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Table of Contents

List of Figures & Tables

Quantum & Classical Mechanics

Kevin Sabaj-Stahl David Sabaj-Stahl

of the Vegetative Complex Health Index

ECHO EFFICACY JOURNAL • VOLUME TWO • JUNE TWO THOUSAND AND FIFTEEN

Abstract: The ecosphere was modeled in a fashion consistent with quantum atomic models utilizing data from vegetative sinusiae of a degraded riparian ecosystem. Analyses revealed the variables community structure, alpha diversity and bio-invasion behaved in a manner whereby the individual state of each could not be fully understood absent their interactions with one another. A highly discrete unification between structure and diversity, modulated by the force of cohesion, was reminiscent of the property of quantum entanglement in atomic theory. The ecosystem was found to exist in all potential microhabitat states simultaneously, apparently a consequence of "orbital decay" and community decoupling induced by changes in hydrology and geomorphology, and overgrazing by livestock. These quantum relationships also revealed an understanding of vegetative structure enabling identification of quantum-dependent and -independent processes of bio-invasion. Vegetative structure was applied as an indirect means to assess geomorphic/ hydrologic attributes of the watershed. Rating systems were developed to assess community structure and bio-invasion. These rating systems allowed for prescription of microhabitat-targeted geomorphic/ hydrologic v. horticultural restoration strategies. The Vegetative Complex Health Index was derived from these ratings in conjunction with Fisher's alpha. The approaches yielded a quantitative, objective and sensitive composite index by which to assess the health and recovery of riparian ecosystems. These proofs of concepts provide additional support for The Theory of Quantum Microbiogeography.

Introduction

Quantum & Classical Mechanics of the Vegetative Complex Health Index expands upon the fundamental concepts presented in *The Theory of Quantum Microbiogeography: Mechanisms of the Priority Site Determination* (Sabaj-Stahl et al. 2013). Herein the ecosphere was modeled in a fashion not entirely unlike quantum mechanics of atomic theory. Previously it was elucidated that the quantized energy of cohesion occurring between community structure and biodiversity evidenced a quantum relationship at the level of the ecosphere (Sabaj-Stahl et al. 2013). The energies associated with microhabitats behaved in a manner somewhat consistent with existing quantum theory of atomic structure and served as the basis of developing conceptual models.

Quantum models of the ecosphere were constructed in a manner reflective of this duality. The lower Dolores watershed was impacted by construction of an impoundment in the city of Dolores, Colorado in 1986. Currently, 40 to 50% of surface water is diverted from the reservoir for irrigation of croplands. It appeared this disturbance facilitated degradation and decoupling of community structure to such an extent that it was reflected via this duality. Thus the quantum models facilitated a conceptualization of the environmental process via community structure and biodiversity. Further, community structure coupled with rates of bio-invasion and juxtaposed with biodiversity provided a solid Newtonian mechanistic framework for construction of an objective, quantitative and sensitive composite ecometric: the *Vegetative Complex Health Index* (VCHI).

Community structural patterns were also compared to invasive species structure and appeared to reveal discrete processes of bio-invasion. Those processes were assessed within the context of ecospheric disease processes and identified as such. Description of community- and species-based ecospheric disease sequelae enabled prescription of therapeutic remedies within the rubric of sustained, long term recovery. These nuanced approaches arose from the conceptual models and therefore provide additional proofs of concepts of The Theory of Quantum Microbiogeography. Taken together, elucidation of the Gehrt-Mueller Priority Site Determination (Sabaj-Stahl et al. 2013), novel ecospheric disease states, and the Vegetative Complex Health Index demonstrate the philosophical tenets of theory translate to sound applied and biologically relevant outcomes.

While the entire research system was developed using just one season's data from the lower Dolores watershed, the dataset in of itself was powerful concerning myriad analyses. The fidelity of the relationships observed across a spectrum of ecologic variables suggests the theoretical and applied discoveries shall apply to a variety of settings beyond the riparian vegetative communities of the arid southwestern United States. Currently the Edwin James Society is engaged in evaluating long term monitoring data from a diversity of systems and anticipates a logarithmic expansion of these bedrock principles. In general it would seem application of mechanistic approaches to ecologic data offer alternative routes of discovery perhaps not available via established and standardized methodology.

Methods

All methods utilized in the current manuscript are described herein. A full description of related methods concerning the 2010 lower Dolores project were published in: *The Theory of Quantum Microbiogeography: Mechanisms of the Priority Site Determination* (Sabaj-Stahl et al. 2013). A PDF of the manuscript is available at: http://www.academia.edu/6665360/ECHO_EFFICACY_JOURNAL_Volume_I_Issue_I.

Thirty-nine monitoring sites, described as microhabitats throughout the manuscript, were selected by the Edwin James Society Division of Research, in cooperation with the Dolores River Restoration Partnership, the Tamarisk Coalition, and the United States Bureau of Land Management. Sites were located within riparian corridors of the lower Dolores watershed, with specific locations mapped in Sabaj-Stahl et al. (2013). Sites were chosen with respect to salt cedar (i.e. *Tamarix chinensis*) density (low, medium and high stem density) and stand age (early-, mid- and old-growth). Microhabitat plots consisted of 80 x 50 meter quadrats (approximately one square acre) parallel the channel banks.

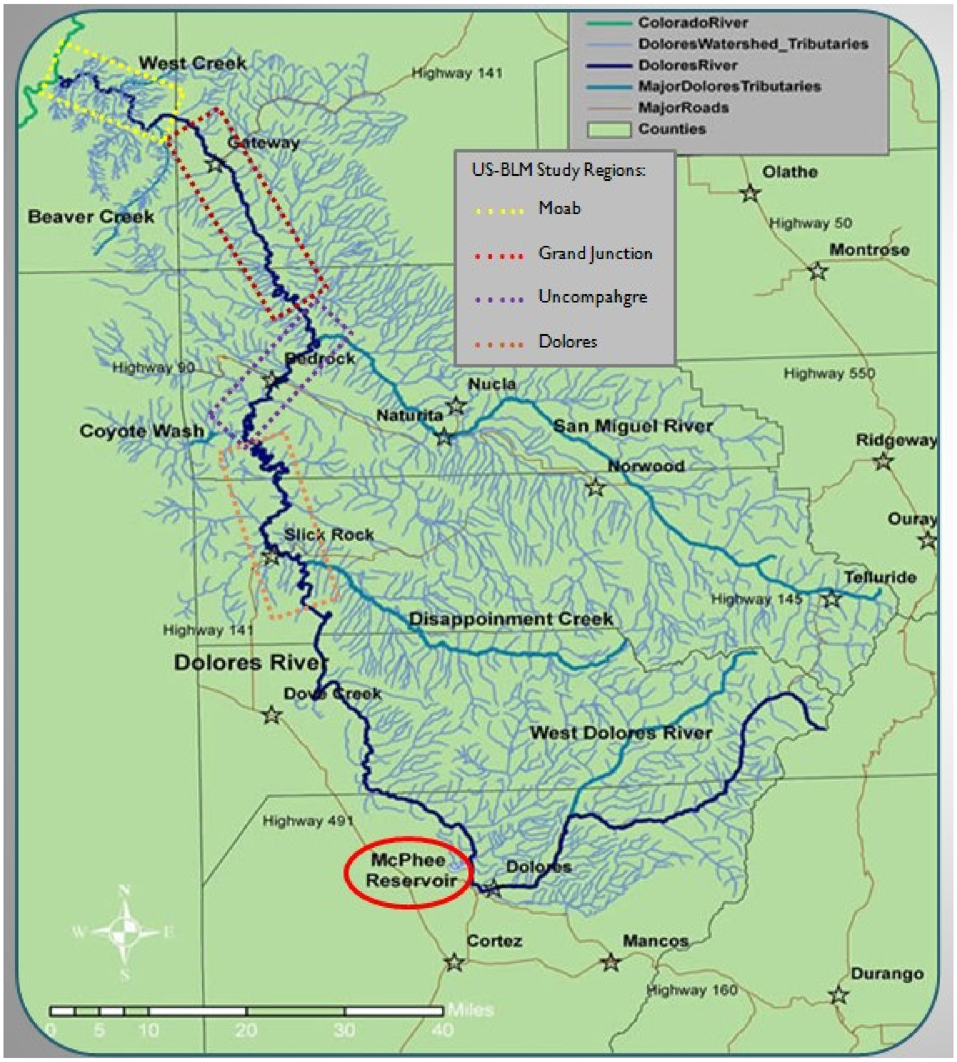


Figure 1: Sampling Locations

Thirty-nine, 80 x 50 M (approximately one square acre) study sites were established parallel to channel banks along the lower Dolores watershed. Ten sites each were located within the Moab, Grand Junction & Uncompahgre US-BLM Field Districts. Nine sites were located within the Dolores District.

50 M distances perpendicular to channel banks were sufficient to intersect historic primary & secondary floodplains.

Original map was prepared by the Dolores River Restoration Partnership (http://ocs.fortlewis.edu/drrp/).

During the first week of August 2010, five, 50-meter transects perpendicular to the channel bank were established in a stratified random design within each microhabitat by randomly placing each transect within 16-meter intervals. Vegetation was evaluated at ten centimeter intervals along each transect, resulting in 500 points per transect and 2500 points per site. Each species intersecting the point was recorded, however, multiple intersections at a given point by individual species were not (i.e., data indicated presence/ absence only). Plants bearing green foliage were counted. While salt cedar had been ubiquitously defoliated throughout the watershed in late June/ early July by the biocontrol beetle (*Diorhabda carinulata*) (Sabaj-Stahl et al. 2013), the vast majority of trees had resprouted green foliage by the time sampling was conducted and thus were included in percent cover estimates. Percent cover of individual species and community types (i.e., native, exotic, woody and herbaceous) were estimated for each transect by dividing the number of occurrences of that species or community type along each transect by the total number of points assessed.

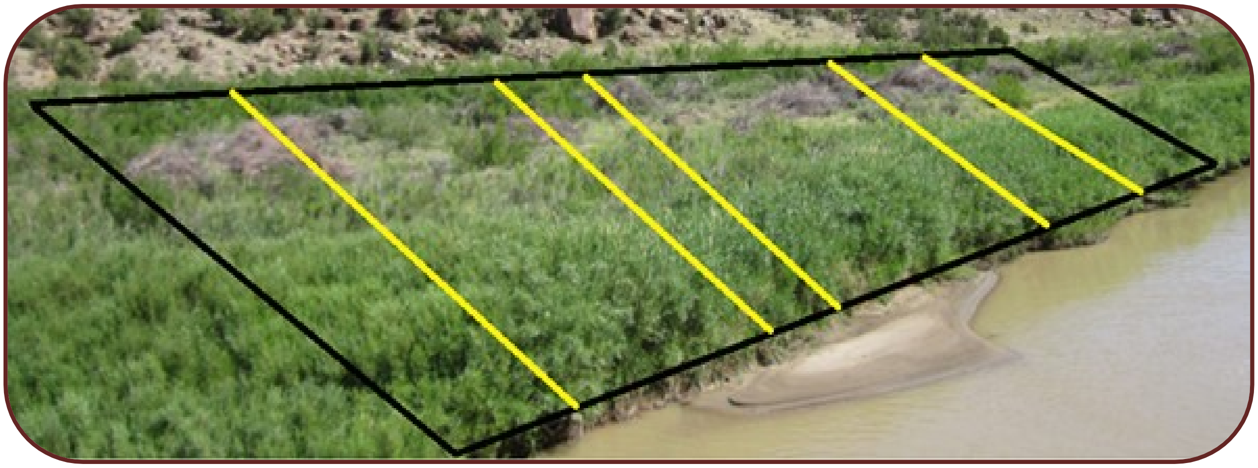


Figure 2: Transect Lay-out

Transects (yellow lines) were randomly placed within 16 M intervals of 80 x 50 M quadrats (black rectangle). Dimensions in figure are not exact, but approximate the generalized pattern. Original photo was produced by the Dolores River Restoration Partnership (http://ocs.fortlewis.edu/drrp/).

Absolute percent cover estimates of individual species were converted to proportions and then arcsine-square root transformed in Microsoft Excel®. Species estimates were combined to yield percent cover estimates of community types. Arcsine-square root transformed data exhibited improved homoscedasticity (Sabaj-Stahl et al. 2013) and were used in all subsequent analyses. The arcsine-square root transformation was performed for each absolute percent cover estimate of each species *per* transect, rather than at the level of the microhabitat. Then, mean absolute percent cover estimates were derived. The approach allowed for calculation of microhabitat-level standard deviations using transect-level transformed data. Standard deviations were required for determination of sampling effort, via confidence interval estimates, using the equation described below.

Occasionally but infrequently total cover proportions exceeded one when species overlap saturated transects. Proportions greater than 1.00 could not be arcsine-square root transformed, thus a proportion of one was applied, and represented vegetative saturation. Arcsine transformed data were used in conjunction with an equation for determination of sampling efficacy and delineation of vegetative sinusiae (Elzinga et al. 2001). The form of the equation applied was: $ESS = \frac{((Z\alpha)^2 * (s)^2)}{B * \bar{X}}$; where *ESS* = estimated sample size, *Zα* = standard normal deviate, *s* = standard deviation, *B* = desired confidence interval width, and $\bar{X}$ = sample mean.

A *Zα* of 1.64 (alpha = 0.10), a *B* of 0.25 for community types, and a *B* of 0.40 for species were selected. These confidence intervals were considered sufficient to approximate the lower detection range of percent cover change based upon a pilot study conducted the same year and in the same region, but on a different watershed. These confidence interval widths allowed for minimum detection of change in percent cover of 25% for communities and 40% for species, using point-in-time parameter estimates. Estimated sample sizes were adjusted to corrected sample sizes using the sample size correction table published by Elzinga et al. (2001) for point-in-time parameter estimates with a tolerance probability of 0.90. Elzinga et al. (2001) created the correction table applying the algorithm reported by Kupper and Hafner (1989).

Sabaj Efficacy Index (SEI) categorical outcomes for community types were derived using the equation given by Elzinga et al. (2001). The SEI was defined as the proportion of communities (native, exotic, woody and herbaceous) *per* microhabitat adherent to 25% confidence interval estimates. The index ranged from 0.00 (no efficacy) to 1.00 (maximal efficacy), with categories defined as follows: 0.00 = no stratification; 0.25 = one of four stratified communities; 0.50 = two of four stratified communities; 0.75 = three of four stratified communities; 1.00 = four of four stratified communities. The SEI was converted to integer ranks of zero (no stratification) through four (maximal stratification) for use in the Vegetative Complex Health Index (VCHI). Potential dependencies of community structure with diversity were tested using simple linear regression, logistic regression, and multiple regression/ principal component analysis (PCA) correlation (Sabaj-Stahl et al. 2013). Stratification criteria for species (0.40 confidence interval estimates) were applied to salt cedar and knapweed and regressed with community structure for identification of specific patterns of bio-invasion.

The Refined Quinn Cohesion Index (r-QCI) was derived from the PCA correlation matrices, using only those indices shown to a) have linear relationships with efficacy *per* linear regression and b) contribute significant information *per* the Akaike Relative Likelihood method (Sabaj-Stahl et al. 2013). Simpson's dominance, Berger-Parker's dominance, Buzas & Gibson's evenness, and Pielou's J were applied for derivation of the r-QCI. The index is an approximation of the cohesion between the variables used in the PCA correlations. The manner in which the index was applied viewed dominance as the compliment (i.e., mirror image) of evenness. Thus, the r-QCI estimated the force of cohesion occurring between evenness and efficacy/ community stratification.

The r-QCI was defined as the least squares regression of the refined Quinn Quotient (QQ) with Sabaj Efficacy/ community stratification. The index is an approximation of the quantized cohesion occurring between the variables used in the PCA.

The Quinn Quotient was derived as follows:

QQ = (∑(# of efficacy-diversity pairings with no gaps per efficacy group)/ ∑(# of indices assessed)) / (∑(gaps per efficacy-diversity pairing per efficacy group) / ∑(gaps for all efficacy-diversity pairings)) + 1.0.

The Quinn Quotient is the proportion of diversity indices with zero line breaks *per* Sabaj Efficacy group, *per* the sum of diversity indices used *per* Sabaj Efficacy group in the PCA correlation matrices; divided by the proportion of the number of line breaks *per* Sabaj Efficacy-diversity index pairing, *per* the sum of all line breaks in all PCA correlation matrices, plus unity (one). Data points were plotted both as raw values and three-point moving averages. The index ranges from zero (no cohesion) to one (maximum cohesion).

The r-QCI revealed specific and discrete quantized microhabitat energies associated with primary, secondary, tertiary and quarternary Harriman orbitals, as well energies of dimensionless microhabitats. Respective color codes used for microhabitat structures were as follows: gold (primary); blue (secondary); red (tertiary); purple (quarternary); and green (dimensionless), and were applied consistently throughout. Respective color codes used for specific VCHI rankings were as follows: red (poor); yellow (poor); purple (fair); blue (good); and green (excellent), and were consistently applied throughout.

Conceptual models were presented concerning the quantum state of the ecosphere. These models are analogous, but not entirely similar to quantum models of atomic structures. Harriman orbitals (so named after Dr. Neil Harriman, botanist emeritus, University of Wisconsin Oshkosh) were modeled as the ecologic equivalent of electron orbitals. Specific microhabitat structures (primary, secondary, tertiary and quarternary) occupy these orbitals, and were modeled as the equivalent of electrons. Dimensionless microhabitats occupied the Strohmosphere. The Paulson Particle is a hypothetical structure occurring at the intersection of the axes of community structure, alpha diversity and Quinn Cohesion.

The Paulson Particle (so named after Dr. James R. Paulson, biochemist, University of Wisconsin Oshkosh) is a hypothetical entity defining the “ecologic qubit,” and is modeled as the equivalent of the atomic nucleus. The interaction occurring between microhabitats and the Paulson Particle define orbital configurations via potential energies of cohesion. Dimensionless microhabitats exist in the “ground state” and occupy the Strohmosphere (so named for Dr. David Strohmeyer, ecologist, retired). Primary microhabitats occupy one of four Harriman orbitals. Secondary microhabitats simultaneously occupy two of four Harriman orbitals; tertiary microhabitats three of four; and quarternary microhabitats four of four.

The VCHI was modeled within the context of the laws of motion embodied by Newtonian mechanics. These models were based upon a three pulley system communicating the Newtonian principles of force, friction, mass and momentum. The VCHI equation was defined as:

$$\mathbf{VCHI = ((SEI + BRI) / 2) * Ø_F}$$

The SEI (Sabaj Efficacy Index) served as a measure of community structure and stratification. Index values were converted to integer ranks to conform to the range of the Bio-invasion Rank Index (BRI). The mean of the SEI and BRI ranks comprised the central aggregate of the VCHI composite. The SEI rankings correspond to the number of communities (native, exotic, woody & herbaceous) stratified within a microhabitat. The Bio-invasion Rank Index (BRI) ranges from 0 to 4 and assesses the absolute percent cover of exotic species. The BRI ranks were established as follows: 0: ≥ 40% exotics; 1: 30-39% exotics; 2: 20-29% exotics; 3: 10-19% exotics; 4: < 10% exotics.

The status of microhabitats was defined by their respective VCHI values: VCHI ≥ 4.0 = excellent; 3.0-3.99 = good; 2.0-2.99 = fair; 1.0-1.99 = poor; 0.0-0.99 = degraded. Horticultural (H) treatment was recommended for microhabitats having BRIs of 0, 1 or 2 (i.e., exotic cover exceeding 20%). Geomorphic (G) treatment was recommended for microhabitats having SEIs of 0 or 1 (i.e., where none or just one community type became stratified).

Figures 3A; 4A through D; 5A through D; 7A and B; 8A and B; and 14A and B were created using PAST software (Hammer et al. 2001). All remaining figures and tables were created using Microsoft Excel® 2010 software.

All data were entered and managed in Microsoft Excel® 2010 software. Data were exported for specific statistical tests, as indicated, to the PAST software program (Hammer et al. 2001).

Some figures and a table were reproduced from Sabaj-Stahl et al. (2013) with permission of the authors. Those reproductions are indicated as appropriate in the manuscript and were reproduced herein for purposes of continuity and further development of novel concepts.

Results

A

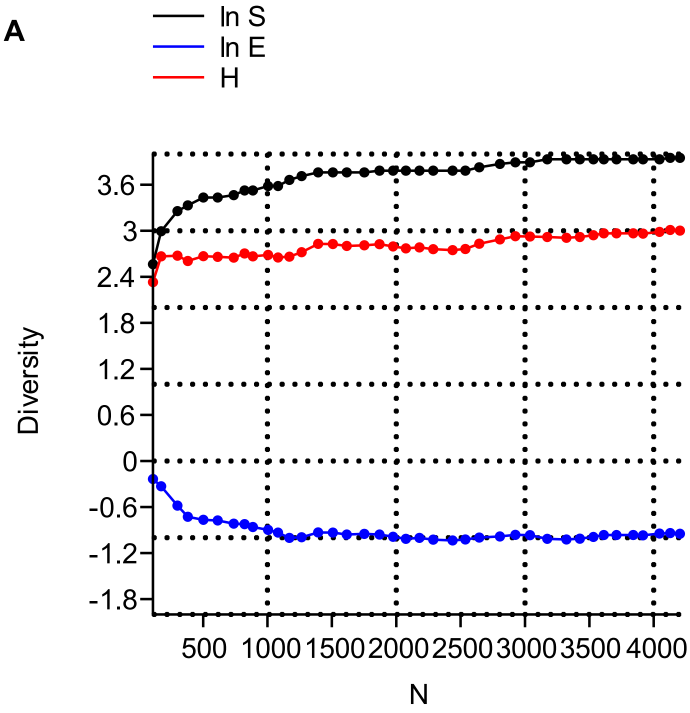
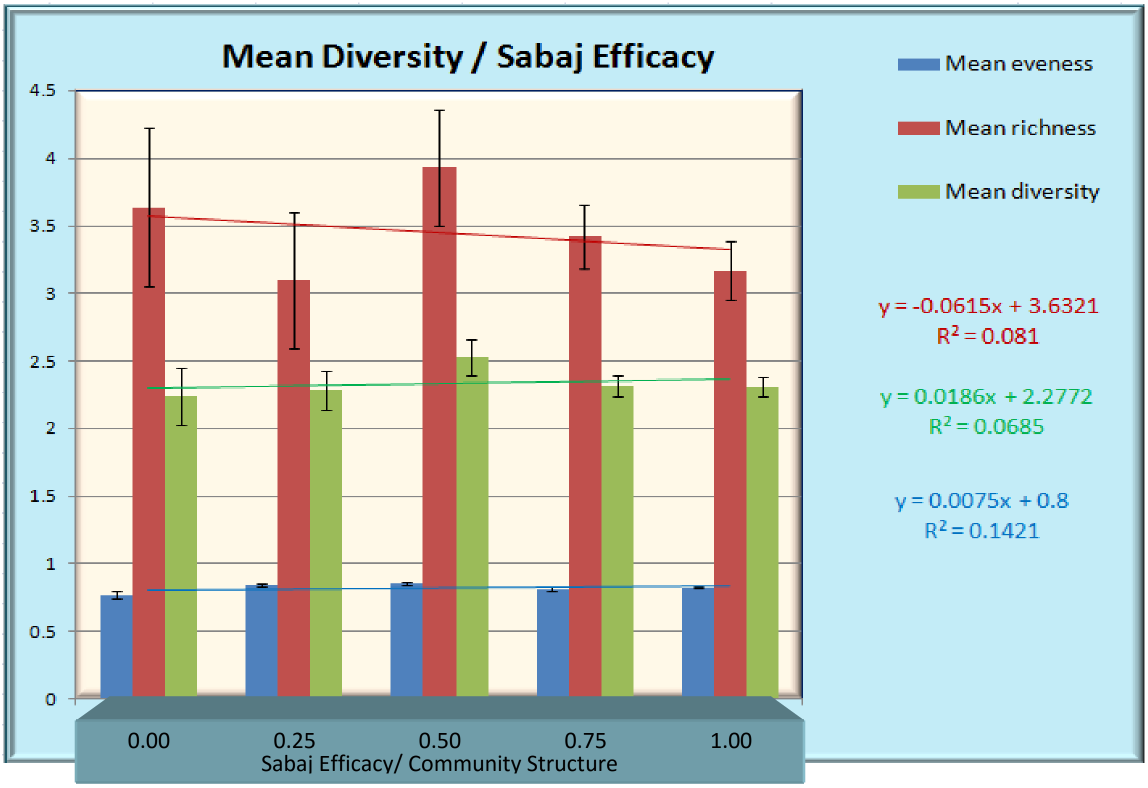


