State of the Estuary Report 2015

Summary

WILDLIFE – Benthic Invertebrates

Prepared by Elizabeth Wells
California Department of Water Resources

1. Brief description of indicator and benchmark

Table 1.1

Attribute	Indicator	Benchmarks
Benthic invertebrates	**1.** Diversity: number of native species **2.** Community composition: percent of all species that are native **3.** Community composition: percent of all individuals that are native	- Benchmark for native diversity is 1981-86. Good ≥ 1981-86 average, “Poor” ≤ historical average -1 standard deviation. - Benchmark for community composition (both by species and individuals) for “Good” ≥ 75% native, “Poor” ≤ 50% native.

2. Indicator status and trend measurements

Table 1.2

Indicator	Status	Trend	Details
1. Benthic invertebrate diversity: number of native species	Good	No change	All sites had “Good” native species diversity and were not significantly different from the historical period.
2. Benthic invertebrate community composition: native/nonnative species	Mixed	No change or deteriorating	The Delta site (D28A) was “Good,” the confluence site (D4) was “Fair” and the Suisun Bay site (D7) was “Poor”. D7 has significantly decreased in proportion of native species since the historical period.
3. Benthic invertebrate community composition: native/nonnative individuals	Fair or poor	No change or improving	The Delta site (D28A) was “Fair”, with a significant increase since historical times. The confluence site (D4) was “Fair” and the Suisun Bay site (D7) was “Poor”; neither had a significant trend in the proportion of native individuals

3. Brief write-up of scientific interpretation

The benthic invertebrate indicators give a summary of the status and trends of the community composition and native species diversity of the benthic (i.e. bottom-dwelling) community of the upper part of the San Francisco Estuary. The data used to construct these indicators is EMP benthic monitoring data from the three longest-sampled sites (D28A in the Delta, D4 at the confluence, and D7 in Suisun Bay) from 1981-2013. The three sites were analyzed independently because of the large differences in benthic communities between regions (Peterson and Vayssieres 2010, Thompson 2013). The data analyzed for the indicators comes from benthic grab samples, which have been collected, identified to species, and counted in the same way for the whole period of the monitoring program.

Benthic invertebrate indicators are important because the benthic community is a key part of estuary foodweb dynamics and nutrient cycling, and because benthic species are a classic bioindicator of estuary health (Gibson *et al.* 2000). The filter and deposit feeders of the San

Francisco Estuary and Sacramento-San Joaquin Delta have a large effect on how phytoplankton either continues into the fish food supply, or is diverted into the benthic community, with potentially large community effects (Alpine and Cloern 1992; Jassby 2008; Kimmerer and Thompson 2014). Benthic invertebrates are more localized indicators of estuary health than plankton or fish, and are sufficiently sensitive and have quick enough life cycles that changes in benthic community patterns can indicate large recent changes in nutrient loading, toxic substances, or sedimentation patterns (Gibson *et al.* 2000).

We chose our three indicators because they are unambiguous indicators of environmental health. Loss of native diversity has been associated with ecosystems that are less productive, have less ecological function and provide fewer ecological services, and are less resilient in the face of stresses (Worm *et al.* 2006, Cardinale *et al.* 2012). Similarly, ecosystems that have higher proportions of non-native species and individuals are characterized by lower environmental health and services than more intact ecosystems, and an increase in non-native species may lead to lower native biodiversity (Pimentel *et al.* 2005, Butchart *et al.* 2010, but see Gurevich and Padilla 2004).

The benchmark for native diversity and community composition was based on the historical period of 1981-86, chosen because 1981 was the earliest year-round monitoring at all sites, and the 1986-87 invasion of the Asian overbite clam (*Potamocorbula amurensis*), along with several other non-native species at roughly the same time, marked a drastic community shift at D4 and D7. Current (2009-2013) native diversity that was equal to or higher than the historical average was counted as “Good”, and the upper boundary for “Poor” native diversity was set at one standard deviation below the historical average, with “Fair” all values between these two. For community composition, the upper boundary of the “Poor” status was set at 50% native for both species and individuals (following the example of the 2011 State of the Bay Fish indicators), and the lower boundary of “Good” was set at or above 75% native in order to give equally sized intervals to “Good” and “Fair”. Trends for all three indicators were determined by determining whether the current status differed significantly from the historical benchmarks.

The status and trends for the various benthic indicators are variable but give a generally worrying overall picture. While native diversity has remained good, and has remained steady compared with 1981-86 historical levels (Figure 1), a large proportion of the community’s species and individuals are now non-native species at some sites (Figure 2). This is especially true at site D7 in Suisun Bay, a major site of *Potamocorbula amurensis* invasion, and where over the last five years native species were 50% of the species diversity but native individuals were only 5% of the total count. The current community composition was considerably better at D4 in the confluence (74% of species and 74% of individuals were native) and at D28A in the Delta (88% of species and 67 % of individuals were native.

The patterns we see in the benthic invertebrate indicators are important because they are a clear indication that the estuary and Delta are not in a pristine state, and are extremely unlikely to return to anything like a pristine state. The San Francisco Estuary is one of the most invaded in the world (Cohen and Carlton 1998, Ruiz *et al.* 2011), and with the addition of many non-native species we can expect changes to ecological services and functions such as food web dynamics that support valued fish, nutrient cycling, and water filtration that removes sediment and contaminants. We do not know exactly how the current benthic community functions differently from the historical one: many of the non-native species were introduced long before regular monitoring. While it is heartening that there has been no large net loss of native diversity at the species level, management of species such as salmonids and smelt should take into account the potential changes in benthic-pelagic food web interaction compared with historical conditions, as assumptions of similar function in the current and historical benthic community may be deeply flawed (Sommer *et al.* 2007).

