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Restoration ecology: Ecological fidelity, restoration metrics, and a systems perspective

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ABSTRACT

Although the importance of ecosystem services associated with estuarine wetlands and their functional linkages to other estuarine habitats have been increasingly recognized in the past 60 years, the approach to "restoration" and "rehabilitation" of degraded wetland habitats has largely lacked the application of systems thinking and scientific rigor; and has resulted in a "disconnect" between the science and practice of wetland restoration. Examples of coastal wetland restoration science are discussed in the context of wetland functions that promote secondary production, ecological fidelity and their "connectedness" to both adjacent waters and the coastal zone. A means to integrate restoration science and practice to inform policy, and the quantification of restored functions in a systems framework is also described in the context of a sample case history.

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1. Introduction

Mankind's activities in the Anthropocene have pushed the Earth system outside of its normal operating range into new equilibrium states (Steffen et al., 2005). Not only do many ecosystems differ in pattern and process from those in the past, but the ecosystem concept itself is becoming increasingly framed in the context of climate change, land use, invasive species, reduced biodiversity and other outcomes of human endeavors. These new ecosystem states, often less desirable, are described as "novel, no-analog, or emerging" states (Hobbs et al., 2009; Higgs, 2012). As a consequence, the challenges of ecosystem restoration and rehabilitation have reached new levels of complexity.

There are two broad themes addressed in this paper; first we distinguish between *restoration ecology*, the 'science' of restoring degraded habitats, and the broader inclusion of cultural aspects and practices in what we refer to as *ecological restoration* (Weinstein, 2007). In reality, the line between restoration ecology and practice is oftentimes "fuzzy" (Falk et al., 2006), but both approaches and their integration are critical for the future success of restoration science, and while there is no one single, fixed, "correct" restoration for any particular site, functional criteria can provide tight guidelines for success (Higgs, 1997). Secondly, we link the designs for wetland restoration to the consideration of linkages of the wetland to the estuary as a whole, including the coastal zone; i.e., wetlands should be viewed as interactive components of the broader mosaic of habitats that exchange materials and organisms and which together interactively support the secondary production of marine transients.

2. Restoration ecology: the emerging research paradigm

Although the importance of ecosystem services associated with estuarine wetlands has been increasingly recognized in the past 60 years, the approach to "restoration" and "rehabilitation" of degraded ecosystems has often lacked scientific rigor. The science of restoration ecology manages for change, fosters biodiversity and emphasizes the return of system functions, connectivity, and the production of goods and services to degraded ecosystems. But while "the time is ripe for basic researchers to ask if current ecological theory is adequate for establishing new principles of restoration ecology" Palmer et al. (1997) and Hildebrand et al. (2005) cautioned that "the incredible complexity of nature forces us to simplify the (complex landscapes) we study in order to develop theory and generalities by reducing them to understandable subsets". Because

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ecosystems are inherently dynamic and exhibit non-linearities and behavioral surprises, the ability to predict and manage restoration trajectories has been particularly vexing (Mitsch et al., 1998; Anand and Desrochers, 2004; Ruiz-Jaen and Aide, 2005). Hildebrand et al. (2005) assert further that realistic goals must include multiple scientifically defensible endpoints of functional equivalence. In defining these endpoints, ecologists are seeking new ways to assess acceptable levels of variability in restored ecosystems, most appropriately in a regional or landscape context and within some "bound of expectation" (White and Walker, 1997; Weinstein et al., 1997; SER, 2004; French, 2005). There are also questions related to community stability, resilience and persistence; all central to understanding/predicting whether a restored system will be self-sustaining. Additionally, individual metrics of restoration success must be better defined, quantified, integrated, and raised to levels compatible with measuring ecosystem functions, self-organization and ecological resilience.

Scientists generally agree that the evaluation of restored functions should include measures of *processes* such as primary or secondary production, but may also reflect considerations of biogeochemical cycling, food-web structure, food quality, habitat connectivity, biological interactions, including the presence of invasive species, refuge from predators, key-stone species, donor control (Polis and Strong, 1996; Weinstein et al., 2005), micro-habitat structure, and access to resources. Many species exhibit complex life histories that place them in different parts of the landscape at different times, but their overall success may depend on the quality of specific habitats at critical bottlenecks in their life history. For example, marine transient finfish at mid-latitudes are characterized by life histories that invoke a "coastal conveyor belt" with adults spawning offshore and near estuaries, and young spending their first year of life in estuarine habitats including tidal wetlands (Weinstein, 1981; Deegan, 1983; Weinstein et al., 2009a). Young-of-year complete the cycle by accompanying the adults offshore during their autumn migration to overwintering areas. It is likely that the quality of the estuarine habitats, especially tidal wetlands at mid-latitudes is reflected in growth, condition and survival of young-of-the-year marine transients and is a critical aspect of their successful recruitment to the adult stage.

2.1. Ecological restoration

From a practical standpoint, the human dimensions of ecosystem restoration and rehabilitation place limits on the application of restoration ecology principles; especially ecological fidelity in restoration designs (Higgs, 1997). More than 35 years ago, Cairns et al. (1975) distinguished between the public perception of restoration practices and scientific knowledge: "the characteristics of restored ecosystems are bound by two general constraints, the publicly perceived restoration and the scientifically documented restoration. For example, recovery may be defined as restoration to usefulness as perceived by the users of the resource. This is significantly different than restoration to either the original structure or the original function (or both) as rigorously determined by scientific methodology." Cairns (1995) noted also that societal constraints place practical limits on the outcomes of restoration efforts.

Thus, restoration success comes in at least two fundamental forms, (1) projects that restore ecological fidelity and longevity (self-organizing traits) to sites through the application of best scientific principles; and (2) projects that rest on cultural foundations, restoring sites to some practical use as perceived by society. For some restoration efforts, what constitutes a "natural ecosystem" is being redefined in the context of the density of humans in the landscape and shifting baselines, but what we want to avoid are

impressions of wetland restoration practices that are devoid of ecological fidelity like these examples:

> [Restoration may] *be seen as a sort of gardening with wild species in natural mosaics . . . an expensive self-indulgence for the upper classes, a New Age substitute for psychiatry* (Allen and Hoekstra, 1992). *It distracts intelligent and persuasive people from systematic initiatives* (Kirby, 1994) . . . *to many industrialists and global environmental negotiators . . . ecological restoration appears a fair and benign, Western middleclass, pastoral practice, the kind of activity that harms no one and fills in the gaps among the really big problems* (Higgs, 1997).

2.2. Integrating restoration ecology and ecological restoration

The challenge then is to build a stronger foundation for the science of restoration based on methods that go beyond simple structural criteria, or population parameters (e.g., catch per unit effort) to metrics of restored functions and/or processes. Habitats and whole ecosystems are being restored nationwide, but the fundamental question remains, what *kinds* of ecosystems are being restored? Previous restoration paradigms, e.g., those appearing in the national framework embodied in the US Clean Water Act, managed by the US Army Corps of Engineers, and overseen by federal "coordinating" agencies, have been woefully inadequate (Turner et al., 2001). A critical aspect of the integration process is to gain acceptance of the science (and the need for scientific rigor) by practitioners who will design and implement the projects. A concrete example of one such effort is found in Restore America's Estuaries (RAE), *Principles of Wetland Restoration*; derived through a partnership of scientists and practitioners (RAE, 2001; Weinstein et al., 2001).

Notwithstanding that processes/functions are difficult and rarely measured in restoration projects because of time/funding constraints restoration science must advance to a point where technology transfer of basic research becomes practical in the practitioner/resource manager's tool kit. Whether in the form of a "bound of expectation", "probabilistic laws" (Ehrenfeld, 2000) or other goal-setting paradigm, the asymptotic endpoint(s) of the restoration effort must be established early so that practitioners can answer the simple question: was the restoration project successful? The scientific basis for determining this success is currently, at best, "thin" (Henry and Amoros, 1995; Stanturf et al., 2001), and the "myths" that these and other authors refer to have been variously described (e.g., Cabin, 2007; Hildebrand et al., 2005). Zedler (2007) has gone so far as to challenge the very use of the term "success", a point well taken, but for the moment, we will simply note her suggestion for "abstinence" or "rendering opinions" when the term is used, and revert to the bad habit here. Because the scope of restoration science is so broad and encompasses such a wide range of ecosystems, we present a case study to describe how restoration science and practice can be integrated to better inform policy, stakeholders and decision makers. We focus on coastal wetland ecosystems and their role in supporting secondary production of marine and estuarine nekton and their forage base.

2.3. "Donor Control" and restoration planning

Marine transient species that are largely marine as adults, benefit from tidal salt marshes and their production *with or without directly occupying these habitats* (Litvin and Weinstein, 2003; Weinstein et al., 2005). Many are highly mobile, and tend to cross habitat boundaries in their quest for food and refuge. Species of interest include taxa of estuarine resident and marine transient species considered to be of "value" to mankind, but includes work
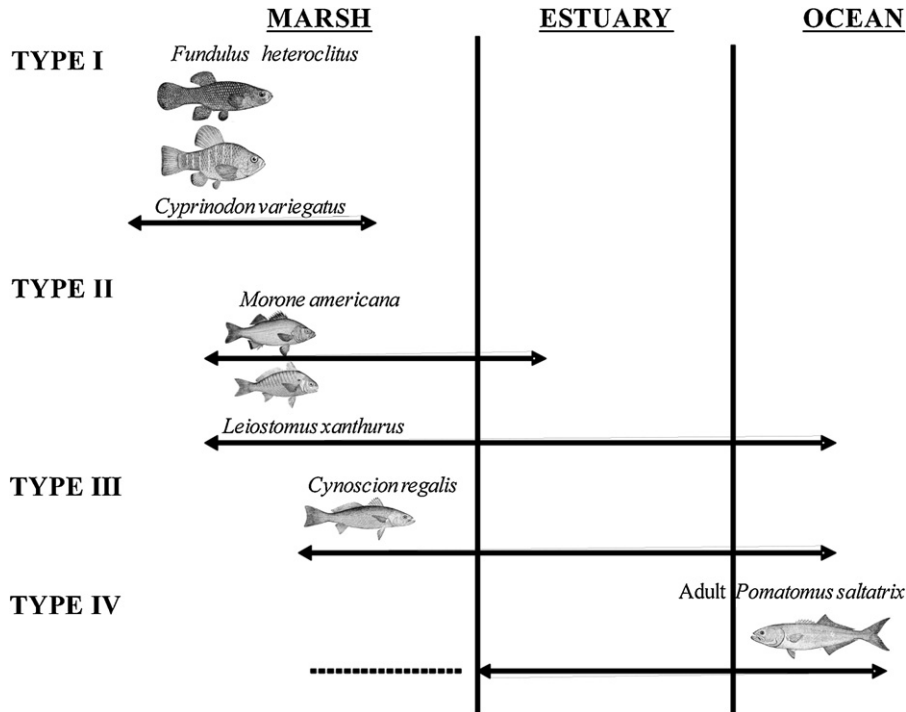
Fig. 1. Spatial distribution of nekton that use tidal salt marsh habitats during all or a part of their life history.

on their forage base as well. Some adult marine transients migrate to estuaries to feed or spawn, but in any case vast numbers of their young spend most of their first year in estuarine habitats (Fig. 1; adapted from Litvin and Weinstein, 2003).