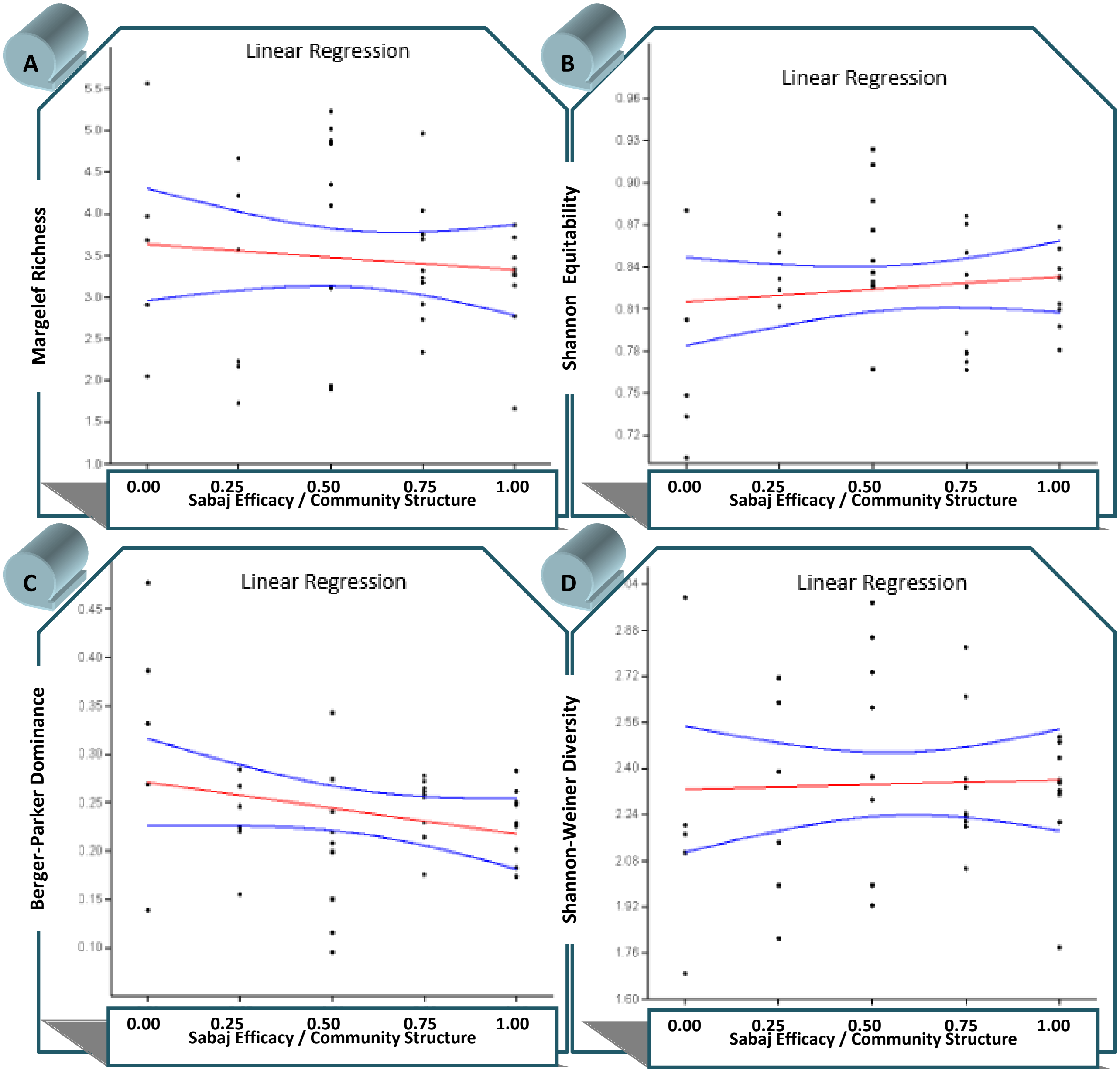
Figure 3: Diversity Profiles

A: SHE analysis was performed using arcsine transformed data for absolute, mean percent cover of all taxa across all sites. SHE analysis assessed contributions of richness (ln S) and evenness (ln E) to changes in Shannon-Wiener's diversity (H) with increasing sampling effort. (Content reproduced from Sabaj-Stahl et al. (2013).

B: Assessment of patterns in mean evenness (Pielou's J), richness (Margalef's index) and total diversity (Shannon-Weiner) per Sabaj Efficacy Index (SEI) group. SEI is a measurement of community level stratification. 0.00 = no stratification; 0.25 = one stratified community; 0.50 = two stratified communities; 0.75 = three stratified communities 1.00 = four communities stratified.

B





Regression Outcomes

	Richness	Equitability	Dominance	Diversity
Slope:	-0.306 (-0.396)	0.018 (0.007)	-0.053 (-0.048)	0.032 (-0.016)
r:	-0.100 (-0.041)	0.122 (-0.139)	-0.254 (0.144)	0.032 (0.064)
r^2:	0.010 (0.002)	0.015 (0.019)	0.064 (0.021)	0.001 (0.004)
t_{stat}:	-0.612 (-0.252)	0.750 (-0.855)	-1.596 (0.884)	0.196 (0.392)
p:	0.544 (0.803)	0.458 (0.398)	0.119 (0.383)	0.846 (0.697)
p(a=1):	0.013 (7.96E-08)	7.73E-33 (2.56E-30)	2.40E-28 (9.39E-07)	6.78E-07 (2.08E-16)

Figure 4: Simple Regression of Biodiversity with Community Structure
Richness (A), evenness (B), dominance (C) and total alpha diversity (D) were regressed with community structure (i.e., Sabaj Efficacy). Red font indicates weighted regression outcomes wherein homoscedasticity was improved using the natural log transformation. One-tailed (p) and two-tailed (p(a=1)) pertain to related t-tests. Data reproduced from Sabaj-Stahl et al. (2013).

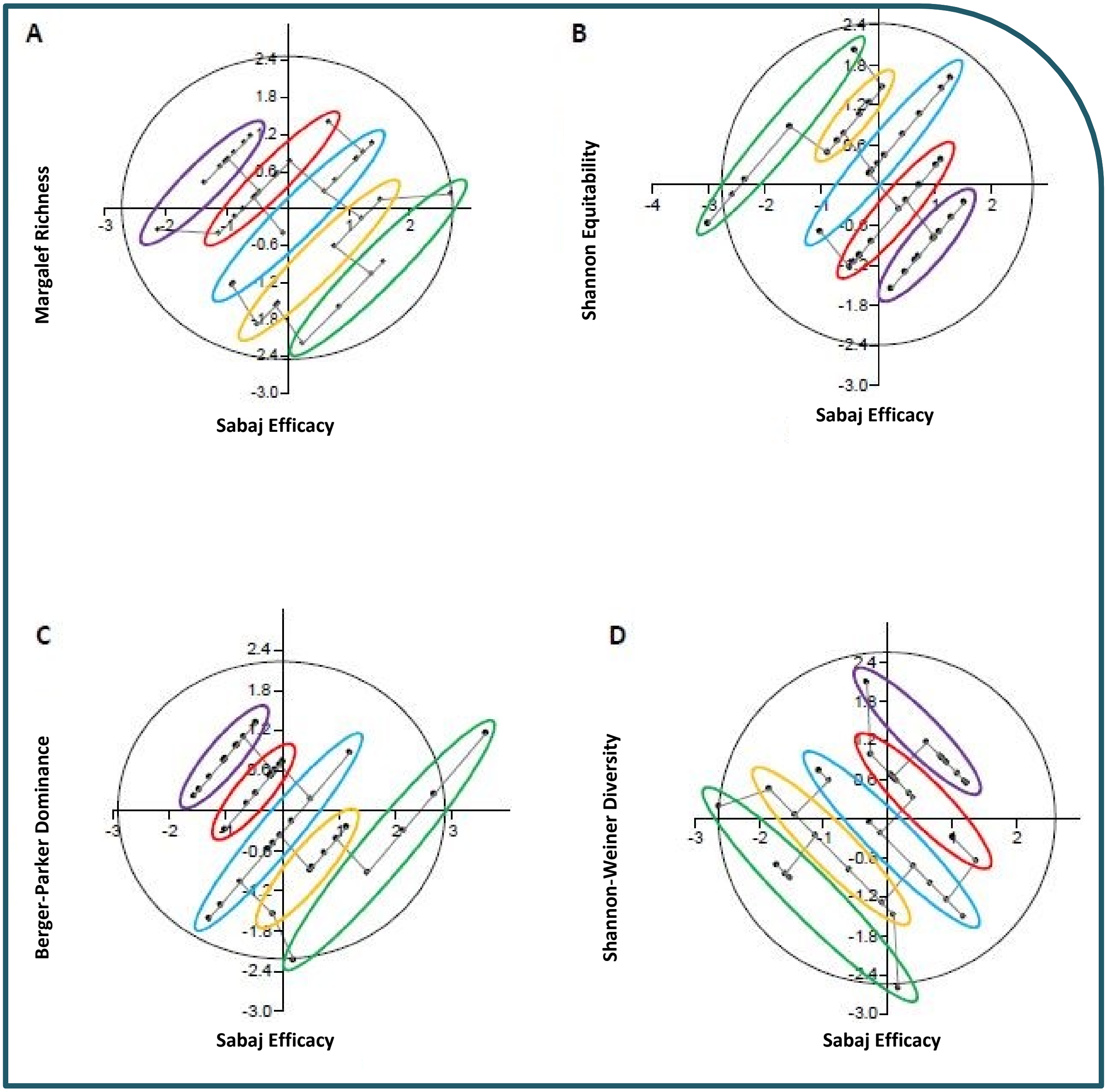
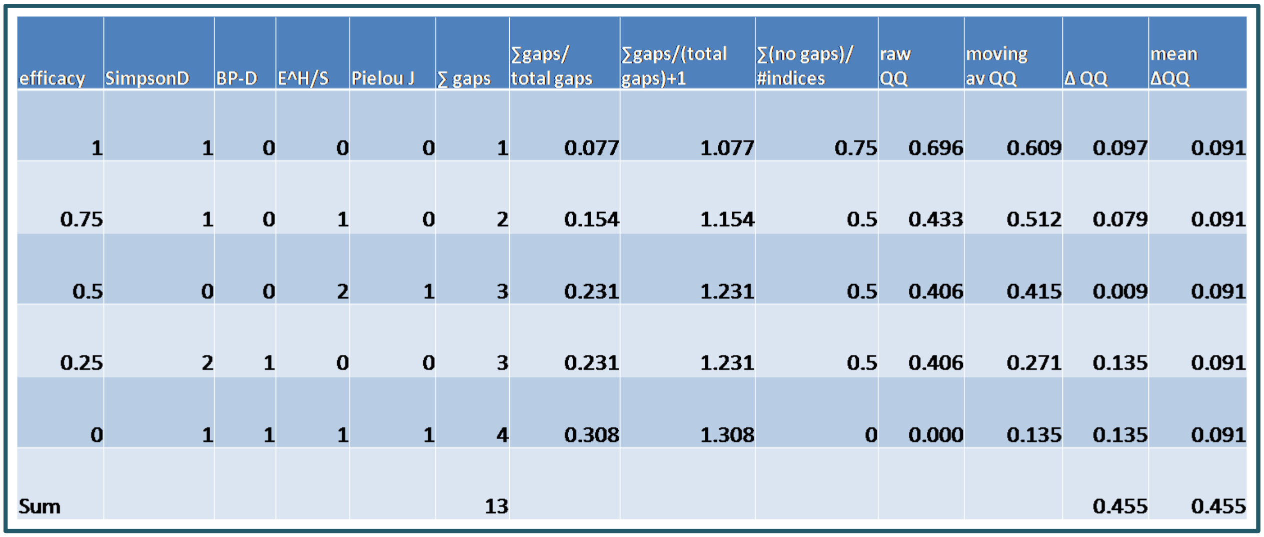


Figure 5: PCA Correlation Matrices of Biodiversity with Structure

Arcsine-square root transformed data were used to derive Sabaj Efficacy (community structure) and Margalef's Richness (A); Shannon's Equitability (B); Berger-Parker's Dominance (C); and Shannon-Weiner's Diversity (D); *per* microhabitat. Sabaj Efficacy and all diversity indices were uniformly distributed. Sabaj Efficacy was defined as the proportion of community types (native, exotic, woody & herbaceous) requiring 5 transects; 500 points per transect & 2500 points per site to detect a minimum 25% change in community structure, relative to baseline percentages, at 90% confidence (i.e.,10% error). Ring Codes: purple = quarternary community structure; red = tertiary community structure; blue = secondary community structure; gold = primary community structure; green = dimensionless community structure. Quarternary = four stratified communities; tertiary = three stratified communities; secondary = two stratified communities; primary = one stratified community; dimensionless = none stratified. (Note: Content reproduced from Sabaj-Stahl, et al. (2013)).

Table 1: Calculation of the Refined Quinn Quotient



efficacy	SimpsonD	BP-D	E^H/S	Pielou J	∑ gaps	∑gaps/ total gaps	∑gaps/(total gaps)+1	∑(no gaps)/ #indices	raw QQ	moving av QQ	Δ QQ	mean ΔQQ
1	1	0	0	0	1	0.077	1.077	0.75	0.696	0.609	0.097	0.091
0.75	1	0	1	0	2	0.154	1.154	0.5	0.433	0.512	0.079	0.091
0.5	0	0	2	1	3	0.231	1.231	0.5	0.406	0.415	0.009	0.091
0.25	2	1	0	0	3	0.231	1.231	0.5	0.406	0.271	0.135	0.091
0	1	1	1	1	4	0.308	1.308	0	0.000	0.135	0.135	0.091
Sum					13						0.455	0.455

The refined Quinn Quotient (r-QQ) was derived from the PCA correlation matrices, using only those indices shown to a) have linear relationships with efficacy *per* linear regression (two-tailed t-test) and b) contribute significant information *per* Akaike Relative Likelihood (Sabaj-Stahl et al. 2013). Simpson's dominance, Berger-Parker's dominance, Buzas & Gibson's evenness, and Pielou's J were used.

QQ = (∑(# of efficacy-diversity pairings with no gaps per efficacy group)/ ∑(# of indices assessed)) / (∑(gaps per efficacy-diversity pairing per efficacy group) / ∑(gaps for all efficacy-diversity pairings)) + 1.0.

The analysis treated dominance as the compliment of evenness (i.e., mirror image). As such, the r-QQ assessed microhabitat relatedness on the basis of evenness and community structure.

Legend: Efficacy = Sabaj Efficacy/ Community Structure. Dominance Indices = Simpson's & Berger-Parker's (BP-D). Evenness Indices = E^H/S & Pielou's J. Raw QQ = Raw Refined Quinn Quotient. Moving Av QQ = Three Point Moving Average of Refined Quinn Quotient. Δ QQ = Difference occurring between efficacy levels for the Three Point QQ. Mean Δ QQ = Mean Difference occurring between efficacy levels for the Three Point QQ.

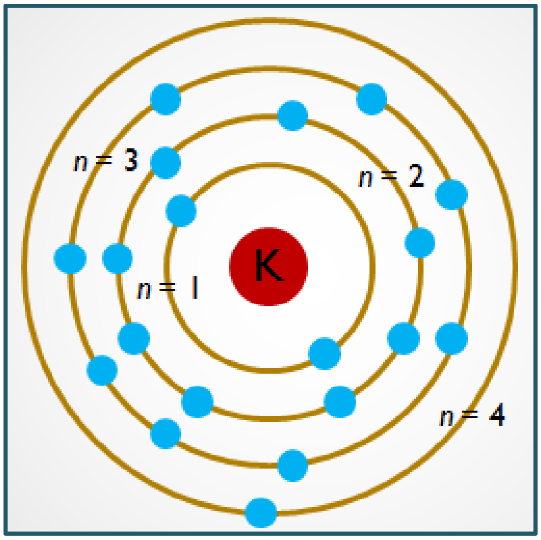


Figure 6: Bohr's Model of the Quantized Atom

Quantum theory is based upon the assumption of the variable integer *n*. The larger the value of *n*, the larger the radius of the electron orbit, and the greater the potential energy of the electron.

Legend:

Red Sphere:	Atomic Nucleus
Blue Spheres:	Electrons
Gold Rings:	Electron Orbitals
n:	Quantum Number
K:	Potassium

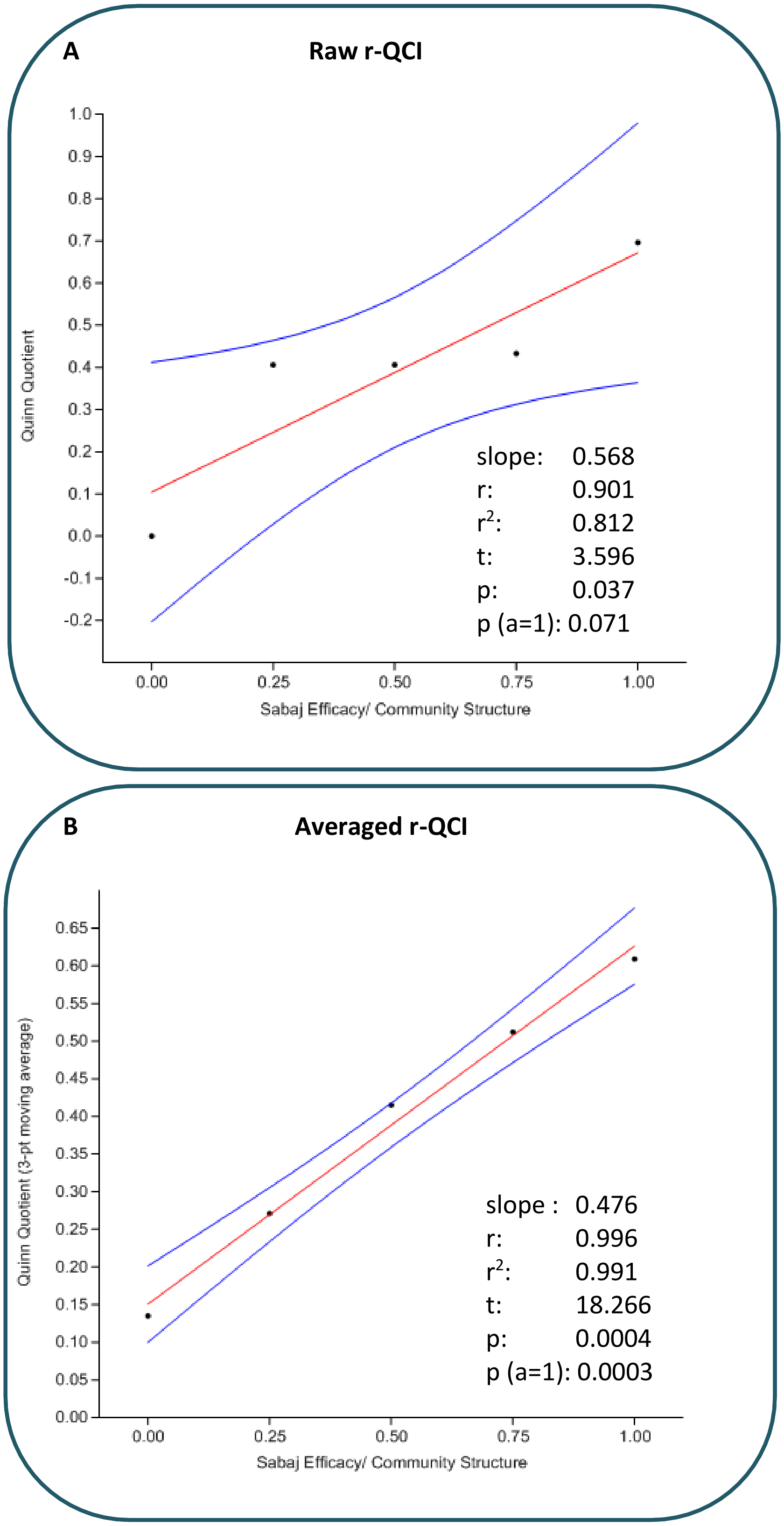


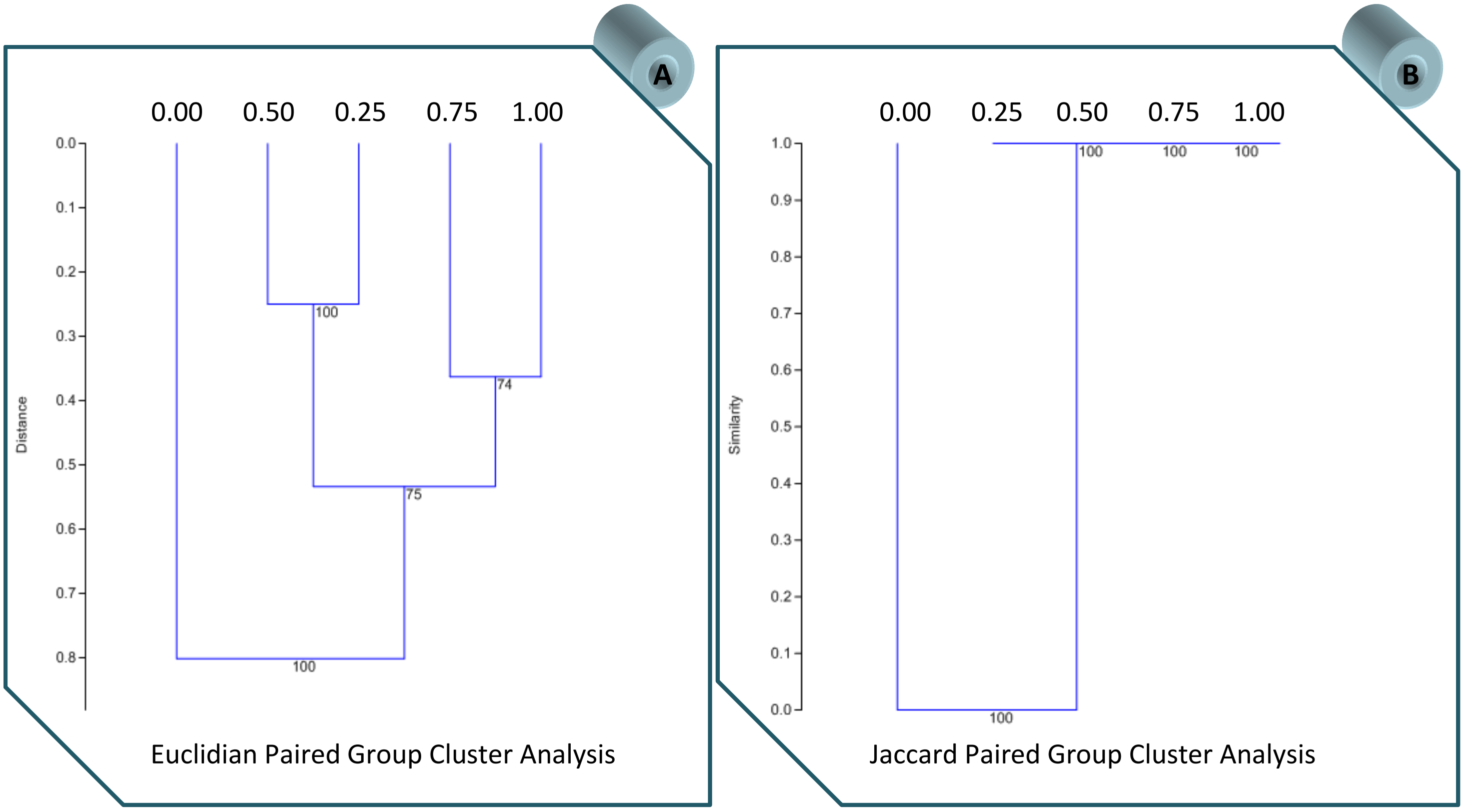
Figure 7: The Refined Quinn Cohesion Index

The r-QCI was defined as the least squares regression of the refined Quinn Quotient (QQ) with Sabaj Efficacy. The r-QCI was derived from the PCA correlation matrices, using only those indices shown to a) have linear relationships with efficacy *per* linear regression and b) contribute significant information *per* Akaike Relative Likelihood (Sabaj-Stahl et al. (2013)). Simpson's dominance, Berger-Parker's dominance, Buzas & Gibson's evenness, and Pielou's J were applied. The index is an approximation of the cohesion between the variables used in the PCA.

