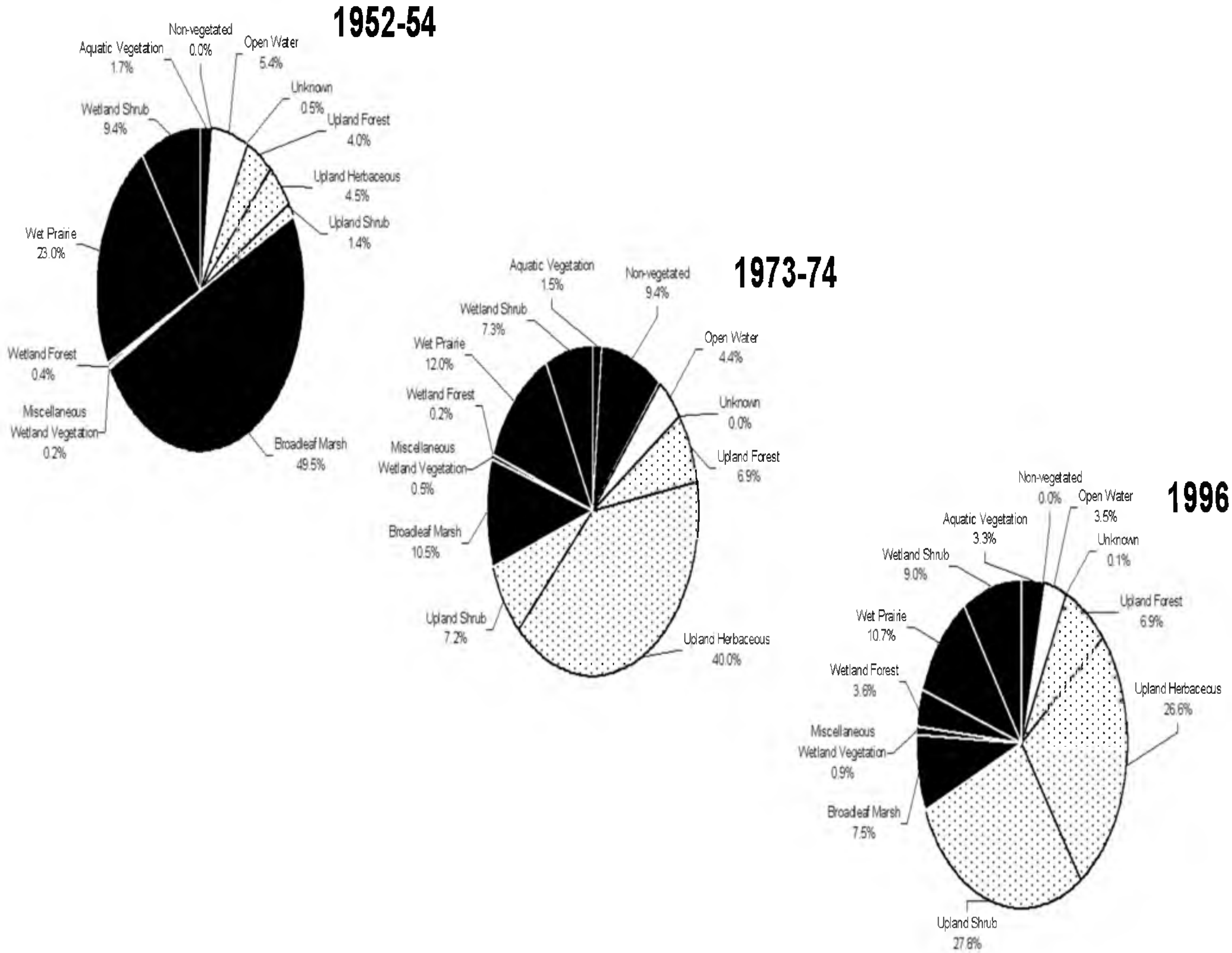

Figure 10-2. Percentages of floodplain cover of Bcode Groups in Pool C in the pre-channelization (1952–1954), early post-channelization (1973–1974), and 1996 vegetation maps.

Table 10-3. Areal extent of Community types (Bcodes) in the Kissimmee River Pool C floodplain, 1996.

Habitat	Bcode Group	Bcode Group name	Bcode	Bcode name	Area (ha)	Percent of Bcode Group	Percent of Pool C
Aquatic	AQ	Aquatic Vegetation	H.EC	Eichhornia crassipes herbaceous aquatic vegetation	2.0	1.4	0.0
			H.EC-PST	Eichhornia crassipes-Pistia stratiotes herbaceous aquatic vegetation	0.1	0.0	0.0
			H.HU	Hydrocotyle umbellata herbaceous aquatic vegetation	0.3	0.2	0.0
			H.MFM	Miscellaneous herbaceous floating mat vegetation	4.4	2.9	0.1
			H.MxFA	Miscellaneous aquatic vegetation dominated by floating species	1.9	1.3	0.0
			H.MxM	Miscellaneous littoral marsh vegetation	12.5	8.3	0.3
			H.MxSV	Miscellaneous submergent aquatic vegetation	2.2	1.5	0.0
			H.NL	Nuphar lutea herbaceous aquatic vegetation	16.7	11.1	0.4
			H.PD	Polygonum densiflorum herbaceous aquatic vegetation	27.4	18.2	0.6
			H.PST	Pistia stratiotes herbaceous aquatic vegetation	0.2	0.2	0.0
			H.SCF	Scirpus cubensis herbaceous floating mat vegetation	32.7	21.7	0.7
			H.SS	Sacciolepis striata herbaceous aquatic vegetation	5.5	3.7	0.1
			S.LSF	Ludwigia spp. floating mat shrubland	29.0	19.3	0.6
			S.MCF	Myrica cerifera floating mat shrubland	6.9	4.6	0.2
			S.MxFS	Miscellaneous floating mat shrubland	8.7	5.8	0.2
Non vegetated	NVBG	Non Vegetated, Bare Ground	NVBG	No vegetation - bare ground	1.2	0.7	0.0
			NVH	No vegetation - human-made structures, roads, etc.	3.2	1.9	0.1
			NVOW	No visible vegetation - open water	161.9	97.4	3.5
Unknown	UN		XUNCL	Unknown - Unclassified	4.9	78.4	0.1
			XUNK	Unknown - Uninterpretable signature	1.3	21.6	0.0
Upland	UF	Upland Forest	F.MxF	Miscellaneous upland forest	1.7	0.5	0.0
			F.QS	Quercus virginiana (-Sabal palmetto) forest	295.7	94.2	6.5
			F.SP	Sabal palmetto forest	16.6	5.3	0.4
	UP	Upland Herbaceous	H.AF	Axonopus fissifolius herbaceous vegetation	64.4	5.3	1.4
			H.MxE	Miscellaneous exotic herbaceous vegetation	74.7	6.1	1.6
			H.MxN	Miscellaneous native herbaceous vegetation	132.1	10.8	2.9
			H.MxW	Miscellaneous invasive herbaceous vegetation	109.5	9.0	2.4
			H.PN	Paspalum notatum herbaceous vegetation	838.0	68.8	18.3
	US	Upland Shrub	S.MC	Myrica cerifera shrubland	981.4	77.0	21.4
			S.MxUS	Miscellaneous upland shrubland	203.3	16.0	4.4
			S.PG	Psidium guajava shrubland	7.1	0.6	0.2
			S.SR	Serenoa repens shrubland	8.3	0.7	0.2
			S.ST	Schinus terebinthifolius shrubland	73.7	5.8	1.6
Wetland	BLM	Broadleaf Marsh	H.PS	Pontederia cordata-Sagittaria lancifolia herbaceous vegetation	159.6	46.7	3.5
			H.PS-CO	Pontederia cordata-Sagittaria lancifolia-Cephalanthus occidentalis herbaceous vegetation	25.2	7.4	0.6
			H.PS-HG	Hibiscus grandiflorus-Pontederia cordata-Sagittaria lancifolia herbaceous vegetation	1.0	0.3	0.0
			H.PS-PH	Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon herbaceous vegetation	69.6	20.4	1.5
			H.PS PH CO	Pontederia cordata Sagittaria lancifolia Panicum hemitomon Cephalanthus occidentalis herbaceous vegetation	86.1	25.2	1.9
	MW	Miscellaneous Wetland Vegetation	H.CJ	Cladium jamaicense herbaceous vegetation	0.9	2.3	0.0
			H.HG	Hibiscus grandiflorus herbaceous vegetation	15.3	37.5	0.3
			H.SB	Spartina bakeri herbaceous vegetation	13.4	33.0	0.3
			H.TY	Typha domingensis herbaceous vegetation	11.1	27.3	0.2
	WF	Wetland Forest	F.AR	Acer rubrum (-Nyssa silvatica var. biflora) forest	120.8	73.6	2.6
			F.FC	Fraxinus caroliniana forest	1.9	1.1	0.0
			F.MTF	Mixed transitional forest	22.0	13.4	0.5
			F.MV	Magnolia virginiana forest	3.1	1.9	0.1
			F.TD	Taxodium distichum forest	16.4	10.0	0.4
	WP	Wet Prairie	H.AG	Andropogon glomeratus herbaceous vegetation	14.0	2.9	0.3
			H.CS	Cyperus spp. herbaceous vegetation	4.6	0.9	0.1
			H.ES	Eleocharis spp. herbaceous vegetation	12.2	2.5	0.3
			H.IV	Iris virginica herbaceous vegetation	18.0	3.7	0.4
			H.JEu	Juncus effusus herbaceous vegetation (upland depressions)	12.0	2.5	0.3
			H.JEp	Juncus effusus herbaceous vegetation (wet prairies)	110.8	22.6	2.4
			H.LF	Luziola fluitans herbaceous vegetation	81.6	16.6	1.8
			H.MxWT	Miscellaneous transitional herbaceous wetland vegetation	142.6	29.1	3.1
			H.MxWT	Miscellaneous native wetland graminoid vegetation	39.8	8.1	0.9
			H.PH	Panicum hemitomon herbaceous vegetation	24.5	5.0	0.5
			H.PP	Polygonum punctatum herbaceous vegetation	20.8	4.2	0.5
			H.PR	Panicum repens herbaceous vegetation	0.8	0.2	0.0
			H.RN	Rhynchospora spp. herbaceous vegetation	8.7	1.8	0.2
	WS	Wetland Shrub	S.CO	Cephalanthus occidentalis shrubland	10.3	2.5	0.2
			S.CO-PS	Cephalanthus occidentalis-Pontederia cordata-Sagittaria lancifolia shrubland	28.5	6.9	0.6
			S.CO-PS-PH	Cephalanthus occidentalis-Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon shrubland	156.6	37.9	3.4
			S.HF	Hypericum fasciculatum shrubland	1.0	0.2	0.0
			S.LS	Ludwigia spp. shrubland	126.9	30.7	2.8
			S.SC	Salix caroliniana shrubland	90.2	21.8	2.0
Wetland/Upland	VN	Vine	V.LM	Lygodium microphyllum-dominated communities	0.1	3.8	0.0
			V.MxV	Miscellaneous vine-dominated communities	2.9	96.2	0.1
					4,583.0		100.0

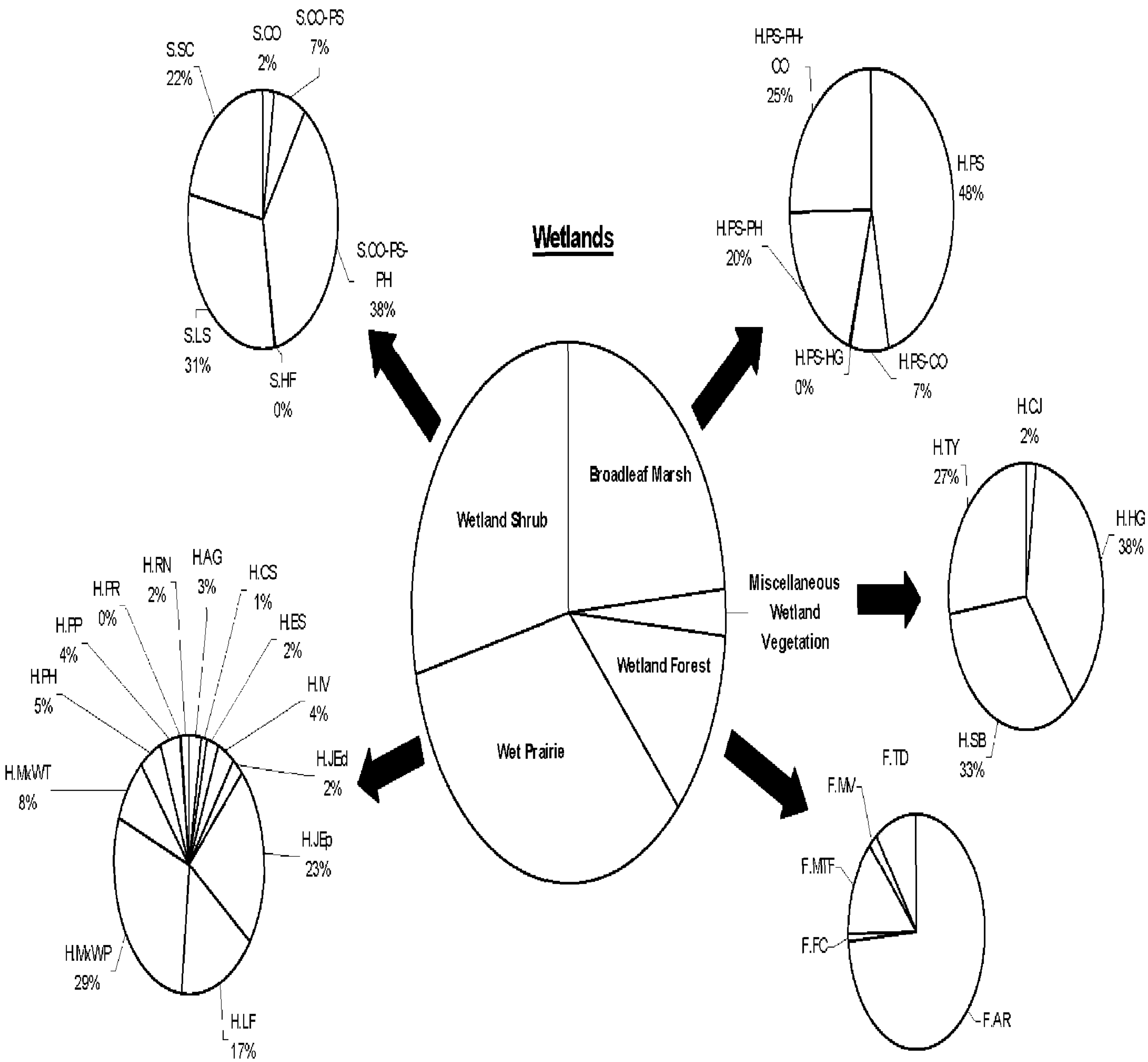

Figure 10-3. Community type percentages (outer circles) of wetland Bcode Groups (central circle) in the 1996 Pool C Vegetation Map. Community type codes are defined in Table 10-3.

The most extensive community type in the UP Group after channelization was pastures dominated by *Paspalum notatum* (bahia grass) (H.PN), which accounted for 69% of this Group and 18% of the total mapped area in Pool C (Table 10-3, Figure 10-3). Second-most common was the Mixed Native Herbacous community type (H.MxN), which covered 11% of Upland Herbaceous areas. This community type has no defined dominant taxa and may include mixtures of native upland graminoids such as *Andropogon virginicus, Panicum angustifolium*, some *Cyperus* spp., non-native and pasture grasses, and other herbaceous communities. Semi-woody annuals and perennials (e.g. *Lantana camara, Sesbania* spp., *Callicarpa Americana)* are often found scattered in these mixed native grasslands. Miscellaneous invasive communities (H.MxW), which are dominated by invasive native species, comprised 9% of the Upland Herbaceous Group in Pool C, and Miscellaneous Exotic Herbaceous communities (H.MxE) occupied 6% of the Group (Table 10-3, Figure 10-3).

Wetland Communities

Wetland Forests. As in other pools during the channelized period, in Pool C, Wetland Forest occurred mostly in the lower third of the pool (Figure 10-1), where hydroperiods were longer due to the backwater effect created by the pool's water control structure and related levees. The largest component of the Wetland Forest Group in channelized Pool C was the *Acer rubrum* (red maple) Forest, which accounted for 74% of the Wetland Forest Group (Table 10-2). *Acer rubrum* communities occurred in dense stands near river channels or mixed with other wetland tree and shrub species. *Taxodium distichum* (cypress) Forest (F.TD) comprised 10% of Wetland Forest, usually in stands in riparian zones of remnant river channels in the lower portion of Pool C (Table 10-3, Figure 10-3). *Magnolia virginiana* (sweetbay) (F.MV) and *Fraxinus caroliniana* (carolina ash) (F.FC) Forests combined, accounted for only 3% of this Group. *Magnolia virginiana* communities typically occurred as domes or "heads" in wet depressions within peripheral Upland Forest. *Fraxinus caroliniana* (carolina ash) communities occurred infrequently in dense clumps within wet depressions. The Miscellaneous Transitional Forested (F.MTF) community made up 13% of Wetland Forest in Pool C (Table 10-3, Figure 10-3). This type is typically composed of combinations of upland (e.g. *Quercus virginiana, Fraxinus caroliniana*) and wetland (e.g. *Persea* spp., *Taxodium distichum*, *Acer rubrum*) species, occurring in various situations but more often in wetland habitats than upland habitats. The F.MTF community was most abundant along the Istokpoga canal, which was excavated before channelization of the Kissimmee River (Figure 10-1).

Wetland Shrub Communities. The Wetland Shrub Group (WS) accounted for 9% of Pool C area in 1996 (Table 10-2, Figures 10-1 and 10-2). This group includes several community types with at least 30% cover of *Cephalanthus occidentalis* (buttonbush) (S.CO, S.PS-CO, S.PS-PH-CO, Table 10-3), which combined, made up over 47% of the Wetland Shrub Group (Table 10-3). These communities typically occurred with Broadleaf Marsh understories, in several associations differentiated by percent cover of the several dominants (Bousquin and Carnal 2005). These communities have a marsh-like appearance with a thin overstory of shrubs, occurring mainly in the south-central portion of the pool, west of C-38. *Salix caroliniana* (coastal plains willow) communities (S.SC) made up 22% of this Group (Table 10-3, Figure 10-3). *Ludwigia* spp. (primrose willow) communities (S.LS) accounted for 31% of the Wetland Shrub Group in Pool C. The majority of *Ludwigia* species in the river system is *L. peruviana* (Peruvian primrose willow) commonly found in or along abandoned channels, ditches, and at lower elevations. The species has benefited from stabilized hydrology and often occurs where willow has declined. *Ludwigia, Salix,* and *Myrica* shrub communities growing on floating mats formed by *Scirpus cubensis* (cuban bullrush) were differentiated separately and are discussed in the Aquatic Vegetation (AQ) Group, below. *Hypericum fasciculatum* (sandweed) communities (S.HF) typically occurred in the outer rings of upland marsh depressions but were not common (0.2% of the Wetland Shrub Group).

Broadleaf Marsh. The Broadleaf Marsh Bcode Group (BLM) includes five combinations of wetland forb and grass mixtures, although the dominant species of all of these types are *Pontederia cordata* (pickerelweed) and/or *Sagittaria lancifolia* (bulltongue arrowhead) (Bousquin and Carnal 2005) (Table 10-3). The *Pontederia cordata/Sagittaria lancifolia* (H.PS) community type contributed the largest area of the Broadleaf Marsh Group, at 47% of the Group (Table 10-3, Figure 10-3). The other major communities contained lesser coverage of these two species, but all include significant cover of *Panicum hemitomon* or *Cephalanthus occidentalis* (i.e. H.PS-PH, H.PS-CO, H.PS-PH-CO) (Table 10-3) and combined, accounted for over 53% of the Broadleaf Marsh Group. The Broadleaf Marsh community contains five combinations of forbs and wetland grass mixtures, although the dominant species for all of these types is *Pontederia* *cordata* (pickerelweed) and/or *Sagittaria lancifolia* (bulltongue arrowhead). The *Pontederia/Sagittaria* (H.PS) community was the largest constituent of the BLM habitat and must contain at least 50% cover of one or both species of *Pontederia* or *Sagittaria* in a polygon to be classified as such (Table 10-3, Figure 10-3). Other species commonly occurring in Broadleaf Marshes include *Cephalanthus occidentalis, Panicum hemitomon,* and *Hibiscus grandiflorus.*

Wet Prairie. The Wet Prairie Group includes communities with various combinations of graminoid and forb species. *Panicum hemitomon* (maidencane) communities (H.PH) and *Rhynchospora* spp. (beakrushes) communities (H.RN) were common Wet Prairie components in the pre-channelization system (Pierce et al. 1982), but together accounted for only 7% of Pool C Wet Prairie in 1996 (Table 10-3, Figure 10-3). *Panicum hemitomon* dominated a large region of MacArthur Impoundment in a west-central portion of Pool C prior to channelization (Figure 10-2). The ditch and levee system of this impoundment likely shortened hydroperiods and led to the dominance of maidencane in this area, which is surrounded by Broadleaf Marsh. *Polygonum punctatum* (dotted smartweed) (H.PP) and *Juncus effusus* (soft rush) communities (H.JEp and H.JEd) often occurred in wet depressions within pastures. The two *Juncus* community types accounted for 25% of Wet Prairie coverage in Pool C in 1996. The *Polygonum* community type was common in agricultural ditches and accounted for 4% of Pool C Wet Prairie. Like the *Juncus* communities, the *Iris virginica* (Virginia iris) community type (H.IV) was found around pasture depressions, but was less common and seasonal in occurrence. *Iris virginica* accounted for 4% of the Wet Prairie. It was mainly distributed in lower elevation pastures near Oak Creek (Figure 10-1). *Luziola fluitans*- (southern watergrass) dominated communities (H.LF) covered almost 17% of the Wet Prairie habitat in baseline Pool C. Stabilized water levels, pasture grass seeding, and grazing led to replacement of Wet Prairie species by other species including *Paspalum notatum, Axonopus* spp., and various species of weeds. In addition, some forbs with low forage value (e.g. *Pontederia cordata, Juncus effusus*) for cattle consumption (Pruitt et al. 1976) established where conditions were favorable. This was evident in pasture depressions containing remnant wetlands with mixed species of broadleaf marsh, and in wet prairies surrounded by heavily grazed pasture grasses.

Several Wet Prairie species that occurred in the channelized system (e.g., *Juncus effusus, Luziola fluitans, Phyla nodiflora, Centella asiatica, Iris virginica, Eleocharis spp., Andropogon glomeratus*) apparently did not occur as dominants in the pre-channelization wet prairies described by Pierce et al. (1982). These plants probably occurred infrequently in the pre-channelized system.

Luziola fluitans (southern watergrass) was common at lower elevations of pastures and depressions, often associated with *Polygonum punctatum* (dotted smartweed), smaller species of *Eleocharis* (spikerush), *Bacopa* spp. (hyssops), *Phyla nodiflora* (turkey tangle frogbit), *Hydrocotyle umbellata* (manyflower marshpennywort), *Centella asiatica* (spadeleaf), and occasionally with *Pontederia cordata, Sagittaria lancifolia,* and *Juncus effusus. Andropogon glomeratus* (bushy bluestem) is a grass preferring moist soils, but was found throughout the channelized system in pastures, floating mats, Upland Shrub communities, and disturbed areas. *Panicum repens* (torpedo grass) and *Leersia hexandra* (southern cutgrass), which often form dense mats in shallow water, also occurred in very small amounts on the floodplain.

Miscellaneous Wetlands. The Miscellaneous Wetland Group (MW) includes communities dominated by *Cladium jamaicense* (sawgrass) (H.CJ), *Typha domingensis* (southern cattail) (H.TY), *Spartina bakerii* (sand cordgrass) (H.SB), *Hibiscus grandiflorus* (swamp rosemallow) (H.HG), and a fern-dominated community (H.MxFN). The MW Group comprised only 0.9% of the total mapped area in Pool C (Table 10-2, Figure 10-2). *Hibiscus* communities were the largest component of this category, comprising about 38% of all MW communities (Table 10-3, Figure 10-3). *Cladium* communities made up 2%, *Typha* communities accounted for 27%, and *Spartina* communities comprised 33% of the MW Group. *Cladium jamaicense* communities occurred mostly in small patches within Broadleaf Marsh communities and was rare on the Pool C floodplain during baseline evaluation. *Typha domingensis* occurred in small areas in often dense clumps across many landscape zones (Appendix 10-1A), particularly in spoil or road ditches. *Spartina bakeri* prefers moist soils, and dominated communities found primarily on the periphery of the floodplain between wetland and upland habitats, where it often occurred in sparse linear expanses. No fern-dominated communities were mapped during baseline evaluation, although ferns are often abundant in the understory of shrub and Wetland Forest communities.

Mixed communities of grass and forb species, which occurred under various hydrologic conditions, and in which dominance is ambiguous or composition does not fit Community type decision rules, were grouped as miscellaneous transitional Wet Prairie (H.MxWP). Trends in species composition within this community may be further evaluated to more clearly define types of transitional Wet Prairies. A

Miscellaneous Wetland grass category (H.MxWT) was used to capture graminoids of mixed dominance, where identification was unclear, or the community is too rare to warrant a separate category, such as *Phragmites australis* (common reed), which occurred in small patches along remnant river channels and C-38.

Aquatic Communities

Aquatic Communities were defined as communities of plants that grow in permanently deep aquatic conditions, as opposed to wetlands which are inundated for only part of the year or that occur in shallow water or wet soil and are dominated by hydrophytic species (Cowardin et al. 1979). An exception to this definition is the communities that develop on floating mats that occur in the lower portions of pools, nonflowing remnant river channels, and abandoned channels under channelized conditions. These communities are difficult to characterize. In some cases, floating mats support normally upland (e.g., *Myrica cerifera*) or wetland species (e.g., various shallow-water rooted emergents), interspersed with fully aquatic species in mat openings of open water (such as the floating species *Pistia stratiotes* or *Salvinina minima*), resulting in recurring communities of species that confound aquatic/wetland/terrestrial distinctions. The Aquatic Vegetation Group (AQ) includes continuous floating mats formed by *Scirpus cubensis* (cuban bullrush) (Pierce et al. 1982), on which occur rooted aquatic vegetation, free floating plants, various marsh species, and shrubs. *Scirpus cubensis*- dominated floating mats (H.SCF, H.MFM) accounted for 25% of the Aquatic Vegetation Group.

Shrub-dominated floating mat community types dominated by *Ludwigia* spp. (S.LSF), *Myrica cerifera* (S.MCF), and occasionally *Salix caroliniana*, which are included in the Aquatic Vegetation Bcode Group, were found in the lower sections of pools, abandoned channels, and occasionally in remnant river channels.

Collectively, floating mat communities made up 3% of the total mapped area of Pool C (Table 10-3, Figure 10-3). The *Ludwigia* spp. type was the most common, accounting for 19% of the Aquatic Community Group and occurring mainly in abandoned channels. The aquatic emergent *Polygonum densiflorum* (denseflower smartweed) Community type (H.PD) made up approximately 18% of the Aquatic Vegetation Group, and *Nuphar lutea* (spatterdock) (H.NL), a rooted floating-leaf emergent, accounted for 11% of the Aquatic Group.

Non-vegetated, Human-Influenced, and Problematic Categories

Approximately 4% of the floodplain was unvegetated open water (Table 10-3, Figure 10-3). Approximately 72% of open water in the baseline 1996 data was located in C-38, while remnant river channels and other natural water habitats (e.g. abandoned river channels and depressions) made up almost 27% of the 1996 open water habitat.

The Non-Vegetated Bare Ground category (NVBG) was used to classify areas of sand or mud and the Non-Vegetated Human-Influenced Group (NVH), was used to represent features that were constructed, such as water control structures, houses and lawns, roads, farm complexes, and rip rap. The combined cover of these categories accounted for only 0.09% of the mapped area in Pool C. Polygons that were uninterpretable, or composed of rarely occurring species that do not fit community type decision rules, were grouped together as Unknowns (UN). This category was needed for only 0.2% of the mapped area in Pool C.

CONCLUSIONS

Floodplain vegetation shifted from dominance by wetland vegetation to dominance by upland communities as early as 1973-74 (Table 10-2, Figures 10-1 and 10-2), two to three years after the C-38 canal was completed. Prior to channelization, wetland vegetation occurred on over 80% of the floodplain's total area. By 1974, three years after completion of channelization, wetlands had declined to about 29% of the floodplain. Pre-channelization wetlands were dominated by herbaceous marshes, primarily Broadleaf Marsh and Wet Prairie, which occurred on 46% and 21% of the floodplain, respectively. Wetland Shrub communities (WS) covered 13% of the floodplain prior to channelization. By 1974, Broadleaf Marsh occurred on only 7% of the floodplain, Wet Prairie on 13%, and Wetland Shrub communities had declined to 8% of the floodplain. Much of the gross-level vegetation change in wetlands that took place in Pool C following channelization had occurred by the time the 1974 aerial photography was taken; little additional change in wetland plant communities had occurred in Pool C by 1996. The similarity of areal vegetation cover in Pool C, compared to the entire floodplain in the 1952 and 1974 maps, suggests that extrapolation of this finding to the entire floodplain is not unreasonable.

Much of the loss of wetlands described in this chapter is accounted for by conversion of marshes to upland pastures. These drained areas were used as improved (human-modified) or unimproved grazing lands. Opportunistic Upland Shrub species increased. *Myrica cerifera* occupied higher elevations of formerly long-hydroperiod marshes and sections of lower pools where dryer substrates of floating vegetation formed in permanently wet areas. *Schinus terebinthifolius* colonized banks of the canal and river channels.

These changes were largely a result of lost seasonal inundation of the floodplain marsh communities that had dominated the floodplain prior to channelization. Less important factors affecting the distribution and extent of vegetation included increases in the elevations of former wetland areas where spoil was dumped, loss of flow in riparian and other river channel habitats, development of "floating" substrates for non-aquatic species, and directly human-mediated factors such as introductions of cattle and forage grass species and suppression of shrubs in drained marshes.

Restoration Expectations

Three expectations were developed to predict vegetation change resulting from restoration (Figure 10-4). The restoration expectations are presented in Carnal (2005a, 2005b, and 2005c) by restoration construction phase. These predictions are based on coverage in the 1954 pre-channelization reference vegetation map, overlaid with restoration phase areas (Table 10-4, Map Appendix 9A). Wetland plant communities are expected to eventually comprise approximately 80% of the area restored in restoration Phases I–IV. Broadleaf Marsh communities are expected to cover 50% or more of the Phase I–IV area, and Wet Prairie communities are expected to cover at least 17% of the Phase I–IV area.

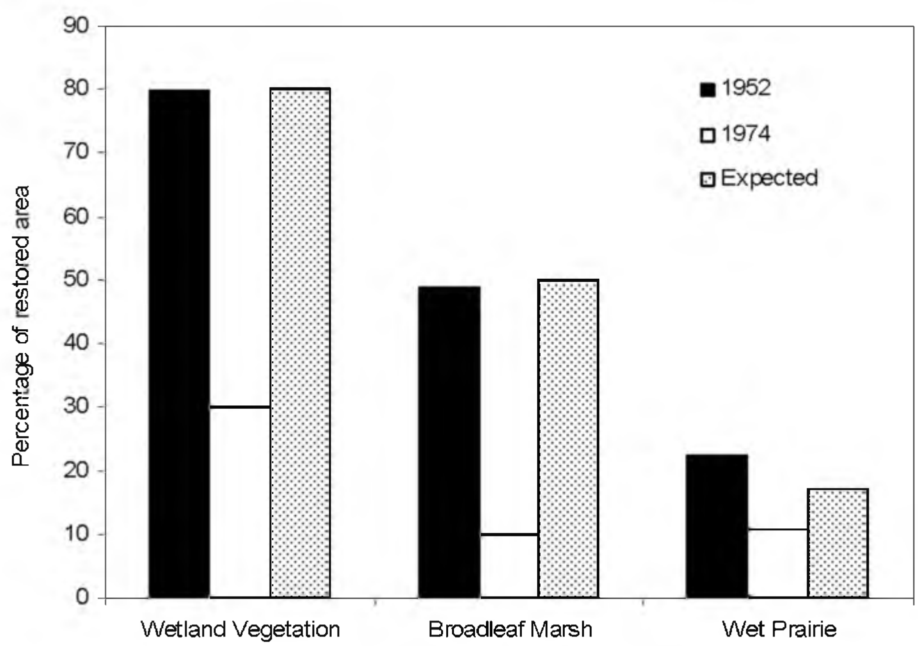

Figure 10-4. Reference, baseline, and predicted area of wetland, Broadleaf Marsh, and Wet Prairie in the restoration project area.

Table 10-4. Areal extent of Bcode Groups by restoration construction phase. The total areas shown are the total area affected by the restoration project. The 1952 pre-channelization reference estimates were used for predictions of restored areal extent of floodplain vegetation.

			Area (hectares)		Percent of restoration area	
Restoration phase	**Status**	**Bcode Group**	**1952**	**1974**	**1954**	**1974**
Phase I	Aquatic	Aquatic Vegetation	61.3	35.9	0.6	0.3
	Non-vegetated	Non-Vegetated: Bare Ground		379.3	0.0	3.6
		Non-Vegetated: Human		5.7	0.0	0.1
		Non-Vegetated: Open Water	209.7	176.0	2.0	1.7
	Unknown	Unknown	20.0		0.2	0.0
	Upland	Upland Forest	148.2	269.7	1.4	2.6
		Upland Herbaceous	198.0	1840.6	1.9	17.6
		Upland Shrub	55.6	303.9	0.5	2.9
	Wetland	Broadleaf Marsh	1672.3	174.9	16.0	1.7
		Miscellaneous Wetland	8.6	26.7	0.1	0.3
		Wet Prairie	1185.3	524.7	11.3	5.0
		Wetland Forest	11.6	5.7	0.1	0.1
		Wetland Shrub	276.1	103.5	2.6	1.0
Phase I Total			3846.6	3846.6		
Phase II/III	Aquatic	Aquatic Vegetation	115.5	68.4	1.1	0.7
	Non-vegetated	Non-Vegetated: Bare Ground	0.3	572.9	0.0	5.5
		Non-Vegetated: Human	20.3	34.0	0.2	0.3
		Non-Vegetated: Open Water	440.4	353.9	4.2	3.4
	Unknown	Unknown	16.6	1.3	0.2	0.0
	Upland	Upland Forest	227.6	337.2	2.2	3.2
		Upland Herbaceous	59.2	1384.3	0.6	13.2
		Upland Shrub	102.5	297.5	1.0	2.8
	Wetland	Broadleaf Marsh	2504.2	565.4	23.9	5.4
		Miscellaneous Wetland	32.9	32.4	0.3	0.3
		Wet Prairie	514.5	181.6	4.9	1.7
		Wetland Forest	55.5	35.7	0.5	0.3
		Wetland Shrub	297.5	522.3	2.8	5.0
Phase II/III Total			4386.9	4386.8		
Phase IV	Aquatic	Aquatic Vegetation	25.5	49.9	0.2	0.5
	Non-vegetated	Non-Vegetated: Bare Ground		141.9	0.0	1.4
		Non-Vegetated: Open Water	120.1	77.6	1.1	0.7
		Unknown	6.2		0.1	0.0
	Upland	Upland Forest	66.3	153.1	0.6	1.5
		Upland Herbaceous	64.9	307.0	0.6	2.9
		Upland Shrub	54.3	201.0	0.5	1.9
	Wetland	Broadleaf Marsh	673.5	123.9	6.4	1.2
		Miscellaneous Wetland	16.5	66.0	0.2	0.6
		Wet Prairie	471.3	401.9	4.5	3.8
		Wetland Forest	2.1	7.8	0.0	0.1
		Wetland Shrub	190.3	161.3	1.8	1.5
Phase IV Total			1691.0	1691.4		
Phase IVA	Aquatic	Aquatic Vegetation	7.6	11.8	0.1	0.1
	Non-vegetated	Non-Vegetated: Bare Ground		73.4	0.0	0.7
		Non-Vegetated: Open Water	64.5	48.9	0.6	0.5
		Unknown	2.3		0.0	0.0
	Upland	Upland Forest	25.7	39.2	0.2	0.4
		Upland Herbaceous	3.7	134.9	0.0	1.3
		Upland Shrub	4.0	11.1	0.0	0.1
	Wetland	Broadleaf Marsh	255.6	190.0	2.4	1.8
		Wet Prairie	179.2	16.9	1.7	0.2
		Wetland Forest	2.9	1.9	0.0	0.0
		Wetland Shrub	1.8	19.2	0.0	0.2
Phase IVA Total			547.2	547.2		
Totals of phases			10472	10472	100	100

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CHAPTER 11

AQUATIC INVERTEBRATE COMMUNITY STRUCTURE AND FUNCTIONAL CHARACTERISTICS IN THE KISSIMMEE RIVER-FLOODPLAIN ECOSYSTEM: BASELINE AND REFERENCE CONDITIONS AND EXPECTATIONS FOR RESTORATION

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ABSTRACT: Channelization of the Kissimmee River likely altered aquatic invertebrate community structure and functional characteristics of river channel and floodplain habitats. Remnant river channels are characterized by no flow, low levels of dissolved oxygen, abundant emergent, submergent, and floating vegetation, and thick accumulations of organic matter overlaying pre-channelization sand substrates. The channelized floodplain is characterized primarily by upland pasture, although small areas of remnant, but altered Broadleaf Marsh occur near the southern end of each pool. In order to determine baseline (pre-restoration) conditions, multiple sampling methods were used to determine aquatic invertebrate community structure, functional characteristics, and production in seven river channel and three floodplain habitats. Results indicate that aquatic invertebrate community structure and functional characteristics of the channelized Kissimmee River ecosystem are atypical of unmodified southeastern Coastal Plain blackwater river systems. Aquatic invertebrates of river channel habitats are representative of lentic and depositional habitats rather than flowing water habitats. No flow and isolation of the river channel from the floodplain preclude passive drift and bi-directional exchange of aquatic invertebrates between river channel and floodplain habitats. Floodplain habitats remain dry most of the time, but occasionally support an ephemeral and depauperate aquatic invertebrate community during the wet season. Habitat-specific macroinvertebrate secondary production within the channelized river was highly variable, but generally within the range of values reported for similar habitats in other blackwater river systems. Floodplain macroinvertebrate production was very low, primarily due to sporadic, short-term inundation patterns.

Restoration of the Kissimmee River is expected to alter aquatic invertebrate community structure and secondary production, and reestablish invertebrate drift and food web linkages within and between riverine and floodplain habitats. Shifts in species composition and secondary production, functional feeding and habitat groups, and invertebrate drift will be compared to baseline data and expectations for restoration. Although no historic or baseline data on bi-directional river channel/floodplain exchange exist for the Kissimmee River, post-construction evaluation of this functional attribute will be documented because of its critical role in food web and energy flow dynamics.

INTRODUCTION

Aquatic invertebrates were identified as a critical biological component for assessing restoration of ecological integrity within the Kissimmee River ecosystem (Karr et al. 1991; Harris et al. 1995). Aquatic invertebrates can play an integral role in river ecosystem processes including nutrient cycling (Merritt et al. 1984), decomposition of detritus (Wallace and Webster 1996), and energy flow to higher trophic levels; e.g., amphibians, reptiles, fishes, wading birds, and waterfowl (Weller 1995, Benke et al 2001). Aquatic invertebrates also have a long history of use in biomonitoring (Plafkin et al. 1989, Rosenberg and Resh 1993), and can serve as indicators of biotic integrity and ecological health (Karr 1991).

The pre-channelized Kissimmee River was characterized by a diverse littoral zone composed of submerged, emergent, and floating plants, shifting sand substrate, and minimal amounts of large woody debris (Toth et al. 1995). The river was highly stained with dissolved organic carbon primarily derived from the flanking floodplain and contributing watersheds. Dissolved oxygen levels varied seasonally, but likely ranged from 3–7 mg/L (Colangelo 2005). Discharge exceeded 11 m^3 per second 90–95% of the period of record, with highest discharge generally occurring near the end of the wet season (September–November). Average in-stream velocities ranged from 0.3–0.6 m/second. Pre-channelization stage data indicate that the Kissimmee River experienced a seasonal wet-dry cycle; however, only peripheral areas of the floodplain underwent consistent annual seasonal drying. Most floodplain habitats remained inundated for long periods (e.g., approximately 77% of the floodplain was inundated for 76% of the historical period of record (Toth et al. 1995) with water depths ranging from 0.3–0.7 meters (Koebel 1995). These river channel-floodplain characteristics likely shaped aquatic invertebrate community characteristics and rates of secondary production.

Elimination of flow through remnant channels and conversion of wetlands to pasture likely altered aquatic invertebrate community structure, and disrupted critical food web linkages within and between riverine and floodplain habitats. Under these hydrologic conditions, aquatic invertebrate taxa inhabiting the remnant (non-flowing) river channels are more characteristic of lentic or palustrine systems rather than a flowing river (Vannote 1971, Toth 1993, Warren and Holt 1996). Colonization and production of aquatic invertebrates in remnant Broadleaf Marsh is limited to short periods when summer rains temporarily inundate floodplain habitats, and because exchange of organic matter between the floodplain and the river channel is rare, passive drift by aquatic invertebrates is likely nonexistent.

Restoration of pre-channelization hydrology, including continuous, variable flow and long-term floodplain inundation frequencies, is expected to reestablish historic river channel and floodplain habitats, and aquatic invertebrate community structure characteristics. Specific changes likely will include shifts in functional feeding and functional habitat associations among primary river channel habitats (i.e., large woody debris and sandy benthos), increased macroinvertebrate species richness and diversity among floodplain habitats, and increased passive drift by macroinvertebrates.

Objectives

The objectives of this study are: (1) to assess baseline (pre-restoration) aquatic invertebrate community structure characteristics of the channelized Kissimmee River and floodplain; (2) to estimate rates of aquatic invertebrate secondary production for river channel and floodplain habitats; (3) to document aquatic invertebrate drift within the river channel; (4) to estimate reference conditions for aquatic invertebrate community structure in primary river channel habitats; (5) to estimate reference conditions for floodplain aquatic invertebrate communities; (6) to estimate reference conditions for aquatic invertebrate drift within the river channel; (7) to quantify impacts of channelization by comparing pre-channelization (reference) conditions and baseline conditions; and (8) to define and discuss specific expectations for restoration of aquatic invertebrate community structure and production in primary river channel habitats, aquatic invertebrate community structure in floodplain habitats, and aquatic invertebrate drift within the river channel.

METHODS

Baseline Conditions

Study Site

Aquatic invertebrate community structure characteristics and functional attributes were examined in seven remnant river channel and three floodplain habitats in Pools A, C, and D of the channelized Kissimmee River. Under channelized (baseline) conditions, remnant river channels are characterized by no flow, consistently low levels of dissolved oxygen (generally <2 mg/L) (Colangelo 2005), excessive growth of in-channel vegetation, and large accumulations of organic matter over benthic substrates. Sampled river channel habitats included *Nuphar lutea* (H.NL, *Nuphar lutea* herbaceous aquatic vegetation, Bousquin and Carnal 2005), *Polygonum densiflorum* (H.PD, *Polygonum densiflorum* herbaceous aquatic vegetation, Bousquin and Carnal 2005), *Scirpus cubensis* (H.SCF, *Scirpus cubensis* herbaceous floating mat vegetation, Bousquin and Carnal 2005), *Ceratophyllum/Hydrilla* (H.MxSV, miscellaneous submerged vegetation, Bousquin and Carnal 2005), Mid-channel Benthic (BENT), Mid-channel Water Column (ZOOP), and Woody Debris (SNAG). Snags were defined as any submerged dead wood greater than 1" in diameter. See Bousquin and Carnal (2005) for more detailed vegetation classification scheme.

Sampled floodplain habitats included Broadleaf Marsh (BLM) (Bousquin and Carnal 2005), Woody Shrub (S.MCF, *Myrica cerifera* Floating Mat Shrubland, Bousquin and Carnal 2005), and Woody Debris (FSNAG). Remnant Broadleaf Marsh habitats are spatially homogeneous and dominated by arrowhead (*Sagittaria lancifolia*), pickerelweed (*Pontederia cordata*), and maidencane (*Panicum hemitomon).* Woody Shrub habitats are characterized by dense stands of wax myrtle (*Myrica cerifera*) that exist on a bog-like floating mat. The understory is composed of a diverse mixture of broadleaf marsh, wet prairie, and upland vegetation including broomsedge (*Andropogon glomeratus*), sedges (*Cyperus* spp.), pennywort (*Hydrocotyle umbellata*), spatterdock (*N. lutea*), rushes (*Rynchospora* spp.), and Cuban bulrush (*S. cubensis*). See Bousquin and Carnal (2005) for more detailed vegetation classification scheme.

Aquatic Invertebrate Community Structure

Aquatic invertebrates were sampled quarterly over a two-year period from August 1995–May 1997. Three replicate samples were collected from each river channel and floodplain habitat within Impact and Control sites (when available) on each sample date. Control sites included three remnant river channels (Ice Cream Slough Run, Rattlesnake Hammock Run, and Persimmon Mound Run) and remnant BLM (Latt Maxcy Floodplain) in Pool A. These sites will not be affected by restoration and will serve as long-term Control sites. An additional short-term Control site was established in Pool D Woody Shrub (S.MCF). This site will be impacted by restoration construction during Phase II/III (2008–2010). Impact sites included three remnant river channels (Oxbow 13, Micco Bluff Run, and MacArthur Run), remnant BLM (Pool C Broadleaf Marsh), and Pool C Woody Shrub (S.MCF). These sites will be affected following Phase I construction. Sampling locations within remnant channels were selected by traveling at a constant speed (~ 1000 rpms) for a randomly determined time through the channel, and continuing until the next appropriate habitat type was encountered. Floodplain sample locations were selected by traveling a randomly determined distance ($\leq$400 m) and compass direction from a randomly determined location on the floodplain. All samples were preserved in the field with 5–10% formalin stained with rose bengal. Each sample was located in space and time with a Global Positioning System (GPS) with sub-meter accuracy. For each sample, ancillary data including water temperature, specific conductance, pH, and dissolved oxygen were recorded at a depth of 15 cm below the water surface using a Hydrolab™ or YSI™ multi-probe water quality instrument. In shallow floodplain habitats, water quality parameters were generally recorded within the first 5 cm of the water column. Water depth was recorded at each location with a meter stick or PVC pole calibrated in 5 cm intervals. Current velocity was measured in the river channel with a Marsh-McBirney series 2000 flow meter. A continuous record of river channel surface water temperature at Impact and Control sites was recorded using a HOBO™ temperature logger. Missing values in this record were estimated from a regression developed from this data set, and air temperature records from Archbold Biological Station, Lake Wales, Florida (D. H. Anderson, SFWMD, personal communication).

Preserved samples were sieved into two size classes using 1 mm (coarse fraction) and 125 µm (fine fraction) mesh sieves. All invertebrates were hand-picked from the coarse fraction using a dissecting microscope at 6–12X magnification, and preserved in 70% ethanol. The fine fraction was elutriated to separate organic matter from inorganic matter. The organic matter portion was sub-sampled to a fraction that could be processed in approximately two hours (usually 1/8–1/64). All invertebrates were counted and identified to the lowest taxonomic level using Thorp and Covich (1991), Merritt and Cummins (1996), Epler (1992, 1996), and Thompson (1984). For most taxa, individual biomass was estimated from published length-mass regressions (Benke et al. 1999, Meyer 1989, Culver et al. 1985, Rosen 1981, Anderson and Benke 1994, Anderson et al. 1998a, Lei and Armitage 1980, Fleeger and Palmer 1982). For mites, we used a dry mass of 0.06 mg/individual (D.H. Anderson, unpublished data). For nematodes and leeches, individual mass was estimated volumetrically by assuming a cylindrical shape, a specific density of 1.05, and a dry mass content of 15% (Strayer and Likens 1986). Oligochaetes were dried for four hours at 60°C and weighed to the nearest 0.001 gram.

Abundance and biomass estimates for each taxon in each sample were weighted by sampler area to standardize estimates to numbers/m^2 or grams/m^2, respectively. Mean quarterly density and biomass for each taxon was determined by averaging its sample density and biomass for each replicate on each date. Mean annual density and biomass were determined by averaging the four quarterly estimates of density and biomass. For dates when habitats were not available (e.g., dry floodplain), zeros were averaged to obtain estimates of mean annual density and biomass. Zeros were not included in the calculations when poorly preserved samples were discarded.

Organic matter in the coarse fraction was classified as macrophyte, wood, or detritus, and dried at 60°C for 24 hours. Dried material was weighed to the nearest 0.001 gram, ashed at 450°C for 4 hours, and re-weighed to determine ash-free-dry-mass (AFDM). Ash-free-dry-mass also was determined for organic and inorganic matter from the fine fraction.

Community structure was described by species richness (S = the total number of species present), species diversity (H'), where $H' = -\Sigma(p_i \ln p_i)$ and p_i is the proportion of species belonging to the i^{th} taxa, and community evenness (J'), where $J' = H'/\ln S$ (Price 1984). Taxa were assigned to functional feeding groups according to Merritt and Cummins (1996), Rader (1994), Borror et al. (1989), Merritt et al. (1996) for aquatic insects, and Rader (1994), Gladdon and Smock (1990), and Balcer et al. (1984) for non-insects. Functional feeding group categories included filtering-collectors (FCOLL), gathering-collectors (GCOLL), predators (PRED), scrapers (SC), shredders (including shredders of coarse particulate organic matter and vascular plants) (SHRD), and vascular plant piercers (PRC). Taxa also were classified into four functional habitat groups - LENTIC (only occurring in standing water), LOTIC (only occurring in flowing water), BOTH (occurring in lentic or lotic habitats), and DEP (occurring in lentic or lotic depositional zones). Functional habitat groups were based on the classification in Merritt and Cummins (1996) and supplemented with information from Epler (1996), Tressler (1959), and Thompson (1984).

Analysis of variance (ANOVA; SYSTAT version 8) was used to test for differences in total sample organic matter and mean annual density. These analyses used a randomized block design with site (Impact and Control) as the treatment and habitat blocks. The natural logarithm of total organic matter and total density was used to make the variance independent of the mean. Pairwise comparisons were made using Tukey's HSD test that controls the experiment-wise error rate (Day and Quinn 1989). When sample sizes are uneven, SYSTAT uses the Tukey-Kramer modification that maintains the experiment-wise error rate at or below the nominal level, and is more powerful than most pairwise comparison methods (Day and Quinn 1989). Unless otherwise stated, all statistics are significant at $p < 0.05$.

Secondary Production

Secondary production was estimated using the instantaneous growth rate method, which requires knowledge of individual biomass and growth (Benke 1993). For most taxa, the appropriate length dimension was measured with an ocular micrometer, and individual biomass was estimated from length-mass regressions. Growth rates were estimated from published growth equations (Morin and Dumont 1994, Pickard and Benke 1996, Benke and Jacobi 1994, Hauer and Benke 1991, Anderson and Benke 1994, Anderson et al. 1998a). A growth equation for grass shrimp (*Palaemonetes paludosus*) in the Kissimmee River was developed for this study. A growth equation for crayfish, developed for a congeneric species (*Procambarus alleni*) from wetlands in the Lake Okeechobee basin, also was used. These equations predict daily growth rate from temperature and individual mass.

To estimate annual production, each year of the baseline period was divided into four equal intervals centered on the quarterly sampling date. For each taxon in each sample, secondary production was estimated as the product of biomass, daily growth rate, and number of days in the interval. Production and biomass estimates for each sample were averaged to obtain a mean for each interval. Interval production estimates were summed to obtain annual production, and biomass for each quarter was averaged to obtain mean annual biomass. Annual P/B was obtained by dividing annual production by mean annual biomass.

Aquatic Invertebrate Drift

Aquatic invertebrate drift samples were collected approximately quarterly beginning in January 1998. Paired drift nets (900 cm^2 equipped with 125 µm mesh netting) were placed 15 cm below the water surface and 0.5 m above the substrate at three locations within each of three remnant river channels in Pools A and C. Because there is no flow through remnant channels, there was little risk of nets becoming clogged; therefore, samples were collected at eight-hour intervals ($\pm$ 1 hour) over a 24-hour period. Current velocity at each surface and bottom net opening, wind direction, and wind velocity were measured whenever a net was set or removed. All samples were preserved in the field with 10% buffered formalin stained with rose bengal.

Preserved samples were rinsed through a 125 µm mesh sieve and sub-sampled to a fraction that could be processed in approximately two hours (usually 1/32–1/64). All invertebrates were hand-picked using a dissecting microscope at 12X magnification, and preserved in 70% ethanol.

RESULTS

Habitat Characteristics

Mean annual water temperature in remnant channels was 23°C in year one and 25°C in year two and differed by less than 0.5°C between Control and Impact sites (Figure 11-1). Approximately 90% of all current velocity measurements in all habitats were 0.0 m/s, with only two values >0.2 m/s. Mean annual values for dissolved oxygen, pH, and specific conductance (Figure 11-2) were similar across habitats and sites. Surface dissolved oxygen concentrations were typically low, with a mean baseline value averaged across all habitats and sites, of 2.9 mg/l.

Mean organic matter content of samples (Figure 11-3) was significantly different among habitats (ANOVA, $p < 0.01$) but not between Control and Impact sites (ANOVA, $p > 0.05$). Organic matter composition also varied among habitats, but showed similar patterns at Control and Impact sites (Figure 11-4).

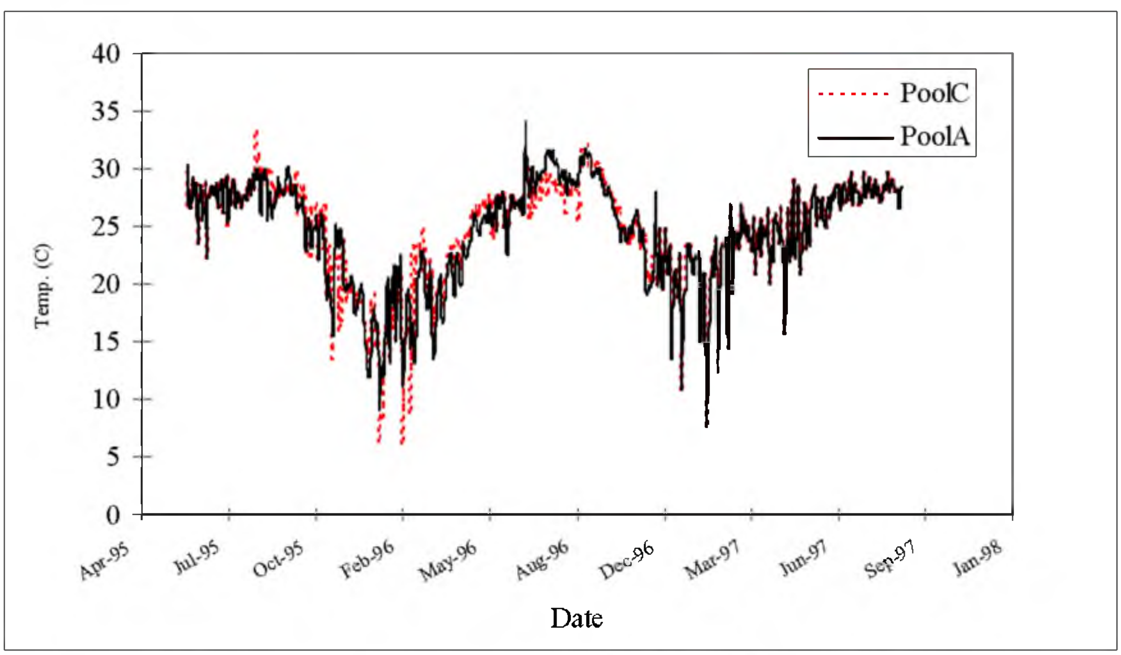


Figure 11-1. Daily water temperature at Impact and Control sites during the baseline study period.

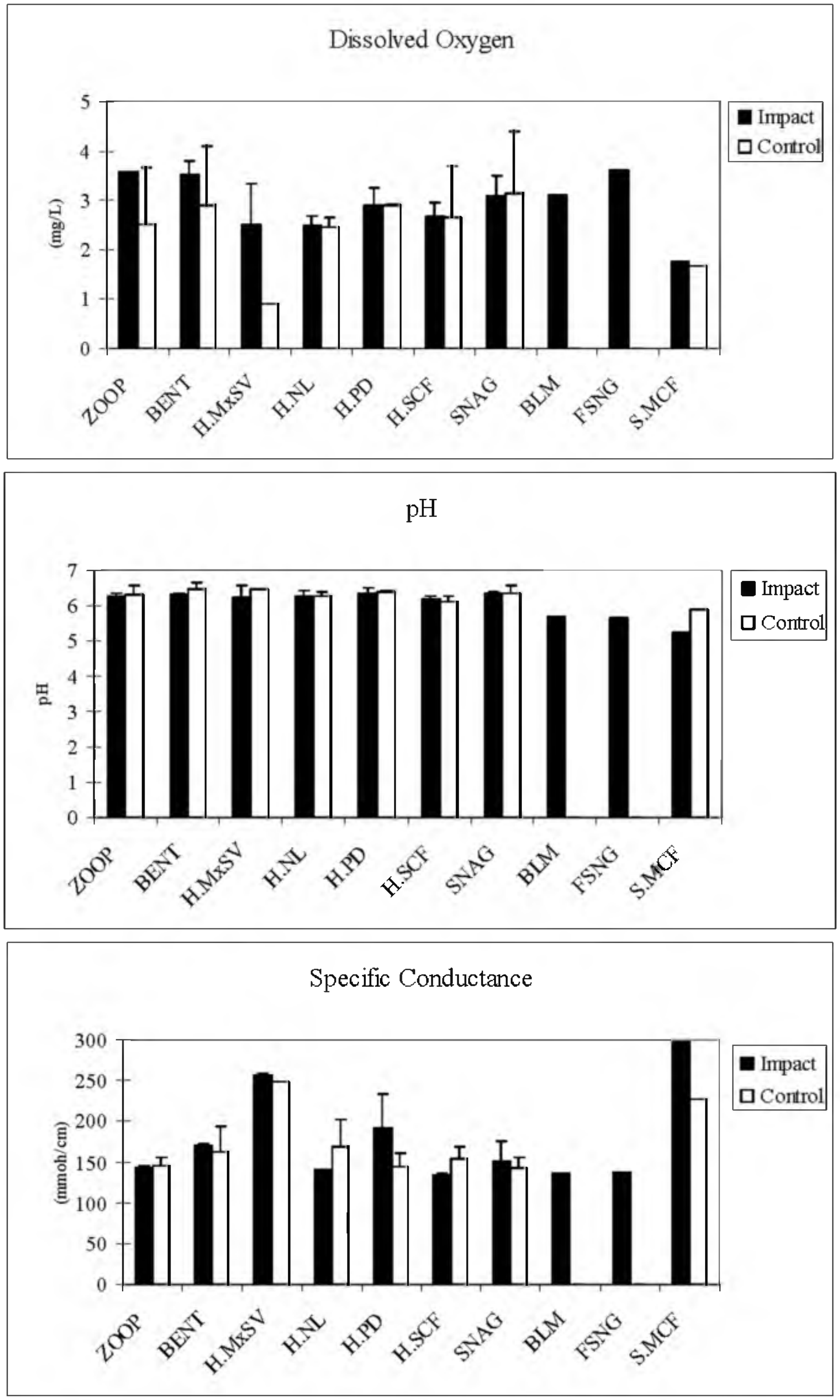


Figure 11-2. Mean annual values for dissolved oxygen (top), pH (middle), and specific conductance (bottom) in Pools A (Control) and C (Impact). ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

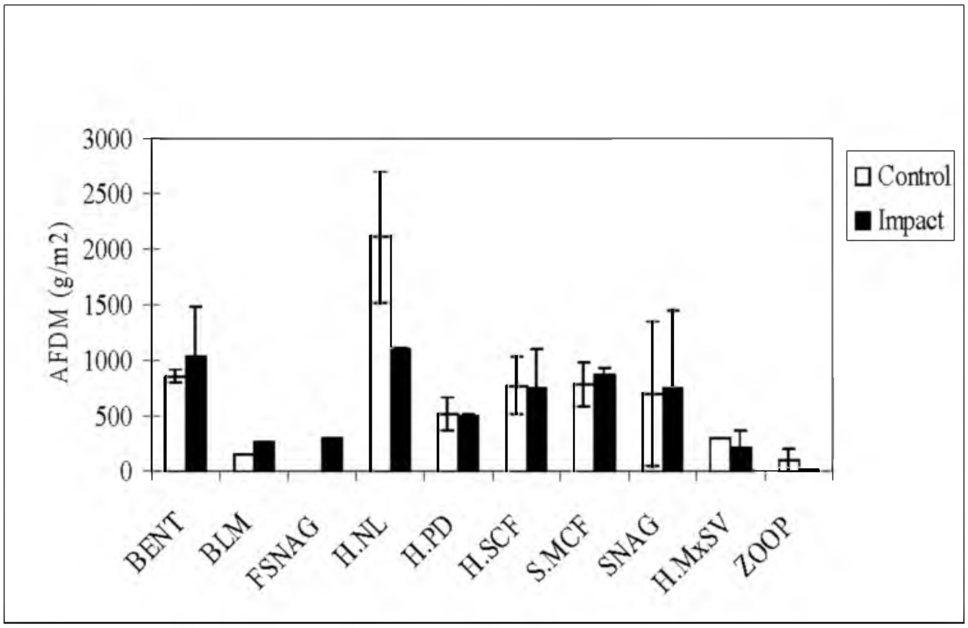


Figure 11-3. Total ash-free-dry-mass (AFDM) of organic matter from replicate samples averaged across dates and years for all habitats. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Aquatic Invertebrate Community Structure

One hundred and eighty-seven taxa of aquatic invertebrates were collected from remnant river channel and floodplain habitats. Coleoptera (48 genera), chironomids (26 genera), and microcrustaceans (42 genera) accounted for 62% of all taxa. Two additional taxa, *Corbicula fluminea*, and the native unionid mussel, *Elliptio buckleyi*, were not sampled quantitatively; however, qualitative collections of both species occurred at several locations along the river.

Taxa richness, taxa diversity, and community evenness varied among habitats and sites, with higher values generally occurring at the Impact site; however, differences among habitats tended to be greater than those between sites (Figure 11-5). Highest richness and diversity occurred in *Nuphar* (H.NL), *Polygonum* (H.PD), *Scirpus* (H.SCF), snag (SNAG), and Woody Shrub (S.MCF) habitats at Impact and Control sites; however, diversity values were usually <2.0. Community evenness exceeded 0.5 for all habitats except Broadleaf Marsh (BLM), Floodplain Snag (FSNAG), and Submerged Vegetation (H.MxSV). Seasonal patterns were not apparent for taxa richness, taxa diversity or community evenness.

Mean annual density (Figure 11-6) ranged from 6049/m^2 in BLM to 134,871/m^2 in H.SCF at the Control site, and 1732/m^2 in FSNAG to 232,997/m^2 in H.SCF at the Impact site. There were no significant differences in the natural logarithm of total density between Control and Impact sites (ANOVA, $p > 0.05$), but there were significant differences for habitat blocks (ANOVA, $p < 0.01$). Mean annual density in SCIR was significantly higher than all other habitats (Tukey's HSD, $p < 0.05$). Density showed no seasonal pattern at either Control or Impact sites.

Core taxa were identified as those that accounted for at least 5% of mean annual abundance in any habitat at either site. Seventeen core taxa (ten at the Control site, 14 at the Impact site, and seven at both sites) were identified, and accounted for 26–86% of mean annual density in each habitat (Tables 11-1 and 11-2). Most core taxa occurred in most habitats, but their relative abundance varied among habitats.

Gathering-collectors accounted for the largest fraction of individuals sampled in most habitats (Figure 11-7). Microcrustacean filtering-collectors were most abundant in mid-channel open water samples, and were well represented in most habitats, often accounting for 20% of total numbers, and over 40% of total number in mid-channel samples (ZOOP). Macroinvertebrate passive filtering-collectors were absent from most habitats, never accounting for >2% of total numbers on any sampling date. Predators and scrapers accounted for most of the remaining individuals in most habitats.

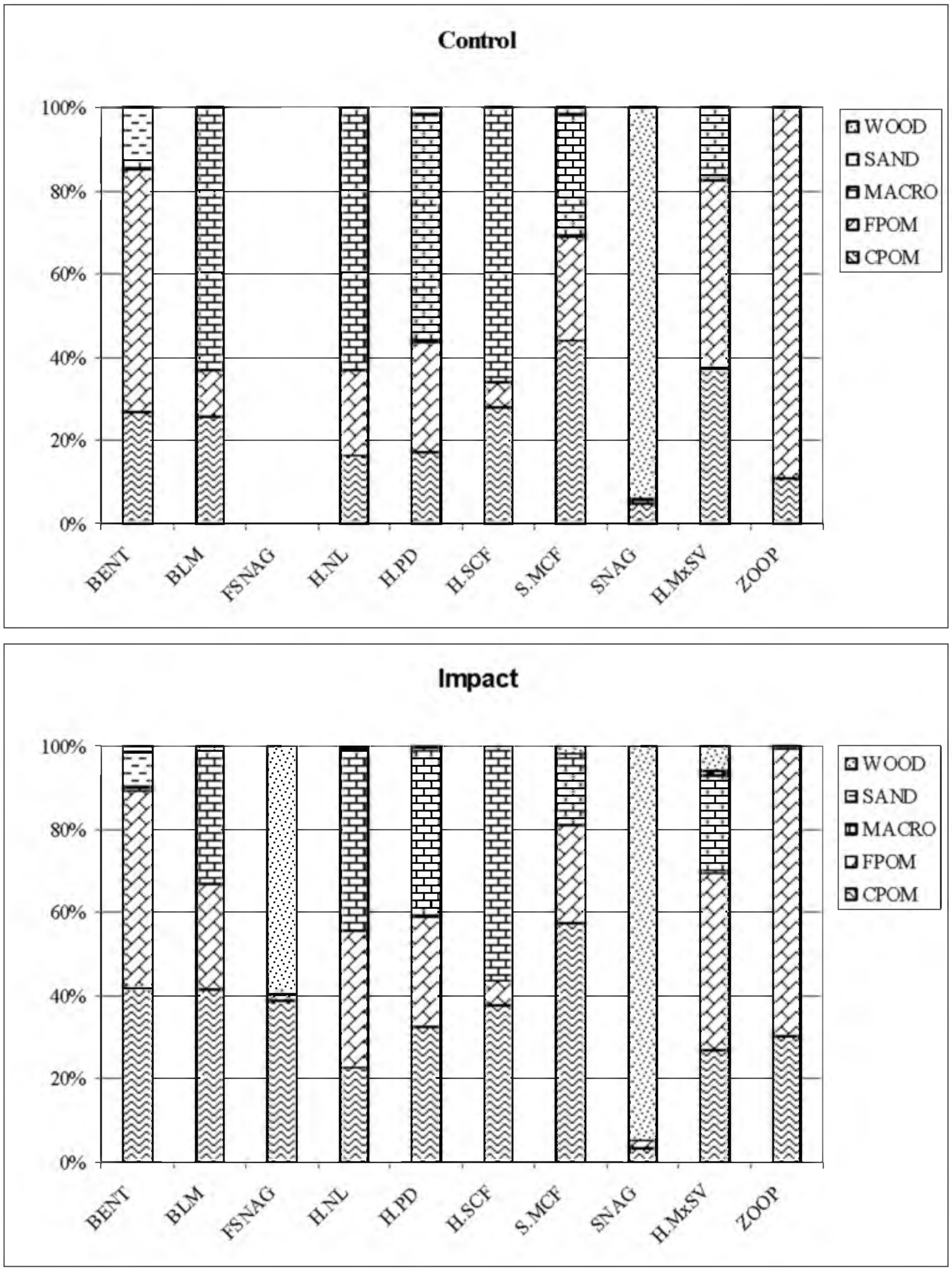


Figure 11-4. Organic matter composition among habitats at Impact and Control sites. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Functional habitat group composition varied among habitats but was similar among sites (Figure 11-8). Taxa typical of lotic habitats were rare and comprised <3% of mean annual density in each habitat. Taxa typical of lentic habitats accounted for the largest fraction of mean annual abundance, often exceeding 50% in most habitats at both Control and Impact sites. Taxa typical of lentic habitats or lotic depositional areas (BOTH) accounted for the next highest fraction.

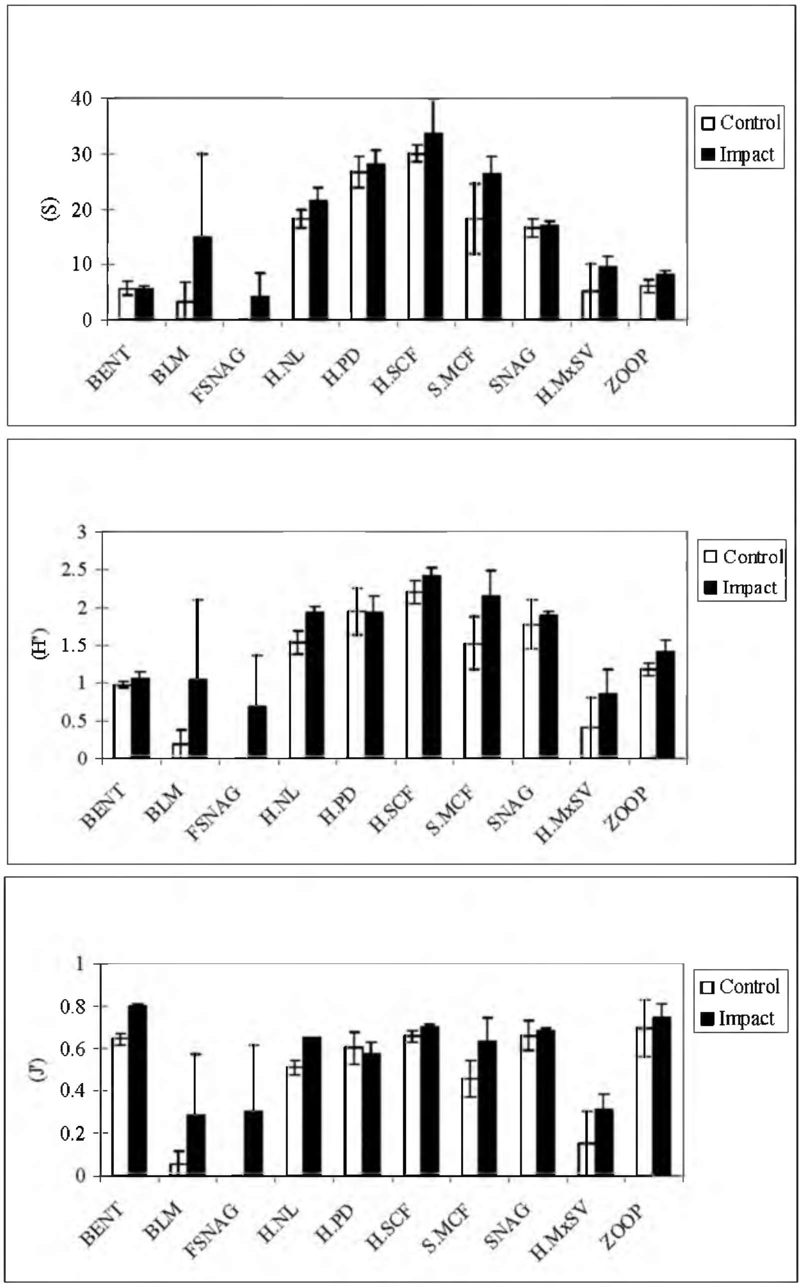


Figure 11-5. Mean taxa richness (top), diversity (middle), and community evenness (bottom) at Control and Impact sites. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

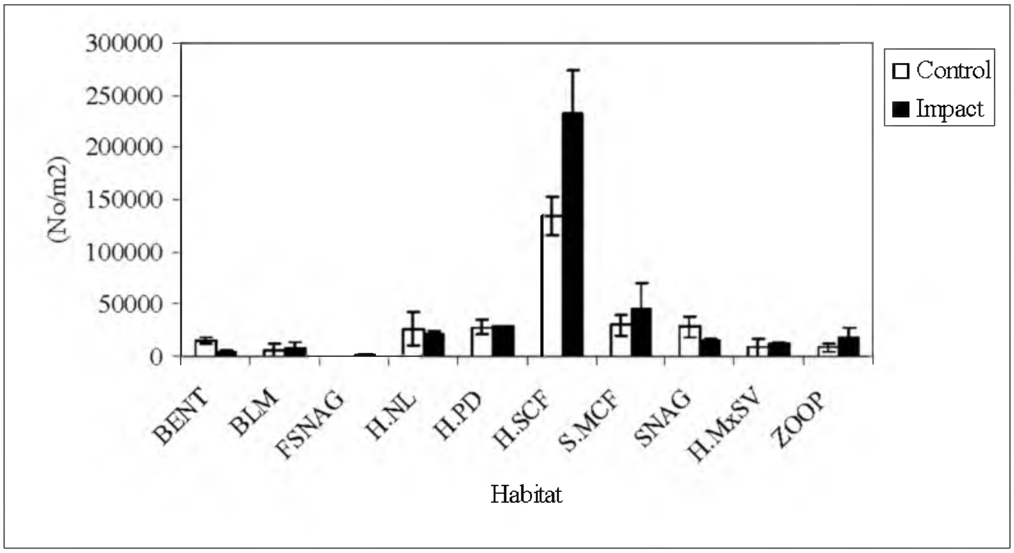


Figure 11-6. Mean annual invertebrate density for each habitat at Control and Impact sites. Bars represent mean ± SE of mean annual density for two baseline years. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Table 11-1. Mean (SE) baseline density (no/m^2) for core taxa (bold type) at the Control site. Habitats are arranged from mid-channel to the edge of the floodplain. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

Taxon	ZOOP	BENT	H.MxSV	H.NL	H.PD	H.SCF	SNAG	BLM	S.MCF
Acari	133	96	284	160	494	2938	654	**3840**	**6366**
	(133)	(93)	(284)	(160)	(185)	(2783)	(154)	(3840)	(546)
Caecidotea	0	0	0	1	88	1409	260	0	**4560**
				(1)	(34)	(1229)	(12)		(2578)
Chironomus/Goeldchironomus	42	245	170	**863**	529	2236	425	0	1801
	(42)	(57)	(170)	(295)	(238)	(1277)	(62)		(466)
Cypria/Physocypria	**2145**	**10322**	**4191**	**6354**	**6122**	**14203**	373	0	38
	(1444)	(2249)	(4191)	(5227)	(1921)	(2791)	(242)		(38)
Dicrotendipes	11	0	362	252	**781**	**8407**	**1476**	0	43
	(11)		(362)	(189)	(362)	(6081)	(502)		(33)
Hyalella azteca	85	19	42	**960**	**2647**	**19836**	**5494**	19	904
	(64)	(19)	(42)	(364)	(841)	(1967)	(3652)	(19)	(347)
Macrocylops	**717**	264	**580**	757	**3844**	**13961**	84	0	**4845**
	(441)	(113)	(580)	(501)	(873)	(4228)	(14)		(1005)
Osphranticum	**122**	76	1	19	429	415	5	**308**	**2027**
	(80)	(76)	(1)	(19)	(71)	(415)	(3)	(308)	(124)
Paracyclops	106	28	0	179	**666**	3288	11	**307**	**5032**
	(85)	(9)		(179)	(666)	(1870)	(2)	(307)	(2648)
Polypedilum	32	57	40	**1162**	302	3693	339	247	762
	(11)	(57)	(40)	(695)	(46)	(1057)	(196)	(247)	(169)
Others	5181	3663	2954	16037	12139	64484	19300	1327	762
	(2145)	(225)	(2954)	(10242)	(4203)	(15268)	(5038)	(1327)	(169)
Total	8573	14770	8623	26744	28040	134871	28424	6049	29944
	(4432)	(2878)	(8623)	(16555)	(6748)	(18582)	(9417)	(6049)	(10149)

Secondary Production

Annual production and mean annual biomass varied with habitat but showed similar patterns at the Control and Impact sites (Figure 11-9). Differences in production across habitats tended to parallel differences in biomass. Production and biomass were much higher in H.SCF than any other habitat. Estimates of baseline annual P/B tended to be more uniform, and generally exceeded 20 for most habitats (Figure 11-9).

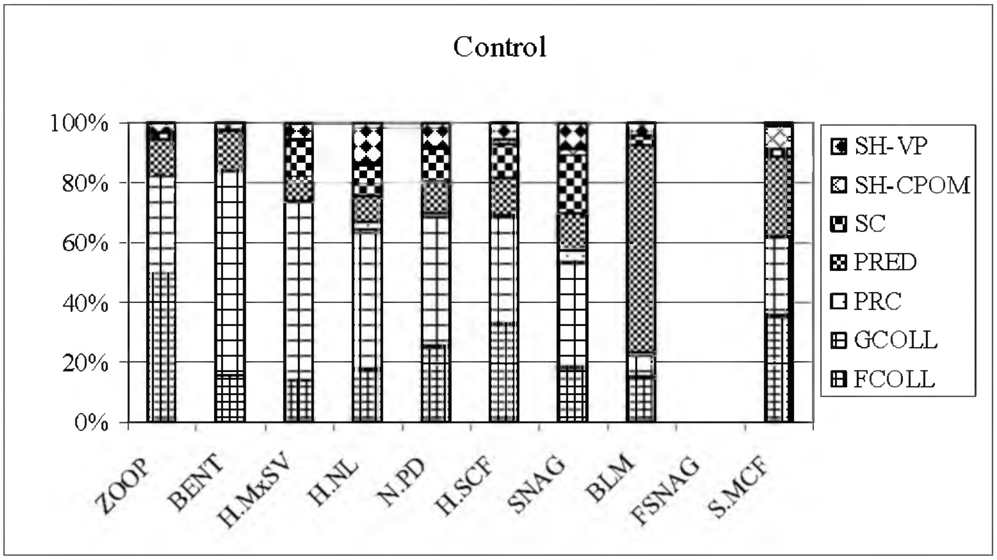
Table 11-2. Mean (SE) baseline density (No/m^2) for core taxa (bold type) at the Impact site. Habitats are arranged from mid-channel to the edge of the floodplain. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

Taxon	ZOOP	BENT	H.MxSV	H.NL	H.PD	H.SCF	SNAG	BLM	FSNG	S.MCF
Acari	133	17	89	289	702	3128	**678**	231	**429**	**6797**
	(48)	(2)	(32)	(236)	(0)	(1552)	(461)	(231)	(429)	(5824)
Bosmina	**1566**	106	10	0	0	0	0	0	0	0
	(802)	(83)	(10)							
Cypria/Physocypria	**5112**	**437**	**5139**	**8314**	**7175**	**45734**	**511**	83	42	386
	(4985)	(3)	(4351)	(4332)	(2772)	(30673)	(470)	(83)	(42)	(181)
Daphnia	**1232**	**111**	0	6	0	0	0	0	0	0
	(403)	(2)		6						
Diaptomidae	**2320**	**217**	0	6	0	0	0	0	0	0
	(664)	(47)		(6)						
Dicrotendipes	16	7	**609**	621	**1721**	**16549**	2526	70	29	370
	(5)	(7)	(556)	(103)	(1159)	(7726)	(288)	(70)	(29)	(193)
Eucyclops	202	80	100	**2336**	224	6555	42	**570**	2	390
	(180)	(14)	(100)	(1862)	(19)	(4161)	(30)	(570)	(2)	(83)
Glyptotendipes	5	85	73	94	210	1658	**1916**	4	42	0
	(5)	(85)	(29)	(76)	(36)	(260)	(20)	(4)	(42)	
Hyalella azteca	5	73	**1081**	**1227**	**1854**	**9767**	**3028**	44	4	737
	(5)	(21)	(1072)	(794)	(145)	(959)	(575)	(44)	(4)	(198)
Macrocyclops	**1529**	**231**	**545**	**1111**	**3502**	**25377**	85	**773**	33	646
	(446)	(109)	(91)	(220)	(1219)	(7740)	(48)	(773)	(33)	(275)
Osphranticum	**287**	47	**415**	118	**2172**	4221	0	**702**	0	451
	(244)	(47)	(415)	(118)	(811)	(1301)		(702)		(144)
Paracyclops	**1396**	52	20	154	307	7220	0	112	**89**	**2227**
	(483)	(33)	(20)	(102)	(205)	(1248)		(112)	(89)	(435)
Simocephalus	239	**1142**	132	**817**	302	3287	100	22	0	0
	(218)	(1142)	(132)	(364)	(215)	(2431)	(81)	(22)		
Tanytarsini	0	47	**769**	**682**	948	8524	**782**	122	40	272
		(47)	(575)	(574)	(132)	(3618)	(600)	(122)	(40)	(67)
Others	4369	1547	2618	4981	9303	100976	5886	4434	1021	34240
	(3010)	(521)	(4)	(1348)	(1994)	(59483)	(1330)	(4434)	(1021)	(18311)
Total	18409	4226	11598	20758	28421	232997	15552	7168	1732	24379
	(9746)	(1984)	(1636)	(3824)	(935)	(41338)	(553)	(7168)	(1732)	(24379)

Twenty core taxa accounted for at least 5% of the baseline annual production across all habitats at the Control site (Table 11-3); twenty-one core taxa were identified at the Impact site (Table 11-4). Twelve of these were core taxa at both sites, but they were not always core taxa in the same habitats. Approximately 75% of core taxa in both pools are characteristic of lentic or depositional habitats (Anderson et al. 1998).

Functional feeding group contributions to annual production varied with habitat, but tended to show similar trends at both the Control and Impact sites (Figure 11-10). Gathering-collectors generally accounted for the largest fraction of production. Filtering-collectors (active and passive) rarely accounted for >10% of annual production except in mid-channel open water habitats (ZOOP), where they accounted for 27% and 51% at the Control and Impact sites, respectively; however, this guild was dominated by active filtering-collector microcrustaceans.

Functional habitat groups show fairly consistent patterns across habitats at both the Control and Impact sites (Figure 11-11). Taxa typical of lentic habitats (LENTIC) account for about half of annual production in most habitats. Taxa typical of depositional zones (DEP) account for the next largest percentage of annual production. The only departures from this pattern are in floodplain habitats, where taxa that can occur in both lentic and lotic habitats (BOTH) account for a larger fraction. This is primarily due to the production of aquatic mites that are common in both lentic and lotic habitats. Taxa typical of lotic conditions (LOTIC) account for a very small fraction of annual production.



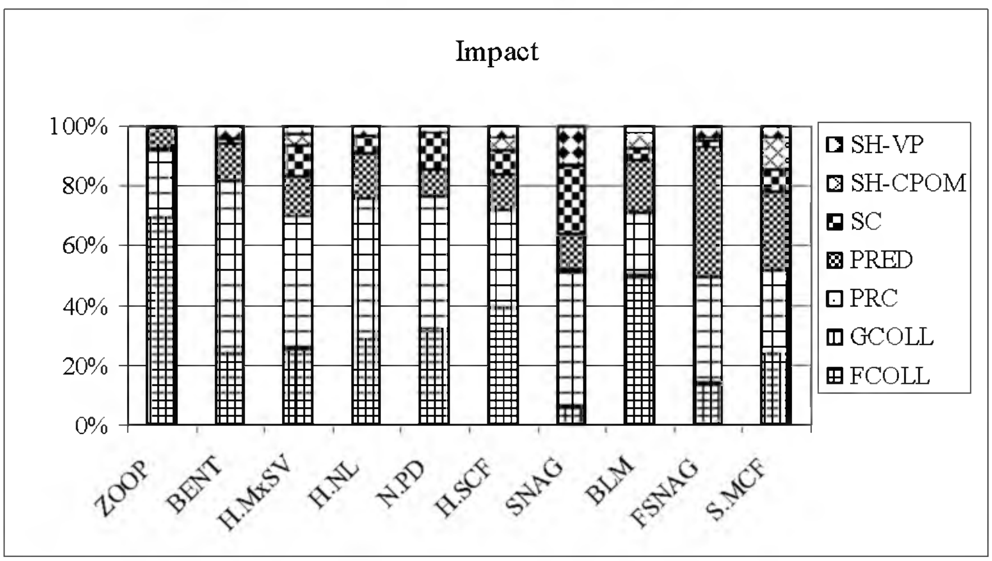


Figure 11-7. Mean functional feeding group composition, based on total abundance, for each habitat at Control and Impact sites. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

DISCUSSION

The sampling strategy used in this study was intended to broadly characterize habitat-specific aquatic invertebrate community structure in remnant river channel and floodplain habitats. Because no quantitative invertebrate data exist for the pre-channelized Kissimmee River, our baseline data is compared to data from unregulated southeastern Coastal Plain rivers, with the understanding that any inferences about impacts of channelization must consider other factors (e.g., introduction of exotics, biogeography) that can influence community structure characteristics. For instance, the channelized Kissimmee supports a guild of scraping invertebrates (e.g., snails and *Hyalella azteca*) that is rare in other Coastal Plain rivers. The presence of scrapers is not an obvious consequence of channelization, but may reflect other differences between these rivers, including a greater abundance of macrophytes and associated periphyton, which provide a surface and food source for grazers. Additionally, high water column calcium concentrations (10–20 mg/L) in the Kissimmee (SFWMD unpublished data) may be more favorable for snail growth than water chemistry in other Coastal Plain rivers (Stites et al. 1995).

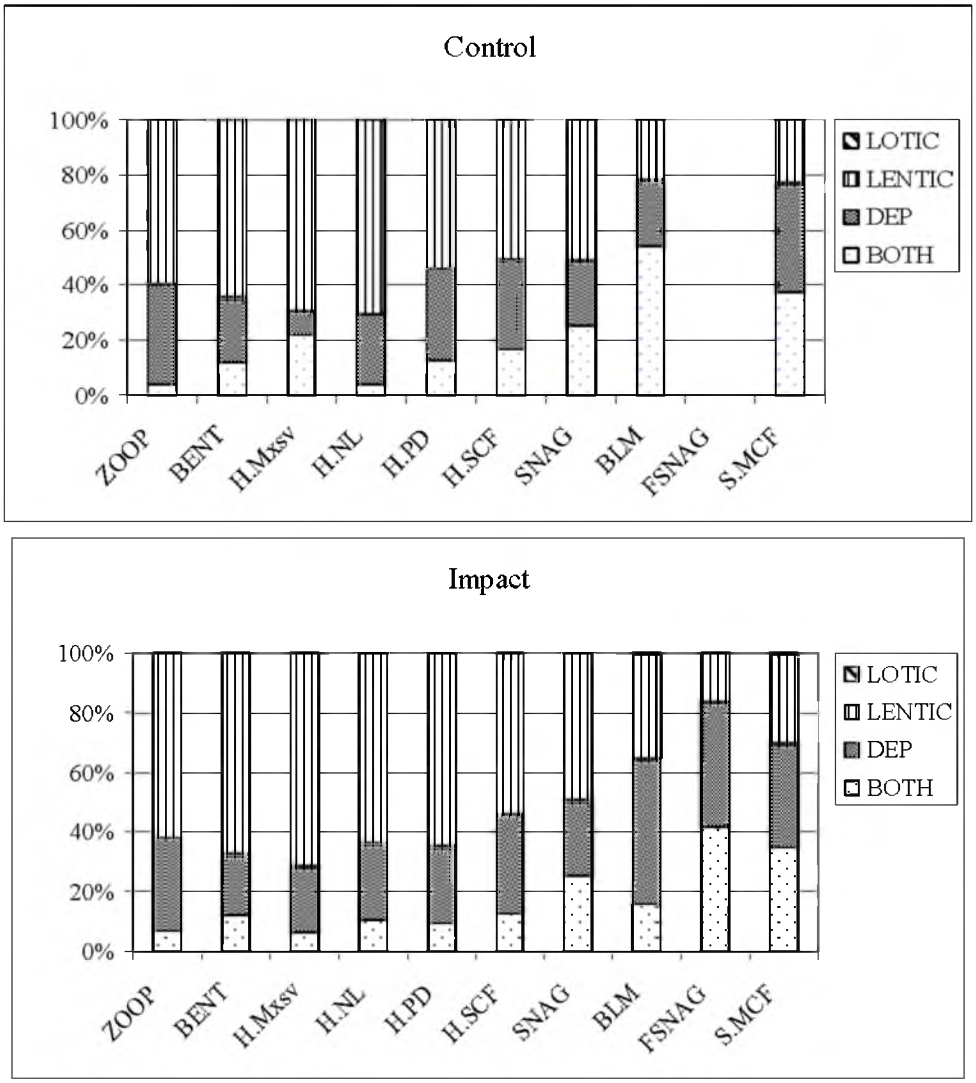


Figure 11-8. Mean annual functional habitat composition, based on total abundance, for each habitat at Control and Impact sites. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

Low sample replication (three) and frequency of collection (quarterly) was necessitated by manpower constraints. Although data collected in this manner may not be optimal for addressing temporal or seasonal patterns of abundance or biomass, we believe it was sufficient for documenting structural characteristics of the invertebrate community that are likely to change as a result of restoration (e.g., shifts in functional feeding and functional habitat groups).

Aquatic Invertebrate Community Structure

Invertebrate density in remnant channels of the Kissimmee River is generally within the range reported for three unimpacted Coastal Plain blackwater rivers (Benke et al. 1984, Smock et al. 1985, Benke and Meyer 1988).

The highest estimates of mean density in the channelized Kissimmee were found in floating mats of H.SCF, which had densities nearly four times greater (130,000–230,000/m^2) than those reported for any habitat in Coastal Plain river systems. Floating H.SCF mats consist of a dense web of highly branched roots located just below the water surface. The roots accumulate large amount of fine particulate organic matter, and provide a highly heterogeneous habitat that supports large numbers of microcrustaceans, *Hyalella azteca*, and several chironomids.

Invertebrate taxa diversity was low in all habitats and rarely exceeded 2.0. These values are in the range for moderately polluted streams, with values <1 typical of heavy pollution (Wilhm 1972). Species richness also is low in the Kissimmee River; however, Warren and Hohlt (1996) found that richness and diversity in Pools A and C bracketed values for Fisheating Creek, a reference (i.e., minimally impacted) stream in the eastern Florida flatwoods region. Although data for Fisheating Creek are limited to one sampling period, and not sufficient to generalize about richness and diversity in undisturbed rivers of central and south Florida, biogeographical factors (peninsular effect, isolation from tropical source pools) may account for low species richness in the Kissimmee River and other lotic system of south Florida.

Core taxa based on density were heavily skewed toward microcrustaceans (Table 11-1 and 11-2). Forty percent of core taxa at the Control site, and 64% of core taxa at the Impact site were microcrustaceans. Although microcrustaceans are likely to be seasonally abundant in some habitats (e.g., BLM), restoration of flow likely will reduce density of many taxa in river channel habitats.

Previous studies in remnant channels of the Kissimmee River have characterized the invertebrate community as typical of standing water (Vannote 1971, Toth 1993, Warren and Hohlt 1996). Our functional habitat classification was developed to quantify this pattern, and showed near complete absence of taxa characteristic of flowing water, and a large proportion of taxa characteristic of lentic habitats.

Snag habitats within remnant channels of the Kissimmee River are dominated by gathering-collectors (primarily midges characteristic of lentic or depositional habitats), shredders (primarily *Glyptotendipes* spp. [Chironomidae]), and scrapers (primarily the amphipod *Hyalella azteca* and several gastropods). The filtering-collector guild is dominated by active filtering-collectors (primarily microcrustaceans); passive filtering-collectors accounted for <3% of total numbers on snags within remnant channels. Benke et al. (1984) report that passive filtering-collectors, including caddisflies (primarily *Hydropsyche* spp.) and blackflies (*Simulium* spp.), were the major consumers on snags in the Satilla River, Georgia, and accounted for 75–80% of mean annual density, 65–75% of mean annual biomass, and 72–79% of mean annual production at two sample locations. Smock et al. (1985) report passive filtering-collectors (primarily *Macronema carolina* [Hydropsychidae] and *Tanytarsus* sp. [Chironomidae]) were the dominant taxa on snags in Cedar Creek, South Carolina, and accounted for 28–39% of mean annual density, 25–65% of mean annual biomass, and 29–34% of mean annual production at two study sites. Benke and Meyer (1988) found that microfiltering-collectors and gathering-collectors strongly dominated invertebrate numbers on snags in the Ogeechee River, Georgia.

