided valuable space for rearing, spawning, migration, and refuge from predators for both aquatic and terrestrial species. The extensive changes to the Delta landscape have reduced, fragmented, and isolated these habitats. While land and water were once intricately connected, in the current Bay-Delta landscape, levees maintain complete separation.

ES4. Changes in Temperature

Although annual trends in water temperature have not been observed in recent decades, significant changes in average monthly temperatures have been observed between 1983 and 2007. Climate change is expected to result in further increases in water temperature in the estuary. Cold-water reservoir releases have been used for decades to provide temperature refugia for salmonids; however, climate change could result in a decrease in cold-water pools in upstream reservoirs as the contribution of snowmelt to mountain runoff declines. Additionally, the number of areas experiencing temperatures above lethal ranges for native species is expected to increase. Increased temperature could adversely affect aquatic invertebrates and alter wetland plant communities by causing changes in available carbon.

Water temperatures provide an important constraint on ecological function, including effects on aquatic invertebrates and effects on fish spawning, swimming performance, metabolism, and mortality. The biological implications of climate change effects on water temperatures may be profound, including increasing risk of extinction of native species and increasing dominance of nonnative species.

Water temperatures in the Bay-Delta are primarily driven by atmospheric influences, although thermal dispersion also influences water temperatures, and bathymetric features can influence site-specific water temperatures. Reservoir releases will be unable to affect water temperatures in the Bay-Delta during the warmer summer and fall seasons when cooler water temperatures are most needed.

ES5. Changes in Turbidity

Monitoring by the California Department of Fish and Game (DFG) and the IEP in the Bay-Delta estuary over the past 35 years has documented trends in increased water clarity, reduced turbidity, and declines in chl-*a*. The decline in turbidity has undergone what has been characterized as a significant step decrease, with turbidity in the San Francisco Bay (expressed by suspended sediment concentration) decreasing 36% between water years 1991-1998 and 1999-2007. That decline is thought to have resulted from a shift from depositional to erosion-generating processes in Suisun, San Pablo, and central San Francisco bays. These changes have significant implications for several fish species, including delta smelt, as turbidity appears to be a critical factor for delta smelt larval feeding. With decreasing turbidity, more light penetration could increase primary productivity.

Reservoirs on the major tributaries have reduced sediment input to the Bay-Delta and the sediment transport capacity of channels below these reservoirs decreases over time as the channels become incised and armored. However, while suspended solids concentrations in the Bay-Delta rise following significant rainfall, releases from upstream reservoirs are not an effective means of delivering suspended sediment to the Delta.

ES6. Changes in Through-Delta Flows

The 2010 State Water Board Flow Criteria Report (2010 Flow Criteria Report) recommends flow standards based on a percentage of unimpaired flows. That approach does not recognize that "unimpaired flow" is a calculation of a hypothetical condition that never existed in the Bay-Delta system. Previous analyses of natural flow indicate that outflow in the historic undeveloped environment are lower, likely substantially, than the unimpaired flows previously considered by the State Water Board as representative of "natural" conditions. Further, the mechanisms through which freshwater flow contributes to desired characteristics in a largely unaltered system are different from than those that would be provided by flows in the highly altered Bay-Delta estuary. In addition, most of the literature and examples of application of a natural flow regime approach are from riverine systems, not systems like the Bay-Delta estuary. Complex ecological and biological processes occur within estuaries, primarily due to their dynamic nature, complex freshwater-seawater mixing processes, random influences, antecedent conditions, and complex ecological linkages.

ES7. Conclusions

There is widespread agreement that adverse changes in the Bay-Delta food web are driven by nutrients, invasive species, changes in landscape attributes at the land-water interface, and potentially contaminants. Several environmental stressors can be feasibly addressed by water quality objectives in the Bay-Delta plan and with implemented actions by the State Water Board and other agencies. Such actions could directly address underlying causes and reduce or reverse adverse changes that have impacted the lower levels of the Delta's food web.

Conversely, the science does not support increasing inflow or outflow as a way to improve the health of the ecosystem. The causal links between flow and fish abundance are largely unknown, and there is insufficient evidence to rely on increased flow as a tool to increase fish abundance.

This Page Intentionally Left Blank

1 Introduction

This paper provides the State Water Board with a technical assessment of available scientific information regarding five aspects of ecosystem change that biologists contend have contributed to the current health and integrity of the Bay-Delta estuary:

- Changes to the composition of the food web,
- Anthropogenic changes to the physical landscape,
- Warming of water temperature,
- Reduced turbidity, and
- Changes to flows and the location of the low-salinity zone in the estuary.

Within the discussion of changes to the composition of the food web, this paper describes six factors that may explain all or a portion of the changes: flow, water diversions, contaminants, invasive species, nutrients, and physical landscape.

The scientific community agrees that the San Francisco Bay-Sacramento/San Joaquin River Delta (Bay-Delta) “is one of the most highly modified and controlled estuaries in the world” (Moyle et al. 2010). The same is true for many of the streams and watersheds that feed the estuary. Over the last 150 years, more than 95 percent of the original wetlands, floodplains, and riparian habitats have been destroyed. Channels have been widened, straightened, deepened, connected, and regulated with levees and gates. Rivers tributary to the Bay-Delta have been dammed and flows manipulated. Hydraulic mining has had lasting effects on sediment dynamics. Non-native species have been introduced and have become well established. The human population has grown considerably with resultant land use changes, increases in the demand for water, and increases in pollutant loads in waterways. The climate has changed and is changing, sea level is rising, and ocean conditions have fluctuated, and many of these elements will continue to change into the future. There have been winners and losers among the Delta’s native plants and animals as a result of these changes. Unfortunately, many of the estuary’s native fish, including delta smelt and longfin smelt, salmonids, and sturgeon have been the losers, while clams, invasive zooplankton, and predatory warm-water fish have been the winners.

Because of the complexity of changes that have occurred in the Bay-Delta estuary, protection of water quality for beneficial uses cannot be realized through water quality objectives that address a single parameter or through authority held by a single regulatory agency. The solution, like the problem, will be need to be comprehensive. So many aspects of the Bay-Delta and its watershed have changed that solutions such as mandating a percent of the “natural flows” will neither restore “natural conditions” nor address many of the key stressors; a more comprehensive approach is necessary. Regulation of water appropriation has been an important tool to provide protection for beneficial uses within the Bay-Delta estuary. This paper does not dispute that. Instead, this paper considers the role of flow (including regulation of water appropriation) in rivers and estuaries. It ultimately reflects the existing science that shows the important function natural flow provides to rivers and estuaries, but that significant changes in the regulation of appropriations should not occur until other physical, chemical, and biological ecosystem changes are addressed.

This Page Intentionally Left Blank

2 Major Changes to the Delta Food Web

Summary: The best available scientific evidence indicates that changes to the composition of the food web in the Bay-Delta are the primary factors that have driven historic and recent changes to the Bay-Delta ecosystem. Published literature and data from the Bay-Delta estuary indicate that addressing food web changes should benefit stressed species. There are a number of possible explanations for the changes.

1. **Flow:** There is general agreement that in an unaltered estuarine ecosystem, freshwater inflow can be an important factor in primary and secondary productivity (plankton production). However, in the highly altered Bay-Delta ecosystem, there is no indication that a linear or simple relationship between flow and plankton productivity exists. And, because the Bay-Delta estuary has been so extensively altered, regulation of flow must be carefully considered in spatial and temporal contexts. Regulation of flow must be based on specific flow functions and considered in context with co-occurring changes to localized functions and processes.
2. **Appropriation of Water:** While water appropriations remove plankton from the ecosystem, there are no studies that demonstrate appropriations of water impact on overall productivity. Water appropriations alter patterns of plankton transport, and models indicate that completely ceasing in Delta appropriation by CVP and SWP will not alter the transport of plankton to food limited areas.
3. **Contaminants:** Many contaminants have been detected in Bay-Delta water and sediment, including pesticides, metals, pharmaceuticals, personal care products, and nutrients. Published literature suggests that contaminants have the potential to impact the ecosystem but must be considered in view of the complex interactions and uncertainties associated with potential exposures. While it is clear from studies to date that potential contaminant effects are important to the health of the Bay-Delta ecosystem, it is equally clear that it is important to recognize the uncertainties and complexity of these relationships.
4. **Invasive Species:** The invasive Amur River clam (*Potamocorbula amurensis*[10]) has reduced plankton biomass levels in Suisun Bay and the lower Sacramento River. However, the Amur River clam cannot explain the entire decline in phytoplankton biomass, which began before the clam became established in these bodies of water. In addition, bivalve abundance in upstream regions and in the spring insufficiently explains phytoplankton biomass declines that have been observed in those areas and times.

   While increasing flow has been proposed as a management tool to reduce the adverse impact of clams, the proposal is not supported by available science. Ecological responses in other ecosystems support the hypothesis that nutrient load reductions may reduce invasive clam abundance and improve ecosystem productivity.
5. **Nutrients**: The scientific literature shows that nutrient loads affect ecosystem dynamics in complex ways that extend beyond our historic understanding of eutrophication. In the Bay-Delta, total loads and the forms and relative proportions of nutrients have changed over time. Ammonium loads have doubled, ammonium-to-nitrate ratios have increased, and nitrogen-to-phosphorus ratios have doubled. A review of the science in this and other estuaries indicates that these changes may have had profound effects on the Bay-Delta ecosystem by altering and suppressing the food web that

---

[10] Also referred to as *Corbula amurensis*, *Corbula,* overbite clam, and Amur River clam.

supports native fish species. Other ecosystems also provide examples of increasing native species and declining invasive species following restoration of balanced nutrient conditions.

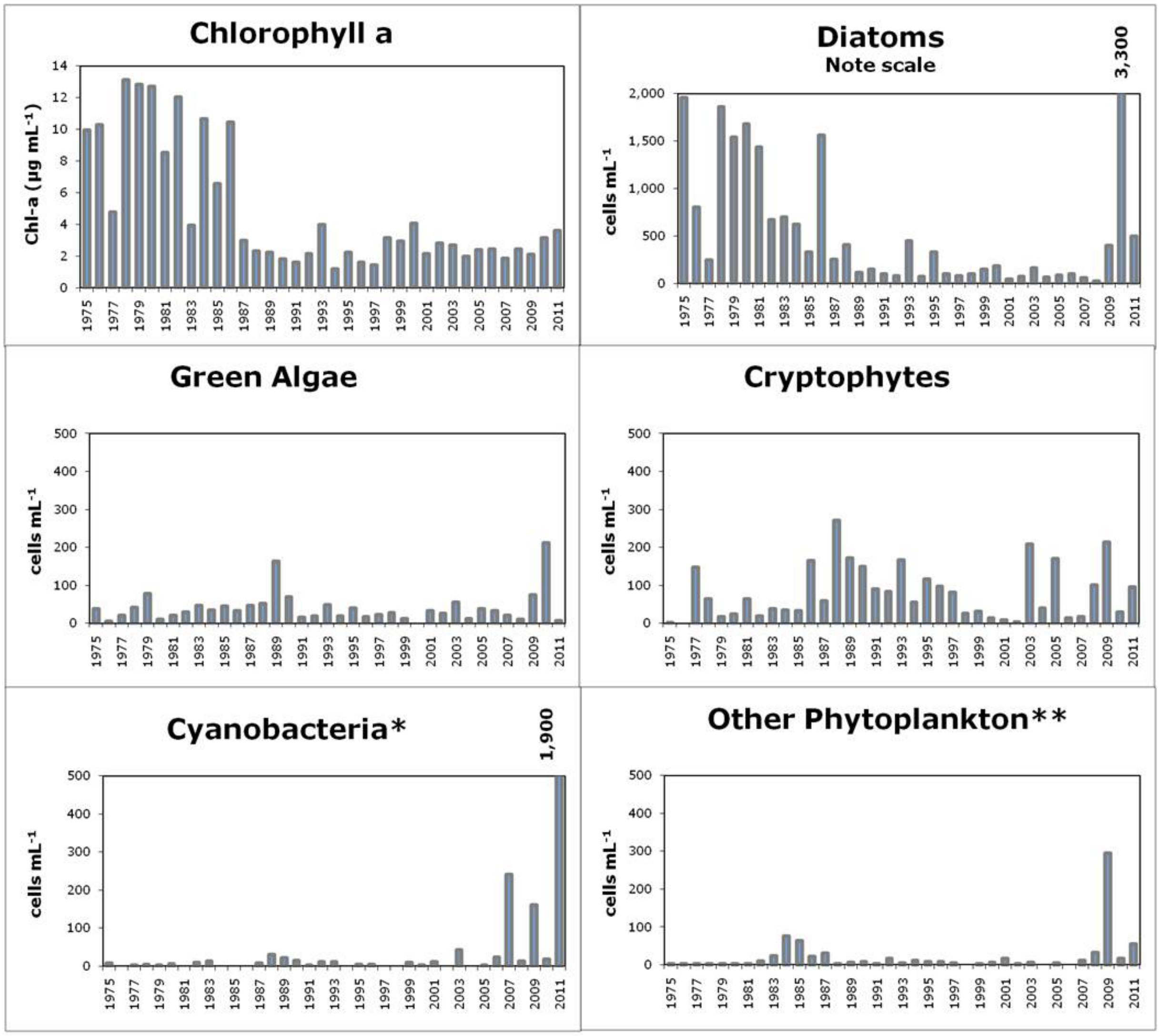
6. **Physical landscape:** The rich diversity and abundance of the aquatic and terrestrial life in the predevelopment Bay-Delta estuary and its tributaries supported dense fish populations. Now the historic connections between land and water have been severed by development, construction of dams, and the draining and diking of wetlands behind a vast network of levees. While changes in the land-water interface from decades ago cannot explain recent fish declines, they do play a role in the overall low productivity.

   The restoration of large floodplain areas could allow for inundation periods that would maximize food web productivity and provide ecosystem benefits at the population level, especially if coupled with measures to restore the nutrient balance.

2.1 Changes to the Food Web

There is agreement in the scientific community that the composition of the food web in the Bay-Delta now differs from that of the past. The dominant phytoplankton species have shifted from diatoms to less nutritious – and sometimes toxic – algal species. The larger calanoid copepods that have been identified as important prey for delta smelt and longfin smelt are now outnumbered by smaller cyclopoid copepods. There is scientific agreement that changes in primary and secondary production (phytoplankton and zooplankton) have had significant effects on the abundance and distribution of several species, such as the northern anchovy (*Engraulis mordax*), mysid shrimp (*Neomysis mercedis*) and striped bass (*Morone saxatilis*). There is also scientific agreement regarding some of the causes of these changes to the food web, such as the invasion by the Amur River clam (*P. amurensis*) in the past, and changes to the Delta landscape in the long term. This report presents the compelling science that reflects that changes at the base of the food web have had detrimental effects on delta smelt and longfin smelt abundance.

There have been changes at all levels of the Delta food web. Primary productivity and phytoplankton biomass in the Delta, as measured by chlorophyll-*a* (chl-*a*) concentrations, are among the lowest of all estuaries studied (Jassby et al. 2002). In fact, chl-*a* decreased significantly in each season except spring (April through June) from 1975 to 1995 (Jassby et al. 2002, 2003), and remains low (Kimmerer 2004). Laboratory experiments suggest that Delta-wide chl-*a* levels are now so low that they are limiting zooplankton abundance (Müller-Solger et al. 2002). Not only has total biomass changed (as measured by chl-*a*), the Delta's algal composition has shifted over time from diatoms to smaller, less nutritious taxa such as dinoflagellates and cyanobacteria (Lehman 2000; Lehman et al. 2005; Lehman et al. 2010; Jassby et al. 2002; Sommer et al. 2007; Glibert et al. 2011; Winder and Jassby 2010). In a retrospective analysis of data collected from 1975-2005 from the Suisun Bay and Sacramento River regions, Glibert et al. (2011) found, "the declines in chlorophyll-*a* and diatoms over the entire time course were significant (R2 = 0.57 and 0.68, p <0.01), as was the increase in dinoflagellates (R2 = 0.30, p 0.05)." Time trends in phytoplankton biomass and species density are shown on Figure 1.



*Cyanobacteria does not include *Microcystis aeruginosa* and other picocyanobacteria that are not sampled accurately by the standard phytoplankton surveys. *M. aeruginosa* blooms were first observed in the Delta in 1999, and have since been occurring with increasing frequency and magnitude (Lehman *et al.,* 2005).

**Other phytoplankton is the sum of Chrysophyta, Euglenophyceae, Haptophytes, and Dynophyceae.

Figure 1. Change in the average March-October concentration of chl-*a* (µg L−1) and abundances of the major classes of phytoplankton and cyanobacteria (cells mL−1) over time. The Delta's algal taxonomic composition has shifted over time from diatoms to smaller, less nutritious species such as dinoflagellates (shown in the other phytoplankton graph) and cyanobacteria (Source: Environmental Monitoring Program, stations D4, D6, D7, D8 in Suisun Bay, lower Sacramento River and lower San Joaquin River).

As reported by Cloern (2001), "changes in phytoplankton species composition can directly impact the population dynamics of metazoan consumers that utilize the phytoplankton as their primary nutritional source." Evidence suggests that this is the case in the Bay-Delta. As shown on Figure 2, the abundance of many zooplankton species has also changed over time, with declines observed in *Eurytemora, Neomysis,* and rotifers, increases in *Acartiella*, and *Limnoithona,* and fluctuating abundances of *Pseudodiaptomus*, *Sinocalanus,* and Harpaticoids. In fact, there are very few zooplankton species that have not experienced a significant change in abundance since monitoring began (Winder and Jassby 2010).

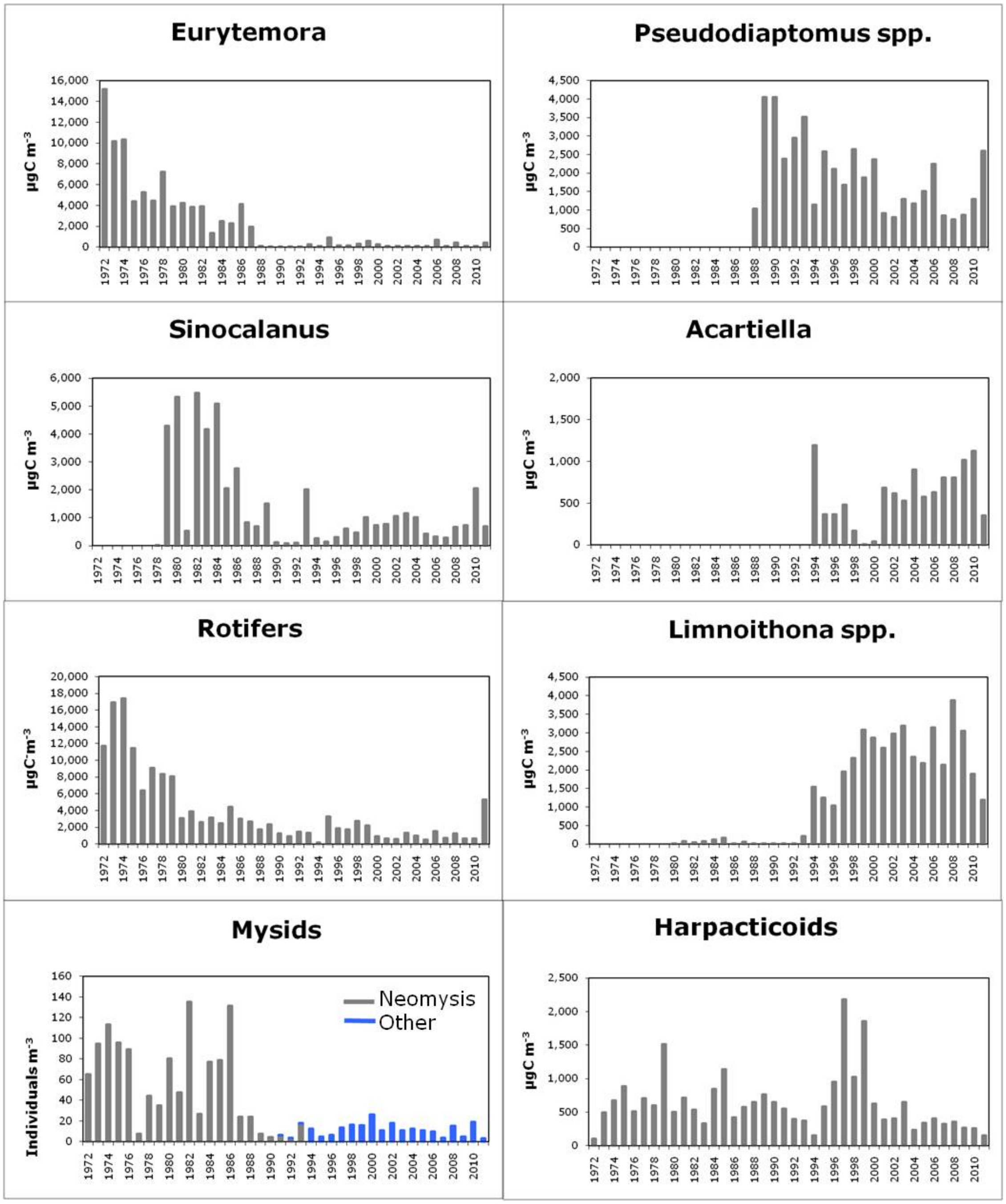


Figure 2. **Average March-October density of zooplankton over time showing significant changes in community composition. (Source: IEP zooplankton survey data from all Delta and Suisun Bay stations with a long continuous record of sampling. Survey counts were converted to biomass using NCEAS C per individual. Biomass was multiplied by volume of region and averaged across regions to obtain average annual volume weighted densities). Note y-axis scale differences.**

When considering trends in Delta fish abundance as a whole, it is apparent that the Delta fish community has not crashed – it has shifted composition. Just as in the lower trophic levels, there have been clear winners and losers in the fish community (Figure 3). Based on California Department of Fish and Game's (DFG's) fall mid-water trawl and summer townet surveys, some native fish species have declined and have been listed as threatened or endangered under the Endangered Species Act. However, the abundances of some fish species have increased over time, including non-native species that are popular for sport fishing, such as largemouth bass and sunfish.

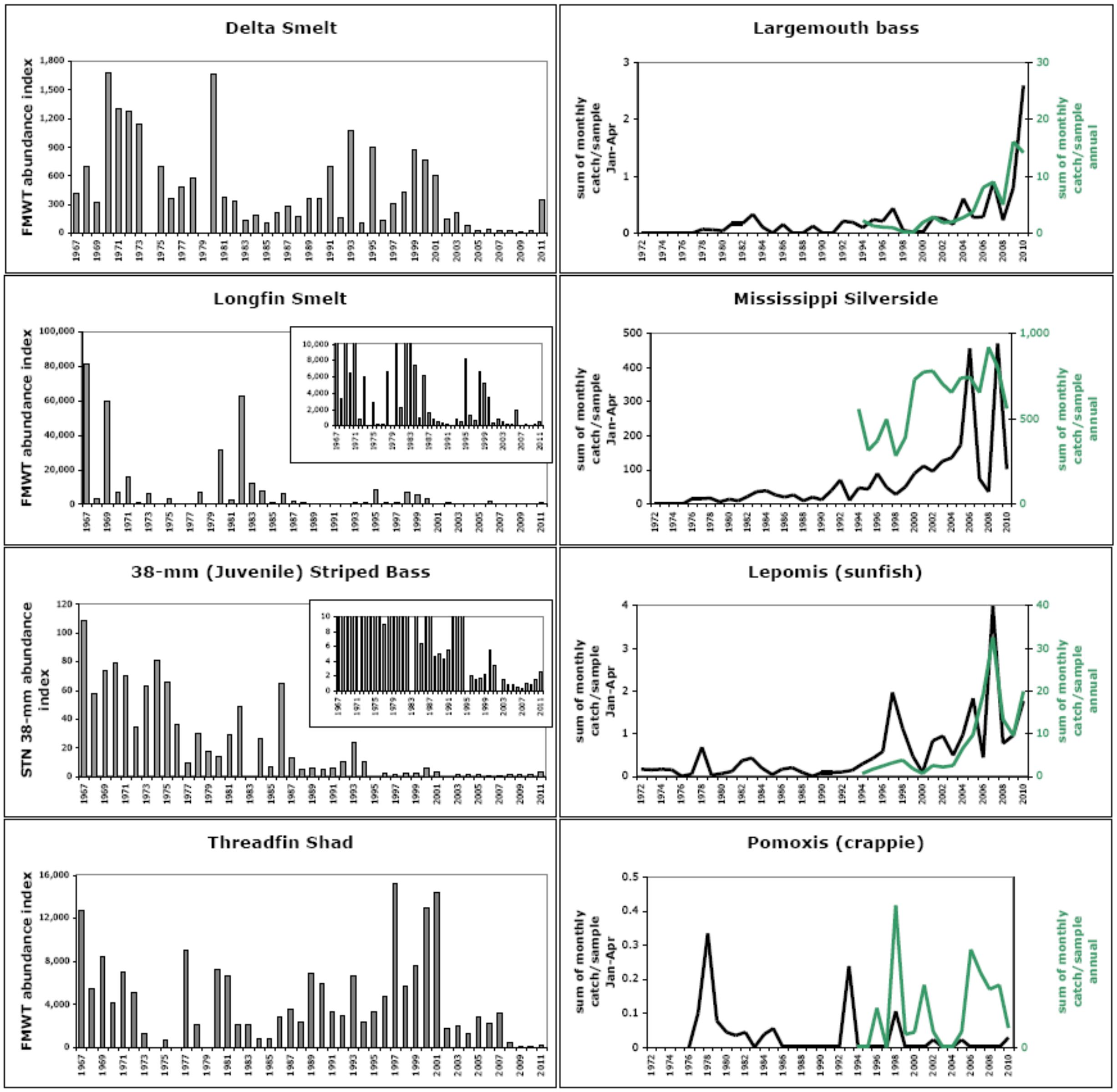


Figure 3. **Trends in fish abundance over time. Just as in the lower trophic levels, clear winners and losers have emerged in the fish community. (Source: Data for delta smelt, longfin smelt and threadfin shad come from DFG's Fall MidWater Trawl Index. Data for juvenile striped bass are from DFG's Summer Tow Net Survey. Data for the panels in the column on the right are from Beach Seine surveys at stations located in the Delta.**

Unfortunately, the winners have predominantly been non-native and littoral fish (fish that occupy near-shore areas), while the native and pelagic fish (fish that occupy open waters) have been the losers. There is evidence that this divergence is due to separate food pathways for the pelagic versus littoral fish (Grimaldo et al. 2009). Results of fish surveys in the Delta have shown a trend of increasing abundance of non-native fish inhabiting the Delta, particularly largemouth bass and sunfish (Figure 4). The Delta is now considered to be a world class largemouth bass fishery, with thousands of anglers and nationally televised tournaments (e.g., Bass Masters), as well as local and regional tournaments occurring throughout the year.

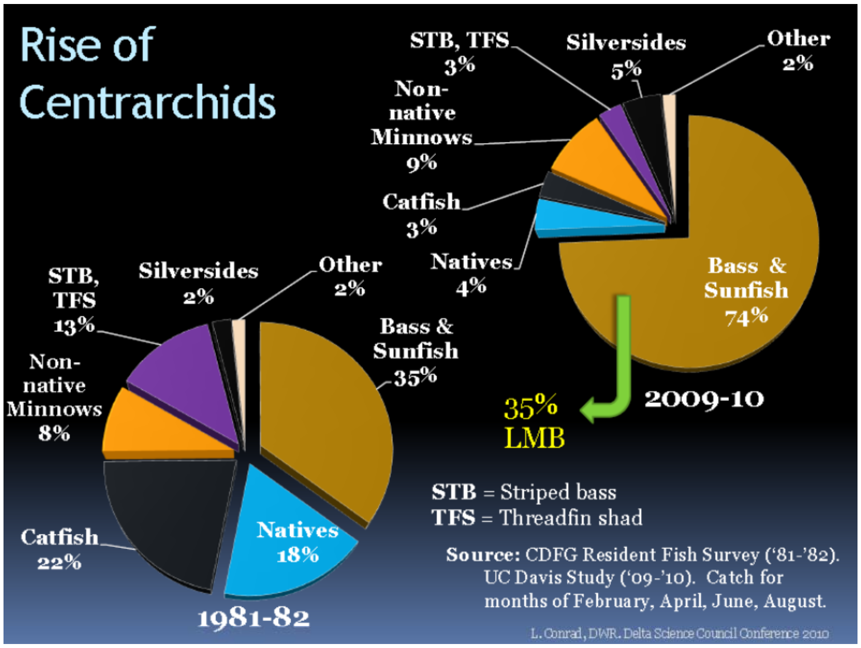


Figure 4. Change in fish species composition in surveys conducted in 1981-1982 and 2009-2010, showing a trend of increasing abundance of non-native fish inhabiting the Delta (Source: Conrad et al. 2010b).

In addition to demonstrating the increasing trend in bass abundance, fish surveys have also shown that the size of largemouth bass inhabiting the Delta has increased in the past decade, with an increase in the occurrence of bass in the size classes from 300 to greater than 500 millimeter (mm) in the population (Figure 5). Increases in both bass abundance and size in recent years reflect the favorable habitat conditions and prey that are available in the Delta for these fish. Increases in submerged aquatic vegetation in the Delta in recent years have been identified as a factor contributing to increased habitat and foraging opportunities for largemouth bass and sunfish (Conrad et al. 2010a, b; Conrad et al. 2011). Using stomach content and stable isotope analyses, Grimaldo et al. (2009) found evidence of two separate food-web pathways, one for pelagic fish and one for littoral fish such as bass and sunfish. They state, "[t]his apparent shift to grazer amphipods may partially explain why centrarchids in the Delta have increased in abundance over the last 2 decades (Brown and Michniuk 2007), whereas declines in pelagic production have apparently had adverse consequences for pelagic fish populations in the estuary (Sommer et al. 2007)."

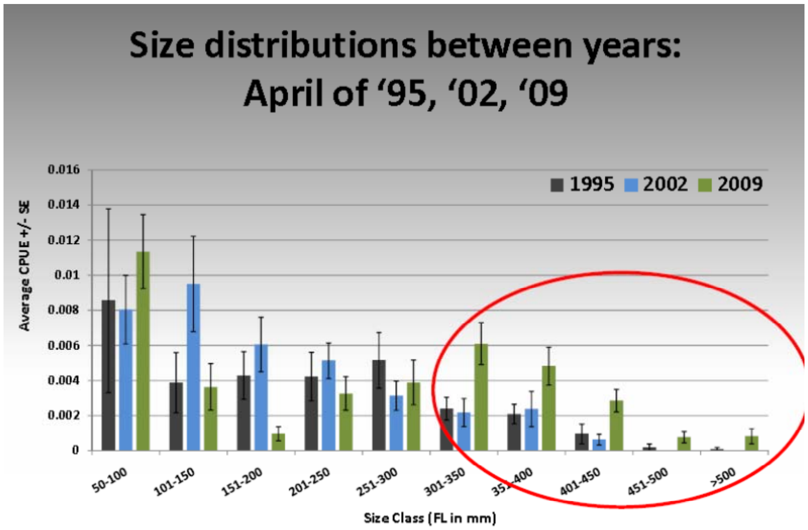


Figure 5. **Length frequency trends in largemouth bass collected in the Delta showing the size of largemouth bass inhabiting the Delta has increased in the past decade with an increase in the occurrence of bass in the size classes from 300 to greater than 500 mm in the population. (Source: Conrad et al. 2010a).**

2.2 Biological Implications of Changes in the Food Web

Research has established that a strong link exists between food availability and the growth and survival of fish species. In recent studies focused on the Bay-Delta, researchers now hypothesize that food quantity and quality are limiting the growth and survival of several fish species in the Bay-Delta. Winder and Jassby (2010) state, “Low food supply combined with changing food quality likely translated into reduced growth and survival of pelagic fish and affected their long-term and more recent recruitment success.” Rosenfield and Baxter (2007) state that, “Food limitation is consistent with our finding of reduced age-class 1 productivity and the disproportionate reduction in age-class 2 recruitment” for longfin smelt. Slaughter and Kimmerer (2010) state, “The combination of low primary production and a long and inefficient food web have likely contributed to declines of pelagic fish.” Jassby et al. (2002) conclude, “Overall, the Delta lost 43% in annual primary production during the period [1975-1995]. Given the evidence for food limitation of primary consumers, these findings provide a partial explanation for widespread Delta species declines over the past few decades.” Kimmerer et al. (2012) explain, “it seems likely that the persistently low productivity at the base of the food web, particularly for larger cells, has affected higher trophic levels.” And, using a multistage life-cycle model, Maunder and Deriso (2011) found that food abundance was one of the most important factors controlling the population dynamics of delta smelt – a result confirmed by Miller et al. (2012).

The link between food availability and fish abundance has been extensively studied. Nixon (1988) reviewed studies from freshwater, marine, and estuary systems around the world and reported a strong relationship between production at the base of the food web (primary production) and production of fish (fishery yield). In fact, Nixon (1988) states, “Although it has long been assumed that there was a strong link between primary production and the yield of fish from the sea, Fig. 6 is the first empirical demonstration that **such a link is strong enough to be seen (at least on a broad scale) against all the other variables that influence fish production**…” [emphasis added]. Figure 6, shows that as primary production increases so does fishery production. The figure shows that most estuaries (solid circles) are high in both primary and fishery production. Suisun Bay, where the Low Salinity Zone is often located, is a striking exception. Figure 6 shows primary productivity data from the Bay-Delta estuary superimposed on Nixon’s (1988) figure, showing low and declining primary productivity in the Bay-Delta.

