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Diversity of arthropods in farmed wetlands in the Gulf of Mexico Coastal Plain and effects of detrital subsidies

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ABSTRACT
Wetland acreage has declined continentally in the USA. With the loss of natural wetlands, farmed wetlands may be a surrogate for natural wetlands. Seasonally, flooded rice fields are commonly managed wetlands on the Gulf of Mexico Coastal Plain. This study had two objectives: (1) catalog arthropod diversity in rice fields; (2) investigate if detrital subsidies could elicit trophic cascades that reduce nuisance organisms. In 2013 and 2014, experimental rice plots were established, and detritus was applied to half of the plots in each year. Floating pitfall traps, aquatic D-net sweeping, Gee crawfish traps and root/sediment corers were used to sample for arthropods. Over the two growing seasons, 143 different species were sampled from 13 orders, totaling 49,251 individuals. Detrital subsidies neither elicited a detectable trophic cascade nor did they significantly alter the rice field arthropod community. Contrary to previous studies, results suggest that macroinvertebrate communities in farmed wetlands are resilient to alterations possibly due to the long-term agricultural use.

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Arthropod diversity; farmed wetlands; detrital subsidies; Gulf of Mexico Coastal Plain; rice fields; trophic cascades

Introduction

Wetlands are habitats for diverse fauna and flora (Hook 1993; Lawler 2001), and provide many ecosystem services, and thus, are a conservation priority (O'Malley 1999; Zedler & Kercher 2005). Wetlands acreage in the USA has declined, with an estimated loss of 260,700 ha from 1986 to 1997, 26% of which was attributed to conversion of wetland to agriculture (Dahl 2000). Farmed wetlands, specifically rice and crayfish ponds, are a potential surrogate for natural wetlands, especially in regions where natural wetlands are less common (Lawler 2001; Huner et al. 2002; Richardson & Taylor 2003; Manley et al. 2004; Wilson et al. 2008; Wyss et al. 2013). Farmed wetlands, however, generally have different biotic and abiotic characteristics than natural wetlands (Huner et al. 2002). Farming requires specific management tactics that often differ from regional wetland hydrologic characteristics (Richardson & Taylor 2003; Wyss et al. 2013) and application of pesticides, and synthetic fertilizers can greatly alter aquatic biota and community dynamics (Cohen et al. 1994; Schoenly et al. 1995; O'Malley 1999; Mize et al. 2008; Stenert et al. 2009; Stenert et al. 2010). Biodiversity in these managed ecosystems is often lower than natural wetlands (Piatti et al. 2010; Chen et al. 2013; Wyss et al. 2013; Moreira & Maltchik 2014). Many current organic farming and integrated pest management strategies are promoting farming methods that require less chemical inputs, which may aid in the ability of farms to function as surrogate wetlands (Lawler 2001; Lou et al. 2013).

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A large portion of the Gulf of Mexico Coastal Plain was historically composed of natural wetlands and is currently composed of farmed wetlands, specifically rice fields (Dahl 2000; Blanche et al. 2012). While not as structurally diverse as natural wetlands, rice fields can provide habitat for wetland biota (Richardson & Taylor 2003; Wilson et al. 2008; Stenert et al. 2010), the composition of which can be greatly influenced by farming practices (Elphick & Oring 1998; Stenert et al. 2009). Agronomic practices in rice vary in the USA, with a wide range of water management practices from continuous flooding to cycles of flooding and drying (Blanche et al. 2012). Rice cultivation in this region relies heavily on chemical control practices for both weed and insect pests, which can be detrimental to biodiversity (Cohen et al. 1994; Blackman et al. 2014). Farmers frequently till the soil before planting, which can have varying effects on insect community composition as well (Lawler 2001; Stenert et al. 2010; Foley 2015).

Detrital subsidies to rice fields as well as other aquatic systems have been found to initiate trophic cascades in which primary producer populations are increased, leading to increases in predator numbers and reductions in primary consumers from the larger predator population (Settle et al. 1996; Cross et al. 2006; Leroux & Loreau 2008). Generating this cascade, especially early in the growing season in rice fields, has been found to reduce the number of nuisance and invasive organisms in rice fields in Asia (Settle et al. 1996; Jiang & Cheng 2004). Biological control via detrital subsidies can reduce the need for pesticide applications that negatively affect wetland biota, making cultivated rice fields more suitable as a surrogate wetland. Hesler et al. (1993) and the theses of Puissegur (1976) and Foley (2015) are the only three known surveys of arthropod diversity in rice in the USA (California (Hesler et al. 1993) and Louisiana (Puissegur 1976; Foley 2015)), and none of these North American studies investigated effects of detrital subsidies. Surveys of Louisiana rice fields indicate that arthropod predators are present (Puissegur 1976; Foley 2015); the use of detrital subsidies may increase their abundance and reduce populations of nuisance organisms (primary consumers of rice).

The objective of this study was twofold. The first objective was to catalog the arthropod community in simulated rice paddies using multiple sampling methods. The second objective was to investigate the effects of detrital subsidies on arthropod communities in the simulated rice paddies. In particular, we were interested in whether detrital subsidies could elicit a trophic cascade in these simulated rice paddies, reducing the number of nuisance organisms. To determine if this was taking place, populations of rice water weevil (RWW), *Lissorhoptrus oryzophilus* (Kuschel), the major nuisance consumer of rice in Louisiana (Stout et al. 2001), were monitored. We expected that populations of RWW would be negatively affected by detrital subsidies due to the trophic cascade.