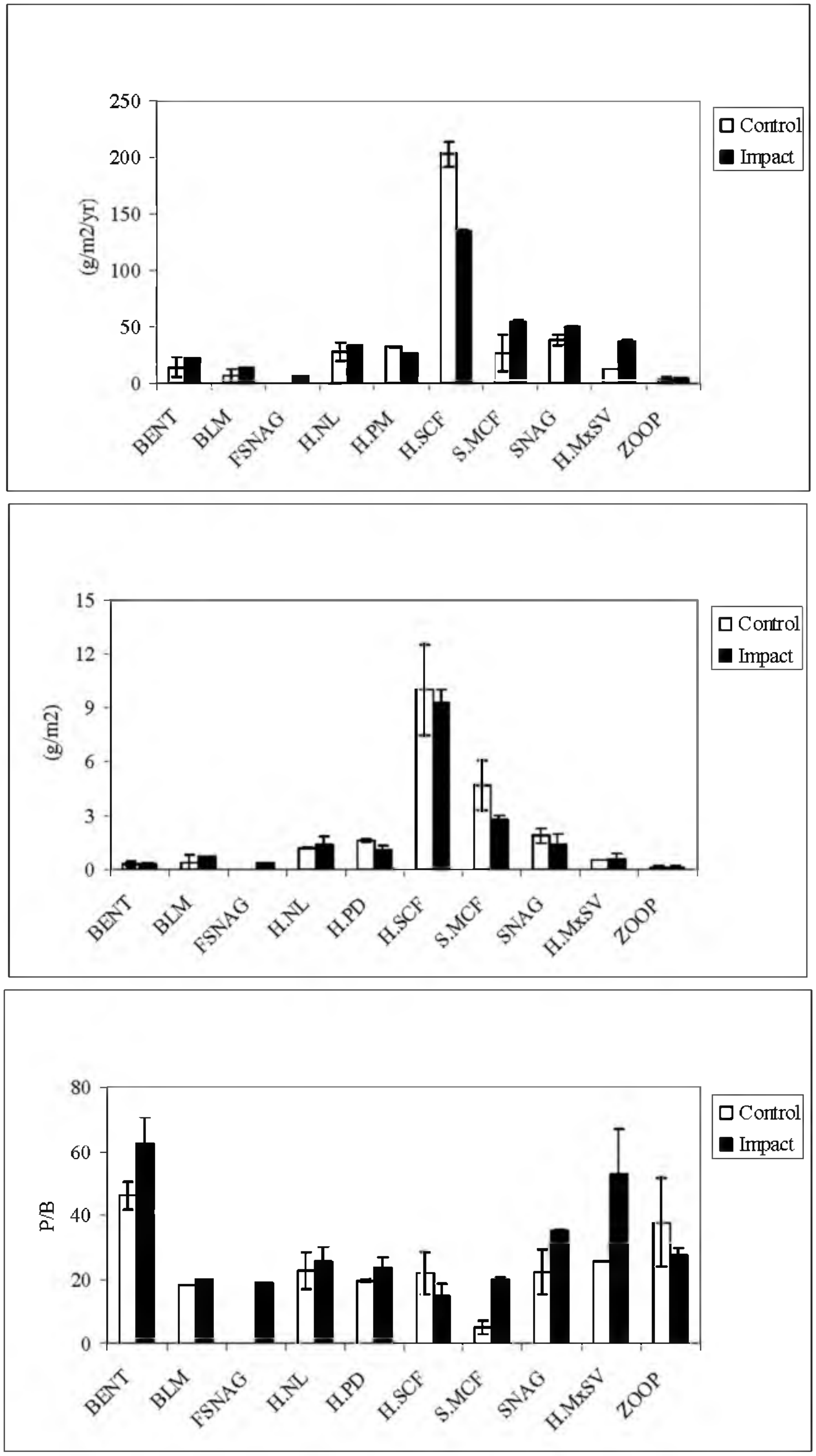


Figure 11-9. Mean annual production, biomass, and P/B ratio for all habitats at Control and Impact site. Estimates were obtained by averaging Year 1 + Year 2/2. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Table 11-3. Annual production (mg m^{-2} yr^{-1}) at the Control site. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Taxon	BENT	BLM	FSNG	H.NL	H.PD	H.SCF	S.MCF	SNAG	H.MxSV	ZOOP
Ablabesmyia	664									194
Acari		4940					2639		559	164
Bezzi/Palpomyia						9404	2141	1974		
Caecidotea							2450			
Caenis	1186				1615					370
Celina				3137						
Chaoborus	1421									
Chironomus	4655			2301	1812	12769			621	
Curculionidae										175
Cypria/Physocypria	2607			1547					1106	632
Dicrotendipes					1770	10397		3456	610	
Glyptotendipes				1937	7739	35254		13336	337	
Goeldichironomus						13930		2354		
Guttipelopia									325	
Helobdella									551	
Hyalella azteca					3200	24990		4746		
Mesocyclops										196
Oligochaete						12494	3374			
Polypedilum		462		7200		15514	2117			
Tipulidae							1243			
Other	3865	1151		12005	15982	68397	12376	12999	2341	1570
Total	14398	6553		28128	32119	203149	26340	38866	6450	3302

Channelization also altered benthic aquatic invertebrate community structure. Mid-channel benthic communities, while not highly diverse, are often composed of several dipteran, ephemeropteran, trichopteran, and molluscan species (Benke et al. 1984, Smock et al. 1985, Stites 1986, Stites and Benke 1989). Dominant species in the channelized Kissimmee include the microcrustacean group *Cypria/Physocypria*, several dipterans, and aquatic mites. Most of these taxa are common and widespread in lentic and lotic systems of the southeast United States, and are generally tolerant of organic pollution and low levels of dissolved oxygen.

Bivalves are probably more abundant in the Kissimmee River than indicated by our samples, and may require more attention in future studies because of national concern about declines in the biodiversity of this group, and because the exotic Asian clam (*Corbicula fluminea*) invaded the Kissimmee during channelization. Prior to channelization, a survey of freshwater mussels (Family Unionidae) in peninsular Florida, including two sampling sites in the Kissimmee River, identified seven species as occurring in the Kissimmee/Everglades drainage basin (Johnson 1972). Only one of these, *Elliptio buckleyi*, was collected in the Kissimmee River. After channelization, Vannote (1971) collected *P. buckleyi*, another unionid *Anodonta couperiana*, and *Corbicula*. We occasionally made qualitative collections of *P. buckleyi*, *A. couperiana*, and possibly a third unionid, *A. imbecilis*, as well as *Corbicula*.

It is difficult to predict how this group of benthic filtering-collectors will respond to restoration, but some insight may be gained from considering data collected during the Demonstration Project (Toth 1991). *Corbicula* populations increased at several river locations with reestablished flow approximately one year after construction of the demonstration project weirs, and attained a maximum density of 2757 m^{-2} at one location. When this location was sampled again in August, after three months of low or no flow, density had decreased to 9 m^{-2}. Similar declines were observed at the other river channel locations. Although density of *Corbicula* may increase within restored river channels, it is not expected to displace any native bivalves or play a major role in the trophic dynamics of the restored system.

Table 11-4. Annual production (mg m^{-2} yr^{-1}) at the Impact site. BENT = Mid-channel, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, , S.MCF = Woody Shrub, SNAG = River Channel Woody Debris, H.MxSV = Miscellaneous Submerged Vegetation, ZOOP = Mid-channel Water Column.

Taxon	BENT	BLM	FSNG	H.NL	H.PD	H.SCF	S.MCF	SNAG	H.MxSV	ZOOP
Acari		326	674				8730			221
Belostoma			172							
Bezzia/Palpomyia	1062						8339			
Caenis	1410			1267						
Chaoborus	9465									
Chironomus	6236			3800	2411				1857	
Collembola			198							
Cypria/Physocypria				3369	1878	6517			1910	1987
Dicrotendipes				4223	2238	15787		4520	3789	
Erythemis								4596		
Fittauimyia		1083	346							
Glyptotendipes					2549	15551		22883		
Hyalella azteca				1677	3359	7796		3731	2447	
Laccophilus		444								
Microtendipes	1724			3061	1286					
Natarsia		314								
Oligochaete			1198				3895			
Polypedilum						7292		2678		
Procambarus		625							14217	
Scirtes							3037			
Tipulidae							6437			
Other	2290	4100	709	15907	12996	81750	24741	11401	13210	1985
Total	22187	6894	3297	33304	26716	134695	55179	49910	37430	4193

Secondary Production

Our estimates of biomass and secondary production rely on estimates of individual mass and growth rates from regression equations developed in other systems, with minimal replication over a broad temporal scale. We expect some error to be associated with the cross-organism and cross-system use of these equations, and from the fact that our estimates of biomass were obtained from a few replicates. However, this error will be applied systematically across habitats at Control and Impact sites, which will allow us to make inferences about changes between sites, and between the baseline and post-construction periods (Benke et al. 1998). Also, we reduce the influence of errors for individual taxa by emphasizing estimates for communities in each habitat, and for guilds such as functional feeding groups (Morin and Dumont 1994).

Annual production in all habitats at Control and Impact sites was dominated by taxa atypical of relatively undisturbed rivers of the southeastern Coastal Plain. Core taxa, based on percent of total production, were dominated by lentic and depositional chironomids, and several larger lentic taxa, including *Hyalella azteca* and coleopterans.

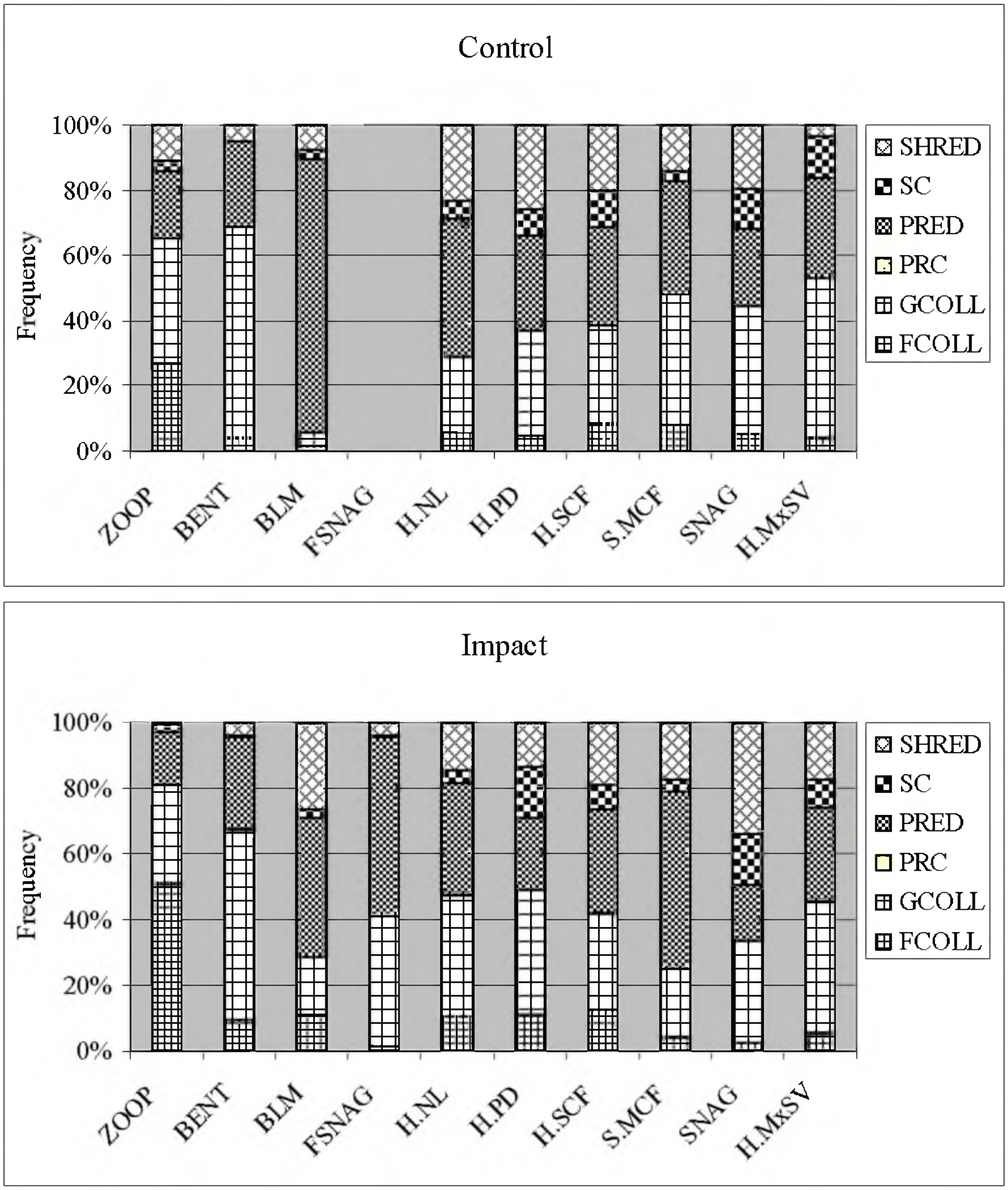
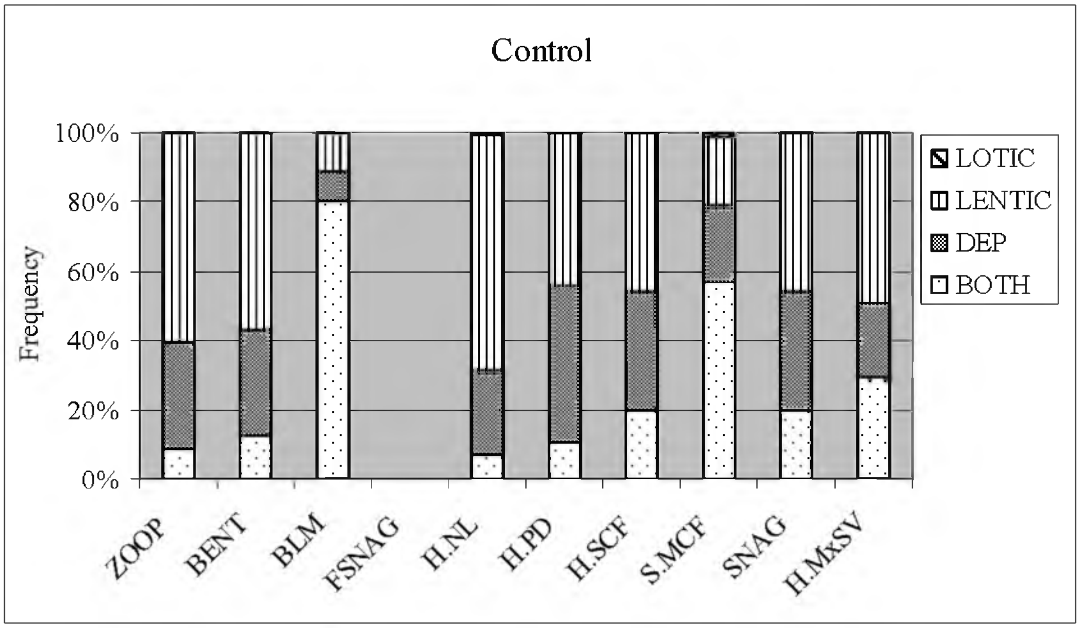


Figure 11-10. Distribution of total production among functional feeding groups at Control and Impact sites. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.



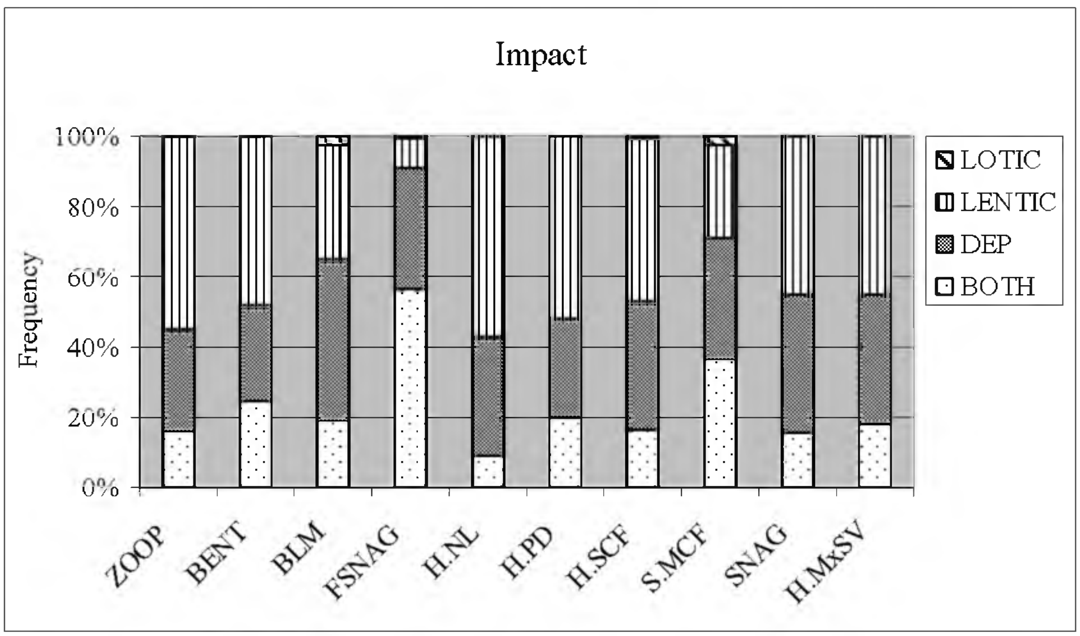


Figure 11-11. Distribution of total production among functional habitat groups at Control and Impact sites. ZOOP = Mid-channel Water Column, BENT = Mid-channel Benthic, H.MxSV = Miscellaneous Submerged Vegetation, H.NL = *Nuphar lutea*, H.PD = *Polygonum densiflorum*, H.SCF = *Scirpus cubensis*, SNAG = River Channel Woody Debris, BLM = Broadleaf Marsh, FSNG = Floodplain Woody Debris, S.MCF = Woody Shrub.

Community production estimates for 40 streams around the world range from 0.6 g m^{-2} to 612 g m^{-2}, but estimates >70 g m^{-2} occur at organically enriched sites, downstream of impoundments, or in warm desert streams (Benke 1993). By averaging baseline production estimates across habitats, we obtain a value of 40 g m^{-2} at the Control site and 37 g m^{-2} at the Impact site. Our estimates are much larger than the 3 g m^{-2} for Cedar Creek, South Carolina, (Smock et al. 1985), 2.4 g m^{-2} and 6.1 g m^{-2}, respectively in

Buzzards Branch and Colliers Creek, Virginia (Smock et al. 1992), which occur in the Coastal Plain but are smaller than the Kissimmee River. We estimated production of 14 and 22 g m^{-2} in the benthos at Control and Impact sites, respectively, while Benke et al. (1984) reported 21 g m^{-2} in sandy benthos and 18 g m^{-2} in mud benthos for the Satilla River. We estimated production of 39 g m^{-2} and 50 g m^{-2} on snags at Control and Impact sites, respectively, while production was 65 g m^{-2} on snags in the Satilla River. Because estimates of secondary production within Kissimmee River channel habitats is within the range of values reported for similar habitats in unmodified Coastal Plain rivers, post-construction estimates of secondary production within these habitats likely will not provide a useful measure of restoration success. However, changes in the distribution of production among functional feeding and functional habitat groups can be used as indicators of restored hydrology and restoration success.

River floodplains are typically highly productive environments that support abundant fish and wildlife resources. Most studies of floodplain macroinvertebrate production have occurred in systems much smaller than the Kissimmee or have focused on a small number of species rather than whole communities (Smock et al. 1985, Gladdon and Smock 1990, Smock et al. 1992, Duffy and LaBar 1994, Pickard and Benke 1996), making comparisons between these studies and our baseline data difficult. Estimates of total secondary production for floodplain macroinvertebrate communities within the channelized system are very low (6.0 and 6.4 g m^{-2} yr^{-1}, respectively for Pool A and C), and are within the range of values reported for single species and small groups of aquatic invertebrates.

Aquatic Invertebrate Drift

Aquatic invertebrate drift is a key functional attribute of flowing water systems. Drift can be an effective way for some aquatic organisms to colonize new areas (Cellot 1989), and can play an important role in energy transfer to higher trophic levels (Benke et al. 1985, Rader 1997). Aquatic organisms can enter the water column in a number of ways, including behavioral (i.e., periodic, for example, to escape from a predator), constant (i.e., background drift due to accidental dislodgement), and catastrophic (i.e., as a result of some major adverse event) drift mechanisms (Waters 1972). In the channelized (non-flowing) Kissimmee River, aquatic macroinvertebrates are rare in the drift. Those that do occur likely enter the water column through active swimming or rafting on floating vegetation (e.g., *Pistia stratiotes*).

Because the channelized Kissimmee River functions more like a lake than a river, and supports an aquatic invertebrate community more typical of a lentic system, drift composition in the channelized Kissimmee River is very different from free-flowing southeastern Coastal Plain blackwater rivers (Benke et al. 1986, 1991). In these systems, larval Coleoptera, Diptera, Ephemeroptera, and Trichoptera are the major contributors to drift numbers and biomass. Microcrustaceans generally account for a small proportion of drifting organisms (Table 11-5).

Reestablishment of an aquatic macroinvertebrate community typical of unmodified southeastern Coastal Plain rivers is a prerequisite for reestablishing invertebrate drift composition typically found in southeastern blackwater rivers. Restoration of continuous flow and in-channel habitat structure will be the impetus for macroinvertebrate (including Coleoptera, Ephemeroptera, and Trichoptera) colonization of restored habitats. Colonization by most river channel macroinvertebrate taxa likely to be found in the drift will occur through adult oviposition. As aquatic invertebrate community structure is restored, seasonal variable flow patterns are expected to result in a shift in macroinvertebrate drift composition from microcrustaceans to one more typical of unmodified Coastal Plain rivers (i.e., macroinvertebrates).

Reference Conditions, Comparisons, & Expectations

Introduction

Channelization of the Kissimmee River likely impacted aquatic invertebrate community structure, functional feeding group associations, productivity, and drift dynamics. Community structure and functional organization on snags and benthic habitats are very different from those of reference sites.

Table 11-5. Major invertebrate groups found in the drift of the Satilla and Ogeechee Rivers, Georgia (Benke et al. 1986, 1991) and Pool C of the channelized Kissimmee River. There was no significant difference between invertebrate drift numbers or biomass between Pools A and C; therefore, only Pool C data is presented. Numbers indicate frequency of occurrence.

	Satilla River		Ogeechee River		Kissimmee River (Pool C)	
Taxonomic Group	Density	Biomass	Density	Biomass	Density	Biomass
Diptera	52.9	53.8	27.3	10.6	< 1	11.2
Coleoptera	11.3	21.5	6.2	27.4	< 1	2.5
Ephemeroptera	5.8	6.2	15.4	34.6	< 1	7.4
Trichoptera	18.6	13.8	11.5	20.2	--	--
Odonata	1.4	4.6	1	5.3	<1	2.4
Crustacea*	10	< 1	31.9	1.9	96.8	54.6
Miscellaneous	--	--	6.7	--	2.7**	21.9**

* Includes macro- and microcrustaceans.
** Includes Hemiptera, Trichoptera, Megaloptera, Lepidoptera, Collembola, Gastropoda, Oligochaeta, and Nematoda

Aquatic invertebrate species richness and diversity in remnant Broadleaf Marsh are likely lower than pre-channelization marshes, and aquatic invertebrate drift is dominated by zooplankton rather than macroinvertebrates. To determine success of the Kissimmee River restoration project, specific comparisons must be made between reference and baseline conditions and between baseline and post-construction conditions. Comparisons between the reference and baseline conditions estimate whether the system has changed as a result of channelization, and to what extent, and provide clues as to what the pre-channelization condition may have been and what the restored condition might be. Following restoration, comparisons between the baseline and post-construction conditions will reveal if the system has responded to restoration efforts, and whether the response is in the expected direction and magnitude. The following sections describe development of reference conditions for habitat-specific aquatic invertebrate communities, compare reference conditions with baseline conditions, and predict how communities are expected to respond to restoration through development of specific habitat-based expectations for restoration.

River Channel Aquatic Invertebrate Community Structure and Production

Pre-channelization data from the lower Kissimmee River basin would provide the best reference conditions for assessing aquatic invertebrate responses to Kissimmee River restoration. However, an extensive literature search found no information on aquatic invertebrate community structure or functional characteristics in the pre-channelized Kissimmee River.

Large Woody Debris

Methods. In order to develop quantitative predictions of aquatic invertebrate responses to Kissimmee River restoration, published studies of invertebrate communities in other southeastern, blackwater Coastal Plain river/floodplain systems were reviewed. Based on this review, data from two Coastal Plain river systems, the Satilla and Ogeechee rivers in Georgia, were selected as appropriate reference sites for developing expectations for restoration of density, biomass, and production of passive filtering-collectors on large woody debris and aquatic invertebrate community structure in sand habitats.

The Satilla River provides the primary source of information on functional feeding group composition, density, biomass, and annual production of aquatic invertebrates on large woody debris within the pre-channelized Kissimmee River (Benke et al. 1984). The Satilla River is a sixth-order, blackwater southeastern Coastal Plain river characterized by a very low gradient, low pH, high organic carbon, and high color (Benke et al. 1986).

In order to quantify aquatic invertebrate community structure on large woody debris in the Satilla River, Benke et al. (1984) sampled snags from two locations for one year. Six samples per site were collected every two weeks from May through August, and monthly for the remainder of the year. Invertebrates were identified and measured. Invertebrate density and standing stock biomass were converted to amount per square meter of habitat surface for each snag sample. Production was estimated using the size-frequency method.

Results. Within the Satilla River, passive filtering-collectors accounted for 75–80% of total numbers, 65–75% of total biomass, and 72–79% of total production at two sample locations (Benke et al. 1984).

Discussion and Comparison with Baseline Condition. Filtering-collectors were selected as an indicator guild because they often account for the largest proportion of mean annual density, standing stock biomass, and production on snags in southeastern river systems. Because most passive filtering-collectors are sedentary and utilize various sieving mechanisms for removing particulate matter from suspension, continuous flows are necessary to transport fine particulate organic matter that can be captured and used as a food source. Additionally, many filtering-collectors respond predictably (decrease) to increased perturbation (e.g., no flow, low dissolved oxygen) (Lenat 1988, Lamberti and Berg 1995, Barbour et al. 1996). Channelization of the Kissimmee River eliminated flow through remnant river channels, reduced levels of dissolved oxygen within the water column (Colangelo 2005), and likely altered density, biomass, and production of passive filtering-collector guild on large woody debris.

Passive filtering-collector taxa are rare on large woody debris in the channelized Kissimmee River, accounting for <2% of mean annual density, <3% of mean annual biomass, and <1% of mean annual production in Pool A, and <1% of mean annual density, <2% of mean annual biomass, and <1% of mean annual production in Pool C. This is very different from the Satilla River, where passive filtering-collectors account for the greatest proportion of these metrics at two sample locations (Benke et al. 1984) (Figure 11-12). Although the Satilla River is the sole reference site for pre-channelization community structure and production on river channel snags, other studies (Thorp et al. 1985, Smock et al. 1985, Benke and Meyer 1988) support the fact that the passive filtering-collectors often make up the largest proportion of density, biomass, and production within this habitat.

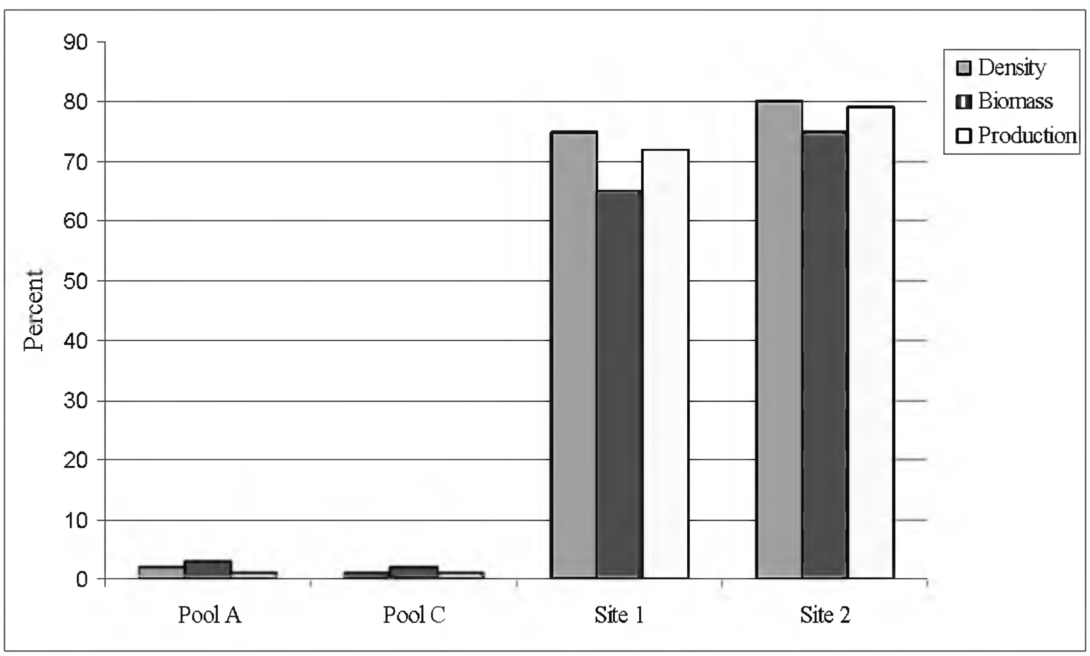


Figure 11-12. Mean annual density, biomass, and production of passive filtering-collectors on snags in the Kissimmee River (Pools A and C), and Satilla River, Georgia (Sites 1 and 2) (Benke et al. 1984).

Based on a comparison of baseline and reference data for mean annual density, biomass, and production of snag-dwelling passive filtering-collectors, restoration of physical and chemical habitat structure within the Kissimmee River likely will result in shifts in functional feeding group composition on snags within the restored river. The following expectation has been developed from baseline data and best available reference data.

<u>Expectation:</u> Increased relative density, biomass, and production of passive filtering-collectors on river channel snags.

Passive filtering-collectors are expected to respond quickly to restored flow and increases in levels of dissolved oxygen within the river channel, and account for the greatest proportion of mean annual density, mean annual biomass, and mean annual production on large woody debris in restored river channels (Koebel 2005a). However, because passive filtering-collector macroinvertebrates are rare in the channelized system, the time frame for redistribution of density, biomass, and production among functional feeding groups is primarily dependent on colonization by filtering-collectors and displacement of existing dominant functional feeding groups, which will depend on distance colonists must travel. It is expected that small and large-bodied filtering-collectors, primarily chironomids, simuliids, and caddisflies will immigrate from lotic systems within the Kissimmee basin (e.g., Fisheating Creek, Tiger Creek, Cypress Creek, Weohykapka Creek) and likely colonize within six to nine months. The potential for high standing stock biomass of several filtering-collectors (primarily caddisflies), and rapid biomass turnover rates for others (Simuliidae and Chironomidae), likely will result in the greatest proportion of mean annual density, mean annual biomass, and mean annual production being attributed to passive filtering-collectors.

Sampling of snags will commence approximately six months following initiation of the revised headwaters regulation schedule and reestablishment of continuous flow. Snag-dwelling macroinvertebrate density, biomass, and production will be analyzed for a minimum of three years following reestablished flow. Post-construction sampling will include collection of monthly, replicate (five) snag samples from randomly selected locations within reconnected channels in Pool C and remnant channels in Pool A. Samples will be analyzed for invertebrate species identity, functional feeding group composition, density, and standing stock biomass. Passive filtering-collectors will be identified according to Merritt and Cummins (1996). Production will be calculated using the instantaneous growth rate (IGR) method. Growth equations for major taxa will be determined experimentally or obtained from scientific literature. Monthly means will be averaged annually to determine mean monthly density and biomass for the filtering-collector guild. The three annual estimates of mean monthly density and biomass will be averaged to obtain a mean annual value. The three estimates of annual production also will be averaged to determine mean annual production. Although values for these metrics may vary from year to year, a multi-year, multi-metric evaluation of changes in macroinvertebrate functional composition and production on snags will provide an objective measure of restoration-related changes that integrate potential intra- and inter-annual variability.

Sand Substrates

<u>Methods.</u> The primary source of information on sand-dwelling macroinvertebrates within the pre-channelized Kissimmee River is derived from published data on community composition in the Ogeechee and Satilla Rivers, Georgia (Benke et al. 1984, Stites 1986). The Ogeechee River, a sixth-order, blackwater river in the lower Coastal Plain of Georgia, is characterized as low gradient (0.02%), with a high level of dissolved organic carbon, mean annual discharge of 66.8 m^3 s^{-1} (44 year period of record), mean annual water temperature ranging from 3–32°C (Stites 1986), and a river channel bottom consisting of 80–90% sand (Stites and Benke 1989). Detailed sampling methods for sand-dwelling macroinvertebrates can be found in Benke et al. (1984) and Stites (1986). Additional information was derived from published reports on the geographic distribution of sand-dwelling macroinvertebrates throughout central Florida (Dunkle 1989, Toth 1991, Epler 1992, Merritt et al. 1996, Berner and Pescador 1988)

<u>Results.</u> The sand-dwelling aquatic invertebrate community of the Ogeechee and Satilla Rivers are quite similar. Dominant macroinvertebrates included the dipterans *Corynoneura* sp., *Cladotanytarsus* sp., *Cryptochironomus* sp., *Parakiefferiella* sp., and *Robackia* sp., Certatopogonidae, and oligochaetes. Other

dominant taxa in the Ogeechee included *Lopescladius* sp., *Rheosmittia* sp., and *Corbicula fluminea* (Stites 1986). Additional dominant taxa in the Satilla River included *Polypedilum* sp., *Tanytarsus* sp., and *Thienemanniella* sp.

Based on habitat preferences and geographic distributions throughout Florida, other taxa likely to be present among the sandy benthos of the restored Kissimmee include Ephemeroptera, including *Stenonema* sp. and *Cercobrachys* sp. (Berner and Pescador 1998); mollusks, including *Musculium/Pisidium* complex (Toth 1991); odonates, including *Dromogomphus spinosus*, *Gomphus minutus*, *Gomphus dilatatus*, and *Stylurus plagiatus* (Dunkle, 1989); and Trichoptera, including *Oecetis* sp. and *Setodes* sp. (Merritt et al. 1996).

Discussion and Comparison with Baseline Condition. Most of the historic sand substrate within mid-channel habitats of remnant river channels is covered with large accumulations of organic matter, primarily derived from dead and decaying aquatic vegetation. The associated aquatic invertebrate community consists of taxa most often associated with organically enriched environments, and are generally tolerant of low levels of dissolved oxygen. Restoration of flow is expected to flush organic deposits, or redistribute existing sand to cover these deposits and form sand bars along the inside margins of meanders. Restoration of flow and reestablishment of a sand substrate is likely to result in increased levels of dissolved oxygen within restored channels by reducing microbial sediment oxygen demand (Colangelo 2005). These shifts in physical and chemical habitat structure are likely to induce changes in aquatic invertebrate community structure within mid- and marginal channel sand habitats.

Because of the lack of historical data, the Ogeechee and Satilla Rivers provide reasonable reference conditions for aquatic invertebrate community structure in sand habitats of the pre-channelized Kissimmee River. Although reference conditions are solely derived from these two systems, other studies (Whitman and Clark 1984, Strommer and Smock 1989) indicate that many of the same taxa dominate sand substrates in other lotic systems of the southern United States (Virginia and Texas). Most taxa occurring in sand habitats of the Ogeechee and Satilla Rivers are considered characteristic, or obligate sand-dwellers (Whitman and Clark 1984). These characteristic taxa are absent or rare in benthic habitats of the channelized Kissimmee River; however, most occur within the lower Kissimmee basin or adjacent watersheds, and many are likely to quickly colonize restored sand substrates (Table 11-6).

Based on a comparison of baseline and reference data for macroinvertebrate community composition in sand habitats, restoration of physical habitat structure (sand habitat) within the Kissimmee River likely will result in colonization of invertebrate taxa considered characteristic of sand habitats. The following expectation has been developed from baseline data and best available reference data.

Expectation: Aquatic invertebrate community structure in river channel benthic habitats.

The macroinvertebrate fauna of river channel benthic habitats will primarily consist of taxa that are common and characteristic of sand substrates (Koebel 2005b).

The expectation for shifts in aquatic invertebrate community structure in sand habitats of the restored Kissimmee River is less rigorously defined; however, sand substrates of many southeastern Coastal Plain rivers support a characteristic and consistent group of aquatic invertebrate taxa. Because many of these taxa appear to be habitat specialists, it is not unreasonable to expect that many of these taxa will colonize sand substrates in the restored Kissimmee River. It is unlikely that all taxa will be present in restored habitats; however, representative taxa (Table 11-6) are expected to show substantive change relative to the baseline condition and therefore be reasonable indicators of habitat restoration.

Sampling of sand habitats will commence approximately six months following initiation of the revised headwaters regulation schedule and reestablishment of continuous flow. Sand-dwelling macroinvertebrates will be collected for a minimum of three years following reestablished flow. Post-construction sampling will include collection of monthly, replicate (five) mid-channel sand samples and five marginal channel sand samples from randomly selected locations within reconnected channels in Pool C. For comparison, mid-channel benthic samples also will be collected in remnant channels in Pool A. Samples will be analyzed for invertebrate species identity. Community composition will be compared to the baseline condition and stated expectation.

Floodplain Macroinvertebrate Community Structure

Methods. A thorough literature search found no information on aquatic invertebrate community structure characteristics of pre-channelization Broadleaf Marshes of the Kissimmee River, or marshes that were structurally similar to pre-channelization marshes. Therefore, in the absence of historical data or suitable reference sites, baseline data collected in remnant, but altered BLM in Pool C, was used to predict a minimal response by aquatic invertebrates to restored hydroperiod and habitat structure.

An attempt was made to collect quarterly, replicate (three) aquatic invertebrate samples from remnant BLM in Pools A and C between August 1995 and May 1997. Each quarter, when water was present on the floodplain, replicate stovepipe (area = 1662 cm^2) samples were collected from randomly selected locations in BLM. Sample locations were determined by traveling a randomly determined distance ($\leq$400 m) and direction (0–360°) from a randomly determined starting point within BLM. Following trap placement, water depth within the trap was recorded and all vegetation was removed. A dip-net equipped with a 118 µm mesh net was used to remove invertebrates. A total of ten "dips" constituted a sample. All invertebrates were identified. Species richness and species diversity were calculated for each replicate on each date. Because pasture habitat in Pools A and C was dry during most of the baseline period, aquatic invertebrates were not quantified in this habitat.

Results. Broadleaf Marsh habitat in Pools A and C was dry during much of the study period. Pool A was sampled only once, and Pool C was sampled only three times. In Pool A, species richness was 21 and species diversity was 0.84. In Pool C, species richness ranged from 15 to 32 (total species richness = 65) and species diversity ranged from 1.86 to 2.75 (mean diversity = 2.37). Species richness and diversity in pasture habitat was assumed to be 0 and 0.00, respectively.

Discussion and Comparisons with Baseline Condition. Documented studies on aquatic invertebrate community structure of subtropical wetland systems are limited (Rader 1994, Evans et al. 1999, Rader 1999), and have focused on systems that are structurally different from pre-channelization Broadleaf Marshes of the Kissimmee River floodplain (i.e., Water Conservation Areas and flatwoods marshes). Rader (1994) found 174 taxa comprise the known aquatic invertebrate community in the Everglades, but indicates that the actual number of taxa may be as great as 250. Diversity estimates for benthic macroinvertebrates in natural flatwoods marshes of central Florida range from 3.94 to 4.50, with a mean of 4.23 (Evans et al. 1999). Although vegetation communities of the Everglades and flatwoods marshes are structurally different from pre-channelized marshes of the Kissimmee River, it is likely that the aquatic invertebrate community of restored Broadleaf Marshes will be species rich and diverse. Although these studies provide insight into the potential for high species richness and diversity within restored or natural marshes of central Florida, they can not be used to predict species richness and diversity in restored BLM. However, assuming that a restored BLM will support an aquatic invertebrate community with at least the same species richness and diversity as remnant marshes, baseline data from Pool C can provide a conservative estimate of species richness and diversity in restored BLM.

Expectation: Aquatic invertebrate community structure in Broadleaf Marsh.

Aquatic invertebrate species richness and species diversity will be $\geq$65 and $\geq$2.37, respectively in restored Broadleaf Marsh (currently pasture in the channelized system) (Koebel 2005c).

Unpredictable hydroperiods and homogeneous vegetation communities in remnant Broadleaf Marsh likely limit aquatic invertebrate species richness and diversity. Although data on pre-channelization species richness and diversity of floodplain wetlands do not exist for the pre-channelized Kissimmee, reestablishing long-term hydroperiods and associated development of a diverse, heterogeneous wetland plant community likely will allow for development and persistence of a diverse macroinvertebrate community.

Initial sampling of existing Broadleaf Marsh and future Broadleaf Marsh (existing pasture) will coincide with sampling of large-bodied fish and wading bird use of floodplain habitats (i.e., approximately one year after initiating the revised headwaters regulation schedule). Although this time frame is not sufficient to reestablish historic aquatic invertebrate community structure characteristics, these data may be useful for interpreting the initial response and distribution of large-bodied fishes and wading birds within floodplain habitats. Methods will be identical to those outlined in Anderson et al. (1998b), and include monthly, replicate (five) stovepipe (area = 0.105 m^2) or throwtrap (area = 0.25 m^2) samples from randomly selected locations within Pools A and C. Additional focus will be on density and biomass of "keystone" taxa (e.g., crayfish, grass shrimp, dragonflies, and snails) likely to serve as high quality prey items for higher trophic levels (e.g., wading birds and fishes). Sampling for these taxa will correspond with floodplain fish sampling and consist of monthly, replicate (ten) throwtrap (1 m^2) samples from existing BLM and pasture habitats undergoing transition to BLM in Pool C and remnant BLM and improved pasture in Pool A. Sampling will continue for at least three years.

Aquatic Invertebrate Drift

<u>Methods</u>. An extensive literature search found no information on aquatic invertebrate drift in the pre-channelized Kissimmee River. In order to develop quantitative predictions of aquatic invertebrate responses to Kissimmee River restoration, published studies of aquatic macroinvertebrate drift in other southeastern, blackwater Coastal Plain river/floodplain systems were reviewed. Based on this review, data from two southeastern Coastal Plain rivers were selected as appropriate reference sites for developing an expectation for restoration of aquatic macroinvertebrate drift composition in the restored Kissimmee River.

Reference conditions have been developed based on macroinvertebrate drift data from the Satilla and Ogeechee Rivers, Georgia (Benke et al. 1986, 1991). In order to characterize macroinvertebrate drift density and biomass in the Satilla River, Benke et al. (1984) collected samples from the water column using two nets (mesh = 400 µm, net opening = 0.135 m^2). One net was positioned 10–50 cm above the sand bottom, while the second net was placed just below the water surface. Current velocity was measured at each net in order to determine the volume of each sample. Samples were collected at two to four-week intervals just after dark for a period of one year. All organisms were identified and measured. Numbers per volume of water were converted to biomass per volume of water using taxon-specific length-mass relationships. Ogeechee River drift was characterized by Benke et al. (1991) in a similar manner, although mesh size (234 µm) and net opening (89.4 cm^2) differed between studies, and the Ogeechee River study was conducted for two years.

<u>Results</u>. These studies indicate larval Coleoptera, Diptera, Ephemeroptera, and Trichoptera are the major contributors to drift numbers and biomass in these three systems (Table 11-5).

<u>Discussion and Comparison with Baseline Conditions</u>. Because the channelized Kissimmee River is characterized by no flow, aquatic invertebrate drift is primarily due to active swimming or rafting on floating aquatic vegetation. Drift composition within the channelized Kissimmee consists primarily of zooplankton which is very different from the Satilla River. Drift community structure from the Satilla and Ogeechee Rivers provide reasonable reference conditions for macroinvertebrate drift in the restored Kissimmee. Reestablished continuous flow and restoration of habitat structure will be the impetus for changes in aquatic invertebrate community structure, as well as the subsequent shift in invertebrate drift density and biomass from dominance by zooplankton to dominance by macroinvertebrates. The following expectation has been developed from baseline data and the best available reference conditions.

<u>Expectation:</u> River channel Macroinvertebrate drift composition.

Macroinvertebrate drift composition will be dominated by Coleoptera, Diptera, Ephemeroptera, and Trichoptera (Koebel 2005d).

Invertebrate drift will be sampled monthly beginning two years after implementation of the revised headwaters regulation schedule, assuming that this time period is sufficient to reestablish river channel invertebrate communities typical of unmodified southern Coastal Plain rivers. Drift will be quantified monthly from two sites (upper and lower) in Micco Bluff Run. Paired drift nets (net opening = 900 cm2, mesh size = 125 µm), facing into the flow, will be placed at the water surface and 0.5 m above the channel substrate. Samples will be collected for a period of four hours beginning one-half hour after dusk. Flow will be measured at each net opening when nets are set or retrieved. All invertebrates will be identified to Order (minimally), and an appropriate length measurement will be taken to determine length-mass relationships. Numbers and biomass per volume of water will be calculated for each taxonomic group. Sampling will occur for a minimum of two years. Post-construction data will be compared to baseline data and the expectation in order to determine changes in drift density and biomass.

Table 11-6. Sand-dwelling taxa in reference sites and the channelized Kissimmee River, and taxa likely to colonize restored sand habitats of the Kissimmee River.

Taxon	Satilla River[1]	Ogeechee River[2]	Kissimmee-Pool A	Kissimmee-Pool C	Restored Kissimmee	Reference
Diptera						
Corynoneura	X***	X			X	Merritt et al. 1996
Cladotanytarsus	X**	X				
Cryptochironomus	X**	X			X	Merritt et al. 1996
Lopescalidius		X			X	Epler 1992
Parakiefferiella	X***	X			X	Epler 1992
Paracladoplelma					X	Epler 1992
Polypedilum	X**	X	X#	X#	X	Merritt et al. 1996
Rheosmittia		X			X?	Epler 1992
Robackia	X***	X			X	Epler 1992
Tanytarsus	X**				X	Merritt et al. 1996
Tanytarsini group			X#	X#	X	Merritt et al. 1996
Thienemaniella	X**				X	Epler 1992
Orthocladinae		X			X	Epler 1992
Ceratopogonidae	X***	X			X	Merritt et al. 1996
Ephemeroptera						
Stenonema					X	Berner&Pescador 1988
Cercobrachys					X	Berner&Pescador 1988
Mollusca						
Musculium					X	Toth 1991
Pisidium					X	Toth 1991
Corbicula fluminea		X			X	Toth 1991
Trichoptera						
Nectopsyche					X	Pescador et al. 1995
Oecetis					X	Merritt et al. 1996
Setodes					X	Merritt et al. 1996

** = frequent
*** = abundant
# = rare

1 = Benke et al. 1984, 2 = Stites 1986

Bi-directional Exchange of Aquatic Invertebrates between River Channel and Floodplain

Reliable reference conditions for bi-directional exchange of aquatic invertebrates between river channel and floodplain habitats do not exist; therefore, a specific expectation for restoration of this component can not be developed. However, because this functional attribute is a key characteristic of healthy river-floodplain systems, and critical to the productivity of higher trophic levels in the river channel and floodplain, it will be evaluated as part of the comprehensive restoration evaluation program to determine restoration of ecological integrity within the Kissimmee river-floodplain system.

Methods. A review of the literature revealed only one study that documented the bi-directional exchange of aquatic invertebrate numbers and biomass between river channels and floodplains (Smock 1994). Drift into and out of two first-order blackwater streams (Colliers Creek and Buzzards Branch) in Virginia was conducted between 1990–1991. Specific sampling methods can be found in Smock (1994).

Results. Because channelization eliminated stage fluctuations within remnant channels of the Kissimmee River, movement of invertebrates to and from the floodplain was considered zero for the baseline condition.

For Colliers Creek, total input of invertebrates to the floodplain over the year by drifting was 1.47 X 10^6 individuals and 0.25 kg dry mass; total output to the channel was 2.68 X 10^6 individuals and 0.15 kg dry mass. Therefore, net exchange through drift was 1.21 X 10^6 individuals to the channel and 0.10 kg dry mass to the floodplain. Copepods, chironomids, and ostracods accounted for most of the net output of individuals from the floodplain, while net input of biomass to the floodplain was primarily by Ephemeroptera, Trichoptera, and Isopods as well as several rare but large taxa (e.g., Odonata and Megaloptera). A total of 2.10 X 10^5 and 0.66 kg dry mass moved onto the floodplain by crawling, with total output to the channel of 0.40 X 10^5 individuals and 0.05 kg dry mass. Therefore, net movement by crawling was 1.70 X 10^5 individuals and 0.61 kg dry mass. Drift and crawling accounted for a net export of 1.04 X 10^6 individuals from the floodplain over the year, but an import of 0.71 kg of biomass from the channel.

For Buzzards Branch, drift densities, biomass concentration, and biomass drift rates were significantly higher in water flowing into than out of the Buzzards Branch floodplain. Copepods and chironomids were the most abundant taxa drifting between the floodplain and channel. Very few individuals crawled between the channel and floodplain at Buzzards Branch. Results of this study indicate that while there may be substantial exchange of organisms across the river-floodplain boundary in these two systems, the floodplains, which produce 67–95% of annual invertebrate production in the two stream systems, retained most of that production.

Discussion and Comparison with Baseline Condition. Although no specific expectation for bi-directional exchange of aquatic invertebrates between the river channel and floodplain has been developed due to lack of reference data, restoration of pre-channelization discharge and floodplain hydroperiod is expected to result in a net movement of invertebrate number and biomass from the river channel to the floodplain during the rising hydrograph (initial flood-pulse), and a net influx of invertebrate numbers and biomass from the floodplain to the river channel during the falling hydrograph.

Sampling of invertebrate exchange will begin approximately two years after initiating the revised headwaters regulation schedule. Paired, replicate (three) drift nets (900 cm^2), equipped with 125 μm mesh netting, will be placed at pre-determined locations at the interface between the floodplain and river channel, in order to capture invertebrates moving onto and off of the floodplain during the rising hydrograph, and onto and off of the floodplain during the falling hydrograph. Nets will be set for three-hour intervals, at four time periods, over a 24-hour period. Nets will be checked and replaced every one hour (or as necessary) to prevent clogging. Current velocity and water depth will be measured at the opening of each net prior to setting and upon retrieval to determine the volume of water sampled. Sampling will occur four times annually, twice on the rising hydrograph, and twice on the falling hydrograph. Actual sampling dates will be determined from daily river channel stage data and visual observations of overbank flow and recession of water from the floodplain. This sampling routine is designed to evaluate temporal variability of import and export from the floodplain over a 24-hour period, and may be adjusted following analyses of initial data.

Secondary Production of Floodplain Aquatic Invertebrates

Methods. A literature review found no information on macroinvertebrate production in pre-channelization marshes of the Kissimmee River or in marshes with similar characteristics as pre-channelization marshes. Therefore, baseline data collected in floodplain habitats (pasture and remnant Broadleaf Marsh) in Pool C was used to predict the minimum level of macroinvertebrate productivity in restored Broadleaf Marsh (currently pasture).

In order to estimate production of aquatic invertebrates in remnant marshes of the Kissimmee River floodplain, replicate (three) stovepipe samples were collected quarterly between August 1995 and May 1997 in pools A and Pool C. Samples were analyzed for species identity, density, and biomass. Production was calculated using the instantaneous growth rate (IGR) method.

Results. Pasture (UP, upland herbaceous vegetation, Bousquin and Carnal 2005) habitats in the channelized system were dry most of the year; therefore, aquatic macroinvertebrate community production within this habitat was assumed to be 0 $g/m^2/yr$. Production of aquatic invertebrates in altered Broadleaf Marsh of the Kissimmee River is low. Remnant BLM in Pools A and C was dry over much of the sample period. Annual invertebrate community production in Pool A and C was 6.4 and 6.0 $g/m^2/yr$, respectively.

Discussion and Comparison with Baseline Condition. Because production of aquatic invertebrates is critical to energy flow pathways in aquatic systems, and production of floodplain invertebrate communities can be several orders of magnitude greater than river channel production, it is important to estimate production of floodplain aquatic invertebrates in order to predict the amount of biomass available for transfer to higher trophic levels.

The expectation for increased aquatic macroinvertebrate production above that of the reference condition is based on expectations for restored aquatic invertebrate community structure, including an increase in species richness, year-round persistence of a diverse aquatic invertebrate community, increases in mean annual biomass for most taxa, and the potential for high biomass turnover rates (annual P/B ratios) for many taxa. Because the magnitude of production depends on standing stock biomass and biomass turnover rates, factors affecting one, or both, will influence rates of production (Benke 1984). Dipterans may account for >30% of all taxa and >50% of total individuals in natural flatwoods marshes of central Florida (Evans et al. 1999). Assuming a cohort P/B ratio of 5 (Waters 1969) and a mean developmental time of 21 days, annual P/B ratios for many dipterans can approach 90, which means biomass turnover time may be as short as four days. Annual P/Bs in this range and greater have been reported for numerous Diptera from a variety of aquatic systems (Benke 1998), and indicates the potential for high turnover rates for some taxa to contribute to high rates of annual production. Densities of large invertebrates (e.g., crayfish, grass shrimp, amphipods, and odonates) can be high in natural marshes of central and south Florida (Jordan et al. 1996a, 1996b, Milleson 1976, J.W. Koebel, personal observation). Mean crayfish density within a Broadleaf Marsh of the channelized Kissimmee River approached 40/m^2 when the marsh was inundated to a depth >20 cm (J.W. Koebel, personal observation). Moderate mean annual density and associated biomass of crayfish and other large invertebrates is expected in restored Broadleaf Marsh habitats, and likely will contribute to a high rate of annual invertebrate community production.

Sampling of remnant Broadleaf Marsh and reestablished Broadleaf Marsh (pasture in the channelized system) will commence approximately two years after initiating the revised headwaters regulation schedule. This time frame should be sufficient for reestablishing pre-channelization floodplain vegetation characteristics. Methods will be similar to those outlined in Anderson et al. (1998b), and include collection of monthly, replicate (five) throwtrap (area = 0.25 m^2) samples from randomly selected locations within remnant and restored Broadleaf Marsh in Pools A and C. Samples will be analyzed for species identity, density, and standing stock biomass. Production will be calculated using the instantaneous growth rate method (IGR). Sampling in remnant and restored marsh will continue for three years. The three independent estimates of annual production will be averaged to determine mean annual production, which will be compared to baseline data and the expectation.

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CHAPTER 12

AMPHIBIAN AND REPTILE COMMUNITIES OF THE LOWER KISSIMMEE RIVER BASIN PRIOR TO RESTORATION: BASELINE AND REFERENCE CONDITIONS AND EXPECTATIONS FOR RESTOATION

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ABSTRACT: To characterize baseline (channelized) conditions in the Kissimmee River ecosystem, herpetofauna were surveyed using multiple sampling techniques within several altered floodplain habitats. Amphibian and reptile species richness within the channelized lower Kissimmee basin was similar to that of other disturbed wetland sites of south-central Florida. Many taxa characteristic of undisturbed wetland and upland habitats of central Florida were absent from the baseline surveys. Data were compared to distributions of amphibians and reptiles in central Florida, and with data collected from undisturbed wetlands on the Avon Park Bombing Range, to define reference conditions and evaluate whether channelization altered herpetofaunal community structure and patterns of amphibian reproduction in floodplain habitats. Comparisons suggest that herpetofaunal community structure and patterns of amphibian reproduction in floodplain habitats were severely impacted by channelization. Expectations of changes predicted to result from restoration were developed based on the data presented in this report. The expectation for restoration of community structure predicts that at least 24 amphibian and reptile taxa considered "characteristic" or "frequently occurring" in natural broadleaf marshes (BLM) of central Florida will recolonize restored floodplain habitats within three years of reestablishing hydroperiod and vegetation characteristics similar to the pre-channelization period. The expectation for amphibian reproduction predicts that larval amphibians will be present in restored BLM for at least seven months each year.

INTRODUCTION

Amphibian and reptile (herpetofauna) communities can serve as indicators of the health of aquatic ecosystems, especially wetlands. Adult and larval herpetofauna play an integral role in food web dynamics and energy flow through aquatic and terrestrial ecosystems. They are major consumers of invertebrates and algae (Blaustein and Wake 1990) and, in turn, are consumed by a variety of invertebrates (Travis et al. 1985, Roth and Jackson 1987), fishes (Azevedo-Ramos et al. 1999), birds (Ogden et al. 1976, Collopy and Jelks 1989, Beissinger 1990), and other amphibians and reptiles (Morin 1983, Wilbur et al. 1983, Ashton and Ashton 1988).

Amphibians are of particular interest because of their complex life cycle which includes obligate association of larvae with water and may include a terrestrial or semi-terrestrial adult stage. Thus, environmental conditions within aquatic and terrestrial habitats must be favorable for reproduction, development, and survival. Adult and larval amphibians are vulnerable to low temperature, drought, and shifts in wetland hydrology (Pechmann et al. 1989, Stebbins and Cohen 1995).

Conversion of wetlands to uplands combined with shortened and unpredictable hydroperiods in remnant wetlands following the channelization of the Kissimmee River are likely to have altered herpetofaunal communities. Restoration of pre-channelization hydrology, including long-term floodplain inundation through the Kissimmee River Restoration Project, is expected to reestablish historic floodplain wetland plant communities in the central portion of the Kissimmee river/floodplain ecosystem. Herpetofauna are important biological components for assessing restoration of ecological integrity within the Kissimmee River ecosystem.

Objectives

The objectives of this study are to:

(1) Assess baseline (channelized, pre-restoration) amphibian and reptile community structure in of the Kissimmee River and floodplain;
(2) Assess temporal patterns of amphibian reproduction during the baseline period;
(3) Estimate pre-channelization conditions for amphibian and reptile community structure characteristics and patterns of anuran reproduction using reference data;
(4) Quantify impacts of channelization by comparing pre-channelization (reference) conditions and baseline conditions; and
(5) Develop specific expectations for restoration of herpetofaunal community structure and amphibian reproduction.

BASELINE CONDITIONS

Methods

Study Site

Sampling for herpetofaunal community structure characteristics and patterns of amphibian reproduction was stratified by habitat (plant community). Sampled habitats included Broadleaf Marsh (BLM); Woody Shrub (S.MCF); Upland Herbaceous plant communities (UP); Wetland Forest (WF); and Upland Forest (UF). Broadleaf Marsh habitats are spatially homogeneous, primarily consisting of arrowhead *(Sagittaria lancifolia)*, pickerelweed (*Pontederia cordata*), and maidencane (*Panicum hemitomon*). Woody Shrub is characterized by dense stands of wax myrtle (*Myrica cerifera*) that exist on a bog-like floating mat. The understory is composed of a diverse mixture of broadleaf marsh, wet prairie, and upland vegetation including broomsedge (*Andropogon glomeratus*), sedges (*Cyperus* spp.), pennywort (*Hydrocotyle umbellata*), spatterdock (*Nuphar lutea*), rushes (*Rhynchospora* spp.), and Cuban bulrush (*Scirpus cubensis*). Upland herbaceous communities (pasture) are characterized by upland and mesic grasses, forbs, and shrubs. Wetland Forest habitats are characterized by the presence of red maple (*Acer rubrum*), cabbage palm (*Sabel palmetto*), live oak (*Quercus virginiana*), Pteridophyta, American cupscale (*Sacciolepis striata*), and greenbriar (*Smilax* spp.), while Upland Forest is characterized by *Q. virginiana* and *S. palmetto*. More explicit definitions of these plant communities can be found in Bousquin and Carnal (2005). Sample methods and sample habitats varied according to what metric was being measured.

Visual Encounter Surveys

Visual encounter surveys (VES) (Donnelly et al. 1998a) were conducted monthly over a 31 month period in BLM (Pools A and C) and S.MCF (Pools C and D) habitats, and a 15 month period in WF (Pools B and C) habitats, beginning in August 1996. Surveys were conducted over a 12 month period in UP (Pools A and C) habitats beginning in March 1998. One group of three 50 meter long permanent line transects, divided into five-meter intervals, was established within each habitat approximately 100 meters from and adjacent to the river channel. Transects were set perpendicular to the river channel and separated by 20 meters. In March 1998, in order to more accurately characterize the herpetofaunal community in BLM and S.MCF, six additional 50 meter transects (two groups of three) were established in BLM and

S.MCF. Nine 50 meter transects (three groups of three) also were established at this time in UP habitats. No additional transects were established in WF due to the limited areal extent of this habitat. The specific location of each transect group within each habitat was based on habitat availability, habitat size, and ease of access.

Each transect was surveyed once per sampling event. Sampling events began approximately 30 minutes after sunset. Head lamps or bright flashlights were used to illuminate a one-meter wide strip on each side of the transect line. For every amphibian and reptile encountered, species identity, age class (larva, juvenile, adult), perch height, and substrate association were recorded. Water depth was recorded at 0, 25, and 50 m on each transect using a permanently mounted stream gauge or meter stick.

Community structure was described by species richness (S = the total number of species present); relative abundance (the proportion of individuals of species *i* in relation to the total number of individuals); species diversity (H'), where $H' = -\Sigma(p_i \ln p_i)$ and p_i is the proportion of individuals belonging to the i^{th} taxa; and community evenness (J'), where $J' = H'/\ln S$ (Price 1984). A coefficient of community similarity (CCS) calculated as

$$\frac{\Sigma\,(2m_i)}{\Sigma\,(a_i + b_i)},$$

where a_i is the abundance of species *i* in community *a* (Control site), *b* is the abundance of species *i* in community *b* (Impact site), and m_i is the minimum value for that species in community *a* or *b* (Bray and Curtis 1957), also was calculated for each habitat. A species accumulation curve was developed for each habitat in each pool. An accumulation curve shows the cumulative number of species observed during successive sampling periods. Accumulation curves usually rise sharply during the initial sampling periods but approach an asymtote as the species list for an area or habitat nears completion (Heyer et al. 1994).

Drift Fence Arrays

Drift fence arrays (Donnelly et al. 1998b) were sampled monthly in UP and oak/cabbage palm (upland forest, UF) hammocks in Pools A (hammock only) and C from February–March 1997 through September 1998. Replicate (three), cross-shaped arrays consisting of four, 15 meter long sections of aluminum flashing were partially sunk into the soil. Each array was separated by at least 20 meters. Each fence had one pit-fall trap (plastic 19 L bucket) at each end (n=4). In the middle of each side of the fence were either funnel traps, which were constructed of flexible window screen, or pit-fall traps. Pit-fall traps were buried in the soil so that the bucket lip was approximately 2.5 cm below the soil surface. Funnel traps were held against each fence with duct tape. Holes were drilled into the bottom of each pit-fall trap to provide drainage. A damp sponge was placed in each trap to prevent desiccation of captured animals and each trap was shaded with a tempered Masonite© board. Traps were opened for 24–96 continuous hours and checked daily. Species identity was recorded for each captured animal. Species richness, relative abundance, species diversity, and community evenness were calculated for each habitat within each pool. Community similarity was calculated for each habitat between pools.

Larval Amphibians

Larval amphibians within BLM and S.MCF habitats were sampled monthly from March 1997 through February 1999 with a 1 m^2 aluminum throwtrap. Larval amphibians within UP habitats were sampled monthly from April 1998 through March 1999. Five replicates (first nine months) or ten replicates (last 14 months) were collected from randomly selected locations within each habitat type on each sampling date. Sample locations were determined by traveling a randomly determined distance ($\leq$400 m) and direction (0–360°) from a randomly determined starting point within each habitat. Following trap placement, all vegetation within the trap was identified and counted. Water depth within the trap was recorded at each corner and at the center of the trap. All vegetation was removed and larvae were dip-netted from the trap. Dip-nets were equipped with 1 mm mesh netting. Dip-netting continued until no larvae were collected from ten consecutive dips. All larvae were preserved in 10% buffered formalin and stored for future identification.

In the laboratory, larval salamanders were identified using Altig and Ireland (1984) and Conant and Collins (1991). Ronald Altig (Mississippi State University) identified larval anurans. Body length and total length of all larvae were measured to the nearest 0.1 mm. Developmental stage of larval anurans was determined from Gosner (1960). Larval amphibian species richness was calculated for each habitat within each pool on each sampling date.

River Channel Turtle Community Structure

River channel turtles were sampled monthly from January 1997–September 1998 within remnant river channels and C-38 in Pools A and C. During the first seven months, 1 m diameter, 2.5 m long, single-throated hoopnets were used, but proved inefficient at capturing turtles. Consequently, 1.3 m diameter, 5 m long, double-throated hoopnets were used for the remainder of the study. Three hoopnets and three aluminum frame box traps were set in randomly selected locations in each of three remnant channels and in C-38 on each sampling date. Sample locations were selected by traveling at a constant boat speed (~ 1000 rpm) for a randomly determined time period through each remnant channel. Box traps were baited with sardines and placed along the deep-water edge of littoral vegetation, or within open water areas. Hoopnets were baited with salt pork or raw chicken and placed in deeper sections of each channel adjacent to emergent or floating vegetation. Nets contacted the substrate and were supported with 5 cm diameter PVC poles anchored to the substrate. Traps were set for a maximum of 96 hours during each month; however, time of deployment usually was less than 12 hours. Additionally, if time permitted, turtles were captured using a long-handled dip-net. Each turtle was identified to species, weighed, and marked with a unique coded tag or carapace mark (Cagle 1939), and released. Testudine species richness was calculated for each pool.

Casual Observations

Opportunistic observations of amphibians and reptiles also were recorded during this and other non-herpetological studies within the restoration project area from August 1995 through March 1999. When possible, amphibians and reptiles were captured and identified to the lowest possible taxonomic level, and released.

Results

A total of 48 taxa (Table 12-1; see Appendix 12-1A for common names of taxa), including 20 amphibians and 28 reptiles, were captured or encountered with all sampling methods. Nine taxa were encountered only once, and four are introduced species (Wilson and Porras 1983). Species richness was highest in Upland Hammock (20), followed by Broadleaf Marsh (19), Woody Shrub (17), Upland Herbaceous (14), and Wetland Forest (5). Species diversity and community evenness were low in all floodplain habitats in all pools (Table 12-2).

Visual Encounter Surveys

Visual encounter surveys detected 14 amphibian and reptile species within four floodplain habitats of the channelized Kissimmee River (Table 12-3). The number of species observed quickly accumulated in WF, with all species encountered within two months (Figure 12-1). Accumulation of species in UP was slower, with all species encountered within eight months. Species accumulated even more slowly in S.MCF and BLM, with all species encountered after 23 months (Figure 12-1).

Species richness was highest in S.MCF habitats with eight and eleven species present in Pools C and D, respectively (Table 12-2). Seven species were encountered in both pools; one species *(Rana grylio)* was found only in Pool C, and four species (*Gastrophryne carolinensis, Nerodia fasciata, Notopthalmus viridescens piaropicola,* and *Elaphe guttata*) were found only in Pool D (Table 12-3). Eight and six species were observed within BLM habitats of Pools A and C, respectively. Five species were present in both pools, three species (*Thamnophis sauritus, G. carolinensis*, and *Rana sphenocephala*) were found only in Pool A, and one species (*Agkistrodon piscivorous*) was found only in Pool C (Table 12-3). Three and five species were observed in WF habitats of Pools B and C, respectively. Three species were present in both pools, with two additional species (*Hyla femoralis* and *G. carolinensis*) found only in Pool C (Table 12-3). Pasture habitat within Pools A supported four species (19 total encounters) while UP habitat in Pool C supported one species (three total encounters) (Table 12-3).

Table 12-1. Herpetofauna captured or encountered within surveyed habitats in the lower Kissimmee basin. BLM = Broadleaf marsh, S.MCF = Woody Shrub, WF = Wetland Forest, UH = Upland Hammock, UP = Upland Herbaceous, KR = Kissimmee River, C38 = C-38 canal, and B = Building.

REPTILES	BLM	S.MCF	WF	UH	UP	KR	C38	B
Emydidae:								
Pseudemys floridana						X	X	
Pseudemys nelsoni						X	X	
Kinosternidae:								
Kinosternon baurii				X		X		
Kinosternon subrubrum						X		
Sternotherus odoratus						X		
Testudinidae:								
Gopherus polyphemus				X		X*		
Trionychidae:								
Apalone ferrox						X	X	
Alligatoridae:								
Alligator mississippiensis		X				X	X	
Anguidae:								
Ophisaurus attenuatus				X				
Gekkonidae:								
Hemidactylus sp.								X
Iguanidae:								
Anolis carolinensis	X	X	X	X	X			
Anolis sagrei								X
Scincidae:								
Eumeces inexpectatus					X			
Scincella lateralis				X	X			
Colobridae:								
Coluber constrictor				X	X			
Diadophis punctatus				X	X			
Drymarchon corais				X				
Elaphe guttata					X			
Elaphe obsoleta		X						X
Nerodia fasciata	X	X				X	X	
Opheodrys aestivus					X			
Regina alleni						X		
Seminatrix pygaea				X				
Storeria dekayi					X			
Thamnophis sirtalis				X				
Thamnophis sauritus	X	X		X				
Viperidae:								
Agkistrodon piscivorus	X	X				X		
Crotalus adamanteus							X	
AMPHIBIANS								
Amphiumidae:								
Amphiuma means	X					X		
Plethodontidae:								
Eurycea quadridigitata	X	X						
Salamandridae:								
Notopthalmus viridescens	X	X						
Sirenidae:								
Pseudobranchus a. axanthus	X							
Siren intermedia	X					X		
Siren lacertina	X	X				X		

* Observed swimming across river channel.

Table 12-1. Continued.

	BLM	S.MCF	WF	UH	UP	KR	C38	B
Bufonidae:								
Bufo terrestris				X				
Bufo quercicus				X				
Hylidae:								
Acris gryllus	X	X						
Hyla cinerea	X	X	X	X	X	X	X	
Hyla femoralis	X		X	X				
Hyla squirella	X	X		X				
Osteopilus septentrionalis							X	
Pseudacris nigrita	X			X				
Pseudacris ocularis	X	X		X				
Leptodactylidae:								
Eleutherodactylus planirostris				X				
Microhylidae:								
Gastrophryne carolinensis	X	X	X	X	X			
Ranidae:								
Rana catesbeiana	X							
Rana grylio	X	X						
Rana sphenocephala	X	X	X	X	X			

Table 12-2. Community structure indices calculated from total encounters and captures during visual encounter surveys and drift fence sampling within baseline floodplain habitats. BLM = Broadleaf Marsh, S.MCF = Woody Shrub, WF = Wetland Forest, and Up = Upland Herbaceous.

Visual Encounter Survey

	BLM		S.MCF		WF		UP	
Metric:	Pool A	Pool C	Pool C	Pool D	Pool B	Pool C	Pool A	Pool C
Species Richness (S')	8	6	8	11	3	5	4	1
Diversity (H')	0.43	1.25	1.19	1.19	0.21	0.31	1.11	0.00
Evenness(J')	0.21	0.70	0.57	0.50	0.19	0.19	0.80	0.00
Coefficient of Similarity	0.22		0.51		0.94		0.27	

Drift Fence Array

	UH		UP	
Metric:	Pool A	Pool C	Pool B	Pool C
Species Richness (S')	10	14	7	5
Diversity (H')	0.99	1.95	0.69	0.81
Evenness(J')	0.41	0.74	0.35	0.5
Coefficient of Similarity	0.42		0.45	

Hyla cinerea was the most frequently observed species in each habitat at all times during this study (Table 12-3), accounting for 52.4, 60.4, 84.0, and 94.4% of total numbers within UP, S.MCF, BLM, and WF, respectively. Only four other species, *Eurycea quadridigitata*, *Anolis carolinensis*, *Pseudacris ocularis*, and *R. sphenocephala* accounted for greater than 5% of total numbers within any habitat.

Species diversity was low in all habitats (Table 12-2). Values of community evenness were low in Pool C UP (0.0), Pool A BLM (0.21), and Pools B and C WF (0.19 and 0.19, respectively), moderate in Pools C and D S.MCF (0.57 and 0.50, respectively) and high in Pool C BLM (0.70) and Pool A UP (0.80). A coefficient of community similarity, which was calculated for each habitat, indicated that WF habitats are very similar between Control and Impact pools, S.MCF habitats are moderately similar, and BLM and UP habitats are dissimilar in species abundance (Table 12-2).

Table 12-3. Total herpetofaunal observations during 31 monthly visual encounter surveys (VES) in Broadleaf Marsh (BLM) and Woody Shrub (S.MCF), 15 monthly VES in Wet Forest (WF), and 11 monthly VES in Upland Herbaceous (UP) habitats.

	BLM		S.MCF		WF		UP	
	Pool A	Pool C	Pool C	Pool D	Pool B	Pool C	Pool A	Pool C
Taxon:								
Acris gryllus dorsalis	1	5	0	0	0	0	0	0
Agkistrodon piscivorous conti	0	3	2	4	0	0	0	0
Anolis carolinensis	50	40	54	77	9	13	2	0
Elaphe guttata	0	0	0	1	0	0	0	0
Eurycea quadridigitata	11	1	20	169	0	0	0	0
Gastrophryne carolinensis	5	0	0	3	0	2	7	0
Hyla cinerea	1006	72	163	480	318	294	9	3
Hyla femoralis	0	0	0	0	0	5	0	0
Nerodia fasciata	0	0	0	1	0	0	0	0
Notopthalmus viridescens piaropicola	0	0	0	1	0	0	0	0
Pseudacris ocularis	24	54	3	17	0	0	1	0
Rana grylio	0	0	1	0	0	0	0	0
Rana sphenocephala	1	0	30	27	6	1	0	0
Thamnophis sauritus	9	0	1	11	0	0	0	0
Totals	1107	181	274	791	333	315	19	3

Drift Fence Arrays

Drift fence arrays revealed a combined total of nine amphibian and reptile species in UP habitats of Pools B and C, and a combined total of 18 species in oak hammock (UF) habitats of Pools A and C (Table 12-4). The number of captured species quickly accumulated in UP habitats, with four of five species in Pool C captured within four months, and all species in Pool B captured within five months (Figure 12-2a). The number of species accumulated more slowly in (UF) hammock habitats, with all species in Pools A and C captured after 17 months (Figure 12-2b).

Gastrophryne carolinensis accounted for 84% and 77% of total numbers in Pool B and C UP, respectively. *Rana sphenocephala* was the only other taxon accounting for greater than 5% of total numbers in UP habitats. Three species, *Bufo quercicus, B. terrestris*, and *H. cinerea* only occurred in Pool B, while *Eumeces inexpectatus* and *Diadophis punctatus* were collected only in Pool C.

Gastrophryne carolinensis accounted for 73% and 35% of total numbers in Pool A and C hammocks. *Rana sphenocephala*, *Scincella lateralis*, and *E. inexpectatus* also accounted for greater than 5% of total numbers in oak hammocks (UF).

Species diversity was low in both habitats, ranging from 0.69 in Pool B UP, to 1.95 in Pool C hammock. Community evenness was variable, ranging from 0.35 in Pool B UP to 0.74 in Pool C hammock (Table 12-2). A coefficient of community similarity indicates moderately dissimilar communities in UP habitats of Pools B and C, and upland hammocks of Pools A and C (Table 12-2).

Larval Amphibians

Larval amphibians occurred sporadically in BLM, S.MCF, and UP habitats of Pools A, C, and D. When there was water on the floodplain in Pool A BLM, larvae were present seven of nine months in 1997–1998 and one of seven months in 1998–1999. When there was water on the floodplain in Pool C BLM, larvae were present six of nine months in 1997–1998 and one of seven months in 1998–1999. When there was water on the floodplain in Pool C S.MCF, larvae were present six of 12 months in 1997–1998 and three of nine months in 1998–1999. When there was water on the floodplain in Pool D S.MCF, larvae were present seven of 12 months in 1997–1998 and five of nine months in 1998–1999. One larval *Rana* *sphenocephala* was found in both Pool A and C UP habitat during one month, which was the only month that water was present during the 1998–1999 sampling period.

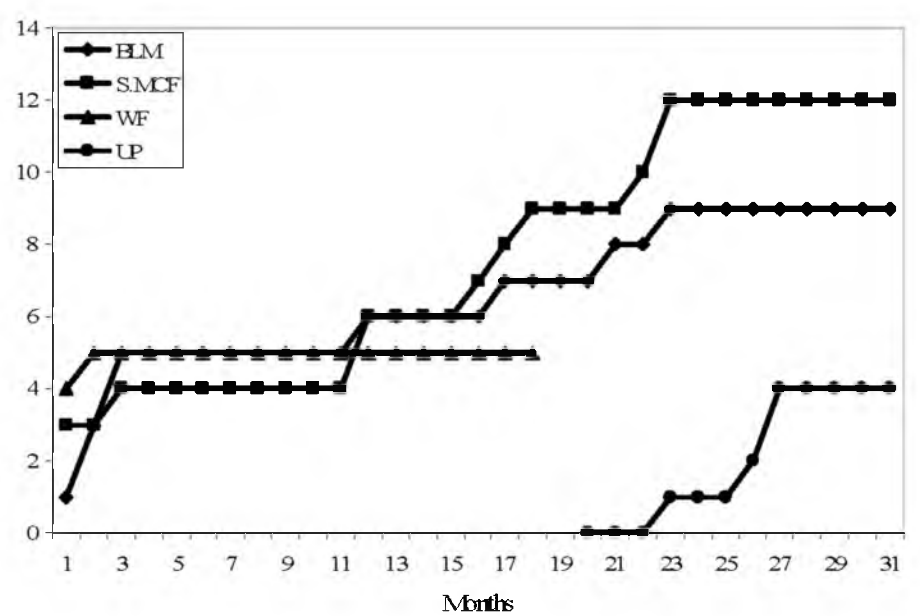


Figure 12-1. Species accumulation curves for floodplain visual encounter surveys. Accumulation curves show the cumulative number of species observed during successive sampling periods. BLM = Broadleaf Marsh, S.MCF = Woody Shrub, WF = Wetland Forest, and UP = Upland Herbaceous.

<u>Salamanders</u>

A total of five larval salamander taxa were collected from S.MCF, BLM, and UP habitats (Table 12-5). Species richness (4) and composition were identical between BLM habitat in Pools A and C. Species richness (4) was identical between S.MCF habitat in Pools C and D; however, these habitats had only three species in common. Pasture habitat in Pools A and C supported two and one larval salamander taxa, respectively.

Eurycea quadridigitata was collected most frequently and was most abundant in S.MCF habitats. Larvae first appeared in December 1997. Mean snout-vent (S-V) length increased from 11.0 mm to 19.1 and 19.3 mm, in Pools D and C respectively, between December 1997 and March 1998. Only adults were captured between May and December 1998, with larvae (mean S-V length = 13.0 and 15.7 mm in Pools D and C, respectively) reappearing in January 1999 in both pools. Larvae (mean S-V length = 16.4) also were collected in Pool C S.MCF in March 1999. *Eurycea quadridigitata* was less common in BLM habitats, although the seasonal pattern of reproduction was similar to S.MCF. *Eurycea quadridigitata* was not collected from UP habitats.

Larval *Siren lacertina* were collected from UP (Pool A), BLM (Pools A and C), and S.MCF (Pools C and D) habitats between December 1997 and April 1998. Other taxa rarely collected from any habitat included *Amphiuma means*, *Notopthalmus viridescens piaropicola*, *Siren intermedia intermedia*, and *Pseudobranchus axanthus axanthus*.

Table 12-4. Total herpetofaunal captures in drift fence, pit-fall trap, and funnel trap arrays. UH = Upland Hammock, UP = Upland Herbaceous.

	UH		UP	
Taxon:	Pool A	Pool C	Pool B	Pool C
Anolis carolinensis	1	1	0	0
Bufo quercicus	3	0	5	0
Bufo terrestris	2	0	2	0
Coluber constrictor	0	1	0	0
Diadophis punctatus	0	5	0	1
Drymarchon corais	0	1	0	0
Eleutherodactylus planirostris	9	0	0	0
Eumeces inexpectatus	0	8	0	2
Gastrophryne carolinensis	155	33	90	27
Hyla cinerea	5	5	2	0
Hyla femoralis	0	2	0	0
Kinosternon baurii	0	1	0	0
Ophisaurus attenuatus	0	1	0	0
Pseudacris nigrita verrucosa	0	0	1	0
Rana sphenocephala	29	19	6	4
Scincella lateralis	5	14	1	1
Seminatrix pygaea cyclas	1	0	0	0
Thamnophis sauritus sackenii	1	2	0	0
Thamnophis s. sirtalis	0	1	0	0
Total	211	94	107	35

Anurans

Ten larval anuran taxa were collected from floodplain habitats between April 1997 and February 1999 (Table 12-5). Overall larval anuran species richness was highest in BLM (10), followed by S.MCF (5), and UP (1); however, most taxa including *Acris gryllus*, *G. carolinensis*, *H. cinerea*, *H. femoralis*, *H. squirella*, *Pseudacris nigrita*, and *Rana catesbeiana* were captured infrequently.

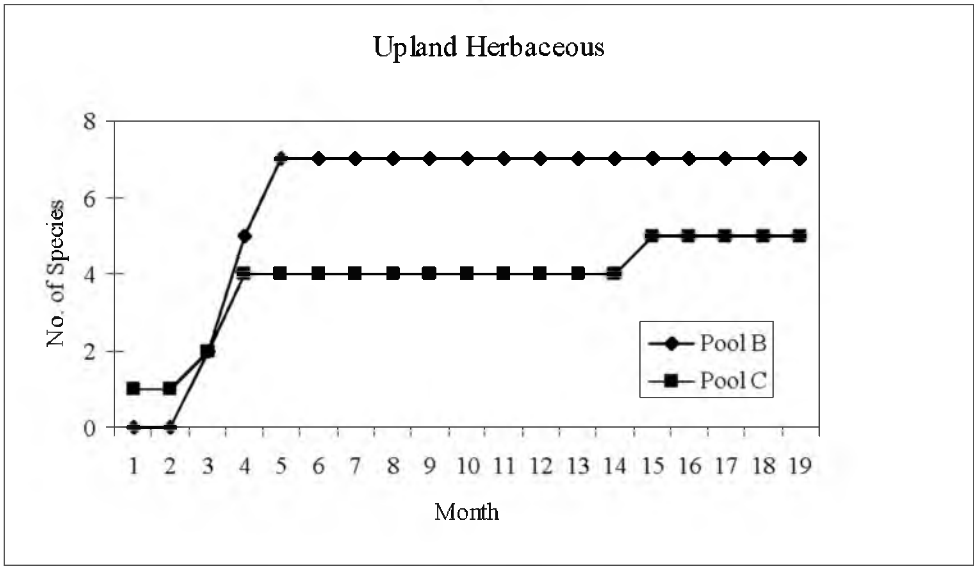
Pseudacris ocularis occurred monthly from October 1997 through January 1998 in Pool C BLM. Developmental stages of *P. ocularis* ranged from 27–36 in October to 39 in January. Larval *Rana sphenocephala* were captured on three dates between December 1997 and March 1998. Developmental stages ranged from 25 in December to 28–44 in March.

Within S.MCF habitats, mid-summer and spring patterns of development were apparent for *R. sphenocephala* and *R. grylio*, with larvae present in July–August (1997), December–April (1997–1998), and July–August (1998). Larvae collected in July–August (1997) were at developmental stage 25. Developmental stage of individuals collected in December–April ranged from 25–42. Individuals collected in July–August (1998) had attained a developmental stage of 25–26. Within UP habitats of Pools A and C, larval *R. sphenocephala* were each captured on one date. No other larval anurans were collected from UP habitats.

River Channel Turtle Community Structure

A total of 81 turtles (46 and 35 in Pools A and C, respectively), representing six taxa, were captured by hoopnet, box trap, or dip-net from remnant river channels and C-38 over a 20 month period beginning in January 1996. Captures occurred during approximately 6000 trap hours in Pool A and 6200 trap hours in Pool C. Seventy-nine percent of all turtles were captured in remnant river channels. In Pool A, *Pseudemys floridana peninsularis* accounted for 45.6% of total numbers and 50.1% of total mass, followed by *Pseudemys nelsoni* (43.5% and 35.8%, respectively), and *Apalone ferox* (8.7% and 14%, respectively). In Pool C, *P. floridana peninsularis* accounted for 34.3% of total numbers and 44.9% of total mass, followed by *P. nelsoni* (28.6% and 25%, respectively), and *A. ferox* (22.8% and 29.7%, respectively). Other less frequently captured turtles included *Stenotherus odoratus*, *Kinosternon bauri*, and *Kinosternon subrubrum steindachneri*.

A



B

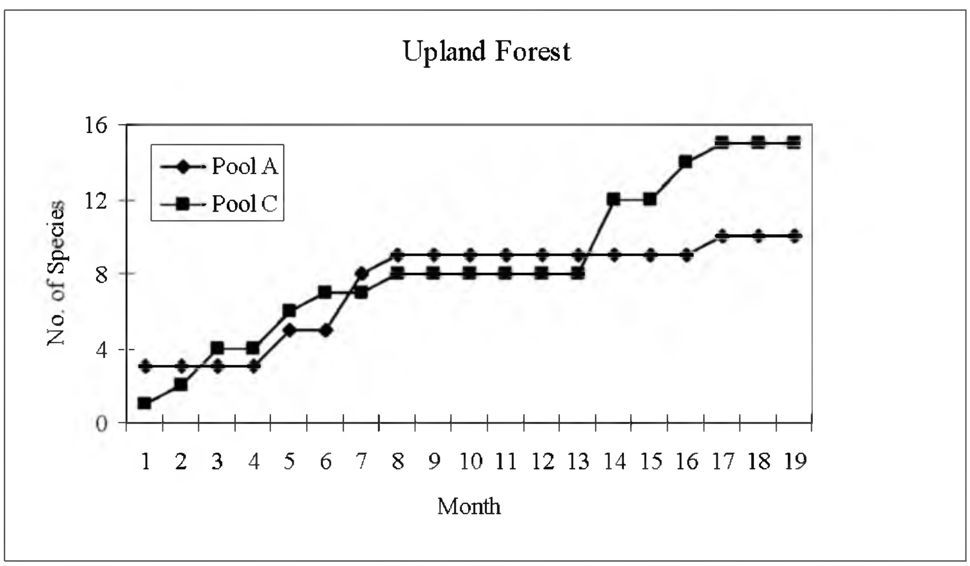


Figure 12-2. Species accumulation curves for drift fence, pit-fall, and funnel trap arrays.

Table 12-5. Habitat-specific occurrence of larval amphibians on the channelized Kissimmee River floodplain. BLM = Broadleaf Marsh, S.MCF = Woody Shrub, and UP = Upland Herbaceous.

	BLM		S.MCF		UP	
Taxa:	Pool A	Pool C	Pool C	Pool D	Pool A	Pool C
Salamanders:						
Eurycea quadridigitata	X	X	X	X		
Notopthalmus viridescens	X	X	X	X		
Pseudobranchus a. axanthus	X	X			X	
Siren i. intermedia			X			X
Siren lacertina	X	X	X	X	X	
Anurans:						
Acris gryllus dorsalis		X				
Gastrophryne carolinensis	X					
Hyla cinerea	X		X	X		
Hyla femoralis	X					
Hyla squirella	X	X	X	X		
Pseudacris nigrita verrucosa		X				
Pseudacris ocularis		X		X		
Rana catesbeiana	X					
Rana grylio		X	X	X		
Rana sphenocephala	X	X	X	X	X	X

Discussion

Visual Encounter Surveys

Visual encounter surveys can be an effective and economical means to determine species richness, species composition, and relative abundance of amphibians and reptiles within similar habitats. In addition, VES is an appropriate technique for both inventory and monitoring studies (Heyer et al. 1994, Pearman et al. 1995). The nine taxa observed in BLM and 12 taxa observed in S.MCF habitats over the baseline study period represent approximately 36% and 48% of all taxa likely to occur in natural wetlands of central Florida, respectively (Carr 1940, Franz et al. 2000). Although rare or cryptic taxa likely were overlooked during baseline surveys, data clearly indicate that remnant BLM and S.MCF habitats within the channelized Kissimmee River ecosystem support a depauperate wetland herpetofaunal community that is dominated in numbers by two or three species.

Visual encounter survey data from UP habitats (former broadleaf marsh) of the channelized river system indicate a severely impacted wetland herpetofaunal community. Although a total of 14 taxa were observed in UP habitats over the course of the baseline period (all methods), only seven wetland taxa were recorded, and only four (22 observations) were encountered by VES. These seven taxa represent 16% of taxa considered “characteristic” or “frequently occurring” in natural wetland habitats of central Florida (Carr 1940).

No historical or reference data on amphibian and reptile relative abundance, evenness, or diversity are available from the Kissimmee River ecosystem; therefore, no specific expectation for change in these metrics has been developed. However, based on expectations for hydrologic and habitat restoration, and knowledge of the occurrence of characteristic wetland herpetofaunal taxa within the lower Kissimmee basin (Franz et al. 2000), it is reasonable to hypothesize that species richness will increase within restored habitats (UP and BLM).

Drift Fence Arrays

Although some sampling bias is associated with drift fence sampling (Dodd 1991), drift fences combined with pitfall traps and funnel traps can be an effective technique for quantifying some animal populations, and usually capture some individuals of most species (Gibbons and Semlitsch 1981, Greenberg et al. 1994, Heyer et al. 1994). If one assumes that capture rates are similar between similar habitats, these data can be used to compare relative abundance of species among study areas (Heyer et al. 1994).

Pasture: Drift fence data clearly indicate that existing UP supports a depauperate herpetofaunal community dominated in numbers by one taxon and uncharacteristic of natural wetlands of central Florida (Carr 1940, Franz et al. 2000). Restoration of historic hydrologic patterns is expected to result in shifts in species richness, relative abundance, diversity, and evenness in UP habitats as they revert to BLM. Taxa characteristic of terrestrial habitats (e.g., *Bufo quercicus*, *B. terrestris*, *Eumeces inexpectatus*, and *Diadophis punctatus*) should emigrate to upland habitats while aquatic and semi-aquatic taxa colonize restored wetlands.

Upland Hammock: Taxa captured in pit-fall and funnel traps within oak hammocks (UF) represent approximately 33% of the species known to occur in upland hammocks of central Florida (Tennant 1997, Bartlett and Bartlett 1999). Although data indicate a somewhat depauperate community in upland hammocks, several factors may have contributed to low capture rates. Optimally, drift fences and pit-fall traps should be run continuously, with captured animals removed daily (Heyer et al. 1994). Available resources during the baseline period only allowed us to run traps for 24–96 hours per month. Extreme rainfall events associated with an El Niño Southern Oscillation Event (November 1997–March 1998) flooded hammocks and made sites inaccessible and pit-fall traps inoperable for approximately four months. The absence of most serpentines, which are often major components of the herpetofauna in upland habitats, may have been influenced by funnel trap design. Double-ended funnel traps used in this study were composed of lightweight window screen that had a tendency to collapse when taped to the drift fence. This likely prevented or deterred entrance by snakes, especially large-bodied individuals.

Because of the potential biases cited above, rare taxa likely were overlooked; however, taxa considered common and conspicuous in upland habitats within the Florida peninsula (Carr 1940) including *Elaphe guttata guttata*, *Elaphe obsoleta quadrivittata*, *Masticophis flagellum flagellum*, *Micrurus fulvius fulvius*, *Terrapene carolina bauri*, *Ophisaurus ventralis*, *Cnemidophorus sexlineatus*, *Scaphiopus h. holbrookii*, *Hyla gratiosa*, and *Hyla squirella* were never captured in upland habitats of the channelized Kissimmee River, indicating that channelization, or post-channelization impacts to uplands, may have altered population numbers and/or spatial patterns of distribution for some taxa.

Although no specific expectation for restoration of upland herpetofaunal communities has been developed, post-construction changes in community composition are likely to occur. We suggest, if sufficient resources are available, that these populations be monitored biannually (wet and dry season) to determine seasonal patterns of richness and abundance. Because seven taxa (~15% of the total) were unique to upland hammocks, these data are important in developing an accurate herpetofaunal inventory, which may serve as a useful indicator of biodiversity within the lower Kissimmee basin. Because post-construction data will not be directly compared to baseline data, additional sampling techniques including coverboards and PVC pipes should be incorporated into the sampling design to potentially encounter cryptic species. We also recommend that drift fences with pit-fall traps and rigid funnel traps be run for a minimum of 30 consecutive days during each season.

Larval Amphibians

Salamanders: Six salamander species are known to occur within the lower Kissimmee River basin (Table 12-2), and may be seasonally abundant in suitable habitats (Bartlett and Bartlett 1999). The dwarf salamander, *Eurycea quadridigitata*, was the most abundant salamander encountered during the baseline sample period. Increased visual observations of adult *E. quadridigitata* between August 1997 and February 1998 within S.MCF preceded a sharp increase in the occurrence of larval *E. quadridigitata* from January through April 1998 and again in January 1999. This correlation between increased adult and larval abundance corresponds well with breeding migrations and reproduction of *E. quadridigitata* on the upper Coastal Plain of South Carolina and in Alabama (McMillan and Semlitsch 1980, Trauth 1983).

Less frequently encountered taxa including *Amphiuma means*, *Notopthalmus v. piaropicola*, *Siren i.intermedia*, *S. lacertina*, and *Pseudobranchus a. axanthus* are likely more common in the Kissimmee River ecosystem than the results of this survey indicate. All are typical of shallow, heavily vegetated, soft-bottom habitats including littoral margins of remnant channels and long hydroperiod wetlands (e.g., S.MCF); however, they are often undetected due to their nocturnal and cryptic behavior (Bartlett and Bartlett 1999). Little is known about the reproductive habits of these taxa; however, it is likely that they will persist, reproduce, and become more obvious in the restored system as long-term floodplain hydroperiods and suitable habitat are restored.

Anurans: In central Florida, most anurans can breed during any month (Conant and Collins 1991). Given the prolonged floodplain inundation frequencies within the pre-channelized system, it is likely that anuran reproduction and larval recruitment occurred during most of the year. Although larval amphibians likely were present year-round, community structure characteristics (e.g., species richness and relative abundance) within pre-channelization marshes likely were heavily influenced by the presence of avian predators during periods of low water, and piscine predators during periods of high water.

Within the channelized Kissimmee River system, the availability of suitable habitat likely is the critical factor influencing reproduction by adult anurans (and salamanders), and the development and recruitment of larvae. Channelization eliminated seasonal, long-term floodplain inundation frequencies and fluctuating stage, thereby eliminating much of the historic breeding habitat for anurans. Under channelized conditions, floodplain habitats are often only inundated during the rainy season (typically June–September) with hydroperiods varying from days to months, depending on frequency and amount of rainfall. During this study, atypical floodplain inundation patterns resulted from rainfall associated with the 1997–1998 El Niño Southern Oscillation event. During this period, larvae from at least five taxa were collected from floodplain habitats, with several taxa collected consistently over a seven-month period. The presence of at least one larval anuran taxa in ten of the 16 months (~62%) in which water was present on the floodplain, indicate the potential for extended anuran reproduction.

River Channel Turtle Community Structure

Turtles are common and often conspicuous inhabitants of slow-flowing rivers and marshes of the southeastern Coastal Plain of the United States (Meylan et al. 1992), and often represent the majority of vertebrate biomass in aquatic systems (Iverson 1982, Congdon et al. 1986). Predatory fish, large wading birds (Ernst et al. 1994), and raptors (Cagle 1950, Beissinger 1990, Walley 1993, Means and Harvey 1999) occasionally consume hatchling and juvenile turtles, whereas adult turtles have few natural enemies except *Alligator mississippiensis* (Valentine et al. 1972, Delany and Abercrombie 1986).

Turtles were observed along river channel margins during most times of the year, and were frequently observed basking on floating vegetation and small woody debris. A total of six taxa (Table 12-2) were captured during this study. *Chelydra serpentina osceola* and *Deirochelys reticularia chrysea* were not observed or captured within the lower Kissimmee basin although their presence is likely.

All turtle species present in the Kissimmee River ecosystem are typical of large river systems of the southeastern United States (Ernst et al. 1994) and are expected to remain a highly visible component of the restored system. Although there is no intent to measure shifts in testudine community structure following restoration, opportunistic observations of river channel turtles will be recorded. Specific attention will be given to restored floodplain habitats that should become primary sites for foraging and reproduction by aquatic turtles. Additionally, all turtles observed in upland habitats will be recorded to determine seasonal shifts in habitat use.