Figure 1. Indicator 1: Native diversity

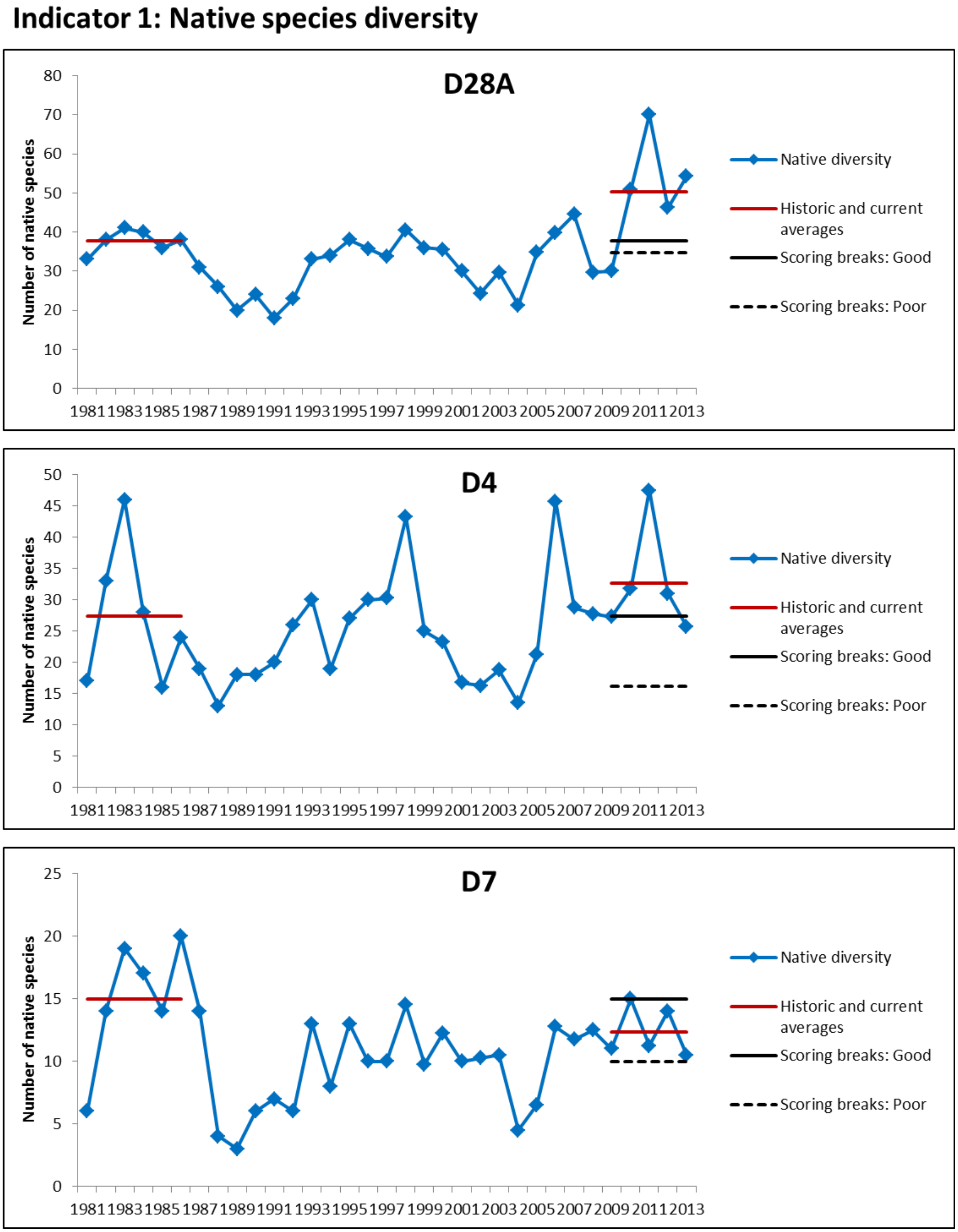
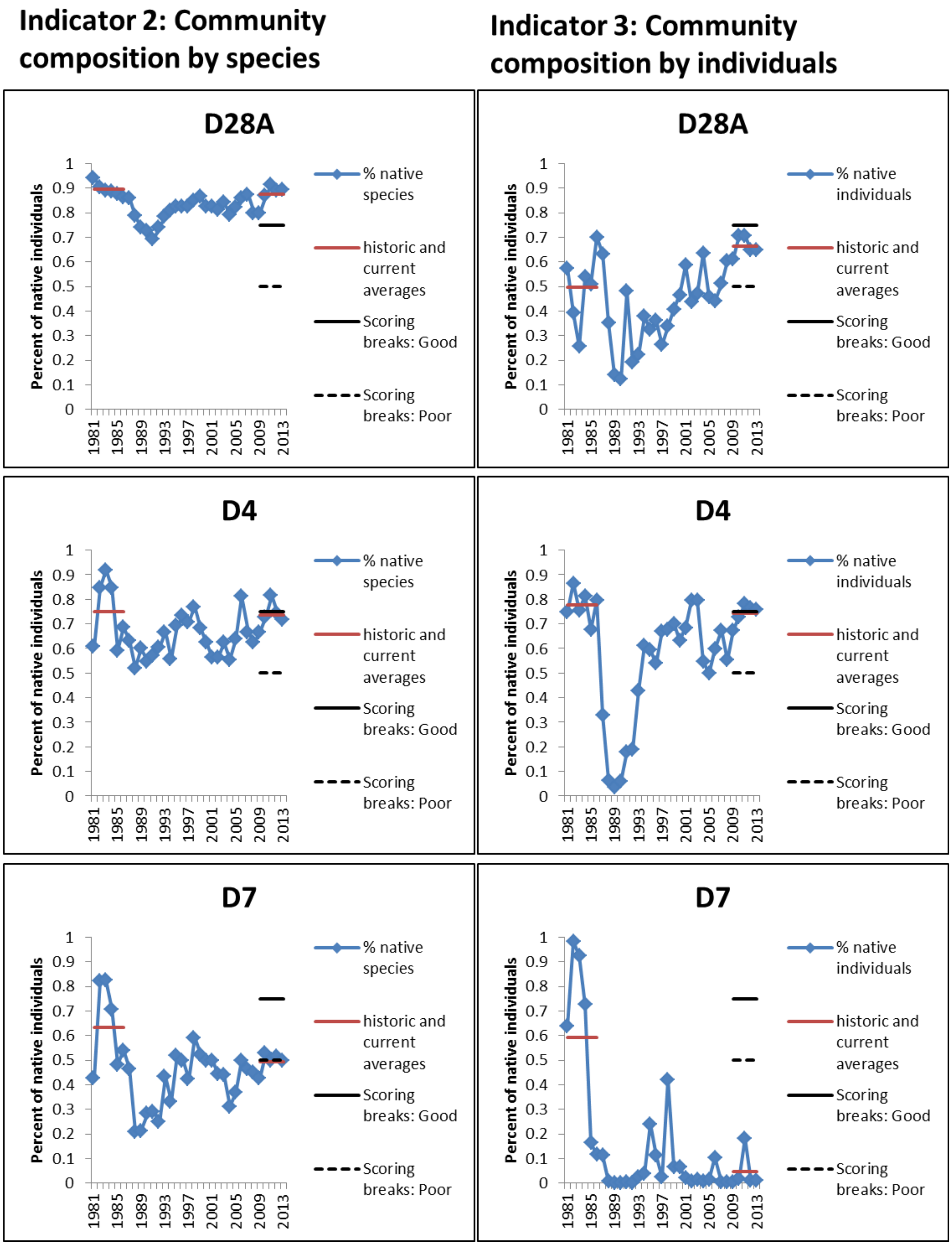