With reference to the Nixon (1988) relationship, Kimmerer et al. (2012) conclude, "…the lack of a substantial commercial fishery in the San Francisco Estuary probably reflects the overall low productivity in this system."

A similar analysis to Nixon's that compares total fishery yield to primary production has not been conducted in the Bay-Delta estuary, but would likely add further support to the role of food in the decline of native fish. The State Water Board should commission such a study.

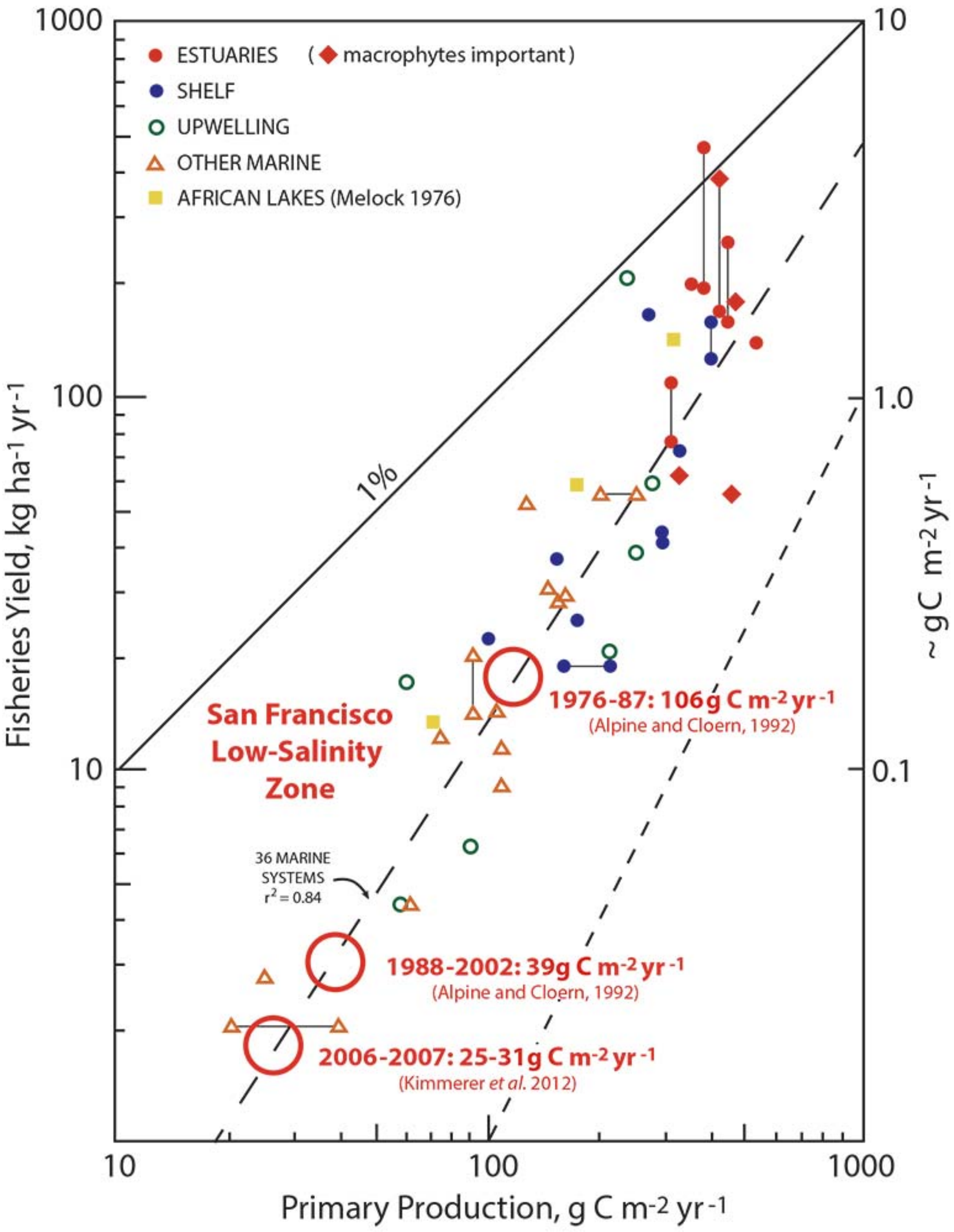


Figure 6. **Relationship between primary production and fisheries yield. Fishery yield increases with increasing productivity. (Source: Figure 6 from Nixon 1988, modified with data from Alpine and Cloern 1992 and Kimmerer et al. 2012).**

There are numerous examples in the Bay-Delta estuary and elsewhere demonstrating how species abundance and distribution changes in response to changing food resources, including species moving to new locations, declines in total abundances, or changes in feeding habits.

Higher trophic-level species, such as the northern anchovy, have been observed moving to a new location. In response to a significant and abrupt drop in phytoplankton biomass in the Suisun Bay area following the invasion by the Amur River clam, a redistribution of northern anchovy farther downstream occurred, reducing its summer abundance in the Suisun Bay and Delta by as much as 94% (Kimmerer 2006). Kimmerer explored several possible explanations for the dramatic and rapid redistribution of northern anchovy in 1987 and thereafter, including climate variability and biomass, catch, and abundance on the California coast. He concluded that the most parsimonious explanation for the change in anchovy distribution was a direct or indirect response to the decline in chl-*a*.

The shift in distribution of a fish population away from a region that had become inhospitable is not surprising. In the lower Hudson River, several open-water fish species shifted seaward following a reduction in chl-*a* concentration due to the introduced zebra mussel, *Dreissena polymorpha* (Strayer et al. 2004, cited in Kimmerer 2006). Similar behavioral shifts of northern anchovy in apparent response to chl-*a* concentration (or its covariates) have been noted off Baja California (Robinson 2004, cited in Kimmerer 2006).

The shift in distribution can also be vertical or lateral within the same areas. Work by Sommer et al. (2011) examines the distribution shift in striped bass in the Bay-Delta, stating:,

> “The survey data suggest a substantial long-term distribution shift away from channels and toward shoal areas. The hypothesis that young striped bass are under sampled by mid-water trawls is supported by modeling of demographic patterns, which showed that the decline in numbers of age-0 fish was not consistent with increasing trends in age-1 fish. We hypothesize that reduced food availability in pelagic habitat is a major cause of apparent behavioral shifts by age-0 striped bass and some native fishes.”

Declines in species population abundance in response to food limitation have also been observed. Orsi et al. (1996) concluded that food limitation is the primary mechanism of decline in the native mysid, *Neomysis mercedis*. Feyrer et al. (2003) observed that 8 of 13 fish species declined in abundance during the study period (1979-1983 vs. 1998-1999) had mysids as important components of their diets in the earlier time period, but not in the latter. Kimmerer et al. (2000) found evidence that carrying capacity for striped bass declined in relation to mysid abundance declines.

Some species change feeding habits. Feyrer et al. (2003) found that striped bass switched to piscivory at a smaller size when mysid abundance declined. Before chlorophyll-*a* declined, striped bass shifted to piscivory at 140 mm FL; after chlorophyll-*a* declined, striped bass shifted to piscivory at about 80mm.

In summation, a change in food availability can and does affect the production and distribution of organisms at all levels of the food web. This linkage between food availability and species abundance and distribution has been demonstrated in the Bay-Delta estuary, as well as aquatic systems elsewhere. This view is strongly reflected in a review of the scientific literature.

2.3 Possible Explanations for Observed Changes in the Bay-Delta Food Web

A number of possible explanations have been suggested to account for the observed changes to the composition of the food web at all trophic levels in the Bay-Delta. The most commonly mentioned are changes in flows and flow patterns, invasive species (particularly the Amur River clam), contaminants, changes in nutrients, and changes in the land-water interface. In this section, the paper explores those explanations and the available scientific information. This section of the paper also provides evidence that two or more of these explanations could operate in concert, thereby reinforcing biological and biogeochemical feedbacks.

2.3.1 Possible Explanation 1: Changes in flows and flow patterns affect primary and secondary productivity

There is general agreement that freshwater river flow is an important factor in primary and secondary productivity within an estuary. Freshwater flow into an estuary serves to: (1) provide bulk transport of river nutrients to the estuary, (2) resuspend nutrients within the estuary through gravitational circulation, and (3) enhance dispersion of nutrients down the estuary due to salinity stratification.

In this context, the distinction between rivers and estuaries is important; estuaries do not function the same way rivers do. Unlike rivers, “the physical, chemical, sediment, water quality and ecological processes within estuaries are exceedingly complex primarily due to their dynamic nature, complex mixing processes, stochastic influences, strong antecedent effects and the vast number of complex ecological linkages.”(Pierson et al. 2002). The complexity of the Bay-Delta estuary is no different.

While a number of regulatory requirements in the Bay-Delta are based on net flows, aquatic species inhabit environments dominated by “instantaneous” flows, which are far greater than net flows and largely beyond human control. Figure 7 shows the typical maximum flows over a 25-hour tidal cycle in summer conditions. Figure 7 also shows that tidal flows overwhelm the relatively small net inflow. The tidal flows can be greater than 300,000 cfs in the western Delta compared to net outflow of 5,000-10,000 cfs. Because of these very large tidal flows, fish and other aquatic organisms have adapted strategies that allow them to maintain position or move around in the estuary, as evidenced by their continual presence in the face of large instantaneous flows. This is even the case for “weak swimmers,” like delta smelt and their prey items.

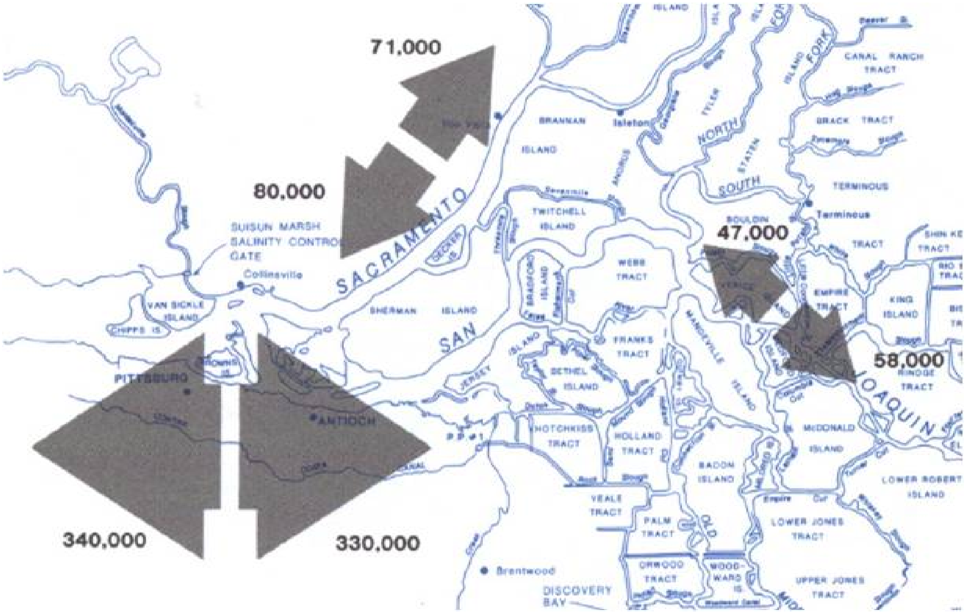


Figure 7. **Typical maximum flows over a 25-hour cycle in summer conditions, measured in cubic feet per second (cfs). The flow near Pittsburg during a typical tidal cycle can vary from 330,000 cfs upstream to 340,000 cfs downstream. The “net” summer Delta outflow is a very small amount of the total water movement, generally 5,000**

to 10,000 cfs. (Source: DWR 1995, p. 21).

Relationship between Freshwater Inflow on Primary and Secondary Production

In the Bay-Delta estuary, the relationship between freshwater inflow and primary and secondary production has been studied for decades. In the highly altered Bay-Delta estuary, neither a linear nor simple relationship between flow and productivity exists. In one of the early reports on the "Entrapment Zone", Ball and Arthur (1979) reported on the historic understanding of the association between phytoplankton and flow and flow patterns.

> "The quantity of river inflow to the Delta is important to phytoplankton growth in that it regulates nutrient concentrations, affects water transparency, determines water and phytoplankton residence times, and regulates the extent of salinity intrusion and the location of the entrapment zone. These and other factors all interact to determine the quantity and quality of the phytoplankton."

They also reported on associations between residence time and phytoplankton. In many estuaries, the effect of residence time on algal production is described as contributing to eutrophication (Pierson et al. 2002). Ball and Arthur found evidence of reduced residence time associated with increased algal growth in the south Delta in the San Joaquin River. They also found the same association in the north Delta, but at much lower chl-*a* concentrations.

In 1995, Jassby et al. reported a positive correlation between primary production in Suisun Bay and outflow, as measured by X2, from 1975 to 1989. However, they cautioned that the actual mechanisms were understood for only a few species, and they also noted uncertainty:

> "What are the causal mechanisms underlying these relationships? A variety of potential mechanisms deserves a detailed consideration that is beyond the scope of this study…"

They concluded:

> "By ignoring variables other than X2(or Q_{out}) we could therefore be in danger of imposing inappropriate standards, either too stringent or too lenient. The mere fact of a correlation between some ecosystem property and an indicator such as X, is therefore not sufficient grounds for using the indicator as a policy variable. The presence of much unexplained variation is one signal that an existing model can lead to unacceptably biased management policies, and should result in a search for alternative and additional variables."

Kimmerer (2002a) reexamined those relationships in Suisun Bay, expanded the data set to 1999, and incorporated the invasion of the Amur River clam as an additional environmental stressor. Kimmerer found no correlation between freshwater flow and chl-*a*. He found weak correlations between flow and several zooplankton species, and he observed chlorophyll-*a* and zooplankton populations decreases after the invasion of the clam around 1987.

Although both studies used chlorophyll-*a* data from the Suisun Bay location, their results are ambiguous for the time period prior to the invasion of the clam. One reason may be the year intervals used; 1975 to 1995 by Jassby et al. compared to 1975 to 1987 by Kimmerer. Another reason may be the interannual time period used; an annual time period by Jassby et al., compared to spring time and summer time periods by Kimmerer. Similar to Jassby et al., Kimmerer noted uncertainty:

> "Abundance or survival of several estuarine-dependent species also increases with freshwater outflow. These relationships to flow may be due to several potential mechanisms, each with its own locus and period of effectiveness, but no mechanism has been conclusively shown to underlie the flow relationship of any species."

Subsequently, Jassby (2008a), reported on phytoplankton abundance and growth in Suisun Bay and the Delta over the period 1975–2005. The author drew four major conclusions:

1. A long-term decrease in phytoplankton biomass occurred over the period 1975–2005.
2. A shorter-term increase in phytoplankton biomass occurred in the Delta, but not in Suisun Bay, from 1996–2005.
3. A change in the relationship between outflow and Suisun Bay phytoplankton biomass and productivity occurred after 1986 when "…Suisun Bay phytoplankton exhibited relatively low responsiveness to flow variability. This behavior differs from earlier chlorophyll-flow relationships reported in the literature. The reason appears to be the invasion of Suisun Bay by a clam—*Corbula amurensis*—in 1986, which has since maintained the phytoplankton community mostly at low levels by vigorous filter-feeding. In the past, flows into Suisun Bay generally diluted the higher phytoplankton concentrations within the bay; now they bring in higher phytoplankton concentrations from upstream."
4. The increase in phytoplankton biomass and productivity in the Delta was associated with reduced inflow to the Delta from 1996–2005. "The main source of interannual phytoplankton variability in the Delta during 1996–2005, including the upward trend, appears to have been freshwater flow variability and its effect on particle residence time." This is similar to the findings of Ball and Arthur (1979) three decades earlier, but is inconsistent with the majority of estuaries where increased flow is associated with increased phytoplankton production (Pierson et al. 2002; Kimmerer et al. 2012).

In a recent study, Kimmerer et al. (2012) were the first to examine the response of primary productivity within the Bay-Delta estuary across a wide range of freshwater flows during the spring-summer period of 2006–2007. Kimmerer et al. found that temporal and spatial variability in productivity was small. When the study combined data from 2006–2007 with long-term monitoring data, the data did not show any persistent patterns in production rates or biomass accumulation. More importantly, Kimmerer et al. (2012) found that, "**[p]roduction within the low-salinity zone was unresponsive to variation in freshwater flow,** in contrast to findings in other estuaries where nutrient loading drives variability in production and other regions of the [Bay-Delta estuary] where production responds to residence time or to stratification" (emphasis added). With respect to "other regions of the estuary where production responds to residence time", Jassby et al. (2002) and Jassby (2008a) found that greater productivity is related to longer residence time (lower flows).

There are also findings that longer water residence time does not necessarily translate to greater phytoplankton biomass; it depends on the phytoplankton growth and loss rates (Lucas et al. 2009). Loss rates are affected by numerous factors, including nutrient balance, abundance of grazers, turbidity, and temperature, among others. With high loss rates, lower residence time results in greater biomass accumulation. The complicated nature of the relationship between flow and productivity is reinforced by Kimmerer (2002a) who found that, "[i]n contrast with the higher trophic levels, chl-*a* and several species of zooplankton declined markedly after 1987, and had either weak responses to flow or responses that changed after 1987." Elsewhere, Kimmerer (2002b) states, "available evidence does not support flow responses of lower trophic levels." Figure 8 supports this finding; there does not appear to be a clear relationship between different measures of monthly flow and monthly chl-*a* concentrations.

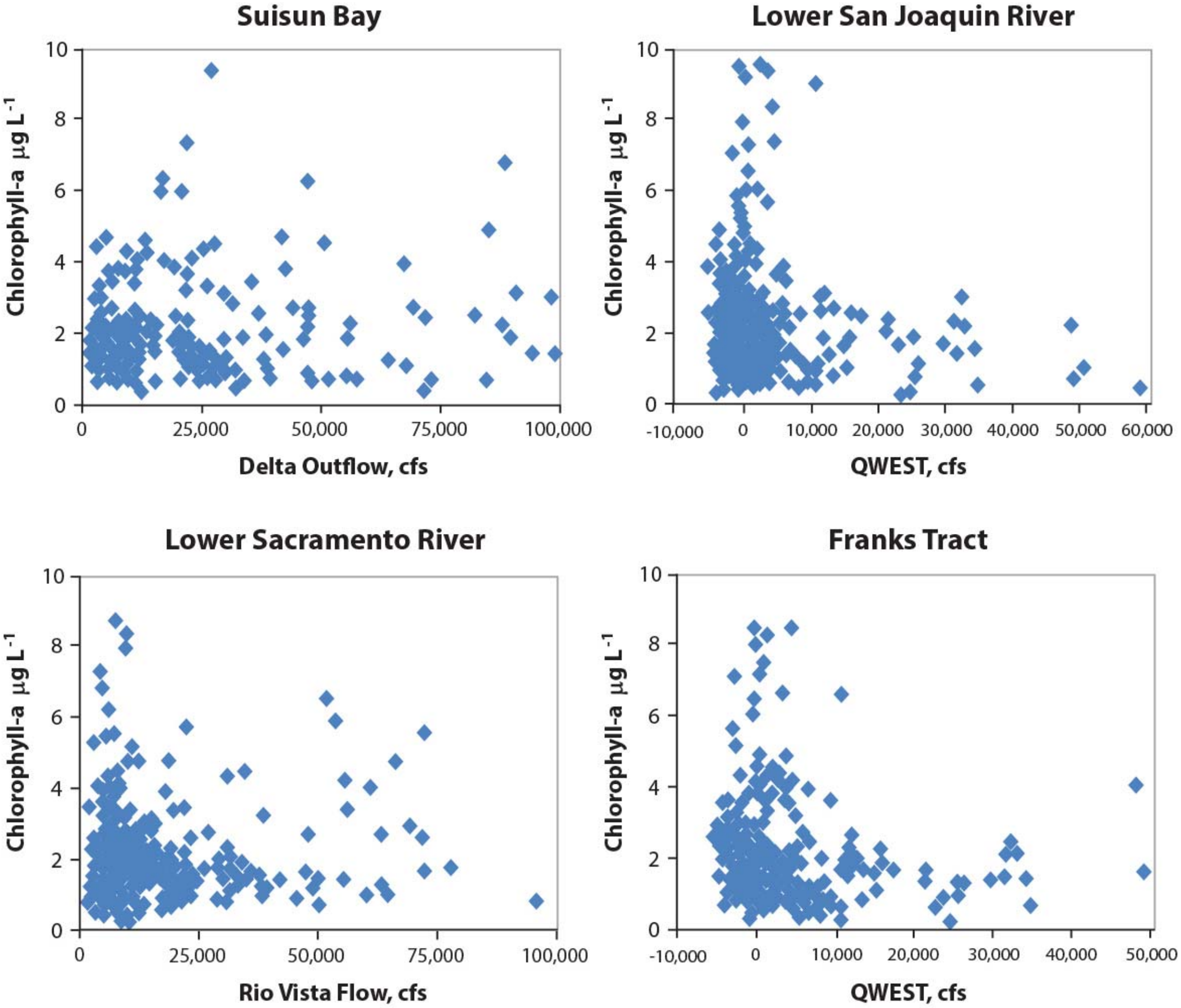


Figure 8. **Average monthly chl-*a* concentration plotted with different average monthly flow measures in four regions of the Delta, showing no apparent relationship (Source: Chl-*a* data from EMP monitoring, flow data from DAYFLOW, 1987-2011).**

Based on the long history of studies, significant uncertainty exists as to how flow affected primary production within the Bay-Delta estuary, and how flow will affect primary production in the future. What is certain is that a complex relationship exists between flow and productivity. Furthermore, any manipulation of flows must be carefully considered in its spatial and temporal contexts. Manipulation of flows should not occur for the sake of restoring "natural flows" per se; rather, such manipulation should be premised on restoration of localized functions and processes.

Importance of the Estuarine Turbidity Maximum to Primary and Secondary Production

In many estuaries, freshwater flow regulates the location of the estuarine turbidity maximum. The estuarine turbidity maximum, often referred to as the “Entrapment Zone,” is where suspended particles including phytoplankton cells and zooplankton individuals accumulate. It is a biologically rich aquatic environment (Ball and Arthur 1979; Kimmerer 1992). According to Schoellhamer (2001), several factors can contribute to the formation of estuarine turbidity maxima, including (1) gravitational circulation or tidal asymmetry of velocity, (2) cycles of local deposition, bed storage, and resuspension, and (3) suppression of turbulence by salinity stratification. The location of the Low Salinity Zone is just one of several factors that can influence the location of the estuarine turbidity maximum in an estuary. They can also be located at fixed locations.

The State Water Board established the Spring X2 outflow objective ndard in D-1641 to a large degree because it was believed that the location of the estuarine turbidity maximum and peaks in abundance of several species in the Bay-Delta occurred at 2 practical salinity units (psu) (Jassby et al. 1995). However, work published since that time by Burau (1998) and Schoellhamer (2001) note that locations of the estuarine turbidity maxima in the Bay-Delta are not defined by salinity. Schoellhamer (2001) concludes:

> “Salinity, bottom topography, and tides affect the locations of estuarine turbidity maxima (ETM) in northern San Francisco Bay. **ETMs are not associated with a singular salinity**.
>
> Bottom suspended-solids concentration (SSC) during cruises and tidally averaged **SSC did not show any maxima associated with a particular salinity**.
>
> Bottom topography, especially sills in the channels, is another factor controlling the location of ETMs in northern San Francisco Bay. Locations of ETMs are related to bottom topography because salinity stratification and gravitational circulation are enhanced seaward of sills.
>
> Wind-wave resuspension of bed sediment in shallow water subembayments is another topographically controlled source of suspended solids.
>
> Bottom shear stress and SSC are greatest during spring tides and smallest during neap tides. “[emphasis added]

These researchers found there are multiple areas of maximum turbidity that are primarily associated with the bathymetries of channels (Figure 9). Because bottom topography is important in determining the location of higher concentrations of suspended particles, these locations (“entrapment zones” or “null zones”) are more or less stationary, and cannot be greatly moved by flow, provided that the null zone is landward to the location in question (Burau 1998). In other words, unlike some other estuaries, locations of estuarine turbidity maxima in the Bay-Delta estuary are largely decoupled from freshwater inflow.

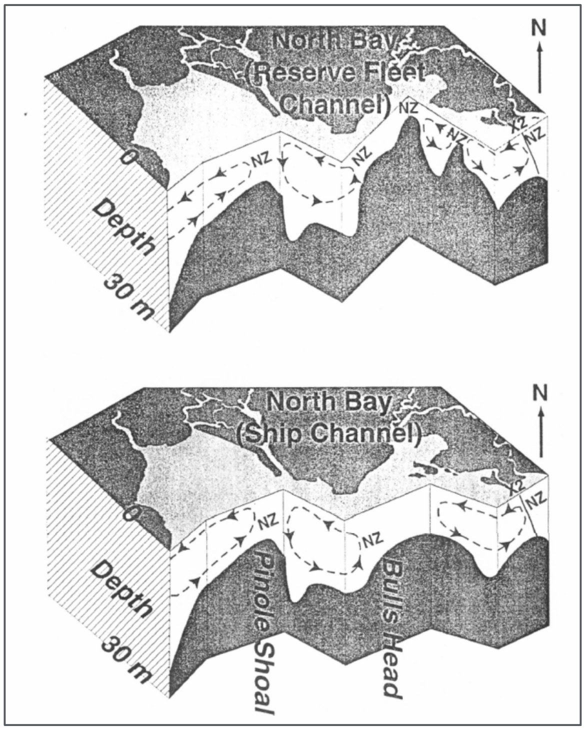


Figure 9. Conceptual model of gravitational circulation in the North Bay, based on the overturning time scale. Longitudinal slices are shown along the axis of (a) Reserve Fleet/Suisun Cutoff channel in the top graph and (b) the ship channel of Suisun Bay and the San Joaquin River in the bottom graph. Salient features of this revised conceptual model of gravitational circulation include the possibility of multiple gravitational circulation cells that terminate near restrictions in depth and a modulation of gravitational circulation cell strength with the spring/neap cycle. (NZ= Null Zone, M= Meters) Source: Burau 1998.

Affect of Freshwater Inflow on Primary and Secondary Production in Floodplains and Wetlands

Flow also affects productivity through inundation of floodplains and wetlands. Numerous, published studies link inundation of floodplains and wetlands to increased productivity, which lead to improvements in growth and survival of some fish species. As described in Section 3, the historic connections between these land areas and water have largely been prevented. In today's Bay-Delta, increased flow sends water through highly altered riverine systems (i.e., rock-lined channels), which, to prevent flooding, prevents water in the rivers from reaching floodplains or wetlands.

In summation, the relationship between flow and productivity is not unidirectional, is often weak, varies over time and space, and is complicated by the many alterations in the Bay-Delta estuary and its tributaries. The lack of response in primary and secondary productivity relative to freshwater inflow, including those observed by Kimmerer et al. (2012), contrasts with results from other systems. The lack of response demands careful consideration of the physical, chemical, and biological attributes of the Bay-Delta, specifically how freshwater inflow may affect food webs and subsequently higher trophic level organisms. Further, there is no science that considers the relative influence that changes in water appropriations may have on primary or secondary productivity in the Bay-Delta estuary, the location or extent of Estuarine Turbidity Maximum, or floodplain and wetland habitats.

2.3.2 Possible Explanation 2: Appropriation of water adversely affects the abundance of plankton (phytoplankton and zooplankton) in the Bay-Delta estuary

Appropriation of water can affect productivity through three mechanisms: (1) directly through removal of biomass, (2) indirectly by altering residence time, and (3) indirectly by altering transport pathways. Plankton biomass (phytoplankton and zooplankton) is certainly removed from the Bay-Delta via appropriation of water; the screens that reduce entrainment of fish at the State Water Project and Central Valley Project fish facilities are far too coarse to filter out plankton, and most other appropriations from the Delta are not screened at all. Jassby et al. (2002) conducted a phytoplankton mass balance assessment and determined that net transport loss, which included losses from outflow to Suisun Bay and with appropriated water, accounted for only 6 tons C day^{-1} compared to net production of 44-53 tons C day^{-1} and within Delta consumption of 38 to 47 tons C day^{-1}.

First, while appropriation of water may remove some of the Delta's carbon biomass, it appears to have little effect on the concentration of the remaining phytoplankton. Figure 10 presents the relationship between average March to June chl-*a* at plankton sampling stations in central Delta locations and export levels during this same time period. These data show no apparent relationship between phytoplankton densities, measured by chl-*a*, and export rates.

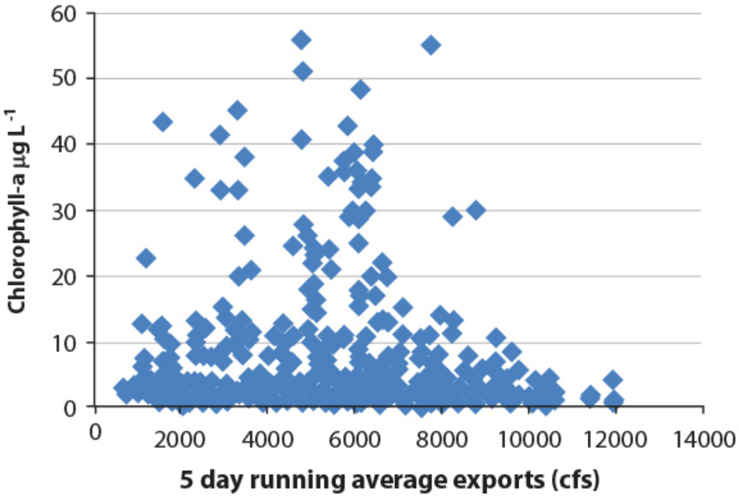


Figure 10. March – June average chl-*a* at zooplankton survey stations NZ080, NZ086, NZD16, NZD19, NZD28 versus 5-day running average export rates. These stations represent central Delta stations north of the export pumps during the period of the year when delta smelt are frequently present. These data show no apparent relationship between phytoplankton levels and export rates.

A second possible effect of appropriations of water is on residence time. Appropriations of water, particularly upstream of the Delta impact water residence time in the Bay-Delta estuary by increasing flow rates through the Bay-Delta. Transport time can affect phytoplankton biomass through the integration of growth and loss rates during the period of transport through the Delta. However, the relationship is not simple. Longer residence time does not necessarily translate to greater phytoplankton biomass; it depends on phytoplankton growth and loss rates (Lucas et al. 2009). The spatial and temporal variation in growth and loss rates within the Bay-Delta are too great to predict with any accuracy the overall effect of changes in residence time.

A third possible effect of water appropriations is on transport of plankton and other small particles from one area to another. For example, appropriations from the south Delta might prevent phytoplankton or zooplankton in the south Delta from reaching Suisun Bay. One hypothesis is that plankton, originating in the San Joaquin River part of the Delta, are removed by diversions and, therefore, do not subsidize plankton in downstream areas such as Suisun Bay. This hypothesis is based on the assumption that south Delta and San Joaquin River water with its higher plankton biomass is diverted directly to the pumps and therefore does not reach downstream areas.

The DSM2-QUAL model was used to simulate Delta flows under three scenarios: (1) historical hydrology, operations, and SWP and CVP exports (“historical exports”), (2) historical hydrology, operations, and no in delta SWP or CVP appropriations (“no exports”), and (3) historical hydrology, operations, and in-delta SWP and CVP appropriations limited to 50% of the Delta inflow (“reduced exports”). The percent of water originating in the eastern rivers (San Joaquin, Mokelumne, Cosumnes, and Calaveras rivers) that reaches Mallard Island, just upstream of Suisun Bay, was estimated for the 2001-2010 period (Figure 11).

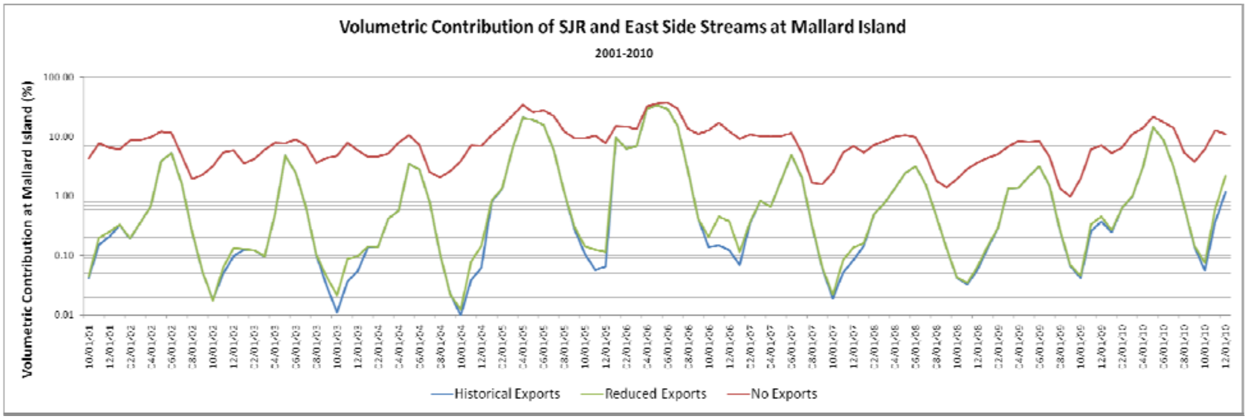


Figure 11. **Volumetric contribution of San Joaquin River and east-side streams at Mallard Island between 2001-2010. While the difference between the no exports scenario and the two with export scenarios is significant, even with exports completely shut off, over 90% of the time the San Joaquin River and other eastern tributaries combined contribute less than 10% of the flow at Mallard Island.**

Even with in-Delta appropriations by Central Valley Project and State Water Project completely shut off, over 90% of the time the San Joaquin River and other eastern tributaries combined contribute less than 10% of the flow at Mallard Island (percentage data not shown). The “no exports” scenario is included to emphasize the point that in-Delta appropriations have little effect. Reducing them to the minimum of historical exports or 50% of inflow makes very little difference. Assuming no loss of plankton due to predator grazing and senescence during transport, under the “no exports” scenario, plankton densities in the San Joaquin River region would have to be an order of magnitude greater than in the Sacramento River region to make a significant contribution to food availability in the confluence and Suisun Bay regions. Otherwise, increasing the contribution from this area would not have much effect.