REFERENCE CONDITIONS

Methods

Amphibian and Reptile Community Structure and Amphibian Reproduction

Samples collected during the baseline study period from remnant but altered BLM in Pool C provide some insight into wetland herpetofauna taxa richness and amphibian reproduction in pre-channelization BLM habitats.

In order to locate additional potential sources of reference conditions for amphibian and reptile community structure and patterns of amphibian reproduction within BLM habitats, a thorough literature search was conducted using the State Library of Florida Online Computer Library Center FirstSearch service.

Results

Baseline Surveys

Fourteen amphibian and reptile taxa considered characteristic or frequent inhabitants of permanent wetlands of central Florida were captured or observed in remnant marshes of Pool A and C during the baseline study period (Table 12-6) (Carr 1940, Franz et al. 2000). These taxa represent approximately 56% of taxa most likely to occur in broadleaf marsh habitats in central Florida, and are expected to occur in restored marshes within the Kissimmee River ecosystem.

Table 12-6. Potential wetland taxa for indicating restoration of amphibian and reptile community structure in reestablished broadleaf marsh habitats of the Kissimmee River ecosystem. These taxa occur in natural marshes of the Avon Park Bombing Range (APBR) and are considered characteristic or frequent inhabitants of natural marshes of central Florida (Franz et al. 2000). Taxa that are underlined were collected from remnant, but altered, Broadleaf Marsh (BLM) in Pools A and C.

Amphibians	Reptiles
	Agkistrodon piscivorus conanti
Acris gryllus dorsalis *	*Alligator mississippiensis*
Amphiuma means	*Anolis carolinensis* *
Eurycea quadridigitata	*Farancia abacura abacura*
Hyla cinerea	*Nerodia floridana*
Hyla squirella *	*Pseudemys floridana penninsularis*
Notophthalmus viridescens piaropicola	*Pseudemys nelsoni*
Pseudacris nigrita verrucosa	*Regina alleni*
Pseudacris ocularis	*Seminatrix pygaea cyclas*
Rana grylio	*Sistrurus miliarius barbouri*
Rana sphenocephala spp.	*Storera dekayi victa*
Siren intermedia intermedia	*Thamnophis sauritus sackenii*
Siren lacertina	*Thamnophis sirtalis sirtalis*
	Apalone ferox

* Although these taxa are not considered characteristic or frequent inhabitants of APBR marshes, they do occur in remnant marshes of the Kissimmee River and are likely to occur in restored BLM.

Reference Site

Pre-channelization data on herpetofaunal community structure from the Kissimmee River ecosystem are limited. Our primary source of information on herpetofaunal species richness of pre-channelization Kissimmee River marshes is herpetofaunal surveys of permanent wetlands of APBR. The APBR borders the Kissimmee River in Pools A and B (Highlands and Polk Counties) and contains over 54,000 acres of natural wetlands, of which less than 5% have been directly disturbed or impacted. Franz et al. (2000) surveyed the APBR for sensitive herpetofaunal species between October 1996 and May 1998. Data from these surveys indicate that 24 wetland amphibian and reptile taxa are characteristic or frequently occur in permanent wetlands of the APBR (Table 12-6). Because these relatively undisturbed habitats are directly adjacent to the Kissimmee River, it is likely that these taxa also occurred in pre-channelization marshes of the Kissimmee River (Table 12-6). Additionally, Carr (1940) presents a comprehensive review of amphibian and reptile habitat distributions throughout Florida, and lists species that are characteristic or frequently occur within each habitat. Based on this review, 25 amphibian and reptile taxa likely inhabited BLM habitats of the pre-channelized Kissimmee River during some portion of their lifetime. Although reference conditions are solely derived from Franz et al. (2000), information from Carr (1940) provides supporting information on herpetofaunal taxa likely to occur in post-channelization marshes. Taxa that occur in marshes of the APBR were judged likely to occur in pre-channelization marshes of the Kissimmee River, and are expected to occur in restored floodplain marshes. Table 12-6 lists taxa that are characteristic or frequently occur in permanent wetlands of APBR (Franz et al. 2000).

Larval Anurans

No data on temporal patterns of amphibian reproduction within the pre-channelized Kissimmee River exist; however, baseline data collected from remnant BLM in Pools A and C provide some indication of the possible temporal patterns of reproduction by amphibians in the pre-channelized system. Data indicate the presence of larval amphibians in seven of nine months (78%) in 1997–1998 and one of seven months (14%) in 1998–1999 when water was present on the floodplain in Pool A remnant BLM (Table 12-7). Larval amphibians were present in six of nine months (67%) in 1997–98 and one of seven months (14%) in 1998–1999 when water was present on the floodplain in Pool C remnant BLM (Table 12-7). Overall, larval amphibians were present in 11 of 16 months (69%) when water was present on the floodplain in either Pool A or C (Table 12-7).

Discussion

Amphibian and Reptile Community Structure

Based on reference condition data, it is possible to estimate species richness of amphibian and reptile taxa inhabiting pre-channelization Kissimmee River floodplain marshes. Although data do not provide insights into temporal patterns of abundance or diversity, they do provide enough information to develop an expectation for the occurrence of amphibians and reptiles in restored (currently UP) floodplain marshes of the Kissimmee River. This expectation is based on reestablishing a full range of hydrologic variation within floodplain UP habitats, including floodplain hydroperiod and variable depth patterns. Restoration of pre-channelization hydrologic patterns will be the impetus for reestablishing BLM vegetation and an aquatic invertebrate community necessary for colonization and persistence of amphibians and reptiles. Adult colonists likely will emigrate from existing wetland depressions within the UP, or from the river's littoral zone.

Larval Anurans

Specific data on anuran reproduction and larval development in pre-channelization marshes of the Kissimmee River do not exist. However, this does not preclude the development of an expectation for temporal patterns of anuran reproduction in restored BLM. Several studies (Blair 1961, Brooks 1980, Diaz-Paniagua 1988) have documented the reproductive phenology of multiple-anuran species assemblages over several years. In each of these studies, reproduction by individual species was partitioned over many months, often encompassing spring, summer, fall and winter. In these cases, larvae of at least one species were present during the entire year. Given the subtropical climate and prolonged floodplain inundation frequencies within the pre-channelized Kissimmee River system, it is likely that anuran reproduction and larval recruitment occurred during most of the year. Table 12-8 presents the known breeding periods of anurans likely to occur in pre-channelization marshes of the Kissimmee River.

GENERAL DISCUSSION

Baseline Conditions

The herpetofaunal community of the lower Kissimmee River basin is moderately species rich (48); however, numerous taxa characteristic of natural wetlands and upland hammocks were rare or not recorded during the baseline period. Dalrymple (1988) and Meshaka (1997) encountered 51 and 53 species of amphibians and reptiles from four habitats on Long Pine Key, Everglades National Park, and a five-year study of seven habitats at a disturbed wetland site in central Florida, respectively. Enge and Wood (1998) captured or identified 64 taxa (25 amphibians and 39 reptiles) from 12 habitats in the Big Bend Wildlife Management Area, Taylor County, Florida, while Franz et al. (2000) identified 68 taxa from wetland and upland sites on the Avon Park Air Force Range, Highlands and Polk Counties, Florida.

Hydrology and habitat quality are two critical factors influencing species composition, distribution, and reproduction in herpetofaunal communities (Skelly 1997, Adams 1999, Bodie and Semlitsch 2000). Loss of floodplain habitat combined with irregular and unpredictable hydroperiods following channelization, likely has altered patterns of abundance, distribution, and reproduction for many taxa within the channelized system. However, without historical records, it is difficult to reach any conclusions regarding shifts in species composition of amphibian and reptile species from the Kissimmee basin following channelization.

Table 12-7. Seasonal distribution of larval amphibians in altered broadleaf marsh and pasture habitats of the Kissimmee River. Months underlined indicate months when water was present on the floodplain. BLM = Broadleaf Marsh, UP = Upland Herbaceous.

Pool A BLM																							
					1997												1998						
	A	M	<u>J</u>	J	<u>A</u>	<u>S</u>	<u>O</u>	<u>N</u>	<u>D</u>		<u>J</u>	<u>F</u>	<u>M</u>	<u>A</u>	M	J	<u>J</u>	<u>A</u>	<u>S</u>	<u>O</u>	<u>N</u>	D	<u>J</u>
<u>Anurans</u>																							
Gastrophryne carolinensis			X																				
Hyla cinerea						X																	
Hyla femoralis													X										
Hyla squirella					X																		
Hylidae			X																				
Rana catesbeiana			X																				
Rana sphenocephala									X			X											
<u>Salamanders</u>																							
Eurycea quadridigitata									X		X	X	X										X

Pool C BLM																							
					1997												1998						
	A	M	<u>J</u>	J	<u>A</u>	<u>S</u>	<u>O</u>	<u>N</u>	<u>D</u>		<u>J</u>	<u>F</u>	<u>M</u>	<u>A</u>	M	J	<u>J</u>	<u>A</u>	<u>S</u>	<u>O</u>	<u>N</u>	D	<u>J</u>
<u>Anurans</u>																							
Acris gryllus																		X					
Hyla cinerea													X										
Hyla femoralis																		X					
Hylidae													X										
Pseudacris nigrita							X																
Pseudacris ocularis							X	X	X	X													
Rana grylio							X						X										
Rana sphenocephala									X			X	X										

Pool A UP												
						1998						
	<u>M</u>	A	M	J	J	A	S	O	N	D	J	F
<u>Anurans</u>												
Rana sphenocephala	X											

Pool C UP												
						1998						
	<u>M</u>	A	M	J	J	A	S	O	N	D	J	F
<u>Anurans</u>												
Rana sphenocephala	X											

Taxa collected or observed during the study (excluding introduced species) represent approximately 65% of native taxa likely to occur within wetland and upland habitats of the lower Kissimmee basin. The rarity or absence of characteristic and common taxa from floodplain habitats suggests that channelization and loss of habitat contributed to the decline or temporary elimination of some taxa.

Reference Conditions

Amphibian and Reptile Community Structure

Pre-channelization data from the lower Kissimmee River basin would provide the best reference conditions for assessing amphibian and reptile responses to Kissimmee River restoration. However, in the absence of pre-channelization data, records of amphibian and reptile distributions in natural wetlands of the APBR provide reasonable reference conditions for comparing pre- and post-restoration herpetofaunal communities (Franz et al. 2000). Additionally, historical records on the distribution and habitat preferences of amphibians and reptiles of central Florida provide additional information on potential taxa that may occur following restoration (Carr 1940).

S.MCF habitats will be excluded from initial post-construction studies. Although herpetofaunal community structure characteristics in S.MCF habitats are eventually expected to change as BLM vegetation becomes reestablished, this change is not expected for several (three–five or more) years. Once S.MCF habitats revert to BLM, post-construction sampling will commence.

Table 12-8. Florida breeding periods of amphibian species likely to colonize existing Broadleaf Marsh, Woody Shrub, and restored Broadleaf Marsh habitats currently characterized as pasture. Breeding periods are from Mount (1975) and Conant and Collins (1991).

Indicator Species	Spring	Summer	Autumn	Winter
Anurans:				
Acris gryllus dorsalis	X	X	X	X
Gastrophryne carolinensis	X	X	X	
Hyla cinerea	X	X	X	
*Hyla femoralis**	X	X	X	
*Hyla gratiosa**	X	X		
*Hyla squirella**	X	X	X	
Pseudacris nigrita verrucosa	X	X	X	X
Pseudacris ocularis	X	X	X	
Rana catesbeiana	X	X	X	
Rana grylio	X	X	X	X
Rana sphenocephala	X	X	X	X
Salamanders:				
Amphiuma means	X			
Eurycea quadridigitata	X		X	X
*Pseudobranchus a. axanthus***				
Siren i.intermedia	X			
Siren lacertina	X			

* Likely to occur near upland edge of floodplain.
** Breeding habits unknown.

Larval Amphibians

Reference conditions for the presence of larval amphibians in restored floodplain marshes are less rigorously defined. However, assuming that adult amphibians colonize restored marshes, there are no known factors that should prohibit adults from initiating breeding activities. Because of the potential for temporal partitioning of breeding among a multi-species assemblage, it is likely that larval amphibians will be present at least seven months each year.

The presence of larval amphibians in restored BLM will be determined from replicate, monthly throwtrap samples collected in the same BLM and UP habitats sampled during the baseline period in Pools A and C. Sampling of larval amphibians will commence approximately three years after reestablishing pre-channelization floodplain hydroperiods, and continue for a period of three years.

Comparisons and Expectations

Channelization of the Kissimmee River and subsequent draining of wetlands, severely impacted amphibian and reptile community structure and temporal patterns of anuran reproduction in floodplain habitats. Species richness in UP habitats (formerly BLM) is approximately five times lower than natural marshes of the APBR (Franz et al. 2000) and natural marshes of central Florida, as described by Carr

(1940). Periods of anuran reproduction in the channelized system appear to be governed by floodplain inundation patterns, which are highly unpredictable. Based on comparisons of baseline and reference data for community structure characteristics and patterns of amphibian reproduction, restoration of the Kissimmee River ecosystem should result in increased amphibian and reptile species richness ($\geq$24) in restored BLM, and near year-round reproduction by amphibians. The following expectations have been developed from baseline data and best available reference data.

<u>Expectation:</u> Number of amphibians and reptiles using the floodplain

Herpetofaunal taxa were rare in sampled UP habitats, all of which were BLM habitat prior to channelization. Five taxa (22 individuals) were observed over the 12 month sample period in Pool A and C, and represent approximately 20% of all wetland taxa considered characteristic or frequently occurring in BLM throughout central Florida. Additionally, these five taxa account for approximately 21% of the wetland taxa occurring in natural marshes of the APBR (Figure 12-3). Restoration of pre-channelization hydrologic characteristics within the lower Kissimmee basin will be the impetus for reestablishing BLM communities in areas that currently exist as UP. Our expectation for restoration of amphibian and reptile community structure in restored BLM, which currently exist as UP, predict the presence of at least 24 taxa. A community composed of 24 taxa represents nearly all taxa that are throughout undisturbed wetlands of central Florida (Carr 1940, Franz et al. 2000), and a >400% increase over the number of wetland taxa currently found in UP habitats of Pool C. This expectation does not imply the continuous presence of 24 taxa; rather, 24 taxa will be observed cumulatively within these habitats three years after restoration of pre-channelization hydrologic characteristics (Koebel 2005a).

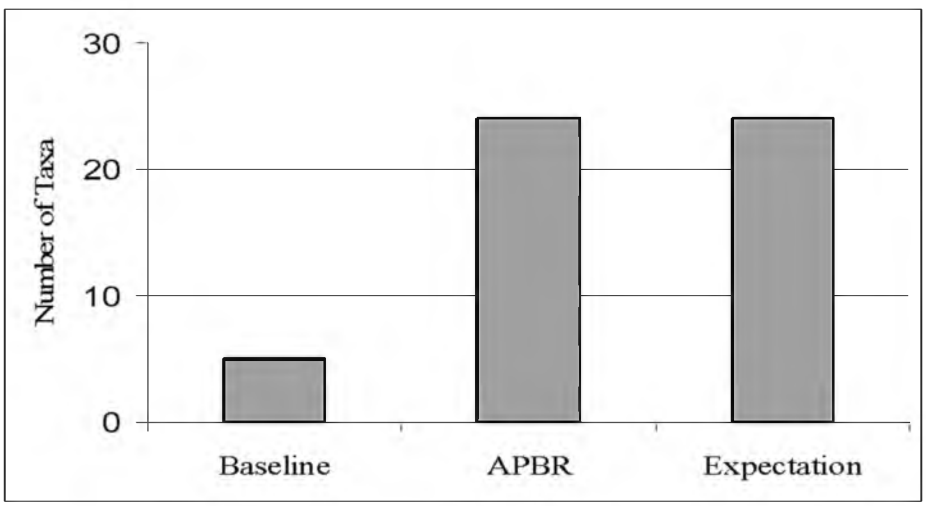


Figure 12-3. Number of taxa occurring in pasture habitats during the baseline period and number of taxa expected to occur in restored BLM following restoration. The expectation is based on the number of characteristic or frequently occurring wetland taxa in natural marshes of Avon Park Bombing Range, Highlands and Polk Counties, Florida (Franze et al. 2000).

Monthly visual encounter surveys, larval amphibian sampling, and casual observations (aural and visual) will commence in the same BLM and UP locations sampled during the baseline period within one year of reestablishing pre-channelization floodplain hydroperiods, and continue for a period of three years. Visual encounter surveys repeated at regular intervals (monthly) over several years and a variety of environmental conditions likely will detect a large percentage of total taxa present. We anticipate that the use of multiple sampling techniques will be sufficient to document changes in species composition, species richness, and relative abundance within restored wetlands, and that these changes will be useful indicators of restoration success.

<u>Expectation:</u> Use of floodplain for amphibian reproduction and larval development

Adult amphibians should respond quickly to restored hydrologic patterns and increased plant community heterogeneity within restored marshes, and are likely to begin breeding shortly after colonizing. Because amphibian breeding activity in subtropical climates may occur during most of the year (Stebbins and Cohen 1995), larval amphibians are likely to be present year-round. However, because reference conditions documenting amphibian breeding periods are not available for the pre-channelized Kissimmee River, our expectation for the presence of larval amphibians is based on the occurrence of larval amphibians in remnant but altered BLM habitat during the baseline study period. During this period, when water was present on the floodplain, larval amphibians were collected a maximum of seven months during either year in Pool A or C remnant marsh. Assuming that a restored marsh will support larval amphibians at least as often as remnant marsh, a conservative estimate predicts the presence of larval amphibians for at least seven of 12 months in restored marshes in Pool C (Koebel 2005b).

The presence of larval amphibians will be determined from replicate throwtrap samples collected in the same BLM and UP habitats sampled during the baseline period in Pools A and C. Table 7-8 lists amphibians likely to use floodplain habitats for reproduction within the restored system, and typical breeding periods.

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CHAPTER 13

STATUS OF FISH ASSEMBLAGES OF THE KISSIMMEE RIVER PRIOR TO RESTORATION: BASELINE CONDITIONS AND EXPECTATIONS FOR RESTORATION

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ABSTRACT: Fish surveys addressing multiple metrics were conducted within severely altered habitats of the Kissimmee River following channelization. Attributes of baseline fish assemblages were compared to pre-channelization assemblages, where data were available, to determine if channelization-related impacts have occurred. Comparisons indicate that floodplain and river channel fish assemblage structure has shifted and that respective assemblages are dominated by taxa or guilds more characteristic of lentic and/or degraded conditions. Fishing effort for largemouth bass *Micropterus salmoides* has decreased by approximately 30% and catch rates for sport fishes are varied. Expectations for restoration-related change in specific fish assemblage metrics were developed to evaluate restoration success. Floodplain fish assemblages are characterized by guild according to macrohabitat use and, based on reference data, are expected to be dominated by off-channel dependent taxa in the restored system. Mean annual density of small fishes (< 10 cm total length) in floodplain habitats is expected to be greater than 18 fish/m^2. The expectation for river channel fish assemblages describes changes in the mean annual relative abundance of specific taxa and families and predicts that less than 1% bowfin *Amia calva* and 3% Florida gar *Lepisosteus platyrhincus*, greater than 16% redbreast sunfish *Lepomis auritus*, and greater than 58% centrarchids will be present in the post-restoration assemblage.

INTRODUCTION

Fishes are ecologically important components of large river-floodplain ecosystems (Welcomme 1979). Fish taxa representing a range of trophic categories (herbivore, piscivore, omnivore, invertivore, planktivore, detritivore) consume foods from aquatic and terrestrial environments (Karr et al. 1986) and serve as a critical link in the energy pathway between primary producers and higher trophic level consumers, including amphibians, reptiles, and birds (Karr et al. 1991, Gerking 1994). Fishes are used often as bioassays for contaminants within aquatic environments (Sprague 1973, USEPA 1977). Because freshwater fishes are relatively long-lived (Carlander 1977) and can travel considerable distances within their watershed (Gent et al. 1995, Furse et al. 1996), they integrate aspects of aquatic ecosystems across broad temporal and spatial scales (Karr et al. 1986). Fishes are therefore useful indicators of aquatic ecosystem health or integrity (Karr et al. 1986, Ohio EPA 1987, Oberdorf and Hughes 1992, Gammon and

Simon 2000). For these reasons, fishes were chosen as a biotic component of the Kissimmee River Restoration Evaluation Program.

Channelization of the Kissimmee River through the construction of the C-38 canal in 1962-1971 dramatically altered the hydrology of the system and resulted in drainage or obliteration of approximately 8,000 ha of floodplain wetlands, elimination of instream and overbank flow, and isolation of the river from its floodplain (Koebel 1995). These hydrologic alterations propagated changes in physical, chemical, functional, and biological aspects of the ecosystem that influence fish assemblages. These characteristics include depressed levels of dissolved oxygen, re-structuring of the food web, and habitat loss or degradation (Welcomme 1979, Junk et al. 1989, Gladden and Smock 1990).

Restoration of pre-channelization hydrologic characteristics through the Kissimmee River Restoration Project is expected to restore the physical habitat template, as well as reestablish chemical and functional attributes of the ecosystem that influence fish assemblages. Reestablishment of the pre-channelization river channel/floodplain linkage is critical for restoring food web pathways through transport of fish prey and organic inputs to the river channel and for providing essential nesting, nursery and foraging habitat. Reintroduction of flow is projected to alleviate seasonally low levels of dissolved oxygen and increase heterogeneity of in-channel microhabitat. Fish assemblages are expected to respond favorably to restored conditions and should approximate pre-channelization conditions or those of natural systems within the region (Trexeler 1995).

Objectives

The objectives of this study are: (1) to assess the baseline condition of floodplain and river channel fish assemblage structure, fish reproductive effort and larval fish assemblage structure, fish diets from nine taxa representing a range of trophic levels, angling effort and catch rate for specific sport fish taxa, largemouth bass and bluegill movement patterns, and methylmercury bioaccumulation in largemouth bass, (2) to estimate the reference condition of floodplain and river channel fish assemblage structure and angling effort and catch rate, (3) to quantify impacts of channelization by comparison of estimated pre-channelization and baseline conditions for floodplain and river channel fish assemblage structure and angling effort and catch rate, and (4) define and discuss specific expectations for selected attributes of floodplain and river channel fish assemblage structure.

STUDIES WITH ASSOCIATED RESTORATION EXPECTATIONS

I. FLOODPLAIN FISH ASSEMBLAGE STRUCTURE

Baseline Condition

Methods

Floodplain fishes were sampled with a 1-m^3 aluminum throw trap, which provides accurate estimates of density, size structure, and relative abundance of small-fish (<10 cm total length) populations within heavily vegetated habitats (Kushlan 1981, Chick et al. 1992, Jordan et al. 1997). Sampling was conducted quarterly between August 1996 and April 1997, and monthly from August 1997 through January 1999. Two habitat units (one Control unit and one Impact unit) were sampled in three vegetation types each, which included Broadleaf Marsh (BLM, Bousquin and Carnal 2005), Woody Shrub (*Myrica cerifera* Floating Mat Shrubland Bcode group; S.MCF), and Pasture (Upland Herbaceous Bcode group; UP) (Figure 13-1). Ten replicate samples were collected in randomly selected locations in each habitat on each sampling date. Following trap placement, all vegetation within the trap was removed.

Water depth was recorded at each corner and at the center of the trap. All vegetation within the trap was removed, and fishes were removed with a dip-net (1-mm mesh). Dip-netting continued until no fish were collected in 10 consecutive attempts. All fishes were preserved in 10% buffered formalin. In the laboratory, all fishes were identified to species, counted, and measured to the nearest mm (total length).

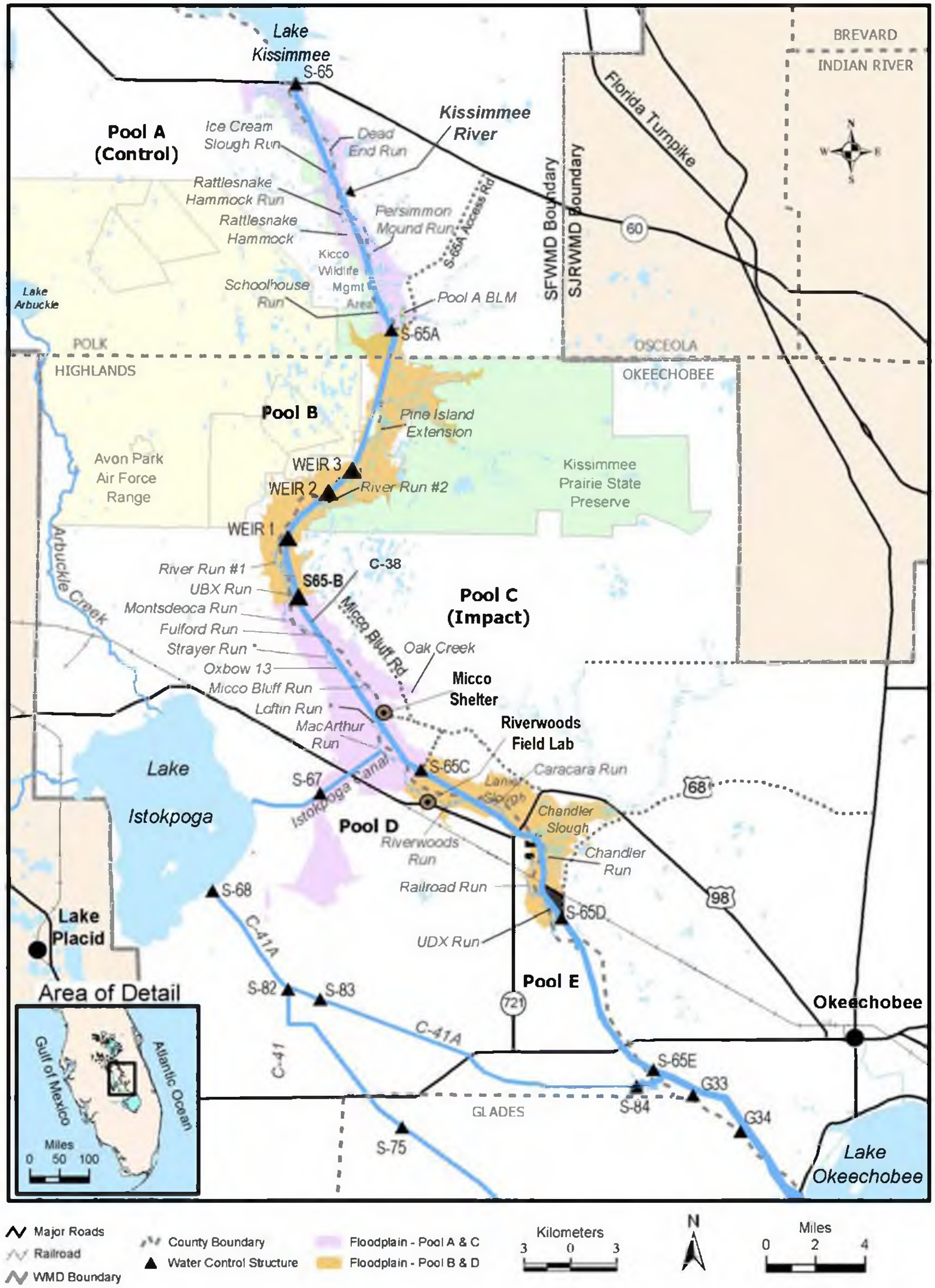


Figure 13-1. Map of the Kissimmee River showing the locations of pools and remnant river runs where fish were sampled.

Two metrics were used to develop restoration expectations for floodplain fish assemblages — relative abundance according to macrohabitat guild and fish density (number of fish/m^2). The macrohabitat guild structure developed by Bain (1992) used to assess guild relative abundance was augmented to include two new guild categories based on fish dependence on off-channel habitats (Figure 13-2).

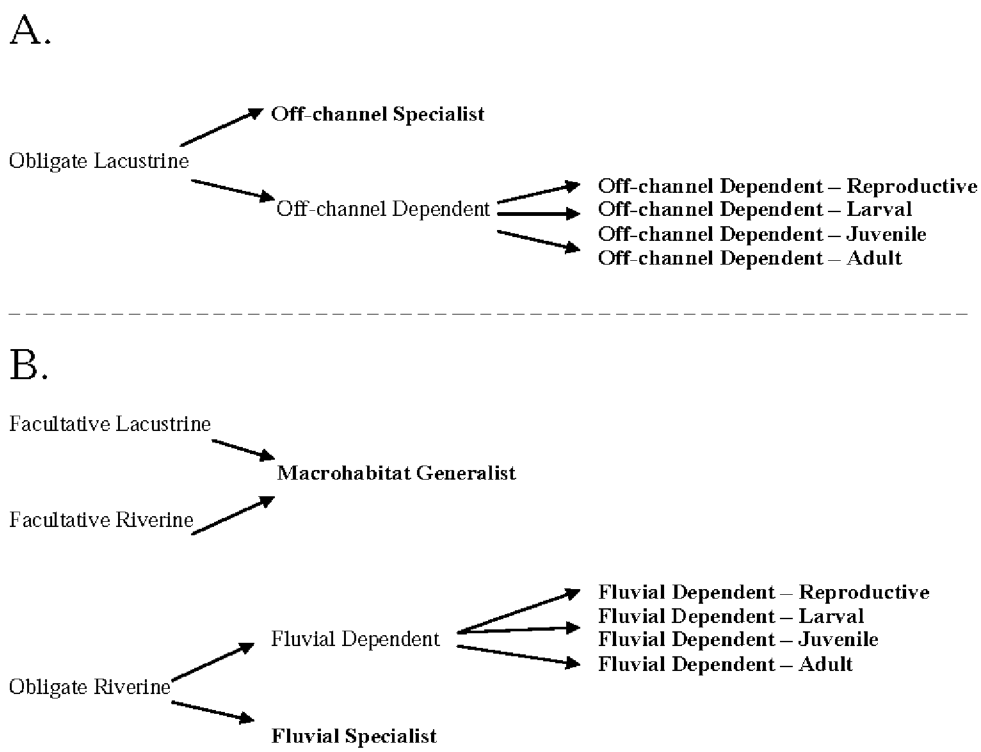


Figure 13-2. Schematic representation of modified macrohabitat guild structure derived by Bain (1992). (A) New guild categories based on dependence of associated taxa on off-channel habitat. The new category termed off-channel dependent includes species that are found in a variety of habitats, but require access or use of off-channel habitats, or are limited to nonflowing, vegetated waters at some point in their life cycle. These species may have significant riverine populations during particular life history stages. The off-channel specialist category refers to species that are almost always found only in off-channel habitats or species that are limited to non-flowing, vegetated habitats throughout life. Occasionally, individuals may be found in the river channel, but the vast majority of information on these fishes pertains to off-channel habitat. (B) Original macrohabitat guild classification developed by Bain (1992).

The new guild categories were constructed based on habitat required for reproduction according to Balon (1975), general habitat use listed by Lee et al. (1980), Eenier and Starnes (1993), and Mettee et al. (1996), and from results of a literature review (Appendix 13-1A) conducted to identify off-channel habitat use by Kissimmee River fishes and their life-history stage(s). All terms follow Bain (1992), with the addition of "off-channel" (of, or related to, any habitat not included in the open water portion of the river channel). These areas include river channel littoral vegetation and any floodplain habitat. Guild relative abundance is defined as the proportion of individuals of guild *i* in relation to the total number of individuals recorded (Bain 1992).

Mean annual fish density was calculated for each habitat by first calculating a sample mean for each month by averaging the ten monthly replicates for each habitat, and then calculating a monthly mean by averaging sample means. Finally a mean annual value was determined by averaging monthly means for each year of study. Mean annual fish density was compared among habitats using ANOVA (SAS Institute 1990). Relationship between mean monthly density and water depth was tested using Linear Regression.

Results

The augmented macrohabit guild structure classified fish taxa known to occur in the Kissimmee River as follows: 29% off-channel specialist, 52% off-channel dependent, 10% habitat generalist, 0% fluvial dependent, and 8% fluvial specialist (Table 13-1).

A total of 3159 fishes representing ten species, six families, and three guilds were collected from floodplain habitats during the baseline (1996–1999) survey (Table 13-2).

Table 13-1. Macrohabitat guild classification of fishes occurring in the Kissimmee River. The off-channel dependent guild includes classification according to dependence on off-channel habitat for reproduction (R) or by life history stage (larval - L or juvenile - J).

Scientific name	Common name	Off-channel specialist	Off-channel dependent	Habitat generalist	Fluvial dependent	Fluvial specialist
Amia calva	bowfin	X				
Esox americanus	redfin pickerel	X				
Esox niger	chain pickerel	X				
Ameiurus natalis	yellow bullhead	X				
Ameiurus nebulosus	brown bullhead	X				
Noturus gyrinus	tadpole madtom	X				
Aphredoderus sayanus	pirate perch	X				
Jordanella floridae	flagfish	X				
Lucania goodei	bluefin killifish	X				
Gambusia holbrooki	mosquitofish	X				
Heterandria formosa	least killifish	X				
Poecilia latipinna	sailfin molly	X				
Elassoma evergladei	Everglades pygmy sunfish	X				
Elassoma okefenokee	Okefenokee pygmy sunfish	X				
Enneacanthus gloriosus	bluespotted sunfish	X				
Lepisosteus osseus	longnose gar		R			
Lepisosteus platyrhincus	Florida gar		R			
Dorosoma cepedianum	gizzard shad		L			
Dorasoma petenense	threadfin shad		J			
Cyprinus carpio	common carp		R			
Ctenopharyngodon idella	grass carp		R			
Notemigonus crysoleucas	golden shiner		L			
Notropis maculatus	taillight shiner		R, L, J			
Notropis petersoni	coastal shiner		J			
Opsopoedus emiliae	pugnose minnow		J			
Erimyzon sucetta	lake chubsucker		J			
Ameiurus catus	white catfish		R			
Ictalurus punctatus	channel catfish		R			
Clarius batrachus	walking catfish		R			
Hoplosternum littorale	brown hoplo		R, L, J			
Fundulus seminolis	Seminole killifish		J			
Labidesthes sicculus	brook silverside		L			
Lepomis auritrus	redbreast sunfish		R, L, J			
Lepomis gulosus	warmouth		R, L, J			
Lepomis machrochirus	bluegill		R, L, J			
Lepomis microlophus	redear sunfish		R, L, J			
Lepomis punctatus	spotted sunfish		R, L, J			
Micropterus salmoides	largemouth bass		R, L, J			
Pomoxis nigromaculatus	black crappie		R, L, J			
Astronotus ocellatus	oscar		J			
Oreochromis aureus	blue tilapia		R			
Fundulus chrysostus	golden topminnow			X		
Fundulus lineotus	lined topminnow			X		
Fundulus rubifrons	redface topminnow			X		
Menidia beryllina	tidewater silverside			X		
Etheostoma fusiforme	swamp darter			X		
Anguilla rostrata	American eel					X
Strongylura marina	Atlantic needlefish					X
Percina nigrofasciata	blackbanded darter					X
Mugil cephalus	stripped mullet					X
Pterygoplichthys disjunctivus	sailfin catfish					

Table 13-2. Fishes collected from Kissimmee River floodplain habitats in a 1957 survey (FGFWFC 1957) and during the baseline period between 1996 and 1999. Habitats sampled included Broadleaf Marsh (BLM), Woody Shrub (S.CMF) and Pasture (UP). (Ψ denotes off-channel specialist taxa, Φ denotes off-channel dependent taxa, and Λ denotes habitat generalist taxa).

Species	1957	Number collected 1996-1999 BLM Site 1	BLM Site 2	S.CMF Site 1	S.CMF Site 2	UP Site 1	UP Site 2
Esocidae							
Ψ Redfin pickerel *Esox americanus*	5						
Cyprinidae							
Φ Golden shiner *Notemigonus crysoleucas*	363						
Φ Tailight shiner *Notropis maculatus*	96						
Φ Coastal shiner *Notropis petersoni*	2						
Catostomidae							
Φ Lake chubsucker *Erimyzon sucetta*	13						
Ictaluridae							
Φ White catfish *Ameiurus catus*	2						
Ψ Brown bullhead *Ameiurus nebulosus*	1						
Φ Channel catfish *Ictalurus punctatus*	1						
Ψ Tadpole madtom *Noturus gyrinus*	18						
Clariidae							
Φ Walking catfish *Clarias batrachus*			2				
Aphredoderidae							
Φ Pirate perch *Aphredoderus sayanus*	1						
Fundulidae							
Λ Golden topminnow *Fundulus chrysotus*	6			12	13		
Ψ Bluefin killifish *Lucania goodei*	15	1					
Poeciliidae							
Ψ Eastern mosquitofish *Gambusia holbrooki*	14	50	120	123	263	3	5
Ψ Least killifish *Heterandria formosa*	3	83	47	468	712	13	1
Atherinidae							
Φ Brook silverside *Labidesthes sicculus*	12			1	29		
Elassomatidae							
Ψ Everglades pygmy sunfish *Elassoma evergladei*	7	304	226	361	94	16	16
Ψ Okefenokee pygmy sunfish *Elassoma okefenokee*		64	12	70	44	3	
Centrarchidae							
Φ Bluespotted sunfish *Enneacanthus gloriosus*	28	1	1				
Φ Redbreast sunfish *Lepomis auritus*	298						
Φ Warmouth *Lepomis gulosus*	7						
Φ Bluegill *Lepomis machrochirus*	1				1		
Φ Redear sunfish *Lepomis microlophus*	9						
Φ Largemouth bass *Micropterus salmoides*	8						
Φ Black crappie *Pomoxis nigromaculatus*	1						
Percidae							
Λ Swamp darter *Etheostoma fusiforme*	11						
Total	922	503	408	1035	1156	35	22

All fishes, except three individuals (bluegill *Lepomis macrochirus* and walking catfish *Clarias batrachus*), were small-bodied fishes. Large-bodied fishes were collected only during the wet season. Distribution of taxa according to guild included five off-channel specialists (50%), four off-channel dependents (40%), and one habitat generalist (10%) (Table 13-2). The assemblage was dominated in abundance by off-channel specialists (98%), especially least killifish *Heterandria formosa* (42%), Everglades pygmy sunfish *Elassoma evergladei* (32%), and eastern mosquitofish *Gambusia holbrooki* (18%) (Table 13-2). The remainder of the assemblage was comprised of off-channel dependents (1%) and generalists (1%) (Table 13-2). Only a single immature, large-bodied off-channel dependent (bluegill) individual was collected. Guild composition was similar among sampling periods for each habitat over the period of study and was dominated by off-channel specialist (Figure 13-3).

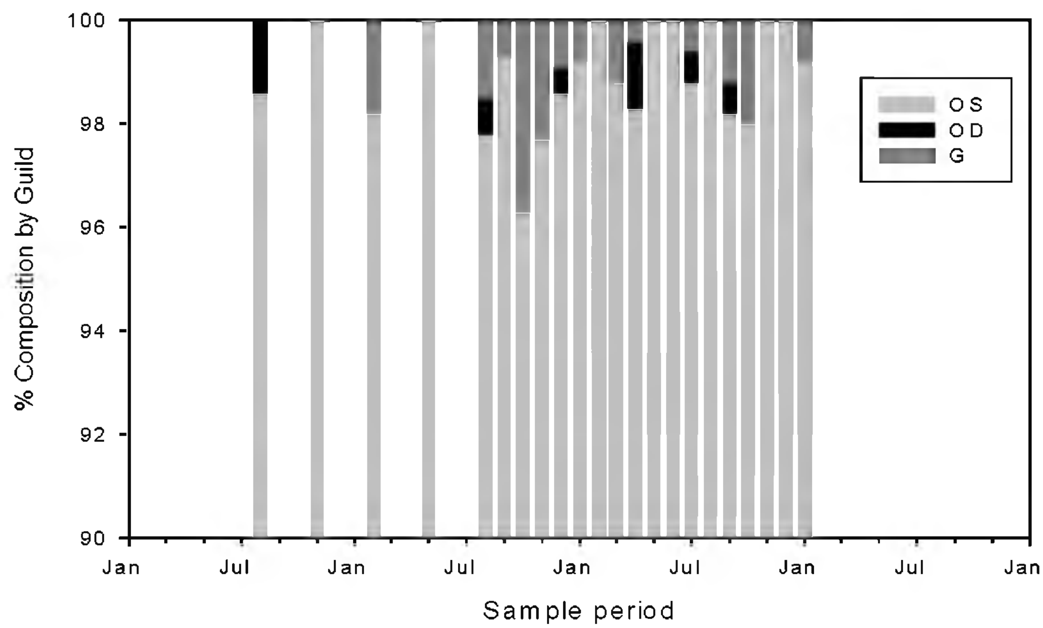


Figure 13-3. Percent composition of fishes collected in floodplain habitats by macrohabitat guild for each sampling period during the baseline survey (1996–1999). Guilds include off-channel specialist (OS), off-channel dependent (OD), and habitat generalist (G).

Mean annual density was low in all habitats (Table 13-3). Mean annual density was highest in S.CMF habitats (3.93–5.35 fish/m^2) and did not differ significantly (ANOVA; $p = 0.6314$) between pools. Broadleaf Marsh had lower mean annual densities (1.49–1.70 fish/m^2), which also were not significantly different between Control and Impact sites (ANOVA; $p = 0.9123$). Mean annual densities were lowest within UP sites (not exceeding 0.30 fish/m^2 for either pool) and were not significantly different between Control and Impact sites (ANOVA; $p = 0.7457$).

Regression analysis showed a weak, but not significant, relationship between monthly fish density and water depth at BLM sites (Figure 13-4; Pool A $R^2 = 0.21$, Pool C $R^2 = 0.18$). This relationship was stronger, but not significant, at S.CMF sites (Figure 13-5; Pool C $R^2 = 0.37$, Pool D $R^2 = 0.45$).

Reference Condition

Methods

Between 1956 and 1957 the Florida Game and Fresh Water Fish Commission (FGFWFC) sampled fish assemblages of the pre-channelized Kissimmee River to provide consideration and guidance to the Army Corps of Engineers (ACOE) for the planned channelization of the river. The sampling method employed and habitat characteristics of the sample area are unclear. Fishes were collected from a single 0.1 ha sample of floodplain marsh to which rotenone was applied. Water depths in the sample area ranged from "shallow" to 1.0 m (FGFWFC 1957). Sampling was conducted in June 1957, one year following an extreme drought. Floodplain fish assemblage structure was described by guild relative abundance according to criteria outlined under baseline conditions.

Table 13-3. Mean (± SE) annual density (number of fish/m^2) of fishes collected from Broadleaf Marsh (BLM) and Woody Shrub (S.CMF) habitats at Control and Impact sites during baseline sampling. Density values for Pasture (UP) habitat are monthly sample means because data were collected only over a single year.

Habitat	Control	Impact
BLM	1.7 ± 1.5	1.5 ± 1.1
S.CMF	3.9 ± 2.5	5.4 ± 1.1
UP	0.3 ± 0.3	0.2 ± 0.2

Fish density data for marshes of south and central Florida were compiled and summarized from published papers, theses, technical reports, and unpublished data (Jordan 1999). A total of 5314 independent samples were synthesized strictly from enclosure methods with clearly defined sampling areas capable of providing quantitative density estimates. Sample locations included marshes of the Everglades, marshes associated with lakes (including Lake Okeechobee) and canals, and marshes associated with rivers (including the upper St. Johns River). Sample methods included throw traps, Wegner rings, and block nets. Habitat types at sample locations were defined according to dominant vegetation taxa present and only data for marshes characterized by emergents (i.e., *Pontedaria* sp., *Sagittaria* sp., *Peltandra* sp.) were included for deriving the reference condition for Kissimmee River marshes. Mean fish density was calculated by averaging sample density across studies.

Results

The Florida Game and Freshwater Fish Commission (1957) collected 922 individual fish representing 24 taxa, 11 families, and three guilds (Table 13-2). This assemblage included large (adults >80 mm SL) and small-bodied fishes. Distribution of taxa according to guild included seven off-channel specialists (29.1%), 15 off-channel dependents (62.5%), and two habitat generalists (8.3%). The assemblage was dominated in abundance by off-channel dependents (88.1%), especially golden shiner *Notemigonus crysoleucas* (39%) and redbreast sunfish *Lepomis auritus* (32%) (Table 13-2). The remainder of the assemblage was comprised of off-channel specialists (10.1 %) and habitat generalists (1.8%) (Table 13-2). Of the 812 off-channel dependents collected, 39.7% were juvenile or young of the year centrarchids and esocids. Mean density of fishes in emergent marshes of south and central Florida was 23.4 (± 0.9) fish/m^2.

Discussion

Although collection methods and sample sizes differed between surveys, it is clear that dramatic changes have occurred in fish use of floodplain habitats since channelization. Approximately 60% of all species documented in the Kissimmee River during the pre-channelization survey (FGFWFC 1957) were found to use floodplain habitats, which is supported by previous studies indicating facultative use of floodplain habitats by a majority of fish taxa in river-floodplain systems (Guillory 1979, Welcomme 1979, Kwak 1988, Bayley et al. 1991, Leitman et al. 1991). Timing, depth, and duration of flood events are the critical factors regulating fish use of floodplain habitats. Results of pre-channelization surveys indicate that hydrologic conditions on the floodplain were capable of supporting a large proportion of taxa inhabiting the river-floodplain system. Also, the pre-channelization assemblage comprised both juvenile and adults of off-channel dependent taxa, implicating the floodplain's function as a nursery area.

The augmented macrohabitat guild structure reclassifies 41 taxa (82%) that would have been categorized as habitat generalist to either off-channel dependent or off-channel specialist (Table 13-1), thereby illustrating the importance of off-channel habitat availability to Kissimmee River fishes. However, fish assemblages of the channelized floodplain were dominated exclusively by small-bodied, off-channel specialist taxa. These fishes typically are not limited by minimal inundation depths, and were able to exploit floodplain habitats year-round. Large-bodied individuals, including juvenile and especially adult off-channel dependent taxa, would not be expected within floodplain habitats when depths are less than 50 cm, a depth generally required for immigration of large-bodied fishes from the river channel to the floodplain (F. Jordan, Jacksonville University, personal communication). During the baseline sampling period, mean monthly water depths on the floodplain exceeded 50 cm only once (February 1998 - Pool A BLM).

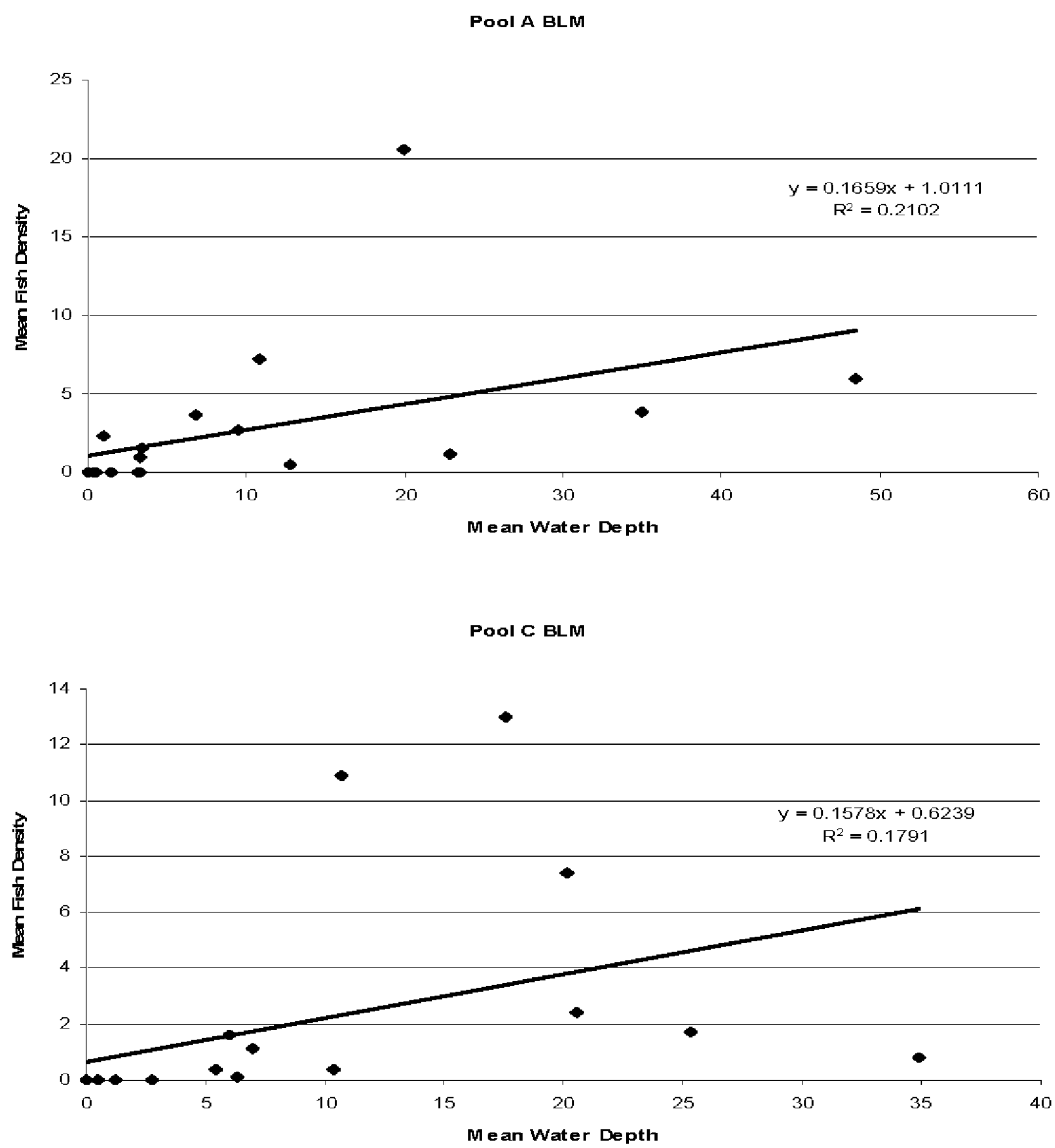
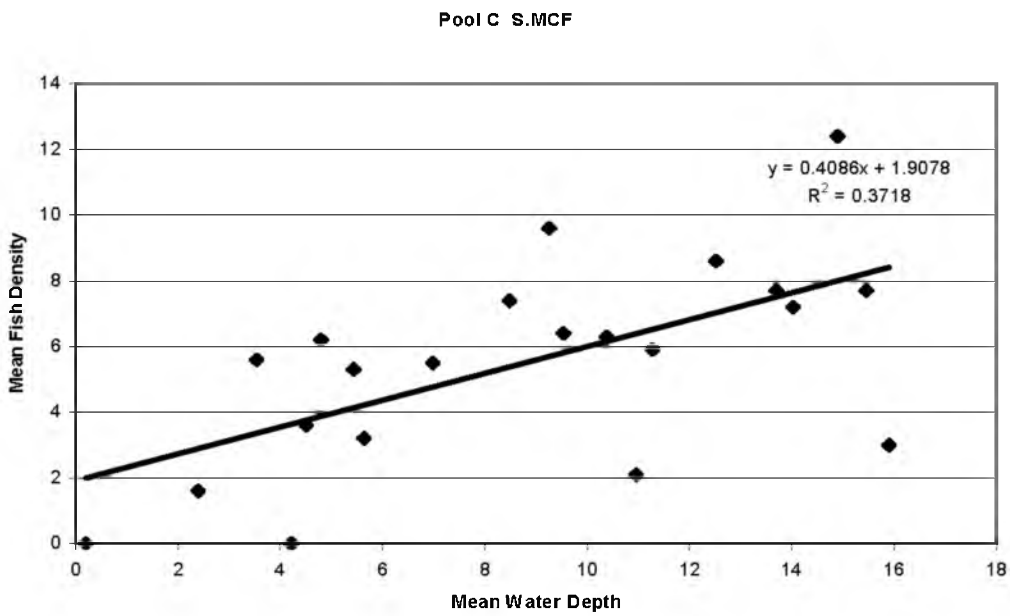


Figure 13-4. Relationship between mean monthly fish density and mean monthly water depth at Broadleaf Marsh (BLM) sites during the baseline period.

Although members of the off-channel dependent guild require access to off-channel habitat during a particular life history stage, most are also capable of using these habitats during non-dependent life history stages when conditions are favorable (Lee et al. 1980) (Appendix A). Bayley (1991) argues that species capable of using inundated floodplains benefit from increased production associated with a moving littoral zone and gain a competitive advantage (i.e., flood-pulse advantage) over taxa that cannot. Facultative fish use of floodplains is common in unaltered river systems (Welcomme 1979, Leitman et al. 1991), due in part to the temporal availability of floodplain habitats and resources associated with climatic cycles (e.g., wet and dry seasons), and is believed to have occurred frequently in the pre-channelization Kissimmee River, due to protracted floodplain inundation. Results of the baseline study suggest that the habitat requirements necessary to support off-channel dependent taxa were not present under channelized conditions.



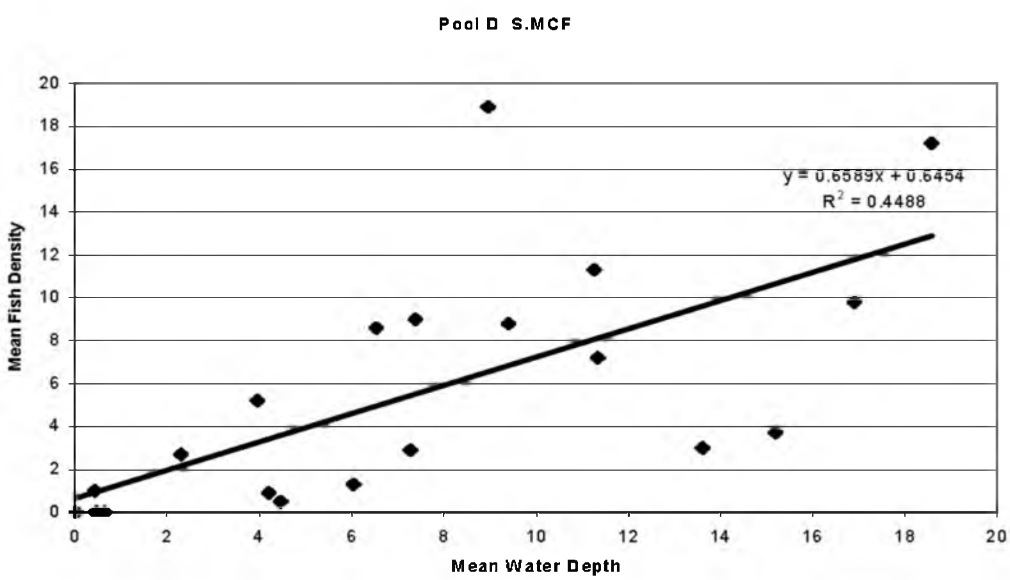


Figure 13-5. Relationship between mean monthly fish density and mean monthly water depth at Woody Shrub (S.CMF) sites during the baseline period.

The shift in numerical dominance from off-channel dependent taxa (88%) under pre-channelization conditions to dominance by off-channel specialist taxa (98%) under channelized conditions, coincides with loss of the seasonal flood pulse and associated floodplain accessibility. Even though the single sample from the 1957 survey depicts floodplain fish community structure as only a snapshot in time, it is believed to accurately portray, at a minimum, seasonal use by off-channel dependent taxa. No seasonal change in guild composition was indicated from monthly sampling over two years in the latter survey (Figure 13-3). Fishes that dominate biomass and production in river–floodplain systems depend on periodically inundated floodplain habitats for reproduction (Shaeffer and Nickum 1986, Copp 1989), foraging (Gladden and Smock 1990), and refugia (Savino and Stein 1982, Welcomme 1985) at some life history stage, unlike off-channel specialist, which are able to complete their entire life history on the floodplain. Pre-channelization data indicate that 37% of off-channel dependent fishes collected were juvenile or young-of-the-year (YOY) centrarchids, which are the dominant taxa in most peninsular Florida rivers (Bass and Cox 1985). The results suggest that the hypothesized nursery function afforded to centrarchids, which are off-channel dependent, in the pre-channelized system was compromised due to channelization, as only a single immature centrarchid was collected under channelized conditions.

Although off-channel dependent taxa were represented by only one individual in the floodplain, members of this guild were abundant in remnant river channels (see Section II below). Several factors may account for the limited use of floodplain habitats by immature off-channel dependent taxa under channelized conditions: (1) adult access to floodplain habitats for spawning was limited by inundation depth or dense vegetation; therefore, these species were restricted to littoral habitats within the river channel; (2) floodplain habitats under the baseline condition do not receive a seasonal flood-pulse due to hydrologic regulation of the system and therefore the cue for initiating lateral migration is absent; or (3) elimination of flow and resulting increased coverage of littoral vegetation in remnant river channels (Bousquin 2005) provided the necessary habitat structure within remnant channels.

The observed shift in numerical dominance by off-channel specialists, especially poeceiliids (59%) and elassomatids (38%), in floodplain fish assemblages also may indicate decline in floodplain macrohabitat quality. Members of this guild are capable of completing their entire life cycle in non-flowing environments and often possess adaptations for harsh conditions that may occur in altered floodplain habitats. Poeciliids and elassomatids dominant in channelized floodplain habitats are tolerant of protracted shallow inundation depths and of low levels of dissolved oxygen, and can exist in highly degraded habitats (Meffee and Snelson 1989). Poeciliids often remain dominant under these conditions due to the high reproduction rates associated with their reproductive mode (live bearer) (Meffee and Snelson 1989).

Additionally, degraded floodplain habitats within the channelized system likely lack the heterogeneity required to support diverse fish communities (MacArthur and Wilson 1967, Trexler 1995). The principle factors affecting habitat heterogeneity within floodplain habitats are hydroperiod, inundation depth, areal extent of inundation, and macrophyte and emergent vegetation type and density (Lowe 1986, Copp 1989, Chick and McIvor 1997). These factors create niches capable of supporting greater numbers of species than can be supported in more homogenous habitats within the channelized system.

Expectations

Restoration of the physical form and pre-channelization hydrology of the Kissimmee River is expected to reestablish ecological integrity to over 100 km^2 of river-floodplain ecosystem (Toth 1993). Floodplain fish assemblage composition is expected to shift and more closely resemble that occurring before channelization, notably with the off-channel dependent guild reestablishing dominance. Potential evidence for this shift is illustrated by increased use of “enhanced” floodplain habitat in Pool B of the Kissimmee River by off-channel dependent taxa. Hydroperiod and inundation depths in floodplain habitats at the southern end of Pool B have been enhanced by the Demonstration Project (Toth 1993). Limited throw trap sampling (n=10 samples) of BLM within this area produced juveniles of two off-channel dependent taxa (bluegill and warmouth), which comprised approximately 8% of the total number of fishes collected. These results suggest that floodplain use by juvenile centrarchids and other large-bodied off-channel dependent species is likely to increase following restoration of pre-channelization hydrologic conditions. Increases in floodplain use will result from reproduction and population expansion by resident fishes, lateral migrations of small and large-bodied riverine fishes during periods of overbank flow (flood pulse), and from increased areal coverage of both temporary and permanently inundated floodplain habitat. Concurrent increases in primary and secondary production within floodplain habitats will provide the necessary food base to support increased fish populations.

Expectation for floodplain fish assemblages. Applying guilds to biotic community data has been found to simplify analyses and predictions of community change (Austen et al. 1994). The benefit of using guilds rather than individual indicator taxa to indicate environmental change is that guilds function as a "super-species" (Austen et al. 1994) that uses a particular resource similarly. The presence of one or more guild members is indicative that at least a minimal amount of the resource in question is available (Austen et al. 1994). If the dramatic decline in floodplain use by members of the off-channel dependent guild depicts elimination of floodplain connectivity or degradation of floodplain habitat quality, then the expected increase in floodplain use by the same guild infers reestablishment of that resource, especially if the magnitude of change in use is great. The expectation for floodplain fish assemblages states that following restoration, the off-channel dependent guild will constitute >50% of the assemblage and will be comprised of $\geq$12 taxa. Young-of-the-year or juveniles will comprise $\geq$30% of the off-channel dependent guild. Figure 13-6 shows pre-channelization and baseline values of percent composition and number of taxa of off-channel dependent guild members in floodplain habitats. Dashed line indicates expected value for each metric following restoration.

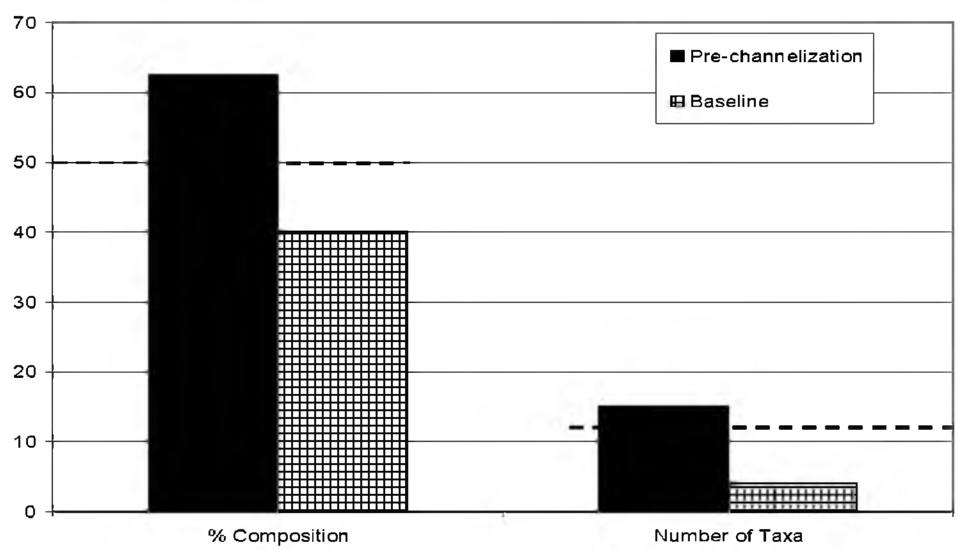


Figure 13-6. Baseline percent composition and number of taxa of off-channel dependent guild members in floodplain fish assemblages of the Kissimmee River. Dashed line indicates expected value for each metric following restoration.

All success criteria for expectation metrics of guild relative abundance are approximately 80% of historic values for the 1957 GFC sample (12 taxa and 50% relative abundance). Although conservative, these expected values account for the natural variability of floodplain fish assemblages, potential use of the floodplain by non-indigenous taxa that were introduced since channelization, and limited quantity of historic data on which the expectation is based.

Expectation for floodplain fish density. Mean annual fish density within floodplain habitats was low (<5.4 fishes/m^2 in all sampled habitats; range 0.2–5.35 fishes/m^2) during the baseline period. Fish density within floodplain habitats is related to prey availability, composition of predator assemblages, heterogeneity of floodplain vegetation, areal coverage of floodplain inundation, and depth and duration of floodplain inundation (Welcomme 1979, Lowe 1986, Heck and Crowder 1991, Connolly 1994, Loftus and Ekland 1994, Jordan et al. 1996, 1998). Fish density is expected to increase following restoration through reestablishment of these features, but is projected to fluctuate with inundation patterns. Fish densities within restored floodplain habitats are likely to be greater during periods of floodplain recession, due to concentration within topographic depressions scattered throughout the landscape. Although baseline sampling results indicate mean fish density was greater during the dry season, this increase likely was attributable to uncharacteristic floodplain inundation patterns associated with the 1997–1998 El Niño event. At S.CMF sites, mean monthly density increased with water depth. The expectation for density of fish in inundated floodplain habitats states that mean annual density of small fishes (fishes <10 cm total length) within restored BLM habitats will be ≥ 18 fish/m^2 (Figure 13-7).

The success criterion for the expectation metric of fish density is approximately 80% of the reference value for freshwater marshes of central and south Florida. Although conservative, these expected values account for the natural variability of floodplain fish assemblages and limited quantity of historic data on which the expectation is based.

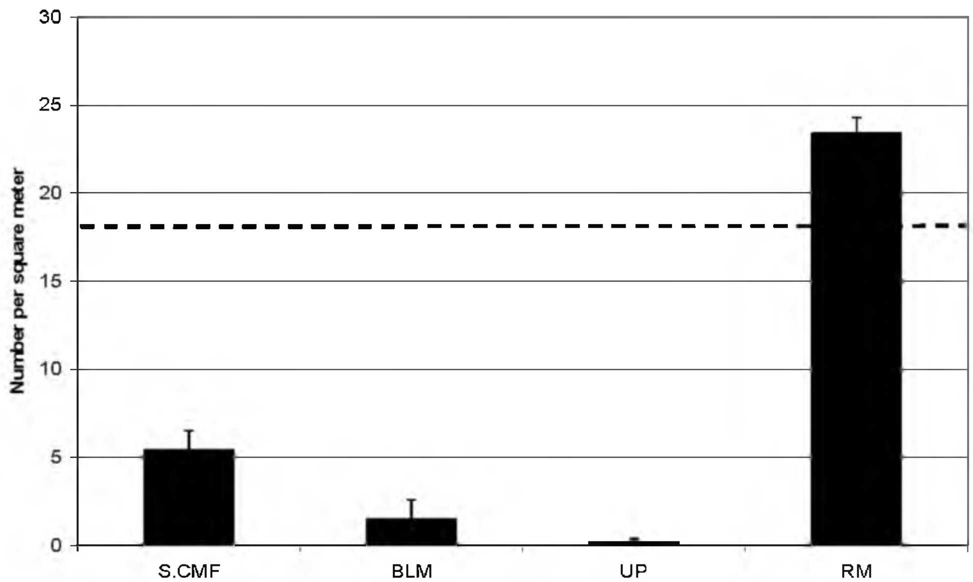


Figure 13-7. Mean density of fishes collected from Broadleaf Marsh (BLM), Woody Shrub (S.CMF), and Pasture (UP) habitats of the Kissimmee River under baseline conditions and from reference marshes (RM) of south and central Florida. Dashed line indicates expected value of small fishes within floodplain marshes following restoration.

Expectation evaluation

Throw trap sampling will be used to evaluate post-restoration floodplain fish assemblages at the same locations as baseline sampling. Sampling will begin immediately following inundation of floodplain habitats associated with implementing the Final Headwaters Regulation Schedule. Methods will be identical to those utilized for baseline studies, including monthly collection of ten random samples in each habitat. Sampling will be conducted in three-year periods beginning on the first and sixth years following implementation of the Final Headwaters Regulation Schedule.

Samples will be analyzed for composition, age class, and relative abundance of small- and large-bodied taxa according to macrohabitat guild. These metrics will document restoration of river channel-floodplain exchange and use of floodplain habitats as spawning and nursery grounds. Age classes of centrarchids and esocids will be based on total body length (Table 13-4). Mean annual relative abundance for all taxa will be based on each three-year block of post-restoration data. Annual means will be derived by averaging monthly relative abundance, generated from total numbers pooled from ten replicates each month. Seasonal effects (especially prolonged floodplain inundation during the wet season) on relative abundance are expected to be reflected in yearly means. Although this expectation is based on mean annual relative abundance, data also will be analyzed by season to evaluate the potential significance of seasonality.

II. RIVER CHANNEL FISH ASSEMBLAGE STRUCTURE

Baseline Condition

Methods

River channel fish communities inhabiting areas within and adjacent to littoral vegetation were sampled annually in June between 1992 and 1994 by the FGFWFC using electrofishing gear. Electrofishing adequately samples fish populations in shallow, vegetated habitats and does not alter community composition, as collected individuals are released alive following work-up. Sampling gear consisted of a 5.5 meter jon boat outfitted with a 5-kilowatt generator, Coffelt electrofishing unit (Model #VVP-15), and cable electrodes, with the boat serving as the anode. Pulsed AC current varied between 200–240 volts and 4–8 amperes. Triplicate 15 minute shocking episodes were conducted along fixed transects within C-38 and remnant river channel. Electrofishing was conducted in C-38 and three remnant river runs in Pools A (Ice Cream Slough Run, Persimmon Mound Run, and School House Run) and C (Montsdeoca Run, Micco Bluff Run, and MacArthur Run) (Figure 13-1). Sampling was conducted by two-person crews (one driver and one dip-netter) along the deep water edge of littoral vegetation as the boat traveled downstream. Fish were identified to species, counted, and weighed. All fishes except Florida gar *Lepisosteus platyrhincus* and bowfn *Amia calva* (due to difficulty in handling) were measured to the nearest millimeter. Body lengths for unmeasured gar and bowfin were derived from length-weight regressions generated from a subset of measured and weighed fishes.

Table 13-4. Body lengths for age class determination of centrarchid and esocid taxa in the Kissimmee River (modified from Carlander 1977 and Lee et al. 1980).

Taxa	Common Name	Young-of-the-year	Juvenile
Esox ameicanus	redfin pickerel	--	<250 mm
Esox niger	chain pickerel	--	<300 mm
Micropterus salmoides	largemouth bass	0–64 mm	65–120 mm
Lepomis auritrus	redbreast sunfish	0–35 mm	36–60 mm
Lepomis gulosis	warmouth	0–32 mm	33–75 mm
Lepomis machrochirus	bluegill	0–45mm	46–90 mm
Lepomis microlophus	redear sunfish	0–56 mm	57–134 mm
Lepomis punctatus	spotted sunfish	--	<55 mm (SL)
Pomoxis nigromaculatus	black crappie	0–51 mm	52–130 mm

Catch per unit effort (CPUE) was calculated for abundance data. Catch per unit effort is the number or weight of organisms captured within a defined unit of sampling or fishing effort (e.g., fish/min). Mean annual relative abundance was calculated as the average of replicate samples for each pool for each year. Mean annual CPUE for abundance was calculated similarly for individual taxa and centrarchids. Mean annual relative abundance CPUE was compared between years and sites using ANOVA (SAS Institute 1990) and associated means separation test.

Results

A total of 6247 fishes representing 32 species were collected by electrofishing (Table 13-5). Dominant species (>5% of mean annual relative abundance) at Control sites in Pool A included *L. platyrhincus* (36.8%), *L. macrochirus* (19.9%), *A. cavla* (8.4%), and *Micropterus salmoides* (7.9%) (Table 13-5). Assemblage composition at Impact sites (Pool C) was similarly dominated by *L. platyrhincus* (19.6%), *L. macrochirus* (16.5%), and *M. salmoides* (9.5%), but also included *G. holbrooki* (16.9%) and

Notemigonus crysoleucas (11.7%) (Table 13-5). Centrarchids accounted for only 31.8% and 38.3% of the fish assemblages in Pools A and C, respectively (Table 13-5). Centrarchid mean annual CPUE was significantly greater than that for lepisostids/amiids at canal sites in both Pools (ANOVA; $p < 0.05$ in all cases), however no difference occurred between groups at river channel sites in either pool (ANOVA; $p > 0.05$ in all cases).

Reference Conditions

Methods

Annual river channel fish sampling was conducted between 1983 and 1990 by FGFWFC in the lower St. Johns, Withlacoochee, and Oklawaha Rivers using electrofishing gear. Sampling was conducted in the lower St. Johns River between 1984 and 1988, in the Oklawaha River between 1983 and 1990, and in the Withlacootchee during 1983, 1985, 1986, and 1989. Sampling gear consisted of a 5.5 meter jon boat outfitted with a 5-kilowatt generator, Coffelt electrofishing unit (Model #VVP-15), and cable electrodes, with the boat serving as the anode. Pulsed AC current varied between 200--240 volts and 4--8 amperes. Duplicate 15 minute shocking episodes were conducted at fixed transects along each river. Four sites were sampled in the Oklawaha and lower St. Johns Rivers and six were sampled in the Withlacootchee. Sampling was conducted by two-person crews (one driver and one dip-netter) along the deep water edge of littoral vegetation as the boat traveled downstream. Fishes were identified to species, counted, and weighed.

Table 13-5. Mean ± SE annual relative abundance (percentage of total numbers) of fish species sampled during baseline conditions within remnant river channels of the Kissimmee River by electrofishing.

Species	Common Name	FGFWFC Electrofishing 1992-1994	
		Pool A	Pool C
Ameiurus natalis	yellow bullhead	--	0.5 ± 0.2
Ameiurus nebulosus	brown bullhead	0.07 ± 0.07	0.3 ± 0.1
Amia calva	bowfin	8.3 ± 2.5	4.4 ± 0.7
Clarias batrachus	walking catfish	0.4 ± 0.4	1.4 ± 0.4
Dorosoma cepedianum	gizzard shad	0.2 ± 0.2	--
Dorosoma petenense	threadfin shad	0.06 ± 0.06	--
Elassoma okeefenokei	Okeefenokee pygmy sunfish	--	0.1 ± 0.1
Ennecanthus gloriosus	bluespotted sunfish	0.1 ± 0.1	0.5 ± 0.2
Erimyzon sucetta	lake chubsucker	1.4 ± 0.5	3.9 ± 1.2
Esox niger	chain pickerel	0.3 ± 0.1	0.3 ± 0.1
Etheostoma fusiforme	swamp darter	--	0.1 ± 0.05
Fundulus chrysotus	golden topminnow	0.3 ± 0.2	0.4 ± 0.3
Gambusia holbrooki	mosquitofish	4.5 ± 2.4	16.9 ± 9.0
Heterandria formosa	least killifish	0.2 ±0.2	0.7 ± 0.6
Jordanella floridae	flagfish	--	0.2 ± 0.2
Labidesthes sicculus	brook silverside	0.2 ± 0.2	0.1 ± 0.1
Lacania goodei	bluefin killifish	--	0.2 ± 0.2
Lepisosteus osseus	longnose gar	--	0.1 ± 0.05
Lepisosteus platyrhincus	Florida gar	36.8 ± 2.9	19.6 ± 3.0
Lepomis gulosus	warmouth	1.6 ± 0.4	4.8 ± 1.6
Lepomis macrochirus	bluegill	19.1 ± 4.8	16.5 ± 4.0
Lepomis marginatus	dollar sunfish	--	0.3 ± 0.1
Lepomis microlophus	redear sunfish	2.6 ± 1.0	4.4 ± 0.9
Lepomis punctatus	spotted sunfish	0.1 ± 0.1	1.5 ± 0.7
Micropterus salmoides	largemouth bass	7.9 ± 3.5	9.4 ± 0.7
Notemigonus crysoleucas	golden shiner	14.4 ± 5.5	11.7 ± 4.3
Poecilia latipinna	sailfin molly	0.1 ± 0.1	0.2 ± 0.1
Pomoxis nigromaculatus	black crappie	0.3 ± 0.1	0.9 ± 0.02

Catch per unit effort for individual taxa was calculated for each year of study by dividing the total number of fishes collected at all sites (site data were pooled) by total pedal time (total amount of electrofishing effort). Mean annual CPUE was calculated by summing yearly CPUE values and dividing by the number of sample years.

Results

Lepomis auritus and *L. macrochirus* were dominant in all reference rivers, with mean annual relative abundances exceeding 18% (range: 18.7–23.2%) and 14% (range: 14.8–35.0%), respectively (Table 13-6). Other centrarchids contributing greater than 5% mean annual relative abundance included *L. punctatus, L. microlophus, L. gulosus*, and *M. salmoides* (Table 6). *Gambusia holbrooki* and *Notropis petersoni* were the remaining dominant species in the Withlacoochee River, while *N. crysoleucas* and *Fundulus seminolis* contributed greater than 5% in the St. Johns River (Table 13-6). Centrarchids collectively comprised ≥ 70% of the river channel fish community in all three reference rivers (Table 13-7).

Discussion

Results of electrofish sampling data indicate mean annual relative abundance of centrarchids at Control and Impact sites was 31.8% and 38.3%, respectively. Centrarchids are abundant in most freshwater river systems in Florida and are dominant in several (Bass and Cox 1985, Bass 1990). The relative contribution of centrarchid species to fish populations within peninsular Florida rivers is great when compared to the rest of the southeastern United States (Swift et al. 1986, Gilbert 1987). Members of the family Centrarchidae (sunfishes) made up more than 70% of CPUE relative abundance in the three reference rivers. Thus, decreased relative abundance of centrarchids in the channelized system is a likely indication that riverine habitat is no longer suitable for sustaining the abundance of centrarchids typical of the region.

Reestablishment of continuous flow will facilitate increased mean annual relative abundance of *L. auritus* and *L. punctatus* in restored river channels. *Lepomis auritus* is considered to be a predominantly stream-dwelling species (Lee et al. 1980, Aho and Terrell 1986). Abundance of *L. auritus* increased in Pool B river channels following implementation of the Demonstration Project and was believed to reflect a response to reestablished flows (Wullschleger et al. 1990). Although *L. punctatus* occurs in more diverse habitats than *L. auritus* (Loftus and Kushlan 1987), it is common in moderately flowing waters with vegetation or other cover (Lee et al. 1980). Abundance of *L. punctatus* also increased in Pool B following reintroduction of flow (Wullschleger et al. 1990). Centrarchid relative abundance will increase as a result of restoration and will be due, in part, to increased abundance of *L. auritus* and *L. punctatus*.

Abundance of tolerant species (least affected by seasonally low levels of dissolved oxygen) in river channel habitats at Control sites suggests this group has increased by 900% since channelization, and is an indication of decreased habitat quality in the channelized system. Florida gar (*Lepisosteus platyrhincus*), bowfin (*Amia calva*), and mosquitofish were the dominant tolerant species at Impact sites. These taxa typically increase in relative abundance in rivers with reduced water quality, especially in those rivers exhibiting chronically low levels of dissolved oxygen (Bass and Cox 1985, Bass 1990, Champeau 1990).

Dissolved oxygen levels were typically low within remnant river channels under channelized conditions, especially during summer months when water temperatures were high. Relative composition of fishes in the river channel is expected to significantly change following restoration, as relative abundance of tolerant species declines. Electrofishing conducted to evaluate effects of the Demonstration Project indicated revitalized runs in Pool B supported greater species richness, and centrarchids contributed a higher percentage of the total catch (numbers and biomass) than in a stagnant run in Pool E (Wullschleger et al. 1990). Increased levels of dissolved oxygen will allow centrarchids and other less tolerant species to better compete with tolerant species for available resources.

<u>Expectation for River Channel Fish Assembleages.</u> Four relative abundance metrics *(L. platyrhinchus, A. calva, L. auritus*, and centrarchids) show strong differences between baseline and reference conditions (Figure 13-8) were used to develop the expectation for assessing change in river channel fish assemblage

structure following restoration. Relative abundances of *L. platyrhincus* and *A. calva* are typically higher in river systems with degraded water quality (Champeau 1990, Bass 1991). Relative abundance of *L. auritus* is positively correlated with increased flow (Aho and Terrell 1986). Relative abundances of *L. platyrhincus* and *A. calva* are influenced by flow dependent habitat availability, and both species prefer little to no flow and abundant aquatic vegetation. (Lee et al 1980, Mettee et al. 1996). Reestablishment of historic sand substrate and sandbars will increase spawning habitat for *L. auritus* and other centrarchids (Carlander 1977, Aho and Terrell 1986), with increased recruitment resulting from reestablishment of river channel–floodplain linkage that historically provided floodplain habitat as refugia for juveniles (FGFWFC 1957).

Table 13-6. Mean ± SE annual relative abundance of fishes collected by electrofishing by Florida Game and Fresh Water Fish Commision between 1983 and 1990 in the St. Johns (STJ), Oklawaha (OKL), and Withlacoochee (WIT) Rivers

Species	Common Name	STJ	OKL	WIT
Alosa sapidissima	American shad	0.02 ± 0.01	0.3 ± 0.04	--
Ameiurus catus	white catfish	0.3 ± 0.2	0.1 ± 0.04	0.1 ± 0.01
Ameiurus natalis	yellow bullhead	0.1 ± 0.01	0.5 ± 0.2	0.1 ± .06
Ameiurus nebulosus	brown bullhead	0.3 ± 0.1	0.1 ± 0.03	0.04 ± 0.02
Amia calva	bowfin	0.6 ± 0.2	0.8 ± 0.1	1.3 ± 0.4
Anguilla rostrata	American eel	0.2 ± 0.1	--	0.1 ± 0.05
Aphredoderus sayanus	pirate perch	0.03 ± 0.01	2.0 ± 0.4	0.9 ± 0.4
Centrarchus macropterus	flier	0.01 ± 0.01	--	--
Dorosoma cepedianum	gizzard shad	0.9 ± 0.4	0.3 ± 0.2	0.03 ± 0.02
Dorosoma petenense	threadfin shad	0.3 ± 0.2	0.05 ± 0.02	0.04 ± 0.03
Elassoma evergladei	Everglades pygmy sunfish	--	0.01 ± 0.01	0.07 ± 0.02
Elassoma zonatum	banded pygmy sunfish	--	0.01± 0.01	--
Ennecanthus gloriosus	bluespotted sunfish	0.03 ± 0.02	0.02 ± 0.01	0.5 ± 0.2
Erimyzon sucetta	lake chubsucker	0.6 ± 0.1	2.5 ± 0.3	1.6 ± 0.4
Esox americanus	redfin pickerel	--	0.03 ± 0.01	0.2 ± 0.1
Esox niger	chain pickerel	0.08 ± 0.01	0.6 ± 0.1	0.1 ± 0.03
Etheostoma fusiforme	swamp darter	--	0.6 ± 0.2	0.2 ± 0.08
Fundulus chrysotus	golden topminnow	--	0.01 ± 0.01	0.1 ± 0.06
Fundulus seminolis	Seminole killifish	6.0 ± 1.8	0.1 ± 0.07	0.1 ± 0.04
Gambusia holbrooki	mosquitofish	0.3 ± 0.2	0.5 ± 0.1	6.4 ± 2.3
Heterandria formosa	least killifish	0.03 ± 0.03	--	0.1 ± 0.04
Ictalurus punctatus	channel catfish	0.1 ± 0.06	0.02 ± 0.01	0.03 ± 0.02
Jordanella floridae	flagfish	0.03 ± 0.03	--	0.01 ± 0.01
Labidesthes sicculus	brook silverside	0.4 ± 0.1	1.5 ± 0.3	2.7 ± 1.2
Lacania goodie	bluefin killifish	0.1 ± 0.05	0.03 ± 0.01	0.2 ± 0.1
Lepisosteus osseus	longnose gar	0.1 ± 0.03	0.2 ± 0.04	0.2 ± 0.03
Lepisosteus platyrhincus	Florida gar	2.4 ± 0.4	1.3 ± 0.2	2.9 ± 0.9
Lepomis auritus	redbreast sunfish	18.7 ± 1.2	23.2 ± 1.6	19.2 ± 2.9
Lepomis gulosus	warmouth	1.3 ± 0.2	4.9 ± 0.5	6.1 ± 0.4
Lepomis macrochirus	bluegill	35.0 ± 1.1	27.7 ± 2.4	14.8 ± 2.8
Lepomis marginatus	dollar sunfish	0.03 ± 0.03	0.1 ± 0.04	2.5 ± 0.7
Lepomis microlophus	redear sunfish	8.1 ± 1.1	9.3 ± 0.6	6.7 ± 1.8
Lepomis punctatus	spotted sunfish	3.4 ± 0.3	10.7 ± 1.5	18.5 ± 2.1
Lucania parva	rainwater killifish	0.05 ± 0.03	--	--
Menidia beryllina	inland silverside	0.7 ± 0.3	0.01 ± 0.01	--
Menidia peninsulae	tidewater silverside	0.5 ± 0.4	--	--
Micropterus salmoides	largemouth bass	4.8 ± 0.2	5.3 ± 0.4	5.8 ± 2.3
Morone saxatilis	striped bass	0.02 ± 0.02	--	--
Morone sp.	sunshine bass	0.1 ± 0.1	--	--
Mugil cephalus	striped mullet	2.7 ± 0.3	0.1 ± 0.04	0.1 ± 0.07
Myrophis punctatus	speckled worm eel	--	--	0.01 ± 0.01
Mugil curema	white mullet	0.03 ± 0.03	--	--
Notemigonus crysoleucas	golden shiner	6.3 ± 0.8	1.7 ± 0.3	0.5 ± 0.1
Notropis maculates	taillight shiner	1.5 ± 2.4	0.8 ± 0.2	0.6 ± 0.1
Notropis petersoni	coastal shiner	0.01 ± 0.01	2.0 ± 0.6	5.6 ± 2.3
Noturus gyrinus	tadpole madtom	--	0.04 ± 0.01	0.3 ± 0.1
Noturus leptacanthus	speckled madtom	--	0.06 ± 0.01	--
Opsopoedus emilidae	pugnose minnow	0.1 ± 0.1	0.01 ± 0.01	--
Oreochromis aureus	blue tilapia	0.05 ± 0.02	0.01 ± 0.01	--
Percine nigofasciata	blackbanded darter	--	1.3 ± 0.4	--
Poecilia latipinna	sailfin molly	0.03 ± 0.03	0.1 ± 0.05	0.5 ± 0.1
Pomoxis nigromaculatus	black crappie	2.1 ± 0.3	0.5 ± 0.1	0.3 ± 0.2
Strongylura marina	Atlantic needlefish	0.8 ± 0.3	0.05 ± 0.01	0.08 ± 0.04
Trinectes maculates	hogchoker	0.03 ± 0.02	0.02 ± 0.01	0.2 ± 0.1

Table 13-7. Percent contribution by centrarchids collected using electrofish sampling within three reference rivers between 1983 and 1990 and the Kissimmee River between 1992 and 1994. (KIS = Kissimmee River, STJ = St. Johns River, OKL = Oklawaha River , WIT = Withlacoochee River).

Species	KIS	STJ	OKL	WIT
Centrarchus macropterus	--	0.01 ± 0.01	--	--
Ennecanthus gloriosus	0.5 ± 0.2	0.03 ± 0.02	0.02 ± 0.01	0.5 ± 0.2
Lepomis auritus	--	18.7 ± 1.2	23.2 ± 1.6	19.2 ± 2.9
Lepomis gulosus	4.8 ± 1.6	1.3 ± 0.2	4.9 ± 0.5	6.1 ± 0.4
Lepomis macrochirus	16.5 ± 4.0	35.0 ± 1.1	27.7 ± 2.4	14.8 ± 2.8
Lepomis marginatus	0.3 ± 0.1	0.03 ± 0.03	0.1 ± 0.04	2.5 ± 0.7
Lepomis microlophus	4.4 ± 0.9	8.1 ± 1.1	9.3 ± 0.6	6.7 ± 1.8
Lepomis punctatus	1.5 ± 0.7	3.4 ± 0.3	10.7 ± 1.5	18.5 ± 2.1
Micropterus salmoides	9.4 ± 0.7	4.8 ± 0.2	5.3 ± 0.4	5.8 ± 2.3
Pomoxis nigromaculatus	0.9 ± 0.02	2.1 ± 0.3	0.5 ± 0.1	0.3 ± 0.2
TOTAL	38.3	73.4	81.7	74.4

The remaining metric, percent centrarchid composition, was chosen because peninsular Florida river systems are typically dominated by centrarchids (Swift et al. 1986, Gilbert 1987) (Table 13-4). The restoration expectation for river channel fish assemblages states that mean annual relative abundance of fishes in the restored river channel will consist of ≤ 1% bowfin *Amia calva*, ≤ 3% Florida gar *Lepisosteus platyrhincus*, ≥ 16% redbreast sunfish *Lepomis auritus*, and ≥ 58% centrarchids (sunfishes) (Figure 13-8).

Restoration of pre-channelized discharge patterns will increase levels of dissolved oxygen due to reaeration through turbulent mixing, flushing of accumulated organic deposits, and reduction in associated biological oxygen demand (Toth 1993, 1996). Baseline dissolved oxygen regimes persist at the tolerance threshold (2.0 ppm) for many fish species (Moss and Scott 1961, Davis 1975) and periodically reach critically low levels (<0.5 ppm) during summer months (Toth 1993, Koebel 1995). Depressed levels of dissolved oxygen negatively affect survivorship of all life history stages of most large-bodied species currently inhabiting the system, and may be the primary factor influencing decreased densities of large-bodied fish since channelization. Dissolved oxygen profiles are expected to become less stratified (especially during summer months), with higher levels of dissolved oxygen throughout the water column. Increased levels of dissolved oxygen will allow for increased survivorship of all life history stages of large-bodied fishes, especially species intolerant (i.e., centrarchids) of low levels of dissolved oxygen, thus allowing them to better compete with tolerant species (i.e., *L. platyrhinchus* and *A. calva*).

Numerous physical changes within restored river channels will confer change in river channel fish assemblage structure. Changes in river channel geomorphology also will affect riverine fish diversity and density. Existing cross sections impede community partitioning through lack of depth diversity and decreased availability of instream microhabitats. Geomorphic features including erosion and deposition zones provide a range of flow velocities that are used differently by dissimilar species and life history stages (Lobb and Orth 1991, Sheldon and Meffe 1995). Reintroduction of instream flow will flush accumulated organic deposits and provide the topographic diversity necessary to produce a range of flow velocities useful to a larger consort of species and life history stages (Bain et al. 1988, Lobb and Orth 1991, Sheldon and Meffe 1995). Newly created zones of erosion and deposition will include scour areas (providing deep-water habitat), point bars (creating back eddies and slower current velocities), and shoals (creating spawning grounds and shallow water habitat). River channel depth diversity can be positively correlated with fish community attributes including biomass, species richness, density, and mean size (Lobb and Orth 1991, Sheldon and Meffe 1995). Erosional processes also will create snags as riparian trees are displaced into the river. Snags provide relief from high velocities, as well as an abundance of prey items such as aquatic invertebrates, which use woody debris as a substrate for attachment and feeding (Benke et al. 1985, Lobb and Orth 1991). These physical attributes and processes will be responsible to some degree for influencing changes in the metric developed for river channel fish assemblages.

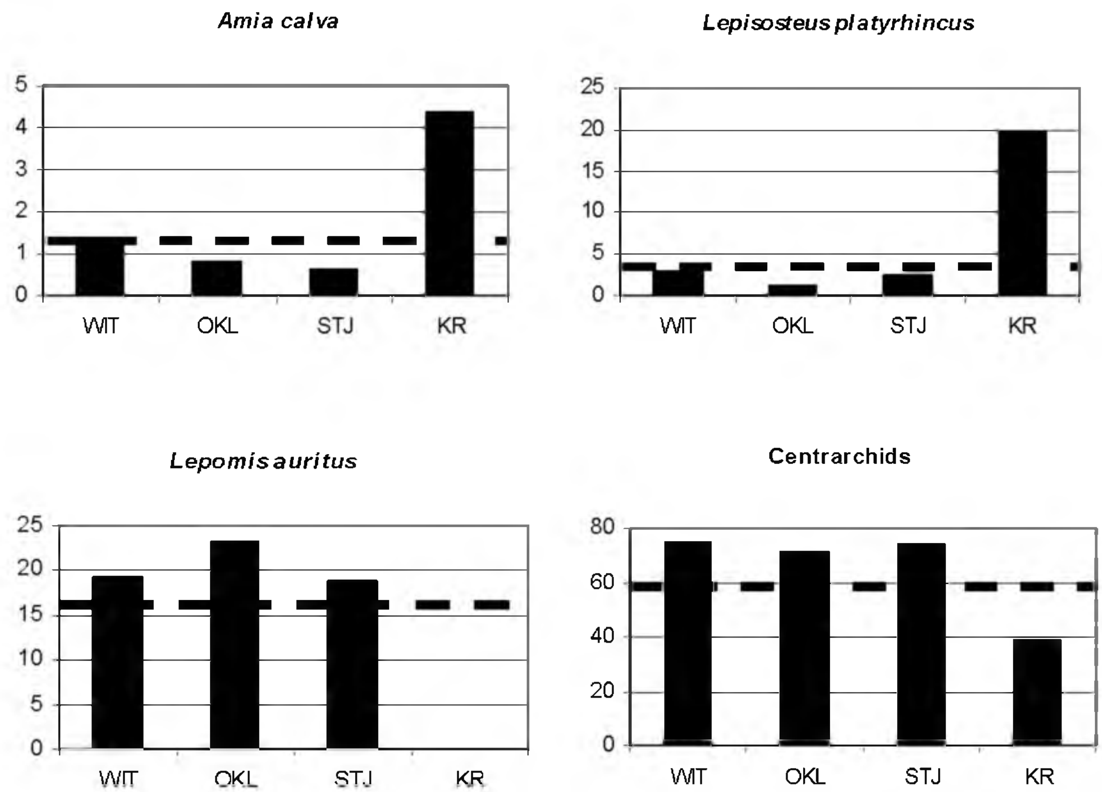


Figure 13-8. Baseline mean annual relative abundance of fish taxa or family that will be used as metrics to evaluate restoration success in reestablishing river channel fish assemblage structure. Dashed line indicates expected value for each taxon or family following restoration. (WIT = Withlacoochee River, OKL = Oklawaha River, STJ = St. Johns River, KR = Baseline data from Kissimmee River).

Similar effects of channelization on fish assemblages have been documented in other systems. Tarplee et al. (1971) found channelized Coastal Plain streams in North Carolina had reduced biomass, diversity, carrying capacity, and number of harvestable sized game fishes, notably centrarchids. They also noted that channelization adversely affected game fish to a greater degree than nongame fish. Hortle and Lake (1983) attributed decreased abundance and species richness of fishes in Australian streams after channelization to loss of suitable habitat (i.e., area of snags, area of slack water, length of bank fringed with vegetation). Other studies attribute reduced standing crop, density, and diversity of stream fish assemblages to decreased habitat, as well as decreased cover and shelter, prey or other food items, and available spawning areas (Guillory 1979, Welcomme 1985, Scheaffer and Nickum 1986, Copp 1989, Junk et al. 1989). Karr and Schlosser (1978) suggested that as much as 98% of the standing crop of fishes in a river may be lost when the flood regime is altered by channelization.

Sampling will be conducted annually, for three year-periods, beginning on the second year following implementation of the Final Headwater Regulation Schedule. Sample methods will be identical to baseline studies (FGFWFC 1996).

STUDIES WITHOUT ASSOCIATED RESTORATION EXPECTATIONS

I. FLOODPLAIN FISH ASSEMBLAGE STRUCTURE

Baseline Condition

Methods

Floodplain fish assemblage metrics included species richness (S = the total number of species present), species diversity (H'), where H' = $-\Sigma\ p_i \ln p_i$ and p_i is the proportional abundance of the ith species, and community evenness (J'), where J = H'/lnS (Price 1984). Species richness was calculated for each habitat seasonally and for the entire baseline period. Seasons were defined as wet (June through November) and dry (December through May). Values of evenness were compared between like habitats to better understand results of the Shannon index, as it evaluates both species richness and evenness within a community. Mean species diversity and evenness were calculated for the baseline period and seasonally by summing monthly values and dividing by the number of months sampled over each period. Mean species diversity was compared between Control and Impact sites for all habitats using General Linear Models (ANOVA; SAS Institute 1990). These metrics were not used in restoration expectation development.