Figure 2. Indicators 2 and 3: Community composition by species and by individuals



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Technical Appendix

WILDLIFE – Benthic Invertebrates

Prepared by Elizabeth Wells
California Department of Water Resources

Benthic Invertebrates Technical Appendix

I. Background and Rationale

Benthic (bottom-dwelling) invertebrate indicators are an important part of assessing estuary health because the benthic community is a key part of estuary foodweb dynamics and nutrient cycling, and because benthic species are classic bioindicators (Gibson *et al*. 2000, Holt and Miller 2010). The filter and deposit feeders of the San Francisco Estuary and Sacramento-San Joaquin Delta have a large effect on how phytoplankton either continues into the fish food supply, or is diverted into the benthic community, with potentially large community effects (Alpine and Cloern 1992; Jassby 2008; Kimmerer and Thompson 2014). San Francisco Bay and the Delta comprise one of the most invaded estuaries in the world (Cohen and Carlton 1998, Ruiz *et al*. 2011) as well as having experienced major changes and degradation in the forms of altered water flow, channelization and hardening, pollution, agriculture, and development. Benthic invertebrates are more localized indicators of estuary health than plankton or fish, and are sufficiently sensitive and have short enough life cycles that changes in benthic community patterns can indicate large recent changes in nutrient loading, toxic substances, or sedimentation patterns (Gibson *et al*. 2000, Gomez Gesteira and Dauvin 2000).

The benthic invertebrate indicators give a summary of the status and trends of the native species diversity and community composition of the benthic community in the Sacramento – San Joaquin Delta and the upper part of the San Francisco Estuary. One indicator measures the native species diversity, or “how many species?” are found in the estuary. Two indicators assess the community composition, or “what kinds of species?”, comparing the number of native vs. non-native species and individuals.

Because the San Francisco Estuary and Sacramento – San Joaquin Delta covers conditions from marine to completely fresh water, there are distinct groupings of invertebrate communities along the salinity gradient sites (Peterson and Vayssieres 2010, Thompson 2013). These completely distinct communities displayed different patterns and cannot be compared directly, so all indicators were analyzed separately for each of three long-term monitoring sites: D28A (on Old River in the south Delta), D4 (at the confluence of the Sacramento and San Joaquin Rivers), and D7 (in Suisun Bay).

III. Data Source

The data used to construct these indicators is EMP benthic monitoring data from 1981-2013, which was derived from analysis of benthic grab samples. A standard-sized PONAR grab sampler (152mm x 152mm, or 6 inches x 6 inches) was used to take 3 replicate grabs at each site (1981-1995), which was increased to 4 replicate grabs at each site in later years (1996-present). The samples were sieved over an 0.5mm sieve in the field, preserved in 10% formalin and transferred to 70% ethanol, and were then identified to species and enumerated by Hydrozoology. For further details about the sampling protocols, please see the California Department of Water Resources page on benthic sampling methods: http://www.water.ca.gov/bdma/meta/benthic.cfm

The stations used are the three longest continuously sampled sites in the EMP benthic monitoring program. While seven other sites are currently monitored, and several others have been monitored historically, including them in this analysis proved difficult statistically due to the varying periods of study and conclusions from the analysis could not be interpreted unambiguously. The sites used for this analysis are listed in Table 1 and are placed on a map in Figure 1.