The U.S. Fish and Wildlife Service’s (USFWS) biological opinion (BiOp) for continued operations of the Central Valley Project and State Water Project (2008) and DFG’s Ecosystem Restoration Program Conservation Strategy suggest that water appropriations may remove a disproportionately large share of plankton from the Bay-Delta, but neither source offers scientific support for this suggestion. A subsequent independent peer review of the effects analysis prepared for the BiOp to be unconvincing on this point (PBSJ 2008, p. 9). Court-appointed experts in litigation over the BiOp also addressed the suggestion and found that the BiOp showed no quantitative effects nor showed that entrainment of productivity could have effects at the population level in delta smelt (Quinn and Punt 2010). Based on the findings of the Court-appointed experts, the Court concluded that the assertion by the USFWS was unsupported. (See e.g., Memorandum Decision Regarding Cross Motions for Summary Judgment, Doc. No. 757, p. 167.)

In summation, the findings above refute any suggestion that significant productivity is lost to appropriation of water. While appropriations of water remove plankton from the Bay-Delta, evidence is insufficient to support a conclusion that regulation of water appropriations will improve plankton productivity. The concentration of plankton in the Bay-Delta is unaffected by water appropriations. Water appropriations alter the transport of productivity in the system. However, modeling indicates that completely ceasing in-Delta appropriations will make little difference.

2.3.3 Possible Explanation 3: Increases in contaminants have direct and indirect effects on the survival and health of aquatic organisms

A wide array of contaminants including pesticides, metals, pharmaceuticals, personal care products and nutrients have been detected in Bay-Delta estuary water and sediment. (Nutrient contamination will be discussed in Section 1.3.4). These have been documented through a number of studies conducted over the last 25 years, primarily by the Central Valley Regional Water Quality Control Board and the Interagency Ecological Program. The U.S. Environmental Protection Agency's (USEPA's) 2010 CWA 303(d) List of Water Quality Limited Segments provides a succinct summary of the known impacts of contaminants on aquatic life in the Bay-Delta estuary (USEPA 2010).

Published literature suggests that contaminants have the potential to impact the ecosystem but must be considered in view of the complex interactions and uncertainties associated with potential exposures (Baxter et al. 2010; NRC 2010; 2011, 2012). While it is clear from studies to date that potential contaminant effects are important to the health of the Bay-Delta ecosystem, it is equally clear that it is important to recognize the uncertainties and complexity of these relationships.

2.3.3.1 Potential Sources of Contaminants in the Bay-Delta

Contaminants enter the Bay-Delta estuary and its tributaries by runoff from urban and agricultural land uses, atmospheric deposition, municipal and industrial water treatment effluent, recreational and commercial boating activities, and from historic mining operations. Contaminant levels vary both spatially and temporally and many are highest following rain events (Kuivila and Hladik 2008). Several contaminants, including pesticides, metals, pharmaceuticals, personal care products and nutrients, have been detected in Bay-Delta estuary water and sediment and continue to be the focus of concern by regulatory agencies.

2.3.3.2 Pesticides

Exposures of fish to sublethal concentrations of pyrethroids have resulted in decreased growth (Baldwin et al. 2009), and impaired swimming performance (Connon et al. 2009), increased susceptibility to viral infection (Clifford et al. 2005), and impacts to olfactory response (Sandahl et al. 2004). Acute exposure to some pesticides can have measurable population level effects on larval survival and development rates (Baldwin et al. 2009).

The Bay-Delta Advanced Notice of Proposed Rulemaking identifies the significant data gaps regarding pesticide use, sources, toxicity, and contributions to the Bay-Delta Estuary ecosystem collapse (USEPA 2011, p. 39). However, recent studies indicate that the contribution from urban sources may be significant (Weston et al. 2005; Weston and Lydy 2010, 2012). Several of these studies also indicate that the contribution from agricultural sources may not be as significant as previously indicated (Weston and Lydy 2010) and are improving (Hall 201Oa; Hall 201Ob; Hall 201Oc).

Use of the organophosphates diazinon and chlorpyrifos has been significantly reduced in agriculture and eliminated from urban use; however, pyrethroid pesticides have largely taken their place (Weston and Lydy 2010). In addition, there has been a significant shift to more toxic pyrethroid pesticides in the last decade (Amweg et al. 2005). Weston and Lydy (2010) detected pyrethroids in all but one of 33 urban runoff samples and observed toxicity over at least a 30 km reach of the American River, and at one site in the San Joaquin River. Similarly, Ensminger and Kelley (2011) sampled 13 urban sites in 2009-2010 and detected pesticides in 95% of the samples; 75% of the samples had 2 or more pesticides. Weston and Lydy (2010) occasionally detected pesticides in agricultural discharges.

Similarly, a USGS study detected 23 different dissolved pesticides in Bay-Delta water samples between 1998 and 2000 (Kuivila and Moon 2004). All water samples contained at least two and up to 14 different pesticides in each sample (with a median of five). Several pesticides overlapped temporally and spatially with the period of peak densities of larval and juvenile smelt.

2.3.3.3 Metals

A diverse array of metals, both naturally occurring and anthropogenic, has been detected in the Sacramento and San Joaquin rivers and the Bay-Delta estuary (SWRCB 1990). Numerous laboratory and field studies have demonstrated that exposure to metals can exert a range of adverse effects Exposure to low levels of copper can affects the olfactory mechanism of many fish species (Sandahl et al. 2007; Tierney et al. 2010; Raloff 2007). The olfactory system conveys critical information to fish, enabling activities such as mating, locating food, discriminating kin, avoiding predators and homing. In a review of studies on contaminant effects to fish olfactory systems, Raloff (2007) cited one researcher that noted, “pesticides and copper at concentrations similar to those in the environment knock out olfactory communications in every species tested to date – whether water fleas, leeches, or fish.” Sandahl, et al. (2007) observed a 40% reduction in olfactory response in juvenile coho salmon exposed to copper concentrations as low as 2 $\mu g\ L^{-1}$ for only 3-hours. This loss in olfactory sensitivity led to a failure to initiate predatory avoidance behaviors in response to chemical alarm cues.

Mercury toxicity in fish is also well documented and includes decreased appetite, ability to catch food, visual activity, and growth; lethargy; loss of equilibrium; gill hyperplasia and reduced respiration; neurotoxicity; nephrotoxicity; and teratogenic and reproductive effects (Reimschuessel 2001, Rodgers and Beamish 1982, Weis and Weis 1991, others). Reproductive effects from mercury exposure are a particularly sensitive endpoint, and can begin with the maternal transfer of mercury to embryos via the yolk (Weis and Weis 1995). Experimental studies have shown that embryo survival can be substantially reduced by very low concentrations of mercury from waterborne exposure or maternal transfer (Birge et al. 1979).

2.3.3.4 Pharmaceuticals and Personal Care Products (PPCPs)

The presence of PPCPs in the Bay-Delta is also a concern for the health of zooplankton and fish. PPCPs enter the watershed by treated wastewater discharges, septic tanks, urban and agricultural runoff. Sublethal adverse effects, such as impaired growth and reproduction, behavioral changes, and even population collapse are well documented in numerous aquatic organisms in response to chronic exposure to low concentrations of one or more PPCPs (Munoz et al, 2009; Cripe et al. 2009; Iwanowicz et al. 2009; Martinovic et al. 2007; Kidd et al. 2007; Dussault et al. 2008). Exposure to multiple compounds can have additive or synergistic adverse effects, which may be compounded by other environmental stressors. There is extensive documentation of these sublethal and synergistic effects on other aquatic organisms (Ward et al. 2007, Oros et al. 2005, Clifford et al. 2005).

The National Water Research Institute (NWRI) recently released a report entitled, "Source, Fate, and Transport of Endocrine Disruptors, Pharmaceuticals, and Personal Care Products in Drinking Water Sources in California" (NWRI, 2010). This study included the collection and analysis of samples from upstream and downstream of the Sacramento Regional Wastewater Treatment Plant. The concentrations of caffeine, carbamazepine, DEET, gemfibrozil, primidone, sulfamethoxazole, dilantin, and TCEP were higher in the Sacramento River at Hood downstream of the Plant than upstream in the Sacramento River at the West Sacramento Water Treatment Plant Intake and at the Fairbairn Water Treatment Plant intake on the American River.

Schaefer and Johnson (2009) also conducted monitoring up and downstream of the largest POTW in the Delta and detected caffeine, trimethoprim, sulfamethoxazole, gemfibrozil, fluoxetine, ibuprofen, carbamazepine, xylene, nonylphenol, and nonylphenol ethoxylates at one or more of the downstream monitoring sites. None of these compounds were detected in the upstream samples. Schaefer and Johnson (2009) state, "All of the compounds detected in the monitoring effort have been shown to have an adverse effect on one or more aquatic species." In fact, ibuprofen was detected at concentrations far greater than those observed to reduce activity in Gammarus pulex (Aquatic Ecosystems Analysis Laboratory 2009).

Kolpin et al. (2002) sampled 139 streams in 30 states and detected one or more organic wastewater contaminants in 80% of the streams. Half the streams contained seven or more chemicals. Six of the sites are in the Central Valley. Samples from the Sacramento River at Freeport had detectable levels of cholesterol, acetaminophen, and mestranol. Acetaminophen was also found in the San Joaquin River at Vernalis and upstream tributaries. The Turlock Irrigation District Lateral 5 that drains to the San Joaquin River had detectable concentrations of 17 β-estradiol, estriol, 19-norethisterone, 1 cholesterol, coprostanol (measure for the presence of human fecal matter), acetaminophen, caffeine, diltiazem (potent vasodilator of peripheral and coronary vessels), 1,7-Dimethylxanthine (caffeine metabolite), and codeine.

Several studies within the Bay-Delta estuary have observed endocrine disruption. Brander and Cherr (2008) observed choriogenin induction in male silversides from Suisun Marsh. Riordan and Biales (2008) reported endocrine disruption in male fathead minnows following in-situ exposures below the Sacramento Regional Wastewater Treatment Plant. Sommer (2008) reported that the sex ratio of young of the year striped bass in the Bay-Delta is heavily skewed toward male (90:10 male:female). While the cause of this skewed sex ratio is unknown at this time, exposure to endocrine disrupting chemicals cannot be ruled out. And, Connon et al. (2009) reported that “exposure to water from Hood elicited significant transcriptional differences of genes involved predominantly in neuromuscular functions, suggesting that contaminants originating from the SRWTP effluent may impact on swimming performance, growth and development of larval delta smelt. Down-regulation of structural muscle genes may also indicate physiological damage.”

In summation, a wide array of contaminants including pesticides, metals, pharmaceuticals, personal care products and nutrients have been detected in Bay-Delta estuary water and sediment. Several recent reviews conclude that contaminants have the potential to impact the ecosystem but must be considered in view of the complex interactions and uncertainties associated with potential exposures. While it is clear from studies to date that potential contaminant effects are important to the health of the Bay-Delta ecosystem, it is equally important to recognize the uncertainties and complexity of these relationships.

2.3.4 Possible Explanation 4: Increases in invasive species affect productivity

Agreement is universal that the invasion by the Amur River clam has had a significant effect on chl-*a* levels in Suisun Bay (Alpine and Cloern 1992; Kimmerer et al. 1994; Jassby et al. 2002; Kimmerer 2006; Greene et al. 2011) with its influence extending into the lower Sacramento and San Joaquin Rivers (Jassby et al. 2002). As described above, the clams have been blamed for the decline in *N. mercedis* (Orsi et al. 1996; Feyrer et al. 2003), the shift in distribution of anchovies (Kimmerer 2006) and young-of-the-year striped bass (Kimmerer et al. 2000; Feyrer et al. 2003; Sommer et al. 2007), as well as the decline in diatoms (Kimmerer 2005) and several zooplankton species (Kimmerer et al. 1994). The impact of the clams on chl-*a* and the Bay-Delta ecosystem is also reflected by a shift in many of the original correlations between species abundance and X2, that occurred after the establishment of the clams (Kimmerer 2002a; Sommer et al. 2007).

However, invasive clams cannot explain all of the chl-*a* decline. For example, Peterson and Vayssieres (2010) report at the lower Sacramento River station "the downward trend in phytoplankton biomass is coincident with significant reductions in overall benthic organism abundance." Nor can the invasive Amur River clam explain chl-*a* declines in the south Delta. In the south Delta the brackish-adapted *P. amurensis* have never been observed and that area is well beyond the influence of tidal dispersion, yet IEP benthic and phytoplankton data demonstrate that chl-*a* has also declined there (Figure 12). Presence of the invasive freshwater clam, *Corbicula fluminea,* can explain localized declines in chl-*a*, it cannot entirely explain the widespread drop in chl-*a* in the Delta because there is no apparent relationship between clam densities and chl-*a* concentrations after the clam invasion (Figure 12).

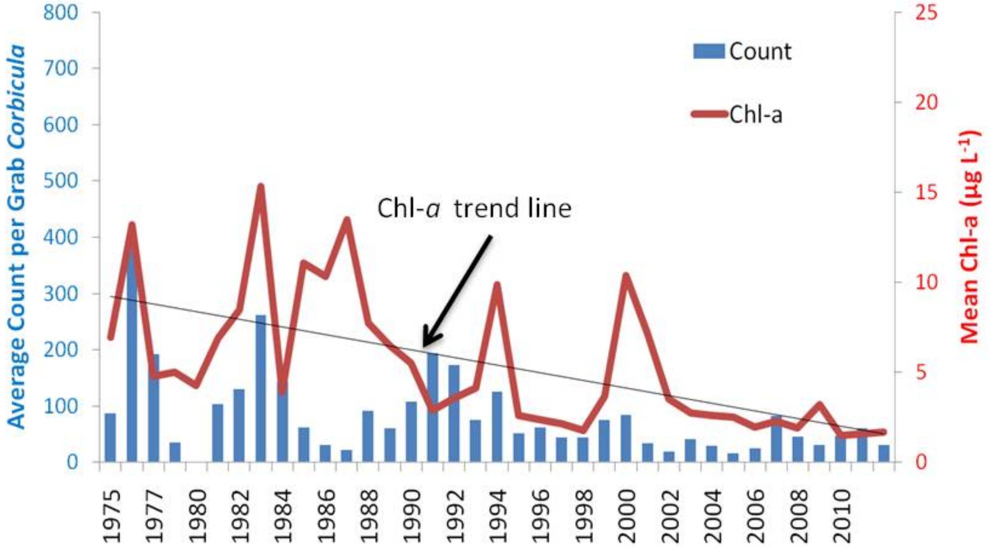


Figure 12. Average annual (March-November) chl-*a* concentration in Old River (station D-28A) and the average annual (March-November) abundance of Corbicula fluminea, showing no apparent relationship between clam abundance and chl-*a* concentration. (Source: IEP benthic and phytoplankton data).

Further evidence that some factor in addition to the clam is responsible for low phytoplankton biomass comes from observations that the decline in chl-*a* and diatoms in the Suisun Bay region began prior to establishment of *P. amurensis*, (Figure 13, Alpine and Cloern 1992; Jassby et al. 2002; Dugdale et al. 2007; Glibert 2010; Glibert et al. 2011). That further evidence also comes from the fact that *P. amurensis* is not abundant in spring (Dugdale et al. 2007), yet IEP benthic and phytoplankton data demonstrate chl-*a* concentrations have declined in spring (Figure 14). As described below, changes in nutrient concentrations and ratios might explain both the increase in clams and the drop in chl-*a* and diatoms.

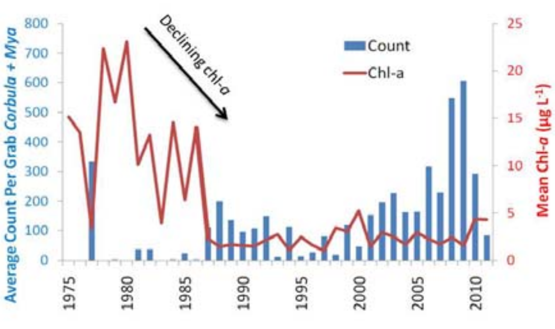


Figure 13. Average annual (March-November) chl-*a* concentration in Grizzly Bay (Station D7) and the average annual (March-November) abundance of *Potamocorbula amurensis* and *Mya arenaria*, showing a drop in chl-*a* concentration in the early 80s, prior to the increase in clam abundance in 1987 (Source: IEP benthic and phytoplankton data).

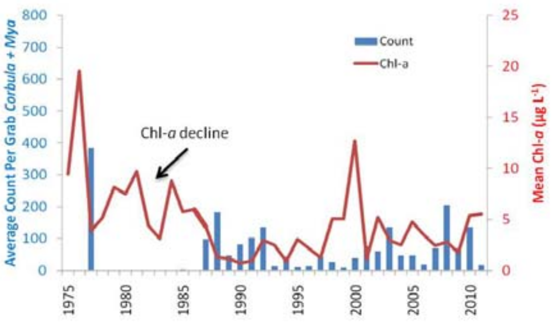


Figure 14. Average spring (March-May) chl-*a* concentration in Grizzly Bay (Station D7) and the average spring (March-May) abundance of *Potamocorbula amurensis* and *Mya arenaria*, showing a drop in chl-*a* concentration prior to the increase in clam abundance and little apparent relationship thereafter. (Source: IEP benthic and phytoplankton data).

Suggestions have been made that increased outflow can be used as a tool to control *P. amurensis* abundance. However, there have been no scientific support for this approach. *P. amurensis* distribution is affected by freshwater flows. But, increasing outflows only offers a temporary and localized solution. Peterson and Vayssieres (2010) report that "[a]ssemblages moved down-estuary in years with high delta outflow, and up-estuary during years with low delta outflow." In other words, high outflow merely moves the problem downstream and unless high outflow is maintained in every year, the clam assemblage may move back when flows decline. In addition, the invasive freshwater clam, *C. fluminea* moves down estuary with high delta outflow as well, occupying the space vacated by *P. amurensis.* Parchaso and Thompson (2002) do not provide any support for this strategy either. They report that "[t]he success of *P. amurensis* in this system is therefore related to its…apparent lack of temperature or salinity control on reproductive activity." In fact, Parchaso and Thompson (2002) found that "[p]opulations of *P. amurensis* at the upstream sites in Suisun Bay and Carquinez Strait were more reproductively active during wet years than dry years." This finding suggests that *P. amurensis* may be able to compensate for the detrimental effects of increased outflow on their abundance and distribution by increasing reproductive activity.

It is also worth noting that large-scale climatic variations may reduce the negative impacts of *P. amurensis* on the Bay-Delta food web. Cloern et al. (2007) observed declines in the abundance of *P. amurensis* in South San Francisco Bay and San Pablo Bay that they attributed to "a state change in the California Current System characterized by increased upwelling intensity, amplified primary production, and strengthened southerly flows."

The invasive aquatic plant, *Egeria densa*, may also affect productivity in the Delta. *Egeria* has been present in the Delta for about 50 years, but the area that it impacts began to expand significantly in the mid-1990s and is now spreading at a rate of 10-20% per year (Department of Boating and Waterways 2006). In addition to blocking channel access for boaters, and clogging diversion pumps, Anderson (1999) describes its effects on the food web as:

> "Other impacts of egeria are less obvious. These include the displacement of native pondweed species (*Potamogeton spp.*), impairment of access for waterfowl, and severe shading of the upper water column. The dense upper "canopy" formed by egeria blocks light that would normally be available to microscopic algae (phytoplankton). The lack of

> primary production by the phytoplankton inhibits fisheries production since these organisms form the base of the food chain."

Egeria may affect more than phytoplankton. Conrad (2010a) reports that submerged aquatic vegetation best explained juvenile largemouth bass abundance. Townsend (2010) developed an ecosystem model for testing the potential causes of POD. In preliminary simulations, chl-*a* and submerged aquatic vegetation biomass best explained both the POD fish declines and the increase in several non-native fish populations.

In summation, there is universal agreement that the invasion by the Amur River clam has had a significant effect on chl-*a* levels in Suisun Bay, with its influence extending into the lower Sacramento and San Joaquin Rivers. However, it appears that the arrival of invasive clam species does not fully explain the decrease in the Bay-Delta's productivity. The invasive aquatic plants, Egeria densa, may be one of the other factors suppressing productivity in the Delta.

2.3.5 Possible Explanation 5: Changes in nutrient concentrations, forms and ratios cause changes in species composition and abundance at all trophic levels

Historically, scientists have described primary productivity in the Bay-Delta as not being limited by nutrients (Cloern 2001; Lopez et al. 2006) and not experiencing signs of classic eutrophication (Cloern 2001). However, the consensus within the scientific community is changes in nutrient loads are affecting ecosystem dynamics in complex ways that extend beyond our historic understanding of the process of eutrophication. Total nutrient load sets the upper limit on total primary production, and ultimately secondary production- and increases in nutrient loading are commonly associated with eutrophication. The most common effects of eutrophication are increased chl-*a* in the water column, development of hypoxia or anoxia (low or no dissolved oxygen, respectively), loss of native submerged aquatic vegetation, increased harmful algal blooms, and changes in biodiversity, including loss of certain fisheries (e.g., Cloern 2001; Anderson et al. 2002).

Nutrient effects on aquatic systems are far more complex and subtle than those normally associated with eutrophication. Changes in nutrient form (chemical state, oxidized vs. reduced, organic vs. inorganic, dissolved vs. particulate) and the proportion of different elements (including carbon (C), nitrogren (N), phosphorous (P), and silicon (Si), among others) also have effects on ecosystems at both the scale of the primary producers (the algae) and throughout the ecosystem. In the Bay-Delta, the total loads, the forms, and the relative proportions of nutrients have been changing over time. These changes have had profound effects on ecosystem structure of this system, as documented below. This paper illustrates the effects of two such changes, the proportion of ammonium to nitrate (NH_4:NO_3) and that of nitrogen to phosphorus (N:P).

2.3.5.1 Ammonium/Nitrate

For decades, researchers have explored the relative use – or relative preference for -- different forms of nitrogen (N) by phytoplankton. Ammonium (NH_4) is generally considered to be the form of nitrogen preferred by phytoplankton due to the more favorable energetics associated with its assimilation compared to that of nitrate (NO_3). It is also well documented that NH_4 can inhibit the uptake of NO_3, but the relative effect of this inhibition is a function of species composition and other environmental factors (Dortch 1990). When NH_4 inhibits the uptake of NO_3 by phytoplankton, it can also exhibit a strong negative control on total productivity (Yoshiyama and Sharp 2006; Dugdale et al. 2007). NH_4 suppression of NO_3 uptake when both nutrients are in ample supply should not be confused with the preferential use of NH_4 by phytoplankton when N is limiting. Under the latter conditions, phytoplankton will use NH_4 preferentially because it requires less energy than NO_3. Under the former conditions, the cells must cope with an excess; and in doing so, their metabolism is less capable of assimilating NO_3.

Within the Bay-Delta's aquatic ecosystems, Wilkerson et al. (2006), Dugdale et al. (2007), and Parker et al. (2012a, b) show that "bloom levels of chlorophyll-*a* are evident only when NO_3 uptake occurs and that NO_3 uptake only takes place at lower ambient NH_4 concentrations." The authors note that sufficient light must also be present to support a bloom. They conclude that NH_4 concentrations greater than 4 micromols per liter [$\mu mol\ L^{-1}$] (0.056 mg L^{-1}) inhibit N uptake by diatoms and thus suppress bloom formation and reduce primary productivity. This level of NH_4 is in line with previous inhibitory level estimates (Lomas and Glibert 1999a; Yoshiyama and Sharp 2006) and is exceeded a majority of the time in the Sacramento River and Suisun Bay (Figure 15). As described in more detail below, the estimate of 4 $\mu mol\ L^{-1}$ as an inhibitory threshold will vary as a function of species composition and environmental factors, such as temperature, which affects metabolism. When monthly data of chl-*a* and diatom cell count are plotted against NH_4 levels for the period 1975 to 2012 there is a marked decreasing trend in both as NH_4 levels rise with a increasing effect around 0.056 mg L^{-1} (~4 $\mu mol\ L^{-1}$) NH_4 (Figures 16 and 17).

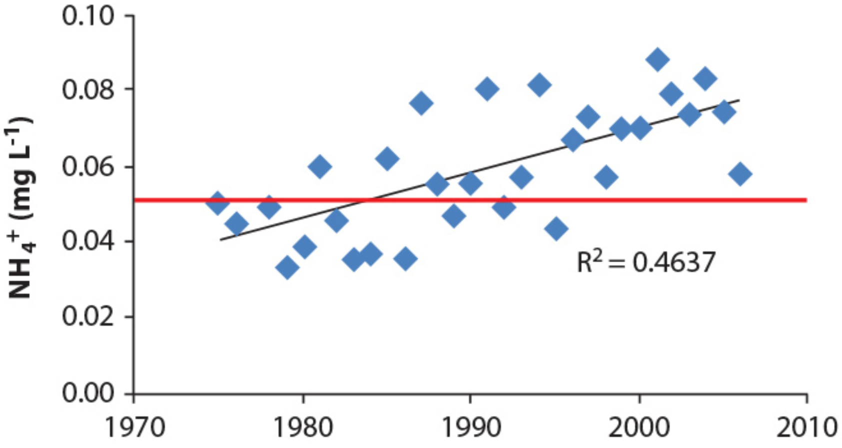


Figure 15. **Change in concentration of NH_4^+ over time on an annual basis for stations sampled from the confluence to Suisun Bay. The horizontal red line at 0.056 mg L^{-1} is the concentration at which inhibition of N uptake by diatoms is observed in the Bay-Delta. Coefficient of determination was significant at $p<0.01$.**

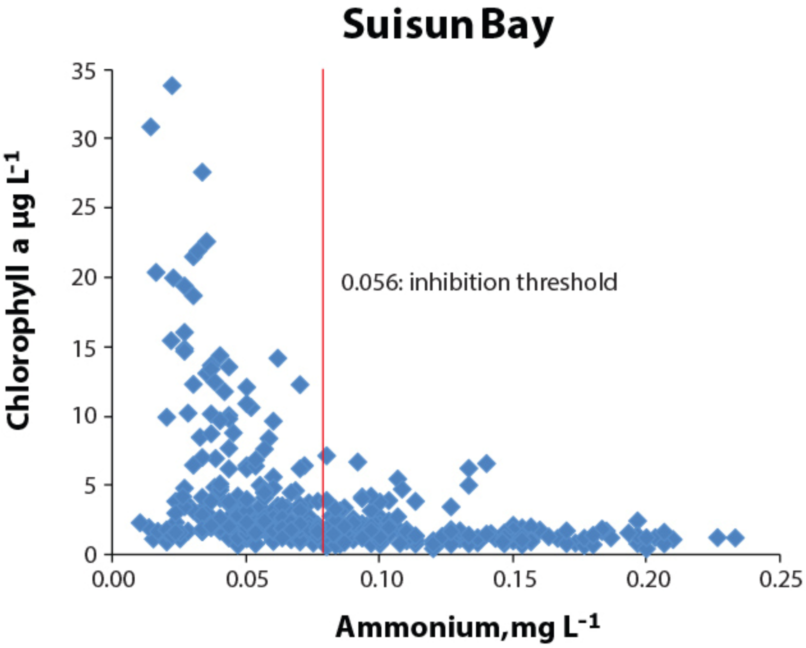


Figure 16. **Chlorophyll-*a* concentration plotted with ammonium concentration in Suisun Bay. As NH_4 concentration increases above the level where inhibition is observed (0.056 mg L^{-1} NH_4), chl-*a* levels decline. (Source: Environmental Monitoring Program data)**

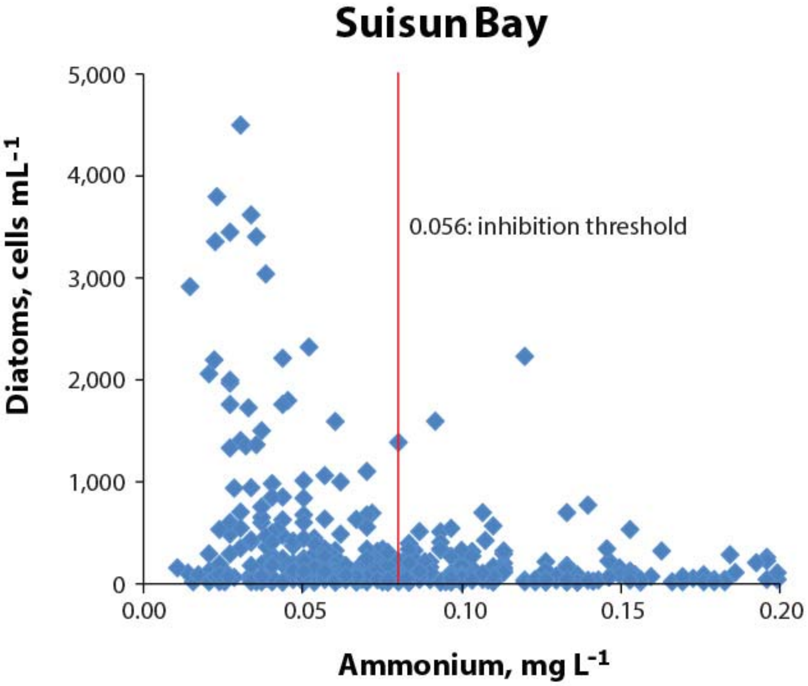


Figure 17. **Diatom cell count plotted with ammonium concentration in Suisun Bay. As NH_4 concentration increases above the level where inhibition is observed, chl-*a* levels decline. (Source: Environmental Monitoring Program data)**

The effects of changes in the proportion of NH_4:NO_3 have been shown for the Bay-Delta in both field observations and laboratory experiments. Parker et al. (2012a) observed a 60% decline in primary production in the Sacramento River below the Sacramento Regional Wastewater Treatment Plant, where NH_4 is discharged, compared to production above the Treatment Plant's outfall. Also supporting this finding, Parker et al. (2012b) found that "[b]y tracing both carbon (C) and N uptake we provide clear evidence that high rates of C uptake are linked to phytoplankton NO_3, and not NH_4, use." They conclude that the increased proportion of NH_4 "may help explain some of the reduced primary production and phytoplankton biomass observed [in the San Francisco Estuary] since the 1970s."

In enclosure experiments with samples from Central Bay, Suisun Bay, and the Sacramento River at Rio Vista, representing a gradient of both nutrient concentrations and proportions of different forms of N, Wilkerson et al. (in preparation) observed "a gradient of decreasing phytoplankton physiological rates in the upstream direction as far as Rio Vista." Phytoplankton productivity rates (both carbon and nitrogen uptake) decreased with increasing concentrations of NH_4.

In a series of recent experiments conducted with natural samples from the Sacramento River the effects of altered proportions of NH_4:NO_3 were also apparent (Glibert et al. 2012). In these experiments, the proportions of these nutrients were manipulated, and both short-term N uptake rates and longer-term N production rates were assessed. Two findings are of note. First, when 20 μmol NH_4 was added to the sample, a concentration commonly observed in the Sacramento River, and the rate of NO_3 uptake was measured across a concentration gradient, the rate of uptake of NO_3 decreased significantly compared to unamended rates measured over a period of < 1 hour (Figure 18). Second, when samples were enriched with NH_4, NO_3, or urea (at the molar equivalent dose) for a period of 24-48 hours, and then rates of uptake of all N forms measured, the summed rate of N uptake in the NH_4-added treatment was significantly lower than that in the NO_3-added or urea-added treatments (Figure 19).

