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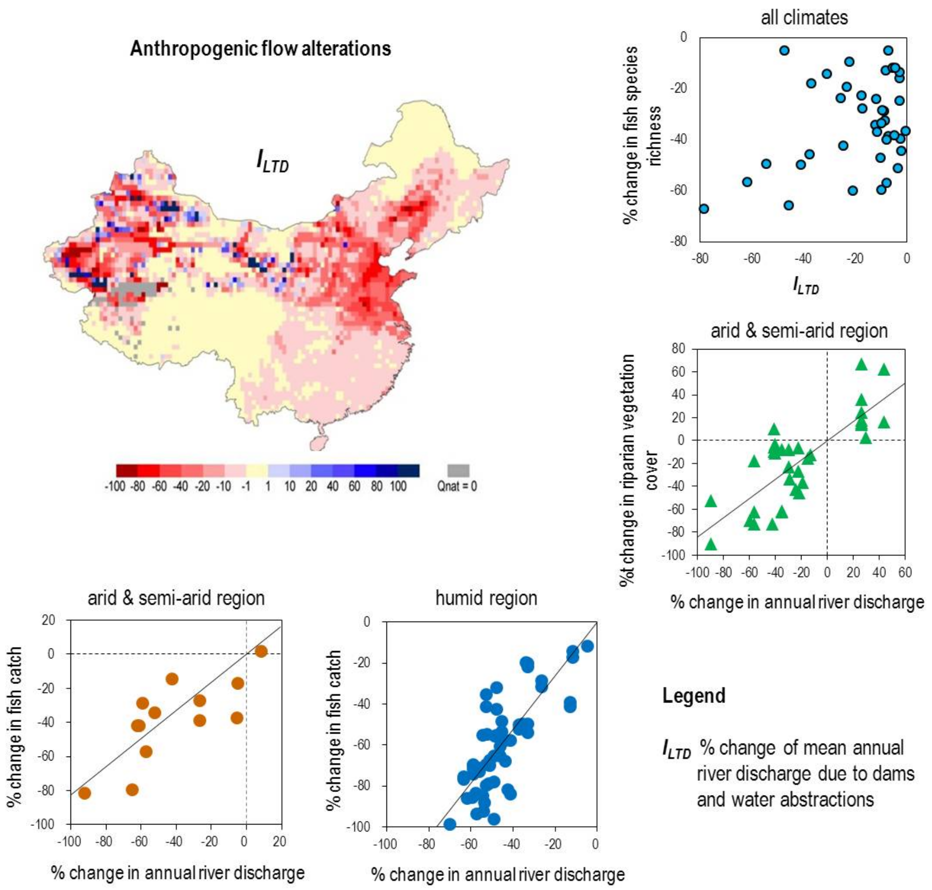
Anthropogenic river flow alterations and their impacts on freshwater ecosystems in China



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Summary

In the past six decades, as China's economy and population booms, excessive water withdrawals and dam construction have significantly affected the natural flow regimes and surface freshwater ecosystems in the whole of China, and thus resulted in serious environmental problems. In order to balance the competing water demands between human and environment as well as provide knowledge for sustainable water management, assessment on anthropogenic flow alterations and their impacts on aquatic and riparian ecosystems in China were needed.

The major objective of this research was to develop quantitative relationships between anthropogenic flow alterations and ecological responses in China's aquatic and riparian ecosystems. To fulfill the goal, the first step was to quantify the degree of anthropogenic impacts on natural flow regimes in China. Thus, a comprehensive assessment of river flow alterations due to human water use and dams for the whole of China, with particular emphasis on changes of flow magnitude, was conducted by using an improved version of the global hydrological and water use model WaterGAP, which combines 731 artificial reservoirs and 2 regulated lakes in China. Natural and anthropogenically altered conditions for five ecologically relevant flow indicators were then quantified and compared. The results showed that the total annual river discharge into oceans and internal sinks as well as discharge at international boundary for the whole of China has been decreased by 6%. At macroscale level, long-term average river discharge and statistical low flow Q_{90} have decreased by more than 10% on 25% and 35% of China's total land area, mainly due to irrigation. Statistical high flow Q_{10} has strongly decreased by 31% of the total land area, mainly due to dam operation, while low flow Q_{90} has increased by 12% of land area downstream of the reservoirs. Q_{10} has increased on only 3% of the total land area as a result of return flow from groundwater abstraction. Seasonal flow amplitude has decreased significantly on one third of China's land area, while seasonal regime changed significantly on two fifth of the total land area. Generally, large flow alterations occurred in many regions of northern China and only minor changes were found in most of southern China.

Flow alterations have occurred in most of China's surface water bodies to a certain extent. Although the overall reduction in average discharge is relatively small, the low- and high-flows as well as seasonal variability have been largely affected in China due to water withdrawals and river flow regulation by dams. Such flow alterations may have caused significant impacts on aquatic and riparian ecosystems in China, thus rational planning and development of water resources should be considered in the future management.

After determining impacts of human activities on river flow regimes, the following step was to quantify impacts of anthropogenically altered river flow regimes on freshwater ecosystems in China. To do this, a total of 61 published Chinese studies with related to environmental water requirements and/or sustainable water management were extensively reviewed. Observed hydrological and ecological data under both reference (the earliest records that were reported in the studies) and altered conditions for eleven river basins and watersheds in China were extracted. Based on these datasets, the first estimation on quantitative relationships between anthropogenic flow alterations and ecological response in China was performed. According to the literature review, most of the ecological variations were associated with the alterations in flow magnitude, particularly decreases in average river discharge. Ecological responses were largely expressed as negative responses of the most ecological groups, i.e. fish, macrophyte and riparian vegetation, while positive responses to reduced flow metrics were reported for planktons and waterbirds. Linear relationships between ecological responses and alterations in flow metrics in China were developed among fish, riparian vegetation and plankton. Fish diversity and weight or amount of fish catch decreased consistently in response to reduced flow magnitude in China. Generally, about 40% of changes in fish were associated with alterations in average river discharge as well as low- and high-flows (r^2 = 0.43). Moreover, 4.8-92% decreases in flow magnitude might have resulted in 6.9-99.9% losses in fish diversity and fish catch in China, while an increase of 8.4% in average river discharge might lead to 1.8 % increase in fish catch.

Vegetation cover and biomass of riparian vegetation responded mostly negatively to decreased flow magnitude, while vegetation cover and growth rate responded positively to increased average river discharge. More than 60% of variations in riparian vegetation could be explained by altered flow magnitude. Generally, 12-89% reductions in flow magnitude were likely resulted in 4-90.3% decreases in riparian vegetation cover, while 26.4-171% increases in average river discharge might have led to 2.5-172.2% of increases in both vegetation cover and growth rate of riparian vegetation in China.

No clear relationship was found between response of plankton and altered flow metrics. Diversity and abundance of most sensitive plankton species reduced in response to either increased or decreased river flows, while some tolerant species showed significantly positive response (113-2354% increases) to reduced high flow (12-83% decreases) and increased low flow (6% increase).

Because general relationships could not be developed from all responses of reported ecological categories to flow alterations in China based on current literature review, a supplementary analysis was conducted with respect to responses of specific ecological categories to climate-driven and anthropogenically altered flow metrics. Consequently, quantitative relationships of variations in riparian vegetation and fish to alterations in average river discharge in arid and semi-arid region and/or humid region were performed. The results showed that riparian vegetation cover was significantly correlated with altered average river discharge ($r = 0.79$) in arid and semi-arid region, and about 63% of the changes in vegetation cover could be explained by alterations in average river discharge. Moreover, around 53% and 58% variations in fish catch in arid and semi-arid region as well as humid region might be associated with altered average river discharge and seasonal low- and high-flows. According to the findings, fish are more sensitive than other species, when flow alterations occur. The findings of this study indicated that direction and magnitude of ecological responses to flow alterations depend largely on characteristics of ecological categories and types of flow alteration. Therefore, relationships between responses of specific ecological groups or species-specific responses and flow metrics can provide a better solution in quantifying the impacts of anthropogenic flow alterations on freshwater ecosystems in China and worldwide. Additionally, robust relationships can be developed by including more data points for the whole range of changes in flow regimes, particularly the alterations with respect to low to moderate range.

The final step of this research was to determine the responses of fish species richness to the impacts of flow alterations in China. linear relationships between fish extinction rate and impacts of five flow indicators (i.e. long-term average annual discharge, statistical low flow Q_{90}, statistical high flow Q_{10}, seasonal flow amplitude and seasonal regime) in 34 river basins and/or sub-basins in China were developed based on the fish data, which were reported by 49 published Chinese studies. As only a few observed hydrological data could be extracted from published papers, all flow indicators included in this study were calculated based on discharge data which was simulated by applying an improved version of global hydrological and water use model WaterGAP.

Reference and impact conditions for both fish diversity and flow metrics were compared. The results clearly indicated that long-term average annual discharge (I_{LTD}) was an important flow indicator in quantifying the responses of fish species to flow alterations in China. Changes of fish species richness were positively correlated to changes (original changes) in long-term average annual discharge, while other indicators implicated in this analysis were not able to provide any meaningful information due to high degree of multicollinearity. Indicator of I_{LTD} was dominant over the other flow indicators included in this analysis.

Furthermore, two datasets which include the number of native freshwater fish species at two or three time periods and dynamics of the five flow indicators in 34 river basins and/or sub-basins in China were created. The fish diversity dataset was integrated based on the fish records which were extracted from 49 published Chinese literature, while the flow dataset was generated according to the river discharges that were simulated by the global hydrological model WaterGAP.

The study could not provide clear evidences for quantitative estimation of relationships between reduction of fish species richness and changes in flow components other than average discharge mainly due to three aspects of reasons: 1) the fish species data might not be precise enough; 2) the WaterGAP model cannot simulate monthly discharge accurately and 3) inappropriate application of flow indicators that were highly collinear. However, it did not imply that such relationships would not be detected by using other indicators. Thus, some better flow indicators that represent regimes in other flow components and are not highly collinear should be taken into account in future studies.

All data points included in this study were with respect to alterations in flow magnitude. However, in reality, aquatic and riparian species are influenced by multiple hydrological drivers simultaneously. Thus, the magnitude-oriented flow indicators might add bias to quantitative analysis and lead to overestimation of the impacts of those indicators on freshwater fish species. Therefore, impacts of other flow indicators, such as low flow and high flow duration, frequency and rate of change should be considered in future research in determination of further hydro-ecological relationships by using an improved hydrological model, which can generate daily discharge data in a good manner. Moreover, environmental factors, e.g. pollutant concentrations and sediment discharge could be taken into account in future analysis by conducting more extensive literature review on published and unpublished studies in China. Such an approach has the potential to provide environmental flow guidelines for the sustainable water resources management in rivers with high risk of diversity loss in China.

Zusammenfassung

In den letzten sechzig Jahren haben die übermäßige Wassernutzung und der Bau von Staudämmen das natürliche Durchflussregime in Fließgewässern und die Süßwasserökosysteme in China stark beeinflusst und somit zu schweren Umweltproblemen geführt. Um den Wasserbedarf von Mensch und Umwelt gleichermaßen zu befriedigen sowie die Wissensgrundlagen für ein nachhaltiges Wassermanagement zu schaffen, war eine Bewertung der anthropogenen Änderungen der Durchflussdynamik und deren Auswirkungen auf Süßwasserökosysteme und die Ufervegetation in China erforderlich.

Ziel dieser Forschungsarbeit war die Entwicklung quantitativer Beziehungen zwischen anthropogenen Änderungen der Durchflussdynamik und deren ökologischen Auswirkungen auf Süßwasserökosysteme und die Ufervegetation von Fließgewässern in China. In einem ersten Schritt wurde das Ausmaß der anthropogenen Änderungen des Durchflussregimes quantifiziert. Zu diesem Zweck wurde eine umfassende Abschätzung der Durchflussänderungen, insbesondere der veränderten Durchflussmenge, in Fließgewässern in ganz China infolge von Wasserentnahmen und Staudämmen mithilfe einer verbesserten Version des globalen Wassernutzungs- und Wasserressourcenmodells WaterGAP durchgeführt. In dem Modell sind 731 Stauseen und 2 bewirtschaftete natürliche Seen in China vorhanden. Es wurden fünf ökologisch relevante Durchflussindikatoren unter natürlichen und anthropogen veränderten Durchflussbedingungen berechnet und verglichen. Die Ergebnisse zeigten, dass der gesamte jährliche Durchfluss, der in den Ozean, in Inlandssenken und in Nachbarländer abgeleitet wurde, um 6 % abgenommen hat. Der langjährige mittlere Durchfluss und der statistische monatliche Niedrigwasserdurchfluss Q_{90} haben sich auf 25 % bzw. 35 % der Landflächen Chinas vor allem durch Bewässerung um mehr als 10 % verringert. Der statistische monatliche Hochwasserdurchfluss Q_{10} verringerte sich aufgrund der Bewirtschaftung von Stauseen auf 31 % der Landflächen, während sich Q_{90} auf 12 % der Landflächen im Unterstrom von Stauseen erhöhte. Auf nur 3 % der Landflächen ist eine Erhöhung von Q_{10} infolge der Rückflüsse von entnommenem Grundwasser zu verzeichnen. Auf einem Drittel der Landflächen hat sich die saisonale

Durchflussamplitude stark verringert, während auf zwei Fünfteln der Landflächen deutliche Änderungen des saisonalen Durchflussregimes zu verzeichnen waren. Generell traten in vielen Regionen Nordchinas starke Änderungen des Durchflussregimes auf, wohingegen in den meisten Teilen Südchinas eher geringe Änderungen zu verzeichnen waren.

In den meisten Fließgewässern in China traten bis zu einem gewissen Grad Durchflussänderungen auf. Obwohl die Verringerung des mittleren Durchflusses insgesamt relativ gering ist, wurden die Niedrig- und Hochwasserdurchflüsse sowie die saisonale Durchflussvariabilität durch Wasserentnahmen und Staudämme stark beeinflusst. Da diese Durchflussänderungen die Süßwasserökosysteme in China stark beeinträchtigt haben können, sollten bei der zukünftigen Bewirtschaftung der Fließgewässer nachhaltige Bewirtschaftungsmaßnahmen berücksichtigt werden.

Nachdem zunächst die Auswirkungen menschlicher Aktivitäten auf das Durchflussregime bestimmt wurden, erfolgte im nächsten Schritt eine Quantifizierung der Auswirkungen des veränderten Durchflussregimes auf die Süßwasserökosysteme in China. Hierfür wurden 61 veröffentlichte chinesische Studien, die den Wasserbedarf aquatischer Ökosysteme und/oder nachhaltiges Wassermanagement behandelten, detailliert ausgewertet. Für elf Flusseinzugsgebiete in China wurden hieraus die hydrologischen und ökologischen Daten unter Referenz (die frühesten Aufzeichnungen, die in den Studien berichtet wurden) und anthropogen veränderten Bedingungen zusammengetragen. Diese Daten ermöglichten eine erstmalige Abschätzung quantitativer Beziehungen zwischen anthropogenen Durchflussänderungen in China und deren ökologischen Auswirkungen. Die Literaturrecherche ergab, dass die meisten ökologischen Beeinträchtigungen auf Änderungen der Durchflussmenge, insbesondere auf eine Verringerung des mittleren Durchflusses, zurückzuführen sind. Die meisten ökologischen Gruppen wie Fische, Makrophyten und Ufervegetation reagierten negativ auf Veränderungen der Durchflussdynamik, während sich eine Verringerung des Durchflusses bei Plankton und Wasservögeln positiv ausgewirkte. Für die Änderungen der Durchflussdynamik und die ökologischen Auswirkungen auf Fische, Plankton und die Ufervegetation wurden lineare Beziehungen entwickelt. Sowohl die Diversität als auch die relative Abundanz von Fischen (Gewicht oder Menge des Fischfangs) verringerten sich infolge der geringeren Durchflussmenge in China. Generell konnten ca. 40 % dieser Änderungen durch Änderungen des mittleren Durchflusses sowie der Niedrig- und Hochwasserdurchflüsse erklärt werden (r^2=0,43). Des Weiteren konnten Verringerungen der Fischdiversität und des Fischfangs um 6,9-99,9 % auf eine Abnahme der Durchflussmenge um 4,8-92 % zurückgeführt werden, während bei einem Anstieg des mittleren Durchflusses um 8,4 % eine Erhöhung der relativen Abundanz von Fischen um 1,8 % zu verzeichnen war.

Die Vegetationsdecke und die Biomasse der Ufervegetation reagierten größtenteils negativ auf eine Verringerung der Durchflussmenge, während sich eine Erhöhung des mittleren Durchflusses positiv auf die Vegetationsdecke und die Wachstumsrate auswirkte. Mehr als 60 % der Veränderungen der Ufervegetation konnten auf eine Änderung der Durchflussmenge zurückgeführt werden. Generell weisen die Ergebnisse darauf hin, dass Durchflussverringerungen um 12-89 % wahrscheinlich Verringerungen die Vegetationsdecke und der Ufervegetation um 4-90,3 % hervorgerufen haben, während Anstiege des mittleren Durchflusses um 26,4-171 % zu einer Erhöhung die Vegetationsdecke und der Wachstumsrate der Ufervegetation um 2,5-172,2 % geführt haben.

Zwischen Veränderungen des Planktons und der veränderten Durchflussdynamik wurde kein klarer Zusammenhang festgestellt. Die Diversität und die Abundanz der meisten empfindlichen Planktonarten sanken infolge von verringerten oder erhöhten Durchflüssen, während einige tolerante Arten deutlich positiv (113-2354 %) auf einen verringerten Hochwasserdurchfluss (um 12-83%) und einen erhöhten Niedrigwasserdurchfluss (um 6 %) reagierten.

Da auf Grundlage der Literaturrecherche nicht für alle ökologischen Indikatoren allgemeine Beziehungen zu Änderungen der Durchflussdynamik von Fließgewässern in China entwickelt werden konnten, wurde eine ergänzende Analyse durchgeführt, die Reaktionen von spezifischen ökologischen Gruppen auf klimabedingten und anthropogenen Änderungen der Durchflussdynamik. Es wurden hierin quantitative Beziehungen zwischen Veränderungen der Ufervegetation und der Fischpopulation und Änderungen des mittleren Durchflusses in ariden und semi-ariden und/oder humiden Regionen untersucht. Die Ergebnisse zeigten eine deutliche Korrelation (r = 0,79) zwischen der Ufervegetation und den Änderungen des mittleren Durchflusses in ariden und semi-ariden Gebieten, wobei 63 % der Änderungen der Vegetationsdecke durch Änderungen des mittleren Durchflusses erklärt werden konnten. Des Weiteren konnten ca. 53 % bzw. 58 % der Schwankungen der Fischfangmengen in ariden und semi-ariden sowie in humiden Gebieten auf Veränderungen des mittleren Durchflusses und des hohen und niedrigen Durchflusses zurückgeführt werden. Gemäß diesen Ergebnissen reagieren Fische empfindlicher auf Durchflussänderungen als andere Arten. Die Ergebnisse dieser Studie weisen darauf hin, dass die ökologischen Auswirkungen der Änderungen der Durchflussdynamik in ihrer Art und in ihrem Ausmaß sehr stark von den Eigenschaften der ökologischen Gruppen sowie der Art der Durchflussänderungen abhängen. Daher könnten die funktionalen Beziehungen zwischen Durchflussindikatoren und Beeinträchtigungen bestimmter ökologischer Gruppen oder Arten eine bessere Quantifizierung der Reaktion von Süßwasserökosystemen auf anthropogene Änderungen der Durchflussdynamik in China sowie auf globaler Skala ermöglichen. Ferner könnten robuste funktionale Beziehungen entwickelt werden, indem mehr Datenpunkte einbezogen werden, wodurch die gesamte Bandbreite der Änderungen des Durchflussregimes, insbesondere geringe bis mittlere Änderungen, erfasst werden kann.

Abschließend wurden die Reaktionen Der Fischartenreichtum auf verschiedene Änderungen der Durchflussdynamik in China bestimmt. Da Fisch empfindlich auf Änderungen des Durchflussregimes reagieren, und die Fischdiversität als guter Indikator für Langzeiteffekte gilt, wurden lineare Beziehungen zwischen der Aussterberate von Fischen und den Auswirkungen mehrerer hydrologischer Größen (Der langjährige mittlere Durchfluss, der statistische monatliche Niedrigwasserdurchfluss Q_{90}, der statistische monatliche Hochwasserdurchfluss Q_{10}, der saisonalen Durchflussamplitude und der saisonalen Durchflussdynamik) in 34

Flusseinzugsgebieten und/oder Teileinzugsgebieten in China basierend auf Daten aus 49 veröffentlichten chinesischen Studien entwickelt. Da aus diesen Studien nur wenige hydrologische Messdaten entnommen werden konnten, beruhen die Durchflusswerte der vorliegenden Studie auf Modellergebnissen einer verbesserten Version des globalen Wasserressourcen- und Wassernutzungsmodells WaterGAP.

Sowohl die Fischdiversität als auch die Durchflusscharakteristika wurden unter Referenz und anthropogen veränderten Bedingungen verglichen. Die Ergebnisse zeigten deutlich, dass der langjährige mittlere Durchfluss war ein wichtiger Indikator für die Quantifizierung der Reaktionen die Fischvielfalt auf die Änderungen der Durchflussdynamik in China. Der Rückgang des Fischartenreichtums hat eine positive Korrelation mit den Verlust der langjährigen mittleren Durchfluss, während andere in der Analyse einbezogen Indikatoren konnten keine aussagekräftigen Informationen bestimmen aufgrund der zu hohen Multikollinearität. Der Indikator I_{LTD} war dominierend über die anderen Indikatoren in der Analyse.

Des weiteren wurden zwei Datensätze generiert. Ein Datensatz beinhaltete die Diversität der einheimischen Süßwasserfische in zwei oder drei Zeitabschnitten basierend auf Daten aus 49 veröffentlichten chinesischen Studien. Der andere Datensatz enhält die Modellergebnisse des globalen hydrologischen Modells WaterGAP sowie daraus errechnete Dynamik der fünf hydrologischer Größen.

Es konnte keinen klaren Zusammenhang festgestellt werden für die quantitative Beziehungen zwischen Rückgängen der Fischartenreichtum und Veränderungen in mehrerer Flusskomponenten außer dem mittleren Durchfluss hauptsächlich wegen drei Gründe: 1) die Daten über Fischarten sind möglicherweise nicht genau genug; 2) das WaterGAP modell kann die monatlichen Abflüsse nicht genau simuliert werden und 3) die unangemessenen Anwendung von hydrologischen Größen, die sehr collinear sind. Das bedeutet jedoch nicht, dass solche Beziehungen nicht durch die Verwendung anderer Indikatoren nachgewiesen werden. Daher sollten einige bessere hydrologischer Größen, die nicht sehr kollinear sind, in zukünftigen Studien berücksichtigt werden könnten.

Alle Datenpunkte in dieser Studie bezogen sich auf Änderungen der Durchflussmenge. In der Realität werden aquatische Ökosysteme und die Ufervegetation jedoch durch eine Vielzahl von hydrologischen Parametern beeinflusst. Die Menge-orientierte Indikatoren könnten zur Voreingenommenheit auf die quantitative Analyse führen und zu einer Überschätzung der Auswirkungen dieser Indikatoren auf Süßwasserfischarten führen. Daher sollten in zukünftigen Analysen zur Bestimmung weiterer hydro-ökologischer Beziehungen andere Indikatoren wie beispielsweise die Dauer von Niedrig- und Hochwasserdurchflüssen, und die Häufigkeit und Änderungsrate von Durchflussänderungen berücksichtigt werden, indem ein verbessertes hydrologisches Modell angewendet wird, das tägliche Durchflusswerte auf bessere Art und Weise generiert. Des Weiteren sollten Umweltfaktoren wie z.B. die Schadstoffkonzentrationen und des Sedimenttransport in der Analyse berücksichtigt werden durch Literaturrecherche auf veröffentlichte und unveröffentlichte Studien in China. Solche Ansätze haben das potenzial um Richtlinien zur Umweltfluss für das nachhaltige Wasserresourcenmanagment in Flüssen mit hohem Risiko vom Diversitätenverlust in China bereitzustellen.