III. Benchmarks

The benchmarks for all three indicators were based on a historical period of 1981-86. While monitoring began in 1975 at some sites, 1981 was the earliest year-round monitoring at all sites, and the 1986-87 invasion of the Asian overbite clam (*Potamocorbula amurensis*), along with several other non-native species at roughly the same time, marked a drastic community shift at D4 and D7.

More details about indicator calculation and analysis can be found below in discussion of the individual indicators' Methods sections.

IV. Peer Review

Peer review for the benthic invertebrate indicators was performed in several different venues. The first line of consultation and revision was fellow State of the Estuary contributors April Hennessey and Hildie Spautz (both from the California Department of Fish and Wildlife), as well as Jon Rosenfield and Alison Stover-Weber (both from The Bay Institute). Drafts of the indicator ideas, calculations, and results were presented at State of the Estuary meetings as well as at several California Estuary Monitoring Workgroup meetings, and were discussed in meetings of the the Living Resources section of the California Estuary Monitoring Workgroup. Further discussion on the indicator benchmarks and scoring was conducted with Letitia Grenier and Amy Richey (both of the San Francisco Estuary Institute), as well as with April Hennessey and Hildie Spautz.

In addition, Karen Gerhts (Department of Water Resources) and Jan Thompson (USGS), who have both worked with the EMP benthic data and familiar with the dataset's scope and limitations, were consulted about the indicators' calculation and interpretation. They reviewed drafts of the summary and technical appendix, which were amended accorded to their recommendations.

V. Indicator Rationales, Methods, and Results

A. Indicator 1: Native Diversity

1. Rationale

Diversity is one of the key indicators of a community's health, and tends to be highest in systems that have not experienced as much human alteration and degradation (Butchart *et al*. 2010, Cardinale *et al*. 2012). Native diversity in particular is an important component of measuring ecosystem health, since endemic or rare native species with narrow environmental tolerances and specific developmental or trophic requirements may be lost due to habitat degradation.

In the course of 40 years of monitoring at all of its current and historic sites, the EMP benthic program has identified approximately[1] 397 native species to date (although note that three known cryptogenic species were counted as "native" for this analysis). These species span a salinity gradient that extends from completely fresh water in the Delta to near-marine conditions in the summers of very dry water years in San Pablo Bay. This high benthic invertebrate diversity provides a responsive tool to measure diversity responses to ecosystem health over a relatively long period of record.

2. Methods and Calculations

The native diversity indicator was measured as simple species richness at each site in each year. We had to calculate native diversity differently for the years 1981-1995 (when we took three replicate benthic grabs) with the years 1996-2013 (when we took four replicate benthic grabs). We calculated native diversity for 1981-1995 as:

Equation 1

$$1981 - 1995 \text{ native diversity} = \#\text{ of native species identified in a calendar year}$$

For 1981 -1995, data from all three replicate benthic grabs was used, and the native diversity used for calculation of the indicator status and trend was the same as the total number of native species observed in those grabs.

However, for 1996 we used an effort-adjusted measurement of native diversity since an increased number of sampling events increases the total diversity count (assuming that all species were not completely detected by three replicate grabs). Since we had four replicate grabs

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[1] The exact number of species is constantly in flux by 5-10 species at any time, as unidentified specimens counted as separate species are determined by taxonomists either to be truly new species or to belong to previously identified species.

(identity numbers were randomly assigned), the calculation process was to repeatedly subsample with replacement:

1. Exclude all data from replicate grab #1 for all sampling events and calculate total native diversity for that site in that year. This diversity = A.
2. Exclude all data from replicate grab #2 for all sampling events and calculate total native diversity for that site in that year. This diversity = B.
3. Exclude all data from replicate grab #3 for all sampling events and calculate total native diversity for that site in that year. This diversity = C.
4. Exclude all data from replicate grab #4 for all sampling events and calculate total native diversity for that site in that year. This diversity = D.

Equation 2

$$1996 - 2013 \text{ native diversity} = \text{Average of } (\mathrm{A}, \mathrm{B}, \mathrm{C}, \mathrm{D})$$

This replicate-adjusted native diversity provided a metric of native diversity that did not inflate total diversity from the increased sampling effort of later years, and was comparable to the 1981-1995 native diversity.

It should also be noted that we took a conservative approach to native vs. non-native designation. Only species that had been specifically denoted as non-native in the database were counted as such, and cryptogenic species or those with uncertain status were counted as native. The findings of this indicator, and indeed all three benthic invertebrate indicators, may therefore be slightly more optimistic with regards to native species presence and abundance than if cryptogenic species were examined separately.

Including the cryptogenic species as natives was done for logistical reasons, because we wanted to count the cryptogenic species in some way, and creating their own category for either indicator or for Indicators 2 and 3 was not feasible. Two cryptogenic species (*Grandofoxus grandis*, an amphipod, and *Macoma* sp. A, a clam) were each seen only a handful of times, in low numbers, while the third (*Macoma petalum*, a clam seen in consistent numbers across the monitoring period in Suisun Bay) was likely a trans-Arctic invasion of Atlantic *Macoma balthica* in the Early Pliocene (Nikula *et al*. 2007). The majority of the “cryptogenic” individuals were therefore more similar to natives than non-natives, and were grouped accordingly.