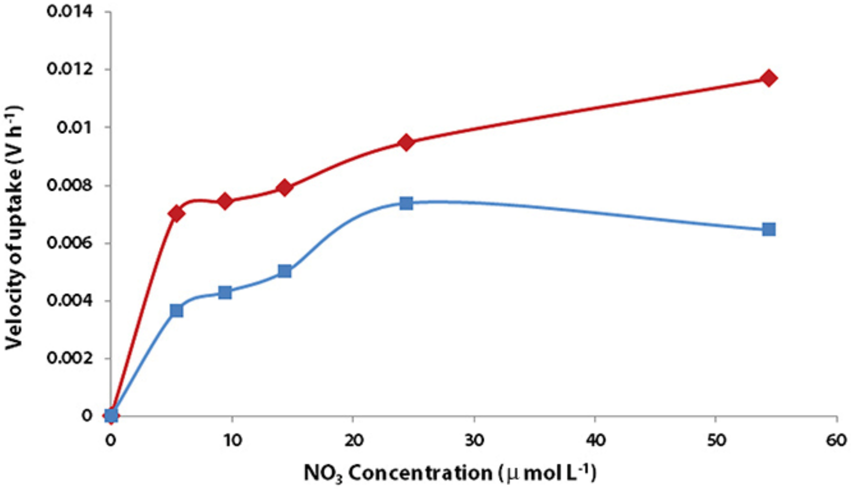


Figure 18. **Velocity of uptake of NO_3 as a function of added NO_3 enrichment (red curve), and the same relationship but with a constant addition of 20 μmol L^{-1} NH_4 (blue curve). Nitrate uptake is reduced when NH_4 is added. Experiment was conducted with water collected from the Sacramento River. Data from Glibert et al. (2012).**

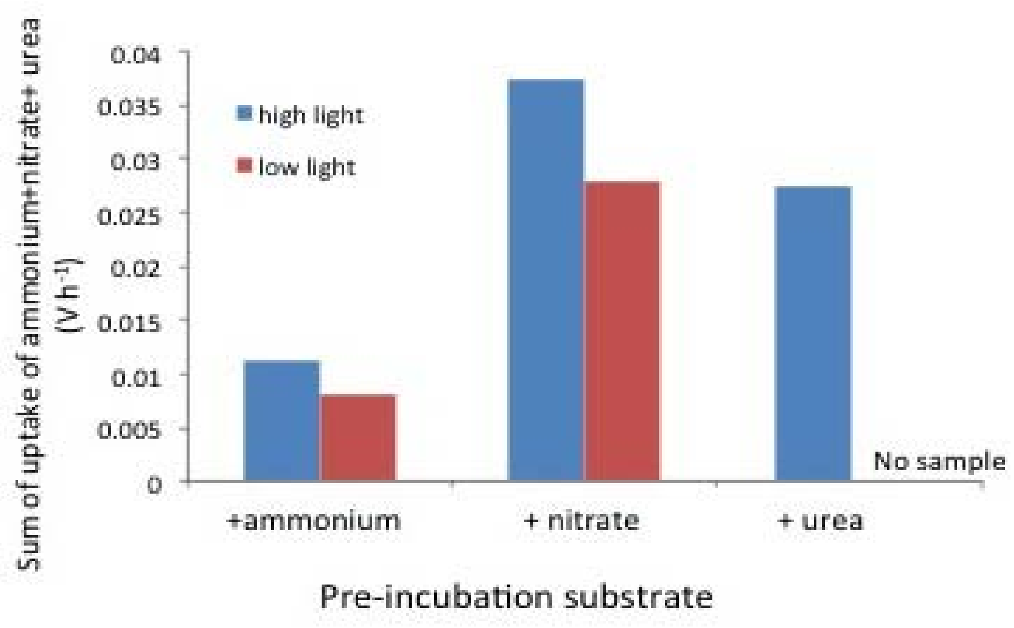


Figure 19. **Summed rate of uptake of nitrate+ ammonium+ urea for samples collected from the Sacramento River and pre-incubated with the substrate indicated, after which short-term uptake rates were measured using ^{15}N tracer techniques. The experiment was conducted under both high (blue bars) and low (red bars) irradiance levels. The summed rate of N uptake in the NH_4-added treatment was significantly lower than that in the NO_3-added or urea-added treatments. From Glibert et al. (2012).**

These observations of NH_4 suppression are not new in or unique to the Bay-Delta. A large body of scientific research describes NH_4 suppression of algae productivity (*e.g.* Ludwig 1938; Harvey 1953). Some of the early field demonstrations of this phenomenon were carried out by MacIsaac and Dugdale (1969, 1972), followed by research in the Chesapeake Bay by McCarthy et al. (1975). Maestrini et al. (1982) showed that only after NH_4 concentrations were reduced to < 7 µmol L^{-1} (0.098 mg L^{-1}) was NO_3 uptake sufficient to match that of NH_4 uptake. Price et al. (1985) showed that the rate of NO_3 uptake was reduced ~50% in samples that also received an NH_4 spike compared to those receiving a NO_3 spike. Lomas and Glibert (1999a) described the threshold for inhibition of NO_3 uptake at NH_4 levels of approximately 1 µmol L^{-1} (0.014 mg L^{-1}). Yoshiyama and Sharp (2006) saw a “striking decline in production at NH_4 levels above a low threshold (around 10 µmol L^{-1})” (0.14 mg L^{-1}). The importance of NH_4 inhibition of NO_3 uptake was considered to be a necessary interaction to include in a recent model of the emergent phytoplankton community in the California Current System (Goebel et al. 2010). In recent experiments conducted in the tidal freshwater estuarine zone of the Guadiana Estuary (Spain and Portugal), it was also found that NO_3 consumption decreased with increasing NH_4 uptake, and these findings were most pronounced during the most productive period. Total primary productivity was suppressed as a result (Domingues et al. 2011). The now well-supported notion that NH_4 may be inhibitory not only to NO_3 uptake but to total productivity is particularly problematic for the Bay-Delta as it is already a comparatively low producing estuary (Jassby et al. 2002).

The form of N available to a phytoplankton community affects more than just rates of uptake and productivity. Their proportions also affect phytoplankton species composition. The physiological literature strongly supports the concept that different algal communities use different forms of N. Diatoms, once the dominant algal group in the Bay-Delta, generally have a preference for NO_3; dinoflagellates and cyanobacteria generally prefer more chemically reduced forms of N (NH_4, urea, organic nitrogen) (e.g., Berg et al. 2001; Glibert et al. 2004, 2006; Brown 2009). Under some circumstances, diatoms have a physiological requirement for NO_3 (Lomas and Glibert 1999a,b). Moreover, diatoms usually show no evidence of NO_3 uptake saturation under very high NO_3 conditions (Collos et al. 1992*,* 1997; Lomas and Glibert 1999a), in contrast to the generally accepted saturating uptake kinetic relationships that are used to describe the relationship between nutrients and uptake rate. Cyanobacteria have been shown to preferentially use chemically reduced forms of N, like NH_4, over NO_3. Evidence comes from measurements of enzyme activities in the cells (Solomon et al. 2010), directly determined rates of N uptake using isotope tracer techniques (Glibert et al. 2004; Kendall et al. 2011), direct growth studies (Berman and Chava 1999; citations within Meyer et al. 2009), and observations of changes in community composition with enrichment with different forms of N (Domingues et al. 2011).

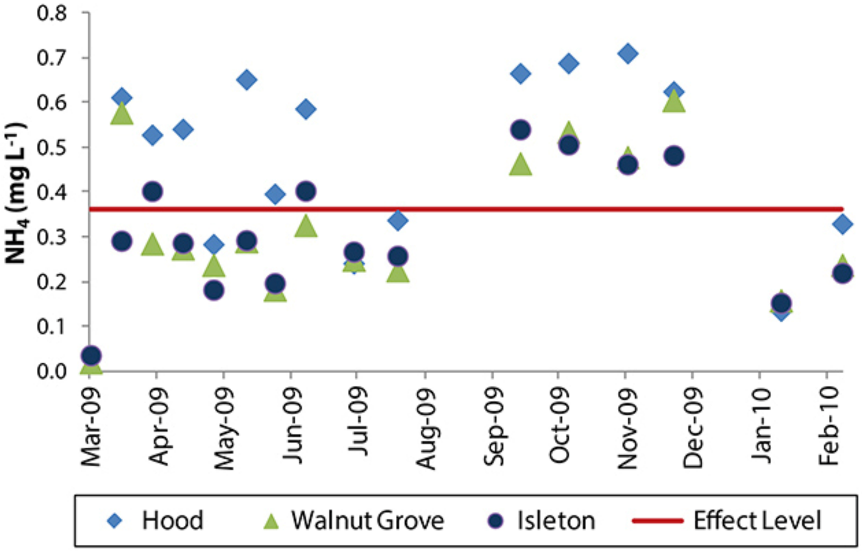
There is also evidence that the increase in aerial coverage by the invasive aquatic plant *Egeria densa* may be attributed to the ratio of nutrient inputs to the system. Feijoo, et al. (2002) experimentally found that *E. densa* absorbed more nitrogen from water when it was present in the form of NH_4 than when it occurred as NO_3.

Thus, although there are many factors that regulate the relative contribution of a nutrient source to different phytoplankton groups, and even species-specific differences within groups, it has generally been established that NO_3 disproportionately contributes to diatoms' uptake and production, while reduced forms of N (both NH_4 and urea) disproportionately contribute to the uptake and growth of cyanobacteria. As stated by Domingues et al. (2011), "…increased inputs of N as NH_4 due to urban waste effluents may result in a shift in phytoplankton community composition, towards a dominance of cyanobacteria and green algae." And, as stated by Lehman et al. (2010), "Recent increases in NH_4 concentration in the western delta may give a competitive advantage to Microcystis which rapidly assimilates NH_4 over NO_3." The phytoplankton community composition in the Bay-Delta estuary has shifted in just this manner. (refer to Figure 1).

Moreover, there is recent evidence that diatom blooms may be restored in the Bay-Delta estuary if NH_4 loading is reduced. In Suisun Bay, an unusual diatom bloom in Spring 2000 reached chl-*a* concentrations of 30 µg L^{-1} when NH_4 concentrations declined to 1.9 µmol L^{-1} (0.027 mg L^{-1}) (Wilkerson et al. 2006). Similarly, chl-*a* concentrations in Suisun Bay reached 35 µg L^{-1} during spring 2010 when NH_4 concentrations declined to 0.5 µmol L^{-1} (0.007 mg L^{-1})(Dugdale et al. 2011; Dugdale et al., in press). These blooms are comparable to spring chl-*a* levels from 1969 to 1977 (Ball and Arthur 1979) when NH_4 concentrations were 1.8 µmol L^{-1} (0.025 mg L^{-1}) during summer and 4.0 µmol L^{-1} (0.056 mg L^{-1}) during winter (Cloern and Cheng 1981).

In addition to altering phytoplankton community structure, growth rates and abundance, NH_4 is also toxic to some higher trophic level organisms. Scientists at UC Davis have investigated the effects of NH_4 to the calanoid copepod *Pseudodiaptomus forbesi* using a full life-cycle bioassay approach (Teh et al. 2011). *P. forbesi* is an important prey item for the young of many fish species in the Bay-Delta including delta smelt and longfin smelt (Nobriga 2002; Hobbs et al. 2006; Feyrer et al. 2003). Teh et al. (2011) found that total NH_4 at 0.36 mg L^{-1} (25.7 µmol L^{-1}) significantly affects the recruitment of new adult copepods and total NH_4 at 0.38 mg L^{-1} (27. 1 µmol L^{-1}) significantly affects the number of newborn nauplii surviving to 3 days (Teh et al. 2011). For comparison, monthly water samples collected between 2009-2010 from the Sacramento River between Hood and Isleton, approximately 30 miles downstream from the point of discharge, exceeded this level of NH_4 44% of the time (Figure 20).

Figure 20. Ammonium concentrations in the Sacramento River measured at Hood, Walnut Grove, and Isleton between 2009 and 2010. Hood and Isleton are approximately 8 and 30 miles downstream of the discharge, respectively. The horizontal line at 0.36 mg L^{-1} is the level at which significant toxicity to copepods is observed. Data from Foe et al. (2010).



2.3.5.2 *Nitrogen:Phosphorus*

Extensive research has found that the N:P ratio also has profound effects on community structure. The N:P ratio of nutrients has doubled in the Bay-Delta estuary over the last 35 years as is apparent from the data on Figure 21. These increases are a result not only of the increasing total N load (due to increasing effluent as well as other sources), but also as a function of declining P loads (Van Niewenhuyse 2007; Glibert 2010, Glibert et al. 2011).

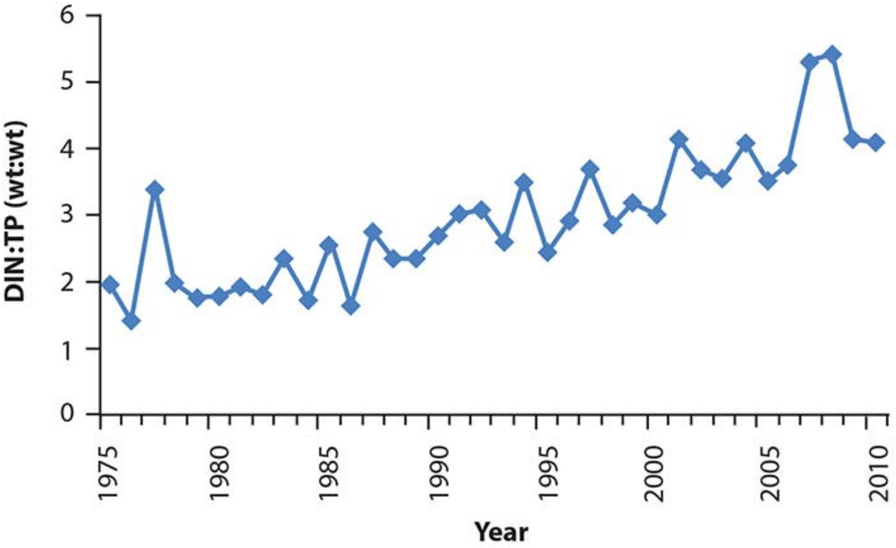


Figure 21. **Average annual (March-November) ratio of dissolved inorganic nitrogen to total phosphorus has doubled in the Bay-Delta. (Source: IEP monitoring data from stations in Suisun Bay and the confluence (D4, D6, D7, D8)).**

There are a number of strategies available to different types of phytoplankton for coping with an environment where nutrient ratios are not in proportion to their internal requirements (Glibert and Burkholder 2011). For example, cell size is an important determinant of elemental composition (Harris 1986; Finkel et al. 2010). Small cells have a lower requirement for P due to the smaller need for structural components in the cell (Finkel et al. 2010). In comparison to diatoms, very small cyanobacteria such as *Synechococcus* have a much larger cellular ratio of carbon to phosphorus (C:P), on average (Finkel et al. 2010). This explains why small cells, such as *Synechococcus*, have been found to thrive in waters that are comparatively P poor, as is the case in Florida Bay (Glibert et al. 2004).

There is strong support in the scientific literature for the proposition that the N:P ratio influences phytoplankton community composition. For example, in the Seto Inland Sea of Japan, removal of phosphorus also led to a shift in phytoplankton community structure from “nonharmful diatoms to harmful raphidophytes…and then finally to harmful/toxic dinoflagellates” (Yamamoto 2002). In this case the reduction in phosphorus which increased N:P, led to a change in phytoplankton community composition and was suggested to be “the major cause of the reduction in fishery production” (Yamamoto 2002).

In a retrospective analysis of 30 years of data from the Bay-Delta estuary, Glibert et al. (2011) found that the variation in these nutrient concentrations and ratios is highly correlated to variations in the total amount and composition of phytoplankton. This analysis revealed relationships between biological parameters and nutrients and/or nutrient ratios using both the original data and data that were adjusted for autocorrelation. At the phytoplankton level, as described earlier, there has been a decline in total chl-*a* and a decline in total diatoms over the past several decades in proportion to the increase in total inorganic N to total P (Figure 22). The change in chl-*a* with N:P is apparent in different regions of the Bay Delta; as N:P increases, chl-*a* declines (Figure 23).

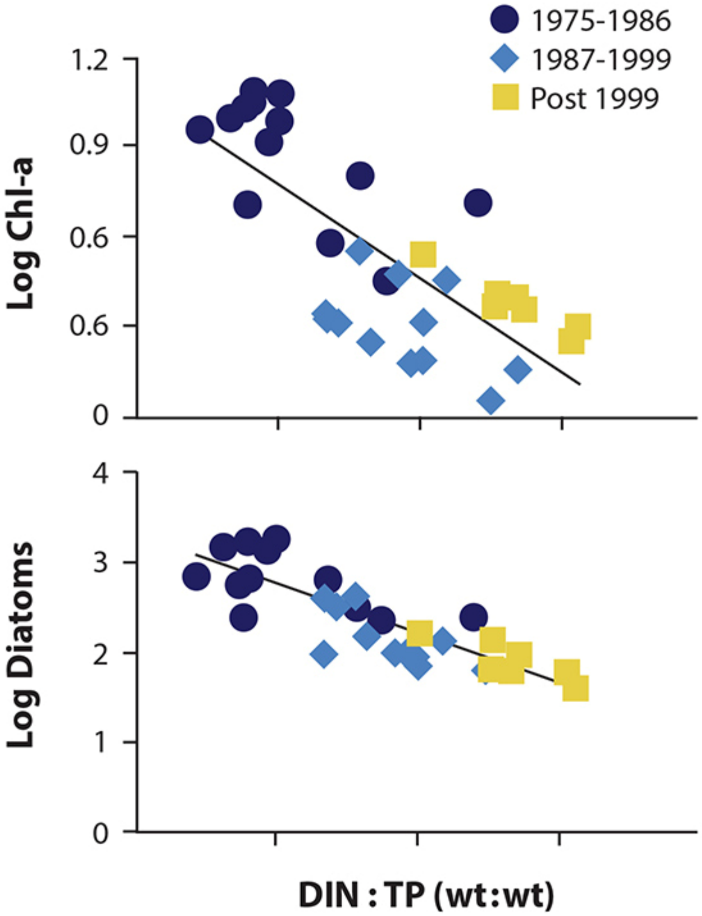


Figure 22. **Change in the concentration of chl-*a* (µg L^{-1}) and abundance of diatoms (Bacillariophyceae, cells mL^{-1}) as a function of dissolved inorganic N to total phosphorus. A loss of total chl-*a* and a loss of total diatoms in the phytoplankton community have occurred over the past several decades in proportion to the change in total inorganic N to total P. The relationship is significant at $p<0.05$. Different periods of time are represented by different symbols: 1975-1986, filled circles; 1987-1999, diamonds; post-1999- filled squares. Data shown are for the years 1975-2005 and cover the region from the confluence to Suisun Bay. All data log-transformed. Data from Glibert et al. (2011).**

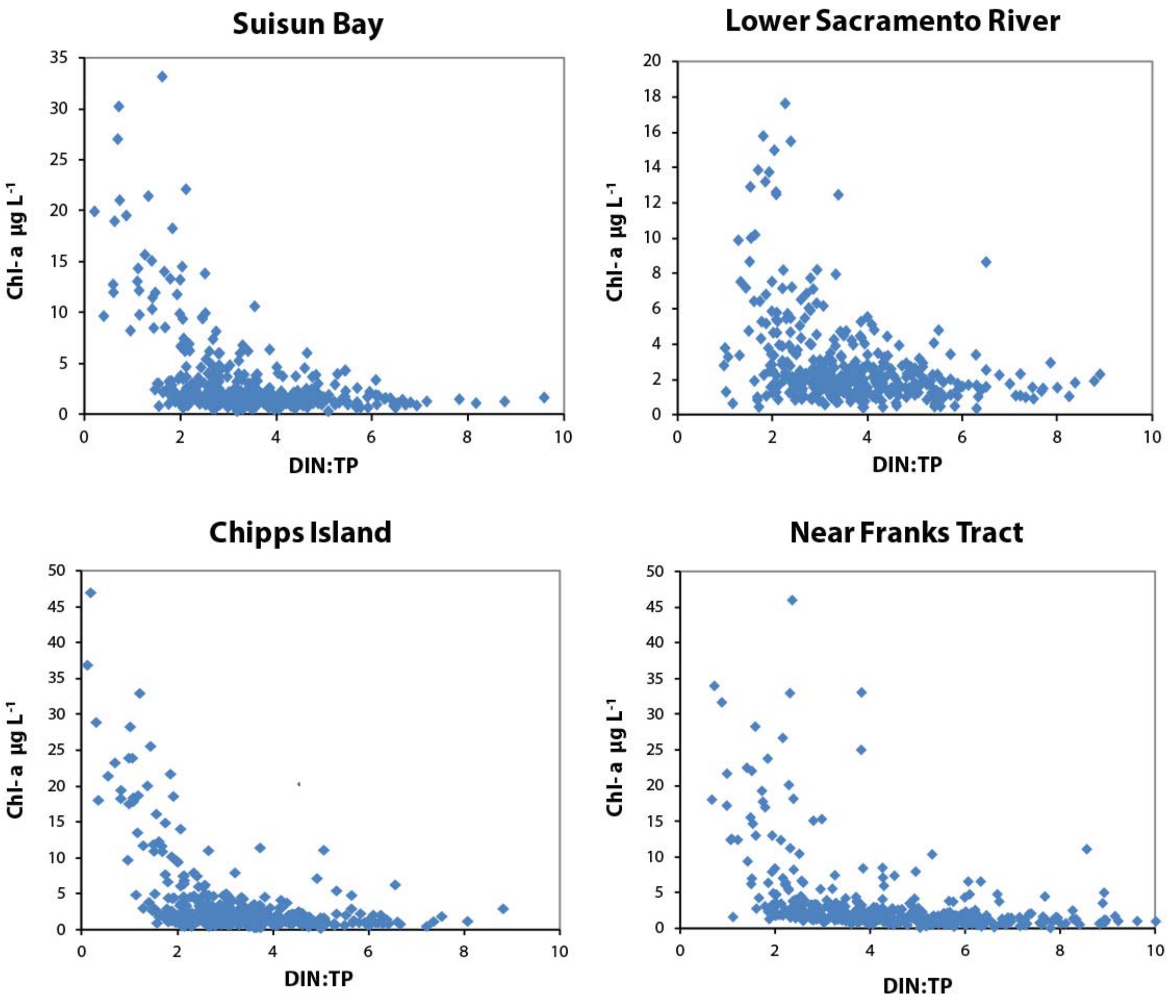


Figure 23. **Chl-*a* concentration plotted against DIN:TP for subregions of the Delta, 1975-2011. As DIN: TP increases, there is a loss of chl-*a*. (Source: Environmental Monitoring Program data).**

Fast-growing phytoplankton require proportionately more P to satisfy metabolic demands. Diatoms are typically fast-growing, and thus require proportionately more P to meet this metabolic demand. In ecological terms, they are considered a *r*-selected group, would be expected to have a low N:P biomass ratio (due to the high P cellular demand), and thus **would be expected to be outcompeted if N:P in the environment increases**. So-called *r*-selected species are out-competed when the environment changes (e.g., Heckey and Kilham 1988). In contrast, many cyanobacteria are considered to be *k*-selected, implying a slower growth rate and a higher metabolic N:P. In fact, “Reynolds (1984) singled out *Microcystis* as an example of a *k*-selected phytoplankter because it grows slowly in nature” (Heckey and Kilham 1988).

The balance of N:P can also affect other metabolic aspects of phytoplankton besides growth, including toxin production, cell membrane thickness, and other chemical constituents that have been considered to turn good food “bad” (Mitra and Flynn 2005). For example, toxin production by numerous harmful algae has been shown to increase when the cells are grown under nutrient-imbalanced conditions and when there is a change in N or P availability (Flynn et al. 1994; Johansson and Granéli 1999; Granéli and Flynn 2006). In Daechung Reservoir, Korea, researchers found that toxicity of cyanobacteria was related not only to an increase in N in the water, but to the cellular N content as well (Oh et al. 2000). A recent report by van de Waal (2009) demonstrated in chemostat experiments that under high carbon dioxide and high N conditions, microcystin (an algal toxin) production was enhanced in *Microcystis*. Similar relationships were reported for a field survey of the Hirosawa-no-ike fish pond in Kyoto, Japan, where the strongest correlations with microcystin were high concentrations of NO_3 and NH_4 and the seasonal peaks in *Microcystis* blooms were associated with extremely high N:P ratios (Ha et al. 2009). Thus, not only is *Microcystis* abundance enhanced under high N:P, but its toxicity appears to be as well (Oh et al. 2000).

It is well accepted that the nutritional value of phytoplankton differs from one species to another. Toxin production can inhibit grazing. Some phytoplankton species are rejected by grazers due to their size. Others vary in their nutritional quality. For example, some diatom species produce certain highly unsaturated fatty acids that are essential for zooplankton reproduction (reviewed by Kilham et al. 1997) while flagellates generally produce different fatty acids than diatoms (Olsen 1999). Many trophic interactions, such as rates of growth or fecundity, are dependent on the acquisition of particular fatty acids, as a measure of the food quality of algae (e.g., Ahlgren et al. 1990; Coutteau and Sorgeloos 1997; Weers and Gulati 1997; Brett and Müller-Navarra 1997). In feeding experiments, Ger et al. (2010) observed reduced survival of the copepods, *Pseudodiaptomus* and *Eurytemora,* even when *Microcystis* was only a small portion of their available diet. Brett and Müller-Navarra (1997) developed a food quality rank for 10 species from 5 major phytoplankton groups based on the average of the observed change in the abundance of individual zooplankters that preyed upon these phytoplankton in growth bioassays. They and others (see Park et al. 2003) have applied a 0-1 scale of phytoplankton food quality in which cyanobacteria ranks at 0.2; green algae, 0.525; diatoms, 0.7; and cryptomonads, 0.95. Thus, a trend of decreasing diatoms and increasing cyanobacteria in the Bay-Delta would suggest, based on these rankings, a decrease in food quality for higher trophic levels. Jassby (2008a) states:

> A decrease in percentage of diatom biovolume occurred during 1975–1989, caused by both a decrease in diatoms and an increase in green algae, cyanobacteria, and flagellate species biovolume (Kimmerer 2005; Lehman 1996), i.e., probably in the direction of declining nutritional value per unit biomass. In principle, the total nutritional value of a community could decrease even as its biomass increases. Moreover, changes in size, shape, and motility of species comprising the phytoplankton community could also affect their availability as food particles for crustacean zooplankton and other consumers.

Cloern and Dufford (2005) state, “[t]he efficiency of energy transfer from phytoplankton to consumers and ultimate production at upper trophic levels vary with algal species composition: diatom-dominated marine upwelling systems sustain 50 times more fish biomass per unit of phytoplankton biomass than cyanobacteria-dominated lakes.”

For species that prey on phytoplankton (e.g., zooplankton), stoichiometry affects all aspects of behavior, such as growth rate, fecundity, and ultimately the success of different populations (Jeyasingh and Weider 2005, 2007), but may affect various life stages differently (Moe et al. 2005, p.31): “[a]n organism’s requirements for different elements may vary throughout its life cycle, and thus certain life stages may be more sensitive than others to variation in the stoichiometry of its resource.” For example, copepod juveniles have a relatively high demand for C, N, and P, but at a later stage, while C is still needed for metabolism, more P must be allocated to eggs. Therefore, P-poor food sources can disproportionately affect egg production while not affecting survival (Faerovig and Hessen 2003; Laspoumaderes et al. 2010). In a laboratory study where *Acartia tonsa* was fed diatoms grown on different N concentrations, Kiørboe (1989), confirmed that this zooplankter changes its feeding rate in response to phytoplankton of different chemical composition – thus, in response to food quality. Moreover, egg production followed the variation in algal N content and increased with increasing algal N.

In a review of field and laboratory-based research on stoichiometry in food webs, Hessen (1997) showed that a shift from copepods to *Daphnia* tracked N:P; copepods retain proportionately more N, while *Daphnia* are proportionately more P rich. Often, those organisms that are most able to retain the nutrient in limited supply, in this case P, have the competitive advantage in an unbalanced system. Glibert et al. (2011) illustrated a finding similar to Hessen's, that the decline in calanoid copepods in the Bay-Delta, and the invasion of cyclopoids tracked N:P over time. Variation in proportional densities of the calanoid copepod *Eurytemora* with the cyclopoid copepod *Limnoithona* over time has followed changes in the DIN:TP (dissolved inorganic nitrogen to total phosphorus) ratio (Figure 24), a pattern consistent with these grazers being responsive to changes in elemental stoichiometry and maintenance of altered dynamic equilibria on a long-term scale. In fact, Glibert et al. (2011) found relationships between many of the shifts in zooplankton community composition seen on Figure 2 and shifts in nutrient composition. Results from whole-lake experimentation suggest that the N:P ratio is linked to alterations in zooplankton size, composition, and growth rate, as those animals with increased RNA allocation (more P available for growth) will grow at higher rates due to increased protein synthesis rates (Sterner and Elser 2002). Similar findings were reported from annual studies in the Baltic Sea (Walve and Larsson 1999).

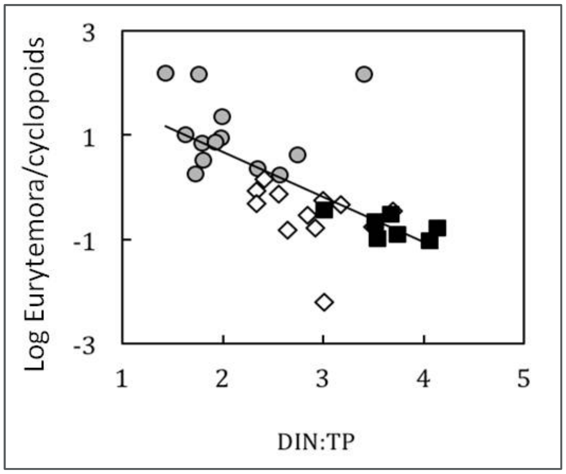


Figure 24. Change in the ratio of Eurytemora to cyclopoid copepods (all data log transformed) as a function of DIN:TP for annually averaged data from 1975-2005 for samples collected between the confluence and Suisun Bay. As DIN:TP increases, the proportion of Eurytemora to Cyclopoids decreases. Different periods of time are represented by different symbols: 1975-1986, filled circles; 1987-1999, diamonds; post-1999 filled squares. The correlation for these and for data that were detrended (not shown) are significant ($p<0.05$). From Glibert et al. 2011.

Superimposed on these empirical observations is consideration of whether substrate quality or food quality is altered on an episodic basis, or whether changes are long-term and sustained. As conceptualized by Hood and Sterner (2010), a change in predator growth rate depends on the extent to which a diet is sustained or switches between low-quality food and high-quality food as defined by the relative P content.

2.3.5.3 Higher Trophic Level Effects

Disproportionate N and P loads are now recognized to have effects at all scales, from genomic to ecosystems that need further empirical resolution (Peñuelas et al. 2012). When N:P availability changes, food webs change, biogeochemical cycling can change, and these changes can be positively reinforcing. Sterner and Elser (2002) state: "[s]toichiometry can either constrain trophic cascades by diminishing the chances of success of key species, or **be a critical aspect of spectacular trophic cascades with large shifts in primary producer species and major shifts in ecosystem nutrient cycling**" [emphasis added]. Just as different elemental ratios may affect the composition of the primary producers, different nutrient requirements of organisms occupying higher trophic levels will have an impact on their ability to thrive as community composition changes at the base of the food web. At the ecosystem scale, the total load and balance of nutrient elements have effects that propagate through the food web, with the potential of transforming ecosystems to new stable states. Although the shift in algal community composition in terms of diatoms and cyanobacteria has been emphasized above, this shift in the Bay-Delta estuary has been far more complicated. With the decline in water column chl-*a* and an increase in light availability, other primary producers have increased in abundance, including invasive macrophytes such as *Egeria densa* (Sommer et al. 2007; Nobriga et al. 2005; Glibert et al. 2011). *E. densa* may be particularly well suited to the low DIP:DIN environment of the Bay-Delta since it is able to access sediment bound phosphorus through its roots. In fact, similar increases in macrophytes were observed in many other systems in which N:P increased following N enrichment and P reduction, including the Potomac River, Chesapeake Bay, Ebro River in Spain, and the Rhine River in Germany (Glibert et al. 2011; Glibert 2012). Such macrophyte invasions can have profound impacts on ecosystems, not only because they alter the flow of C and the overall productivity of the system, but they also serve as “ecological engineers,” decreasing nutrients through uptake, reducing turbidity by trapping sediments, and providing refuge for zooplankton and habitat for other species, including fish (Yarrow et al. 2009; Glibert 2012).

The interplay between nutrient stoichiometry and biogeochemistry is well illustrated when a system is driven to higher macrophyte productivity. Macrophytes can be highly productive, which can result in elevation of pH due to carbon drawdown in the process of photosynthesis. As noted by Glibert (2012), once pH is elevated, the fundamental physical–chemical relationships related to P adsorption–desorption in sediments change, as does N biogeochemistry (Jordan et al. 2008; Gao et al. 2012). Moreover, under increased pH conditions, the biogeochemistry of calcification is altered, increasing the potential for calcification and the growth of calcifying organisms. Thus, the change in the abundance of the clam *Potamocorbula amurensis* from the time of its introduction in the mid-1980s to 2005 has been shown to be highly and positively correlated to the increase in total N:total P ($r^2 = 0.46$; $n = 20$; $p < 0.01$; all data log transformed), and the average annual abundance of this species has also been found to be highly and positively correlated with mean annual average pH in the estuary ($r^2 = 0.64$; $n = 19$; $p < 0.01$; species abundance data log transformed) (Glibert et al. 2011). Interestingly, the Potomac River, Rhine River and the Ebro River have had similar invasions of macrophytes and *Corbicula* clams that relate to increases in N:P loading (Ibanez et al. 2008; Glibert et al. 2011; Glibert 2012).

