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Presiones antropogénicas en la cuenca baja del río Grijalva en Tabasco, México

Tesis

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Por

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para obtener el grado de **Maestra en Ciencias en Recursos Naturales y Desarrollo Rural**

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Resumen

La cuenca del río Grijalva ha sido sometida a diversas presiones antropogénicas como son la construcción de cuatro presas hidroeléctricas sobre el cauce del río y el enriquecimiento de nutrientes debido a la agricultura, ganadería y urbanización. Estas presiones alteran la estructura y función de los ríos, lo cual pone en riesgo la integridad ecológica de estos ecosistemas y por ende los servicios ambientales que ofrecen. Por ello, el objetivo de esta tesis consistió en evaluar los efectos de las presiones antropogénicas en ríos de la cuenca baja del Grijalva, centrándose principalmente en la alteración del flujo por la construcción de represas y la contaminación de nutrientes por cambios en el uso de la tierra. Se compararon variables hidrológicas entre el río Carrizal (represado) y el río de la Sierra (sin represar) utilizando el Índice de Alteración Hidrológica (IHA) a fin de determinar los efectos sobre el caudal debido a la construcción y operación de las presas. Empleando datos históricos, se analizaron los patrones temporales en la concentración y transporte de nitratos (NO_3) y fósforo total (PT). Asimismo, se compararon las concentraciones y el transporte del año 2016, de las diferentes formas de nitrógeno (N) y fósforo (P) entre tributarios de la cuenca baja del Grijalva y entre las temporadas de secas y lluvias. Se encontró que la construcción de represas en el Grijalva, modificó el régimen del caudal del río, afectando patrones estacionales así como diversas características hidrológicas. Se obtuvo una tendencia significativa al incremento de NO_3 en tres de los cuatro sitios, mientras que en el caso de PT la tendencia es decreciente en dos de los cuatro sitios. Los resultados indican que los valores de las concentraciones están influenciados por el uso de suelo aguas arriba de los sitios muestreados y por la temporalidad, registrándose las mayores concentraciones de N y P en temporada de lluvias. Las mediciones y resultados presentados en esta tesis podrían ayudar a la mejor estimación de los servicios ecosistémicos y a evaluar los impactos negativos que tienen la construcción de presas y la contaminación por nutrientes, viéndose afectadas fuentes de agua potable, pesquerías, así como la zona costera.

Palabras claves: fósforo, nitrógeno, presas hidroeléctricas, régimen hidrológico, uso de suelo.

Capítulo 1. Introducción

Los ríos son de vital importancia para el ser humano, ya que proporcionan diversos servicios ambientales, tales como: agua para el consumo humano, disposición de desechos, energía eléctrica, mantenimiento del gradiente de salinización, fuente de alimentación y recreación, control de inundaciones, transporte de nutrientes y como medio de transporte (Baron et al. 2003; Garrido et al. 2010). Además, alrededor del 60% del agua que se utiliza en los campos agrícolas proviene de las aguas superficiales (Parker y Oates 2016). Se calcula que los servicios ecológicos que ofrecen estos ecosistemas se valoran en cerca de 6.5 billones de dólares al año (Costanza et al. 1997). Sin embargo, el aprovechamiento de los servicios ecosistémicos obtenidos de los ríos pueden generar graves impactos a estos ecosistemas como la degradación de la calidad de agua, pérdida de biodiversidad, alteración del régimen hidrológico y alteración de la geomorfología (Garrido et al. 2010).

Los ríos albergan 9.5% de las especies conocidas a nivel mundial y se ha encontrado que la densidad de especies en estos ecosistemas son mayores en comparación con los ecosistemas terrestre y marino (Postel y Richter 2010; Strayer y Dudgeon 2010). Sin embargo, en los últimos años la biodiversidad de los ríos ha disminuido a causa de las actividades antropogénicas que se realizan en sus cuencas (Poff 2014), como los cambios de uso de suelo para la agricultura, ganadería y urbanización. La construcción de presas y los cambios en el uso del suelo se consideran las mayores presiones antropogénicas a las que se ven sometidos los ecosistemas fluviales (Maavara et al. 2015).

1.1 Presiones antropogénicas por la construcción de presas

Alrededor del 60% de los ríos más grandes del mundo se encuentran obstruidos por presas u otras construcciones civiles (Van Cappellen y Maavara 2016). Se calcula que en el mundo existe alrededor de 40,000 presas mayores a 15 m de altura y conforme pasen los años este número irá en aumento, en especial en los países en vías de desarrollo (Poff 2014). De acuerdo con la Comisión Nacional del Agua (2010), en

México existen 667 presas catalogadas como grandes presas ya que tienen una altura igual o mayor a 15 m.

La construcción de presas sobre los cauces de los ríos provoca múltiples cambios como alteración del régimen hidrológico, retención de sedimentos, pérdida de biodiversidad, fragmentación de los ríos, inundaciones, cambios en el transporte de nutrientes, entre otros (Anderson et al. 2006; Syvitski y Kettner 2011; Al-Faraj y Scholz 2014). Estos a su vez, traen otras series de problemas que se extienden hasta las costas como el hundimiento del delta de los ríos, el aumento de la tasa de erosión y el retroceso de la línea costera (Syvitski y Kettner 2011). Ya que todo funciona como un gran sistema, la fragmentación de los ríos por las presas afecta gravemente todo lo que se encuentre aguas abajo de estas construcciones. No obstante, los impactos de la construcción de presas pueden mitigarse aplicando estrategias de manejo integral en toda la cuenca, ya que todas las actividades realizadas en las cuencas afectan la integridad ecológica de los ríos (Cotler y Caire 2009). También se puede modificar el manejo y la operatividad de las presas para restaurar el caudal estacional y así garantizar el caudal ecológico (Garrido et al. 2010; Kennedy et al. 2016; Poff et al. 2016).

1.2 Presiones antropogénicas a causa del cambio de uso de suelo

Así como las presas, el cambio de uso de suelo trae consigo consecuencias ambientales en los ríos, debido a la conectividad que existen entre los ecosistemas terrestres y acuáticos (Lambin et al. 2003). Los cambios de uso de suelo están asociados principalmente al crecimiento de la población y por lo tanto a un incremento en la demandan de alimentos (agricultura y ganadería), así como de espacios para establecerse (urbanización) (Balvanera 2012). El cambio de uso de suelo está vinculado, además, a amenazas de contaminación y modificación del caudal, las cuales repercuten sobre los ecosistemas acuáticos provocando alteraciones en la calidad del agua, pérdida de la biodiversidad y con ello de los servicios ambientales que nos proveen los ríos. Por otra parte, el cambio de uso de suelo es acompañado generalmente del incremento en las concentraciones de nutrientes, influyendo en los procesos físicos, químicos y biológicos que ocurren en los ecosistemas fluviales (Dourojeanni y Jouravlev 1999).

La urbanización afecta a los ríos modificando su hidrología, conectividad, hábitat y la calidad del agua (Ramírez et al. 2014). En la actualidad las áreas urbanas han aumentado en todo el mundo, ejemplo de esto es que en el año 1900 sólo se tenían cuatro ciudades con más de 2,000,000 de habitantes, mientras que ahora se cuentan con más de 100 ciudades que superan los 10,000,000 de habitantes, por lo que las áreas donde se ubican estas ciudades también han crecido (Dourojeanni y Jouravlev 1999; Latham 2002; Departamento de Asuntos Económicos y Sociales de la Secretaria de las Naciones Unidas 2014). Las ciudades requieren un gran número de servicios (agua para el consumo humano, irrigación, disposición de desechos, etc.) (Baron et al. 2003), y por lo tanto tienen mayores impactos en los ríos y otros cuerpos de agua. Uno de los principales impactos es el enriquecimiento de nutrientes a causa de las descargas de aguas residuales, lo cual presenta una gran amenaza a la integridad ecológica de estos ecosistemas (Figueroa-Nieves et al. 2014).

Aproximadamente, cerca del 40% de la superficie mundial se encuentra destinada a la agricultura (Park 2015), la cual se ha expandido en lo que antes eran bosques, sabanas y estepas a causa del crecimiento de la población y una mayor demanda de alimentos, fibras y biocombustibles (Lambin et al. 2003; Laurance et al. 2014). Después de los años cincuenta, el crecimiento de la agricultura fue en aumento y se fue intensificando cada vez más, empleándose fertilizantes y agroquímicos para el control de las plagas y malezas, para obtener una mayor producción de los cultivos (Park 2015). En las zonas tropicales la expansión agrícola está creciendo rápidamente a causa de la demanda de alimentos y de biocombustibles, por lo cual los hábitats terrestres y acuáticos se verán gravemente afectados (Laurance et al. 2014). De acuerdo con Lin et al. (2015) se espera que los pastizales y zonas de conservación se conviertan en campos agrícolas a causa de la demanda de biocombustibles, la cual se prevé que para algunos cultivos como el de la palma de aceite incrementará hasta en un 100% para el año 2050 (Park 2015). Laurance et al. (2014) pronostica que los efectos de la expansión de la agricultura en las zonas tropicales traerá graves consecuencias a los ríos, como la alteración del caudal, disminución de oxígeno, aumento de la temperatura, incremento de nutrientes, sedimentos y contaminantes, y por lo tanto se alterarán los servicios ambientales que brindan estos cuerpos de agua.

Como ya se ha mencionado el aumento de áreas urbanas y agricultura trae como consecuencia un aumento en las concentraciones de nutrientes. Los principales nutrientes en los ecosistemas acuáticos son el N y P ya que generalmente son considerados el factor limitante en la producción primaria y su exceso puede provocar eutrofización, afectando severamente a los ecosistemas (Schlesinger y Bernhardt 2013; Khatri y Tyagi 2014). La eutrofización altera la estructura y función de los ecosistemas acuáticos, derivando en la degradación de la calidad del agua y en la alteración de las comunidades biológicas, a causa del crecimiento acelerado de las algas bentónicas y fitoplantónicas (Dodds 2006). Esto lleva a una disminución de oxígeno disuelto generando zonas de hipoxia (<2 mg/L) (Vaquer-Sunyer y Duarte 2008) y anoxia (≤0.5 mg/L) (Chan et al. 2008). Algunas repercusiones que trae consigo la hipoxia son la pérdida de hábitat para peces y fauna bentónica, mortandad de organismos, disminución de los recursos alimentarios y la alteración de la trama trófica (Rabalais et al. 2014). Otros cambios relacionados con la hipoxia se refieren a alteraciones en la composición del fitoplancton, así como de las funciones del ecosistema (p. ej. el ciclaje de nutrientes), y que puede ocurrir en arroyos, ríos, lagos, lagunas y en la zona costera (Rabalais et al. 2014). Al ser degradados estos ecosistemas, los servicios ecológicos se ven afectados y traen consecuencias directas a la sociedad entre los que destacan la degradación de la calidad de agua y el consecuente aumento en los costos de tratamiento, enfermedades, pérdida de lugares de recreación, pérdida de especies comerciales y de fuentes de alimento (Meyer et al. 2005). Las causas de la eutrofización y la hipoxia están relacionadas con los aportes de nutrientes de las actividades agropecuarias en la cuenca así como con las descargas de aguas residuales de centros urbanos, ya que los ecosistemas acuáticos están directamente relacionados con los procesos que ocurren en la cuenca (Cotler y Caire 2009).