To find the current status of native diversity, we found the average of the last five years (2009-2013) of native diversity at each site and compared it to the benchmark average diversity of the historic period (1981-86). Native diversity that was equal to or higher than the historical average was counted as “Good”, and the upper boundary for “Poor” native diversity was set at one standard deviation below the historical average, with “Fair” all values between these two (Table 2).

Trends in community composition by species were identified by performing a two-sided two-sample t-test comparing the years in the benchmark historic period to the years of the current period. A significant result ($p<0.05$) was counted as a significant trend in native diversity up or down from historic levels. We used this approach rather than a linear regresssion of diversity on year because diversity is not expected to behave in a linear manner and does not meet the assumptions of linear regressions. For example, decreases in biodiversity may dramatically decrease following a catastrophic disturbance, which would be better assessed with to a change-point or step analysis than with a linear regression. A t-test such as the one we used still captures the signal of change, while not assuming a linear rate of change. In addition, each year is not independent of other years, a requirement for linear regression's independent variable; a species' persistence in each year (and thus total biodiversity) is not independent of whether it was found at a site in the previous year.

Results

At all sites, the native diversity is currently at "Good", with no significant trends up or down (Figure 2). The current (2009-2013) native diversity average at D28A (Old River, in the south Delta) was 50.25 species, which was not statistically different from the 1981-86 average of 37.7 species (Figure 2). (Note that 50.25 species is the effort-adjusted species richness; current observed species diversity using all four replicate grabs was 54.2 species). The current native diversity average at D4 (confluence of Sacramento and San Joaquin Rivers) is 32.7 species (effort-adjusted; observed diversity was 36.2 species), which did not differ significantly from the 1981-86 average of 27.3 species. The current native diversity average at D7 (Suisun Bay) is 12.4 species (effort-adjusted; observed diversity was 14 species), which did not differ significantly from the 1981-86 average of 15 species.

The steady maintenance of native diversity at a level close to or slightly above historical levels is an encouraging sign of health in the benthic invertebrate community. Loss of biodiversity is often cited as a cause or correlation with decrease in environmental services and functions (Butchart *et al*. 2010, Cardinale *et al*. 2012). We can conclude that the benthic community has not responded to the stresses and disturbances of the last 30 years with a crisis of native biodiversity loss.

One reason for confidence in these results is that there have been no changes in identification methods, which have been performed in the same way by the same person the same since throughout the length of the monitoring effort. Nor has any real loss of biodiversity been disguised by changes in taxonomic classification, e.g. one original species now identified as two or more; very few of those taxonomic splits have happened with the species in this dataset (Wayne Fields of Hydrozoology, personal communication).

One caveat in interpreting these results is that even though over thirty years of monitoring is often considered to be a respectably long-term dataset, the start of the EMP benthic monitoring used for this analysis happened centuries after the beginning of human influence in the region There may have been much earlier losses to native biodiversity that we do not see in this analysis because of our shifted baseline of comparison. Indeed, considering the scale of alterations to water flow and sediment loading from agriculture, mining, and development that affected the Delta, it would be surprising if there were not early losses to the native diversity. We cannot estimate the size of any earlier decreases in native diversity, but this indicator at least reassures us that decreases are not currently ongoing.

B. Indicator 2: Community composition by species
Indicator 3: Community composition by individuals

1. Rationale

The relative abundances of native and non-native species and individuals are another key component of ecosystem health. Since non-native species may not have the same relationships with other species in the community as natives, the addition of non-native species (and in some cases, their replacement of native species) may affect food web dynamics and overall ecosystem function. While non-native species may increase the total diversity, they are associated with ecosystem disturbance and may actually increase environmental degradation (MacDougall and Turkington 2005, Didham *et al*. 2007), which indicate lower overall ecosystem health.

Community composition by species (Indicator 2) is similar to native diversity (Indicator 1), which both look at status and trends of native species numbers. The difference is that Indicator 2 explictly examines native species diversity in the context of all diversity in each year, which is important since a majority of the species found may not be native to the area, but should be considered when assessing how ecosystem function may have changed.

In addition to examining the relative proportions of native and non-native species, looking at proportions of native and and non-native individuals gives a more nuanced perspective of community composition than either alone. We present two indicators: community composition by species (Indicator 2) measures what proportion of total species diversity consists of native species, while community composition by individuals (Indicator 3) measures what proportion of all the individual organisms belong to native species. Each indicator is analyzed separately for each long-term monitoring site, since the three sites display very different patterns.

2. Methods and Calculations

Note that by “native” species, we are counting all species not designated as “introduced” as native, including cryptogenic species. For the reasoning behind this decision, please see “Methods and Calculations” for Indicator 1.

Community composition by species was calculated as the percentage of native species in the total annual species diversity in each region, for each year. The percentage of non-native species could of course be easily calculated as 100%-percentage of native species.

Equation 3

$$\text{Annual community composition by species} = \frac{\#\text{ native species}}{\#\text{ of all species}} \times 100$$

Community composition by individuals was calculated as the total number of native individuals as a proportion of all individuals collected, within each region for each year.