QQ = (∑(# of efficacy-diversity pairings with no gaps *per* efficacy group)/ ∑(# of indices assessed)) / (∑(gaps *per* efficacy-diversity pairing *per* efficacy group) / ∑(gaps for all efficacy-diversity pairings)) + 1.0.

Selection criteria (simple regression + Akaike Relative Likelihood) reduced to four (from eleven) the total number of indices assessed *per* the QQ. The averaged r-QCI was derived by plotting the QQ as a 3-point moving average *per* efficacy group.

For purposes of the analysis, dominance was treated as the compliment of evenness, thus providing two additional assessments of evenness. PCA correlation matrices bore that assumption out. Thus, r-QCI measured the force of cohesion occurring between evenness and community structure.



Euclidian Paired Group Cluster Analysis

Jaccard Paired Group Cluster Analysis

Figure 8: Distance & Similarity

Cluster analyses were performed using Sabaj Efficacy (community structure) as the independent (*X*-axis) variable and the Quinn Quotient as the dependent (*Y*-axis) variable. Boot N = 10,000 for both analyses. The analyses assessed potential degrees of relatedness based upon these variables using distance (A) and similarity (B). SEI values (0.00 through 1.00) correspond to one through four stratified communities per microhabitat, respectively.

Table 2: Quantum Numbers of the Ecosphere

Utilizing the trend line produced from the simple regression of the raw refined Quinn Quotient with Sabaj Efficacy/ community structure, quantized energy states were identified for efficacy groupings of 0.00, 0.25 through 0.75, and 1.00. Adjusted r-QCI is the raw refined Quinn Quotient value (*Y*-axis) occurring on the trend line for efficacy (*X*-axis) groups of 0.00, 0.50 & 1.00. The efficacy group of 0.50 was selected to represent the three efficacy groups of 0.25, 0.50 & 0.75; as these three groups exhibited equivalent r-QCI values. Absolute and assumed (rounded) ratios are displayed in the table for r-QCI. These ratios (1.00:3.75:6.50) represent the quantized Quinn Cohesion potential energies assigned to microhabitats based upon the relationship of community structure with diversity.

efficacy	adjusted r-QCI	absolute ratio	assumed ratio	Quantum #
0.00	0.104	1.00	1.00	1
0.50	0.388	3.73	3.75	2
1.00	0.672	6.46	6.50	3

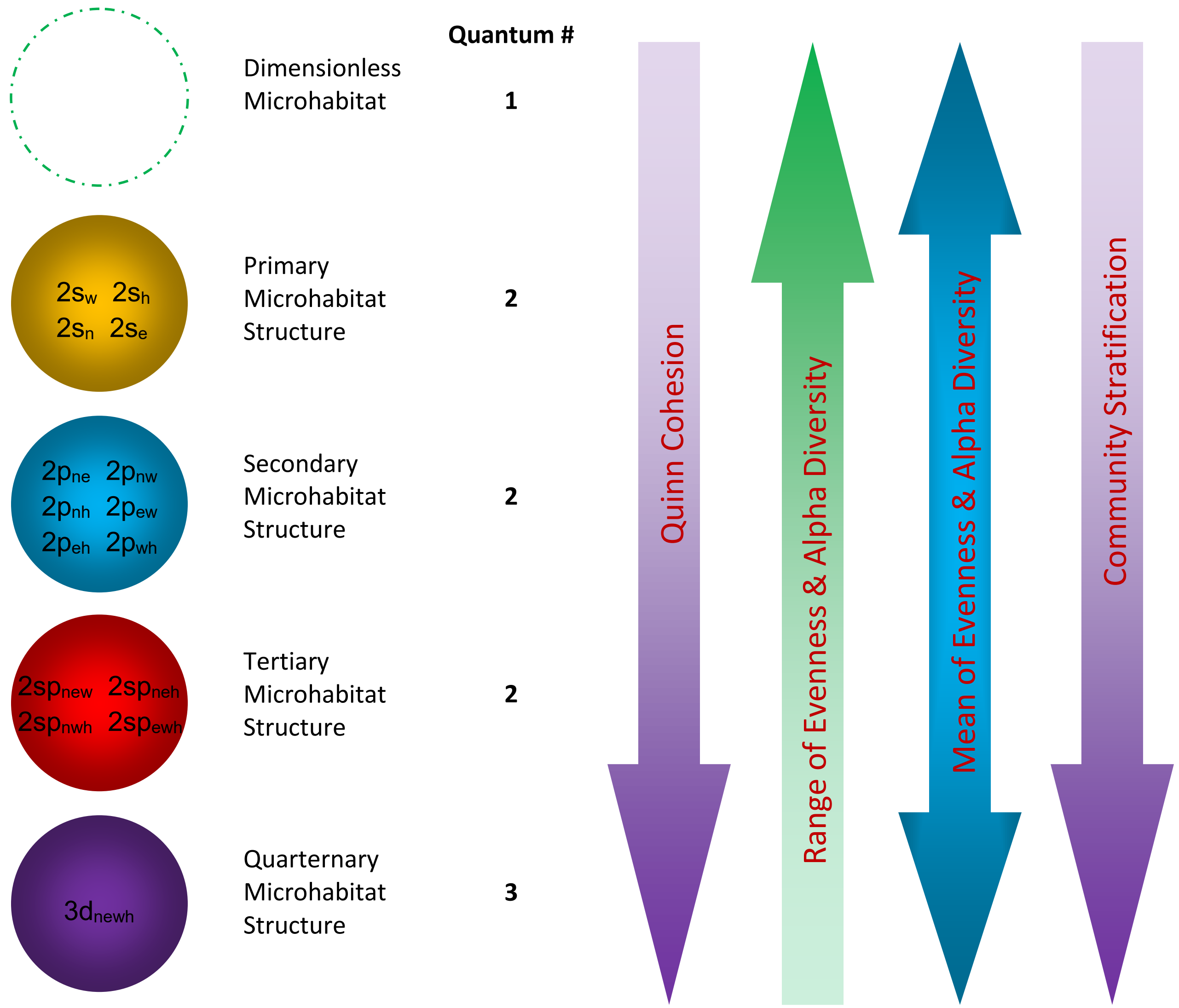


Figure 9: Microhabitats & the Quantum Model

Dimensionless microhabitats contain no stratified communities and occupy the Strohmosphere. Primary microhabitats were stratified for one of four community types and occupy primary (s) orbitals. Secondary microhabitats were stratified for two of four community types and occupy secondary (p) orbitals. Tertiary microhabitats were stratified for three of four community types and occupy tertiary (sp) orbitals. Quarternary microhabitats were stratified for four of four community types and occupy the quarternary (d) orbital. Community types were native, exotic, woody & herbaceous. The force of cohesion increased linearly with increasing degrees of community stratification. Harriman orbital configurations are indicted within each microhabitat sphere. Subscripts (n, e, w and h) indicate those communities (native, exotic, woody and herbaceous) stratified within specific orbital configurations. Quantum numbers (one through three) correspond to quantized energies revealed by the raw refined Quinn Cohesion Index. Multiple orbitals corresponding to the same quantum number are degenerate. Ground state energy corresponds to dimensionless microhabitats.

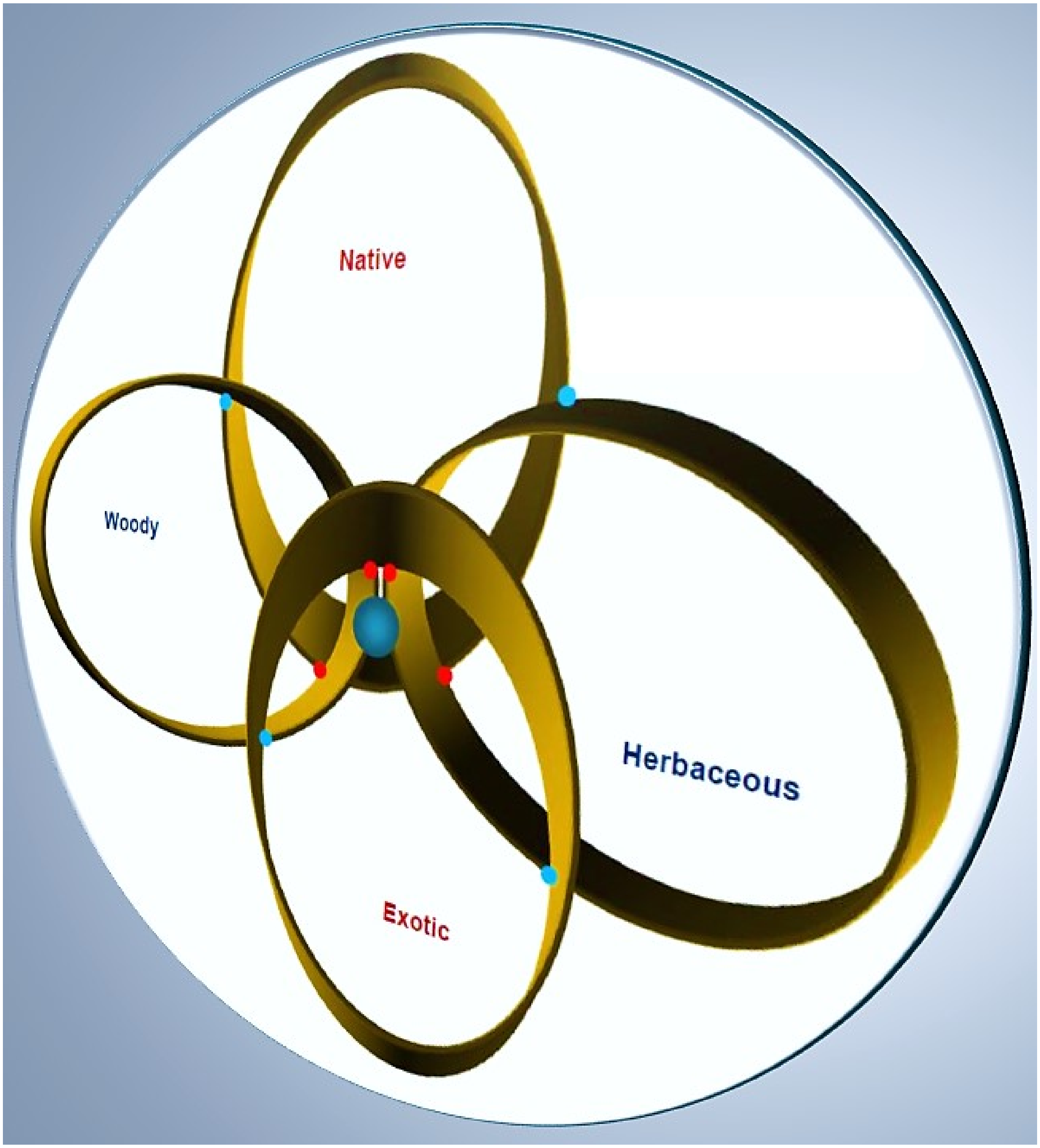


Figure 10: Harriman Orbital Rings of the Quantum Ecosphere

Gold rings indicate primary (s) orbitals for microhabitats with one stratified community type. Primary orbital overlap (blue spheres) creates secondary (p) orbitals for microhabitats containing two stratified communities. Primary and secondary orbital overlap forms hybrid tertiary (sp) orbitals for microhabitats evidencing three stratified communities. Secondary orbital overlap (red spheres) generates the quarternary (d) orbital for microhabitats with four stratified communities. Orbital arrangements reflect overlap of the native community with woody & herbaceous communities; and the exotic community with woody & herbaceous communities. The model illustrates native & exotic communities; and woody & herbaceous communities are mutually exclusive of one another. The Paulson Particle (teal green sphere) represents the confluence of community structure, alpha diversity and Quinn Cohesion.

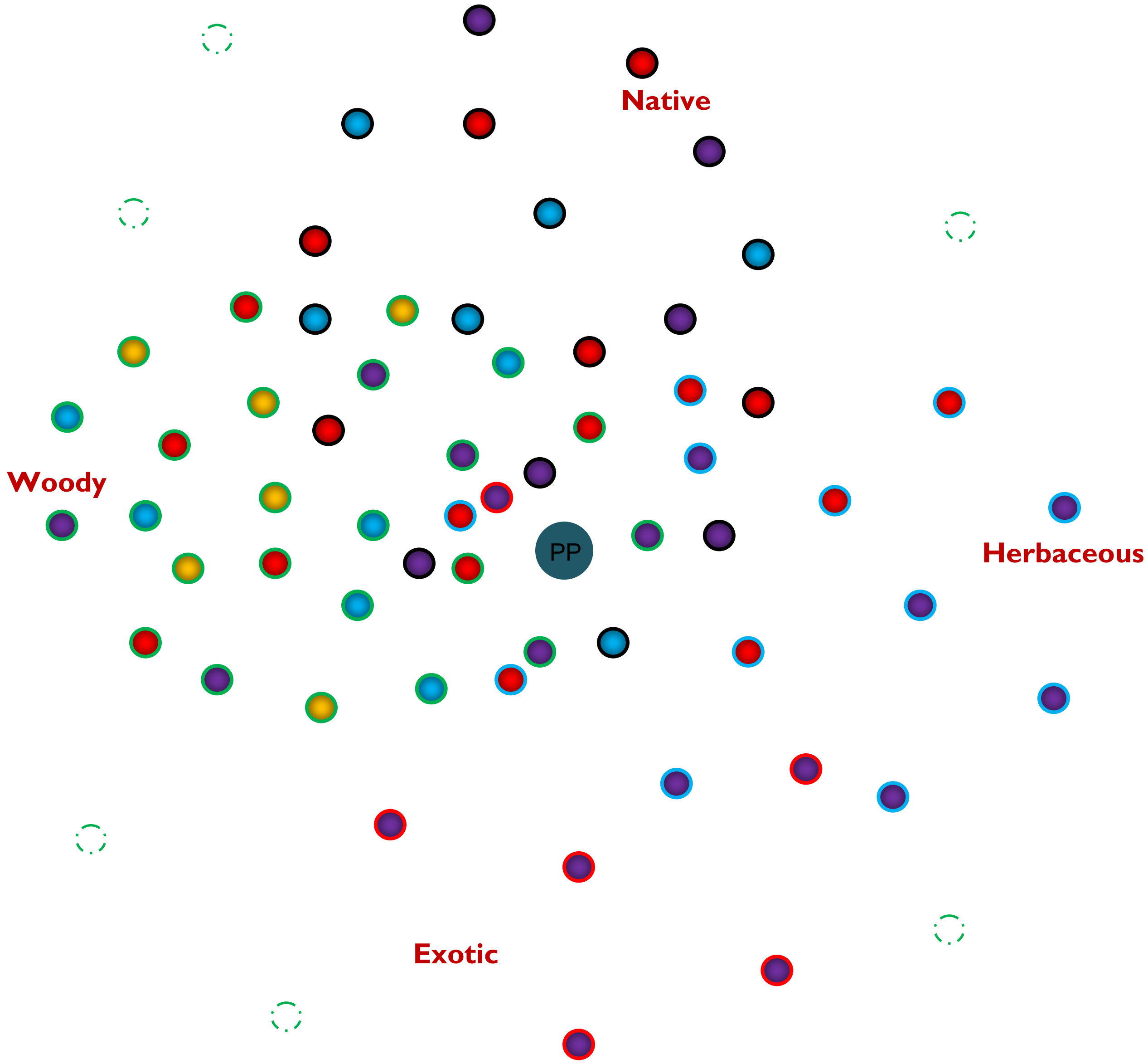


Figure 11: The Quantum Ecosphere: Hypothetical Model

Structure:

- 🟡 = Primary Microhabitat Structure.
- 🔵 = Secondary Microhabitat Structure.
- 🔴 = Tertiary Microhabitat Structure.
- 🟣 = Quarternary Microhabitat Structure.
- ◌ =Dimensionless Microhabitat Structure.

Orbital:

- ○= Woody Harriman Orbital.
- ○= Native Harriman Orbital.
- ○= Herbaceous Harriman Orbital.
- ○= Exotic Harriman Orbital.

The hypothetical quantum model is based upon 36 microhabitats; with six microhabitats per structure. 30 microhabitats are represented in the woody Harriman orbital cloud. Then, moving clockwise, the lowest order structure is sequentially eliminated from subsequent orbital clouds. Thus, microhabitats with primary structure are absent from the native, herbaceous & exotic Harriman orbital clouds. Microhabitats with secondary structure are absent from herbaceous & exotic Harriman orbital clouds. Microhabitats with tertiary structure are absent from the exotic Harriman orbital cloud. Quartnerary microhabitats are present in all four orbital clouds. Dimensionless microhabitats occupy the Strohmosphere. PP = Paulson Particle.

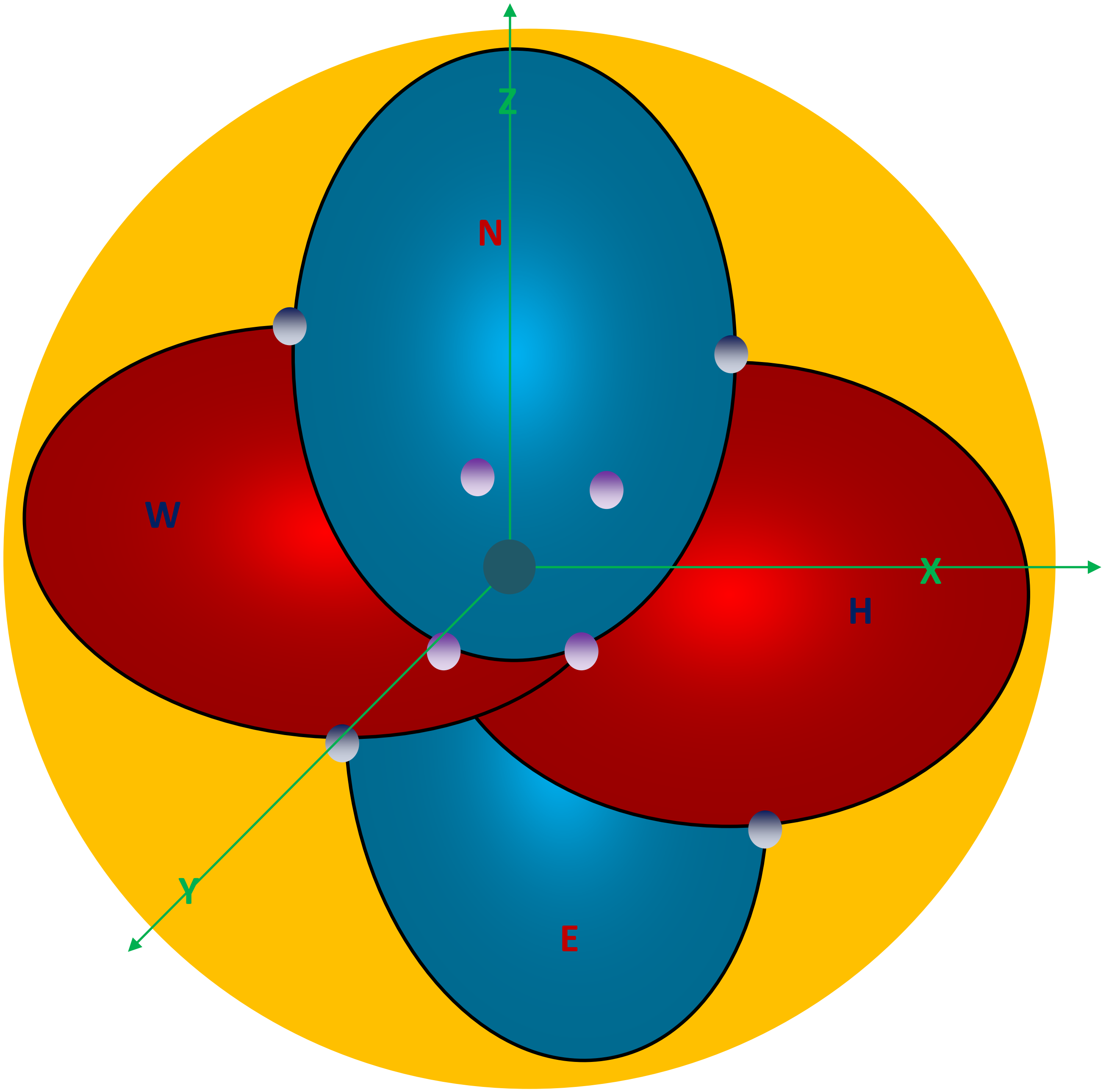


Figure 12: The Quantum Ecosphere

Primary Harriman orbital clouds: N=native; E=exotic; W=woody; H=herbaceous. Secondary Harriman orbital clouds are modulated by primary ● orbital nodes. Tertiary Harriman orbital clouds are modulated primary and secondary ● orbital nodes. Quarternary Harriman orbitals are modulated by secondary orbital lobes. *X*-axis = community structure; *Y*-axis = alpha diversity; *Z*-axis = Quinn Cohesion. Axes intersect at the Paulson Particle ●. The Strohmosphere (yellow sphere) is occupied by dimensionless microhabitats.

Table 3: Percent Cover Estimates & Sampling Effort

Confidence intervals estimated number of transects required to detect 25% change per cover type. GJ = Grand Junction sites; MO = Moab sites; UC = Uncompahgre sites; DL = Dolores sites. Font color codes: purple = quarternary structure; red = tertiary structure; blue = secondary structure; gold = primary structure; green = dimensionless structure. Site codes highlighted in dark grey were diagnosed with Ecospheric Late Stage Disease; those also outlined in red with Ecospheric End Stage Disease.