El control de la entrada y salida de nutrientes hacia los cuerpos de agua es fundamental para evitar la degradación de los ecosistemas. Los aportes provenientes de cultivos y actividades ganaderas pueden reducirse a través del análisis de los suelos para determinar los requerimientos necesarios para los cultivos, pero también se podrían reforestar las zonas ribereñas para disminuir la entrada de nutrientes en los sistemas (Altieri 1994; Biggs et al. 2000; Huerta et al. 2014). Por otra parte, los impactos de los centros urbanos sobre los cuerpos de agua y específicamente sobre los niveles de nutrientes pueden disminuirse a través del tratamiento de aguas residuales (Meyer et al. 2005; Alianza por el Agua 2008; Yee-Batista 2013). Sin embargo, en muchas ocasiones la capacidad instalada de las plantas de tratamiento no es el adecuado para el área en que se localizan, en algunos casos la capacidad se ve superada por el rápido crecimiento de las ciudades, y en otros casos en lo largo de su vida útil no reciben el mantenimiento y manejo adecuado lo cual también compromete su funcionamiento (Mays 2009). En Latinoamérica alrededor del 30% de las aguas de alcantarillado de las ciudades reciben tratamiento, es decir que 70% de las aguas residuales domésticas e industriales son vertidas en cuerpos de agua sin previo tratamiento lo cual es un riesgo para la salud humana, así como para la capacidad natural de asimilación y dilución de contaminantes, afectando por tanto el sistema receptor (Ramírez y Espejel 2001; Yee-Batista 2013). El porcentaje de aguas residuales municipales que son tratadas en México es del 50.2% de las aguas colectadas (Comisión Nacional del Agua 2014), por lo que muchos cuerpos de agua reciben aguas residuales sin tratar.

1.3 La cuenca baja del río Grijalva y sus presiones antropogénicas

La cuenca del río Grijalva es una de las áreas más importantes del país debido a su caudal hídrico, a su extensión y porque en esta zona se localizan las áreas más lluviosas del país (Plascencia-Vargas et al. 2015). Tiene un escurrimiento medio anual de 45,842 hm^3 (Toledo y Navarro 2011). El río Grijalva nace en Guatemala y recorre los estados de Chiapas y Tabasco antes de unirse con el río Usumacinta para desembocar en el Golfo de México (Plascencia-Vargas et al. 2015). Aguas abajo de la unión del río Grijalva con el Usumacinta se localiza una de las regiones de humedales más importantes de Mesoamérica, ya que su productividad primaria neta se encuentra entre las 75 ton/ha/año (Toledo 2003). En general, la cuenca del río Grijalva es una zona con alta biodiversidad, en esta área habitan alrededor del 20% de las especies de vertebrados terrestres conocidas en México, y el 36% de las aves endémicas del país (Toledo 2003).

Sin embargo, debido a su paso por poblaciones, zonas ganaderas, agrícolas y de desarrollo industrial, el Grijalva se considera uno de los ríos más contaminados, a causa de los contaminantes que son aportados por dichas áreas (Toledo 2003). De acuerdo con De la Peña et al. (2013), el río Grijalva se encuentra clasificado como fuertemente contaminado dado sus valores de demanda bioquímica de oxígeno (DBO_5) demanda química de oxígeno (DQO) y sólidos suspendidos totales (SST). Además, Borbolla-Sala et al. (2005) encontró que los indicadores bacterianos (coliformes fecales y totales, *vibrio cholerae*) de las aguas de lagunas, ríos, agua potable y aguas negras del estado de Tabasco superaron los límites máximos permisibles establecidos por la norma mexicana (NOM-127-SSA1-1994), lo cual puede traer consecuencias graves en la salud de los habitantes. Esto se debe principalmente a la descarga de aguas residuales a los cuerpos de agua ya que en los estados de Chiapas y Tabasco solo se trata alrededor del 22% de las aguas residuales municipales (De la Peña et al. 2013), aunado a la presencia de grandes centros urbanos como Tuxtla Gutiérrez y Villahermosa. Además de la amenaza por descarga de contaminantes, el río Grijalva se encuentra dentro de la categoría de alta alteración ecohidrológica (Garrido et al. 2010), ya que sobre su cauce se encuentra el mayor desarrollo hidroeléctrico del país, el cual representa el 41.6% (CFE 2014) de la capacidad hidroeléctrica total en operación y está integrado por cuatro grandes presas: Netzahualcóyotl “Malpaso” (construida entre 1955-1969), Belisario Domínguez “Angostura” (construida entre 1969-1974), Manuel Moreno Torres “Chicoasén” (construida entre 1974-1980) y Ángel Albino Corzo “Peñitas” (construida entre 1979-1987) (Díaz Perera, 2015).

Las presiones en la cuenca baja del río Grijalva producto de la operación de cuatro presas hidroeléctricas y de los cambios de uso de suelo, pueden estar afectando la hidrología y la calidad de agua, que a su vez puede alterar los servicios ambientales del río Grijalva, como las pesquerías y las fuentes de agua para la población. Además estas presiones también pueden afectar las zonas costeras cercanas a la desembocadura. Por ejemplo, en la desembocadura del Grijalva-Usumacinta, Signoret et al. (2006), encontraron valores menores a 1.75 mg/l de oxígeno disuelto, indicando la presencia de una zona de hipoxia, la cual puede estar relacionada con las descargas de nutrientes de los ríos.

Considerando lo anterior, este estudio tiene como finalidad evaluar las alteraciones al caudal y a los niveles de nutrientes producto de las presiones antropogénicas, específicamente la construcción de presas y el cambio de uso de suelo en la cuenca del río Grijalva. Los resultados generados por este estudio pretenden ser una contribución para entender los cambios temporales y espaciales en los nutrientes, sus fuentes y así apoyar en el diseño de medidas de control de contaminantes. Además que servirá para futuros estudios relacionados con el enriquecimiento de nutrientes, calidad de agua, y alteración hidrológica en la región.

Pregunta de investigación

¿Cómo las presiones antropogénicas han afectado el caudal y las concentraciones de nutrientes en los ríos de la cuenca baja del río Grijalva?

Hipótesis

1. La construcción de presas sobre el río Grijalva altera la variabilidad temporal del caudal.
2. Las concentraciones y el transporte de los nutrientes en la cuenca baja del río Grijalva aumentan a través del tiempo producto de los cambios en el uso de suelo.
3. Aguas abajo de la ciudad de Villahermosa se encontrarán las concentraciones más altas de N y P.

Objetivo general

Evaluar los efectos de las presiones antropogénicas en la cuenca baja del Grijalva, centrándose principalmente en la alteración del caudal y la contaminación de nutrientes.

Objetivos específicos

- Identificar las alteraciones del caudal por la construcción de presas en la cuenca baja del río Grijalva.
- Examinar los patrones temporales a través de los años en las concentraciones y transporte de nutrientes empleando datos históricos.
- Comparar las concentraciones y transporte actuales de nutrientes entre tributarios de la cuenca baja ubicados aguas arriba y aguas abajo de la ciudad de Villahermosa.
- Comparar las variaciones entre las temporadas de secas y lluvias en las concentraciones y transporte actuales de nutrientes en tributarios de la cuenca baja del río Grijalva.

Capítulo 2. Artículo: Temporal changes in the hydrological and chemical characteristics of a large tropical river: anthropogenic influence in the lower Grijalva River, Mexico

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ABSTRACT

1. Dam construction and nutrient loading are among the greatest threats to freshwater ecosystems, altering ecological processes and the provisioning of ecosystem services. Temporal change in hydrology and ambient chemical concentrations were studied on the Grijalva, a large tropical river in southern Mexico, where four hydroelectric dams operate and where land conversion from forest into agriculture, pasture, and urban areas has impacted the chemical environment.
2. Temporal changes in discharge and in river chemistry were examined by analyzing long-term discharge and nutrient data. Furthermore, additional water chemistry samples were collected in order to compare seasonal patterns in nutrient concentrations and nutrient loads among tributaries in the lower Grijalva.
3. Long-term discharge data indicated dam construction has severely altered seasonal patterns in discharge and other hydrological characteristics in regulated tributaries.
4. The watershed has also experienced significant temporal changes in water chemistry. Specifically, ambient nitrate concentration and the nitrate load carried by the river have increased through time, which may be attributed to the expansion of agricultural and urban areas in the watershed. Conversely, total phosphorus seemed to decline in the sites influenced by dam construction, suggesting dam operation may also influence water chemistry in the river.
5. In general, lower nutrient concentrations were collected upstream from the city of Villahermosa, suggesting that inputs from urban areas contributed to the nutrient loading. Additionally, higher nitrate and total phosphorus concentrations were detected in tributaries draining intensive agricultural and suburban areas.
6. Collectively, the results from the study suggest that dam construction and land conversion in the lower Grijalva have produced chemical and hydrological changes in the river, which may negatively impact small-scale and subsistence fisheries, drinking water sources, and the integrity of coastal zones on the Gulf of Mexico.

Keywords: river, monitoring, water quality, fish, hydropower, nutrient enrichment

INTRODUCTION

Globally, anthropogenic stressors, such as nutrient pollution and hydroelectric development, alter the structure and function of freshwater ecosystems. Habitat loss, channel fragmentation, biodiversity decline, changes in productivity and biogeochemical cycles, threatened human water security and a decline in fisheries are among the impacts caused by these stressors in freshwater ecosystems (Allan et al., 2005; Dudgeon et al., 2006; Gangloff, Edgar, & Wilson, 2016; Malmqvist & Rundle, 2002; Vörösmarty et al., 2010; Winemiller et al., 2016). Though many studies have examined these impacts on lotic ecosystems, there are still a relatively limited number of studies examining anthropogenic stressors in large, tropical rivers (Pelicice, Pompeu, & Agostinho, 2015; Pringle, Freeman, & Freeman, 2000). Land conversion for agriculture and urbanization and the construction of large dams currently threaten the ecological integrity of tropical rivers, their biodiversity and the provision of ecosystem services (Laurance, Sayer, & Cassman, 2014; Winemiller et al., 2016). Therefore, there is a great need to understand the influence of these activities on the quality and quantity of freshwater resources produced by tropical rivers.

Tropical rivers are ecologically unique. They are often characterized by a strong hydrological seasonality, which is driven by seasonally high rainfall and runoff (Boulton et al., 2008; Lewis, 2008), and are among the largest rivers in the world in terms of discharge. Tropical rivers are important sources of energy and nutrients for marine systems. They transport between 20-25% of the sediments that reach the oceans (Latrubesse, Stevaux, & Sinha, 2005). Rivers in the tropics are also frequently characterized by high biodiversity, and are home to many migratory species that rely on both freshwater and salt water environments (Anderson, Pringle, & Rojas, 2006; Pearson & Boyero, 2009; Winemiller et al., 2016). Additionally, many tropical rivers are located in lower-income economies and they provide important ecosystem services, such as food, drinking water, and employment in economically-marginalized human populations (Allan et al., 2005; Neiland & Béné, 2008). Hence, changes in hydrology, water quality and aquatic habitat may have particularly strong ecological and socioeconomic impacts in tropical watersheds.

Though the influence of dam construction and the conversion of large tracts of forest for agricultural and urban development may be declining in many higher-income countries in the temperate zone, this is not common in countries with emerging economies in the tropics (Laurance et al., 2014; Zarfl, Lumsdon, Berlekamp, Tydecks, & Tockner, 2015). For instance, considerable numbers of hydroelectric projects have been proposed in large tropical watersheds such as the Amazon, Congo and Mekong, threatening ecological processes, socioeconomic stability, and the conservation of biodiversity (Fearnside, 2016; Winemiller et al., 2016). Additionally, tropical forest conversion is predicted to intensify to support increasing global demands for biofuel, agriculture, and cattle (Laurance et al., 2014). Increasing urbanization in the tropics also presents a great threat to the ecological integrity of running waters. This is especially true when wastewater is discharged without proper treatment, which is common in lower-income economies. For example, in Latin America only approximately 20-30% of wastewater is treated prior to discharge (Ramírez, Rosas, Lugo, & Ramos-González, 2014; WWAP, 2017).