Equation 4

$$\text{Annual community composition by individuals} = \frac{\#\text{ native individuals}}{\#\text{ of all individuals}} \times 100$$

Current (2009-2013) community composition was found in the same way for both species and individuals. The upper boundary of the “Poor” status was set at 50% native for both species and individuals, since an ecosystem with under 50% native species or individuals is generally considered to be in poor ecological health (per 2011 State of the Bay Fish indicators). The lower boundary of “Good” was set at or above 75% native in order to give equally sized intervals to “Good” and “Fair” (Table 2).

Trends in community composition by species were identified by performing a two-sample t-test comparing the years in the benchmark historic period to the years of the current period. A significant result ($p<0.05$) was counted as a significant trend in native diversity up or down from historic levels.

3. Results

The current (2009-2013) community composition by species of site D28A (Old River, in the south Delta) has a status of “Good” with an average of 87.5% native species, with no significant trend from its historic (1981-86) average of 89.5% native species (Figure 3). The community composition by individuals at D28A was “Fair” with 66.5% native individuals, which was was a significant upward trend increase from its historic average of 49.6% native individuals. Most of the numerically dominant species at D28A have remained constant in identity while fluctuating in abundance through the monitoring record. The difference observed between the historic and current period appears to be due largely to a decrease in density in the non-native clam *Corbicula fluminea* from historic highs, and a recent sharp increase of the native amphipod *Americorophium spinicorne*.

At D4 (confluence of Sacramento and San Joaquin Rivers), the current community composition by species is 73.5% native, with a status of “Fair” and no significant difference from the historic community composition of 75% native species. D4 is currently composed of 74.1% native individuals, with a status of “Fair” and not different from its historic composition of 77.6% native individuals. While various species have fluctuated in abundance throughout the period of monitoring, the native amphipod *Americorophium spinicorne* and the native oligochaete worm *Varichaetodrilus angustipenis* have consistently made up much of the total abundance of the community at D4 through time, both in the historic and current time periods.

The current community composition by species at D7 (Suisun Bay) is just under the line for “Poor” at 49.5% native species, which is not significanlty lower than the historic mean of 63.5% native species. The community composition by individuals at D7 is well into the “Poor” category at 4.6% native individuals, a sharp downward trend from the historic average of 59.3% native individuals. The change to a high proportion of non-native individuals is due in large part to the 1986 arrival of the non-native clam *Potamocorbula amurensis* as well as the non-native amphipod *Corophium alienense*, whose rise in numbers at D7 can be dated to the late 1980s and which is especially dominant in dry water years. These two species are by far the most numerically dominant species in the estuary, while formerly dominant native species like the arthropod *Americorophium stimsoni* and the oligochaete worm *Limnodrilus hoffmeisteri* have both declined since the historic period. These dominant non-natives have added massively to the number of non-native individuals, and may have also replaced some of the native individuals through competition for space or other resources.

For many of the species in the benthic community, too little is known about their natural history (either observationally or experimentally) to compare the role of non-native species with the roles of native species.The community composition indicators are therefore not necessarily an indication of lower ecological health in all systems. However, in the Delta, the advent of non-natives, especially clams has been identified as a contributing factor in the Pelagic Organism Decline (Sommer *et al*. 2007), and the dramatic changes seen, particularly in the proportion of native and non-native individuals at site D7, are an effective indicator of major shifts in the community that has had effects on the Delta and Suisun Bay food webs.

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Table 1. Sites used for benthic invertebrate data source

Region	**Site**	**Latitude and longitude**	**Period of sampling**
Suisun Bay	D7	38.1171292 N, 122.0395539 W	1981-present
Delta	D28A	37.9701652N, 121.5741188 W	1981-present
Confluence	D4	38.0581151 N, 121.8193499 W	1981-present

Table 2. Benchmarks and scoring for benthic invertebrate indicators

Indicator	**Quantitative reference condition**	**Evaluation and Interpretation**
1. Native diversity	≥ historical period average	"Good"
	< historical period average and > historical period average – 1 standard deviation	"Fair"
	≤ historical period average – 1 standard deviation	"Poor"
2. Community composition (species)	≥ 75 % native species	"Good"
	<75% and >50% native species	"Fair"
	≤50% native species	"Poor"
3. Community composition (individuals)	≥ 75 % native individuals	"Good"
	<75% and >50% native individuals	"Fair"
	≤50% native individuals	"Poor"