Results

Species richness was highest within Pool C S.CMF (7), followed by Pool D S.CMF and both BLMsites (6), Pool A UP (4), and Pool C UP (3) (Table 13-8). A similar trend was observed for species richness during wet and dry seasons; however, no species were collected within UP sites during the wet season. Species diversity (H') was low in all habitats over the baseline period and ranged from 0.64 in Pool C UP to 0.77 in Pool C C.MCF (Table 13-8). Mean diversity of all floodplain samples during the baseline period was not significantly different between Control and Impact sites in any habitat (ANOVA; $p > 0.05$). Seasonal species diversity showed similar ranges (wet: 0.00–0.77; dry: 0.00–0.86). Mean wet and dry season diversity in similar habitats also was not significantly different ($p > 0.05$). Community evenness (J') was low to moderate in all habitats (range: 0.00–0.57) (Table 13-8). Evenness showed greater seasonal variability and was higher during the dry season for both S.CMF sites and Pool C UP.

Table 13-8. Community structure indices for baseline floodplain fish assemblages. Results for the entire study period are summarized in Section A. Section B lists indices calculated for each habitat during wet (w) and dry (d) seasons. (S=Species Richness, H'=Shannon index, J'=Evenness). Habitats sampled included Broadleaf Marsh (BLM). Woody Shrub (S.CMF), and Pasture (UP).

		BLM		S.CMF		UP	
A.	Index	Pool A	Pool C	Pool C	Pool D	Pool A	Pool C
	S	6	6	7	6	4	3
	H'	0.43 ± 0.11	0.29 ± 0.09	0.77 ± 0.08	0.73 ± 0.11	0.17 ± 0.12	0.06 ± 0.06
	J'	0.39 ± 0.09	0.25 ± 0.08	0.56 ± 0.05	0.57 ± 0.08	0.17 ± 0.11	0
B.				Control			
		Pool A (w)	Pool A (d)	Pool D (w)	Pool D (d)	Pool A (w)	Pool A (d)
	S	5	5	6	5	0	4
	H'	0.42 ± 0.16	0.43 ± 0.15	0.62 ± 0.15	0.86 ± 0.15	0	0.37 ± 0.24
	J'	0.40 ± 0.14	0.37 ± 0.11	0.47 ± 0.12	0.68 ± 0.12	0	0.37 ± 0.23
				Impact			
		Pool C (w)	Pool C (d)	Pool C (w)	Pool C (d)	Pool C (w)	Pool C (d)
	S	6	4	7	6	0	3
	H'	0.32 ± 0.15	0.25 ± 0.13	0.77 ± 0.13	0.77 ± 0.10	0	0.14 ± 0.14
	J'	0.28 ± 0.13	0.22 ± 0.10	0.51 ± 0.08	0.61 ± 0.05	0	0

Reference Condition

Methods

Between 1956 and 1957 the Florida Game and Fresh Water Fish Commission (FGFWFC) sampled fish assemblages of the pre-channelized Kissimmee River to provide consideration and guidance to the U. S. Army Corps of Engineers (ACOE) for the planned channelization of the river. The sampling method employed and habitat characteristics of the sample area are unclear. Fishes were collected from a single 0.1 ha sample of floodplain marsh to which rotenone was applied. Water depths in the sample area ranged from "shallow" to 1.0 m (FGFWFC 1957). Sampling was conducted in June 1957, one year following an extreme drought.

Results

Pre-channelized floodplain marsh supported 24 species (Table 13-2) and was reasonably diverse (H' = 2.53). Community evenness was 1.86.

Discussion

Although reference data on floodplain fish assemblage structure come from a single sample, it is evident that pre-channelization floodplains supported, at least periodically, a relatively diverse fish community. Because species richness within Control and Impact sites was similar under channelized conditions, any increases within Impact sites will be clearly linked to restoration, if species richness within Control sites remain similar to baseline values following restoration. Pool A BLM is expected to be inundated a greater portion of the year as a result of elimination of a secondary drainage ditch in this region; however, this change in hydrology will not be as great as in Pool C, and the Pool A BLM will not be adjacent to a restored river reach. The degree of change in species richness from the baseline condition within Impact sites will be significant (>300%), if species richness approximates that found prior to channelization (24 species).

Species diversity for the baseline period was low (see Margalef 1972, Magurran 1988) and varied little seasonally among habitats. Increased fish species diversity within floodplain habitats following restoration will require reestablishment of specific system functions and microhabitats. Reestablishment of a fluctuating hydrograph and spatial and temporal variability in inundation depth across the floodplain will lead to restoration of backwater lakes and ponds (for supporting large-bodied species), deep and shallow marsh, and a peripheral, shallow wet prairie (nursery and refuge areas for small-bodied fish, and young-of-the-year and juvenile large-bodied species). Diversity will increase significantly (>100%) following restoration of these floodplain habitats if it approximates that found in pre-channelized marsh (FGFWFC 1957). Species diversity is likely to exhibit seasonal trends following restoration of a seasonal flood pulse. Diversity values are likely to be higher during the wet season when hydrologic conditions favor use by the majority of fish taxa in the system.

II. RIVER CHANNEL FISH ASSEMBLAGE STRUCTURE

Baseline Conditions

Methods

Attributes of river channel fish assemblage structure were studied using block nets coupled with a fish toxicant (5% emulsified rotenone) and hoopnets. Each sampling method evaluated specific metrics of fish assemblage structure, because neither method was free of bias for all metrics. These metrics were not used in restoration expectation development.

Block Net Sampling. Block net samples are one of the few sampling methods that estimate fish density directly (Bettoli and Maceina 1996). Collection sites for single block net samples (0.4 acre) were selected randomly within three remnant river channels of each pool (Pool A: Ice Cream Slough Run, Rattlesnake Hammock Run, Persimmon Mound Run, Figure 13-1; Pool C: Oxbow 13, Micco Bluff Run, MacArthur Run, Figure 13-1) by driving a randomly determined number of seconds (obtained by random number generator) at 2000 rpm into each river run. The three samples in each pool were considered replicates.

Sampling was conducted in late spring months (May, June) to maximize toxicant effectiveness (Bettoli and Maceina 1996) while minimizing its persistence in the environment (Gilderhus et al. 1988), and to coincide with time periods exhibiting minimum to no flow within the pre-channelized system. Block net sampling was conducted over a ten-day (five days per pool) period in 1997 and 1998.

> Day One - On the first day of sampling, paired block nets (60.9 m x 6.9 m) were deployed perpendicular to the river bank, spaced 50 meters apart, and extending across the river channel. Nets extended from the water surface to the river channel substratum by floats and lead lines. Water column depth was recorded at 10 m intervals along three transects (center of channel and half distance to shore on each side of centerline) within each enclosure. Mean water depth within the sample area was calculated using depths obtained from each transect. Total volume of aqueous habitat sampled was estimated as: Volume = shoreline length X river width X mean water depth. Sample locations were recorded in space and time using GPS (Trimble® GPS Pathfinder Pro XL).

> Day Two - Prior to rotenone application, 50 fishes of various species were captured within the enclosed area using electroshocking gear (Coffelt® model VVP-15), marked by fin clipping or fine fabric Floy® tags (depending upon fish size), and released into the enclosed area for recapture to determine sampling efficiency. Five percent emulsified rotenone was applied within each block net to achieve concentration levels of 2–3 ppm (e.g., 2.466 l rotenone/ 4046.854 m^2 = 2 ppm).

> Days Three through Five - Poisoned fish were collected, identified to species, measured (standard and total lengths; mm), weighed (grams), and counted. Weights for fish collected subsequent to Day Three were assigned from length-weight regressions developed from first day collections. Three 15 minute electroshocking episodes were conducted (subsequent to collection of poisoned fishes) within each blocked off area to collect any fish unaffected by rotenone application (Day Five only). Shocked fishes also were identified to species, measured, weighed and counted. Shocked fishes were included in all analyses. Dense littoral vegetation precluded efficient collection of small-bodied fishes; therefore, analyses of block net samples were limited to large-bodied species.

Fish species richness, relative abundance, density, and biomass were calculated for block net samples. Mean sample abundance was calculated for each pool by summing relative abundance for each run and dividing by three. Mean annual relative abundance was generated by averaging sample means for each year. Mean annual relative abundance was compared between sites using ANOVA (SAS Institute 1990) and associated means separation test. Mean annual sample density (#/0.4 acre) and mean annual sample biomass (g/0.4 acre) were calculated and similarly compared. Mean annual sample density and mean annual sample biomass were converted to #/acre and g/acre for comparisons with pre-channelization data.

<u>Hoopnet Sampling.</u> Hoopnet gear is selective for centrarchid species and yields relative species composition, abundance, and biomass for most game fish species. This sampling method is not as resource-consumptive as block netting and is easily replicated throughout the year without negatively affecting local fish populations because collected fishes are released. Thus, it provides data on temporal distributions.

Three hoopnets were deployed at random locations within C-38 and each of three remnant river runs (Pool A: Ice Cream Slough Run, Rattlesnake Hammock Run, Persimmon Mound Run; Pool C: Oxbow #13, Micco Bluff Run, MacArthur Run) in each pool (n = 12 nets/pool) (Figure 13-1). Sample sites in remnant river runs were selected by driving a randomly determined number of seconds at 2000 rpm from the entrance of the run being sampled (first net), and from each previously deployed net (second and third nets). Sample sites for hoopnets deployed in C-38 were similarly selected by driving a randomly determined number of seconds from the entrance of the previously sampled remnant river run. Direction traveled in C-38 from the remnant river run entrance (North, South) was determined using a random number generator (odd numbers = North, even numbers = South). All hoopnets were set at a distance greater than 50 m from the entrance of remnant river runs to minimize any bias associated with fish populations using the area near the confluence of C-38 and remnant river runs. The side of the river on which each hoopnet was deployed was determined using a random number generator (odd numbers = East, even numbers = West). At each sample location, nets were placed along the deep-water edge of the littoral zone, at a depth <300 centimeters. Hoopnets were deployed perpendicular to the riverbank, with the mouth (opening) facing downstream, and supported with 5 cm diameter PVC poles anchored to the substrate.

Hoopnet sampling was conducted monthly from September 1997 to August 1998. Hoopnets were deployed for approximately seven hours (0900–1600 hrs.) during each sampling event. Captured fish were identified to species, measured (mm; standard length, total length), weighed, and released. Analyses of hoopnet samples were limited to large-bodied species due to sampling net mesh size (5 cm x 5 cm).

Species richness was calculated for each pool. Catch per unit effort (CPUE) was calculated for abundance and biomass data. Mean seasonal abundance and biomass were calculated for each run based on each replicate taken over each season. Seasons were defined as wet (June through November) and dry (December through May). Mean seasonal abundance and biomass were compared between runs and sites using ANOVA (SAS Institute 1990) and associated means separation test.

Catch per unit effort (CPUE) was calculated for biomass data. Mean annual CPUE for biomass was calculated similarly for groups and all species combined. Mean annual biomass CPUE was compared between years and sites using ANOVA (SAS Institute 1990) and associated means separation test.

Results

Block Net Sampling. A total of 2242 fishes representing 18 species were collected during block net sampling (Table 13-9). Species richness for the baseline period was identical at Control (16) and Impact (16) sites. Total numbers of fishes sampled also was similar at Control (1112) and Impact sites (1130). Mean sample density did not differ significantly between years within Control (ANOVA; $p = 0.6898$) or Impact (ANOVA; $p = 0.0700$) sites (Table 13-10). Mean density also did not differ significantly between Control and Impact sites within Year 1 (ANOVA; $p = 0.9352$) or Year 2 (ANOVA; $p = 0.9230$). No significant difference ($p > 0.05$) was found in mean annual density between Control and Impact sites.

Mean annual relative abundance within Control sites was dominated by centrarchids (69%), followed by lepisostids (14.7%), ictalurids (6%), and exotic fish (1%). Similar mean annual relative abundance was found at Impact sites (centrarchids 75%, lepisostids 8%, ictalurids 9%, exotic fish 0.3%). Mean annual relative abundance was not significantly different between years within sites or between Control and Impact sites within years ($p > 0.05$). However, these relative abundance values may not accurately reflect river channel fish community composition because small-bodied fishes were not collected, and therefore not included in analyses.

A total of 293,011 g (live mass) of fish biomass was collected during block net sampling (Table 13-11). Total sample biomass was similar at Control (166,084 g) and Impact (126,927 g) sites. Mean sample biomass was not significantly different between years at Control (ANOVA; $p = 0.9801$) or Impact sites (Impact; $p = 0.3078$; Table 13-12). Mean sample biomass also did not differ between Control and Impact sites during Year 1 (ANOVA; $p = 0.6150$) or Year 2 (ANOVA; $p = 0.6304$). However, mean annual biomass was greater at Control sites (ANOVA; $p = 0.0504$). Lepisostids and amiids had the highest mean biomass within Control sites (38.4%), followed by centrarchids (33.4%), ictalurids (17.5%), and exotics (3.2%). Within Impact sites, mean biomass of centrarchids (38.6%) was greater than lepisostids and amiids (30.2%), ictalurids (25.4%), and exotics (2.1%).

Hoopnet Sampling. A total of 1099 fishes representing 16 species were collected by hoopnet sampling (Table 13-13). Species richness for the baseline period was similar at Control (16) and Impact (14) sites. Total numbers were similar between sites (Control 518, Impact 581). Total numbers of fishes collected within canal and remnant river runs also were similar for both Control (canal 274; river channel 244) and Impact (canal 260; river channel 321) sites; however, sampling effort was three times greater in river channels. Species composition was similar between pools, and was dominated by centrarchids.

Mean seasonal abundance was not significantly different among river runs within each pool during dry (ANOVA; Control $p = 0.3631$, Impact $p = 0.6061$) and wet (ANOVA; Control $p = 0.1115$, Impact $p = 0.0935$) seasons. Therefore, mean seasonal abundance data for all river runs within Control and Impact sites were pooled for each season, and was not significantly different between Control and Impact sites during dry (ANOVA; $p = 0.9049$) and wet (ANOVA; $p = 0.6909$) seasons (Table 13-14).

Total numbers of fishes collected differed between seasons at Control sites (dry = 355, wet = 163) and Impact sites (dry = 399, wet = 182). Mean seasonal abundance was significantly different between seasons in Ice Cream Slough (p = 0.0020), Rattlesnake Hammock (p = 0.0460), and MacArthur (p = 0.0175) river runs (Table 13-13). Therefore, data could not be pooled for comparisons between sites.

Over 440,000 g (live mass) of fish biomass was collected during hoopnet studies. Total sample biomass was similar at Control (204,788 g) and Impact sites (239,265 g). Mean seasonal biomass was not significantly different among river runs (excluding C-38 canal) within each site during dry (ANOVA; Control p = 0.3282, Impact p = 0.6826) and wet (ANOVA; Control p = 0.2397, Impact p = 0.2464) seasons (Table 13-15). Therefore, mean seasonal biomass data for all river runs within Control and Impact sites were pooled for each season, and was not significantly different between Control and Impact sites during dry (ANOVA; p = 0.6160) and wet (ANOVA; p = 0.0700) seasons (Table 13-16).

Table 13-9. Total numbers of fishes collected per river run during 1997 (A) and 1998 (B) baseline block net sampling.

SPECIES	CONTROL			IMPACT		
1997	ICS	PM	RSH	OX13	MB	MAC
Amia calva	8	7	4	4	1	2
Ameiurus natalis	1	0	0	0	0	0
Ameiurus nebulosus	7	1	0	7	4	7
Dorosoma petenense	0	0	0	0	0	3
Erimyzon sucetta	12	11	1	1	11	23
Esox niger	0	1	0	0	0	0
Ictalurus punctatus	0	0	0	0	1	0
Lepisosteus platyrhincus	14	3	7	19	30	18
Lepomis gulosus	151	12	4	90	166	58
Lepomis macrochirus	181	16	16	57	34	69
Lepomis microlophus	57	1	1	10	2	1
Lepomis punctatus	10	0	0	5	16	4
Micropterus salmoides	12	1	9	2	16	25
Oreochromis aureus	3	0	0	0	0	2
Pomoxis nigromaculatus	110	17	0	7	25	3
TOTAL	566	70	42	202	306	215
1998	ICS	PM	RSH	OX13	MB	MAC
Amia calva	3	3	5	6	5	3
Ameirus nebulosus	20	15	2	21	12	23
Clarias batrachus	0	1	0	2	5	4
Erimyzon sucetta	1	0	1	3	2	1
Esox niger	0	0	0	0	1	0
Hoplosternum littorale	0	0	2	0	0	1
Lepisosteus osseus	1	0	0	0	0	0
Lepisosteus platyrhincus	35	17	16	3	5	19
Lepomis guolosis	76	11	15	11	53	46
Lepomis macrochirus	120	14	11	34	57	58
Lepomis microlophus	12	2	5	1	10	4
Lepomis punctatus	18	0	0	0	1	4
Micropterus salmoides	5	0	0	0	2	1
Pomoxis nigromaculatus	20	2	1	1	3	5
TOTAL	311	65	58	82	156	169

ICS = Ice Cream Slough, RSH = Rattlesnake Hammock Run, PM = Persimmon Mound Run, Ox-13 = Oxbow 13 Run, MB = Micco Bluff Run, MAC = MacArthur Run.

Mean seasonal biomass was not significantly different between wet and dry seasons in each run except Ice Cream Slough, which was significantly higher during the dry season (ANOVA; $p = 0.0146$). Therefore, data could not be pooled for comparisons between sites. Total biomass estimates were distributed similarly among fisheries categories at both Impact (centrarchids 31%, lepisostids and amiids 28%, and ictalurids 38%) and Control (centrarchids 28%, lepisostids and amiids 36%, ictalurids 36%) sites.

Table 13-10. Mean densities of fishes collected in block net samples at Control and Impact sites. (A) Mean sample densities (± SE) of fishes collected at Control and Impact sites (n=3 at both sites for both years). (B) Mean annual density (± SE) at Control and Impact sites.

YEAR	Mean Sample Density (± SE)	
A.		
	Control	Impact
Year 1	565 ± 425	602 ± 80
Year 2	360 ± 207	338 ± 68
B.		
	Control	Impact
	462 ± 144	470 ± 186

Reference Conditions

Methods

A single 0.38 acre fish sample was collected by FGFWFC in July of 1957 using block nets and 5% emulsified rotenone. The exact methods used by FGFWFC are unclear. The sample area was chosen "to include boils, whirlpools, and eddies" found in the center of a river bend (FGFWFC 1957). The sample location also included a shallow beach area in which there was a backward movement of current.

Results

Pre-channelized river channels contained 26 freshwater fish taxa belonging to 12 families (Table 13-17). Ictalurids (61.1%) dominated community composition, but small-bodied species (28.9%), centrarchids (8.8%), and catostomids (1.2%) also were present. Density of fishes within the river channel was 937 fish/0.2 ha; however, severe drought conditions occurring the previous year may have affected fish density through stress-related mortality, or alternatively, by leading to downstream emigration into Lake Okeechobee.

Discussion

Based on results of baseline block net sampling, density of river channel fishes appears to have declined by approximately 50% since channelization. Pre-channelization data indicate a density of 937 fish per 0.2 ha (FGFWFC 1957), while samples from Impact and Control sites yielded a mean of 462 and 470 fish per 0.2 ha, respectively. Results of hoopnet sampling suggest fish density and biomass vary seasonally with greater mean abundance and biomass during the dry season. This trend might be expected in river systems with a seasonal river channel-floodplain linkage because densities within the main channel are likely to decrease during the wet season as riverine species migrate onto inundated floodplain to exploit temporarily abundant resources (Welcomme 1979, Bayley 1991). However, floodplain habitats on the channelized Kissimmee were not available (or available on a very limited basis) to fishes during the baseline period. Instead, fishes may have responded to seasonal differences in dissolved oxygen concentrations within river channels. Mean dissolved oxygen concentrations in river channels were greater during the dry season (1.9–3.7 mg/L) than the wet season (0.8–1.4 mg/L) (Colangelo and Jones 2001). Fishes may have dispersed throughout the system during the wet season to minimize constraints of low levels of dissolved oxygen.

Mean annual density of river channel fish is expected to increase following restoration. Increases in densities of large-bodied fishes require restoration and maintenance of riverine habitats that match the habitat requirements of the pre-channelized community (Sheldon and Meffe 1995). Restoration of pre-channelization hydrologic characteristics, especially river channel/floodplain connectivity, will be the mechanism driving restoration of the river channel fish community. Increased export of vertebrate and invertebrate biomass from the floodplain to the river channel during the receding hydrograph should supplement fish diets (Welcomme 1979, Harris et al. 1995), thereby increasing growth and reproductive rates of most river channel species. The availability of protective floodplain habitats should lead to increased survivorship and recruitment of juveniles into breeding populations.

Table 13-11. Total biomass (g/0.4 acre) of fishes collected per river run during 1997 and 1998 baseline block net sampling.

SPECIES	CONTROL			IMPACT		
1997	ICS	PM	RSH	OX13	MB	MAC
Amia calva	11889	9017	3121	4130	1456	823
Ameiurus natalis	260	0	0	0	0	0
Ameiurus nebulosus	4341	755	0	2752	1008	3684
Dorosoma cepedianum	0	0	0	0	0	806
Erimyzon sucetta	3458	577	455	595	83	218
Esox niger	0	474	0	0	0	0
Ictalurus punctatus	0	0	0	0	838	0
Lepisosteus platyrhincus	9116	1328	1885	3121	7611	4848
Lepomis gulosus	1590	135	42	2032	2116	1326
Lepomis macrochirus	10028	2468	1203	3916	2370	5162
Lepomis microlophus	2002	258	67	1018	120	210
Lepomis punctatus	415	0	0	205	865	308
Micropterus salmoides	2075	429	3412	367	3321	722
Oreochromis aureus	5045	0	0	0	0	2383
Pomoxis nigromaculatus	7851	37	0	31	254	244
TOTAL	58078	15478	10185	18167	20042	20734
1998	ICS	PM	RSH	OX13	MB	MAC
Amia calva	4167	3743	4544	2690	2840	1911
Ameirus nebulosus	14257	7911	1475	9559	3494	10950
Clarias batrachus	0	200	0	848	694	1108
Erimyzon sucetta	104	0	460	1970	1101	159
Esox niger	0	0	0	0	524	0
Hoplosternum littorale	0	0	230	0	0	63
Lepisosteus osseus	4	0	0	0	0	0
Lepisosteus platyrhincus	7222	7037	7606	567	2354	6021
Lepomis guolosis	2677	601	567	352	2064	1408
Lepomis macrochirus	10812	1671	440	2841	6699	3682
Lepomis microlophus	591	575	98	119	519	371
Lepomis punctatus	941	0	0	0	39	158
Micropterus salmoides	1837	0	0	0	893	181
Pomoxis nigromaculatus	2144	396	33	75	383	1275
TOTAL	44756	22134	15453	19021	21604	27359

ICS = Ice Cream Slough, RSH = Rattlesnake Hammock Run, PM = Persimmon Mound Run, Ox-13 = Oxbow 13 Run, MB = Micco Bluff Run, MAC = MacArthur Run.

Relative abundance of small-bodied fishes was similar under baseline and pre-channelization conditions (pre-channelization 28.9%; baseline - Control 20.4%, Impact 33.1%). In the pre-channelization system, small-bodied fish composition was dominated by taillight shinner *Notropis maculatus* (73.8%), eastern mosquitofish *Gambusia holbrook* (8.2%), tadpole madtom *Noturus gyrinus* (5.5%), and golden shinner *Notemigonous crysoleucas* (4.8%). Mosquitofish was dominant at Impact sites (51.1%), while golden shiner was dominant at Control sites (70.6%) and abundant at Impact sites (35.6%) during baseline conditions. Dominance of mosquitofish at Impact sites likely is attributable to increased vegetative cover and decreased water quality within the channelized system.

Table 13-12. Mean biomass of fishes collected in block net samples at Control and Impact sites. (A) Annual mean biomass (± SE) of fishes collected at Control and Impact sites with block net sampling (n=3 at both sites for both years). (B) Mean biomass (± SE) at Control and Impact sites over both years of block net sampling (n=6 at both sites).

A.	Mean Sample Biomass (± SE)	
	Control	Impact
Year 1	27,918 ± 15,160	19,652 ± 767
Year 2	27,452 ± 8866	25,852 ± 3565

B.	Mean Annual Biomass (± SE)	
	Control	Impact
	27,685 ± 7854	22,738 ± 2136

Although relative abundance of small-bodied fishes was similar between baseline and pre-channelization conditions, differences in density could not be evaluated due to different sampling methods. Electrofishing does not estimate the number of fish per unit area, but provides an estimate of catch per unit effort. Electrofishing also has an inherent bias for larger fishes, and may not have provided a complete inventory of smaller individuals, including small-bodied fish species.

Species richness is not expected to change significantly following restoration. The number of large-bodied species inhabiting the Kissimmee is consistent with distributions of fishes occurring in other rivers of pennisular Florida, including the Peace, Caloosahatchie, Manatee, Alafia, Hillsborough, and Withlacoochee Rivers (Trexler 1995). Species richness of large-bodied fishes has increased since channelization due to the introduction of the following exotic species: walking catfish *Clarias batrachus*, Oscar *Astronotus ocellatus*, blue tilapia *Oreochromis aureus*, and most recently, brown hoplo *Hoplosternum littorale* in 1997 and suckermouth catfish *Pterygoplichthyes disjunctivus* in 2001. New exotic species may become established within the Kissimmee River over the next 20 years, as they work their way through the interconnected waterways of south and central Florida. Recolonization by species believed to be extirpated from the system (Perrin et al. 1982) may occur if restored conditions are amenable and a source population has access to the basin.

Small-bodied fish relative abundance likely will be higher in restored river channels due to increased production on the floodplain and subsequent transport to the river channel. Forage fish inhabited both river channel and floodplain habitats in the pre-channelization system. Forage fishes are particularly important components in the piscine food web and are a primary food item of large piscivorous species. Most piscivorous fishes undergo an ontogenetic shift from a diet of invertebrates to fishes. Fishes able to make this shift earlier exhibit faster growth rates, higher overwinter survival, and greater reproductive success (Mittlebach and Persson 1998).

Post-restoration evaluation of river channel fishes will be conducted using electrofish and hoopnet sampling; however, block net sampling will be eliminated. Block net sampling is costly, time consuming, and not amenable to high temporal and spatial replication without negatively impacting the fish population. Also, this method is permitted by the Florida Fish and Wildlife Conservation Commission only in non-flowing waters. No-flow conditions are not likely to occur within the river following restoration. All sampling will be initiated two years subsequent to initiation of the revised headwaters regulation schedule, which will provide continuous flow through the restored river channel. Electrofishing will be conducted annually for three consecutive years following two years of continuous flow within Impact sites, and will begin on the third and eighth 8th years.

Table 13-13. Total number of fishes collected during baseline hoopnet collections at Control and Impact sites in the Kissimmee River under channelized conditions.

			Control	Impact
Lepisosteidae (gars)				
Lepisosteus platyrhincus	Florida gar		8	6
Amiidae (bowfins)				
Amia calva	bowfin		4	15
Clupeidae (herrings)				
Dorosoma cepedianum	gizzard shad		14	4
Esocidae (pikes)				
Esox niger	chain pickerel		1	2
Catostomidae (suckers)				
Erimyzon sucetta	lake chubsucker		31	39
Iclaluridae (catfishes)				
Ameiurus natalis	yellow bullhead		2	1
Ameiurus nebulosus	brown bullhead		112	130
Callichthyidae (armored catfishes)				
Hoplosternum littorale	brown hoplo		4	4
Centrarchidae (sunfishes and basses)				
Lepomis gulosus	warmouth		3	3
Lepomis macrochirus	bluegill		199	212
Lepomis microlophus	redear sunfish		37	59
Lepomis punctatus	spotted sunfish		3	4
Micropterus salmoides	largemouth bass		18	20
Pomoxis nigromaculatus	black crappie		79	85
Cichlidae (cichlids)				
Oreochromis aureus	blue tilapia		3	3
		TOTAL	518	581

Analysis of condition indices and growth rates might be useful in detecting restoration associated change and may be incorporated as a metrics for post-restoration evaluation. Growth exponent b and growth rates were determined under baseline conditions by Arrington and Jepsen (2001). Growth exponent b measures length-weight relationships in fishes and provides information on the relative health or "plumpness" of fishes. Growth rates were determined using linear and von Bertalanffy growth function relationships of standard length on age (determined through otolith analysis).

III. CREEL SURVEYS

Baseline Condition

Methods

Estimates of angler effort and success were evaluated for the Kissmmee River/C-38 system by the Florida Game and Fresh Water Fish Commission (FGFWFC) for the period of March 1992 through March 1994 via stratified roving creel surveys with non-uniform sampling probabilities (FGFWFC 1994). The Kissimmee River was divided into three units on the basis of access and time required to survey each unit. Pools A and B, Pools C and D, and Pool E were treated as individual units (Figure 13-1). Fishing success in each unit was assumed to be equal. Proportional fishing effort in each unit, and for each month, were estimated from a year-round aerial survey of boats in the channelized system.

Table 13-14. Mean seasonal abundances (± SE) of fishes in hoopnet samples from river channels at Control and Impact sites.

Control		Impact	
		Dry Season	
ICS (n = 12)	3.25 ± 0.69	Ox-13 (n = 15)	4.86 ± 1.01
RSH (n = 12)	3.40 ± 0.83	MB (n = 15)	6.53 ± 3.28
PM (n = 12)	7.40 ± 3.63	Mac (n = 15)	3.60 ± 1.00
		Wet Season	
ICS (n = 18)	1.05 ± 0.24	Ox-13 (n = 18)	3.16 ± 0.88
RSH (n = 18)	1.72 ± 0.23	MB (n = 18)	2.61 ± 0.58
PM (n = 18)	1.27 ± 0.17	Mac (n = 18)	1.27 ± 0.13

ICS = Ice Cream Slough, RSH = Rattlesnake Hammock Run, PM = Persimmon Mound Run, Ox-13 = Oxbow 13 Run, MB = Micco Bluff Run, MAC = MacArthur Run.

Table 13-15. Mean seasonal abundances (± SE) of fishes in hoopnet samples from river channels (pooled data) at Control and Impact sites.

	Control	Impact
Dry Season	4.78 ± 1.35 n = 42	5.00 ± 1.17 n = 45
Wet Season	1.35 ± 0.13 n = 54	2.35 ± 0.36 n = 54

Twenty-six contiguous periods consisting of one weekday sample and one weekend sample were scheduled during each year. Peak sampling intensity was scheduled during the months of peak fishing effort (June through November) and a minimum of two samples per month were scheduled during the months of least fishing pressure. The starting point of each sample (north and south end of the unit), the order of creel tasks (instantaneous count of anglers or angler interviews), and the actual date of sampling (one weekday sample and one weekend sample per period) were chosen with uniform probability. The unit and time of date to be sampled were chosen randomly with non-uniform probability. The probability of selecting a unit to be sampled was based on the estimated proportional fishing effort in that unit, and the probability of selecting the time of day was 0.60 for the a.m. period and 0.40 for the p.m. period based on the proportional amount of fishing expected during each time period. The sample units were divided into two areas, remnant river channels and the C-38 canal. Instantaneous angler counts were conducted by boat within C-38 and remnant river runs longer than 0.8 km. For angler interviews, hours fished for all species, hours fished for particular species, and catch were recorded.

Reported values for fishing effort and success come directly from a FGFWFC completion report (FGFWFC 1994), which did not provide raw data. Fishing effort and success were determined for C-38 canal, remnant river runs, and both areas combined. Also, results by sampling unit were not provided, but instead were reported as overall values. Therefore, differences between units could not be determined. Annual estimates of effort and success are presented with corresponding percent coefficient of variation as compiled in FGFWFC (1994). Species categories include largemouth bass, black crappie, sunfishes (*L. gulosus, L. macrochirus, L. microlophus, and L. punctatus*), catfish (*A. catus, A. natalis, A. nebulosus, and I. punctatus*), and general fish.

Table 13-16. Mean seasonal biomass (± SE) of fishes in hoopnet samples from river channels at Control and Impact sites.

Control		Impact	
	Dry Season		
ICS (n = 12)	1377 ± 355	Ox-13 (n = 15)	2135 ± 488
RSH (n = 12)	983 ± 294	MB (n = 15)	2837 ± 1742
PM (n = 12)	2773 ± 1398	Mac (n = 15)	1473 ± 590
	Wet Season		
ICS (n = 18)	462 ± 1623	Ox-13 (n = 18)	1214 ± 495
RSH (n = 18)	694 ± 242	MB (n = 18)	972 ± 223
PM (n = 18)	268 ± 84	Mac (n = 18)	442 ± 171

ICS = Ice Cream Slough, RSH = Rattlesnake Hammock Run, PM = Persimmon Mound Run, Ox-13 = Oxbow 13 Run, MB = Micco Bluff Run, MAC = MacArthur Run.

Results

Total fishing effort over the period of study was 284,160 hours, and 292,188 fish were caught. Largemouth bass was the most sought after species. Total estimated effort for largemouth bass was 101,527 hours and comprised 35.7% of fishing effort. Sunfishes (30.8%) were the next most sought after group, followed by black crappie (18.5%), and catfish (5.6%). The remainder of effort (9.4%) targeted general fish. Catch rate was highest for sunfishes (1.86 fish/hour), followed by black crappie (0.79 fish/hour) and catfish (0.48 fish/hour), and was lowest for largemouth bass (0.36 fish/hour) (Table 13-18).

Table 13-17. Numbers and percent composition of fishes collected by GFC (1957) in the historic river channel using block nets and 5% emulsified rotenone. Fishes present in historic river channel, but collected using other methods, are represented with an asterisk (*) and were not used to generate percent composition.

		Number Collected	Percent Composition
Large-bodied Taxa			
Catostomidae			
Erimyzon sucetta	lake chubsucker	6	1.2
Centrarchidae			
Micropterus salmoides	largemouth bass	6	1.2
**Lepomis auritus*	redreast		
Lepomis macrochirus	bluegill	26	5.2
Lepomis gulosus	warmouth	3	0.6
Lepomis microlophus	readear sunfish	9	1.8
**Pomoxis nigromaculatus*	black crappie		
Clupeidae			
**Dorosoma cepedianum*	gizzard shad		
Ictaluridae			
Ameiurus catus	white catfish	3	0.6
**Ameiurus natalis*	yellow bullhead		
Ameiurus nebulosus	brown bullhead	3	0.6
Ictalurus punctatus	channel catfish	300	59.9
Small-bodied Taxa			
Atherinidae			
Labidesthes sp.	silverside	3	0.6
**Menidia beryllina*	inland silverside		
Clupeidae			
Dorosoma petenense	threadfin shad	1	0.2
Cyprinidae			
Notemigonous crysoleucas	golden shiner	7	1.4
Notropis maculatus	tailight shiner	107	21.3
**Notropis petersoni*	coastal shiner		
**Opsopoedus emilidae*	pugnose minnow		
Cyprinodontidae			
**Jordanella floridae*	flagfish		
Fundulidae			
**Fundulus chrysostus*	golden topminnow		
Fundulus seminolis	seminole killifish	3	0.6
Lucania goodei	bluefin killifish	1	0.2
Ictaluridae			
Noturus gyrinus	tadpole madtom	8	1.6
Percidae			
Etheostoma fusiforme	swamp darter	3	0.6
Poeciliidae			
Gambusia holbrooki	eastern mosquitofish	12	2.4
	TOTAL	500	100

Reference Condition

Methods

Estimates of angler effort and success were evaluated for the Kissimmee River by the U. S. Fish and Wildlife Service (USFWS) from June 1955 through May 1956. Fishing pressure on the Kissimmee River was determined from fishing camp records of boats rented and private boats launched. For survey purposes, the river was divided into three areas. The "upper river" included the stretch north of Dougherty Dike (exact location unknown) and Lake Kissimmee. The "middle river" included the stretch from Dougherty Dike south to Highway 70. The "lower river" included Highway 70 south to Lake Okeechobee, excluding Government Cut. Several stations in each river section were to be creeled one day each month over the study period. However, due to manpower limitations and extremely low water levels caused by severe drought, surveys were conducted on only nine dates. Survey stations were creeled by boat and each fisherman was interviewed by asking the following questions: (1) catch, (2) time fished, (3) target species, and (4) reason for choosing fishing location. Survey data is reported as percentage of total fishing effort by taxa and catch rate (number of fish/hour) by taxa.

Results

An estimated 17,066 anglers fished the lower and middle Kissimmee River during the survey period. This estimate accounted for 22% of the total fishing effort in the Kissimmee Basin. Also, the observed estimate is considered to be conservative due to limited angler access and negative angler success resulting from severe drought. Interviews with camp operators indicated that fishing pressure was off approximately 50% from the previous year.

Table 13-18. Recent fishing effort and catch rates from creel surveys conducted under pre-channelization and baseline conditions. (* Denotes estimated angler effort for largemouth bass under pre-channelization conditions when not affected by severe drought).

	Pre-channelization Survey		Baseline Survey	
	% effort	Catch rate	% effort	Catch rate
Largemouth bass	56 (75*)	0.21	35.7	0.36
Sunfish	17	0.79	31.8	1.96
Black crappie	11	0.95	18.5	0.79
Catfish	---	---	5.6	0.48
General fish	16	0.66	9.4	---

Creel data indicated that largemouth bass (56%) was the taxa most targeted by anglers (Table 13-18). Sunfishes (17%) were the next most sought after group, followed by black crappie (11%). The remainder of effort (16%) did not target individual taxa and is described as general fish. Catch rate was highest for black crappie (0.95 fish/hour), followed by sunfishes (0.79 fish/hour) and general fish (0.66 fish/hour), and was lowest for largemouth bass (0.21 fish/hour)(Table 13-18).

Comparisons and Discussion

Largemouth bass *Micropterus salmoides* was the most sought after species (56%) prior to channelization (USFWS 1959). However, this estimate is considered conservative because the survey was conducted during a severe drought, when fishing pressure was reduced by 50% from the previous year (USFWS 1959). It was estimated that greater than 75% of the total fishing effort would be directed at largemouth bass during normal water conditions. Actual fishing pressure on the river is underestimated because fishing effort in the river portion of the upper river segment could not be separated from fishing effort on Lake Kissimmee. The catch rate for largemouth bass was considered to be an all time low for the river during the pre-channelization survey period because of the severity of the drought. Because largemouth bass catch rates declined during the drought, it is believed many anglers switched their effort to more easily caught sunfishes. Most effort for black crappie was expended in the lower portion of the river during the spawning migration when large concentrations of crappie moved from Lake Okeechobee into the Kissimmee River.

Comparisons of pre-channelization and baseline creel data suggest that the focus of angling effort has changed dramatically. Most angling effort expended in the channelized Kissimmee River system was equally focused on largemouth bass and sunfishes, whereas over 50% (and possibly as much as 75%) of effort was directed at largemouth bass prior to channelization. The primary focus on largemouth bass prior to channelization is believed to be a result of the river's reputation for producing many exceptionally large individuals (Miller 1988). Comparisons of catch rates for bass under pre-channelized and baseline conditions are suspect, since catch rates in the pre-channelization study were greatly reduced as a result of extreme drought conditions. The trend of increased angler success for sunfishes following channelization reflects their concurrent increase in relative abundance, and demonstrates increased populations of adult, harvestable fish. Conversely, low catch rates of largemouth bass and black crappie indicate decreased populations of adult, harvestable fish.

A restoration expectation was not derived for angler effort and success since pre-channelization data were negatively impacted by extreme drought and do not reflect typical conditions. Also, angler effort is contingent on numerous factors other than reestablishment of ecological integrity to the river system and, therefore, is not suitable for use as an indicator of restoration success. Post-restoration evaluation of angler effort and success will be conducted using baseline methods. A three-year creel investigation will commence on the second year following implementation of the Headwaters Revitalization Schedule.

IV. FISH DIETS

Methods

Fish feeding habits were studied by examining gut contents of nine fish taxa that were selected based on trophic categories and included *Micropterus salmoides* (piscivore), *Lepomis gulosus* (invertivore/piscivore), *Lepisosteus platyrhincus* (piscivore), *L. machrochirus* (omnivore), *L. microlophus* (invertivore), *Pomoxis nigromaculatus* (invertivore/piscivore), *Erimyzon sucetta* (invertivore), *Notemigonus chrysoleucas* (omnivore), and *Gambusia holbrooki* (omnivore). Fishes were collected in and around littoral vegetation of remnant channels and C-38 canal in Pools A, B, and C using boat-mounted electrofishing gear to determine if location affected fish diets. Sample locations within each pool were selected by driving a randomly determined number of seconds at 2000 rpm from a randomly chosen point on C-38 or remnant river run. Fishes were collected during daytime hours in both winter (December 1996 and January 1997) and summer (June 1997) to include a range of environmental conditions. Fishes were placed in a mixture of ice and fresh water to arrest metabolism. In the field, standard length of fishes >100 mm was measured to the nearest mm. Stomachs were removed and preserved in buffered formalin. Smaller fishes were preserved whole in buffered formalin and returned to the laboratory for identification, measuring, and removal of stomach contents.

For large fishes, stomach contents were rinsed, separated by prey type into individual aluminum tins, and dried for 24 h at 100°C. For small fishes, stomach contents were rinsed through a series of nested sieves (0.500, 0.250, 0.150, 0.075 mm) with distilled water to sort prey items into different size categories (Livingston 1982, 1984, 1988). A 0.5 ml sub-sample from each sieve fraction was then examined under a dissecting microscope to identify prey to the lowest taxonomic level possible, and calculate relative abundance. The contents of each sieve fraction were placed into separate aluminum tins and dried for 24 h at 100°C.

For large fishes, relative prey biomass was calculated by dividing dry weights of prey species by the total dry weight of stomach contents. For small fishes, dry weights were multiplied by prey relative abundance to calculate relative prey weight for each sieve fraction. Analysis of prey data by absolute or relative weight is preferred (Wallace 1981). Data for large predators were categorized in the 0.500 mm sieve fraction. However, data across size classes were pooled to simplify interpretation.

The metrics analyzed were number of prey types eaten (prey richness) and prey weight. Analysis of variance (ANOVA) was used to test for the effect of sample location on species prey weight and relative abundance of fish prey. Tukey-Kramer post hoc tests were used for pairwise comparisons. Similarity in relative abundance of prey categories between pools was determined using cluster analysis. Prey categories included filamentous algae, annelids, microcrustaceans (e.g., ostracods, copepods), detritus, fishes, herpetofauna, aquatic insects, mollusks, grass shrimp, plant remains, crayfish, sand, and terrestrial arthropods. Simple linear regression was used to determine the relationship between length of predatory fishes and prey length.

Results

Number and relative abundance (%) of different prey items and prey richness for each taxa studied are listed in Appendices 13-1A to 13-9A. Predator length explained 22% of the variation in prey length (ANOVA, $F1,1879 = 5333.3$, $p = 0.0001$) (Figure 13-9). Prey weight (ANOVA; $F2,1923=45.4$, $p=0.0001$) varied significantly among pools (Figure 13-10). Post-hoc comparisons indicate that weights of prey taken by Pool B predators were greater than prey items of predators from Pools A and C. Prey quantity (number of prey items) was higher in Pool B predators after adjusting for differences in body length. Cluster analysis indicates that major prey groupings in Pools A and C are more closely related to one another than to Pool B prey (Figure 13-11). Relative abundance (%) of fish prey was not significantly different in fishes collected from Pools A, B, and C (ANOVA; $p > 0.05$). Fish prey comprised the greatest percentage of food items in the diets of Florida gar, bowfin, and largemouth bass (Figure 13-12). Grass shrimp (*Palaemonetes* sp.) dominated the diet of warmouth in Pool A, while crayfish was the dominant food item in Pool C (Figure 13-12).

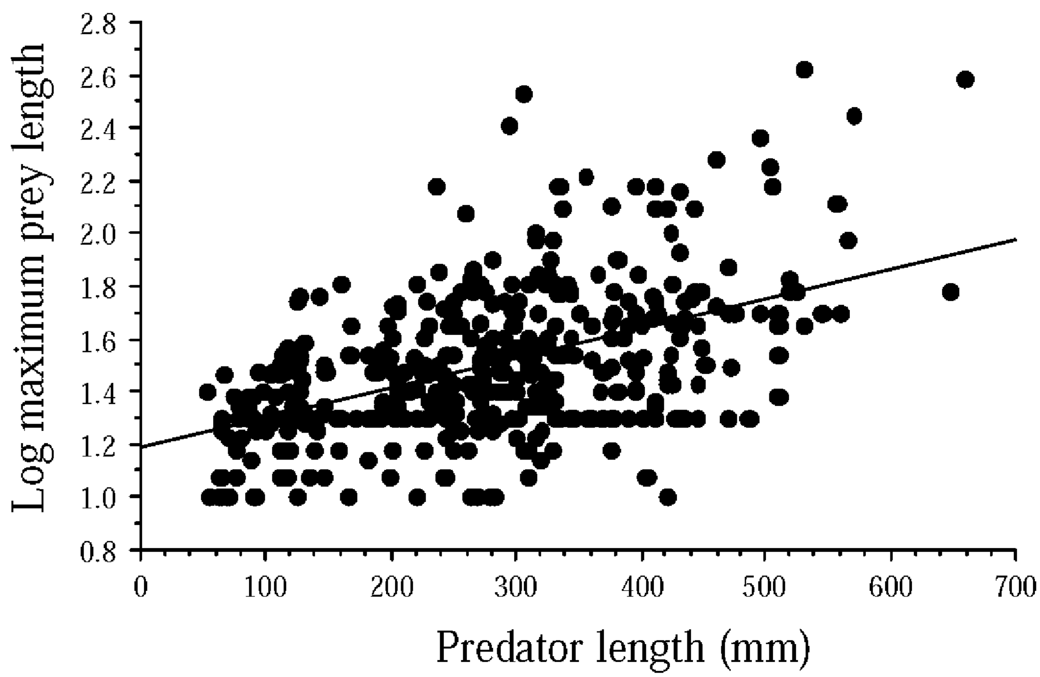


Figure 13-9. Scatterplot indicating relationship between length of fish predator and log maximum prey length. Predator length explains 24% of the variation in prey length ($F_{1,423}$= 132.1, $p<0.05$).

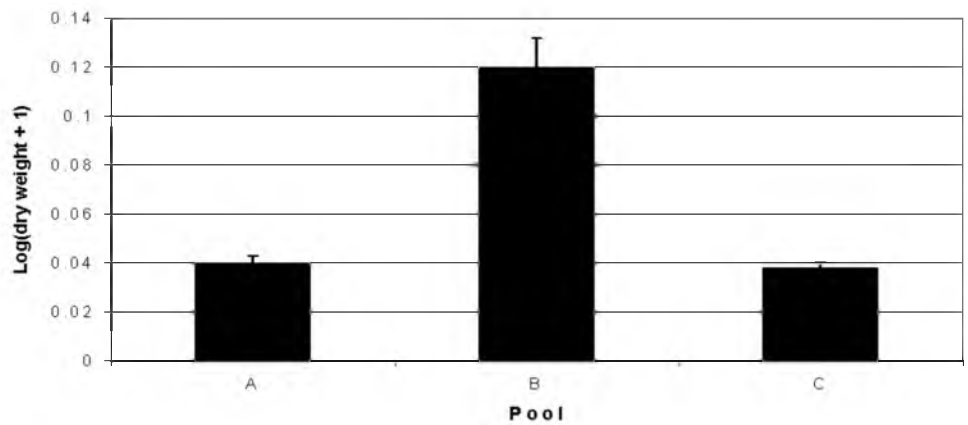


Figure 13-10. Mean (± 1 standard error) log transformed dry weight of stomach contents of predators collected from Pools A, B, and C of the Kissimmee River.

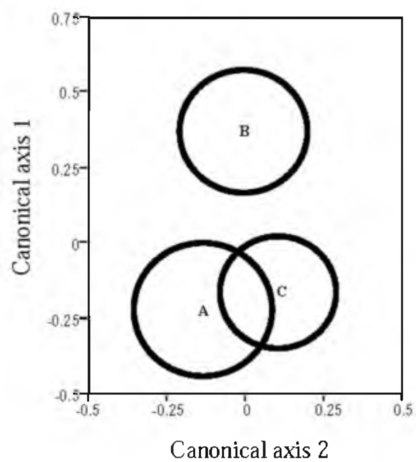


Figure 13-11. Multivariate least-squares means and 95% confidence ellipses of dominant prey types (see Table 13-4) from Pools A, B, and C of the Kissimmee River.

Discussion

Flow through three remnant river runs in Pool B was enhanced by placement of notched weirs in C-38, asscociated with the Kissimmee River Demonstration Project, located immediately downstream of the northern confluence of each river run with C-38 canal. The weirs functioned to back up water during periods of high flow, forcing water through remnant river runs and occasionally out on to limited portions of adjacent floodplain. Reintroduced flow flushed accumulated organic sediments and reduced the width of emergent vegetation along the littoral edge. Dissolved oxygen levels increased in these runs as a result of decreased sediment oxygen demand and reparation through turbulent mixing (Toth 1991). Sampled fishes in enhanced Pool B were significantly longer than their counterparts in Pools A and C, so it was not unexpected that the total weight of food in their stomachs also was greater. However, prey quantity (total number of prey items) was still higher in Pool B predators even after adjusting for differences in body length. Increased food quantity may reflect enhanced foraging opportunities that have arisen since enhancement of Pool B (Jordan and Arrington 2001).

Although the amount of prey is an important indicator of habitat quality, the types of prey available may be even more important. For example, most piscivorous fishes start life feeding on invertebrates and later undergo ontogenetic shifts to piscivory (Gerking 1994). Fish prey are apparently more energetically profitable than invertebrate prey, and fishes that switch to piscivory have faster growth rates, higher overwinter survival, and potentially greater reproductive success (e.g., Mittlebach and Persson 1998). Restoration of the Kissimmee River will result in increased connectivity between river channel and floodplain habitats and may result in more fish prey becoming available (Trexler 1995). Jordan and Arrington (In review) found that large predatory fishes in enhanced Pool B consumed greater proportions of fish prey. Although piscivory was mostly limited to large-bodied fishes, smaller fishes fed on scales and larvae. The amount of fish in a predator's diet reflected both taxonomy and foraging location. At least 90% of the diet of Florida gar was comprised of fishes, whereas the proportion of fishes in the diets of bowfin, warmouth, and largemouth bass varied considerably among Pools A, B, and C (Figure 13-12). Similarly, the relative importance of crayfish and grass shrimp also varied with fish species and foraging location. However, the similarity in prey community composition between Pools A and C indicates they should serve as good Control and Impact sites.

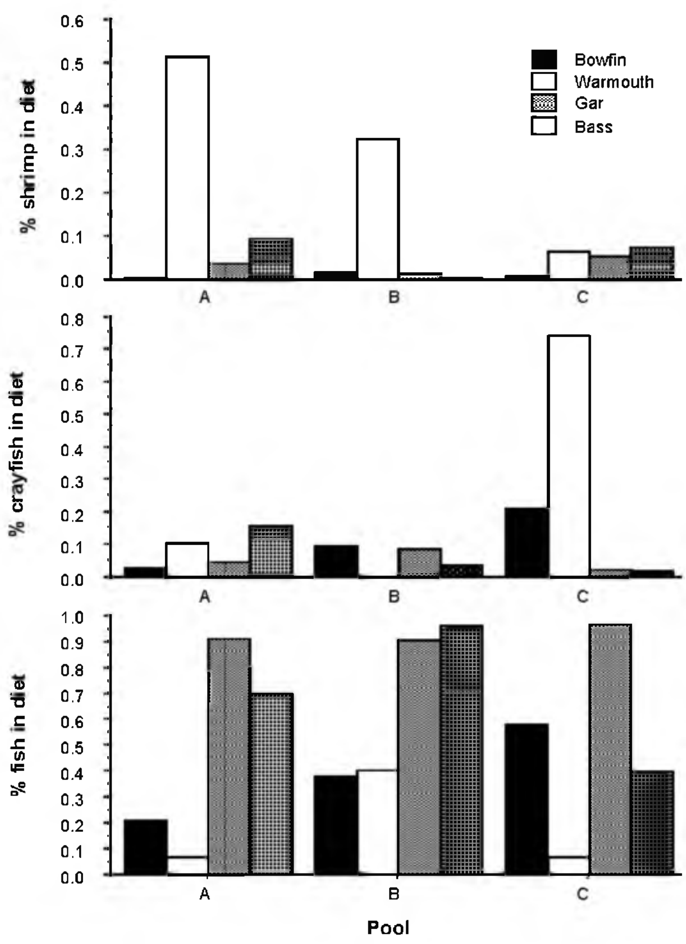


Figure 13-12. Relative abundance (%) of (a) grass shrimp, (b) crayfish, and (c) fishes in the diets of large predatory fishes collected from Pools A, B, and C of the Kissimmee River.

Analysis of fish feeding habits will be repeated during post-construction evaluations. It is expected that large predatory fishes in Pool C will consume greater proportions of fish prey than similar taxa in Pool A. Sampling should be initiated no sooner than three years following initiation of the revised Headwaters

Regulation Schedule to allow for sufficient change in river channel fish community structure and reestablishment of the aquatic food web. Ontogenetic changes in feeding habits may be an important metric to include in post-restoration analyses. A study of fish feeding habits using stable isotopes to identify energy pathways within the aquatic food web also may be incorporated during post-restoration evaluations.

V. REPRODUCTIVE EFFORT AND LARVAL FISH ASSEMBLAGE STRUCTURE

Methods

Fish larvae were sampled bi-weekly between March 11, 1997 and June 26, 1997 (eight sampling events) and between January 13, 1998 and April 8, 1998 (seven sampling events) to evaluate baseline larval fish assemblage structure within the channelized Kissimmee River. Push net sampling was conducted at fixed sites using paired, bow-mounted 505-micron plankton nets pushed just below the water's surface. Sampling effort was stratified within lower, middle, and upper zones within each pool to address the hypothesis that spawning occurs only in southern (lower) reaches of the channelized system (Trexler 1995). Two replicate samples were collected in lower, middle, and upper reaches of three remnant river runs of Pools A (Persimmon Mound Run, Kicco Run, and Ice Cream Slough Run; Figure 13-1) and C (MacArthur Run, Micco Bluff Run, and Oxbow 13; Figure 13-1). Only middle reaches of Ice Cream Slough Run and Oxbow 13 were sampled due to limited sampling area by encroachment of emergent vegetation into the center of the channel. Two replicate samples also were collected from mid-channel and littoral zones of C-38 in lower, middle, and upper regions of each pool, resulting in a total of 26 samples per pool. Mid-channel zones were sampled using replicate, side-by-side plankton nets, while littoral zones were sampled using two consecutive single net pushes.

Water quality data was collected at each site prior to sampling. Dissolved oxygen, specific conductance, pH, turbidity, and water temperature were measured using a Hydrolab™ multiprobe water quality instrument. Current velocity was measured using a Marsh-McBirney Flowmate 2000 portable flow meter. Mechanical flow meters were suspended inside each plankton net to calculate total water volume sampled. All samples were preserved in the field with 10% buffered formalin.

Fishes without a full complement of fin rays were classified as larval. Larval fish from each sample (replicate samples were not pooled) were sorted, identified to lowest possible taxonomic unit, and measured (total length) to the nearest millimeter. Species richness and relative abundance were calculated. Differences in total larval fish density within each riverine category (remnant river channel, C-38 pelagic, C-38 littoral) across three regions of each pool (lower, middle, and upper) were tested using repeated-measures analysis of variance (R-M ANOVA). Differences in longitudinal distribution of numerically significant taxa within each pool also were tested using R-M ANOVA. Differences in total larval fish density among the three riverine categories among and between pools were tested using R-M ANOVA. Correlations between larval fish density and environmental factors (levels of dissolved oxygen, pH, turbidity, water temperature, and flow rate) within each pool were tested using Pearson Correlation Coefficient (SAS Institute 1990).

Results

A total of 23 taxa were collected during the study (Table 13-19). Species richness was similar at Control ($n = 22$) and Impact sites ($n = 19$). Species richness ranged between 15 and 20 taxa in river channels and 16 and 17 taxa in C-38 (Table 13-20). However, species richness was 25% greater at river channel Control sites during 1998 than 1997 (Table 13-20). Unidentified sunfishes (*Lepomis* spp.) and shad (*Dorosoma* spp.) were numerically dominant in both pools and comprised 69.1% and 80.9% of larval fishes collected in Control and Impact pools, respectively.

Mean sample density was significantly greater during 1997 than 1998 at both Control (ANOVA; $p < 0.0001$) and Impact (ANOVA; $p = 0.0001$) sites, and was significantly greater at Control sites than Impact sites during 1997 (ANOVA; $p = 0.0059$), but not during 1998 (ANOVA; $p = 0.53$; Table 13-21). Mean sample density varied between lower, middle, and upper remnant river channels at Control (R-M ANOVA; $p = 0.0050$) and Impact (R-M ANOVA; $p = 0.0030$) sites. Densities typically were greater at lower sites (i.e., Kicco Run) in the Control pool, but consistently greater at middle sites (i.e., Micco Bluff

Run) in the Impact pool during 1997 (Figure 13-13). Mean sample density also differed between lower, middle, and upper regions for mid-channel (pelagic) C-38 sites at Control (R-M ANOVA; $p = 0.0115$) and Impact sites (R-M ANOVA; $p = 0.0092$); however, there was no clear pattern of differences among locations, nor was there any consistency between years (Figure 13-14). No difference was found in mean sample density between lower, middle, and upper regions among C-38 littoral sites at Control (R-M ANOVA; $p = 0.1631$) or Impact (R-M ANOVA; $p = 0.6595$) sites (Figure 13-15).

There also were differences in larval fish density among river channel, C-38 littoral, and C-38 pelagic sites. At lower Control sites, mean sample density was significantly greater at riverine sites (R-M ANOVA; $p = 0.0176$) (Figure 13-16). Densities also differed among site types at middle (R-M ANOVA; $p = 0.0062$) and upper (R-M ANOVA; $p = 0.0085$) locations of Control sites; however, mean sample density was lowest at riverine sites within both of these regions (Figure 13-16). Differences were not significant among site types at middle (R-M ANOVA; $p = 0.2002$) and upper (R-M ANOVA; $p = 0.1431$) regions of Impact sites. However, mean sample density was significantly greater at lower C-38 littoral sites (R-M ANOVA; $p = 0.0298$), than river channel or C-38 pelagic sites (Figure 13-17). Larval fish density was positively correlated, but not statistically significant, with water temperature; however, the degree of correlation varied among sites within pools (Table 13-22).

Table 13-19. Total numbers of larval fishes collected at Control and Impact sites.

		Control	Impact
LARGE-BODIED TAXA			
Catostomidae (suckers)			
Erimyzon sucetta	lake chubsucker	852	593
Centrarchidae (sunfishes)			
Lepomis spp.	unidentified sunfishes	24,142	24,897
Micropterus salmoides	largemouth bass	118	63
Pomoxis nigromaculatus	black crappie	2116	574
Clupeidae (herrings)			
Dorosoma spp.	unidentified shad	10,238	3974
Dorosoma cepedianum	gizzard shad	229	255
Cichlidae (cichlids)			
Oreochromis aureus	blue tilapia	14	8
Esocidae (pikes)			
Esox niger	chain pickerel	2	--
Ictaluridae (bullhead catfishes)			
Ameiurus natalis	yellow bullhead	9	1
Ameiurus nebulosus	brown bullhead	7	4
Ictalurus punctatus	channel catfish	1	--
Lepisosteidae (gars)			
Lepisosteus osseus	longnose gar	1	--
Lepisosteus platyrhincus	Florida gar	--	2
SMALL-BODIED TAXA			
Aphredoderidae (pirate perches)			
Aphredoderus sayanus	pirate perch	14	34
Atherinidae (silversides)			
Labidesthes sicculus	brook silverside	2538	1851
Belonidae (needlefishes)			
Strongylura marina	Atlantic needlefish	15	--
Cyprinidae (carps and minnows)			
Notemigonous crysoleucas	golden shiner	740	339
Notropis maculatus	tailight shiner	1992	80
Fundulidae (killifishes)			
Fundulus chrysostus	golden topminnow	31	60
Fundulus lineotus	lined topminnow	7	5
Lucania goodei	bluefin killifish	304	176
Percidae (perches)			
Etheostoma fusiforme	swamp darter	648	1443
Poeciliidae (livebearers)			
Gambusia holbrooki	eastern mosquitofish	1547	719
Heterandria formosa	least killifish	293	443
Poecilia latipinna	sailfin molly	1	9
	TOTAL	45,859	35,530

Table 13-20. Larval fish species collected in pushnet samples at Control and Impact sites. (cl = C-38 littoral, cp = C-38 pelagic, rc = river channel).

		Control Sites		Impact Sites	
		1997	1998	1997	1998
Ameiurus natalis	yellow bullhead	cl, cp, --	--, --, --	--, cp, --	--, --, --
Ameiurus nebulosus	brown bullhead	cl, cp, --	--, --, --	--, cp, rc	--, --, --
Aphredoderus sayanus	pirate perch	--, --, --	cl, cp, rc	--, --, rc	cl, cp, rc
Dorosoma spp.	unidentified shad	cl, cp, rc	--, --, --	cl, cp, rc	--, --, --
Dorosoma cepedianum	gizzard shad	--, --, --	cl, cp, rc	--, --, --	cl, cp, --
Erimyzon sucetta	lake chubsucker	--, --, --	cl, cp, rc	--, --, --	cl, cp, rc
Esox niger	chain pickerel	--, --, --	--, --, rc	--, --, --	--, --, --
Etheostoma fusiforme	swamp darter	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Fundulus chrysostus	golden topminnow	cl, cp, rc	cl, cp, rc	cl, cp, rc	--, cp, rc
Fundulus lineotus	lined topminnow	--, --, --	--, cp, rc	cl, --, --	--, --, --
Gambusia holbrooki	eastern mosquitofish	cl, cp, rc	--, cp, rc	cl, cp, rc	cl, cp, rc
Heterandria formosa	least killifish	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Ictalurus punctatus	channel catfish	--, cp, --	--, --, --	--, --, --	--, --, --
Labidesthes sicculus	brook silverside	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Lucania goodei	bluefin killifish	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Lepisosteus osseus	longnose gar	--, --, --	--, --, rc	--, --, --	--, --, --
Lepisosteus platyrhincus	Florida gar	--, --, --	--, cp, --	--, --, --	--, --, --
Lepomis spp.	unidentified sunfish	cl, cp, rc	--, cp, rc	cl, cp, rc	cl, cp, rc
Menidia berylina	inland silverside	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Micropterus salmoides	largemouth bass	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Notemigonous crysoleucas	golden shiner	cl, cp, --	cl, cp, rc	cl, cp, rc	cl, cp, rc
Notropis maculatus	tailight shiner	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Oreochromis aureus	blue tilapia	--, --, rc	--, --, rc	--, --, --	cl, cp, rc
Poecilia latipinna	sailfin molly	--, --, rc	--, --, --	cl, --, rc	--, --, --
Pomoxis nigromaculatus	black crappie	cl, cp, rc	cl, cp, rc	cl, cp, rc	cl, cp, rc
Strongylura marina	Atlantic needlefish	--, cp, rc	--, --, rc	--, --, --	--, --, --

Table 13-21. Mean annual density (± SE) of larval fishes in pushnet samples (all habitats combined) at Control and Impact sites. Values are expressed as fish/m^3.

	Control	Impact
Year 1	5.63 ± 0.71	3.03 ± 0.59
Year 2	0.60 ± 0.20	0.46 ± 0.12

Discussion

Studies on larval fish assemblages have shown that the number of species and their relative composition generally do not reflect similar attributes of adult fish communities within the same system (Holland and Sylvester 1983, Holland 1986, Turner et al. 1994, Scheidegger and Bain 1995). Early life stages that are buoyant are collected more easily by most widely used sampling methods (e.g. push nets, seines, and towed plankton nets), and dominance of these taxa may result in misrepresentation of community structure of larval fish assemblages (Holland 1986). However, of the taxa generally collected by common sampling methods, dominance of specific taxa can be used to characterize aquatic systems as either lentic or lotic, based primarily on habitat requirements of larvae (Scheidegger and Bain 1995).

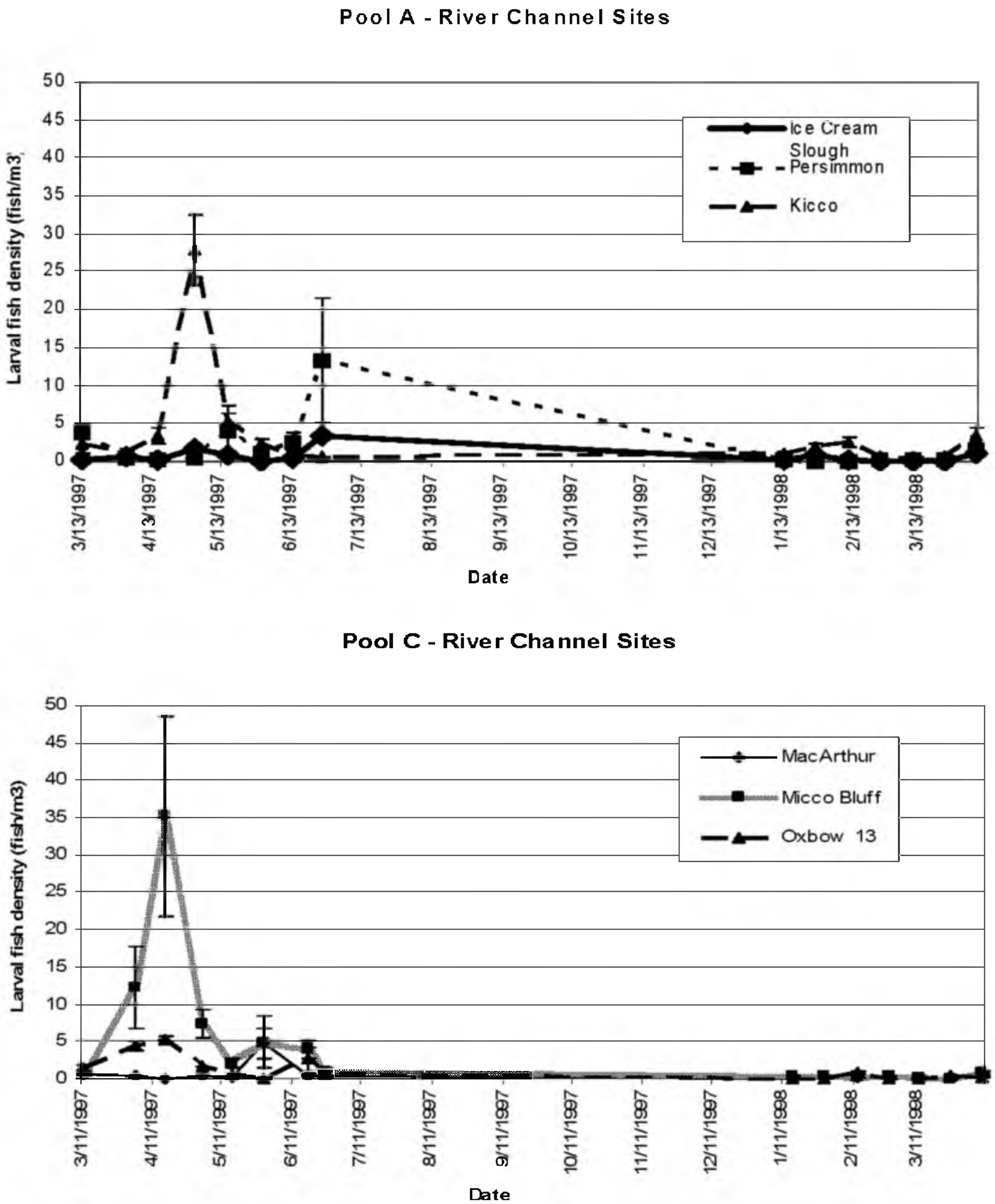
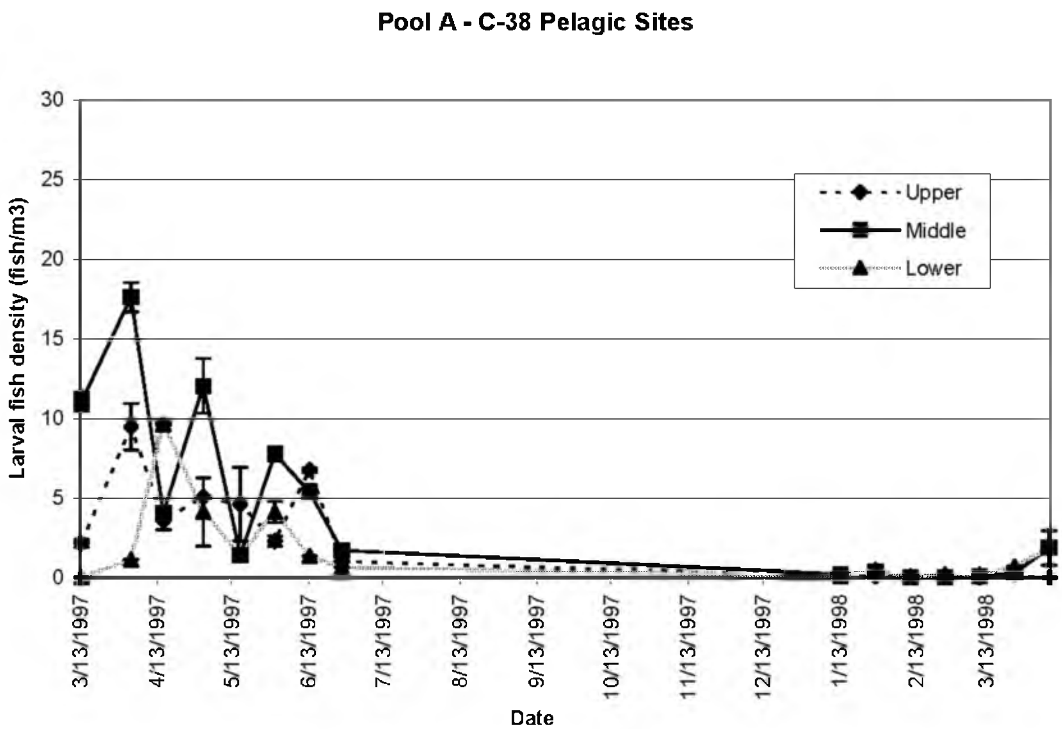


Figure 13-13. Mean larval fish density for each sampling date within remnant river channels at Control (Pool A) and Impact (Pool C) sites under channelized conditions.



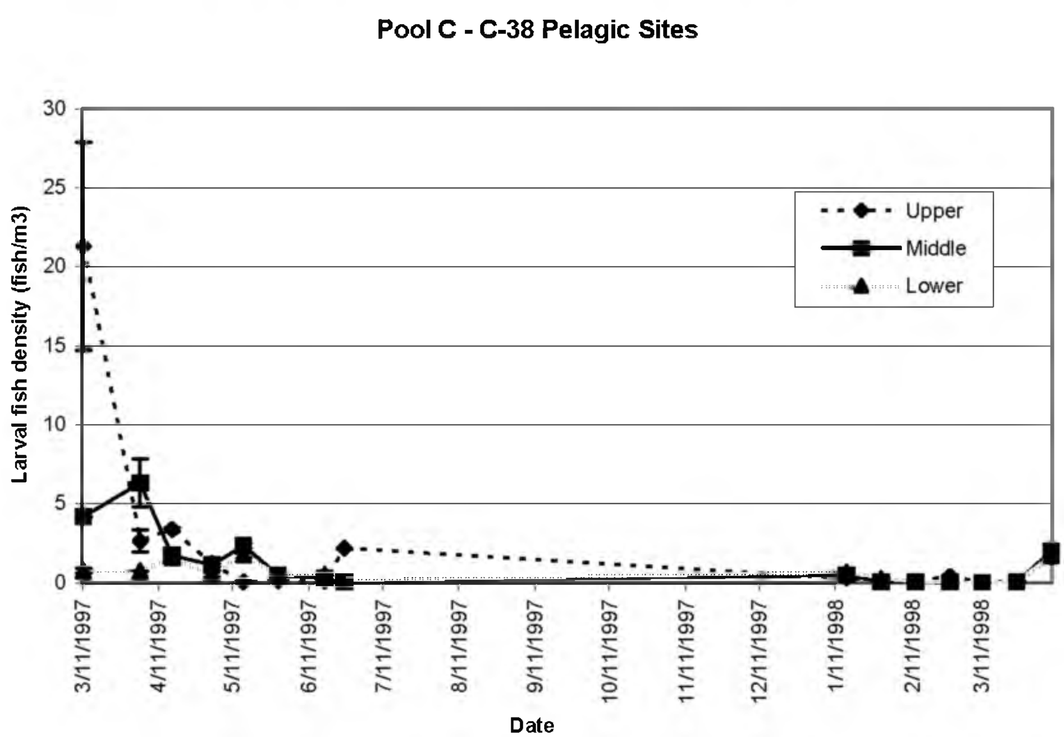
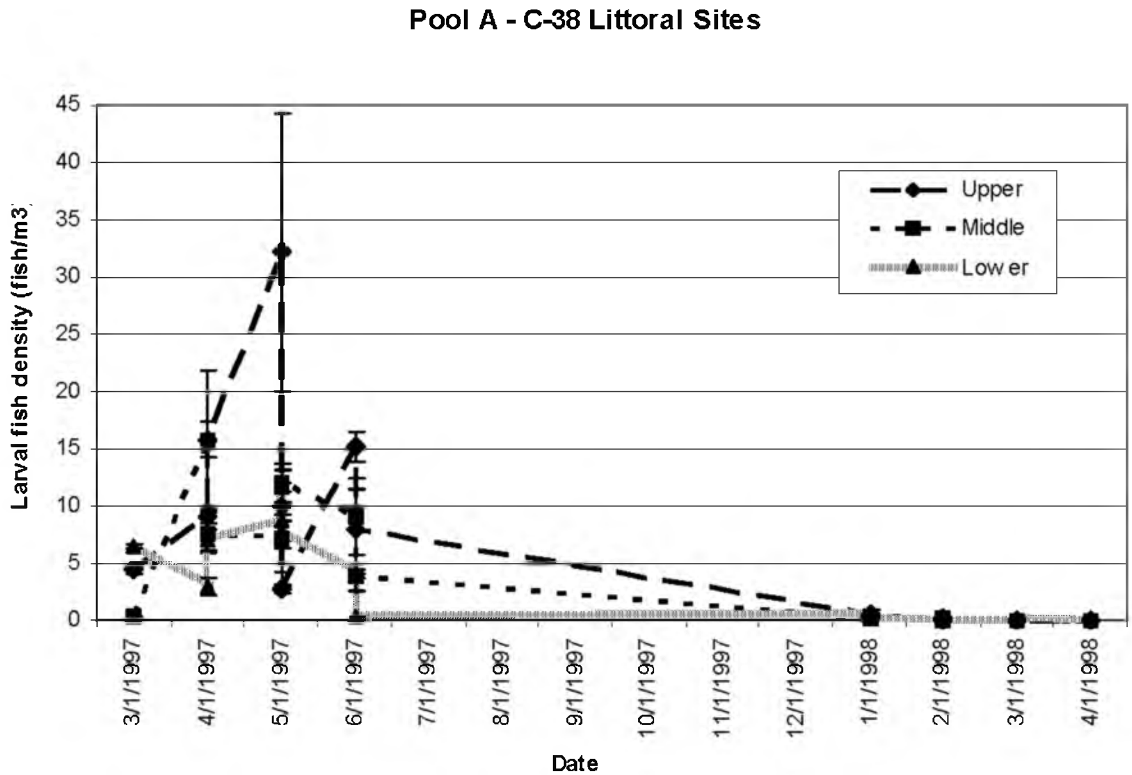


Figure 13-14. Mean larval fish density for each sampling date in upper, middle, and lower C-38 pelagic zones at Control (Pool A) and Impact (Pool C) sites under channelized conditions.



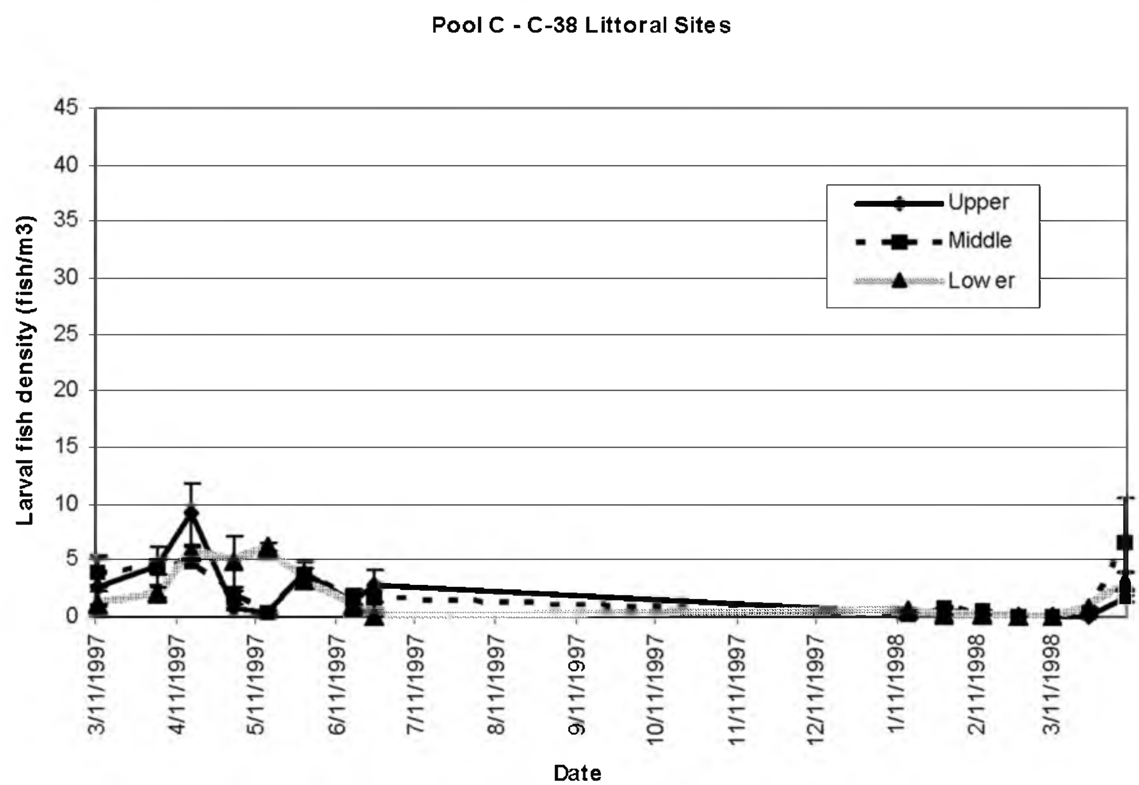


Figure 13-15. Mean larval fish density for upper, middle, and lower C-38 littoral zones at Control (Pool A) and Impact (Pool C) sites under channelized conditions.

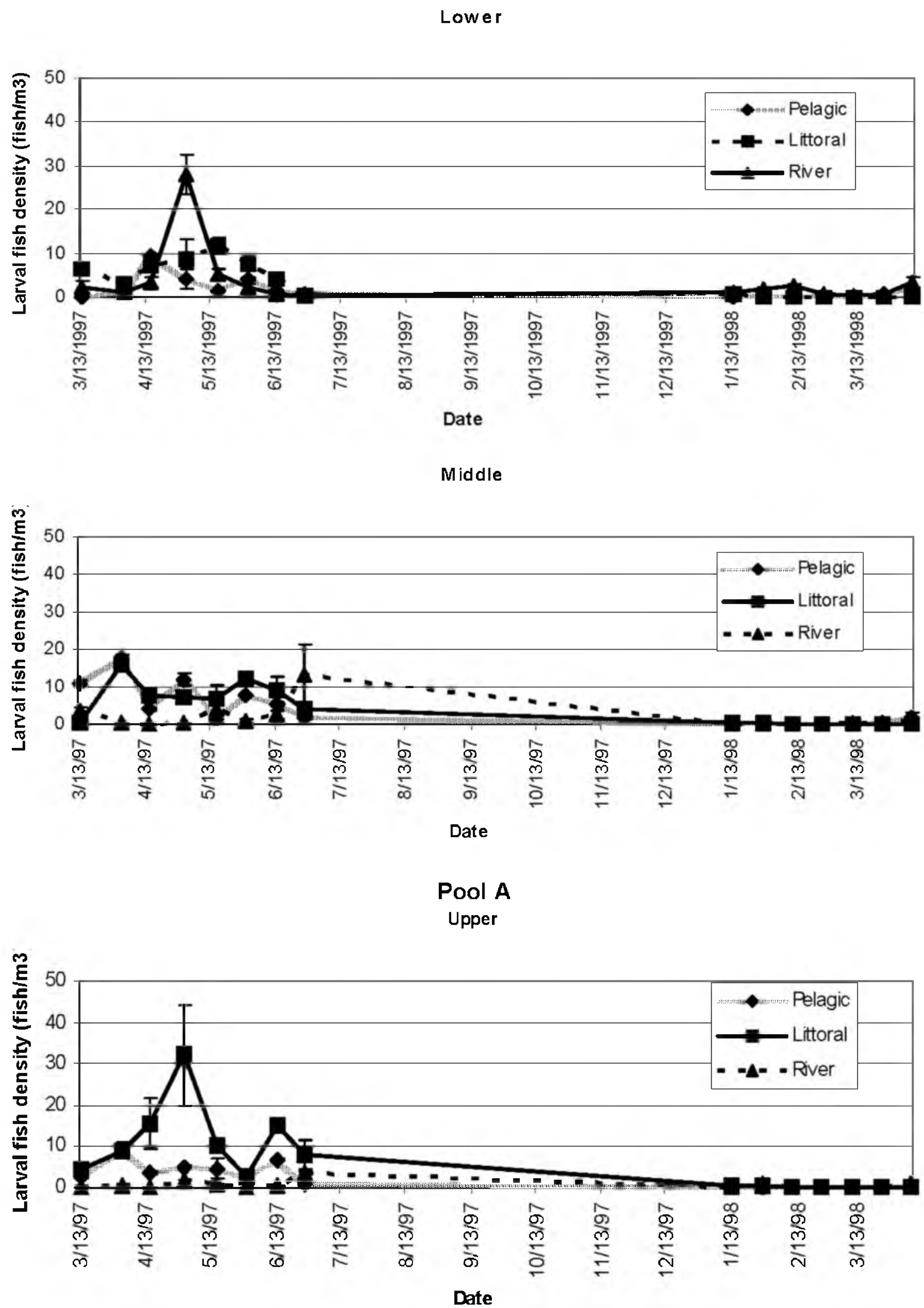


Figure 13-16. Mean larval fish density in upper, middle, and lower regions at C-38 pelagic, C-38 littoral, and river channel sites in Pool A (Control Site) under channelized conditions.

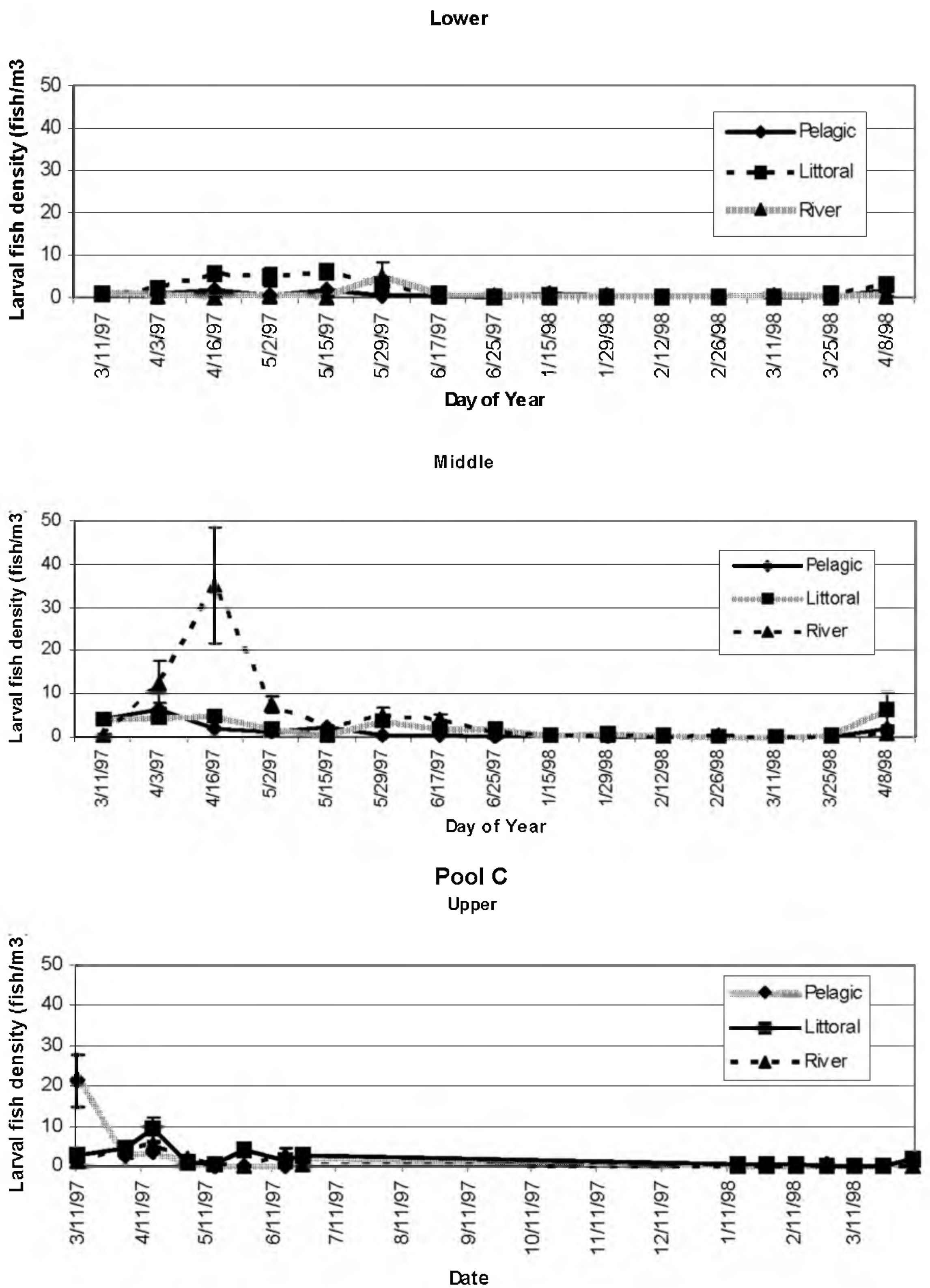


Figure 13-17. Mean larval fish density within upper, middle, and lower regions at C-39 pelagic, C-38 littoral, and river channel sites in Pool C (Impact site) under channelized conditions.

Table 13-22. Pearson's Correlation Coefficients for larval fish densities in relation to environmental variables at Control and Impact sites.

Pool A

	Combined Data	C-38 Littoral	C-38 Pelagic	ICS	Persimmon Mound	Kicco
Water temperature	.49	.74	.61	.42	.50	.14
Dissolved oxygen	.10	-.24	-.22	-.30	-.09	.05
pH	-.16	-.28	-.17	-.33	-.09	-.16
Turbidity	-.18	-.70	-.35	-.62	.40	-.57
Current velocity	-.17	.23	-.07	-.66	-.14	-.36

Pool C

	Combined Data	C-38 Littoral	C-38 Pelagic	Oxbow-13	Micco Bluff	MacArthur
Water temperature	.27	.46	.32	.26	.29	.30
Dissolved oxygen	-.12	-.31	-.17	.16	-.29	.01
pH	-.12	-.19	-.19	-.10	-.22	-.04
Turbidity	-.19	-.38	-.29	-.36	-.28	-.03
Current velocity	.08	-.41	-.25	-.57	-.08	-.17

Studies have shown that sunfishes (Lepomis spp.) and shad (Dorosoma spp.) dominate collections in aquatic systems exhibiting little to no-flow (Holland 1986, Turner et al. 1994, Scheidegger and Bain 1995). In a comparative study of larval fish assemblages in the Tallapoosa (regulated river) and Cahaba Rivers (free-flowing river), both located in Alabama, Scheidegger and Bain (1995) found sunfishes and shad were predominant in regulated reaches exhibiting little to no-flow. Conversely, they found cyprinids (carps and minnows) and catostomids (suckers), taxa more indicative of riverine conditions, were dominant in both regulated and free-flowing reaches with daily flow. Similarly, sunfishes and shad are the dominant larval taxa in backwaters of the Mississippi, Missouri, and Tallahatchie Rivers (Holland and Sylvester 1983, Holland 1986, Brown and Coon 1994, Turner et al. 1994), which are characterized by shallow depths and absence of flow. Results of larval fish studies conducted on lakes and ponds provide further evidence of larval sunfish and shad dominance as an indicator of lentic systems (Holland and Huston 1985, Conrow et al. 1990, Sabo and Kelso 1991).

The dominant larval fish taxa collected within Control and Impact pools of the channelized system were sunfishes (Lepomis spp.) and shad (Dorosoma spp.), which collectively comprised greater than 69% of all fishes collected. Dominance of sunfish larvae within the channelized system likely is attributable to lentic conditions. Dominance by sunfish larvae within the drift is expected to decrease at Impact sites following restoration due to the reestablished flow. Although sunfish larvae should remain abundant, their relative abundance should decrease due to increased abundance of more riverine taxa (e.g. silversides - Atherinidae, minnows - Cyprinidae).

Significantly lower larval fish density during the second year of study was likely due to differences in sampling periods between years. Sampling was initiated later in the year and extended further into the summer during 1997, and included an additional sampling event. The spawning season for most fish species in the Kissimmee River extends from early spring into summer months and is driven by increasing water temperature (Carlander 1969, Lee et al. 1980). Peak densities during 1997 occurred subsequent to the first week in April, when sampling concluded during 1998. Larval fish densities were greatest in both pools on the last sample date in 1998, and likely would have increased with rising water temperature as summer progressed.

Larval fish density varied along longitudinal gradients and between habitats within Control and Impact pools. However, patterns of density varied between pools. Channelization significantly decreased the amount of floodplain wetlands available to fishes for spawning (Carnal and Bousquin 2005). It is likely that areas appropriate for spawning are not uniformly distributed throughout Control and Impact sites, leading to the lack of trends in larval fish density between habitats and longitudinal zones under channelized conditions. Jurajada (1995) concluded that reduced reproduction and recruitment of 0+ (young-of-the-year) fish following channelization was primarily due to isolation of inundated floodplain from the main channel, resulting in loss of spawning habitat and refugia.

Larval fish density is expected to be greater within floodplain habitats and backwater areas following restoration. Numerous studies have shown greater larval fish density within backwater areas compared to the river channel (Holland and Sylvester 1983, Holland 1986, Brown and Coon 1994), with minimal flow being the primary regulatory factor. Densities also should be greater within the ecotone between littoral vegetation and mid-channel than within mid-channel riverine sites. Paller (1987) found greater larval fish densities within this region in Steel Creek, South Carolina and attributed it to emigration from littoral macrophyte beds, where larval fish densities were approximately 160 times greater than the river channel.

Sampling of larval fish assemblages will be conducted during post-construction evaluations. Taxa dominance appears to be a potential indicator for evaluating restoration-associated change in the system. However, a restoration expectation was not developed for this metric since suitable reference data were not available. Commencement of sampling for this study should be delayed a minimum of three years following initiation of the Headwaters Regulation Schedule to allow for sufficient changes in age structure of the river channel fish community. Increased recruitment is expected for most fish species following restoration, which will potentially increase the numbers of adults capable of spawning.

VI. FISH MOVEMENTS

Floodplains of large river systems provide essential habitat for fishes during some life history stages. Species that dominate fisheries biomass and production in river–floodplain systems depend on periodic inundation of floodplain habitats (Welcomme 1979, Bayley 1981). The extent to which riverine fishes utilize floodplain habitats in modified river–floodplain systems is determined by the magnitude of change in the flood regime (Ward and Stanford 1989). Channelization of the Kissimmee River eliminated overbank flow and severed the historic river channel–floodplain linkage (Anderson 2005). Loss of this linkage precluded river channel and floodplain fishes from exploiting resources in floodplain habitats.

Enhancements within Pool B, due to pool stage fluctuation, Kissimmee River Demonstration Project weirs, and the 1994 test-fill, produced limited areas of river channel connectivity with the floodplain (Koebel 1995). Largemouth bass and bluegill sunfish were tracked within Pool B using radiotelemetry to determine the extent of floodplain utilization within this enhanced portion of the channelized river.

Methods

Twenty-five bluegill and 12 largemouth bass were collected at random locations within Pool B between October and December 1997, fitted with radio transmitters, and released at the same locations where they were collected. Largemouth bass ranged in size from 258–508 mm (TL) and bluegill size ranged from 203–241 mm (TL). Minimum individual body mass of largemouth bass and bluegill was 321.1 g and 159.2 g, respectively, and conform with Winter's (1977) recommendation for a maximum transmitter-mass to body-mass ratio of 2%.

Fishes were tracked Monday through Friday for a 12 week period during winter 1997. Each time a fish was located, its position was recorded using GPS. Water depth and water quality data including water temperature, dissolved oxygen, pH, and specific conductance were collected at each fish location. Water depth was measured with a calibrated (10 cm units), 3 m section of PVC pipe. Fish were considered to be on the floodplain when the GPS-fixed positions fell outside the geographically referenced river channel margin and within floodplain boundaries. Floodplain habitats were available to fish throughout the study period.

Results

Based on the total number of coordinate locations (largemouth bass, n = 90 locations; bluegill, n = 68 locations), largemouth bass and bluegill were on the floodplain approximately 45% and 55% of the time, respectively. Water depths on the floodplain ranged between 0.19–2.30 m over the study period, with a mean of 1.15 m (n=75)(SFWMD DBHydro database). When present in the river channel, largemouth bass and bluegill occurred within the vegetated littoral zone 74% and 79% of the time, respectively. Open water habitats were used by each species less than 3% of the time, while channel margins with large woody debris were utilized approximately 23% and 18% of the time.