Monitoring the impacts of anthropogenic activities in lower-income economies presents an additional challenge as long-term environmental data are rarely available due to limited financial resources, inadequately trained personnel, and a lack of resources needed to conduct environmental analyses (Chouler & Di Lorenzo, 2015; Kostyla, Bain, Cronk, & Bartram, 2015).The purpose of this study was to assess the effects of anthropogenic pressures, including hydroelectric development and nutrient pollution, on a large tropical river. This work was conducted in the Grijalva River, which is part of the Grijalva-Usumacinta Watershed. The Grijalva-Usumacinta is the sixth largest river in Latin America and the longest river in Mesoamerica (Hamann & Ankersen, 1996). It flows through the northernmost tropical rainforest in the Western Hemisphere. The watershed produces 30% of Mexico's freshwater, and drains 42% of Guatemala (Hamann & Ankersen, 1996). The Grijalva-Usumacinta is also one of the most biologically diverse areas in Mesoamerica (Dickinson & Lawton, 2001). It contains approximately 112 fish species, 50% of which are thought to be endemic (Hamann & Ankersen, 1996). The watershed is a mosaic of disturbed and pristine habitats (Toledo, 2003). Natural resource extraction, including oil and forest products, agricultural development, and cattle ranching are an important part of the economy in this region (Manuel-Navarrete, Slocombe, & Mitchell, 2006) and are leading to the degradation of the watershed (Hamann & Ankersen, 1996). In addition to changes in land use, the ecological integrity of the river has also been compromised by the construction of four hydroelectric dams, which operate on the mainstem of the Grijalva (moving from upstream to downstream): Belisario Domínguez "Angostura" (built between 1969-1974), Manuel Moreno Torres "Chicoasén" (built between 1974-1980), Netzahualcóyotl "Malpaso" (built between 1955-1969), y Ángel Albino Corzo "Peñitas" (built between 1979-1987) (Díaz Perera, 2015). Research on the physicochemical condition of the river is relatively limited, and few studies have employed long-term datasets to examine changes in river hydrology and ecology (but see Muñoz-Salinas & Castillo (2015)).

METHODS

Study Area

The Grijalva headwaters are located in Guatemala and the river runs through the Mexican states of Chiapas and Tabasco, before joining the Usumacinta River to flow into the Gulf of Mexico. The watershed area of the Grijalva is 56,895 km^2, and approximately 10% is located in Guatemala and 90% in México. Average annual precipitation ranges from 1500-2000 mm near the coast to 4500 mm in the Northern Mountains of Chiapas (Plascencia-Vargas, González-Espinosa, Ramírez-Marcial, Álvarez-Solís, & Musálem-Castillejos, 2015). Average annual temperature varies from 12°C in the Highlands of Chiapas to 26°C in the lowlands of Tabasco. The Mexican portion of the watershed has a human population of 3,739,937 and includes large urban centers like Tuxtla Gutiérrez in Chiapas and Villahermosa in Tabasco.

Site selection

Study sites were selected based on the availability of historic discharge and water quality data from the National Water Commission (CONAGUA) of the Mexican Government (Table 1; Figure 1). Daily discharge data were obtained for sites 1, 3, 5 y 6 from ftp://ftp.conagua.gob.mx/Bandas/Bases_Datos_Bandas (Table 1). Water quality data was available for sites 1, 2, 5 and 7 and were requested from CONAGUA. These data corresponded primarily for the period 2000-2016 and showed generally a bimonthly frequency. Analysis was focused on total phosphorus and nitrate-N, because these variables showed the higher number of entries in the selected records. In addition, sites 1, 3 and 6 were sampled in 2016 to compare concentrations and loads of different forms of phosphorus and nitrogen between the dry and the wet season and among the study sites. The Pichucalco River (site 4), which drains the suburban areas located south of the city of Villahermosa, Tabasco, was also sampled.

Land Use

Drainage areas for the 2016 sampling sites were defined based on delineations of the Grijalva sub-basins from the National Institute for Statistics and Geography (INEGI). The resulting polygons were used to estimate land use and cover for the drainage area upstream from each sampling site (Table 2). This information was obtained from land use and vegetation maps (1:250,000) for year 2000 (series III) and 2011 (series V) from INEGI. Population data was obtained from http://www.conabio.gob.mx/informacion/gis/.

Nutrient sampling and load estimates

Monthly sampling was conducted from March through October 2016 at sites 1, 3, 4 and 6. At each site, three replicate samples were collected across the channel at 0.5 m of depth using a Van Dorn bottle and transported to the laboratory. Nitrite-N was determined by the colorimetric method, nitrate-N by cadmium reduction, ammonium-N by the phenate method, total nitrogen (TN) by persulfate digestion, soluble reactive phosphorus (SRP) by the ascorbic acid method and total phosphorus (TP) by persulfate digestion (APHA, 2012). TP and nitrate-N instantaneous loads (g/s) for sites 1, 2, 5 and 7 were estimated as the product of discharge and nutrient concentration from the data provided by CONAGUA. No discharge data were available for sites 2 and 7, so discharge records from upstream sites 1 and 6 were used, respectively. Load for 2016 sampling sites was estimated by combining the CONAGUA discharge measurements with the results of the concentration determinations for the nitrite-N, nitrate-N, ammonium-N, TN, SRP and TP. As there was not gauging station at site 4, discharge was estimated by subtracting the sum of the discharge from the upstream sites 1 and 3 from the discharge of the downstream site 6. Notably, these estimates include discharge from the Viejo Mezcalapa, hence discharge at site 4 may have been overestimated.

Statistical Analyses

The impact of dam construction on the lower Grijalva flow was assessed by comparing discharge patterns between site 1 (Carrizal River), located downstream from the hydroelectric dams and site 3 (De la Sierra River), an unregulated tributary of the Grijalva. By using the software Indicators of Hydrologic Alteration (The Nature Conservancy, 2009), 33 flow parameters were calculated for both sites and compared for two times periods, before (1957-1968) and after (1970-2016) the first dam was built on the Grijalva. These parameters are classified in five groups as described by Richter, Baumgartner, Powell and Braun (1996). In addition, Mann-Kendall tests, a non-parametric method used to detect monotonic changes over time (Helsel & Hirsch, 2002), were used to identify trends in discharge over the years at sites 1 and 3 (1970-2016). Mann-Kendall was also used tests to estimate changes in discharge in sites 1, 3 and 6 between 1997 and 2016, and to examine temporal changes in TP and nitrate-N concentrations and loads for sites 1, 2, 5 and 7. The analyses for nutrient concentrations and loads were conducted separately for the pooled data (both seasons), the dry season (February-May), and the wet season (June-January) data.

To test the null hypothesis of no differences in the concentration and load of each form of nitrogen and phosphorus among 2016 sampling sites (1, 3, 4 and 6) and seasons (dry and wet), results were analyzed by two-way ANOVA (site by season). Based on discharge patterns for 2016, the dry season was defined from March through June and the wet season from July through October. If interactions between site and season were significant, treatment means between seasons and sites were compared using Tukey HSD. Normality was tested by Shapiro-Wilk and homoscedasticity by the Levene's test. When departures from normality or homoscedasticity were observed, a logarithmic transformation was applied. All statistical analyses were performed using R Core Team (2015) with a 0.05 level of significance.

RESULTS

Historic discharge data

Before dams were constructed on the Grijalva, the Carrizal River (site1) exhibited seasonal patterns in discharge, with low flows occurring between February and May and higher flows between June and October. However, after the first dam was constructed, predictable changes were evident in the system. Initially, less seasonal variation in flow was observed in the river (Figure 2A); flow increased 32% on average during the dry season and it declined approximately 42% in the wet season (Table 3). Annual extreme conditions in flow were also altered. Specifically, the magnitude of minimum flows (1, 3, 7, 30 and 90-day minimums, calculated as moving averages for each period) decreased after the construction of the first dam as well as maximums (1, 3, 7, 30 and 90-day; Table 3). Additionally, the base flow index, which is the relationship between the 7 day minimum flow and annual average flow, increased in 33% after dam construction. The date of minimum flows shifted from May (the end of the dry season) to November (the month after the wet season ended). The number of low pulses (discharge below 25th percentile) also increased in 211% while its duration decreased in 75% after dam construction. Furthermore, the number of high pulses (discharge above 75th percentile) decreased by 60% and the duration of the high-discharge pulses declined by 37.5% in the post-dam period. Finally, the number of times flow moved between rising and falling increased by 48.8% after dam installation.

Although changes were also evident in the discharge patterns of undammed De la Sierra River (site 3) post 1970, the site did not experience similar, extreme changes in flow relative to the dammed site (Figure 2B; Table 3). For example, there were significant increases in discharge through time in impounded and unmodified reaches (Figures 2C, 2D), though the correlation coefficient was higher in the impounded site. Notably, though there were no declines in discharge at the undammed focal site (site 3; tau, 0.056, $p<0.001$), discharge did decline in the 1997-2006 time period in sites 1 and 6 which are located downstream from the dams (tau, -0.436, $p<0.0001$ and tau, -0.145, $p=<0.0001$, respectively).

Historical nutrient data

Nitrate-N concentration increased through time at sites 1, 2, 3, and 7 (Figure 3; pooled data: site 1, tau, 0.49, $p<0.001$; site 2, tau, 0.266, $p=0.02$; site 5, tau, 0.486, $p<0.001$; site 7, tau, 0.55, $p<0.001$). Concentrations were higher in the wet than in the dry season at all sites except at site 2, where averages were similar for both seasons (Figure 3). TP concentrations did not display similar patterns. There were significant declines in TP at sites 1 and 2 through time (Figure 4; pooled data: site 1, tau, -0.404, $p<0.001$; site 2, tau,-0.484, $p<0.001$) and values were similar between the wet and dry seasons at these sites (Figure 4). No significant trends were detected in sites 5 and 7, but TP values were lower in the dry season relative to the wet season (Figure 4). Significant increases in nitrate-N loading were documented in site 7 (tau, 0.251, $p=0.02$) and for the wet season in site 5 (tau, 0.451, $p=0.01$; Figure 5). Instantaneous nitrate-N loads were similar for the wet and dry season at site 1. However, other sites exhibited greater average values for the wet season. This was especially true at site 5, where average load was 4 times higher in the wet season (Figure 5). TP loads were greater in the wet than in the dry season in most sites, with the exception of site 2, where averages were similar in both seasons. The TP load declined in sites 1 and 2 in both the wet and dry seasons (Figure 5).

2016 sampling data

Nitrate-N concentrations varied among sampling dates and sites (Figure 6). Lowest average concentrations for both seasons were observed upstream from urbanizing areas at site 1. In the dry season, concentrations at site 1 were significantly lower than concentrations at sites 3 and 4, which drain agricultural and suburban areas upstream of Villahermosa, respectively (Tukey, $p<0.006$). In the wet season, nitrate-N concentrations at sites 1 and 4 were significantly lower than at sites 3 and 6 (Tukey, $p<0.02$; Tukey, $p<0.01$). Within sites 3 and 6, significantly higher nitrate concentrations were observed in the wet season (Tukey, $p<0.0001$). Notably, site 3, located upstream from Villahermosa in a region of intensive agriculture, had the highest average dry and wet season nitrate concentrations (Figure 6). There was a range of ammonium-N concentrations among sites, but there were no significant effect of season on ammonium concentrations. Upstream, site 1 had significantly lower ammonium concentrations that sites 4 and 6 (Tukey, $p<0.001$). Nitrite-N concentrations were very low relative to the other forms of nitrogen in the watershed. Again, the upstream site on the Carrizal River, site 1, had lower average concentrations than sites 3, 4, and 6 (Tukey, $p<0.002$). Site 4, had higher concentrations than sites 3 and 6 in the dry season (Tukey, $p<0.00001$), but this difference was not evident in the wet season. Significant seasonal differences in nitrite concentrations were also observed at site 6 (Tukey, $p<0.001$). Average TN concentration also varied among sites and seasons (Figure 6). In the wet season, concentrations were higher than in the dry season at sites 3 and 6 (Tukey, p<0.001). Within sampling events, nitrate-N represented between 3.1 and 72.4% of TN, while ammonium contribution varied from 0.34 to 52.13%.