Many marine transients and some estuarine residents are generally not habitat specialists but rather are opportunistic in utilizing resources and habitats across the entire estuarine landscape. Restoration planners should and must, therefore, view restoration goals in the context of the full estuarine mosaic and the processes that exchange materials and organisms between adjacent habitats (e.g., salt marshes and the open waters of the estuary). Stated simply, *salt marshes do not function in isolation when supporting estuarine secondary production*, but rather are integrated components of larger systems (Childers et al., 2000; Weinstein et al., 2005). Moreover, the open waters of the estuary may be donor-controlled, i.e., systems in which the rate of import, availability, or dynamics of allochthonous resources (such as products of the salt marsh), is controlled by external donor systems rather than by consumers. Indeed, consumers may be more abundant when supported by allochthonous resources than if supported solely by the in situ resources of open waters (Polis et al., 1995). The latter concept is critical in the context of restoration ecology, because failure to account for trophic subsidies in the open estuary may result in restoration designs that have negative feedback on the recruitment success of numerous marine transients.

In the following section, the restoration precepts discussed thus far are summarized in the context of research we have conducted in Delaware Bay and other estuaries. An attempt is made to synthesize available data in a framework linking restoration ecology to ecosystem services, but focusing on the role of tidal salt marshes in subsidizing fisheries production in the estuary. Specific consideration is also given to impacts of the invasive haplotype of *Phragmites australis* on marsh processes and functions. A systems view (i.e., a "whole estuary" approach) is adopted to help extend our findings in specific habitats and regions to the entire ecosystem. Although the narrative centers primarily on the Delaware Bay, it is

supplemented by studies at other locations. The general approach focuses on the flow of nutrients from primary producers to finfish using stable isotope analysis, with the added use of the latter method as 'biomarkers' to infer degrees of site fidelity in these otherwise mobile taxa (Litvin and Weinstein, 2004). We also use biochemical condition, principally the presence of specific lipids and lean protein mass, to infer the quality of habitat types. The fundamental premise is that levels of fat reserves reflect the "well-being" of individuals, and may be indicative of the overall value of habitats to consumers in secondary production. Several questions were addressed in this long-term research: (1) what are the trophic linkages between primary producers and estuarine finfish; (2) what are the relative contributions of the primary producers to the estuarine food web; (3) does *P. australis* contribute to the trophic spectrum of marine fishes; (4) is biochemical condition a sensitive indicator of essential fish habitat; and (5) what are the allometric relationships among body constituents and survival, growth and reproduction?

3. Delaware Bay coastal wetlands—restoration ecology in a "whole" estuary (systems) perspective

The Delaware Bay estuary shoreline is fringed by approximately 200,000 acres (81,000 ha) of nearly contiguous tidal salt marshes, but marshes in the oligohaline-tidal freshwater portions of the estuary below Philadelphia, PA are dominated by an introduced variety of *P. australis* comprising ∼40,000 acres (16,000 ha; Fig. 2) (Weinstein and Balletto, 1999; Weinstein et al., 2000a; Saltonstall, 2002). One of the most expansive ecotones of its type in the mid-Atlantic region, Delaware Bay tidal salt marshes play a critical role in the production and recruitment success of commercially and recreationally valuable species and their forage base. Many of the Bay's wetlands, however, have been degraded by anthropogenic activities, nearly back to colonial times, by dredge and fill to reclaim lands for living space, impounded and/or diked for agricultural purposes and wildlife management including waterfowl
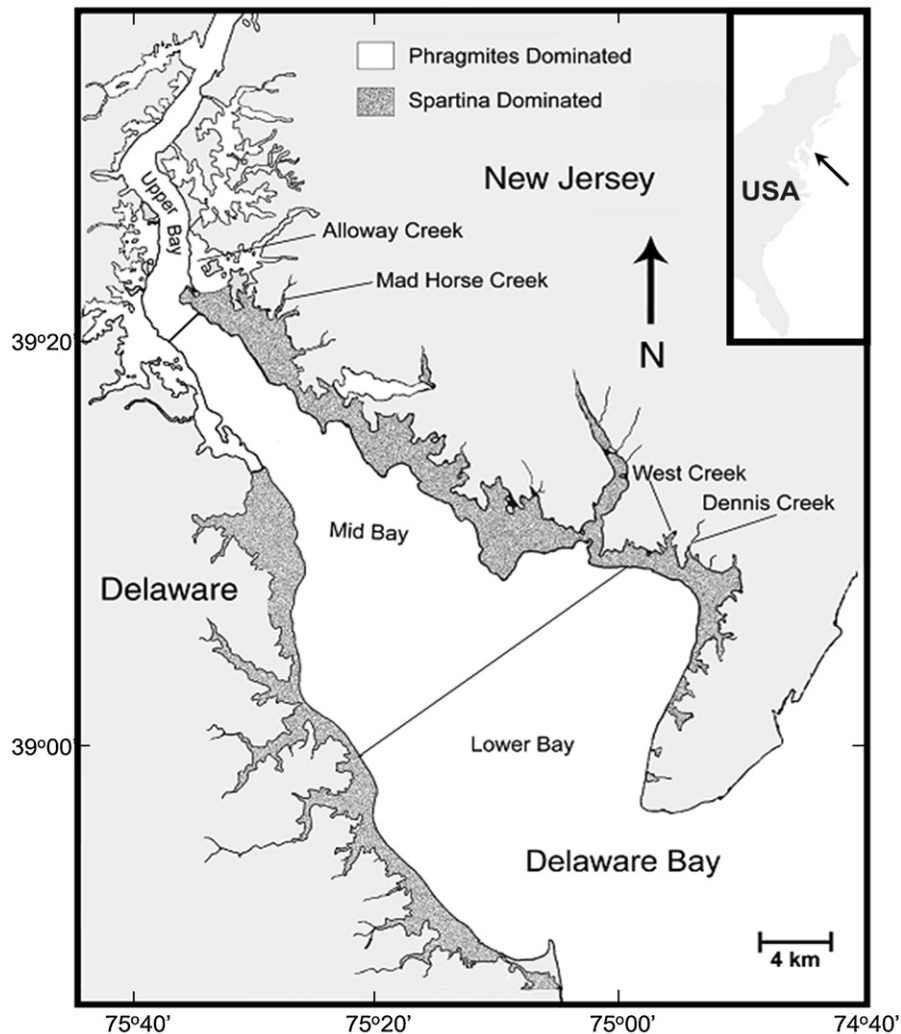
Fig. 2. Locations of the three sub-regions and affiliated marsh creeks that formed part of these studies in Delaware Bay, USA (lower, mid and upper Bay, Alloway Creek, Mad Horse Creek, West Creek, and Dennis Creek) and the extent of tidal salt marshes dominated by *Spartina alterniflora* and *Phragmites australis*.

and muskrats (*Ondatra zibethicus*), polluted, and/or reduced to virtual wetland monocultures by invasion of *P. australis* (16,000 ha). Large-scale efforts to restore these degraded wetlands have been undertaken in the past several decades, including the restoration of more than 14,000 acres (5666 ha) of formerly diked salt hay farms and *Phragmites* degraded marshes known as the Estuary Enhancement Program (EEP) (Teal and Weinstein, 2002). Primary production in the Delaware Bay water column is also light limited (Pennock and Sharp, 1986) resulting in little or no bottom coverage by seagrasses or benthic macroalgae thus making it easier to sort out the end-members of primary production, and track nutrient flux. For these reasons and others, the Delaware Bay is an ideal 'laboratory' for examining the links between wetland restoration, the overall mosaic of estuarine habitats, and secondary production of marine transient finfishes.

Since 1996, we have conducted nearly continuous research throughout the system, divided into six regions of interest-open waters of the lower, mid and upper Delaware Bay and their adjacent marshes in each of these bay regions. Supplemented by projects in the Hudson River estuary, and the Cape Fear River estuary, the work has focused on four taxa representing at least three trophic levels, weakfish (*Cynoscion regalis*), white perch (*Morone americana*), mummichogs (*Fundulus heteroclitus*) and bay anchovy (*Anchoa mitchilli*). Additional data were collected on primary producers

including C_3 (*P. australis*) and C_4 (*Spartina* spp.) macrophytes, benthic microalgae and phytoplankton (reported as suspended particulate matter; SPM). Field and laboratory procedures have been presented in previous publications and will not be reproduced here; but for details see Wainright et al. (2000), Weinstein et al. (2000b, 2009b) and Litvin et al. (2011).