In the Bay-Delta estuary, data show top-down grazing of phytoplankton by the clam *P. amurensis* exerts a strong control on phytoplankton biomass, as is also the case for other systems when invaded by bivalve mollusks. Prior interpretations, emphasizing stochastic invasions largely via ballast water exchange imply that the invasive event was the ultimate cause of the change in top-down control of phytoplankton. The ecological stoichiometric interpretation does not preclude strong top-down control of selected component organisms, nor ballast water exchange as the mechanism of introduction. The distinction is that, at the overall ecosystem level, the structuring of species is affected by alterations in nutrients and ecosystem biogeochemistry.

The arguments presented here make the case that bottom-up control contributed to the conditions that allowed *P. amurensis* to become a dominant regulator of phytoplankton production. In other words, invasive species effects and nutrient effects are interrelated. This interpretation is consistent with Ware and Thompson’s (2005) insights from a broad survey of the relative contributions of “bottom-up” vs. “top-down” factors that potentially control fish catch in the coastal waters of the western U.S.; they, too, reported that bottom-up factors were more important.

Several recent reviews have investigated the stoichiometry of fish (Sterner and George 2000; Hendrixson et al. 2007; McIntyre and Flecker 2010). Not only does a strong shift in body N:P occur with growth stage (Pilati and Vanni 2007), but strong differences between taxonomic families also do. In fact, Hendrixson et al. (2007) demonstrated, for 20 families of fish, that a phylogenic tree could be developed based on the body nutrient composition.

In the Bay-Delta estuary, numerous changes in fish community composition occurred in relation to phytoplankton and zooplankton changes, and to N:P (Glibert 2010; Glibert et al. 2011) (Figure 25). Glibert et al. (2011) also found that total P "explained at least as much of the variability in delta smelt as did the [Feyrer et al. 2011] habitat index, and dinoflagellate abundance explained even more." Unlike the X2 correlations where the underlying mechanisms driving the correlations are largely unknown, the nutrient relationships have a strong mechanistic explanation in ecological stoichiometry and stable state principles. For this reason, there is relatively low uncertainty that changes in nutrient stoichiometry in the Bay-Delta estuary, achieved through both external forces (altered land-based nutrient loads) and internal, organism- driven, assimilative and dissimilative processes, are related to community compositional changes (Glibert et al. 2011; Glibert 2012).

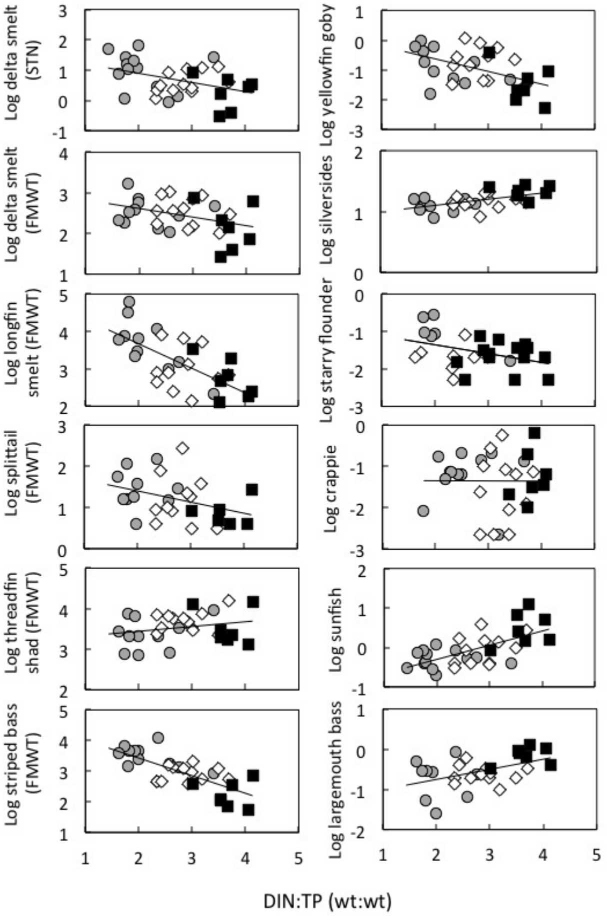


Figure 25. **Changes in the abundance of major fishes in relation to ratio of dissolved inorganic nitrogen to total phosphorus from 1975-2005. Different periods of time are represented by different symbols: 1975-1986, filled circles; 1987-1999, diamonds; post-1999 filled squares. All data were log-transformed. The correlations for all fish except crappie were significant (p<0.05) in these data as well as in data that were detrended. Source: Glibert et al. (2011).**

Eutrophication, ecological stoichiometry and alternate stable state theories combine to serve as a unifying framework for understanding the complexity of responses not only in the Bay-Delta estuary but also, more generally, in many comparative systems. This interpretation does not negate the importance of ecological invasions, habitat changes, multiple stressors and food-web complexities, but adds an explanatory mechanism to those interpretations through biogeochemistry and organismal stoichiometry. Ecological stoichiometry affects systems by setting elemental constraints on the growth of organisms. This, in turn, affects food quality and the relationships between predators and prey.

A growing body of literature documents improvements in ecosystem functions where nutrient loading is reduced and stoichiometric balance is restored. Reducing nutrient loading in the Chesapeake Bay, Tampa Bay, and coastal areas of Denmark has proven to be effective at reversing the harmful effects of previously undertreated discharges and restoring the native food webs. For example, within several years of increasing nutrient removal at the Blue Plains treatment plant in Washington DC, N:P ratios in the Potomac River declined, the abundance of the invasive *Hydrilla verticillata* and *Corbicula fluminea* began to decline (Figure 26 showing *C. Fluminea Halic*), and the abundance of native grasses increased (Ruhl and Rybicki 2010).

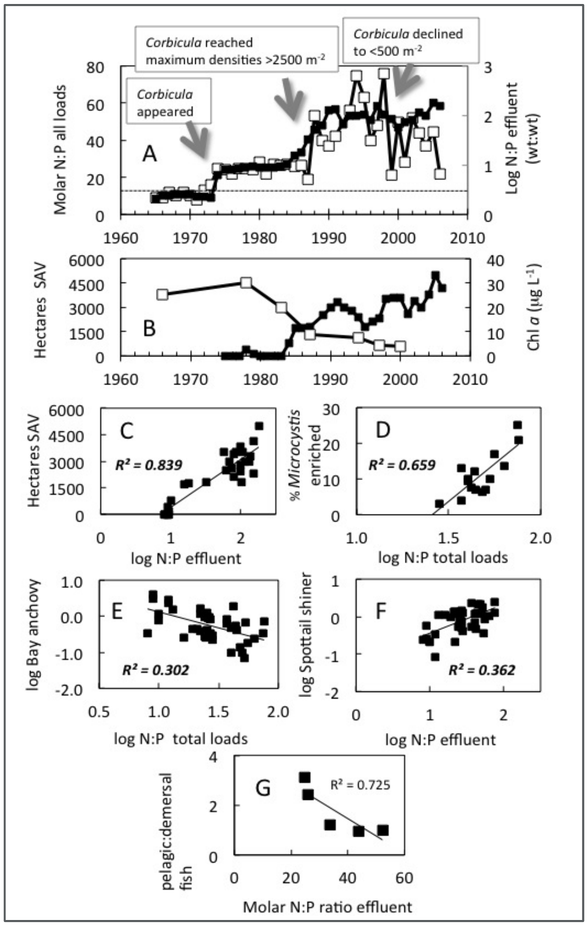


Figure 26. Comparative relationships for the Potomac River. Panel A shows the change in effluent N loading and the relative abundance of the invasive clam, *Corbicula fluminea*. C. fluminea appeared coincident with a sharp increase in N:P and increased in abundance as N:P increased. When N:P decreased sharply around 1999, *C. fluminea* abundance also declined sharply from >2500 m^{-2} to <500 m^{-2} Data derived from Dresler and Cory (1980), Jaworski et al. (2007), and Cummins et al. (2010). Figure reproduced from Glibert et al. (2011).

Tampa Bay provides another important example. Eutrophication problems in Tampa Bay were severe in the 1970s, with N loads approximating 24 tons per day, about half of which was due to point source effluent (Greening and Janicki 2006). Several years after nitrogen and phosphorus reductions were achieved, native seagrass began to increase. Lower nutrient discharges also had positive effects on the coastal waters around the island of Funen, Denmark (Rask et al. 1999). Since the mid 1980s, there has been a roughly 50% reduction in the loading of N and P in the region due to point source reductions. Again, native grasses returned and low oxygen problems were reversed.

Cloern (2001) provides additional examples of recovery following reductions in nutrient and waste inputs (Figure 27). Citing other researchers, Cloern (2001) shows improvements in dissolved oxygen levels in the Forth Estuary in Scotland following improvements in wastewater treatment. Citing a second study, Cloern (2001) shows increases in fish diversity in the Thames Estuary following improvements in wastewater treatment there (Figure 27).

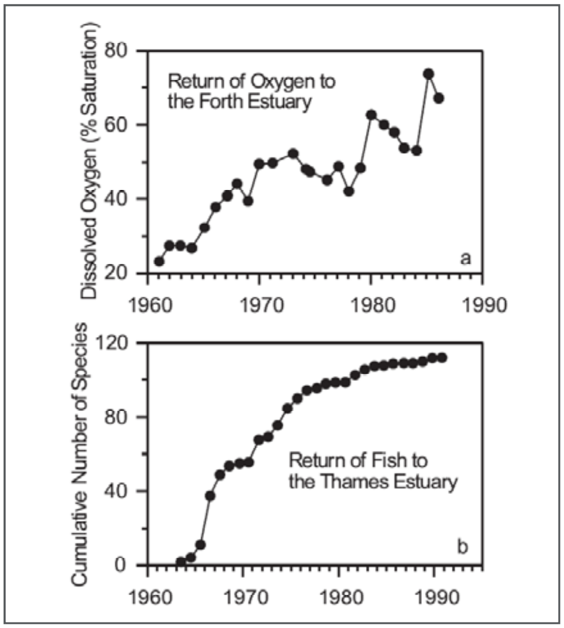


Figure 27. Two examples of recovery following actions to restore water quality in estuaries impacted by nutrient and waste inputs: (a) trend of increasing dissolved oxygen concentration (summer months) in the Forth Estuary, Scotland, following Improvements in wastewater treatment; (b) trend of increasing diversity of fishes in the Thames Estuary following implementation of advanced wastewater treatment and increases in oxygen concentrations (Source: Figure 20 from Cloern 2001).

Release of stored water will not contribute to restoring the N:P balance. As demonstrated by Figure 28, outflow, as measured by the location of the X2 isohaline in the estuary, is correlated with concentrations of individual nutrients (e.g., total phosphorus and phosphate concentration are correlated with the amount of outflow); however, because the sources and geochemical processes governing nitrogen and phosphorus differ, there is no relationship between outflow and the N:P ratio (Glibert et al. 2011).

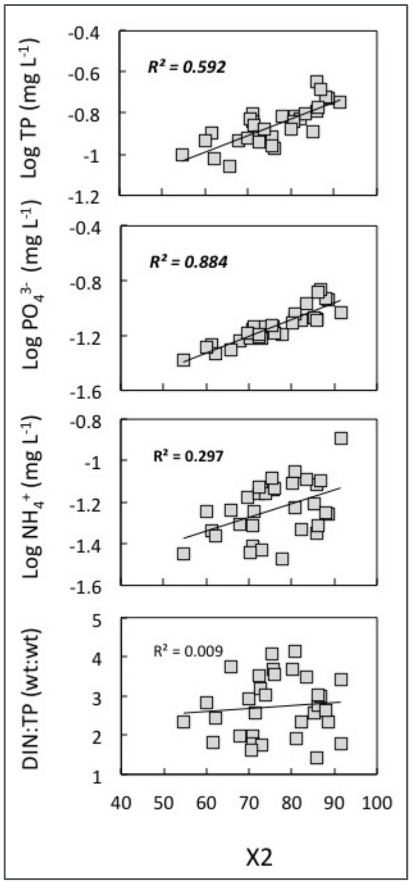


Figure 28. **Comparison of the relationship between nutrients and X2 for the time course from 1975–2005. Outflow, as measured by X2, does affect concentrations of individual nutrients (top three graphs), but does not affect N:P ratio (bottom graph) (Source: Figure 36 from Glibert et al. 2011).**

In summation, changes in nutrient loads are clearly impacting Bay-Delta ecosystem dynamics in complex ways that extend beyond eutrophication. In addition to increases in nutrients, changes in the form of available nutrients (chemical state, oxidized vs. reduced, organic vs. inorganic, dissolved vs. particulate) and the proportion of different nutrients produce adverse effects at both the scale of the primary producers and the entire ecosystem.

2.3.6 Possible Explanation 6: Changes in the land-water interface have negatively impacted Bay-Delta productivity?

Moyle et al. (2010) believe the rich diversity and abundance of aquatic and terrestrial life in the predevelopment Delta imply,

> "…high productivity, which was likely generated by nutrients from the extensive riparian corridors, marshes and seasonal floodplains. High connectivity among the habitats allowed the dispersion of these nutrients throughout the system and into the estuarine food webs, supporting the dense fish populations."

The predevelopment Bay-Delta was an extensive, complex and diverse environment (The Bay Institute 1998). The pre-settlement Sacramento River was narrow, meandering, and sinusoidal, but the massive dredging of the Sacramento River for flood control in the 1920s deepened, widened, and straightened the river, resulting in profound changes in the bathymetry and regional hydrodynamics (James and Singer 2008). From 1860 to 1930, approximately 400,000 acres of tidal marsh were leveed from the channels and converted to Delta farm land, thereby cutting off tidal prisms (Lund et al. 2007). Miles of dendritic channels were eliminated and replaced with deep channels with far less bathymetric diversity (James and Singer 2008). These profound changes to the physical environment resulted in the hardening of the land-water interface and isolating it from natural tidal action and flood events. The loss of floodplain and wetlands habitats has reduced primary production (Baxter et al. 2010), and this is an arena where management actions can be taken to improve the food web.

2.3.6.1 Floodplains

Natural floodplains are among of Earth's most productive and biologically diverse ecosystems (Tockner and Stanford 2002). Floodplains benefit species that directly access them, such as fishes that spawn or forage on floodplains when they are inundated (Moyle et al. 2007). In addition, floodplains can potentially provide regional benefits by exporting food resources such as phytoplankton to downstream systems (Sommer et al. 2004; Ahearn et al. 2006; Lehman et al. 2008).

However, most floodplains in the Bay-Delta estuary and in the watersheds of the Sacramento and San Joaquin Rivers have been severed from their rivers by levees, channelization and flow regulation (Mount 1995). This disconnect affects functional attributes of floodplains, including reduced nutrient replenishment and associated food-web development, and decreased variability of flood-dependent habitats (Jeffres et al. 2008; Opperman et al. 2010).

Historically, floodplains produced high levels of phytoplankton and other algae, particularly during long-duration flooding that occurred in the spring (Sommer et al. 2004; Ahearn et al. 2006). The shallow water depth and long residence time in floodplains facilitated settling of suspended solids, resulting in reduced turbidity and increased total irradiance available for phytoplankton growth in the water column (Tockner et al. 2000). At the Cosumnes River Preserve, the inundated floodplain progressed from a physically driven system when connected to the river floods to a biologically driven pond-like system with increasing temperature and productivity once inflow ceased (Grosholz and Gallo 2006). Periodic small floods boosted aquatic productivity of phytoplankton by delivering new pulses of nutrients, mixing waters, and exchanging organic materials with the river (Ahearn et al. 2006). Aquatic productivity was greater in floodplain ponds than in river sites (5-10 times greater chl-*a* values and 10-100 times greater zooplankton biomass) (Ahearn et al. 2006; Grosholz and Gallo 2006). Zooplankton biomass increased rapidly following each flood event to a peak approximately 7–25 days after disconnection from the river, with highest observed values (approximately 1,000–2,000 mg/m^3) at approximately 21 days (Grosholz and Gallo 2006).

In addition, as reviewed by Lehman et al. (2008), phytoplankton produced on the floodplains are often higher in nutritional quality than phytoplankton found in rivers because they have a wider spherical diameter and thus higher carbon content (Hansen et al. 1994; Lewis et al. 2001). Diatoms and green algae, which are the dominant algal species in the Yolo Bypass (Lehman et al. 2008), have the highest cellular carbon content in the San Francisco Estuary phytoplankton community (Lehman 2000; Hansen et al. 1994).

Studies of the Yolo Bypass provide evidence of the incremental value of floodplains to the Bay-Delta estuary (Sommer et al. 2001a, 2001b, 2008). Chl-*a* levels were significantly higher in the floodplain than in the river and were negatively associated with flow. These results were consistent with longer hydraulic residence times, increased surface area of shallow water, and warmer water temperatures. Copepod and cladoceran densities were similar in the river and its floodplain, and were mostly negatively associated with flow. Chironomid fly abundances were positively correlated with flow (discharge and flow velocity);

these organisms were one to two orders of magnitude more abundant in the Yolo Bypass floodplain than the adjacent Sacramento River channel (Sommer et al. 2001a).

Providing river–floodplain connectivity can rapidly enhance production of lower trophic level organisms (Sommer et al. 2004). In the Yolo Bypass, some food-web organisms respond within days and attain very high densities soon after inundation, including smaller fast-growing algae (e.g., picoplankton, small diatoms, nanoflagellates), vagile organisms such as drift insects, and organisms associated with wetted substrate such as chironomid flies. These organisms, particularly chironomids, provide a food source to fish that is available prior to the development of food-web productivity associated with long residence times (e.g., phytoplankton and zooplankton responses to inundation) (Sommer et al. 2004).

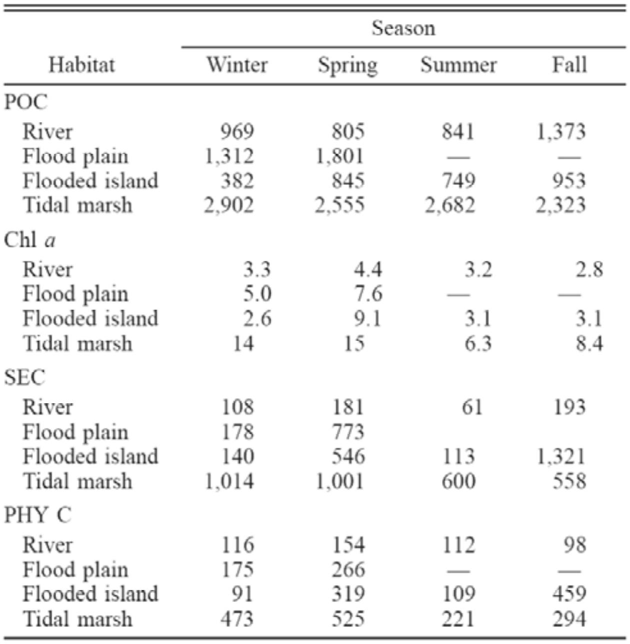
Floodplains have been proposed as "productivity pumps" (Junk et al. 1989) that can export food resources, especially algae, to support food webs in downstream aquatic ecosystems (Sommer et al. 2001b; Ahearn et al. 2006; Lehman et al. 2008). By periodically pulsing small "floodplain activation floods," it may be possible to pump high concentrations of algae to downstream waters (Ahearn et al. 2006). Analysis of suspended algal biomass in the Cosumnes River channel and floodplain by Ahearn et al. (2006) documented an increase in chl-*a* concentrations on the floodplain during periods of river-floodplain disconnection, and subsequent increase in chl-*a* in the river when connection was restored. Lehman et al. (2008) suggested that the quantity and quality of riverine phytoplankton biomass available to the aquatic food web could be enhanced by passing river water through a floodplain such as the Yolo Bypass during the flood season.

2.3.6.2 Wetlands

Over the last 150 years, approximately 95% of the tidal wetlands in the Bay-Delta estuary have been lost due to local development (The Bay Institute 1998). In an investigation of the ecological values of shallow-water areas contiguous and adjacent to tidal wetlands and freshwater marshes, Lopez and her colleagues found that such areas in the estuary support high phytoplankton growth rates (Lopez et al. 2006). Mueller-Solger et al. (2002) conclude, "From a restoration perspective, the results of this study emphasize the importance of tidal marsh and floodplain preservation and restoration as relatively food-rich areas for pelagic primary consumers." Table 1 reproduces Table 1 from their report and shows high levels in all measures of food availability for tidal marsh and floodplains. These findings are consistent with the prevailing belief that wetlands in areas such as Suisun Marsh and the Cache Slough complex contribute to primary production. In a summary of a workshop on Suisun Marsh restoration, Brown (2004) reported that tidal marsh restoration would fuel the estuarine food web. At the same time, such production can be variable and is related to connectivity in aquatic ecosystems driven by advection, dispersion, and gravitational circulation (Brown 2004; Lopez et al. 2006).

Cloern (2007) used a nitrogen-phytoplankton-zooplankton model to illustrate how shallow water bodies sustain fast phytoplankton growth and net autotrophy (photosynthesis exceeds community respiration), whereas deep, light-limited areas within the Delta channels sustain low phytoplankton growth (Jassby et al. 2002) and net heterotrophy. Lopez et al. (2006) found that surplus primary production in shallow areas provided potential subsidies that likely supported zooplankton in neighboring areas, except in areas heavily colonized by the invasive clam *Corbicula fluminea*.

Table 1. Median POC and Chl-*a* concentrations in the <243 µm Seston size fraction and related estimates for Delta habitats and seasons. Scenedesmus equivalent carbon (SEC) concentrations associated with observed Daphnia growth rates; phytoplankton carbon concentrations (PHY C) estimated from Chl-*a* concentrations. All concentrations are in mg L^{-1}. High levels in all measures of food availability are apparent for tidal marsh and floodplain habitats.



	Season			
Habitat	Winter	Spring	Summer	Fall
POC				
River	969	805	841	1,373
Flood plain	1,312	1,801	—	—
Flooded island	382	845	749	953
Tidal marsh	2,902	2,555	2,682	2,323
Chl *a*				
River	3.3	4.4	3.2	2.8
Flood plain	5.0	7.6	—	—
Flooded island	2.6	9.1	3.1	3.1
Tidal marsh	14	15	6.3	8.4
SEC				
River	108	181	61	193
Flood plain	178	773		
Flooded island	140	546	113	1,321
Tidal marsh	1,014	1,001	600	558
PHY C				
River	116	154	112	98
Flood plain	175	266	—	—
Flooded island	91	319	109	459
Tidal marsh	473	525	221	294

Source: Table 1 from Mueller-Solger et al. 2002

In summation, major changes to the Bay-Delta landscape over the past 150 years have resulted in the hardening of the land-water interface, thereby isolating large geographic areas from natural tidal action and flood events. Reduction in primary productivity resulting from losses of wetlands across much of the Bay-Delta estuary is recognized by agency biologists as a key determinant of declines in zooplankton and the native fish that prey on them. Reestablishment of emergent wetlands will contribute to food production for desired fish, provide fish spawning and rearing areas, refugia from predators, and aid in migration and within-Delta dispersal.

3 Changes to the Landscape

In an unaltered state, the physical landscape of an estuary provides valuable spaces for rearing, spawning, migration, and refuge from predators. There is no disagreement that the changes to the Delta landscape have been extensive and have reduced or eliminated many of those functions. Where land and water were once intricately connected, now levees maintain substantial if not complete separation.

Historically, the flood basins of the Bay-Delta and its tributaries were the home to vast tracts of permanent tule marsh or swamps. Areas now with levees were populated with riparian forests of cottonwood, willows, sycamores, elders, ash, walnut, and Valley oak and a thick understory of grape, wild rose, blackberry, poison oak, and other vines (Katibah 1984 as cited in Garone 2011; Kooser et al. 1861). Luxuriant native grasses, often 3 to 5 feet high, covered much of the plains area outside of these flood basins (Garone 2011; Holmes et al. 1915 as cited in Katibah 1984).

Before European settlement began in the early 1800s, the rivers flowed through approximately 400,000 acres of wetlands and other aquatic habitats in the Delta (Delta Stewardship Council 2012). The primary landscapes in the historical Delta included flood basins in the north Delta, tidal islands in the central Delta, and a complex network of dendritic channels formed by riverine processes in the south Delta. Over the last 160 years, 1,335 miles of levees have been constructed to drain wetlands and convert them into farmland. Today, over 95 percent of the wetlands once present in the Delta are gone (Moyle et al. 2012).

3.1 Biological Implications of Changes to the Landscape

Several studies support the notion that access to wetlands is important to the success of many of the Delta’s desired fish species (Moyle et al. 1992; Lindberg and Marzuola 1993; McIvor et al. 1999). Both the DFG and USFWS have made restoration of wetlands a condition of authorizations issued for the CVP and SWP for longfin smelt and delta smelt, respectively.

Floodplain inundation provides spawning and rearing habitat for fish that take advantage of the high productivity on the floodplain (Poff et al. 1997; Sommer et al. 2001a, b; Feyrer et al. 2004; Schramm and Eggleton 2006; Grosholz and Gallo 2006). During these periods of connection to the river, fish can move on and off the floodplain to forage or spawn (Moyle et al. 2007). The low-velocity, shallow, and vegetated conditions of the floodplain serve as a refuge from the fast, turbid waters of the river during high flows (Sommer et al. 2001a; Jeffres et al. 2008).

Large floodplain areas such as the Yolo Bypass (24,000 ha) have the capacity to influence fish abundance and survivorship at the population scale. The duration of inundation of the Yolo Bypass is a strong predictor of year-class strength for Sacramento splittail (*Pogonichthys macrolepidotus*) for the entire Central Valley and Delta system (Sommer et al. 1997; Feyrer et al. 2005).

The Sacramento splittail is perhaps the most floodplain-dependent species in the Delta (Sommer et al. 1997). Adults migrate onto the inundated floodplain to spawn on vegetation in February-March at both the Cosumnes floodplain (Moyle et al. 2007) and the Yolo Bypass (Sommer et al. 2004). Juveniles rear on the floodplain and depart when it drains in April-May, achieving better condition on the floodplain than in river habitats (Ribeiro et al. 2004).

Juvenile Chinook salmon also benefit from floodplains as foraging and refuge habitat. Juveniles migrate downstream onto floodplains in February to March to forage on the abundant invertebrates in the flooded vegetation, prior to emigrating to the sea (Moyle et al. 2007; Grosholz and Gallo 2006). At the Cosumnes

River, growth rates of juveniles reared in enclosures are faster on ephemeral floodplains than in the river (Jeffres et al. 2008) (Figures 29 and 30).

Figure 29. **Comparison of juvenile Chinook salmon reared 54 days at the Cosumnes River Preserve in (1) intertidal river habitat below the floodplain (left) and (2) floodplain vegetation (right). Growth rates of juveniles reared in enclosures were faster on floodplain habitats than in the river (Source: Jeffres et al. 2008).**

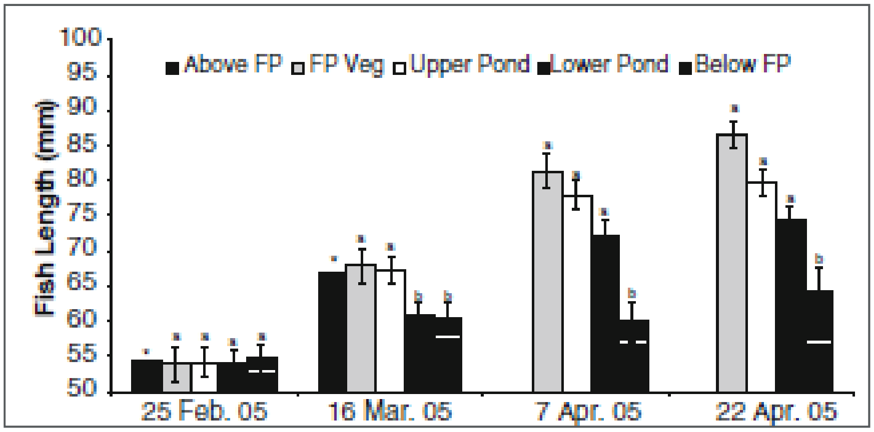


Figure 30. **Size (mean fork length ± standard error) of juvenile Chinook at the Cosumnes River Preserve reared in floodplain habitats (FP Veg,Upper Pond, and Lower Pond) and river channel sites (Above FP and Below FP) over four sampling sessions during the 2005 flood season. Growth rates of juveniles reared in enclosures were faster on ephemeral floodplain habitats than in the river habitats with different letters are statistically different. Asterisks indicate habitats not included in the statistical analysis (Source: Jeffres et al. 2008).**

At the Yolo Bypass, juvenile Chinook salmon grow larger and are in better condition than those in the river (Sommer et al. 2001a). Drifting macroinvertebrates, such as chironomids and terrestrial invertebrates, are an important food resource for fish. Yolo Bypass salmon have significantly more prey in their stomach than salmon collected in the Sacramento River (Sommer et al. 2001a, 2004). Chironomids, the primary food resource for juvenile Chinook, are 1 to 2 orders of magnitude more abundant in the floodplain than the adjacent Sacramento River channel (Sommer et al. 2001a).

Wetlands also provide benefits to delta smelt. Recently, researchers carried out a habitat affinity analysis for delta smelt. While reduced turbidities and unacceptably high water temperatures in summer and early autumn render much of the south and east Delta unacceptable to delta smelt, remaining portions of the Delta where bays, embayments, and larger channels abut shallow waters, marshes, and wetlands provides habitat for the species (Hamilton and Murphy, in review). Hamilton and Murphy conclude that efforts to conserve delta smelt will best be realized, not by altering contemporary flow regimes in the estuary, but by carrying out strategically located marsh and wetland restoration efforts intended to restore functions and processes, where those areas are adjacent to open waters and shallow circumstances that support spawning. The work by Hamilton and Murphy suggests that inattention to the importance of wetlands has contributed to the degraded status of native, pelagic fish in the Delta.

In summation, floodplains, wetlands, and riparian habitat provide valuable landscape features that are used by desired fishes for rearing, spawning, migration, dispersal, and refuge from predators. Physical improvements are needed to restore these features and the functions they provided.

This Page Intentionally Left Blank

4 Changes in Temperature

Another ecosystem change that must be considered is temperature. Jassby (2008b) states,

> "Water temperature trends are of particular interest because the success of both desirable and undesirable species can be highly temperature-sensitive. For example, the nuisance cyanobacterium Microcystis aeruginosa, which has been spreading in the estuary since 1999, has a high optimum temperature over 25°C, depending on the strain (Nalewajko and Murphy 2001). In contrast, the endangered native delta smelt Hypomesus transpacificus appears to have a lethal limit near 25°C (Swanson et al. 2000)."

Jassby (2008b) did not find an overall long-term trend in the mean, minimum or maximum daily water temperature from 1983 to 2007. However, he did find significant monthly trends that ranged from -0.09 to 0.09°C y^{-1}, or more than 2°C over the 25-year period from 1983 to 2007, based on Theil-Sen slopes.

Climate change modeling suggests that water temperatures in the estuary will increase (Cloern et al. 2011; Wagner et al. 2011), although effects throughout the system may not be even (Wagner et al. 2011). A potentially serious effect on water temperature from climate change could be a decrease in the cold water pool of upstream reservoirs as the snowmelt contribution to runoff declines (Cloern et al. 2011). Areas experiencing thermal maxima at or above lethal ranges for native species, such as delta smelt, will increase.

Source of Temperature Changes

Water temperatures in the Bay-Delta are primarily driven by atmospheric influences (Kimmerer 2004; Cloern et al. 2011; Wagner et al. 2011; Jassby 2008a, b). On shorter timescales, flow can affect water temperature in the estuary during high flows that result from storm events (Wagner et al. 2011; Cloern et al. 2011); however, these high flow events occur during the cooler, winter-spring months when water temperatures are not reaching critical levels in the Bay-Delta. Thermal dispersion also influences water temperatures. Monismith et al. (2009) found effects of tidally driven thermal dispersion in the San Joaquin River near its confluence with the Sacramento River. Bathymetric features can also influence site-specific water temperatures by causing upwellings of deeper, cooler oceanic water. Schoellhamer (2001) has reported on the effect of several sills in and around Suisun Bay where gravitational circulation creates turbidity maxima.

Jassby (2008a) conducted a trend analysis and found,"[f]or both the Delta and Suisun Bay, the resulting negative Kendall's rank correlation between flow and temperature during 1996–2005 was not statistically significant: *tau* = -0.33 (*p* = 0.21) and *tau* = -0.022 (*p* = 0.99), respectively." Jassby (2008b) found similar results using temperature data on a shorter time step, stating, "[t]he significant water temperature trends, as one would expect, appear to be driven primarily by corresponding trends in air temperature." And, a model developed by Cloern et al. (2011) was able to predict water temperatures in the Delta using air temperature, insolation, and the previous day's water temperature with an r^2 of 0.964 for the verification period.

4.1 Biological Implications of Temperature Changes

Water temperatures provide an important constraint on ecological function. Examples include effects on fish spawning (Myrick and Cech 2011), swimming performance (Myrick and Cech 2000), metabolism (Myrick and Cech 2011), and mortality (Parker et al. 2011) as well as effects on aquatic invertebrates

(Vannote and Sweeney 1980). Specific examples of species of concern within the estuary that are sensitive to water temperatures at various points in the life cycles include salmonids (Myrick and Cech 2011), the Sacramento splittail (Moyle et al. 2004), and the delta smelt (Bennett 2005). Aquatic plants in tidal wetlands are also sensitive to water temperature (Parker et al. 2011). Temperature changes can alter wetland plant communities, causing changes in available carbon.