Material and methods

Study area

This study was conducted during the 2013 and 2014 rice growing season at the H. Rouse Caffey Rice Research Station, Crowley, LA, USA (30°14′23.9″N, 92°20′44.0″W) (Figure 1) on silt-loam soils (fine, montmorillonitic, thermic Typic Albaqualf). The research area is located in a subtropical region and has been used for rice cultivation and experimentation since 1908. Specific information on land use and cover prior to 1908 is lacking; however, most of this region was converted from tall grass prairie to agriculture between 1870 and 1900 (Allen et al. 2001), presumably including this site. Once converted to a research station the fields were on a rice-cattle rotation for some time, followed by a rice soybean rotation and then switched to a rice-fallow rotation. The experimental plots in which studies were conducted are in a two-year rice-fallow rotation for over 20 years. Earthen levees surrounded the plots with access to a lateral channel for irrigation.

2013 Field survey and detrital subsidy study

The experimental area was divided in 2013 into 16 leveed areas (plots) measuring 24.2 m × 4.9 m. Each plot had separate access to a lateral pipe for irrigation. Due to space constraints, plots were

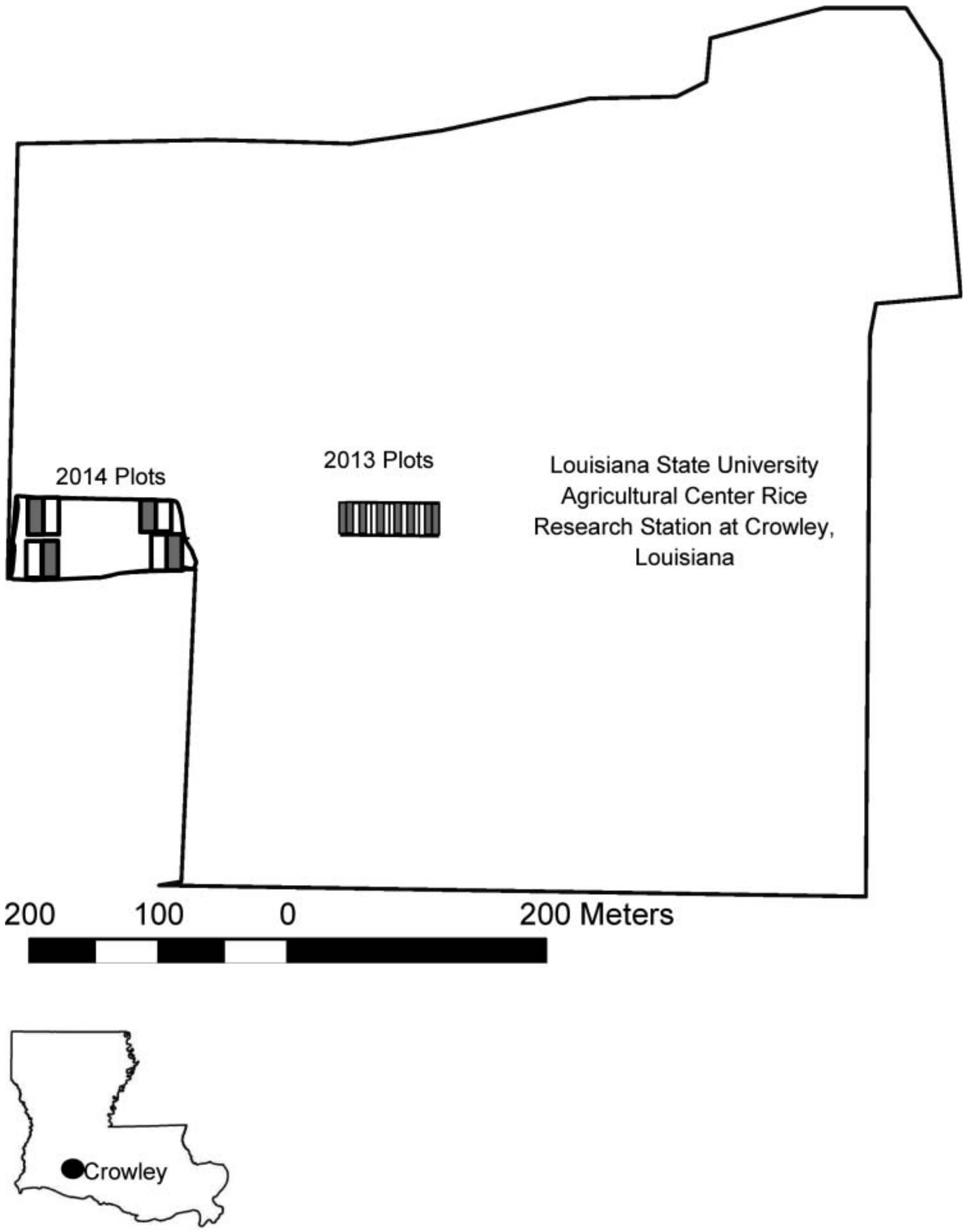

Figure 1. Louisiana State University AgCenter Rice Research Station, Crowley, LA, USA (30°14′23.9″N, 92°20′44.0″W. Areas containing research plots are marked for 2013 and 2014. Detrital subsidy plots are gray, control plots are white.

separated only by a 1 m wide earth levee that prevented water transfer between plots. Detrital subsidy or control status was randomly assigned, yielding eight control and eight detrital subsidy plots.