3.1. Flux of nutrients from primary producers to finfish

A principal species in our work, juvenile weakfish were collected throughout Delaware Bay between 1998 and 2001 in tidal salt marsh creeks, open waters (Fig. 2), and at the bay mouth in late fall, at a time when they were preparing to move offshore to overwinter. Canonical discriminant analysis was used to extract several prominent features in these data (Figs. 3(a) and (b) and 4) (for details, see Litvin and Weinstein, 2004). As noted in Fig. 3a, the canonical functions classified young weakfish to their location of collection at an average rate of 84%, while the cross validation ('jackknife') success rate was 80%. In all, 102 of 141 fish collected at the bay mouth in the fall were classified into the lower bay category (Fig. 3a) (Litvin and Weinstein, 2004). Site fidelity in two related species, Atlantic croaker (*Micropogonias undulatus*) (Miller and Able, 2002) and spot (*Leiostomus xanthurus*) (Weinstein et al., 1984a) were reported in Delaware Bay and Chesapeake Bay, respectively. In the
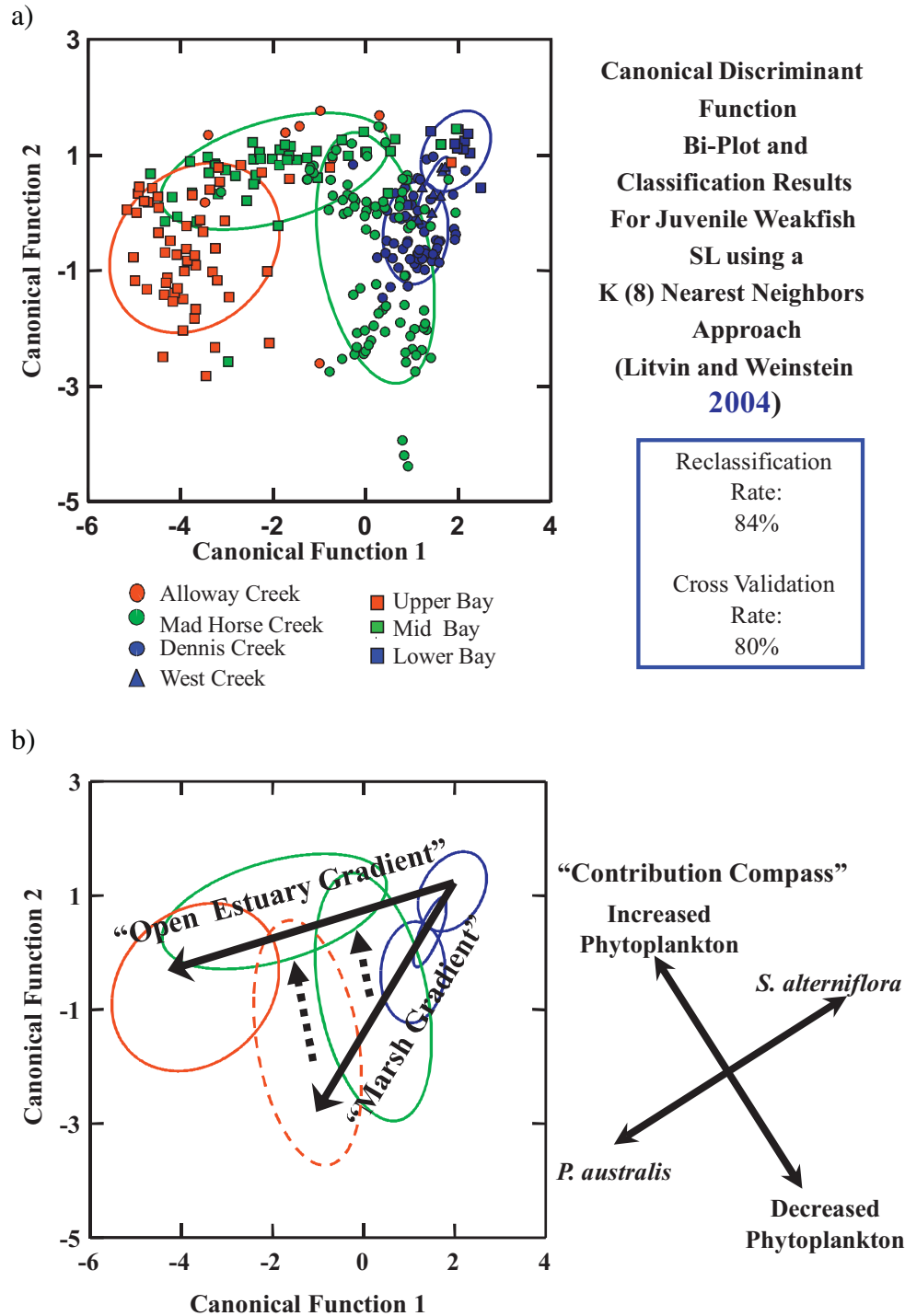
a)

Canonical Discriminant
Function
Bi-Plot and
Classification Results
For Juvenile Weakfish
SL using a
K (8) Nearest Neighbors
Approach
(Litvin and Weinstein
2004)

Reclassification
Rate:
84%

Cross Validation
Rate:
80%

- ● Alloway Creek
- ● Mad Horse Creek
- ● Dennis Creek
- ▲ West Creek
- ■ Upper Bay
- ■ Mid Bay
- ■ Lower Bay

b)

"Contribution Compass"

Increased
Phytoplankton

S. alterniflora

P. australis

Decreased
Phytoplankton

Fig. 3. (a) Plot of standardized canonical function scores based on the carbon, nitrogen, and sulfur isotopic signatures of weakfish collected in Delaware Bay open waters and tidal salt marsh creeks. (b) Primary producer source contributions comprising juvenile weakfish (*Cynoscion regalis*) tissue values. The two gradients shown reflect the end-member contributions from *Phragmites australis* in the upper Bay and *Spartina* spp. in the lower Bay. Mid Bay values (dotted ellipse) are intermediate reflecting contributions from both macrophytes. Fish collected in tidal marshes were supported by greater contributions from benthic microalgae whereas fish collected in open waters had tissue values reflecting contributions from phytoplankton. A 'contribution compass' summarizing these results is also shown.

former study, the authors noted that most returns in their year-long mark-recapture effort came from the same tidal creek where micro-coded wire tags were applied, whether restored or reference marshes, for approximately two months until fall emigration commenced. More than 95% of recaptures came from what Miller and Able (2002) referred to as "home creeks". Earlier, Weinstein (1983) and Weinstein et al. (1984b) reported residency periods for

juvenile spot of up to 90 days in individual tidal creeks of the Guinea Marshes, York River, Virginia. Thus, site fidelity may be a more common trait exhibited by juvenile marine transients during their first year in the estuary than heretofore thought.

The tidal-marsh and open-water gradients depicted by these analyses underscore the importance of locally produced organic matter in the trophic spectrum of Delaware Bay (Wainright et al.,
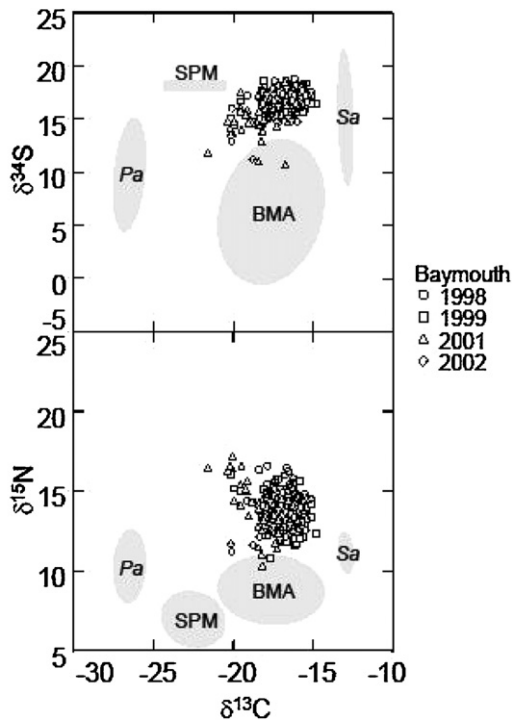
Fig. 4. Stable isotope signatures of juvenile weakfish captured at the mouth of Delaware Bay during a four-year period just prior to offshore migration to over-wintering grounds.

Fig. 5. Water column feeding strategies in four marine finfish, bay anchovy (*Anchoa mitchilli*), white perch (*Morone americana*), weakfish (*Cynoscion regalis*) and common mummichog (*Fundulus heteroclitus*). Mummichog data included values derived from tidal salt marsh collections in the Hudson River Estuary (Weinstein et al., 2009b).

2000; Weinstein et al., 2000a; Litvin and Weinstein, 2003) and other estuarine systems (Peterson et al., 1986; Deegan and Garritt, 1997; Deegan et al., 2000). Moreover, the stable-isotope composition and resulting canonical scores of juvenile weakfish captured in tidal salt marshes of Delaware Bay reflected their underlying dependence on local dominant vegetation for extended periods, either *P. australis* or *S. alterniflora*, supplemented by nutrients from phytoplankton and benthic microalgae. Together, the isotopic signatures of the macrophytes defined the end-members of what we described in Litvin and Weinstein (2004) as the marsh or open-water gradients, and ultimately to what we called the "contribution compass" from plants to the tissues of these species (Fig. 3b).

Another striking feature of these results reflects the importance of marsh productivity and the links between secondary production in the marsh-open water complex and coastal food webs (Fig. 4). The isotopic signatures collected in weakfish as they congregated at the bay-mouth in the fall prior to offshore migration and overwintering are telling in two ways. First as suggested by Haines (1979), the presence of juvenile weakfish that are classified as recent inhabitants of salt marshes indicates not only that marsh macrophyte production is exported for utilization in open-estuary food webs (Eldridge and Cifuentes, 2000) but that additional export occurs via the juvenile weakfish themselves. Secondly, the majority of juvenile weakfish collected just prior to emigration in the four years of collection (Fig. 4) were reclassified as being from either marsh creeks or open waters within the lower bay (133 of 141); all locations where marsh macrophytes accounted for a significant proportion of the organic matter entering the food web (Litvin and Weinstein, 2004). Therefore, a large proportion of the organic matter exported from Delaware Bay via juvenile weakfish had its origins in salt-marsh macrophyte production.