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Jing Zhang
Frankfurt am Main
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Chapter 1: Introduction

1.1 Background

Freshwater ecosystems, including rivers, lakes, wetlands, floodplains and estuaries, which provide essential services for human well-being (Millennium Ecosystem Assessment, 2005) and habitat for over 100,000 species (Hawksworth and Kalin-Arroyo, 1995), have been degrading more rapidly than terrestrial or marine ecosystems (Jenkins, 2003; Sala et al., 2000). On a global scale, between 1970 and 2000, population of freshwater species (combined in the Living Planet Index) declined by 50%, while marine and terrestrial species both declined by around 30%. Moreover, about 20% of the world's 10,000 described freshwater fish species have been listed as threatened, endangered, or extinct in the last few decades (Millennium Ecosystem Assessment, 2005). The main impacts on global inland water biodiversity can be characterized in five aspects: overexploration, water pollution, destruction or degradation of habitat, invasion by exotic species and flow alteration (Dudgeon et al., 2006). Among these factors, alterations in flow regimes due to climate change and human activities are considered to be the most critical threats to the ecological sustainability of rivers (Bunn and Arthington 2002) and will further affect freshwater biodiversity in the future. Flow regime is regarded as a master variable (Power et al., 1995) in determining biotic composition, diversity and processes within riverine ecosystems (Richter et al., 1996; Poff et al., 1997; Arthington and Pusey, 1993). Surface water and groundwater withdrawals, dam construction, water diversion and land exploration are the major drivers of alterations in flow regimes. Globally, several studies have quantified ecologically relevant flow alterations due to dams and human water use. Döll et al. (2009) demonstrated that long-term average annual discharge has decreased by more than 10% on one sixth of the global land area, and consequently, the number of fish species decreased by at least 10% on 10% of the total land area. Vörösmarty et al. (2010) indicated that 65% of global river discharge, and associated aquatic habitat, are under moderate to high threat as result of impoundment and depletion of river flows.

In the past 60 years, anthropogenic hydrological changes, such as reductions of river discharge, flow stabilization and shift in low flow or high flow duration resulting from human water use and dam operation, have profoundly influenced freshwater ecosystems in China. According to ICOLD (1998), between 1949 and 1990, the number of large dams, with a dam height above 15m, increased from only eight to more than 19,000 in China, and have caused great modification in physical and chemical environment, to which freshwater species have adapted. Zhao et al. (2008) reported that population size of the finless porpoise has dramatically decreased from 2700 to 1800 between 1990s and 2006 in the Yangtze River, while population of the Yangtze River dolphin has significantly reduced from 400 to 150 between 1980 and 1990 (Ellis et al., 1993) mainly due to impacts of dams. In addition, by 2000, approximately 22% of total wetland area in China has disappeared (An et al., 2007), and more than 10% of the 860 recorded freshwater fish species have been listed as endangered (China's Red Data Book of Endangered animal: Pisces, 1998). Zhao et al. (2007) reported that fish diversity in Baiyangdian wetland of the Haihe River has declined from 54 to 27 between 1958 and 2007 as the result of reduced river discharge. Moreover, seven native fishes have been extinct in the upper reaches of the Yellow River due to dam construction (Zhang et al., 2009). According to the former studies, China is facing perilous freshwater crisis and requires solutions in allocating uneven water resources to social and environmental purposes. Thus, in order to balance human and environmental water demands, to protect ecological functions associated to hydrology, and to adjust human activities within the proper limits, assessment with respect to impacts of anthropogenic flow alteration on freshwater ecosystems in China is urgently needed. Clear and transferable relationships between ecological responses and river flow regimes can provide guidelines to sustainable water management and contribute to the conservation and restoration of important ecological elements in regulated flow systems in China.

Globally, many researchers have provided recommendations for environmental flow determination and have increasingly recognized the importance of the 'natural flow paradigm', which indicates that full range and natural variability of flow regimes should be maintained to protect native biodiversity and provide ecological services (Richer et al., 1996; Poff et al., 1997; Lytle and Poff, 2004). However, translating general hydro-ecological principles and knowledge into specific management rules for particular river basins remains a daunting challenge (Arthington et al. 2006). Therefore, environmental flow assessment should consider the entire natural flow variability in terms of different flow indicators, instead of focusing on habitat requirements for specific species (Arthington et al., 2012) and determination of minimum flows. A comparison between natural flow regimes and anthropogenically altered flow regimes can provide an indication for quantifying the degree of human impacts on freshwater ecosystems, and a number of ecologically relevant flow indicators that reflect the well-being of the biotic components of the freshwater ecosystem are required. The Indicators of Hydrologic Alteration (IHA) method of Richter et al. (1996) has been broadly applied for its ability in characterizing flow alterations, by using a suite of 32 ecologically relevant hydrological indicators, which include alterations in flow magnitude, duration, frequency and rate of change. The Dundee Hydrological Regime Alteration Method (DHRAM) of Black et al. (2005) applies IHA method further to rank the risk of damage to river ecosystems using a five-class scheme.

Many case studies compile a wealth of information on various ecological responses result from different types of flow alterations (Bunn and Arthington, 2002). The information provides a foundation for determining a general understanding on quantitative relationships between hydrological changes and responses of aquatic ecosystems (Poff et al., 2003). Such understanding is needed to define ecological limits of flow alteration and support guidelines for development of flow standards to rivers which lack sufficient data (Arthington et al., 2006; Poff et al., 2010). Approaches that concern developing hydro-ecological relationships based on data extracted from published studies have been performed by several researches. Bunn and Arthington (2002) reviewed a number of literatures worldwide and summarized four principles regarding influence flow regimes on aquatic biodiversity. Lloyd et al. (2003) reviewed 70 studies and indicated that 86% of these studies demonstrated ecological responses to reduction in flow magnitude. Poff and Zimmerman (2010) extensively reviewed 165 papers globally and found that 92% studies reported negative responses to flow alterations, while 13% of them reported positive responses. Unfortunately, either simple thresholds (Loyd et al., 2003) or general quantitative relationships (Poff and Zimmerman, 2010) were not able to be obtained from current literature review.

Other methods with related to linear relationships between fish species richness and hydrological characteristics were conducted at global scale as well. Xenopoulos et al. (2005) predicted future declines of fish diversity as the result of decreased river discharge. Applying a similar method, Xenopoulos and Lodge (2006) suggested that 20-90% reductions in river discharge would result in 2-38% loss in fish diversity in southern rivers in United States. Iwasaki et al. (2012) performed a statistical analysis on relationships between 14 hydrological metrics and fish species richness in 72 rivers worldwide and indicated that low flow and high flow events are critical indicators that affect fish diversity.

Little information could be extracted from previous studies with respect to ecological responses to anthropogenic flow alterations in China due to lack of sufficient information for both hydrology and ecology. Even though numerous case studies have shown threats of altered flow regimes to specific species in different geographic regions in China, however, at the macroscale, general knowledge on quantitative hydro-ecological relationships in China's freshwater ecosystems is still unknown. Thus, compilation of a body of knowledge on hydro-ecological relationships is required to quantify the degree of hydrological degradation and its effects on freshwater ecosystems, to establish ecological threshold of flow alterations, and to provide environmental guidelines for sustainable water management in China.

1.2 Research questions

The foregoing sector has illustrated the background and the related problems with respect to quantification of ecological responses to anthropogenic flow alterations in

China's rivers, wetland and associated floodplains. The following research questions, which aimed at those problems, were raised and answered during the thesis process.

1. How are natural flow regimes altered in China's rivers and other freshwater bodies?
2. What is known about ecological responses to flow alterations at the macroscale in China?
3. Is it possible to obtain quantitative hydro-ecological functions based on the evaluation of hydrological and ecological data which provided by Chinese studies?
4. Is it possible to develop general relationships between ecological categories and flow metrics in China, according to observations reported in the literature?
5. If general relationships between ecological groups and flow metrics cannot be determined based on current observations, what steps can be taken to quantify hydro-ecological relationships in China?

1.3 Research objectives

The main research objectives of this study were to:

1. Provide a quantitative assessment of how the natural flow regimes in China have been altered by human water use and dams, and degrees of the anthropogenic alterations.
2. Summarize existing information regarding ecological responses to flow alterations in river basins, wetlands and associated floodplains in China through literature review.
3. Develop quantitative relationships between anthropogenic flow alterations and ecological responses in China according to the information that could be extracted from the published literature.
4. Define linear relationships of specific ecological responses (e.g. fish diversity) to anthropogenic flow alterations at sub-basin scale in China.
5. Provide supplementary datasets, which combine hydrological and ecological data of China's river basins for future studies.

1.4 Thesis outline and methodology

The research questions and objectives are addressed and presented in the Chapters 2-4, in addition to the introduction chapter.

To answer the first research question, an assessment of flow alterations due to human water withdrawals and dams in China, with particular emphasis on changes of flow magnitude, was performed in Chapter 2. Moreover, five ecologically relevant flow indicators were identified and then quantified for each 0.5 degree grid cell in China, using an improved version of global hydrological and water use model WaterGAP.

For the research questions two to four, general linear relationships of responses of different ecological categories to a variety of river flow alterations, as well as responses of fish and riparian vegetation to average river discharge at different climate regions in China were developed and analyzed based on the information that could be extracted from 61 published Chinese studies in Chapter 3. Furthermore, a database that combines hydrological and ecological observations, and information of the main drivers of flow alterations and ecological responses, was established.

For the final research question, an evaluation on quantitative relationships of reduction in fish species richness to flow alterations in 34 river basins and sub-basins in China was performed in Chapter 4. Four flow metrics that are of specific relevance for biodiversity were quantified using the global hydrological model WaterGAP, while fish diversity for different time periods in 34 river basins were obtained from 49 published Chinese studies. Additionally, a dataset that integrates historical records of fish species richness as well as simulated flow metrics at basin and sub-basin scale in China were provided.

In Chapter 5, the main findings and the contributions of the previous chapters were summarized, together with the future research direction.

Chapter 2: Assessment of ecologically relevant anthropogenic flow alterations in China

Abstract

As China's economy booms, increasing water use has significantly affected hydro-geomorphic processes and thus the ecology of surface waters. A large variety of hydrological changes arising from human activities have been sustained throughout China and resulted in severe ecological degradation. In order to balance the water requirements between human and ecosystems and provide knowledge on sustainable water management, general information on anthropogenically altered flow regimes is needed to define the regions where freshwater ecosystems are to be strongly affected. This study performed a comprehensive assessment of hydrological changes due to water withdrawals and reservoirs for all over China, with particular emphasis on change of flow magnitude. Using an improved version of the global hydrological and water use model WaterGAP, natural and anthropogenically altered flow conditions were calculated for five ecologically relevant flow indicators by taking into account impacts of human water consumption, as well as 731 large reservoirs and 2 regulated lakes. Long-term average river discharge, statistical low flow Q_{90} and high low Q_{10} have decreased by more than 10% on 25%, 35% and 31% of China's total land area, mainly due to irrigation. Q_{90} has increased significantly by 12% of the total land area, downstream of reservoirs, while Q_{10} has increased on 3% of the land area as a result of return flow from groundwater abstraction. Due to both water withdrawals and reservoirs, seasonal flow amplitude has decreased strongly on 30% of China's land area, while seasonal regime has changed on 40% of the total land area in consequence of irrigation and dams. Areas most affected by anthropogenic flow alterations are north-western China, the Liaohe River, the Haihe River, the middle and lower reaches of the Yellow River basin and northeastern part of the Yangtze River basin. These large flow alterations would threaten the sensitive freshwater ecosystems in China and are likely to have caused significant ecological impacts.

2.1 Introduction

Flow regimes play a profound role in determining the biotic composition, structure, function and diversity within river ecosystems (Richter et al., 1996; Arthington and Pusey, 1993). Alteration of flow regimes is often claimed to be the most serious threat to the ecological sustainability of rivers and floodplain wetlands (Bunn and Arthington 2002). In the last 60 years, human-driven flow alterations, such as reduction in river flow discharge and change in seasonal regimes due to human water withdrawals and dam operation, have significantly affected freshwater ecosystems in China. By the year 2000, 21.6% of total wetland area in China has disappeared (An et al., 2007) and more than 10% of the 860 recorded freshwater fish species have been listed as endangered (China's Red Data Book of Endangered animal: Pisces, 1998). Therefore, it is urgent to protect the natural functions of water resources in relation to hydrology, biology and chemistry, and adjust human activities to within the limits of nature and implement the sustainable development impact assessment system in the decision-making for water resource management (White Paper on China's Strategy for Population, Environment and Development in the 21st Century, chapter 14, 1994). To fulfill these goals, an assessment of hydrological changes due to human impacts that integrates ecological aspects is needed for all of China.

In recent years, the importance of flow variability for river ecosystems has been well documented (Poff and Ward, 1989; Poff et al., 1997; Richter et al., 1996, 1997; Puckridge et al., 1998; Clausen and Biggs, 2000). A "natural flow paradigm" is suggested by accumulated research on the relationship between hydrological variability and river ecosystem, stating that "the full range of natural intra- and interannual variability of hydrological regimes, and associated characteristics of flow magnitude, frequency, duration, timing and rate of change, are critical in sustaining the full native biodiversity and integrity of aquatic ecosystems" (Richter et al., 1997; Poff et al., 1997). Comparing natural river flow regimes with anthropogenically altered river flow regimes can provide an indication for quantifying the degree of human impacts on freshwater ecosystems. A suite of biologically relevant hydrological indicators that reflect the well-being of the biotic components of the freshwater ecosystem are required. The Indicators of Hydrologic Alteration (IHA) approach of Richter et al. (1996) has been widely adopted because of its comprehensive ability to characterize ecologically relevant hydrological changes. In this method, two sets of flow time series representing natural and altered conditions at the same site are compared using 32 indicators spanning the five characteristics mentioned above.

Many case studies have shown that how human activities affected river flow regimes at basin scales, but only a few regarding their impacts on the aquatic components of freshwater ecosystem. According to those studies, the main drivers of ecologically relevant flow alterations in China can be grouped under three categories: withdrawals of surface and groundwater, dam construction and loss of channel-floodplain connectivity. Low instream flow and groundwater depletion due to excessive water withdrawals have caused negative impacts on aquatic ecosystems in northern and western China. In the Haihe River basin, the annual average discharge into ocean was decreased by 95.8% from 1950s to 2001 and led to extinction of many estuarine species (Xia et al., 2004). In the lower reaches of Tarim River basin, annual river discharge has dropped by 59% from 1958 to 1978 due to irrigation water use, thus resulted in severe groundwater depletion and 69% loss of Populus cover (Feng et al., 2005). Over construction of dams has greatly affected seasonal and interannual flow variability, with negative impacts on biodiversity in river and riparian ecosystems (Poff et al., 2007). According to ICOLD (1998), the number of large dams (with a dam height of more than 15 m or have a storage capacity of more than 3 million m^3) in China increased from only eight to more than 19,000 from 1949 to 1990. Those large dams have caused significant changes in physical and chemical environment, to which freshwater fish species have adapted. WWF (2004) reported that frequent and large variation in water levels in the middle and lower reaches of the Yangtze River are reduced after dam construction and discharge regulation. In consequence, population size of the finless porpoise *Neophocaena phocaenoides*, the only freshwater adapted porpoise, has dramatically decreased from around 2700 in the early 1990s (Zhang et al, 1993) to 1800 in 2006 (Zhao et al, 2008). The Yangtze River dolphin or Baji (*Lipotes* *vexillifer*), which is the most threatened cetacean in the world, is also at risk. By 1980 an estimated 400 individuals remained and by 1993 only 150 remained with their range substantially reduced (Ellis et al., 1993). Channel-floodplain disconnection has negatively affected biodiversity of lakes and wetlands in some river basins in China. During 1950s-1970s, sluice gates were constructed in almost all lakes, which were interlaced with the mainstream and tributaries of the Yangtze River and brought about the decline in natural fish stocks in the river and associated wetlands, especially the decline in species richness and abundance of migratory fish (Xie and Chen, 1999). Migratory fish abundance of Lake Bohu in the lower reaches of the Yangtze River decreased from 56% of the total catch before the building of sluice gates in 1956 to 20% of the total catch after the gates construction (Zeng, 1990).

In order to balance the water requirements between human and freshwater ecosystems and provide a guideline for sustainable water management in China, general knowledge on quantification of anthropogenically altered flow alterations is required. Döll et al. (2009) conducted an analysis on river flow alterations due to water withdrawals and reservoirs for global scale (including China). Natural and anthropogenically altered flow regimes were compared for six hydrological indicators. The results show that northern China has been the most affected area in the world mainly due to water withdrawals. However, due to lack of capacity on modeling groundwater withdrawals, decrease of river discharge was overestimated in semi-arid area and northeastern China.

In this study, a comprehensive assessment of ecologically relevant hydrological alterations due to human water use and reservoirs, with emphasis on changes of flow magnitude, was performed for all over China. Due to lack of consistent and reliable observed data, both natural and anthropogenically altered flows were simulated by an improved version of Global Hydrological and Water Use Model WaterGAP (Alcamo et al., 2003; Döll et al., 2003; Döll and Fiedler, 2008; Döll et al., 2012), which takes into account groundwater abstraction, as well as 731 large reservoirs and 2 regulated lakes in China. The anthropogenic flow alterations were described by changes in the long-term average discharge, statistical low flow Q_{90}, high flow Q_{10}, seasonal regime and seasonal amplitude for each 0.5 degree grid cell.

2.2 Methods

2.2.1 Simulation of natural and anthropogenically altered flows using WaterGAP: model description and data preparation

In order to analyze the impacts of human water withdrawals and reservoirs in China, an improved version of Global Hydrological and Water Use Model WaterGAP 2.2 was used to compute natural flow (NAT) and anthropogenically altered flows (ANT) for the time period of 1971-2000 overall China. With a spatial resolution of 0.5° by 0.5° (55 km by 55 km at the equator), the WaterGAP model simulates water availability and human water use globally excluding Antarctica (Alcamo et al., 2003b). It combines a global hydrological model (WGHM) and several water use models, which compute water withdrawals and consumptive water uses in sectors of irrigation, household, manufacturing, cooling for thermal power plants and livestock. WGHM, in the standard approach, is driven by daily reanalysis-based WFD/WFDEI climate data, i.e. a combination of the daily WATCH Forcing Data based on ERA40 for the year 1901-1978 (WFD), and the WATCH Forcing Data based on ERA-Interim for the year 1979-2009 (WFDEI). and is calibrated against long-term average river discharge at 1319 stations world-wide, by adjusting 1–3 model parameters individually in each of the 1319 upstream basins (Müller Schmied et al., 2014). In the former version of WGHM 2.1g (Döll et al., 2009; Döll et al., 2010), groundwater withdrawals were not considered due to lack of knowledge on which part of the water use coming from groundwater or surface water, thus all water was withdrawn from surface water resources. For proper estimating the impact of surface water and groundwater withdrawals on water flows, WGHM 2.2 included a new model component GWSWUSE, which calculates the total net water abstraction from groundwater and from surface water in each 0.5° grid cell, based on sectorial water withdrawals and consumptive use as computed by the five water use models (Döll et al., 2012).

An updated reservoirs and regulated lakes dataset was implemented in WGHM2.2.

The dataset was derived by adding 5733 additional reservoirs from the GRanD database (Lehner et al., 2011) to the 886 reservoirs that were included in previous version WGHM 2.1h. Since GRanD data does not distinguish between regulated lakes and reservoirs, all reservoirs with an area larger than 100 km^2 or a maximum storage capacity of at least 0.5 km^3 were checked to decide whether they are regulated lakes. Reservoirs of this size are defined as “global” in WGHM and they are fed by river discharge from the upstream cells, while smaller reservoirs are defined as “local” and they are only fed by the runoff generated within the grid cell. As a result, 6619 reservoirs and 43 regulated lakes were included in WGHM2.2 at global scale.

The reservoirs and regulated lakes dataset used in this study contains 731 reservoirs and 2 regulated lakes throughout China (Table 2.1 and Fig. 2.1). All reservoirs are scattered in the Heilongjiang River (Chinese part of the Amur River), the Liaohe River, the Luanhe River, the Haihe River, the Yellow River, the Yangtze River and the Pearl River and most of the reservoirs are located in north-eastern and southern China. The Three Gorges Dam is also included in this dataset. The surface area of the reservoirs and regulated lakes is 11014.3km^2 and 12204.6km^2. Maximum storage capacity of reservoirs and regulated lakes is 434.2km^3 and 28.9km^3 (Table 2.1).

Table 2.1 Reservoirs and regulated lakes included in this study.

	Number			Surface area (km^2)			Storage capacity[1] (km^3)		
	WG2.1f[2]	WG2.2[3]		WG2.1f[2]	WG2.2[3]		WG2.1f[2]	WG2.2[3]	
		reservoirs	regul. lakes		reservoirs	regul. lakes		reservoirs	regul. lakes
China	47	731	2	435.2	11014.3	2204.6	166.2	434.2	28.9
Global	886	6619	43	254301	296811.4	182005.5	4642.0	6061.4	8557.4

[1]At global scales, only 672 reservoirs in ‘WG2.1f’ were used to calculate the storage capacity because of lack of data for storage capacity.

[2]Included in previous WGHM 2.1f (Hunger and Döll, 2008)

[3]Included in current WGHM 2.2 (Döll et al., 2012)

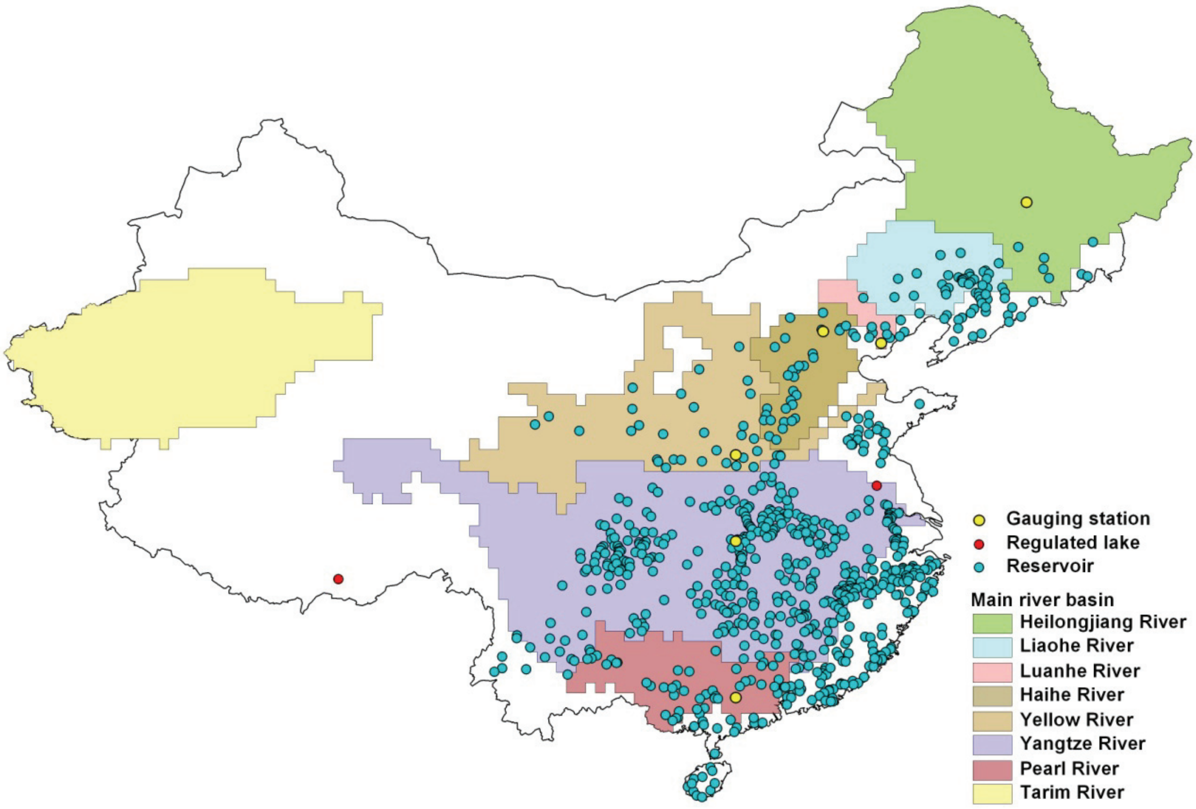

Fig. 2.1. Distribution of reservoirs, regulated lakes, major river basins and gauging stations included in this study.