Detrital subsidies were applied to treatment plots in the form of Hopi-Gro Cow Manure and Compost Mix (Hopi-Gro, Hope Agri Products Inc., Hope, AR, USA). Detritus was hand cast into treatment plots on 26 March and 16 April at a rate of 18.5 kg per plot (1554 kg/ha) per date. All plots were flooded on 28 March with well water and floods were maintained for the duration of the experiment. On 25 April, all plots were seeded by hand casting with 1.4 kg of rice seed (cv. 'Cheniere') that had been pre-germinated following Blanche et al. (2012). While not a typical agronomic practice in southern Louisiana, water seeding with continuous flooding used to be consistent with the experimental practices of other detrital subsidy studies (Settle et al. 1996; Jiang & Cheng 2004).

The 29 d delay between flooding and planting was employed to allow aquatic predators to become established as RWW only oviposit under flooded conditions (Settle et al. 1996; Stout et al. 2002). No pesticides were used.

Sampling of plots for arthropods began on 4 April and was repeated every two weeks thereafter using four different methods: Gee crayfish traps, sweeps with D-nets, floating pitfall traps and root/soil corer sampling. A 42 cm × 23 cm 1.27 cm mesh cylinder Gee crayfish trap with 2.5 cm indented coned openings at each end was baited with a pellet of Purina Southern Pride crawfish bait (Purina Southern Pride, Shreveport, LA, USA) and placed in a random location within the plot 48 hrs before sampling. Aquatic netting was performed with a standard 30 cm, 500 micron mesh, D-net with one 2 m sweep in each plot. Invertebrates collected by these two methods were placed in 80% ethanol and taken back to the lab where they were sorted under a dissecting scope. Identification was done to the lowest taxonomic level practical using relevant keys and voucher specimens in the Louisiana State University Arthropod Museum (Arnett & Thomas 2000; Arnett et al. 2000; Triplehorn & Johnson 2005; Merritt et al. 2008).

Floating pitfall traps (see Parys & Johnson 2011 for trap design) were deployed at the time of flooding every 6 m along the midline of the plot and tethered to a 0.6 m landscape stake that was hammered into the sediment yielding three traps per plot. Traps were left out for the duration of the experiment and only removed for servicing. Each pitfall trap contained 2.5 cm of ethylene glycol (Prestone® green anti-freeze). Floating pitfall traps were serviced by pouring their contents into plastic bags (Whirl-Pak® (Nasco©), refilling them with ethylene glycol and placing them back in the field. Bag contents were identified and stored using the same methods as Gee traps and D-netting.

Coring was done at a single random location within each plot with a 9.2 cm × 7.6 cm metal corer. Cored samples were washed through a 40 mesh (0.47 mm) screen sieve bucket, and the remaining material was collected, taken back to the lab and stored at −20 C until sorted. Sorting sediment samples was done in a 5 cm deep pan filled with 2 cm of water and arthropods were identified using the same methods as described previously. These four invertebrate sampling methods resulted in six samples per plot for a total of 96 samples per sampling date. Labeled voucher specimens from all samples were deposited in the Louisiana State Arthropod Museum at Louisiana State University.

Once rice plants had reached the third leaf stage and emerged from the flood water, monitoring of RWW populations began by taking an additional root/sediment core during sampling (see Stout et al. 2001 for RWW larvae sampling and counting methods). One of these cores from each plot was randomly chosen and recollected for arthropod sampling using the previously mentioned methods.

2014 Field survey and detrital subsidy study

The 2014 field experiment was conducted in a manner similar to the previous year with some modifications. The experiment was performed at a different location within the same station that had been fallowed the previous growing season. Plot size was increased to 62.5 m × 5.8 m. The eight plots were divided into two blocks, with four plots on the north side of a central lateral and four plots on the south side with a minimum of 24.4 m between plots. Plots in each block were assigned to detrital subsidy or control treatments, with two plots of each treatment in each block.

Methods for applying detritus were similar to those used in 2013, but the number of applications was increased from two to four starting on 22 April, then on 16 and 30 May, and 12 June. Amount of detritus applied per date was increased from 18.5 to 72.6 kg per plot (2074 kg/ha) reflecting the increase in plot size, roughly 3× larger, as well as to increase ratio of detritus to plot area. All plots were flooded permanently on 22 April. On 23 May, plots were seeded using the same methods as the previous year, except 3.47 kg per plot was planted to keep rice density similar to 2013. A single application of a commercial herbicide and surfactant mixture was applied to each plot on 17 June due to heavy weed infestation, as the objective of this study was to quantify the biota of rice fields, not grass wetlands. The herbicide and surfactant mixture consisted of 0.0167 ml 3-(1-methylethyl)-

1H,1,3-benxothiadiazin-4(3H)-one 2, 2-dioxide (Basagran) and 0.0079 ml Alkyl phenol ethozylate (Voyager) per liter of water with a total rate of 6.3 l per plot. No other pesticides were applied.