Figure 1. Map of benthic monitoring sites used for State of the Estuary analysis

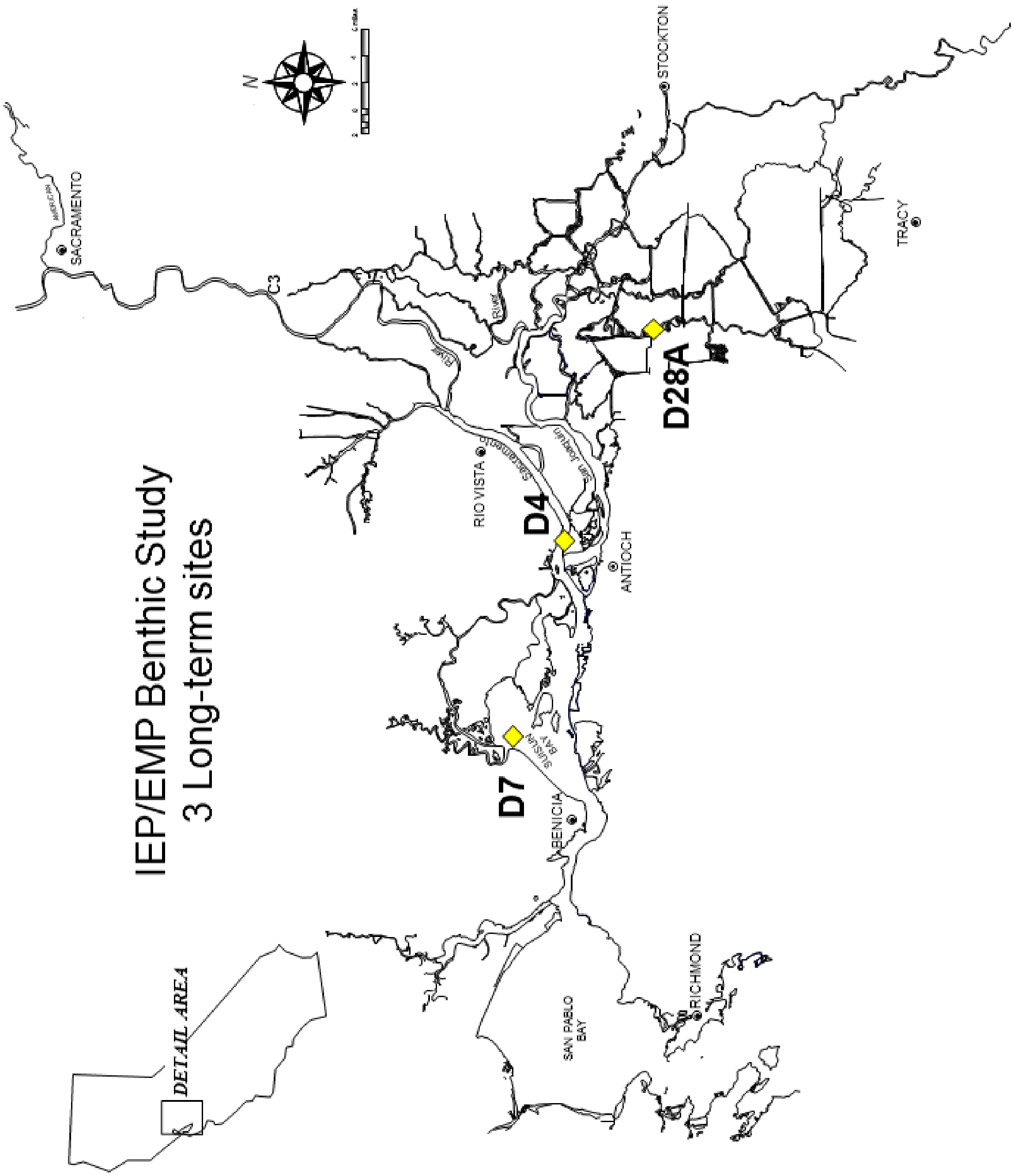


Figure 2. Indicator 1: Native species diversity.