	Absolute Percent Covers				Efficacious Transect Number Estimates			
Site	Native	Exotic	Woody	Herb	Native	Exotic	Woody	Herb
GJ1	35.65	29.12	30.98	33.65	5	5	5	5
GJ2	24.84	20.63	27.25	17.89	6	8	6	5
GJ3	50.44	38.69	59.55	30.38	5	5	5	5
GJ4	35.41	47.81	33.74	49.40	8	5	6	5
GJ5	55.11	37.28	60.21	33.11	5	5	6	8
GJ6	39.19	37.95	41.06	35.99	12	12	11	9
GJ7	35.12	53.66	50.82	37.95	8	5	5	5
GJ8	16.74	46.57	24.21	41.72	6	5	5	5
GJ9	16.42	42.17	40.09	20.11	15	6	6	9
GJ10	84.10	45.32	63.58	31.66	5	5	5	6
MO1	20.73	46.36	36.64	32.49	6	9	6	5
MO2	21.12	47.66	36.05	35.31	11	5	6	5
MO3	41.66	21.06	23.68	39.76	5	9	9	5
MO4	57.32	22.13	43.92	37.42	5	5	5	5
MO5	30.54	43.99	39.94	30.93	6	6	11	5
MO6	35.99	50.52	65.28	20.39	6	12	8	12
MO7	52.75	37.27	68.17	23.53	6	11	8	5
MO8	32.68	41.47	36.75	37.16	5	5	5	5
MO9	20.41	62.64	53.22	29.88	8	16	12	6
MO10	34.82	46.07	39.69	41.32	5	5	5	5
UC1	17.21	41.11	30.09	31.46	9	5	5	5
UC2	32.43	56.66	50.46	38.44	5	5	5	5
UC3	49.92	52.49	82.10	24.59	5	6	5	5
UC4	50.44	19.34	64.38	24.84	5	5	5	5
UC5	37.52	19.83	23.95	35.16	5	6	8	5
UC6	60.89	22.73	57.90	26.77	6	17	8	11
UC7	31.31	39.15	32.89	37.74	5	9	9	5
UC8	26.68	42.58	32.77	27.37	12	6	5	5
UC9	34.38	61.05	58.21	35.23	13	11	5	6
UC10	64.12	41.86	83.35	26.36	5	5	5	5
DL1	43.91	27.96	25.71	45.18	5	5	12	5
DL2	44.17	10.34	37.20	23.15	5	6	5	5
DL3	34.23	0.73	30.80	15.21	5	215+	5	5
DL4	39.95	22.31	41.77	19.84	5	12	5	15
DL5	43.64	34.31	31.81	45.91	5	5	5	5
DL6	19.11	40.19	29.86	31.63	5	5	6	5
DL7	48.32	9.28	38.02	26.72	5	6	5	5
DL8	44.54	8.05	21.31	38.09	5	29	8	5
DL9	25.63	36.27	30.68	31.52	5	9	6	6

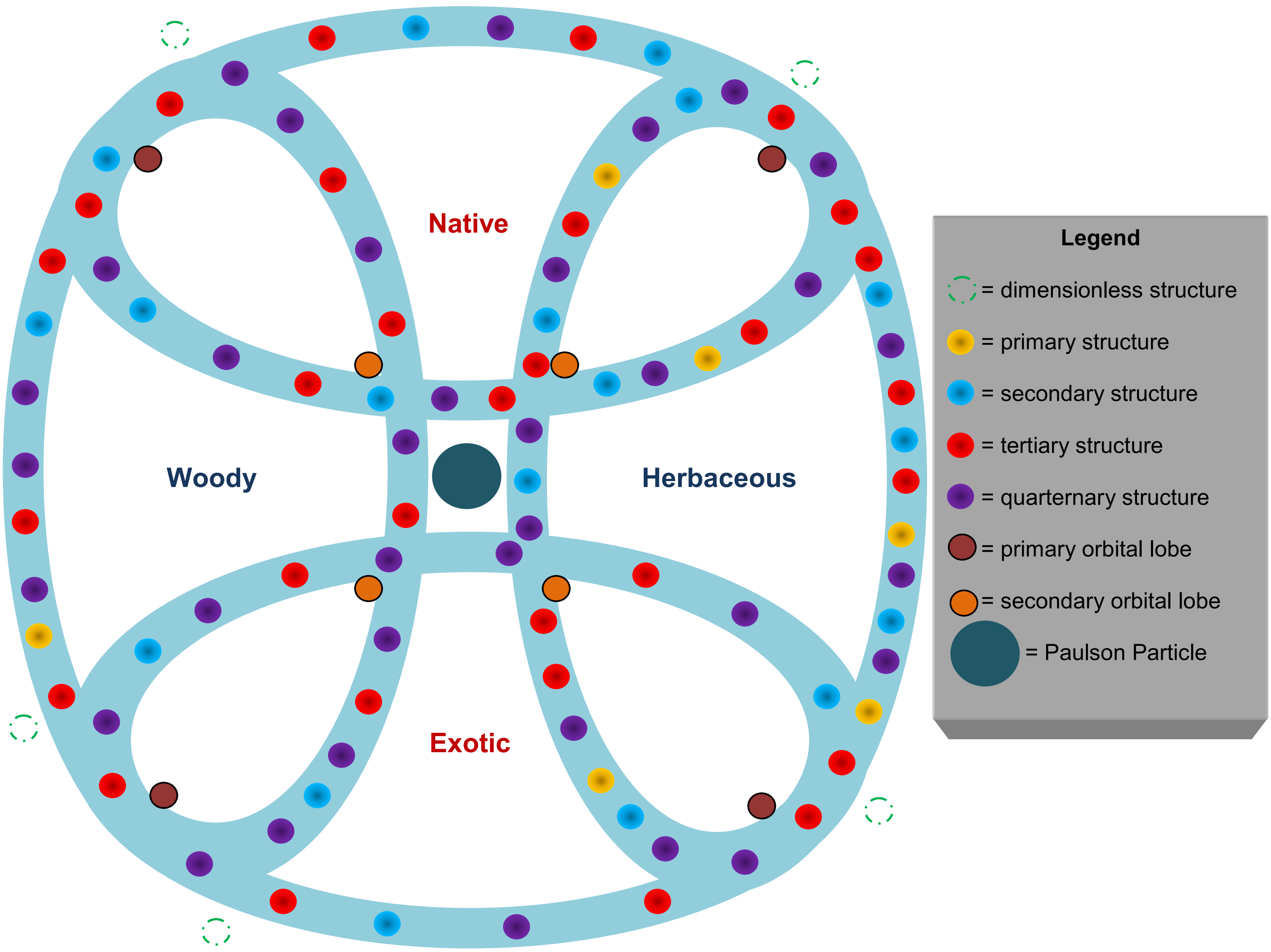
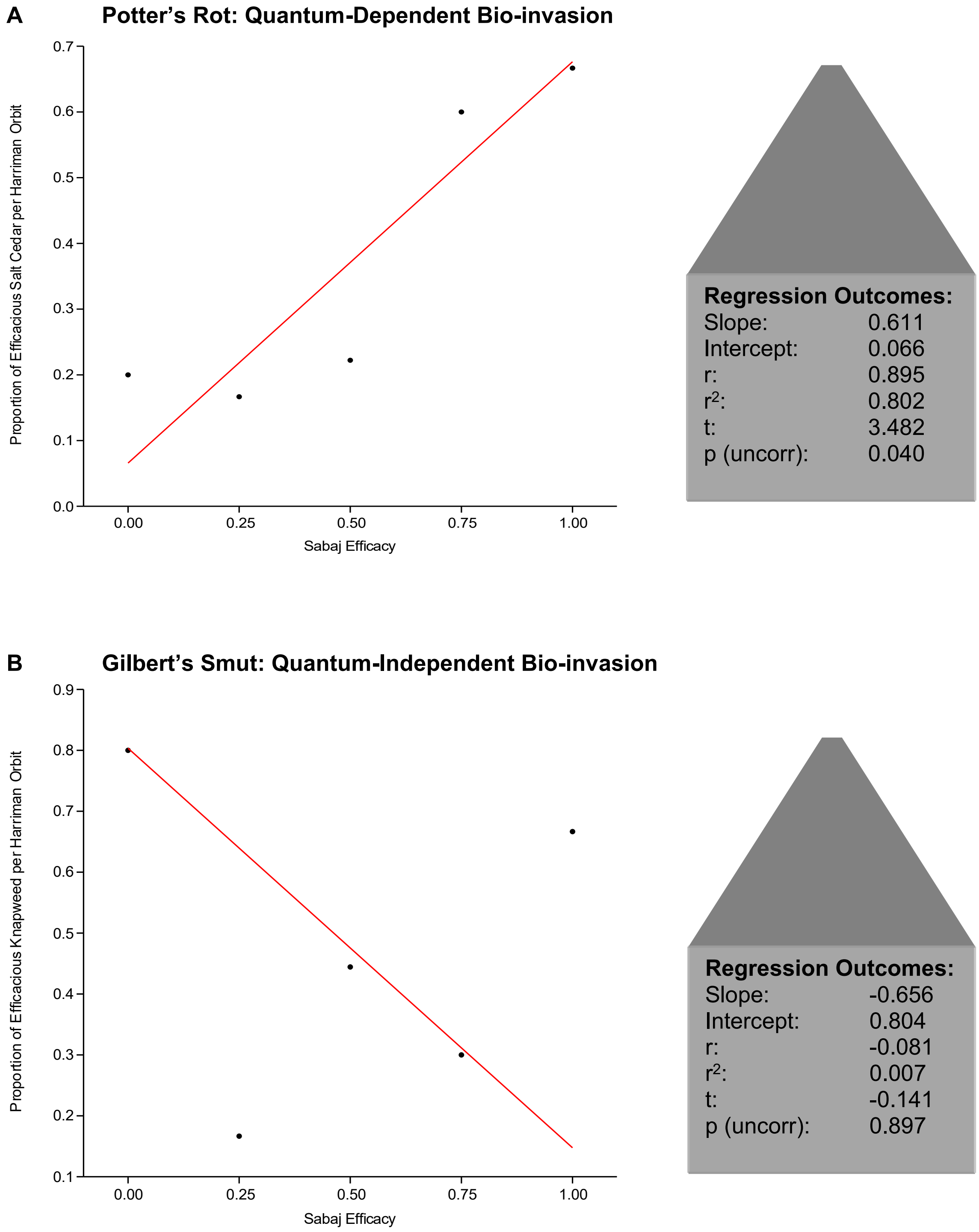


Figure 13: The Lower Dolores Quantum Ecosphere

Gold spheres indicate microhabitats evidencing primary structure (one of four communities stratified) (i.e., Native, Exotic, Woody or Herbaceous). Blue spheres represent microhabitats adherent to secondary structure (two of four communities stratified). Red spheres indicate microhabitats with tertiary structure (three of four communities stratified). Purple spheres indicate microhabitats evidencing quarternary structure (four of four communities stratified). The Paulson Particle occurs at the intersection of the axes of community structure, diversity and cohesion. Microhabitats are depicted in their actual frequencies *per* stratification criteria, *per* community type, as assessed in the lower Dolores Watershed during the summer of 2010. The quantum state reflects discrete energies associated with each microhabitat based upon the duality occurring between structure and diversity. Microhabitats with multiple stratified communities are depicted as occurring in multiple Harriman orbitals. Dimensionless microhabitats occur in the Strohmosphere.



A Potter's Rot: Quantum-Dependent Bio-invasion

Regression Outcomes:	
Slope:	0.611
Intercept:	0.066
r:	0.895
r^2:	0.802
t:	3.482
p (uncorr):	0.040

B Gilbert's Smut: Quantum-Independent Bio-invasion

Regression Outcomes:	
Slope:	-0.656
Intercept:	0.804
r:	-0.081
r^2:	0.007
t:	-0.141
p (uncorr):	0.897

Figure 14: Orthogonal Regression of Invasiveness & Community Structure

Invasive species structure was estimated as the proportion of microhabitats within specific Harriman orbitals wherein salt cedar (A) and knapweed (B) adhered to 40% confidence interval limits. Community structure was defined as Sabaj Efficacy (25% confidence interval limits for community types). Quantum-dependent processes of bio-invasion (A) were consistent with community structure. Quantum-independent processes of bio-invasion (B) were not consistent with community structure.

Table 4: The Vegetative Complex Health Index

$$VCHI = ((SEI + BRI) / 2) * \O_F$$

Site	SEI	BRI	Fisher α	VCHI	Status	Treatment
GJ1	4	2	4.988	2.61	fair	H
GJ2	1	2	9.252	2.42	fair	GM+H
GJ3	4	1	5.933	2.59	fair	H
GJ4	2	0	2.627	0.46	degraded	H
GJ5	2	1	6.745	1.76	poor	H
GJ6	0	1	4.482	0.39	degraded	GM+H
GJ7	3	0	4.874	1.28	poor	H
GJ8	3	0	3.428	0.90	degraded	H
GJ9	0	0	3.001	0.00	degraded	GM+H
GJ10	3	0	6.054	1.58	poor	H
MO1	1	0	2.377	0.21	degraded	GM+H
MO2	2	0	7.751	1.35	poor	H
MO3	2	2	8.783	3.06	good	H
MO4	4	2	5.156	2.70	fair	H
MO5	1	0	7.204	0.63	degraded	GM+H
MO6	0	0	6.061	0.00	degraded	GM+H
MO7	1	1	3.156	0.55	degraded	GM+H
MO8	4	0	6.458	2.25	fair	H
MO9	0	0	6.848	0.00	degraded	GM+H
MO10	4	0	2.237	0.78	degraded	H
UC1	3	0	6.425	1.68	poor	H
UC2	4	0	4.083	1.42	poor	H
UC3	3	0	5.105	1.34	poor	H
UC4	4	3	5.202	3.18	good	None
UC5	2	3	8.930	3.89	good	None
UC6	0	2	10.19	1.78	poor	GM+H
UC7	2	1	9.787	2.56	fair	H
UC8	2	0	4.886	0.85	degraded	H
UC9	1	0	5.680	0.50	degraded	GM+H
UC10	4	0	5.394	1.88	poor	H
DL1	3	2	4.155	1.81	poor	H
DL2	3	3	9.347	4.89	excellent	None
DL3	3	4	4.733	2.89	fair	None
DL4	2	2	2.723	0.95	degraded	H
DL5	4	1	5.130	2.24	fair	H
DL6	3	0	5.285	1.38	poor	H
DL7	3	4	6.805	4.15	excellent	None
DL8	2	4	9.197	4.81	excellent	None
DL9	1	1	3.156	0.55	degraded	GM+H
System	2.31	1.08	5.734	1.69	poor	Varied

$\O_F$ = Fisher's Disambiguous Coefficient = Fisher α_{site} / Fisher α_{system}.

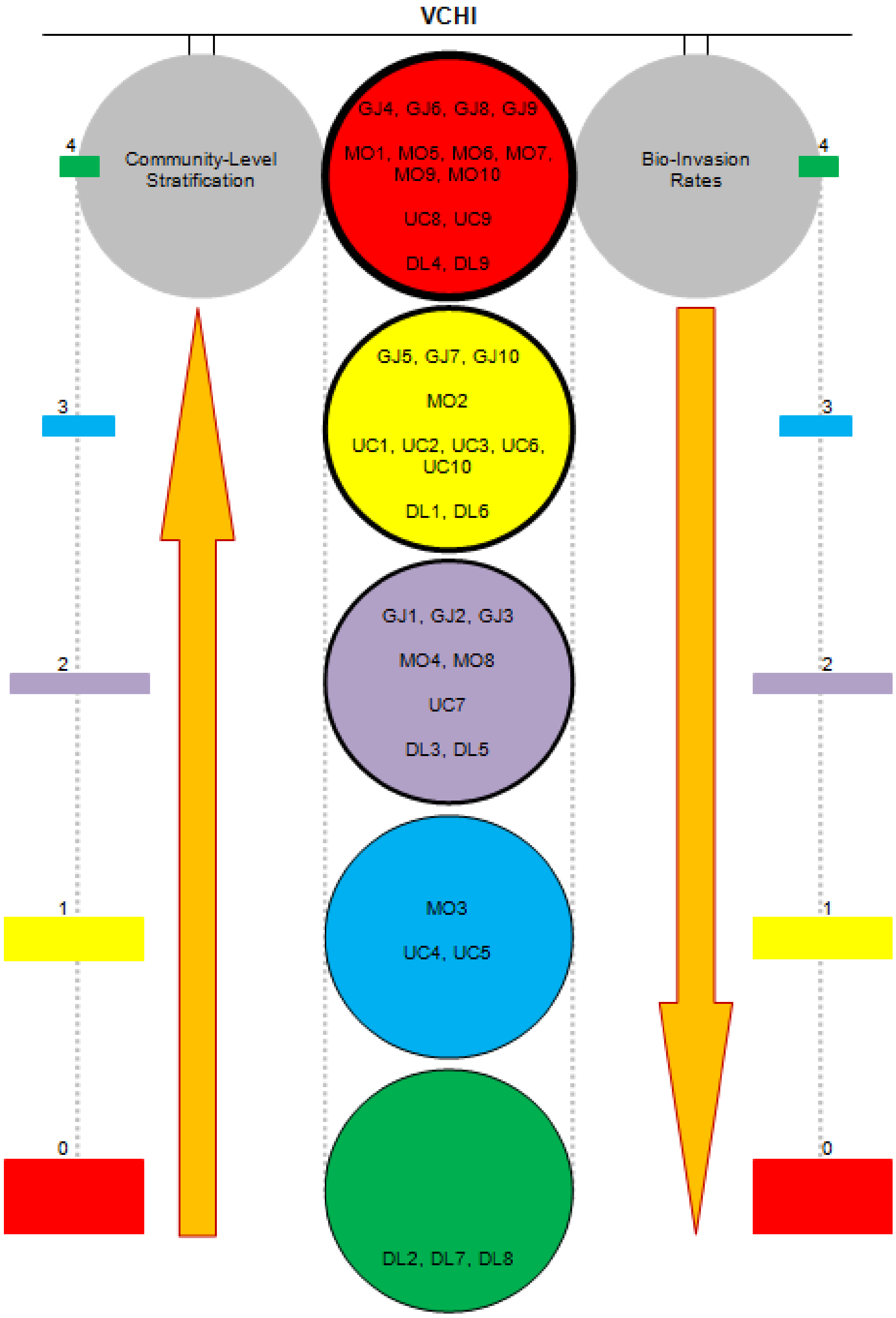
SEI = Sabaj Efficacy Index rankings. Index values were converted to rankings of 0, 1, 2, 3 and 4. These rankings correspond to the number of communities (native, exotic, woody & herbaceous) stratified within a microhabitat.

Bio-invasion Rank Index (BRI) ranges from 0 to 4. 0: ≥ 40% exotics; 1: 30-39% exotics; 2: 20-29% exotics; 3: 10-19% exotics; 4: < 10% exotics.

Status of microhabitats was defined by their respective VCHI: VCHI ≥ 4.0 = excellent; 3.0-3.99 = good; 2.0-2.99 = fair; 1.0-1.99 = poor; 0.0-0.99 = degraded.

Horticultural (H) treatment was recommended for microhabitats having BRIs of 0, 1 or 2 (i.e., exotic cover exceeding 20%). Geomorphic (G) treatment was recommended for microhabitats having SEIs of 0 or 1 (i.e., where none or just one community type became stratified).

Font color site codes: purple = quarternary structure; red = tertiary structure; blue = secondary structure; gold = primary structure; green = dimensionless structure. These codes correspond to the number of communities stratified within a microhabitat.



Legend:

VCHI = Vegetative Complex Health Index.

Grey spheres = fixed pulleys for community level stratification (SEI) & bio-invasion (BRI).

Colored bars = relative weights for community level stratification (SEI) & bio-invasion ratings (BRI) (table 4).

Red, yellow, purple, blue & green spheres = pulley free to move; colors correspond to VCHI ratings of degraded, poor, fair, good & excellent, respectively (table 4).

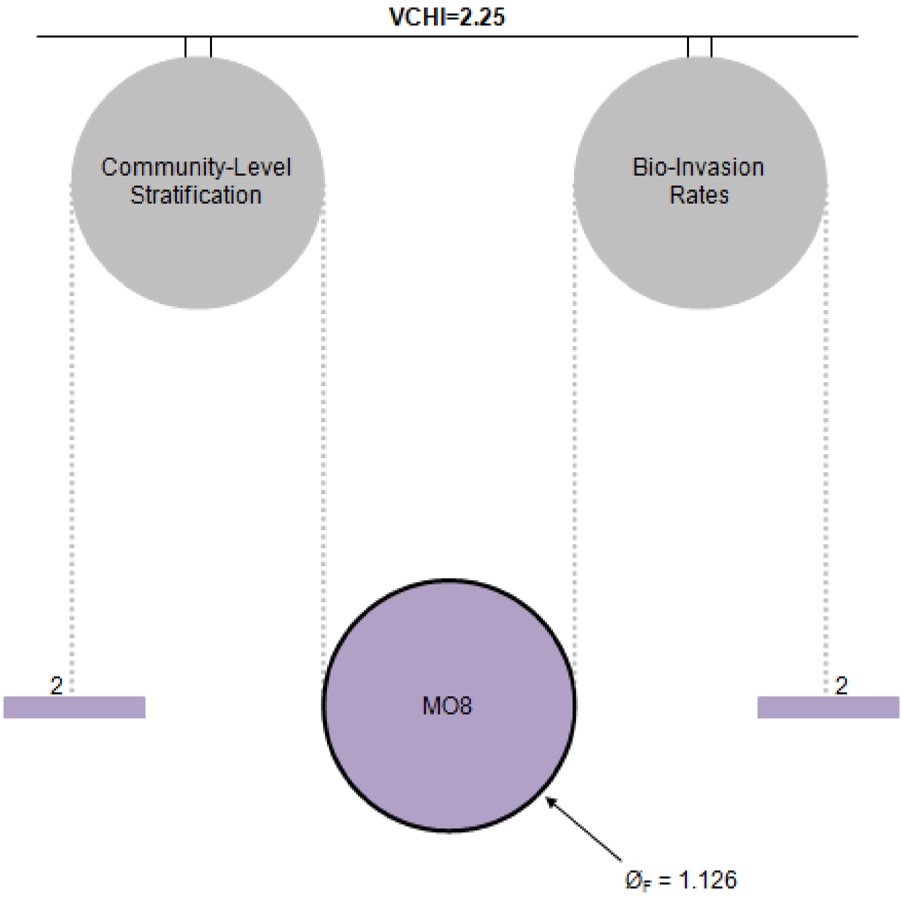
Black outer rings of free pulleys indicate friction of Fisher's Disambiguous Coefficient ($\O_F$).

Gold arrows indicate direction of increasing community-level stratification & bio-invasion.

Dotted grey line = chain attached to weights & associated with fixed and free pulleys.

Figure 15: Overall Mechanics of the Vegetative Complex Health Index

The three components of the Vegetative Complex Health Index as revealed *per* Newtonian mechanics via a three pulley system. Community stratification (SEI) and bio-invasion rates (BRI) are shown as fixed pulleys. The chain extending from each has variable weights attached contingent upon SEI and BRI ratings. The pulley that is free to move represents the mass of individual microhabitats with associated frictional force (i.e., Fisher's Disambiguous Coefficient). Friction is shown to increase from higher to lower VCHI rankings, however the trend was not universal.



Legend:

VCHI = Vegetative Complex Health Index.

Grey spheres = fixed pulleys for community structure and bio-invasion.

Purple bars = relative weights for community level stratification (SEI) & bio-invasion (BRI) ratings (table 4).

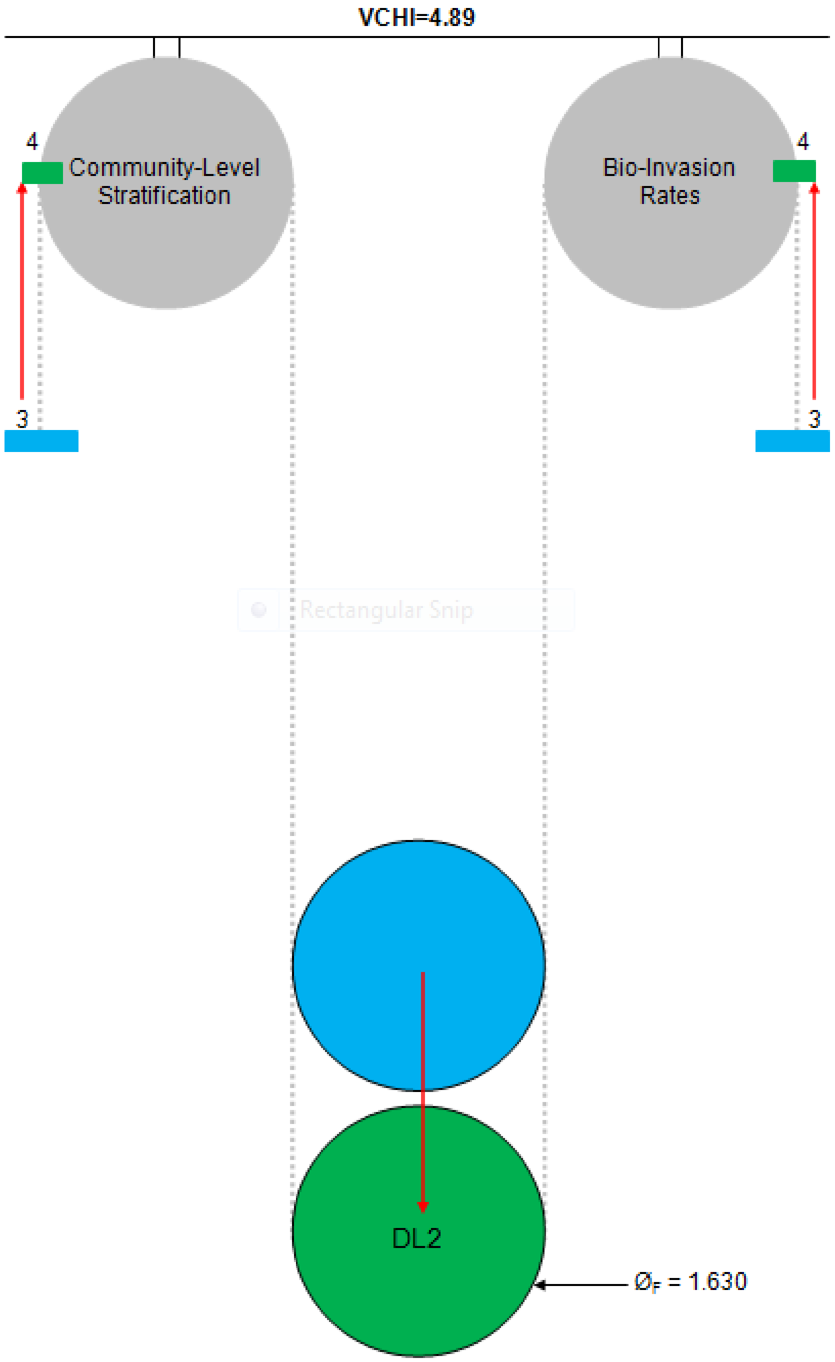
Purple sphere = mass of microhabitat MO8 & pulley free to move; color corresponds to VCHI rating of fair (table 4).