Differences among sites and between seasons were also evident for phosphorus. In the dry and wet seasons, average SRP concentrations measured in sites upstream from Villahermosa (sites 1 and 3; Figure 6) were lower than in downstream sites. Upstream sites (1 and 3) also had greater SRP concentrations in the wet than in the dry season (Tukey, $p<0.0001$). Greater TP concentrations for the dry and wet seasons were measured at site 4 than for the upstream sites (1 and 3; Tukey, $p<0.0001$). Notably, significant differences in TP concentrations between seasons were detected at site 3 (Tukey, $p<0.0001$), where there was a 172% increase in TP in the wet season relative to dry season values (Figure 6). SRP comprised between 14 and 33% of TP among sites and seasons.

The patterns documented in nutrient loads were also site- and season-dependent. The greatest nitrate-N loads were reported in the dry and wet seasons at the farthest site downstream (6). Values at site 6 were significantly higher (Tukey, $p<0.001$) than sites 1 and site 4 in both seasons (Figure 7) and site 3 in the dry season (Tukey, $p<0.001$). Site 1 had a smaller nitrate load than site 3 in the wet season (Tukey, $p<0.001$). Sites 3 and 4 were the important contributors of the nitrate transported to site 6 during the dry season and wet seasons, respectively. Greater nitrate loads were estimated in the wet season for sites 3 and 6 (Tukey, $p<0.001$). Higher ammonium-N, nitrite-N and TN loads were estimated for sites 4 and 6 (Figure 7). No differences between dry and wet seasons were observed in ammonium loads (Tukey, $p>0.1$). Site 4 contributed a large proportion of the ammonium and nitrite transported to site 6 (Figure 7). Conversely, seasonal differences were evident for nitrite and TN in sites 3 and 4 where loads were greater in the wet season (Tukey $p<0.001$; Tukey, $p<0.006$, respectively). Interestingly, site 1 had a significantly greater average TN load in the dry season (Tukey, $p<0.009$) and was an important supplier to the transported TN at site 6, while in the wet season, sites 3 and 4 were the main contributors. SRP load was also greatest at site 6 in the dry season (Tukey, $p<0.03$). In the wet season, sites 4 and 6 showed significant higher averages than sites 1 and 3 (Tukey, $p<0.001$). Relative to the wet season, sites 3 and 4 were estimated to have significantly lower SRP loads in the dry season (Tukey, $p<0.001$). Site 3 experienced a 700% increase in SRP load in in the wet season (Figure 7). Similarly, greater TP loads were measured at site 6. In both seasons, site 6 had greater loads than site 1 and site 3 (Tukey, $p<0.003$; Figure 7). Differences in TP load between seasons were observed at sites 3 and 4, where higher TP load was found in the wet than in the dry season (Tukey, $p<0.01$). In the dry season, sites 1 and 4 were important SRP contributors to site 6, while in the wet season the relative contributions of sites 3 and 4 were higher.

DISCUSSION

Myriad studies have documented the negative impacts anthropogenic activities have had on rivers and streams throughout the globe. Here, these relationships were examined in a relatively understudied tropical river system using long-term discharge and physicochemical data. The development of a series of dams in the mainstem of the Grijalva River and land conversion within the Grijalva Watershed has fundamentally altered the system through time. In particular, this work demonstrated that dam construction has altered seasonal patterns in discharge in a tropical river system. Moreover, the results suggest that patterns in nutrient concentrations and nutrient loading in the system, especially nitrate, are strongly influenced by land use and vary through both space and time.

Previous work has demonstrated the strong influence of dam construction can have on natural discharge patterns (Magilligan & Nislow, 2005; Wang, Rhoads, & Wang, 2016; Zhang, Zhai, Shao, & Yan, 2015). The study on the Grijalva adds to this body of research and demonstrated that the construction of impoundments in the mainstem of the Grijalva changed the flow regime of the river. Before the first dam (Nezahualcoyotl) was constructed, there was strong, seasonal variation in discharge in the site below the dam (site 1). However, the patterns changed significantly after the dam was constructed and now the downstream sites experience much more uniform flows throughout the year (Figure 2A). Changes in discharge also included reductions in higher monthly flows during the wet season, particularly in September and October. Additionally, the lower flows that were naturally apparent between April and May were also altered, and much higher flows were documented during the dry season.

Despite an increase in average monthly flows during the dry season and in the base flow index, the magnitude of minimum flows (1-90 day) decreased below the dam through time. Although this effect on the extreme minimum flow conditions has been observed in some regulated rivers in USA and Asia (Magilligan & Nislow, 2005; Yan, Yang, Liu, & Sun, 2010; Zhou, Zhang, & Lu, 2013), the most common impact is an increase in minimum flows (Poff, Olden, Merritt, & Pepin, 2007; Puig, Olguín Salinas, & Borús, 2016; Wang et al., 2016; Zhang et al., 2015). The different responses between rivers can be related to dam operation and hydrology. Hydropower dams and stable base flow are two conditions related to a decline in the magnitude of minimum flows (Magilligan & Nislow, 2001; McManamay, Orth, & Dolloff, 2012). In the Carrizal, the upstream operation of 4 hydroelectric dams, which probably have to store water during dry periods can be related to the decrease of minimum flows in the lower part of the watershed as has occurred in other regulated rivers (Magilligan & Nislow, 2001; McManamay et al., 2012). Alteration in the minimum flows can have effects on water quality, in-stream and riparian habitats (Lian, You, Sparks, & Demissie, 2012; Richter, Baumgartner, Braun, & Powell, 1998).

Change in the date of minimum discharge from May (during the dry season) to November (the end of the wet season) was also documented, providing additional evidence that seasonal patterns in discharge were fundamentally altered by dam construction in the Grijalva. Additionally, the results presented here mirrored findings in studies in other regulated rivers that have documented increases in the number of reversals, or the times when water levels in the river changed between rising and falling (Magilligan & Nislow, 2005; Wang et al., 2016; Zhou et al., 2013). These data suggest that the regulated portion of the watershed is experiencing much more unstable flows. Changes in the date of minimum flow and the number of reversals can disrupt life cycles of riparian plants and fishes because habitat conditions and cues for growing and spawning can be altered (Lian et al., 2012; Richter et al., 1998).

In addition to the aforementioned changes in discharge in response to dam construction, there were also changes in interannual variation in discharge between the dammed and undammed sites. Overall, the discharge in the regulated portion of the river (site 1) increased through time after the dam was constructed. Though a similar pattern was evident in the unregulated portion of the river, the increase in flow was not as great (Figure 2D). Most likely, the trend was being driven by the higher flows in site 1 between August 1996 and October 2007. Higher flows during the same period were not evident in the unregulated site; therefore, the increase in flow was probably due to the operation of the four dams upstream from the site (Muñoz-Salinas & Castillo, 2015). Additionally, there were major flooding events in 1999 and 2007 in the Lower Grijalva which may have influenced dam operation and produced higher flows. Moreover, a decrease in flow after 1997 detected at site 1 and 6 but not at site 3 likely are related to flow patterns produced by the operation of dams.

Though the changes in hydrology documented here are likely influencing ecosystem processes of the lower Grijalva, few studies have addressed these questions. For instance, changes in sediment load (Muñoz-Salinas & Castillo, 2015), channel configuration, and the location and size of riparian habitat has probably occurred in the Grijalva, as has been detected in similarly affected systems (Anderson, Pringle, & Freeman, 2008; Le, Al-Juaidi, & Sharif, 2014). Furthermore, alteration in flow regime likely influences the interaction between the river and its floodplains (Stevaux, Martins, & Meurer, 2009; Zurbrügg, Wamulume, Kamanga, Wehrli, & Senn, 2012). Such changes can affect nutrient and sediment retention (Hudson et al., 2005) and the migration of aquatic biota from the river main channel into floodplain aquatic habitats (Jourdan et al., 2016).

Hydrologic regulation also influences the structure of aquatic communities and the behavior of aquatic organisms. In the Grijalva, the alteration of flow patterns and longitudinal connectivity due to dam construction have likely affected economically and ecologically important migratory species including, but not limited to the common snook (*Centropomus undecimalis*), the tarpon (*Megalops atlanticus*), and the prawns (*Macrobrachium carcinus*). For example, several piscivorous snook species migrate from the ocean upstream into tributaries and floodplain lakes, which are important breeding habitat for these species (Perera-Garcia, Mendoza-Carranza, Contreras-Sanchez, Huerta-Ortiz, & Perez-Sanchez, 2011). Although *C. undecimalis* is still very common in the unregulated Usumacinta River, it is scarce in the Grijalva (Gómez-González, Velázquez-Velázquez, Anzueto Calvo, & Maza-Cruz, 2015; Zequeira & Castillo, 2015). Physical barriers, alteration of seasonal changes in flow that may have acted as signaling mechanism to migrate, and the modification of river-floodplain interactions due to dam construction have likely negatively affected population numbers of snook in the Grijalva (Andrade, Santos, & Taylor, 2013; Hernández-Vidal, Chiappa-Carrara, & Contreras-Sánchez, 2014).

Small scale fisheries are an important component of many of the smaller, lower-income communities within the Grijalva-Usumacinta River (Mendoza-Carranza, Arévalo-Frías, & Inda-Díaz, 2013; Perera-Garcia et al., 2011). Small fishing cooperatives and subsistence fishing are commonplace along the unregulated Usumacinta, but appear to be much less common in the Grijalva. Therefore, dam construction in the Grijalva may have substantially altered the economic and ecological relationships that local communities have with the river. Additionally, fish production in the Grijalva is now dominated, in large part, by industrial fishing operations devoted to tilapia (*Oreochromis niloticus*) production that have been developed in the reservoirs behind the dams (Pérez, Cabrera, Bermúdez, & Gutiérrez, 2002). Tilapia is a non-native cichlid in a region that is known for its incredible cichlid diversity (González-Espinoza & Ramírez-Marcial, 2013). Hence, in addition to the ecological threats posed by physicochemical changes from dam construction, the dam-supported fishing industry may present an additional threat to the ecological integrity of aquatic communities and the evolutionary trajectory of fish populations.