The foregoing observations are also documented in our studies on other species, but there is an interesting example of tissue stable isotope composit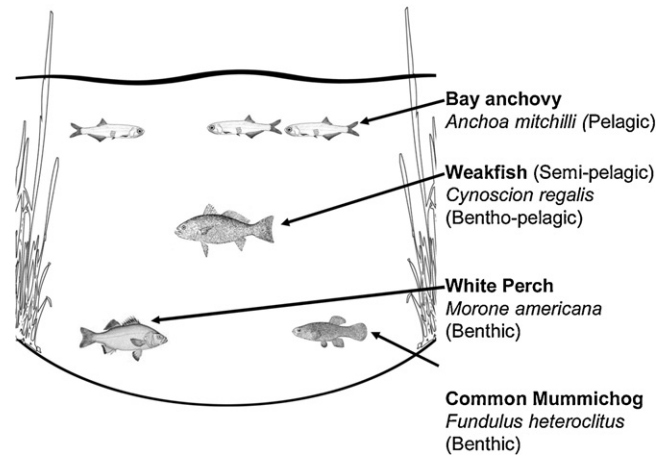ion that appears to reflect life history patterns, and highlights the remarkable 'resolving' power of the stable isotope approach. Mummichogs and white perch are generally considered to be bottom (benthic) feeders, weakfish bentho-pelagic (or semi-pelagic), and bay anchovy pelagic (Fig. 5) (Able and Fahay, 1998). In salt marshes, there was a transition in whole-body isotopic composition in all four species that could be associated with changes in the dominant macrophytes from *Spartina* spp. in the lower bay through a transition zone in mid Bay to the upper Bay where *P. australis* formed virtual monocultures (Fig. 6a). Superimposed on the vegetation dominance pattern was the relative contribution of benthic microalgae in the order of mummichog = white perch > juvenile weakfish > bay anchovy. This pattern mirrored the increasing benthic-to-water column orientation of the three taxa (Fig. 6a). Conversely, and as expected, phytoplankton (as SPM) contributed more to bay anchovy and weakfish biomass than did benthic microalgae, although the latter appeared to make a greater contribution to weakfish (Fig. 6a). These patterns generally held in open waters with bay anchovy and weakfish partially separated by the relative contributions of *Spartina* spp. and *P. australis* in the transition from lower to upper bay (white perch and mummichogs were not collected in the open bay). However, juvenile weakfish appeared to be less dependent on benthic microalgae when in open waters where they utilized phytoplankton to approximately the same degree as bay anchovy (Fig. 6b). It is clear from the results described above, and our earlier work (Wainright et al., 2000; Weinstein et al., 2000b; Currin et al., 2003; Litvin and Weinstein, 2003), as well as that of others (Deegan and Garritt, 1997), that marine transients benefit from the presence of salt marshes *without necessarily having to occupy them*. Thus the estuarine "commissary" serves up different recipes in different regions, and the many estuarine-dependent species apparently take full advantage of these varied combinations of primary producers when seeking energy for rapid growth and survival.

These observations demonstrate that the estuary as a whole is characterized by gradients in available organic matter, a boundary-less state, rather than sharp transitions from one food source to the next (Fig. 7). For bay anchovy and juvenile weakfish, both marine transients, we have observed a gradual change in whole-body stable isotope composition, trending from higher proportions of marsh macrophytes and benthic microalgae in and near tidal creeks to higher proportions of phytoplankton organic sources as one moves toward open waters (Litvin and Weinstein, 2003).
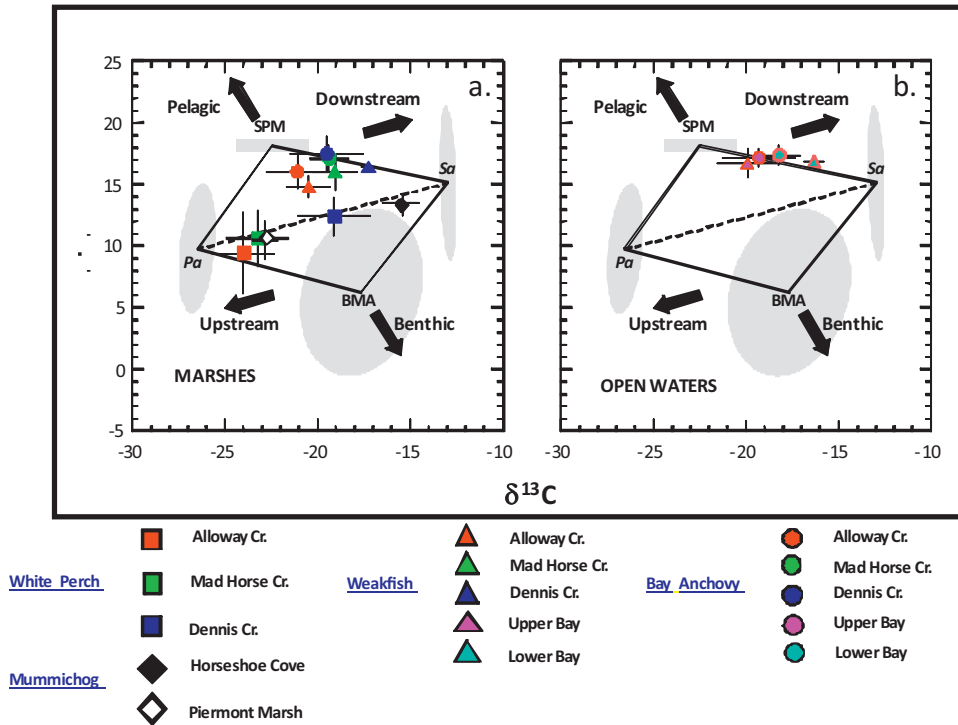
Fig. 6. Stable isotope signatures based on capture location (marshes; open waters), of the Delaware and Hudson River estuaries. In addition to the influence of dominant macrophytes (*Spartina alterniflora*, Sa and *Phragmites australis*, Pa) stable isotope values of benthic feeders were also influence by benthic microalgae (Bμa), while those of bentho-pelagic and pelagic species have a greater proportionate signature associated with phytoplankton (as SPM).

Similarly, along the long axis of the estuary, *P. australis* tends to supply a greater proportion of the nutrition of marine transients in and near brackish salt marshes, whereas *Spartina alterniflora* and benthic microalgae dominate the nutrition of these taxa in and near marshes in the mid and lower estuary (at salinities greater than about 10‰). Yet there is rarely a time when the signatures of virtually all primary producers are *not* detected in most taxa whether they are marsh resident (Wainright et al., 2000; Currin et al., 2003), estuarine resident, or marine transient (Weinstein et al., 2000a; Litvin and Weinstein, 2003). We are not fully certain of the specific export routes for salt marsh primary production—as tidal flux of particulate organic matter, as dissolved organic carbon especially in those portions of the estuary where the marsh:open water ratio >0.8 (Eldridge and Cifuentes, 2000), as allochthonous inputs, in pulsed events, or in the tissues of emigrating marine transients. What is clear, however, is that nutrients from the marsh are reaching marine transients *whether or not they spend time in the marsh*. As noted earlier (Weinstein et al., 2005) we appear to have come full circle, and perhaps Teal (1962) was correct all along when he suggested that about 45% of the primary production in Georgia salt marshes was available to estuarine consumers. However, rather than detrital export driving the mechanics of the process, as Teal suggested, the process appears to be far more complex, and it is essential that these considerations are incorporated into restoration planning and designs.

3.1.1. Inferred movements

There is another, equally important outcome in the sum total of the data presented. The ability of canonical discriminant analysis to correctly identify the collection location of small juvenile weakfish (≤60 mm standard length [SL]) validated the pattern of unique isotopic signatures associated with short-term residency in open-water and tidal-marsh habitats and allowed us to infer the movements of larger (>60 mm SL), more mobile individuals

within Delaware Bay (Hobson, 1999). The classification results for medium-sized fish (61–100 mm SL) collected in the open waters of the upper Bay implied the same pattern of site fidelity found in the small individuals. Conversely, for mid- and lower bay open waters, greater than half of medium-sized individuals collected appeared to be recent arrivals from up-bay habitats or adjacent tidal marshes, while in tidal marshes there was little evidence of emigration. With the exception of Mad Horse Creek, we collected few large juveniles (>100 mm SL) in the mid- and upper bay, indicating that they had previously emigrated from these habitats. Large juveniles collected in the lower Bay open waters, like medium-sized individuals, appeared to be a mixture of lower Bay residents and recent arrivals from other habitats, although a higher proportion of large individuals, 73% versus 30%, exhibited the stable-isotope signatures characteristic of extended residency in the lower Bay. These results indicate that a proportion of the large juveniles classified as "lower Bay open" arrived as medium-sized individuals and acquired the stable-isotope signature of the lower bay as they grew. Our premise is further supported by the high growth rates exhibited by juvenile weakfish in Delaware Bay. An individual arriving in the lower Bay at 75 mm SL will increase its biomass by a factor of nearly 5 at 125 mm SL (Litvin, unpublished data). At this new biomass, fish will have an isotopic signature that reflects in situ trophic transfers and will dilute any previously acquired isotopic signature even in the absence of metabolic turnover (Hesslein et al., 1993; Herzka and Holt, 2000; MacAvoy et al., 2001). A similar proportion of weakfish collected at the mouth of Delaware Bay in late October and early November (Fig. 4) accumulated the bulk of their biomass in lower Bay open waters, leading us to conclude that movements from up-bay habitats and the adjacent tidal marshes followed by continued growth in the lower bay continued through the time of emigration. Of note were the few (6 of 176) medium-sized or large juveniles collected in the lower bay open waters or at the mouth that had directly emigrated from upper bay habitats, rather
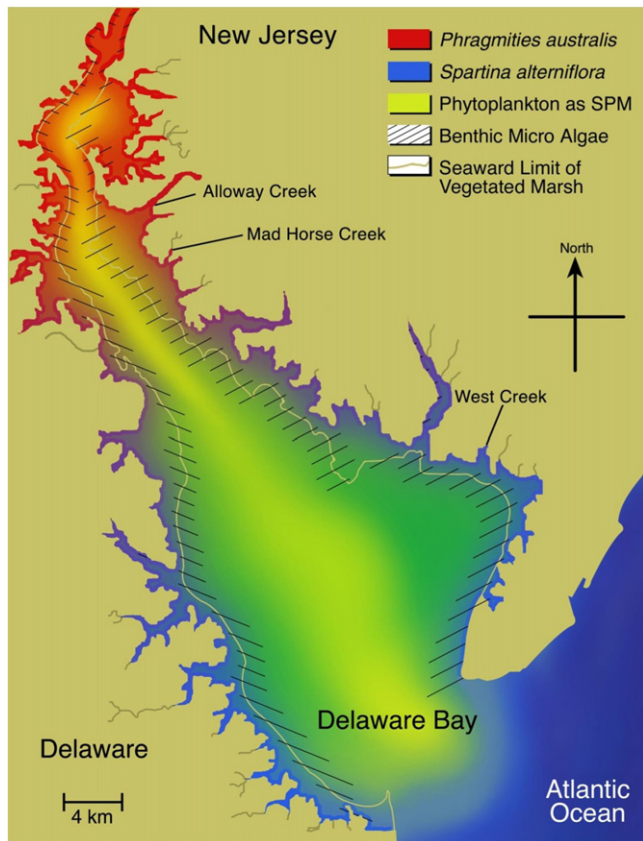
Fig. 7. Conceptual diagram of nutrient gradients developed from data on primary producer sources in Delaware Bay (from Weinstein et al., 2005). Primary colors (blue, red, yellow) are used to depict the main *local* source of organic matter: blue. *Spartina* spp.; red, *Phragmites australis*, yellow, phytoplankton; and diagonal hatching, benthic microalgae. Each organic matter source is "blended" with the others to depict the gradients of general availability to secondary consumers in the food web. Nutrient flux and the establishment of the gradients likely result from a combination of pulsing events, trophic relays, spawning migrations, dissolved organic matter export, and detrital infusion. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of the article.)

than making intermittent stopovers at tidal marshes during their down-bay migration.