Discussion

Within the channelized Kissimmee River, floodplain and main channel littoral zone habitats may provide equivalent resources for bass and bluegill. Due to hydrologic regulation, floodplain habitats within the channelized system do not receive a seasonal flood-pulse, and therefore they do not experience the seasonal "boom" in production associated with re-inundation, so production levels are likely to be less variable and lower. Additionally, cues for lateral migration conferred during the onset of the flood-pulse are likely not present within the channelized system. In this study, largemouth bass and bluegill used inundated floodplain habitats of the Kissimmee River approximately 50% of the time. Floodplain habitat utilization by fish is expected to increase following restoration due to increases in floodplain production and areal extent and availability of floodplain habitats.

River channel/floodplain exchange will be documented in post-restoration studies with modified fyke nets, fitted with 6 mm netting. A series of paired nets will be deployed along the river channel/floodplain interface parallel to the river channel to provide data on direction of fish movement (onto/off of floodplain). Annual sampling will be conducted during the first and second years immediately following implementation of the Headwaters Regulation Schedule. Sampling will be conducted during the rising and falling legs of the hydrograph, when floodplain habitats are inundated to a minimum inundation depth of 40 cm. Sampling will be conducted in Pool C. Post-restoration radio telemetry studies will depend upon monetary constraints and staff availability. If initiated, this study will be completed in conjunction with the study mentioned above, but will be conducted in Pool B to simplify comparisons with Baseline data.

VII. MERCURY BIOACCUMULATION

Methods

Eighteen largemouth bass were collected from pools A, B, and C under channelized conditions in October of 1995 for analysis of mercury bioaccumulation and biomagnification. Mercury bioaccumulation is of societal concern, since concentration at specific levels is considered a health hazard (Wiener 1987). Fishes were collected using electrofishing gear. Sampling gear consisted of a 5.5 meter jon boat outfitted with a 5-kilowatt generator, Coffelt electrofishing unit (Model #VVP-15), and cable electrodes, with the boat serving as the anode. Pulsed AC current varied between 200–240 volts and 4–8 amperes. In each pool, six fish of harvestable size (>14 inches) were collected and placed on ice. Collected fish were weighed, measured (TL), filleted, and otoliths were extracted for age analysis. Skinless fillets were analyzed for total mercury (mg/kg) using the automated cold vapor technique (see Lange et al. 1994) by the Florida Department of Environmental Protection Central Laboratory. Mean total mercury concentration was compared between pools using ANOVA.

Results

Mean total mercury concentration for all collected fishes was 0.83 (± 0.09) mg/kg. Mean total mercury concentration was highest in Pool C (1.07 ± 0.25 mg/kg) and ranged between 0.65 and 2.30 mg/kg (Table 13-22). Fishes in Pools A and B showed similar mean total mercury concentrations at 0.69 (± 0.09) mg/kg and 0.71 (± 0.07) mg/kg, respectively (Appendix 13-6A). Total mercury concentrations ranged between 0.31 and 0.95 in Pool A, and between 0.52 and 0.95 in Pool B (Table 13-23). All fishes collected had total mercury concentrations less than 1.0 mg/kg, except for the largest fish collected, which had a total mercury concentration of 2.3 mg/kg. Mean total mercury levels were not significantly different between pools ($p = 0.1998$).

Table 13-23. Total mercury concentration from largemouth bass collected in Pools A, B, and C of the Kissimmee River under channelized conditions. Age was determined by otolith analysis. Consumption of fish with mercury levels between 0.5 and 1.5 mg/kg should be limited (suggested one meal per week). Fish with mercury levels above 1.5 mg/kg should not be consumed.

Total length (mm)	Weight (g)	Age	Pool	Hg concentration
493	1871	5	A	0.72
449	1457	3	A	0.88
534	2614	5	A	0.95
376	795	3	A	0.59
412	1071	2	A	0.74
414	1029	5	A	0.31
489	2014	3	B	0.66
449	943	4	B	0.95
534	1071	2	B	0.58
376	1229	2	B	0.69
412	669	2	B	0.52
414	736	3	B	0.83
564	3057	4	C	2.30
379	813	2	C	0.99
354	522	2	C	0.65
364	681	2	C	0.91
415	1129	4	C	0.89
395	913	2	C	0.69

Discussion

In recent years, the presence of organic mercury in Florida's natural environment has become recognized as a potential threat to the health of humans and wildlife. Unsafe levels of methylmercury have been found in predatory fishes in the Everglades and other areas of Florida (Ware et al. 1990). At this time, the factors and processes involved in the methylation and magnification of mercury in the food web are uncertain, but have generated a large amount of research (Ware et al. 1990, Spalding et al. 1994, Sepulveda et al. 1995). Research in more temperate regions of the world has shown that contaminated fish are usually restricted to waters with organic sediments, low productivity, and slight acidity (McMuurty et al. 1989, Spry and Wiener 1991). Periodic drying and flooding of wetlands and croplands tend to mobilize mercury in the soil and are thought to contribute to the problem (Bodaly et al. 1984). Soil disturbance and wetland creation can mobilize mercury (Bodaly et al. 1984, Verta et al. 1986, Verdon et al. 1991). For these reasons, the potential effect of the restoration process on mercury dynamics in the Kissimmee River following inundation of newly created floodplain wetlands is unknown.

Mercury concentrations ranged between 0.31 and 2.30 mg/kg under baseline conditions. The highest mercury concentration was found in the largest fish collected, which is consistent with findings of other studies that larger fishes tend to have higher mercury concentrations, as bioaccumulation is an additive process, and levels are magnified in higher trophic levels and larger individuals (Gardner et al. 1978). Total mercury concentrations found in largemouth bass collected in the channelized Kissimmee River are similar to those of fishes collected within the region (Lange et al. 1993). Mercury concentrations in largemouth bass from Lake Kissimmee ranged mostly between 0.5 and 1.0 mg/kg (Hand and Friedman 1990). Mercury concentrations from the Kissimmee River and Lake Kissimmee fall within levels of concern. The Florida Department of Health and Rehabilitative Services (HRS) issues a health advisory when mercury levels in fish tissue are between 0.5 and 1.0 mg/kg. The suggested rate of consumption of fish with mercury levels in this range is one meal per week. Fish tissue having a mercury concentration greater than 1.5 mg/kg is not suggested to be consumed at all by HRS.

Results from this study will be compared to those following restoration to determine if mercury bioaccumulation in fish has changed. Equivalent numbers of similar size class largemouth bass will be collected from Pools A, B, and C three years following inundation of floodplain wetlands and similarly analyzed for total mercury. The three year time period will potentially allow wetland soils to be inundated and dried down several times, which is suggested to cause mercury mobilization (Bodaly et al. 1984, Verta et al. 1986, Verdon et al. 1991).

VIII. INDICATORS OF PHYSIOLOGICAL STRESS RELATED TO HYPOXIA

Three specific sets of indictors of physiological stress in fishes (brain catecholamines, tissue heat shock proteins (Hsp) and blood cortisol) are to be tested to determine their usefulness as indicators for evaluating restoration success in the Kissimmee River. It is well known that stress induces changes in brain monoamines. Stresses include social stress (Artic char *Salvelinus alpinus*, Anders et al. 1998), long-term anoxia (Crucian carp *Carassius carassius*, Nilsson 1990), and hydrocarbon pollution (Gray snapper *Lutjanus griseus*, Brager 1997). The general response is decrease in brain norepinephrine, dopamine, and serotonin concentration and a decrease in the turnover rate of brain catecholamines. The hypothesis to be tested in this study is that stress resulting from seasonal exposure to low dissolved oxygen levels in the channelized Kissimmee River produces a greater brain catecholamine stress response in less hypoxia tolerant fishes (i.e., centrarchids).

In almost all organisms, exposure to environmental stressors induces a molecular response at the cellular level, in which Hsps are produced to ameliorate the stressed condition (Parsell and Lindquist 1993), including hypoxia (Lutz and Prentice 2002). Heat shock proteins are chaperones that assist in refolding thermally or otherwise denatured proteins, thereby returning the misfolded protein's functional state and restoring cellular homeostasis (Currie et al. 2000; Hofmann et al. 2000). Heat shock proteins have been isolated in numerous fish species and have been to shown to respond to a variety of biotic and abiotic stressors, including hypoxia (Iwama et al. 1998). Fish species less tolerant of seasonally low levels of dissolved oxygen may show a greater induction of Hsps than tolerant species during hypoxic exposure in channelized portions of the Kissimmee River. On the other hand, the Hsp scope for increase may be greater in tolerant fish. Nakano and Iwama (2002) have observed that the levels of constitutive Hsp70 and the Hsp scope for increase correlates with the ability of tidepool sculpins (*Oligocottus maculosus*) to handle the marked environmental swings that occur over tide changes. The second hypothesis to be tested is that hypoxia tolerant and hypoxia intolerant fish differ in the Hsp response to seasonal exposure to low dissolved oxygen levels.

In teleost fish, the general stress response involves the principal messengers of the brain-symapthetic-chromaffin cell axis, plasma cortisol being one component of this general response (Wendelaar Bonga 1997). Elevated cortisol can quickly result from many stresses, including handling, hypoxia, and pollution, but it may also quickly decline (Wendelaar Bonga 1997). High persistent levels of cortisol can have harmful effects (Wendelaar Bonga 1997), including the inhibition of testicular pubertal development (Consten et al. 2001). Interestingly, recent work indicates a possible interaction between cortisol and Hsp stress response. Basu et al. (2001) found that stress provoked elevated levels of cortisol significantly suppressed the heat stress-induced levels of gill Hsp70 in trout and tilapia, and DeBoeck et al. (2003) report that in copper exposed carp, cortisol elevation results in a lower Hsp70 response. This suggests that cortisol may be mediating Hsp70 levels in fish tissues during this period.

Methods

Blood cortisol, brain catecholamine, and tissue Hsp stress responses will be tested in two fish groups, one tolerant and the other intolerant of low levels of dissolved oxygen, under differing dissolved oxygen regimes related to habitat condition and season. The tolerant group will include Florida gar and bowfin. Both tolerant species are capable of gulping atmospheric oxygen (Lee et al. 1980) and are not believed to be negatively affected by oxygen minima. The intolerant group will include largemouth bass *Micropterus salmoides* and bluegill *Lepomis macrochirus*. The lower tolerance threshold for these centrarchids is approximately 2 mg/L, below which survivorship of all life history stages may be negatively affected (Moss and Scott 1961, Davis 1975, Knights et al. 1995). Fishes from both groups will be collected bi-monthly over a one year period from Pools A, C, and D to provide a range of dissolved oxygen conditions. Three individuals of each species will be collected at each site using electrofishing gear.

Physiological stress indicators will be tested across groups and compared between treatments (restored versus channelized) to determine their ability to detect differences in fish physiological stress under different oxygen regimes. Fishes collected in Pool A will serve as the control group, since this pool will remain channelized. Fishes collected in Pool D will serve as the impact group, since this pool is channelized currently, but will be resorted. Fishes collected in Pool C will serve as the restored group, since this pool has undergone physical restoration. A controlled study of the effect of electrofishing on these stress indicators is required. For this objective five individuals of each species will be collected by hook and line or gill net, and blood and tissue samples will be quickly taken for analyses.

<u>Electrofishing and tissue collection.</u> Sampling gear will consist of a 5.5 meter jon boat outfitted with a 5-kilowatt generator, Coffelt electrofishing unit (Model #VVP-15), and cable electrodes, with the boat serving as the anode. Pulsed DC current will be used and should range between 200–240 volts and 4–8 amperes. Sampling will be conducted by a three-person crew (one driver and two dip-netters) along the deep water edge of littoral vegetation as the boat travels downstream. Three individuals of each species will be collected from Pools A, C, and D.

Water quality data including dissolved oxygen and water temperature will be collected at each sampling location using a Hydrolab® multi-parameter water quality logger. Recordings will be taken along a depth gradient at 0.5 meter intervals extending from the water surface to the river channel substratum. These data will be used to determine temporal variation in oxygen availability to fishes collected. Linear Regression will be used to determine the relationship of Hsp, blood cortisol, and catecholamine concentrations to dissolved oxygen concentration.

Upon capture, fish will be immediately decapitated, and the brain, liver, and muscle will be dissected out, wrapped in aluminum foil and placed in liquid nitrogen until they can be returned to the laboratory.

<u>Determination of Cortisol</u>. Cortisol will be determined on blood plasma using an Assay Designs' Correlate-EIA™ Cortisol kit. This is an ELISA competitive immunoassay for the quantitative determination of Cortisol in biological fluids (Basu et al 2001). It uses a monoclonal antibody to Cortisol to bind, in a competitive manner, to Cortisol in a body fluid sample. After a simultaneous incubation at room temperature, the excess reagents will be washed away and substrate will be added. After a short incubation time, the enzyme reaction will be stopped and the yellow color generated will be read on a microplate reader at 405nm. The intensity of the bound yellow color is inversely proportional to the concentration of Cortisol in either standards or samples. The measured optical density will be used to calculate the concentration of Cortisol using standards as reference.

<u>Determination of Catecholamine Concentration.</u> Brain samples will be processed according to the method of Nilsson (1989). In essence, brain samples will be weighed while frozen, and homogenized in ice-cold (32°C) perchloric acid (PCA 4% w/v) containing 0.2% EDTA and 0.05% sodium bisulfite, using a variable speed Tissue Tearor from Biospec Products, Inc. The volume of PCA will be adjusted to obtain a 20% (w/v) homogenate. The PCA homogenate will be then centrifuged for 15 min at 13000g at 4 °C and the supernatant collected. The supernatant will be kept at -80 °C until the chromatographic analysis. Monoamine (norepinephrine, dopamine, and serotonin) and monoamine metabolite (DOPAC and HIAA) standards will be obtained from Sigma Chemicals (St. Louis, MO).

The concentrations of monoamine and monoamine metabolites present in aliquots of PCA extracts (volume varied from 250 to 750 mL) of tissue will be quantified using reverse-phase high performance liquid chromatography (HPLC) coupled with electrochemical detection (Nilsson 1989). The HPLC system consists of a Waters 510 HPLC pump and a Rheodyne 7725i Manual Injector (both obtained from Waters, Milford, MA), a reversed phase column (4.6 x 100 mm, catecholamine, C18, 3 μm obtained from Alltech), and an electrochemical detector with a glassy carbon working electrode set at +750mV vs an Ag/AgCl reference electrode (obtained from Bioanalytical Systems, West Lafayette, IN). This system will then be connected to a computer integration unit, Macintegrator (available from Ranin Industries). The mobile phase flow rate will be 1.3 mL/min. For the assay of norepinephrine, epinephrine, and dopamine, the mobile phase will consist of 100 mmol/L NaH_2PO_4, 9% (v/v) methanol, 0.63 mmol/L sodium octylsulfate, and 0.2 mmol/L EDTA, pH 3.60. For the assay of serotonin, and 5-hydroxyindole acetic acid the mobile phase will consist of 105 mmol/L citric acid, 2.5% (v/v) methanol, 20 μmol/L sodium octylsulfate, and 0.2

mmol/L EDTA, pH 2.20. Chromatograms will be analyzed using the Macintegrator software package and catecholamine and indoleamine levels will be reported in ng per g of wet weight tissue.

Determination of heat shock protein. Proteins will be extracted from brain, liver, and heart tissues according to methods adapted from Ramaglia (2004). For protein extraction, brain, liver, and heart samples will be ground to a powder in liquid nitrogen with a mortar and pestle, resuspended in TRIzol reagent (Life Technologies), and then homogenized in a hand held glass homogenizer. Protein extracts will be obtained according to the manufacturer's instructions. Briefly, after precipitation of DNA with ethanol, proteins will be precipitated from the phenol-ethanol supernate with isopropyl alcohol and sedimented by centrifugation (12,000g, 10 min, 4 degrees C). Following washing with 0.3M guanidine hydrochloride/95% ethanol, the protein pellet will be stored for 20 min at 15 degrees C, recentrifuged at 7500xg for 5 min and vacuum dried. Pellets will be redissolved in 1% SDS, and insoluble material is removed by centrifugation prior to analysis of proteins by Western blotting.

Gel electrophoresis and immunodetection protocol. Proteins will be separated electrophoretically according to size according to Locke and Tanguay (1996). Twenty-five micrograms of total protein will be loaded per lane on an SDS-polyacrylamide (12%) gel and separated at 100V for 2h. Molecular weight markers (Rainbow, Amersham) will be employed to determine the mobility of specific proteins on the gel. Subsequently, proteins will be transferred for 1h at 100V onto nitrocellulose membranes (Hybond ECL, Amersham) on a BioRad Protean apparatus. Membranes will be blocked overnight at 4 degrees C in 5% non-fat dried milk in Tris buffered saline (TBS; 25 mmoll-1 Tris-Cl, pH7.5 at 20 degrees C, 150 mmoll-1 NaCl) and then incubated for 1h with a rabbit polyclonal antibody against Hsp72 diluted 1;1000 in 5% milk with TBS/Tween (SPA-812, StressGen, Victoria, BC) or with a rat monoclonal antibody against Hsc73 diluted 1:1000 with 5% milk in TBS/Tween (SPA-815, StressGen, Victoria, BC). After washing in TBS/Tween, the membranes will be incubated for 1h with a goat anti-rabbit secondary antibody (1;1000 dilution, Santa Cruz) or a goat anti-rat antibody (1:1000 dilution, Santa Cruz) both of which are horseradish peroxidase (HRP) conjugated. For actin controls, after blocking blots will be incubated with a monoclonal antibody against actin (1;1000 dilution, Chemicon) in 5% milk in TBS/Tween, washed, and then incubated with an HRP conjugated goat anti-mouse antibody (1:1000, Sigma). The protein antibody complex will be detected by chemiluminscence (ECL, Amersham) for visualization. For quantification of band intensities, digital camera photographs will be analyzed with image-analysis software (NIH Image 1.60).

Results

Sampling began in January 2005 and will be completed in December 2005.

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CHAPTER 14

STUDIES OF BIRD ASSEMBLAGES AND FEDERALLY-LISTED BIRD SPECIES OF THE CHANNELIZED KISSIMMEE RIVER, FLORIDA

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ABSTRACT: Birds are integral to riverine and wetland ecosystems and can be useful indicators of their ecological integrity. Here, we use baseline and reference data to (1) analyze the combined effects of channelization and headwater regulation of the Kissimmee River on wading birds and waterfowl — two bird groups that are excellent candidates for measuring restoration success; and (2) develop expectations for their responses to the restoration project. In addition, other studies for which expectations were not developed are described, including monitoring of federally listed bird species. Quantitative data were not available for bird assemblages of the pre-channelized Kissimmee River, so aerial surveys from the Kissimmee River Demonstration Project were used to estimate reference conditions for densities of wading birds and waterfowl using the floodplain. Aerial surveys also were employed to estimate baseline (1996–1998) densities of long-legged wading birds and waterfowl within the 100 year floodline. During baseline surveys, mean annual dry season (December–May) density of aquatic long-legged wading birds in the Impact area (area to be restored) varied between years ($t = 3.05$, $P = 0.03$), averaging 3.58 ± 0.86 birds/km^2 in 1997 and 14.29 ± 3.37 birds/km^2 in 1998; baseline density from both years was substantially below the reference density of 30.6 birds/km^2. Following completion of the restoration project, dry season densities of long-legged wading birds are expected to be ≥30.6 birds/km^2. Winter waterfowl surveys conducted during the baseline period found low species richness ($n = 4$) and densities (0.44 ± 0.09 birds/km^2) of ducks within the Impact area. By contrast, reference species richness and density of waterfowl were 14 and 3.9 ducks/km^2 respectively. Following completion of the restoration project, winter density of waterfowl is expected to be ≥3.9 birds/km^2 and species richness is expected to be ≥13. Four federally listed bird species were known to occur along the Kissimmee River and surrounding uplands prior to channelization: wood stork (*Mycteria americana*), bald eagle (*Haliaeetus leucocephalus*), snail kite (*Rostrhamus sociabilis*), and Audubon's crested caracara (*Caracara cheriway*). All four species were monitored during the baseline period. Wood stork densities in the Impact area were uniformly low (<0.7 birds/km^2) throughout baseline surveys. Three bald eagle territories were active within the 100 year floodline of the river. No snail kites were documented during 13 monthly baseline surveys. Fifteen Audubon's crested caracara territories were found within the Kissimmee River floodplain and adjacent uplands during 1996–1999, at least 12 of which were active each year. The restoration project is expected to reestablish hydrologic characteristics that typified the pre-channelized system, including a flood pulse that regularly inundates a substantial portion of the floodplain. These changes are expected to provide improved habitat conditions for the wood stork, bald

eagle, and snail kite. These same changes are likely to render the floodplain less suitable for occupancy by Audubon's crested caracara. Monitoring of avian responses to the restoration project will continue for five years following project completion.

INTRODUCTION

Birds are integral to riverine and wetland ecosystems and can be useful indicators of their ecological integrity (Weller 1993, Weller 1995, Austin et al. 2001, Bryce et al. 2002). Within these systems, bird species assemblages are key components of food webs, acting as consumers at multiple trophic levels (Kushlan 1978) and also serving as prey for mammals, fish, reptiles, and other bird species (Bellrose 1980, Frederick and Spalding 1994). Birds also provide transport of nutrients within wetlands and among wetlands and uplands (Frederick and Powell 1994) and influence spatial distributions of plant and invertebrate species via dispersal of propagules (reviewed in Figuerola and Green 2002). Wetland birds respond to multiple classes of environmental variables, including hydrology (Collopy and Jelks 1989, Frederick and Collopy 1989), vegetation structure (Johnson and Montalbano 1984, Kaminski and Prince 1981, Weller and Spatcher 1965), and food availability (Draulens 1987, Gawlik 2002). Additionally, with their high degree of mobility, responses by birds to changes in food resources (Hafner and Britton 1983, Butler 1994, Lefebvre et al. 1994) and other habitat conditions are typically rapid (Custer and Osborn 1977, Weller 1979, Temple and Wiens 1989). Thus, the avian community is a valuable tool for assessing ecosystem change, including the effects of restoration (Weller 1995, Kingsford 1999).

The primary goal of the Kissimmee River Restoration Project is to reestablish the structure and function of the central region (approximately 1/3) of the river/floodplain by reintroducing fluctuating water levels and seasonal hydroperiods, and reconstructing the physical form of the river (Loftin et al. 1990, USACE 1991). Prior to channelization of the Kissimmee River through the construction of the C-38 canal, natural intra - and interannual variability in hydrologic characteristics interacted with local geology to produce a variety of dynamic floodplain and riverine habitats (Toth 1993; Anderson 2005; Anderson et al. 2005, Bousquin 2005, Carnal and Bousquin 2005). These habitats supported a diverse and abundant faunal assemblage, including many wetland birds (Perrin et al. 1982, National Audubon Society 1936–1959). Construction of the C-38 canal and control structures produced a channelized system of five impounded reservoirs (Pools A–E; see Chapter 1, Figure 1-1).Channelization combined with regulation of headwater inflows resulted in the drainage of the majority of Kissimmee River floodplain wetlands and drastically reduced flows in remnant river channels (Obeysekera and Loftin 1990, Toth 1991). These hydrologic changes led to shifts in river channel and floodplain vegetation, along with other shifts in the physical and biotic characteristics of the system (Anderson et al. 2005, Bousquin 2005, Carnal and Bousquin 2005, Colangelo 2005, Koebel et al. 2005a, Koebel et al. 2005b, Perrin et al. 1982). In short, channelization and headwater regulation fundamentally altered the types of habitats available to birds.

Objectives

This chapter has two primary objectives. The first is to use baseline and reference data to analyze the combined effects of channelization and headwater regulation on wading birds and waterfowl — two important bird groups of the Kissimmee River/floodplain that, due to their specific habitat associations and sensitivity to changes in habitat quality, are excellent candidates for measuring restoration success (Weller 1995). From these analyses, restoration expectations (predicted responses of birds to restoration) are developed that define key aspects of wading bird and waterfowl communities in a restored ecosystem. A second task is to describe baseline studies and/or outline monitoring needs for key species and taxonomic groups for which restoration expectations were not developed. Bird taxa in this second task include shorebirds and the federally-listed bald eagle (*Haliaeetus leucocephalus*), Audubon's crested caracara (*Caracara cheriway*), and snail kite (*Rostrhamus sociabilis*).

Chapter Outline

Since multiple studies were conducted to assess the past and present status of bird communities and threatened and endangered bird species of the Kissimmee River/floodplain, the remainder of this chapter has been organized by study, with each having a separate methods, results, and discussion section. Studies associated with restoration expectations are presented first, followed by monitoring studies. Following below is the chapter outline.

1. Introduction
    a. Chapter Outline
2. Wading Bird Density, Relative Abundance, and Reproduction
    a. Methods
    b. Results
    c. Discussion, Expectation Development, and Additional Monitoring Needs
3. Winter Waterfowl Use of the Floodplain
    a. Methods
    b. Results
    c. Discussion and Expectation Development
4. River Channel Waterbird Surveys
    a. Methods
    b. Results and Discussion
5. Bald Eagle Nesting
    a. Methods
    b. Results and Discussion
6. Crested Caracara Territories and Reproduction
    a. Methods
    b. Results and Discussion
7. Snail Kite Surveys
    a. Methods
    b. Results and Discussion
8. Conclusions
9. Literature Cited

WADING BIRD DENSITY, RELATIVE ABUNDANCE, AND REPRODUCTION

Methods

Baseline Data Collection

From June 1996–December 1998, aerial surveys were employed to measure the wet season (June–November) and dry season (December–May) densities of long-legged wading birds and to search for breeding colonies. East-west strip transects (n = 216) that spanned the 100 year floodline of the floodplain were established at 200 m intervals from the S-65 structure south to the S-65D structure (Figure 14-1). Each month, nonadjacent transects were randomly selected without replacement (Krebs 1999) until ≥15% of the floodplain in Control (will not be restored) and Impact (will be restored) areas was included in the survey. Transects were flown by helicopter navigating with Trimble NavPak™ software for aircraft GPS navigation systems. Start direction was alternated for consecutive transects. In 1996/1997, surveys were flown at 61 m and 130 km/hour. In 1997/1998, survey height was decreased to 30.5 m, which improved visibility for concurrent waterfowl surveys, but did not appear to affect visibility of long-legged wading birds (S. Melvin, personal observation). The 1998/1999 sample year was terminated after January 1999 due to increased military activity on Avon Park Air Force Bombing Range, which prevented surveys over a large portion of the western floodplain. Therefore, baseline-period wading bird surveys included two dry seasons and three wet seasons.

Prior to each survey, reference marks were made on helicopter windows that corresponded to the 200 m width of strip transects. Species and numbers of long-legged wading birds within the 200 m transect strip were recorded by a single observer into a handheld microcassette recorder. The observer also searched for evidence of wading bird breeding colonies. If a colony was located, the species and numbers of nests were estimated. Because it is not always possible to distinguish tricolored herons (*Egretta tricolor*) from adult little blue herons (*Egretta caerulea*) during aerial surveys (Bancroft et al. 1990), the two were combined into the category Small Dark Herons. Likewise, snowy egrets (*Egretta thula*) and immature little blue herons were classified as Small White Herons (Bancroft et al. 1990).

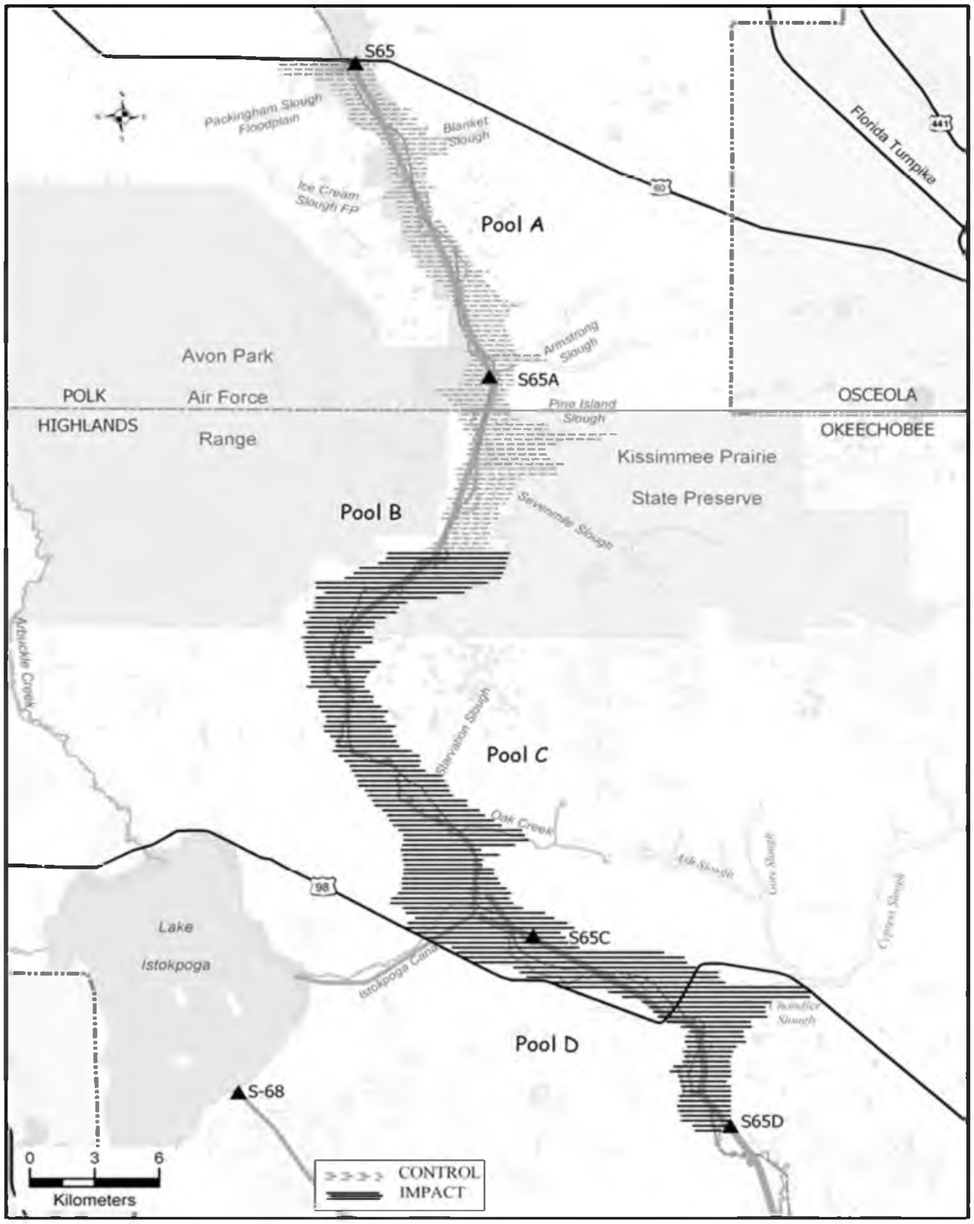

Figure 14-1. Map of transects used for baseline aerial surveys of wading birds and waterfowl. Transects spanned the 100 year floodline, were oriented east-west, and were spaced at 200 m intervals. Data from aerial surveys were summarized separately for the Control (northern portion) and Impact (southern portion) of the study area.

For data summaries, an additional category, Aquatic Wading Birds, was also created and included all long-legged wading bird species except the cattle egret (*Bubulcus ibis*). Because transect lengths varied with the width of the floodplain, densities (birds/km^2) and standard errors were estimated using the ratio method for unequally sized units (Jolly 1969, Caughley and Grigg 1981). Density estimates were generated for each monthly survey and then averaged to produce annual wet season and dry season densities. Densities of each wading bird species and species grouping were estimated separately for

Control and Impact areas (Figure 14-1). It should be noted that Control areas for this project are not controls in the traditional sense; since this study was conducted prior to commencement of restoration, no treatment has been applied. However, by monitoring the differences in wading bird densities over time between areas that will (Impact) and will not (Control) be restored, inferences will eventually be made regarding the effects of restoration (Stewart-Oaten et al. 1986). Analysis of variance (ANOVA) and Welch's t-tests were used to test for differences in seasonal wading bird densities among years in Control and Impact areas. Differences in means were considered significant at an alpha of 0.05.

Reference Data Collection

No quantitative data are available for densities or relative abundances of long-legged wading birds of the pre-channelized Kissimmee River. Audubon Society game wardens noted the approximate numbers of wading bird nests and sizes of foraging flocks while conducting ground-based patrols of the Kissimmee River/floodplain and the surrounding dry prairie/wetland complex during the pre-channelization years of 1936 to 1959 (Audubon Society 1936–1959). No standardized survey protocols were used, however, so estimates of densities cannot be obtained from these data. Approximate locations and minimum numbers of breeding colonies can be determined from the Audubon data, however. It is unknown whether wardens were able to effectively search the entire floodplain for colonies, due to its width and the difficulty of accessing its more remote areas. Thus, it is possible that some colonies, especially small or remotely located ones, were not counted.

Additional reference data are available from wading bird surveys of a flow-through marsh in Pool B that was built as part of the Kissimmee River Demonstration Project, and for floodplain areas along Paradise Run, a portion of the Kissimmee River near Lake Okeechobee that still retains some channel flow and periodic floodplain inundation (Toland 1990, Perrin et al. 1982). The 3.5 km^2 flow-through marsh was constructed just south of the S65-A tieback levee during 1984 and 1985, and was manipulated to simulate inundation and overland flow that were typical of the pre-channelized Kissimmee River floodplain (Toth 1991). While the Demonstration Project was conducted, seasonal inundation and overland flow within the marsh were attained by: (1) installing culverts through the S-65A tieback levee to provide a source of water flow into the marsh and (2) building a berm flanking one side of the marsh to prevent overland drainage into the C-38 canal. Inundation of the flow-through marsh was first achieved during June 1986 (Toth 1991) and aerial surveys of long-legged wading birds were conducted monthly from February 1987–May 1987 and from October 1987–May 1988 (Toland 1990). Thus, with the exception of surveys conducted during October and November, 1987, all surveys were conducted during the dry season. Aerial transects were 400 m wide and covered areas of 3.5 km^2 and 5.2 km^2 in the flow-through marsh and Paradise Run floodplain respectively. Transects were flown in a fixed wing aircraft at an altitude of 25–46 m and airspeed of approximately 145 km/h. Densities were calculated and summarized by species by dividing the number of birds counted by the area of the transect. Mean densities were calculated by averaging the densities from each survey. No measures of variability were reported.

Results

Baseline Surveys

Eleven species of wading birds were observed during 27 monthly aerial surveys from June 1996 through December 1998 (Table 14-1). During monthly surveys within the Impact area, cattle egrets frequently outnumbered all other wading bird species combined (Figure 14-2). Within the Impact area, either the great egret or white ibis was the most numerous aquatic wading bird species in six of ten dry season surveys and 15 of 17 wet season surveys (Figure 14-3). The endangered wood stork (*Mycteria americana*) was uncommon throughout baseline surveys in both Impact and Control areas, with seasonal densities never exceeding 0.72 birds/km^2.

Mean annual dry season density of aquatic wading birds in the Impact area varied between years ($t = 3.05$, $P = 0.03$), averaging 3.58 ± 0.86 birds/km^2 in 1997 and 14.29 ± 3.37 birds/km^2 in 1998; no significant differences were found in the Control area ($t = 0.11$, $P = 0.91$, Figure 14-4). During the wet season ($n = 3$), densities of aquatic wading birds did not vary with year in either the Impact ($F = 2.85$, $P = 0.09$) or Control ($F = 0.74$, $P = 0.49$) areas. In Control–Impact comparisons of within-season densities of aquatic wading birds, no significant differences were found during any season (Table 14-2). Aerial surveys indicated no active breeding colonies on the floodplain in 1996. One colony of cattle egrets and little blue herons was present in Pool B in 1997, and one colony composed of great egrets and anhingas was present in Chandler Slough (outside the 100 year floodline) in Pool D in 1998. Both colonies were small, with less than 100 pairs.

Table 14-1. Baseline estimates of seasonal densities (± SE) of long-legged wading birds of the channelized Kissimmee River, 1996–1999. Densities are expressed as birds/km^2 and were derived from monthly aerial surveys of the floodplain within the 100 year floodline. Densities are reported separately for Impact (to be restored) and Control (not to be restored) areas. Tricolored herons and adult little blue herons were combined to form the Small Dark Heron group. Snowy egrets and juvenile little blue herons were combined as Small White Herons.

	1996 Wet		1997 Dry		1997 Wet		1998 Dry		1998 Wet	
Species	Control	Impact	Control	Impact	Control	Impact	Control	Impact	Control	Impact
Black-crowned Night-heron (*Nycticorax nycticorax*)	0.00 (0.00)	0.00 (0.00)	0.02 (0.02)	0.01 (0.01)	0.09 (0.10)	0.00 (0.00)	0.66 (0.42)	0.03 (0.03)	0.16 (0.13)	0.04 (0.02)
Cattle Egret (*Bubulcus ibis*)	16.00 (8.14)	31.22(14.19)	5.81 (3.69)	6.09 (5.23)	27.19(10.31)	32.92(11.67)	11.20 (9.50)	4.52 (2.35)	10.00 (4.44)	22.87 (3.75)
Great Blue Heron (*Ardea herodius*)	0.45 (0.16)	0.11 (0.06)	0.69 (0.16)	0.07 (0.07)	0.40 (0.12)	0.13 (0.04)	0.37 (0.14)	0.27 (0.03)	0.41 (0.08)	0.20 (0.06)
Glossy Ibis (*Plegadis falcinellus*)	0.00 (0.00)	0.03 (0.03)	1.19 (1.19)	0.01 (0.01)	0.50 (0.55)	0.26 (0.29)	1.31 (0.50)	0.52 (0.42)	0.00 (0.00)	0.00 (0.00)
Great Egret (*Ardea alba*)	2.02 (0.30)	2.44 (0.58)	2.30 (0.43)	0.94 (0.31)	2.34 (0.38)	1.17 (0.30)	3.40 (1.08)	1.43 (0.26)	2.57 (0.74)	1.53 (0.42)
Small Dark Heron (*Egretta tricolor + E. caerulea*)	0.57 (0.21)	0.45 (0.19)	0.72 (0.27)	0.42 (0.26)	1.01 (0.29)	0.31 (0.06)	0.48 (0.11)	0.72 (0.43)	0.68 (0.31)	0.52 (0.19)
Small White Heron (*Egretta thula* + juv. *E. caerulea*)	3.47 (2.28)	0.86 (0.30)	4.17 (3.11)	0.51 (0.20)	1.06 (0.28)	2.61 (2.37)	1.29 (0.41)	0.66 (0.41)	0.67 (0.17)	0.89 (0.52)
White Ibis (*Eudocimus albus*)	0.43 (0.40)	0.66 (0.25)	4.01 (2.65)	1.04 (0.39)	25.31(24.52)	5.08 (2.07)	6.29 (1.80)	9.94 (2.78)	7.54 (3.44)	1.41 (0.50)
Wood Stork (*Mycteria americana*)	0.67 (0.54)	0.18 (0.16)	0.13 (0.08)	0.11 (0.06)	0.02 (0.02)	0.07 (0.08)	0.00 (0.00)	0.72 (0.43)	0.03 (0.03)	0.15 (0.11)
Yellow-crowned Night-heron (*Nyctanassa violacea*)	0.03 (0.03)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.00 (0.00)	0.01 (0.01)	0.00 (0.00)	0.00 (0.00)	0.03 (0.03)	0.00 (0.00)

Reference Conditions

Aerial surveys (n = 12) of the floodplain recorded 11 species of wading birds each in the flow-through marsh and Paradise Run during 1986–1987 (Toland 1990, Table 14-3). Ten of twelve surveys were conducted during the dry season. Data were pooled across surveys and no measures of variability were reported. White ibis had the highest relative abundance in both the flow-through marsh (40%) and Paradise Run (47%), followed by cattle egret (29% and 32%, respectively). Great egret and glossy ibis were the only other species with >5 % relative abundance in either the flow-through marsh or Paradise Run. Aquatic wading birds averaged 27.4 birds/km^2 in the flow-through marsh and 33.8 birds/km^2 in Paradise Run, while cattle egrets averaged 10.9 birds/km^2 and 15.7 birds/km^2 in those same areas. Densities of wood storks were low, averaging 0.6 and 0.3 birds/km^2 in the flow-through marsh and Paradise Run, respectively. Densities were not reported for other individual species of long-legged wading birds.

Wading bird breeding colony information from Audubon warden patrols is available for the Kissimmee River floodplain, tributary sloughs, and Kissimmee Prairie wetland complex for 17 years between 1936 and 1959 (National Audubon Society 1936–1959). The number of active colonies per year varied from zero to four (Figure 14-5). Nesting species were often only reported as “herons” or “egrets”. However, white ibis, great egret, snowy egret, little blue heron, great blue heron, and black-crowned night heron were all recorded as nesting in at least one colony. Number of nesting pairs was inconsistently reported, but seven of the 26 colonies reported had at least 500 pairs, and three other colonies were listed as “large”. Nesting colonies were reported throughout the year, but the majority of nesting occurred during the December–May dry season.

Discussion, Expectation Development, and Additional Monitoring Needs

Prior to channelization, the Kissimmee River experienced an annual (or nearly so) flood-pulse that usually inundated substantial portions of its floodplain (Anderson 2005). While there was considerable variability among years, floodplain inundation tended to peak in early to mid winter and was typically followed by a gradual (~ 30 cm/mo) recession event that ended in early summer (Anderson 2005).

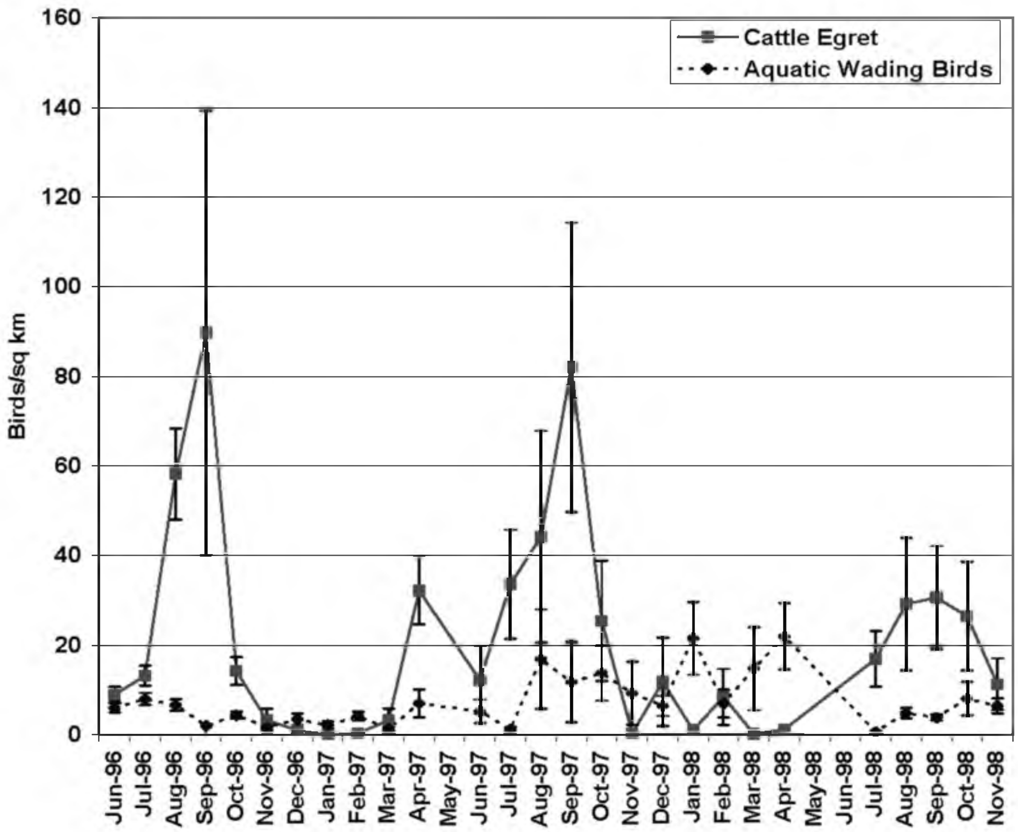

Figure 14-2. Monthly density (± SE) estimates of cattle egrets and aquatic wading birds within the Impact area during the Baseline period, 1996–1998. The category Aquatic Wading Birds includes all long-legged wading bird species observed during aerial surveys, with the exception of the primarily terrestrial cattle egret. Cattle egrets frequently outnumbered all other wading bird species combined.

The gentle slope of the floodplain, along with topographic variability, interacted with variability in river stages to produce a continually changing mosaic of appropriate and inappropriate foraging depths for the suite of wading bird species present. When water levels were highest, it is likely that fewer wading birds utilized the floodplain due to a general lack of appropriate foraging depths and dispersion of prey items (Kushlan 1978). As waters receded, abandoned channels, floodplain depressions, and microtopographical features likely served as refugia for fish, crayfish, and other wading bird prey items, trapping and concentrating them (Kushlan 1986, also see Gawlik 2002).

Channelization and headwater regulation of the Kissimmee River essentially eliminated the annual flood-pulse cycle, and converted the majority of floodplain wetlands into terrestrial communities (Anderson 2005, Carnal and Bousquin 2005), greatly reducing the amount of foraging habitat and nesting substrate for wading birds. While quantitative data are not available for pre-channelization densities of aquatic long-legged wading birds, a post-channelization decrease in use of the floodplain by aquatic wading birds would be expected. Comparisons of dry season reference data from both the Pool B flow-through marsh and

Paradise Run (Toland 1990) with baseline results, supports this supposition. Both reference sites had substantially higher densities of long-legged wading birds (excluding cattle egrets) than were found during baseline aerial surveys. It should be noted, however, that there were multiple methodological differences between Toland's surveys and those used for baseline data collection (altitude, aircraft type, observer) and at least a portion of the difference between baseline and reference densities may be an artifact of these differences in methodology. Further, since Toland (1990) did not report variability estimates for his surveys, it is unknown whether influential observations skewed density estimates upward or downward.

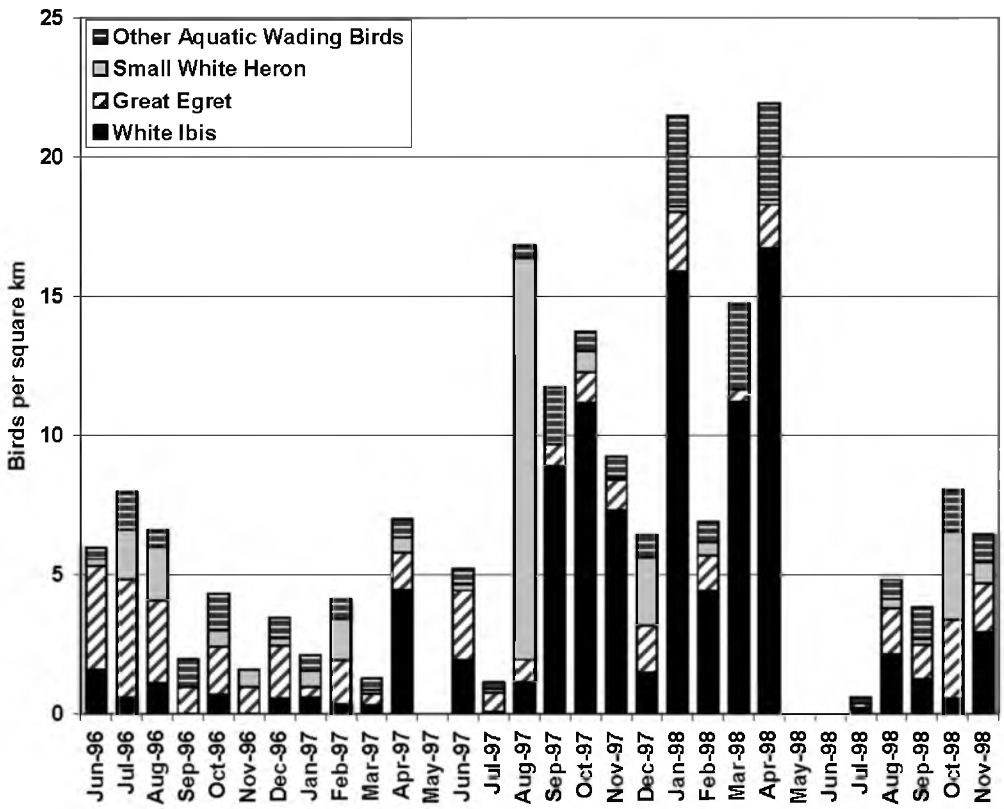

Figure 14-3. Stacked bar chart of total densities of aquatic wading bird species that were most commonly encountered within the Impact area during the Baseline period of 1996–1998. Great egrets were the most commonly observed species in 14 of 27 surveys, while the white ibis was the most common in seven of 27 surveys.

However, the fact that Paradise Run and the flow-through marsh both had similar densities that were substantially higher than baseline surveys is strongly suggestive of an effect of channelization.

Following the restoration of a regular flood-pulse cycle between the river and floodplain, it is expected that wetland communities and the fish and invertebrates that they support will become reestablished (Carnal and Bousquin 2005, Glenn 2005, Koebel et al. 2005a). Once established, these habitats should provide appropriate foraging habitat to support long-legged wading birds (excluding cattle egrets). It is expected that annual dry season densities of long-legged wading birds will be $\geq$30.6 birds/km^2, the mean of the density values from Paradise Run and Pool B flow-through marsh studies (Figure 14-6). Habitat conditions outside the Kissimmee floodplain may influence the magnitude of response by wading birds, however. For example, if foraging conditions are excellent elsewhere, the response may be less than expected.

Factors unrelated to the restoration project, such as prolonged drought, have the potential to affect the speed with which wading birds respond to the restoration project. Furthermore, even under ideal hydrologic conditions, reestablishment of wetland vegetation and aquatic fauna may take several years.

For these reasons, wading bird densities will be monitored for five years after completion of the restoration project. Pre- and post-restoration aerial surveys were designed so that each phase of the restoration project may be examined separately, thus facilitating comparisons between restored and unrestored portions of the floodplain, and allowing measurement of the initial responses and trajectory of recovery within newly restored areas. The expectation for wading bird density will be evaluated across the entire restoration area. The relative contribution of the restoration project to changes in wading bird densities in the Impact area will be assessed by comparing these changes to concurrent surveys of the Control area (Stewart-Oaten et al. 1986). The same aerial survey and data analysis protocols employed for baseline surveys will also be used to measure post-restoration responses.

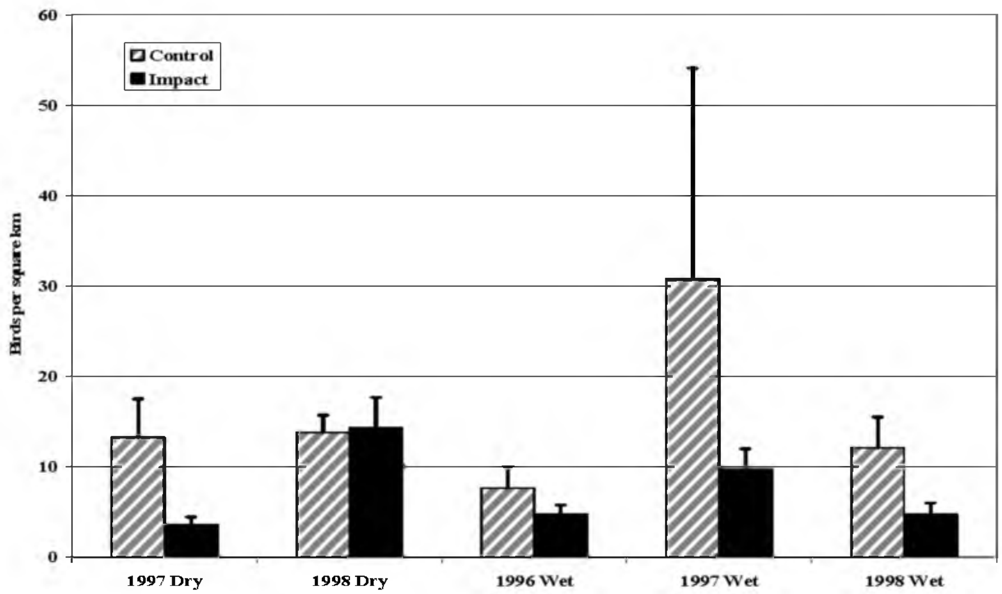

Figure 14-4. Seasonal densities (± SE) of aquatic wading bird in Control and Impact areas during 1996–1998 Baseline surveys. Densities in the Control area during the 1997 wet season (June–November) were strongly influenced by a single observation of a foraging flock of approximately 1000 white ibis.

Table 14-2. Results of within season comparisons of densities (birds/km^2) of aquatic wading birds in Impact and Control areas. Paired two-sample t-tests were used for all comparisons.

Season	Mean Density (birds/km^2) Impact	Control	df	t	P
1996 Wet	4.73	7.65	4	-1.82	0.14
1997 Dry	3.58	13.24	4	-2.13	0.10
1997 Wet	9.65	30.73	4	-0.90	0.42
1998 Dry	14.29	13.79	3	0.68	0.54
1998 Wet	4.74	12.08	3	-1.56	0.22

While an expectation for aquatic wading bird density could be developed from surveys conducted at reference sites (Toland 1990), a lack of appropriate reference data precluded the development of a restoration expectation for wading bird nesting colonies. The only pre-channelization data that are available for colonies come from Audubon warden patrols of the Kissimmee River and surrounding area (National Audubon Society 1936–1959). The Audubon data, while valuable, are inappropriate for developing a restoration expectation for wading bird nesting effort because (1) it is unknown whether wardens were able to effectively and consistently search the entire Kissimmee River floodplain, tributary sloughs, and Kissimmee Prairie wetland complex for colonies, (2) locations of many reported colonies are unknown, making it impossible to determine whether they were within foraging distance of the floodplain, and (3) just because a colony was within foraging distance of the floodplain does not guarantee that the colony was dependent on it.

Table 14-3. Total counts and relative abundances of wading birds in the Pool B flow-through marsh and Paradise Run during 1986–1987 (modified from Toland 1990).

	Flow-through marsh		Paradise Run	
Species	Total count	Rel. abund.	Total count	Rel. abund.
Great blue heron	23	0.01	20	0.01
Great egret	179	0.11	244	0.08
Snowy egret	57	0.04	36	0.01
Little blue heron	70	0.04	41	0.01
Tricolored heron	15	0.01	8	0.00
Cattle egret	460	0.29	980	0.32
White ibis	639	0.40	1443	0.47
Glossy ibis	138	0.09	292	0.09
Black-crowned night-heron	0	0.00	1	0.00
Yellow-crowned night-heron	1	0.00	0	0.00
Wood stork	27	0.02	20	0.01

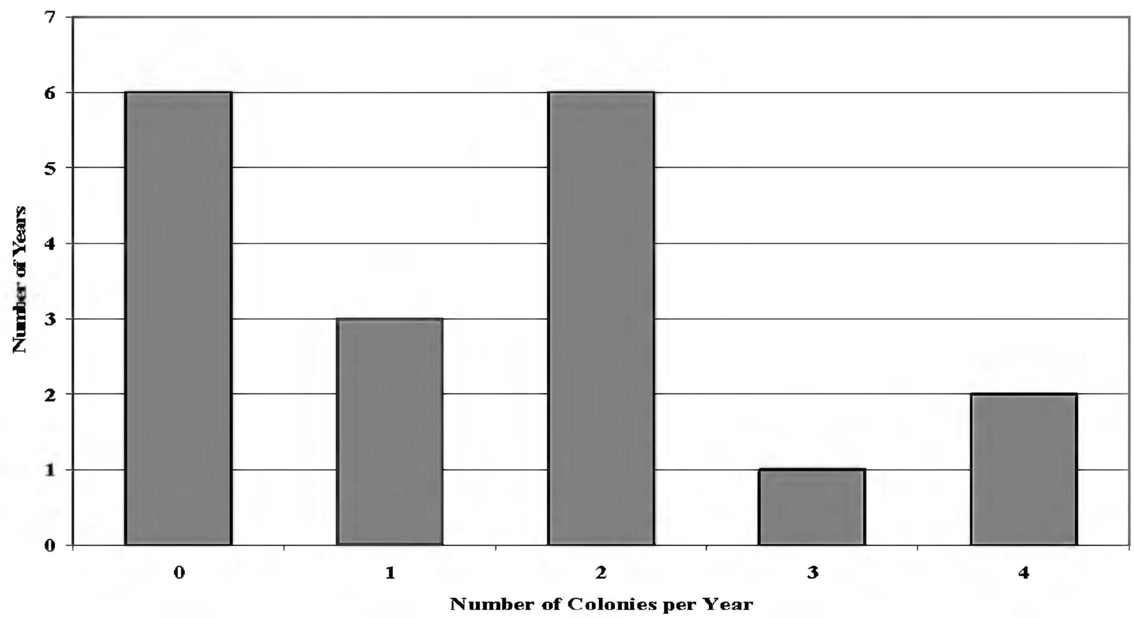

Figure 14-5. Minimum number of wading bird nesting colonies on the Kissimmee River floodplain, tributary sloughs, and Kissimmee Prairie wetland complex prior to channelization (Audubon Society 1936–1959). Data summarized represent 17 years of patrols by Audubon Society game wardens between the years 1936 and 1959. It is unknown whether wardens were able to search the entire area each year, so colony totals represent minimums per year. While colony sizes were inconsistently reported, four colonies were estimated to contain at least 1000 nests each.

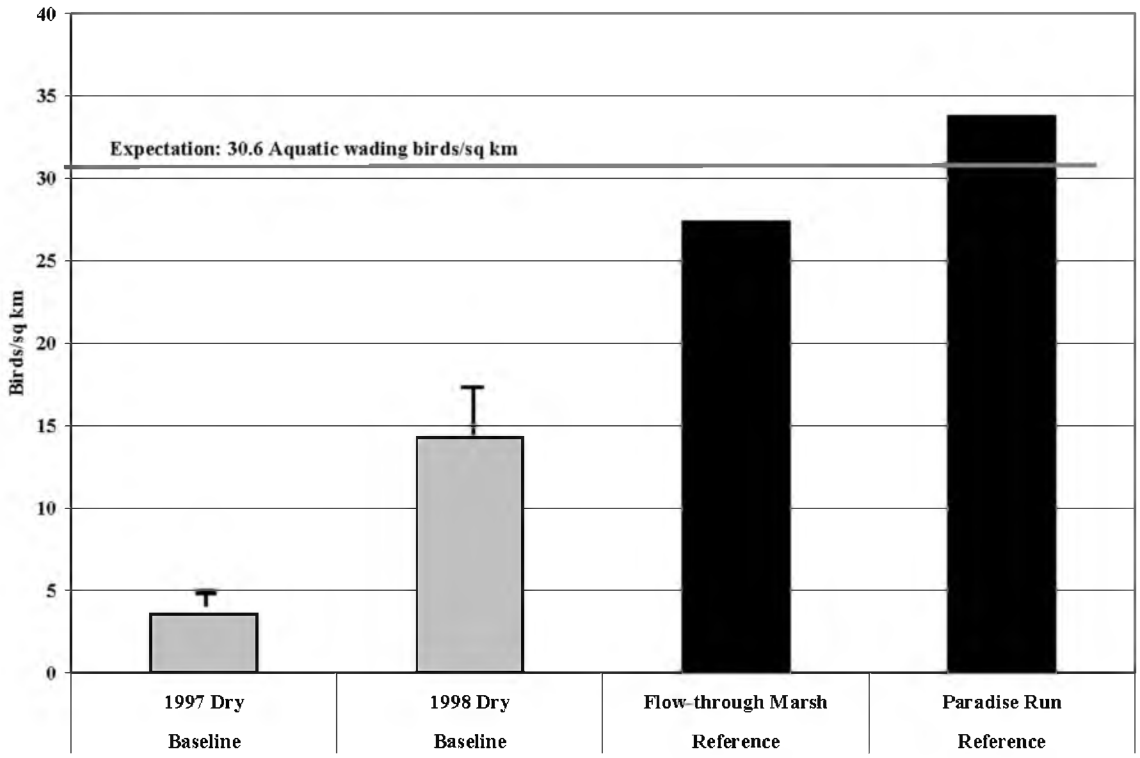

Figure 14-6. Expectation for dry season (December–May) densities (± SE for baseline surveys) of aquatic wading birds in the Impact area following restoration. The expectation is based on the average density from surveys of the flow-through marsh of the Kissimmee River Demonstration Project and of Paradise Run during 1986–1987 (Toland 1990).

Even without an associated restoration expectation, however, continued monitoring of wading bird reproduction is a vital component of the restoration evaluation program because nesting colonies are considered to be excellent indicators of wetland ecosystem integrity (Ogden 1994, Crozier and Gawlik 2003). Long-legged wading birds in the Everglades typically initiate nesting during the dry season and depend on a prolonged recession throughout the nesting cycle to provide the concentrations of prey required to successfully fledge young (Frederick and Collopy 1989, Frederick and Spalding 1994). Changes in the numbers, timing, locations, and success of Everglades wading bird colonies are considered indicative of the quality of habitat available within the ecosystem. Prior to channelization, wading bird nesting within or near the floodplain of the Kissimmee River also occurred primarily during the dry season (National Audubon Society 1936–1959), suggesting that colonial nesting in the area was initiated using similar cues and sustained by similar mechanisms as those of the Everglades. Thus, wading bird nesting colonies within or near the restoration area can provide valuable information regarding prey densities, prey availability, and whether the managed hydrology of the system is conducive to successful reproduction. Aerial searches for nesting colonies will be conducted during project construction and for five years following completion of the restoration project. When a colony is located, the number of visible nests and species of nesters will be estimated. Aerial observers frequently underestimate the number of nests in colonies (Frederick et al. 2003) however, so ground surveys of colonies located from the air will be employed whenever feasible.

WINTER WATERFOWL USE OF THE FLOODPLAIN

Methods

Baseline Surveys

Waterfowl density and species richness were measured within the Kissimmee River floodplain during the winters (November–March) of 1996/1997, 1997/1998, and 1998/1999. Aerial surveys were conducted using the methodology described in the previous section. Densities of each species were estimated separately for Control (will not be restored) and Impact (will be restored) areas. Density estimates were generated for each monthly survey and then averaged to produce annual densities. Annual densities were then averaged to generate mean density estimates for the baseline period. Analysis of variance (ANOVA) was used to test for differences in mean waterfowl densities among years in Control and Impact areas.

Reference Conditions

Eight years (1949–1957) of pre-channelization winter waterfowl data were collected by the Florida Game and Freshwater Fish Commission (1957) and are summarized in two reports (U. S. Fish and Wildlife Service 1959, Perrin et al. 1982). Aerial surveys of the Upper and Lower Basins were conducted approximately biweekly to monthly from November–March using fixed wing aircraft flying at approximately 145 km/h. Transects varied in length and averaged 400 m in width. Survey altitudes were not reported. Flight paths of transects were varied between counts "due to changes in water levels, concentrations of birds, etc." (Florida Game and Freshwater Fish Commission 1957). Since transect paths were altered in an effort to locate concentrations of ducks, it would be inappropriate to extrapolate survey results into densities (see Bancroft and Sawicki 1995 for discussion of transect sampling theory). These surveys can provide reference data for species richness, however.

Reference data for waterfowl densities are available from the Kissimmee River Demonstration Project (Toth 1991). Waterfowl densities were measured during 1987–1988 in a flow-through marsh that was constructed to simulate hydrologic characteristics of the pre-channelized Kissimmee River floodplain (Toland 1990; see Wading Bird Density and Relative Abundance section for a description of the flow-through marsh and aerial survey methods). Surveys were conducted monthly from February 1987–May 1987 and from October 1987–May 1988 (n = 12, Toland 1990). A single mean density estimate was generated for the entire survey period. No measures of variability were reported.

Results

Baseline Surveys

Four duck species, blue-winged teal (*Anas discors*), green-winged teal (*Anas crecca*), mottled duck (*Anas fulvigula*), and hooded merganser (*Lophodytes cullulatus*) were recorded during baseline aerial surveys. Duck densities were quite variable, but nearly uniformly low throughout the baseline survey period, with zero ducks observed in five of 13 surveys in the Impact area and four of 13 Control area surveys (Figure 14-7, Table 14-4). Mean annual density was 0.44 ± 0.09 ducks/km^2 in the Impact area and 0.61 ± 0.24 ducks/km^2 in the Control area, and no clear within-season pattern was observed. While density estimates trended higher in 1997/1998 (Figure 14-8), no significant differences in annual densities were detected within either the Impact (ANOVA, F = 1.99, P = 0.19) or Control (ANOVA, F = 3.08, P = 0.09) areas. Teal (primarily blue-winged teal) accounted for 76% of all observations, followed by mottled ducks (20%), and hooded mergansers (<1%); unidentified species comprised 4% of observations. Casual observations of wood ducks (*Aix sponsa*) were made three times during 1997 while conducting ground-based surveys of other bird taxa (S. Melvin, personal observation).

Reference Conditions

Density of ducks within the flow-through marsh averaged 3.9 ducks/km^2; no measures of variability were reported (Toland 1990). Three species of ducks were encountered during surveys, with blue-winged teal accounting for 78% of all observations (identities of the other two species were not reported). Pre-channelization surveys of the Kissimmee River Basin identified 19 species of waterfowl using the

Kissimmee River and lakes in the Upper Basin (Florida Game and Fresh Water Fish Commission 1957) (Table 14-5). Mean annual species richness averaged 14.6 and at least 11 species were observed each year.

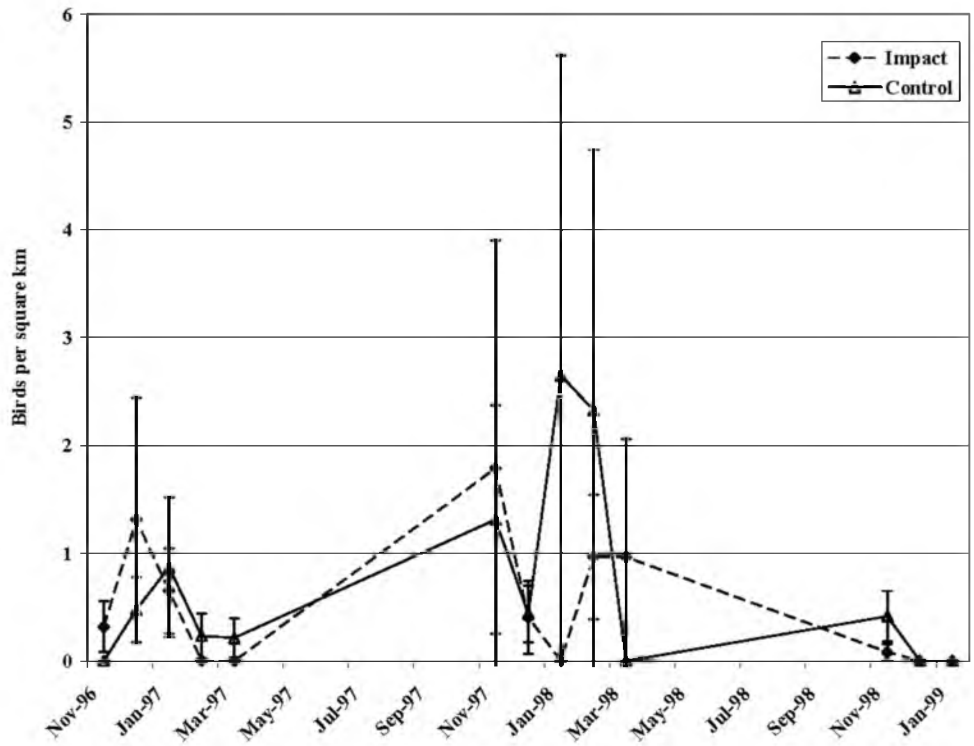

Figure 14-7. Mean (± SE) monthly winter (November–March) densities of ducks in Control and Impact areas during 1996–1999 Baseline surveys.

Discussion and Expectation Development

Baseline surveys of the Kissimmee River floodplain documented a winter waterfowl community with low densities and few species of ducks. Prior to channelization, 19 species of waterfowl were found within the Kissimmee Basin (Florida Game and Fresh Water Fish Commission 1957) (Table 14-5). However, these surveys pooled data from the Kissimmee River floodplain and the lakes in the Upper Basin. Of the 19 species, redhead (*Aythya Americana*), canvasback (*Aythya valisineria*), and ruddy duck (*Oxyura jamaicensis*) prefer lakes and open water (Bellrose 1980) and were probably rarely found on the Kissimmee River floodplain. Bufflehead (*Bucephala albeola*) and common goldeneye (*Bucephala clangula*) were only documented during one survey, and perhaps only occasionally utilized the floodplain. Taking these factors into account, 14 species of waterfowl were likely to have been regular users of the Kissimmee River floodplain prior to channelization. Approximately ten years after channelization was completed, surveys of Pools A–D noted six species of waterfowl (Perrin et al. 1982) and during baseline surveys, only four species were found, an estimated 69% reduction in species richness from pre-channelization levels.

Toland (1990) conducted his flow-through marsh surveys during February–May, 1987 and October–May 1988. Thus, some of the surveys were conducted during non-winter months (April, May, and October) when the majority of migrant ducks are not in Florida (Bellrose 1980). For this reason, the density of 3.9 ducks/km^2 reported in the study is likely to be a conservative estimate of duck densities of the pre-channelized system. The low species richness reported by Toland (1990) is another factor that suggests that duck densities of the flow-through marsh underestimate pre-channelization levels. In a discussion of a conceptual model of duck responses to floodplain restoration, Weller (1995) noted that timing of responses by waterfowl would be linked to the times that preferred foods became available, and went on to note that species that depend on annual plants would be expected to respond more rapidly than those that depend on perennials. Similarly, since many fishes and invertebrates would presumably take longer than plants to reach high densities in restored habitats, duck species that strongly prefer animal foods would be expected to respond more slowly to restoration than herbivorous species. Since Toland (1990) collected his data eight to 24 months after initial inundation of the flow-through marsh, it is likely that there was not sufficient time for the marsh to develop its full complement of waterfowl foods, particularly perennial plants and animals. Since there is variability in diet among duck species (Bellrose 1980), higher species richness would presumably lead to higher densities.

Table 14-4. Monthly densities (ducks/km^2) of resident and overwintering duck species. Data are from Baseline aerial surveys of the 100 year floodline of the Kissimmee River and were collected between November and March each year.

		Mottled duck		Blue-winged teal		Green-winged teal		All teal[1]		All ducks[2]	
Date	Group	Density	SE	Density	SE	Density	SE	Density	SE	Density	SE
Nov-96	Control	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Dec-96	Control	0.19	0.17	0.29	0.27	0.00	0.00	0.29	0.27	0.48	0.30
Jan-97	Control	0.65	0.62	0.00	0.00	0.00	0.00	0.00	0.00	0.87	0.65
Feb-97	Control	0.00	0.00	0.23	0.21	0.00	0.00	0.23	0.21	0.23	0.21
Mar-97	Control	0.21	0.18	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.18
Nov-97	Control	0.00	0.00	0.13	0.11	0.00	0.00	1.31	1.06	1.31	1.06
Dec-97	Control	0.22	0.20	0.22	0.20	0.00	0.00	0.22	0.20	0.44	0.26
Jan-98	Control	0.00	0.00	2.65	2.97	0.00	0.00	2.65	2.97	2.65	2.97
Feb-98	Control	0.00	0.00	2.32	2.42	0.00	0.00	2.32	2.42	2.32	2.42
Mar-98	Control	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nov-98	Control	0.10	0.10	0.00	0.00	0.00	0.00	0.31	0.22	0.42	0.23
Dec-98	Control	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Jan-99	Control	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nov-96	Impact	0.32	0.23	0.00	0.00	0.00	0.00	0.00	0.00	0.32	0.23
Dec-96	Impact	0.00	0.00	1.31	1.13	0.00	0.00	1.31	1.13	1.31	1.13
Jan-97	Impact	0.00	0.00	0.00	0.00	0.00	0.00	0.33	0.30	0.65	0.40
Feb-97	Impact	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Mar-97	Impact	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Nov-97	Impact	0.45	0.46	0.00	0.00	0.00	0.00	1.34	1.49	1.79	2.11
Dec-97	Impact	0.41	0.34	0.00	0.00	0.00	0.00	0.00	0.00	0.41	0.34
Jan-98	Impact	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Feb-98	Impact	0.24	0.19	0.32	0.30	0.40	0.32	0.72	0.39	0.96	0.58
Mar-98	Impact	0.00	0.00	0.96	1.09	0.00	0.00	0.96	1.09	0.96	1.09
Nov-98	Impact	0.08	0.07	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.07
Dec-98	Impact	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Jan-99	Impact	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

[1]Sum of blue-winged and green-winged teal.
[2]Combined sum of all ducks.

Species richness of ducks using the floodplain dropped almost immediately following channelization, and there is some evidence that overall numbers of ducks declined at the same time (Perrin et al. 1982). Given the timing of these decreases and the fact that they have persisted, there is strong evidence that channelization directly led to lower species richness and density.

Restoration of the physical characteristics of the central region of the Kissimmee River and floodplain along with the hydrologic characteristics of headwater inputs is expected to produce hydropatterns and hydroperiods on the floodplain that will lead to the development of extensive areas of wet prairie and broadleaf marsh, two preferred waterfowl habitats (Chamberlain 1960, Bellrose 1980). Given that waterfowl are able to search wide areas for suitable habitat, it is likely that individual species will begin using the restoration area soon after appropriate amounts of preferred food items become available. Thus, changes in the species richness and density of waterfowl within the restoration area are expected to be directly linked to the rate of development of floodplain plant communities and the faunal elements they support. Extrinsic factors such as annual reproductive output on summer breeding grounds, and local and regional weather patterns, may also play a role in the speed of recovery of the waterfowl community. For these reasons, waterfowl will be monitored until five years after completion of the restoration project. Based on species richness estimates from the pre-channelized system and the likely conservative density estimates from the Kissimmee River Demonstration Project, it is expected that waterfowl species richness will be 13 and densities will be at least 3.9 ducks/km^2 (Figure 14-9). The species richness metric (Figure 14-10) was decreased from 14 species estimated for pre-channelization to 13 species estimated for post-construction because the American black duck (Anas rubripes) no longer winters in significant numbers in central Florida (Stevenson and Anderson 1994). Species richness will be calculated as the total number of species encountered in any three periods, and densities will be evaluated as three year averages.

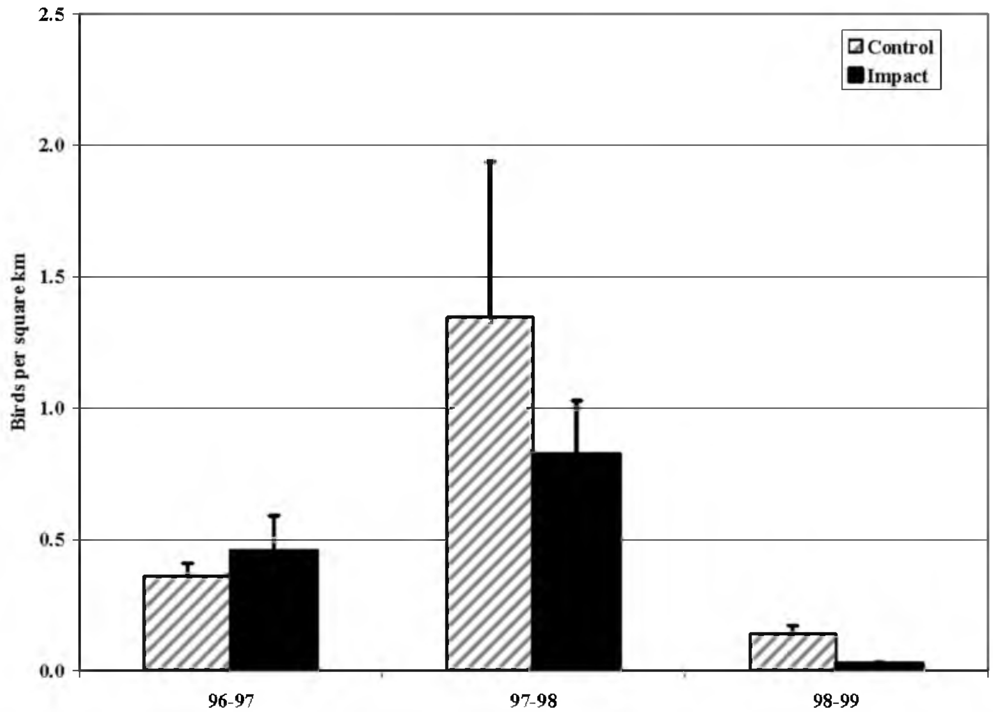

Figure 14-8. Mean densities (± SE) of ducks in Control and Impact areas for the winters of 1996/1997, 1997/1998, and 1998/1999. Surveys were conducted November–March each year except 1998/1999 when February and March surveys were not conducted due to military training activities within Avon Park Air Force Base.

RIVER CHANNEL WATERBIRD SURVEYS

Methods

Airboat surveys were employed to determine abundance and diversity of waterbirds using littoral and open water habitats in remnant (not destroyed by C-38 canal construction) river channel sections of the channelized Kissimmee River. Information from this study is intended to complement aerial surveys of the floodplain and, as such, river channels were surveyed separately from floodplain habitats. The survey area for this study was defined as the river channel and associated littoral habitat located between the top edges of opposite channel banks. The group Waterbirds was defined as all species that are generally considered to be dependent upon aquatic habitats from the orders Anseriformes, Charadriiformes, Ciconiiformes, Coraciiformes, Gruiformes, Podicipediformes and Pelecaniformes.

Three sections of remnant river channel were chosen for study from each of three pools in the channelized system (Figure 14-11). Selection criteria included length (longest stretches of remnant channel in each pool) and connection at both ends to the C-38 canal. All nine remnant river sections chosen for survey were flowing (i.e. were not abandoned oxbows) prior to channelization. Remnant river sections were surveyed monthly from May 1996 through June 1998. A survey was defined as one visit to one section of remnant river channel. Surveys were conducted on three consecutive days (one pool per day) each month, and within three hours of sunrise. Observations were made from 1.6 m above the water surface aboard an airboat traveling at 38 km/hr. An airboat was required for conducting surveys because most river sections were impassible by powerboat and the distance traveled each day prevented the use of a non-motorized boat. The moderately high rate of speed was used in an attempt to increase detection rate by minimizing the time available for birds to flush or move into the cover of littoral vegetation before they were seen. Each survey was 10 minutes in length.

Table 14-5. Species richness and preferred foods of waterfowl detected during pre- and post-channelization aerial surveys of the Kissimmee River floodplain. Pre-channelization surveys include data from both the Kissimmee River floodplain and Upper Basin lakes. Preferred foods were derived from a literature summary in Weller (1995).

Species	FWC 1954 - 1957[1]	FWC 1978 - 1980	Baseline 1996 - 1999	Preferred foods[5]
Mottled duck	x	x	x	I, S
Green-winged teal	x	x	x	I, S
Blue-winged teal	x	x	x	I, S
Hooded merganser	x	x	x	Fi, I
Mallard	x			I, S, M
Gadwall	x			Fo, S, I
Northern pintail	x			I, S
American wigeon	x	x		Fo, G, I, S
Ring-necked duck	x	x		S, I, Fo
Northern shoveler	x			I, S
Scaup sp.	x			I, Fo
Wood duck	x			I, S, M
Red-breasted merganser	x			Fi, I
Black duck[2]	x			I, S
Redhead[3]	x			T, Fo, I, S
Canvasback[3]	x			T, Fo, I
Ruddy duck[3]	x			I, Fo
Bufflehead[4]	x			I
Common goldeneye[4]	x			I, Fi, T

[1] Includes Upper Basin lakes.
[2] No longer a regular winter resident of central Florida.
[3] Species prefers open water/lakes.
[4] Species was only recorded during one survey.
[5] Reproduced from Weller (1995). I = invertebrates, S = Seeds, M = Mast, Fo = Foliage, G = Graminoids, Fi = Fish, T = Tubers.

Remnant river channel sections chosen as study sites were variable in length. In order to allow randomization of starting points for surveys, a timed run through each river channel was made at 38 km/hr prior to the beginning of the study. By subtracting the 10 minute duration of a survey from the time it took to travel the length of a remnant channel, the maximum amount of travel time possible prior to starting the survey could be determined. Starting points for each survey were determined by choosing a random number between zero and the maximum number of seconds that could be traveled in a particular channel while still having enough time to complete a 10 minute survey. A 10 minute boat survey resulted in an average of 6.3 km of river section traveled. Distance traveled on each survey was slightly variable due to the difficulty in maintaining steady speed around curves in the river channel.

Survey data were separated into two sample years, July 1996–June 1997 (1996/1997) and July 1997–June 1998 (1997/1998). Surveys were also grouped into seasons using the following definitions: winter (December, January, February), spring (March, April, May), summer (June, July, August), and fall (September, October, November). Initial analyses showed no significant difference in abundance among sites, so each visit to a site was considered a replicate for that month and sample year (i.e. all sites were averaged for each month). Seasonal analysis included all visits within the three month period, resulting in 27 replicate surveys per season within a sample year.

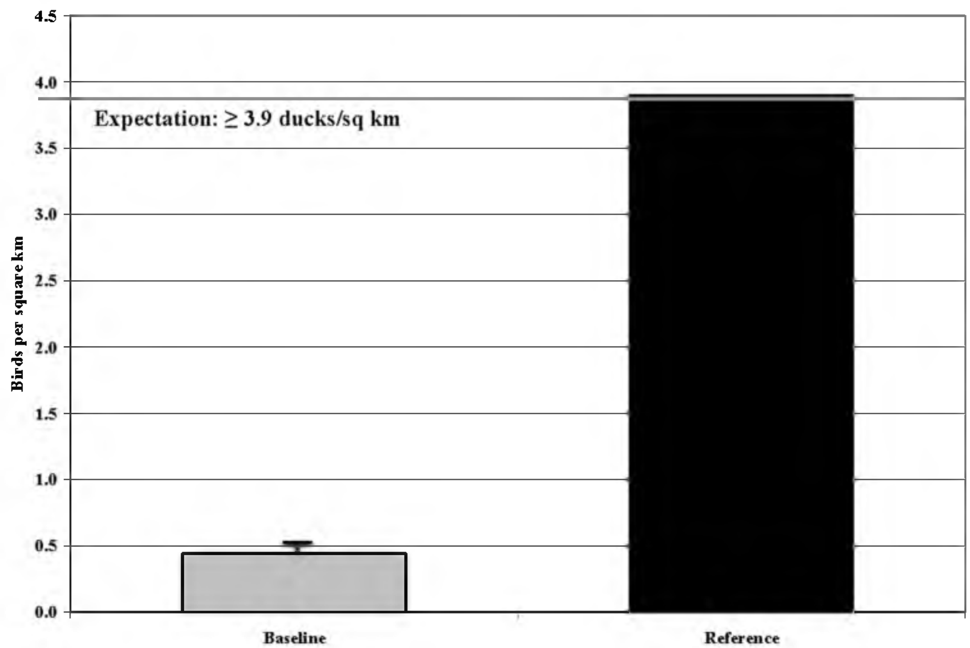

Figure 14-9. Summary of baseline and reference surveys of duck densities within the 100 year floodline of the Kissimmee River. Baseline data are reported as density (± SE); measures of variability were not reported for reference data (Toland 1990). The expectation of 3.9 ducks/km^2 is based on densities reported for the flow-through marsh of the Kissimmee River Demonstration Project.

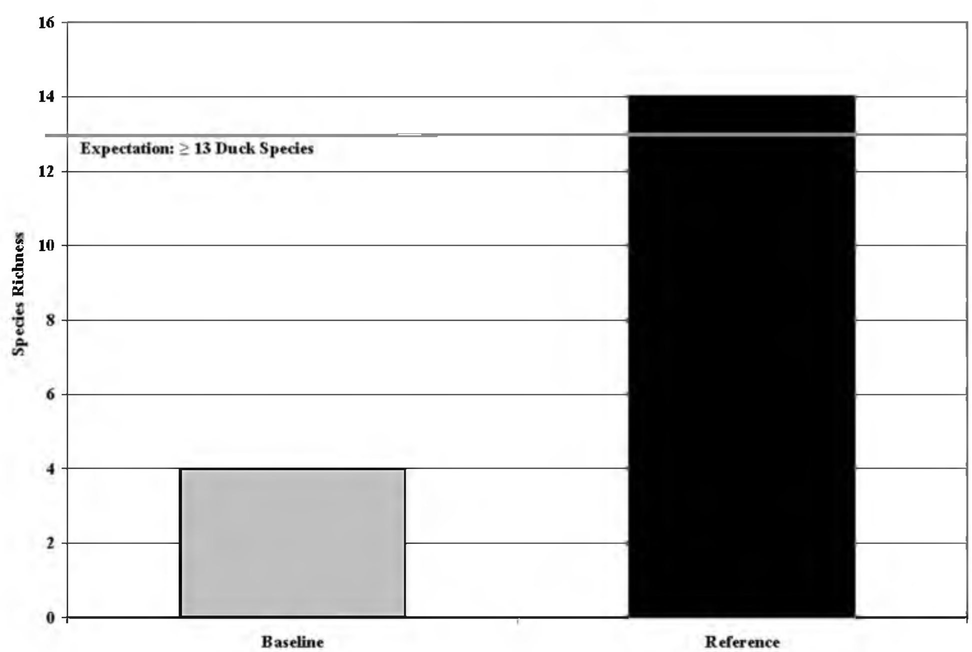

Figure 14-10. Summary of aerial surveys of duck species richness within the 100 year floodline of the Kissimmee River before and after channelization. Baseline and reference species richness were calculated as the total number of duck species encountered across all surveys. The expectation of 13 species is based on the estimated pre-channelization species richness minus the black duck, which no longer overwinters in Central Florida in significant numbers.

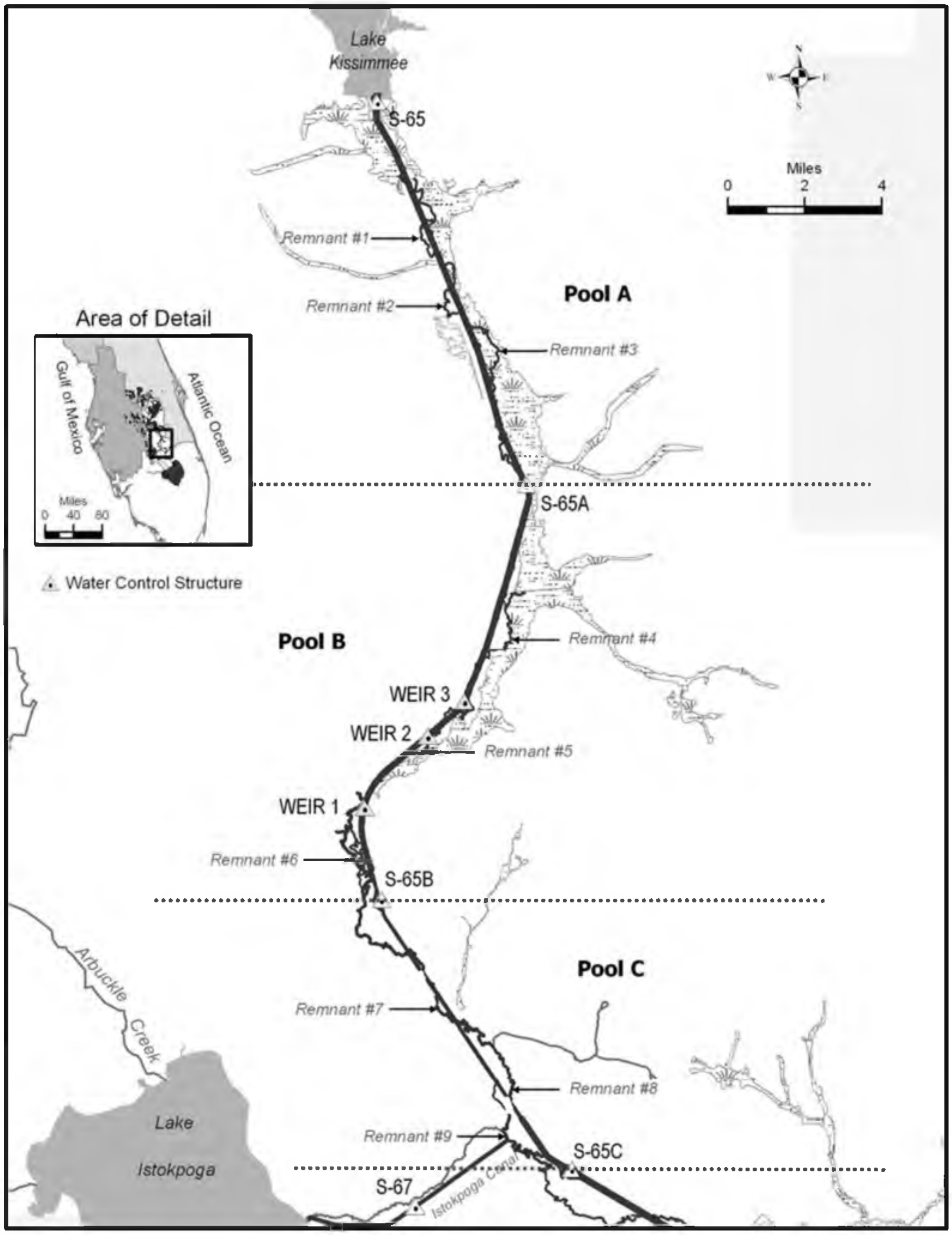

Figure 14-11. Locations of remnant river channels used for baseline surveys of waterbirds. Three remnant channels each were chosen from Pools A, B, and C.

Two-way analysis of variance for unbalanced data was used to compare mean number of birds per survey by season and sample year. Differences in means were considered significant at an alpha of 0.05. If the overall model was significant, a means separation test (Least Squared Mean) was performed to further evaluate differences. Species richness was the maximum number of species recorded per survey.

Results and Discussion

A total of 2015 waterbirds was observed during 177 surveys of remnant river sections. Mean birds per survey was 10.8 ± 1.3 in 1996/1997 and 11.9 ± 1.4 in 1997/1998, and did not differ significantly between years (*P* = 0.56). Thus, both years were combined for seasonal analysis. The interaction of sample year and season was not significant for mean abundance (*P* = 0.79). However, there was a significant difference in mean abundance among seasons (*P* = 0.02). Mean abundance in spring was significantly higher than fall (*P* = 0.01) and summer (*P* = 0.01; Table 14-6). Fall and summer mean abundance were not significantly different (*P* = 0.88). Winter mean abundance was not significantly different from fall (*P* = 0.11), summer (*P* = 0.33), or spring (*P* = 0.10).

Table 14-6. Mean number of birds and mean species richness per baseline survey of remnant river channels.

Year	# of Surveys	Mean (SE) Birds/Survey	Mean (SE) Species Richness/Survey
96/97	87	10.8 ± 1.3	2.9 ± 0.02
97/98	90	11.9 ± 1.4	3.2 ± 0.02
Fall	47	8.8 ± 1.4	2.9 ± 0.3
Spring	44	15.2 ± 2.3	3.6 ± 0.4
Summer	40	11.6 ± 1.8	2.5 ± 0.2
Winter	46	12.8 ± 1.9	3.1 ± 0.3

Mean species richness was 11.5 + 1.3 in 1996/1997 and 12.3 + 1.4 in 1997/1998 (Table 14-6). No significant differences existed among seasons for species richness (P = 0.07; Table 14-6). Twenty-six species of waterbirds representing six orders (Table 14-6) were observed during surveys of remnant river sections. Common moorhen (*Gallinula chloropus*) was the most commonly observed species in both sample years, making up 36% of total waterbird abundance (Table 16-7). During both sample years, birds from the order Ciconiiformes (wading birds) comprised the majority of waterbird observations (49%). Gruiformes (cranes, moorhens, gallinules) contributed nearly as much to the overall observations (41%), while Anseriformes (waterfowl) represented only 2%. Charadriiformes (gulls, terns, shorebirds), Coraciiformes (kingfishers), and Pelicaniformes (pelicans, cormorants, anhingas) were represented scarcely (<1% each). No birds from the order Podicipediformes (grebes) were observed. Interspecific differences in detectibility may have influenced estimates of relative abundance, however. For example, counts of some secretive bird species, such as American bittern (*Ixobrychus exilis*) or king rail (*Rallus elegans*) may have been underestimated.

Channelization of the Kissimmee River essentially eliminated flow of water in river channels, creating stagnant conditions that led to expansion of littoral vegetation, thick layers of accumulated organic matter on channel bottoms, fewer exposed sandbars, and low dissolved oxygen levels (Anderson 2005, Anderson et al. 2005, Bousquin 2005, Colangelo 2005). These physical, chemical, and biological changes in turn, led to low population levels and decreased availability of many fish and invertebrate species preferred by waterbirds (Glenn 2005, Koebel et al. 2005a). When considered in light of these changes, the low species richness of waterbirds and prevalence of some species is not surprising. Common moorhen, for example, which accounted for 36% of all observations, is a species that prefers slow- or non-moving water. Lack of flow in remnant river channels has likely led to increases in channel use by moorhens, and the return of flow following restoration should precipitate a decrease.

Among the results of this study, the lack of shorebirds is perhaps the most notable. Historical accounts of the pre-channelized Kissimmee River, its floodplain, and surrounding wetlands noted at least ten species of shorebirds (National Audubon Society 1936–1959) (Table 14-8). While some of these species, such as least sandpiper (*Calidris minutilla*), would be expected to be more common on the floodplain, nearly all of them would be expected to utilize the periphery of river channels, especially sandbars (Stevenson and Anderson 1994). Sandbars support a diverse invertebrate prey base and also provide loafing areas for shorebirds (Koebel et al. 2005a). In remnant river sections of the channelized Kissimmee, sandbars do not exist, or are completely covered with organic deposition, and the lack of flowing water precludes the formation of new sandbars (Anderson et al. 2005).

Table 14-7. Relative abundances of species encountered during baseline surveys of remnant river channels. Common moorhen, a species that prefers slow-moving or non-moving water, was the most abundant.