Black outer ring of free pulley indicates friction of Fisher's Disambiguous Coefficient ($Ø_F$).

Dotted grey line = chain attached to weights & associated with fixed and free pulleys.

Figure 16: Neutral Impact of the FDC

The three components of the Vegetative Complex Health Index as revealed *per* Newtonian mechanics via a three pulley system. Community stratification (SEI) and bio-invasion rates (BRI) are shown as fixed pulleys. The chain extending from each has relative weights of 2.0 attached *per* SEI and BRI ratings (table 4). The pulley that is free to move represents the mass of microhabitat MO8 with associated frictional force $Ø_F$ = 1.126. Friction tends toward that of the ecosystem (i.e., 1.00), thus exhibiting neutral effects upon force, and not impacting the VCHI ranking.



Legend:

VCHI = Vegetative Complex Health Index.

Grey spheres = fixed pulleys for community structure & bio-invasion.

Blue & green bars = relative weights for community level stratification (SEI) & bio-invasion (BRI) ratings (table 4).

Blue & green spheres = pulley free to move; colors correspond to VCHI ratings of good & excellent, respectively (table 4).

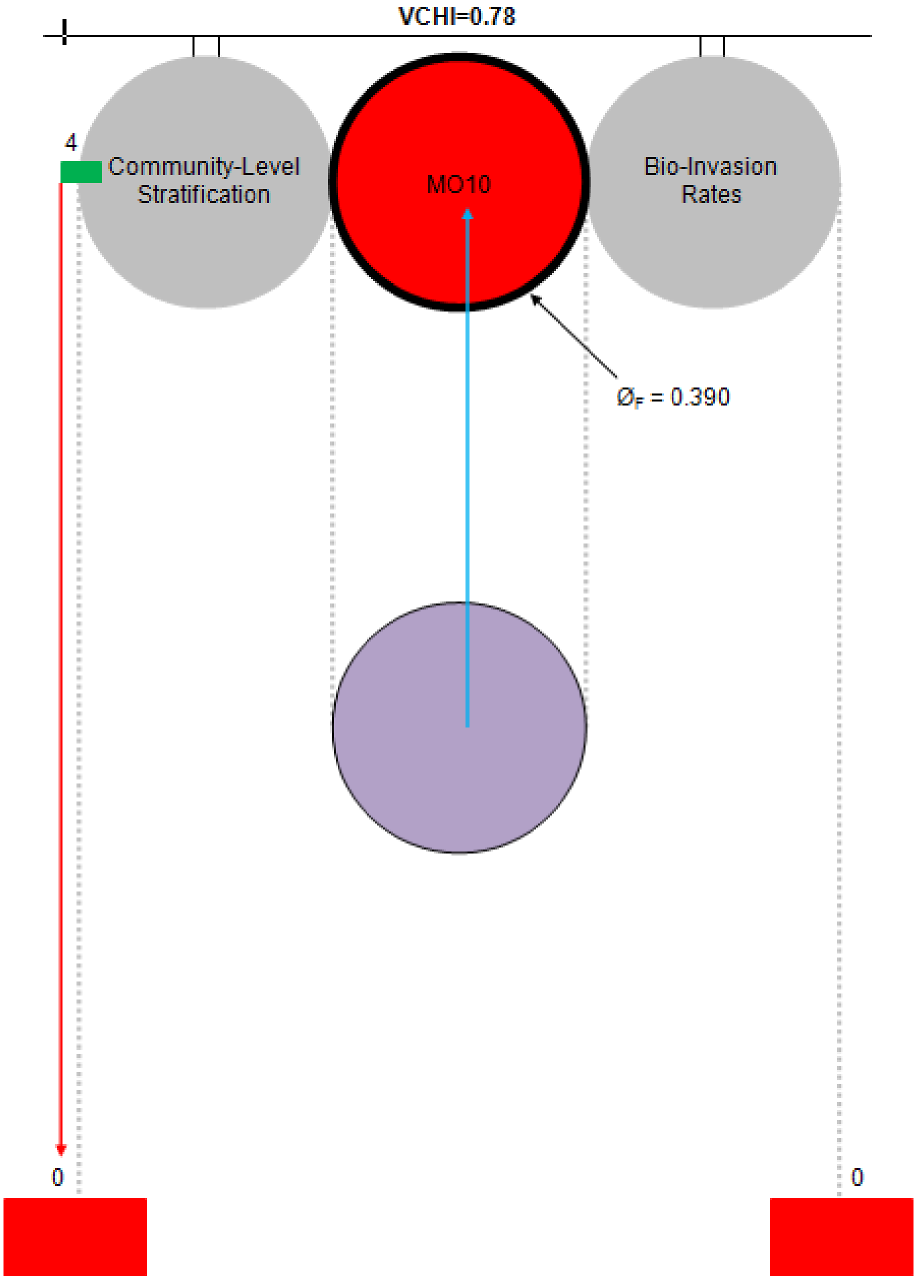
Black outer rings of central pulley indicate friction of Fisher's Disambiguous Coefficient ($Ø_F$).

Dotted grey line = chain attached to weights and associated with fixed and free pulleys.

Red arrows indicate net effects of increased friction ($Ø_F$) upon chain weights and VCHI rating.

Figure 17: Positive Impact of the FDC

The three components of the Vegetative Complex Health Index as revealed *per* Newtonian mechanics via a three pulley system. Community stratification (SEI) and bio-invasion rates (BRI) are shown as fixed pulleys. The chain extending from each has relative weights of 3.0 attached *per* SEI and BRI ratings (table 4). The pulley that is free to move represents mass of microhabitat DL2 with associated frictional force $Ø_F$ = 1.630. Friction was less than that of the ecosystem (i.e., 1.00), thus generating a positive impact upon force, causing the VCHI to increase from good to excellent.



Legend:

VCHI = Vegetative Complex Health Index.

Grey spheres = fixed pulleys for community structure & bio-invasion.

Red & green bars = relative weights for community level stratification (SEI) & bio-invasion (BRI) ratings (table 4).

Purple & red spheres = pulley free to move; colors correspond to VCHI ratings of fair & degraded, respectively (table 4).

Black outer rings indicate friction of Fisher's Disambiguous Coefficient ($Ø_F$).

Dotted grey line = chain attached to weights and associated with fixed and free pulleys.

Red & blue arrows indicate net effects of increased friction ($Ø_F$) upon chain weights and VCHI rating, respectively.

Figure 18: Negative Impact of the FDC

The three components of the Vegetative Complex Health Index as revealed *per* Newtonian mechanics via a three pulley system. Community stratification (SEI) and bio-invasion rates (BRI) are shown as fixed pulleys. The chain extending from each has relative weights of 4.0 and 0.0 attached *per* SEI and BRI ratings, respectively (table 4). The pulley that is free to move represents mass of microhabitat MO10 with associated frictional force $Ø_F = 0.390$. Friction was greater than that of the ecosystem (i.e., 1.00), thus generating a negative impact upon force, causing the VCHI to decrease from fair to degraded.

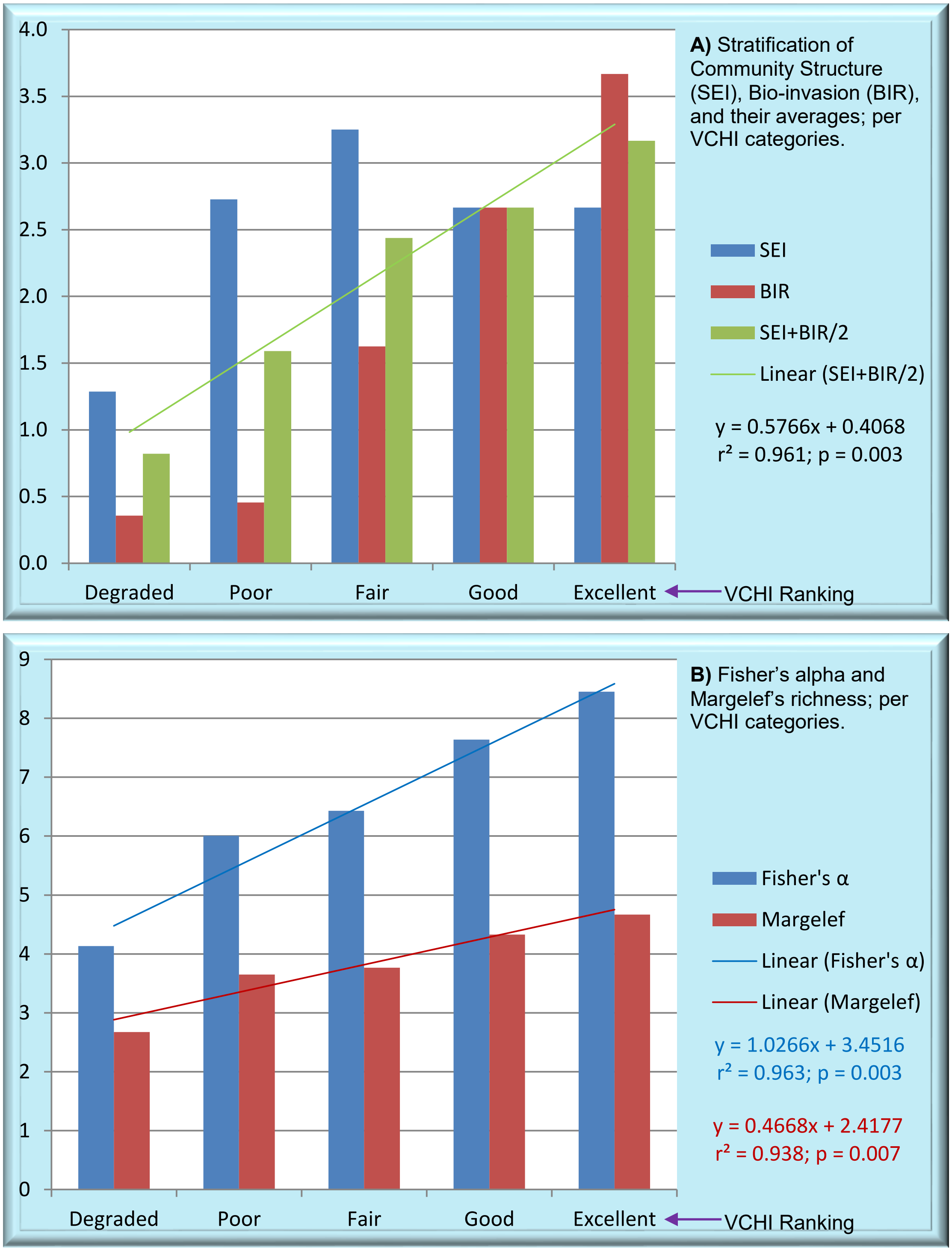


Figure 19: Least Squares Regressions of VCHI Hypotheses

Least squares regression revealed a highly discrete linear relationship for the mean of SEI + BRI (*Y* axis) *per* VCHI ranking (*X* axis) (A). Similar trends were exhibited for Fisher's alpha and Margelef's richness (B). Note how the reordering of microhabitats *per* the VCHI imparted linearity to total diversity and richness; whereas these variables were non-linear when ordered *per* the SEI (figure 3B).

Discussion

I. The Ecospheric Quantum Model:

The Theory of Quantum Microbiogeography was initially applied for identification of a *parsimonious quantum* of highly efficacious study sites (i.e., microhabitats as defined herein) with regard to quantification of ecologically relevant community types, species of interest, and mean alpha diversity (Sabaj-Stahl et al. 2013). Prior analyses revealed a relatively small and closely related set of microhabitats were necessary and sufficient for approximation of ecosystem parameters for these variables. It was speculated these outcomes were at least in part due to the *force of cohesion* occurring between the variables of community structure and alpha diversity, and that the energy of cohesion existed in and modulated a quantum state. Briefly, quantum states are defined as dualities that cannot be explained by classical (i.e., Newtonian) physics. Cohesion appeared to explain the quantum relationship occurring between alpha diversity and community structure.

Observations suggested microhabitats harboring more stratified communities were more closely related to one another than those with fewer stratified communities (Sabaj-Stahl et al. 2013). Relatedness appeared to be modulated by the force of cohesion occurring between diversity and structure. Furthermore, cohesion increased linearly with increasing stratification. Curiously, regression analyses suggested community structure was independent of alpha diversity, a condition initially described as the *McIsaac Paradox* (Sabaj-Stahl et al. 2013). Investigation of this paradox ultimately elucidated an apparent quantum duality of the ecosphere. The quantum mechanics of the ecosphere were modeled herein, in a manner somewhat consistent with quantum mechanics of the atom. Understand these models are conceptual in nature and do not communicate the level of information contained in atomic quantum models. Rather, the modeling exercise was performed to a) communicate the nature of the quantum relationship using an existing (atomic) quantum model and b) to have a model upon which to build a functional, mechanical and predictive composite ecometric assessing ecospheric health.

Development of the quantum model permitted identification of apparent quantum-dependent and -independent processes of bio-invasion, coupling and decoupling of community structure, and integration of quantum–dependent and –independent variables (community structure, alpha diversity and bio-invasion) for derivation of the Vegetative Complex Health Index (VCHI). The VCHI appeared to objectively evaluate the health of microhabitats and ecosystems on the basis of these three indices. Figures 3A, 4, 5 and 7B, and table 3 were reproduced from Sabaj-Stahl et al. (2013) for purposes of clarity and continuity concerning development of the quantum model of the ecosphere.

The Quantum Relationship

SHE analysis (Figure 3A) was suggestive of a log normal species distribution and of a community demonstrating the least evenness (Hayek & Buzas 1997). The SHE plot revealed evenness plateaued after the sampling of just 12, 80 x 50 m microhabitats utilizing the point intercept protocol as described. Richness and total diversity continued to increase beyond the 39 sites assessed, but declined in rate of increase after the sampling of 26 microhabitats. Plant communities were dominated by two invasive species (*Tamarix chinensis* and *Acroptilin repens*). About thirteen species were intermediate in occurrence, with the remainder relatively rare. The ranked species list for the study was published in Sabaj-Stahl et al. (2013).

Figure 3B displays trend lines, with related r^2 values, for regressions of mean evenness, richness and total diversity with community structure. Community structure was characterized using the Sabaj Efficacy Index (SEI) (Sabaj-Stahl et al. 2013). The SEI was defined as the proportion of community types (native, exotic, woody and herbaceous) adherent to 25% confidence interval estimates using point-in-time parameter estimates (Elzinga et al., 2001). Thus, communities were assumed stratified (i.e., to have formed uniform bands of vegetation parallel to channel banks) when data were sufficient to estimate 25% or greater change in percent cover estimates. Given five, 50 m transects were placed randomly within 16 m intervals perpendicular to channel banks, *per* microhabitat, 25% confidence intervals were assumed sufficient to detect formation of uniform bands of native, exotic, woody and herbaceous communities.

Trend lines of these regressions evidenced marginal line slopes with relatively little variation in the models explained by the relationships of evenness, richness and total diversity with community structure, *per* coefficients of determination. Figure 4 displays linear regression plots using Pielou's J (Shannon's equitability), sample size independent Margalef's richness, Berger Parker's dominance, and Shannon Weiner's diversity; regressed with community structure. Rather than applying system means of evenness, richness and diversity (as in figure 3B), estimates from each of the 39 microhabitats were used for these regressions. Again, line slopes and r^2 values were marginal, with p values for one tailed t-tests insignificant ($p > 0.05$). The exceptionally small p values for two tailed t-tests were likely a consequence of a large data set with unequal variances, and were discussed at length in Sabaj-Stahl et al. (2013). The apparent conundrum of independence of diversity from structure was probed in depth within the guise of the *McIsaac Paradox*. In a nutshell, the paradox appeared to exist in the apparent independence of diversity from structure, while the relationship of diversity with structure seemed to account for degrees of microhabitat relatedness (Sabaj-Stahl et al. 2013).

The notion of independence is counter-intuitive, given diversity and structure were derived from identical data (i.e., percent cover estimates of species). Simple inspection of principle components analysis (PCA) scatter plots (Figure 5) indicated linear regression analyses were insufficient for detection of another form of dependency between these variables. Specifically, the ranges of richness, evenness, dominance and total diversity were apparently restricted with increasing stratification of community structure. This effect was also apparent in the standard error bars displayed in Figure 3B. In effect, the variances of richness, evenness, dominance and total diversity were reduced as community structure increased, while their respective means were preserved across a spectrum of microhabitats with divergent stratification. The relationship of diversity with structure, and associated cohesive energy served as the basis of the quantum models.

Minimum line distances of PCA matrices suggested microhabitats became more closely related to one another on the basis of diversity and structure, as structure became more ordered (Figure 5). Thus, community stratification not only resulted in increased order (i.e., decreased entropy) of structure, but an increased order of diversity. Elimination of microhabitats from the extremes of distributions for evenness, dominance, richness and total diversity among the most stratified of microhabitats represented an increase in order (reduced variability) of diversity. This was not an especially surprising outcome because of a tendency for communities to become less even as they formed distinct bands parallel to channel banks. So for instance, woody-exotic communities were dominated by salt cedar (*Tamarix chinensis*); herbaceous-exotic communities by Russian knapweed (*Acroptilin repens*); and woody-native communities by sandbar willow (*Salix exigua*) and New Mexico privet (*Forestiera neomexicana*) (Sabaj-Stahl et al. 2013). However, the preservation of mean diversity accompanied by reductions in variance with increased structure was not anticipated.

Ultimately this effect was judged a consequence of the patterns revealed by evenness and dominance (Sabaj-Stahl et al. 2013). In this context, dominance was treated as the compliment (i.e., the mirror image) of evenness. The PCA scatter plots (Figures 5B, C) supported that supposition, with dominance appearing as the “left hand” of evenness, much in the manner of a molecular enantiomer (Schmidt et al. 2012). Evenness was thought to be the culprit concerning the relationship of total diversity with community structure on two grounds: a) Evenness plateaued after the sampling of just 12 microhabitats, while richness and total diversity continued to increase after all 39 sites were sampled (SHE analysis, Figure 3A); and b) The “outlier” observed among the PCA scatter plots (Figure 5) for the most efficacious of microhabitats for richness and total diversity did not behave as an outlier in the PCA scatter plots for evenness and dominance.

That particular data point (microhabitat MO10) was fortuitous in explaining these outcomes. Unlike the other 38 sites in the study, MO10 had all salt cedar removed by chainsaw one year prior to monitoring of the site. Resprouting of salt cedar shoots was robust in this microhabitat. However, while removal of the salt cedar canopy, by comparison to other microhabitats in this diversity-structural pairing appeared to facilitate significant changes in richness and total diversity, mean evenness remained constant within the context of all other microhabitats in the most stratified group. Thus the behavior of evenness relative to community level stratification was considered the primary influence causing the range and variance of total diversity to shrink while its mean remained constant with increasing stratification of community structure. Within these contexts, simple regression outcomes and PCA scatter plots supported the assertion of evenness serving as the *driver*, community structure the *passenger*, and cohesion the *fuel* in the relationship occurring between diversity and structure.

Quinn Cohesion & Quantum Numbers

These observations compelled further analysis of the relationship occurring between diversity and structure. A variety of approaches, including but not limited to ANOVA and MANOVA (data not shown), did not appear to generate outcomes providing more information within this context. In an attempt to further probe the apparent issue of relatedness among microhabitats, the refined Quinn Cohesion Index (r-QCI) was developed to assess microhabitat relatedness on the basis of diversity (specifically, evenness) and stratification of community structure (table 1; Figures 7A and 7B) (Sabaj-Stahl et al. 2013). The r-QCI assessed microhabitat relatedness in a manner not entirely unlike parsimony analysis of molecular data to construct phylogenies, albeit far more simplistic than molecular approaches (Steel & Penny 2000; Heinrichs et al. 2013).

The refined Quinn Quotient (r-QQ) was derived from PCA correlation matrices, using those indices shown to a) have linear relationships with efficacy *per* linear regression, *per* the two-tailed t-test, and b) contribute significant information *per* the Akaike Relative Likelihood Method (Sabaj-Stahl et al. 2013). Simpson's dominance, Berger Parker's dominance, Buzas Gibson's evenness and Pielou's J met these criteria (table 1). When the raw and averaged derivatives (i.e., three point moving averages) of the r-QQ were plotted with community structure (Figures 7A and 7B, respectively), significant linear relationships ($p < 0.05$) were revealed. Furthermore, the degree of variation explained in both models by coefficients of determination was noteworthy, with 81% and 99% for the raw and averaged cohesion indices, respectively. Thus it appeared the general patterns evidenced in the PCA scatter plots were indicative of increasing microhabitat relatedness on the basis of evenness and community structure.

The behavior revealed by the r-QCI was identified as the source of the outcome wherein a *parsimonious quantum* of highly efficacious study sites (i.e., microhabitats as defined herein) yielded quantification of ecologically relevant community types, species of interest, and mean alpha diversity (Sabaj-Stahl et al. 2013). It was felt at the time, based upon the outcomes observed for the r-QCI, that a quantum relationship existed within this rubric. Specifically, the r-QCI evidenced a source of energy occurring between diversity and community structure at the level of the microhabitat and ecosystem. In short, as communities became more organized via the process of stratification, and the range/ variance of diversity was concurrently constrained, an innate energy force maintained higher levels of organization (i.e., decreased entropy) within microhabitats and ecosystems.

A search ensued to identify an atomic or molecular model that could be applied as a template to model the quantum ecosphere. None were judged sufficient for these purposes, although the carbon atom and carbon-hydrogen molecular series came close to approximating relationships observed in these data. As such these were used only as points of reference, and ecologic data were modeled in a manner consistent with the trends they evidenced. Figure 6 illustrates Bohr's model of the atom, using potassium as an example. While since shown to be insufficient for describing many atomic structures, Bohr's model remains useful for modeling the quantum states of atoms like hydrogen and lithium (Bohr: 1913a, 1913b & 1913c). The essential principles of Bohr's model pertaining to atomic nuclei, atomic orbitals, potential energies, and electrons were preserved in elucidating quantum models of the ecosphere.

Figure 7A illustrates the r-QCI and suggested the potential energies of microhabitats were linear in nature, but not perfectly so. For those microhabitats where one, two or three community types became stratified, these energies were approximately equal. At first glance that outcome did not seem congruent with quantum mechanics, as discrete and relatively linear changes in potential energies occur when electrons jump from one atomic orbital to another (Einstein 1917; Bohr 1921). However, community arrangements represented two levels of *independence* and four levels of *overlap*. Woody is independent of herbaceous, and native independent of exotic. Woody and herbaceous communities overlap with native and exotic species. Thus, when moving from a microhabitat with no stratified communities to a microhabitat containing one stratified community, one level of independence is attained. Alternatively, when transitioning from a microhabitat with one stratified community to a microhabitat with two or three, it is not possible to attain another level of independence. Two levels of independence are only realized for microhabitats containing four stratified communities, wherein woody is independent of herbaceous, and native independent of exotic. Thus the changes observed in quantum energy states appeared to be a function of transitions to and from levels of independence. Addition or subtraction of overlapping communities did not appear to alter potential energies of microhabitats.

Distance, or dissimilarity (Euclidian paired group cluster analysis), and similarity (Jaccard paired group cluster analysis) were applied to further assess potential trends of relatedness among microhabitats (Figure 8). In each of these analyses, microhabitats devoid of stratification appeared least related to all other microhabitats. This observation was especially compelling with regard to similarity, insomuch as it suggested these microhabitats no longer behaved in a manner consistent with riparian ecosystems. On the one hand, their inability to maintain uniform bands of vegetation parallel to channel banks implied they were not riparian in nature. On the other, because they harbored drought intolerant species excluded from the uplands, they possessed traits consistent with riparian ecosystems. Encroachment of upland species into these microhabitats also suggested they bore attributes consistent with upland ecosystems. Thus, in the modeling the quantum ecosphere, these microhabitats were relegated to the "Strohmosphere" (so named for David Strohmeyer, PhD; ecologist; retired). In this sense they are analogs of valence shell electrons shared in covalent bonds between atoms (Tsuchida & Kobayashi 1939; Gillespie & Nyholm 1957). The analogy is not perfect insomuch as valence shell electrons possess the highest potential energy compared to all others surrounding an atom, while microhabitats of the Strohmosphere have the lowest potential energy compared to all others in the ecospheric quantum model. These microhabitats were modeled as belonging to neither riparian nor upland ecosystems, but shared based upon characteristics inherent to each.