These observations are consistent with the expected behavior of young marine transients (Chao and Musick, 1977; Weinstein and Brooks, 1983; Szedlmayer et al., 1990); small juveniles recruit throughout the estuary early in the season and spend sufficient time in a given open-water region or marsh creek to acquire a distinct isotopic signature, i.e., they exhibit substantial site fidelity (Deegan and Garritt, 1997). As they grow, they begin to move out of the marsh creeks and (or) upper Delaware Bay, sometimes moving directly to lower bay habitats, but more often exhibiting "saltatory" behavior, i.e., moving partially down-bay while spending sufficient time in the intermediate regions to acquire the dominant isotopic signature of that locality. Juvenile weakfish ultimately arrive in lower Delaware Bay, though apparently not in unison. Early arrivals are resident for a period that allows accumulation of sufficient biomass in the lower bay to be reclassified as having originated there, while others retain the signature of other locations until the time they emigrate from Delaware Bay.

3.2. Biochemical condition

Carnivorous fishes are reliable indicators of the condition of complex ecosystems as they are the tertiary link in the food web,

therefore, the magnitude of protein and fat deposition and the level of fat reserves can not only be used to assess the "degree of well being" of individuals but can also serve to integrate the overall value of habitats to their production. Using this premise, the concept of habitat quality for a marsh resident, the mummichog, *F. heteroclitus*, was examined in relatively undisturbed and *Phragmites*-dominated tidal salt marshes along the mid-Atlantic coast of the USA. In addition to our earlier work on the trophic spectrum of this species (Wainright et al., 2000; Currin et al., 2003), an interesting "natural experiment" was available to us because (1) the species spends its entire life cycle within the confines of the marsh and has an extremely small home range (Valiela et al., 1977; Meredith and Lotrich, 1979; Teo and Able, 2003), and (2) many tidal salt marshes, particularly those with brackish salinities, have become dominated by virtual monocultures of the invasive variety of the *P. australis* that is perceived to reduce habitat quality for *F. heteroclitus*, and general access to the marsh plain by nekton (Weinstein and Balletto, 1999; Saltonstall, 2002; Hagan et al., 2007). By adopting a whole system approach, we essentially had a "captive audience", one in each of two isolated marsh complexes located on the Hudson River estuary (Weinstein et al., 2009b), a polyhaline system dominated by *S. alterniflora* and a meso-oligohaline system dominated by an invasive variety of *P. australis*. In addition to others, the following questions were addressed in this study: (1) were there any differences in biochemical condition, principally the deposition of energy reserves, in mummichogs captured seasonally in the *S. alterniflora*-dominated "natural" and the *P. australis*-invaded salt marshes; (2) were any differences related to size distributions of individuals in the populations; and (3) could biochemical condition ultimately serve as a success criterion to evaluate the functional success of wetland restoration? (Weinstein et al., 2009b).

3.2.1. The use of biochemical condition as a metric of restoration success

Biochemical condition of individual *F. heteroclitus* was evaluated on the basis of triacylglycerol (TAG), free fatty acid (FFA), and phospholipid (PL) composition and concentration (for details of the methods, see Weinstein et al., 2009b; Litvin et al., 2011). Other lipid classes were not examined in detail but were included in the calculation of total lipid mass. It is generally accepted that the size of lipid stores and their composition can be used to predict whether a fish is ready to migrate, preparing to overwinter, or is likely to have future reproductive success (Ackman, 1980; Shulman and Love, 1999). Previous results of lipid class dynamic studies in young teleosts suggest that TAG was the main form of lipid used in energy storage; therefore, this lipid class was used as our primary indicator of biochemical condition (Ackman and Eaton, 1976; Lochmann et al., 1995, 1996; Lochmann and Ludwig, 2003; Heintz et al., 2004). Because, free fatty acids and phospholipids can contribute to energy metabolism (Ross and Love, 1979; Yuneva et al., 1991; Henderson and Tocher, 1987) and may be important in the reproductive cycle (Ackman, 1980), they were also examined in individual fish.

3.3. Morphometric–biochemical condition comparisons

The tradeoffs between energy allocation for growth, reproduction, and the laying down of sufficient storage reserves for periods of resource scarcity as "competing demands" in pre-reproductive organisms living in seasonal environments have been described by numerous authors (Walters and Juanes, 1993; Fullerton et al., 2000; Post and Parkinson, 2001). This is especially important in north-temperate fishes because experimental and field data suggest that energy availability is often limiting, i.e., fish in their natural environments tend to grow at less than their physiological optimum at a given temperature (Post and Parkinson, 2001).
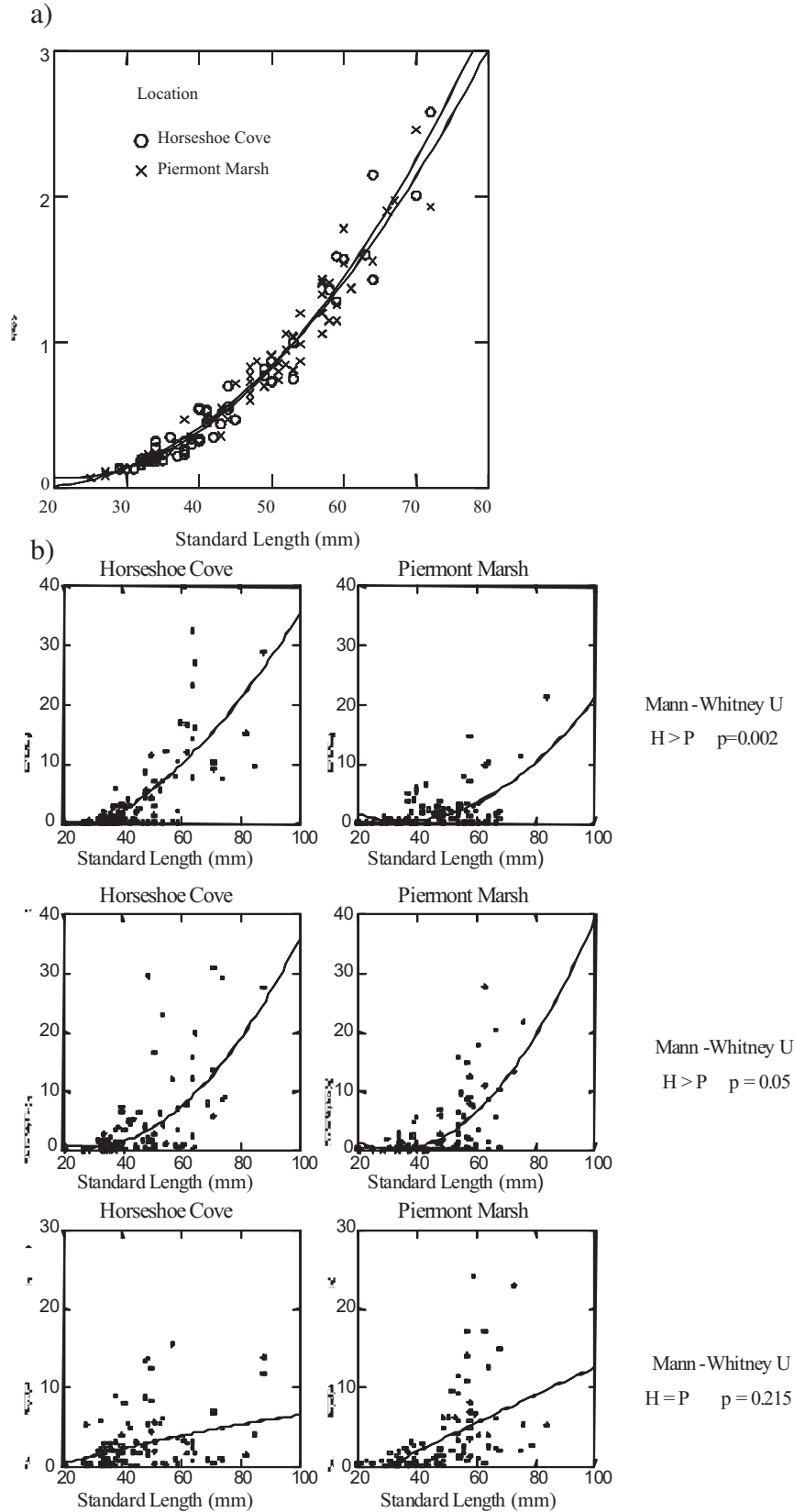
Fig. 8. (a) Somatic condition, dry weight by location for mummichogs *F. heteroclitus* captured in two tidal salt marshes, Horseshoe Cove and Piermont Marsh on the Hudson River estuary. (b) Total free fatty acids, triacylglycerol (TAG), and phospholipids versus standard length (millimeters) in individual mummichogs (*F. heteroclitus*) captured at Horseshoe Cove (H) and Piermont (P) Marsh. All lipid values expressed in milligrams (mg).

Although our comparisons on a dry weight (a morphometric indicator) basis alone *did not detect differences* in somatic condition of *F. heteroclitus* populations in the two marshes (Fig. 8a), the examination of energy reserves in these fish after removing the potential confounding influences of the reproductive cycle and parasitization clearly indicated that significant differences occurred in TAG and free fatty acids levels (milligrams per gram dry weight; Fig. 8b). Thus, supplementing morphometric data with higher order biochemical condition was a more sensitive measure of the condition of individuals produced in these habitats. Similar results were observed by Dibble and Meyerson (2012) for lipid concentrations in *F. heteroclitus* that were lower in fish captured in flow restricted marshes dominated by *P. australis*, compared to levels in fish captured in *S. alterniflora* dominated marshes. This conclusion is also supported by Mommsen (1998) who suggested that a 100 g fish acquiring 1 g of lipid was unlikely to change in length, and its weight gain was hardly detectable in the statistical noise, yet the fish has added a statistically significant amount of energy.

By focusing on energy reserves, principally TAG, we have been able to demonstrate that mummichogs residing in a polyhaline *S. alterniflora*-dominated tidal salt marsh were better able to acquire energy reserves for reproduction and overwintering survival than fish residing in a *Phragmites*-dominated marsh. Thus, *Phragmites* invasion and its consequent impacts may be contributing to lower quality habitat for mummichogs (Weinstein and Balletto, 1999; Hagan et al., 2007; Weinstein et al., 2009b; Dibble and Meyerson, 2012).