2.2.2 Development of indicators of river flow alterations

Five ecologically relevant indicators of river flow alterations that represent changes in flow magnitudes (Table 2.2) were taken from the indicator set of Döll et al. (2009) and the Indicators of Hydrologic Alteration (IHA) approach of Richter et al. (1997). The indicator set of Döll et al. (2009) was developed based on the IHA indicators and the Dundee Hydrological Regime Alteration Method (DHRAM) of Black et al. (2005), for the purpose of analyzing the impact of anthropogenic flow alteration on freshwater ecosystems worldwide. Most of the IHA indicators rely on daily discharge data, which cannot be calculated by the previous version of WaterGAP (2.1g) model driven by monthly climate input data. Although the current version WGHM2.2 can generate daily water flows, but how well do observed and modeled results match has not been tested. Therefore, only indicators on the basis of monthly and annual discharge data were considered.

Table 2.2 Five ecologically relevant indicators of river flow alterations included in this study.

Indicators	Flow characteristics	Description	Ecological relevance
I_{LTD}	Long-term average annual discharge	percent change in long-term average annual river discharges between anthropogenically impacted and natural conditions	fish species richness[1], floodplain vegetation
I_{Q90}	Statistical low flow	percent change in Q_{90} (monthly river discharge that is exceeded in 9 out of 10 months) between anthropogenically impacted and natural conditions	habitat conditions, connectivity of channel or floodplain
I_{Q10}	Statistical high flow	percent change in Q_{10} (monthly river discharge that is equaled or exceeded for 10% of the specified term) between anthropogenically impacted and natural conditions	habitat conditions, species richness[2], floodplain vegetation
I_{SA}	Seasonal amplitude	percent change in seasonal amplitude (maximum minus minimum long-term average monthly river discharge) between anthropogenically impacted and natural conditions	habitat availability in particular on floodplains
I_{SR}	Seasonal regime	mean over 12 monthly values of absolute differences between long-term average monthly river discharges under anthropogenically impacted and natural conditions , in % of natural discharge	habitat conditions, compatibility with life cycle of organisms

[1]Xenopoulos et al. (2005)

[2]Poff and Zimmerman (2010)

2.2.3 Specification of model runs

Time series of two monthly discharge datasets ANT and NAT were simulated by WGHM2.2 over the time period 1971-2000 per 0.5 degree grid cell for the whole of China. The datasets were generated to quantify the indicators of river flow alterations mentioned in Sector 2.2.2. In the analysis, ANT represents the river flow regime affected by human water withdrawals as well as by reservoirs and regulated lakes. NAT was computed by assuming that, in WGHM model run, there are no human water withdrawals and all reservoirs are removed, while all regulated lakes are not treated as reservoirs but as natural lakes.

2.3 Results

2.3.1 Anthropogenic alteration of long-term average annual discharge

Total annual river discharge into oceans and internal sinks as well as discharge at international boundary for the whole of China was calculated to be 3445.25km³/year under anthropogenically altered condition (ANT), compared to 3666.78km³/year under natural condition (NAT). River discharge in China has decreased by 6% due to water withdrawals and reservoirs.

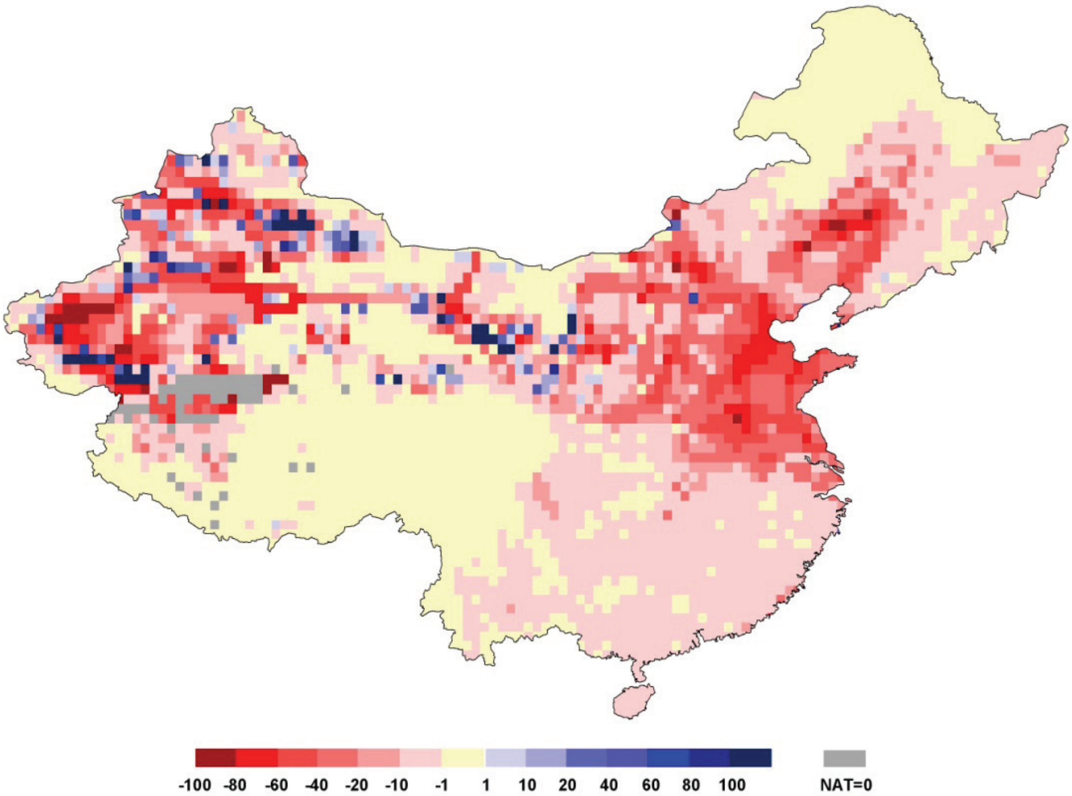

Fig. 2.2. Alteration of long-term average annual river discharge due to water withdrawals and reservoirs as compared to natural discharge in China, in % of natural flows.

Fig. 2.2 shows anthropogenic alterations of long-term average annual discharges (1971–2000) due to water use and dams in China per 0.5 degree grid cells. The most affected regions are northeastern part, northwestern part, northern part and eastern part. In many grid cells in the Liaohe River basin, the Haihe River basin, the Luanhe River basin, middle and lower reaches of the Yellow River Basin, northeastern part of the Yangtze River basin and Tarim River basin, annual river discharge under ANT has reduced by more than 40 % as compared to NAT. Those significant alterations are contrasted by the only small reductions of less than 10% in the Heilongjiang River basin and in the southern part of China, including most of the Yangtze River basin, the Pearl River basin and the southern parts of western China. In some cells in upper reaches of the Yellow River basin and northwestern part of China, river discharge has increased more than 20% due to return flows from irrigation. Generally, long-term average annual discharge has decreased by more than 10% within 25% of China's total land area. In the regions where the river discharge decreases more than 40%, riparian vegetation and aquatic animals are likely to have been severely affected.

2.3.2 Anthropogenic alteration of statistic monthly low flow Q_{90}

Statistical monthly low flow Q_{90} has strongly decreased in large parts of China as a result of reservoir and water withdrawals, but also increased along some rivers. In Fig. 2.3, Q_{90} is significantly decreased by more than 60% in many grid cells in the central Heilongjiang River basin, the Liaohe River basin, the middle and lower reaches of the Yellow River basin, northeastern part of the Yangtze River basin and the Tarim River basin, mainly due to irrigation water withdrawals.

Increased Q_{90} appears mostly along the rivers where reservoirs or regulated lakes are located in upstream. For the purpose of power generation, flood control or water supply, reservoirs balance river flow by taking in water during high flows and releasing it during low flows. Flow balancing leads to stabilized flow regime and causes increases of Q_{90}. In the Yangtze River downstream of the Three Gorges Dam and several other large dams, the upper reaches of the Yellow River, which is regulated by three large dams, as well as the Pearl River, low flows have increased by more than 10%, 20% and 40%. In some cells in northwestern part of China, Q_{90} has increased due to irrigation return flow.

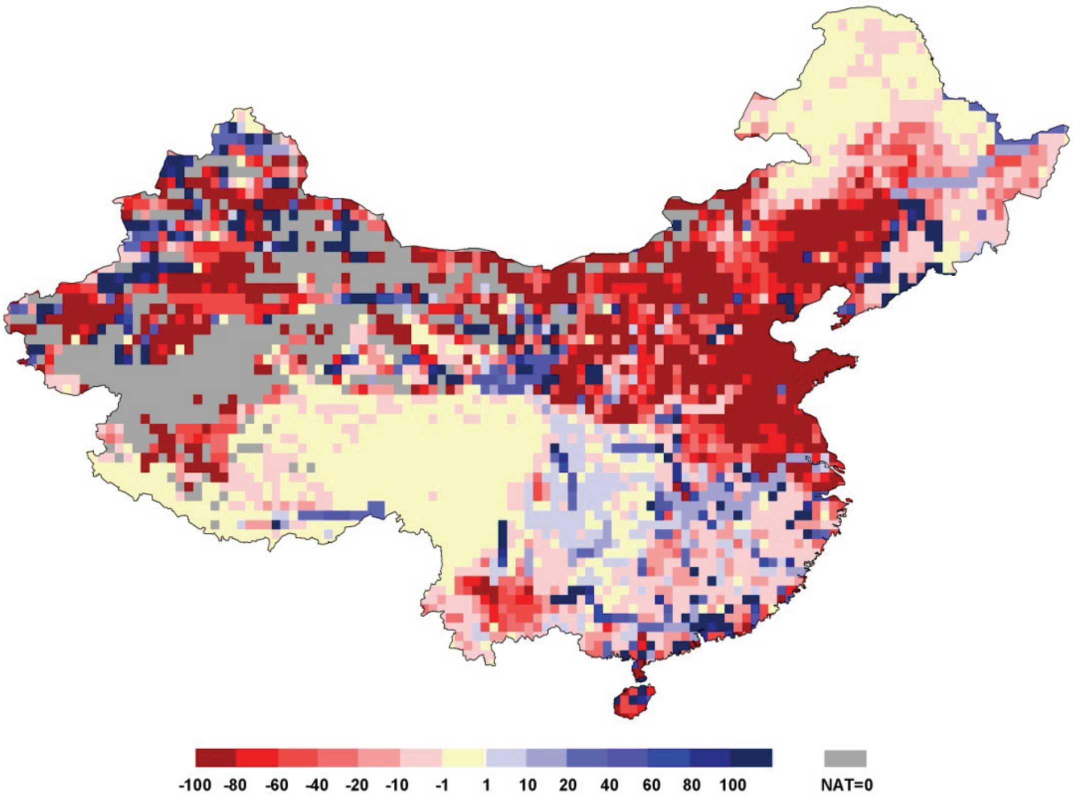

Fig. 2.3. Alteration of statistical monthly Q_{90} due to water withdrawals and reservoirs as compared to natural Q_{90} in China, in % of natural flows.

Totally, Q_{90} has decreased by at least 10% within 35% of China's total land area and has increased significantly by 12% of total land area. The significant changes in low flow may have resulted in negative changes in riparian and aquatic habitats, and thus lead to loss of biodiversity in freshwater ecosystems.

2.3.3 Anthropogenic alteration of statistical high flow Q_{10}

The spatial patterns of changes in high flow Q_{10} are very similar to the patterns of alterations in long-term average annual discharge. The comparison of ANT and NAT (see Fig. 2.4) expresses significant decreases in the Liaohe River basin, the Haihe River basin, the Yellow River basin, the northeastern part of the Yangtze River basin, the Tarim River basin and the northwestern China, due to both reservoirs and high consumptive water use. Q_{10} has decreases in some cells in the Pearl River basin as a consequence of dam operation.

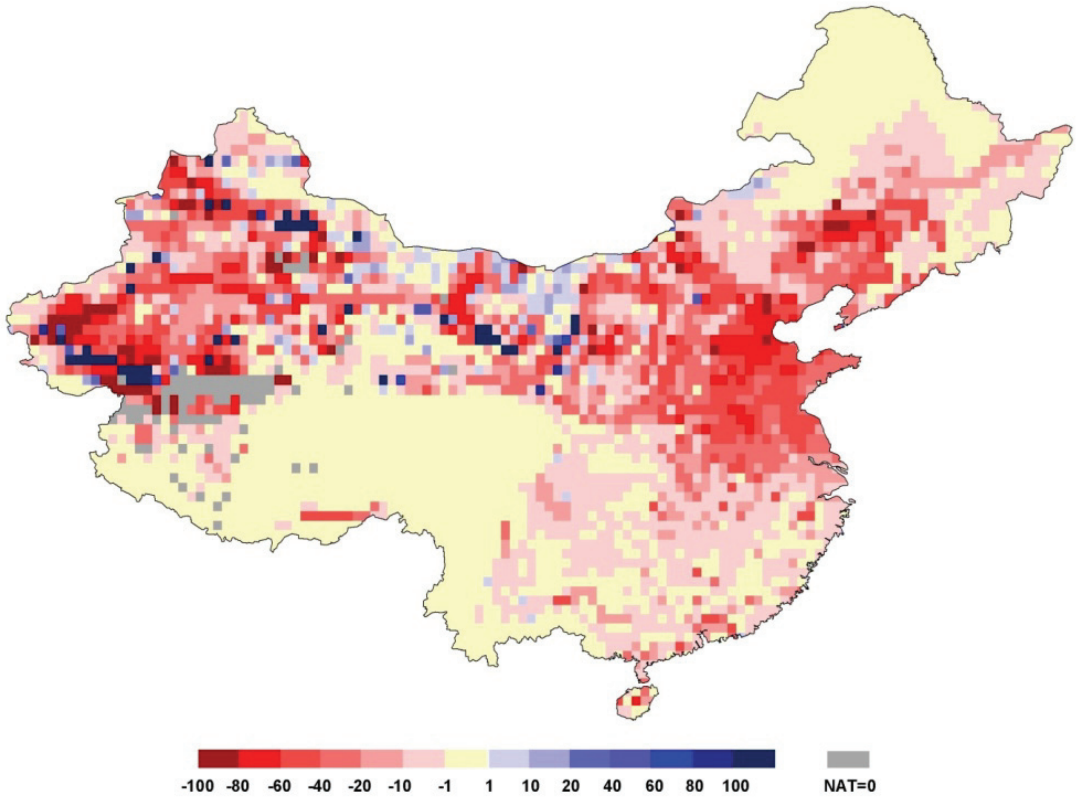

Fig. 2.4. Alteration of statistical monthly Q_{10} due to water withdrawals and reservoirs as compared to natural Q_{10} in China, in % of natural flows.

At macroscale level, high low Q_{10} has decreased on 31% of China's total land area by at least 10%, and has increased on 3% of the land area as a result of return flow from groundwater abstraction. As some native fish species rely on high flows during spring to start migration and spawning, reduced seasonal high flows can severely affect their breeding. Thus, large changes in Q_{10} are likely to have caused declines in the fish abundance and diversity in China.

2.3.4 Anthropogenic alteration of seasonal flow amplitude

Due to water withdrawals and reservoirs, the difference between the maximum and the minimum long-term monthly discharge is decreased downstream of the reservoirs and the regulated lakes, and in regions with high consumptive water use (Fig. 2.5). It increases in regions with low consumptive water use. Outflow reduction by dams and high water withdrawals during high flow seasons leads to reduced high flow, and thus results in decreased seasonal flow amplitude. Such flow stabilization may have negative impacts on flora and fauna of riparian and aquatic ecosystems.

Generally, seasonal flow amplitude has strongly reduced on 30% of China's land area by more than 10%, and has increased by at least 10% on 3% of total land area.

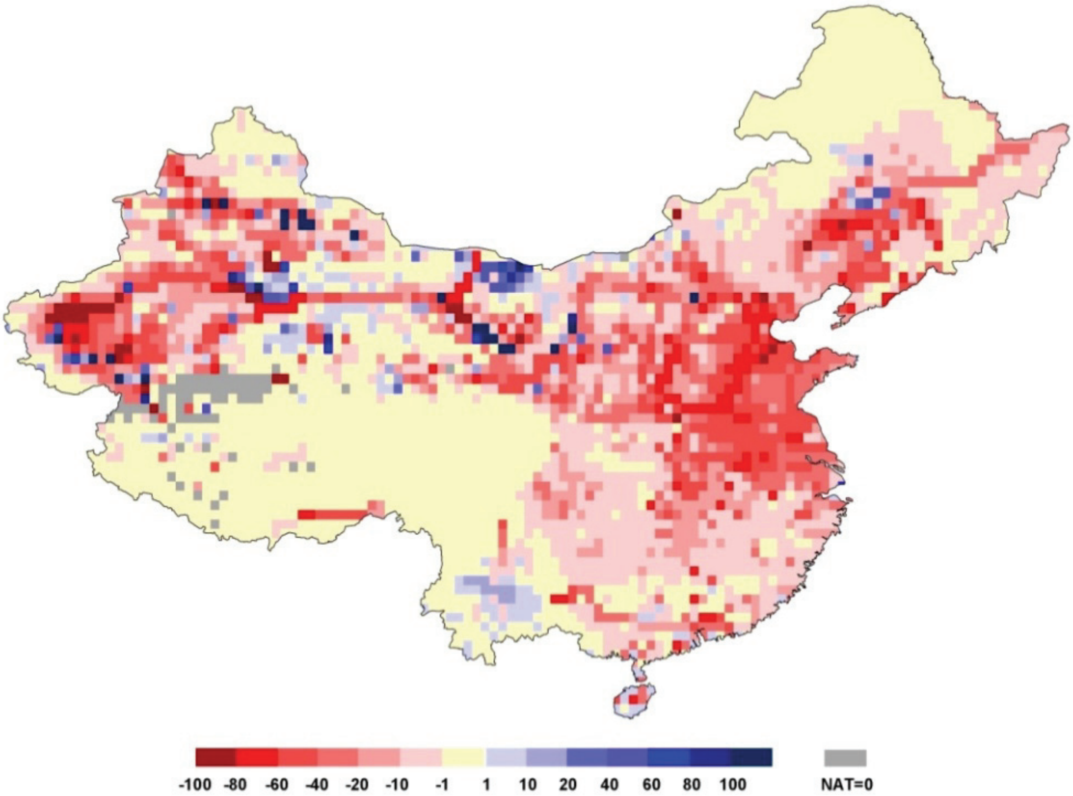

Fig. 2.5. Alteration of seasonal amplitude due to water withdrawals and reservoirs as compared to natural amplitude in China, in % of natural flows.

2.3.5 Anthropogenic alteration of seasonal flow regime

Change in seasonal flow regime reflects how the seasonal variability of the monthly discharge is affected. It considers not only alterations of extreme flows, but also the anthropogenic alterations of all twelve long-term average monthly river discharge values. Figure 2.6 shows the spatial patterns of anthropogenic changes of seasonal flow regime in China. Natural seasonal flow variability has been significantly changed by more than 10% on 40% of China's total land area due to high water withdrawals and reservoirs, and thus leads to negative impacts on habitat availability and the compatibility with the life cycle of riparian and aquatic organisms.

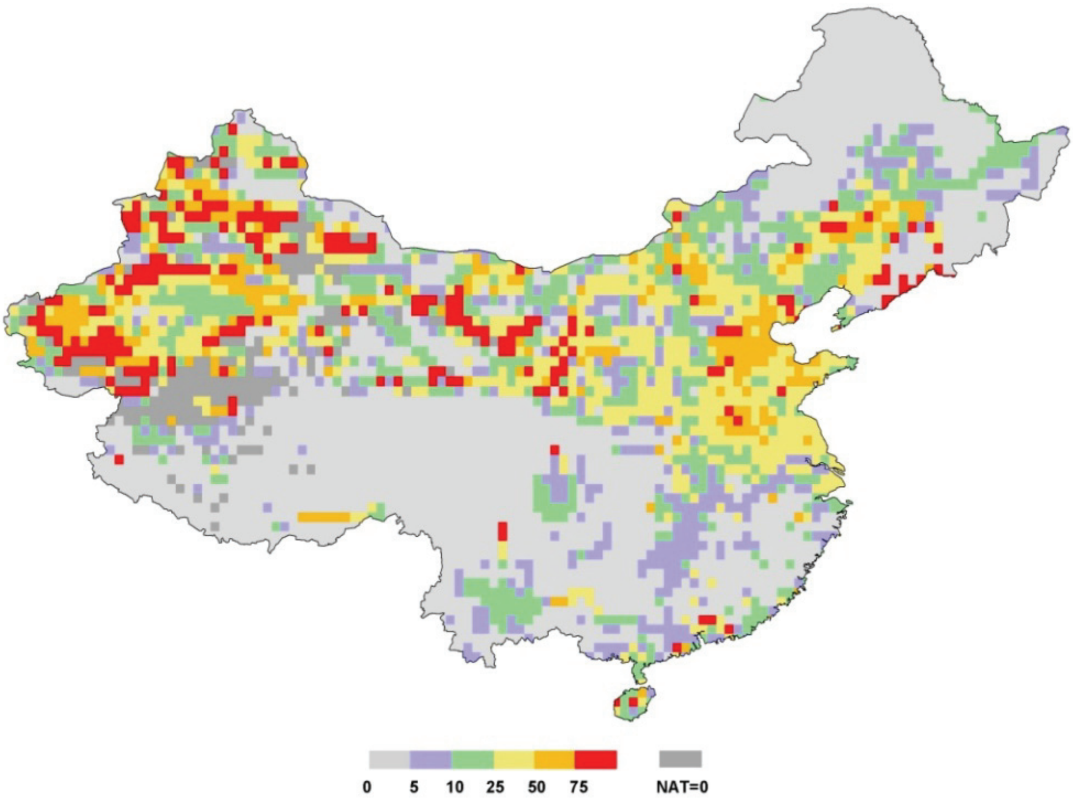

Fig. 2.6. Alteration of seasonal regime due to water withdrawals and reservoirs as compared to natural regime in China, in % of natural flows.