In addition to changes in hydrology, the lower Grijalva has also experienced significant changes in nutrient concentrations and nutrient loads through time. In particular, an increase in nitrate concentrations was documented in most of the sites (Figure 3). In other river systems, long-term increases in nitrate have been related to the expansion of agricultural lands and human population growth (Hatfield, McMullen, & Jones, 2009; Lassaletta, García-Gómez, Gimeno, & Rovira, 2009; Stets, Kelly, & Crawford, 2015). In the Grijalva, increases in the amount of land devoted to agriculture (between 3.3 and 17.1%) and in the human population (between 22.2 and 54.1%) within the study catchments (Table 2) are likely related to the observed increase in nitrate concentrations. Increases in nitrate loads were observed in sites 5 and 7, but not site 1 in the Carrizal River. This pattern suggests that the decreases in flow in the regulated reaches may have counteracted the effects of increasing nitrate concentrations on nitrate loading. In contrast, TP concentration and TP load tended to decline in sites 1 and 2 below the dammed portions of the stream. This is unsurprising considering a large proportion of riverine phosphorus is transported in particulate form (Meybeck, 1982) and that there is typically a strong relationship between water column sediment concentrations and discharge (Dodds & Whiles, 2004; Muñoz-Salinas & Castillo, 2015). Thus, the declines in TP concentrations and load that were observed in sites 1 and 2 may be related to the decrease in discharge detected post 1997. Collectively, these data suggest that dam operation, in addition to the retention of phosphorus in reservoirs (Maavara et al., 2015), may influence long-term trends in phosphorus availability in the Grijalva. This may be especially important as primary productivity in many streams in the region have been classified as phosphorus-limited (Capps & Flecker, 2013), and changes in phosphorus availability in such systems may alter the quality and quantity of basal food resources.

In general, lower nutrient concentrations were documented upstream from Villahermosa, a major urban center on the Grijalva. Lower nutrient concentrations were measured in 2016 at site 1 on the Carrizal River; however, this pattern was not evident in the other upstream site (site 3) that was sampled in the same year. Though land use in the catchments of sites 1 and 3 were similar, there were no major areas of agricultural or urban development above site 1. In contrast, large banana plantations were located upstream from site 3, and may have generated the higher nitrate concentrations measured at this site. In drainage ditches in these banana plantations, Aryal, Geissen, Ponce-Mendoza, Ramos-Reyes and Becker (2012) documented higher nitrate concentrations than in pasture areas, particularly during the wet season (median 3-4 mg/L). They also found high levels of nitrite (up to 14 mg/L) in subsurface and groundwater in plantations (Aryal et al., 2012). Higher concentrations of nitrate in site 6 downstream from Villahermosa were likely influenced by upstream agricultural development and the input of wastewater from the city of Villahermosa, where only 13.89% of its wastewater undergoes primary treatment before it is released into the Grijalva (Carrillo, Félix, & Gutiérrez, 2009). In addition, wastewater discharge may have also been the cause of higher concentrations of ammonium and nitrite at sites 4 and 6, which were influenced by urban and suburban development. Wastewater discharge is also likely related to the high concentrations of P observed at site 4, located in Pichucalco River, which drains a large suburban area located south of Villahermosa (Figure 1). Most likely, this river generates large proportion of the phosphorus transported by the Grijalva River at site 6 (Figure 7).

Increase in nutrient concentration from land use change and urban population expansion can impair freshwater ecosystems of the lower Grijalva and alter the ecosystem services they provide, such as drinking water. Several drinking water treatment plants for Villahermosa and the surrounding suburban areas are located near the study sampling sites (CONAGUA, 2015). Thus, increase in nutrient concentrations may enhance algal abundance, particularly in the dry season, which can negatively affect the purification process (Bond, Huang, Templeton, & Graham, 2011). Excess nutrient can also create “dead zones” or hypoxic areas that threaten the ecological integrity of estuaries (Rabalais et al., 2014). A hypoxia zone has already been identified at the Grijalva-Usumacinta mouth in the Gulf of México (Signoret, Monreal-Gómez, Aldeco, & Salas-de-León, 2006). Therefore, it is important to enhance the regulations and enforcement controlling point and non-point pollution sources in the Grijalva Watershed. This is especially salient now as, the expansion of agriculture has been currently launched as a state strategy to enhance the economy (Gobierno del Estado de Tabasco, 2013).

The threats identified in the Lower Grijalva are representative of the threats faced by many tropical watersheds. Dam construction is predicted to increase markedly in the tropics and land conversion into agriculture and urban areas in tropical regions will persist as human population growth continues, subsequently increasing demand for freshwater resources (Anderson et al., 2006; Laurance et al., 2014; Lees, Peres, Fearnside, Schneider, & Zuanon, 2016; Poff et al., 2007; Winemiller et al., 2016).The combined effects of these activities will threat freshwater ecosystems and their services unless careful analysis in the design of dams and the implementation of sustainable practices such as environmental flows, agroecological production and pollutant control are considered (Poff et al., 2007; Winemiller et al., 2016). Long term monitoring of hydrology, water quality and biodiversity in large tropical rivers is necessary to detect alterations and evaluate the outcomes of mitigation and restoration practices. Furthermore, development agencies and environmentally-focused organizations should be encouraged to support the collection and distribution of long-term ecological data to support management efforts to sustain the ecosystem services provided by tropical rivers.

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Table 1. Monitoring sites in the Grijalva Watershed. Historical discharge and water quality data were available for sites 1-3 and 5-7. Water quality data were also collected in 2016 as part of this study for sites 1, 3, 4 and 6. The time periods listed indicate when data were collected.

Site Number	River	Monitoring Site	Discharge	Water Quality	This study
1	Carrizal	González	1957-2016	2000-2016	March-October 2016
2	Carrizal	Frigorífico	-	2000-2016	-
3	De la Sierra	Pueblo Nuevo	1957-2016	-	March-October 2016
4	Pichucalco	Pichucalco	-	-	March-October 2016
5	De la Sierra	Gaviotas	1997-2011	2000-2011	-
6	Grijalva Mainstem	Porvenir	1997-2016	-	March-October 2016
7	Grijalva Mainstem	Rosarito	-	2000-2016	-

Table 2. Land cover (%) within the catchment upstream of the 2016 sampling sites. Land cover was estimated using data collected by National Institute of Statistic and Geography (INEGI) in Mexico. Montane forest includes pine, pine-oak, oak, oak-pine, and mountain cloud forest. Moist forest includes high and low evergreen forest, riparian forest and semi-evergreen forest.

	Site 1		Site 3		Site 4		Site 6	
	2001	**2011**	**2001**	**2011**	**2001**	**2011**	**2001**	**2011**
Catchment area (km^2)	32,746.73		4,463.05		1,313.86		39,226.95	
Discharge [a] (m^3/s)	283.68 ± 143.67		223.60 ± 214.79		-		551.06 ± 299.07	
Population size [b]	1,869,942	2,320,231	429,022	524,251	79,940	123,176	2,806,411	3,448,446
Agriculture (%)	24.89	25.71	23.6	25.1	11.83	13.86	24.09	25.03
Montane forest (%)	14.47	12.22	6.33	4.23	1.36	1.25	12.83	10.72
Moist forest (%)	2.22	1.88	2.67	2.58	8.15	5.5	2.44	2.06
Secondary vegetation (%)	34.69	34.57	37.04	39.21	8.73	9.6	33.45	33.64
Pasture (%)	21.31	21.68	29.37	28.1	63.82	64.4	24.61	24.74
Wetlands (%)	0.05	0.04	0.86	0.52	5.66	4.93	0.43	0.33
Urban (%)	0.63	0.87	0.13	0.24	0.19	0.19	0.63	0.86
Water bodies (%)	1.65	2.92	-	0.02	0.27	0.27	1.43	2.51
Bare soils (%)	0.1	0.1	-	-	-	-	0.09	0.1

[a] Means and standard deviation of discharge for the period 2000-2016 are shown. No discharge data is available for site 4.

[b] Population data was obtained from INEGI 2000 and 2010 census.

- Information not available.

Table 3. Comparison of IHA parameters at sites 1 (Carrizal River; impounded) and 3 (De la Sierra River; not impounded) for the period before (1957-1968) and after (1970-2016) the construction of the Nezahualcoyotl Dam. Group a: Median values for the monthly discharge; Group b: Mean values; Group c: Julian date; Group d: Median values; Group e: Median values.

	Site 1			Site 3		
	Pre	**Post**	**% Change**	**Pre**	**Post**	**% Change**
Parameter Group a						
January	208.2	145.4	-30.16	152.9	143.2	-6.34
February	149.3	138.2	-7.43	83.32	108.3	29.98
March	133.9	141.2	5.45	66.57	70.32	5.63
April	104.5	166.3	59.14	42.29	50.8	20.12
May	95.96	177.2	84.66	34.85	48.6	39.45
June	169.5	191.2	12.8	118	141.3	19.75
July	281.3	159.6	-43.26	139.9	149.6	6.93
August	255.3	171.5	-32.82	144.3	170.1	17.88
September	339.3	191.2	-43.65	314.3	318.9	1.46
October	444.1	167.7	-62.24	324.9	412.2	26.87
November	285.9	154.9	-45.82	171.2	230.4	34.58
December	217.6	142.4	-34.56	134.7	171.2	27.1
Parameter Group b						
1-day minimum	83.67	49.3	-41.08	27.66	28.11	1.63
3-day minimum	85.06	59.72	-29.79	27.72	30.55	10.21
7-day minimum	86.87	70.06	-19.35	28.06	32.57	16.07
30-day minimum	92.25	87.96	-4.65	34.77	40.09	15.3
90-day minimum	112.1	107.3	-4.28	57.31	64.52	12.58
1-day maximum	853.5	473.2	-44.56	861.4	848.6	-1.49
3-day maximum	770.4	444.4	-42.32	801.3	804.7	0.42
7-day maximum	682.8	362.2	-46.95	705.3	733.7	4.03
30-day maximum	556.5	262.1	-52.9	493.3	528.7	7.18
90-day maximum	442.8	212	-52.12	355.1	412.5	16.16
Number of zero days	0	0		0	0	
Base flow index	0.332	0.441	33.05	0.156	0.179	14.38
Parameter Group c						
Date of minimum	145.5	326	124.05	131.5	126	-4.18
Date of maximum	286.5	298.5	4.19	277.5	278	0.18
Parameter Group d						
Low pulse count	4.5	14	211.11	7.5	6	-20
Low pulse duration	12	3	-75	7	5.5	-21.43
High pulse count	7.5	3	-60	13	13	0
High pulse duration	4	2.5	-37.5	4	4	0
Low Pulse Threshold	128.3			64.74		
High Pulse Threshold	323.9			251.3		
Parameter Group e						
Rise rate	13.53	12.36	-8.65	21.81	18.56	-14.9
Fall rate	-9.26	-11.21	21.03	-13.04	-13.13	0.69
Number of reversals	105.5	157	48.82	94.5	102	7.94

Figure 1. Location of the study area and sampling sites. A) The Grijalva Basin in southern Mexico. B) states of Chiapas and Tabasco in the Grijalva Basin. The grey area represents the study drainage area. C) Historical and 2016 study sites by number. In Tabasco, the river has two branches, the Samaria and the Carrizal, which run through the city of Villahermosa. González (site 1) station is located at the Carrizal River, upstream from Villahermosa. Frigorífico (site 2) is also on the Carrizal but within the urban reaches of the river. Pueblo Nuevo (site 3) is located on De la Sierra River, upstream from Villahermosa. Pichucalco (site 4) is a tributary of De la Sierra River and runs through suburban areas located south of Villahermosa. Gaviotas (site 5) is located downstream from the confluence of De la Sierra and Pichucalco Rivers. Porvenir (site 6) is on the Grijalva River, 4 km downstream from Villahermosa. Rosarito (site 7) is also located on the Grijalva downstream from Villahermosa and 50 km upstream from the mouth of the Grijalva-Usumacinta on the Gulf of Mexico.

Figure 2. Flow regime comparison between impounded (site 1; Carrizal) and unregulated (site 3; De la Sierra) rivers in the Grijalva watershed. Annual variation in monthly flows in the Carrizal River (A) and the De la Sierra River (B) before (1957-1968) and after (1970-2016) the construction of the first hydroelectric dam on the Grijalva River (1969). Monthly median values for each month are shown. The error bars represent the limits of the variability ranged at 17 percentiles from the median. Daily discharge between 1970 and 2016 for the Carrizal River (C) and the De la Sierra River (D); Mann-Kendall tests indicated significant positive trends in discharge through time, although the correlation coefficients were low for both sites.