Common name	Scientific name	Relative abundance
American bittern	*Botaurus lentiginosus*	<1
American coot	*Fulica americana*	<1
Anhinga	*Anhinga anhinga*	7
Belted kingfisher	*Ceryle alcyon*	<1
Black-crowned night-heron	*Nycticorax nycticorax*	2
Blue-winged teal	*Anas discors*	<1
Cattle egret	*Bubulcus ibis*	7
Common moorhen	*Gallinula chloropus*	36
Double-crested cormorant	*Phalacrocorax auritus*	<1
Glossy ibis	*Plegadis falcinella*	5
Great blue heron	*Ardea herodias*	5
Great egret	*Casmerodius albus*	6
Green heron	*Butorides striatus*	6
Least bittern	*Ixobrychus exilis*	<1
Least tern	*Sterna antillarum*	<1
Little blue heron	*Egretta caerulea*	9
Limpkin	*Aramus guarana*	3
Mottled duck	*Anas fulvigula*	<1
Purple gallinule	*Porphyrula martinica*	5
Sandhill crane	*Grus canadensis*	<1
Snowy egret	*Egretta thula*	1
Sora	*Porzana carolina*	<1
Tricolored heron	*Egretta tricolor*	4
White ibis	*Eudocimus albus*	7
Wood duck	*Aix sponsa*	<1
Yellow-crowned night-heron	*Nycticorax violaceus*	1

Restoration of the Kissimmee River will facilitate the formation of sandbars, especially at curves in the river, fostering an increase in hyporheic invertebrates (Harris et al. 1995). Probing shorebirds such as greater and lesser yellowlegs should benefit from this reestablished prey source (Elphick and Tibbitts 1998, Tibbitts and Moskoff 1999).

Using airboat surveys to quantify shorebird use of the channelized system has proven to be problematic for two reasons. First, although shorebirds are expected to make extensive use of restored river channel habitats, especially sandbars, they also use, and, in the case of some species, prefer, shallow floodplain wetlands (Stevenson and Anderson 1994). Thus, river channel surveys alone are inadequate to document shorebird responses to restoration. Second, most shorebird species that are expected to use the restored river/floodplain system are small and cryptically colored. Species with these characteristics are often difficult to detect during airboat surveys. For these reasons, survey protocols must be modified to fully document shorebird responses to the restoration project. Pre- and post-restoration surveys that include both river channel and floodplain habitats will be designed specifically for shorebirds and will be conducted within the area to be included in Phase II/III of the restoration project. There may be opportunities to document Anseriformes, Ciconiiformes, Coraciiformes, Gruiformes, Podicipediformes and Pelecaniformes while conducting these surveys, but the focus will be on shorebirds. The aerial surveys described in previous sections of this report will remain the primary method of evaluating Anseriform and Ciconiiform responses to the restoration project.

Table 14-8. Shorebird species reported from Audubon Society game warden patrols of the pre-channelized Kissimmee River and floodplain, tributary sloughs, and Kissimmee Prairie wetland complex (Audubon Society 1936–1959).

Common Name	Scientific Name
Black-necked stilt	*Himantopus mexicanus*
Dowitcher sp.	*Limnodromus* sp.
Greater yellowlegs	*Tringa melanoleuca*
Lesser yellowlegs	*Tringa flavipes*
Killdeer	*Charadrius vociferus*
Least sandpiper	*Calidris minutilla*
Ruddy turnstone	*Arenaria interpres*
Common snipe	*Gallinago gallinago*
Solitary sandpiper	*Tringa solitaria*
American woodcock	*Scolopax minor*

BALD EAGLE TERRITORIES AND REPRODUCTION

Methods

Surveys of literature and unpublished data from the Florida Fish and Wildlife Conservation Commission (FWC) were used to examine use of the Lower Basin by the federally threatened bald eagle (*Haliaeetus leucocephalus*). Bald eagle nesting data from the Lower Basin are available from ground surveys for the years 1959–1971, which approximately coincide with construction of the C-38 canal and associated water control structures (G. Heinzman, unpublished field notes, summarized in Shapiro et al. 1982a, b). The geographic extent of the data summarized by Heinzman included the Upper Kissimee, Lower Kissimmee and Istokpoga Basins. More recent data are available from the FWC, which has conducted statewide aerial surveys of bald eagle nesting activity since 1972. The goal of the FWC surveys is to monitor the Florida bald eagle population and document annual productivity (Nesbitt 2003). Each year, aerial surveys of the entire state are conducted twice between December and April. Surveys include checks of all previously active nesting territories as well as searches for new territories. During surveys, the status (active/inactive) of each territory is noted.

Results

During surveys conducted from 1959–1971, while the flood control project was under construction, an average of 22.7 bald eagle territories per year were active in the Lower Basin (Shapiro et al. 1982 a, b). No coordinates of these territory sites are known to exist, so it is impossible to determine which territories were inside or within foraging range of the 100 year floodline of the Kissimmee River. Shapiro et al. (1982 a, b) summarized bald eagle activity within the Lower Basin during the post-channelization period 1977–1979, and noted an average of six active territories per year. During the baseline period of 1996–1998, there were a total of eight active bald eagle territories in the Lower Basin each year (FWC, unpublished data courtesy of J. White). Of these eight nests, three were located within or in close proximity to the 100 year floodline.

Discussion

Considered alone, the reduction in the number of Lower Basin bald eagle nesting territories that occurred after channelization does not necessarily implicate the flood control project as a cause for the decline. Bald eagle populations within most of the species' range were quite low through the 1960s–1970s, primarily due to the effects of DDT and other persistent organochlorine pesticides on reproduction (Buehler 2000). However, population data from the basins to the north and west of the Lower Basin tend to support the idea of a strong channelization effect. Shapiro et al. (1982 a, b) compared trends in the numbers of bald eagle territories in the Lower Basin with those in the adjacent Upper and Istokpoga Basins and noted that the post-channelization decline in bald eagle territories in the Lower Basin actually coincided with modest increases in the number of territories in the Upper and Istokpoga Basins. Further, while the number of active bald eagle territories statewide increased steadily from 353 to 1043 nests between 1979 and 1999, Lower Basin territories were essentially unchanged, with six active nests in 1979 and eight in 1999 (Nesbitt 2000). Thus, it appears that channelization and the resulting loss of wetlands at least contributed to the decline and continued low number of territories in the Lower Basin.

In a review of bald eagle diet studies, Stalmaster (1987) found that fish and birds were the most commonly taken prey items. A Florida study made similar conclusions, finding that that fish (79%) and birds (17%) comprised 96% of the total items taken (McEwan 1977). The study also noted that American coots (*Fulica americana*) were the most common avian prey. Bald eagles typically forage over water in areas within 500 m of perches, and may have greater capture success in areas of shallow water, where fish are located closer to the water surface (Buehler 2000). Thus, the pre-channelization timing, depth, and extent of inundation of the floodplain of the Kissimmee River (Anderson 2005) were likely to have frequently provided expansive areas of suitable foraging conditions for nesting bald eagles. Restoration of flooding regimes and hydroperiod will promote reestablishment of floodplain wetlands (Toth 1991), which will increase the amount of foraging habitat available to breeding bald eagles. Increased river–floodplain interactions should lead to greater prey abundance in restored wetlands by allowing fish to immigrate onto the floodplain (Trexler 1995), and spring recessions will concentrate prey in drying wetlands. The increase in the availability of foraging habitats within the restoration area combined with an expanding population of bald eagles statewide (Nesbitt 2000) should lead to increased nesting effort along the restored portion of the Kissimmee River floodplain.

CRESTED CARACARA TERRITORIES AND REPRODUCTION

Methods

Audubon's crested caracara (*Caracara cheriway*) is listed as threatened by the U.S. Fish & Wildlife Service, and the Lower Basin falls within the heart of its range (Morrison 1996). Baseline surveys were conducted to determine distribution, abundance, and reproductive success of the species within the 100 year floodline and adjacent uplands of the Kissimmee River (Morrison 1997a, Morrison 1997b, Morrison 1998, Morrison unpublished data). Surveys were conducted during January–December 1996 and during each succeeding breeding season (approximately January–April) from 1997–1999. Beginning in January 1996, all accessible areas of suitable habitat within the restoration project area (100 year floodline between the S-65 and S-65-D structures) were searched for occupied caracara territories. Suitable habitat (Morrison 1996) was identified using aerial photographs. Once an area of suitable habitat was identified, ground searches were conducted using a combination of systematic searches and observations of adult behaviors. If a nest was located, mirror poles were used to determine nest contents. Nests were monitored approximately monthly until they fledged young or failed. Each year, existing territories were searched for active nests, and unoccupied suitable habitats from the previous year were searched for new nesting territories. Coordinates of all nest sites were placed in a GIS database.

During 1997, home ranges of radiotagged adult caracaras were estimated using RANGESV software (Kenward and Hodder 1996). Cluster analyses within RANGESV were used to eliminate outlying telemetry locations from home range analyses. The fixed kernel estimator in RANGESV was used to estimate home ranges, which were defined as a 99% contour calculated using a smoothing factor of 0.85. Habitat composition of caracara territories was estimated using vegetation coverages from the GAP Analysis Project at the University of Florida (Pearlstine et al. 2000). Habitat composition was calculated in two ways. For caracaras whose home ranges were estimated using telemetry, habitat composition was measured using mapped home ranges. For two additional territories, habitat composition was estimated within a 1300 ha circle (mean home range size for radiotagged birds) centered on the nest.

Results and Discussion

Fifteen separate caracara nesting territories were identified from 1996–1999 and at least 12 fledged young each year (Table 14-9). Of the 15 territories, 11 were located within or upland of the area to be restored (Figure 14-12). Pairs typically initiated nesting in November and fledged young by February. The average number of fledged young per nesting attempt ranged from 1.40 in 1996 to 1.75 in 1999 and averaged 1.57 ± 0.08, which is similar to the results of other studies in the area (Humphrey and Morrison 2000). Home range sizes of seven radiotagged adult caracaras (five males, two females) averaged 1547 ± 523 ha and ranged from 900–2800 ha. Habitat composition within territories (n = 9) was dominated by improved pasture (39.48 ± 6.55 %). Other common habitat types included saw palmetto (17.39 ± 5.24 %), pine forest (13.17 ± 1.60 %), shrub and woodland (9.92 ± 2.55 %), and marsh (8.90 ± 2.65 %).

Table 14-9. Number of Audubon's crested caracaras fledged per territory during baseline surveys of the Kissimmee River floodplain and surrounding uplands.

Territory ID	1996	1997	1998	1999
4E-66	1	0	?[1]	1
4K-3	1	2	3	2
APAFR-31	1	2	2	2
D621-49	?	?	?	2
GH-43	1	3	0	2
HR-36	2	2	0	3
HYATT-61	2	?	?	?
KICCO-47	0	?	?	?
MONTSN-34	2	1	2	?
MONTSS-35	2	2	2	2
OXBOW-58	2	2	?	0
PUTN-1	1	0	1	2
UH-7	1, 2[2]	2	1	1
KE1-83	1	2	2	2
KE2-84	2	2	?	2

[1] No nest was located within the territory during this year.
[2] Pair fledged two broods this year.

Audubon's crested caracara is a species that likely realized a net gain in available habitat in response to channelization. The Lower Basin falls within the heart of the species' range and the grassland/palm/wetland complex that replaced floodplain wetlands following channelization is typical of its preferred habitat (Morrison 1996, Humphrey and Morrison 2000). Caracara territories from this study contained more than four times as much improved pasture as marsh. Though caracaras will forage in wetlands (Morrison 1996), the restoration project will change the floodplain from an area dominated by pasture to an area dominated by wetlands, making it less suitable as caracara habitat (U. S. Fish and Wildlife Service 1991). While the restoration project might affect individual caracaras, it is not expected to jeopardize the continued existence of the species (U. S. Fish and Wildlife Service 1991). The locations and status of caracara territories and nests will be monitored before, during, and after each phase of construction for the restoration project.

SNAIL KITE ABUNDANCE AND REPRODUCTION

Methods

Baseline surveys of the numbers and distribution of the federally endangered snail kite (*Rostrhamus sociabilis*) were conducted monthly within Pools A–D during February–July, 1996 and March–August, 1997 (Dreitz 1996–1997, unpublished reports to SFWMD). Surveys were performed via airboat using two trained observers. During each survey, all remnant (not destroyed during C-38 construction) river channels in each pool were traversed at idle speed and visually searched for adult snail kites. If an adult was located, behavioral cues (Bennetts et al. 1988) were used to locate nest sites. If a nest was found, each of the following characteristics was recorded: latitude and longitude, nest substrate, nest height, water depth, and status (eggs and/or young). If a non-nesting kite was observed, sex and age were recorded.

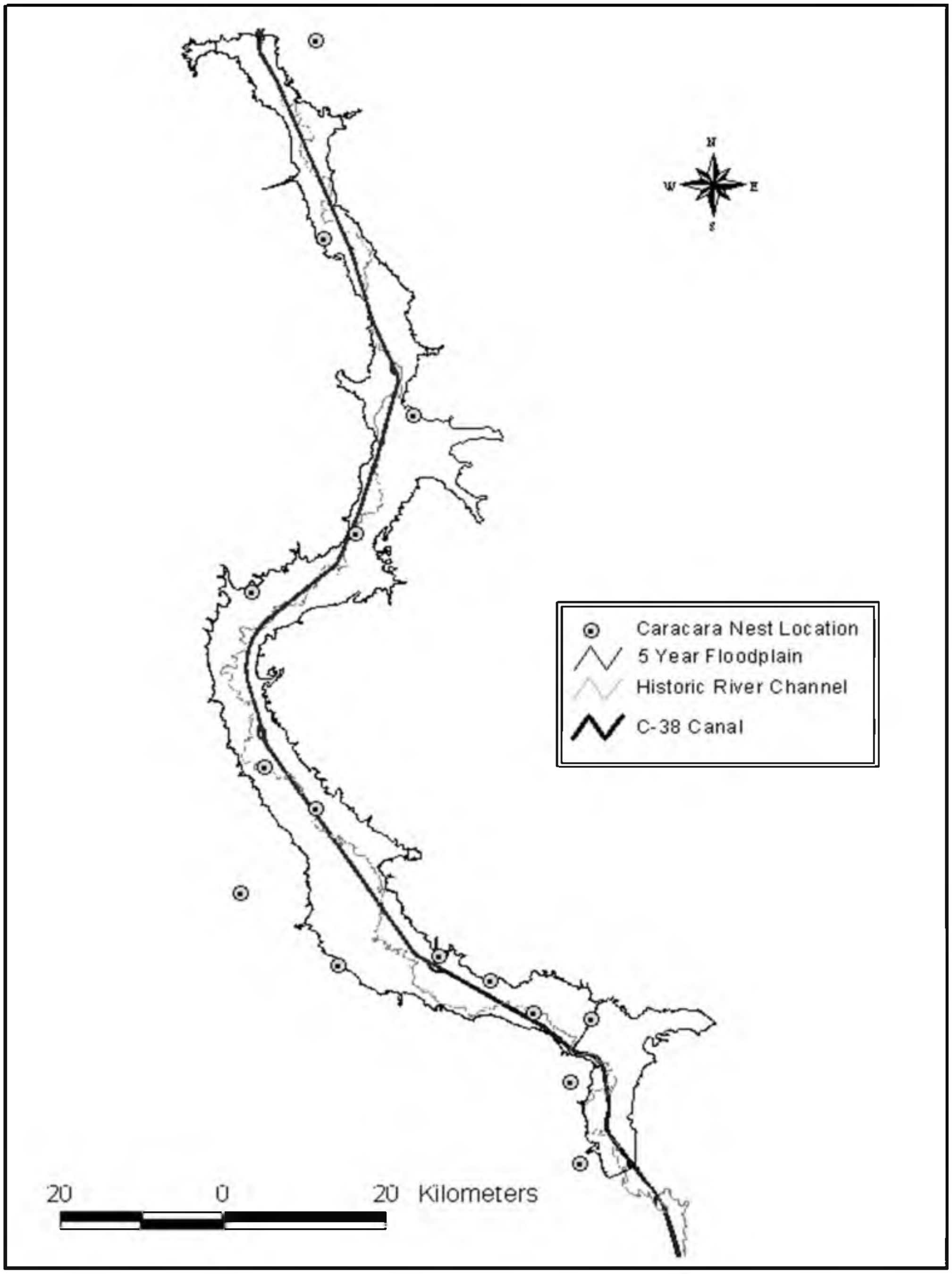

Figure 14-12. Audubon's crested caracara nest locations within and upland of the 100 year floodline of the Kissimmee River. A total of 15 active territories were found during baseline surveys from 1996–1999 and all had at least one nesting attempt during these years.

Results and Discussion

No snail kites were observed during any of the 1996–1997 surveys (n = 13). During this same time period, a single casual observation was made of one snail kite foraging over the Kissimmee floodplain (S. Melvin, personal observation). While the Kissimmee River falls within the current range of the snail kite (Stevenson and Anderson 1994), the lack of snail kite observations is not surprising due to the scarcity of available kite habitat in the channelized system. Snail kites have a highly specialized diet and are largely limited to freshwater marshes and littoral zones of lakes where their preferred prey, the apple snail (*Pomacea paludosa*), is found (Sykes et al. 1995). Nests are constructed in vegetation over water, with shrubs and small trees such as willow (*Salix* sp.) preferred (Sykes et al. 1995). Following channelization, the majority of broadleaf marsh and wetland shrub habitats of the floodplain were replaced by terrestrial communities (Carnal and Bousquin 2005) that were inappropriate for snail kite foraging and reproduction. Flat water profiles within each pool combined with decreases in land elevation from north to south allow some wetlands to persist near the tieback levees at the southern end of each pool (Carnal and Bousquin 2005). However, the vegetation of remnant marshes in these areas of stabilized water levels tends to grow in dense stands, with little open water (Bousquin 2005). Snail kites prefer a mixture of emergent vegetation and open water for foraging (Sykes et al. 1995), and may not be able to forage efficiently in remnant marshes.

Prior to channelization, the floodplain of the Kissimmee River was regularly inundated and contained substantial areas of willow and buttonbush (*Cephalanthus occidentalis*), as well as large expanses of broadleaf marsh (Anderson 2005, Carnal and Bousquin 2005). Thus, appropriate foraging and nesting habitat was available for snail kites. The Kissimmee River Restoration Project is designed to reestablish the flood-pulse cycle and restore large areas of broadleaf marsh and wetland shrub (U. S. Army Corps of Engineers 1991). Given the position of the Kissimmee River floodplain between known nesting areas in Upper Basin lakes and Lake Okeechobee (U. S. Fish and Wildlife Service 1999), it is likely that snail kites will use the system for foraging, as a travel corridor, and perhaps for nesting once it is restored.

CONCLUSIONS

Based on comparisons of reference and baseline information, channelization and headwater regulation of the Kissimmee River had profound impacts on its avifauna, including sharp decreases in the densities of aquatic long-legged wading birds and waterfowl, and decreased waterfowl species richness. Restoration should lead to increases in the densities of both of these groups and increased species richness of waterfowl. Of the four federally-listed bird species that utilized the pre-channelized system (wood stork, snail kite, bald eagle, Audubon's crested caracara), it is expected that the restoration will provide a net benefit for all but the caracara. Restoration expectations could not be developed for some important aspects of the Kissimmee River bird assemblage, including nesting effort by long-legged wading birds and level of use by migratory shorebirds. However, because of their importance as indicators of the health and ecological integrity of the restored river, monitoring for both will be continued. The restoration project is expected to reestablish hydrologic characteristics that typified the pre-channelized system, including a flood pulse that regularly inundates a substantial portion of the floodplain. Reestablishment of the plant and animal communities typical of the pre-channelized system is dependent on these hydrologic changes, and the length of time required for their recovery will also be linked weather conditions (e.g., drought) following project completion. Therefore, evaluation of avian responses to restoration will continue until 2017, five years following the project completion date of 2012.

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APPENDICES

CONTENTS

APPENDIX 2-1A

CHANGES TO THE KISSIMMEE WATERSHED
This appendix summarizes anthropogenic changes in the Kissimmee Basin that may have influenced hydrologic conditions.

Year	Changes	Source
1837	Fort Gardner built.	
	Fort Basinger built on the Kissimmee River.	
Late 1830s	Fort Kissimmee constructed.	
1856	Yates family is first family to settle in Shingle Creek.	4
1881	February 26, Hamilton Disston contracts with the State of Florida to drain lands in exchange for ownership of half the reclaimed land.	6
1882	January - Disston's company completes canal to connect Lake Okeechobee with the Caloosahatchee River.	
	July - Disston's company completes Southport Canal between Lake Tohopekaliga and Lake Cypress.	
1883	January - Disston's company begins work on St. Cloud Canal between Lake Tohopekaliga and East Lake Tohopekaliga.	
	Settlement of Allendale becomes Kissimmee City.	2
1884	September - St. Cloud Canal completed. Over a 30 day period, water levels approximately 3 feet exposing a sand beach between the cypress and the new waterline.	
1884	Canal from Lake Tohopekaliga to East Lake Tohopekaliga completed; East Lake Toho stages fall 36 inches in 30 days. Canal from Lake Tohopekaliga to Lake Cypress completed. Kissimmee River was streamlined by cutting off number of bends. Snag boat in operation on the river.	2
1885	June 5 - Regular steamship service from Fort Meyers to Kissimmee begins.	5
1909	Corps of Engineers completes navigation project to dredge a three foot navigation channel in the Kissimmee River to Istokpoga Creek; snag removal operations.	3
1921	Completion of railroad to Fort Meyers brings steamship era to an end.	5
1927	Last Federal maintenance for Kissimmee River navigation authority.	
	Last steam boat operation on the upper basin lakes.	
1938	During the Herbert Hoover Dike Project for Lake Okeechobee, U. S. Army Corps of Engineers creates a 6.5 mile levee from Lake Okeechobee along the east side of the Kissimmee River. Part of the flow was diverted through the eight mile barrow canal. The canal became known as Government Cut and the remnant river channel as Paradise Run.	3
	Istokpoga Canal dredged to create Istokpoga Canal.	
1947	G-85 sheet pile weir on Istokpoga Canal	8
1947	Zipprer Canal excavated to connect Lake Rosalie with Lake Kissimmee.	7
1962-71	Excavation of the C-38 canal.	10
1963	S-59 installed to regulate outflow from East Lake Tohopekaliga.	9
1963	S-61 installed to regulate outflow from Lake Tohopekaliga.	9
1964	S-65 installed in August to regulate the outflow from Lake Kissimmee.	9
1965	Installation of the S-68 on Lake Istokpoga in December.	
1970	C&SF construction completed in the upper basin lakes and interim operating schedules adopted for water control structures.	11
1971	Lake Tohopekaliga drawdown (Feb–Nov).	1
1976	Adoption of regulation schedules outlined in Report to the Governing Board on Regulatory Levels in the Upper Kissimmee Basin.	11
1977	Lake Kissimmee drawn down (Jan–Dec).	

APPENDIX 2-1A

Continued

Year	Changes	Source
1979	Lake Tohopekaliga drawdown (Jan-May).	
	Weir installed in Zipprer Canal.	7
1982	April - revised regulation schedules implemented.	11
1984	Sheet pile Weir 3 installed (Oct 1 - Nov 6) for Pool B Demonstration Project.	12
1985	Sheet pile Weir 2 installed (Feb 5 - Mar 16) for Pool B Demonstration Project.	12
	Sheet pile Weir 1 installed (May 2 - Jun 9) for Pool B Demonstration Project.	12
	Pool B stage fluctuation initiated on October 28. (Note Obeysekera and Loftin 1990 use September 1985).	12
1987	Lake Tohopekaliga drawdown (Jan-Sep) with muck removal.	
1990	Drawdown in East Lake Tohopekaliga.	
1992	Water Resources Development Act authorizes Kissimmee River Restoration Project.	
1994	Drawdown in Lake Jackson.	
1995	Drawdown in Lake Jackson.	
1996	Drawdown in lake Kissimmee.	
1997	Drawdown in Lake Jackson.	
2000	Drawdown in the Alligator Chain of Lakes.	
2001	June - Interim regulation schedule for S-65 implemented.	
2003-2004	Deviation to regulation schedules at S-61 and S-65 for Lake Toho drawdown project.	

1 = Dierberg and Williams 1989. 2 = Mueller 1966. 3 = U. S. Army Corps of Engineers General Design Memo 1969. 4 = Hetherington, A. 1980, The river of the long water. Mickler House Publications, Chuluota, Florida. 5 = Casselberry 1984. 6 = Blake 1980, 7 = FDEP 1998 Lake Kissimmee State Park Management Plan, Approved. 8 = Abtew 1992. 9 = Guardo 1992. 10 = U. S. Army Corps of Engineers 1991. 11 = U. S. Army Corps of Engineers 1996. 12 = Toth 1991.

APPENDIX 2-2A

Hydrologic data used for the analyses were obtained from the South Florida Water Management District's hydrologic database DBHYDRO. This table lists the sites, general location, type of data, the dbkey that identifies the data series in DBHYDRO that was used, and the start data for collecting data at a site.

Site	Location	Data type[1]	dbkey	Start date
S-65	Outlet of Lake Kissimmee	discharge	H0289	10/1/1933
Fort Kissimmee[2]				
PC62	East bank of UBX Run just north of the S-65B tieback	Stage		
		Discharge		
PC61	Backfilled canal just north of former location of S-65B	Stage	OB442	4/16/2002
		Rainfall	OH522	4/17/2002
PC55	West floodplain	Stage	J8927	11/24/1998
KRDR (PC54)	West bank of Montsdeoca Run	Stage	H7666	7/23/1997
PC53	East floodplain	Stage	J8929	10/7/1998
PC52	East floodplain	Stage	IV155	10/17/1998
PC51	East floodplain	Stage	J8931	9/3/1998
PC45	West floodplain	Stage	J8933	1/12/1999
PC44	West floodplain	Stage	J8935	11/12/1998
KRBN (PC43)	West bank of Oxbow13	Stage	FZ599	8/6/1997
PC42	East floodplain	Stage	J8937	9/14/1998
PC41	East floodplain	Stage	J8939	9/14/1998
PC35	West floodplain	Stage	J8941	10/29/1998
PC34	West floodplain	Stage	J8943	10/30/1998
PC33[3]	East bank of Micco Bluff Run	Stage	G6526	10/17/1997
		Discharge	G6527	11/25/1997
PC32	East floodplain	Stage	J8945	9/30/1998
PC31	East floodplain in Oak Creek	Stage	J8947	8/28/1998
PC22	West floodplain	Stage	J8949	11/24/1998
PC21	West floodplain	Stage	E9681	8/22/1996
PC12	West floodplain	Stage	J8951	10/12/1998
PC 11R	West bank of MacArthur Run	Stage	G6532	10/30/1997
S-65C	C-38 canal	Headwater	6957	4/29/1966
Fort Basinger[4]		Stage		
		Discharge		
S-65E	C-38 canal	Stage	240	1/1/1930
		Discharge	241	10/1/2028

APPENDIX 2-2A

Continued

[1]For data type, stage is mean daily stage, and discharge is mean daily discharge.

[2]Fort Kissimmee stage was recorded for 12/9/1941 - 9/30/1967 until the stage recorder was deactivated. Stage recorder was reactivated in 9/12/1984 and continued through the present. The three dbkeys were combined to create a record of stage at this location. Data prior to October 1, 1967, except for January 14 - 29, 1952, are reported in DBHydro in relative feet with measurements ranging from 0.03 feet to 12.14 feet. Previous unpublished analyses appear to add an offset of 37.98 feet to these data to convert relative stage to stage NGVD29. Division of Water Survey & Research (1952) report a gage elevation of 38.03 ft msl. Relative stage was recorded at Fort Kissimmee and these values were converted to absolute stage (ft NGVD) by adding an offset of 37.98 ft at Fort Kissimmee (J. Chamberlain, unpublished notes).

[3]PC33 discharge below 25 cfs were set to 0 cfs (J. Chamberlain, unpublished notes).

[4]Fort Basinger combines data from C38.Bas (from the original location) and C38Bas (from a new location). The original location of this station was destroyed by the excavation of the C-38. According to the coordinates given in DBHydro the location of C38.BAS would be approximately midway across the C-38 under the State Highway 98 bridge. C38BAS was located in a remnant channel approximately 1000 ft downstream from the original location of C38.BAS. Relative stage was recorded at Fort Basinger and these values were converted to absolute stage (ft NGVD 1929) by adding an offset of 24.64 ft to the values at Fort Basinger (J. Chamberlain, unpublished notes). Note that Division of Water Survey & Research (1952) reports a gage elevation of 24.73 ft msl for this station.

APPENDIX 2-3A

Characteristics of hurricanes and tropical storms passing over the Kissimmee Basin.

Year	Month	Name	Type	Category	Max wind	Type over basin	Comments
1873			H				
1878	Sept 7-11		H				
1887			TS			TS	
1891			TS			TS	
1892			TS			TS	
1896	Oct		H			H	
1897			TS			TS	
1898			TS			TS	
1909			TS			TS	
1909			TS			TS	
1925			TS			TS	
1928	Sept 6-20		H	4	100	H	
1933	July-Aug		H	1	95	TS	
1933	Sept 6-20		H	3	125	H	
1934	Aug		TS			TS	
1939	Aug		H	1	125	TS	
1945	15-Sep		H	3	196	H	8 inches rain
1947	17-Sep		H	4	155	Not over basin	
1947	12-Oct		H	1		Not over basin	
1948	Sep 21-22		H	3	122	Not over basin	
1948	Oct 4-8		H	3	100	Not over basin	
1949	Aug		H	3	153	H	
1950	Oct	King	H	3	150	H	
1951	Oct	How	H			TS	
1953	Oct	Hazel	TS			TS	
1959	Oct	Judith	TS			TS	
1964	Aug 26	Cleo	H	2	138	TS	
1968	June 4-5	Abby	H	1	90	TS	heavy rain
1981	Aug 17-18	Dennis	TS		55	TS	10-20 inches rain
1983	25-Aug	Barry	TS			TS	
1988	Nov 17-24	Keith	TS		65	TS	heavy rain, tornadoes
1994	Nov 16-17	Gordon	TS		50	TS	heavy rain
1995	2-Aug	Erin	H		85	TS	10 inches rain
1995	Aug 23-24	Jerry	TS			TS	15 inches rain
2001	Sep 14-15	Gabrielle	TS			TS	heavy rain

Note that Perrin et al. (1982, page 101) attribute high water levels to heavy rainfall associated with Hurricanes David and Frederick in September 1979.

APPENDIX 2-4A

Number of years, tropical storms and hurricanes during the reference and baseline periods based on Appendix 2-3A. Values in parentheses are the number of events per year.

	Period of record	Years	Tropical Storms	Hurricanes
Reference	1873 - 1961	88	11 (0.13)	15 (0.17)
Baseline	1962 - 1999	38	5 (0.13)	3 (0.08)

APPENDIX 2-5A

Methods for determining discharge.

Ultrasonic Velocity Meters

Velocity, discharge, and stage were recorded from November 1997 - May 1999 using an Acoustic Flowmeter for Remote Areas (AFFRA) at one remnant river sampling site in Pool C (PC33). Acoustic velocity meters are a reliable method to measure discharge in rivers, canals, and culverts (Laenen 1985). The AFFRA is well suited for monitoring small discharges in narrow channels with velocities varying from 0 to $\geq$ 9 ft/s (3 m/s) and water depth $\geq$ 1 foot (30 cm). The AFFRA uses acoustic principles to measure average velocity at the elevation of the acoustic path. The speed of sound is measured between two transducers with electrical pulses transformed into acoustic pulses and vice versa. The sound pulse travels in both directions along a known path length, diagonal to the streamflow. The average line velocity parallel to the streamflow path is calculated as

$$V_{line} = B/2\cos\Phi \; (1/t_{CA} - 1/t_{AC}),$$

where

V_{line}	average velocity at the elevation of the acoustic path,
Φ	angle of departure between streamflow and acoustic path,
t_{AC}	traveltime from A to C (upstream),
t_{CA}	traveltime from C to A (downstream),
B	length of acoustic path from A to C.

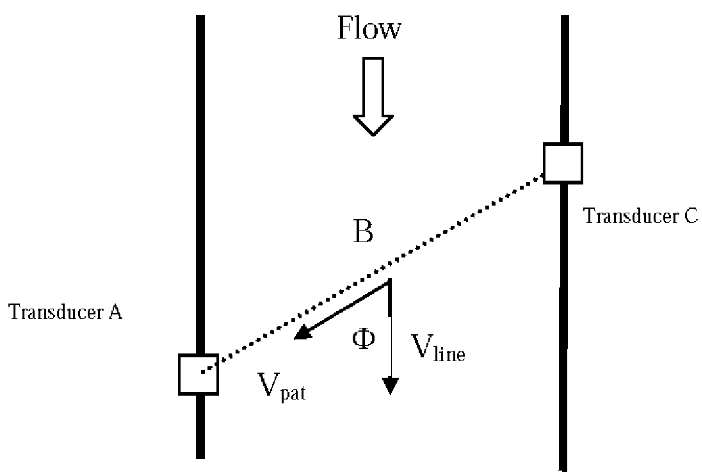


Figure 1. Schematic of UVM setup.

The average line velocity is then related to the average velocity of the cross section. Discharge is calculated by

$Q = K * V_{line} * A\,(d)$,

where,

Q	mean channel discharge,
K	calibration coefficient,
A (d)	area as a function of depth.

Discharge, line velocity, stage, automatic gain, speed of sound, and success rate values were stored in a data logger. The last three data items are used for quality assurance. The calibration coefficient, K, is determined through linear regression analysis and is discussed under the ADCP section.

UVM site selection was based on channel geometry, depth of water, presence of weeds, and construction constraints. To improve the reliability of the data collected, a straight reach of channel approximately 200 ft (60 m) long with a fairly uniform cross section was selected. This station will serve as a long-term monitoring site and provide information representative of other similar areas in the Kissimmee River. Aquatic weeds were controlled by spraying herbicides along the sides of the channel between transducer platforms to minimize fouling of the transducers.

Channel cross section data were collected when the site was established in November 1997 and at the end of the baseline period (May 1999). This frequency was adequate during baseline sampling when flows were minimal. The data showed negligible changes in cross sectional area and shape.

Acoustic Doppler Current Profiler

Discharge and cross section data were collected at four other permanent sampling locations using an acoustic doppler current profiler. Site selection was based on channel geometry, depth of water, presence of weeds, and longitudinal location within the pool. Where possible, sites were established near existing water quality, vegetation, and hydrogeomorphic monitoring stations. One site was established in the Ice Cream Slough remnant river run in the middle of Pool A, but attempts to measure flow at this site with the ADCP were unsuccessful due to very low flows and extensive submerged vegetation. Thus, no discharge data were collected in remnant river channels in Pool A. Three sites were established in Pool C, lower and upper segments of the Micco Bluff Run and in the Monstdeoca Run. Additional sites were investigated in Pool C, but ADCP measurements could not be collected due to extensive submerged vegetation.

To collect discharge data, a tag line was strung across the river between two permanent poles. The ADCP was mounted over the bow of the boat with the acoustic transducers submerged. Data were collected while the boat was pulled along the tag line, bow facing upstream. The ADCP transmits bursts of sound into the water column, which are scattered back to the instrument by particulate matter suspended in the flowing water. The ADCP listens for the returning signal and assigns depth and velocity to the signal based on the change in the frequency caused by the moving particles. This change in frequency is referred to as a Doppler shift. Communication with the ADCP for set up and data recording is accomplished with a portable computer using manufacturer supplied software, hardware, and communication cables.

At each transect, a minimum of three passes across the channel were completed. If discharge measurements during these passes were within 5-10% of each other, no more passes were taken; if they were not, additional passes across the channel were conducted until three measurements were within 5-10% of each other. During each pass the following data were recorded: discharge, start time, stop time, distance to left bank, distance to right bank, make good (distance of good ADCP measurements), configuration file name, and raw data file name.

Bathymetric surveys of the discharge monitoring sites, accurate to one tenth of a foot, were completed twice during the baseline period. Channel bottom profile data was used to calculate average velocities and to document changes in cross sectional area and shape over time. Additional data, including wind direction, wind speed, flow visibility, weather, and presence of weeds were collected at each transect to describe general conditions at the time of data collection.

The ADCP site 14.062 is located approximately 100 ft (30 m) upstream of the UVM site, and was used to calibrate discharge from the UVM. The UVM calibration coefficient was calculated using linear regression of flows measured with the ADCP and the UVM. Flows from the UVM were available during four of the eight ADCP sampling events. During the remaining four events, the UVM was missing data and flow could not be calculated, or values from the ADCP were unreliable due to windy conditions. A simple linear regression model with y intercept equal to zero was derived and used to adjust discharge at the UVM site (Figure 1). Zero flow events are not used in the calibration, but are accounted for by setting the y intercept equal to zero. Although the UVM and ADCP are capable of accurately recording very low flows, data can be less accurate at near zero discharge. Due to uncertainties associated with very low flow conditions, discharges at the UVM that were $\leq$ 25.0 cfs were considered to be zero.

APPENDIX 3-1A

Characteristics of geomorphology transects in the Kissimmee River. Pattern indicates if transect is located in a curved (C) or straight (S) section of channel. Width (m) is the distance between transect markers, which approximates channel width. All transects will be affected by Phase I of restoration except those identified as being affected by Phase II, or that were destroyed during Phase I construction.

Area	Pool	Run	Transect	Pattern	Width	Comment
Impact	C	MacArthur Run	9	C	41.2	Phase II
Impact	C	MacArthur Run	9.1	C	35.4	Phase II
Impact	C	MacArthur Run	9.2	C	42	Phase II
Impact	C	MacArthur Run	9.3	C	33.3	Phase II
Impact	C	MacArthur Run	9.4	S	30.3	Phase II
Impact	C	MacArthur Run	9.5	C	37.8	Phase II
Impact	C	MacArthur Run	10	S	42.7	Phase II
Impact	C	MacArthur Run	10.1	S	36.8	Phase II
Impact	C	MacArthur Run	10.2	C	42.2	
Impact	C	MacArthur Run	11.1	C	37.1	
Impact	C	MacArthur Run	11.2	C	34.9	
Impact	C	MacArthur Run	11.3	C	31.4	
Impact	C	MacArthur Run	11.4	S	38	
Impact	C	MacArthur Run	11.5	S	37	
Impact	C	MacArthur Run	11.6	S	39.1	
Impact	C	MacArthur Run	11.7	C	36.9	
Impact	C	MacArthur Run	11.8	C	40	
Impact	C	MacArthur Run	12	C	20.3	
Impact	C	MacArthur Run	13	S	42.1	
Impact	C	MacArthur Run	13.1	C	32.6	
Impact	C	MacArthur Run	13.2	C	34	
Impact	C	MacArthur Run	13.3	S	34.4	
Impact	C	MacArthur Run	14	S	35.4	
Impact	C	MacArthur Run	14.05	S	26.9	Destroyed
Recarved	C	Loftin Run	14.0501	S	45.2	
Recarved	C	Loftin Run	14.0502	C	41.5	
Recarved	C	Loftin Run	14.0503	C	42.5	
Recarved	C	Loftin Run	14.0504	C	47.6	
Recarved	C	Loftin Run	14.0505	S	45.9	
Recarved	C	Loftin-Micco Connector	14.0506	S	88	
Recarved	C	Loftin-Micco Connector	14.0507	S	54	
Impact	C	Micco Bluff Run	14.06	S	41.2	
Impact	C	Micco Bluff Run	14.061	C	44.8	
Impact	C	Micco Bluff Run	14.062	S	39.2	
Impact	C	Micco Bluff Run	14.063	C	62	
Impact	C	Micco Bluff Run	14.064	C	45.7	
Impact	C	Micco Bluff Run	14.065	S	40.2	
Impact	C	Micco Bluff Run	14.066	C	36.7	
Impact	C	Micco Bluff Run	14.068	C	43.2	
Impact	C	Micco Bluff Run	14.069	S	37.5	
Impact	C	Micco Bluff Run	14.07	C	39.2	
Impact	C	Micco Bluff Run	14.071	C	32.2	
Impact	C	Micco Bluff Run	14.072	C	14	
Impact	C	Micco Bluff Run	14.073	S	26.5	
Impact	C	Micco Bluff Run	14.074	S	27.4	

APPENDIX 3-1A

Continued.

Area	Pool	Run	Transect	Pattern	Width	Comment
Impact	C	Micco Bluff Run	14.075	S	31.7	
Impact	C	Micco Bluff Run	14.076	C	31	
Impact	C	Micco Bluff Run	14.077	C	38.4	
Impact	C	Micco Bluff Run	14.078	S	29.2	
Impact	C	Micco Bluff Run	14.079	S	37.6	
Impact	C	Micco Bluff Run	14.08	S	29	
Impact	C	Micco Bluff Run	14.081	C	39.9	
Impact	C	Micco Bluff Run	14.082	C	25.6	
Impact	C	Micco Bluff Run	14.083	C	29.3	
Impact	C	Micco Bluff Run	14.084	S	31.9	
Impact	C	Micco Bluff Run	14.085	S	28.1	
Impact	C	Micco Bluff Run	14.086	C	35.4	
Impact	C	Micco Bluff Run	14.087	C	31.7	
Impact	C	Micco Bluff Run	14.088	C	32.5	
Recarved	C	Oxbow13-Micco Connector	14.08801	S	68.8	
Impact	C	Oxbow13 Run	14.089	C	38.7	
Impact	C	Oxbow13 Run	14.09	C	41.2	
Impact	C	Oxbow13 Run	14.091	C	38.6	
Recarved	C	Oxbow13 (recarved)	14.09101	C	42.3	
Recarved	C	Oxbow13 (recarved)	14.09102	C	58.5	
Recarved	C	Oxbow13 (recarved)	14.09103	C	50.1	
Impact	C	Oxbow13 Run	14.092	S	34.8	
Impact	C	Oxbow13 Run	14.093	S	38.7	
Impact	C	Oxbow13 Run	14.094	S	37.2	
Impact	C	Oxbow13 Run	14.095	S	42.4	
Impact	C	Oxbow13 Run	14.096	C	58.6	Destroyed
Impact	C	Oxbow13 Run	14.097	C	46	Destroyed
Impact	C	Oxbow13 Run	14.098	C	32.6	Destroyed
Impact	C	Oxbow13 Run	14.099	C	29	Destroyed
Recarved	C	Strayer Run	14.09901	C	39.9	
Recarved	C	Strayer Run	14.09902	C	38.1	
Recarved	C	Strayer Run	14.09903	S	39.5	
Recarved	C	Strayer Run	14.09904	S	39.1	
Recarved	C	Strayer Run	14.09905	S	46.1	
Recarved	C	Strayer Run	14.09906	C	40.1	
Recarved	C	Strayer-Fulford Connector	14.09907	S	53.5	
Recarved	C	Fulford Run	14.09908	C	42.4	
Recarved	C	Fulford Run	14.09909	C	53	
Recarved	C	Fulford Run	14.0991	C	50.8	
Recarved	C	Montsdeoca-Fulford Connector	14.09911	S	87.5	
Impact	C	Montsdeoca Run	14.1	C	37	Destroyed
Impact	C	Montsdeoca Run	14.2	C	30.5	
Impact	C	Montsdeoca Run	15.1	C	33.3	
Impact	C	Montsdeoca Run	15.2	S	28.7	
Impact	C	Montsdeoca Run	15.3	C	31	
Impact	C	Montsdeoca Run	15.4	C	33.5	
Impact	C	Montsdeoca Run	16	S	36.1	

APPENDIX 3-1A

Continued.

Area	Pool	Run	Transect	Pattern	Width	Comment
Impact	C	Montsdeoca Run	16.1	C	23.7	
Impact	C	Montsdeoca Run	16.2	C	28.5	
Impact	C	Montsdeoca Run	16.3	C	33.4	
Impact	C	Montsdeoca Run	17	C	30.2	
Impact	C	Montsdeoca Run	17.1	S	36.8	
Impact	C	Montsdeoca Run	17.2	C	34.5	
Impact	C	Montsdeoca Run	17.3	S	29	
Impact	C	Montsdeoca Run	18	S	34	
Impact	C	Montsdeoca Run	18.1	C	27.9	
Impact	C	Montsdeoca Run	18.2	S	16.5	
Impact	C	Montsdeoca Run	18.3	S	11.9	Destroyed
Impact	B	UBX Run	19.1	S	37.8	Destroyed
Impact	B	UBX Run	19.2	S	41.3	
Impact	B	UBX Run	19.3	S	39	
Impact	B	UBX Run	19.4	C	37.8	
Impact	B	UBX Run	19.5	C	38.8	
Control	A	Persimmon Mound Run	65	S	44.5	
Control	A	Persimmon Mound Run	66	C	44.5	
Control	A	Persimmon Mound Run	67	C	39.2	
Control	A	Persimmon Mound Run	68	C	34	
Control	A	Persimmon Mound Run	69	S	47.1	
Control	A	Persimmon Mound Run	70	S	33.7	
Control	A	Persimmon Mound Run	71	S	42.2	
Control	A	Persimmon Mound Run	72	C	41	
Control	A	Persimmon Mound Run	73	C	40.7	
Control	A	Persimmon Mound Run	74	S	46	
Control	A	Rattlesnake Hammock Run	75	S	39.5	
Control	A	Rattlesnake Hammock Run	76	S	33.5	
Control	A	Rattlesnake Hammock Run	77	C	36	
Control	A	Rattlesnake Hammock Run	78	S	41.4	
Control	A	Rattlesnake Hammock Run	79	S	36.5	
Control	A	Ice Cream Slough Run	81	S	36.1	
Control	A	Ice Cream Slough Run	82	C	36.4	
Control	A	Ice Cream Slough Run	83	C	35.9	
Control	A	Ice Cream Slough Run	84	C	30.9	
Control	A	Ice Cream Slough Run	85	S	34.1	
Control	A	Ice Cream Slough Run	86	S	31.3	

APPENDIX 3-2A

Reference data used to evaluate the effects of channelization.

Oxbow	Transect	n	Thick (cm)	SAND (%)	Position(m)	Zthal	Thal (cm)
Lower	1.0	27	25	22	16.5	337	51
Lower	2.0	21	4	67	22.5	398	30
Lower	2.5	18	0	89	12	479	0
Lower	3.0	26	1	77	28.5	276	0
Lower	3.5	21	4	67	21	446	0
Lower	4.0	29	8	52	31.5	419	0
Middle	1.0	18	5	28	18	200	0
Middle	1.5	20	1	65	7.5	444	1
Middle	2.0	16	8	56	13.5	234	0
Middle	2.5	19	0	89	18	375	0
Middle	3.0	19	3	32	19.5	244	1
Middle	4.0	30	2	87	15	341	0
Upper	2.0	20	17	35	21	324	40
Upper	3.0	22	3	82	16.5	282	0
Upper	4.0	30	3	73	25.5	363	29
Upper	5.0	22	0	68	16.5	182	0
Upper	6.0	18	1	56	21	320	0
Upper	7.0	17	6	24	16.5	419	12
Upper	8.0	17	7	35	10.5	435	43
Upper	9.0	17	6	12	7.5	276	2
Upper	10.0	14	10	21	10.5	304	1
Upper	11.0	15	2	40	6	187	0
Upper	12.0	13	2	92	7.5	262	1
Upper	13.0	17	0	65	6	313	1

APPENDIX 5-1A

Summary statistics for Kissimmee River water quality monitoring stations, March 18, 1996 to June 8, 1999.

	Station	Statistic	Turbidity (NTU)	Tot. Susp. Solids (mg/L)	Chlor. a (mg/m^3)	Color (Pt-Co units)	Tot. Org. C (mg/L)	Dis. Org. C (mg/L)	Total P (mg/L)	Sol. React. P (mg/L)
Pool A	Ice Cream Slough	Median	2.5	< 3.0	17.3	84	19.9	18.8	0.069	0.022
	Run (KREA 97)	Mean	2.5	3.4	27.5	88	19.9	19.1	0.078	0.029
		Std. Dev.	1.1	2.8	28.2	31	3.1	3.0	0.037	0.025
	Pool A	Min.	0.9	< 3.0	1.8	38	14.2	14.3	0.025	0.002
		Max.	6.5	11.0	120.7	172	26.2	26.0	0.185	0.106
		N	31	31	28	30	30	31	28	28
	Rattlesnake	Median	2.2	< 3.0	9.2	151	22.5	22.2	0.040	0.015
	Hammock Run	Mean	2.3	2.2	12.1	165	23.6	23.4	0.051	0.018
	(KREA 91)	Std. Dev.	1.0	1.4	10.8	85	5.9	5.7	0.028	0.019
		Min.	0.9	< 3.0	1.0	75	17.2	14.8	0.018	0.002
	Pool A	Max.	4.5	7.0	50.9	561	49.8	46.4	0.123	0.096
		N	31	31	31	31	31	30	28	29
	Schoolhouse	Median	2.4	< 3.0	9.9	101	17.2	16.7	0.067	0.013
	Run (KREA 92)	Mean	3.5	3.7	13.4	113	18.6	18.3	0.075	0.029
		Std. Dev.	3.2	5.1	11.7	71	4.6	4.4	0.037	0.035
	Pool A	Min.	0.9	< 3.0	2.0	32	13.4	13.5	0.026	0.002
		Max.	17.3	25.0	54.8	318	31.6	31	0.206	0.162
		N	35	35	33	35	34	34	32	32
	C-38 at S-65A	Median	3.0	3.0	12.0	99	17.2	17.1	0.067	0.015
		Mean	5.1	5.7	18.5	117	18.4	18.5	0.073	0.025
	Pool A	Std. Dev.	9.5	6.2	34.4	70	4.4	4.4	0.034	0.024
		Min.	1.1	< 3.0	0.5	30	12.9	13.3	0.036	0.002
		Max.	87.0	30.0	308.6	292	32.1	30.1	0.296	0.085
		N	85	84	83	84	85	84	79	80
Pool C	Montsdeoca	Median	1.2	< 3.0	3.3	88	17.8	17.8	0.034	0.012
	Run (KREA 98)	Mean	1.3	1.6	8.3	94	17.9	17.6	0.038	0.016
		Std. Dev.	0.8	0.4	13.4	32	2.3	2.4	0.023	0.011
	Pool C	Min.	0.6	< 3.0	0.5	47	14.0	14.0	0.017	0.002
		Max.	3.6	3.0	52.4	158	22.4	22.1	0.122	0.05
		N	17	18	17	17	18	17	18	16
	Oxbow 13	Median	1.9	< 3.0	11.8	129	20.0	18.5	0.048	0.014
	(KREA 93)	Mean	2.1	2.3	12.6	149	22.0	21.1	0.056	0.020
		Std. Dev.	0.8	2.2	8.8	92	7.2	7.3	0.025	0.017
	Pool C	Min.	1.0	< 3.0	1.0	40	12.3	11.6	0.018	0.002
		Max.	3.7	13.0	45.4	358	41.4	44.9	0.121	0.078
		N	32	33	33	32	33	31	31	30
	Micco Bluff	Median	1.6	< 3.0	6.5	142	22.8	22.3	0.071	0.038
	Run (KREA 94)	Mean	1.9	2.5	13.6	164	24.0	23.6	0.094	0.057
		Std. Dev.	1.4	3.1	20.1	77	6.0	5.6	0.081	0.066
	Pool C	Min.	0.6	< 3.0	0.5	48	16.1	15.9	0.029	0.004
		Max.	5.5	18.0	82.0	373	40.9	37.7	0.411	0.318
		N	31	32	31	31	32	31	30	29
	MacArthur	Median	1.6	< 3.0	5.9	87	18.5	18.3	0.047	0.012
	Run (KREA 95)	Mean	1.8	1.9	8.0	133	21.1	21.3	0.055	0.025
		Std. Dev.	1.2	0.9	7.2	106	9.3	9.2	0.033	0.030
	Pool C	Min.	0.5	< 3.0	0.5	32	10.3	9.8	0.015	0.002
		Max.	6.3	5.0	29.8	394	41.8	42.5	0.152	0.105
		N	34	35	35	34	34	34	33	32
	C-38 at S-65C	Median	2.0	< 3.0	8.0	103	17.0	17.5	0.056	0.020
		Mean	2.5	2.8	11.5	117	18.0	18.4	0.062	0.027
	Pool C	Std. Dev.	1.4	2.5	13.7	59	3.6	3.7	0.023	0.025
		Min.	0.9	< 3.0	0.5	34	13.4	12.9	0.035	0.002
		Max.	7.0	15.0	105.9	273	29.5	29.6	0.196	0.123
		N	85	84	83	85	85	84	79	81

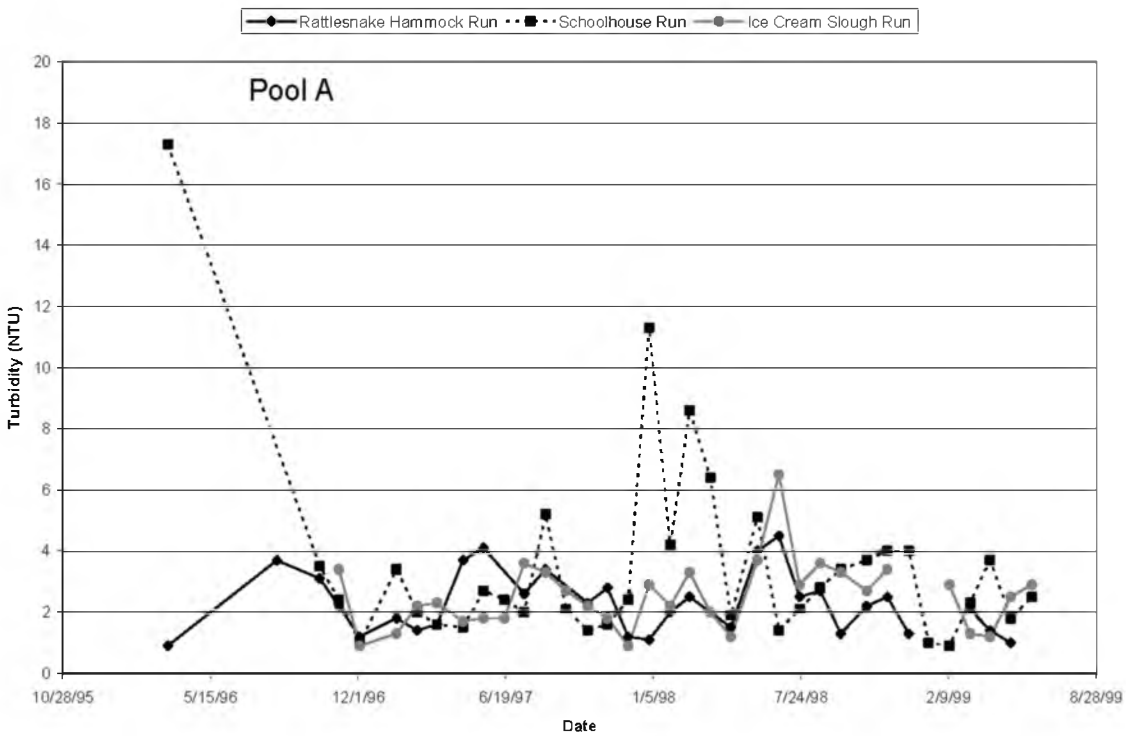
APPENDIX 5-1A

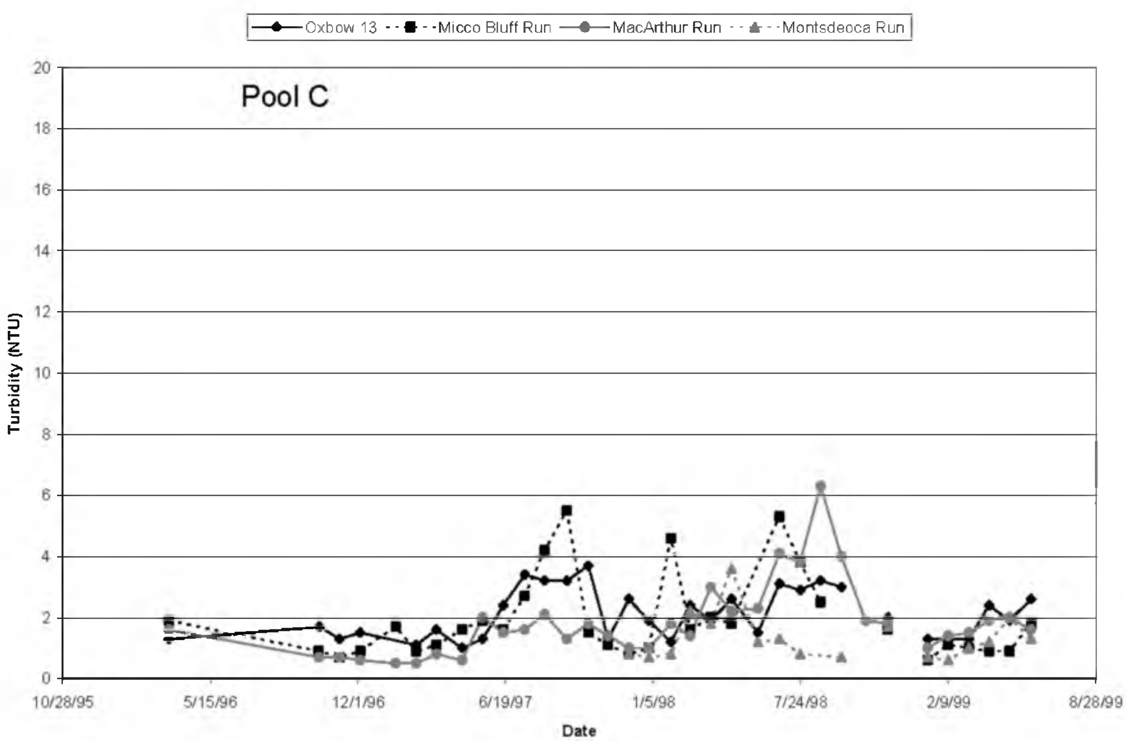
Continued

	Station	Statistic	Total N (mg/L)	Organic N (mg/L)	Dis. inorg. N (mg/L)	Sp. Cond. (microS/cm)	Chloride (mg/L)	pH	Alkalinity (mg $CaCO_3$/L)
Pool A	Ice Cream Slough	Median	1.33	1.27	0.03	187	15.7	6.50	61.9
	Run (KREA 97)	Mean	1.30	1.27	0.05	216	14.7	6.47	72.4
		Std. Dev.	0.40	0.42	0.05	82	3.2	0.28	45.8
	Pool A	Min.	0.55	0.51	0.01	111	8.5	5.96	19.1
		Max.	2.00	1.96	0.18	413	19.4	7.32	185.8
		N	24	22	22	30	29	30	31
	Rattlesnake	Median	1.16	1.09	0.03	110	11.0	5.88	34.0
	Hammock Run	Mean	1.18	1.08	0.05	135	10.9	5.95	38.5
	(KREA 91)	Std. Dev.	0.38	0.40	0.04	76	3.5	0.32	27.0
		Min.	0.50	0.25	0.01	54	6.2	5.50	13.9
	Pool A	Max.	1.98	1.88	0.18	332	16.6	7.12	124.7
		N	26	24	24	31	30	30	31
	Schoolhouse	Median	1.15	1.07	0.07	120	15.3	6.31	22.3
	Run (KREA 92)	Mean	1.20	1.08	0.07	118	14.2	6.36	22.6
		Std. Dev.	0.29	0.29	0.05	24	3.5	0.32	6.1
	Pool A	Min.	0.51	0.25	0.01	73	0.9	5.89	11.9
		Max.	1.79	1.65	0.17	162	20.6	7.06	37.3
		N	28	25	25	34	34	33	34
	C-38 at S-65A	Median	1.13	1.04	0.09	130	15.7	6.87	22.2
		Mean	1.25	1.14	0.10	125	15.2	6.86	23.0
	Pool A	Std. Dev.	0.54	0.54	0.07	27	3.2	0.47	6.6
		Min.	0.52	0.25	0.01	59	7.0	4.80	11.2
		Max.	5.00	4.95	0.34	175	23.3	7.74	43.3
		N	69	73	68	82	83	83	85
Pool C	Montsdeoca	Median	1.12	0.95	0.16	277	42.2	6.35	29.8
	Run (KREA 98)	Mean	1.22	0.95	0.27	308	40.4	6.31	30.1
		Std. Dev.	0.34	0.18	0.31	118	17.6	0.28	14.6
	Pool C	Min.	0.83	0.64	0.01	124	15.2	5.79	8.7
		Max.	1.87	1.25	0.94	552	78	6.76	53.9
		N	16	16	16	17	17	17	18
	Oxbow 13	Median	1.26	1.18	0.04	135	15.5	6.11	23.6
	(KREA 93)	Mean	1.26	1.20	0.05	137	16.8	6.11	24.3
		Std. Dev.	0.34	0.33	0.05	51	7.8	0.34	5.8
	Pool C	Min.	0.73	0.70	0.01	54	6.7	5.42	10.4
		Max.	2.26	2.18	0.24	382	56.0	6.81	35.9
		N	26	26	26	32	33	32	33
	Micco Bluff	Median	1.18	1.13	0.04	148	16.7	6.32	31.7
	Run (KREA 94)	Mean	1.31	1.24	0.05	137	16.3	6.34	28.8
		Std. Dev.	0.36	0.37	0.03	34	3.5	0.30	8.3
	Pool C	Min.	0.86	0.62	0.01	52	7.6	5.53	10.9
		Max.	1.96	1.93	0.13	187	22.1	6.94	44.3
		N	23	25	23	31	32	31	32
	MacArthur	Median	1.17	1.13	0.09	214	25.8	6.23	18.1
	Run (KREA 95)	Mean	1.28	1.15	0.11	228	31.4	6.20	18.5
		Std. Dev.	0.40	0.33	0.10	82	14.0	0.42	5.7
	Pool C	Min.	0.78	0.67	0.01	90	14.2	5.42	8.9
		Max.	2.34	2.09	0.38	428	64.8	6.97	30.0
		N	27	28	27	34	35	33	35
	C-38 at S-65C	Median	1.13	0.99	0.14	133	15.7	6.85	23.0
		Mean	1.14	0.99	0.14	133	15.1	6.82	23.7
	Pool C	Std. Dev.	0.21	0.23	0.07	27	3.3	0.47	6.2
		Min.	0.51	0.25	0.01	63	0.7	4.84	11.4
		Max.	1.75	1.48	0.36	245	20.9	8.15	57.2
		N	70	72	68	81	84	82	85

APPENDIX 5-2A

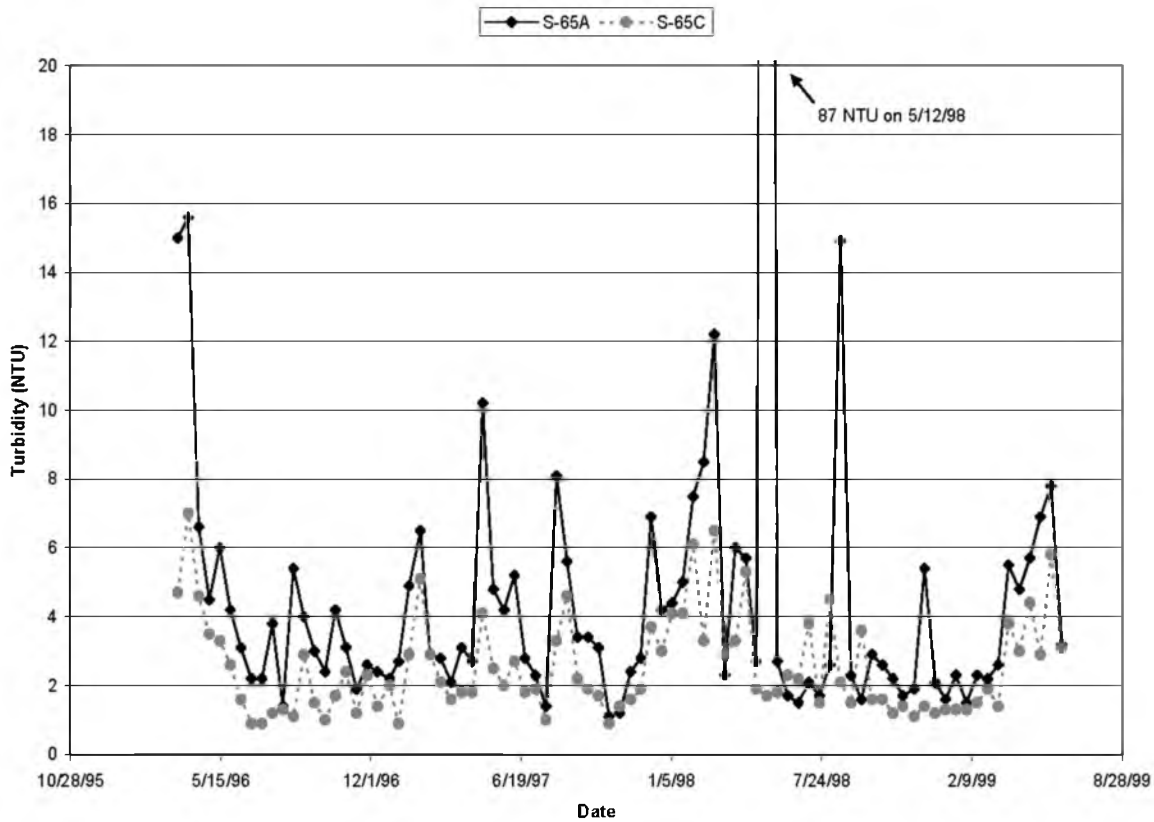
Turbidity in Pool A and Pool C remnant runs (0.5 m depth).





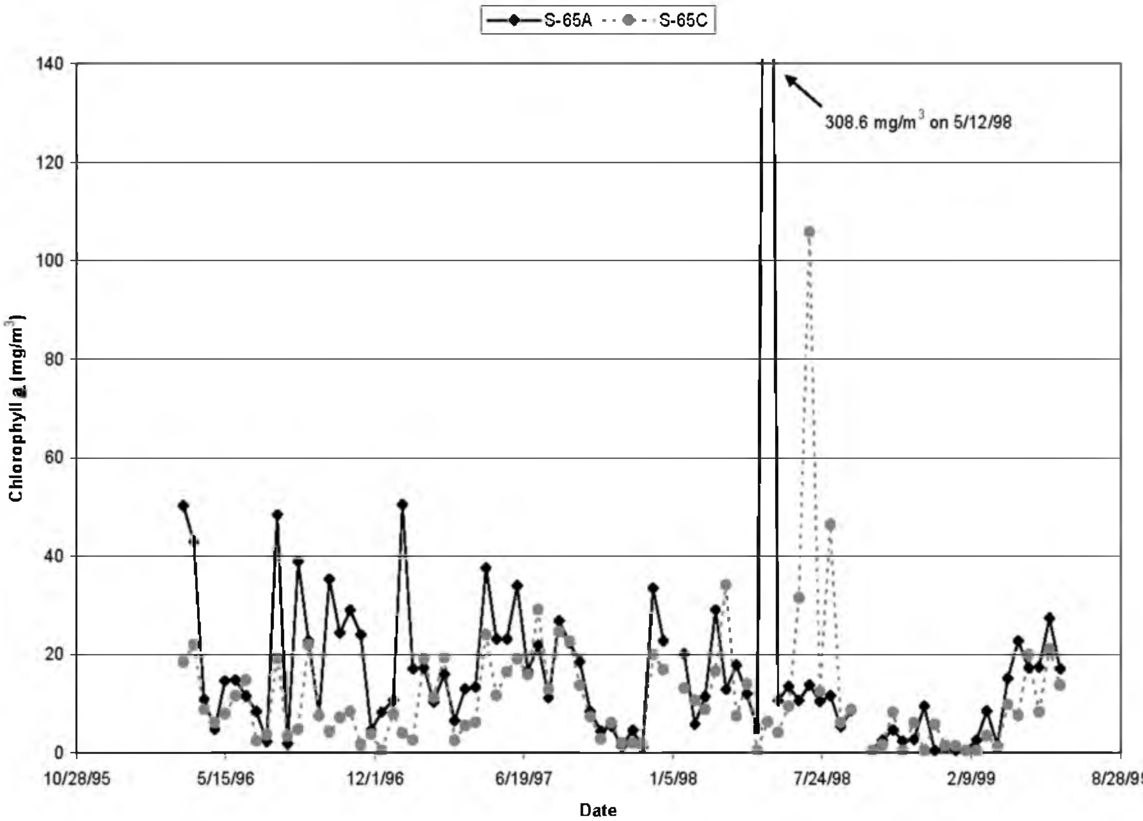
APPENDIX 5-3A

Turbidity in C-38 (0.5 m depth).



APPENDIX 5-4A

Chlorophyll *a* concentrations in C-38 (0.5 m depth).



APPENDIX 5-5A

Comparison of S-65A and S-65C water quality data from different periods.

Station	Statistic	Turbidity (NTU)	Tot. Susp. Solids (mg/L)	Chlor. a (mg/m^3)	Color (Pt-Co units)	Tot. Org. C (mg/L)	Dis. Org. C (mg/L)	Total P (mg/L)	Sol. React. P (mg/L)
S-65A:	6/13/73 - 6/8/99								
	Median	2.7	3.0	13.2	93	17.4	17.2	0.045	0.007
	Mean	3.7	8.4	23.0	111	18.3	18.9	0.056	0.017
	Std. Dev.	5.7	72.6	42.6	71	5.1	5.1	0.036	0.025
	Min.	0.4	0.5	0.5	16	4.9	13.3	0.010	0.001
	Max.	87	1447	309	409	42.6	42.8	0.333	0.243
	N	452	397	88	444	142	90	473	475
S-65A:	6/13/73 - 3/5/96								
	Median	2.5	3.0	----	91	17.6	22.1	0.041	0.006
	Mean	3.3	9.1	----	109	18.0	23.9	0.052	0.016
	Std. Dev.	4.4	81.7	----	71	6.0	9.8	0.035	0.025
	Min.	0.4	0.5	----	16	4.9	15.1	0.010	0.001
	Max.	72	1447	----	409	42.6	42.8	0.333	0.243
	N	367	313	5*	360	57	6	394	395
S-65A:	3/19/96 - 6/8/99								
	Median	3.0	3.0	12.0	99	17.2	17.1	0.067	0.015
	Mean	5.1	5.7	18.5	117	18.4	18.5	0.073	0.025
	Std. Dev.	9.5	6.2	34.4	70	4.4	4.4	0.034	0.024
	Min.	1.1	1.5	0.5	30	12.9	13.3	0.036	0.002
	Max.	87	30	309	292	32.1	30.1	0.296	0.085
	N	85	84	83	84	85	84	79	80
S-65C:	6/13/73 - 6/8/99								
	Median	1.9	1.5	8.4	100	17.1	17.4	0.047	0.011
	Mean	2.5	3.6	13.1	115	17.8	18.4	0.054	0.018
	Std. Dev.	1.9	10.7	15.8	65	4.4	4.6	0.067	0.020
	Min.	0.5	0.5	0.5	20	5.6	7.2	0.016	0.001
	Max.	22	206	105.9	431	37.1	41.6	1.418	0.125
	N	451	394	88	443	141	91	475	474
S-65C:	6/13/73 - 3/5/96								
	Median	1.9	2.0	----	100	17.7	16.5	0.044	0.010
	Mean	2.4	3.8	----	114	17.4	18.9	0.053	0.016
	Std. Dev.	2.0	12.0	----	66	5.5	11.2	0.073	0.018
	Min.	0.5	0.5	----	20	5.6	7.2	0.016	0.001
	Max.	22	206	----	431	37.1	41.6	1.418	0.125
	N	366	310	5*	358	56	7	396	393
S-65C:	3/19/96 - 6/8/99								
	Median	2.0	1.5	8.0	103	17.0	17.5	0.056	0.020
	Mean	2.5	2.8	11.5	117	18.0	18.4	0.062	0.027
	Std. Dev.	1.4	2.5	13.7	59	3.6	3.7	0.023	0.025
	Min.	0.9	1.5	0.5	34	13.4	12.9	0.035	0.002
	Max.	7.0	15.0	105.9	273	29.5	29.6	0.196	0.123
	N	85	84	83	85	85	84	79	81

* Insufficient data for comparison.

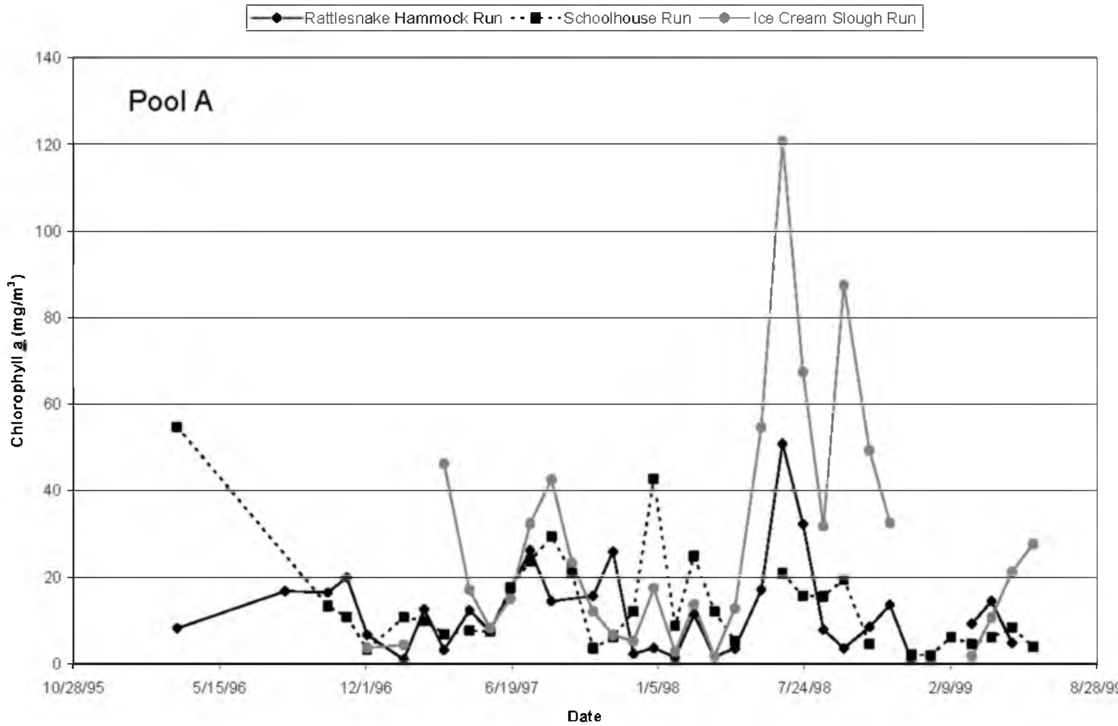
APPENDIX 5-5A

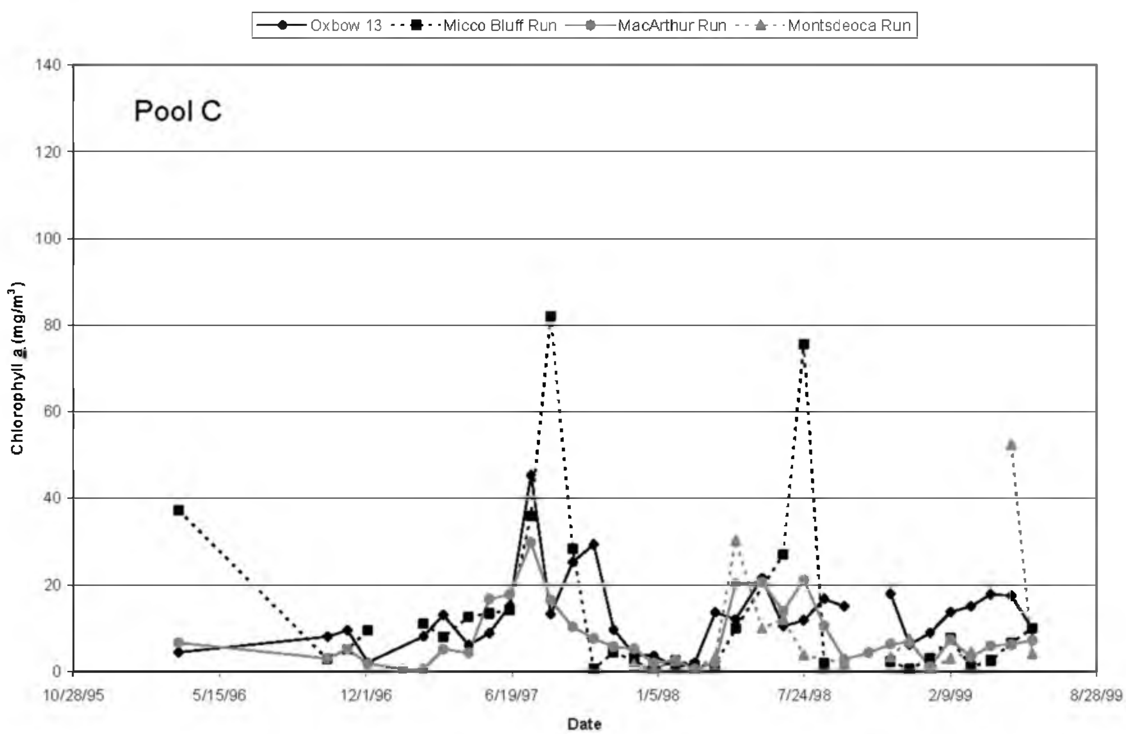
Continued

Station	Statistic	Total N (mg/L)	Organic N (mg/L)	Dis. Inorg. N (mg/L)	Sp. Cond. (microS/cm)	Chloride (mg/L)	pH	Alkalinity (mg $CaCO_3$/L)
S-65A:	*6/13/73 - 6/8/99*							
	Median	1.20	1.11	0.07	144	18.6	6.77	25.0
	Mean	1.29	1.20	0.09	154	19.2	6.73	25.2
	Std. Dev.	0.47	0.47	0.10	98	6.0	0.56	9.5
	Min.	0.13	0.20	0.01	59	7.0	4.66	2.5
	Max.	5.00	4.95	0.95	1213	63.9	10.6	57.7
	N	463	467	462	461	480	453	478
S-65A:	*6/13/73 - 3/5/96*							
	Median	1.21	1.13	0.06	149	20.0	6.74	26.0
	Mean	1.30	1.21	0.09	161	20.1	6.70	25.7
	Std. Dev.	0.46	0.45	0.11	107	6.1	0.58	10.0
	Min.	0.13	0.20	0.01	62	7.8	4.66	2.5
	Max.	4.31	4.24	0.95	1213	63.9	10.6	57.7
	N	394	394	394	379	397	370	393
S-65A:	*3/19/96 - 6/8/99*							
	Median	1.13	1.04	0.09	130	15.7	6.87	22.2
	Mean	1.25	1.14	0.10	125	15.2	6.86	23.0
	Std. Dev.	0.54	0.54	0.07	27	3.2	0.47	6.6
	Min.	0.52	0.25	0.01	59	7.0	4.80	11.2
	Max.	5.00	4.95	0.34	175	23.3	7.74	43.3
	N	69	73	68	82	83	83	85
S-65C:	*6/13/73 - 6/8/99*							
	Median	1.17	1.04	0.11	148	17.5	6.74	25.5
	Mean	1.26	1.13	0.13	159	18.4	6.72	26.7
	Std. Dev.	0.45	0.44	0.14	100	6.5	0.52	9.7
	Min.	0.22	0.25	0.01	57	0.7	4.70	2.5
	Max.	3.77	3.69	1.51	1267	90.6	9.84	66.0
	N	467	469	465	459	481	452	478
S-65C:	*6/13/73 - 3/5/96*							
	Median	1.20	1.07	0.10	153	18.3	6.70	27.0
	Mean	1.28	1.15	0.13	165	19.1	6.70	27.4
	Std. Dev.	0.48	0.46	0.15	109	6.8	0.53	10.3
	Min.	0.22	0.25	0.01	57	8.0	4.70	2.5
	Max.	3.77	3.69	1.51	1267	90.6	9.84	66.0
	N	397	397	397	378	397	370	393
S-65C:	*3/19/96 - 6/8/99*							
	Median	1.13	0.99	0.14	133	15.7	6.85	23.0
	Mean	1.14	0.99	0.14	133	15.1	6.82	23.7
	Std. Dev.	0.21	0.23	0.07	27	3.3	0.47	6.2
	Min.	0.51	0.25	0.01	63	0.7	4.84	11.4
	Max.	1.75	1.48	0.36	245	20.9	8.15	57.2
	N	70	72	68	81	84	82	85

APPENDIX 5-6A

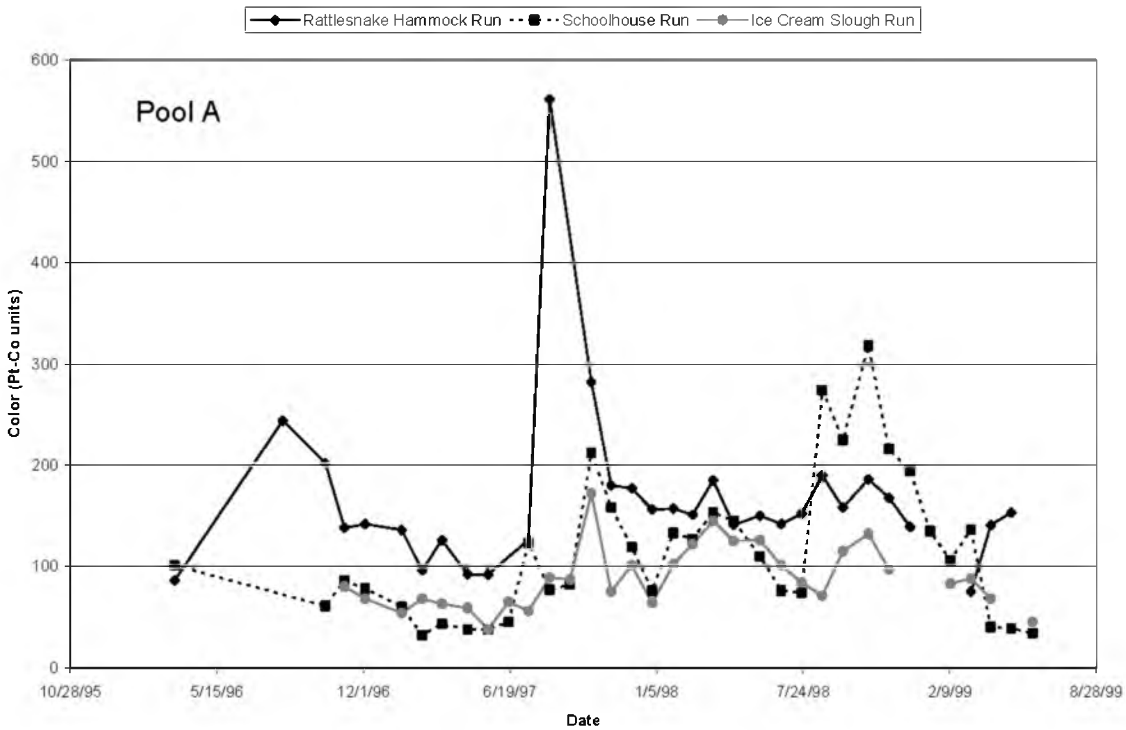
Chlorophyll *a* concentrations in Pool A and Pool C remnant runs (0.5 m depth).

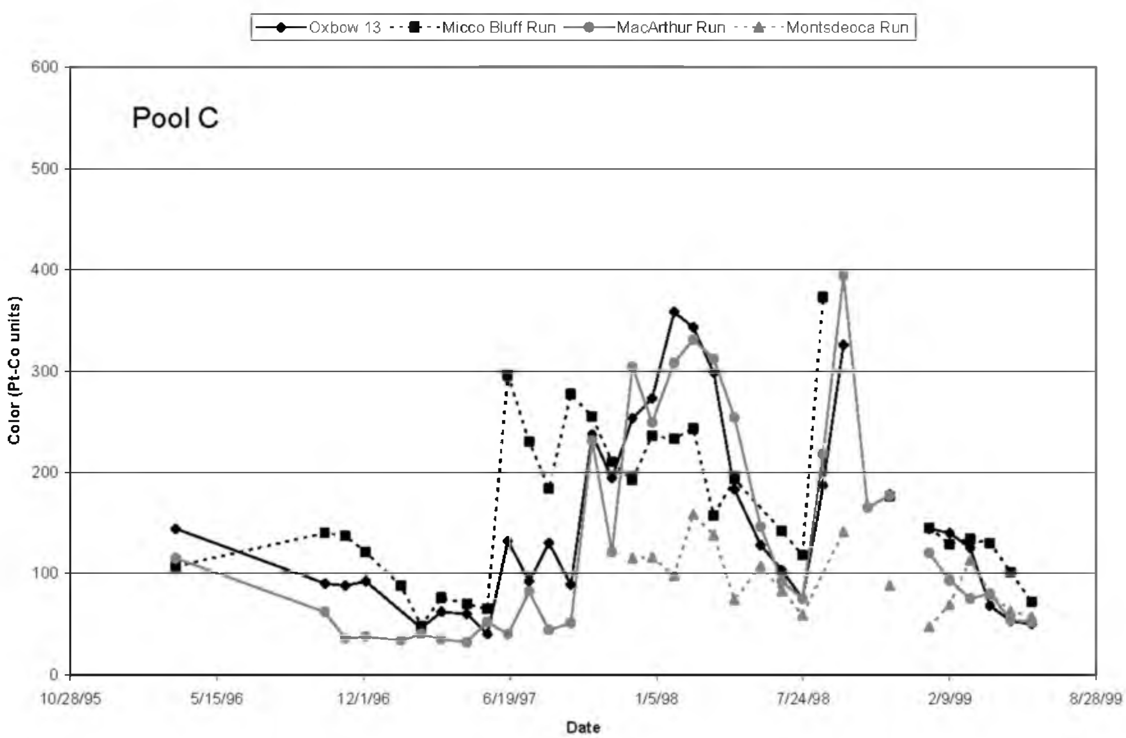




APPENDIX 5-7A

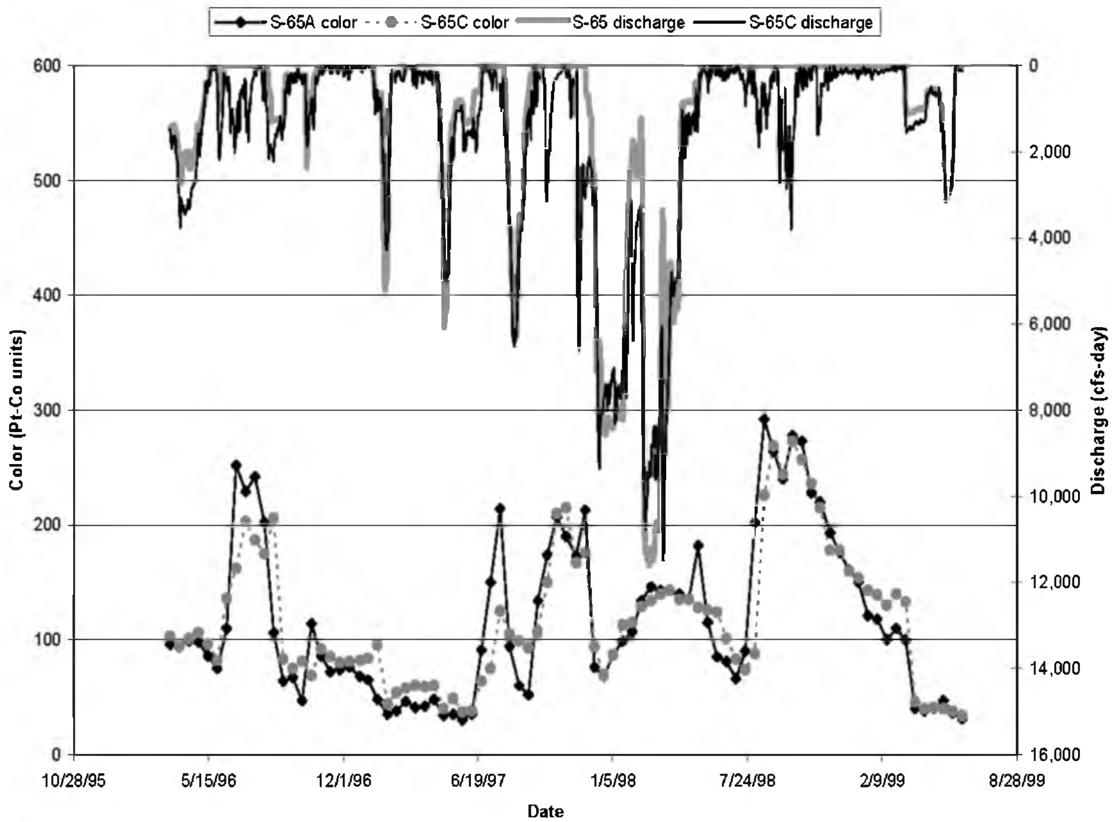
Color in Pool A and Pool C remnant runs (0.5 m depth).





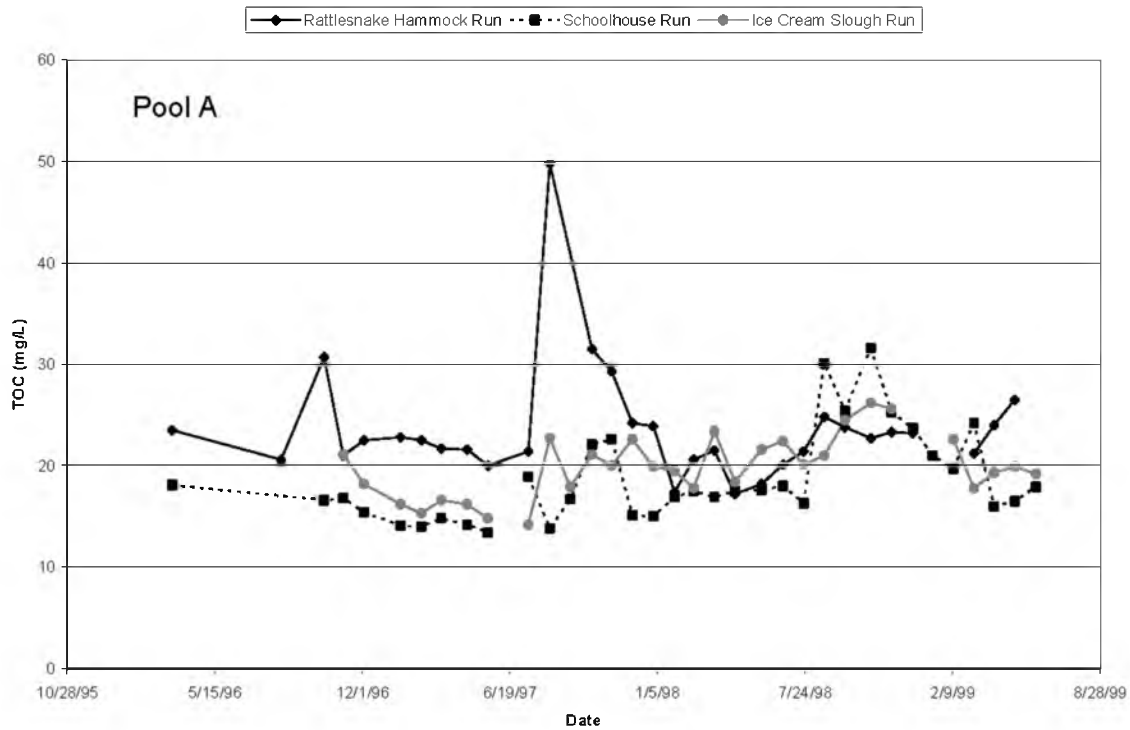
APPENDIX 5-8A

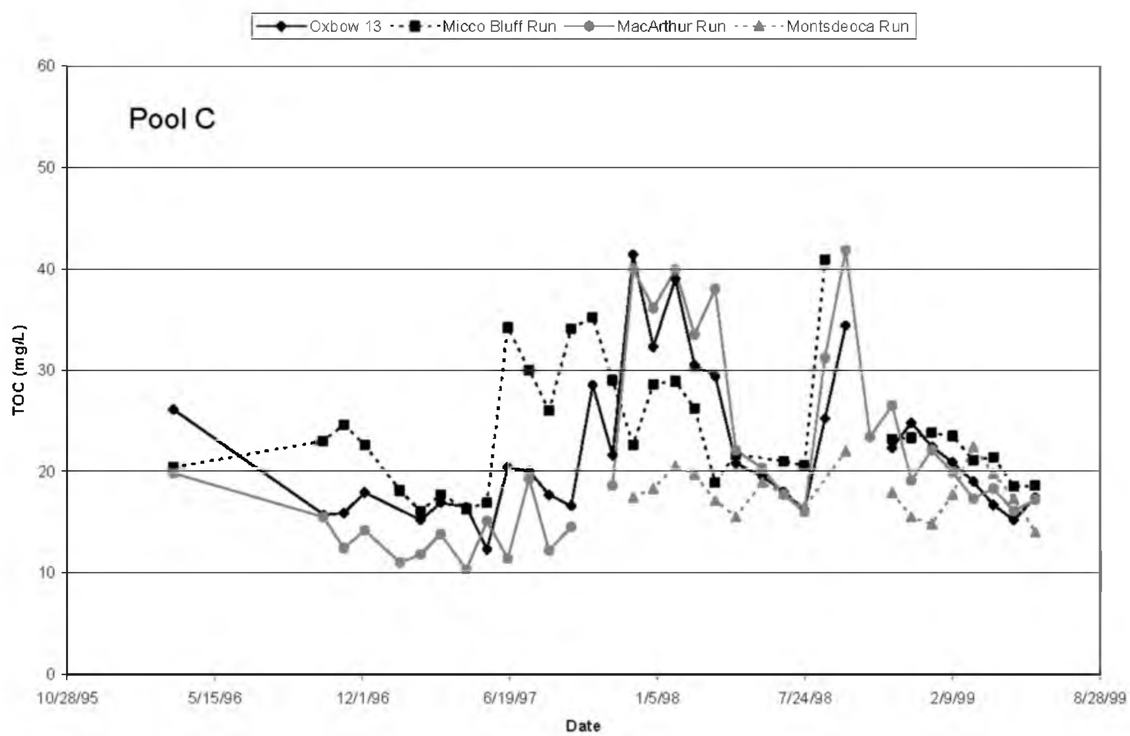
Color in C-38 (at 0.5 m) compared to daily discharge.