2.4 Discussion

The study indicated that natural flow regimes has been significantly modified in areas with high water withdrawals (i.e. irrigation areas) and downstream of reservoirs in China. The results are consistent with the findings in Döll et al. (2009) who analyzed flow alterations due to water withdrawals and reservoirs using WGHM2.1g at global scale. In order to test how well the improved version of WGHM2.2 is able to estimate the impacts of reservoirs and water use on river discharge, modeled and observed mean monthly river discharges at six gauging stations in China were compared. All six stations (Luanhe at Luanxian, Xijiang at Wuzhou3, Yangtze at Yichang, Songhuajiang at Haerbin, Yellow at Sanmenxia and Yongding at Guanting) are located in the regions with high consumptive water use and/or downstream of large dams.

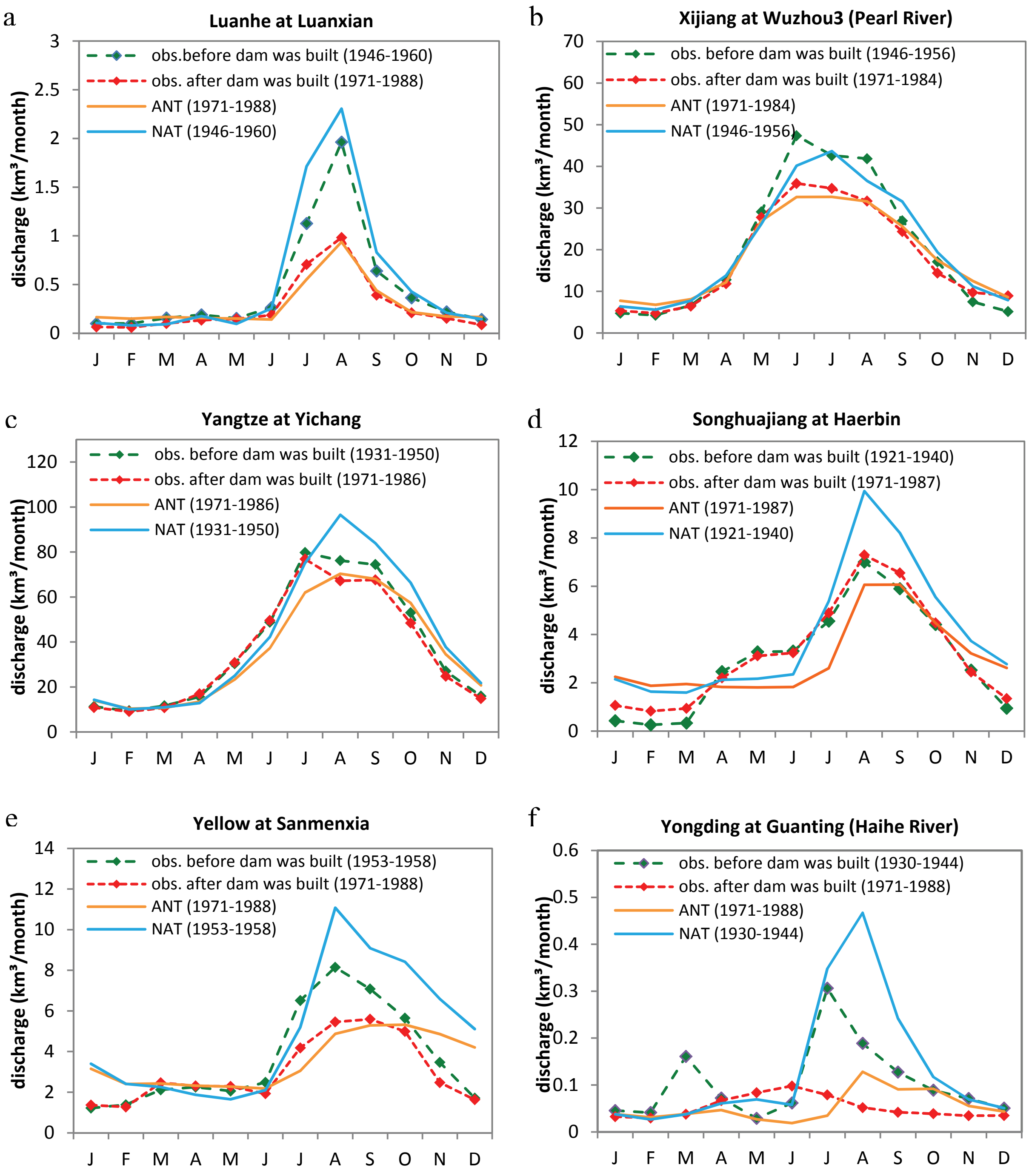

Fig.2.7. Long-term average monthly river discharge at six selected gauging stations: comparison between observed and modeled anthropogenically altered (ANT) and natural (NAT) conditions. Discharge observations were obtained from the Global Runoff Data Centre (www.bafg.de).

For the Luanhe River at Luanxian station, modeled natural mean monthly discharge (NAT) for the time period of 1946-1960 (the time before dam construction) was compared with values that were observed during the same time period. In addition, simulated anthropogenic discharge (ANT) was compared to observational data for the same time period 1971-1988, i.e. for the time after dam construction. Fig. 2.7a shows that seasonal variations in discharge before and after dam construction are captured quite well by WGHM2.2. Anthropogenically altered discharge in dry season (December to March) is slightly overestimated, while the simulated peak flows under natural condition are higher than the observations. Before dam construction, both simulated and observed discharges have much higher seasonal variability than anthropogenically altered discharges, with a peak in August. Therefore, for this station, WGHM can estimate seasonal amplitude and seasonal regime correctly.

For the Xijiang River at Wuzhou3 station, the observed high flows occurs from June to August during the time period 1946-1956 before dam construction, which were modeled rather well by NAT, even though river discharge is underestimated in June and August, and is somehow overestimated from September to April (Fig. 2.7b). For the time period 1971-1984 after dam construction, ANT captures the seasonal variations better than NAT, although the peak flow is slightly underestimated in June. As compared to NAT, discharge of ANT increases by 10% from November to March, and decreases by 20% from June to August. The comparison indicates that the impact of reservoir on discharge is strong in wet season and small in dry season at this station.

The seasonal variability of the natural and the anthropogenically impact river discharge is captured well by WGHM at Yangtze River at Yichang even through the modeled peak flow occur one month later than the observations (Fig. 2.7c). The natural peak flow is somewhat overestimated by the model and thus results in an overestimation in seasonal amplitude, while the seasonal regime of the river discharge is estimated quite well by WGHM at this station although the flows in dry season (November to January) is somehow overestimated.

Similar to the condition for Luanxian, natural and anthropogenically altered seasonal peak flows for the Songhuajiang at Haerbin are well modeled by WGHM as well as the seasonal variations (Fig. 2.7d). Low flows during the period of November to March under ANT and NAT conditions are overestimated, while the peak flow under ANT condition is underestimated. Seasonal amplitude of observed discharge is rather small at this station; however, simulated seasonal amplitude is much higher than actual.

The natural peak flow occurs in August, which is well modeled for the Yellow River at Sanmenxia, while the anthropogenically altered discharge peaks one month later than the observed one (Fig. 2.7e). The seasonal variability of both ANT and NAT is correctly simulated by WGHM even though the low flows are somewhat overestimated from November to February.

The hydrographs for the Yongding River at Guanting (Fig. 2.7f) show that WGHM simulates the low seasonal variability of anthropogenically impacted discharge well even though natural seasonal peak flow is overestimated. The maximum natural discharge occurs one month later as compare to observed value, while the modeled peak flow under ANT occurs two months later than the observation. The observed discharge increases in March before dam construction which is likely due to discharge of snow melting, but this signal is not captured by the model.

Conclusions can be drawn from the analysis of the six stations that WGHM-based analysis estimated the actual anthropogenic impact on seasonal river flow regimes properly and succeeded in catching the seasonal amplitude of the river flows pre- and post-dam construction for the most stations. For five out of six stations, WGHM simulated higher winter low flows and lower summer high flows than the observations. Thus, the model might overestimate the reservoir impact on the river discharge. However, modeling the impacts of reservoirs on discharge dynamics contains many uncertainties. The major uncertainty is associated with the fact that reservoirs are operated in a very site-specific manner in reality, which can hardly be simulated perfectly by implementing a general algorithm in a macro-scale model. In addition, the real number of reservoirs, especially the number of small reservoirs, is much higher than the number of reservoirs included in this study. Regarding the effects of water withdrawals, the anthropogenic flow changes in the semi-arid areas and the eastern part of China, where deep groundwater was highly withdrawn for irrigation purpose, were estimated much better by WGHM2.2 as compare to the results of WGHM2.1g (Döll et al., 2009). In WGHM2.2, water withdrawals are assumed to be taken from both surface water and deep groundwater, such that river flows are increased in those areas due to the return flow to surface water body, while in WGHM2.1g, water withdrawals are taken from surface water or shallow groundwater and all water withdrawals lead to a river flow reduction. In order to test how well the sub-module GWSWUSE calculates groundwater abstraction, modeled groundwater withdrawals for irrigation in six provinces that have large irrigated areas and high groundwater abstraction for the year 2004 and 2005 were compared to an estimation of provincial groundwater withdrawal for agriculture which was provided by Wang et al. (2012) (Fig. 2.8). According to Wang et al, groundwater withdrawals for agriculture for 2004 was calculated based on field survey data, while the values for 2005 were estimated from unpublished and published data in the China Groundwater Level Yearbook from GEO-Environmental Monitoring Institute (China GEO Environmental Monitoring Institute 2006). GWSWUSE computes much higher groundwater withdrawals for irrigation in Hebei and Henan and quite lower values in Liaoning and Shandong as compared to the data provided by Wang et al. (2012). In Beijing and Xinjiang, modeled values are very close to the estimated data. As the estimated data for 2004 were calculated based on surveyed groundwater pumping data at selected villages, the actual groundwater use for irrigation is likely to be underestimated in Hebei and Henan. In addition, water use estimation contains many uncertainties, in particular for irrigation water use, because the location of areas equipped for irrigation is rather uncertain in many areas (Siebert et al., 2005). Therefore, it is difficult to draw a conclusion from the comparison that GWSWUSE module cannot simulate impacts of groundwater abstraction for irrigation in a good manner.

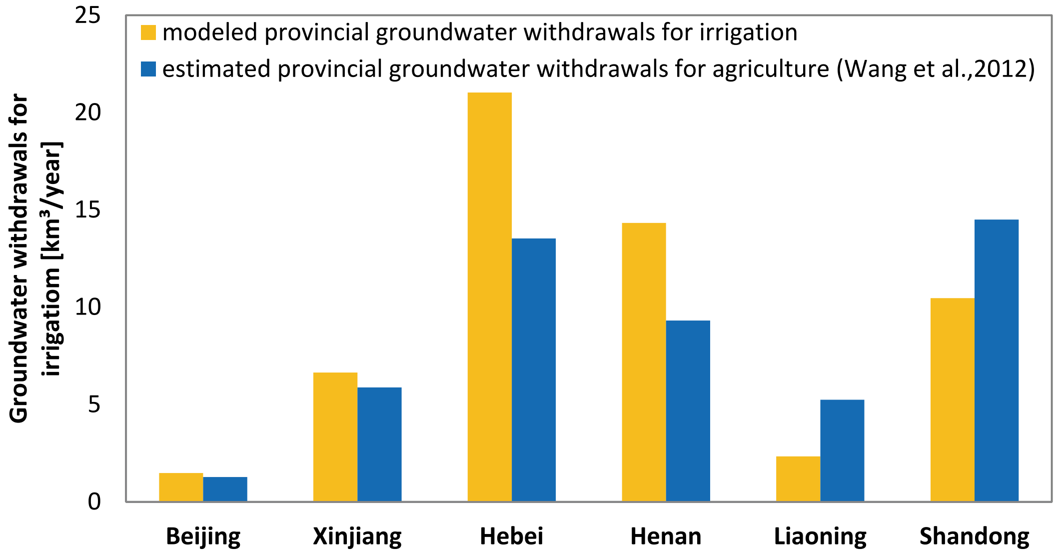

Fig. 2.8. Modeled groundwater withdrawals for irrigation as compared to estimated provincial groundwater withdrawals for agriculture which is provided by Wang et al (2012), in six provinces for the year 2004 (Hebei, Henan, Liaoning, Shandong) and 2005 (Beijing and Xinjiang).

2.5 Conclusion

This study has performed a comprehensive assessment of hydrological changes due to water withdrawals and reservoirs for the whole of China, with particular emphasis on changes of flow magnitude. Five flow indicators that are relevant with the health of the biotic components of freshwater ecosystems in China have been developed. Each indicator represents a type of anthropogenic flow alteration which concerns aquatic organisms in surface water bodies or groundwater-dependent vegetation in floodplain. Using an improved version of the global hydrological and water use model WaterGAP, which takes into account impacts of human water consumption, as well as 731 large reservoirs and 2 regulated lakes, the five ecologically relevant flow indicators were quantified for each 0.5 degree grid cell. Total annual river discharge into oceans and internal sinks as well as discharge at international boundary for the whole of China has been decreased by 6% due to water withdrawals and reservoirs. 25% of China's total land area has suffered strong decrease of long-term average river discharge (more than 10%). Statistical low flow Q_{90} has decreased significantly in 35% of the total land area in China mainly due to water use and has increased by 12% of the total land area downstream of reservoirs. High low Q_{10} has decreased by more than 10% on more than one third of China’s total land area and has increased on 3% of the land area as a result of return flow from groundwater abstraction. Seasonal flow amplitude has decreased significantly on 30% of China’s land area due to both water withdrawals and reservoirs, while seasonal regime has strongly changed on 40% of the total land area in consequence of irrigation.

After identifying anthropogenic flow alterations, quantitative relationships between flow indicators and ecological characteristics of freshwater ecosystems, such as species richness, abundance, assemblage structure and recruitment will be developed to detect limits that would be useful in sustainable water management in China. Poff et al (2003) and Poff et al. (2010) suggested that comprehensive synthesis of case studies can provide generalized quantitative relationships between ecological response and specific type of flow alteration and can also support development and implementation of regional environmental flow standards. Those standards are urgently required to determine an environmental guideline that balances ecosystem and human water requirements in China.

Chapter 3: Developing quantitative relationships between anthropogenic flow alterations and ecological responses in China based on published data

Abstract

In the past decades, a large variety of river flow alterations due to human water use and dam operation have significantly affected biotic processes in river and riparian ecosystems in China and have caused severe environmental problems. In order to define ecological limits of flow alteration and environmental flow guidelines, knowledge on relationships between flow and ecology is needed to quantify the degree of anthropogenic impacts on freshwater ecosystems in China. This study has conducted the first attempt in developing quantitative relationships between river flow alterations and ecological responses in seven main river basins and four watersheds in China based on the data that could be extracted from published case studies with respect to environmental flow or ecosystem management. Quantitative relationships between percent change in flow magnitude (average annual discharge, seasonal low flow and seasonal high flow) and percent change in ecological indicators (fish diversity, fish catch, vegetation cover, vegetation biomass, vegetation growth rate, plankton diversity and abundance) were analyzed based on 190 data points that were extracted from 42 published literature. The results showed that changes in coverage and biomass of riparian vegetation as well as changes in fish diversity and fish catch were strongly correlated with the changes in flow magnitude ($r = 0.77, 0.66$), especially with changes in average river discharge, but no robust relationship was found between flow alterations and plankton response. In the supplementary analysis, 117 data points with respect to changes in riparian vegetation cover and fish catch as consequences of alterations in average annual river discharge were characterized according to classification of two climatic regions (arid and semi-arid region, humid region) in China. The quantitative analysis showed that riparian vegetation cover was highly correlated with the alterations in average river discharge in arid and semi-arid

regions in China ($r = 0.79$). Coefficient of determination ($r^2 = 0.63$) denoted that more than half of the variations in vegetation cover could be explained by changes in average annual river discharge. Fish catches showed robust correlations to alterations in average annual river discharge in both arid and humid regions ($r = 0.78, 0.77$) and roughly 50 % changes in arid and semi-arid region and 60% changes in humid region could be determined by alterations in flow magnitude ($r^2 = 0.53, 0.58$). Vegetation and fish responded sensitively when river discharge has been changed. Based on current literature review, riparian vegetation cover and fish catch might be reasonable ecological indicators in developing quantitative relationships between flow alterations and ecological changes in China.

3.1 Introduction

Inland water ecosystems, including rivers, lakes and wetlands, provide numerous services for human well-being, such as fresh water, food, maintenance of fisheries and biodiversity, recreation, scenic values, and ecosystem function (Millennium Ecosystem Assessment, 2005). As a master variable (Power et al., 1995), flow regime determines basic ecological characteristics of riverine ecosystem (Poff et al., 1997). Full range and natural variability of flow regimes are the key points in maintaining native biodiversity and ecological processes (Richer et al., 1996; Poff et al., 1997). During the past six decades, flow alterations as consequences of human water use and reservoir operation have significantly affected biodiversity and biotic processes in riverine ecosystems in China and worldwide. Many case studies provide a wide range of information on various ecological responses to different type of flow alterations (Bunn and Arthington, 2002), which supports a general understanding on quantitative relationships between changes in flow and responses of aquatic ecosystem (Poff et al., 2003). Such understanding is needed to define ecological limits of flow alteration and environmental flow guidelines, and to quantify the degree of anthropogenic-induced impacts on freshwater ecosystems in China and worldwide.

Determining hydro-ecological relationships requires integration of hydrological and ecological datasets, which provide sufficient information for statistical analysis, nevertheless spatial datasets that cover both hydrologic and ecological data are not available in many regions (Kight et al., 2008). An approach that concerns extracting information from published studies regarding hydro-ecological relationships through extensive literature review has been conducted by several studies. Bunn and Arthington (2002) selectively reviewed some literature worldwide and illustrated four principles with respect to impacts of altered flow regimes on aquatic ecosystem. Lloyd et al. (2003) reviewed 70 peer-reviewed and unpublished studies and reported that 86% of the studies documented ecological changes with related to decrease in flow magnitude. Poff and Zimmerman (2010) extensively reviewed 165 papers at global scale and found that 92% studies reported decreases in ecological metrics in response to anthropogenically altered flows, while 13% of them reported increased numbers. A conclusion were drawn by the latter studies that simple thresholds (Loyd et al., 2003) or general quantitative relationships (Poff and Zimmerman, 2010) could not be developed from current literature review. Another approach that considered linear relationships between fish species richness and hydrological metrics was conducted globally. Xenopoulos et al. (2005) predicted future losses in fish species richness as a consequence of reduced river discharge using a linear regression model, which included published fish data (Oberdorff et al., 1995; Froese and Pauly, 2000) and modeled river discharge, and found that fish diversity would be reduced more than 75% by 2070 due to climate change and human water use. Using a similar method, Xenopoulos and Lodge (2006) estimated that 20-90% decreases in river discharge would cause 2-38% reduction in fish species richness in two regions in United States. Iwasaki et al. (2012) conducted statistical analysis on relationships between fish species richness and 14 hydrological metrics and indicated that low flow and high flow could be important indications that influenced fish diversity.

Previous studies mentioned above did not support enough information regarding ecological responses to anthropogenic flow alterations in China mainly due to lack of baseline data. Although many case studies showed impacts of hydrological changes on various specific ecological characteristics in different geographic regions in China, however, at the macro-scale, general knowledge on quantitative hydro-ecological relationships and thresholds is still lacking. In order to determine ecological threshold of flow alteration and to support environmental flow management, compiling information on relationships between flow and ecology is needed to quantify the degree of anthropogenic-induced impacts on freshwater ecosystems in China. In this study, the first attempt in developing general quantitative relationships between ecological responses and anthropogenic flow alteration in China was conducted based on the information that could be extracted from published case studies with respect to environmental flow and ecosystem management. In addition, a database that focused on all sources of information, such as data of hydrological and ecological metrics that were reported in the literature, was developed.

3.2 Methods

3.2.1 Data construction

A total of 61 studies that provided ecological responses in aquatic or riparian ecosystems to anthropogenic flow alterations were reviewed. Most of the studies were conducted in arid and semi-arid watersheds in northwestern China (18) and in the Yangtze River (16), 6 regarding the Haihe River, 5 representing the Yellow River, 5 about the Huaihe River, 4 from the Heilongjiang River (Chinese part of the Amur River) and only 1 from the Pearl River. Of all sources, 10 were published in international journals and the rest 51 were published in Chinese journals. Ecological responses reported in the studies were categorized into seven ecological groups: fish, riparian vegetation, macrophyte, plankton, bird, macroinvertebrate and mammal (Fig. 3.1). Each group was again characterized by types of responses, such as diversity, abundance and growth rate. Fish was the predominant ecological group in most papers (34), followed by riparian vegetation (19), macrophyte (9), plankton (8) and bird (6). Several studies reported responses of macroinvertebrate (3) and mammal (1) to changes in river flow regimes.

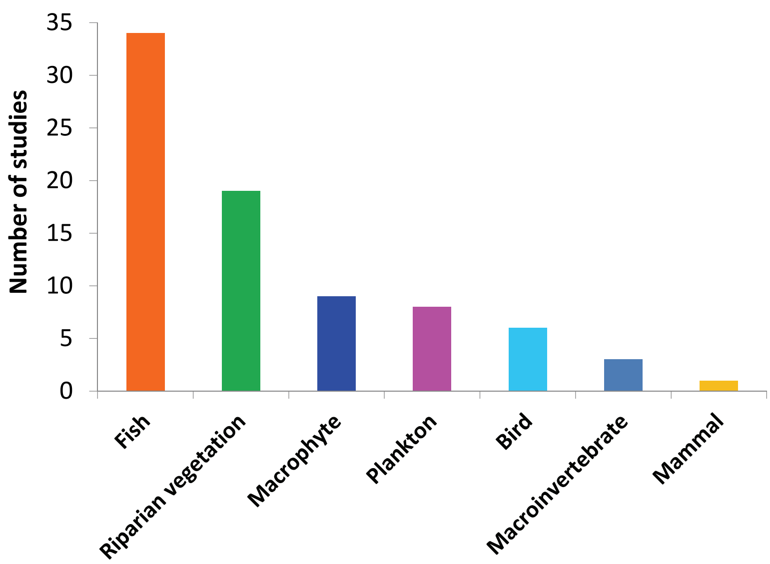

Fig. 3.1. Number of studies that reported responses of different ecological groups to anthropogenic flow alterations. Some studies presented more than one ecological group and therefore the number of papers adds up to more than 61.

Anthropogenic flow alterations associated with different drivers (e.g. water withdrawals, dam operation and water diversion) were categorized according to magnitude, duration and rate of change. For papers that reported multiple flow components, only the primary flow component was considered in this study. 92% of papers focused on flow alterations with respect to changes in magnitude, which included alterations in average river discharge (56%), low flow (21%), high flow (30%) and seasonal variability (10%), and only a few studies reported changes in duration (7%) and in rate of change (5%) (Fig. 3.2). Dam construction was the predominant driver of flow modifications in the reviewed literature (66%), while other flow alterations were caused by human water withdrawals (26%), water diversion (7%) and river cutoff (only one study). Hydrological changes in 13 papers were resulted from both reservoir operation or impoundments and excessive water withdrawals.