4. Synthesis and conclusions: restoration ecology, secondary production and the estuarine landscape

Much like MacArthur's (1958) seminal paper on habitat partitioning by neotropical songbirds (Parulidae) on individual trees in mature boreal forests, coastal estuaries including the Delaware Bay may be viewed as hierarchically patchy environments where habitat selection by often related taxa including marine transients appears to reduce competition and maximize their survival during the first year of life. Moreover, life history strategies appear adaptive, and the system appears highly programmed in the seasonality of spawning and recruitment, feeding strategies, and habitat use patterns. Where the system has been 'disrupted' by an invasive variety of *P. australis*, we have been able to detect differences in habitat quality based on the biochemical condition of species occupying them (Weinstein et al., 2009b, 2010). This higher order metric bridges the gap between an organism's presence or density in a habitat and the level of secondary production that might be anticipated. Moreover, invasion events like that observed for *Phragmites* provide an opportunity to better understand species-habitat associations and may provide clues for developing restoration strategies. For example, by influencing marsh surface elevation and topographic relief, and consequently local hydroperiods, *P. australis* will influence marsh access by nekton and general exchange of materials between the marsh plain and the adjacent drainage network. Lower rates of secondary production and survival rates for marsh resident fishes may result. Restoring hydroperiods and the tributary drainage of the marsh then become critical strategies in the restoration design.

In earlier work in the Cape Fear River, North Carolina, USA we also observed differential habitat use during the earliest life stages of nekton recruited to the estuary as well as habitat related mortality rates in juvenile spot (*L. xanthurus*; Weinstein and Walters, 1981). Young Atlantic croaker (*M. undulatus*) tended to accumulate in the upper reaches of the Cape Fear estuary in early spring, while spot (*L. xanthurus*) although also more abundant up estuary,

occurred in far greater densities in tidal salt marshes throughout the estuary (Fig. 9; reproduced from Weinstein et al., 1980; Weinstein, 1985). These species appeared to have different behavioral strategies for reaching different habitats (Weinstein et al., 1980). A similar pattern was observed in length-frequency distributions with larger croaker (within the limits of gear efficiency; Weinstein et al., 1980) observed at the head of the estuary near the turbidity maximum zone (Fig. 10a) while larger spot appeared to move out of this region, most likely to the adjacent salt marshes (Fig. 10b). We observed similar relationships with spot in the Chesapeake Bay including the marsh creek-seagrass meadows continuum (Weinstein and Brooks, 1983; Smith et al., 1985). Marsh habitats are similarly 'programmed' in terms of preferred habitats utilized by related and other taxa (Fig. 11). Confamilial and congeneric taxa especially appear to minimize spatial and (although not shown in Fig. 11) temporal habitat usage. Examples include *Menidia menidia* and *M. beryllina*, *Mugil cephalus* and *M. curema*, *Paralichthys* spp., several sciaenids and penaeid shrimps.

4.1. Restoration ecology in a landscape context

There are many reasons to adopt a landscape approach to restoration efforts. First, the sustainability of many systems varies on spatio-temporal scales that are linked in both bottom-up and top-down directions (Wu and Loucks, 1995; Wu, 1999, 2012). Secondly, as argued by Wu (2012), landscape represents the most pivotal scale for place-based ecosystem management because "landscape gives identity to place and landscape is where past and present meet" (Phillips, 2007). This is just one of the reasons to meld restoration ecology with ecological restoration. It is beyond the scope of this paper, however, to discuss the various meanings of the term, but suffice it to say that the term "landscape" connotes both ecological and social elements that address spatial scale and the different aspects of a landscape that are emphasized (e.g., Wu and Hobbs, 2007). "Human perceived" landscapes, for example, coincide with geographical units such as watersheds or urban centers. These spatial domains of human perception also resonate well with the public and decision-makers "who are conscious of the environmental setting in which they live, work and play" (Wu, 2012). As noted above, John Cairns made similar comments decades earlier.

Secondly, estuarine ecosystems are highly complex and may be viewed as multi-scale hierarchies; i.e., they constitute spatially heterogeneous areas of various sizes. We have observed that different estuarine species perceive, experience and respond to spatial heterogeneity at different scales, and patterns and processes in this landscape tend to have these different characteristic scales. Some species tend to perceive their environments as coarse grained (e.g., mummichogs) while others perceive a more fine-grained fabric over the year (e.g., juvenile weakfish) (Levins, 1968). At one level, populations of mummichogs may spend their entire life cycle in permanent ponds on the marsh surface (Weinstein et al., 2010), while juvenile weakfish make use of virtually all available habitats during their first year, but in a "programmed" fashion residing for extended periods in specific areas (Litvin and Weinstein, 2004). Restoration science, therefore, may need to consider a variety of scales to accommodate the successful production of *interacting* species at scales ranging from tens to perhaps 10,000 m^2.

4.1.1. A systems view of restoration science in coastal wetlands

Although early investigators proposed important modifications to the role of tidal–marsh organic matter in estuarine secondary production, changes to Teal (1962) original model have largely been refinements rather than outright rejection or dismissal. These refinements demonstrated how organic matter from tidal salt
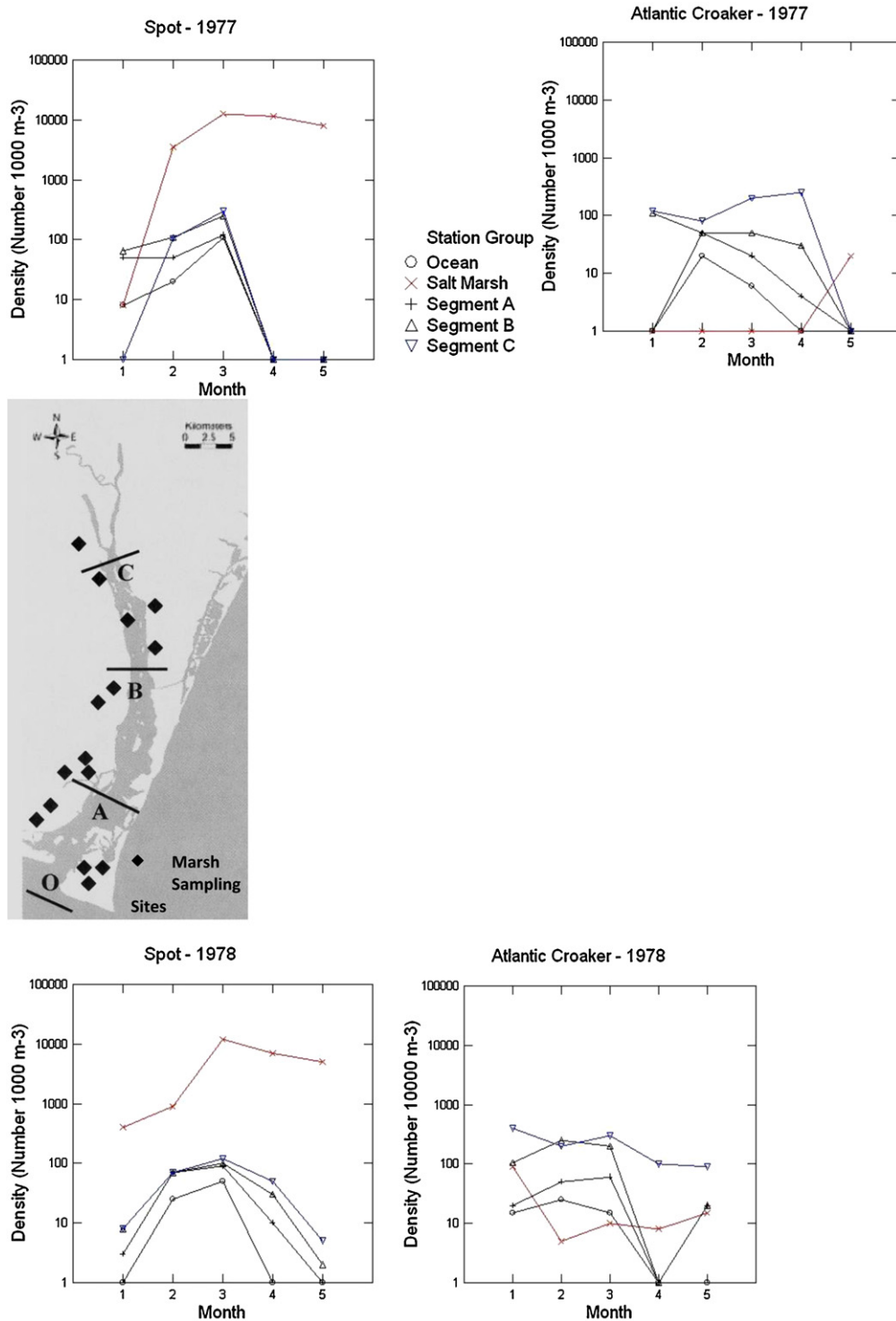
Fig. 9. Peak densities for postarval and early juvenile spot, *Leiostomus xanthurus* (number 1000 m^{-3}) and Atlantic croaker, *Micropogonias undulatus* (number 1000 m^{-3}) captured in the nearshore ocean, estuary and at marsh stations in the Cape Fear River estuary, 1977 and 1978 (map insert), Ocean samples were collected by the Carolina Power and Light Co. All mainstream estuary stations (Groups A, B and C) extending from the river mouth (A), upstream to near Wilmington, North Carolina (C) were sampled by personnel from North Carolina State University.

marshes is made available to juvenile marine transients in open waters of the estuary, and probably in the coastal zone (Turner et al., 1979), through the movement of organisms (Haines, 1979; Deegan, 1983), phytoplankton and benthic microalgal production (Haines, 1979; Sullivan and Montcrieff, 1990), trophic relays (Kneib, 1997), pulsed events (Odum, 2000), export of dissolved organic matter

(Eldridge and Cifuentes, 2000), or microbial processing (Newell and Porter, 2000), or by some as yet undefined route(s).