The biological implications of climate change effects on water temperatures may be profound. Winter-run Chinook salmon eggs develop in summer, when river temperatures reach their highest levels. Summer river temperatures are projected by Cloern et al. (2011) to reach lethal levels as a result of climate change. In fact, the study by Cloern et al. (2011) indicates an increasing risk of extinction of native species and increasing dominance of non-native species due to effects of climate change. MacNally et al. (2010) found lesser effects of warmer summer temperatures and duration of water temperatures during spawning. Wagner et al. (2011) verifies that climate change will increase the number of days above delta smelt’s thermal maxima (especially along the Sacramento River) and a shift to earlier spawning.

Engineered solutions such as reservoir releases (which have been used for decades to provide temperature refugia for the salmonids in the upper stream reaches) would not be effective at controlling Delta temperatures during the warmer summer and fall seasons when cooler water in the Delta is most needed. There is already a delicate balance between providing sufficient cold water pool releases for each of the salmon runs and maintaining a large enough carryover pool for future dry years. And, as reported by Cloern et al. (2011), this cold water pool may diminish further with climate change. Unfortunately for Delta aquatic organisms, we have no control over atmospheric and oceanic influences, and reservoir releases are unable to affect Delta temperatures during the warmer summer and fall seasons when cooler water in the Delta is most needed. Thus future trends in temperature will need to be considered when considering options to improve conditions for desirable aquatic species.

In summation, water temperatures have warmed and will continue to warm. The number of areas experiencing temperatures above lethal ranges for native species is expected to increase. The biological implications of climate change effects on water temperatures may be profound, including increasing risk of extinction of native species and increasing dominance of nonnative species. Water temperatures in the Bay-Delta are primarily driven by atmospheric influences, although thermal dispersion also influences water temperatures, and bathymetric features can influence site-specific water temperatures. Reservoir releases will be unable to affect water temperatures in the Bay-Delta during the warmer summer and fall seasons when cooler water temperatures are most needed.

5 Changes in Turbidity

Another ecosystem change that must be considered is turbidity. Turbidity is a physical characteristic of water and is an expression of the optical property that causes light to be scattered and absorbed by particles and molecules rather than transmitted in straight lines through a water sample. It is caused by suspended matter or impurities that interfere with the clarity of the water. Constituents of turbidity may include clay, silt, finely divided inorganic and organic matter, soluble colored organic compounds, plankton and other microscopic organisms (USEPA 1999). Turbidity is one among many environmental variables that affect the quality of habitat for aquatic organisms. Monitoring by the DFG and IEP in the Bay-Delta over the past 35 years has documented trends of increased water clarity (Moyle and Bennett 2008), reduced turbidity (Schoellhamer 2011), and declines in chl-*a* (Jassby 2008a; Jassby et al. 2002; Kimmerer et al. 1994).

Water clarity, historically measured as Secchi disc depth (the depth to which a white disc lowered into the water is no longer visible), has been measured by the DFG and IEP several times a month at multiple stations for eight different programs for over 35 years. Moyle and Bennett (2008) documented a trend of increased water clarity between 1976 and 2008, coincident with the decline in delta smelt and striped bass (the POD) and increases in invasive species, including inland silverside and centrarchid species. Increasing water clarity has long been attributed to sediment retention behind dams and the proliferation of submerged aquatic vegetation.

Secchi depth data collected by the DFG and IEP has been compiled into a single data set and averaged by month and then by subregion as shown on Figure 31. As illustrated on these figures, water clarity has been increasing across most of the Delta over the past 35 years.

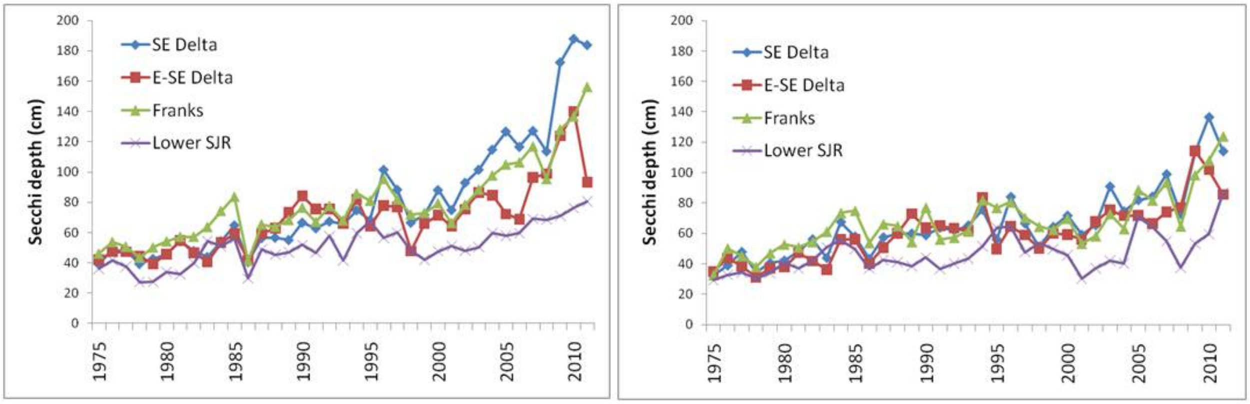


Figure 31. Average July-October (left panel) and March-June (right panel) Secchi depth in the South and Southeast Delta from 1975-2011, showing increasing clarity over the time course. (Data compiled from: Environmental Monitoring Program (conducted by DWR and USBR with assistance from DFG, USGS, and USFWS); Fall Midwater Trawl, Summer Townet Survey, 20-mm Survey, Kodiak Trawl, Monthly Zooplankton Survey, and Bay Study (conducted by DFG); Suisun Marsh Fisheries Monitoring (conducted by UC Davis).

Turbidity in the Bay-Delta has more recently undergone what has been characterized as a “significant step decrease” (Schoellhamer 2011). Turbidity, expressed as suspended sediment concentration (SSC), in the San Francisco Bay decreased by 36% between water years 1991-1998 to 1999-2007 (Schoellhamer 2011). This sudden clearing of the estuarine waters coincided with the decline in abundance of the Delta's desired fishes. Schoellhamer (2011) postulated that the sudden drop in SSC during this period resulted from the depletion in the erodible pool of sediment that had been in the system until the documented step decrease. While sediment supply from the Sacramento River and other tributaries has declined as a result of bank protection, flood bypasses, the construction of dams and reservoirs and resultant trapping of sediment behind dams, and diminishment of sediment pulses from hydraulic mining (Schoellhamer et al. 2005; Wright and Schoellhamer 2004), the step decrease in SSC does not appear to be due to a sudden decrease in river-supplied sediment. The river supply of sediment to San Francisco Bay varies annually and decreased 1.3% per year during the latter half of the 20th century, which does not account for the sudden 36% decrease in suspended mass in 1999. The decrease in turbidity in the estuary is thought to have resulted from a shift from depositional to erosional processes in Suisun, San Pablo, and the Central Bays.

An increase in submerged aquatic vegetation (SAV), including the rapidly expanding *Egeria densa*, also contributes to increased water clarity (Kimmerer 2004). Anderson (1999) states, "In addition to negative impacts on wildlife, *Egeria* acts like a filter and accelerates deposition of suspended materials. Each year, hundreds of tons of organic and inorganic materials are carried downstream with seasonal snowmelt. In areas where *Egeria* has formed dense stands, the water is slowed and materials that are normally transported and spread throughout the Delta and upper Bay become entrapped and settle." Hestir et al. (2010) investigated the relationship between turbidity and SAV cover and found that SAV cover explains an estimated 21-70% of the trend of decreasing turbidity in the Bay-Delta.

Since plankton and other microscopic organisms are constituents of turbidity, the decline in turbidity may also be related to the decline in primary and secondary productivity described in Section 2.0, Major Changes to the Food Web.

While suspended sediment concentrations in the Bay-delta rise following significant rainfall, releases from upstream reservoirs are not an effective means of delivering suspended sediment to the Delta for the following reasons:

- Releases from reservoirs would not transport the sediment impounded behind dams into the rivers downstream of the dams because those sediments have been deposited to the bed of the reservoirs and the flow velocities in the reservoirs are too low to mobilize them.
- Water released from rim dams is very clear. The clear water released from the dams erodes downstream channels. Over time, the sediment transport capacity of the channel is reduced as the channel becomes incised and armored (Schoellhamer 2005). Vegetation encroachment along the channel may also reduce the sediment transport capacity of the channels downstream from dams.
- The Sacramento River is the major contributor of sediment to the Delta, delivering approximately seven times the sediment yield of the San Joaquin River (Oltmann et al. 1999). Wright and Schoellhamer (2004) estimated that during the period 1999-2002, 85% of the suspended sediment entering the Delta came from the Sacramento River (including the Yolo bypass), 13% came from the San Joaquin River and 2% was from eastside tributaries. The well-armored levees in the Delta prevent them from being significant sources of sediment.
- While the Sacramento River delivers much of the sediment transported into the Delta, most of that sediment is not transported into the Delta's network of channels. In a study of sediment budgets for the years 1999-2002, Wright and Schoellhamer (2004) estimated that at least 82% of the sediment entering the Delta from the Sacramento River watershed either deposited along the Sacramento River or moved past Mallard Island and into San Francisco Bay. They estimated that not more than 18% of the Sacramento River sediment moves towards the San Joaquin River. On

the San Joaquin River, there was a significant loss of sediment between Vernalis and Stockton (64% over the period 1999-2002), due to deposition in this reach or its entry into the south delta channel complex through Middle River (Wright and Schoellhamer 2004).

- Although Wright and Schoellhamer did not model sediment transport dynamics associated with reservoir releases, given the transport processes they describe for the Delta, one can infer from their study results that relatively little sediment transported by the Sacramento River from reservoir releases would move towards the San Joaquin River.
- The delivery of suspended-sediment from the Sacramento River has decreased by about one-half between 1957 and 2001 after the two major reservoirs in the watershed (Shasta and Folsom dams) were constructed (Wright and Schoellhamer 2004). This trend is likely caused by the depletion of erodible sediment from hydraulic mining in the late 1800s, reservoir sedimentation and bank protection. Increased releases from Sacramento River watershed reservoirs would not reverse these factors.

5.1 Biological Implications of Turbidity Changes

The changes in water clarity have significant implications for several Bay-Delta fish species, with particular significance for delta smelt. Clearer water with abundant submerged aquatic vegetation favors centrarchid fishes and is less desirable habitat for delta smelt (Moyle and Bennett 2008). Turbidity appears to be a critical factor for delta smelt larval feeding, providing better contrast between prey and their background, enabling larval predators to better locate their prey (Boehloert and Morgan 1985 in Lindberg et al. 2000). The physical components of “green” or turbid water rather than chemical properties of algal filtrate that contribute to turbidity appear to be important (Lindberg et al. 2000).

In laboratory experiments on feeding behavior of delta smelt larvae, Baskerville-Bridges et al. (2004) observed maximum feeding responses at the highest algal concentrations and light intensity, and relatively high prey density, supporting the conclusion that they are “…better able to forage under turbid conditions, which may also provide refuge from predators.” Baskerville-Bridges et al. (2003) found that larval delta smelt in culture were most effective at feeding in the highest light intensity and algal concentrations of the study. A second experiment showed that feeding responses were very low without algae present, but dramatically increased at high concentrations; and high rotifer (prey) density significantly enhanced feeding behaviors.

In considering ramifications of decreased SSC, Schoellhamer (2011) stated:

> “Reduced SSC may be one of several factors that have contributed to a collapse of several San Francisco Bay estuary fish species that occurred around 2000 (Sommer et al. 2007). Abundance of some fish species increases in more turbid waters (Feyrer et al. 2007). The population collapse has had the most serious consequences for delta smelt which require turbid water for successful feeding and predator avoidance. The relation between decreased SSC and fish decline, however, is not well established, and the concurrence of less SSC, more phytoplankton, and fewer fish merits additional study.”

Moyle (2002) describes the importance of turbidity for delta smelt: "…individual fish apparently hang out in the water column and rely on their small size and transparency to hide them from predators in turbid water. The fact that they are rarely found in the stomachs of such predatory fish as striped bass, white catfish, and black crappie, even when they are abundant is a good indication that this strategy is successful."

However, there are also biological benefits of reduced turbidity. Turbidity can lead to light limitation for primary producers. Several researchers have hypothesized that light penetration controls primary productivity in the Bay-Delta (e.g., Cole and Cloern 1984; Kimmerer 2004). With decreasing turbidity, there is more light for phytoplankton. More light also potentially reaches the benthos where benthic primary productivity can also increase, providing another food source.

In summation, water clarity in the Bay-Delta estuary has increased over the past 35 years. This change has significant implications for several fish species, including delta smelt, as turbidity appears to be a critical factor for delta smelt larval feeding. With decreasing turbidity, more light penetration could increase primary productivity. Reservoirs on the major tributaries have reduced sediment input to the Bay-Delta and the sediment transport capacity of channels below these reservoirs decreases over time as the channels become incised and armored. While suspended sediment concentrations in the Bay-delta rise following significant rainfall, releases from upstream reservoirs are not an effective means of delivering suspended sediment to the Delta.

6 Changes in Through-Delta Flows

The final determinant of ecosystem change that has been proffered as important in the decline of the Delta's desired fishes is changes to through-Delta flows and the location of the low-salinity-zone. Enright and Culberson (2010) did an extensive review of trends in Delta outflow and salinity. They examined precipitation, outflow, and salinity trends before and after 1968 to discern outflow and salinity response to CVP and SWP operations (they also include analysis of pre- and post-Suisun Marsh salinity control gate operations, which began in 1988). They conclude that the data do not verify variability reduction; rather, annual and by-month salinity variability is generally greater in the post-project period: coefficients of variability for precipitation, outflow, and salinity increased after the projects were initiated. This increase suggests that more powerful mechanisms are at play including land-use changes and climate, which overpower the homogenizing influence of appropriations of water, including those by the CVP and SWP, when considering long-term trends.

A detailed analysis of historical outflow, including outflow as measured by the location of X2 over the period of record 1922–2011 and a description of some of the causes of identified changes in outflow over time will be submitted for the Analytical Tools for Evaluating Water Supply, Hydrodynamic and Hydropower Effects Workshop. Here, the paper describes the difference between "unimpaired flow" and "natural flow" as well as some of the uncertainties regarding application of a natural flow regime to a highly modified estuary such as the Bay-Delta estuary.

6.1 Difference between "Unimpaired" and "Natural" Flow

The 2010 Flow Criteria Report identifies a percentage of unimpaired flows as an approach to improving conditions in the Bay-Delta estuary. That approach fails to account for the fact that unimpaired flow is a calculation of a hypothetical condition that never existed in the Bay-Delta. This section describes "natural" Delta outflow during the predevelopment era. It shows that unimpaired flows are not natural flows. An extensive investigation of acreages of undeveloped native land cover and associated ET in the Bay-Delta estuary is under development and will be submitted during the Analytical Tools Workshop.

Natural flow approximates the flows under which native species evolved. Historic rim inflows flooded out of the original river channels into wetlands and floodplains that reduced the flood peaks and supported vast acreages of natural vegetation. Natural flow patterns did not approximate flows under the currently configured system, where outflow rushes through rock-lined channels surrounded by levees. Native species did not evolve in an environment with long-term, annual-average unimpaired outflow of 25 MAF/year.

The word "natural" connotes the Central Valley landscape in a predevelopment state. Under natural conditions, the Central Valley functioned as a series of natural reservoirs, located along the major river courses, rather than at the headwaters of the streams. These streamside reservoirs filled and drained every year. Dense tule marshes occupied these reservoirs and evapotranspired significant amounts of the water that flooded into them. The main river channels were lined by wide, naturally formed levees that were much larger and more developed along the Sacramento River than along the San Joaquin River (Hall 1880, Part II, p. 51). Lush riparian forests occupied these naturally occurring levees and uplands.

Under natural conditions, groundwater generally moved from recharge areas along the sides of the valley toward topographically lower areas in the central part of the valley, where it discharged primarily as ET from marshes and riparian forests (The Bay Institute 1998 ,Sec. IV.B.2; Bertoldi et al. 1991, pp. A17, A23, Fig. 14A; Williamson et al. 1989 p. D33; Davis 1959, p. 86). Groundwater was near the surface in much of the Valley (Bryan 1915, Plate 11 and p. 19; Kooser et al. 1861, p. 265). The USGS estimates that under natural conditions, direct evapotranspiration from groundwater would occur in areas where the groundwater table was less than 10 feet below the surface. They estimated that under natural conditions, groundwater levels were less than 10 feet below the surface over about 62 percent or 8,000 square miles of the Central Valley (Williamson et al. 1989, p. D40). The groundwater system was in a state of dynamic equilibrium. Natural recharge was balanced by natural discharge. This balance has been recently confirmed for the San Joaquin Valley (excluding the Tulare Basin) using a physically based, surface-subsurface numerical model (HydroGeoSphere - see Bolger et al. 2011, pp. 322-330).

“Unimpaired” flows, on the other hand, are calculated flows that include various adjustments to remove consumptive use. Unimpaired flow is rim flow, delivered to the Bay-Delta through contemporary river channels, but without any reduction for storage or upstream use. Unimpaired flow is a measure of how much water is available for use, upstream of the reservoirs with current channel configurations.

Unimpaired outflow is greater than natural outflow. The State Water Board proposed essentially the same unimpaired flow approach in its 1987 Bay Delta Plan hearings. Testimony presented in those hearings, which is just as relevant today as it was then, led the DWR to conclude:

> *“Since unimpaired flow estimates assume present channel configurations and levee and flood bypass systems, they are not the same as natural flows (i.e., flows that occurred in a state of nature, before development). Natural flows through the Delta would probably be far smaller than unimpaired flows due to consumptive use by extensive natural marshes and riparian areas that were later leveed and reclaimed. Monthly distribution of flows would also be different”* (California Department of Water Resources (DWR) 1987, p. 10).

In those 1987 hearings, general agreement was reached with the DWR regarding the conclusion that natural outflow was likely lower than current outflow. Dr. Leopold, a member of the National Academy of Sciences, testified that he agreed with DWR’s conclusion that natural flow would be lower than the estimates of unimpaired flow (1987 Bay Delta Plan Reporter’s Transcripts, LVI, p. 60). Dr. Phyllis Fox (1987) presented extensive analysis on the distinction between natural and unimpaired flows, including in State Water Contractor Exhibits 260, 262, 276, 281, 351, 352, and 353 as well as indirect and rebuttal testimony. Dr. Horne, a Professor of Ecology in the Civil Engineering Department of the University of California at Berkeley, concurred and testified that he had made similar natural flow calculations.

6.2 Biological Functions of Natural Flows

As described above, unimpaired flows are not the same as natural flows. This section describes the scientific uncertainty associated with a restoration effort that is based on release or bypass of a percent of “natural” flow in a system like the Bay-Delta estuary, which as described above is a highly modified system.

The natural flow regime is a concept that evolved for riverine systems as a synthesis from the research and concepts of many earlier authors (Poff et al. 1997). Ecologists, geomorphologists, and other scientists studying rivers and streams recognized that human alteration of flows and physical changes to the environment within watersheds were contributors to ecological degradation, declines and extirpation of species, changes in channels, changes in sediment transport and supply, loss of riparian vegetation, declines in water quality and availability, and more frequent and intense flooding. However, while individual causes were recognized as affecting individual components of the rivers by many researchers, little analytical integration occurred due to the complexity and periodicity of flows.

The concept of the natural flow regime, as indicated by the five flow characteristics described by Richter et al. (1996) and Poff et al. (1997), magnitude, frequency, duration, timing (daily and seasonal), and rate of change, represents maintenance of flow characteristics that have **implied** relationships to physical and ecological processes, but not **explicit** relationships. In fact, one of the concerns, expressed by Poff and Zimmerman (2010) and others is that the relationships between ecological processes and flow are not well understood and require further study.

Although the intent behind implementing a natural flow regime is to protect unaltered or largely unaltered systems, or to assist in conservation or restoration of altered systems, much of the literature surrounding the application of the natural flow regime paradigm addresses measures to restore ecosystem and physical processes in largely unaltered systems. Poff et al. (1997) asked and answered,

> *"Can reestablishing the natural flow regime serve as a useful management and restoration goal? We believe that it can, although to varying degrees,* ***depending on the present extent of human intervention and flow alteration affecting a particular river****"* [emphasis added].

Poff and Zimmerman (2010) reviewed 165 papers related to the natural flow regime. A narrative summary of the reported results strongly corroborated previous, less comprehensive, reviews by documenting strong and variable ecological responses to all types of flow alteration. Study results revealed some sensitivity of different ecological groups to alterations in flow magnitudes, but robust statistical relationships were not supported. The Poff and Zimmerman results revealed:

> *"Macroinvertebrates showed mixed responses to change in flow magnitude, with abundance and diversity both* ***increasing*** *and* ***decreasing*** *in response to elevated flows and to reduced flows. Fish abundance, diversity and demographic rates consistently declined in response to both* ***elevated*** *and* ***reduced*** *flow magnitude. Riparian vegetation metrics both* ***increased*** *and* ***decreased*** *in response to reduced peak flows, with increases reflecting mostly enhanced non-woody vegetative cover or encroachment into the stream channel."* (emphasis added).

Of the 165 papers, 92 percent concluded adverse response of ecological metrics to flow alterations, whereas 13 percent reported beneficial ecological metric responses. The authors state that their analyses did not support use of the existing literature to develop general, transferable quantitative relationships between flow alteration and ecological response. They did believe that the results support the inference that flow alteration is associated with ecological change and that the risk of ecological change increases with increasing magnitude of flow alteration. However, it is clear from the review that these ecological changes resulting from flow alteration include both adverse and beneficial outcomes to ecological metrics. The authors point out the need for careful monitoring, especially before and after flow alterations.

As Poff and Zimmerman (2010) explained, "Given that alteration of flow regimes is typically confounded with other environmental factors, we would not necessarily expect unambiguous relationships between single measures of flow alteration and ecological response." These confounding relationships have also been observed by other researchers. Bunn and Arthington (2002) describe the uncertainties associated with attempting to restore "natural" flow to promote ecological restoration.

> *"In writing this review, we often encountered reports of river systems affected by multiple stressors and were unable to definitely separate the impacts of altered flow regimes from those of the myriad of other factors and interactions. How much of an observed decline in species diversity can be attributed directly to modified flow compared with diffuse inputs of nutrients and other contaminants? A similar problem occurs in our attempt to unravel the cause and effect of exotic species on aquatic diversity. Is an observed decline in native fish species the result of a modified flow regime or direct impact of an introduced*

> *species (or both)? Alternatively, is the proliferation of exotic species a direct response to the modified flow or the decline in native species (or both)? Ecological science is not yet able to answer these questions, important as they are.*
>
> *Ecologists still have much to learn about the ecological significance of individual flow events and sequences of events, and descriptive science can take us only so far in unraveling these linkages. The advice from aquatic ecologists on environmental flows might be regarded at this point in time* ***as largely untested hypotheses about the flows that aquatic organisms need and how rivers function in relation to flow regime****"*[emphasis added].

In largely unaltered river systems, the importance of flow in sustaining biodiversity and ecological integrity is well established (Poff et al. 1997; Hart and Finelli 1999; Bunn and Arthington 2002; Poff and Zimmerman 2010). However, establishing an appropriate flow regime for an estuary is not as simple (Pierson et al. 2002). Within an estuary, freshwater inflow mixes with seawater through physical mechanisms of density gradients, gravitational circulation, tidal action, and wind energy, creating variable brackish conditions for a variety of species during all or a part of their life cycles. Complex ecological and biological processes occur within estuaries, primarily due to their dynamic nature, complex freshwater-seawater mixing processes, random influences, antecedent conditions, and complex ecological linkages. Key differences between riverine and estuarine systems are summarized in Pierson et al. (2002) and presented in Table 2.

Table 2. Key Relevant Differences between Fluvial and Estuarine Systems (Source: Pierson et al. 2002)

Characteristic	Rivers	Estuaries
1. Flow Direction	Unidirectional	Reversing
2. Depth determined in	Flow	Primarily tides
3. Flow cross section determined by	Sedimentary regime	Sedimentary regime, flocculation, littoral drift
4. Water masses	Fresh only	Fresh and salt
5. Pollutant flushing	Rainfall runoff	Rainfall runoff and tidal flows
6. Water Quality changes	Downstream of source	Both upstream and downstream of source
7. Antecedent effects in relation to physical and chemical character	Moderate	Potentially very important
8. Biota	Limited diversity	More diverse
9. Ecological interactions	Less complicated	Much more complicated
10. Size of literature pertinent to environmental flows	Large	Small
11. Understanding of environmental flow effects	Limited	Very limited

Mechanisms through which freshwater flow contributes to desired characteristics in a largely unaltered system are summarized below by Pierson et al. (2002), followed by a discussion of how these mechanisms would respond in the highly altered Bay-Delta estuary.

- Freshwater inflow influences the volume and distribution of brackish water: the variability in freshwater inflow influences the variability in brackish water both longitudinally and vertically, and within and between years.

In a largely unaltered system, brackish water variability affects the distribution of estuarine organisms (plants, invertebrates, and vertebrates), triggers changes in their behavior such as daily, seasonal, or interannual spatial distribution or movements within the estuary, positioning within the water column, or reproductive behavior (spatial distribution of suitable spawning and/or rearing habitat conditions, mates), and affects their ability to forage by changing the daily, seasonal, and interannual distribution of nutrients and/or food organisms.

In the highly altered Bay-Delta, the efficacy of freshwater inflow to contribute to these characteristics is reduced due to changes in the volume and variability based not only on human water demands, but current regulations for protected species. Human water demands reduce the volume of freshwater inflow on an annual basis, and change the in-season hydrograph for agriculture and urban use. Layered on that are the regulations for protected species.

California native fishes evolved in a complex and dynamic environment, thereby promoting a competitive advantage over non-native species. Richness of the fauna has been shown to increase as habitat complexity increases (Bunn and Arthington 2002). In the highly altered Bay-Delta this function cannot be restored with reservoir releases. Varying flows in leveed, riprapped channels will not increase the complexity of species habitat, nor will it cause the hydrodynamics to be more dynamic at a variety of spatial scales. ***However, the BDCP can restore this flow function through habitat restoration that includes strategic levee breaches and tidal marsh restoration. This flow function will be discussed in greater detail in the submittals for the pelagic and salmonid workshops.***

- Freshwater inflow provides nutrient enrichment[11]: in an estuary, enrichment mechanisms include (1) bulk transport of river nutrients (both soluble and particulate), (2) resuspension of nutrients within the estuary through gravitational circulation, and (3) enhanced dispersion of nutrients down the estuary due to salinity stratification. The nutrient enrichment of estuaries from freshwater inflow makes them some of the most productive aquatic ecosystems (Pierson et al. 2002).

In the highly altered Sacramento-San Joaquin River watersheds, transport of river nutrients is diminished due to diking of floodplain habitat, removal of riparian habitat, and channelizing, deepening, and widening of waterways for flood control, navigation, and agricultural and urban land reclamation.

Similarly, within the highly altered Bay-Delta, nutrient input is diminished due to diking of tidal perennial aquatic habitat, tidal mudflats, tidal marsh, floodplain and seasonal wetlands and riparian habitats, and channelizing, deepening, and widening of estuarine waterways for flood control, navigation, and agricultural and urban land reclamation.

The nutrient enrichment function of river flows has not been corroborated by phytoplankton and zooplankton production in the Bay-Delta as previously described in Section 2. In the waters of the Bay-Delta, analyses that have relied primarily on biomass measurements have shown weak effects of freshwater flow on primary and secondary production (Kimmerer 2002a; Kimmerer et al. 2009). In a recent study, Kimmerer et al. (2012) were the first to examine the response of primary productivity within the San Francisco Estuary across a wide range of freshwater flows during the spring-summer period of 2006–2007. They found that temporal and spatial variability in production was small. When they combined and compared data from 2006–2007 with long-term monitoring data, they did not

---

[11] It should be noted that nutrient enrichment in this context refers to both soluble and particulate nutrients, including nutrients bound in primary and secondary productivity, nutrients in detrital material and microbial nutrients.

discern patterns in production and biomass, even though time and salinity levels were variable. More importantly, they found that primary production within the low-salinity zone was unresponsive to variation in freshwater flow, in contrast to findings in other estuaries (Kimmerer et al. 2012).

The lack of response in primary productivity relative to freshwater flow observed by Kimmerer et al. (2012) contrasts with results from other systems and emphasizes the need to carefully consider the physical, chemical, and biological attributes of estuaries, and the Bay-Delta in particular, which influence how freshwater flow affects their food webs and subsequently higher-trophic-level organisms.

- Freshwater inflow transports depositional material: depositional material is transported in an estuary through the same three mechanisms described above: bulk transport, resuspension, and longitudinal transport. Depositional material is important to the maintenance of tidal perennial aquatic areas, tidal mudflats, tidal marsh, floodplain and seasonal wetlands, and riparian areas through the seasonal and spatial distribution of sediment deposition and/or erosion (Collier et al. 1997; Poff et al. 1997; Flannery et al. 2002). These landscape cover types are very productive and diverse, providing food and protection to many species of plants and animals.

In the highly altered Bay-Delta, sediment transport from rivers is reduced and/or altered due to dam emplacement and channel hardening. In addition, the function of deposition is reduced by alterations in the estuary that disconnect the rivers from the landscapes dependent on the deposition. This function cannot be restored with reservoir releases. Unlike natural rain events, reservoir releases do not wash sediment down from the upper watersheds and into the Delta and they do not overflow levees into tidal and wetland areas needing the sediment. Compounding altered sediment dynamics, decreased sediment loads may increase erosion in the Bay-Delta and contribute to remobilization of buried contaminants (McKee et al. 2006). Estimates of sediment inputs to the San Francisco Estuary indicate a shifting equilibrium dynamic. Spanning years 1957 through 2001, Wright and Schoellhamer (2004) show that the delivery of suspended sediment from the Sacramento River to San Francisco Bay has decreased by approximately one-half. The Sacramento River is considered the primary source of new sediment input into the estuary, and delivers approximately 7 times the sediment yield of the San Joaquin River (Wright and Schoellhamer 2004). See Section 5 for more discussion on turbidity.

Deposition is reduced by alterations in the estuary including diking of tidal perennial aquatic habitat, tidal mudflats, tidal marsh, floodplain and seasonal wetlands and riparian habitats, and channelizing, deepening, and widening of estuarine waterways for flood control, navigation, and agricultural and urban land reclamation. Tidal marshes have been diked and drained to create farmland, evaporation ponds for salt, and residential and industrial land. Such land reclamations were largely complete by the end of the 1920s. Today, approximately 125 square kilometers of the original 2,200 square kilometers of tidal marsh remains in the estuary (Nichols et al. 1986). Throughout the estuary, overall, an estimated 79 percent of historic tideland marshes have been lost to diking and development (Goals Project 1999). Historical land reclamation is evident today; the Delta is comprised of around 70 island tracts, surrounded by 1,100 miles of levees and 700 miles of interspersed waterways. Over the subsequent decades, decomposition of exposed peat soils resulted in profound subsidence of reclaimed land with most of the Delta now below sea level and a large portion of the western and central Delta at least 15 to 25 feet below sea level (USGS 2008).

- Freshwater inflow inundates seasonal floodplain and wetland and riparian habitats: Floodplain habitats and seasonal wetlands offer a variety of relatively shallow water habitats, some with submerged vegetation. Riparian habitat within an unaltered estuary includes all successional stages of woody riparian vegetation and provides important cover in the form of shaded riverine habitat. Floodplain habitats are particularly important to the growth of important migratory fish species such as juvenile salmon. (Pierson et al. 2002).

In the highly altered Bay-Delta and upper watershed, the combination of dam construction, channel deepening and widening, land reclamation, and levee building reduce the frequency, magnitude, and duration of inundation of floodplains and seasonal wetlands. Those functions cannot be restored with reservoir releases. Physical intervention is needed to alter the landscape to restore this connection. However, the BDCP can restore this flow function through modifications to the landscape and changes to water control structures, like notching the Fremont Weir, to extend periods of flooding in the Yolo-Bypass. See Section 2.3.6.1 for more discussion of floodplains.

- Freshwater inflow promotes native vegetation cover and composition, and leaf litter deposition and decomposition: In an unaltered system, water table depth and periodic natural flooding promotes vegetation growth (Rood et al. 1995; Molles et al. 1995). In the highly altered Bay-Delta this function cannot be restored with reservoir releases. Most of the large rivers are largely bordered by rock-lined levees. The levees would have to be set back and the riprap removed before riparian vegetation could propagate. However, the BDCP can restore this flow function through levee modification and habitat restoration.
- Freshwater inflow contributes to turbidity: In a largely unaltered system, freshwater inflow contributes to turbidity in an estuary through bulk transport, resuspension, and longitudinal transport of sediments and depositional material. A secondary mechanism to increase turbidity is through the effect of freshwater inflow on nutrient enrichment and phytoplankton production. In addition to providing depositional material, turbidity enhances the ability of many prey organisms to avoid predation. As previously described, these functions are diminished in the highly altered Bay-Delta and cannot be restored with reservoir releases. See Section 5 for further discussion on turbidity.
- Freshwater inflow facilitates suspension, dispersion, and transport of eggs and larvae: In an unaltered system, freshwater inflow contributes to suspension, dispersion, and transport of eggs and larvae through recirculation, seasonal variability in freshwater inflow, and longitudinal transport. Suspension prevents eggs and larvae from anoxic conditions, typically found in the sediments. Dispersion and transport of eggs and larvae may be necessary for semidiadromous species (Pierson et al. 2002).