Methods for sampling arthropods were similar to those used in 2013 with some exceptions, mostly to account for the larger plot size. The number of floating pitfall traps per plot was increased from three to four and traps were spaced 12.2 m apart instead of 6 m. A second Gee trap was added as well as an additional D-net sweep. Thus, a total of eight samples were taken from each of eight plots on each sampling date. The sediment corer was not used in 2014 as it was time consuming and yielded almost no arthropods (0.007% of individuals found in 2013). Moreover, those arthropods found in core samples in 2013 were not unique to that sampling method and were unlikely to normally inhabit the sediment (e.g. Corixidae).

Concerns were raised during the 2013 field season that detrital subsidies were altering water quality, potentially confounding the experiment. Chlorophyll *a* and dissolved oxygen measurements were added in 2014 as a measure of water quality (King & Brazner 1999; Cochard et al. 2014). Chlorophyll *a* concentration was measured by taking a 1 l water sample from each plot in the early afternoon of the sample day. Water filtration was performed by the Freshwater Ecology Lab, School of Renewable Natural Resources, by filtering through 0.45 micro-m filter (Standard Method 10200 H; APHA 2012) under vacuum, and water samples were analyzed by fluorometry (TD 700, Turner Designs, Inc., San Jose, CA, USA) for chlorophyll *a* content (minimum 0.02 micro-g/l; EPA [Environmental Protection Agency] method 445.0) by the Wetland Biogeochemistry Institute Analytical Services, Louisiana State University, Baton Rouge, LA, USA. Measurement of dissolved oxygen was taken using a HACH Dissolved Oxygen Test Kit Model OX-2P. RWW monitoring was the same in 2014 as in 2013 with the exception that three instead of two root/substrate cores were taken. Cores were used solely for RWW monitoring in 2014.

Data analysis

Principal component analysis, detrended correspondence analysis and non-metric multidimensional scaling (NMDS) were used to compare arthropod communities between detrital treatments. To select the ordination most appropriate for the data, axes length and goodness of fit (STRESS) from these analyses were assessed based on criteria described in ter Braak (1995), Leps and Smilauer (2003), and Hirst and Jackson (2007). Multi-response permutation procedure (MRPP) was then performed to determine differences between treatment and controls plots.

A sample-based rarefaction curve was computed for each treatment using EstimateS v. 9.1.0 (Colwell 2013). EstimateS was performed with 100 randomizations of species orderings without replacement and 10 individuals as the cut-off for rare species. Differences among treatments in predator populations (determined by Merritt et al. (2008) and Triplehorn and Johnson (2005)) were analyzed by a general linear model (abundance was log transformed) with correlated errors to account for sequential sampling using R v 3.0.3 with the package GEE v 4.13-18 (R Core Team 2014; Carey et al. 2015). RWW populations, chlorophyll *a* and dissolved oxygen were compared among treatments with analysis of variance in R v 3.03 (R Core Team 2014).

Results

Rice field community composition

Over 14 total sampling dates in two growing seasons, 1057 samples (687 in 2013 and 370 in 2014) were taken, yielding 143 different species spanning 13 orders for a total of 49,251 individuals (Table 1). Of the 13 orders found in the 1057 samples, Collembola represented 57.76% (27,616) of the individuals collected, with Diptera being the next most abundant order at 24.13% (11,886). Of the Diptera collected, 87.35% (10,383) were Chironomidae. Araneae (5.83%, 2873) and Coleoptera (5.87%, 2889) were the next most abundant orders (Table 1). Floating pitfall traps caught the greatest number of individuals over both years, 44,297, while aquatic netting caught 3898 and Gee traps found 981.

Table 1. Arthropod taxa sampled from rice fields in 2013 and 2014 and their total abundance found in each year in both their manure and control plots. The differences in numbers of Collembola, Carabidae and Staphylinidae in 2013 and 2014 are attributable to differences in the resolution to which taxa were identified in 2013 and 2014.

Order	Family	Genus	Species	2013 Control	2013 Manure	2014 Control	2014 Manure
Collembola				5022	5141	0	0
	Isotomidae			0	0	5812	5358
	Sminthuridae			0	0	3424	2859
Ephemeroptera	Baetidae	*Callibaetis*		9	13	10	4
				234	259	166	192
	Caenidae	*Caenis*		3	2	2	2
Odonata	Aeshnidae			0	0	1	2
		Anax		28	17	30	32
	Coenagrionidae			39	42	35	40
		Nehalennia		72	111	6	11
	Corduliidae			0	0	5	3
	Lestidae	*Lestes*		0	0	3	1
	Libellulidae			0	0	0	2
		Erythemis		0	0	2	0
		Orthemis		0	0	1	1
		Pantala		6	3	21	47
Orthoptera	Nemobiinae (SF)			1	0	0	2
	Gryllotalpidae	*Neocurtilla*	*hexadactyla*	4	9	0	2
Hemiptera	Aphididae			10	8	6	14
	Belostomatidae			71	34	19	18
		Belostoma	*fusciventre*	26	31	16	15
			lutarium	19	24	12	16
		Lethocerus		0	0	12	0
			uhleri	7	11	12	7
	Cicadellidae		australis	16	15	11	9
	Corixidae			40	40	11	17
		Trichocorixa		71	80	183	152
		Ramphocorixa		28	17	8	16
	Delphacidae			1	4	0	1
	Gerridae	*Neogerris*	*hesione*	1	1	0	0
	Hebridae	*Hebrus*	*consolidus*	1	1	2	2
	Hebridae	*Lipogomphus*		0	0	1	0
		Merragata	*brunnea*	0	0	1	0
			hebroides	0	0	2	0
	Hydrometridae	*Hydrometra*		6	1	0	0
	Largidae			2	0	2	2
	Lygaeidae			0	0	0	1
	Mesoveliidae	*Mesovelia*	*mulsanti*	2	1	5	7
	Nepidae	*Ranatra*		0	0	0	1
			australis	5	6	3	3
	Notonectidae	*Buenoa*		116	151	51	121
		Notonecta		119	111	64	93
	Pentatomidae			1	3	0	0
	Reduviidae			0	0	0	1
	Saldidae	*Micracanthia*	*humilis*	3	5	3	4
		Pentacora		0	0	0	1
	Scutelleridae			0	0	2	0
	Tingidae			0	0	1	0
	Veliidae	*Microvelia*		6	9	12	9
		Platyvelia	*brachialis*	0	2	1	2
Thysanoptera				24	31	36	38
Psocoptera				8	4	1	1