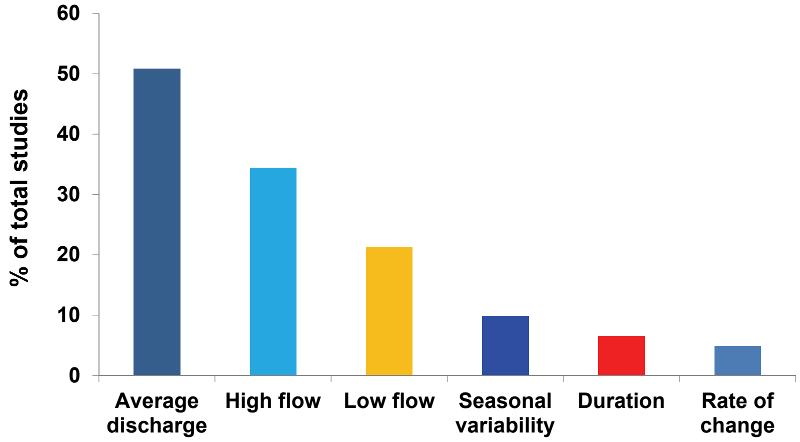

Fig. 3.2. Frequency of ecologically relevant primary flow components reported in 61 literatures.

3.2.2 Data analysis

3.2.2.1 Analysis of responses of different ecological groups to flow alterations in China

In order to determine whether the published literature could provide enough data, which was required in developing relationships between flow alterations and ecological changes, studies that reported quantitative changes in both flow regimes and ecological groups were identified. In these studies, hydrological records and ecological variables under natural (or reference) and altered conditions have to be reported. Natural conditions referred to the time period of records before human impacts occurred, and for the studies that reported measures of ecological changes after anthropogenic impacts occurred, the earliest hydrological and ecological records were adopted as the reference conditions. For ecological data that was collected within a short time period (e.g. in a specific month), the data was considered as the record for the entire year. Normally, period-of-record flow regimes overlapped with time period of ecological records, but in some studies, time period of flow records were longer or shorter than that of ecological data. Nevertheless, in these cases, hydrologic records were assumed to be reasonable to formulate relationships between flow and ecology due to lack of accurate data. For variables that were reported as ranges, mean values of the ranges were adopted in this study. Similar to Poff and Zimmerman (2010), alterations in flow regimes and ecological groups in each study were presented as percent changes between reference and altered conditions.

Of all 61 papers, only 21 of them supported quantitative changes in both flow regimes and ecological components. In order to increase the sample size, 21 studies that only reported ecological data in quantitative units were included by obtaining necessary flow data from additional 15 studies and GRDC Runoff Data Center regarding the same time periods and study sites (See Appendix A1 for the complete summary of anthropogenic flow alterations and hydrological data based on a literature review of 61 published papers and 15 additional studies in China). A total of 42 papers provided 190 data points that represented various ecological responses to flow alterations, and among them, 8 papers reported changes in multiple ecological groups. For all of 42 papers, some of them reported changes in more than one flow components, but only the primary type of flow alteration was considered. Of the 190 data points, 187 were with respect to changes in flow magnitude and the rest three were spilt on changes in duration and rate of change. Data points regarding changes in flow magnitude were grouped into the following three sub-categories: average discharge (113 points), low flow (18 points) and high flow (56 points). Due to lack of variables for duration and rate of change, those two flow components were not included in this study. Therefore, flow alterations were characterized as percent changes in any of those three sub-categories. Ecological responses were expressed as percent changes in different types of response (abundance, diversity or growth rate) of the following six ecological groups: fish, macroinvertebrate, riparian vegetation, macrophyte, plankton and bird (See Appendix A2 for detailed information of ecological responses to anthropogenic flow alterations and ecological data based on a literature review of 61 published papers in China). As only a few data was available for macroinvertebrate (3 points), macrophyte (8 points) and bird (8 points), they were not considered in data analysis.

One of the goals of this study was to define whether a linear or non-linear relationship existed between anthropogenic flow alterations and ecological response that were recorded in the literature. To do this, percent change in flow magnitude and percent change in different ecological groups were included in simple regression models, where ecological responses were treated as the dependent variables and flow changes were referred as the independent variables.

3.2.2.2 Analysis of responses of riparian vegetation and fish to flow alterations in different climatic regions in China

Magnitude and direction of ecological responses to flow alterations depended largely on characteristics of ecological groups and types of flow alterations (McManamay et al., 2013). Different ecological groups may have diverse responses to the same type of hydrological change (e.g. plankton and riparian vegetation have both negative and positive responses to reduced seasonal high flows in China). Thus, general relationships could not be derived between altered flow regimes and all types of response in entire six ecological groups. In this study, fish and riparian vegetation were selected as the optimum ecological indicators, because both of them could provide enough data for analysis and were sensitive to flow alterations (Poff and Zimmerman, 2010). Responses of fish were presented as percent change in weight or amount of fish catch, while reactions of riparian vegetation were expressed as percent change in riparian vegetation cover. Flow alterations referred to percent change in average annual river discharge. As a result, 117 out of 187 data points with respect to changes in riparian vegetation cover (34 points) and fish catch (83 points) as consequences of flow alterations were selected and spilt into 11 river basins and watersheds in China (Fig. 3.3, see Appendix B1, B2 and B3 for detailed information).

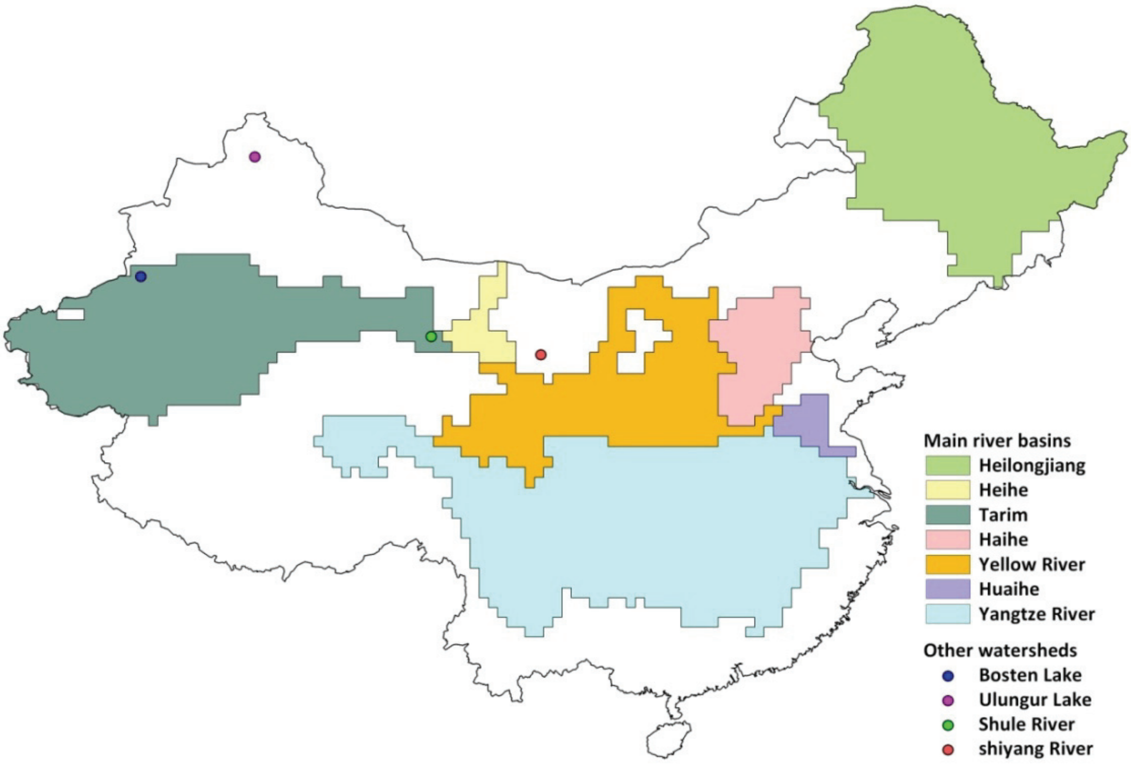

Fig. 3.3. Main river basins and watersheds included in the regression analysis.

Other factors such as hydroclimate and geomorphy are expected to affect ecological responses to flow changes as well (Poff and Ward, 1989; Arthington et al., 2006). In China, due to influence of Asian monsoon, annual precipitation decreases from southeast coast to northwest inland at the macro scale and results in uneven distribution of water resources. In arid and semi-arid regions, river discharge is much smaller than the volume of flows in humid regions, thus, natural flow regimes in those regions are very sensitive to climate change and human impacts. As a result, aquatic and riparian ecosystems in arid and semi-arid regions are more fragile than ecosystems in humid regions. Li et al. (2004) reported that natural fish catch decreased 37% from 1950s to 1960s in Haihe River, when annual river discharge decreased only 5%. Liu et al. (1983) showed that 5% decrease in annual discharge led to 17% decrease of fish catch in Bosten Lake. In order to analyze responses of fish and riparian vegetation to flow alterations under different climate conditions in China, two climatic regions were characterized according to values of aridity index (ratio of precipitation to potential evapotranspiration). Region with values between 0.05 and 0.5 was classified as arid and semi-arid region, while area with aridity index larger than 0.5 was classified as humid region. 117 data points that represent responses of changes in fish catch and riparian vegetation to flow alterations in 11 river basins and watersheds were finally spilt in two climatic regions (Fig. 3.4).

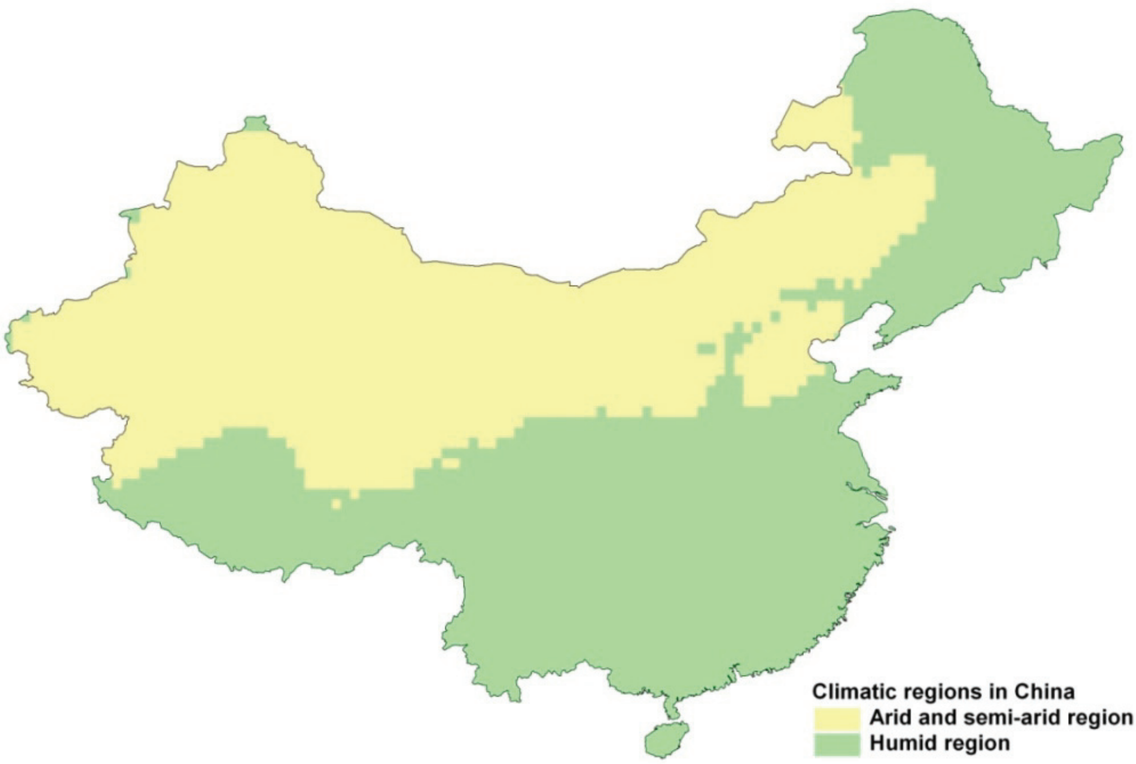

Fig. 3.4. Climatic regions classified in this study.

3.3 Results

3.3.1 Data summary

Data from 61 published studies that were used to evaluate impact of anthropogenic flow alterations on aquatic and riparian ecosystems was summarized in Table 3.1. According to the literature, the majority of ecological changes were caused by modifications in flow magnitude, most commonly as reductions in average river discharge. Ecological responses were largely reported as negative responses for most ecological categories, such as fish, macrophyte and riparian vegetation, while some positive responses to decreased flow regimes were recorded for planktons, birds and riparian vegetation.

Table 3.1 Summary of total studies reporting negative and positive ecological responses to alterations in different flow components due to anthropogenic effects based on a literature review of 61 published papers in China. Some studies reported both negative and positive ecological responses, thus the number of papers adds up to be greater than 61 papers.

Flow component	Organism	Total No. of studies	No. of studies reporting negative ecological response	No. of studies reporting positive ecological response	Primary flow alteration	No. of studies	Ecological response: Increase	Ecological response: Decrease
Magnitude	Aquatic	2 2	22	5	Stabilization (reduced high and/or low flow, increased low flow, decreased seasonal variability)	20	Macrophytes growth rate Macrophytes abundance Diversity and abundance of plankton Abundance of macro-invertebrate Bird diversity	Fish diversity and abundance Shift in fish community composition Fish spawning habitat Reduced reproduction and altered recruitment Change in fish assemblage Disruption of life cycle Diversity and abundance of sensitive macroinvertebrate Diversity and abundance of plankton Macrophytes abundance
					Decreased average river discharge	2		Floodplain connectivity Wetland habitat Fish abundance

Table 3.1 (Continued)

Flow component	Organism	Total No. of studies	No. of studies reporting negative ecological response	No. of studies reporting positive ecological response	Primary flow alteration	No. of studies	Ecological response: Increase	Ecological response: Decrease
Magnitude	Riparian	38	28	8	Stabilization (increased low flow, reduced low flow, decreased water level)	9	Macrophytes diversity and abundance	Macrophytes diversity and abundance Riparian vegetation cover Bird diversity Fish diversity and abundance Change in fish life cycle Loss of floodplain connectivity Diversity of river dolphin
					Decreased average river discharge	25	Plankton diversity Macrophytes abundance	Fish diversity and abundance Change in fish assemblage Bird diversity Plankton diversity Shift in plankton community composition Macrophytes diversity and abundance Macroinvertebrate diversity Altered riparian community composition Loss of habitat Riparian vegetation cover Loss of floodplain connectivity
					Increased average river discharge	4	Riparian vegetation cover Growth rate of riparian forest Vegetation diversity	

Table 3.1 (Continued)

Flow component	Organism	Total No. of studies	No. of studies reporting negative ecological response	No. of studies reporting positive ecological response	Primary flow alteration	No. of studies	Ecological response: Increase	Ecological response: Decrease
Duration	Aquatic	3	3		Decreased peak flow duration Increased inundation duration and water level	3		Loss of floodplain connectivity Loss of fish spawning habitat Reduced recruitment success of lowland river fish Fish diversity and abundance Disruption of fish life cycle Disruption of cues for fish migration and spawning Bird diversity
	Riparian	1			Increased no-flow period Shift in no-flow period	1		Riparian vegetation cover Loss of wetland habitat
Rate of change	Aquatic	1			Increased rise rate	1		Loss of cues for fish spawning and migration
	Riparian	2			Changes in rates of water level fluctuation and disturbance frequency	2		Fish abundance Loss of fish spawning habitat Shift in fish community composition Macrophytes diversity and abundance

3.3.2 Relationships between flow alterations and responses of different ecological groups in China

Fish was the predominant ecological group in most literatures and contributed 95 data points for quantitative analysis. Fish diversity and fish catch reduced consistently in response to decreased flow magnitude. One sample from a study of Bosten Lake showed positive response to increased annual river discharge due to water diversion (Fig. 3.5). The regression analysis showed that diversity and fish catch strongly correlated with changes in flow magnitude ($r = 0.66$). Moreover, coefficient of determination ($r^2 = 0.43$) indicated that about 40% of reduction in fish diversity and fish catch could be explained by alterations in flow magnitude. Decreases of 4.8-92% in flow magnitude could have resulted in losses of 6.9-99.9% for fish diversity and fish catch in China, while an increase of 8.4% in average annual discharge might lead to 1.8 % of increases in fish catch in the Bosten Lake (Xinjiang).

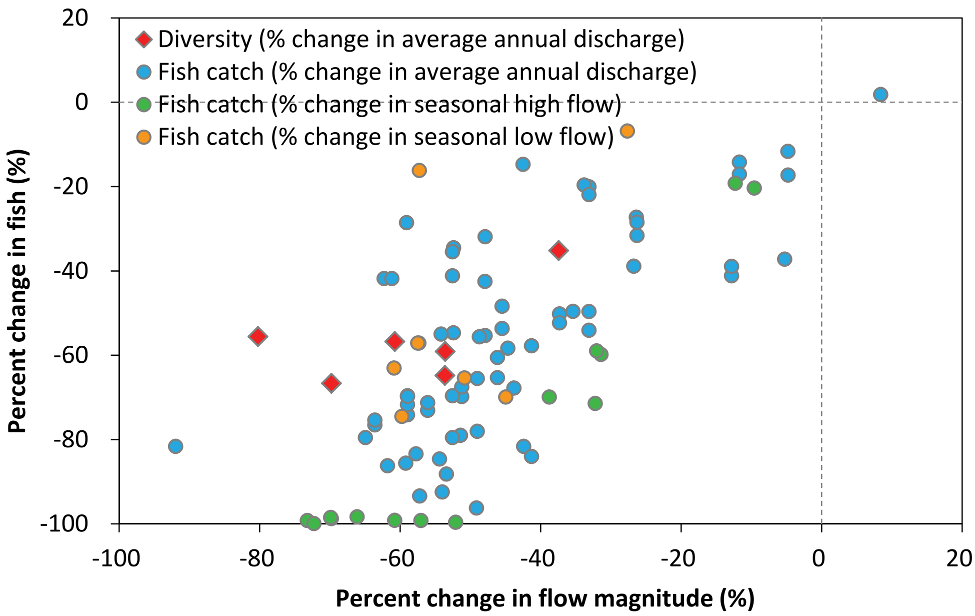

Fig. 3.5. Percent change in fish diversity and fish catch with respect to percent change in flow magnitude in China. Alterations in flow magnitude referred to changes in average annual discharge, seasonal high flow and seasonal low flow. Percent change in responses of fish and flow magnitude was calculated as the difference between impact and reference (the earliest records that were reported in the studies) conditions, in % of reference condition.

Estimation of responses of riparian vegetation to anthropogenically altered river flows were performed for 43 data points. Vegetation cover and biomass of riparian vegetation showed almost negative responses to reduced average discharge. However, a study in Tarim River recorded increased vegetation cover to reduced annual average river discharge. Vegetation cover and growth rate of riparian vegetation positively responded to increased annual river discharge (Fig. 3.6). With a correlation coefficient equals to 0.77, percent changes in coverage, biomass and growth rate of riparian vegetation were highly correlated with the changes in average discharge. In addition, coefficient of determination (r^2 = 0.60) indicated that more than 60% of changes in riparian vegetation were related to the changes in average discharge. In general, decreases of 3.6-90.3% in cover of the riparian vegetation were likely caused by 12.3-89% decreases in average discharge, while 26.4-171% increases in average discharge might lead to 2.5-172.2% of increased responses of riparian vegetation in China.

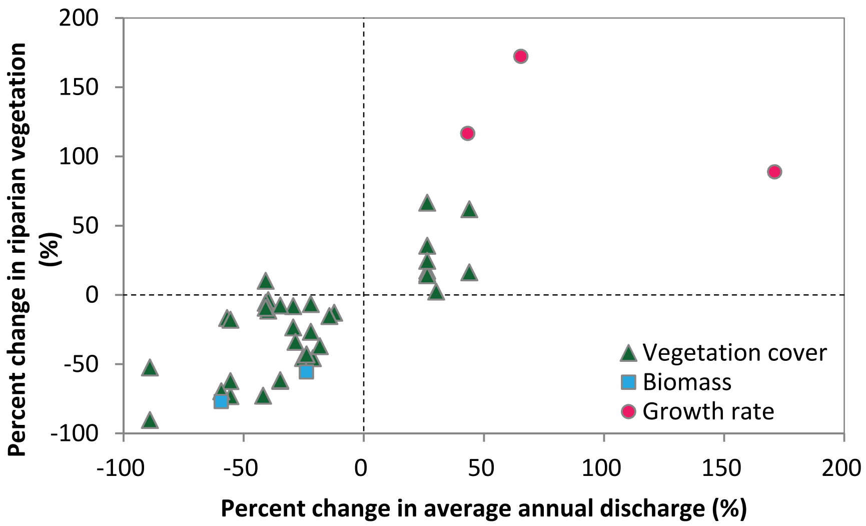

Fig. 3.6. Percent change in riparian vegetation cover, biomass and growth rate as response to percent change in average annual discharge in China. Percent change in both riparian vegetation and flow magnitude was calculated as the difference between impact and reference (the earliest records that were reported in the studies) conditions, in % of reference condition.

No clear relationships could be drawn between plankton responses and alterations in flow magnitude (Fig. 3.7). Diversity and abundance of most sensitive plankton species decreased with respect to either increased or reduced flow regimes, while some tolerant species showed significantly positive response to reductions in high flow and increases in low flow. Percent change of these species ranged from 113% to 2354% corresponding to 12-83% decreases in seasonal high flow and 6% increases in seasonal low flow. For four points represented changes in diversity and abundance larger than +200%, values were plotted at +200%.

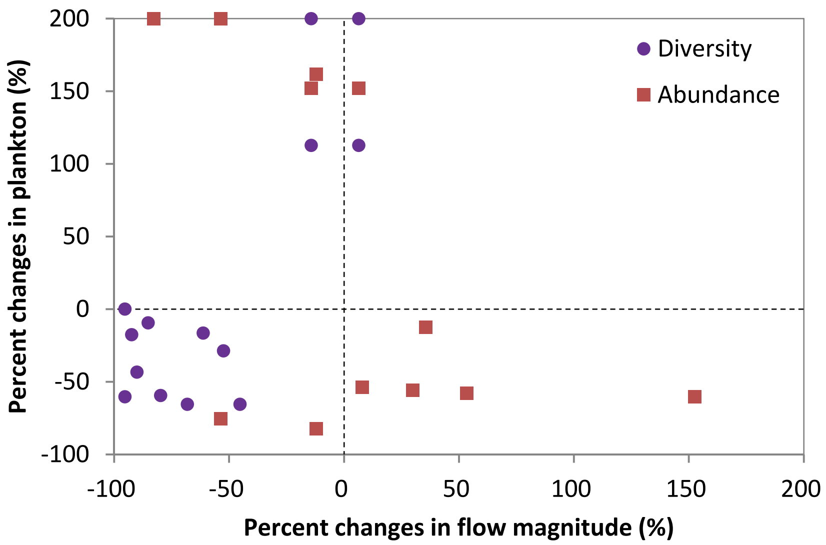

Fig. 3.7. Percent change in plankton diversity and abundance as response to percent change in flow magnitude in China. Alterations in flow magnitude referred as changes in annual average river discharge, seasonal high flow and seasonal low flow. Percent change in both plankton and flow magnitude was calculated as the difference between impact and reference (the earliest records that were reported in the studies) conditions, in % of reference condition. Please note that four values for changes in plankton due to flow alterations were plotted at +200%, but the real numbers are larger than 200%.