APPENDIX 5-9A

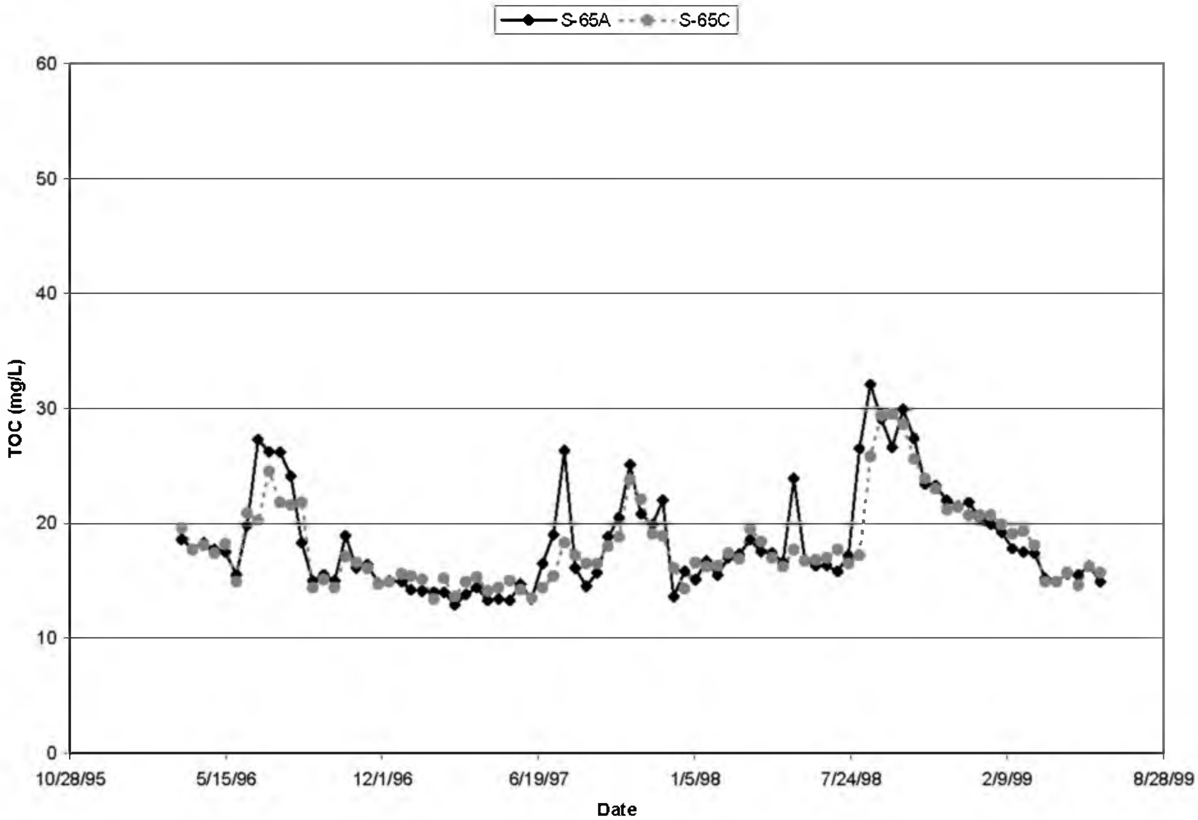
Total organic carbon concentrations in Pool A and Pool C runs (0.5 m depth).





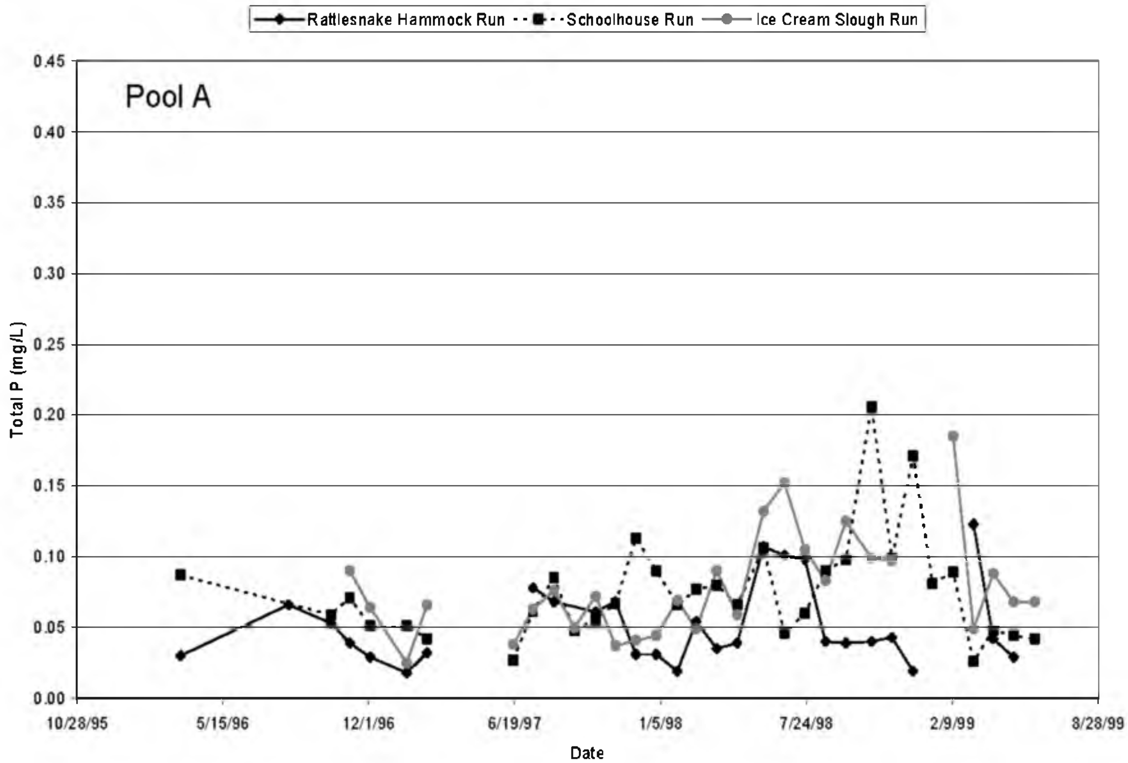
APPENDIX 5-10A

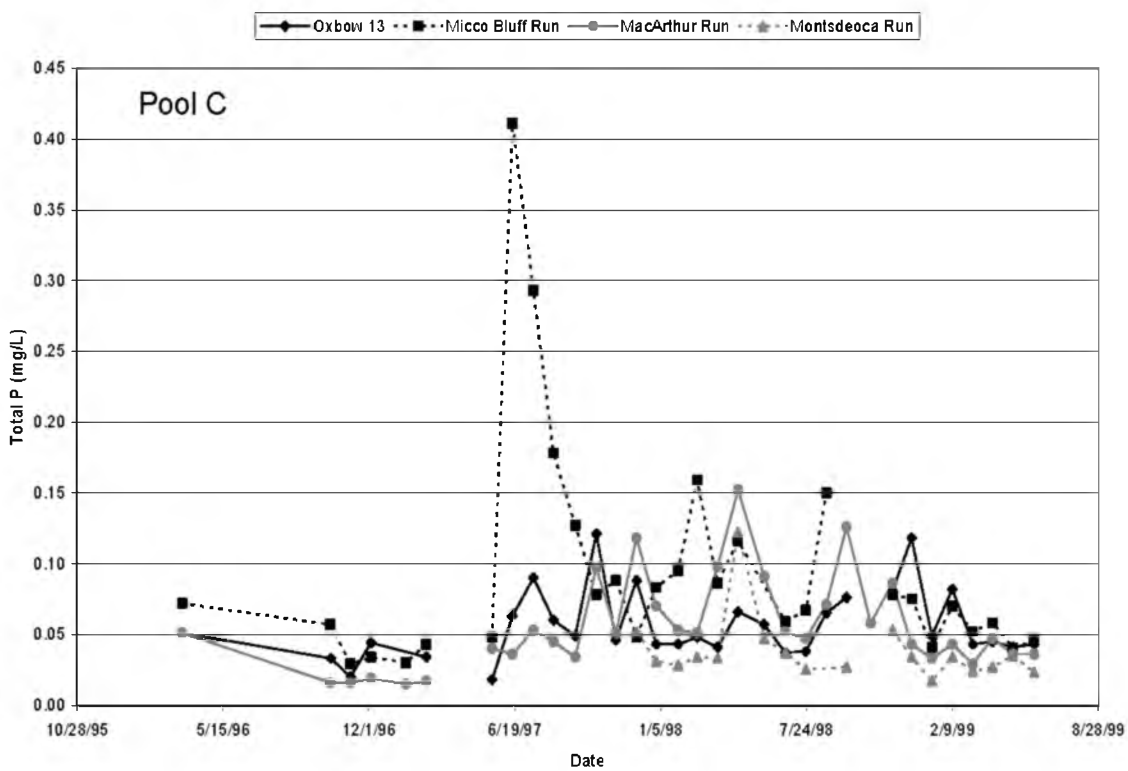
Total organic carbon concentrations in C-38 (0.5 m depth).



APPENDIX 5-11A

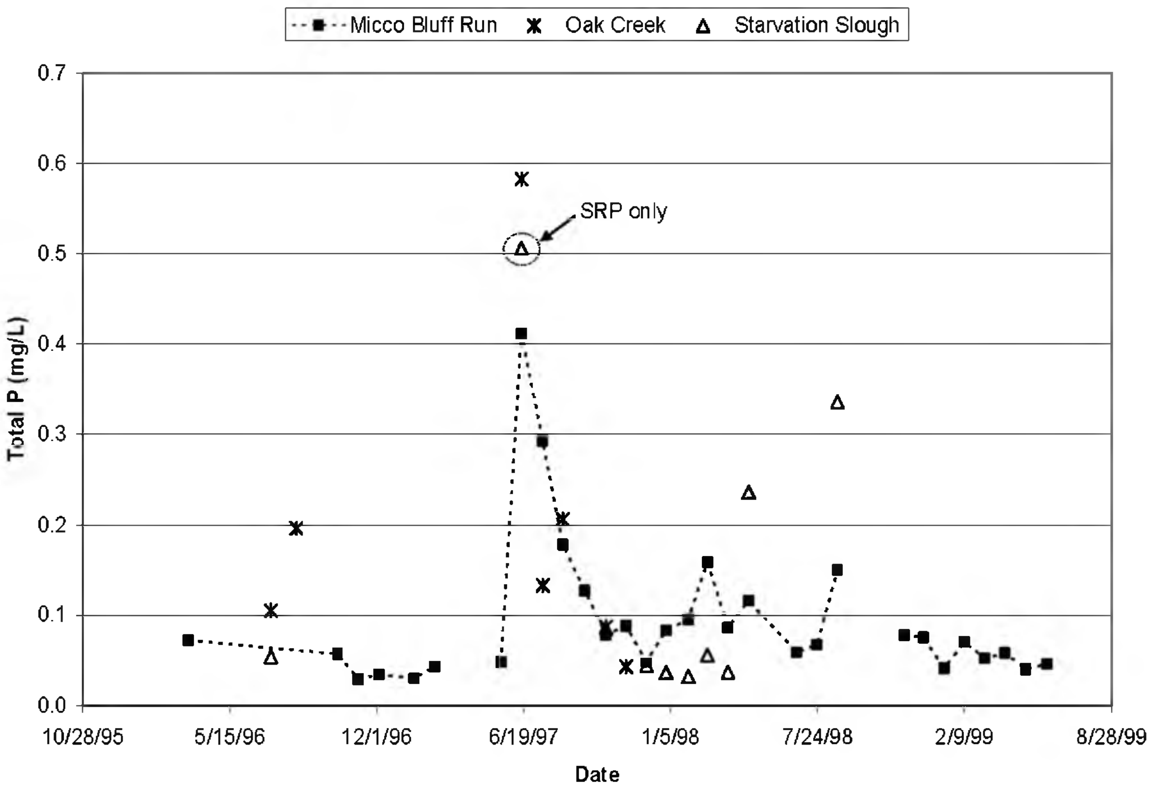
Total phosphorus concentrations in Pool A and Pool C remnant runs (0.5 m depth).





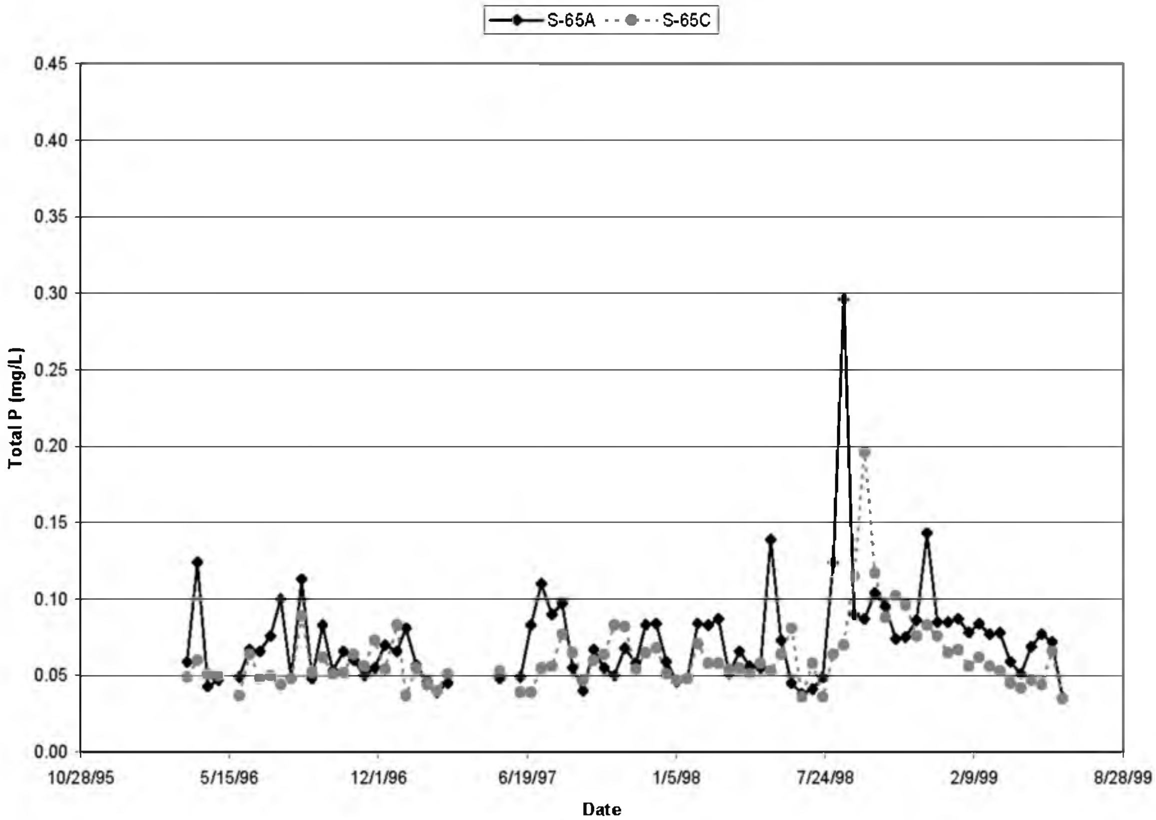
APPENDIX 5-12A

Comparison of total phosphorus concentrations (at 0.5 m) in Micco Bluff Run (Pool C) and its tributaries. Only soluble reactive phosphorus (SRP) was measured from Starvation Slough on 6-16-97.



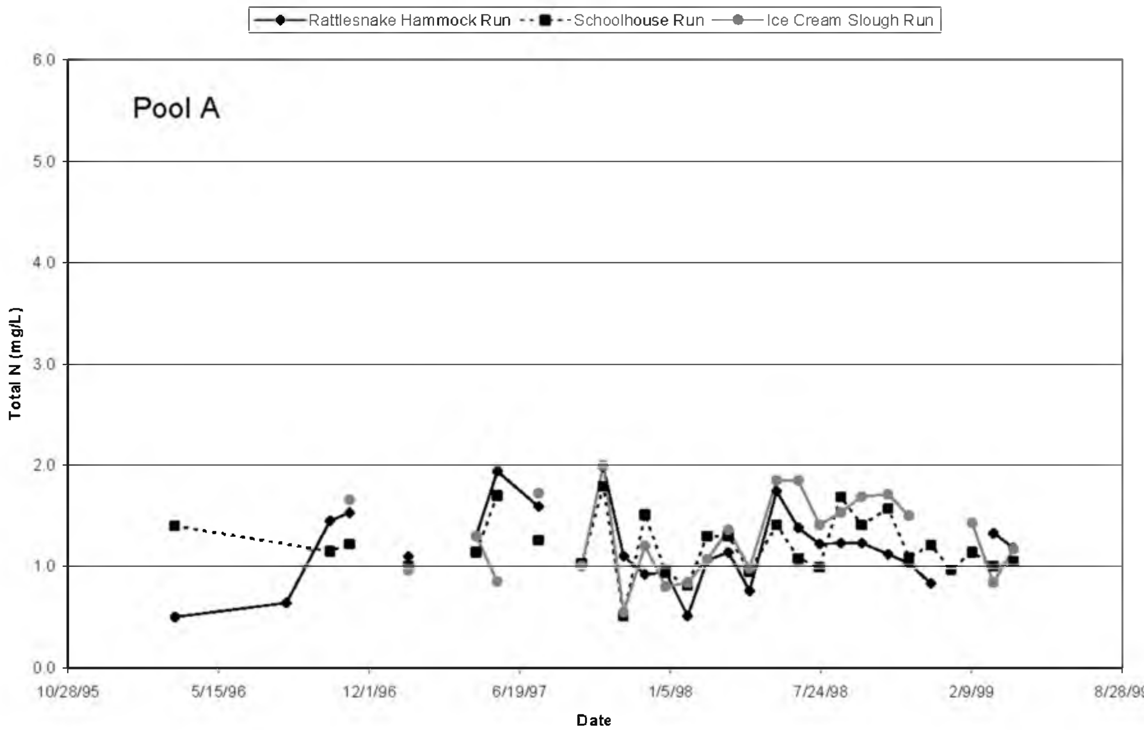
APPENDIX 5-13A

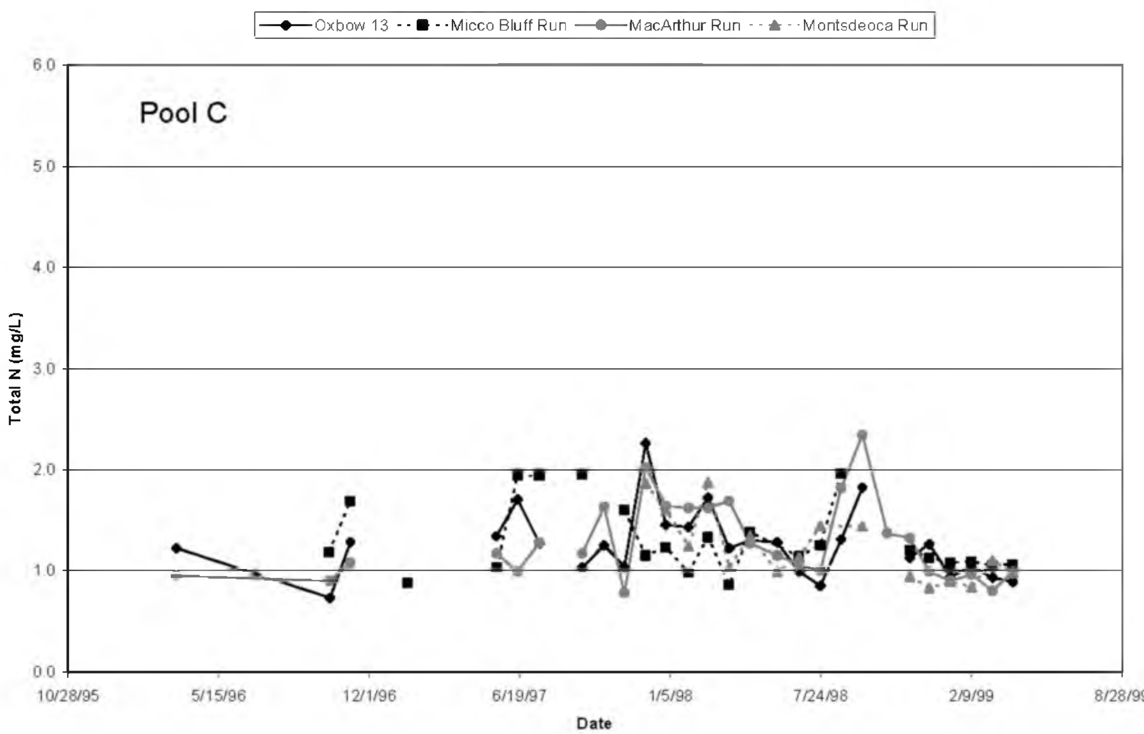
Total phosphorus concentrations in C-38 (0.5 m depth).



APPENDIX 5-14A

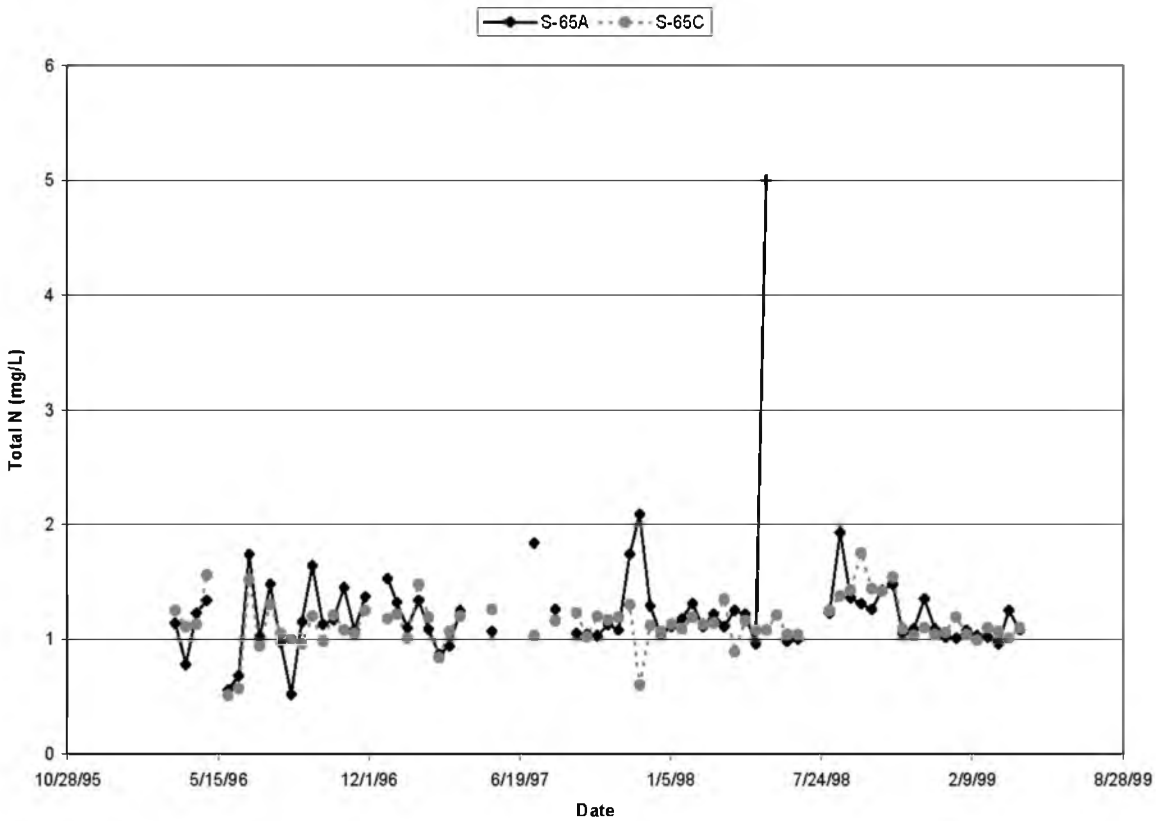
Total nitrogen concentrations in Pool A and Pool C remnant runs (0.5 m depth).





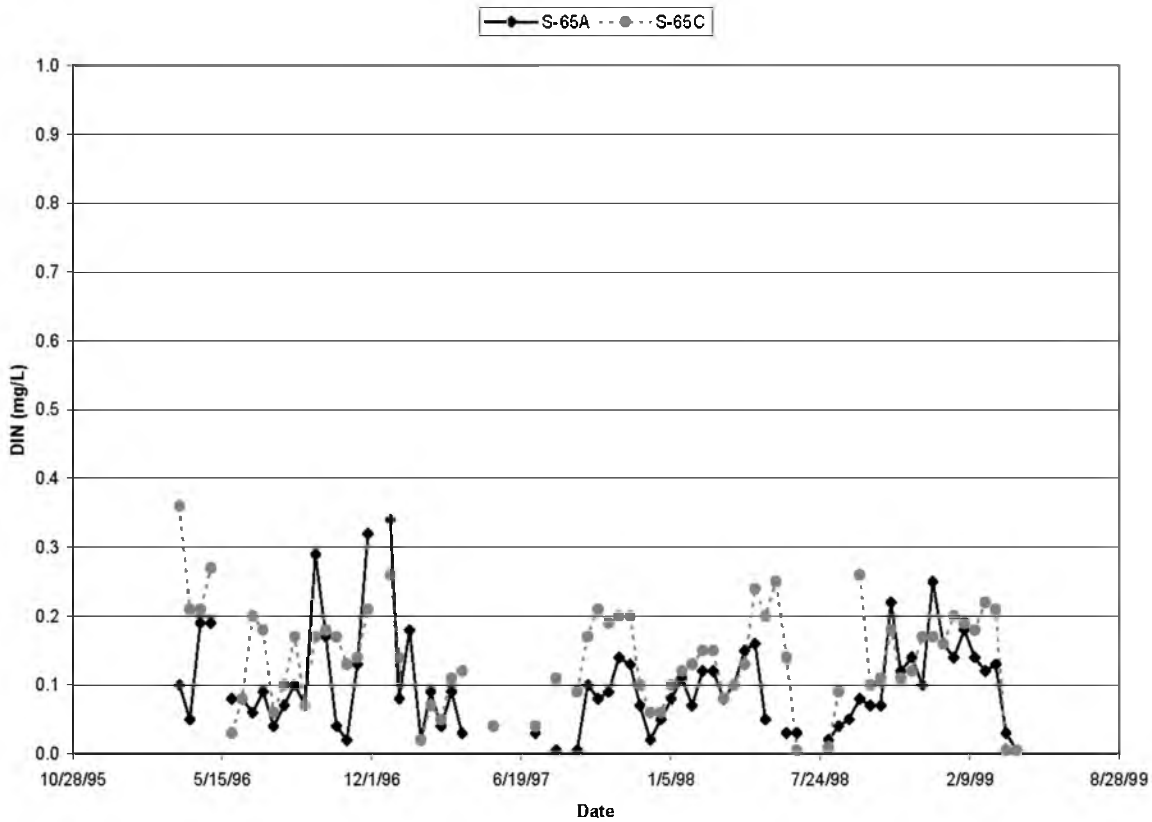
APPENDIX 5-15A

Total nitrogen concentrations in C-38 (0.5 m depth).



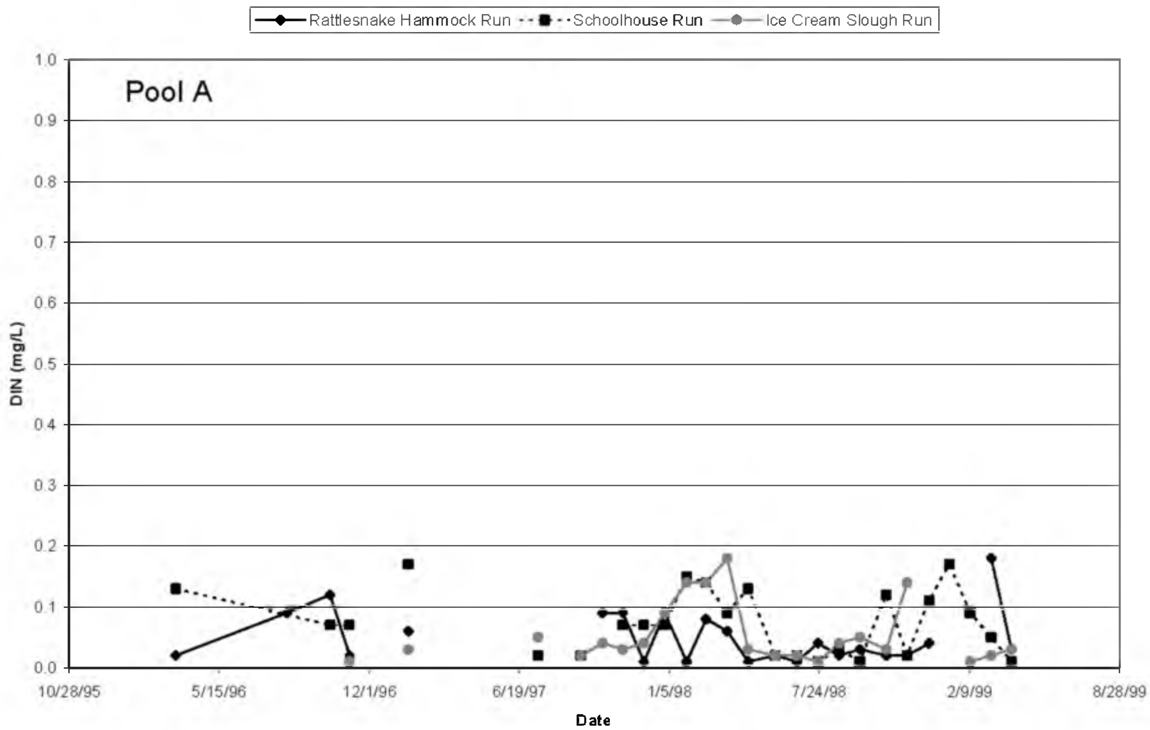
APPENDIX 5-16A

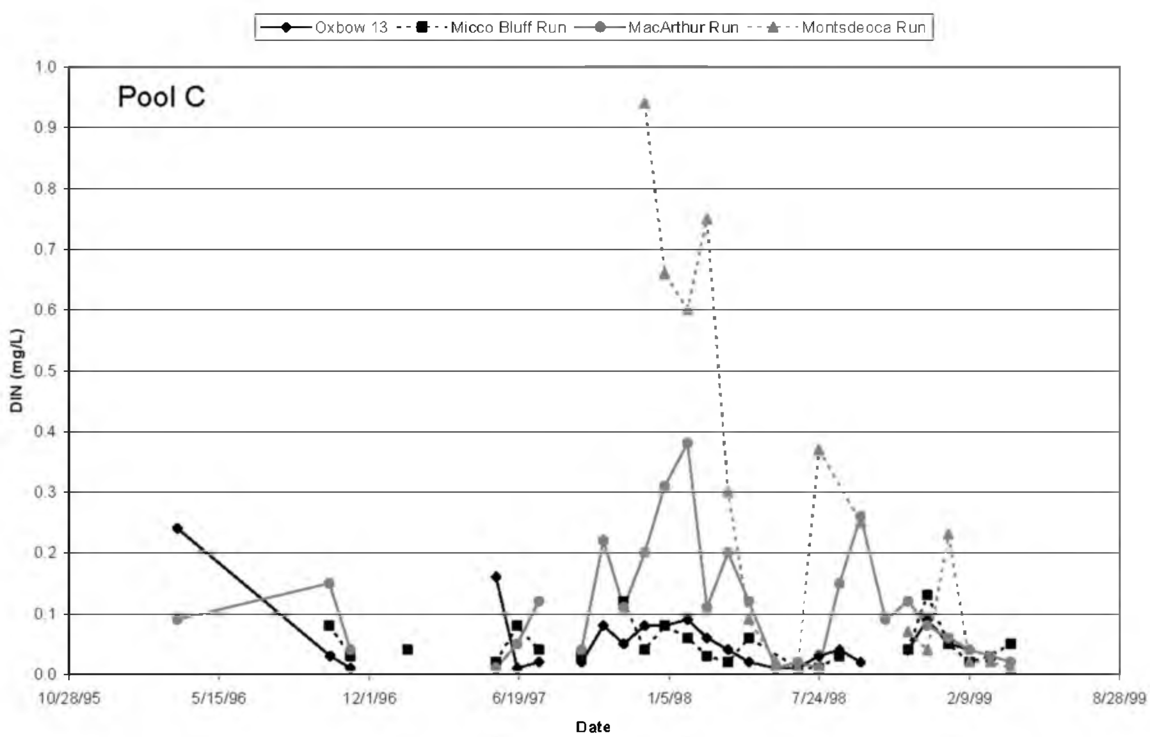
Dissolved inorganic nitrogen concentrations in C-38 (0.5 m depth).



APPENDIX 5-17A

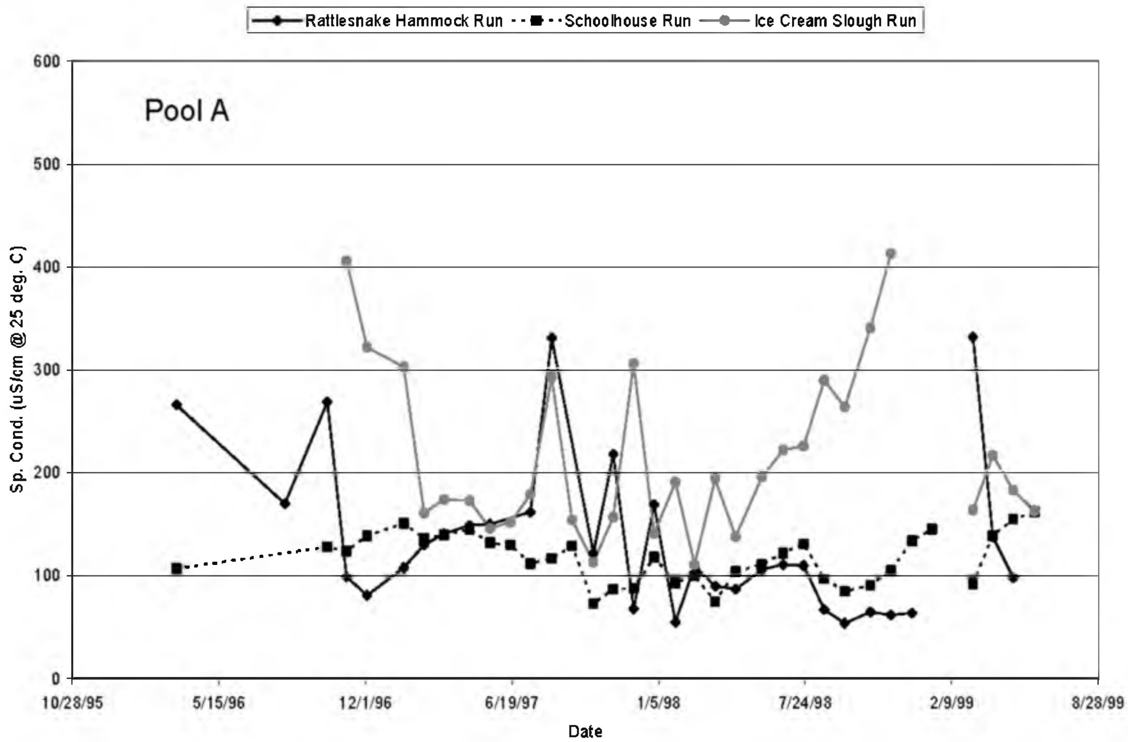
Dissolved inorganic nitrogen concentrations in Pool A and Pool C remnant runs (0.5 m depth).

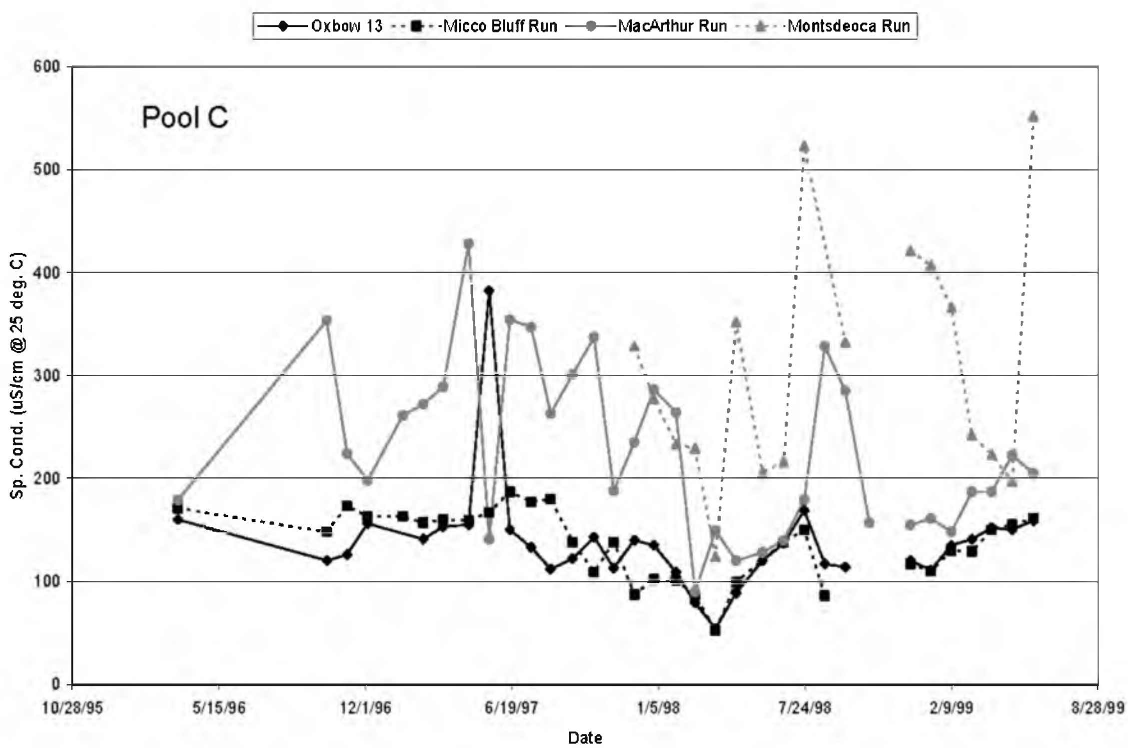




APPENDIX 5-18A

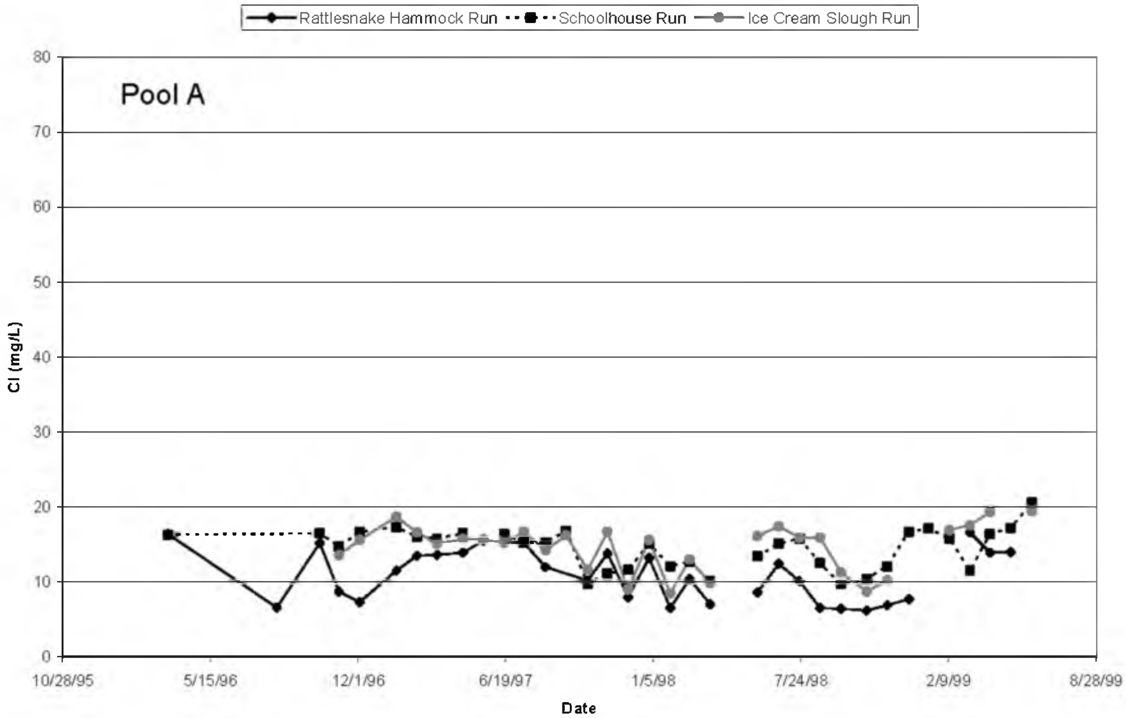
Specific conductance in Pool A and Pool C remnant runs (0.5 m depth).

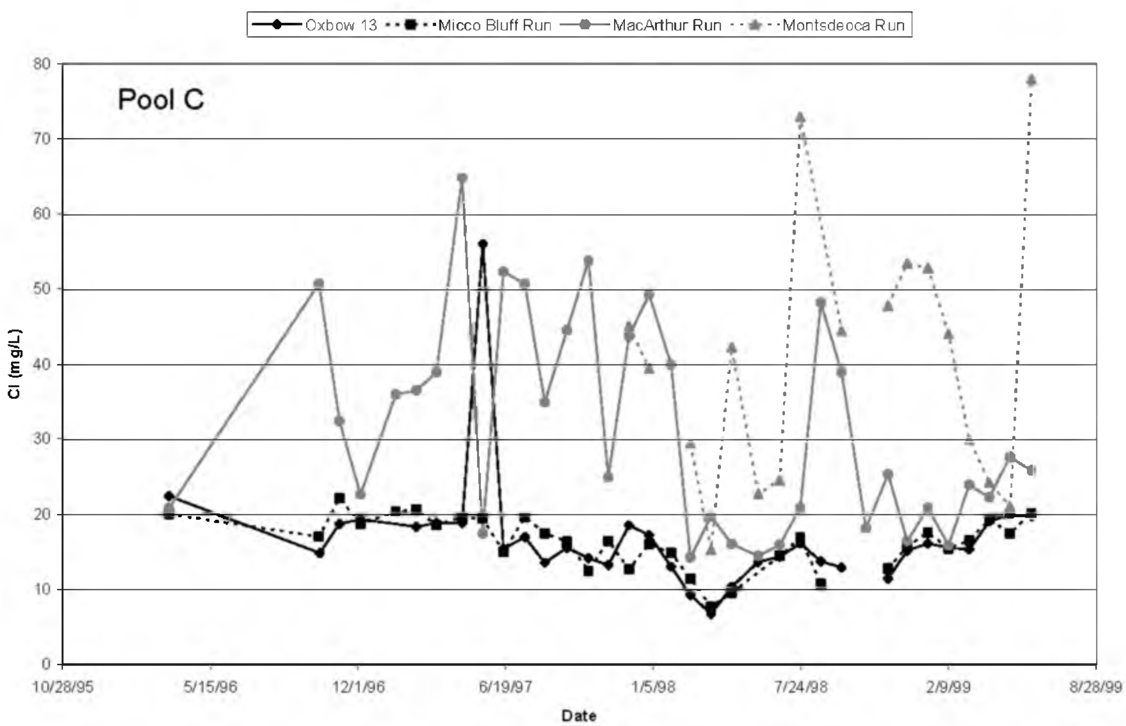




APPENDIX 5-19A

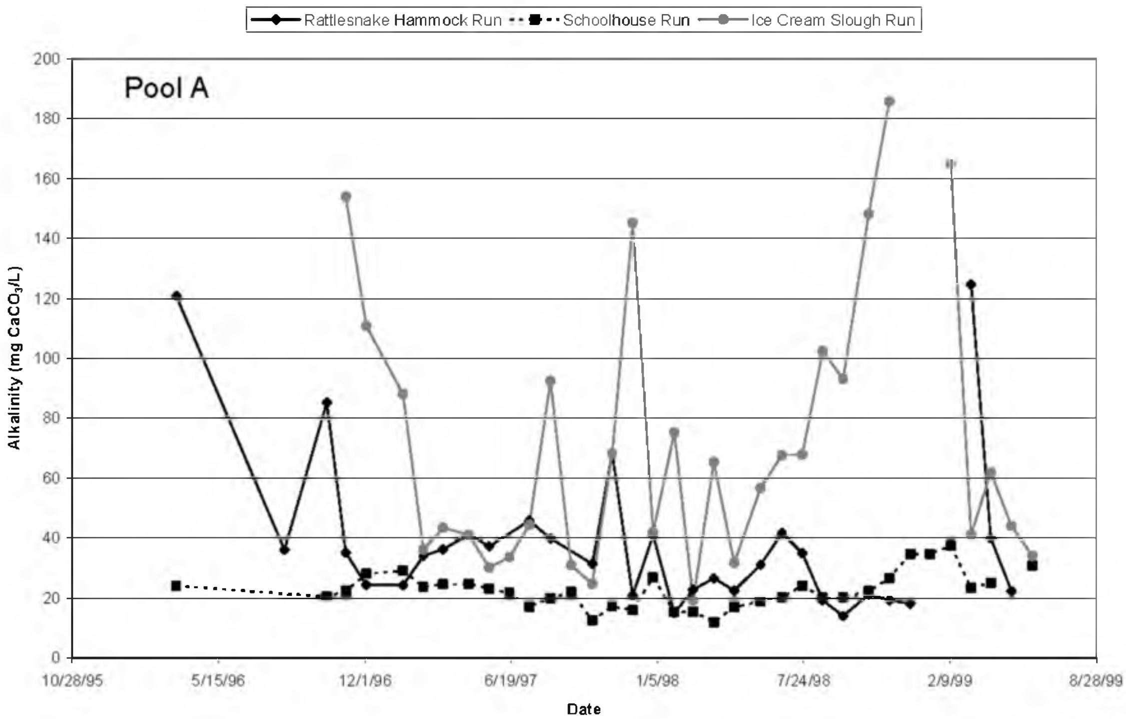
Chloride concentrations in Pool A and Pool C remnant runs (0.5 m depth).

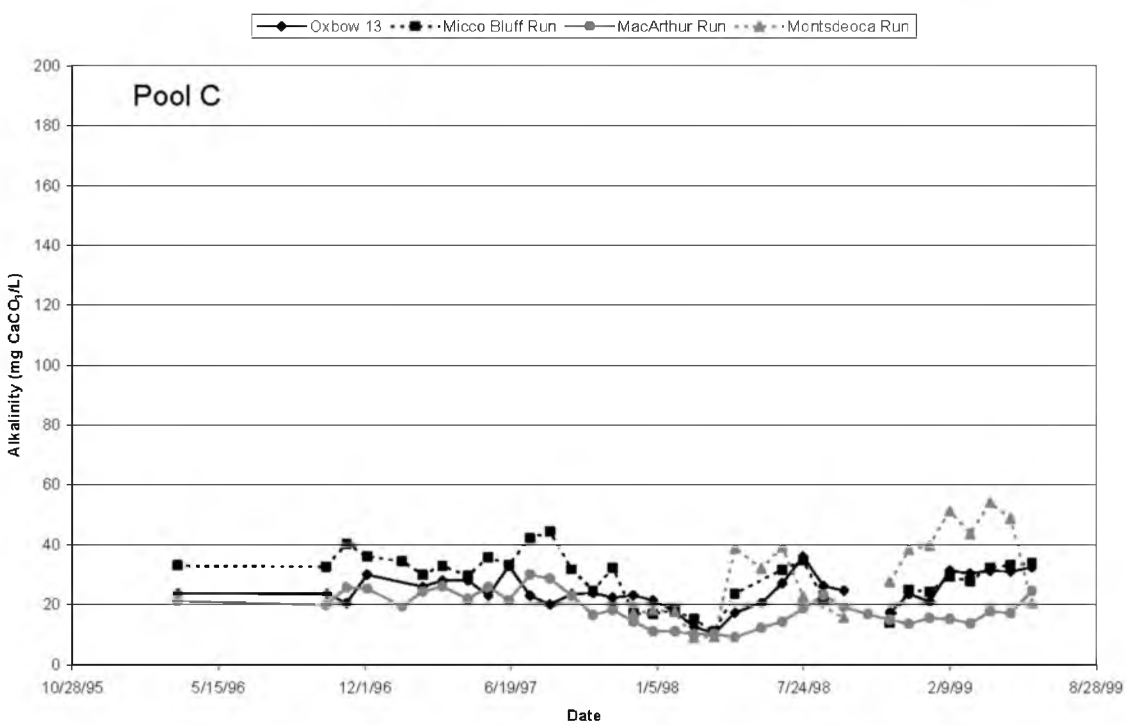




APPENDIX 5-20A

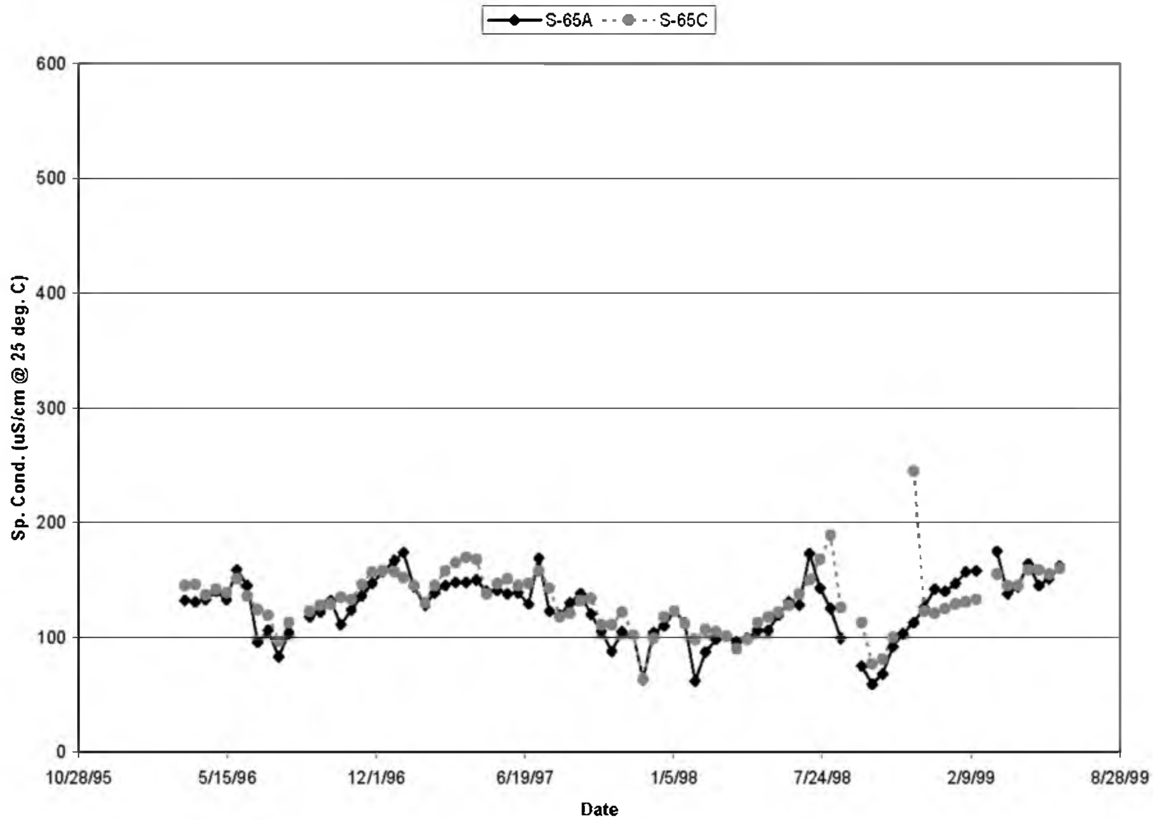
Alkalinity in Pool A and Pool C remnant runs (0.5 m depth).





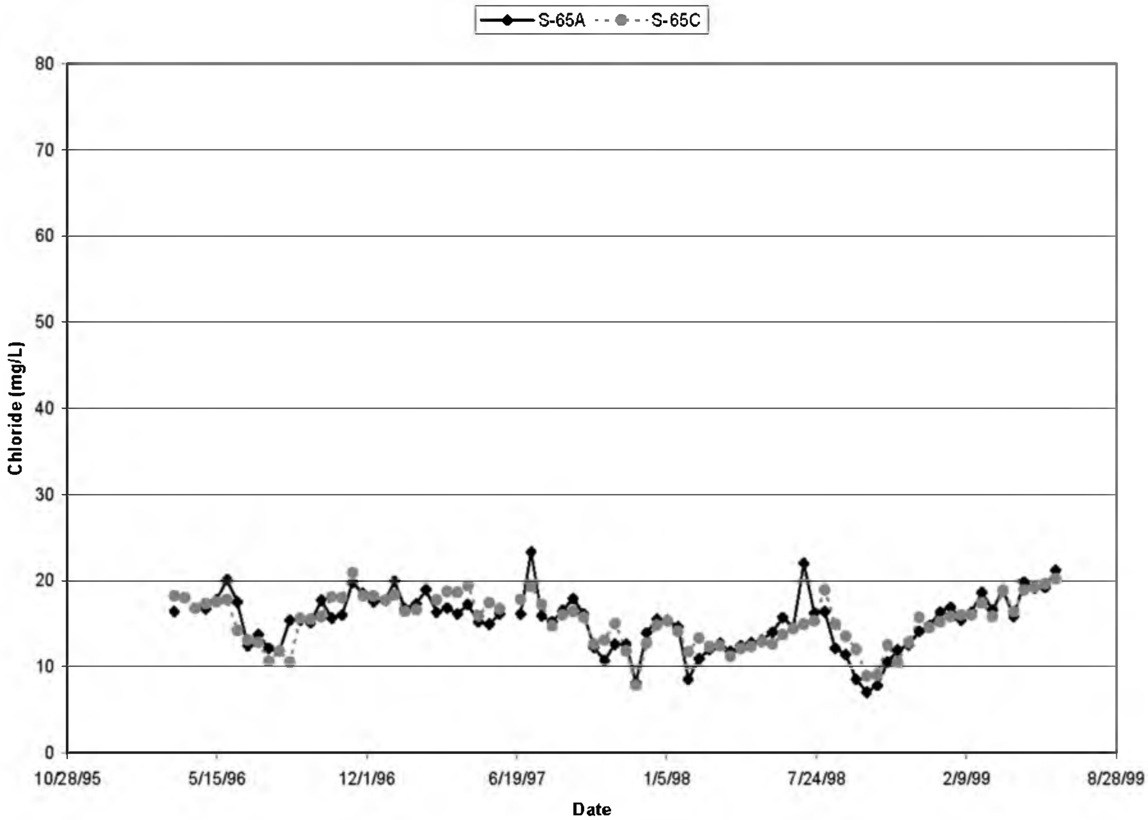
APPENDIX 5-21A

Specific conductance in C-38 (0.5 m depth).



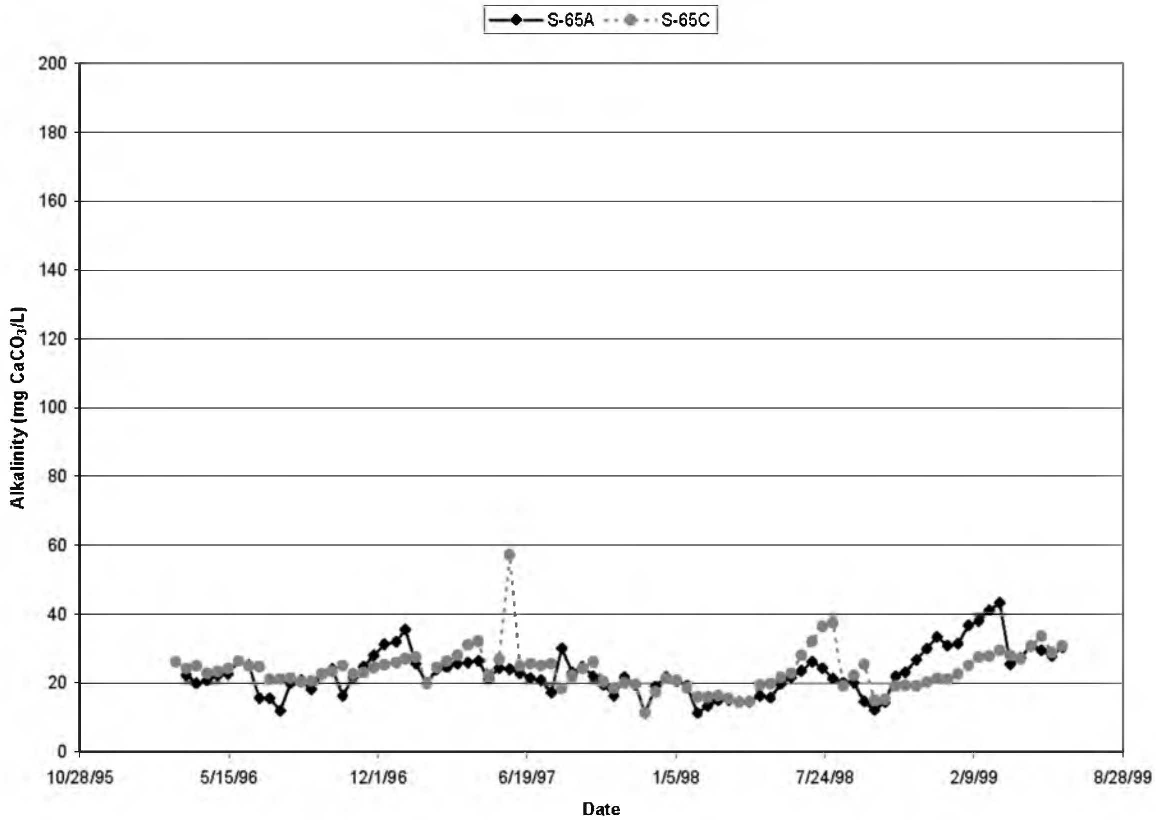
APPENDIX 5-22A

Chloride concentrations in C-38 (0.5 m depth).



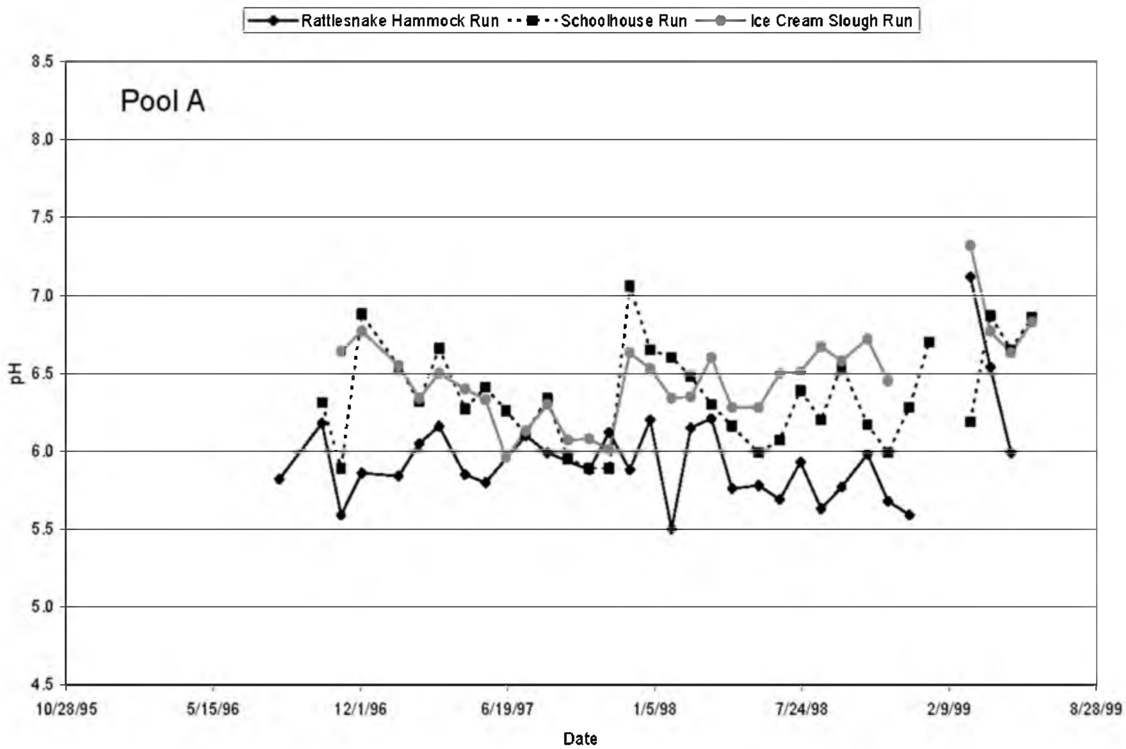
APPENDIX 5-23A

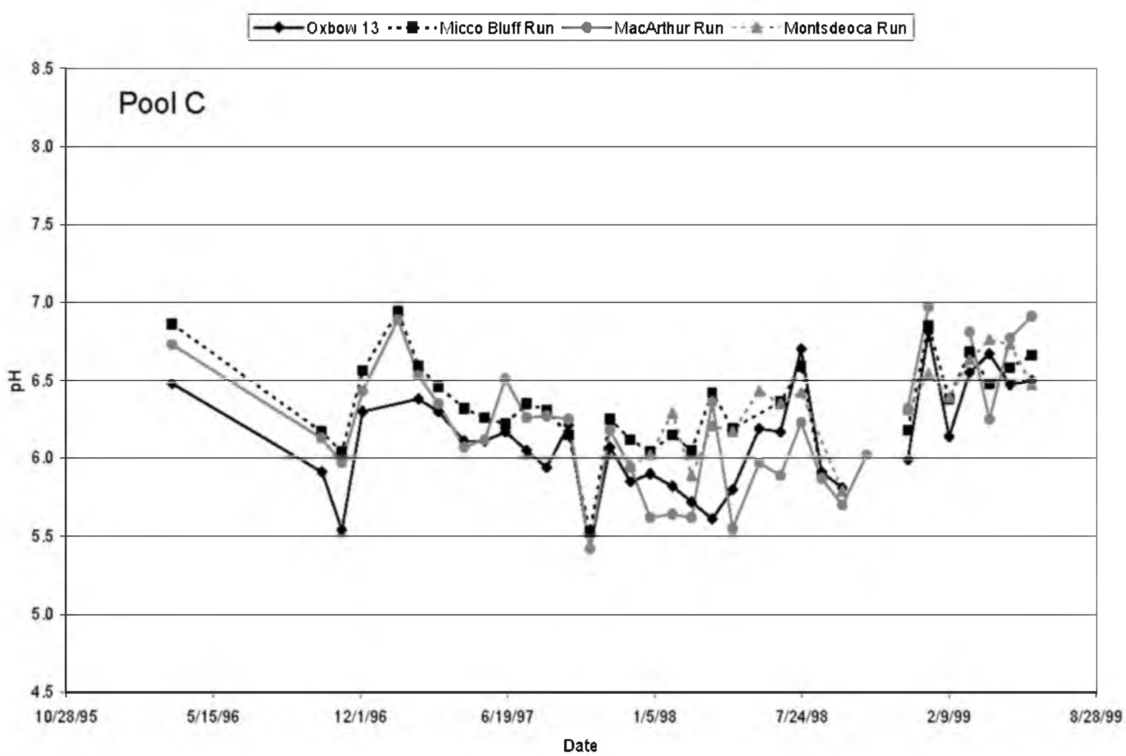
Alkalinity in C-38 (0.5 m depth).



APPENDIX 5-24A

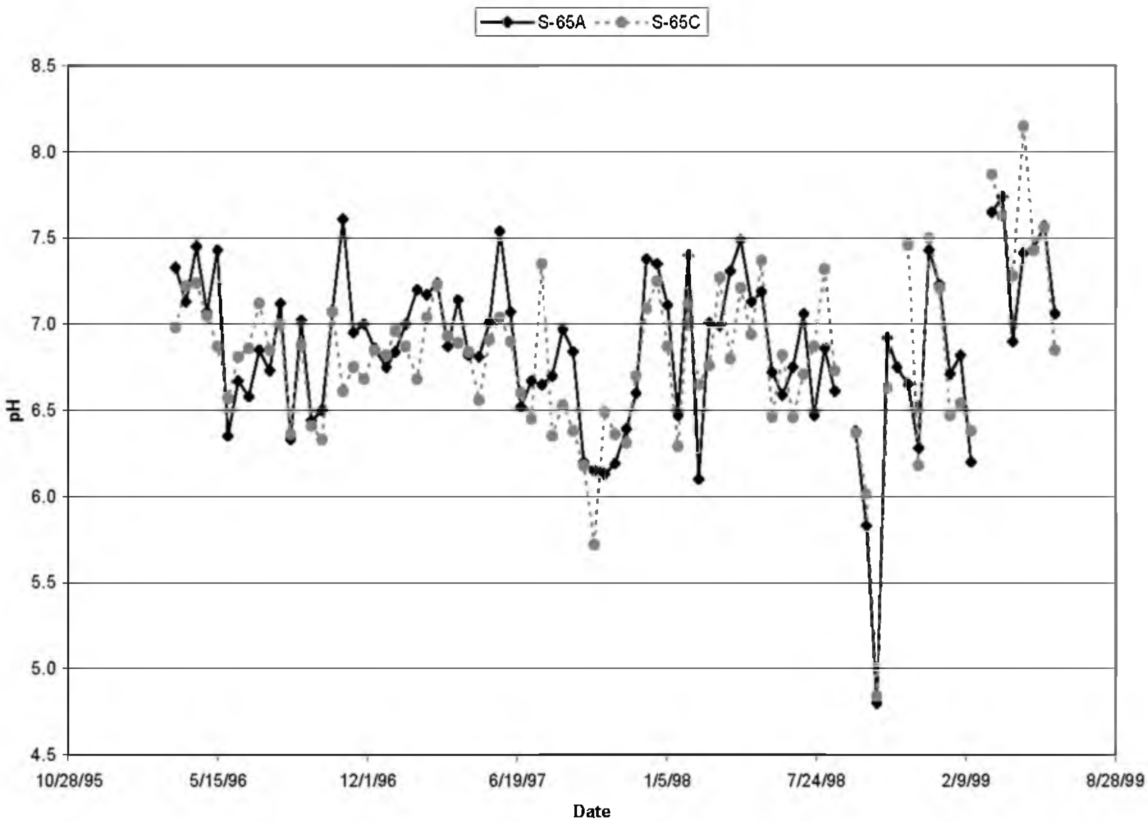
pH in Pool A and Pool C remnant runs (0.5 m depth).





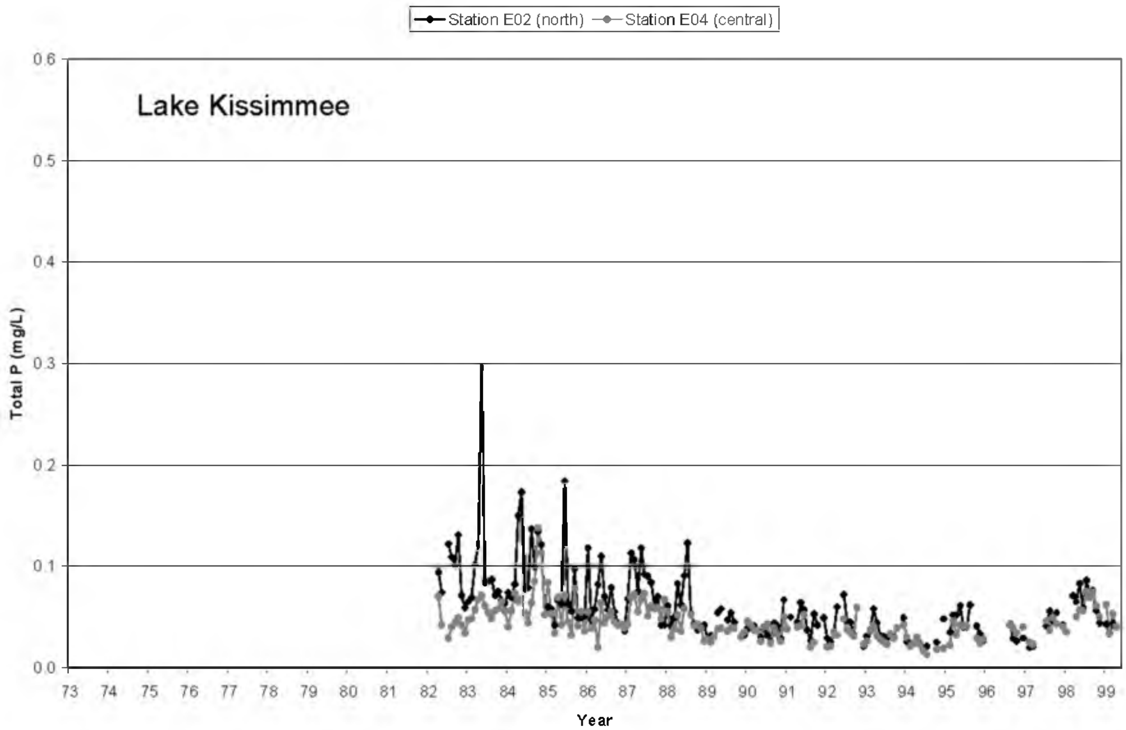
APPENDIX 5-25A

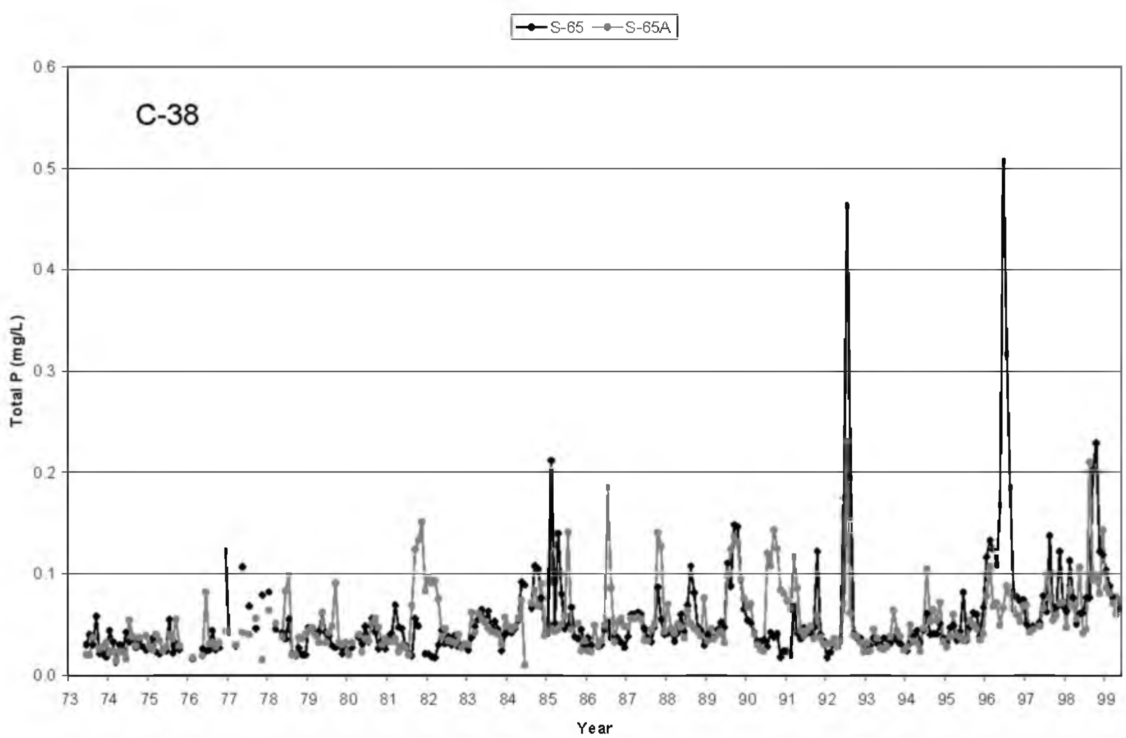
pH in C-38 (0.5 m depth).



APPENDIX 5-26A

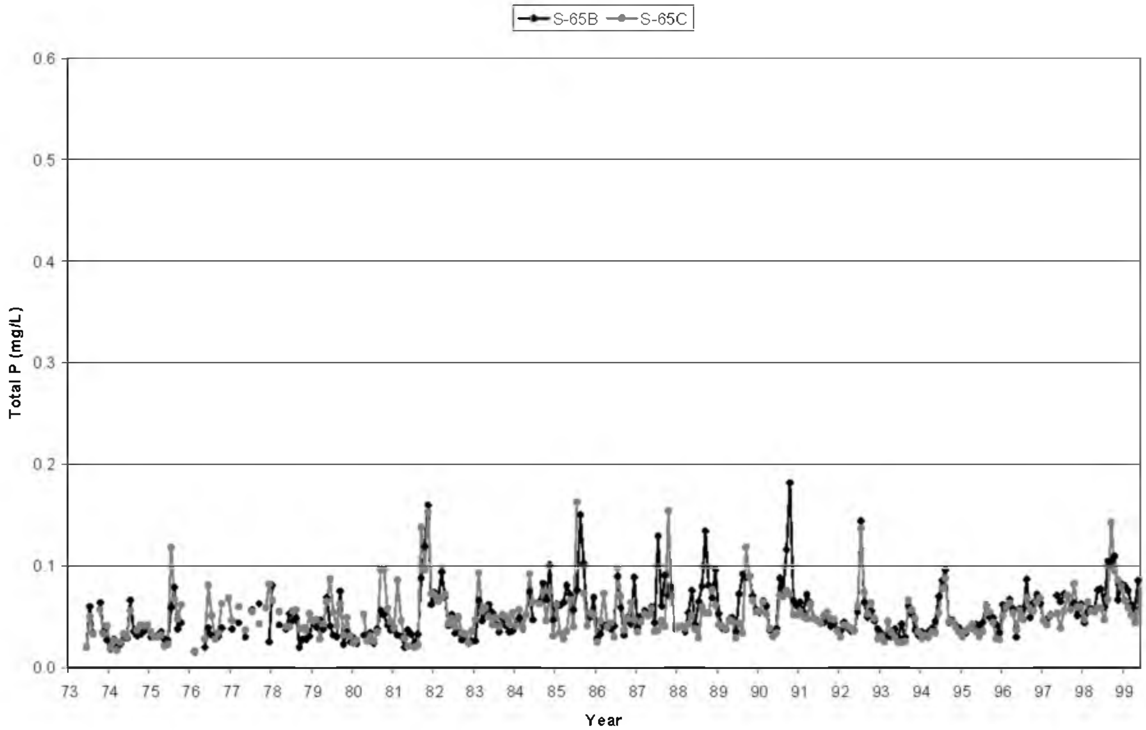
Monthly mean total phosphorus concentrations in Lake Kissimmee and at S-65 and S-65A (0.5 m depth). Lake Kissimmee was not sampled during the 1996 drawdown, so comparison of mid-lake TP concentrations and S-65 concentrations is not possible during that period.

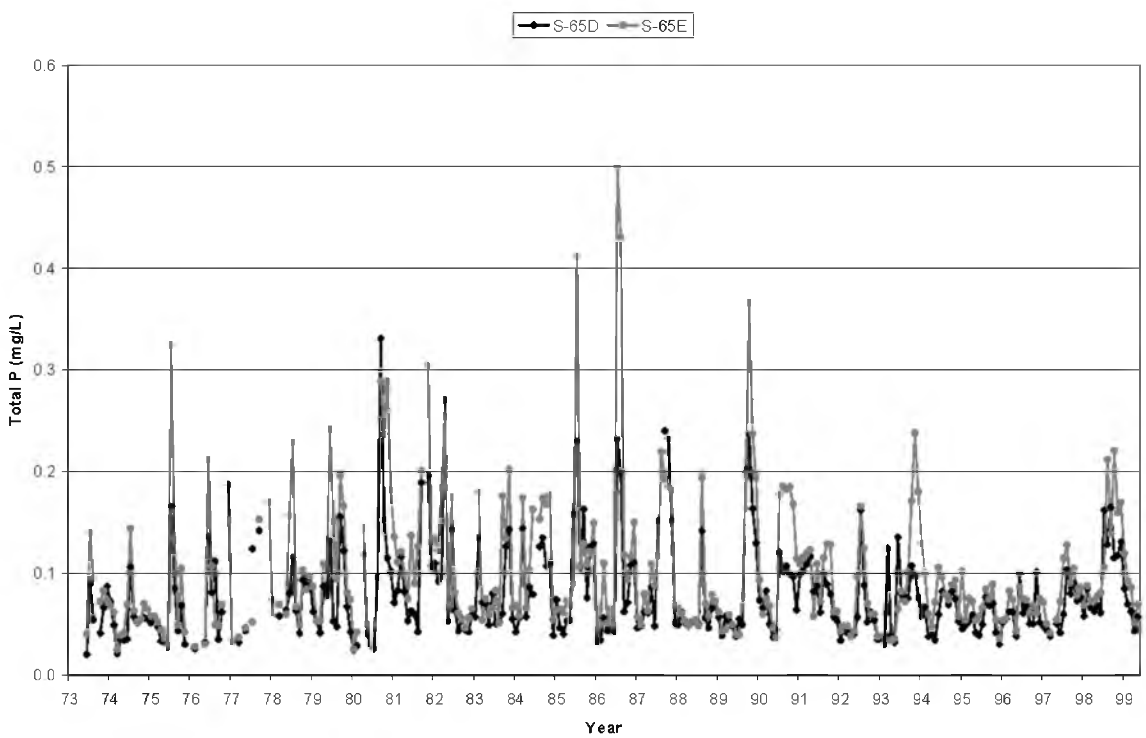




APPENDIX 5-27A

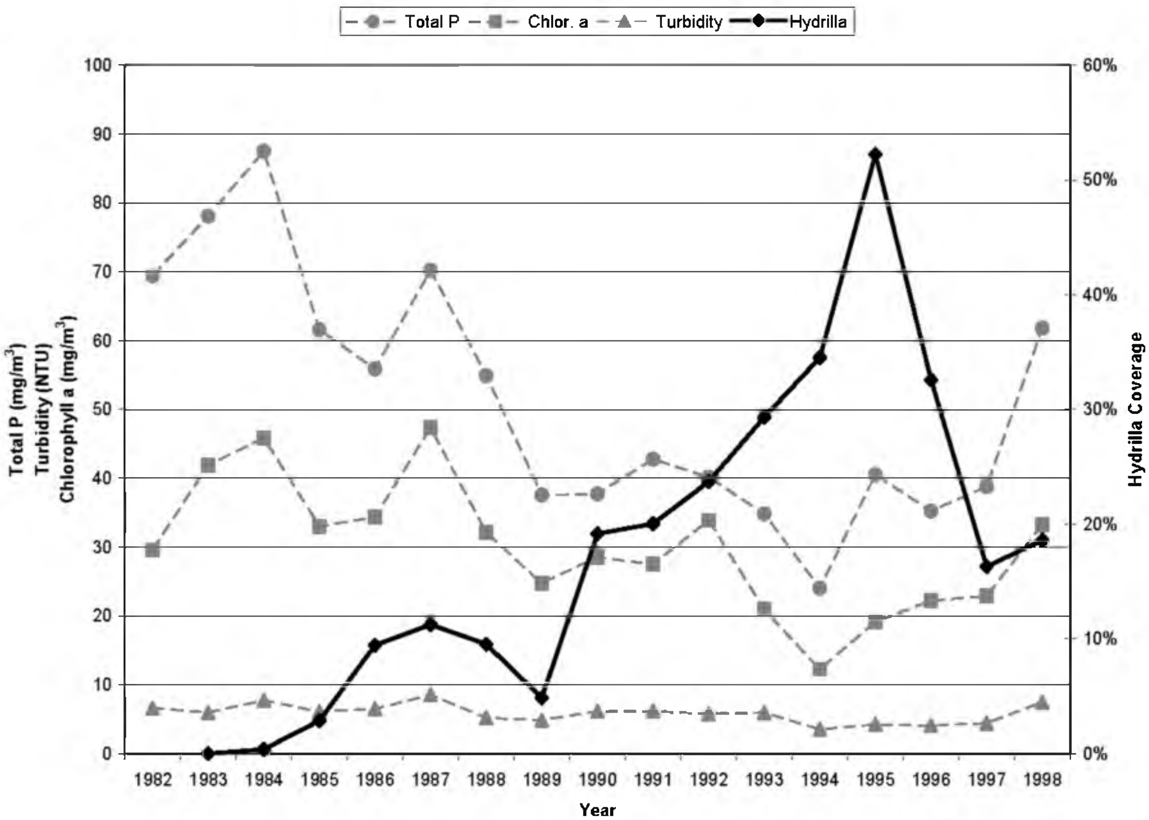
Monthly mean total phosphorus concentrations at S-65B, S-65C, S-65D, and S-65E (0.5 m depth).





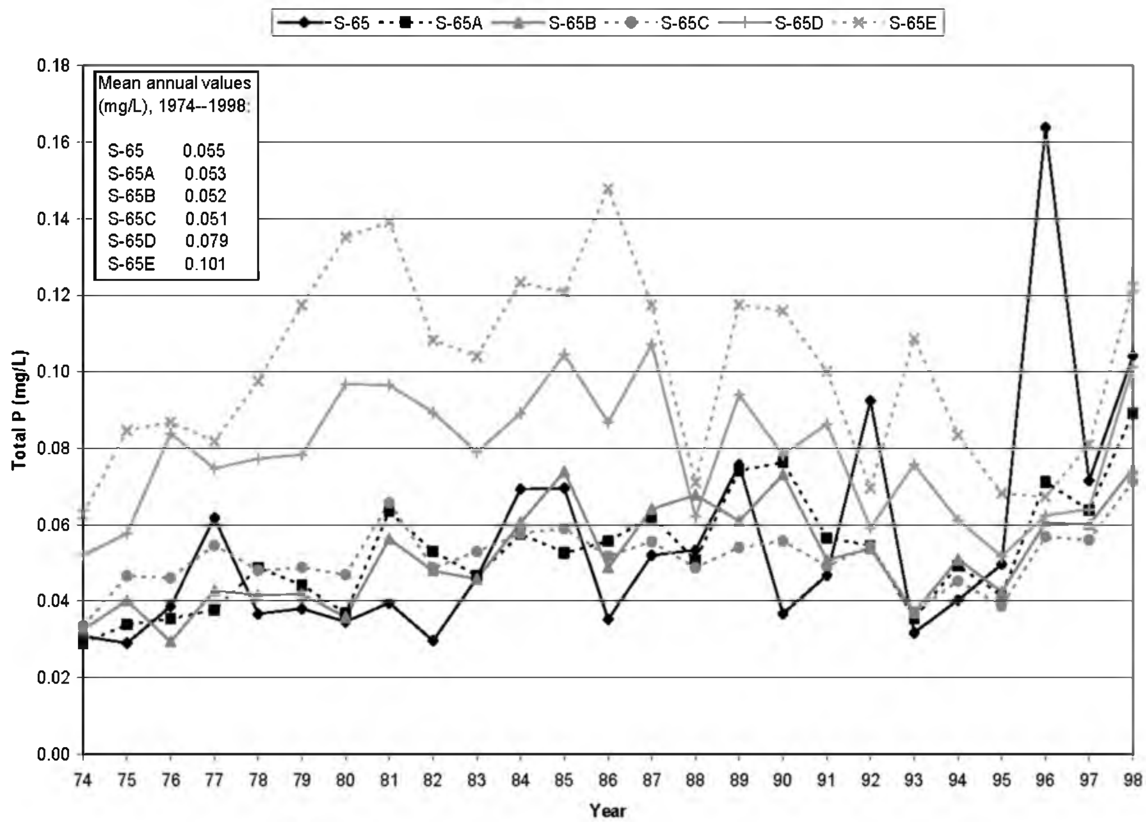
APPENDIX 5-28A

Percent hydrilla coverage and annual mean total phosphorus, chlorophyll a, and turbidity in Lake Kissimmee. Hydrilla coverage was estimated once per year in summer or fall. Water quality was monitored monthly at two stations (E02 and E04).



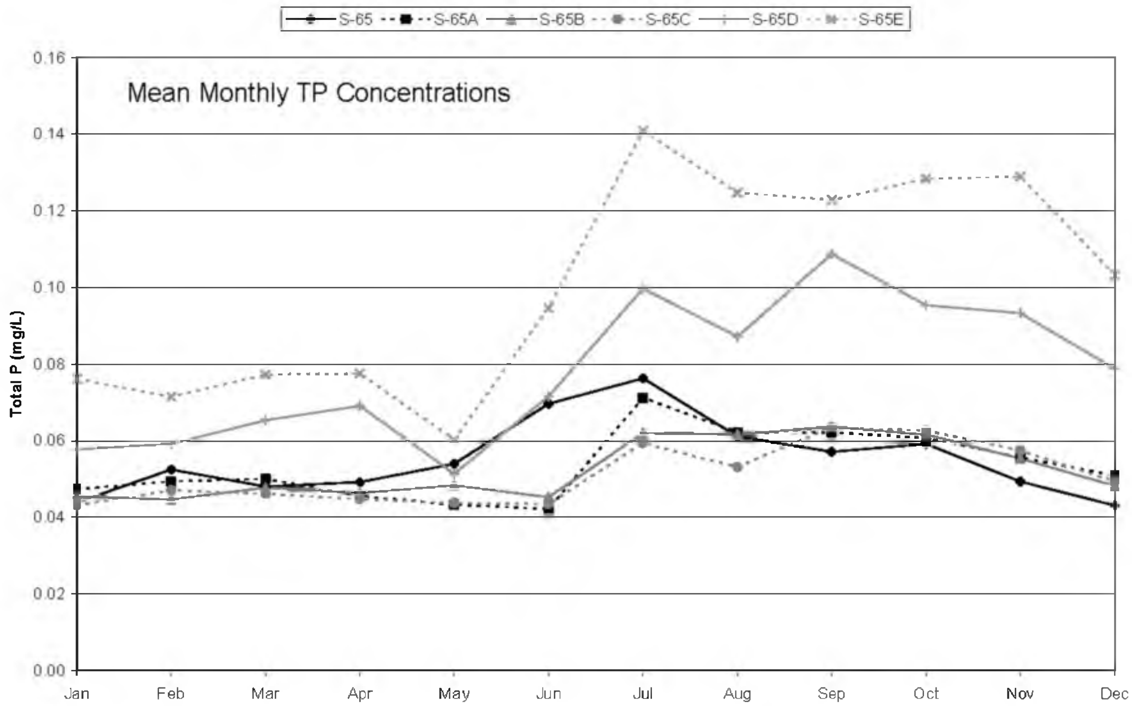
APPENDIX 5-29A

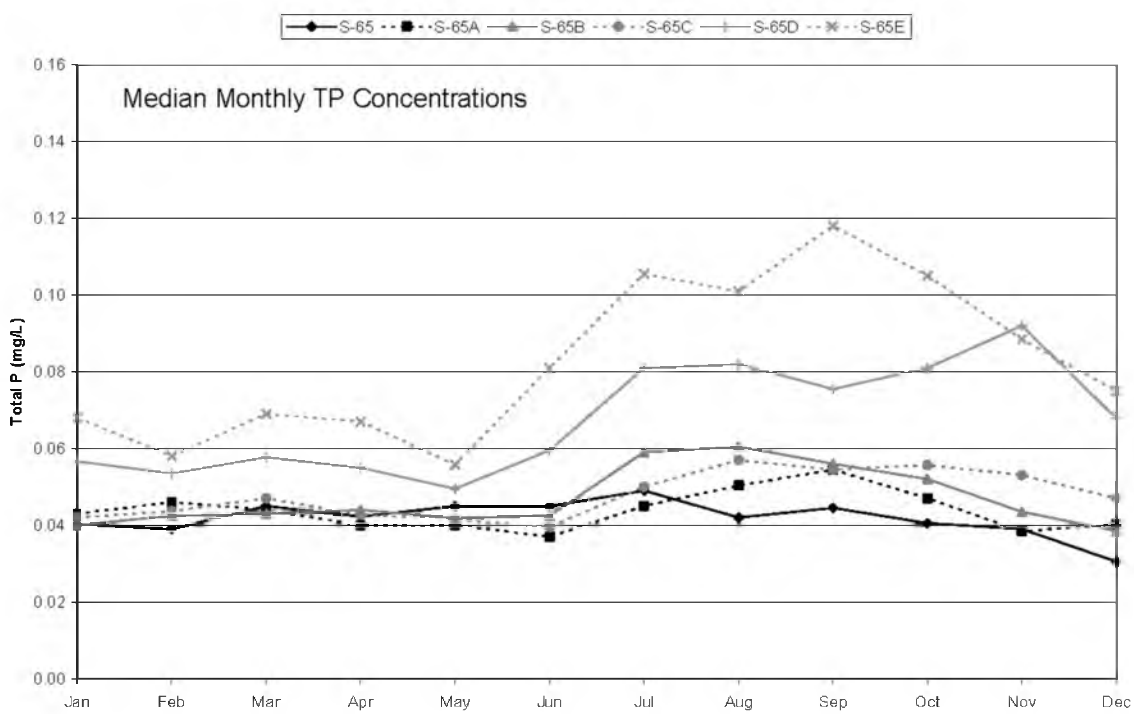
Annual mean total phosphorus concentrations in C-38.



APPENDIX 5-30A

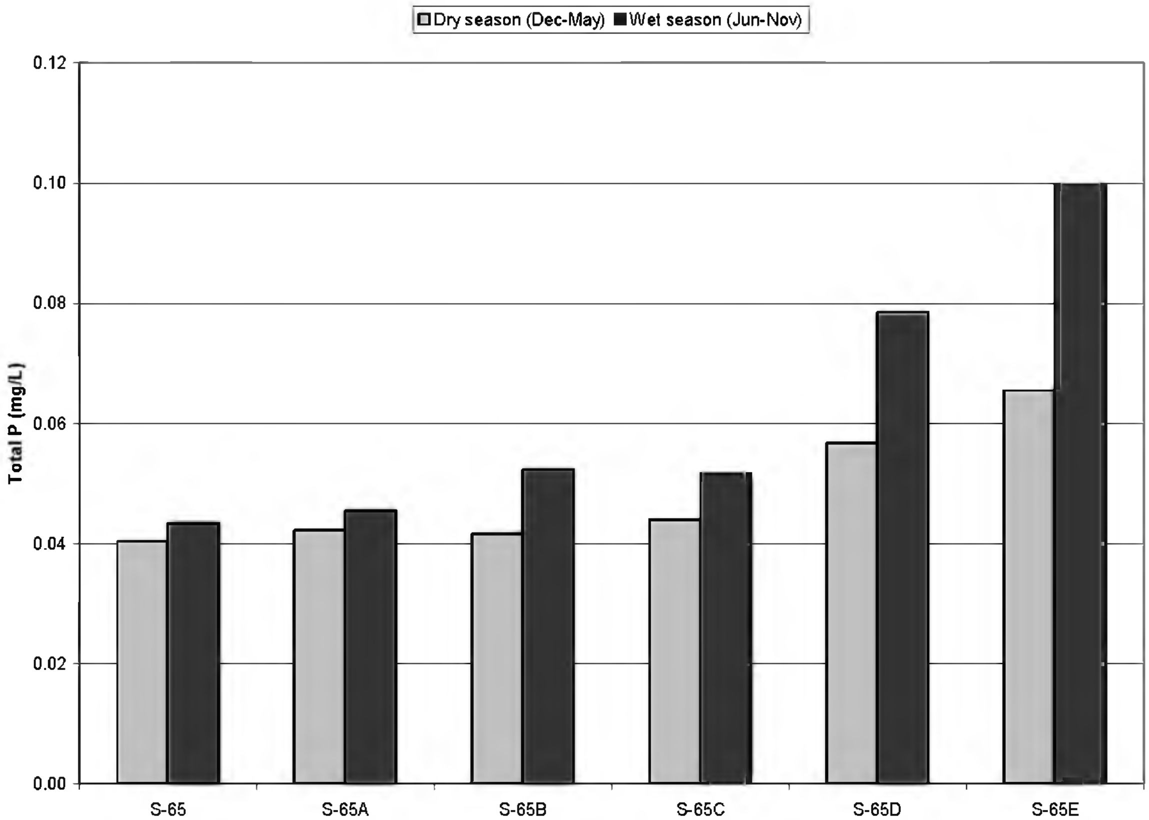
Comparison of mean and median monthly total phosphorus (TP) concentrations in C-38.