Figure 3. Temporal variation in nitrate concentrations for the dry and wet seasons at the historical sites (site 1 (A, B), site 2 (C, D), site 5 (E, F), and site 7 (G, H)). Blue lines represent the trend of the data and grey area 95% confidence interval. Significant, positive trends in nitrate concentration were obtained using Mann-Kendall tests for sites 1, 5 and 7. No significant trends were observed at site 2 ($p>0.05$). Numbers of observations are in parentheses.

Figure 4. Temporal variation in total phosphorus (TP) concentrations for the dry and wet seasons at the historical sites (site 1 (A, B), site 2 (C, D), site 5 (E, F), and site 7 (G, H)). Blue lines represent the trend of the data and grey area 95% confidence interval. Significant, negative trends in TP were obtained using Mann-Kendall tests for sites 1 and 2. No significant trends were found at sites 5 and 7 ($p>0.05$). Numbers of observations are in parentheses.

Figure 5. Temporal variation in nitrate-N and total phosphorus (TP) loads at the historical sites (site 1 (A, B), site 2 (C, D), site 5 (E, F), and site 7 (G, H)). Significant trends over time were only obtained for site 5 nitrate loading during the wet season (tau, 0.451, $p=0.01$) and for TP loading in site 1 and site 2 in both seasons (site 1: dry season, tau, -0.510, $p=0.0007$, wet season, tau, -0.61, $p<0.0001$; site 2: dry season, tau, -0.500, $p=0.007$, wet season, tau, -0.708, $p<0.0001$). Please note that data were only available through 2012 for site 5.

Figure 6. Nutrient concentrations at the 2016 sampling sites for nitrate-N, ammonium-N, nitrite-N, total nitrogen (TN), soluble reactive phosphorus (SRP), and total phosphorus (TP). Mean values and ±1 standard deviation are shown for each sampling date. The dry season runs from March through June and the wet season runs from July through October.

Figure 7. Instantaneous nutrient loads at the 2016 sampling sites for nitrate-N, ammonium-N, nitrite-N, total nitrogen (TN), soluble reactive phosphorus (SRP), and total phosphorus (TP). Mean values and ±1 standard deviation are shown for each sampling date. The dry season runs from March through June and the wet season runs from July through October. Loads were estimated from nutrient concentrations determined in this study and daily discharge data from CONAGUA monitoring sites.

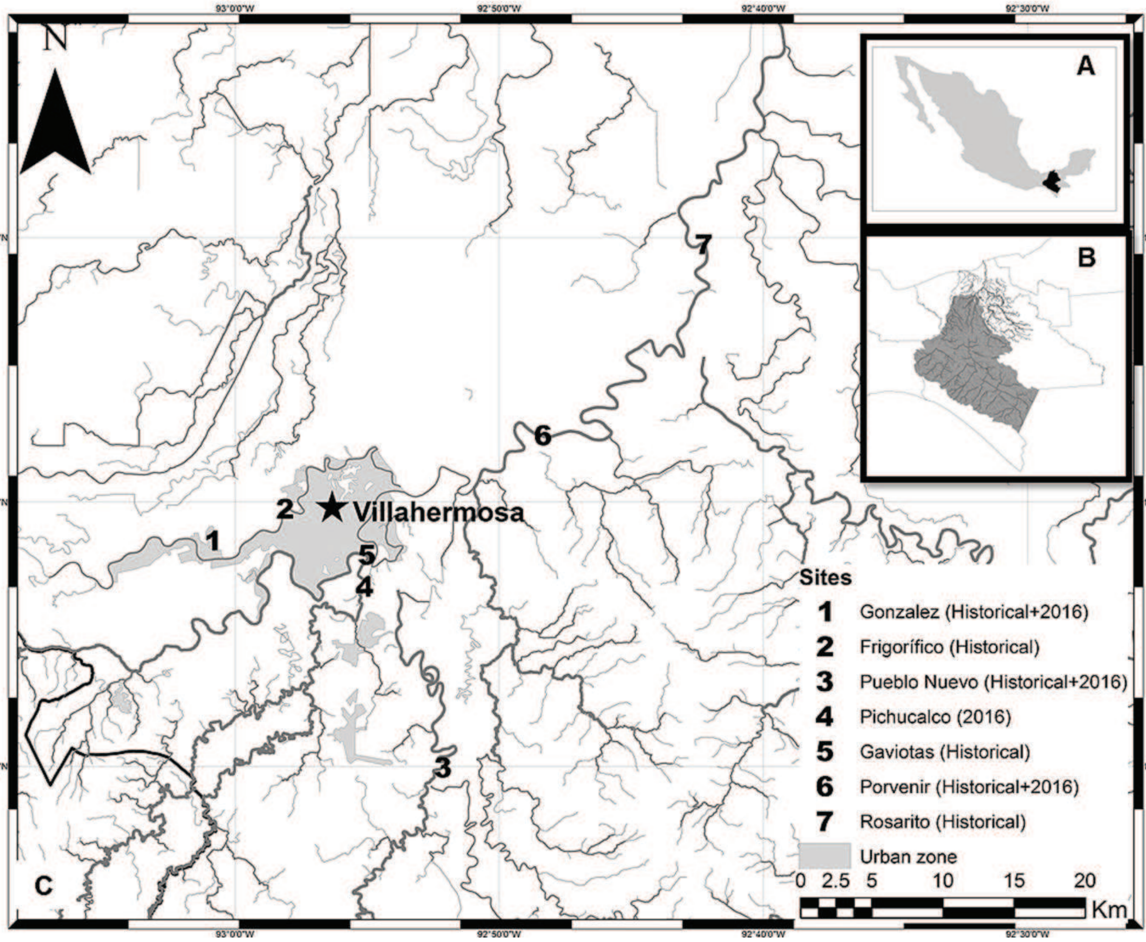

N
A
B
93°0'0"W
92°50'0"W
92°40'0"W
92°30'0"W
18°10'0"N
18°0'0"N
17°50'0"N
7
6
2
Villahermosa
1
5
4
3
C
Sites
1 Gonzalez (Historical+2016)
2 Frigorífico (Historical)
3 Pueblo Nuevo (Historical+2016)
4 Pichucalco (2016)
5 Gaviotas (Historical)
6 Porvenir (Historical+2016)
7 Rosarito (Historical)
Urban zone
0 2.5 5 10 15 20
Km

Figure 1

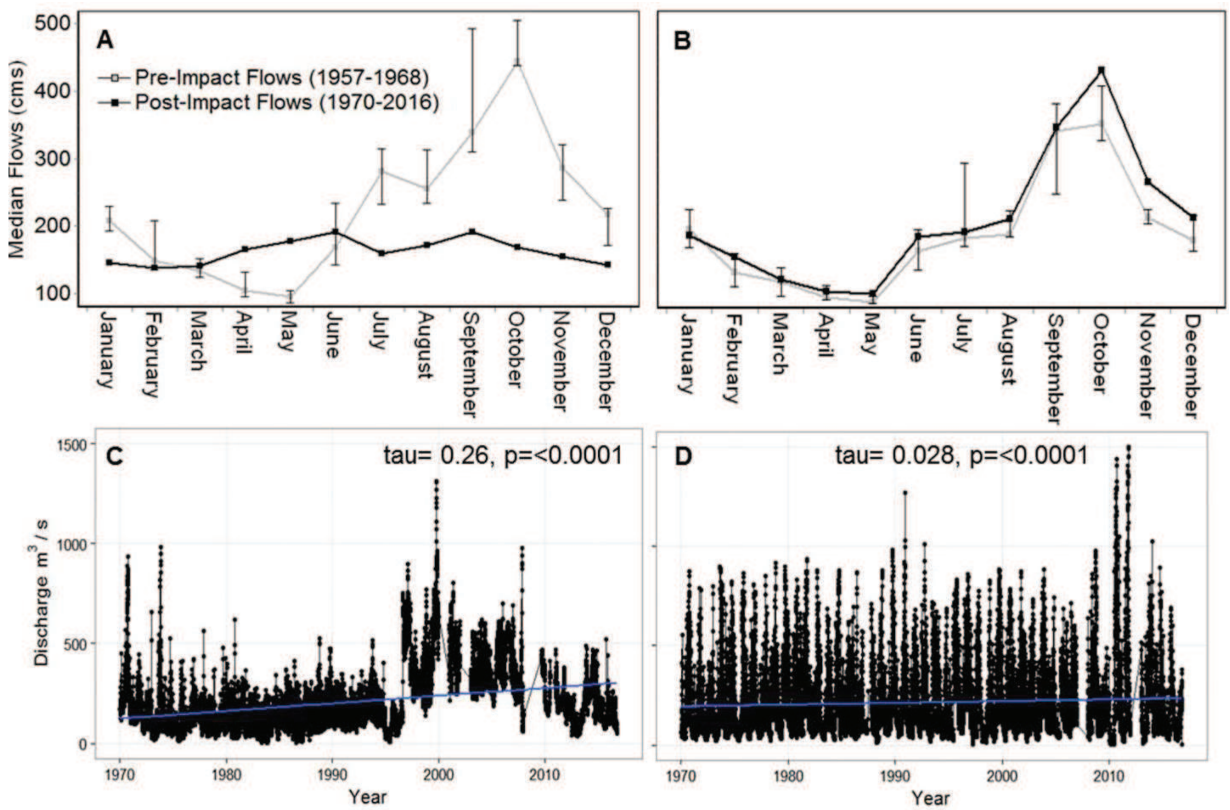
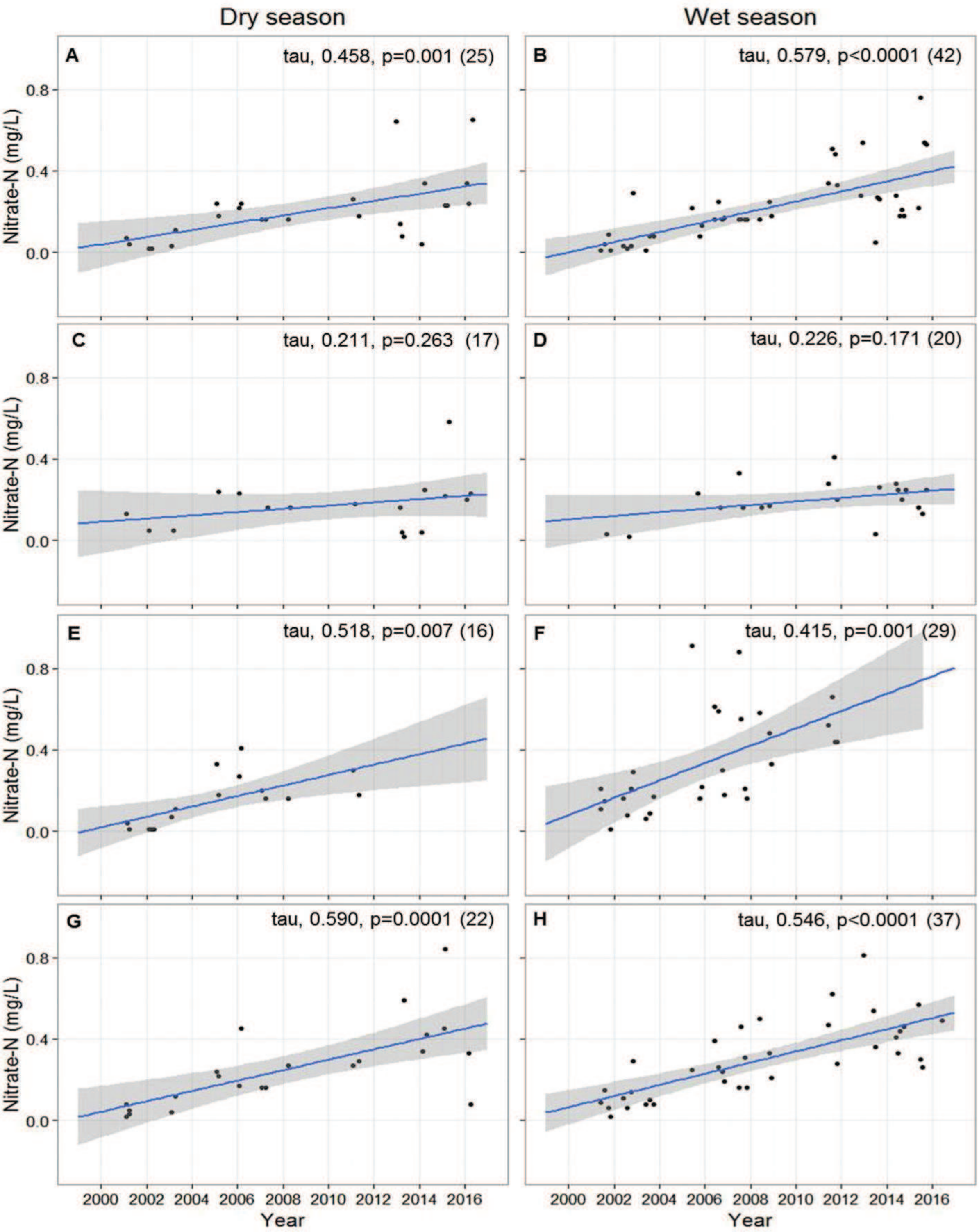