(*continued*)

Table 1. (*Continued*)

Order	Family	Genus	Species	2013 Control	2013 Manure	2014 Control	2014 Manure
Coleoptera	Carabidae			26	33	0	0
		Acupalpus		0	0	24	2
			indistinctus	0	0	22	21
		Aspidoglossa	*subangulata*	0	0	0	2
		Bembidion		0	0	44	51
		Chlaenius	*perplexus*	0	0	1	2
		Clivinia		0	0	2	3
		Elaphropus		0	0	0	2
		Semiardistomis	*viridis*	0	0	2	1
		Spp. 1		0	0	1	1
		Spp. 2		0	0	0	1
		Spp. 3		0	0	5	1
	Chrysomelidae			9	10	0	9
	Chrysomelidae	*Lysathia*	*ludoviciana*	0	0	16	12
		Chmorph1		0	0	0	2
	Curculionidae	Spp. 1		6	5	16	7
		Lissorhoptrus	*oryzophilus*	219	261	95	73
		Scolytinae		0	0	2	0
	Dytiscidae	*Bidessonotus*		2	1	4	2
		Copelatus		2	1	4	2
		Cybister		4	6	0	0
			fimbriolatus	10	10	14	11
		Desmopachria		0	1	1	1
		Hydroporus		1	4	0	0
		Laccophilus		41	19	15	19
		Liodessus		0	1	0	0
		Neoporus		3	1	1	0
		Thermonectus		6	2	6	9
			basillaris	5	3	19	24
			nigrofasciatus	2	1	15	12
	Elateridae			0	1	1	1
	Endomychidae			0	0	0	1
	Haliplidae	*Haliplus*		2	0	0	1
	Heteroceridae	*Tropicus*	*pusillus*	3	3	9	6
	Hydrophilidae	*Berosus*		27	37	45	63
		Enochrus		2	4	22	18
		Helophorus		45	39	46	38
		Hydrobius	*tumidus*	0	0	0	1
		Hydrophilus		1	0	0	1
			triangularis	19	25	12	16
		Paracymus		0	1	2	2
		Phaenonotum		3	2	0	0
		Tropisternus		73	77	40	38
			blatchleyi	0	0	0	1
			collaris	25	25	56	39
			lateralis	45	97	103	150
	Lampyridae			0	0	1	0
	Latridiidae			2	0	3	0
	Nitidulidae			2	0	0	0
	Noteridae	*Hydrocanthus*	*oblongus*	8	4	5	1
	Scarabaeidae			0	3	3	5
	Silvanidae			0	0	0	3
	Staphylinidae			30	25	0	0
		Spp. 1		0	0	1	2
		Athetini (T)		0	0	1	2
		Bisnius		0	0	7	4
		Carpelimus		0	0	75	25
		Philonthus		0	0	2	2
	Tenebrionidae			1	0	2	3

(*continued*)

Table 1. (*Continued*)

Order	Family	Genus	Species	2013 Control	2013 Manure	2014 Control	2014 Manure
Neuroptera	Chrysopidae			0	1	0	0
Hymenoptera	Spp. 1			0	0	0	1
	Argidae			1	0	0	0
	Braconidae			3	6	1	5
	Diapriidae			5	15	6	2
	Encyrtidae			2	0	1	0
	Eulophidae			4	4	2	2
	Figitidae			1	1	2	0
	Formicidae			12	8	31	19
	Ichneumonidae			0	3	1	0
	Mymaridae			12	18	2	1
	Platygastridae			11	8	10	7
		Baeus		0	0	7	10
	Tiphiidae			0	2	0	0
	Vespidae			0	0	1	0
Trichoptera	Hydroptilidae			7	12	21	23
Lepidoptera	Pyralidae			0	0	11	9
Diptera	Spp. 1			0	0	1	1
	Calliphoridae			2	0	3	1
	Cecidomyiidae			0	0	1	0
	Ceratopogonidae			38	24	19	15
		Alluaudomyia		0	0	1	1
	Chaoboridae	*Chaoborus*		0	1	6	92
	Chironomidae			2585	3219	2152	2427
	Culicidae			11	16	7	7
	Dolichopodidae			5	9	18	9
	Muscidae			17	19	46	54
	Mycetophilidae			0	0	1	0
	Phoridae			1	5	2	4
	Psychodidae			2	1	14	5
	Scatopsidae			0	1	1	0
	Simuliidae			1	0	1	0
	Stratiomyidae	*Odontomyia*		233	221	98	132
	Sciaridae			2	3	0	0
	Syrphidae			0	1	0	1
	Tabanidae			11	4	70	37
	Tachinidae			1	2	2	0
	Thaumaleidae			0	1	0	0
	Tipulidae			11	93	7	3
		Antocha		26	27	15	9
Araneae				792	680	189	213
	Dictynidae			0	0	0	1
	Lycosidae			598	528	288	252
		Pirata	*insularis*	0	0	115	86
			piraticus	0	0	58	42
	Salticidae			0	0	2	1
	Tetragnathidae	*Glenognatha*	*foxi*	7	4	58	62
		Tetragnatha	*laboriosa*	14	5	20	24