In the highly altered Bay-Delta, the efficacy of freshwater flow to contribute to dispersion and distribution of eggs and larvae is also affected by the significant landscape modifications. This flow function will be discussed in greater detail in the submittals for the pelagic and salmonid workshops.

- Freshwater inflow provides migration cues: Anadromous salmonids use freshwater cues to return to their natal spawning grounds. Other diadromous or semidiadromous species use freshwater cues to disperse or migrate. In the highly altered Bay-Delta, changes in the magnitude and timing of freshwater inflow may affect the ability of salmonids to find their natal spawning grounds. Existing regulations attempt to rectify this situation. However, the efficacy of freshwater inflow to cue salmonid migration to natal streams is diminished due to other mitigating factors including dam blockage, hatchery practices, and contaminants. Dams block most salmonids from reaching all or part of their natal spawning grounds, hatchery practices have accelerated straying rates, and contaminants affect salmon olfactory senses. This flow function will be discussed in greater detail in the submittals for the pelagic and salmonid workshops.

The organisms that occur in California rivers and streams are the evolutionary result of physical, chemical and biological characteristics of the regional environment that have been shaped by the sequential, predictable, seasonal flooding, and extended base flows that occur over each annual cycle. California's hydrological patterns are the result of its Mediterranean climate (Gasith and Resh 1999; Bonada et al. 2008), which differs markedly from other locations discussed in the literature regarding the results of restoring natural flows. The adaptation of the resident organisms to the conditions of wet season high flows and dry season low flows, whether they were stable or continuing to decrease until the onset of rains in autumn, influences organism response to departures from this Mediterranean hydrologic pattern.

In the 2010 Flow Criteria Report, the State Water Board proposed a percent of unimpaired flow approach only for the months January through June (State Water Board 2010, p. 96.) and, subsequently, contradicts the approach by seeking to artificially maintain high outflow during the reminder of the year, during the dry season, when historically Delta inflows would be lower. By doing so, the 2010 Flow Criteria Report proposes an unimpaired flow approach that increases outflow, year-round, without regard for natural interannual flow patterns.

Implementation of the natural flow regime in California has generally involved partial application of the concepts for the purpose of achieving specific ecological goals (Cain et al. 2003; Brown and Bauer 2009; Kiernan and Moyle 2010). They have included combinations of minimum instream flows for habitat for various life stages, passage flows, flows for channel maintenance, flows for riparian vegetation maintenance, and more rarely channel-forming flows. However, while the steps of assessing the natural flow regime have taken place in those proceedings, the information was primarily used to negotiate flows that would have specific ecological benefits, rather than for a broad implementation of a natural flow regime absent identification of specific desired outcomes as is currently contemplated by the 2010 Flow Criteria Report.

In general, in the literature, the function that is being targeted is clearly articulated and the specific ecological benefit is identified. Rather than targeting a general goal like “fish abundance” as was done in the 2010 Flow Criteria Report, in most cases the mechanism or function of flow is identified and flows are set to target the function. This is particularly important in the Bay-Delta estuary where there are relatively few intact remnants of the natural Bay-Delta environment. There have been some limited scientific investigations that estimate historic or “natural” conditions of the Bay-Delta, prior to land reclamation, dam construction, and river dredging and straightening. As described above in Section 6.2, several prior analyses indicate that flows in the historic undeveloped environment were lower, likely substantially, than the unimpaired flows described in the 2010 Flow Criteria Report..Many of the general functions that have been targeted in other ecosystems, and in the literature, cannot be achieved in the highly altered Bay-Delta with reservoir releases.

There is little scientific support for application of a natural flow regime, as it is currently proposed, to the highly modified Bay-Delta. Alternate actions, with a lower degree of uncertainty are supported by the science and by the literature. Based on (1) the non-explicit nature of the natural flow regime, (2) natural flow regime has only been partially implemented in highly modified systems, and (3) the uncertainty of success where it has been implemented, little scientific support exists for application of a natural flow regime, as it currently exists, to the highly modified Bay-Delta. Therefore, it is necessary for the State Water Board to consider regulations that address nonflow factors, specifically nutrient pollutants, other toxicants, habitat restoration, and invasive species to enhance the efficacy of flow on ecosystem functions prior to modifying current flow regulations.

7 Conclusions

The physical landscape of the Bay-Delta bears little resemblance to the system that existed more than 160 years ago; a system tha cannot feasibly be fully restored.

The ecosystem will continue to change and evolve. However, without intervention, the decline of certain species, such as the native salmonids and smelt will continue, or even accelerate. Local intervention to date has not worked. Science does not support increasing inflow or outflow as a way to improve the health of the ecosystem. No causal link between flow and fish abundance has been identified, and there is insufficient evidence to rely on increased flow as a tool to increase fish abundance.

Contrary to flow, the scientific literature is clear; the ecosystem’s food web has changed substantially, including decreases in primary and secondary productivity and abundance of native fish species. The literature is equally clear that those adverse changes have been driven by nutrients, invasive species, changes in the land-water interface, and potentially contaminants. Each of these can be addressed by water quality objectives in the Bay-Delta plan and the associated implementation actions by the State Water Board and other agencies.

This Page Intentionally Left Blank

8 References

Ahearn, D. S., J. H. Viers, J. F. Mount, and, R. A. Dahlgren. 2006. Priming the productivity pump: flood pulse driven trends in suspended algal biomass distribution across a restored floodplain. Freshwater Biology 51:1417–1433.

Ahlgren, G., L. Lundstedt, M. Brett, and C. Forsberg. 1990. Lipid composition and food quality of some freshwater phytoplankton for cladoceran zooplankters. Journal of Plankton Research 12: 809-818.

Alpine, A.E., and J.E. Cloern. 1992. Trophic Interactions and Direct Physical Effects Control Phytoplankton Biomass and Production in an Estuary. American Society of Limnology and Oceanograph, Inc. 37 (5) 946-955.

Amweg, E., D. Weston., and N. Ureda. 2005. Use and Toxicity of Pyrethroid Pesticides in the Central Valley, California, USA. Environmental Toxicology and Chemistry, Vol. 24, No. 4, pp. 966-972.

Anderson, L.W.J. 1999. Egeria invades the Sacramento-San Joaquin Delta. Aquatic Nuisance Species Digest 3:37-40.

Anderson, D. M., P. M. Glibert, and J. M. Burkholder. 2002. Harmful algal blooms and eutrophication: Nutrient sources, composition, and consequences. Estuaries 25: 704-726.

Aquatic Ecosystems Analysis Laboratory, UC Davis. 2009. Pharmaceuticals and Personal Care Products in Surface Water- Occurrence, Fate and Transport, and Effect on Aquatic Organisms.

Baldwin, D., J. Spromberg, T. Collier, and N Scholz. 2009. A fish of many scales: extrapolating sublethal pesticide exposures to the productivity of wild salmon populations. Ecological Applications, 19(8), pp. 2004-2015

Ball, M.D. and J. F. Arthur. 1979. Planktonic chlorophyll dynamics in the Northern San Francisco Bay and Delta. Pacific Division of the American Association for the Advancement of Science c/o California Academy of Sciences Golden Gate Park San Francisco, California 94118.

Baskerville-Bridges, B., J.C. Lindberg, and S.I. Doroshov. 2004. The Effect of Light Intensity, Alga Concentration, and Prey Density on the Feeding Behavior of Delta Smelt Larvae. American Fisheries Society Symposium. 39:219-227.

Baxter, R., R. Breuer, L. Brown, L. Conrad, F. Feyrer, S. Fong, K. Gehrts, L. Grimaldo, B. Herbold, P. Hrodey, A. Mueller-Solger, T. Sommer, and K. Souza. 2010. Interagency Ecological Program 2010 Pelagic Organism Decline Work Plan and Synthesis of Results. Interagency Ecological Program for the San Francisco Estuary report dated December 2010.

The Bay Institute. 1998. From the Sierra to the Sea: The Ecological History of the San Francisco Bay-Delta Watershed, San Francisco, July.

Bennett, W.A. 2005. Critical assessment of the delta smelt population in the San Francisco estuary, California. San Francisco Estuary and Watershed Science 3(2).

Berg, G.M., P.M. Glibert, N.O.G. Jorgensen, M. Balode, and I. Purina. 2001. Variability in inorganic and organic nitrogen uptake associated with riverine nutrient input in the Gulf of Riga, Baltic Sea. Estuaries 24: 176-186.

Berman, T., and S. Chava. 1999. Algal growth on organic compounds as nitrogen sources. Journal of Plankton Research 21:1423-1437.

Bertoldi, G., R. Johnston, and K. Evenson. 1991. Ground Water in the Central Valley, California -- A Summary Report. U.S. Geological Survey Professional Paper 1401-A.

Birge, W., J. Black, and D. Bruser. 1979. Toxicity of Organic Chemicals to Embryo-Larval Stages of Fish. Prepared for U.S. Environmental Protection Agency, Office of Toxic Substances. Report No. EPA-560/11-79-007.

Boehloert, G.W. and J.B. Morgan, 1985. Turbidity enhances feeding abilities of larval Pacific herring, Clupea harengus pallasi. Hydrobiologia 123: 161-170.

Bolger, B.L., Y.J. Park, A.J.A. Unger, E.A. Sudicky. 2011. Simulating the Pre-development Hydrologic Conditions in the San Joaquin Valley, California, Journal of Hydrology, v. 411, pp. 322-330.

Bonada, N., M. Rieradevall, H. Dallas, J. Davis, J. Day, R. Figueroa, V. Resh, and N. Prat. 2008. Multi-scale assessment of macroinvertebrate richness and composition in Mediterranean-climate rivers. Freshwater Biology 53, 772-788.

Brander, S.M. and G.N. Cherr. 2008. Endocrine disruption in the Sacramento-San Joaquin Delta: the responses of a resident fish species. Poster presentation at 5th Biennial CALFED Science Conference: Global Perspectives and Regional Results: Science and Management in the Bay-Delta System, Sacramento, CA, October 22-24, 2008.

Brett M.T. and D.C. Müller-Navarra. 1997. The role of highly un-saturated fatty acids in aquatic food-web processes. Freshwater Biology 38: 483–499.

Brown, L., and M. Bauer. 2009. Effects of Hydrologic Infrastructure on Flow Regimes of California's Central Valley Rivers: Implications for Fish Populations. River Res. Applic. 2009.

Brown, R.L. 2004. Summary of 2004 workshop: making science work for Suisun Marsh. San Francisco Bay-Delta Science Consortium.

Brown, T. 2009. Phytoplankton community composition: The rise of the flagellates. IEP Newsletter 22(3): 20-28.

Brown, L.R. and D. Michniuk. 2007. Littoral fish assemblages of the alien-dominated Sacramento-San Joaquin Delta, California, 1980-1983 and 2001-2003. Estuaries and Coasts 30:186-200.

Bryan, K. 1915. Groundwater for Irrigation in the Sacramento Valley, California, U.S. Geological Survey Water-supply, Paper 375-A.

Bunn, S.S., and A. H. Arthington. 2002. Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. *Environmental Management 30: 492-507.*

Burau, J.R., J.W. Gartner, and M.T. Stacey. 1998. Results from the hydrodynamic element of the 1994 entrapment zone study in Suisun Bay. In: Report of the 1994 entrapment zone study, edited by

Wim Kimmerer. Technical Report 56. Interagency Ecological Program for the San Francisco Bay-Delta Estuary. 55 pp.

Cain, J., R. Walking, S. Beamish, E. Cheng, E. Cutler, and M. Wickland. 2003. San Joaquin Basin Ecological Flow Analysis. Prepared for the Bay Delta Authority by the Natural Heritage Institute.

California Department of Water Resources. 1987. Closing Brief for Phase I Sacramento-San Joaquin/ San Francisco Bay Estuary Hearing.

California Department of Water Resources (DWR). 1995. Sacramento San Joaquin Delta Atlas.

California Department of Water Resources (DWR). 2007. California Central Valley Unimpaired Flow Data: Fourth Edition, Bay-Delta Office, DRAFT, May.

California Department of Water Resources (DWR). 2011. California Data Exchange Center. Available from http://cdec.water.ca.gov.

California Department of Water Resources (DWR) 2012. DAYFLOW Database. Available from http://www.water.ca.gov/dayflow.

California Department of Water Resources (DWR) and United States Bureau of Reclamation (USBR). 1958. 1957 Joint Hydrology Study: Sacramento River and Sacramento-San Joaquin Delta. California Department of Water Resources, Division of Resources Planning.

Clifford, M., K. Eder, I. Werner, and R. Hedrick. 2005. Synergistic Effects of Esfenvalerate an Infectious Hematopoietic Necrosis Virus on Juvenile Chinook Salmon Mortality. Environmental Toxicology and Chemistry, Vol. 34, No. 7, pp. 1766-1772.

Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecological Progress Series 210: 223-253.

Cloern, J.E. 2007. Habitat connectivity and ecosystem productivity: implications from a simple model. American Naturalist 169(1):E21-E33.

Cloern, J.E and R.T. Cheng. 1981. Simulation model of *Skeletonema costatum* population dynamics in Northern San Francisco Bay, California. Estuarine, Coastal and Shelf Science 12:83-100.

Cloern, J. E., A.D. Jassby, J. K. Thompson, and K. A. Hieb. 2007. A cold phase of the East Pacific triggers new phytoplankton blooms in San Francisco Bay. Proceedings of the National Academy of Sciences 104(47): 18561–18565.

Cloern J.E., N. Knowles, L.R. Brown, D. Cayan, M.D. Dettinger, T.L. Morgan, D.H. Schoellhamer, M.T. Stacey, M. van der Wegen, R.W. Wagner, and A.D. Jassby. 2011. Projected evolution of California’s San Francisco Bay-Delta river system in a century of climate change. PLoS ONE 6(9).

Cloern, J.E. and R. Dufford. 2005. Phytoplankton community ecology: principles applied in San Francisco Bay. Marine Ecology Progress Series 285:11-28.

Cole, B.E., and D.J.E. Cloern. 1984. Significance of biomass and light availability to phytoplankton productivity in San Francisco Bay. Mar Ecol. Prog. Ser 17:15-24.

Collier, M.P., R.H. Webb, E.D. Andrews. 1997. Experimental Flooding in the Grand Canyon. *Scientific American.*

Collos, Y., M.Y. Siddiqi, M.Y. Wang, A.D.M. Glass, and P.J.Harrison. 1992. Nitrate uptake kinetics by two marine diatoms using the radioactive tracer 13N. Journal of Experimental Marine Biology and Ecology 163: 251–260.

Collos, Y., A. Vaquer, B. Bibent, G. Slawyk, N. Garcia, and P. Souchu. 1997. Variability in nitrate uptake kinetics of phytoplankton communities in a Mediterranean coastal lagoon. Estuarine, Coastal and Shelf Science 44: 369–375.

Connon, R., J. Geist, J. Pfeiff, A. Loguinov, L. D'Abronzo, H. Wintz, C. Vulpe, and I. Werner. 2009. Linking mechanistic and behavioral responses to sublethal esfenvalerate exposure in the endangered delta smelt; *Hypomesus transpacificus* (Fam, Osmeriadae). BMC Genomics 10:608.

Conrad, J.L., K.L. Weinersmith, M.J. Young, D. de Carion, P.Crain, D.J. Harris, M.C. Ferrari, E. Hestir, M. Santos, S. Ustin, P.B. Moyle, A. Sih. 2010a. Rising abundance of largemouth bass in the littoral zone of Sacramento –San Joaquin Delta: the role of *Egeria densa.* Oral Presentation at IEP Workshop, California State University, May 26.

Conrad, J.L., K.L. Weinersmith, M.J. Young, D. de Carion, E. Hestir, M. Santos, P. Crain, S. Ustin, P.B. Moyle, A. Sih. 2010b. More big bass: Understanding the role of largemouth bass as top predators in the littoral zone. Oral Presentation at Delta Science Council Conference, Sacramento, California, September.

Conrad, L., Weinersmith, K., Young, M., de Carion, D., Bibian, A., Moyle, P., and Sih, A. 2011. Invaders Helping Invaders: Expansion of Largemouth Bass in the Sacramento- San Joaquin Delta Facilitated by Brazilian Waterweed, Egeria Densa. AFS 2011- Seattle, 58-13.

Coutteau P. and P. Sorgeloos. 1997. Manipulation of dietary lipids, fatty acids and vitamins in zooplankton cultures. Freshwater Biology 38: 501–512.

Cripe, G.M., B.L. Hemmer, L.R. Goodman, J.W. Fournie, S. Raimondo, J.C. Vennari, R.L. Danner, K. Smith, B.R. Manfredonia, D.H. Kulaw, and M.J. Hemmer. 2009. Multigenerational exposure of the estuarine sheepshead minnow (Cyprinodon Variegatus) to 17b-estradiol. I. organism-level effects over three generations. Environmental Toxicology and Chemistry, 28(11): 2397–2408.

Davis, G.H. 1959. Groundwater Conditions and Storage Capacity in the San Joaquin Valley, California. U.G. Geological Survey Water Supply Paper 1469.

Delta Stewardship Council. Final Staff Draft Delta Plan, May 14, 2012. http://deltacouncil.ca.gov/sites/default/files/documents/files/DeltaPlan_05-14-2012.pdf

Department of Boating and Waterways. 2006. *Egeria densa* Control Program (EDCP). December 8, 2006.

Domingues, R.B., A.B. Barbosa, U. Sommer, and H.M. Galvao. 2011. Ammonium, nitrate and phytoplankton interactions in a freshwater tidal estuarine zone: potential effects of cultural eutrophication. Aquatic Science 73: 3331-343.

Dortch, Q. 1990. The interaction between ammonium and nitrate uptake in phytoplankton. Marine Ecological Progress Series 61:183-201.

Dugdale, R.C., F.P. Wilkerson, V.E. Hogue, and A. Marchi. 2007. The role of ammonium and nitrate in spring bloom development in San Francisco Bay. Estuarine, Coastal and Shelf Science 73: 17-29.

Dugdale, R., F. Wilkerson, and A. Parker. 2011. Brief report in response to selected issues raised by Sacramento Regional County Sanitation District in petition for review of discharge permit issued by the Central Valley Regional Water Quality Control Board. May 4.

Dugdale, R., F. Wilkerson, A.E. Parker, A. Marchi, and K. Taberski. In press. River flow and ammonium discharge determine spring phytoplankton blooms in an urbanized Estuary. Estuarine and Coastal Shelf Science.

Dussault, E.B., V. K. Balakrishnan, E. Sverko, K. R. Solomon, And P. K. Sibley. 2008. Toxicity of human pharmaceuticals and personal care products to Benthic invertebrates. Environmental Toxicology and Chemistry, 27(2): 425–432.

Enright, C. and S.D. Culberson. 2010. Salinity Trends, Variability and Control in the Northern Reach of San Francisco Estuary, San Francisco Estuary and Watershed Science 7(2).

Ensminger, M. and Kelley, K. 2011. Monitoring Urban Pesticide Runoff in Northern California, 2009-2010. California Environmental Protection Agency Report 264.

Færøvig, P.J. and D.O. Hessen. 2003. Allocation strategies in crustacean stoichiometry: the potential role of phosphorus in the limitation of reproduction. Freshwater Biology 48: 1782-1792.

Feijoó, C., M.E. García, F. Momo and and J.Toja. 2002. Nutrient absorption by the submerged macrophyte *Egeria densa* planch.: effect of ammonium and phosphorus availability in the water column on growth and nutrient uptake. *Limnetica* 21(1-2): 03-104

Feyrer F., B. Herbold, S.A. Matern, and P.B. Moyle. 2003. Dietary shifts in a stressed fish assemblage: Consequences of a bivalve invasion in the San Francisco Estuary. Environmental Biology of Fishes 67:277-288.

Feyrer, F., T.R. Sommer, S.C. Zeug, G. O'Leary, and W. Harrell. 2004. Fish assemblages of perennial floodplain ponds of the Sacramento River, California, U.S.A., with implications for the conservation of native fishes. Fisheries Management and Ecology 11:335-344.

Feyrer, F., T. Sommer, and R. Baxter. 2005. Spatial-Temporal Distribution and Habitat Associations of Age-O Splittail in the Lower San Francisco Estuary Watershed. Copeia, 1:159-168.

Feyrer, F., M.L. Nobriga, T.R. Sommer. 2007. Multidecadal trends for three declining fish species: habitat patterns and mechanisms in the San Francisco Estuary, California, USA. Canadian Journal of Fisheries and Aquatic Sciences. 64: 723-734.

Finkel Z.V., J. Beardall, K.J. Flynn, A. Quiqq, T.A. Rees, J.A. Raven. 2010. Phytoplankton in a changing world: Cells size and elemental stoichiometry. *J. Plankt. Res.*2010*,* 32: 119-137

Flannery, M.S, Peebles, E.B, Montgomery, Ralph T. 2002. A Percent-of-flow Approach for Managing Reductions in Freshwater Inflow from Unimpounded Rivers to Southwest Florida Estuaries. *Estuaries, 25: 1318-1331*.

Flynn, K., J.M. Franco, P. Fernández, B. Reguera, M. Zepata, G. Wood, and K.J. Flynn, 1994. Changes in toxin content, biomass and pigments of the dinoflagellate *Alexandrium minutum* during nitrogen refeeding and growth into nitrogen and phosphorus stress. Marine Ecological Progress Series 111: 99-109.

Foe, C., A.Ballard, and S. Fong. 2010. Nutrient concentrations and biological effects in the Sacramento-San Joaquin Delta. Report of the Central Valley Regional Water Quality Control Board. July 2010.

Fox, J.P. 1987. Freshwater Inflow to San Francisco Bay Under Natural Conditions. State Water Contractors Exhibit No. 262, California State Water Resources Control Board Bay-Delta Hearings, Sacramento, CA.

Fox, J.P., T.R. Mongan, and W.J. Miller. 1990. Trends in Freshwater Inflow to San Francisco Bay from the Sacramento-San Joaquin Delta. Water Resources Bulletin 26(1): 101-116.

Gao, Y., J. Cornwell, D. Stoecker, and M. Owens. 2012. Effects of cyanobacterial-driven pH increases on sediment nutrient fluxes and coupled nitrification-denitrification in a shallow fresh water estuary. Biogeosciences, 9:2697-2710.

Garone, P. 2011. The Fall and Rise of the Wetlands of California's Great Central Valley, University of California Press, Berkeley .

Gasith, A., and V. Resh. 1999. Streams in Mediterranean Climate Regions: Abiotic Influrences and Biotic Responses to Predictable Seasonal Events. Annual Review of Ecology and Systematics 30:51-81.

Ger, K.A., S.J. Teh, D.V. Baxa, S. Lesmeister, and C.R. Goldman. 2010. The effects of dietary *Microcystis aeruginosa* and microcystin on the copepods of the upper San Francisco Estuary. *Freshwater Biology*, 55:1548-1559.

Glibert and Burkholder. 2011. Harmful algal blooms and eutrophication: Strategies for nutrient uptake and growth outside the Redfield comfort zone. *Chinese J. Limnol. Oceanogr*. **29**: 724-738

Glibert, P.M. 2010. Long-term changes in nutrient loading and stoichiometry and their relationships with changes in the food web and dominant pelagic fish species in the San Francisco Estuary, California. Reviews in Fisheries Science 18(2):211-232.

Glibert, P.M. 2012. Ecological stoichiometry and its implications for aquatic ecosystem sustainability. *Current Opinion Envir. Sustainability* 4:272-2

Glibert, P.M., C.A. Heil, D. Hollander, M. Revilla, A. Hoare, J. Alexander, and S. Murasko. 2004. Evidence for dissolved organic nitrogen and phosphorous uptake during a cyanobacterial bloom in Florida bay. Marine Ecological Progress Series 280:73-83.

Glibert, P.M., D. Fullerton, J.M. Burkholder, J.C. Cornwell, and T.M. Kana. 2011. Ecological stoichiometry, biogeochemical cycling, invasive species, and aquatic food webs: San Francisco Estuary and comparative systems. Reviews in Fisheries Science 19:358-417.

Glibert, P.M., J. Harrison, C.A. Heil, and S. Seitzinger. 2006. Escalating worldwide use of urea – a global change contributing to coastal eutrophication. Biogeochemistry 77: 441-463.

Glibert. P.M., R.C. Dugdale, A.E. Parker, F. Wilkerson, J. Alexander, S. Blaser, E. Kress, and S.Murasko. 2012. Elevated ammonium concentrations inhibit total nitrogen uptake and growth, not just nitrate uptake. Poster presentation at Interagency Ecological Program Annual Workshop, April 2012, Folsom, CA.

Goals Project. 1999. Baylands Ecosystem Habitat Goals. A report of habitat recommendations prepared by the San Francisco Bay Area Wetland Ecosystem Goals Project. U.S. Environmental Protection Agency, San Francisco, Calif./S.F. Bay Regional Water Quality Control Board, Oakland, Calif.

Goebel, N.L., C.A. Edwards, J.E. Zehr, and M.J. Follows. 2010. An emergent community ecosystem model applied to the California Current System. Journal of Marine Systems 83: 211-242.

Granéli E., and K. Flynn. 2005. Chemical and physical factors influencing toxin content. In: Granéli E, Turner, J.T. (Eds.), Ecology of Harmful Algae. Springer, Heidelberg, pp. 229-241.

Greene, V.E., L.J. Sullivan, J.K. Thompson, W.J. Kimmerer. 2011. Grazing impact of the invasive clam Corbula amurensis on the microplankton assemblage of the northern San Francisco Estuary. Mar Ecol Prog Ser 431: 183–193.

Greening, H. and A. Janicki. 2006. Toward reversal of eutrophic conditions in a subtropical estuary: Water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. Environmental Management 38(2):163-178.

Grimaldo, L. F., A.R. Stewart, and W. Kimmerer. 2009. Dietary segregation of pelagic and littoral fish assemblages in a highly modified tidal freshwater estuary. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 1:200–217.

Grosholz E., and E. Gallo. 2006. The influence of flood cycle and fish predation on invertebrate production on a restored California floodplain. Hydrobiologia 568:91-109.

Ha, J.H., T. Hidaka, and H. Tsuno. 2009. Quantification of toxic *Microcystis* and evaluation of its dominance ratio in blooms using real-time PCR. Environmental Science and Technology 43: 812-818.

Hall, L. 2010a Brief Summary of Water Column and Sediment Toxicity Temporal Trends Analysis for the Westside Coalition from 2004 -2009. Report to Westside Coalition.

Hall, L. 2010b Brief Summary of Diazinon Temporal Trends Analysis from Surface Water Monitoring Data for the Westside Coalition from 2004 -2009. Report to Westside Coalition.

Hall, L. 2010c Brief Summary of Chlorpyrifos Temporal Trends Analysis from Surface Water Monitoring Data for the Westside Coalition from 2004 -2009. Report to Westside Coalition.

Hamilton, S.A. and D.D. Murphy. In review. Habitat affinity analysis as a tool to guide environmental restoration for an imperiled estuarine fish: the case of the delta smelt in the Sacramento-San Joaquin Delta.

Hansen B.H., P. K. Bjornsen, and P. J. Hansen. 1994. The size ratio between planktonic predators and their prey. Limnol Oceanogr 39:395–403

Harris, G.P. 1986. Phytoplankton Ecology: Structure, Function and Fluctuation. Cambridge University Press, Cambridge.

Hart, D. and Finelli, C. 1999. Physical-Biological Coupling in Streams: The Pervasive Effects of Flow on Benthic Organisms. Annual Review of Ecology and Systematics 30:363-395.

Harvey, H.W. 1953. Synthesis of organic nitrogen and chlorophyll by *Nitzschia closterium*. Journal of the Marine Biological Association of the United Kingdom 31: 477-487.

Hecky, R.E. and P. Kilham. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: A review of recent evidence on the effects of enrichment. Limnol. Oceanogr. 33:796-822.

Hendrixson, H. A., R. W. Sterner, and A. D. Kay. 2007. Elemental stoichiometry of freshwater fishes in relation to phylogeny, allometry and ecology. Journal of Fish Biology 70: 121-140.

Hessen, D.O. 1997. Stoichiometry in food webs – Lotka revisted. Oikos 79: 195-200.

Hestir, E. 2010. Trends in Estuarine Water Quality and Submerged Aquatic Vegetation Invasion.

Hobbs, J.A., W.A. Bennett, and J.E. Burton. 2006. Assessing nursery habitat quality for native smelts (Osmeridae) in the low-salinity zone of the San Francisco estuary. Journal of Fish Biology, 69: 907–922.

Hood, J.M., and R.W. Sterner. 2010. Diet mixing: Do animals integrate growth or resources across temporal heterogeneity? The American Naturalist 176: 651-663.

Ibanez, C., N. Prat, C. Duran, M. Pardos, A. Munne, R. Andreu, N. Caiola, N. Cid, H. Hampel, R. Sanchez, and R. Trobajo. 2008. Changes in dissolved nutrients in the lower Ebro River: Causes and consequences. Limnetica 27(1):131-142.

Iwanowicz, L.R., V. S. Blazer, C.P. Guy, A.E. Pinkney, J. E. Mullican, And D. A. Alvarez. 2009. Reproductive health of bass in the Potomac, USA, drainage: Part 1. Exploring the effects of proximity to wastewater treatment plant discharge. Environmental Toxicology and Chemistry, 28(5): 1072–1083.

James, L.A. and M.B. Singer. 2008. Development of the lower Sacramento Valley flood-control system: An historical perspective. Natural Hazards Review, 9(3): 125-135.

Jassby, A. 2008a. Phytoplankton in the Upper San Francisco Estuary: recent biomass trends, their causes, and their trophic significance. San Francisco Estuary and Watershed Science 6(1): Article 2, February.

Jassby, A. 2008b. Temperature trends at several sites in the upper San Francisco Estuary. Unpublished report dated, June 2, 2008.

Jassby, A.D., W.J. Kimmerer, S.G. Monismith, C. Armor, J.E. Cloern, T.M. Powell, J.R. Schubel, and T.J. Vendlinski. 1995. Isohaline Position as a Habitat Indicator for Estuarine Populations, Ecological Applications 5(1): 272-289.

Jassby, A.D., J.E. Cloern, B.E. Cole. 2002. Annual primary production: Patterns and mechanisms of change in a nutrient-rich tidal ecosystem. Limnology and Oceanography 47(3): 698-712.

Jassby AD, Cloern JE, Mueller-Solger A. 2003. Phytoplankton fuels Delta food web. *California Agriculture*, 57:4

Jeffres, C. A., J. Opperman, and P.B. Moyle. 2008. Ephemeral floodplain habitats provide best growth conditions for juvenile Chinook salmon in a California river. Environmental Biology of Fishes 83(4):449-458.

Jeyasingh, P. D., and L.J. Weider. 2005. Phosphorus availability mediates plasticity in life-history traits and predator-prey interactions in *Daphnia*. Ecology Letters 8: 1021-1028.

Jeyasingh, P. D. and L.J. Weider. 2007. Fundamental links between genes and elements: evolutionary implications of ecological stoichiometry. Molecular Ecology 16: 4649-4661.

Johansson, N. and E. Granéli. 1999. Cell density, chemical composition and toxicity of Chrysochromulina polylepis (Haptophyta) in relation to different N:P supply ratios. Marine Biology 135: 209 - 217.

Jordan, T. E., J. C. Cornwell, W. R. Boynton, and J. T. Anderson. 2008. Changes in phosphorus biogeochemistry along an estuarine salinity gradient: the iron conveyer belt. Limnology and Oceanography 53: 172-184.

Junk W. J., P. F. Bayley, and R. E. Sparks. 1989. The flood pulse concept in river-floodplain systems. Canadian Journal of Fisheries and Aquatic Sciences 106:110-27.

Katibah, E. 1984. A Brief History of Riparian Forests in the Central Valley of California. Berkeley: University of California Press.

Kendall, C., P. Lehman, S.R. Silva, M.B. Young, and M. Guerin. 2011. Tracing sources of nutrients fueling *Microcystis* blooms in the Sacramento-San Joaquin Delta using a multi-fingerprinting approach. Draft report to California Department of Water Resources (unpublished).

Kidd, K.A., P. J. Blanchfield, K.H. Mills, V. P. Palace, R. E. Evans, J. M Lazorchak, and R.W. Flick. 2007. Collapse of a fish population after exposure to a synthetic estrogen. PNAS, 104(21):8897-8901.

Kiernan, J., and P. Moyle. 2012. Flows, droughts, and aliens: factors affecting the fish assemblage in a Sierra Nevada, California, stream. Ecological Applications 22:1146-1161.

Kilham, S. S., D.A. Kreeger, C.E. Goulden, and S.G. Lynn. 1997. Effects of nutrient limitation on biochemical constituents of *Ankistrodesmus falcatus*. Freshwater Biolology 38: 591-596.