A
B
C
D
Pre-Impact Flows (1957-1968)
Post-Impact Flows (1970-2016)
Median Flows (cms)
January
February
March
April
May
June
July
August
September
October
November
December
tau= 0.26, p=<0.0001
tau= 0.028, p=<0.0001
Discharge m^3 / s
Year

Figure 2



Dry season
Wet season
A
tau, 0.458, p=0.001 (25)
B
tau, 0.579, p<0.0001 (42)
C
tau, 0.211, p=0.263 (17)
D
tau, 0.226, p=0.171 (20)
E
tau, 0.518, p=0.007 (16)
F
tau, 0.415, p=0.001 (29)
G
tau, 0.590, p=0.0001 (22)
H
tau, 0.546, p<0.0001 (37)
Nitrate-N (mg/L)
0.0
0.4
0.8
2000
2002
2004
2006
2008
2010
2012
2014
2016
Year

Figure 3

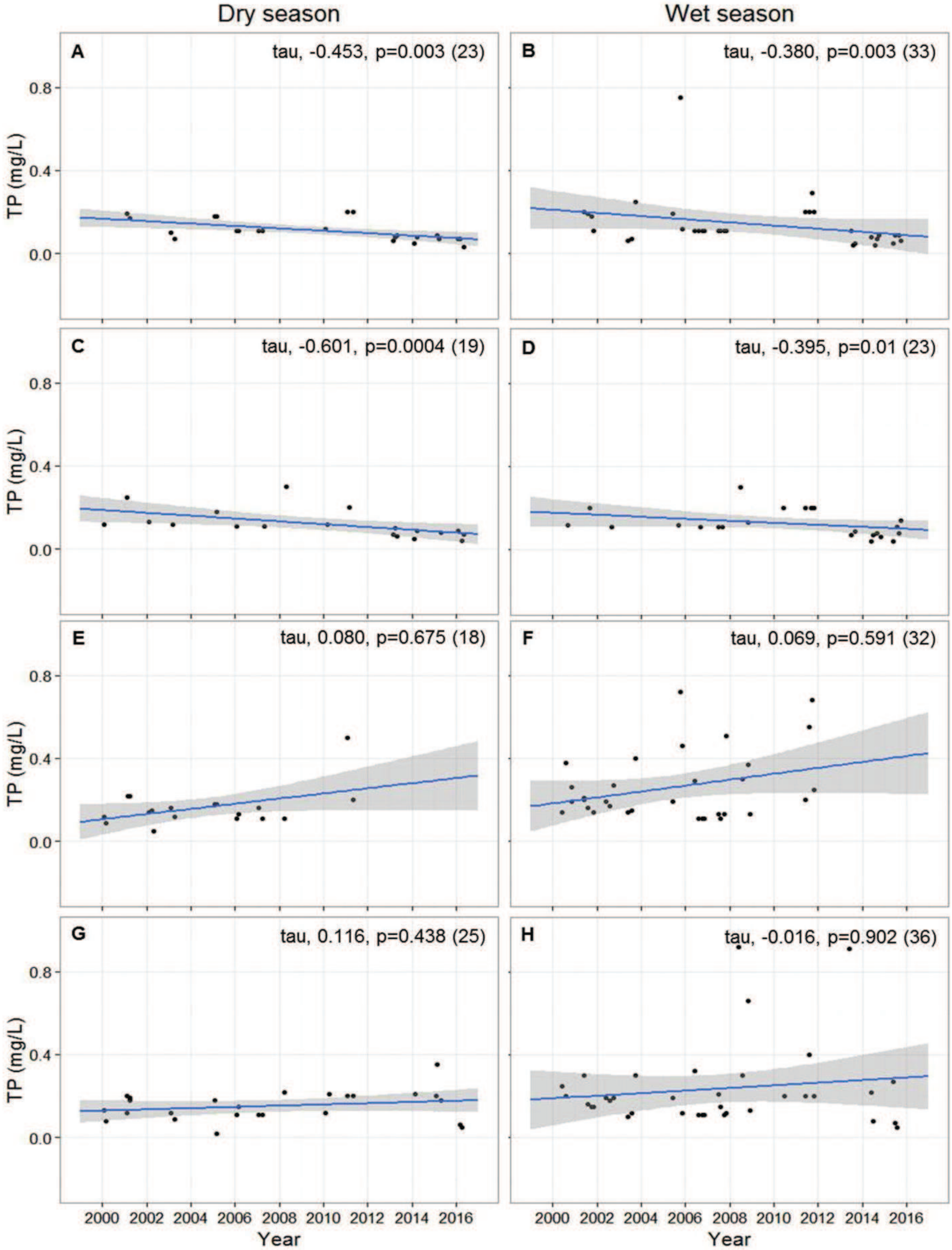

Dry season
Wet season
A
tau, -0.453, p=0.003 (23)
B
tau, -0.380, p=0.003 (33)
C
tau, -0.601, p=0.0004 (19)
D
tau, -0.395, p=0.01 (23)
E
tau, 0.080, p=0.675 (18)
F
tau, 0.069, p=0.591 (32)
G
tau, 0.116, p=0.438 (25)
H
tau, -0.016, p=0.902 (36)
TP (mg/L)
0.0
0.4
0.8
2000 2002 2004 2006 2008 2010 2012 2014 2016
Year

Figure 4

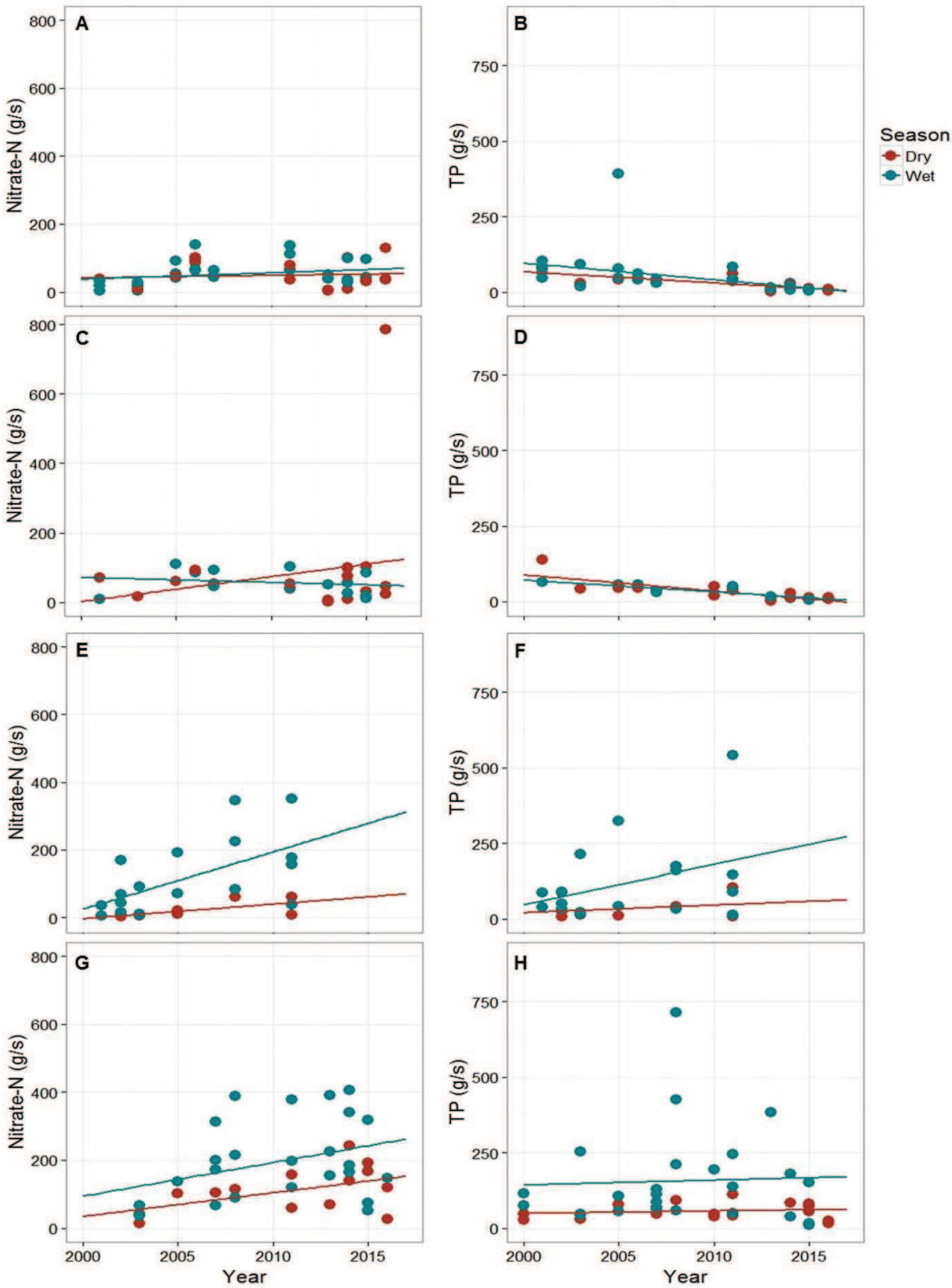

A
B
C
D
E
F
G
H
Nitrate-N (g/s)
TP (g/s)
Season
Dry
Wet
Year
800
600
400
200
0
750
500
250
2000
2005
2010
2015