Note: SF indicates subfamily level identification and T indicates tribe level identification.

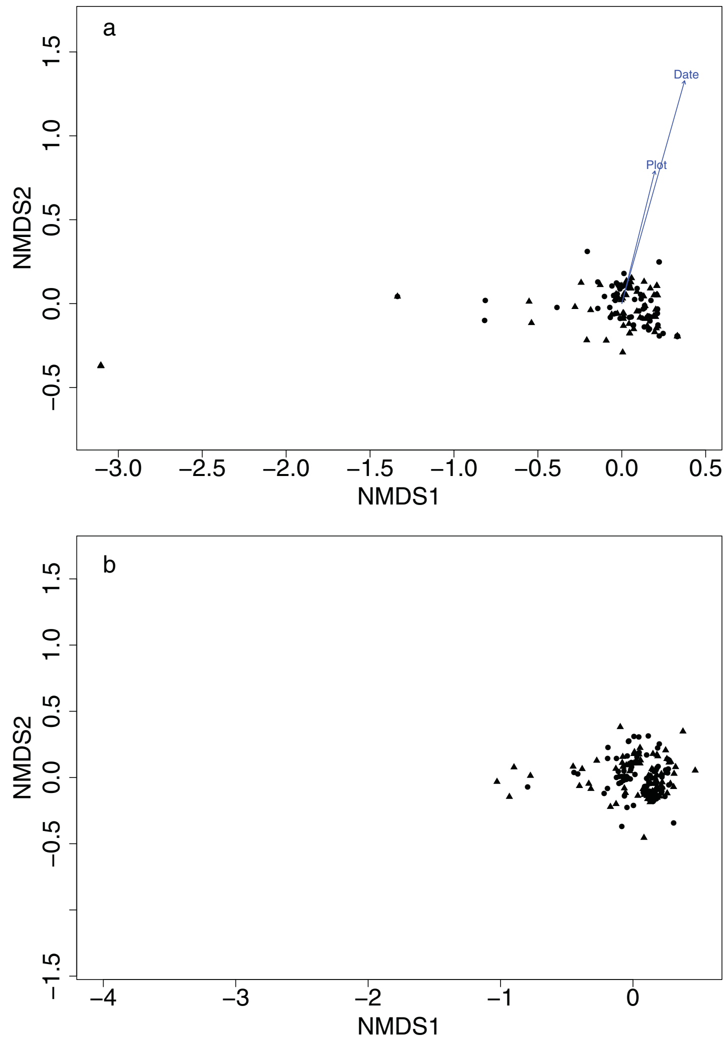

Figure 2. Non-metric multidimensional scaling of sites in rice fields between detrital subsidy plots (▲) and no-treatment (●) plots at the LSU AgCenter Rice Research Station, Crowley, LA, USA in 2013 (a) and 2014 (b). Date and Plot (arrows) indicate that the date and the plot the sample was taken is a strong predictor of community composition.

2013 Detrital subsidy study

Samples were taken on eight dates from 4 April to 16 July in 2013. NMDS found no differences in arthropod communities among control and detrital subsidy plots (STRESS = 0.11) (Figure 2(a)). MRPP agreed with this assessment, finding no significant differences in communities between treatments ($F_{1,2} = 0.69$, $p = 0.83$). Date and plot were significant predictors of community composition, irrespective of treatment, meaning that some taxa were more strongly associated with certain plots and dates (Figure 2(a)).

To generate sample-based rarefaction curves, the level of taxonomic resolution of adult insects and their immature stages must be consistent. For analysis of data in this study, higher-resolved taxa were downgraded to match less resolved life stages. For example, adult *Tropisternus* specimens were initially identified to species (*T. lateralis* and *T. collaris*), whereas immature *Tropisternus* sp. were not identified past genus. Consequently, for rarefaction curves, all *Tropisternus* counts were