3.3.3 Responses of riparian vegetation and fish to average annual discharge in two climatic regions in China

In order to estimate responses of specific ecological groups to climate-driven and anthropogenically altered flow regimes, 117 data points with respect to changes in riparian vegetation cover (34 points) and fish catch (83 points) as consequences of altered average annual discharge were characterized according to classification of two climatic regions (arid and semi-arid region, humid region) in China. As expected, the regression analysis demonstrated that riparian vegetation cover was highly correlated with the alterations in average river discharge ($r = 0.79$) in arid and semi-arid regions in China. Coefficient of determination ($r^2 = 0.63$) indicated that about 60% of the variations in vegetation cover could be explained by changes in average river discharge (Fig. 3.8). The expected regression model for relationships between vegetation cover and average river discharge in four river basins was constructed as:

% change of riparian vegetation cover = 0.843 × % change of average annual discharge,
$r^2 = 0.63$, $p < 0.0001$ (1)

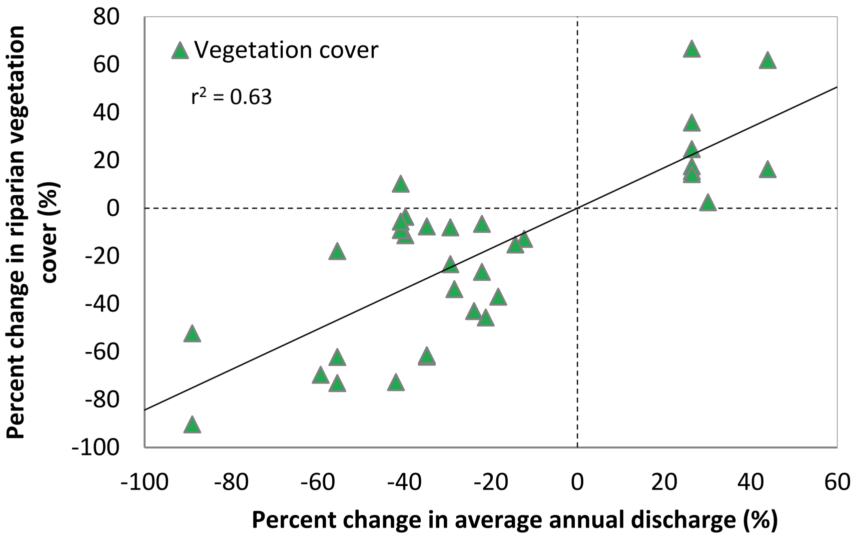

Fig. 3.8. Percent change in riparian vegetation cover as response to percent change in average annual discharge in arid and semi-arid region in China. Percent change in both riparian vegetation cover and average river discharge was calculated as the difference between impact and reference (the earliest records that were reported in the studies) conditions, in % of reference condition.

Fish catch showed robust correlations ($r = 0.78$, 0.77) to alterations in average river discharge in arid regions and humid regions. Approximately half of changes (Fig. 3.9) in arid and semi-arid region and over 50% changes (Fig. 3.10) in humid region could be explained by changes in average annual discharge ($r^2 = 0.53$, 0.58). Fishes showed highly sensitive responses when average annual discharge has been changed.

Relationships between fish catch and average river discharge in arid and semi-arid region was modeled as follow:

% change of fish catch = 0.816 × % change of average annual discharge,
$r^2 = 0.53$, $p = 0.002$ (2)

while regression model for response of fish catch to impact of average river discharge, in four main river basins in humid region was formed as:

% change of fish catch = 1.312 × % change of average annual discharge,
$r^2 = 0.58$, $p < 0.0001$ (3)

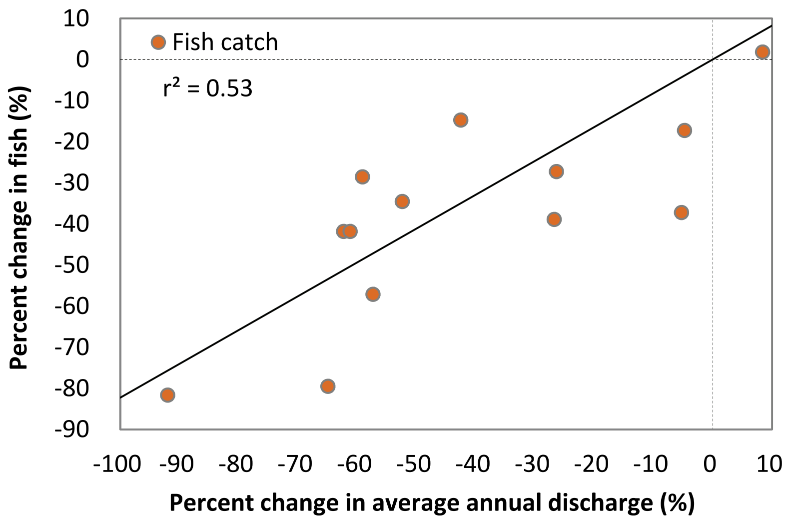

Fig. 3.9. Percent change in fish catch as response to percent change in average annual discharge in arid and semi-arid region in China. Percent change in both fish catch and average river discharge was calculated as the difference between impact and reference (the earliest records that were reported in the studies) conditions, in % of reference condition.

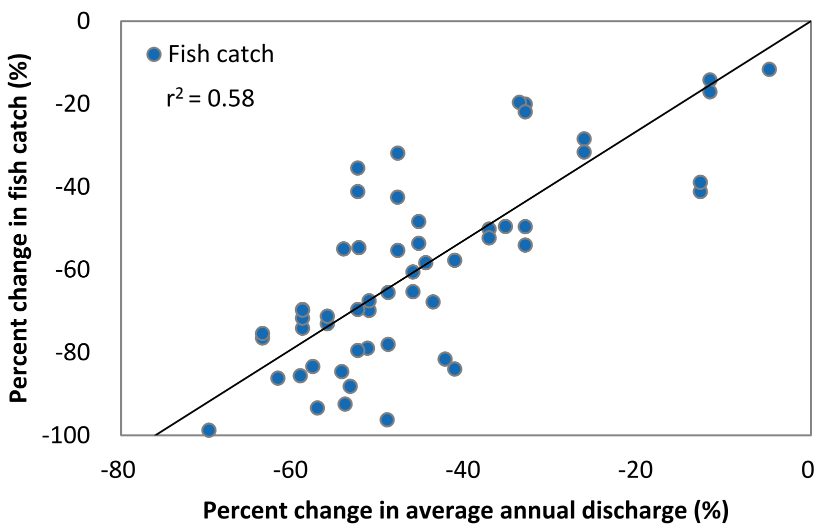

Fig. 3.10. Percent change in fish catch as response to percent change in average annual discharge in humid region in China. Percent change in both fish catch and average annual discharge was calculated as the difference between impact and reference (the earliest records that were reported in the studies) conditions, in % of reference condition.

3.4 Discussion

The main goal of this study was to determine relationships between ecological responses and anthropogenic flow alterations in China according to sources of information that could be extracted from published literatures. The research hypothesis was trying to define ecological limits of flow alteration and provide environmental guidelines for China's sustainable water management. As expected, it was possible to derive general relationships between changes in flow regimes and variations in ecological categories of fish, riparian vegetation and plankton from literature review, moreover, robust linear relationships ($r^2 = 0.60$) were extracted between alterations in average discharge and different responses of riparian vegetation. Responses of fish strongly correlated with altered flow magnitude (i.e. average annual discharge, seasonal high flow and seasonal low flow), and about 40% changes in fish catch and diversity could be explained by flow alterations. The analysis indicated that improvements have been achieved by including more data points within the range of 0-50% as compared to the study of Poff and Zimmerman (2010), which could not derive any strong relationship between flow alterations and ecological responses at global scale due to lack of samples in the low- to mid-range and to the work of Lloyd et al. (2003), which failed in developing thresholds regarding flow-ecology relationships according to information of 70 published studies. Based on current literature review, the analysis suggests that general relationships could not be derived from all responses of different ecological categories to flow alterations in China. Developing relationships between specific response of individual ecological group and flow components could be a better solution in determining impacts of anthropogenically altered flows on freshwater ecosystems in China.

Magnitude and direction of ecological response to changes in flows greatly depend on characteristics of ecological categories and types of flow alteration (McManamay et al., 2013) and other drivers like hydroclimate and geomorphy may affect ecological responses to flow changes as well (Poff and Ward, 1989; Arthington et al., 2006). The results of the subsequent analysis provided supplements in these contexts. In the latter analysis, coefficient of determination between riparian vegetation cover and average river discharge was increased to 0.63, as compared to the value of 0.60 in the previous analysis, which analyzed responses of coverage, biomass and growth rate of riparian vegetation to changes in flow magnitude. Estimation of responses of fish diversity and fish catch to altered flow magnitude resulted in a R-squared value of 0.43, while in subsequent evaluation, the values have been significantly increased to 0.53 and 0.58 between variations of fish catch and alterations of average river discharge in arid and semi-arid region, and in humid region. To a certain extent, the results could provide some supports for the previous studies, such as Poff and Zimmerman (2010), Lloyd et al. (2003) and Bunn and Arthington (2002), which were not able to take into account hydroclimate and regional differences among study sites.

According to regression analyses, fish responded rapidly to either reduced or increased flow magnitude. 5-13% reduction in flows might lead to 12-41% decrease in fish catch in early impact period (e.g. a few years after dam construction), additionally, both fish catch and diversity strongly decreased more than 55% in most cases where decline of flow magnitude exceeded 50% (Fig. 3.5, Fig. 3.9 and Fig. 3.10). Thus, it can be concluded that fish can serve as good ecological indications with respect to modified flow regimes. The finding is similar to the suggestion of Poff and Zimmerman (2010) that fish are sensitive indicators of flow alteration.

For riparian vegetation, 18-89% reduction in river flows due to water withdrawals and dam construction were likely has resulted in 36-90% decline in vegetation cover from 1950s to 1970s and 1980s. Seven data points represented small decreases (4-11%) in vegetation cover corresponding to large reduction (29-41%) in flow magnitude from 1987 to 1996 and 2000. However, the earliest vegetation observations were recorded 30 years after the occurrence of the human-induced flow alterations, thus the degree of the impacts on riparian vegetation were likely underestimated. Nine samples recorded increased vegetation cover with respect to increases in river discharge due to water diversion. Dissimilar to the finding of Poff and Zimmerman (2010), those increased responses were mostly from forest or shrub instead of non-woody vegetation. Nevertheless, the results are not robust enough due to limited sample size.

A dataset that included all information of ecological and hydrological metrics in China was developed based on 61 publish studies, moreover, 190 data points that expressed responses of ecological groups of fish, riparian vegetation, plankton, macrophyte, bird and macroinvertebrate to alterations of average discharge, low flow, high flow, flow duration and rate of change were extracted and analyzed. As few information regarding Chinese studies was included in previous research, the finding of the present study can provide additional knowledge on determination of relationships between ecological responses and flow alterations at global scale, especially those changes in low- to mid- ranges.

Several limitations to this study need to be acknowledged, and have to be considered in the future research. First, lack of sufficient and consistent observations of flow alteration limited the ability to include more samples regarding ecological responses. Of all 61 reviewed studies, only 42 of them were involved in analysis, while the rest 19 papers that provided 76 data points with respect to various ecological changes could not be considered. Additionally, 90% studies included in this study estimated flow alteration only as changes in one or two flow components (e.g. average discharge, seasonal low flow and/or seasonal high flow), however, in reality, aquatic and riverine species are influenced by a variety of hydrological drivers simultaneously (Poff and Zimmerman, 2010). Thus, application of macro-scale hydrological models such as WaterGAP (Alcamo et al., 2003a) can generate simulated information of multiple flow components for study sites lacking observed flow data, and hence provide a solution to the future estimation of the impacts of multiple hydrological drivers on freshwater ecosystems in China. Second, most of the data points (89 of 95 points) that represent fish responses focused on weight or quantity of fish catch. Nevertheless, in addition to effects of flow alterations, fish catches are largely influenced by fishing technology, fishing power and fishing intensity. For example, Yi and Wang (2009) reported that weight of fish catch in Lake Dongting was 41.25 tons in 1970 and 48.75 tons in 1971, but annual average discharge was 105.19 km^3/year in 1970 and 79.63 km^3/year in 1971. Consequently, introducing fish diversity into further assessment will improve accuracy of the analysis, because fish diversity is sensitive to changes in flow regimes, in particular changes in river discharge (Xennopoulos et al., 2006; Xennopoulos and Lodge, 2006; Iwasaki et al, 2012) and can be a good indicator for long-term effects. Third, impacts of other environmental factors, such as block of dam, hydraulic structure, flow velocity, water temperature, sediment transport, water pollution and non-native species invasion were not able to be considered in this study due to insufficient information for those characteristics. Furthermore, effects of some human-induced factors, such as land expansion, overfishing and deforestation were not included either, because it is generally difficult to measure their impacts precisely.

3.5 Conclusion

This study conducted the first estimates of general quantitative relationships between responses of different ecological categories and a variety of river flow alterations, as well as responses of fish and riparian vegetation to average river discharge in China based on the information that could be extracted from published studies. A total of 61 literatures reported quantitative ecological responses to anthropogenic-induced flow alterations regarding study sites of six main river basins and seven watersheds in

China. The results indicated that coverage, biomass and growth rate of riparian vegetation as well as diversity and catches of fish showed strong correlation to percent change in flow magnitude (r = 0.77, 0.66), particularly to changes in average river discharge. Roughly 60% of alterations in riparian vegetation and 40% of changes in fish were likely caused by modified flow magnitude (r^2 = 0.60, 0.43). Except for riparian vegetation and fish, robust relationships between flow alterations and responses of other ecological groups were unable to be derived. Analysis of relationships between riparian vegetation cover and average river discharge showed that in arid and semi-arid region, more than half of the variations in vegetation cover could be explained by changes in average river discharge. Estimation denoted strong linear relationships between fish catch and modified average discharge in arid and semi-arid region, and humid region as well. Approximately over 50% changes in arid and semi-arid region and humid region were contributed by alterations in flow magnitude (r^2 = 0.53, 0.58). Overall, it is possible to derive quantitative relationships between ecological responses and flow alterations in China based on current sources of literature, even though the general relationships could not be drawn from all responses of different ecological categories. In addition, riparian vegetation cover and fish catch might be reasonable ecological indicators in developing quantitative relationships between flow alterations and ecological changes in China.

As the aquatic and riparian species respond to multiple confound hydrological drivers in reality (Poff and Zimmerman, 2010), estimation of relationships between more representative ecological indicators such as fish diversity and modeled ecological relevant flow indicators in China will be performed in the future study.

Supporting Information

Appendix A

A1. Summary of changes in different flow components due to anthropogenic impacts and hydrological data based on a literature review of 61 published papers and 15 additional studies in China.

A2. Summary of ecological responses to anthropogenic flow alterations and ecological data based on a literature review of 61 published papers in China.

Appendix B

B1. Summary of quantitative relationships between riparian vegetation cover and average river discharge in arid and semi-arid region in China.

B2. Summary of quantitative relationships between fish catch and average river discharge in arid and semi-arid region in China.

B3. Summary of quantitative relationships between fish catch and average river discharge in humid region in China.

Appendix C

C1. Reference list of 61 published studies included in this study.

C2. Reference list of 15 studies reported additional flow data.

Chapter 4: Estimation of changes in fish species richness as consequence of anthropogenic flow alterations at basin and sub-basin scale in China

Abstract

Anthropogenically altered flow regimes, such as reduced river flow discharge and flow stabilization due to dam operation and human water use, have greatly influenced biodiversity in freshwater ecosystems in China during the past 60 years. Compiling knowledge with related to the relationships between flow alterations and risk of biodiversity loss into ecological impact assessment could provide suggestions for ecological conservation and sustainable water management in China. This study has presented the first estimation on quantitative relationships between decreases in native fish species richness and anthropogenic flow alterations due to water use and dam construction in 34 river basins and/or sub-basins in China. Five ecologically relevant flow indicators were quantified using the discharge data which were modeled by an improved version of the global hydrological model WaterGAP, while fish species richness for different time periods were extracted from 49 published Chinese studies. A total of 360 data points that represent relationships between losses of fish species and the five flow indicators were analyzed by single and multiple regression models. For the single regression analysis, significant linear relationships ($p < 0.05$) were detected for the indicators of long-term average annual discharge and statistical low flow Q_{90}. For the multiple regressions, coefficients of determination (R^2) of most models ranged from 0.10 to 0.31. The indicator of long-term average annual discharge was detected in all of the best-fitting models ($\Delta AIC_c \leq 2$) and has very significant relationships ($p < 0.01$) with changes in number of fish species. The indicator of statistical low flow Q_{90} (I_{Q90}) was found in one best-fitting model, but showed no significant effect ($p > 0.1$) on changes of fish species richness due to the influence of collinearity. Two conclusions emerged from the analysis: 1) losses of fish species were positively correlated with changes in long-term average annual discharge in China and 2) indicator of I_{LTD} was dominant over other flow indicators included in this

research for the given dataset. These results can provide environmental flow guidelines for the sustainable water resources management in rivers with high risk of fish extinction in China.

4.1 Introduction

Surface fresh waters, such as rivers, lakes and wetlands, occupy 0.08% of Earth's surface and account for only 0.01% of the global water resources (Gleick, 1996), however this small proportion of water provides habitats for around 100 000 species (Hawksworth and Kalin-Arroyo, 1995). Thus, freshwater ecosystems are the most endangered ecosystems in the world (Dudgeon et al., 2006), and freshwater biodiversity decreases much faster than terrestrial and marine biodiversity (Jenkins, 2003; Sala et al., 2000). The major causes of loss in freshwater species can be characterized in four aspects: overexploration, flow alteration, water pollution and exotic species invasion (Dudgeon et al., 2006), and among these factors, flow alterations as consequence of climate change and human activities are considered to be the most critical factor causing declines in freshwater biodiversity (Postel and Richter, 2003). Flow regimes are the key points in determining the biotic composition, function and diversity within river ecosystems (Richter et al., 1996; Arthington and Pusey, 1993), meanwhile, alterations in flow regimes are claimed to be the most serious threats to the ecological sustainability of rivers (Bunn and Arthington 2002) and will further influence freshwater biodiversity in the future.

During the past six decades, anthropogenic-induced flow alterations, such as decline in river flow discharge and flow stabilization due to dam construction and increasing human water withdrawals have significantly affected freshwater biodiversity in riverine ecosystems in China. Fish species in the Yangtze River basin have decreased rapidly since 1950s (Zeng 1990) and 25 fish species have been identified to be endangered by 1998 (Yue and Chen 1998), mainly due to dam construction. Huang and Xie (1996) reported that fish species in Lake Donghu (floodplain in the middle of the Yangtze River) decreased from 67 before 1971 to 39 in 1994 due to channel-floodplain disconnection. Zhang et al. (2007) demonstrated that diversity of macrophytes in Lake Nansi (wetland of the Huaihe River) reduced from 116 in 1983 to 46 in 1989 as consequence of diminished inflows. Fish diversity in the middle and lower reaches of the Yellow River has decreased dramatically from 66 and 81 in 1980s to 30 and 41 in 2008 as a result of decreased river discharge (Ru et al., 2010).

Li et al. (2007) reported that fish species richness in the Lianjiang River (a tributary of the Pearl River) decreased from 101 in 1960s to 87 in 2005 due to reduction in peak flow. In view of the increasing impact of anthropogenic flow alterations on freshwater biodiversity in China, research on quantitative relationships between changes in flow regimes and loss of species in China's riverine ecosystems is urgently needed for ecological conservation and sustainable management of freshwater resources.

Reduced river discharge was regarded as the major environmental driver of biodiversity loss (Postel and Richter, 2003), therefore a series of studies that focused on evaluation of relationships between fish species richness and river discharge were conducted. A pilot study of prediction of future losses in fish species richness as consequence of reduced river discharge was carried out by Xenopoulos et al. (2005), using a log-linear analysis based on published fish data (Oberdorff et al., 1995; Froese and Pauly, 2000) and simulated river discharge. The resulted species-discharge relationships indicated that fish diversity would be decreased more than 75% by 2070. By applying a method similar to the former study, Xenopoulos and Lodge (2006) anticipated that 2-38% reduction in fish diversity would occur within 33 southeastern rivers in United State as result of 20-90% decrease in river discharge. Mcgarvey and Ward (2008) estimated quantitative relationships between fish diversity and river discharge within three large rivers in southeastern United State and divided each river into three longitudinal zones. Their study suggested that the significance of species-discharge relationship was greatly improved by using longitudinal zones as sampling units instead of the complete river basin (Xenopoulos et al., 2005; Xenopoulos and Lodge, 2006) because species-discharge is scale dependent.

In addition to river discharge, other characteristics of flow regimes such as magnitude, frequency, duration, timing and rate of change, are critical in sustaining the full native biodiversity and integrity of aquatic ecosystems (Richter et al., 1997; Poff et al., 1997). Ecological responses to above flow metrics were evaluated by several studies through literature review (Bunn and Arthington, 2002; Lloyd et al., 2003; Poff and Zimmerman, 2010). Poff and Zimmerman (2010) extensively reviewed 165 papers at global scale and suggested that fish are sensitive indicators of flow alteration**.** Iwasaki et al. (2012) estimated relationships between fish species richness and 14 hydrological metrics using logistic regression and found that except mean river discharge, low flow and high flow might be critical factors that influence fish diversity globally. Generally, quantitative ecological responses to flow metrics other than mean river discharge is still unknown (Poff and Zimmerman, 2010).

Little information was provided by the former studies with respect to hydro-ecological relationships in China. In Chinese academia, even though many case studies reported responses of various specific ecological categories to the impacts of flow alterations, general knowledge on quantitative hydro-ecological relationships is still lacking. In thesis Chapter 2, the author extensively reviewed 61 published Chinese studies regarding ecological changes due to different hydrological drivers and developed significant relationships between changes in flow magnitude and alterations in riparian vegetation cover and fish catches with in seven main river basins in China based on information that was extracted from the reviewed literature. In the discussion sector (see Sector 3.4), the author indicated that fish catches can be affected by other factors other than flow alterations, and fish diversity could be a better ecological indicator because it is sensitive to changes in flow regimes and suitable for analysis of long-term impacts.

The aim of this study was to figure out fish extinction rates with respect to changes in flow regimes at macroscale in China. Therefore a first estimation of quantitative relationships between anthropogenic alterations in different flow metrics and loss of fish species richness was carried out at basin and sub-basin scale in China using a multiple linear regression model. Due to lack of observed flow data, values of flow metrics were simulated by a macroscale hydrological model WaterGAP (Alcamo et al., 2003a, 2003b; Döll et al., 2003; Döll and Fiedler, 2008; Döll et al., 2012), while fish species data was collected from the published Chinese papers.

4.2 Methods

4.2.1 Fish data preparation

Number of fish species was collected from 49 published Chinese studies (see Table 4.1 and Appendix D1 for the summary of fish species richness and the list of the literature included in this study), which reported different time period of fish records and fish extinction mainly due to anthropogenic flow alterations within 34 river basins/sub-basins in China (see Fig. 4.1 for the names and geographic distribution of these basins and sub-basins). As many non-native fish species are more tolerant to alterations in flow regimes and have impacts on the biological integrity of native aquatic ecosystems (Kennard et al., 2005), only native fish species were taken into account in this study to avoid systematic bias.