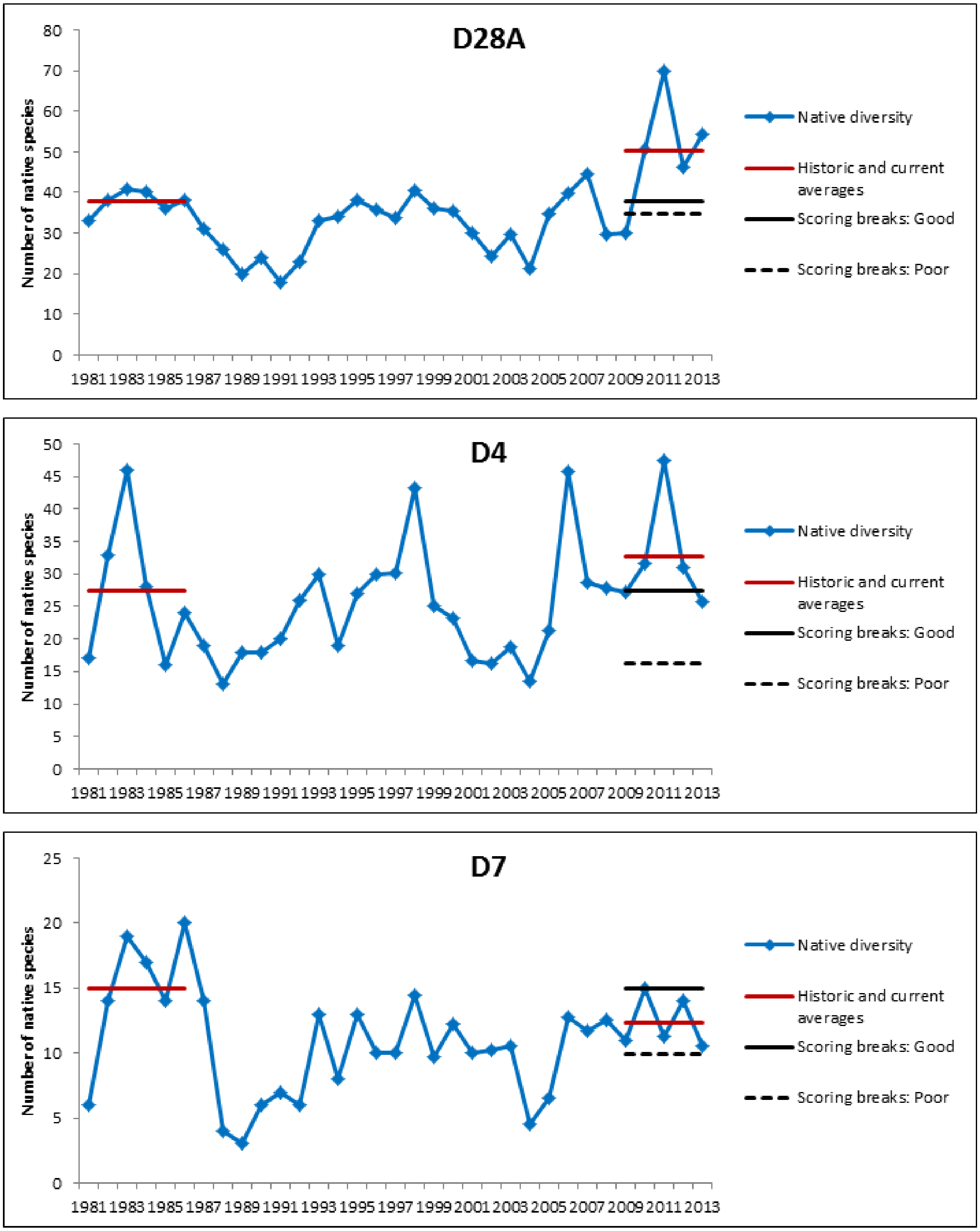
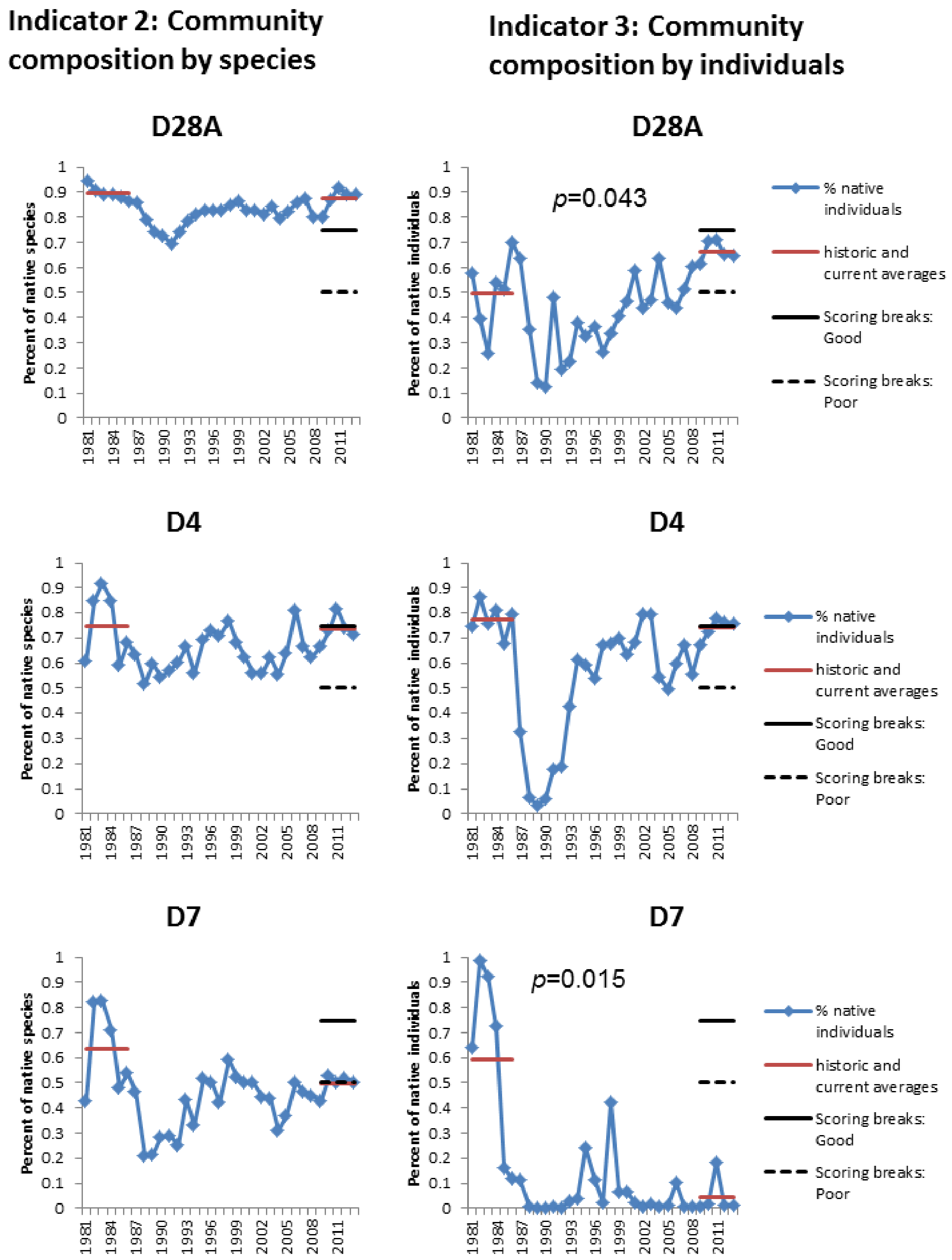


Figure 3. Indicators 2 and 3: Community composition by species, by region. Significant trends are marked with p-values.



Indicator 2: Community composition by species

Indicator 3: Community composition by individuals

D28A

D4

D7