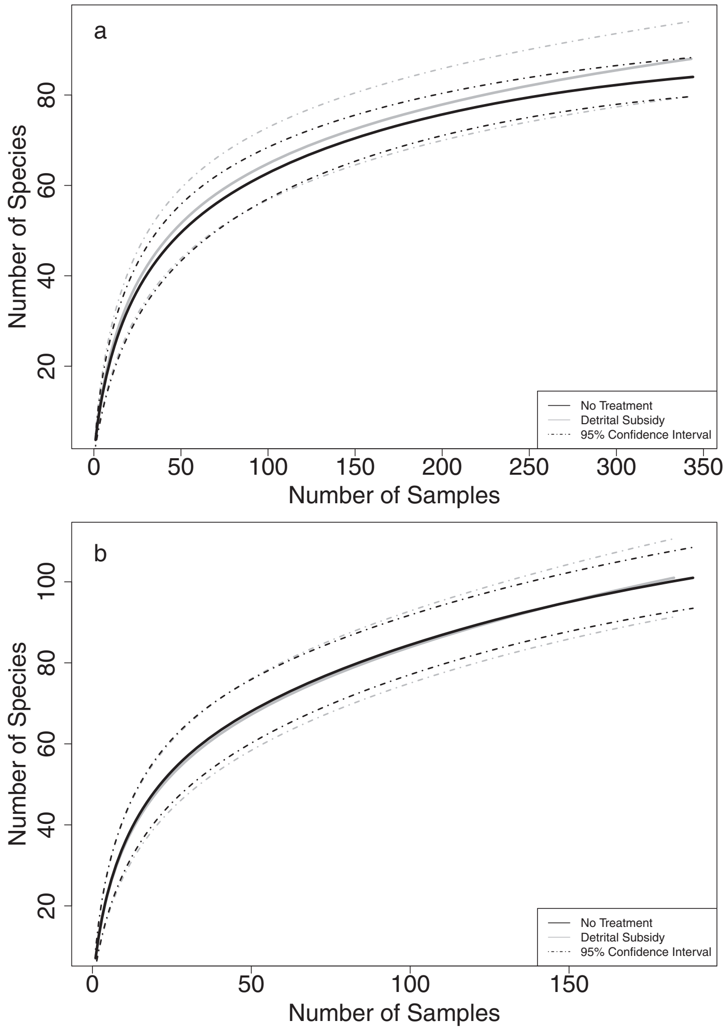

Figure 3. Sample based rarefaction curves for 2013 (a) and 2014 (b), comparing number of species expected to be found in a given number of samples in detrital subsidy and control plots. Note the difference in *x*- and *y*-axis scale.

combined to the genus level. Comparison of control and treatment curves found no difference between treatments (Figure 3(a)).

Predator populations, following the classification of Merritt et al. (2008), did not differ between detrital subsidy and control plots ($Z = -0.23$, $p = 0.22$). RWW larvae counts were taken on the last four sampling dates, from 6 June to 16 July. RWW larvae numbers did not vary significantly among detrital subsidy and control plots on any sample date (6 June 2013 – Detritus 5.4 (±0.84 SE), Control 7.5(±1.1 SE); 20 June 2013 – Detritus 15.64 (±1.65 SE), Control 15.38 (±2.26 SE); 5 July 2013 – Detritus 12.38 (±1.08 SE), Control 9.82 (±0.98 SE); 18 July 2013 – Detritus 5.44 (±1.19 SE), Control 4.87 (±0.87 SE)).

2014 Detrital subsidy study

Samples were taken on six dates from 7 May to 16 July in 2014. NMDS and MRPP again found no differences among arthropod communities in control and detrital subsidy plots (STRESS = 0.14; MRPP, $F_{1,2} = 0.83$, $p = 0.58$) (Figure 2(b)). Chlorophyll *a* concentrations did not differ between control and treatment plots ($F_{1,2} = 0.97$, $p = 0.33$). Similarly, dissolved oxygen levels were consistently 7 mg/l across plots of both treatments for all sampling dates. Monitoring chlorophyll *a* concentrations and dissolved oxygen ended on 18 June because compost-manure was no longer being added.

In 2014, the same taxonomic resolution adjustments were made as in 2013 for sample-based rarefaction curves. Comparison of treatment and control plots found no difference between rarefaction curves (Figure 3(b)). Due to heavy weed pressure during the 2014 season, even after herbicide application, rice plants did not grow well and RWW larvae monitoring found only a few scattered larvae. Effects of detrital subsidies on abundance of RWW larvae were not analyzed for 2014. As in 2013, there was no effect of detrital subsidy on predators between treatments in 2014 ($Z = -1.23$, $p = 0.66$).

Discussion

The objectives of this study were to catalog arthropod diversity in farmed wetlands and to examine the influence of detrital subsidy on insect assemblages in rice fields. Samples from control plots indicate that insect families found in farmed wetlands are highly variable from year to year. Detrital subsidy did not substantially alter insect assemblages in either study year. These results suggest that rice fields are resilient to change from outside inputs, such as detritus.

RWW numbers, predator numbers, chlorophyll *a* concentration and dissolved oxygen levels were also unaffected by detrital subsidy. The failure to observe an effect of detrital subsidy was unlikely to be the result of insufficient sampling, as four different sampling methods were used and a large number of samples were taken (Meyer et al. 2011). The lack of response to detritus was unexpected, as subsidies to aquatic systems have resulted in alterations to the community in prior studies, including rice (Settle et al. 1996; King & Brazner 1999; Jiang & Cheng 2004; Cross et al. 2006; Ogren & King 2008). Moreover, the two most prevalent groups collected in this study, Collembola and Chironomidae, are largely composed of detritivores and were expected to respond to subsidies (Merritt et al. 2008). There are a few possible explanations for the failure of detrital subsides to elicit a trophic cascade. One possible explanation is that not enough detrital subsidy was applied to the field, as previous studies in rice have found a trophic cascade (Settle et al. 1996; Jiang & Cheng 2004). The total subsidy applied in 2014, however, exceeded quantities used in either of the previously mentioned studies which may indicate that another factor is limiting arthropod response.