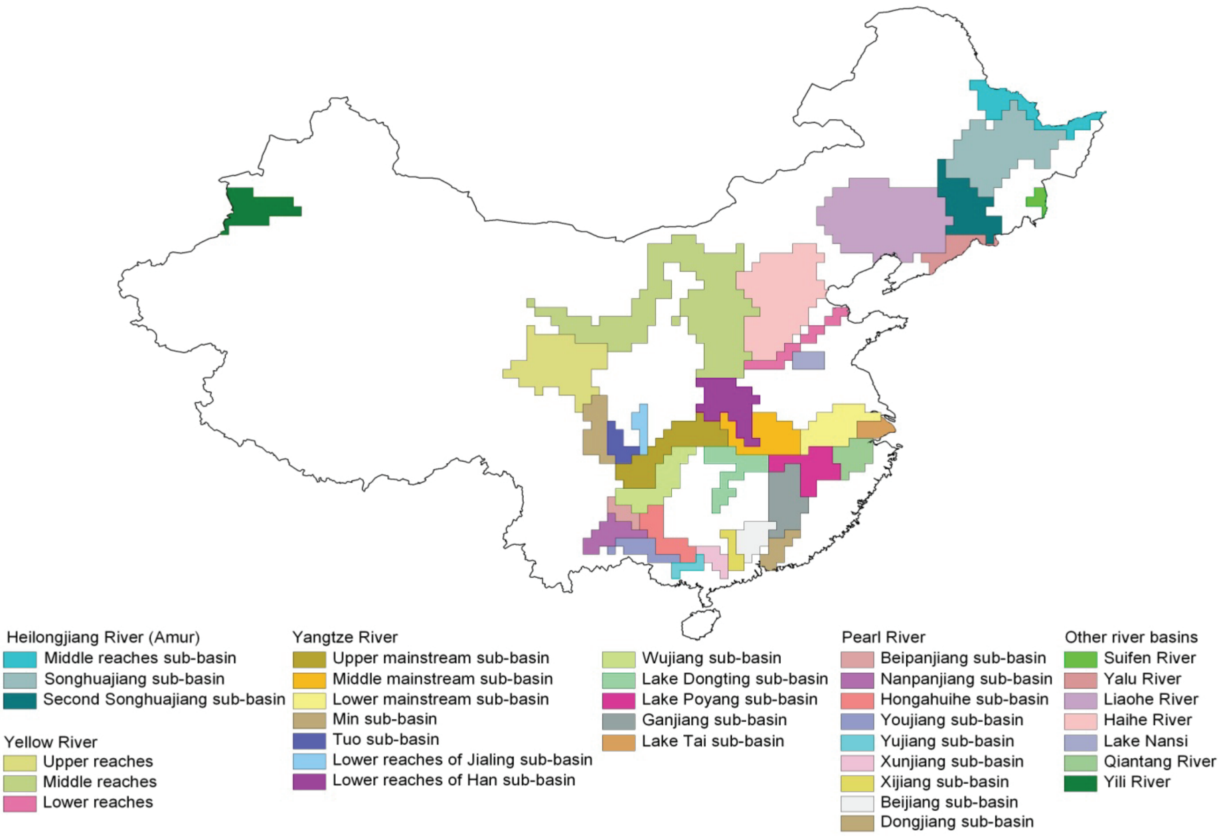

Fig. 4.1. Name and geographic distribution of the 34 river basins /sub-basins included in this study.

For each basin and sub-basin, the earliest record of fish species richness was considered as the reference condition (no change or only slight change occurred in number of fish species and in river flow regimes due to human impacts) of fish diversity, while the fish records that were reported during the latter time periods were considered as the altered conditions of fish diversity caused by anthropogenic river flow alterations. The changes of fish species richness in 34 river basins/sub-basins in China were calculated as the difference between altered and referenced number of fish species, in percent of the referenced number of fish species. As two time periods of altered fish records were reported for 6 out of 34 basins/sub-basins, in consequence, a total of 40 data points with respect to the loss of fish diversity were obtained.

4.2.2 Computation of reference and anthropogenically altered river discharge using WaterGAP

4.2.2.1 Description of WaterGAP 2.2

In order to calculate the anthropogenically altered and the naturalized (or referenced) river flows in China, an improved version of Global Hydrological and Water Use Model, WaterGAP 2.2, was used to compute river discharge in each river basin and sub-basin. The WaterGAP model contains a global hydrological model (WGHM) and several water use models for the sector irrigation, livestock, manufacturing, cooling of thermal power plants and households. Irrigation water consumption is calculated by the global irrigation model (GIM) as a function of climate, irrigation area and crops, which are distinguished as only rice and non-rice (Döll and Siebert, 2002). Based on the outputs as computed by the water use models, a submodel GWSWUSE, which calculates the total net water abstraction from groundwater and from surface water in each 0.5° grid cell (Döll et al., 2012) was introduced in version 2.2 of WGHM to estimate the impact of surface water and groundwater withdrawals on river flows. Compared to Döll et al. (2012), irrigation water use efficiencies (ratio of net irrigation consumption to total water abstractions) differ between surface water and groundwater use in WaterGAP 2.2. While for surface water irrigation, country-specific values are still used, and irrigation water use efficiency was set to 0.7 worldwide (Döll et al., 2014a). Return flows from irrigation to either groundwater or surface water are computed as a function of the cell-specific artificial drainage fraction (Döll et al., 2012). In WaterGAP 2.2, the fraction of irrigation return flows that recharge groundwater was increased as compared to Döll et al. (2012) and is computed as 0.95–0.75 times the cell-specific artificial drainage fraction (Döll et al., 2014a).

Table 4.1 Number of fish species and changes in fish diversity included in this study.

Drainage basin		Year or time period of fish species data records: Reference	Altered	No. of native fish species: Reference	Altered	% change	Source of fish species data
Heilongjiang River (Amur River)	Middle reaches	1980-1983	2009-2010	69	49	-28.99	Zhang 1995; Xia et al., 2012
	Songhuajiang River	1980-1983	2010	82	69	-15.85	Zhang 1995; Zhao et al., 2011
	The second Songhuajiang River	1957	1975-1983	73	66	-9.59	Wang et al., 1959; Yu and Zhang, 1984
Yalu River		1961-1964	1980-1983	84	72	-14.29	Zhang 1986; Xie 1986
Liao River		before 1964	1977-1978	76	72	-5.26	Xie 1981;
			2009		25	-67.11	Pei et al., 2010
Suifen River		1961-1964	1980-1983	31	25	-19.35	Zhang 1985
Haihe River		before 1950	before 1979	59	45	-23.73	Zhang et al., 2011
Lake Nansi		1959	1995-1998	74	32	-56.76	Zhou and Chen, 1997; Li et al., 2005
Yellow River	Upper reaches	1961-1965	2005-2007	18	11	-38.89	Zhang et al., 2009
	Middle reaches	1981-1983	2008	68	34	-50.00	He et al., 1986; Ru et al., 2010
	Lower reaches			81	41	-49.38	
Yangtze River	Min sub-basin	before 1959	1984-1997	40	16	-60.00	Deng and Wu, 2001
	Tuo sub-basin	1975	1980-1984	122	106	-13.11	Ding 1989
	Lower reaches of Jialing sub-basin	1976	2003	105	79	-24.76	Shi and Deng, 1980; Jiang and He, 2008
	Lower reaches of Han sub-basin	before 1960	1976-1978	79	75	-5.06	Yu et al., 1981; Li et al., 2005
			2003-2004		61	-22.78	
	Middle mainstream sub-basin	1973-1975	2001-2003	146	59	-59.59	Zeng 1990; Liu et al., 2005
	Lower mainstream sub-basin	1973-1975	2004	140	74	-47.14	Zeng 1990; Duan et al., 2007
	Upper mainstream sub-basin	1973-1975	2005-2006	146	96	-34.25	Wu et al., 2007
	Wujiang sub-basin	1964-1984	2004-2008	120	81	-32.50	Yang et al., 2010
	Lake Dongting sub-basin	1974	2004-2005	104	69	-33.65	Ru et al., 2005
	Lake Poyang sub-basin	before 1980	1982-1990	117	103	-11.97	Zhang and Li, 2007
			1997-2000		101	-13.68	
	Ganjiang sub-basin	1982-1990	2008-2009	118	71	-39.83	Gao and Liu, 1995; Zou et al., 2010
	Lake Tai sub-basin	1970s	2002-2006	103	56	-45.63	Ni and Zhu, 2005; Zhu et al., 2007
Qiantang River		1982-1987	1995-2001	144	127	-11.81	Zheng and Jia, 1988; Ge 2005
Pearl River	Beipanjiang sub-basin	1994	2008-2010	71	45	-36.62	Zhou et al., 2011
	Nanpanjiang sub-basin	before 1989	2000-2008	137	59	-56.93	Wang et al., 2011
			2008-2010		47	-65.69	
	Hongshuihe sub-basin	1981-1986	1996-1997	70	43	-38.57	Li 2006
			2002-2003		42	-40.00	
	Youjiang sub-basin	1974-1977	2008	73	42	-42.47	Anonymous, 2006; Zhou et al., 2011
	Liujiang sub-basin	1974-1976	2006	117	57	-51.28	Zhu et al., 2007
	Yujiang sub-basin	before 1989	2004	74	53	-28.38	Zhou et al., 2006
	Xunjiang sub-basin	1974-1976	2004	83	63	-24.10	Anonymous, 2006; Zhou et al., 2011
	Xijiang sub-basin	1981-1985	2005-2008	136	84	-38.24	Li et al., 2010
	Beijiang sub-basin	1981-1983	2005-2006	140	78	-44.29	Pan et al., 1984; Guo et al., 2008
	Dongjiang sub-basin	1981-1983	2009-2010	124	78	-37.10	Ye et al., 1991; Liu 2011
Yili River		1963-1965	1995-1997	11	9	-18.18	Anonymous, 1979; Ren 1998

The standard WaterGAP 2.2 model version is driven by daily reanalysis-based WFD/WFDEI climate data, a combination of the daily WATCH Forcing Data based on ERA40 for the year 1901-1978 (WFD), and the WATCH Forcing Data based on ERA-Interim for the year 1979-2009 (WFDEI). With a spatial resolution of 0.5° by 0.5°, WaterGAP 2.2 generates daily water flows and storages at the global scale, and was calibrated against measured long-term average annual river discharge at 1319 gauging stations, and the adjusted calibration factors is regionalized to grid cells outside the calibration basins (Müller Schmied et al., 2014).

4.2.2.2 Specification of model runs

According to Kennard et al. (2010), the minimum time period for a hydrological analysis should be no less than 15 years. Therefore, in this analysis, two 15-year time series (reference and altered) of gridded monthly river discharge at the outlet of each river basin and sub-basin were calculated by WGHM2.2, which were then used to compute the ecologically relevant indicators of river flow alterations described in Sect. 4.2.2.3. Under the reference condition, the model run computed river flow for a period of 15 years, in which the end year of the simulation was defined as the year when the earliest number of fish species was reported. If the fish species data were collected during a period of time, then the last year of this period was set as the end year of the simulation. With respect to the altered condition, the year or the time period when the subsequent fish data was reported was considered as the end year of the simulation (see Table 4.1 and Table 4.2).

In WGHM2.2, the impact of human water withdrawals is computed by subtracting the total net abstraction (water abstraction minus return flow) from groundwater and surface water body. In this study, the net abstractions before 1979 are calculated with the daily WATCH Forcing Data based on ERA40 (WFD) as the climate input, while the net abstractions during 1979-2009 are computed using the WATCH Forcing Data ERA Interim (WFDEI) as the climate input. As the climate data end in 2009, WaterGAP cannot be used for the periods after 2009. Therefore, for the model runs with the period of 1996-2010, the net abstraction of 2010 was assumed to be the same as the values in 2009.

In order to compute impact of dams on river discharge, a reservoirs and regulated lakes data set that includes 6619 reservoirs and 43 regulated lakes worldwide was implemented in WGHM2.2. In this analysis, 731 reservoirs and 2 regulated lake were used to simulate river flow regimes under reference and altered conditions in 34 river basins / sub-basins in China, and the years of construction of those dams range from 1909 to 2006. In standard WGHM 2.2, reservoirs and regulated lakes are included as a constant input, which means that no matter what time period will be simulated, the effects of all reservoirs and regulated lakes will be taken into account, i.e. in a model run with the early time series, dams that were built after this period are included as well. Therefore, the real impact of dams on river discharge for this time series can be somehow overestimated. In this study, the reservoirs and the regulated lakes that were constructed after each time period of simulation were not included in the input data of the relative model run in order to avoid uncertainties of overestimation.

4.2.2.3 Indicators of river flow alteration

Five different indicators of river flow alteration that are ecologically relevant and can be calculated by WaterGAP 2.2 in a rather reliable manner were taken from the indicator set of Döll et al. (2009) and the Indicators of Hydrologic Alteration (IHA) approach of Richter et al. (1997) (see Table 4.2 and Table 2.2 in the Sect. 2.2.2 for detailed description). The indicator set of Döll et al. (2009) was developed based on the IHA indicators and the Dundee Hydrological Regime Alteration Method (DHRAM) of Black et al. (2005), and was used to analyze the impact of anthropogenic flow alteration on freshwater ecosystems worldwide. As the most of the IHA indicators rely on daily discharge data, which cannot be calculated by the previous version of WaterGAP (2.1g) driven by monthly climate input data, therefore only indicators on the basis of monthly and annual discharge data were considered in Döll et al. (2009).

The five ecological relevant indicators represent anthropogenic alterations in the river flow characteristics as follow: long-term annual discharge (I_{LTD}), statistical low flow Q_{90} (I_{Q90}), statistical high flow Q_{10} (I_{Q10}), seasonal amplitude (I_{SA}) and seasonal regime (I_{SR}), and were then calculated based on the monthly river discharge data which were computed by WGHM2.2 in the Sect. 4.2.2.2 for 34 river basins / sub-basins in China (Table 4.3).

Table 4.2 Five ecologically relevant indicators of river flow alteration included in this study.

Indicators	Flow characteristics	Description	Ecological relevance
I_{LTD}	Long-term average annual discharge	percent change in long-term average annual river discharges between anthropogenically altered and reference conditions	fish species richness[1], floodplain vegetation
I_{Q90}	Statistical low flow	percent change in Q_{90} (monthly river discharge that is exceeded in 9 out of 10 months) between anthropogenically altered and reference conditions	habitat conditions, connectivity of channel or floodplain
I_{Q10}	Statistical high flow	percent change in Q_{10} (monthly river discharge that is equaled or exceeded for 10% of the specified term) between anthropogenically altered and reference conditions	habitat conditions, species richness[2], floodplain vegetation
I_{SA}	Seasonal amplitude	percent change in seasonal amplitude (maximum minus minimum long-term average monthly river discharge) between anthropogenically altered and reference conditions	habitat availability in particular on floodplains
I_{SR}	Seasonal regime	mean over 12 monthly values of absolute differences between long-term average monthly river discharges under anthropogenically altered and reference conditions , in % of referenced discharge	habitat conditions, compatibility with life cycle of organisms

[1]Xenopoulos et al. (2005)

[2]Poff and Zimmerman (2010)

Table 4.3 Summary of ecologically relevant flow indicators included in this study. All flow indicators were calculated based on simulated monthly river discharges using WGHM2.2.

Drainage basin		Time period of simulation		long-term average annual discharge (km³/year)			Q_{90} (km³/month)			Q_{10} (km³/month)			Seasonal amplitude (km³/month)			Seasonal[1] regime I_{SR} (%)
		Reference	Altered	Reference	Altered	I_{LTD} (%)	Reference	Altered	I_{Q90} (%)	Reference	Altered	I_{Q10} (%)	Reference	Altered	I_{SA} (%)	
Heilongjiang River (Amur River)	Middle reaches	1969-1983	1996-2010	128.70	117.24	-8.90	6.713	6.513	-2.98	29.104	25.260	-13.21	23.037	20.690	-10.19	6.30
	Songhuajiang River	1969-1983	1996-2010	52.40	50.91	-2.84	1.964	1.995	1.58	8.167	8.380	2.61	6.842	7.007	2.41	6.27
	The second Songhuajiang River	1943-1957	1969-1983	18.06	14.04	-22.26	1.280	1.344	5.00	7.622	5.134	-32.64	7.390	4.120	-44.25	22.75
Yalu River		1950-1964	1969-1983	26.82	18.53	-30.91	1.239	0.933	-24.70	4.105	2.099	-48.87	5.755	1.293	-77.53	18.60
Liao River		1949-1963	1964-1978	15.81	8.36	-47.12	0.288	0.120	-58.33	3.453	1.475	-57.28	4.183	1.865	-55.41	41.18
			1995-2009		3.45	-78.18		0.002	-99.31		0.513	-85.14		1.060	-74.66	72.73
Suifen River		1950-1964	1969-1983	1.38	1.06	-23.19	0.012	0.011	-8.33	0.325	0.218	-32.92	0.221	0.156	-29.41	25.06
Haihe River		1935-1949	1965-1979	11.38	8.48	-25.48	0.118	0.007	-94.07	2.152	1.736	-19.33	2.790	1.756	-37.06	71.61
Lake Nansi		1945-1959	1984-1998	4.79	1.84	-61.59	0.052	0.002	-96.15	0.600	0.432	-28.00	0.950	0.273	-71.26	57.24
Yellow River	Upper reaches	1951-1965	1993-2007	19.95	16.54	-17.09	0.755	1.024	35.63	3.303	1.617	-51.04	2.032	0.644	-68.31	28.09
	Middle reaches	1969-1983	1994-2008	34.80	20.56	-40.92	1.710	0.404	-76.37	4.955	2.316	-53.26	2.300	0.519	-77.43	37.32
	Lower reaches	1969-1983	1994-2008	31.46	14.44	-54.10	0.095	0.001	-98.95	4.906	2.787	-43.19	4.539	1.766	-61.09	56.71
Yangtze River	Min sub-basin	1944-1958	1983-1997	74.72	59.18	-20.80	1.114	0.958	-14.00	13.840	10.747	-22.35	15.501	11.151	-28.06	18.98
	Tuo sub-basin	1961-1975	1970-1984	16.59	15.25	-8.08	0.427	0.436	2.11	3.051	2.583	-15.34	2.663	2.378	-10.70	6.02

Table 4.3 (Continued)

Drainage basin		Time period of simulation		long-term average annual discharge (km³/yr)			Q_{90} (km³/month)			Q_{10} (km³/month)			Seasonal amplitude (km³/month)			Seasonal[1] regime I_{SR} (%)
		Reference	Altered	Reference	Altered	I_{LTD} (%)	Reference	Altered	I_{Q90} (%)	Reference	Altered	I_{Q10} (%)	Reference	Altered	I_{SA} (%)	
Yangtze River	Lower reaches of Jialing sub-basin	1962-1976	1989-2003	223.57	217.53	-2.70	1.759	1.757	-0.11	14.443	11.246	-22.14	13.741	9.601	-30.13	14.74
	Lower reaches of Han sub-basin	1945-1959	1964-1978	56.31	52.38	-6.98	1.32	2.363	79.02	10.11	7.296	-27.83	8.82	4.815	-45.41	36.62
			1990-2004		46.47	-17.47		2.113	60.08		6.233	-38.35		3.795	-56.97	34.03
	Middle mainstream sub-basin	1961-1975	1989-2003	745.31	673.07	-9.69	24.630	26.107	6.00	108.182	95.712	-11.53	76.040	68.800	-9.52	11.01
	Lower mainstream sub-basin	1961-1975	1990-2004	990.49	890.59	-10.09	37.751	40.271	6.68	130.684	115.255	-11.81	91.631	83.722	-8.63	11.09
	Upper mainstream sub-basin	1961-1975	1992-2006	443.87	390.69	-11.98	10.153	10.501	3.43	73.686	63.849	-13.35	64.089	56.091	-12.48	9.30
	Wujiang sub-basin	1970-1984	1994-2008	53.07	48.62	-8.39	1.278	1.291	1.02	8.345	8.439	1.13	7.058	7.043	-0.21	8.59
	Lake Dongting sub-basin	1960-1974	1991-2005	642.86	580.07	-9.77	5.311	4.603	-13.33	27.129	23.523	-13.29	18.083	16.887	-6.61	14.44
	Lake Poyang sub-basin	1965-1979	1976-1990	167.98	158.70	-5.52	5.17	5.039	-2.53	27.444	24.381	-11.16	23.129	20.285	-12.30	12.43
			1986-2000		163.15	-2.88		5.037	-2.57		24.736	-9.87		20.907	-9.61	12.08
	Ganjiang sub-basin	1976-1990	1995-2009	81.45	79.61	-2.26	2.267	2.088	-7.90	12.371	11.870	-4.05	10.424	10.422	-0.02	15.57
	Lake Tai sub-basin	1965-1979	1992-2006	3.20	2.00	-37.50	0.020	0.011	-45.00	0.612	0.439	-28.27	0.297	0.273	-8.08	38.92

Table 4.3 (Continued)

Drainage basin		Time period of simulation		long-term average annual discharge (km³/yr)			Q_{90} (km³/month)			Q_{10} (km³/month)			Seasonal amplitude (km³/month)			Seasonal[1] regime I_{SR} (%)
		Refe-rence	Altered	Refe-rence	Altered	I_{LTD} (%)	Refe-rence	Altered	I_{Q90v}(%)	Refe-rence	Altered	I_{Q10} (%)	Refe-rence	Altered	I_{SA} (%)	
Qiantang River		1973-1987	1987-2001	35.43	33.93	-4.23	1.203	1.219	1.33	5.418	5.328	-1.66	4.123	4.863	17.95	14.42
Pearl River	Beipanjiang sub-basin	1980-1994	1996-2010	12.43	12.37	-0.48	0.163	0.145	-11.04	2.667	2.547	-4.50	2.390	2.554	6.86	8.54
	Nanpanjiang sub-basin	1974-1988	1994-2008	38.99	36.01	-7.64	1.695	1.485	-12.39	6.078	5.328	-12.34	4.002	3.528	-11.84	6.57
			1996-2010		21.25	-45.50		1.435	-15.34		5.323	-12.42		3.698	-7.60	9.01
	Hongshuihe sub-basin	1972-1986	1983-1997	82.13	76.44	-6.93	2.808	2.662	-5.20	12.742	12.788	0.36	9.178	9.071	-1.17	7.81
			1989-2003		75.70	-7.83		2.622	-6.62		13.069	2.57		10.141	10.49	11.14
	Youjiang sub-basin	1963-1977	1994-2008	16.52	12.47	-24.52	0.189	0.147	-22.22	3.522	2.781	-21.04	3.862	2.840	-26.46	24.78
	Liujiang sub-basin	1962-1976	1992-2006	42.50	41.10	-3.29	1.052	0.918	-12.74	7.669	6.161	-19.66	6.603	6.185	-6.33	14.60
	Yujiang sub-basin	1974-1988	1990-2004	45.21	41.02	-9.27	0.973	0.830	-14.70	8.881	8.144	-8.30	7.182	8.619	20.01	16.76
	Xunjiang sub-basin	1962-1976	1990-2004	230.46	203.01	-11.91	6.620	5.788	-12.57	39.261	34.748	-11.49	31.532	31.097	-1.38	12.41
	Xijiang sub-basin	1971-1985	1994-2008	233.21	221.90	-4.85	6.708	5.941	-11.43	37.850	40.603	7.27	26.443	33.531	26.80	15.01
	Beijiang sub-basin	1969-1983	1992-2006	35.84	35.05	-2.20	0.830	0.804	-3.13	5.982	5.395	-9.81	5.922	4.883	-17.54	13.54
	Dongjiang sub-basin	1969-1983	1996-2010	32.20	28.57	-11.27	1.111	0.917	-17.46	4.943	4.458	-9.81	3.255	4.433	36.19	21.15
Yili River		1951-1965	1983-1997	2.63	1.66	-36.88	0.034	0.032	-5.88	0.511	0.269	-47.36	0.818	0.463	-43.40	26.76

[1]Indicator of seasonal regime (I_{SR}) was calculated as the mean over 12 monthly values of absolute percentage differences between long-term average monthly river discharges under altered and reference conditions, therefore it cannot be presented as "reference" and "altered" values separately as the other flow indicators.