Kimmerer, W.J. 1992. An Evaluation of Existing Data in the Entrapment Zone of the San Francisco Bay Estuary. Technical Report 33 prepared for the Interagency Ecological Studies Program for the Sacramento-San Joaquin Estuary.

Kimmerer, W.J. 2002a. Effects of freshwater flow on abundance of estuarine organisms: physical effects or trophic linkages? Mar Ecol Prog Ser 243: 39–55.

Kimmerer, W.J. 2002b. Physical, biological, and management responses to variable freshwater flow into the San Francisco Estuary. Estuaries 25(6): 1275-1290.

Kimmerer, W.J. 2004. Open water processes of the San Francisco Estuary: From physical forcing to biological responses. San Francisco Estuary and Watershed Science 2(1), Article 1.

Kimmerer WJ. 2005. Long-term changes in apparent uptake of silica in the San Francisco estuary. *Limnology and Oceanography,* 50(3):793-798.

Kimmerer, W.J. 2006. Response of anchovies dampens effects of the invasive bivalve *Corbula amurensis* on the San Francisco Estuary foodweb. Marine Ecology Progress Series 324: 207-218.

Kimmerer, W.J., E. Gartside, and J.J. Orsi. 1994. Predation by an introduced clam as the likely cause of substantial declines in zooplankton of San Francisco Bay. Marine Ecology Progress Series. Volume 113:81-93. October 13.

Kimmerer, W.J., E.S. Gross, and M.L. MacWilliams. 2009. Is the response of estuarine nekton to freshwater flow in the San Francisco Estuary explained by variation in habitat volume? Estuaries and Coasts 32: 375-389.

Kimmerer, W.J., J.H. Cowan Jr., L.W. Miller, and K.A. Rose. 2000. Analysis of an estuarine striped bass (*Morone saxatilis*) population: influence of density-dependent mortality between metamorphosis and recruitment. Canadian Journal of Fisheries and Aquatic Sciences 57: 478-486.

Kimmerer W.J., A.E. Parker, U.E. Lidström, and E.J. Carpenter. 2012. Short-Term and Interannual Variability in Primary Production in the Low-Salinity Zone of the San Francisco Estuary. *Estuaries and Coasts* ,35:913–929.

Kiørboe, T. 1989. Phytoplankton growth rate and nitrogen content: Implications for feeding and fecundity in a herbivorous copepod. Marine Ecological Progress Series 55: 229-234.

Kolpin, D., E. Furlong, M. Meyer, E. Thurman, S. Zaugg, L. Barber, and H. Buxton. 2002. Pharmaceuticals, Hormones, and Other Organic Wastewater Contaminants in U.S. Streams, 1999-2000: A National Reconnaissance. Environ. Sci. Technol. 36:1202-1211

Kooser, B., Mills, J., Sturm, J. and Webster, G. 1861. Committee Report- Committee Nos. 1 and 2 on Farms and Orchards. Transactions of the San Joaquin Valley Agricultural Society, 265-283.

Kuivila, K. M., and Hladik, M. L. 2008. Understanding the occurrence and transport of current-use pesticides in the San Francisco Estuary Watershed. San Francisco Estuary and Watershed Science 6(3): article 2. Available from: http://repositories.cdlib.org/jmie/sfews/vol6/iss3/art2

Kuivila K.M. and Moon, E. G. 2004. Potential Exposure of Larval and Juvenile Delta Smelt to Dissolved Pesticides in the Sacramento–San Joaquin Delta, California. American Fisheries Society Symposium 39:229–241.

Laspoumaderes, C., B. Modenutti and E. Balseiro. 2010. Herbivory versus omnivory: linking homeostasis and elemental imbalance in copepod development. Journal of Plankton Research 32: 1573-1582

Lehman, P.W. 1996. Changes in chlorophyll-a concentration and phytoplankton community composition with water-year type in the upper San Francisco Estuary, pp. 351-374. In: San Francisco Bay: The ecosystem (Hollibaugh, J.T., Ed.) Pacific Division of the American Association for the Advancement of Science, San Francisco.

Lehman, P. W. 2000. The influence of climate on phytoplankton community biomass in San Francisco Bay Estuary. Limnology and Oceanography 45: 580–590.

Lehman, P. W., G. Boyer, C. Hall, S. Waller and K. Gehrts. 2005. Distribution and toxicity of a new colonial *Microcystis aeruginosa* bloom in the San Francisco Bay Estuary, California. Hydrobiologia 541:87-99.

Lehman, P.W., T. Sommer and L. Rivard. 2008. The influence of floodplain habitat on the quantity of riverine phytoplankton carbon produced during the flood season in San Francisco Estuary. Aquatic Ecology 42: 363-378.

Lehman, P.W., S.J. Teh, G.L. Boyer, M.L. Nobriga, E. Bass, and C. Hogle. 2010. Initial impacts of *Microcystis aeruginosa* blooms on the aquatic food web in the San Francisco Estuary. Hydrobiologia 637:229-248.

Lewis, W. Jr., S. Hamilton., M Rodriguez., J. III Saunders, and M. Lasi. 2001. Food web analysis of the Orinoco floodplain based on production estimates and stable isotope data. J N Am Benthol Soc 20:241–254.

Lindberg, J.C. and C. Marzuola. 1993. Delta smelt in a newly created, flooded island in the Sacramento–San Joaquin Estuary, Spring 1993. BioSystems Analysis Inc. Prepared for California Department of Water Resources, Sacramento, CA.

Lindberg, J., B. Baskerville-Bridges, and S. Doroshov. 2000. Update on Delta Smelt Culture with and Emphasis on Larval Feeding Behavior. IEP Newsletter. Vol. 13, Number 1, Winter. Pages 45-49.

Lomas, M.W., and P.M. Glibert. 1999a. Interactions between NH+4 and NO3 uptake and assimilation: comparison of diatoms and dinoflagellates at several growth temperatures. Marine Biology 133: 541-551.

Lomas, M.W. and P.M. Glibert. 1999b. Temperature regulation of nitrate uptake: A novel hypothesis about nitrate uptake and reduction in cool-water diatoms. Limnology and Oceanography 44: 556-572.

Lopez, C.B., J.E. Cloern, T.S. Schraga, A.J. Little, L.V. Lucas, J.K. Thompson, and J.R. Burau. 2006. Ecological values of shallow-water habitats: implications for the restoration of disturbed ecosystems. Ecosystems 9: 422–440.

Lucas, L., J. Thompson., L. Brown. 2009. Why are diverse relationships observed between phytoplankton biomass and transport time? Limnol. Oceanogr., 54(1), 2009, 381-390.

Ludwig, C.A. 1938. The availability of different forms of nitrogen to a green alga (*Chlorella*). American Journal of Botany 25:448-458.

Lund, J., E. Hanak, W. Fleenor, R. Howitt, J. Mount, P. Moyle. 2007. Envisioning futures for the Sacramento–San Joaquin Delta. Public Policy Institute of California.

MacIsaac, J.J. and R.C. Dugdale. 1969. The kinetics of nitrate and ammonium uptake by natural populations of marine phytoplankton. Deep-Sea Research 16:45-67.

MacIsaac, J.J. and R.C. Dugdale, 1972. Interactions of light and inorganic nitrogen controlling nitrogen uptake in the sea. Deep-Sea Research 19:209-232.

MacNally R., J.R. Thomson, W.J. Kimmerer, F. Feyrer, K.B. Kewman, A. Sih, W.A. Bennett, L. Brown, E. Fleischman, S.D. Culberson, and G. Castillo. 2010. Analysis of pelagic species decline in the upper San Francisco Estuary using multivariate autoregressive modeling (MAR). Ecological Applications 20(5): 1417-1430.

Maestrini, S Y., J.M. Robert, I. Truquet. 1982. Simultaneous uptake of ammonium and nitrate by oyster-pond algae. Mar Biol. Lett. 3: 143-153.

Martinovic, D., W.T. Hogarth, R.E. Jones, and P. W. Sorensen. Environmental estrogens suppress hormones, behavior, and reproductive fitness in male fathead minnows. Environmental Toxicology and Chemistry, 26(2): 271–278.

Maunder M.N. and R.B. Deriso. 2011. A state-space multi-stage lifecycle model to evaluate population impacts in the presence of density dependence: Illustrated with application to delta smelt. Canadian Journal of Fisheries and Aquatic Science 68:1285-1306.

McCarthy, J.J., W.R. Taylor and J.L. Taft, 1975. The dynamics of nitrogen and phosphorous cycling in the open water of the Chesapeake Bay. In: T.M. Church (ed.) Marine Chemistry in the Coastal Environment. American Chemical Society Symposium Series 18. Washington D.C., pp. 664-681.

McIntyre, P. B., and A. Flecker. 2010. Ecological stoichiometry as an integrative framework in stream fish ecology. American Fisheries Society Symposium 73: 539-558.

McIvor, C., L.R. Brown and Z. Hymanson. 1999. Shallow water habitat workshop summary. IEP Newsletter 12(1).

McKee, L., Ganju, N., and Schoellhamer, D. 2006. Estimates of suspended sediment entering San Francisco Bay from the Sacramento and San Joaquin Delta, San Francisco Bay, California. Journal of Hydrology 323, 35-352.

Meyer, J.S., P.J. Mulholland, H.W. Paerl, and A.K. Ward. 2009. A framework for research addressing the role of ammonia/ammonium in the Sacramento-San Joaquin Delta and the San Francisco Bay Estuary ecosystem. Report to CalFed Science Program.

Miller, B., B. Manly, D. Murphy, D. Fullerton, and R. Ramey. 2012. An Investigation of Factors Affecting the Decline of Delta Smelt (Hypomesustranspacificus) in the Sacramento-San Joaquin Estuary. Reviews in Fisheries Science, 20:1, 1-19.

Mitra, A. and K.J. Flynn. 2005. Predator-prey interactions: is “ecological stoichiometry” sufficient when good food goes bad? Journal of Plankton Research 27: 393-399.

Moe, S. J., R.S. Stelzer, M.R. Forman, W.S. Harpole, T. Daufresne and T. Yoshida. 2005. Recent advances in ecological stoichiometry: insights for population and community ecology. Oikos 109: 29–39.

Molles, M.C., C.S. Crawfor, L.M. Ellis. 1995. Effects of an Experimental Flood on Litter Dynamics in the Middle of the Rio Grande Riparian Ecosystem. Regulated Rivers Research and Management, 11: 275-281.

Monismith, S.G., J.L. Hench, D.A. Fong, N.J. Nidzieko, W.E. Fleenor, L.P. Doyle and S.G. Schladow. 2009. Thermal variability in a tidal river. Estuaries and Coasts 32: 100-110.

Mount, J. F. 1995. Rivers and streams: the conflict between fluvial process and land use. University of California Press.

Moyle, P.B. 2002. Inland fishes of California. University of California Press. Berkeley, California.

Moyle, P.B., B. Herbold, D.E. Stevens and L.W. Miller. 1992. Life history of delta smelt in the Sacramento-San Joaquin Estuary, California. Transactions of the American Fisheries Society 121: 67-77.

Moyle P.B., R.D. Baxter, T.R. Sommer, T.C. Foin and S.A. Matern. 2004. Biology and population dynamics of Sacramento splittail (*Pogonichthys macrolepidotus*) in the San Francisco estuary: A review. San Francisco Estuary and Watershed Science 2(2).

Moyle, P. B., P. K. Crain, and K. Whitener. 2007. Patterns in the use of a restored California floodplain by native and alien fishes. San Francisco Estuary and Watershed Science. Volume 5.

Moyle P.B. and W.A. Bennett W.A. 2008. The future of the Delta ecosystem and its fish. Technical appendix D to the Public Policy Institute of California report, Comparing Futures for the Sacramento-San Joaquin Delta. Public Policy Institute of California.

Moyle, P.B, W.A. Bennett, C. Dahm, J.R. Durand, C. Enright, W.E. Fleenor, W. Kimmerer, J.R. Lund . 2010. Changing ecosystems: a brief ecological history of the Delta 1. February.

Moyle, P., W. Bennett, J. Durand, W. Fleenor, B. Gray, E. Hanak, J. Lund and J. Mount. 2012. Where the Wild Things Aren’t, Making the Delta a Better Place for Native Species. Public Policy Institute of California.

Müller-Solger, A., A.D. Jassby and D.C. Müller-Navarra. 2002. Nutritional quality of food resources for zooplankton (*Daphnia*) in a tidal freshwater system (Sacramento-San Joaquin River Delta). Limnology and Oceanography 47(5): 1468-1476.

Munoz, I., J.C. Lo Pez-Doval, M. Ricart, M. Villagrasa, R. Brix, A. Geiszinger, A. Ginebreda, H. Guasch, M. J. Lo Pez De Alda, A.M. Romani´,S. Sabater, and D. Barcelo. 2009. Pharmaceuticals and Personal Care Products in the Environment: Bridging levels of pharmaceuticals in river water with biological Community structure in the Llobregat River Basin (Northeast Spain). Environmental Toxicology and Chemistry, 28(12): 2706–2714.

Myrick, C.A. and J.J. Cech. 2000. Swimming performances of four California stream fishes: Temperature effects. Environmental Biology of Fishes 58: 289-295.

Myrick, C.A. and J.J. Cech. 2011. Temperature effects on Chinook salmon and steelhead: A review focusing on California’s Central Valley populations. Technical publication 01-1. Bay-Delta Modeling Forum.

National Water Research Institute 2010 Source, Fate, and Transport of Endocrine Disruptors, Pharmaceuticals, and Personal Care Products in Drinking Water Sources in California. Final Project Report. NWRI Project Number 07-WQ-004.

Nichols F.H., J.E. Cloern, S.N. Luoma and D.H. Peterson. 1986. The modification of an estuary. Science 231: 567-573.

Nixon, S.W. 1988. Physical Energy Inputs and the Comparative Ecology of Lake and Marine Ecosystems. American Society of Limnology and Oceanography, Inc. 33 (4, part 2) 1005-1025.

Nobriga, M.L. 2002. Larval delta smelt diet composition and feeding incidence: environmental and ontogenetic influences. California Fish and Game 88:149-164.

Nobriga, M.L., F. Feyrer, R.D. Baxter and M. Chotkowski. 2005. Fish community ecology in an altered river delta: spatial patterns in species composition, life history strategies, and biomass. Estuaries, 28(5): 776-785.

National Research Council (NRC). 2010. A scientific assessment of alternatives for reducing water management effects on threatened and endangered fishes in California's Bay-Delta. National Academies Press, Washington, D.C.

National Research Council (NRC). 2011. Expert Panels on Sustainable Water and Environmental Management in the California Bay Delta.

National Research Council (NRC). 2012. Sustainable water and environmental management in the California Bay-Delta. National Academies Press, Washington, D.C.

Oh, H-M., S.J. Lee, M-H. Jang and B-D. Yoon. 2000. Microcystin production by *Microcystis aeruginosa* in a phosphorus-limited chemostat. Applied Environmental Microbiology 66: 176-179.

Olsen, Y., 1999. Lipids and essential fatty acids in aquatic food webs: what can freshwater ecologists learn from mariculture? In: Arts, M.T., Wainman, B.C. (Eds).Lipids in Freshwater Ecosystems. Springer-Verlag, New York, pp. 161-202.

Oltmann, R.N., D.H. Schoellhamer and R.L. Dinehart. 1999. Sediment inflow to the Sacramento-San Joaquin Delta and the San Francisco Bay: Interagency Ecological Program Newsletter 12(1):30-33. http://www.iep.ca.gov/report/newsletter/

Opperman, J.J., R. Luster., B.A. McKenney, M. Roberts, and A.W. Meadows. 2010. Ecologically functional floodplains: Connectivity, flow regime, and scale. Journal of the American Water Resources Association 46:211–226.

Oros, D.R. and I. Werner. 2005. Pyrethroid insecticides: an analysis of use patterns, distributions, potential toxicity and fate in the Sacramento-San Joaquin Delta and Central Valley. White Paper for Interagency Ecological Program. SFEI Contribution 415. San Francisco Estuary Institute, Oakland, CA.

Orsi, J.J., W.L. Mecum and J.T. Hollibaugh. 1996. Food limitation as the probable cause of a long-term decline in the abundance of *Neomysis mercedis* the opossum shrimp in the Sacramento-San Joaquin Estuary. San Francisco Bay: The Ecosystem. 1996. San Francisco: Pacific Division, AAAS, California Academy of Sciences. p. 375-401.

Parchaso, F., and J.K. Thompson. 2002. Influence of hydrologic processes on reproduction of the introduced bivalve *Potamocorbula amurensis* in Northern San Francisco Bay, California. Pacific Science, 56(3):329-345.

Park, S., M.T. Brett, E.T. Oshel and C.R. Goldman. 2003. Seston food quality and *Daphnia* production efficiencies in an oligo-mesotrophic subalpine lake. Aquatic Ecology 37: 123-136.

Parker, A.E., R.C. Dugdale and F.P. Wilkerson. 2012a. Elevated ammonium concentrations from wastewater discharge depress primary productivity in the Sacramento River and the Northern San Francisco Estuary. Marine Pollution Bulletin 64: 574–586.

Parker, A.E., V. E. Hogue, F.P. Wilkerson and R.C. Dugdale. 2012b. The effect of inorganic nitrogen speciation on primary production in the San Francisco Estuary. Estuarine, Coastal and Shelf Science 104-105:91-101.

Parker, V.T., J.C. Calloway, L.M. Schile, M.C. Vasey and E.R. Herbert. 2011. Climate change and the San Francisco Bay-Delta tidal wetlands. San Francisco Estuary and Watershed Science 9(3).

PBSJ. 2008. Independent Peer Review of USFWS's Draft Effects Analysis for the Operations Criteria and Plan's Biological Opinion. Prepared for United States Fish & Wildlife Service, Sacramento office. October 23, 2008

Peñuelas, J., J. Sardans, A. Rivas-Ubach and I.A. Janssens. 2012. The human-induced imbalance between C, N and P in Earth's life system. Global Change Biology 18: 3-6.

Peterson, H.A. and M. Vayssieres. 2010. Benthic assemblage variability in the Upper San Francisco Estuary: A 27-year retrospective. San Francisco Estuary and Watershed Science, 8(1).

Pierson, WL., K. Bishop, D. Van Senden, P.R. Horton, and C.A. Adamantidis. Environmental Water Requirements to Maintain Estuarine Processes (2002), Environmental Flows Initiative Technical Report Number 3, Commonwealth of Australia, Canberra.

Pilati, A. and M.J. Vanni. 2007. Ontogeny, diet shifts, and nutrient stoichiometry in fish. Oikos 116: 1663-1674

Poff, N. L., Allan, D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Spaerks, R.E., Stromberg, J.C. 1997. The Natural Flow Regime: A Paradigm for River Conservation and Restoration. *BioScience, 47:11*.

Poff, N.L., and J.K.H. Zimmerman. 2010. Ecological Responses to Altered Flow Regimes: A Literature Review to Inform the Science and Management of Environmental Flows. *Freshwater Biology 55: 194:205*.

Price, N.M., W.P. Cochlan and P.J. Harrison. 1985. Time course of uptake of inorganic and organic nitrogen by phytoplankton in the Strait of Georgia: comparison of frontal and stratified communities. Mar. Ecol. Prog, Ser. 27: 39-53.

Quinn, T. and A. Punt. 2010. Responses to Questions submitted by Judge Wanger in the Delta Smelt Consolidated Cases. Case No. 1:09-CV- 407.

Raloff, J. 2007. Aquatic Non-Scents: Repercussions of water pollutants that mute smell. Science News Online. Available at: www.sciencenews.org

Rask, N., S.E. Pedersen and M.H. Jensen. 1999. Response to lowered nutrient discharges in the coastal waters around the island of Funen, Denmark. Hydrobiologia 393: 69–81

Reimschuessel, R. 2001. A Fish Model of Renal Regeneration and Development. ILAR Journal V42(4).

Reynolds, C. S. 1984. Phytoplankton periodicity: the interactions of form, function and environmental variability. Freshwater Biol., 14, 111–142.

Ribeiro F., P. K. Crain and P. B. Moyle. 2004. Variation in condition factor and growth in young-of-the-year fishes in floodplain and riverine habitats of the Cosumnes River, California. Hydrobiologia, 527:77-84.

Richter, B., J. Baumgartner, J. Powell, and D. Braun. 1996. A Method For Assessing Hydrologic Alteration within Ecosystems. Conservation Biology, Volume 10, Issue 4, 1163-1174.

Riordan, D. and A. D. Biales. 2008. In-situ exposure of fish for biomarker experimentation at Department of Water Resources (DWR) real-time monitoring sites. Presented at 5th Biennial CALFED Science Conference 2008, Sacramento, CA, October 22 - 24, 2008.

Robinson, C.J. 2004. Responses of the northern anchovy to the dynamics of the pelagic environment: Identification of fish behaviours that may leave the population under risk of overexploitation. Journal of Fish Biology 64(4): 1072-1087.

Rodgers, D.W. and F.W.H. Beamish. 1982. Dynamics of dietary methylmercury in rainbow trout, *Salmo gairdneri.* Aquat. Toxicol. 2:271-290.

Rood, S.B., J.M. Mahoney, D.E. Reid, L. Zilm. 1995. Instream Flows and the Decline of Riparian Cottonwoods Along the St. Mary River, Alberta. *Can. J. Bot. 73: 1250-1260*.

Rosenfield, J.A. and R.D. Baxter. 2007. Population dynamics and distribution patterns of longfin smelt in the San Francisco Estuary. Transactions of the American Fisheries Society 136:1577–1592.

Ruhl, H.A. and N.B. Rybicki. 2010. Long-term reductions in anthropogenic nutrients link to improvements in Chesapeake Bay habitat. Proceedings of the National Academy of Sciences of the United States of America. Available from www.pnas.org/cgi/doi/10.1073/pnas.1003590107.

Sandahl, J.F., D.H. Baldwin, J.J. Jenkins, and N.L. Scholz. 2004. Odor-evoked field potentials as indicators of sublethal neurotoxicity in juvenile coho salmon exposed to copper, chlorpyrifos, or esfenvalerate. Canadian Journal of Fisheries and Aquatic Sciences, 61:404-413.

Sandahl, J.F., D.H. Baldwin, J.J. Jenkins, and N.L. Scholz. 2007. A sensory system at the interface between urban stormwater runoff and salmon survival. Environmental Science and Technology, 41:2998-3004.

Schaefer, M. and M.L. Johnson. 2009. Pharmaceuticals and Personal Care Products in the Sacramento River. Final Report: Activities from May – June 2008. Aquatic Ecosystems Analysis Laboratory Center for Watershed Sciences, University of California, Davis. Prepared for the State Water Resources Control Board.

Schoellhamer, D.H. 2001. Influence of salinity, bottom topography, and tides on locations of estuarine turbidity maxima in northern San Francisco Bay. IEP Newsletter, Vol. 14, Number 4, Fall 2001.

Schoellhamer, D.H. 2011. Sudden Clearing of Estuarine Waters upon Crossing the Threshold from Transport to Supply Regulation of Sediment Transport as an Erodible Sediment Pool is Depleted: San Francisco Bay, 1999. Estuaries and Coasts. Volume 34, Number 5 (2011), 885-899.

Schoellhamer, D.H., M.A. Lionberger, B.E. Jaffe, N.K. Ganju, S.A. Wright, and G.G. Shellenbarger. 2005. Bay sediment budgets: Sediment accounting 101. In The pulse of the estuary: Monitoring and managing water quality in the San Francisco Estuary, 58-63. Oakland: San Francisco Estuary Institute.

Schramm Jr. H. L. and M. A. Eggleton. 2006. Applicability of the flood-pulse concept in a temperate floodplain river ecosystem: Thermal and temporal components. River Research Applications. 22:543–553.

Slaughter, A. and W. Kimmerer. 2010. Abundance, composition, feeding, and reproductive rates of key copepod species in the food-limited Low Salinity Zone of the San Francisco Estuary. Poster Presentation at the 6th Biennial Bay-Delta Science Conference, Sacramento, CA, September 27-29.

Solomon, C.M., J.L. Collier, G.M. Berg and P.M. Glibert. 2010. Role of urea in microbial metabolism in aquatic systems: a biochemical and molecular review. Aquatic Microbial Ecology 59: 67-88.

Sommer, T., R. Baxter, and B. Herbold, 1997. Resilience of Splittail in the Sacramento-San Joaquin Estuary. Transactions of the American Fisheries Society 126:961-976.

Sommer, T.R., M.L. Nobriga, W.C. Harrell, W. Batham and W.J. Kimmerer. 2001a. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. Canadian Journal of Fisheries and Aquatic Sciences 58(2):325-333.

Sommer, T.R, W.C. Harrell, M.L. Nobriga, R. Brown, P.B. Moyle, W. Kimmerer, and L. Schemel. 2001b. California’s Yolo Bypass: evidence that flood control can be compatible with fisheries, wetlands, wildlife, and agriculture. Fisheries 28(8): 6-16.

Sommer, T.R., W.C. Harrell, A.Mueller Solger, B. Tom, and W. Kimmerer. 2004. Effects of flow variation on channel and floodplain biota and habitats of the Sacramento River, California, USA. Aquatic Conservation-Marine and Freshwater Ecosystems 14:247-261.

Sommer, T., C. Armor, R. Baxter, R. Breuer, L. Brown, M. Chotkowski, S. Culberson, F. Feyrer, M. Gingras, B. Herbold, W. Kimmerer, A. Mueller-Solger, M. Nobriga and K. Souza. 2007. The collapse of pelagic fishes in the upper San Francisco Estuary. Fisheries 32(6): 270-277.

Sommer, T. R., W. C. Harrell, Z. Matica, and F. Feyrer. 2008. Habitat associations and behavior of adult and juvenile splittail (Cyprinidae: Pogonichthys macrolepidotus) in a managed seasonal floodplain wetland. San Francisco Estuary and Watershed Science. Vol. 6, Issue 2 (June), Article 3.

Sommer, T., Mejia, F., Hieb, K., Baxter, R., loboschefsky, E., Loge, F. 2011. Long-Term Shifts in the Lateral Distribution of Age-0 Striped Bass in the San Francisco Estuary. Transactions of the American Fisheries Society 140:1451-1459, 2011.

State Water Resources Control Board (SWRCB). 1990. Draft Functional Equivalent Document for: Inland Surface Waters of California and Enclosed Bays and Estuaries of California, January 29, 1990. State Water Resources Control Board, Sacramento, CA.

State Water Resources Control Board (State Board). 2010. Development of Flow Criteria for the Sacramento-San Joaquin Delta Ecosystem. Prepared pursuant to the Sacramento-San Joaquin Delta Reform Act of 2009, August 3, 2010.

Sterner, R. W. and N.B. George. 2000. Carbon, nitrogen, and phosphorus stoichiometry of cyprinid fishes. Ecology, 81, 127-140.

Sterner, R.W. and J.J. Elser. 2002. Ecological stoichiometry: The biology of elements from molecules to the biosphere. Princeton University Press, Princeton, New Jersey.

Strayer, D.L., K.A. Hattala and A.W. Kahnle. 2004. Effects of an invasive bivalve (*Dresseina polymorpha*) on fish in the Hudson River estuary. Canadian Journal of Fisheries and Aquatic Science 61: 924-941.

Teh, S., I. Flores, M. Kawaguchi, S. Lesmeister and C. Teh. 2011. Full life-cycle bioassay approach to assess chronic exposure *of Pseudodiaptomus forbesi* to ammonia/ammonium. Final report to Chris Foe and Mark Gowdy at the State Water Board.

Tierney, K. B., D.H. Baldwin., T.J. Hara, P.S. Ross., N.L. Scholz, and C.J. Kennedy. 2009. Olfactory toxicity in fishes. Aquatic Toxicology 96 (2010) 2–26.

Tockner, K., F. Malard, and J. V. Ward. 2000. An extension of the flood pulse concept. Hydrological Processes 14: 2861-2883.

Tockner, K., and J.A. Stanford. 2002. Riverine flood plains: present state and future trends. Environmental Conservation 29: 308-330

Townsend, H., Bauer, M., Brown, L., and Feyrer, F. 2010. “An ecosystem model for testing potential causes of the San Francisco Estuary pelagic organism decline.” Oral presentation at Interagency Ecological Program 2010 Annual Workshop. Sacramento, CA, May 26, 2010.

United States Environmental Protection Agency (USEPA). 1999. Guidance Manual for Compliance with the Interim Enhanced Surface Water Treatment Rule: Turbidity Provisions. EPA 815-R-99-010. April.

United States Environmental Protection Agency (USEPA). 2010. CWA 303(d) List of Water Quality Limited Segments. Available at: http://www.epa.gov/region09/water/tmdl/303d.html

United States Environmental Protection Agency (USEPA). 2011. Advanced Notice of Proposed Rule Making.

United States Fish and Wildlife Service (USFWS). 1996. Recovery Plan for the Sacramento/San Joaquin Delta Native Fishes.

United States Fish and Wildlife Service (USFWS). 2008. Biological Opinion for the Delta Smelt.

United States Geological Survey. 2012. National Water Information System (NWIS). Available from http://waterdata.usgs.gov/ca/nwis.

Van de Waal, D. B., J. M. Verspagen, M. Lurling, E. Van Donk, P. M. Visser and J. Huisman. 2009. The ecological stoichiometry of toxins produced by harmful cyanobacteria: An experimental test of the carbon-nutrient balance hypothesis. Ecology Letters 12: 1326-1335.

Van Nieuwenhuyse, E. 2007. Response of summer chlorophyll concentration to reduced total phosphorus concentration in the Rhine River (Netherlands) and the Sacramento-San Joaquin Delta (California, USA). Canadian Journal of Fisheries and Aquatic Sciences 64: 1529-1542.

Vannote, RL, B.W. Sweeney. 1980. Geographic analysis of thermal equilibria: A conceptual model for evaluating the effect of natural and modified thermal regimes on aquatic insect communities. The American Naturalist 115(5):667-695.

Wagner, R.W., M. Stacey, L.R. Brown, and M. Dettinger. 2011. Statistical models of temperature in the Sacramento–San Joaquin Delta under climate-change scenarios and ecological implications. Estuaries and Coasts 34:544–556.

Walve, J. and U. Larsson. 1999. Carbon, nitrogen and phosphorus stoichiometry of crustacean zooplankton in the Baltic Sea: implications for nutrient recycling. Journal of Plankton Research 21: 2309-2321.

Ward, A., A. Duff, J. Horsfall, and S. Currie. 2007. Scents and scents-ability: pollution disrupts chemical social recognition and shoaling in fish. Proc. R. Soc. B 274:101-105.

Ware, D. M., and R. E. Thompson. 2005. Bottom-up ecosystem trophic dynamics determine fish production in the Northeast Pacific. Science: 308: 1280-1284.

Weers, P.M.M. and R.M. Gulati. 1997. Effects of the addition of polyunsaturated fatty acids to the diet on the growth and fecundity of *Daphnia galeata*. Freshwater Biology: 38, 721-729.

Weis, J. and Weis, P. 1995. Effects of embryonic exposure to methylmercury Fundul Us Heteroclitus on larval prey-capture ability in the Mummichog. Environmental Toxicology and Chemistry, Vol. 14, No. 1, pp. 153-156.

Weston, D.P., R.W. Holmes, J. You, and M.J. Lydy. 2005. Aquatic Toxicity Due to Residential Use of Pyrethroid Insecticides. Environmental Science and Technology, 39:9778-9784.

Weston, D.P., and M.J. Lydy. 2010. Urban and Agricultural Sources of Pyrethroid Insecticides to the Sacramento-San Joaquin Delta of California. Environ. Sci. Technol., 2010, 44 (5), pp 1833–1840.

Weston, D.P., and M.J. Lydy. 2012. Stormwater input of pyrethroid insecticides to an urban river. Environmental Toxicology and Chemistry 31: 1-8.

Wilkerson, F.P, R.C. Dugdale, V.E. Hogue and A. Marchi. 2006. Phytoplankton blooms and nitrogen productivity in San Francisco Bay. Estuaries and Coasts 29(3): 401–416.

Williamson, K., D. Prudic, and L. Swain. 1989. Ground-Water Flow in the Central Valley, California, U.S. Geological Survey Professional Paper 1401-D.

Winder, M. and A.D. Jassby. 2010. Shifts in zooplankton community structure: Implications for food-web processes in the upper San Francisco Estuary. Estuaries and Coasts. 34: 675-690

Wright. S.A. and D.H. Schoelhammer. 2004. Trends in the sediment yield of the Sacramento River, California, 1957-2001. San Francisco Estuary and Watershed Science 2. http://repositories.edlib.org/jmie/sfews/vol2iss2/art2.

Yamamoto, T. 2002. The Seto Inland Sea—eutrophic or oligotrophic? Marine Pollution Bulletin 47(1): 37 – 42.

Yarrow, M., V. H. Marin, M. Finlayson, A. Tironi, L. E. Delgado, and F. Fischer. 2009. The ecology of *Egeria densa* Planchón (Liliopsida: Alismatales): A wetland ecosystem engineer? Revista Chilena de Historia Natural 82: 299-313.

Yoshiyama, K. and J.H. Sharp. 2006. Phytoplankton response to nutrient enrichment in an urbanized estuary: apparent inhibition of primary production by overeutrophication. Limnology and Oceanography 51: 424-434.