Euclidian distance analysis further elucidated relatedness on the basis of diversity and structure. Microhabitats containing one or two stratified communities were more closely related to one another than other microhabitats. The same was observed for microhabitats with three or four stratified communities. Furthermore, microhabitats containing one or two stratified communities were more closely related to each other than any microhabitat was related to any other. Lesser organized microhabitats appeared more related to one another than those more organized due to a greater tendency for the herbaceous community to become stratified among microhabitats with one or two stratified communities (table 3). Thus the range of evenness was highly constrained among microhabitats with one stratified community due to most of these being herbaceous (Figure 5). Jaccard similarity analysis was not as discerning of these organizational trends, but nonetheless illustrated microhabitats with one or more stratified communities were more closely related to one another than those evidencing no stratification.

Quantum numbers were assigned to microhabitats based upon outcomes of the r-QCI (table 2). A quantum duality was observed in the relationship of diversity with community structure. Specifically, evenness was *uneven* in its association with structure. The mean of evenness was conserved across stratification of community structure, but the range was not. The simultaneous dependence and independence of evenness with structure caused diversity to exhibit similar trends with structure. *Verschränkung*, or entanglement, coined by Erwin Schrödinger, was applied "to describe the correlations between two particles that interact and then separate, as in the EPR experiment" (Schrödinger 1935). As with the EPR Paradox of quantum physics (Einstein, Podolsky & Rosen 1935), the McIsacc Paradox (Sabaj-Stahl et al. 2013) speculated an unknown phenomenon accounted for the duality observed in the relationship of diversity with community structure.

Quantum entanglement occurs when pairs of particles interact in ways such that the quantum state of each cannot be described independently. Rather, a quantum state is said to exist for the system as a whole (Schrödinger & Dirac 1936). Furthermore, entanglement describes the condition wherein particles experience both interaction and separation (Bell 1964). The relationship occurring between community structure and diversity can only be understood in terms of the differential associations of evenness with structure. This effect is analogous to quantum entanglement, insomuch as we cannot state the relationship is dependent or independent, but both. The ecosystem is said to exist in a quantum state, as it cannot be understood solely on the basis of dependence or independence of evenness with structure. It can only be understood as a coexistence of both states; of simultaneous dependence and independence.

A duality is therefore established that is analogous to the wave-particle duality of light (Schrödinger 1928). As Einstein lamented upon the paradox of the wave-particle duality of matter, he opined, “It seems as though we must use sometimes the one theory and sometimes the other, while at times we may use either. We have two contradictory pictures of reality; separately neither of them fully explains the phenomena of light, but together they do.” (Einstein & Infeld 1938) The equivalent is said to exist between diversity and structure: they are simultaneously dependent and independent of one another, interacting and separating in a manner that cannot be aptly described by either dependence or independence, but only by both.

Evidence for the proposed quantum state occurring between evenness and structure was provided by the r-QCI. Averaged r-QCI values appeared to reveal discrete energy states for specific evenness/ structural pairings. In fact, as indicated per r-, r^2- and t-values (with associated p value), a nearly perfect linear relationship of these energies was revealed concerning transitions from one specific evenness/ community structure pairing to another. Quantum mechanical systems are constrained to discrete, quantized energies (Einstein 1917; Bohr 1921). Furthermore, ground states are defined as the lowest possible energy level (Bohr 1921). When multiple quantum mechanical configurations are defined for equivalent quantized energies, they are described as degenerate systems (Atkins 1974). These essential features of quantum atomic systems were modeled at the level of the ecosphere.

In quantum mechanics of atomic systems, the principle quantum number (*n*) of ground state energies is set at one (Dirac 1982). The ground state of the ecospheric system was also set at one, and occurred within microhabitats evidencing no stratification of community structure (table 2). Regression of the raw r-QQ with community structure revealed three discrete, quantized energy states, even though five levels of community structure were applied (Figure 7A). Utilizing the trend line produced from this regression, quantized energy states were identified for structural groups of 0.00, 0.25 through 0.75, and 1.00. Adjusted r-QCI was simply the Quinn Quotient value (*Y*-axis) occurring on the trend line in the regression with community structure (*X*-axis) for structural groups of 0.00, 0.50 & 1.00. The structural group of 0.50 was selected to represent the three groups of 0.25, 0.50 & 0.75, as they exhibited equivalent r-QCI values.

These outcomes were assumed a function of levels of independence and not overlap of community structure. Intermediate states of community structure, wherein one, two or three communities were stratified, are equivalent to degenerative systems observed in atomic quantum models. As such, three quantum numbers (1, 2 and 3) were assigned and correspond to zero, one and two levels of independence occurring between stratified communities within microhabitats. The ratio of discrete energies occurring relative to these quantum numbers was 1:3.75:6.50 (table 2). The less than perfect linear relationship among quantized energies apparently arose from the loss of 0.25 units of potential energy *per* increase in non-degenerative orbit *per* microhabitat. These essential relationships concerning community structure and diversity were applied to develop conceptual models of the ecosphere not entirely unlike atomic models of quantum physics. However, because differences exist in their respective quantum relationships, these data were modeled in a manner consistent with the observed trends.

The Quantum Ecosphere

The quantum ecospheric models (Figures 9 through 13) were based upon several key concepts. Microhabitats were viewed as the equivalent of electrons in atomic models and were color-coded to reflect the degree of community stratification occurring within them. Figure 9 illustrates quantum number and associated quantized energy of r-QCI increased with increasing stratification of community structure. Orbital classification was also introduced and is patterned after atomic orbitals, in part to communicate quantized energy states, utilizing the s, p, sp and d configurations (Heisenberg: 1925, 1927). The range of diversity decreased with increasing structure, while the mean of diversity remained constant. This duality elucidated the quantum mechanics of the ecosphere.

Figure 10 introduces Harriman orbitals, orbital lobes and the Paulson Particle. The term orbital, rather than orbit, was selected to reflect that microhabitats do not necessarily follow a specific trajectory, as would a planet orbiting a star. This is an important distinction, for these data were insufficient to project the path any microhabitat might follow concerning future trends of diversity and structure. Harriman orbitals were named in honor of Neil Harriman, PhD (botanist emeritus) and the Paulson Particle for James Paulson, PhD (distinguished professor & biochemist). Harriman orbitals were depicted as ring structures (Figures 10 and 13) and clouds (Figures 11 and 12). Models depict orbital configurations wherein native v. exotic and woody v. herbaceous communities are independent of one another. Orbital overlap is shown where appropriate.

Adopting conventions used to describe electron orbitals, in part to reflect the quantized energy of Quinn Cohesion, these were defined as s, p, sp and d configurations (Heisenberg: 1925, 1927). Primary (s) orbitals were populated by microhabitats exhibiting primary structure. Intersections of orbital lobes occurred in primary and secondary phases. Primary orbital intersections produced secondary (p) orbital paths occupied by microhabitats with secondary structure. Primary and secondary orbital overlap generated hybrid tertiary (sp) orbitals populated by microhabitats with tertiary structure. Lastly, intersection of secondary orbitals yielded the quarternary (d) orbital containing microhabitats with quarternary structure. Some orbital arrangements were degenerate as their associated microhabitats possessed equivalent potential energies and quantum number. The s orbital configuration corresponded to microhabitats wherein one of four community types were stratified. Of these, there were four degenerative subtypes (native, exotic, woody or herbaceous), all of which were assigned a quantum number of two. As such, these were identified as $2s_n$, $2s_e$, $2s_w$ and $2s_h$. Orbitals for microhabitats containing two or three stratified communities were also degenerate in nature, as the quantum number remained at two, but their configurations were different from that of the s orbital.

Microhabitats evidencing two stratified communities occupied p orbitals, characterized with two lobes, with each representing one stratified community. These were identified as $2p_{ne}$, $2p_{nw}$, $2p_{nh}$, $2p_{ew}$, $2p_{eh}$ and $2p_{wh}$. As is the case with electron orbitals, the three-lobe system herein is modeled as a hybrid consisting of an s and p orbital. These are degenerative orbitals as determined by the r-QCI and were assigned the quantum number of two. The sp hybrid consisted of four possible degenerative states: $2sp_{new}$, $2sp_{neh}$, $2sp_{nwh}$ and $2sp_{ewh}$. Last but not least, there was but one d orbital configuration consisting of four lobes, which was not degenerate and was assigned the quantum number of three: $3d_{newh}$. The system as devised enables investigators to classify microhabitats on the basis of community structure and the potential energy occurring between diversity and structure. While that might seem somewhat trivial for the moment, it shall be revealed quantum-dependent and -independent aspects of the ecosphere required nuanced approaches to yield a composite and objective ecospheric health metric.

The Paulson Particle (Figures 9 through13) is a hypothetical structure equivalent to the photon in quantum physics or the qubit in quantum computer theory. In each of these systems, the photon and bit exist as a quantum superposition of two states: vertical and horizontal polarization of light, and the 0 and 1 states of the classical computer bit (Können 1985; Nizovtsev 2005). The particle represents the superposition of independent and dependent states of diversity with structure. Specifically, this quantum duality was described as the simultaneous independence and dependence of evenness with community structure (Sabaj-Stahl et al. 2013). This dynamic appeared to be modulated by the force of Quinn Cohesion. Figure 12 illustrates diversity, structure and the quantized energy of cohesion as three axes converging at the Paulson Particle. The particle is illustrated in a manner analogous to atomic nuclei. In general, increases in Quinn Cohesion were correlated with increased organization and decreased entropy. As potential energy of microhabitats increased, community stratification increased, and the range of diversity decreased (Figure 9). Thus, microhabitats exhibiting high degrees of community stratification were constrained about the mean of diversity. Reduced variability of diversity with increasing structure corresponds to decreased entropy, and is maintained by the potential energy of cohesion.

Figures 11, 12 and 13 model the Strohmosphere, a sort of *no man's land* for lack of a better description. The Strohmosphere is the region occupied by microhabitats displaying no community structure as defined. *As defined* is an important caveat, because these microhabitats were not devoid of structure. The absence of structure, as defined, was a failure to maintain uniform bands of vegetation parallel to channel banks within parameters established for 25% confidence interval estimates for native, exotic, woody & herbaceous communities. Thus, these microhabitats appeared to lack the structure typical of riparian habitats and were in some ways more representative of upland habitats. However, because they still retained many drought intolerant species exclusive of riparian ecosystems, they were not appearing to belong to the upland ecosystem either. These microhabitats were relegated to the Strohmosphere and assigned the ground state quantum number of one. It is believed this condition arose from impoundment of the river and diversion of 40 to 50% of surface water annually for irrigation of croplands (Richards & Leib 2011). Reduction of the groundwater table, coupled with reductions in the degree and frequency of overbank flooding, were assumed to alter soil water profiles to an extent sufficient to obliterate community structural dynamics consistent with riparian ecosystems (Gregory et al. 1991; Chambers & Miller 2004; Mellman-Brown, Roberts & Pugesek 2008; Tercek, Stottlemyer & Renkin 2010; Chambers & Miller 2011). However that assumption was not directly assessed, as the groundwater table, and overbank flood delineation and frequency measurements were not conducted in this study.

The quantum models in of themselves are conceptual in nature, and no particular model is assumed to represent the actual quantum state of the ecosphere. These models are best viewed as initial attempts to conceptualize quantum relationships at the level of the ecosphere. A tremendous amount of information simply remains unavailable and precludes any attempt to fully characterize the quantum state of any ecosystem. While the concepts of diversity and community structure are two fundamental cornerstones of ecology, it is unknown if the quantum nature of their interaction is universal. Many hurdles remain in terms of devising approaches to probe this interaction in other botanic systems, much less other trophic levels. It is also likely other ecosystem variables behave in a quantum manner. To say the least, The Theory of Quantum Microbiogeography is very much in its infancy.

Much remains to be discovered in terms of extending this quantized classification system across a spectrum of variables. In restricting analyses to just four ecologically relevant community types, the potential for elucidating Quinn Cohesion energy states (and by extension, orbitals and quantum numbers) was also restricted. However some essential observations are worthy at this juncture. By definition, each and every community shall have an *anti-community*, as it were. This condition is somewhat analogous to the spin and anti-spin configurations of electrons (Heisenberg 1926). Native, for example, is the antithesis of exotic. It also appears the quantized states of Quinn Cohesion hinge upon antithesis (i.e., independence) of communities, as community overlap did not alter potential energies of microhabitats. While there potentially exist an infinity of community assemblages, inherent constraints would seem to limit these possibilities.

An emphasis was given to *ecologically relevant* communities. Community definitions are somewhat arbitrary and artificial in nature, at least in comparison to populations (Palmer & White 1994; Mucina 1997). The applied classification system was adherent to floristic community concepts, which arise from vegetation plot data (Didita, Nemomissa & Gole 2010). Primary vegetation assemblages (native, exotic, woody and herbaceous) were consistent with sinusiae, allowing for biogeographic assessments concerning community stratification (Gleason 1936). These assemblages were dominated by characteristic floristic taxa, such as *Tamarix chinensis* and *Acroptilin repens* of the exotic community. As such, the potential number of floristic community arrangements is distinctly finite, as limited by not only the number of species occurring in an ecosystem, but the subset of species that define characteristic floristic communities.

Ecological relevance was defined as a community assemblage (group of species) of significant impact upon the function of other such relevant communities. Thus, for example, uniform bands of woody vegetation parallel to channel banks provide critical nesting habitat for any number of bird species (Carothers, Johnson & Aitchison 1974;

Ohmart & Anderson 1982; Croonquist & Brookes 1993; Finch & Stoleson 2000). Absent these woody species, many associated bird species would be extirpated from these microhabitats. Furthermore, edge habitat and the associated *edge effect* formed by abutment of distinct vegetative communities, bankside vegetation with emergent vegetation, and littoral and pelagic zones of lakes are crucial for maintenance of biodiversity (Sweeney 1993; Levin 2009; Vadeboncoeur, McIntyre & Vander Zanden 2011). Thus, ecological relevance was also imparted to woody and herbaceous riparian communities due to formation of edge habitat where these communities were stratified.

Conversely, edge habitat arising from unnatural disturbance such as agriculture can contribute to decreases in biodiversity (Murci, 1995). These disturbances have the effect of fragmenting landscapes and habitats, facilitating bio-invasion, and reducing relative habitat patch size while increasing edge habitat (Adams & Dove 1989; Saunders, Hobbs & Margules 1991; Rosenzweig 1995; Fahrig 2003). As such, ecological relevance was also associated with the community types of native and exotic, as transitions to edges dominated by invasive species appears to reduce biodiversity. Edge habitat is also created by the overlap of riparian and upland ecosystems, wherein species associated with each intermix and give rise to an ecotone (Clements 1905). Ecotones contribute significantly to maintenance of biodiversity, especially concerning species dependent upon habitats from both ecosystems (Hansen & diCastri 1992). Ecotones also harbor unique species that are restricted to the region of ecosystem overlap (Risser 1995). Thus, the interface of riparian and upland ecosystems relative to the four communities assessed also imparts ecological relevance.

The potential maximum occurrence of community types within a microhabitat or ecosystem relative to stratification parameters is also limited by factors of space, competition, energy flow and the trophic pyramid structure of ecosystems (Odum 1971; Chapin, Matson & Mooney 2002; Begon, Townsend & Harper 2006; Schoener 2009). Within the context of riparian ecosystems, community stratification was defined as the formation of uniform bands of vegetation parallel to channel banks. Communities adherent to 25% confidence interval limits were assumed to have formed sinusiae. Within the context of this community definition, limited space, matter and energy within microhabitats and ecosystems precludes the occurrence of all potential ecologically relevant communities achieving stratification standards. In essence, there must exist some minimum critical mass of any species assemblage to achieve the minimum detection limits established for any methodology. In general, the highest and lowest trophic levels will contain fewer stratified communities compared to intermediate trophic levels due to the interaction of richness and productivity, although this relationship is by no means universal (Naeem & Lee 1998; Yee et al. 2007; Savada et al. 2011).

Another aspect of deserving of attention concerns diversity in of itself. Not long after Sabaj-Stahl et al. (2013) was released, an investigator contacted the Edwin James Society and asked if the approaches developed for identification of a parsimonious quantum of highly efficacious study sites, relative to riparian plant communities, could be directly applied to a small group of mammals consisting of four to five species over a broad geographic range. The answer, simply, was no. Given the system hinges upon the relationship of diversity with community structure, there was no means by which to augment these approaches to an extent where quantification was possible for either variable. Thus it shall be important for investigators working with divergent systems to carefully design studies such that enough species, occurring with sufficient frequency, can be assessed using these approaches. Our suggestion to this investigator was to consider augmenting his approaches within the context of plant community dynamics. Occurrences of small mammals are often associated with preferred vegetation types such as relative cover density, overstory composition, grassland dynamics or decaying organic debris (Whitaker & Maser 1976; Doyle 1990; Peffer 2001). Thus by honing in on specific behaviors of foraging, cover and reproduction of small mammals, related plant community constructs could permit use of these approaches within acceptable limits of quantification.

Although unresolved issues persist concerning these models, the approaches and concepts appear to hold the potential for applied uses relative to the processes of disturbance, bio-invasion and ecologic restoration. Thus attempts were undertaken to implement these concepts concerning assessments of disturbance, bio-invasion, ecologic function and ecologic restoration.

II. Patterns of Structure & Ecospheric Disease

Data were initially applied in a manner consistent with the approach advocated for by Shipley and Keddy (1987). Specifically, species and community level data were not applied to initially test any causal hypotheses *per se*, but instead biogeographic patterns were assessed to deduce mechanisms of community organization relative to disturbance and bio-invasion. Then, hypotheses were developed relative to those observations. While pattern to process inferences have not always panned out especially well in community ecology (Gilbert & Bennett 2010), the strength of the correlations observed among these patterns were compelling enough to warrant mechanistic inferences. Those inferences were then tested using a set of *a priori* hypotheses based upon pattern analyses. The approach seemed fruitful, as pattern assessment in of itself elucidated a heretofore undescribed feature of biodiversity relative bio-invasion, and apparently independent of percent cover.

A number of theories have been proposed concerning the mechanics of community organization. Those theories were not necessarily tested nor directly assessed via the current work, however some elements of those existing models were apparent. The nonequilibrium model suggests communities are in constant flux due to regular disturbance events (Chesson and Case 1986; Huston 1994). Riparian ecosystems often serve as good examples of nonequilibrium theory, as overbank flooding tends to uproot vegetation and continually create opportunities for establishment of new flora and floristic communities. Erosion and sedimentation also contribute to nonequilibrium processes, as entire banks can be washed out while sedimentation creates new embankments and islands ripe for colonization. The situation was quite the opposite on the lower Dolores watershed, as impoundment of the river and diversions of surface water eliminated many of the processes attributed to nonequilibrium. Thus, the McPhee dam presented another kind of disturbance and the biogeographical patterns observed were in part attributed to alterations in hydrology.

The keystone theory of community organization proposes a given species has a disproportionate impact upon the system it inhabits, relative to its abundance, to the extent of maintaining community structure (Paine 1995; Barua 2011). While not necessarily a keystone species, as it is not indigenous and does not entirely depend upon the natural environment for survival and reproduction, livestock were a significant component of the lower Dolores watershed. Most of the lands throughout the watershed were regulated under United States Department of the Interior grazing leases, allowing ranchers to graze cattle. Additionally, several large private ranches intersected riparian zones throughout the watershed. Thus in many ways cattle behaved as a keystone species insomuch as they exerted an obvious impact on floral communities. Those impacts were assessed relative to bio-invasion and community structure.

The integrated hypothesis of community structure envisions communities as collections of closely associated species bound by requisite interactions (Clements 1916; Connell & Slatyer 1977). The integrated hypothesis views communities as integrated and functional superorganisms, much along the lines of the Gaia hypothesis (Lovelock & Margulis 1974). The manner in which coupling and decoupling of community structure were assessed and discussed was somewhat in line with the concept of the superorganism. So too were the medicinal approaches adopted to assess and diagnose species-based and community-based ecospheric disease processes. Whether or not communities actually function as superorganisms remains open to debate, however the philosophical underpinnings were useful in elucidating new frontiers of The Theory of Quantum Microbiogeography. Among other things, quantum microbiogeography provides a hypothesis concerning community organization (Sabaj-Stahl et al. 2013). At the crux of it, quantum microbiogeography predicts the relationship occurring between diversity and community structure, and the "organizing energy" associated with that relationship, are responsible for the patterning of community organization and, more specifically, stratification. Philosophically, the quantum microbiogeographical hypothesis views top-down and bottom-up forces acting directly upon biodiversity and secondarily upon community structure, because community structural patterns are a direct consequence of the quantum interaction occurring between biodiversity and community structure.

Stratification in of itself provided clues concerning the health of ecosystems. G.E. Hutchinson once remarked, "The structure which results from the distributions of organisms in, or from, their interactions with, their environments, will be called pattern" (Hutchinson 1953). Hutchinson asserted patterns of structure were fundamental to understanding ecological processes, and it was this philosophy that led to his development of the niche concept and competitive exclusion principle (Hardin 1960; Holt 2009). The patterning of structure, as defined herein, involved a two-step process: assessment of vegetative communities within the context of 25% confidence interval estimates, and development of the Sabaj Efficacy Index as a means to characterize those patterns. Community structural dynamics were then applied to identify specific processes of bio-invasion, characterize ecospheric disease sequelae and related syndromes, and develop a composite ecometric reflective of the mechanics of ecosystem function.

Community Based Disorders of the Ecosphere

The absence of multi-layered stratification of community structure was considered an indication that riparian microhabitats degraded (i.e., had become *decoupled*) to an extent that they no longer functioned within the context of healthy riparian ecosystems. Concerning the lower Dolores watershed, the decoupling of community structure was primarily suspected a consequence of alteration to the bottom-up forces of hydrology and geomorphology (Scott, Wondzell & Auble 1993; Décamps, Planty-Tabacchi &Tabacchi 1995; Hupp & Osterkamp 1996 Naiman & Décamps 1997; Shafroth, Stromberg & Patten 2002; Bagstad, Lite & Stromberg 2006). Hydrology and geomorphology serve as the primary drivers shaping vegetative communities of riparian ecosystems (Rood & Mahoney 1990; Nilsson, Gardfjell & Grelsson 1991; Middleton 1999; Jansson et al. 2000; Nilsson & Berggren 2000; Nilsson & Svedmark 2002). While hydrology was not directly assessed in this study, impoundment of the river, coupled with diversions for irrigation obviously altered ecologic function of the watershed (Décamps et al. 1988; Rood & Mahoney 1990; Smith et al. 1991; Horton, Kolb & Hart 2001; Tickner et al. 2001; Azami, Suzuki & Toki 2004).

With 40 to 50% of Dolores River surface water diverted annually for irrigation, and obliteration of pre-impoundment peak and low river flows, optimal function of the riparian ecosystem was undoubtedly impaired (Richards & Leib 2011). Depleted groundwater tables, coupled with reductions in the frequency and extent of overbank flooding appeared to radically alter community structure (Stromberg, Tiller & Richter

1996; Merritt & Cooper 2000; Richardson et al. 2007). In fact, the nearly ubiquitous invasion of salt cedar within the lower Dolores watershed was a direct consequence of alterations to hydrology and geomorphology (Horton, Mounts & Kraft 1960; Fenner, Brady & Patton 1984; Howe & Knopf 1991; Taylor, Wester & Smith 1999; Bhattacharjee et al. 2009). We describe this process generally as *community decoupling*, evidenced by increases in disorder/ entropy of community structure and biodiversity, and reductions in the potential energy of cohesion. It was suspected soil moisture profiles within these degraded microhabitats were more representative of upland ecosystems, as reflected by their inability to sustain uniform bands of vegetation parallel to channel banks (Swanson et al. 1982; Merritt & Cooper 2000; Horton, Kolb & Hart 2001). It has been well established that groundwater tables and soil moisture profiles are largely responsible for vegetation patterns characteristic of riparian ecosystems (Waring & Major 1964; Griffin 1967; Prichard 1998; Stallins 2001; Chalmers et al. 2012; Stockan, Langan & Young 2012). Microhabitats devoid of community structure required for edge habitat were characterized as suffering from **Ecospheric Late-Stage Disease** (ELSD).