In their review of the salt marsh "paradigm", Childers et al. (2000) incorporated the concept of donor control in their description of interactions among estuarine habitats that support fisheries production. Their model framework posited integrated subsystems
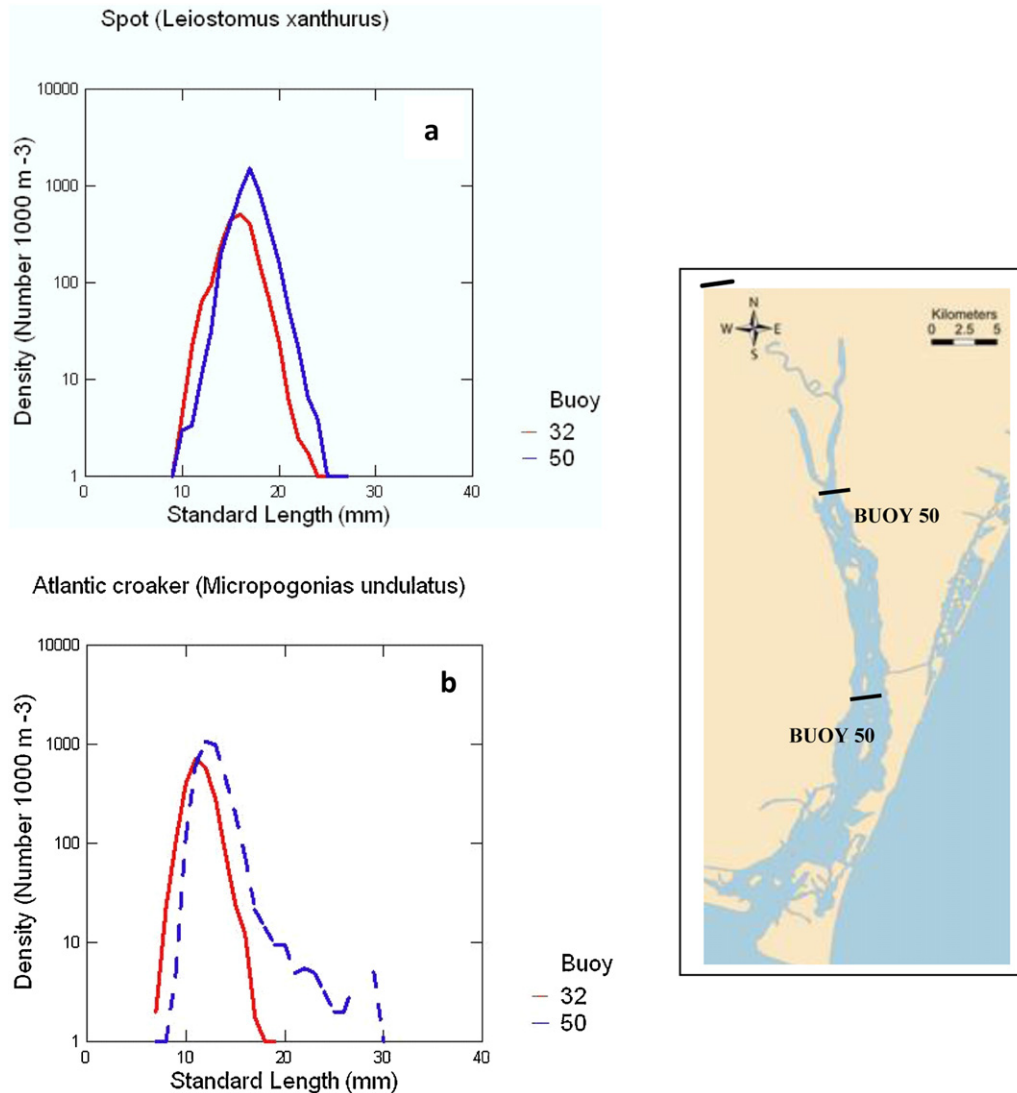
Fig. 10. (a and b) Length frequency distributions for spot (*Leiostomus xanthurus*) and Atlantic croaker (*Micropogonias undulatus*) at two buoy locations, 32 and 50 (near the turbidity maximum) in the Cape Fear River estuary. Data were combined for three collecting dates, 14–15 March, 5–6 April, and 11–12 April 1978.

linked by an overlying water column that mediates functional processes across subsystem boundaries thus extending the scale of the salt marsh paradigm to encompass a greater proportion of the estuarine landscape. Nutrient and organic matter flux associated with the movements of animals, especially juvenile marine transients, were also recognized as important vectors transcending system boundaries (see also Haines, 1979). The question of whether specific habitats confer disproportionate survival advantage to young marine transients is still rigorously debated (Grecay and Targett, 1996; Beck et al., 2001). In our view, trophic subsidies in donor-controlled systems may confer survival advantages on young nekton. In fact, trophic interactions may have important bearing on restoration outcomes yet have failed to take hold in many applied management endeavors such as fisheries and wildlife management (Vander Zanden et al., 2006).

From a restoration perspective then, recognition of landscape contexts, scale, cross-habitat linkages and energy flux across habitat boundaries represent an important component of food-web ecology with potential implications for ecological restoration. The need for this type of systems view was exemplified in our demonstration of export of secondary production from Delaware Bay

salt marshes to offshore habitats via juvenile weakfish (Litvin and Weinstein, 2004). Our results suggested that organic matter entering the near-shore coastal food web had its origins in salt-marsh macrophytes and benthic microalgae (see also Turner et al., 1979; Weinstein, 1981; Deegan, 1983).

While it is clear that the most appropriate interpretation of the stable-isotope data requires an understanding of the seasonal movements of larger and more mobile weakfish, ultimately Delaware Bay salt marshes function as important sources of organic matter driving a significant portion of the production of juvenile weakfish both in the marsh and in the estuary as a whole (Litvin and Weinstein, 2003). Furthermore, juvenile weakfish emigrating from the estuary in fall leave with a significant proportion of their biomass derived from salt-marsh organic-matter sources. Though our results do not quantify the various pathways that make nutrients available to consumers, including young weakfish, they support the notion that marine transients act as conduits of organic-matter export across habitat boundaries within the estuary and coastal zone.

Because spatial heterogeneity is ubiquitous in all ecological systems, and underlies the importance of pattern-process
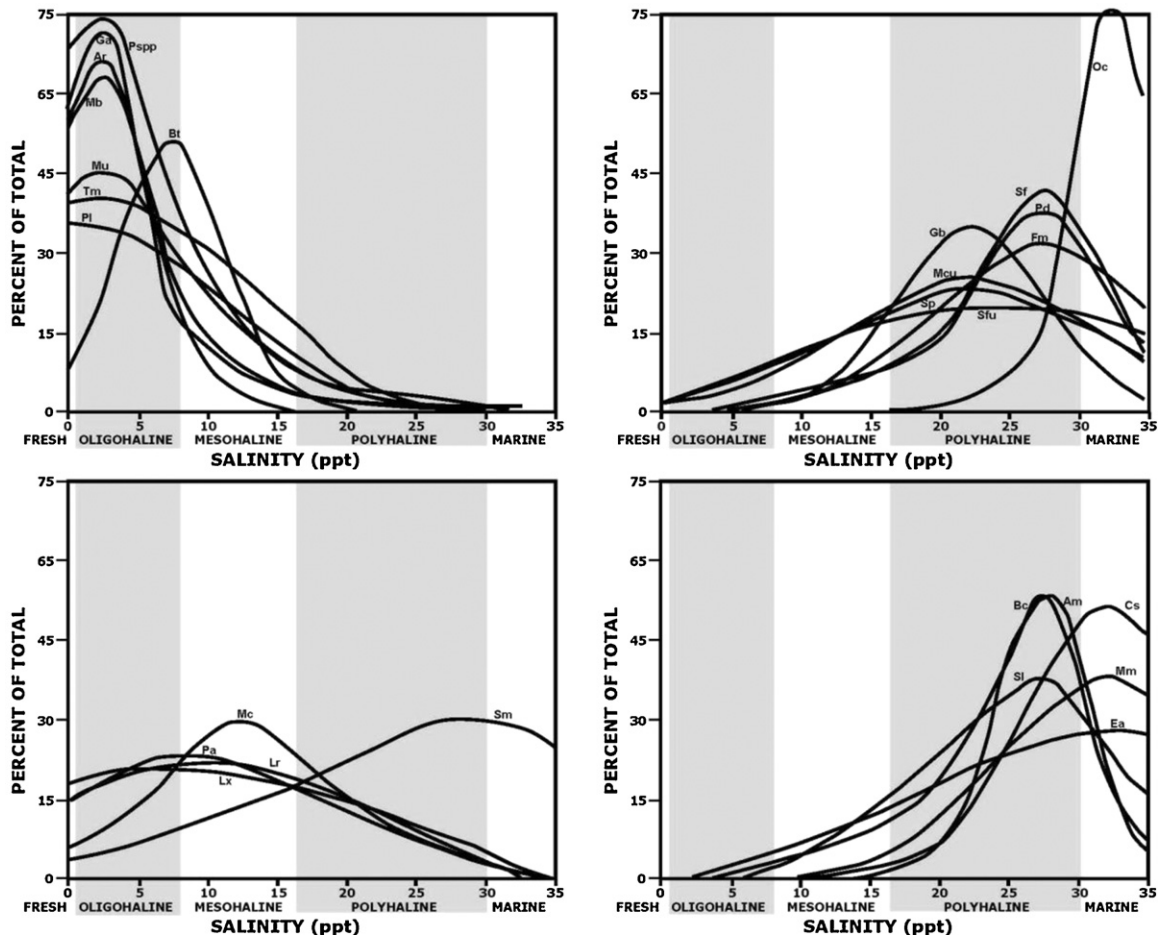
Fig. 11. Direct ordination of species distributions along the estuarine salinity gradient, Cape Fear River estuary, North Carolina, USA. Densities calculated for 5‰ salinity increments and smooth curves fitted to the data. The four panels correspond to species grouping depicted by binary discriminant analysis (see Weinstein et al., 1980b for details). Am = *Anchoa mitchilli*; Ar = *Anguilla rostrata*; Bc = *Bairdiella chrysoura*; Bt = *Brevoortia tyrannus*; Cs = *Callinectes similis*; Ea = *Eucinostomus argenteus*; Fm = *Fundulus majalis*; Ga = *Gambusia affinis*; Gb = *Gobiosoma bosc*; Lr = *Lagodon rhomboides*; Lx = *Leiostomus xanthurus*; Mb = *Menidia beryllina*; Mc = *Mugil cephalus*; Mcu = *M. curema*; Mm = *Menidia menidia*; Mu = *Micropogonias undulatus*; Oc = *Orthopristis chrysoptera*; Pa = *Peneus aztecus*; Pd = *P. duorarum*; Pl = *Paralichthys lethostigma*; Pspp = *Paralichthys* spp. (mostly *P. dentatus*); Sf = *Synodus foetens*; Sfu = *Syngnathus fuscus*; Sl = *S. louisianae*; Sm = *Strongylura marina*; Sp = *Symphurus plagiusa*; Tm = *Trinectes maculatus*.

relationships and scale, the estuarine complex may be viewed as a nested hierarchy in which smaller units (e.g., species populations) or habitat 'patches' (e.g., marsh ponds; tidal creeks and their constituent species assemblages; Figs. 1 and 5) form larger spatial units (e.g., marshes and larger estuarine regions defined), for example, by the salinity gradient (Fig. 11) or perhaps the turbidity maximum (Wu and Loucks, 1995; Wu and David, 2002; Fig. 10a and b herein). Restoration scientists and practitioners would do well to adapt this perspective, because as Forman (1990) notes, "for any landscape, or major portion of a landscape, there exists an optimal spatial configuration of ecosystem land uses to maximize ecological integrity, achievement of human aspirations, or sustainability of an environment". Among the metrics of importance here are the number of patch types and their proportions, patch density, edge density, patch size, patch or landscape shape indices, connectivity indices, and fragmentation indices. Additionally, landscape ecology addresses spatial and temporal interactions and exchanges across heterogeneous landscapes, influences of spatial heterogeneity on biotic and abiotic processes, and management of spatial heterogeneity. Thus, each restoration effort should consciously place the site of interest into its role in the overall system. Restoration of any site should not be undertaken in a vacuum, rather its connectivity to the estuarine complex in its entirety should be considered and ultimately prioritized.