A second possibility is that using another form of detritus, such as mulch, may elicit cascades more effectively than manure (Halaj & Wise 2002; Mathews et al. 2004; Schmidt et al. 2004). Another possibility is that experimental plots were too small for detritus to noticeably affect diversity, and scaling plots up to a whole field may be required to elicit trophic cascades (Garratt et al. 2011). This latter explanation does not seem plausible, as Settle et al. (1996) used plots measuring 20 m × 20 m and found a significant effect. The duration of the experiment may not have been sufficient to observe an effect, and monitoring for a longer period of time may have resulted in a change in arthropod community as secondary consumers may have a delay in their response to detrital subsidies (O'Brien et al. 1992; Cross et al. 2007). In Asia, where studies have found effects of detrital subsidies, rice is cultivated year-round (>1 crop per year). Moreover, rice paddies are flooded for a greater portion of the season and plants are transplanted into flooded fields (Fernando 1993). Due to this, the rice ecosystem in Asia may represent a more permanent habitat. Rice in the USA is much more of a transient habitat, one crop per year, and the long period between seeding and permanent flooding does not allow a permanent community to establish.

Alternatively, detritus may not be limiting in this system; rather, other factors, such as inherent wetland insect resilience to change (e.g. Brock et al. 2003; Batzer 2013) and biotic interactions (Batzer & Rhui 2013), may be structuring diversity (Halaj & Wise 2002; Batzer & Palik 2007). The agricultural research station where this study occurred has been used for rice research since 1908. Continuous use over the years can possibly alter the local arthropod community. Anthropogenic alterations in land use can change a biotic community to one that is composed of species more tolerant of disturbances, such as detrital subsidies (Harding et al. 1998; Baumgartner & Robinson 2015). Harding et al. (1998) looked at past and current watershed use, finding that habitat type (agriculture or forest) in the 1950s was a better predictor of current community structure than current land use, indicating that agriculture causes lasting alterations to the habitat. Baumgartner and Robinson (2015) found that detrital subsidies were insufficient to alter macroinvertebrate stream communities that had a history of agriculture association, most likely because the current community is composed of less sensitive taxa. Evidence that this change may have taken place is provided by Foley (2015), who found Cambaridae (Decapoda) as a common taxon in commercial rice fields. Cambaridae is susceptible to insecticides used in rice (Barbee & Stout 2009) and completely absent from the current study which suggests that the local habitat is no longer suitable. The complete absence of many other non-insect arthropods or non-arachnids, such as Amphipoda and Isopoda, was unexpected and also points to alterations in habitat. Moreover, qualitative comparisons to a recent survey of natural coastal wetlands suggest numerous other taxonomic differences, specifically with natural wetlands supporting four Odonata genera, one Hemiptera genera, seven Coleoptera genera, one Megaloptera genera and two Trichopera genera not found in this study (Kang 2011). Taken together, taxonomic differences between this study with Foley (2015) and Kang (2011) reinforce the premise that habitat may have been altered. Performing this study in commercial fields that have a shorter history of rice cultivation than the current site may give a better indication the suitability of rice fields as surrogate wetlands.

RWW larval populations did not differ between treatments in 2013 and monitoring was unsuccessful in 2014. This study provided no information of the potential for biological control of RWW by aquatic invertebrates, primarily because detrital subsidies failed to generate a trophic cascade. Another possible hindrance to biological control is the late appearance of natural enemies in plots, even with an extended pre-flood. Many of the large predators, which are most likely to prey on adult and larval RWW, were not found in plots until July. RWWs, in contrast, are most abundant and most important as pests early in the growing season, when rice is smaller and more vulnerable to RWW attack (Stout et al. 2012). RWWs are present during mid to late summer (when large predators are present), but they are not as important pests at that time because rice is larger and because they begin to migrate back to overwintering sites at that time (Tindall & Stout 2003). Our results conflict with the majority of findings that detrital subsidies in wetlands generate a trophic cascade (Settle et al. 1996; Jiang & Cheng 2004; Cross et al. 2006; Leroux & Loreau 2008; Ogren & King 2008; Hagen et al. 2012; but see Batzer & Palik 2007; Batzer 2013), although the possibility of generating a cascade with an increase in detritus cannot be ruled out. Trophic cascades are desirable as they can aid in biological control of nuisance and invasive organisms and reduce the need for chemical applications to fields. Reducing the amount of chemicals applied to a rice field would likely increase its ability to act as a surrogate wetland (Lawler 2001) and help to mitigate the loss of unmanaged wetland habitat.

This study constitutes the most comprehensive survey of arthropods in US rice paddies to date, as it used multiple sampling methods that sampled different rice paddy microhabitats (Meyer et al. 2011). This study demonstrated that intensively farmed wetlands may differ from less intensively farmed Gulf coast wetlands and are potentially not capable of supporting a trophic cascade response to detrital additions, unlike less intensively cultivated wetlands in other regions. Further arthropod surveys preferably in commercial rice plots and with increased duration of monitoring and detrital subsidies can elucidate possible ecosystem services.

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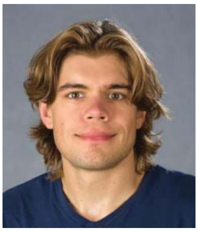

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