The hallmark of ELSD is therefore abolishment of historically stable community structural patterns. Although pre-impoundment community level data were not available for the lower Dolores watershed for means of comparison, it was readily apparent the floodplain had been significantly reduced following impoundment. In fact, elevation gradients defining the historic secondary floodplain suggested these regions no longer experienced inundation, as woody debris caches were not observed in the secondary floodplains of any microhabitats. Channelization of several river reaches, coupled with increased bank elevation caused by salt cedar mounds, appeared to prevent inundation of several primary floodplain reaches (Steiger, James & Gazelle 1998; Wohl 2004; Cohn 2005).

Because microhabitat assessments were conducted within primary and secondary floodplains, the absence of edge habitat was considered symptomatic of ELSD. Note however this process is distinguished from obliteration of structure caused by the natural event of overbank flooding. For riparian ecosystems, ELSD sequelae include perturbations to hydrologic regimes with associated loss of characteristic banding patterns of woody *and* herbaceous communities. These disruptions occur between woody and herbaceous patches within the riparian ecosystem and in the ecotone occurring between the riparian and upland ecosystems. The process is described as a diseased state because the associated loss of edge habitat results in disruptions to connectivity and spatio-temporal community dynamics correlated with decreases in biodiversity (Ward, Tockner & Schiemer 1999; Liu & Cui 2009). These disturbances can also cause ecotones to become reservoirs and conduits of invasive species (Lloyd et al. 2000; Ren et al. 2014).

Microhabitats diagnosed with ELSD (table 3) harbored woody and/ or herbaceous communities exhibiting confidence interval estimates yielding transect numbers greater than six. These are indicated with the study site designation code highlighted in dark grey. Six, rather than five transects were used as a cut-off to provide latitude for those microhabitats where perforations to bands of vegetation gave rise to higher confidence interval estimates. The reason for including microhabitats in the ELSD group where *either* woody *or* herbaceous communities adhered to these confidence interval standards is that the presence of just one uniform band of vegetation did not allow for formation of edge habitat within the primary and secondary floodplains. Next, exotic community structure was assessed for the purpose of identifying **Ecospheric End-Stage Disease** (EESD).

The EESD sequelae include those of ELSD, with the added caveat that the exotic community was stratified within the context of confidence interval estimates yielding five or six transects to assess 25% changes in percent cover estimates. Again, the six transect group was included to allow latitude for perforations in otherwise uniform bands of exotic communities. Exotic communities were in part considered established upon formation of uniform bands. These exotic communities were dominated by salt cedar and/ or Russian knapweed. All other exotic species were rare in occurrence (Sabaj-Stahl et al. 2013). Presence of uniform bands of exotic vegetation was considered an extension of community-based disease and a reflection of alterations to hydrology and overgrazing by livestock. *Populus deltoides* (cottonwood) requires sufficiently timed overbank flooding and recession for seed dispersal and germination (Fenner, Brady & Patton 1984). Cottonwood seedlings must establish contact with the groundwater table to avoid root desiccation and mortality (Fotherby 2012). Thus, alterations of Dolores hydrologic regimes imperiled cottonwood reproduction, allowing invasive salt cedar to replace it. Uniform bands of salt cedar, parallel to channel banks, were very common throughout the watershed while cottonwood had become relatively rare. Microhabitats diagnosed with EESD are indicated with the study site designation code highlighted in dark grey and outlined in red (table 3).

The ecospheric disorders identified as ELSD and EESD provide diagnostic tools for investigators and land managers to assess ecospheric health, prioritization of site- and reach-level restoration efforts, and prescription of disease-specific restoration therapies. It is likely earlier stages of ecospheric disease exist. However, these data did not permit detection of more subtle stages of community-based disease processes because of the inability to calculate trajectory. Understanding of trajectory is essential for knowing whether or not microhabitats have initiated early stages of community decoupling. Percent cover of exotics ranged from about 20 to 44% for microhabitats with EESD, whereas the range for all sites was considerably greater. There were also relatively high (> 50%) percent covers of exotics among microhabitats not diagnosed with ELSD or

EESD. These trends indicated stratification of the exotic community in of itself was not necessarily a good predictor of invasive cover *per se*. Thus efforts were undertaken to better understand the relationship between community stratification and the process of bio-invasion.

Turning attention to salt cedar, one of two highly invasive species in the study, a consistent and perhaps reassuring trend was revealed. Those microhabitats wherein exotic and woody communities established uniform bands also evidenced high degrees of stratification. Of those microhabitats stratified for exotic and woody communities, 69% were also stratified for native and herbaceous communities (quarternary structure) (table 3). The remaining four microhabitats were stratified for three of four communities. Of these, three were not stratified for the native community, while one was not for the herbaceous community. Thus, significant edge habitat was present in **92%** of microhabitats where exotic and woody communities were uniform. Given salt cedar is an exotic woody species in this ecosystem, it seems that its presence did not preclude development of edge habitat. However the development of edge habitat involving invasive species can imperil many native species dependent upon pre-invasion edges for survival and reproduction (McKinney 2008; von der Lippe & Kowarik 2008).

The other highly invasive plant of the Dolores, Russian knapweed, is an exotic herbaceous species. Roughly half (56%) of microhabitats stratified for exotic and herbaceous communities were also stratified for native and woody communities (quarternary structure) (table 3). Of the remainder, 31% exhibited tertiary structure, and 13% secondary structure. Of those microhabitats stratified for exotic and herbaceous communities, 75% were also stratified for the woody community. While not quite as prevalent where exotic and woody communities were stratified, there was nonetheless a significant presence of edge habitat where exotic and herbaceous communities were stratified. Overall, the presence of a stratified exotic community did not appear to appreciably inhibit formation of edge habitat, albeit that edge could be detrimental to biodiversity. The native community evidenced similar trends relative to exotic communities. Among the 18 microhabitats stratified for the exotic community, 72% had a stratified native community.

These outcomes signal that EESD is relatively rare in occurrence within the lower Dolores watershed. While the presence of invasive species poses challenges to maintaining ecosystem services and biodiversity, the retention of edge habitat and significant bands of native species even where exotics became established is somewhat a silver lining for land managers. Active revegetation efforts in the wake of invasive species removal could be limited to those microhabitats where a) edge habitat has been curtailed and b) native communities have not become stratified. However such an approach needs to incorporate hydrologic considerations, as dry and saline soils could inhibit the restoration process (Funk 2013). The presence of robust native communities in proximity to exotic communities could facilitate colonization of native species where invasive species have been eliminated. Five of 39 microhabitats (13%) assessed were diagnosed with EESD, indicating the condition progressed relatively slowly. Construction of the McPhee dam was completed in 1986, indicating 24 years elapsed before the watershed was impacted to this extent.

Species Based Disorders of the Ecosphere

The lower Dolores watershed was dominated by two invasive species (salt cedar and Russian knapweed) (Sabaj-Stahl et al. 2013). Thus invasive species biogeographic structure was assessed within the context of community biogeographic structure (Figures 14A & B). Orthogonal regressions were constructed to probe these relationships. Orthogonal, rather than least squares regression was utilized because: a) symmetrical relationships could occur between species- and community-level biogeographic structures, b) variables on both axes measured similar quantities (i.e., biogeographic structure), c) measurement error was not restricted to the *Y*-axis variable, and d) orthogonal regression makes no assumptions concerning independence/ dependence (Fuller 1987; Carroll & Ruppert 1996). The *Y*-axis variable was assembled using the proportion of microhabitats *per* Harriman orbital where invasive species percent cover estimates adhered to 40% confidence interval limits. A pilot study revealed the 40% limit was the approximate lower detection range of the protocol concerning dominant species (Sabaj-Stahl et al. 2013). As with community stratification standards, species confidence interval limits provided an indication of uniformity throughout microhabitats for those species. Community structure *per* the SEI was placed on the independent (*X*) axis. Regression outcomes were compelling and appeared to provide insights concerning invasion processes.

Concerning orthogonal regression of salt cedar structure with community structure (Figure 14A), results ($r = 0.895$; $r^2 = 0.802$; $t = 3.482$; $p < 0.05$) demonstrated salt cedar biogeographic structure was correlated with community biogeographic patterns. Put another way, orthogonal regression revealed that salt cedar and community biogeographic structure were similar quantities, suggesting salt cedar structure behaved in a manner consistent with community structure. Alternatively, the regression of Russian knapweed structure with community structure (Figure 14B), results ($r = 0.081$; $r^2 = 0.007$; $t = -0.141$; $p >> 0.05$) indicated knapweed biogeographic structure was not correlated with community biogeographic structure. As such, knapweed and community biogeographic structures were unrelated, and in fact measured different quantities.

Coefficients of correlation evidenced a strong positive linear relationship between salt cedar and community biogeographic structure, but a very weak relationship between knapweed and community biogeographic structure. Coefficients of determination revealed 80% of the variation in the orthogonal model was explained by the relationship of salt cedar structure with community structure. Conversely, less than 1% of variation in the other model was explained by the relationship of knapweed structure with community structure. One potential outcome of orthogonal regressions are steep line slopes when coefficients of correlation and t-tests indicate an absence of linearity (Carroll & Ruppert 1996). That effect was observed in the orthogonal regression of knapweed biogeographic structure with community structure. The steep line slope was an apparent result of accounting for errors in observations on both the *X*- and the *Y*-axis (Fuller 1987). Least squares regression does not account for errors of the *X*-axis variable. Least squares regression of knapweed structure with community structure (data not shown) yielded similar coefficients of correlation and determination, but accompanied by a marginal line slope.

The disparate outcomes between salt cedar and knapweed biogeographic structure, relative to community biogeographic structure, were characterized as distinct *ecospheric disease processes*. Salt cedar invasion was considered a *quantum-dependent* process because its biogeographic structure appeared to conform to patterns of community structure. Specifically, those microhabitats stratified for salt cedar exhibited an increasing tendency to be stratified for three or four community types (Figure 14A). Casual field observations supported these outcomes, as it was obvious salt cedar exhibited a strong tendency to form distinct narrow bands parallel to channel banks. Borrowing from existing plant disease terminology, quantum-dependent patterns of bio-invasion were dubbed **Potter's rot**, in honor of Cathryne Potter, PhD; academic dean; Rutgers University. Potter's rot was assumed a function of bottom-up forces (i.e., hydrology). Although hydrology was not directly assessed in this study, many related studies have revealed salt cedar invasion was facilitated by reduced overbank flooding preventing optimal cottonwood seed deposition and germination, and by reduced groundwater tables favoring species with deeply penetrating root systems ((Di Tomaso 1998; Zouhar 2003; Merritt & Poff 2010; Lovich & Hoodle 2011).

Knapweed invasion did not conform to patterns of community biogeographic structure. Although knapweed became stratified most often in microhabitats evidencing no stratification of community structure (Figure 14B), no correlation was observed with community structural trends. Because community structure and biodiversity appeared to interact in a quantum-dependent manner, knapweed biogeography was judged, at least within the context of the lower Dolores watershed, to occur via a *quantum-independent* mechanism. This ecospheric disease state was named **Gilbert's smut**, in honor of Julanna Gilbert, PhD; biochemist emeritus; University of Denver.

Unlike salt cedar, disagreement continues concerning the root cause(s) of knapweed invasion. Some studies suggest grazing by livestock, with associated soil disturbance and nutrient enrichment, facilitate the spread of knapweed (Lacey, Husby & Handl 1990; Di Tomaso 2000; Gaussoin, Knezevic & Lindquist 2010). Others have demonstrated careful alteration of grazing regimens appeared to reduce the prevalence of knapweed (Olson, Wallander & Lacey 1997; Henderson et al. 2012; Alder 2013). Most of the microhabitats under study were subject to livestock grazing under US federal permit. Evidence of grazing (cattle trails, wallows and feces) were observed in a majority of microhabitats. While not assessed directly, the top-down disturbance of grazing by livestock was considered the most likely mechanism facilitating knapweed invasion in the riparian ecosystem. However, knapweed thrives and out-competes many native species on well drained, dry soils (Stannard 2004; Duncan, Story & Sheley 2011). Thus, the Dolores impoundment facilitating reduction of the groundwater table and overbank flood regimes likely contributed to knapweed invasion in lower Dolores riparian habitats. However, because knapweed biogeography did not adhere to community structural trends, hydrology was suspected a secondary cause, and overgrazing the primary cause of knapweed invasion of the Dolores watershed. As such, Gilbert's smut might in general be facilitated by top-down disturbances.

Prescription of Ecospheric Therapies

Taken together, community- and species-based ecospheric disease processes provide an opportunity for prescription of remedial ecospheric therapies. Table 4 specifies horticultural and/ or geomorphic restoration strategies on a microhabitat level. Horticultural restoration strategies generally employ top-down approaches, while geomorphic are bottom-up in nature (Rinaldi & Johnson 1997; Doyle, Boyd & Skidmore 1999; Falk, Palmer & Zedler 2006; Tague, Valentine & Kotchen 2008; Rinella et al. 2009; Wallach et al. 2010; Center et al. 2014). A top-down strategy was well underway when these data were acquired in 2010. Several years prior, the biocontrol beetle, *Diorhabda carinulata*, was released on the lower Dolores in an attempt to control salt cedar invasion (Puckett & van Riper III 2014). While the beetle had defoliated a majority of salt cedar throughout the watershed (Sabaj-Stahl et al. 2013), long term mortality rates were not yet established as consecutive years of defoliation are required to kill salt cedar (Dudley 2005; Hultine et al. 2010). Other horticultural approaches applied on a limited basis included removal of salt cedar by track hoe and chain saw, herbicide treatment of salt cedar and knapweed, and active planting of cottonwood and willow. The long term efficacies of these approaches remain to be determined. Geomorphic restoration strategies include alterations to channel meanderings, active grading of channel bank elevations, and construction of check dams and other reach level impoundments and diversions. Installation of check dams and channel bank remediation appeared to have occurred on a limited basis prior to 2010.

Table 4 recommends horticultural restoration strategies for those microhabitats wherein the Bio-invasion Rank Index (BRI) ranged from zero to two (> 20% exotic species cover). The BRI is a straight forward index assessing arcsine-square root transformed, absolute percent cover of exotic communities. Land managers are encouraged to consider a range of horticultural techniques for removal of invasive species and revegetation of native species. Site-level assessments should be conducted to determine the range of techniques appropriate for treatment of invasive species. These assessments should incorporate variables impacting germination, root establishment, growth and reproduction. In particular, soil nutrient and moisture profiles, coupled with assessments of groundwater profiles and overbank flood regimes should be evaluated to ensure best outcomes related to active revegetation protocols (Harper, Williams & Sagar 1965; Ziemkiewicz & Takyi 1990; Prodgers, Keck & Holzworth 2000; Blew, Jackson & Forman 2003; Mollard & Naeth 2014).

Geomorphic restoration therapies were recommended for microhabitats wherein the Sabaj Efficacy Index (SEI) was zero or one (table 4). These were microhabitats in which just one or none of the four community types assessed were stratified. As community structure of riparian habitats is primarily regulated by hydrology, alterations to channel morphology appear to offer the best chance of long term, self-sustaining restoration (Galat et al. 1998; Mahooney & Rood 1998; Florsheim & Mount 2002; Palmer & Bernhardt 2006). Specifically, there existed several microhabitats where channel bank elevation appeared to have increased as a consequence of salt cedar invasion. Dense thickets of salt cedar produce multiple trunks trapping and accumulating course particulate matter (Kennedy & Hobbie 2004; Brownell 2013). A pronounced mounding effect was evident, and likely restricted overbank flooding in several reaches of the lower Dolores watershed. Several river reaches also appeared deeply incised and channelized, with salt cedar invasion a suspected contributing factor (McKenney, Jacobson & Werthheimer 1995; Grams & Schmidt 2002; Dean & Schmidt 2010). Thus it is recommended channel bank alteration include a knock down salt cedar mounds where accessible by heavy equipment.

Given 40 to 50% of Dolores river water is removed annually for irrigation, and peak and low flows of the historic hydrograph have been obliterated via the McPhee impoundment, it is also recommended reach level impoundments are implemented as a geomorphic restoration technique. Such smaller water control structures would enable land mangers the opportunity to regulate river flows at the level of the reach to an extent representative of historic flows (Bainbridge 2007, Castillo, Pérez & Gómez 2014; Polyakov et al. 2014). A system of check dams could be regulated in a way to facilitate overbank flooding with associated replenishment of the hyporheic zone, without any significant impairment of senior water rights involving ranchers, farmers and municipalities (Bombino, Tamburino & Zimbone 2006; Diaz, Mongil & Navarro 2014; Norman et al. 2015). Check dams can also be manipulated to accumulate sediment and raise river bed elevations to an extent allowing for historic patterns of overbank flooding. Microhabitats MO5 through MO9, where five of six sites appear to require geomorphic restoration, occur along consecutive river reaches. Thus this particular stretch of river seems ideal for implementation of one or more check dams as a means to regulate river flows for the purposes of restoration.

Regardless of the recommendations concerning restoration, land managers are encouraged to acquire additional data within microhabitats prior to restoration protocol development and implementation. While initial assessments of community structure, biodiversity and bio-invasion provided direction relative to restoration, special care should be exercised with regard to active revegetation and geomorphic amendments. So for example, costly and time consuming revegetation practices could be obliterated by untimely flood events, droughts, insect herbivory or ungulate grazing during critical periods of germination and growth (Bleak et al. 1962; Hoffman 1986; Archer & Pyke 1991; Dreesen & Fenchel 2009). Detailed assessment of post-impoundment stream gauge data, overbank flood regimes and groundwater tables should be conducted over a range of dry and wet years prior to engineering of channel bank amendments and check dams. Engineering designs should also assess to the extent possible the trajectories of climate change, anthropogenic water demands and retention/ release regimes of water from the McPhee reservoir, such that engineered geomorphic amendments do not become obsolete.

The widespread and frequent occurrence of restoration failures necessitates a well-thought and nuanced approach to achieve any long term success (Holling & Meffe 1996: Zedler & Callaway 1999; Stanturf et al. 2001; Zampella & Laidig 2003; Hildebrand, Watts & Randle 2005; Palmer 2009; Moreno-Mateos et al. 2012). Restoration failures appear to result from inadequate approaches addressing ecologic structure and function. Thus it is essential both are carefully considered relative to any restoration program (Heckman 1997; Moore, Covington & Fule 1999; Cortina et al. 2006). Current horticultural approaches to restoration of the lower Dolores watershed, which include release of the defoliating salt cedar beetle, mechanical and chemical treatment of salt cedar, chemical treatment of knapweed, and active revegetation are not likely to provide long term benefits absent geomorphic amendments. Thus Dolores River stakeholders are encouraged to invest in geomorphic restoration strategies, including but not limited to channel bank elevation amendments, check dams and other engineered surface water diversions. Absent geomorphic restoration, it is unlikely riparian habitats of the lower Dolores will acquire and retain vegetative community structural elements required for optimal function of the ecosystem.

III. The Vegetative Complex Health Index:

The impetus for pursuit of a composite ecospheric health metric grew out of discussions with experts in the field of ecologic restoration (e.g., Tom Stohlgren, PhD; Amanda Clements, PhD; Richard Alward, PhD; James Kerr, PC; and others). Overwhelmingly, the feedback indicated a frustration concerning the available tools used to evaluate outcomes of restoration efforts. Most communicated that while large sums of money, resources and effort were expended to repair damaged ecosystems, they often were left not knowing to what extent those efforts provided any functional and lasting benefit. Many felt the litany of diversity indices had seen their heyday and were insufficient for characterizing ecologic recovery. Others expressed that measurements of community structure utilizing percent cover, biomass and so on, while useful to a degree, did not necessarily reflect the trajectory of restoration or function. Some explained that assessments of invasive species prevalence, while helpful, likely received too much attention as an indicator of restoration success.

It was also conveyed budgetary constraints usually precluded comprehensive evaluation of broad ecosystem attributes and function. While potential options for assessment of biotic, physical and chemical features were restricted, investigators believed it was possible to characterize the trajectory and relative success of restoration projects. In general there existed a desire for a composite ecometric assessing diversity, structure and function. To that end, *The Theory of Quantum Microbiogeography* (Sabaj-Stahl et al. 2013) was applied to derive an ecometric assessing these variables in a simultaneous, objective and predictable fashion. Development and application of the **Vegetative Complex Health Index** (VCHI) appeared to uniquely combine aspects of structure, function, biodiversity and bio-invasion.

The construction of effective composite indices requires nuanced approaches. Sometimes the procedure is abused to the extent of generating redundancies that reveal little to nothing with regard to the function of systems (Leontief 1936; McGillivray 1991; Coste, Fermanian & Venot 1995). Composites also impart certain advantages and disadvantages. Composite indexing is multi-dimensional in nature as it represents aggregate measures of complex phenomena (Booysen 2002). Composite indices tend to be additive with weighted components. Thus, while compositing reduces subjectivity by integrating variables into a consistent format, subjectivity can increase via ad hoc approaches related to variable selection and weighting criteria (Booysen 2002).

Biodiversity indices in of themselves are composite indices, usually assessing species richness and evenness, and sometimes overall abundance (Rosenzweig 1995). In theory, diversity indices should be able to detect changes in any one of these three variables while the others remain constant. However individual diversity indices vary with respect to these abilities and their sensitivities (Buckland et al. 2005). Simpson's index tends to be less sensitive to shifts in species richness, whereas Shannon's index is less sensitive to changes in species abundance (Rosenzweig 1995). Furthermore, Simpson's and Shannon's indices do not detect changes in overall abundance (i.e., when all species within a region change at a constant rate) (Buckland et al. 2005). Shannon's diversity does not reflect spatial distributions of classes, and neither reflect landscape elements, which are crucial in terms of assessing biodiversity (Filip, Pietsch & Richter 2008; Lamb et al. 2009). These are but a few of the challenges posed when attempting to unify multiple variables into a composite.

Most biological composite indices attempt to assess the integrity of biological systems for the purpose of grading the relative health of the system. Thus, in human medicine, numerous composite indices have been developed to assess the health of patients and populations (Scrambler 2005; Antony & Rao 2007; Jones et al. 2009; Metge et al. 2009). Similar approaches have been developed in ecology for the purpose of assessing environmental health, and are generally referred to as composite measures of biologic integrity (Hilsenhoff 1988; Noss 1999; Barbour et al. 2000; Andreasen et al. 2001; Reza 2014). Composite indices are also popular in the fields of economics, social science and professional sports, among others. In general, all composite indices attempt to achieve a variety of benchmarks. These include: a) summarizing data from multiple variables to reveal relevant trends without obscuring the behavior of individual variables; b) communicate information in a manner that is easily understood by non-scientists, such as policy-makers; c) eliminate the need to qualitatively and subjectively assess multiple variables independently; d) evidence consistent linear trends across space and time; and e) develop a composite metric that accurately predicts the behavior of its individual component variables (Barbour et al. 2000; Moog & Chovanec 2000; Booysen 2002; Young & Sanzone 2002; Zhou, Ang & Poh 2006; Metge et al. 2009; Fongwa 2012).