Figure 5

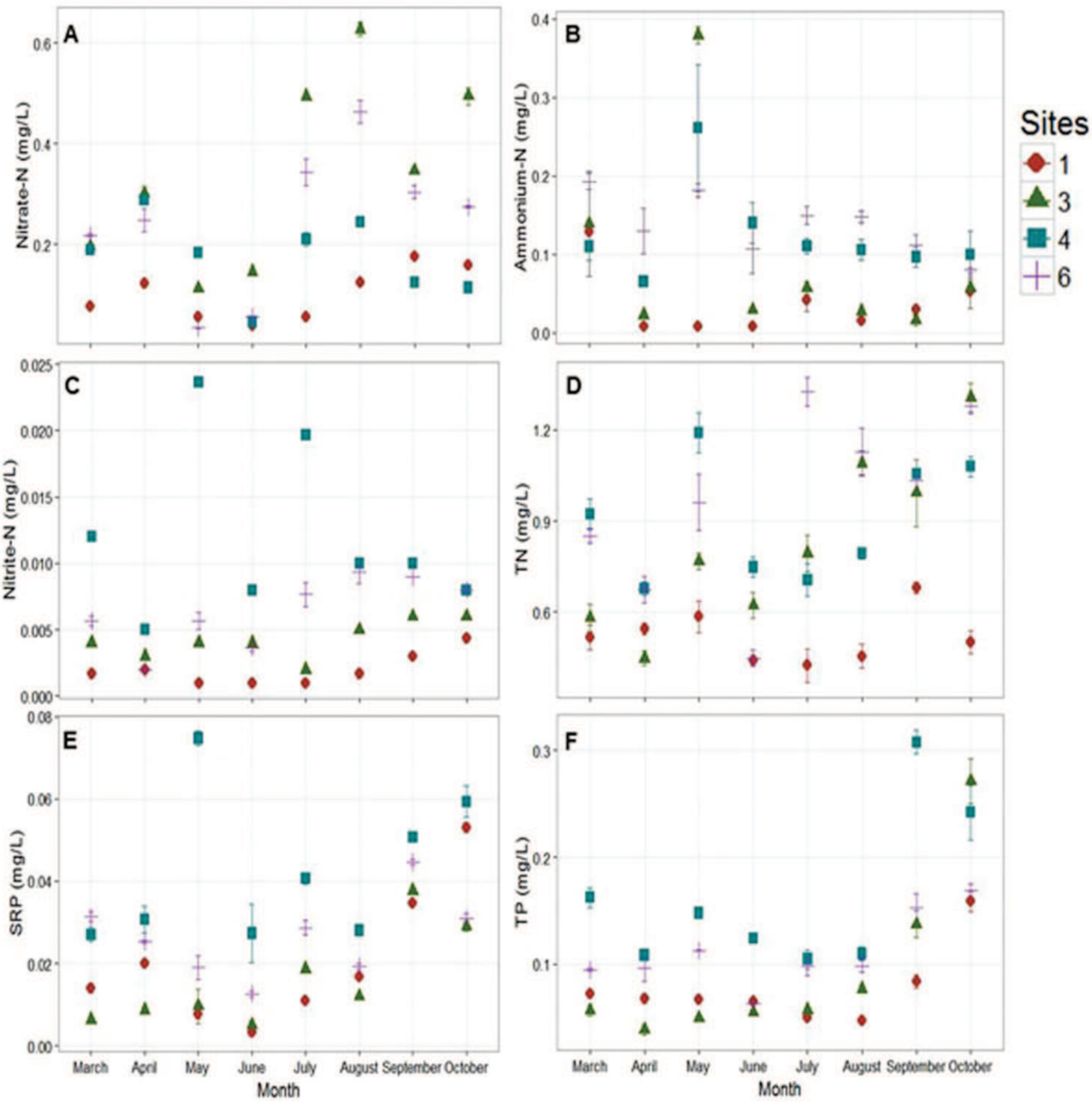

A
B
C
D
E
F
Nitrate-N (mg/L)
Ammonium-N (mg/L)
Nitrite-N (mg/L)
TN (mg/L)
SRP (mg/L)
TP (mg/L)
Sites
1
3
4
6
March
April
May
June
July
August
September
October
Month

Figure 6

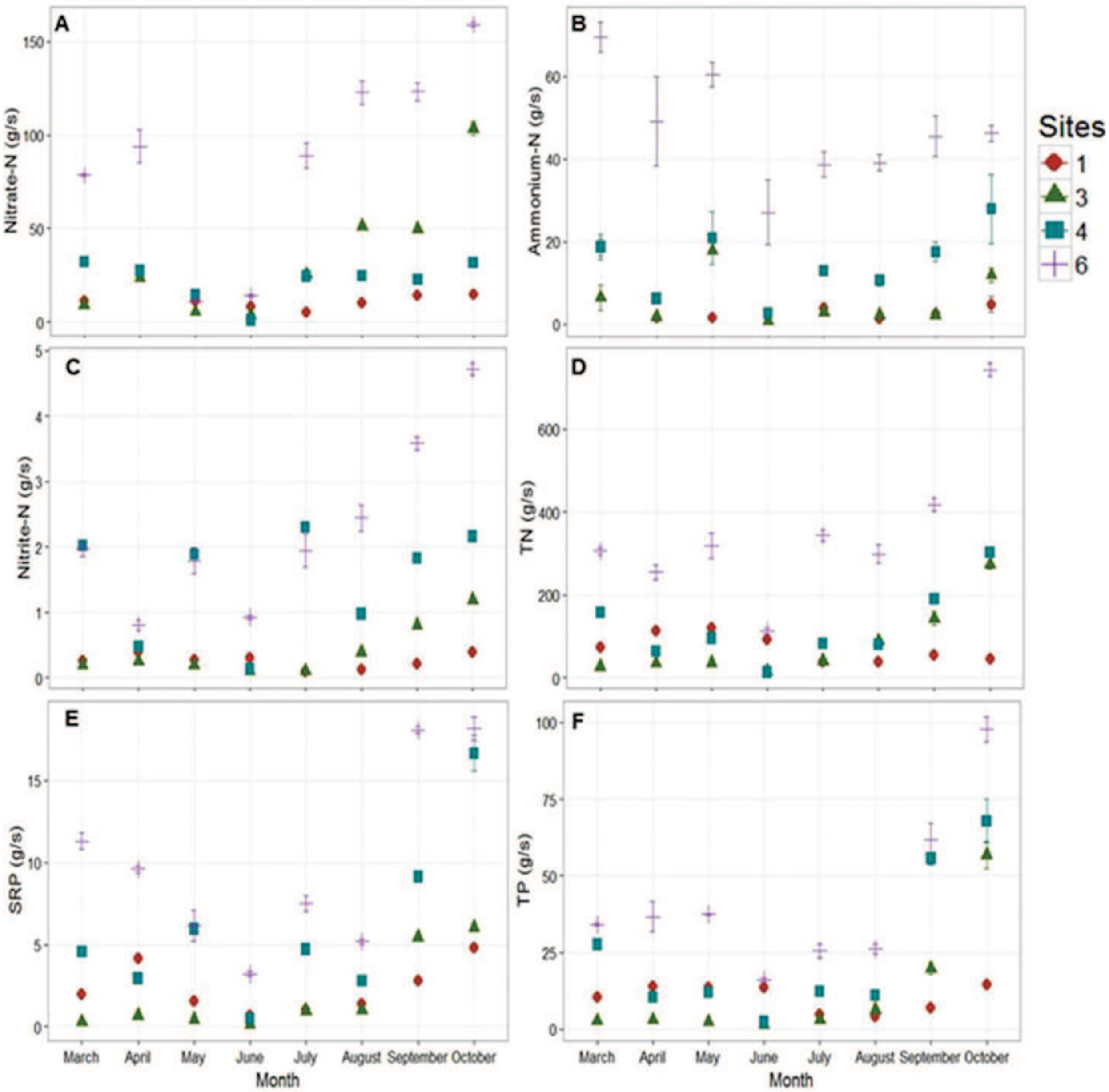

A
B
C
D
E
F
Nitrate-N (g/s)
Ammonium-N (g/s)
Nitrite-N (g/s)
TN (g/s)
SRP (g/s)
TP (g/s)
Sites
1
3
4
6
March
April
May
June
July
August
September
October
Month

Figure 7

Capítulo 3. Conclusiones

La construcción de presas hidroeléctricas y los cambios de uso de suelo, han tenido impactos sobre el caudal y las concentraciones de nutrientes en la cuenca baja del Grijalva, las cuales se han visto modificadas además por el incremento de la población y de actividades como la agricultura y la ganadería. Se encontró que la construcción de las presas en la cuenca alta del río Grijalva han contribuido a alteraciones muy significativas en el caudal del río Carrizal, ubicado aguas abajo de las presas, incluyendo: a) la pérdida de variabilidad estacional en el caudal, b) cambios en la magnitud, duración y momento de las condiciones hidrológicas extremas anuales, así como en c) la frecuencia y duración de los pulsos altos y bajos, y d) en la tasa y frecuencia de los cambios de las condiciones hidrológicas. Por otro lado, se encontró que el río de la Sierra que no ha sido represado conserva esta variabilidad natural, lo cual sugiere que los cambios observados en el caudal del río Carrizal muy probablemente están asociados a la construcción y operación de las presas. Estos cambios en la hidrología del río Grijalva pueden tener efectos negativos sobre la calidad de agua, los organismos acuáticos, y en la interacción del río con las zonas de inundación y las zonas ribereñas. Lo anterior puede afectar principalmente las pesquerías y, por tanto afectar a los habitantes de las comunidades aledañas a los ríos.

Asimismo se encontraron cambios en las concentraciones y transporte de nutrientes a través del tiempo. Los nitratos presentaron una tendencia al aumento, lo cual se vincula a los cambios de uso de suelo que se han llevado en la cuenca a través de los años. Sin embargo las concentraciones de fósforo en el tiempo presentan una tendencia al decremento en el río Carrizal, lo cual puede ser a causa de la alteración del caudal y retención de sedimentos por las presas.

Las concentraciones de nitrógeno y fósforo medidas durante el año 2016 variaron espacial y temporalmente. Las mayores concentraciones de nutrientes se observaron en sitios que presentan agricultura intensiva y desarrollo urbano en sus áreas de drenaje. También se observaron concentraciones más altas aguas abajo de la ciudad de Villahermosa, sugiriendo influencia de las descargas de aguas residuales de esta ciudad. Las concentraciones más altas se encontraron en temporada de lluvias, lo cual puede ser causa de la escorrentía de cultivos y pastizales. En cuanto al transporte de nitrógeno y fósforo calculado durante el año 2016, los valores más altos se registraron en el sitio 6 en ambas temporadas, debido a que en este sitio se registra un mayor caudal en comparación con los registrados en los otros sitios y las concentraciones de los nutrientes también son altas.

Estos cambios en las concentraciones de nutrientes pueden afectar negativamente el suministro de agua a la población, alterar los ecosistemas dulceacuícolas y costeros.
Los cambios observados pueden ser comunes a otros sistemas tropicales, dado a que la economía de estos países se encuentra en desarrollo y por lo tanto demandan un mayor número de servicios.

Por lo cual, es requerido tomar medidas dirigidas hacia la mitigación de los impactos producidos por la construcción de presas y por las actividades antropogénicas desarrolladas en toda la cuenca. Se propone que previo a la construcción de alguna presa, se realicen estudios hidrológicos y ecológicos para determinar la viabilidad de la construcción sin comprometer las funciones ecológicas de los ríos, además de que se elijan sitios donde el daño al ecosistema sea de menor magnitud. Igualmente deben aplicarse controles a las fuentes puntuales y no puntuales de contaminación, para disminuir la entrada de nutrientes a los cuerpos de agua y evitar los efectos de la eutrofización. Es por ello, que el desarrollo de trabajos como el aquí presentado, aunado a un monitoreo continuo de la calidad de agua y de variables hidrológicas de ríos y arroyos, debe fortalecerse a fin de poder evaluar cambios producto de las actividades humanas, y con ello robustecer y hacer más efectivas las medidas de mitigación y restauración.

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