4.1.2. A conceptual model for marsh-estuary-coastal linkages and nekton production

A simple 'stocks and flows' schematic is presented in Fig. 12 that summarizes the linkages between tidal salt marshes and the open estuary-coastal zone. The conceptual model depicts the components of the estuarine complex that supports secondary production of a Type III species (Litvin and Weinstein, 2003) such as juvenile weakfish (*C. regalis*) or spot (*L. xanthurus*). Recruitment to the adult stage follows a spatial-temporal sequence that includes utilization of the intertidal salt marsh plain on the rising tide, subtidal creeks, adjacent open waters of the estuary and ultimately the coastal zone. Previously, we have attempted to characterize the metrics of structure that support secondary production and that should be included in the design of any restoration effort (Weinstein et al., 2001), among them:

- Tidal creek drainages characterized by fourth- or fifth-order stream systems, high drainage density, bifurcation ratios, sinuosity and stream length;
- subtidal refugia for nekton in the highest order streams;
- a hydroperiod characterized by sufficient spatial/temporal inundation (about 4.5 h for the entire low marsh) and intertidal periods sufficient to aerate surficial sediments on the marsh surface and stream bank locations;
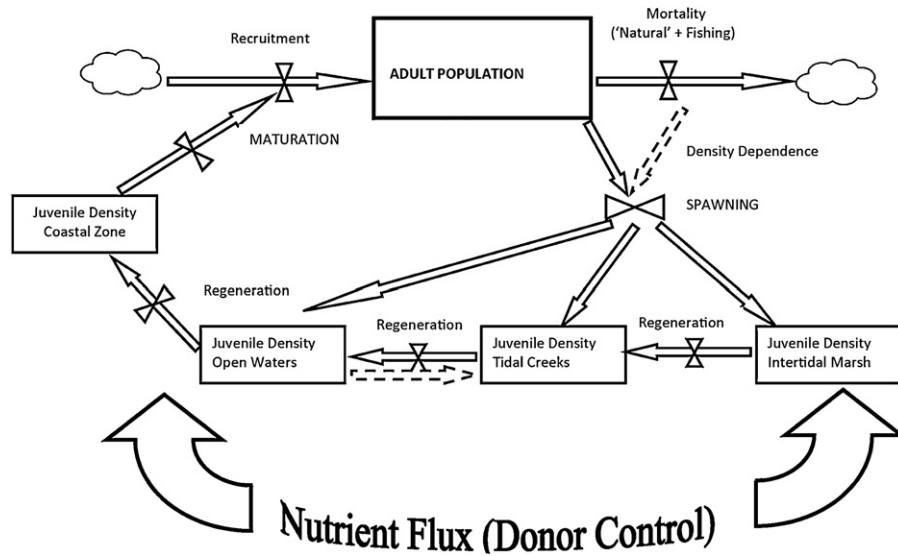
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Fig. 12. Stocks and flows conceptualization of linkages among habitats, species specific life stages and population dynamics for estimating secondary production and recruitment success.

- marsh ponds comprising about 2% of the marsh surface;
- invasive *P. australis* coverage reduced to less than about 10% of the marsh plain; or in older stands, minimally the monoculture broken up to smaller patches, with total coverage less than 20% of the marsh surface;
- natural stream bank slopes; and
- ratios of vegetation: open water of four to one.

But what is different here is our consideration of the site to be restored in terms of how it fits into the overall landscape and its subsequent ecological effects. These considerations include the mechanisms of organismal flows between compartments and the "mass-balance" of energy and materials in the landscape mosaic, as well as knowledge of the life history strategies of individual species and how they interact with landscape structure, spatial scaling (that addresses the translation of information across heterogeneous landscapes), and optimization of landscape patterns for self-organization and sustaining of the restored site. Key elements of the restoration approach and design would, therefore, consider these mechanisms in a landscape context. Size is a principal consideration, both in the context of the sites' support and contribution to secondary production of target species and how population, community and ecosystem processes change with area (Maurer, 2006), especially for restored sites that are small relative to the processes that influence them (Radeloff et al., 2000). Beyond size, processes within a marsh site will be partially governed by the spatial context it occupies within the salinity, topographic, substrate and thermal regimes that define the estuarine mosaic, proximity to open waters, and, in the upper estuary, proximity to the turbidity maximum. These two considerations, size and location within the estuarine landscape, must be considered in terms of the connectivity among sites, i.e., fluxes and flows of waters, materials and organisms across site boundaries. For the species assemblages anticipated to use the site and its various components (marsh plain, tidal creeks, etc.) the age specific distributions of target organisms as well as species specific growth, mortality and production rates must be considered, both in a restored system and the adjacent references systems to which it is connected (Weinstein, 1983; Weinstein et al., 1984a; Weinstein and Turner, 2012).

Consideration of all these mechanisms in a landscape context requires application of system-wide approaches (Fig. 12) that can forecast and monitor the flow of energy (and/or biomass) among habitats and support the development of restoration process beyond trial and error applications (Falk et al., 2006; Weinstein and Turner, 2012).

As Wu (2012) notes, "the science of landscape [systems] ecology focuses on the theoretical basis for understanding the formation, dynamics and effects of spatial heterogeneity, whereas the [practice] of landscape ecology reflects the humanistic and holistic perspectives necessary for integrating ecology, design and planning, socio-economics, and management practices". As noted throughout this paper, these tenets hold equally well for tidal salt marsh restoration. Moreover, "landscapes are arguably the most meaningful [element] in place-based research and provide common ground for ecologists, geographers, planners and designers, and policy makers to work together to shape and improve the society-nature relationship". Continued progress toward understanding the ecological complexity and interrelationships that define and connect the habitats we seek to restore represents a promising future for the integration of restoration ecology and ecological restoration in a systems framework.

5. Post-script

On the basis of our findings and experience over nearly 40 years, we offer the following summary thoughts and suggestions for the future prospects of restoration ecology in a sustainable world (the theme of our Special Session at EcoSummit 2012). First, our work clearly demonstrates that marine transient species (Litvin and Weinstein, 2003; Weinstein et al., 2005) benefit from tidal salt marshes and their production without directly occupying these habitats. Salt marshes do not function in isolation when supporting estuarine secondary production, but rather are integrated components of larger systems (Childers et al., 2000; Weinstein et al., 2005). Restoration planners should, therefore, adopt a systems view in the context of the habitat mosaic and the exchange of materials and organisms between adjacent habitats (e.g., salt marshes and the open waters of the estuary). As stated earlier, failure to account for connectivity among system components may have unintended consequences for estuarine habitat restoration (e.g., neglecting the contributions of marsh-derived trophic subsidies to productivity in

the open estuary may lead to reductions in the recruitment success of numerous marine transients).

Recognition of landscape contexts, scale, cross-habitat linkages and energy flux across habitat boundaries also represent an important component of restoration ecology with profound implications for restoration designs. Along with the notion of donor control that links tidal salt marshes to the estuarine habitat mosaic as a whole (Fig. 6), spatial–temporal partitioning observed in recruitment, habitat selection, and ontogenetic movements of various taxa should be included in the design of wetland restoration efforts. Although the 'KISS' principle has been (by necessity?) applied to most permit-related restoration efforts—e.g., 85% survival of planted vegetation over three years—criteria such as these are demonstrably inadequate to assess the return of functions and processes to the restored sites. If we are to encourage 'bottom up' management of our fisheries, i.e., assessing the value of coastal habitats in recruitment success; then the science of wetland restoration must improve substantially. It is gratifying to see that the National Oceanic and Atmospheric Administration (NOAA) has appeared to adopt this philosophy with the recent emergence of its "Habitat Blueprint". In addressing the question, *Why do we need the Habitat Blueprint now?*, NOAA comments:

> *Protecting our natural infrastructure—our global life support systems—is vital to protecting our communities and their economies as well as fisheries and recreational opportunities along our coasts. With continued widespread loss and deterioration of coastal and marine habitats, we are in danger of losing this infrastructure. Congress has charged the National Oceanic and Atmospheric Administration (NOAA) with protecting habitat for fish, threatened and endangered species, marine mammals, and other natural resources within the coastal zone.*
>
> *Now that we are turning the corner on ending overfishing, we need to increase the sustainability and productivity of our fisheries by focusing on the habitat that fish need to spawn and grow, as well as protecting the coastal resources on which our communities depend. Recognizing the need for more concerted efforts to protect and restore habitat, we are developing the NOAA Habitat Blueprint to guide our future actions.*

Decidedly anthropocentric, but likely acceptable insofar as Congress pays the bill! NOAA also needs to put its money where its mouth is and invest in the necessary science to advance its worthy cause. A plethora of the field's best scientists have raised a clarion call for better science to inform policy. We have touched upon this in our recent book (Weinstein and Turner, 2012) and a companion paper that has been submitted along with the present one for two special issues emanating from the 2012 EcoSummit in Columbus Ohio.

Finally, Holling and Meffe (1996) put it succinctly when they commented, "it is an open question whether ecosystem management will become a passing fad, an expansion of rigid bureaucratic procedures, or a sustaining foundation for learning to deal with interactions between people, nature, and economic activities." We certainly hope that it will be the latter, and that decision-makers let science inform their best policies!

Acknowledgements

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