The Vegetative Complex Health Index was constructed in a manner to optimally reflect these central paradigms of composite measures. The VCHI is also a composite scale, as the index included not only the averaging of ordinal ranks into an aggregate (i.e. community structure and bio-invasion), but also contained a scalar variable expressed as Fisher's Disambiguous Coefficient (FDC). The FDC expressed biodiversity in terms of its relative magnitude occurring between microhabitats and the ecosystem. There has been a proliferation in recent years of merging landscape metrics with biodiversity to better reflect aspects of community structure and variability across a spectrum of spatial heterogeneity (Lapin & Barnes 1995; Xiang 1996; Burnett et al. 1998; Schupbach et al. 1999; Jedicke 2001; Tischendorf 2001; Moser et al. 2002; Zebisch 2002; Ortega, Elena-Rosello & Garcia del Barrio 2004; Zebisch 2004; Sukopp 2007; Schindler, Poirazidis & Wrbka 2008; Sundell-Turner & Rodewald 2008; Walz 2011). Generally it appears the development of composite bio-invasion indicators has not been as robust, although predictive modeling approaches appear to take precedence over composite indices relative to bio-invasion (Carlton & Ruiz 2005; Ferreira et al. 2005; Hulme 2006; Kümpel & Baillie 2006; McGeoch, Chown & Kalwij 2006; Olden, Kennard & Pusey 2008; McGeoch, Spear & Marais 2009; Crossman, Bryan & Cooke 2011). While our review of the literature concerning ecological composite indices was by no means exhaustive, it appears the VCHI is the first composite to combine elements of biodiversity, community structure and bio-invasion in such a manner for assessment of ecosystem health.

The process employed for development of the VCHI was consistent with many of the steps outlined in the *Handbook on Constructing Composite Indicators: Methodology and User Guide* (Nardo et al. 2008). Briefly, this program guides users through the theoretical and applied aspects of composite indicator development, and provides useful information concerning selection of individual indicators, weighting and aggregation, scaling, selection of appropriate data, and model testing and validation. Some portions of this guide were not applicable to the VCHI, but in general the central tenets of theoretical development, hypothesis development and testing, and model validation were applied. Two of three variables (community structure and bio-invasion) of the VCHI were expressed by converting proportional data to integer rank scales. Biodiversity was expressed as a ratio scale. It is important to note all data used for construction of these variables were arcsine-square root transformed. The transformation had the effect of pulling outliers inward, reducing skewness and improving homoscedasticity (Sabaj-Stahl et al. 2013). No additional steps were undertaken to normalize variables of the aggregate (community structure and bio-invasion) as they were each derived from proportional data and then expressed on equivalent integer scales (Nardo et al. 2008).

Variables of the VCHI aggregate were not weighted because: a) Overall community structure was not correlated with diversity (Figures 3B; 4A-D); b) Correlations of salt cedar and knapweed biogeographic structure with community structure were ambiguous (i.e., dependent and independent) (Figures 14A, B); c) Overall bio-invasion rates were not correlated with community structure (Figure 19A); and d) Biodiversity was not correlated with percent cover of invasive species, but was correlated with stratification of the exotic community (discussed below). These independences (a through d) and dichotomies (b and d) appeared to present barriers in terms of determining what, if any appropriate weights ought be applied to structure, biodiversity and bio-invasion. Specifically, principle components and factor analyses applied to derive weights of individual indicators assume variables are correlated (DeCoster 1998; Nardo et al. 2008). Clearly that threshold was not attained for a number of inter-variable assessments concerning the VCHI.

Three VHCI Hypotheses

By way of alternative, biodiversity was applied as a scalar to the mean aggregate of structure and bio-invasion. It was hypothesized the scaling of the aggregate with biodiversity would cause biodiversity, bio-invasion and the mean aggregate of bio-invasion and structure to behave linearly with VCHI rankings. Furthermore, because it appeared community structure was independent of biodiversity, it was also hypothesized that a) the use of biodiversity as a scalar would not linearize structure relative to the VCHI, but b) would instead have the effect of eliminating the stratified exotic community from those habitats with the better (i.e., good and excellent) VCHI rankings, due to the correlation of exotic community stratification with biodiversity.

The SEI, BRI & FDC

Community structure was assessed *per* the Sabaj Efficacy Index (SEI) (Sabaj-Stahl et al. 2013). As previously described, the SEI measures the tendency of communities to form uniform bands of vegetation parallel to channel banks. Thus, the SEI is not a formula for assessment of percent cover *per se*, but instead reflects function in a variety of ways. While it was apparent communities displayed a greater tendency to become stratified with increasing percent cover estimates (table 4), the trend was by no means universal. There were instances where relatively high percent covers did not equate to stratification. Microhabitats MO6 and MO7 evidenced woody percent cover greater than 65%, yet these communities were not stratified. Conversely, microhabitats GJ2 and DL3 revealed herbaceous percent cover of less than 18%. Nonetheless these communities adhered to stratification criteria (table 4). As discussed, the tendency to form stratified bands of vegetation parallel to channel banks is largely a consequence of hydrology (overbank flooding, groundwater table, hyporheic zone) and associated soil moisture profiles. Thus, the SEI provides indications of ecologic function relative to the interaction of vegetative communities with hydrology and soils, via structural analysis. Stratification of communities, particularly native herbaceous and woody types, provides critical edge habitat often enhancing biodiversity. Thus, the SEI provides an assessment of ecologic function considered important for a composite metric evaluating ecospheric health.

However, the SEI in of itself presented a unique challenge concerning assessments of ecospheric health. Community structure as defined prohibited a direct and unbiased ecospheric health ranking of microhabitats on the basis of the SEI alone. While microhabitats absent any stratification (SEI = 0) were judged unhealthy within the context of functional riparian ecosystems, those with maximal stratification (SEI = 1) were stratified for all four community types, including the exotic community. As such, those microhabitats with the highest of SEI values also displayed a significant occurrence of invasive species. This effect precluded an unbiased sorting of microhabitats on the basis of SEI outcomes alone. At best, subjective and qualitative determinations made on a site-by-site basis would be required to grade out overall health on the basis of the SEI.

While the SEI was constructed using the same data as for diversity estimates (i.e., absolute percent cover estimates of taxa), it did not necessarily reflect content concerning diversity, given its independence from diversity (Figures 3B, 4). Salt cedar biogeography was correlated with community biogeographic trends, but knapweed biogeography was not (Figures 14A, B). These limitations necessitated inclusion of additional variables into a composite in order to comprehensively evaluate ecospheric health. The elements of alpha diversity and bio-invasion were considered crucial to any such composite, and were included as components of the VCHI. It was hypothesized a nuanced and informed integration of these variables into a composite metric would overcome the challenges presented by the SEI.

Bio-invasion is among the most noticeable and impactful of processes related to ecologic disturbance and environmental degradation (Di Tomaso 2000; Dale et al. 2001; Didham et al. 2005; Jewett, Hines & Ruiz 2005; MacDougall & Turkington 2005; Stohlgren & Schnase 2006; Didham et al. 2007; Ou, Lu & O'Toole 2008). Thus most restoration programs address bio-invasion as one of the primary objectives of rehabilitation. Programs were already implemented on the lower Dolores watershed for reducing the prevalence of salt cedar and knapweed when these data were acquired. The second component of the VCHI was defined as the **Bio-invasion Rank Index** (BRI). Unlike the SEI, the BRI does not implement stratification criteria, but accounts for arcsine-square root transformed, absolute percent cover of the exotic community. This straight forward ranking system ranges from zero through four, with increasing integers corresponding to reduced percent cover of exotic species (table 4). Notice the inverted relationship of SEI to BRI integer values. Increasing SEI values correspond to increased stratification. Conversely, increasing BRI values correspond to decreased percent cover of invasive species. However, in general, increased SEI and BRI values are considered beneficial to riparian ecosystems, of course with exception to exotic community stratification. The importance of this relationship, as well as the application of comparable integer values for both indices, shall become evident concerning mechanics of the VCHI.

The apparent quantum duality occurring between structure and diversity was evidenced as independence of the mean juxtaposed by dependence of the range of diversity with community structure. It therefore seemed important to reflect that duality in the VCHI. Thus the third component of the VCHI was defined as **Fisher's Disambiguous Coefficient** (FDC). Fisher's alpha was chosen as a measurement of alpha diversity for the VCHI because of its independence with community structure *per* the one- and two-tailed t-tests (Sabaj-Stahl et al. 2013; Figure 4). The FDC is the proportion of Fisher's alpha for a given microhabitat to mean Fisher's alpha for the ecosystem. Applying diversity in this fashion was reflective of the mean and range of biodiversity. The FDC therefore provides an indication of the degree to which diversity within a given microhabitat deviates from mean diversity of the ecosystem.

The VCHI Equation

The equation for the **Vegetative Health Complex Index** (VCHI) was defined as follows:

$$\mathbf{VCHI = ((SEI + BRI)/2) * \text{Ø}_F}$$

The SEI is the Sabaj Efficacy Index, and is the proportion of communities within a microhabitat achieving stratification criteria. In its prior uses, the SEI was directly expressed as a proportion ranging from zero to one. In the VCHI, SEI values were converted to integer rankings of zero through four. These rankings correspond to the number of communities (native, exotic, woody & herbaceous) that were stratified within a microhabitat. The reason for applying integer rankings were to cause them to be within the range of rankings used for the Bio-invasion Rank Index (BRI). The BRI directly assessed percent cover of exotic species and ranged from zero through four. Ranks were based upon arcsine-square root transformed, absolute percent cover of exotic species, *per* microhabitat, as follows: 0: ≥ 40%; 1: 30-39%; 2: 20-29%; 3: 10-19%; 4: < 10%. The FDC (Ø_F) is simply the ratio of Fisher's alpha for a given microhabitat to mean Fisher's alpha for the ecosystem. Theoretically, Ø_F ranges from zero to infinity. In practice, it likely ranges from about 0.1 to 10.

The mean aggregate of community structure and bio-invasion was applied to the VCHI because it afforded a balancing of structural components with the process of bio-invasion. Among those microhabitats stratified for the exotic community, 17 of 18 had BRI rankings of zero to two, indicating relatively high (> 20%) percent cover of exotic species. Thus, the mean aggregate caused those microhabitats to substantially drop in their VCHI ranks, relative to the SEI alone, as all had BRI rankings less than their respective SEI values (table 3). Conversely, only 6 of 21 microhabitats wherein exotic communities *were not* stratified evidenced BRI rankings less than their respective SEI values (table 3). Thus, the BRI exerted no negative impact relative to SEI values for 71% of microhabitats where exotic communities were not stratified. By including a metric for assessment of bio-invasion that was independent of structure, those microhabitats stratified for multiple communities (but not the exotic community), on average, graded out healthier than those stratified for the exotic community. Given salt cedar exhibited quantum-dependent patterns of bio-invasion, but knapweed did not, the BRI also overcame this disparity as a *structure-independent* index. Table 4 demonstrates microhabitats were reshuffled when comparing SEI rankings (site color codes) to VCHI categories (status color codes). Part of that reshuffling was caused by introduction of structure-independent bio-invasion rankings to the VCHI.

Fisher's Disambiguous Coefficient also exerted a significant impact on the rearrangement of microhabitats *per* the VCHI. In fact, as devised, the FDC had the effect of more than halving some of the mean aggregates of SEI and BRI rankings (microhabitats GJ4, MO1, MO10 and DL4); while increasing others by more than an integer (MO3, UC5, UC7, DL2 and DL8) (table 4). Some curious trends were revealed concerning the FDC and stratification of the exotic community. For those microhabitats where exotic communities adhered to stratification criteria, 39% had FDCs greater than one (61% < 1.00). Conversely, in microhabitats where exotic communities were not stratified, 57% had FDCs greater than one (43% < 1.00). Thus there was a *net benefit* imparted by the FDC to microhabitats where exotic communities had not become stratified, compared to microhabitats where exotic communities were stratified. This disparity indicates stratification of exotic communities has a *net negative* effect on ecosystem biodiversity. Juxtaposed with this finding are the average, arcsine-square root transformed, absolute percent covers for exotics from microhabitats where the exotic community was stratified v. those without stratified exotic communities. Eighteen microhabitats stratified for the exotic community averaged 39.85% cover of exotic species. Eighteen microhabitats without stratified exotic communities averaged 37.19% cover for exotic species (three microhabitats were not included in this calculation as invasive salt cedar was removed by chainsaw just prior to data acquisition). Thus, stratification in of itself impacted biodiversity *exclusive of* percent cover. That finding was instrumental as it concerned the third hypothesis of the VCHI.

The disparate outcomes concerning biodiversity v. structure and percent cover v. structure indicate a need for nuanced approaches when evaluating ecospheric health. Protocols assessing only biodiversity, community structure or bio-invasion as a metric to evaluate disturbance, bio-invasion and/ or restoration appear to fall short of fully characterizing ecospheric health and recovery. Attempts to assess biodiversity, bio-invasion and community structure independent of one another ultimately yield a qualitative process that is difficult, if not impossible to replicate. The information contained in table 4 for the SEI, BRI and Fisher's alpha was provided to five biologists. They were asked to grade microhabitats concerning overall health using these metrics (data not shown). Not surprisingly, none concurred in their grades for all microhabitats, nor did any match all outcomes provided by the VCHI. As such, the VCHI provides a *quantitative, objective and consistent* means to assess ecospheric health on the basis of biodiversity, bio-invasion, structure and function. To the best of our knowledge, this has not been accomplished previously, at least within a mechanistic framework.

The VCHI ratings include categories ranging from degraded to excellent (table 4). The VCHI is a continuous variable, so these categories encapsulate ranges and are intended to serve as a guide in assessing ecospheric health and prioritizing land management decisions. Microhabitat status was defined as follows: VCHI ≥ 4.0 = *excellent*; 3.0-3.99 = *good*; 2.0-2.99 = *fair*; 1.0-1.99 = *poor*; 0.0-0.99 = *degraded*. Restoration strategies (horticultural and or geomorphic) were initially recommended using individual BRI and SEI rankings, respectively. These recommendations can be further organized and prioritized based upon VCHI outcomes. No treatments were recommended for any sites with excellent ratings, although salt cedar had been removed from these sites by chain saw just prior to data acquisition. Thus, follow-up treatment with herbicide might be required to suppress salt cedar sprouting from root crowns. Active revegetation coupled with soil amendments might also be necessary to prevent colonization by other invasive species, especially knapweed. Just one of three microhabitats with ranks of good was identified for horticultural restoration. Recommendations for horticultural restoration spiked as VCHI rankings fell, with 88% of fair microhabitats, and 100% of poor and degraded sites identified for these approaches. Geomorphic restoration strategies illustrated similar trends, with no microhabitats in the excellent or good categories recommended for geomorphic restoration. Twelve percent of fair microhabitats, and nine percent of poor appeared in need of geomorphic treatment. Alternatively, 64% of degraded sites were identified for geomorphic amendments.

While the VCHI reordered microhabitats relative to individual SEI and BRI rankings, VCHI rankings appeared to be in agreement with trends evidenced in community- and species-based ecospheric disease processes. Geomorphic restoration approaches rooted in assessments of community dynamics, and horticultural-based strategies tied to invasive species trends evidenced increased need as VCHI ratings worsened. As such, the VCHI provided guidance in terms of balancing restoration needs with specific agenda and available resources. Land managers can assess the status of ecosystems on a microhabitat level using the VCHI, and then allocate resources for implementation of sound restoration practices. Geomorphic amendments, including installation of check dams and diversions, and alterations to channel bank elevation can be prohibitively costly. Given the least healthy of microhabitats indicated a need for geomorphic restoration, land managers might focus upon less costly horticultural techniques initially, and then evaluate the efficacy of those approaches before investing in geomorphic remedies. A note of caution, though, would be that those regions requiring geomorphic amendments may not support many of the drought-intolerant species typical of riparian habitats. Livestock grazing regimens could also preclude establishment of many native species over the long term.

Newtonian Physics & the VCHI

The VCHI was constructed in a manner permitting modeling within the context of Newtonian mechanical principles (Bunge 1966; Nelson 1966; Dugas 1988; O'Donnell 2015). Specifically, the VCHI adhered to the essential Newtonian mechanics of a three-pulley system. Pulley systems also provide good working illustrations of Newton's three laws of motion (inertia, force/ mass/ acceleration, and action-reaction) (Newton 1687; Pople 1987). Figures 13 through 16 illustrate the mechanics of the index within this rubric. Community structure/ stratification (SEI) and bio-invasion (BRI) were modeled as fixed pulleys. The chain extending from each fixed pulley has variable weights attached contingent upon SEI and BRI ratings. These weights are representative of respective index ranks, and are not to be confused with the weighting of composite index variables via PCA and factor analysis. The central pulley is free to move and represents the mass individual microhabitats, with associated frictional force indicated on the outer edge of the pulley. For model purposes, microhabitat mass was assumed an unknown constant.

Friction (disambiguation), defined as Fisher's Disambiguous Coefficient, was displayed as increasing from higher to lower VCHI rankings. While friction generally behaved in this manner, the trend was not universal. Friction opposes the relative motion of two surfaces in contact with one another, such as the chain and pulley. Within the context of Newtonian physics, the *weights* of community structure and bio-invasion exert *force*, biodiversity counteracts force as *friction*, and individual microhabitats are the *mass* that force and friction act upon. *Momentum* is represented by the motion of microhabitats through the series of VCHI rankings. Motion is bidirectional, and is determined by the positive and negative trajectories of microhabitats relative to ecospheric health. Note, however, that because only one season of data were assessed, trajectories cannot be calculated. Thus, motion is defined by the positioning of microhabitats relative to one another. Figure 15 illustrates the overall mechanics of the model as described. Increases in community structure operated in opposition to increases in bio-invasion *per* the relative weights attached to the chain, and exerted force upon the mass of the central pulley. In general, the models illustrated increased stratification of community structure, decreased rates of bio-invasion and increased biodiversity contributed to increased VCHI ranks. As modeled, the VCHI appeared to operate as an excellent ecometric assessing ecospheric health.

Figures 16 through 18 demonstrate the neutral, positive and negative impacts of disambiguation, respectively, upon VCHI rankings. As biodiversity within a microhabitat approached that of the ecosystem (figure 16), friction (defined as the FDC) approached one and had negligible impacts upon the VCHI mean aggregate. Conversely, as the FDC significantly differed from one, it imposed positive (figure 17) or negative (figure 18) impacts upon the VCHI aggregate. The manner by which the index operates potentially imposes limitations concerning assessments of ecospheric health. Specifically, if on average, alpha diversity were to change uniformly across all microhabitats, then it would also change for the ecosystem. However, that pattern of change would not alter the VCHI aggregate, because the proportions of microhabitat diversity to ecosystem diversity would be unchanged. While that specific scenario cannot be excluded from possibility, it is considered unlikely for ecosystems experiencing disturbance, bio-invasion and/ or restoration. So for example, while salt cedar is expected to experience widespread mortality within the lower Dolores watershed, thus eliminating a relatively uniform band of vegetation throughout the ecosystem, subsequent changes in diversity are not expected to be uniform. Factors such as variability in soil moisture and salinity profiles, sun exposure, elevation gradients, and livestock grazing regimens shall likely facilitate variability in recovery to such an extent that changes in biodiversity are not uniform across all microhabitats.

The elements of bio-invasion and community structure as components of the VCHI aggregate also likely preclude uniform changes in biodiversity eluding detection by the composite index. In the example just given concerning salt cedar, changes in biodiversity shall also be accompanied by noticeable decreases in bio-invasion and community structure. Thus it is unlikely the VCHI would fail to detect significant changes in ecospheric health across an ecosystem, even if changes in biodiversity were uniform. The *sixty-four thousand dollar question* (not adjusted for inflation via composite indexing…) concerning the VCHI is: “Does the VCHI reorder microhabitats in a manner such that it supports the hypotheses concerning linearization of biodiversity, bio-invasion, and the aggregate of structure and bio-invasion; whilst remaining independent of structure?” Figures 3B, and 4A through D indicated community structure evidenced no linear trends with richness, abundance, dominance and total diversity. While these outcomes were key in elucidating the quantum duality occurring between diversity and structure, they also presented significant hurdles in developing a comprehensive ecometric incorporating diversity and structure.

Reorganizing the Ecosphere *per* the VCHI

Figures 19A and B revealed the VCHI supported the three hypotheses as stated, and thus was deemed a sufficient composite index for assessment of comprehensive ecospheric health, at least within the confines of the lower Dolores watershed. Figure 19A illustrated stratification of community structure steadily increased across the first three VCHI categories (*degraded*, *poor* and *fair*), then climaxed at 3.25 stratified communities among *fair* microhabitats. Structure declined and plateaued to about 2.5 stratified communities for the *good* and *excellent* VCHI categories. Regression coefficients and the associated p value (not shown) evidenced no linear association between structure and the VCHI. The observed effect upon structure was a consequence of including bio-invasion as part of the mean aggregate. Because percent cover of invasive species was independent of the FDC, but stratification of the exotic community was not, the coefficient of biodiversity therefore exerted the effect of removing the stratified exotic community from the higher VCHI rankings, even though overall stratification was independent of biodiversity. As there was a tendency for biodiversity to decrease within microhabitats where exotic communities were stratified, the FDC offset increases in structure with concurrent increases in friction. Increased friction caused these microhabitats to grade out lower than many of those without stratified exotic communities.

Figure 19A also evidenced bio-invasion steadily improved (became less prevalent) as the VCHI increased. It is notable the highest rates of bio-invasion occurred among those microhabitats with none or just one stratified community. This trend indicates that as community structure decoupled, bio-invasion exploded. However these data provide no indication of which of these processes served as the driver or passenger. Both processes were likely rooted in alterations to hydrology and over-grazing by livestock. Also of note was that bio-invasion rates were lowest among *good* and *excellent* VCHI categories, wherein community stratification was reduced in comparison to the *fair* category. Again, this trend reflected that the index was successful in eliminating stratified exotic communities from the most highly rated microhabitats. It is important to note the improvements observed among the *excellent* microhabitats were not geomorphic in nature. Rather, just weeks prior to monitoring, salt cedar was removed by chain saw with slash stacked outside the monitoring quadrat. That active management strategy had the effect of decoupling exotic communities from stratification criteria, but does not provide any information concerning the long term trajectory of those microhabitats relative to the VCHI.

The mean aggregate of structure and bio-invasion (SEI + BRI/2; Figure 19A) evidenced a strong linear relationship with the VCHI, with 96% of the variation in the model explained by these variables. When Fisher’s alpha and Margelef’s richness were regressed with the VCHI, 96% and 94% of the variation in the models, respectively, were explained by these variables (Figure 19B). Thus, increases in the mean aggregate, richness and Fisher’s alpha were tightly correlated with increases in the VCHI. These outcomes indicate the VCHI is a reliable predictor of ecospheric health and supports application of the index within ecosystems where stratification of community structure, alpha diversity and bio-invasion can be quantified. We predict these relationships will hold for most riparian ecosystems in the southwestern United States, where impoundments and diversions are the norm and invasion by salt cedar and knapweed is widespread. What remains unknown is the extent to which the index will be applicable to upland communities or riparian ecosystems in other biogeographic settings. Efforts are currently underway to assess data from long term monitoring projects in other settings to ferret out these ambiguities.

IV. Conclusions

Three proof of concept approaches have now been verified as it concerns *The Theory of Quantum Microbiogeography*. Application of these concepts were successful in identifying a parsimonious quantum of highly efficacious microhabitats concerning the estimation of a variety of ecosystem variables (Sabaj-Stahl et al. 2013). The current work elucidated the theory was sufficient to identify discrete patterns of bio-invasion (i.e., Potter's rot and Gilbert's smut) and build a mechanistic composite index reflective of ecospheric function and health. This theory rests upon the foundation of the hypothesis that the relationship occurring between biodiversity and community structure regulates community structural patterns. More specifically, the relationship is assumed to exist as a quantum duality, wherein the mean of biodiversity is independent of structure but the range is not. While we have yet to assess that relationship in ecosystems other than the lower Dolores watershed, we expect the relationship to hold in relation to the coupling and decoupling of community structure, given the strength of the correlations observed. While the modeling of the theory within the context of quantum atomic models was largely conceptual in nature, it is expected that as more data are assessed, universal relationships shall be identified within the context of functional mechanical systems.

Habitats, ecosystems and biomes across the globe are increasingly subject to human-based disturbance and bio-invasion, with concurrent decreases in biodiversity. Development of ecometrics sufficient for objective assessment of the impacts of disturbance and bio-invasion, as well as the potential benefits of restoration are necessary for understanding the processes of community and population dynamics, and associated biodiversity and ecologic function. The approaches used for development of the Vegetative Complex Health Index bridged many of the gaps related to quantification and interpretation of multiple ecologic variables. The VCHI arose from an understanding of how those variables *mechanically* interacted with one another, and how they were potentially impacted by the bottom-up forces of hydrology/ geomorphology and top-down impacts such as grazing by livestock. *The Theory of Quantum Microbiogeography* has therefore graduated from a strictly abstract concept to an applied model of considerable impact.

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