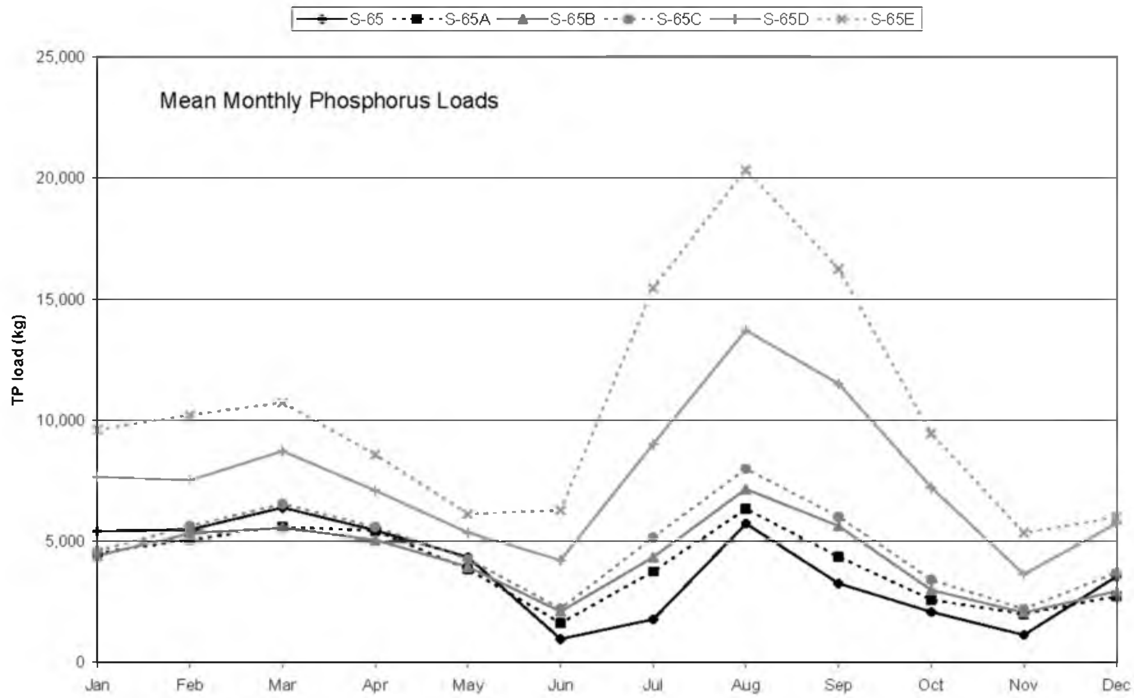
APPENDIX 5-31A

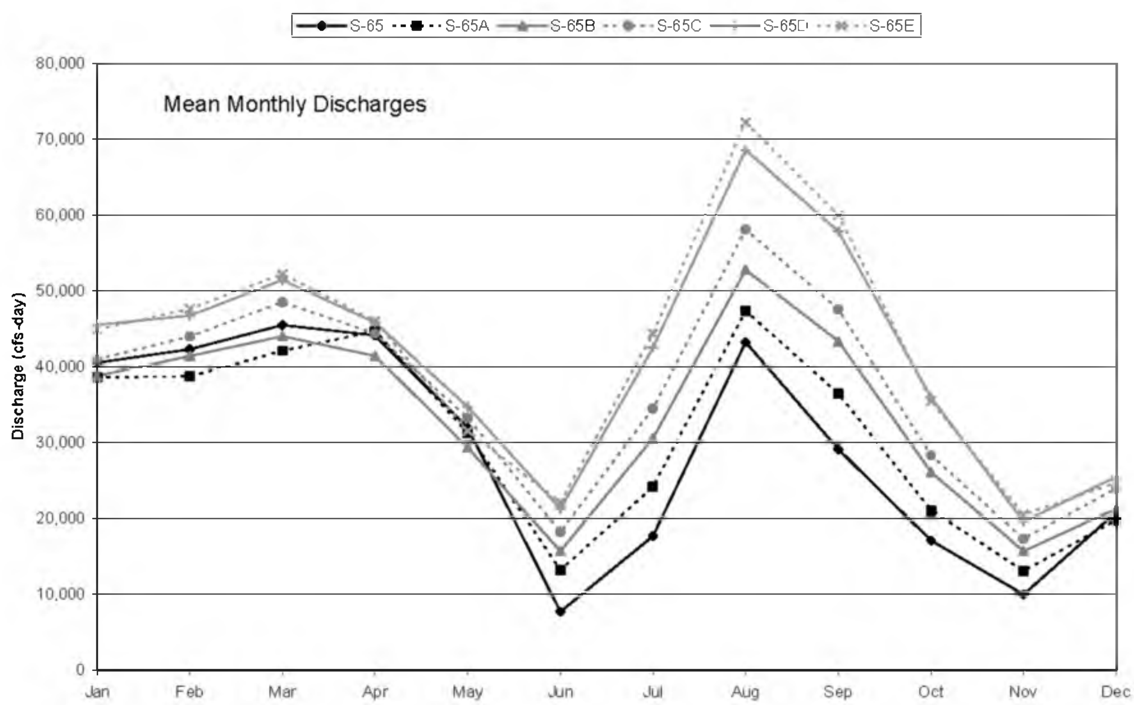
Mean seasonal total phosphorus concentrations in C-38 (computed from median monthly values in Appendix 5-33A).



APPENDIX 5-32A

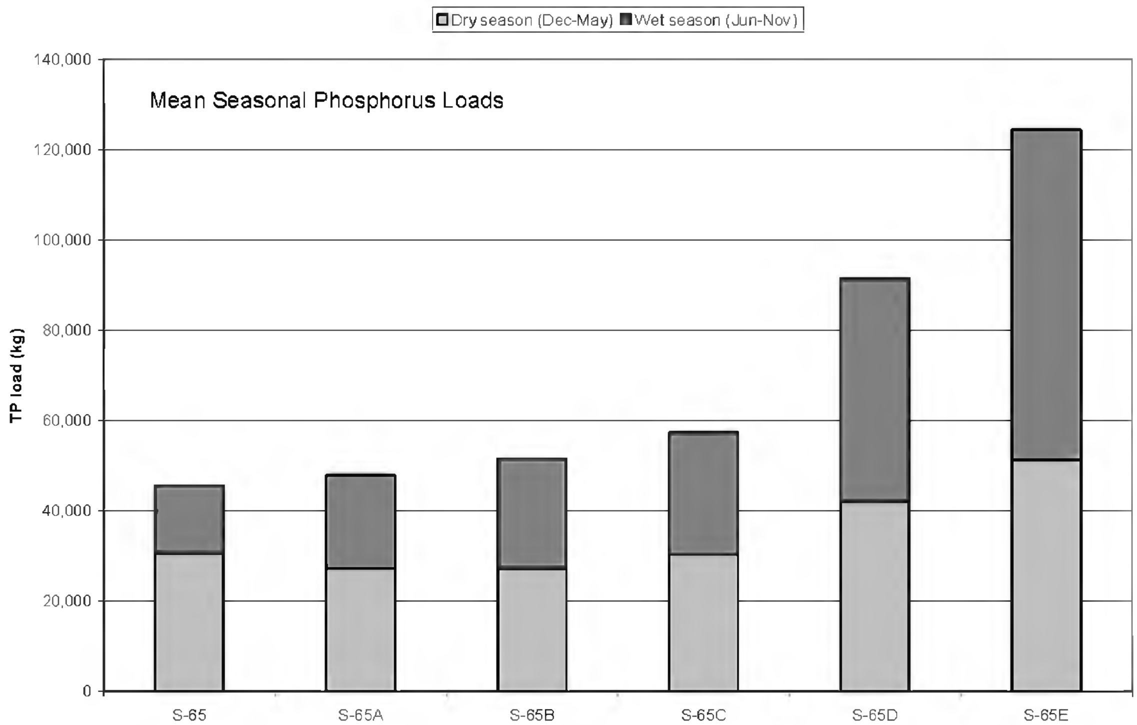
Mean monthly phosphorus loads and discharges at C-38 structures (June 1973 - May 1999).

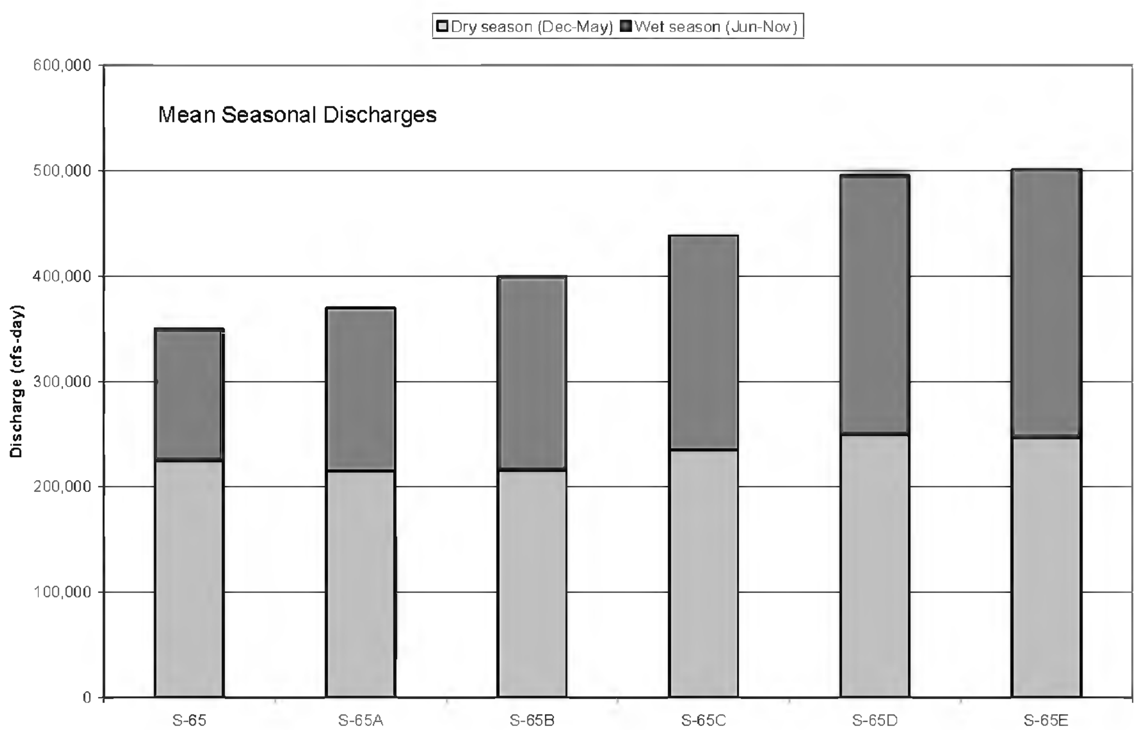




APPENDIX 5-33A

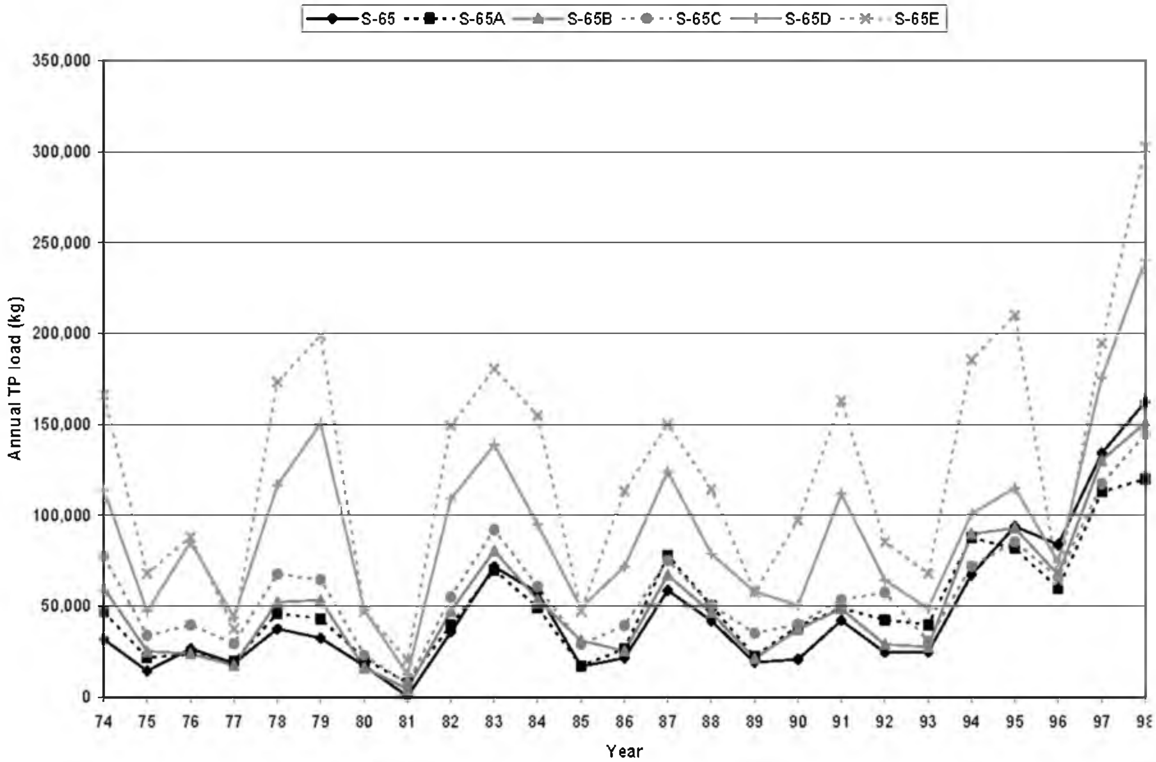
Mean seasonal phosphorus loads and discharges at C-38 structures (June 1973 - May 1999).





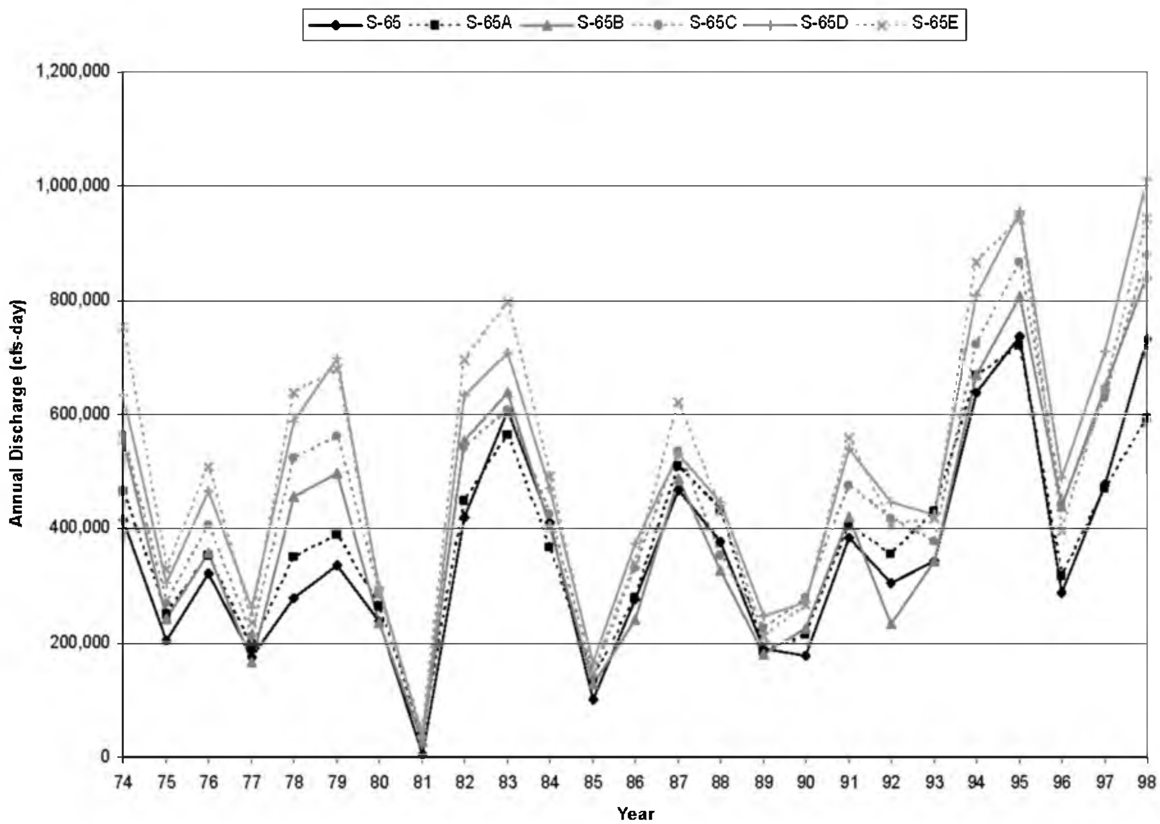
APPENDIX 5-34A

Annual phosphorus loads at C-38 structures.



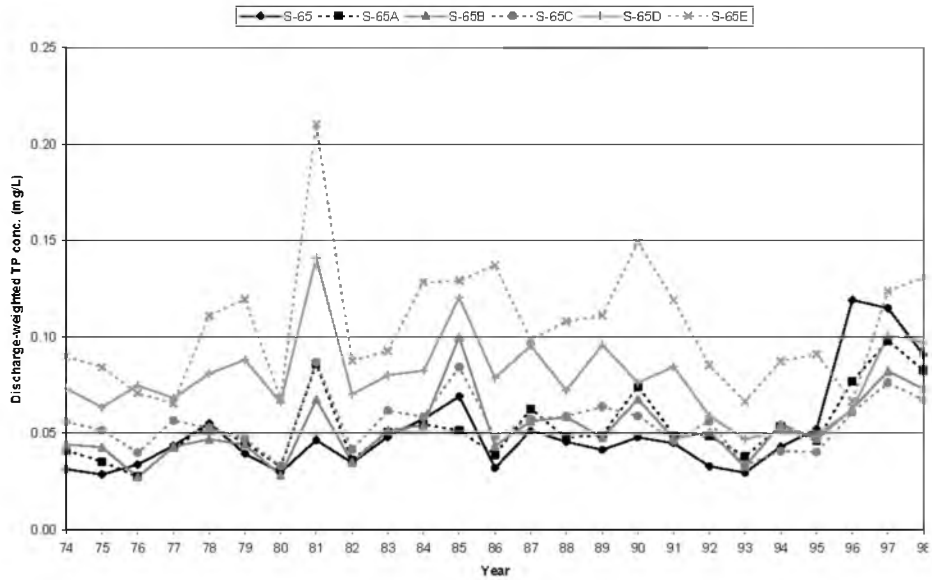
APPENDIX 5-35A

Annual discharges at C-38 structures.



APPENDIX 5-36A

Annual discharge-weighted total phosphorus concentrations at C-38 structures.



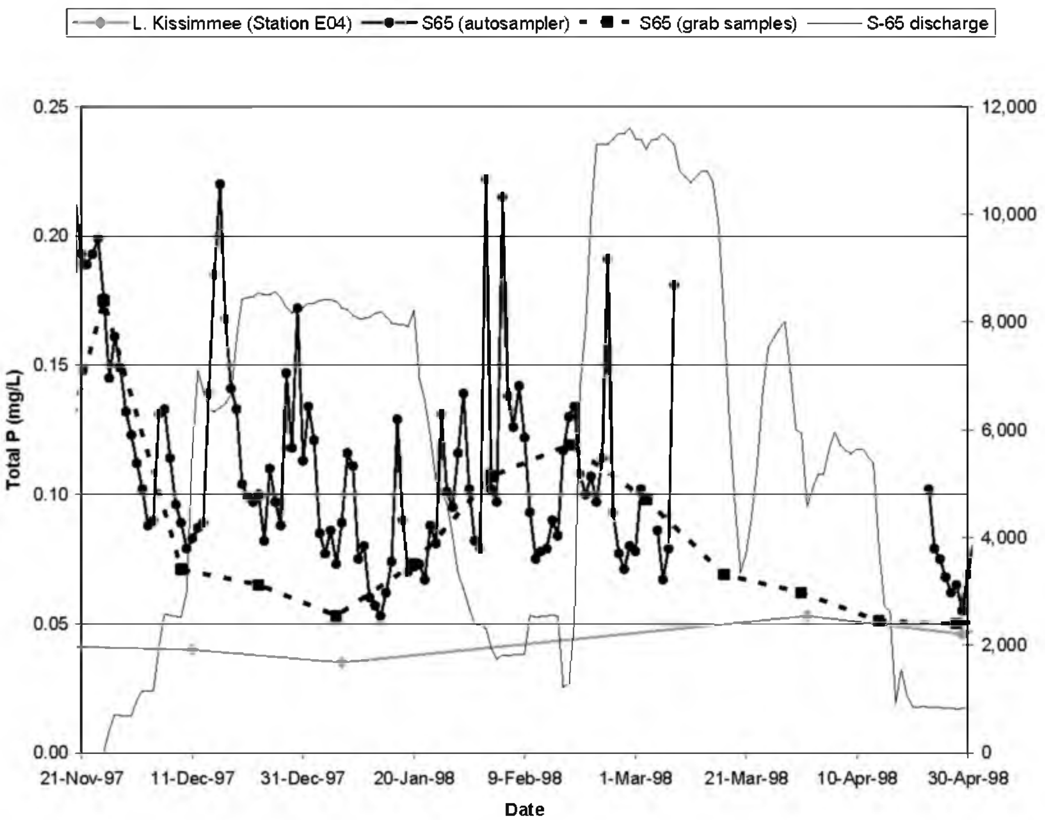
APPENDIX 5-37A

Discharges and total phosphorus loads at C-38 structures before restoration.

Structure	Mean annual 1974-95		Mean annual 1996-98	
	Discharge (cfs-days)	**TP load (metric tons)**	**Discharge (cfs-days)**	**TP load (metric tons)**
S-65	336,627	35	499,209	127
S-65A	364,018	42	460,130	98
S-65B	372,340	43	643,407	116
S-65C	415,846	51	650,052	109
S-65D	468,615	83	737,419	163
S-65E	484,881	117	660,877	187
S-65 as % of S-65E	69%	30%	76%	68%
Pools D&E as % of S-65E	14%	57%	2%	42%

APPENDIX 5-38A

Comparison of total phosphorus concentrations in Lake Kissimmee and at S-65 during winter storms of 1997-1998.



APPENDIX 5-39A

Discharge-weighted concentrations (mg/L) of total phosphorus at C-38 structures before restoration.

Year	S-65	S-65A	S-65B	S-65C	S-65D	S-65E
1974	0.031	0.041	0.044	0.056	0.073	0.090
1975	0.029	0.035	0.043	0.052	0.063	0.084
1976	0.034	0.027	0.027	0.040	0.075	0.071
1977	0.044	0.043	0.043	0.057	0.068	0.065
1978	0.055	0.054	0.047	0.053	0.081	0.111
1979	0.039	0.045	0.044	0.047	0.088	0.120
1980	0.030	0.032	0.028	0.033	0.066	0.068
1981	0.046	0.086	0.068	0.087	0.141	0.210
1982	0.035	0.036	0.035	0.041	0.070	0.088
1983	0.048	0.051	0.051	0.062	0.080	0.093
1984	0.057	0.055	0.054	0.059	0.082	0.128
1985	0.069	0.051	0.100	0.084	0.120	0.129
1986	0.032	0.039	0.043	0.048	0.079	0.137
1987	0.051	0.063	0.056	0.057	0.095	0.099
1988	0.046	0.048	0.059	0.059	0.072	0.108
1989	0.041	0.049	0.048	0.064	0.096	0.111
1990	0.048	0.074	0.068	0.059	0.076	0.149
1991	0.045	0.049	0.047	0.046	0.085	0.119
1992	0.033	0.049	0.051	0.056	0.059	0.085
1993	0.030	0.038	0.033	0.033	0.047	0.066
1994	0.043	0.054	0.055	0.041	0.051	0.088
1995	0.052	0.046	0.047	0.040	0.049	0.091
Mean	**0.043**	**0.048**	**0.050**	**0.053**	**0.078**	**0.105**
Std. Dev.	0.011	0.013	0.015	0.014	0.022	0.033
Std. Error	0.002	0.003	0.003	0.003	0.005	0.007

APPENDIX 6-1A

Periphyton species identified in Pools A and C of the Kissimmee River during baseline sampling (July 1999 - December 1999). * = rheophilic species.

Pool A Periphyton	Pool C Periphyton
Achnanthes delicatula	*Achnanthes delicatula*
Achnanthes exigua	*Achnanthes exigua*
Achnanthes hungaricum	*Achnanthes hungaricum*
Achnanthes lancedata	*Achnanthes linearis*
Achnanthes linearis	*Achnanthes minutissima*
Achnanthes minutissima	*Actinastrum* sp. *
Actinastrum sp. *	*Anabaena aequalis* *
Anabaena aequalis *	*Anabaena* sp. *
Ankistrodesmus falcatus *	*Ankistrodesmus falcatus* *
Anomoneis vitrea	*Anomoneis vitrea*
Aphanizomenon sp. *	*Aphanocapsa* sp.
Aphanocapsa rivularis	*Aphanothece* sp.
Aphanochaete repens	*Asterococcus* sp.
Aphanothece sp.	*Bacteria*
Bacteria	*Caloneis bacillum* *
Caloneis bacillum *	*Calothrix* sp.
Calothrix sp.	*Characiopsis* sp.
Chamaesiphon sp.	*Chlamydomonas* sp. *
Characiopsis sp.	*Chroococcus limneticus*
Chlamydomonas sp. *	*Chroococcus minor*
Chroococcus limneticus	*Chroococcus minutus*
Chroococcus minor	*Closterium lineatum* *
Chroococcus minutus	*Closterium venus* *
Closterium lineatum *	*Coelastrum micro* *
Closterium venus *	*Coelastrum proboscideum*
Cocconeis placentula	*Coelastrum sphaericum*
Coelastrum sphaericum	*Cosmariun phaseolus*
Coloeochaete sp.	*Cosmariun pseudobroomerei*
Cosmariun angul	*Crucigenia crucifera* *
Cosmariun phaseolus	*Crucigenia tetrapedia* *
Cosmariun pseudobroomerei	*Cryptomonas* sp.
Cosmariun trilobulatum	*Cyclotella meneganiana*
Crucigenia crucifera *	*Cymbella minima*
Crucigenia recta *	*Cymbella minuta*
Crucigenia tetrapedia *	*Dactylococcopsis raphidioides*
Cryptomonas tenuis	*Dictyosphaerium* sp. *
Cyclotella meneganiana	*Elactothrix* sp.
Cyclotella steligera	*Eudorina* sp.
Cymbella minima	*Euglena l* sp. *
Cymbella minuta	*Euglena minuta* *
Dactylococcopsis raphidioides	*Eunotia bilunaris*
Desmogonium rabenhorstianum	*Eunotia bilunaris linearis*
Dictyosphaerium sp. *	*Eunotia camelus*
Elactothrix sp.	*Eunotia didyma*
Epithemia argus alpestris	*Eunotia diodon*

APPENDIX 6-1A

Continued.

Pool A Periphyton	Pool C Periphyton
Epithemia sp.	*Eunotia flexuosa*
Euastrum binale	*Eunotia formica*
Euastrum verrucosum	*Eunotia naegelii*
Euglena minuta *	*Eunotia pirla*
Euglena sp. *	*Fragilaria capucina*
Eunotia bilunaris	*Fragilaria construnes*
Eunotia bilunaris linearis	*Fragilaria construnes palmilla*
Eunotia camelus	*Fragilaria construnes vector*
Eunotia carolina	*Fragilaria pinnata*
Eunotia diodon	*Fremyella* sp.
Eunotia flexuosa	*Frustulia rhoboides*
Eunotia formica	*Frustulia rhoboides v. saxonica*
Eunotia naegelii	*Gloecystis* sp.1
Eunotia pirla	*Gloeocystis* sp.2
Fragilaria brebistriate	*Gloeothece rupestris*
Fragilaria capucina	*Gomphonema gracilis*
Fragilaria construnes	*Gomphonema parvalum*
Fragilaria construnes palmilla	*Gomphonema* sp.
Fragilaria construnes vector	*Gomphonema subclavicum*
Fragilaria crotenensis	*Gomphosphaeria* sp.
Fragilaria intermedia	*Gonium* sp.
Fragilaria pinnata	*Kirchneriella subsolitaria*
Fremyella sp.	*Lyngb ter*
Frustulia rhoboides	*Lyngbya limosa*
Frustulia rhoboides v. saxonica	*Lyngbya* sp.1
Gloeocystis sp.	*Lyngbya tenuis*
Gloeothece rupestris	*Melosira distans*
Gomphonema angustatum	*Melosira granulata*
Gomphonema gracilis	*Melosira herzogeii*
Gomphonema intracatum	*Melosira islandica distans*
Gomphonema parvalum	*Merismopedia* *
Gomphonema sp.	*Microcystis* sp. *
Gomphonema subclavicum	*Microthamnion sp.*
Gomphonema turris	*Mougotia* sp.1
Gomphosphaeria sp.	*Mougotia* sp.2
Gonium sp.	*Mougotia* sp.3
Kirchneriella subsolitaria	*Navicula conservacea*
Lyngbya limosa	*Navicula heufleri*
Lyngbya sp.1	*Navicula minima*
Lyngbya sp.2	*Navicula radiosa*
Lyngbya tenuis	*Navicula* sp.1
Melosira distans	*Navicula* sp.2
Melosira granulata	*Navicula* sp.3
Melosira herzogeii	*Navicula* sp.4
Melosira islandica distans	*Navicula* sp.5

APPENDIX 6-1A

Continued.

Pool A Periphyton	Pool C Periphyton
Melosira italica	*Navicula* sp.6
Merismopedia sp. *	*Nephrocytium* sp.1
Microcystis sp. *	*Nitzchia amphibia*
Microthamnion sp.	*Nitzchia archboldii*
Mougotia sp.1	*Nitzchia communis*
Mougotia sp.2	*Nitzchia filiformis*
Mougotia sp.3	*Nitzchia fonticola*
Navicula conservacea	*Nitzchia gracilis*
Navicula cryptocephala	*Nitzchia linearis*
Navicula minima	*Nitzchia obtusa*
Navicula pupula	*Nitzchia palea*
Navicula radiosa	*Nitzchia recta*
Navicula radiosa tenellum	*Nitzchia subacicularis*
Navicula sp.1	*Oedogonium* sp.1
Navicula sp.2	*Oedogonium* sp.2
Navicula sp.3	*Oedogonium* sp.3
Navicula sp.4	*Oocystis parva* *
Navicula sp.5	*Ophiocytium cochleare*
Nitzchia amphibia	*Ophiocytium mucr*
Nitzchia archboldii	*Oscillatoria limnetica* *
Nitzchia communis	*Oscillatoria subbrevis* *
Nitzchia filiformis	*Oscillatoria tenuis* *
Nitzchia fonticola	*Oscillatoria terebriformis* *
Nitzchia gracilis	*Pediastrum bory* *
Nitzchia lacunarum	*Pediastrum obtusum* *
Nitzchia linearis	*Pediastrum tetras 1* *
Nitzchia obtusa	*Pediastrum tetras 2* *
Nitzchia palea	*Phacus curvicauda* *
Nitzchia recta	*Pinnularia acrosphaeria*
Nitzchia subacicularis	*Pinnularia biceps*
Oedogonium sp.1	*Pinnularia subgibba*
Oedogonium sp.2	*Pinnularia subgibba sm*
Oedogonium sp.3	*Quadridula* sp.
Oocystis parva *	*Scenedesmus abundans* *
Ophiocytium cochleare	*Scenedesmus acutiformis* *
Ophiocytium mucr	*Scenedesmus arcuatus*
Oscillatoria limnetica *	*Scenedesmus armatus*
Oscillatoria subbrevis *	*Scenedesmus dimorphus*
Oscillatoria tenuis *	*Scenedesmus quadricauda*
Oscillatoria terebriformis *	*Schizomeris leibleinii*
Pediastrum bory *	*Schizothrix calcicola*
Pediastrum obtusum *	*Selenastrum* sp.
Pediastrum tetras *	*Sphaerocystis* sp.
Phacus curvicauda *	*Spirogyra* sp.1
Pinnularia acrosphaeria	*Spirogyra* sp.2

APPENDIX 6-1A

Continued.

Pool A Periphyton	Pool C Periphyton
Pinnularia subgibba	*Staurastrum* sp.1 *
Pinnularia subgibba sm	*Staurastrum* sp.2 *
Quadridula sp.	*Stigeoclonium* sp. *
Scenedesmus abundans *	*Synedra radians*
Scenedesmus acutiformis *	*Synedra rumpens v. familiaris*
Scenedesmus arcuatus *	*Tetraedron minimum*
Scenedesmus armatus *	*Tetraedron muticum*
Scenedesmus dimorphus *	*Tetrall lageerheimii*
Scenedesmus quadricauda *	*Tetrastrum heteracanthum*
Schizothrix calcicola	*Tetrastrum* sp.
Selenastrum sp.	*Trachelomonas lacustris*
Spirogyra sp.1	*Trachelomonas* sp.1
Spirogyra sp.2	*Trachelomonas* sp.2
Staurastrum sp.1 *	*Ulothrix* sp. *
Staurastrum sp.2 *	*Unknown* sp.1
Stigeoclonium sp. *	*Unknown* sp.2
Synedra filiformis	
Synedra demerarae	
Synedra radians	
Synedra rumpens v. familiaris	
Synedra ulna	
Tetraedron minimum	
Tetraedron muticum	
Tetraedron pentaedricum	
Tetraedron regulare	
Tetrastrum heteracanthum	
Tetrastrum sp.	
Trachelomonas sp.1	
Trachelomonas sp.2	
Unknown sp.1	
Unknown sp.2	

APPENDIX 6-2A

Phytoplankton species identified in Pools A and C of the Kissimmee River during baseline sampling (July 1999 - December 1999). * = truly planktonic species.

Phytoplankton Pool A	Phytoplankton Pool C
Achnanthes delicatula	*Achnanthes delicatula*
Achnanthes exigua	*Achnanthes exigua*
Achnanthes exigua 2	*Achnanthes exigua 2*
Achnanthes hungaricum	*Achnanthes hungaricum*
Achnanthes minutissima	*Achnanthes minutissima*
Achnanthes pinnata	*Achnanthes pinnata*
Achnanthes sp.	*Achnanthes* sp.
Actinastrum sp.	*Actinastrum* sp.
Anab limnetica *	*Anab spiroides* *
Anabaena aequalis *	*Anabaena aequalis* *
Ankistrodesmus falcatus *	*Ankistrodesmus falcatus* *
Aphanizomenon sp.	*Aphanocapsa rivularis* *
Aphanocapsa grevillei *	*Aphanothece* sp. *
Aphanocapsa rivularis *	*Asterococcus* sp.
Aphanothece sp. *	*Bacteria*
Asterococcus sp.	*Botrieococcus* sp.
Bacteria	*Capartogramma crucicula*
Botrieococcus sp.	*Ceratiumsp.*
Capartogramma crucicula	*Chlamydomonas lg* *
Ceratiumsp.	*Chlamydomonas* sp.1 *
Chlamydomonas sp.1 *	*Chlorella* sp. *
Chlorella sp. *	*Chlorochromas* sp.
Chlorochromas sp.	*Chroococcus minor* *
Chroococcus limneticus *	*Chroococcus minutus* *
Chroococcus minor *	*Chroomonas nordstedtii*
Chroococcus minutus *	*Closterium lineatum* *
Chroomonas nordstedtii	*Cocconeis placentula*
Closterium lineatum *	*Coelacium* sp.
Closterium venus *	*Coelastrum proboscideum* *
Cocconeis placentula	*Coelastrum sphaericum* *
Coelacium sp.	*Coelosphaerium* sp.
Coelastrum cambricum *	*Cosmariun angulosum* *
Coelastrum sphaericum *	*Cosmariun phaseolus* *
Coelosphaerium sp.	*Cosmariun sphagnicolum* *
Cosmariun phaseolus *	*Crucigenia crucifera* *
Cosmariun trilobulatum *	*Crucigenia recta* *
Crucigenia apiculata *	*Crucigenia tetrapedia* *
Crucigenia crucifera *	*Cryptomonas erosa* *
Crucigenia tetrapedia *	*Cryptomonas tenuis* *
Cryptomonas erosa *	*Cyclotella comta*
Cryptomonas tenuis *	*Cyclotella meneganiana*
Cyclotella comta	*Cyclotella steligera*
Cyclotella meneganiana	*Cymbella minima*
Cyclotella steligera	*Dactylococcopsis raphidioides*
Cymbella minima	*Dictysphaerium pulchellum* *

APPENDIX 6-2A

Continued.

Phytoplankton Pool A	Phytoplankton Pool C
Dactylococcopsis raphidioides	*Diploneis puella*
Dictysphaerium pulchellum *	*Elactothrix* sp.
Diploneis puella	*Euastrum binale* *
Elactothrix sp.	*Eudorina* sp. *
Euastrum binale *	*Euglena acus var. rigida* *
Eudorina sp. *	*Euglena minuta* *
Euglena acus var. rigida *	*Euglena sp.* *
Euglena minuta *	*Eunotia bilunaris*
Euglena sp. *	*Eunotia formica*
Eunotia bilunaris	*Eunotia pirla*
Eunotia formica	*Fragilaria brebistriate*
Eunotia pirla	*Fragilaria construnes*
Fragilaria brebistriate	*Fragilaria construnes palmilla*
Fragilaria construnes	*Fragilaria construnes vector*
Fragilaria construnes palmilla	*Fragilaria crotenensis*
Fragilaria construnes vector	*Fragilaria pinnata*
Fragilaria crotenensis	*Fremyella* sp.
Fragilaria pinnata	*Frustulia rhoboides v. saxonica*
Fremyella sp.	*Glenodinium* sp.
Frustulia rhoboides v. saxonica	*Gloecapsa* sp.
Glenodinium sp.	*Gloeocystis vericulosa* *
Gloecapsa sp.	*Gomphonema ager*
Gloeocystis vericulosa *	*Gomphonema angestatum*
Gloeothece rupestris *	*Gomphonema gracilis*
Gomphonema ager	*Gomphonema parvulum*
Gomphonema angestatum	*Gomphonema subclavicum*
Gomphonema gracilis	*Gomphosphaeria* sp. *
Gomphonema parvulum	*Gonium* sp.
Gomphonema subclavicum	*Gymnodinium* sp.
Gomphosphaeria sp. *	*Kirchneriella subolitaria* *
Gonium sp.	*Lepto acuta*
Gymnodinium sp.	*Leptocinclis fusiformis* *
Kirchneriella subolitaria *	*Leptocinclis glabra* *
Lepto acuta	*Lyngbya* sp.1
Leptocinclis fusiformis *	*Lyngbya* sp.2
Leptocinclis glabra *	*Mallomonas* sp. *
Lyngbya sp.1	*Melosira granulata*
Lyngbya sp.2	*Melosira islandica 1*
Mallomonas sp. *	*Melosira islandica 2*
Melosira granulata	*Melosira italica*
Melosira islandica 1	*Merismopedia* sp. *
Melosira islandica 2	*Micractinium pusillum*
Melosira italica	*Microcystis* sp.
Merismopedia sp. *	*Microthamnion*
Micractinium pusillum	*Mougotia* sp.1

APPENDIX 6-2A

Continued.

Phytoplankton Pool A	Phytoplankton Pool C
Microcystis sp.	*Mougotia* sp.2
Microthamnion	*Navicula conservacea*
Mougotia sp.1	*Navicula cryptocephala v. veneta*
Mougotia sp.2	*Navicula lanceolat*
Navicula conservacea	*Navicula minima*
Navicula cryptocephala v. veneta	*Navicula radiosa*
Navicula lanceolat	*Navicula radiosa tenellum*
Navicula minima	*Navicula* sp.1
Navicula radiosa	*Navicula* sp.2
Navicula radiosa tenellum	*Navicula* sp.3
Navicula sp.1	*Navicula* sp.4
Navicula sp.2	*Navicula trivialis*
Navicula sp.3	*Nephrocytium obesum**
Navicula sp.4	*Nitzchia acicularis*
Navicula trivialis	*Nitzchia amphibia*
Nitzchia acicularis	*Nitzchia archiboldii*
Nitzchia amphibia	*Nitzchia communis*
Nitzchia archiboldii	*Nitzchia dissipata*
Nitzchia communis	*Nitzchia filiformis*
Nitzchia dissipata	*Nitzchia flexoides*
Nitzchia filiformis	*Nitzchia fonticola*
Nitzchia flexoides	*Nitzchia gracilis*
Nitzchia fonticola	*Nitzchia lacunarum*
Nitzchia gracilis	*Nitzchia linearis*
Nitzchia lacunarum	*Nitzchia obtusa*
Nitzchia linearis	*Nitzchia palea*
Nitzchia obtusa	*Nitzchia recta*
Nitzchia palea	*Nitzchia reversa*
Nitzchia recta	*Nitzchia scalaris*
Nitzchia reversa	*Nitzchia subacicularis*
Nitzchia scalaris	*Ochromonas* sp.
Nitzchia subacicularis	*Oedogonium* sp.2
Ochromonas sp.	*Oocystis parva* *
Oedogonium sp.2	*Ophiocytium cochleare*
Oocystis parva *	*Ophiocytium mucronatum*
Ophiocytium cochleare	*Oscillatoria limnetica* *
Ophiocytium mucronatum	*Oscillatoria subbrevis* *
Oscillatoria limnetica *	*Oscillatoria tenuis* *
Oscillatoria subbrevis *	*Oscillatoria terebriformis* *
Oscillatoria terebriformis *	*Pandor morum*
Pandor morum	*Pedi boryanum* *
Pediastrum obtusum *	*Pediastrum obtusum* *
Pediastrum tetras *	*Pediastrum tetras* *
Peridinium sp. *	*Peridinium* sp. *
Phacus curvicauda *	*Phacus curvicauda* *

APPENDIX 6-2A

Continued.

Phytoplankton Pool A	Phytoplankton Pool C
Phacus longicauda *	*Phacus longicauda* *
Phacus noordstedtii *	*Phacus orbicularis* *
Phacus orbicularis *	*Phacus suecicus* *
Pinnularia subgibba	*Pinnularia subgibba*
Pinnularia subgibba sm	*Pinnularia subgibba sm*
Pinularia biceps	*Pinularia biceps*
Pinularia borealis	*Pinularia borealis*
Pinularia similiformis	*Pinularia similiformis*
Rhizoselium sp.	*Rapphidiopsis curvata*
Scenedesmus abundans *	*Rhizoselium*
Scenedesmus acutiformis *	*Scenedesmus abundans* *
Scenedesmus arcuatus *	*Scenedesmus acutiformis* *
Scenedesmus armatus *	*Scenedesmus arcuatus* *
Scenedesmus dimorphus *	*Scenedesmus armatus* *
Scenedesmus incrassatulus *	*Scenedesmus dimorphus* *
Scenedesmus quadricauda *	*Scenedesmus quadricauda* *
Schizothrix calcicola *	*Schizothrix calcicola* *
Selenastrum westii *	*Selenastrum westii* *
Sorastrum sp. *	*Spondolosium* sp.
Spondolosium sp.	*Spondylomorum quaternarium*
Spondylomorum quaternarium	*Staurastrum* sp.2 *
Staurastrum sp.1 *	*Staurastrum* sp.3 *
Staurastrum sp.2 *	*Stigeoclonium* sp.
Stigeoclonium sp.	*Synedra demerarae*
Synedra demerarae	*Synedra filiformis*
Synedra filiformis	*Synedra radians*
Synedra radians	*Synedra rumpens v. familiaris*
Synedra rumpens v. familiaris	*Synedra ulna*
Synedra ulna	*Synura* sp. *
Tetraedron minimum *	*Tetraedron minimum* *
Tetraedron muticum *	*Tetraedron muticum* *
Tetraedron regulare *	*Tetrallantos lageerheimii*
Tetrallantos lageerheimii	*Tetrastrum heteracanthum*
Tetrastrum heteracanthum	*Tetrastrum* sp.
Tetrastrum sp.	*Tetrastrum staurogeniaeforme*
Tetrastrum staurogeniaeforme	*Trachelomonas girardinna* *
Trachelomonas sp.1 *	*Trachelomonas oblong* *
Trachelomonas sp.2 *	*Trachelomonas* sp.1 *
unknown sp.1	*Trachelomonas* sp.2 *
unknown sp.2	unknown sp.1
	unknown sp.2

APPENDIX 7-1A

List of species recorded in the baseline and reference littoral vegetation surveys.

Code	Species	Growth Form	Origin
AA01	*Ambrosia artemisiifolia*	Emergent	Native
AA05	*Ampelopsis arborea*	Emergent	Native
AC01	*Axonopus compressus*	Emergent	Native
AC02	*Azolia caroliniana*	Floating&Mat-forming	Native
AC10	*Aster carolinianus*	Emergent	Native
AD01	*Symphyotrichum dumosum*	Nonaquatic	Native
AG05	*Andropogon glomeratus*	Emergent	Native
AP01	*Alternanthera philoxeroides*	Emergent	Non-native
AR01	*Acer rubrum*	Emergent	Native
AV01	*Andropogon virginicus*	Emergent	Native
AX99	*Axonopus sp.*	Emergent	Native
BC01	*Bacopa caroliniana*	Emergent	Native
BC05	*Boehmeria cylindrica*	Emergent	Native
BC99	*Bacopa sp.*	Emergent	Native
BH01	*Baccharis halimifolia*	Emergent	Native
BL01	*Bidens laevis*	Nonaquatic	Native
BM01	*Bacopa monnieri*	Emergent	Native
BS01	*Blechnum serrulatum*	Emergent	Native
CA01	*Centella asiatica*	Emergent	Native
CC01	*Cuphea carthagenensis*	Emergent	Non-native
CD01	*Ceratophyllum demersum*	Submergent	Native
CD05	*Commelina diffusa*	Emergent	Native
CD25	*Cyperus distinctus*	Emergent	Native
CG01	*Commelina gigas*	Emergent	Native
CH01	*Cyperus haspan*	Emergent	Native
CL01	*Carex longii*	Emergent	Native
CL02	*Cyperus lanceolatus*	Emergent	Non-native
CO01	*Cephalanthus occidentalis*	Emergent	Native
CO05	*Cyperus odoratus*	Emergent	Native
CP01	*Cyperus polystachyos*	Emergent	Native
CR01	*Cyperus retrorsus*	Emergent	Native
CS99	*Cirsium sp.*	Emergent	Native
DV01	*Diodia virginiana*	Emergent	Native
EC01	*Eichhornia crassipes*	Floating&Mat-forming	Non-native
EC05	*Eupatorium capillifolium*	Emergent	Native
EF01	*Eleocharis flavescens*	Emergent	Native
EH01	*Erechtites hieraciifolia*	Emergent	Native
EI01	*Eleocharis interstincta*	Emergent	Native
EL01	*Eragrostis lugens*	Emergent	Non-native
EO01	*Eleocharis olivacea*	Emergent	Native
EV01	*Eleocharis vivipara*	Emergent	Native
EW01	*Echinochloa walteri*	Emergent	Native
FA01	*Fimbristylis autumnalis*	Emergent	Native
FC01	*Fraxinus caroliniana*	Emergent	Native
FM99	*Fimbristylis sp.*	Emergent	Native
FP01	*Fuirena pumila*	Emergent	Native
GT01	*Galium tinctorium*	Emergent	Native
HA01	*Hemarthria altissima*	Emergent	Non-native
HA02	*Hyptis alata*	Emergent	Native
HC02	*Hypericum cistifolium*	Emergent	Native

Code	Species	Growth Form	Origin
HG01	*Hibiscus grandiflorus*	Emergent	Native
HM01	*Hypericum mutilum*	Emergent	Native
HR01	*Habenaria repens*	Emergent	Native
HR05	*Hydrocotyle ranunculoides*	Emergent	Native
HU01	*Hydrocotyle umbellata*	Emergent	Native
HV01	*Hydrilla verticillata*	Submergent	Non-native
IA01	*Ipomea alba*	Emergent	Native
IC01	*Ilex cassine*	Emergent	Native
IP99	*Ipomea sp.*	Emergent	Native
IS01	*Ipomea sagittata*	Emergent	Native
JE01	*Juncus effusus*	Emergent	Native
JM01	*Juncus marginatus*	Emergent	Native
KB01	*Kyllinga brevifolia*	Emergent	Non-native
KV01	*Kosteletzkya virginica*	Emergent	Native
LC05	*Lachnanthes caroliniana*	Emergent	Native
LF01	*Hydrochloa caroliniensis*	Emergent	Native
LH01	*Leersia hexandra*	Emergent	Native
LL01	*Ludwigia leptocarpa*	Emergent	Native
LM01	*Lygodium microphyllum*	Emergent	Non-native
LM99	*Lemna sp.*	Floating&Mat-forming	Native
LP01	*Ludwigia peruviana*	Emergent	Non-native
LR05	*Ludwigia repens*	Emergent	Native
LS01	*Ludwigia suffructicosa*	Emergent	Native
LS02	*Limnobium spongia*	Floating&Mat-forming	Native
MA01	*Myriophyllum aquaticum*	Emergent	Non-native
MC01	*Myrica cerifera*	Emergent	Native
MP01	*Mitreola petiolata*	Emergent	Native
MP05	*Melothria pendula*	Emergent	Native
MS01	*Mikania scandens*	Emergent	Native
MU01	*Micranthemum umbrosum*	Emergent	Native
MV01	*Magnolia virginiana*	Emergent	Native
NG01s	*Najas guadalupensis*	Submergent	Native
NL01	*Nuphar lutea*	Emergent	Native
OC01	*Osmunda cinnamomea*	Emergent	Native
OR01	*Osmunda regalis*	Emergent	Native
OS99	*Osmunda sp.*	Emergent	Native
PB01	*Persea borbonia*	Emergent	Native
PC01	*Pontederia cordata*	Emergent	Native
PC02	*Ptilimnium capillaceum*	Emergent	Native
PC05	*Paspalum conjugatum*	Emergent	Native
PD01	*Polygonum densiflorum*	Emergent	Native
PD06	*Paspalum dilatatum*	Emergent	Non-native
PG01	*Psidium guajava*	Emergent	Non-native
PG05	*Paspalidium geminatum*	Emergent	Native
PH01	*Panicum hemitomon*	Emergent	Native
PH10	*Polygonum hydropiperoides*	Emergent	Native
PL01	*Paspalum laeve*	Emergent	Native
PL99	*Polygonum sp.*	Emergent	Native
PN01	*Paspalum notatum*	Emergent	Native
PN10	*Phyla nodiflora*	Emergent	Native

Code	Species	Growth Form	Origin
PP01	*Polygonum punctatum*	Emergent	Native
PQ01	*Parthenocissus quinquefolia*	Emergent	Native
PR01	*Panicum repens*	Emergent	Non-native
PR05	*Pluchea rosea*	Emergent	Native
PR10	*Paspalum repens*	Emergent	Native
PS01	*Pistia stratiotes*	Floating&Mat-forming	Non-native
PS05	*Peltandra sagittifolia*	Emergent	Native
RC01	*Rubus cuneifolius*	Emergent	Native
RC02	*Rhynchospora colorata*	Emergent	Native
SB01	*Spartina bakeri*	Emergent	Native
SC01	*Salix caroliniana*	Emergent	Native
SC05	*Scirpus cubensis*	Floating&Mat-forming	Non-native
SC10	*Sarcostemma clausum*	Emergent	Native
SC15	*Sambucus canadensis*	Emergent	Native
SC20	*Scirpus californicus*	Emergent	Native
SC25	*Saururus cernuus*	Emergent	Native
SF01	*Solidago fistulosa*	Emergent	Native
SI01	*Sacciolepis indica*	Emergent	Non-native
SL01	*Sagittaria lancifolia*	Emergent	Native
SL05	*Sagittaria latifolia*	Emergent	Native
SM01	*Salvinia minima*	Floating&Mat-forming	Non-native
SR01	*Serenoa repens*	Emergent	Native
SR05	*Smilax rotundifolia*	Emergent	Native
SR10	*Scleria reticularis*	Emergent	Native
SS01	*Sacciolepis striata*	Emergent	Native
ST01	*Schinus terebinthifolius*	Emergent	Non-native
SV01	*Solanum viarum*	Emergent	Non-native
SV02	*Sesbania vesicaria*	Emergent	Native
TC01	*Teucrium canadense*	Emergent	Native
TD01	*Taxodium distichum*	Emergent	Native
TD02	*Thelypteris dentata*	Emergent	Native
TD05	*Typha domingensis*	Emergent	Native
TI01	*Thelypteris interrupta*	Emergent	Native
TL99	*Thelypteris sp.*	Emergent	Native
TP01	*Thelypteris palustris*	Emergent	Native
TV01	*Triadenum virginicum*	Emergent	Native
UL01	*Urena lobata*	Emergent	Non-native
UT99s	*Utricularia sp.*	Submergent	Native
VL01	*Vigna luteola*	Emergent	Native
VR01	*Vitis rotundifolia*	Emergent	Native
VT99	*Vitis sp.*	Emergent	Native
WA01	*Woodwardia areolata*	Emergent	Native
WD99	*Woodwardia sp.*	Floating&Mat-forming	Native
WG01	*Wolffiella gladiata*	Floating&Mat-forming	Native
WV01	*Woodwardia virginica*	Emergent	Native
XF01	*Xyris fimbriata*	Emergent	Native

APPENDIX 8-1A

Species encountered in floodplain vegetation sampling.

Species	Code	Wetland status	Species	Code	Wetland status	Species	Code	Wetland status	Species	Code	Wetland status
Acalypha gracilens	AG01	UPL	*Digitaria spp.*	DG99		*Myrica cerifera*	MC01	FAC	*Sacciolepis striata*	SS01	OBL
Acer rubrum	AR01	FAC	*Diodia virginiana*	DV01	FACW	*Nuphar lutea*	NL01	OBL	*Sagittaria lancifolia*	SL01	OBL
Alternanthera philoxeroides	AP01	OBL	*Diospyros virginiana*	DV05	FAC	*Osmunda cinnamomea*	OC01	FACW	*Salix caroliniana*	SC01	OBL
Amaranthus spinosa	AS01	FACU	*Drymaria cordata*	DC01	FAC	*Osmunda regalis*	OR01	OBL	*Salvinia minima*	SM01	OBL
Ambrosia artemisiifolia	AA01	FACU	*Echinochloa walteri*	EW01	OBL	*Oxalis corniculata*	OC02	FACU	*Sambucus canadensis*	SC15	FACW
Ampelopsis arborea	AA05	FAC	*Eleocharis vivipara*	EV01	OBL	*Panicum angustifolium*	PA03	FACU	*Sarcostemma clausum*	SC10	FACW
Andropogon glomeratus	AG05	FACW	*Eleusine indica*	EI05	FACU	*Panicum dichotomum*	PD04	FAC	*Saururus cernuus*	SC25	OBL
Andropogon virginicus	AV01	FAC	*Eragrostis lugens*	EL01	FAC	*Panicum hemitomon*	PH01	OBL	*Schinus terebinthifolius*	ST01	FAC
Asclepias incarnata	AI01	OBL	*Erechtites hieraciifolia*	EH01	FAC	*Panicum hians*	PH20	OBL	*Scirpus californicus*	SC20	OBL
Aster carolinianus	AC10	OBL	*Eryngium baldwinii*	EB02	FACW	*Panicum repens*	PR01	FACW	*Scirpus cubensis*	SC05	OBL
Aster elliotti	AE01	OBL	*Eupatorium capillifolium*	EC05	FACU	*Panicum rigidulum*	PR02	FACW	*Scleria reticularis*	SR10	FACW
Asteraceae spp.	AS00		*Euthamia caroliniana*	EC15	FAC	*Panicum sphaerocarpon*	PS03	FACU	*Scoparia dulcis*	SD01	FAC
Axonopus compressus	AC01	FACW	*Fimbristylis autumnalis*	FA01	OBL	*Panicum spp.*	PN99		*Senna obtusifolia*	SO01	FACU
Axonopus fissifolius	AF02	FACW	*Fimbristylis caroliniana*	FC02	FACW	*Panicum verrucosum*	PV01	FACW	*Sesbania vesicaria*	SV02	FAC
Axonopus furcatus	AF01	OBL	*Fimbristylis dichotoma*	FD01	OBL	*Parthenocissus quinquefolia*	PQ01	FAC	*Setaria magna*	SM10	FACW
Baccharis halimifolia	BH01	FAC	*Fimbristylis spp.*	FM99		*Paspalidium geminatum*	PG05	OBL	*Setaria parviflora*	SP02	FAC
Bacopa caroliniana	BC01	OBL	*Galium tinctorum*	GT01	FACW	*Paspalum acuminatum*	PA01	OBL	*Sida acuta*	SA02	FAC
Bacopa monnieri	BM01	OBL	*Habenaria repens*	HR01	OBL	*Paspalum conjugatum*	PC05	FAC	*Sida cordifolia*	SC02	UPL
Bidens mitis	BM02	OBL	*Hemarthria altissima*	HA01	FACW	*Paspalum dilatatum*	PD06	FAC	*Sida elliotti*	SE01	UPL
Blechnum serrulatum	BS01	FACW	*Hibiscus grandiflorus*	HG01	OBL	*Paspalum dissectum*	PD02	OBL	*Sida rhombifolia*	SR02	FACU
Boehmeria cylindrica	BC05	FACW	*Hydrochloa caroliniensis*	LF01	OBL	*Paspalum distichum*	PD11	OBL	*Sisyrinchium angustifolium*	SA01	FAC
Boltonia diffusa	BD01	FAC	*Hydrocotyle umbellata*	HU01	OBL	*Paspalum laeve*	PL01	FACW	*Smilax auriculata*	SA04	FACU
Callicarpa americana	CA15	FACU	*Hypericum cistifolium*	HC02	UPL	*Paspalum notatum*	PN01	FACU	*Solanum americanum*	SA06	FACU
Cardiospermum microcarpum	CM01	FAC	*Hypericum hypericoides*	HH01	FAC	*Paspalum setaceum*	PS02	FAC	*Solanum viarum*	SV01	UPL
Carex longii	CL01	OBL	*Hypericum tetrapetalum*	HT01	FACW	*Paspalum urvillei*	PU01	FAC	*Solidago fistulosa*	SF01	FAC
Centella asiatica	CA01	FACW	*Hyptis alata*	HA02	OBL	*Persea borbonia*	PB01	FACW	*Sporobolus indicus*	SI02	FACU
Cephalanthus occidentalis	CO01	OBL	*Ipomea alba*	IA01	FAC	*Phyla nodiflora*	PN10	FACW	*Symphyotrichium dumosum*	AD01	FAC
Chamaecrista nictitans	CN05	FACU	*Ipomea sagittata*	IS01	FACW	*Phytolacca americana*	PA05	FACU	*Teucrium canadense*	TC01	FACW
Cirsium horridulum	CH05	FAC	*Juncus effusus*	JE01	FACW	*Pluchea foetida*	PF02	OBL	*Thelypteris interrupta*	TI01	FAC
Commelina diffusa	CD05	FACW	*Juncus marginatus*	JM01	FACW	*Pluchea odorata*	PO01	FACW	*Thelypteris kunthii*	TK01	FACW
Commelina gigas	CG01	FACW	*Justicia angusta*	JA01	OBL	*POACEAE*	PC00		*Thelypteris palustris*	TP01	OBL
Conoclinium coelestinum	CC02	FAC	*Kosteletzkya virginica*	KV01	OBL	*Polygonum hirsutum*	PH05	OBL	*Tillandsia spp.*	TN99	
Coreopsis leavenworthii	CL03	FACW	*Kyllinga brevifolia*	KB01	FACW	*Polygonum hydropiperoides*	PH10	OBL	*Triadenum virginicum*	TV01	OBL
Cuphea carthagenensis	CC01	FACW	*Kyllinga odoratus (odorata?)*	KO01	FACW	*Polygonum punctatum*	PP01	FACW	*Urena lobata*	UL01	FAC
Cynodon dactylon	CD10	FACU	*Kyllinga pumila*	KP01	FACW	*Polypremum procumbens*	PP03	FACU	*Urochloa subquadripara*	US01	FAC
Cyperaceae spp.	CP00		*Leersia hexandra*	LH01	OBL	*Pontederia cordata*	PC01	OBL	*Utricularia spp.*	UT99	OBL
Cyperus articulatus	CA05	OBL	*Lemna spp.*	LM99	OBL	*Proserpinaca palustris*	PP02	OBL	*Verbena scabra*	VS01	FACW
Cyperus compressus	CC04	FACW	*Lepidium virginicum*	LV01	FACU	*Psidium guajava*	PG01	FACU	*Vigna speciosa*	VS02	UPL
Cyperus croceus	CC03	FAC	*Ludwigia decurrens*	LD01	OBL	*Psilocarya nitens*	RN05	OBL	*Vitis rotundifolia*	VR01	FAC
Cyperus distinctus	CD25	FACW	*Ludwigia peruviana*	LP01	OBL	*Quercus virginiana*	QV01	FACU	*Woodwardia areolata*	WA01	OBL
Cyperus haspan	CH01	OBL	*Ludwigia repens*	LR05	OBL	*Rhexia mariana*	RM05	FACW	*Woodwardia virginica*	WV01	OBL
Cyperus polystachyos	CP01	FACW	*Ludwigia spp.*	LD99		*Rhus copallinum*	RC03	FACU	*Xyris fimbriata*	XF01	OBL
Cyperus retrorsus	CR01	FACU	*Ludwigia suffruticosa*	LS01	OBL	*Rhynchospora colorata*	RC02	FACW			
Cyperus spp.	CP99		*Lygodium microphyllum*	LM01	FAC	*Rhynchospora fascicularis*	RF01	FACW			
Cyperus surinamensis	CS01	FACW	*Lythrum alatum*	LA01	FACW	*Rhynchospora inundata*	RI01	OBL			
Desmodium incanum	DI01	FAC	*Macroptilium lathyroides*	ML01	FACU	*Rhynchospora microcarpa*	RM01	FACW			
Desmodium triflorum	DT01	FACU	*Magnolia virginiana*	MV01	FACW	*Rhynchospora microcephala*	RM10	OBL			
Dichondra caroliniensis	DC03	FAC	*Melothria pendula*	MP05	FACW	*Rubus cuneifolius*	RC01	FACU			
Digitaria ciliaris	DC02	FAC	*Mikania scandens*	MS01	FACW	*Sabal palmetto*	SP01	FAC			
Digitaria longiflora	DL01	UPL	*Momordica charantia*	MC02	UPL	*Sacciolepis indica*	SI01	FAC			

APPENDIX 9-1A

Key to Bcode Groups and community types (decision rules).

Living vegetation cover equal to or greater than 10%.. Go to **1a.**
Living vegetation not present or very sparse (less than 10% cover), including housing and associated grounds...Go to **1b.**
Problematic communities and signatures .... Go to **1c.**

1a. Living vegetation cover equal to or greater than 10%.
2a. Vine cover less than 50%.
3a. Tree cover equal to or greater than 30%.
4a. Forests in upland habitats...**Upland Forest Bcode Group**

Upland Forest Bcode Group, UF

Pinus elliottii the dominant tree species..*Pinus elliottii* forest **[F.PE]**

Quercus virginiana dominant, often with *Sabal palmetto;* understory often including *Serenoa repens*..... ....*Quercus virginiana (-Sabal palmetto)* forest **[F.QS]**

Sabal palmetto the dominant tree species... *Sabal palmetto* forest **[F.SP]**

Unclassified combinations of upland tree species........................... Miscellaneous upland forest **[F.MxF]**

4b. Forests in wetland habitats...**Wetland Forest Bcode Group**

Wetland Forest Bcode Group, WF

Acer rubrum and/or *Nyssa silvatica* var. *biflora* the dominant tree species:.. ....*Acer rubrum (-Nyssa silvatica* var. *biflora)* forest **[F.AR]**

Fraxinus caroliniana the dominant tree species: .... *Fraxinus caroliniana* forest **[F.FC]**

Magnolia virginiana the dominant tree species .... *Magnolia virginiana* forest **[F.MV]**

Taxodium distichum the dominant tree species .... *Taxodium distichum* forest **[F.TDF]**

Mixtures of upland and wetland species (e.g., *Quercus* spp. with *Acer rubrum, Persea* spp., *Fraxinus caroliniana*, *Taxodium distichum*, and/or *Magnolia virginiana*) .... Mixed transitional forest **[F.MTF]**

3b. Tree cover less than 30%, vine cover less than 50%.
5a. Total shrub cover equal to or greater than 30%.

APPENDIX 9-1A

Continued.

6a. Shrub communities in upland habitats, and successional-transitional shrub communities in wetland-upland transition areas with species composition dominated by mesophytes .... **Upland Shrub Bcode Group**

Upland Shrub Bcode Group, US

Myrica cerifera (waxmyrtle) usually the dominant shrub species, occasionally approximately codominant with *Ludwigia peruviana*, *Baccharis halimifolia*, or other woody mesophytes or hydrophytes; not on floating mat vegetation .... *Myrica cerifera* shrubland **[S.MC]**

Psidium guajava (guava) the dominant shrub species........................ *Psidium guajava* shrubland **[S.PG]**

Schinus terebinthifolius (Brazilian pepper) the dominant shrub species .... *Schinus terebinthifolius* shrubland **[S.ST]**

Serenoa repens (saw palmetto) the dominant shrub species .... *Serenoa repens* shrubland **[S.SR]**

Other upland and successional-transitional shrub communities without significant cover of *Myrica cerifera*; *Baccharis halimifolia* or *Sambucus* spp. (among others) the dominant shrub species .... Miscellaneous upland shrubland **[S.MxUS]**

6b. Shrub communities in wetland habitats, and transitional communities with species composition dominated by wetland species, not on floating mats of aquatic vegetation.

7a. Continuous floating mats with shrubs established, rooted either in or below the mat.. **Floating Mat Shrublands**

Floating Mat Shrublands

(These three communities are in the <**Aquatic Vegetation Bcode Group, AQ**>).

Ludwigia spp. (*L. peruviana* and/or *L. leptocarpa)* dominant .... *Ludwigia* spp. floating mat shrubland **[S.LSF]**

Myrica cerifera the dominant shrub species.................. *Myrica cerifera* floating mat shrubland **[S.MCF]**

Other shrub-dominated communities on floating mats .... Miscellaneous floating mat shrubland **[S.MxFS]**

APPENDIX 9-1A

Continued.

7b. Wetland shrub communities not on floating mats .... **Wetland Shrub Bcode Group**

Wetland Shrub Bcode Group, WS

Cephalanthus occidentalis cover 50% or greater, understory like H.PS herbaceous vegetation.................
.... *Cephalanthus occidentalis* shrubland **[S.CO]**

Cephalanthus occidentalis cover 30%-45% cover in otherwise H.PS herbaceous vegetation ....
.... *Cephalanthus occidentalis-Pontederia cordata-Sagittaria lancifolia* shrubland **[S.CO-PS]**

Cephalanthus occidentalis cover 30%-45% in otherwise H.PS-PH herbaceous vegetation, understory sometimes composed primarily of wet prairie species (e.g., *Panicum hemitomon*) ....
.....................................*C. occidentalis-P. cordata-S. lancifolia-P. hemitomon* shrubland **[S.CO-PS-PH]**

Hypericum fasciculatum the dominant shrub species.............*Hypericum fasciculatum* shrubland **[S.HF]**

Ludwigia spp. (*L. peruviana* and/or *L. leptocarpa*) the dominant shrub, often with *Salix caroliniana, Baccharis halimifolia*, or other shrub species ....*Ludwigia* spp. shrubland **[S.LS]**

Salix caroliniana the dominant shrub species, sometimes associated with *Ludwigia peruviana*................
....*Salix caroliniana* shrubland **[S.SC]**

5b. Tree and shrub cover both less than 30%.

8a. Wetland and terrestrial herbaceous vegetation.

9a. Herbaceous vegetation in upland habitats .... **Upland Herbaceous Bcode Group**

Upland Herbaceous Bcode Group, UP

Axonopus fissifolius dominant, usually with mixtures of *Paspalum notatum* and other species.................
.... *Axonopus fissifolius* herbaceous vegetation **[H.AF]**

Cynodon dactylon dominant...*Cynodon dactylon* herbaceous vegetation **[H.CD]**

Hemarthria altissima dominant.............................. *Hemarthria altissima* herbaceous vegetation **[H.HA]**

Paspalum notatum cover equal to or greater than 50%, usually with mixtures of upland species.............
.... *Paspalum notatum* herbaceous vegetation **[H.PN]**

Invasive exotics dominant (levees, abandoned pastures) ....
.... Miscellaneous exotic herbaceous vegetation **[H.MxE]**

Invasive, weedy native species dominant (e.g., *Eupatorium* spp., *Ambrosia* spp., *Cirsium* spp., *Euthamia* spp., etc.)..Miscellaneous invasive herbaceous vegetation **[H.MxW]**

Native terrestrial grasses dominant, usually with scattered shrubs and upland forbs.................................
.... Miscellaneous native herbaceous vegetation **[H.MxN]**

APPENDIX 9-1A

Continued.

9b. Herbaceous vegetation in wetland habitats, not on floating mats.

10a. Communities with equal to or greater than 50% cover of *Pontederia cordata* and/or *Sagittaria lancifolia* or 10-45% cover of *P. cordata* and/or *S. lancifolia* and less than 50% cover of *Panicum hemitomon* .... **Broadleaf Marsh Bcode Group**

Broadleaf Marsh Bcode Group, BLM

Sagittaria lancifolia and/or *Pontederia cordata* combined or individual cover equal to or greater than 50%. If present, *Cephalanthus occidentalis* cover less than 5% .... *Pontederia cordata-Sagittaria lancifolia* herbaceous vegetation **[H.PS]**

Sagittaria lancifolia and/or *Pontederia cordata,* and/or cover 10-45%, *Panicum hemitomon* cover equal to or greater than 10%; these three species combined making up equal to or greater than 40% cover ....*Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon* herbaceous vegetation **[H.PS-PH]**

Cephalanthus occidentalis cover 5%-25% cover in otherwise H.PS herbaceous vegetation ....*Pontederia cordata-Sagittaria lancifolia-Cephalanthus occidentalis* herbaceous vegetation **[H.PS-CO]**

Cephalanthus occidentalis cover 5%-25% in otherwise H.PS-PH herbaceous vegetation .... *P. cordata-S. lancifolia-P. hemitomon-C. occidentalis* herbaceous vegetation **[H.PS-PH-CO]**

Hibiscus grandiflorus cover 30-45% in otherwise H.PS vegetation ....*Hibiscus grandiflorus-Pontederia cordata-Sagittaria lancifolia* herbaceous vegetation **[H.PS-HG]**

APPENDIX 9-1A

Continued.

10b. Communities not as in (9a) above.

11a. Communities with equal to or greater than 50% cover of *Panicum hemitomon* or dominated by *Panicum repens*, *Rhynchospora* spp., *Cyperus* spp., *Eleocharis* spp., *Iris virginica, Leersia hexandra, Luziola fluitans, Polygonum punctatum, Andropogon glomeratus, Juncus effusus,* or combinations of these species .... **Wet Prairie Bcode Group**

Wet Prairie Vegetation Bcode Group, WP

Panicum hemitomon cover equal to or greater than 50% .... *Panicum hemitomon* herbaceous vegetation **[H.PH]**

Panicum repens dominant .... *Panicum repens* herbaceous vegetation **[H.PR]**

Rhynchospora spp. dominant (usually *R. inundata*).. *Rhynchospora* spp. herbaceous vegetation **[H.RN]**

Juncus effusus cover equal to or greater than 30%, not within isolated ponds or depressions [compare *J. effusus* herbaceous vegetation (upland depressions), above].. .... *Juncus effusus* herbaceous vegetation (wet prairies) **[H.JEp]**

Juncus effusus dominant in ponds or depressions that are inclusions within otherwise upland habitats [compare *Juncus effusus* herbaceous vegetation (wet prairies), below] .... *Juncus effusus* herbaceous vegetation (upland depressions) **[H.JEd]**

Andropogon glomeratus dominant .... *Andropogon glomeratus* herbaceous vegetation **[H.AG]**

Cyperus spp. dominant .... *Cyperus* spp. herbaceous vegetation **[H.CS]**

Eleocharis spp. dominant .... *Eleocharis* spp. herbaceous vegetation **[H.ES]**

Iris virginica dominant .... *Iris virginica* herbaceous vegetation **[H.IV]**

Leersia hexandra dominant .... *Leersia hexandra* herbaceous vegetation **[H.LH]**

Luziola fluitans dominant .... *Luziola fluitans* herbaceous vegetation **[H.LF]**

Polygonum punctatum dominant .... *Polygonum punctatum* herbaceous vegetation **[H.PP]**

Communities composed of mixtures of the species listed above (composition intermediate between other wet prairie types)............... Miscellaneous transitional herbaceous wetland vegetation **[H.MxWP]**

Other mixtures of native wetland grasses (e.g., *Phragmites australis, Paspalidium* spp.) and/or graminoids (*Cyperus* spp., *Scirpus californicus, Juncus* spp.), or dominance not clear............................. .... Miscellaneous native wetland graminoid vegetation **[H.MxWT]**

APPENDIX 9-1A

Continued.

11b. Communities not as in (11a) above .... **Miscellaneous Herbaceous Wetland Bcode Group**

Miscellaneous Wetland Vegetation Bcode Group, MW

Cladium jamaicense (sawgrass) dominant .... *Cladium jamaicense* herbaceous vegetation **[H.CJ]**

Communities dominated by fern species .... Miscellaneous fern-dominated herbaceous vegetation **[H.MxFN]**

Hibiscus grandiflorus cover equal to or greater than 50% .... *Hibiscus grandiflorus* herbaceous vegetation **[H.HG]**

Spartina bakeri (sand cordgrass) cover equal to or greater than 30% .... *Spartina bakeri* herbaceous vegetation **[H.SB]**

Typha domingensis (southern cattail) cover equal to or greater than 50% .... *Typha dom*ingensis herbaceous vegetation **[H.TY]**

8b. Aquatic, littoral, and floating mat herbaceous communities .... **Aquatic Vegetation Bcode Group**
12a. Emergent and floating vegetation .... **Aquatic Vegetation Bcode Group, AQ**

Emergent, floating, and floating mat aquatic vegetation (Aquatic Vegetation Bcode Group)

Eichhornia crassipes dominant .... *Eichhornia crassipes* herbaceous aquatic vegetation **[H.EC]**

Eichhornia crassipes and *Pistia stratiotes* codominant .... *Eichhornia crassipes-Pistia stratiotes* herbaceous aquatic vegetation **[H.EC-PST]**

Pistia stratiotes dominant .... *Pistia stratiotes* herbaceous aquatic vegetation **[H.PST]**

Hydrocotyle umbellata dominant .... *Hydrocotyle umbellata* herbaceous aquatic vegetation **[H.HU]**

Nuphar lutea dominant... *Nuphar lutea* herbaceous aquatic vegetation **[H.NL]**

Polygonum densiflorum dominant .... *Polygonum densiflorum* herbaceous aquatic vegetation **[H.PD]**

Sacciolepis striata dominant .... *Sacciolepis striata* herbaceous aquatic vegetation **[H.SS]**

Scirpus cubensis dominant .... *Scirpus cubensis* herbaceous floating mat vegetation **[H.SCF]**

Scirpus mats with other herbaceous species dominant .... Miscellaneous herbaceous floating mat vegetation **[H.MFM]**

Aquatic communities dominated by combinations of floating species (e.g., *Salvinia* spp., *Azolla* spp., *Lemna* spp., etc.), and where dominance is not clear .... Miscellaneous aquatic vegetation dominated by floating species **[H.MxFA]**

Littoral vegetation dominated by unclassified combinations of species including *Sagittaria lancifolia, Pontederia cordata,* and others .... Miscellaneous littoral marsh vegetation **[H.MxM]**

APPENDIX 9-1A

Continued.

.... **12b.** Submergent vegetation

Submergent vegetation (Aquatic Vegetation Bcode Group)
Aquatic communities dominated by combinations of submergent species (*Ceratophyllum* spp., *Hydrilla* spp., *Utricularia* spp., *Chara* spp., algal *Periphyton)* .... Miscellaneous submergent aquatic vegetation **[H.MxSV]**

2b. Vine cover equal to or greater than 50%. .... **Vines Bcode Group**

Vines Bcode Group, VN
Lygodium microphyllum cover equal to or greater than 30%, typically on living trees or shrubs............... .... *Lygodium microphyllium-dominated* communities **[V.LM]**
Other vine species with cover exceeding 30%, typically on living trees or shrubs.................................... .... Miscellaneous vine-dominated communities **[V.MxV]**

1b. Living vegetation not present or very sparse (less than 10% cover), including housing and associated grounds and open water

13a. Open water .... **Open Water Bcode Group**

Open Water Bcode Group, NVOW.. Open water **[NVOW]**

13b. Not open water.

14a. No vegetation - bare ground... **Bare Ground Bcode Group**

Bare Ground Bcode Group, NVBG.. Bare ground **[NVBG]**

14b. No vegetation - human-made structures, roads, etc., including lawns .... **Human-made Structures and Grounds Bcode Group**

Human-made Structures and Grounds Bcode Group, NVH.............. Human-made structures **[NVH]**

1c. Problematic communities and signatures... **Unknown Bcode Group**

APPENDIX 9-1A

Continued.

Unknown Vegetation Bcode Group, UN
Unclassified combinations of species.......... Unclassified **[X.XUNCL]**
Uninterpretable signatures.......... Uninterpretable **[X.XUNK]**

13b. Not open water.

14a. No vegetation - bare ground .... **Bare Ground Bcode Group**

Bare Ground Bcode Group, NVBG.......... Bare ground **[NVBG]**

14b. No vegetation - human-made structures, roads, etc., including lawns .... **Human-made Structures and Grounds Bcode Group**

Human-made Structures and Grounds Bcode Group, NVH.......... Human-made structures **[NVH]**

1c. Problematic communities and signatures.......... **Unknown Bcode Group**

Unknown Vegetation Bcode Group, UN
Unclassified combinations of species.......... Unclassified **[X.XUNCL]**
Uninterpretable signatures.......... Uninterpretable **[X.XUNK]**

APPENDIX 9-2A

Descriptions and discussions of linkage with the pierce et al. and Milleson et al. categories. Codes in parentheses are Pierce et al. (1982) vegetation codes. Milleson et al. definitions are from the legend of the Milleson et al. (1980) Pool C plant communities map.

Forested Communities

Upland Forest Communities

Oak/Cabbage Palm (OK). Milleson et al. category Oak and Cabbage Palm. No linkage issues with our Quercus virginiana (-*Sabal palmetto*) forest (F.QS) community type. This category is also linked with our *Sabal palmetto* forest (F.SP) community type. Milleson et al. definition: "Terrestrial hammocks dominated by water oak (*Quercus nigra*), live oak (*Quercus virginiana*), or cabbage palm (*Sabal palmetto*). Understory vegetation is usually limited and consists of saw palmetto (*Serenoa repens*), wild berry (*Rubus cuneifolius*), and greenbrier (*Smilax* sp.)."

Pine Forest (PP). Pierce et al. category PP. Milleson et al. do not mention any *Pinus* spp. in any of their categories. No linkage issues of Pierce et al. with our *Pinus elliottii* forest (F.PE) community type.

Wetland Forest Communities

Cypress forest (CY). Milleson et al. category Cypress. Milleson et al. definition: "Elongate strands of bald cypress (*Taxodium distichum*) located throughout the floodplain and many tributaries. A few associated trees include pop ash (*Fraxinus caroliniana*) and buttonbush (*Cephalanthus occidentalis*). Epiphytes may be abundant, and water hyacinth (*Pistia stratiotes*) is occasionally profuse." No linkage problems with our *Taxodium distichum* forest (F.TDF) community type.

Wetland hardwood forest (MP). The category called Wetland Hardwood in the Pierce et al. map was described as forested wetland communities with mixtures of *Taxodium distichum* and/or *Quercus virginiana* and *Sabal palmetto*. However, the type is given the code "MP" (Pierce et al. 1982:5). We have assumed that the definition given in Pierce et al. (1982) is in error and that MP was intended as an abbreviation for "maple."

Milleson et al.'s definition of their Hardwood Trees category is: "Heads or strands of swamp hardwood trees. Major species include red maple (*Acer rubrum*), pop ash (*Fraxinus caroliniana*), and tupelo (*Nyssa silvatica*)." Linked with our *Acer rubrum* (-*Nyssa silvatica*) forest (F.AR) and *Fraxinus caroliniana* forest (F.FC) community types.

Shrub-Dominated Communities

Upland Shrub Communities

Palmetto Prairie (PM). Pierce et al. describe these communities as "extensive stands of dense, impenetrable palmetto" within their Native Uplands upper category. Milleson et al. did not define this type; Pierce et al.'s Palmetto Prairie apparently had converted to Oak/Cabbage Palm by the time of Milleson et al.'s classification (Pierce et al. 1982:11). Pierce's category is linked with our *Serenoa repens* shrubland (S.SR) community type.

Woody shrub (WD). Not defined clearly by Pierce et al., but defined in Milleson et al. (category Woody Shrub) as communities dominated by *Baccharis halimifolia* and *Sambucus simpsonii*; other species present may include *Psidium guajava*, *Ilex cassine*, *Salix caroliniana*, and *Schinus terebinthifolius*. In Pierce et al., WD includes upland waxmyrtle communities (Pierce et al. 1982:8, 12). *Schinus* was not mentioned in the Pierce et al. species list. According to Milleson et al., *Baccharis* and *Sambucus* were most common in the northern valley, while *Schinus* is a dominant in the lower valley, especially in Pools D and E. Milleson et al. point out that this community occurs primarily on drained soils, although it is also found in transition areas. Milleson et al. definition.: "A community which occupies poorly drained soils and is dominated by several shrubby species. The most frequently encountered shrubs are saltbush (*Baccaris halimifolia*), elderberry (*Sambucus simpsonii*), wax myrtle (*Myrica cerifera*), and guava (*Psidium guajava*)." The Pierce et al. WD category is currently linked with our Miscellaneous upland shrubland (S.MxUS) and *Psidium guajava* shrubland (S.PG) community types. We have not linked our *Schinus terebinthifolius* shrubland (S.ST) community type with either previous classification's Woody

Shrub category because *Schinus* is not mentioned as a dominant in their descriptions. In doing so we assume that communities approximating our *Schinus terebinthifolius* shrubland (S.ST) community type did not occur on the floodplain at the time of Pierce et al.'s photography.

Wax Myrtle (no Pierce et al. category). *Myrica cerifera* communities were included by Pierce et al. in their Woody Shrub category (WD, see above) in their Native Uplands upper category. Milleson et al.'s Waxmyrtle category was included in their Terrestrial (upland) Forested upper category. We have defined our *Myrica cerifera* shrubland (S.MC) as an upland or upland-transitional type. Milleson et al. definition: "Uniform dense to sparse stands of wax myrtle (*Myrica cerifera*) shrubs. Understory vegetation is variable and may contain torpedo grass (*Panicum repens*), meadow beauty (*Rhexia* sp.), and dogfennel (*Eupatorium* sp.). Climbing vines such as muscadine grape (*Vitis rotundifolia*) and white vine (*Sarcostemma clausa*) are common." Milleson et al. listed two forms: (a) mature stands on well-drained riverbank sites with Vitis rotundifolia, *Ipomea* sp., *Smilax* sp., and *Sarcostemma clausa*; and (b) immature or stunted stands, occasionally flooded (2–3 in water depth), with diverse understories (*Centella asiatica*, *Hydrocotyle umbellata*, *Panicum repens*, *Lippia nodiflora*, *Alternanthera philoxeroides*, *Eclipta alba*, *Rhexia* sp., *Paspalum notatum*, *Sesbania exaltata*, *Juncus effusus*, and *Eupatorium* sp.). We assume zero distribution of our *Myrica cerifera* shrubland (S.MC) community type at the time of the Pierce et al. photographs; however, our S.MC is linked with Milleson et al.'s Waxmyrtle category.

Milleson et al. noted that waxmyrtle had been observed on "floating tussocks" of *S. cubensis*, a type that we have placed in a separate wetland category, *Myrica cerifera* floating mat shrubland (S.MCF); however, neither authors formally defined a floating mat *Myrica* type.

Wetland Shrub Communities

Buttonbush (BB). Milleson et al. Buttonbush category (in their Marsh upper category). Both authors are clear that the Buttonbush type is characterized by dominance of buttonbush (Pierce et al. 1982:14, Milleson et al. 1980:22). Understory species include *Pontederia cordata*, *Sagittaria lancifolia*, and *Panicum hemitomon*.

Milleson et al. definition: "Dense stands of buttonbush (*Cephalanthus occidentalis*) shrubs with associated vegetation consisting of pickerelweed (*Pontederia lanceolata*), arrowhead (*Sagittaria lancifolia*), and maidencane (*Panicum hemitomon*)." We have defined three categories of Cephalanthus community types: *Cephalanthus occidentalis* shrubland (S.CO), *Pontederia cordata-Sagittaria lancifolia-Cephalanthus occidentalis* shrubland (S.PS-CO), and *Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon-Cephalanthus occidentalis* shrubland (S.PS-PH-CO), all of which are linked with the Buttonbush categories of the previous classifications.

Primrose willow (no Pierce et al. category). This type was not defined in Pierce et al. because they did not encounter it (previously defined in Milleson) in their 1950s photography (Pierce et al. 1982:8). Milleson et al. describe their Primrose Willow type as commonly occurring in stabilized, continuously inundated conditions, especially in the southern portions of impoundment pools. Milleson et al. estimated only 1.8% of the floodplain and 3.4% of Pool C in this type. Milleson et al. definition: "Emergent broadleaf marsh communities which have been invaded and dominated by primrose willow (*Ludwigia peruviana*) and water primrose (*Ludwigia leptocarpa*)." We assume zero distribution of our *Ludwigia* spp. shrubland (S.LS) community type at the time of Pierce et al.'s mapping; the Milleson et al. category is linked with S.LS.

St. John's wort (SJ). Milleson et al. category St. Johns Wort. Milleson et al. found only 0.1% of the floodplain in this type, all in Pools A and B. Milleson et al. definition: "Circular, sandy, upland ponds dominated by a small woody shrub, St. John's Wort (*Hypericum fasciculatum*). Other emergent species, such as spikerush (*Eleocharis* spp.) and yellow-eyed grass (*Xyris* sp.), are usually present." Linked with our *Hypericum fasciculatum* shrubland (S.HF) community type.

Willow (WI). Milleson et al.'s area estimates (their Table 1) subdivide their Willow category into Willows in Floodplain Areas and Willows in Spoil Areas, although their map does not separate these types. Pierce et al. lump these two categories as category WI. Milleson et al. definition: "Willow (*Salix caroliniana*) heads scattered throughout marshes and in spoil retention areas. Associated understory plants include common marsh species, such as pickerelweed (*Pontederia lanceolata*) and arrowhead (*Sagittaria lancifolia*)." Linked with our *Salix caroliniana* shrubland (S.SC) community type.

Herbaceous Communities

Upland Herbaceous Communities

Improved pasture (PI). Milleson et al.'s category is Improved Pasture. Most abundant species is *Paspalum notatum*; others listed by Milleson et al. include *Panicum repens, Juncus effusus*, and *Glottidium vesicaria*.

Milleson et al. definition: "Land specifically managed to provide forage for livestock. Bahia grass (*Paspalum notatum*) is a dominant grass and other common species include torpedo grass (*Panicum repens*), bladderpod (*Glottidium vesicaria*), and small sedges." Linked with our *Paspalum notatum* herbaceous vegetation (H.PN) and *Axonopus fissifolius* herbaceous vegetation (H.AF) community types, both dominated by introduced pasture species.

Switchgrass (SW). Milleson category Switchgrass. Links with our *Spartina bakeri* herbaceous vegetation (SB) type. Milleson et al. definition: "Dominated by large tufts of switchgrass (*Spartina bakeri*). Understory plants include coinwort (*Centella asiatica*), pennywort (*Hydrocotyle umbellata*), and water hyssop (*Bacopa monnieri*)." Linked with our *Spartina bakeri* herbaceous vegetation (H.SB) type.

Unimproved pasture (PU). Pierce et al. used the code PU for any upland, herbaceous-dominated community not apparently altered by cultivation. Milleson et al. describe Unimproved Pasture as "native rangeland" that is "typified by a ground cover of grasses, sedges, and small herbs, with low shrubs ... which is subjected to grazing by range cattle ... ." Species (from Milleson et al.) may include *Lindernia anagallidea* (not on our species list), *Centella asiatica, Panicum repens, Bacopa caroliniana, Juncus effusus, Hydrocotyle umbellata*, and *Carex* spp. At higher elevations, can be dominated by *Serenoa repens* and terrestrial grasses (Milleson et al. 1980).

Milleson et al. define Unimproved Pasture as: "Terrestrial land which provides grazing for range cattle. Vegetation consists of scattered small shrubs, wax myrtle (*Myrica cerifera*), or saw palmetto (*Serenoa repens*), with a variety of secondary and ground cover species including broomsedge (*Andropogon* spp.), dogfennel (*Eupatorium* spp.), coinwort (*Centella asiatica*), and torpedo grass (*Panicum repens*)." The type is important because of its substantial post-channelization representation (10.5% of the floodplain in Milleson et al.'s map). Linked with our Miscellaneous native herbaceous vegetation (H.MxN) and Miscellaneous invasive herbaceous vegetation (H.MxW) community types, both of which are upland types dominated by native species.

Wetland Herbaceous Communities

Broadleaf marsh (PS). We have few linkage difficulties with the Pierce et al. Broadleaf Marsh (PS) and Milleson et al. Broadleaf Marsh categories, although we have defined intermediate community types to encompass gradient vegetation: *Pontederia cordata-Sagittaria lancifolia* herbaceous vegetation (H.PS), *Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon* herbaceous vegetation (H.PS-PH), *Hibiscus grandiflorus-Pontederia cordata-Sagittaria lancifolia* herbaceous vegetation (H.PS-HG), *Pontederia cordata-Sagittaria lancifolia-Panicum hemitomon-Cephalanthus occidentalis* herbaceous vegetation (H.PS-PH-CO), and *Pontederia cordata-Sagittaria lancifolia-Cephalanthus occidentalis* herbaceous vegetation (H.PS-CO). Pierce et al. stated that their communities are heterogeneous and that few species were recognizable from their air photos; they gave a list of species adapted to deep marshes with prolonged inundation (*Nuphar lutea*, *Hydrochloa caroliniensis*, *Pontederia cordata*, *Sagittaria lancifolia*, *Thalia geniculata*, *Panicum hemitomon*, *Cephalanthus occidentalis*, *Polygonum punctatum*, and *Scirpus* spp.). Milleson et al. definition: "Primarily herbaceous, emergent marsh communities characterized by pickerelweed (*Pontederia lanceolata*) and arrowhead (*Sagittaria lancifolia*). Numerous other aquatic species which may be locally abundant include cattail (*Typha latifolia*), bulrush (*Scirpus* spp.), smartweed (*Polygonum* sp.), arrowroot (*Thalia geniculata*), swamp hibiscus (*Hibiscus grandiflorus*), and spatterdock (*Nuphar lutea*)." In addition to the community types listed above, we have linked our *Hibiscus grandiflorus* herbaceous vegetation (H.HG) with the Pierce et al. and Milleson et al. Broadleaf Marsh categories.

Juncus effusus. (no Pierce et al. category). Pierce et al. did not find either of the *Juncus* types we have defined, *Juncus effusus* herbaceous vegetation (upland depressions) (H.Jed) and *Juncus effusus* herbaceous vegetation (wet prairies) (H.Jep), on the 1950s photos (Pierce et al. 1982:8), aside from a small area in Paradise Run. They also do not mention *Juncus effusus* in their species list. Milleson et al. (1980:25) describe two types: (a) Soft Rush Ponds: depression areas in improved and unimproved pastures forming an “outer ring” around the Broadleaf Marsh zone (Soft Rush Pond category), and (b) a type for which they do not provide a discrete classification category, which they describe as occurring in stabilized Kissimmee River pools; in this type, they say, “*J. effusus* may be dense; understory consists of *Hydrochloa caroliniensis*, *Lindernia anagallidea*, *Centella asiatica*, *Hydrocotyle umbellata*, *Bacopa caroliniana*, *Dicromena colorata*, and *Rhexia* spp.” Milleson et al. definition: “Communities characterized by moderate density of soft rush (*Juncus effusus*). Associated species include low-growing herbaceous plants such as false pimpernel (*Lindernia anagallidea*), coinwort (*Centella asiatica*), and aromatic figwort (*Bacopa caroliniana*).”

Sawgrass (CL). Milleson et al. Sawgrass category. Milleson et al. found only 0.2% of the floodplain in this type, all in Pool B. Milleson et al. definition: “Consists of sawgrass (*Cladium jamaicense*) in dense, circular stands among wet prairie plant communities. Other species associated with sawgrass are marsh hibiscus (*Hibiscus grandiflorus*) and arrowhead (*Sagittaria lancifolia*).” According to Milleson et al., associated species may also include *Cephalanthus occidentalis*, *Ludwigia peruviana*, and ferns. However, we have found *Cladium jamaicense* (sawgrass) communities to be virtually monospecific. Linked with our *Cladium jamaicense* herbaceous vegetation (H.CJ) community type.

Wet prairie (WP, MC, RH, TG). Our classification includes two community types that link directly with Pierce et al.’s codes MC (*Panicum hemitomon* wet prairie) and RH (*Rhynchospora* spp. wet prairie). Our comparable community types are *Panicum hemitomon*-(*Pontederia cordata*) herbaceous vegetation (H.PH) and *Rhynchospora* species herbaceous vegetation (H.RN), respectively. We assume both were delineated by Pierce et al. using dominance of the respective namesake species. We also consider the Pierce et al. code TG (mixed aquatic grasses or *Panicum repens* vegetation) as a Wet Prairie type (Pierce et al. 1982:18) and have defined *Panicum repens* herbaceous vegetation (H.PR) to encompass this kind of vegetation. Pierce et al. additionally used a generic, undefined Wet Prairie designation, WP. Pierce et al. admitted to problems of subjectivity in making distinctions among these types from air photographs. Milleson et al. also used Maidencane, *Rhynchospora*, and Aquatic Grasses (primarily *Panicum repens*-dominated) wet prairie categories; however, they did not have a generic Wet Prairie category. Pierce et al. list *Rhynchospora colorata, Scleria spp., Sagittaria lancifolia, Pontederia lanceolata, Hydrocotyle umbellata, Bacopa* spp., *Fuirena scirpoides, Psilocarya nitens,* and *Leersia hexandra* as common components of wet prairies.

Several additional types are considered wet prairie types by Kissimmee staff, included under our Other Wet Prairies Subgroup within the Wet Prairie Group. Note there is no consensus in the literature as to a formal meaning of the term “wet prairie”, and the term is used in south Florida for various types of vegetation. For our purposes, all of these “other wet prairies” are assumed to be linked with the Pierce et al. Wet Prairie category. However, because there is no comparable generic wet prairie category in Milleson et al. with which they can be linked; most of the Other Wet Prairies Subgroup community types remain unlinked with the Milleson et al. classification. Exceptions are *Leersia hexandra* herbaceous vegetation (H.LH), which is linked with the Milleson et al. Aquatic Grasses category (because they list *L. hexandra* as a possible component of this type), and our two *Juncus* wet prairie community types (H.JEp and H.JEd, see *Juncus effusus* types, above).

Floating Mat (FM) and Floating Tussock (TS). Linkage with these types has been difficult in part because of use of the term “floating mat” to mean something distinct from Pierce’s original definition. Based on 1982 reconnaissance of signatures similar to those in their 1950s photography, Pierce et al. assumed for their FM type the presence of some combination of Eichhornia crassipes, Pistia stratiotes, *Hydrocotyle umbellata*, *Sacciolepis striata*, *Azolla caroliniana*, *Lemna* spp., *Scirpus cubensis*, and *Hydrochloa caroliniensis* in abandoned channels and backwaters. They do not have categories for discontinuous communities of these species.

The distinction in Pierce et al. between FM and Floating Tussock (TS) is that the latter is a mat that has been invaded by marsh hydrophytes with a species composition similar to broadleaf marsh. They admit to difficulties distinguishing this type from broadleaf marsh. Milleson et al.’s definition of Floating Tussock appears to include the Pierce et al. FM type but also includes *Ludwigia* spp., *Typha* spp., *Salix* spp., and apparently *Myrica cerifera* (Milleson et al. 1980:15). FM is linked with our community types associated with *Pistia stratiotes*, *Eichhornia crassipes*, and *Scirpus cubensis* . TS is not linked with our classification.

Our *Myrica cerifera* floating mat shrubland (S.MCF) is apparently successional; comparable vegetation was not recorded by Pierce et al. Although Milleson et al. (1980:15) mention observations of wax myrtle growing on floating tussocks of *Scirpus cubensis*, neither authors define categories that include floating mats dominated by *Myrica*. It is possible that any occurrences approximating this type in the