4.2.3 Quantitative analysis of relationships between changes in fish species richness and changes in indicators of river flow alteration

The influences of the five flow indicators on fish species richness in 34 basins and/or sub-basins in China were analyzed using linear regression models. In the analysis, the percent change in number of fish species was included as the dependent variable, while the flow indicators were implemented as the predictor variables. First, single-predictor regression models were created for each flow indicator to test for significant relationships between changes in fish species richness and alterations in specific flow components. Then multiple regression models were built for the combined flow indicators to test the significance and fit of models, i.e. to select the model that best represents the associations between fish species richness and combined flow components. The above regressions were performed by excluding intercepts from the models and by forcing the regression lines to go through the origin, because percent change in flow indicators and in number of fish species were assumed to be equal to zero simultaneously before the human impacts have occurred.

Second, except for the indicators of long-term average river discharge (I_{LTD}) and seasonal regime (I_{SR}), the alterations in low flow (I_{Q90}), high flow (I_{Q10}) and seasonal amplitude (I_{SA}) showed both increased and decreased trends, while the percent change in fish species richness were all negative. In other words, according to the data, no matter what changes occurred in those indicators, they all related to reduction of number of fish species, and this was likely to be a factor that might influence the fit of the regression lines to the data. Therefore, the analyses were performed again based on the absolute changes in I_{LTD}, I_{Q90}, I_{Q10} and I_{SA}, and the changes in I_{SR} (alterations in seasonal regime were already represented as absolute values) using the same methods as used in the first step.

Running a regression without an intercept may lead to a pitfall that the slope estimator might be biased (Hocking 1996). The linear model with the intercept term has the form

$$Y_i = \beta_0 + \beta_1 x_i + e_i \tag{1}$$

where β_0 is the intercept, β_1 is the slope and e_i denotes the *i*th residual. When the intercept is dropped out, the form is transformed into

$$Y_i = \beta_1 x_i + e_i \tag{2}$$

If the data plots are far from the original, the least squares estimator for the slope in a no-intercept model will be systematically shifted towards larger or smaller values and makes the analysis meaningless. Hence, as the third step, responses of fish species richness to the original and the absolute changes in the five flow indicators were estimated by multiple regression models with the intercept terms to test whether they could provide superior fits.

4.2.4 Testing model performance

In this study, the goodness of fit of all regression models was tested using the following measures: coefficient of determination, standard error of estimate and Akaike Information Criterion. Moreover, the p-value was used to test whether each regression model can offer a good fit to the data, and how significant does each of the flow indicators influence fish species richness in the models.

The coefficient of determination (denoted by r^2 in a single regression model and R^2 in a multiple regression model) is a value that indicates how well the data fit a regression model. It is interpreted as the proportion of the variance in the dependent variable that is predictable from the independent variable (Rawlings et al., 1998), thus, it ranges from 0 to 1. An R^2 of 0 means that the dependent variable cannot be predicted from the independent variable and an R^2 of 1 indicates that the dependent variable can be predicted without error from the independent variable. The R-squared value for the regression with intercept is computed as

$$R^2 = \frac{\sum(\hat{Y}_i - \bar{Y})^2}{\sum(Y_i - \bar{Y})^2} \tag{3}$$

or equivalently

$$R^2 = 1 - \frac{\sum(Y_i - \hat{Y}_i)^2}{\sum(Y_i - \bar{Y})^2} \tag{4}$$

where $\bar{Y}$ denotes the mean of the dependent variable, Y_i indicates the ith dependent variable and $\hat{Y}_i$ is the ith fitted value. The term on the top right of the equation (3) is the sum of squares due to regression, the term on the bottom right is total sum of squares and the term on the top right of the equation (4) is the sum of squares due to error. However, for the regression model without an intercept, if the model provides a sufficiently poor fit, the data may exhibit more variation around the regression line than around $\bar{Y}$, in which case $\sum(Y_i - \hat{Y}_i)^2 > \sum(Y_i - \bar{Y})^2$ (Eisenhauer 2003). In this case, applying equation (3) and (4) may result in an implausible negative coefficient of determination. Thus the following equation was developed and has been adopted by many software packages such as SPSS and Excel in calculating R-squared value for regression through origin:

$$R^2 = \frac{\sum \hat{Y_i}^2}{\sum Y_i^2} \tag{5}$$

where $\sum \hat{Y_i}^2$ indicates the sum of squares due to regression and $\sum Y_i^2$ refers to total sum of squares. By applying this equation, the calculated R-squared value can be absurdly large even when the correlation between dependent variable and independent variable is weak, and then makes the estimation meaningless.

For the no-intercept models in this study, the R-squared values were calculated as the square of the correlation between observed and predicted y scores (i.e. percent change in fish species richness) according to the suggestion of Hocking (1996), and they cannot be compared to the values for models which include intercepts. Therefore, standard error of estimate was selected as a good measure in comparing the model fits of the regressions with and without intercepts in this analysis.

The standard error of estimate (SEE) measures how well a least square line equation fits a data set. It is computed as the square root of the sum of squares for error divided by the degrees of freedom. For a multiple regression, SEE is defined as

$$SEE = \sqrt{\frac{\sum(Y-\hat{Y})^2}{n-k-1}} \tag{6}$$

where Y denotes an actual dependent variable, $\hat{Y}$ indicates a predicted dependent variable, n is the number of data points in the sample and k is the number of independent variables in the regression model (Sheskin 2007). The smallest value of SEE is zero which represents all the data points fall along the equation line. The model with the smallest standard error of estimate is the best fit for the sample when compare to other models.

The Akaike Information Criterion (AIC) is a measure of the relative goodness of fit of a statistical model for a given data set. In other words, it provides a way of selecting a model from a set of models for the data by estimating the quality of each model that relative to the other models (Burnham and Anderson 2002). The chosen model is the one that minimizes the loss of information between the model and reality. If only poor models are considered, the AIC will select the best of the poor models (Mazerolle 2006). AIC is generally defined as

$$AIC = 2k - 2ln(L) \tag{7}$$

where k is the number of parameters in the model and L denotes the maximized value of the likelihood function for the model. For the least squares regression models, AIC is computed with the following equation:

$$AIC = n \times ln(\frac{RSS}{n}) + 2k \tag{8}$$

where n is the sample size and RSS is sum of squares for error.

In this study, the model selection was performed by using the second-order Akaike Information Criterion (AIC_c) instead of AIC due to small sample size. The form of AIC_c is defined as:

$$AIC_c = AIC + \frac{2k(k+1)}{n-k-1} \tag{9}$$

As sample size increases, the last term of the AIC_c reaches zero, and the AIC_c tends to yield the same conclusions as the AIC (Burnham and Anderson 2002). All regression models were divided into two groups, with and without intercepts. For these groups, the model with the minimum AIC_c value was chosen as the best model. Furthermore, the ΔAIC_c was adopted to measure how well each model could be relative to the best model within two groups, and it was calculated as

$$\Delta AIC_c = AIC_{ci} - AIC_{cmin} \quad (10)$$

where AIC_{ci} is the AIC_c value for model i, and AIC_{cmin} is the AIC_c value of the best model. As a rule of thumb, a $\Delta AIC_c \leq 2$ suggests substantial support for the model, and value with $2 < \Delta AIC_c \leq 7$ indicates that the model has less support, while a $\Delta AIC_c > 10$ means that the model is very unlikely (Burnham and Anderson 2002).

The p-value is a measure of determining the significance of a model result within a statistical hypothesis test. It is calculated as the smallest level of significance at which the null hypothesis would be rejected. The smaller the p-value, the stronger the evidence supports the alternative hypothesis (Aczel 1993). In other words, the lower the p-value, the more significant the model result is. Generally, one rejects the null hypothesis if the p-value is lower than 0.05 or 0.01. In other words, if the p-value is smaller than 0.01, the impact of the flow indicator is "very significant". If the p-value is between 0.01 and 0.05, the impact of the flow indicator is considered as "significant", while when the p-value is higher than 0.1, the impact of the flow indicator is "not significant".

All estimations and data analysis in the Sect. 4.2.3 as well as evaluation of model performance were performed using XLSTAT 2010 statistical analysis software, which was developed by Addinsoft based on MS Excel interface.

4.3 Results

The relationships between losses of fish species richness and flow indicators in the river basins sampled in this study were estimated by a collection of single and multiple linear regression models with and without intercepts for both original and absolute changes in flow indicators. The goodness of fit tests indicated that the models with intercepts fitted the sampled data better than those without intercepts because they could provide much lower values of standard error of estimation (SEE) and AIC_c (second-order Akaike Information Criterion) as compared to the no-intercept models (see Appendix D1, D2, D3 and D4 for details). Thus, those no-intercept models were not included in the further analysis. Coefficient of determination (R^2) ranged from 0.0004 to 0.31 for the 52 multiple regression models with intercepts, in which 40 models have R-squared values from 0.10 to 0.31. The values are modest but still reasonable for the analysis based on small sample size.

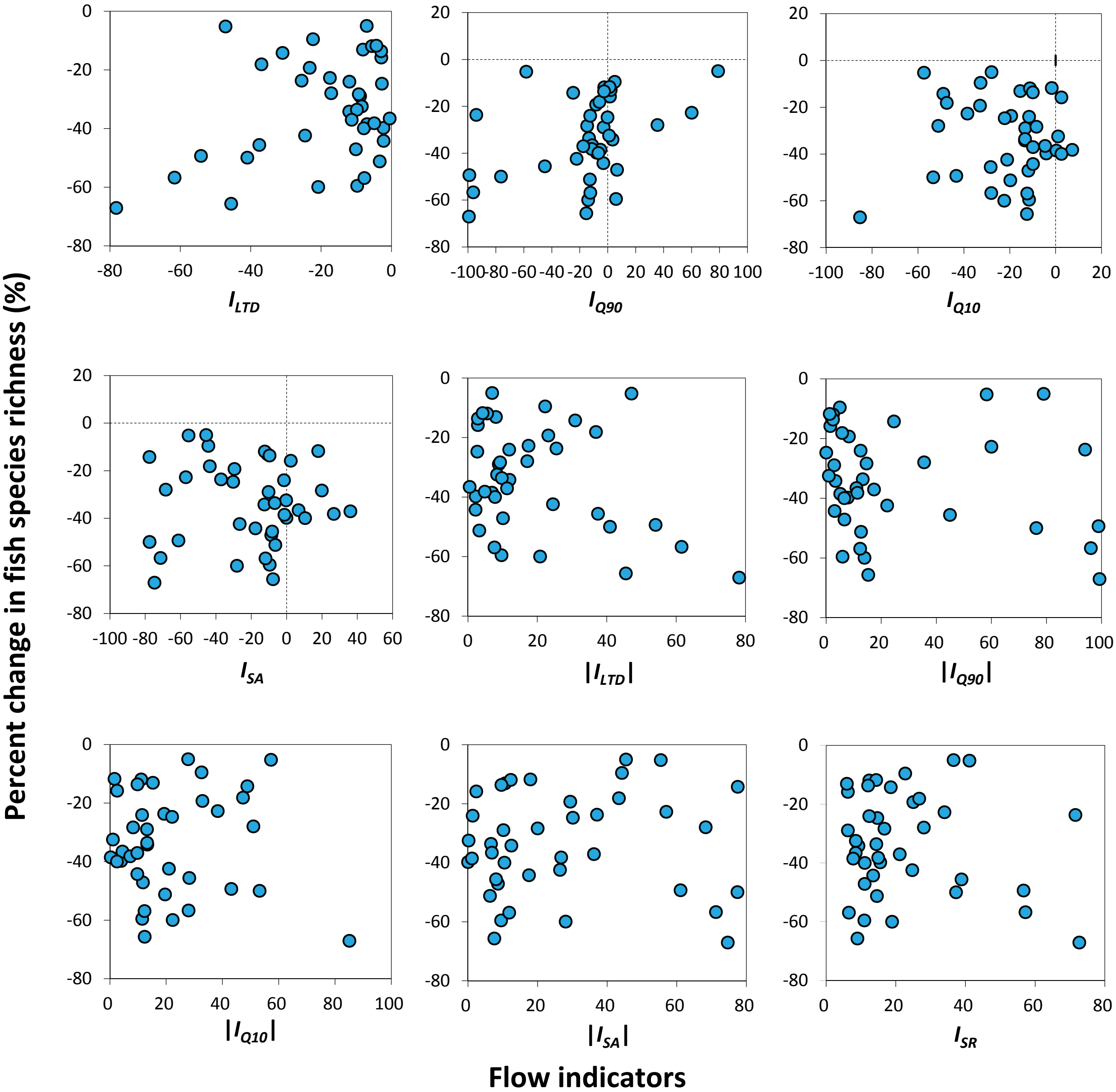

Fig. 4.2. Percent change in fish species richness with respect to percent alterations (both original and absolute) of the five flow indicators in 34 river basins and/or sub-basins in China (see Table 4.2 for the description of each indicator). Percent change of fish species richness and flow indicators denotes alterations relative to the reference condition.

Quantitative estimates of relationships between changes of fish species richness and alterations in each of the flow indicators were shown in Fig. 4.2. Fish species richness consistently declined in response to decreases in long-term average annual river discharge (I_{LTD}), decreases and increases in statistical low flow (I_{Q90}), statistical high flow (I_{Q10}) and seasonal amplitude (I_{SA}), and to absolute changes in I_{LTD}, I_{Q90}, I_{Q10}, I_{SA} and I_{SR} (alterations in seasonal regime). The results of the single linear regression with intercepts indicated that I_{Q90} has the lowest value of standard error of estimate (SEE = 16.00) and the highest value of coefficient of determination ($r^2 = 0.17$) among all the indicators, followed by I_{LTD} and absolute I_{LTD} (SEE = 16.48, $r^2 = 0.11$). The p-values of these variables denoted that, for the given data, the influence of I_{Q90} (original changes) and I_{LTD} (original and absolute changes) on fish species richness were statistically significant ($p < 0.05$, see Appendix D3 and D4), while the rest indicators did not show significant relationships ($p > 0.1$).

For the multiple regressions with intercepts, the models with the best fits to the data were selected from all 52 candidate models according to the AIC_c (second-order Akaike Information Criterion) values and their scores relative to the best model (ΔAIC_c). As a result, absolute changes in long-term average annual discharge (I_{LTD}) and absolute changes in statistical low flow Q_{10} (I_{Q10}) were detected in the best model (Table 4.4). The indicator of I_{LTD} (original and absolute changes) was consistently included in all of the eight best-fitting models ($\Delta AIC_c \leq 2$, marked with a double asterisk) and has consistent plausible regression coefficient signs and very significant effects ($p < 0.01$, except for the model C16 which I_{Q90} has marginal effect on I_{LTD} and resulted in a p-value of 0.07). The rest flow indicators other than statistical low flow (I_{Q90}) consistently showed implausible regression coefficient signs, and therefore could not be considered as the useful indicators in explaining the reduction of fish species richness in this analysis. Percent change in statistical low flow Q_{90} appeared in one best-fitting model, however, with a high p-value ($p = 0.48$), it has no significant effects on losses of fish species compared to the indicator of I_{LTD} for the given data.

Table 4.4 Summary of model selection for the linear models with intercepts

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c	SEE	Regression coefficient	p-value of variable
d6	$\lvert I_{LTD} \rvert$	0.249	0.005	221.846	0.000***	15.379	-0.737	0.001
	$\lvert I_{Q10} \rvert$						0.533	0.014
d7	$\lvert I_{LTD} \rvert$	0.241	0.006	222.267	0.421**	15.460	-0.628	0.002
	$\lvert I_{SA} \rvert$						0.348	0.017
c7	I_{LTD}	0.246	0.005	221.985	0.139**	15.406	0.726	0.001
	I_{Q10}						-0.501	0.015
d18	$\lvert I_{LTD} \rvert$	0.269	0.010	223.106	1.260**	15.382	-0.753	0.001
	$\lvert I_{Q10} \rvert$						0.336	0.248
	$\lvert I_{SA} \rvert$						0.119	0.327
d19	$\lvert I_{LTD} \rvert$	0.262	0.011	223.478	1.632**	15.453	-0.838	0.002
	$\lvert I_{Q10} \rvert$						0.489	0.029
	I_{SR}						0.186	0.427
c8	I_{LTD}	0.217	0.011	223.527	1.681**	15.705	0.596	0.003
	I_{SA}						-0.255	0.034
c20	I_{LTD}	0.260	0.012	223.570	1.724**	15.471	0.832	0.004
	I_{Q10}						-0.460	0.030
	I_{SR}						0.192	0.412
c16	I_{LTD}	0.257	0.013	223.767	1.921**	15.509	0.573	0.065
	I_{Q90}						0.073	0.480
	I_{Q10}						-0.441	0.047

***The best model with the lowest AIC_c value.

** The best-fitting model with $\Delta AIC_c \leq 2$.

In summary, for the given data, indicator of I_{LTD} appeared complete dominance over the other flow indicators included in this study. Similar results were obtained by evaluating the outputs of the rest 44 models with $\Delta AIC_c > 2$ (see appendix D3 and D4). As a conclusion, percent change in fish species richness was positively correlated to original changes in long-term average annual river discharge and negatively related to the absolute changes in I_{LTD}.

4.4 Discussion

The purpose of this study was to quantify the relationships between losses of fish species and impacts of anthropogenic flow alterations in China using simulated flow indicators and fish data obtained from published papers. As expected, the quantitative analysis clearly indicated that the long-term average annual discharge is an important indication in quantifying the relationships between river flow alterations and loss of fish species richness in China. This finding corroborates the earlier conclusions of using average discharge to estimate losses in fish diversity on a large scale (Xenopoulos et al., 2005; Xenopoulos and Lodge 2006; Iwasaki et al., 2012). Furthermore, as a large proportion of the fish data included in this research was extracted at sub-basin scale, it also potentially supports the suggestion of McGarvey and Hughes (2008) that it is preferable to derive species-discharge relationship using individual river reaches rather than entire river basins. However, the conclusion that absolute changes in long-term average annual discharge and losses of fish species richness were negatively related might lead to a tricky, that is, increases in average discharge will result in decreases in fish species richness. With respect to ecological responses to increased discharge, Xenopolous et al. (2005) noted that consequences of increased discharge for freshwater biodiversity are highly uncertain. Hence, such uncertainties should be carefully considered when selecting an absolute value as the predictor variable.

A dataset with related to diversity of native freshwater fishes for different time periods and extinction rates of those fish species in 34 river basins and sub-basins in China was integrated based on an extensive literature review. Besides, another dataset regarding dynamics of different flow indicators in above-mentioned river basins was generated using modeled river discharge for the time periods corresponding to the fish records. As little information is available concerning the Chinese river basins in previous studies (e.g. Xenopoulos et al., 2005; Poff and Zimmerman 2010; Iwasaki et al., 2012), the finding of the present study provides substantial supplements and additional knowledge to the future studies with respect to quantitative estimation of relationships between ecological responses and flow alterations at global scale.

In general, the analysis does not provide direct and clear evidences for developing quantitative relationships between losses of fish species richness and changes in flow components other than average discharge. The possible reasons for this poor performance can be summarized into several aspects.

Firstly, regarding the fish data that were extracted from various studies, even though the authors noted that changes of fish diversity in their research were mainly due to flow alterations, however, in reality, decreases of fish species were likely to be significantly associated with other factors, i.e. increased level of pollutants, changes in sediment transport, loss of connectivity to the wetlands or floodplains and introduction of non-native fish species in addition to river flow alterations. In this case, adoption of the fish data with such noise might strongly influence the regression procedure and thus lead to incorrect estimation of relationships between losses of fish species and flow indicators. For example, in middle mainstream of the Yangtze River, the number of fish species decreased by 60%, while the flow indicators increased or decreased by less than 12% (see Table 4.1 and Table 4.3). In this case, the significant decreases of fish species are likely due to the increased pollution level in this sub-basin. One possible solution for the problem is adding additional parameter with respect to the environmental factors (e.g. pollution concentration or sediment discharge) that might have strong impacts on fish diversity and remove the noisy data which are highly correlated with those factors. Nevertheless, influences of such factors could not be considered in this study due to lack of necessary information.

Secondly, alterations of the five flow indicators represented in this analysis might be certainly underestimated due to underestimation of impacts of reservoirs on discharge regimes. The five flow indicators were calculated based on discharge data which were simulated by global hydrological and water use model WaterGAP2.2, in which the reservoir operation is computed using a general reservoir algorithm. While in reality, reservoir operation is performed in a very site-specific manner that cannot be modeled very well by this algorithm (Döll et al., 2009). In addition, the number of reservoirs and regulated lakes included in this study is much smaller than the actual number of reservoirs in the sampled river basins. As a result, the impacts of the reservoirs on river discharge dynamics are likely to be underestimated and thus lead to underestimates of the changes in the five flow indicators. Such underestimates might lead to biased results during the statistical analysis in this study.

Finally, with respect to the regression analysis in this research, inappropriate predictor variables limited the chances of developing significant relationships between losses of fish species and alterations in flow components. Some of the flow indicators selected in this analysis was highly intercorrelated among each other, and this is referred to as multicollinearity. The major consequences of multicollinearity include two aspects: it may prevent any of the individual predictors (in particular the predictors that are problematic) from being significant (Dewberry 2004), and conflicting conclusions can be obtained from the tests of significance (i.e. wrong signs for the regression coefficient). One case regarding the first consequence is the model performance for the indicator of statistical low flow I_{Q90}. I_{Q90} was detected in one best-fitting model, but it was statistically significant only for the models which the indicator of long-term average annual discharge (I_{LTD}) was excluded (e.g. model c10, c11, c12, c22, c23, c24, and c30; Appendix D3). The correlation between I_{Q90} and I_{LTD} was 0.71 (see the correlation metrics in Appendix D5), which indicated that the two variables were highly correlated and I_{Q90} might become non-significant due to the marginal effect of I_{LTD}. For the second consequence, variables with the wrong signs of regression coefficient were detected in all models (Please note that the models discussed here are all intercept-allowed models). For example, in model c20, the indicator of statistical high flow I_{Q10} is highly correlated with I_{LTD} (r = 0.77) and the indicator of seasonal regime I_{SR} (r = -0.69), therefore, although the correlation between losses of fish species and I_{Q10} is positive, the regression coefficient of I_{Q10} still shows implausible negative sign (Appendix D3). As a conclusion, superior indicators which represent alterations in flow components and are not highly collinear should be adopted instead of I_{Q90}, I_{Q10}, I_{SA} and I_{SR} in future analysis to improve the quality of the estimation. Iwasaki et al. (2012) evaluated relationships of fish species richness to 14 ecologically relevant flow metrics in 72 rivers worldwide and suggested that CV in frequency of low flow and CV in Julian date of annual minimum flow are important low flow indices in quantitative estimates of responses of fish species to flow alterations. Moreover, McGarvey (2014) explored associations between 148 flow indices and fish species richness in 89 rivers in the Pacific Northwest (USA) and noted that three indices of episodic high flow events, i.e. median large flood rise rate, CV of 1-day maximum flow and median high flow timing may be good flow indicators in predicting the changes of fish species richness in the study sites. Nevertheless, these indicators cannot be used in this study due to lack of observed time series of daily discharge. Although the improved version of hydrological model WaterGAP has the capacity to calculate daily flows, but how well do observed and modeled results match has not been tested.

One critical limitation in this study is the inability to take into account the impacts of flow components other than flow magnitudes and environmental effects. Poff and Zimmerman (2010) stated that freshwater species respond to multiple hydrologic drivers and the drivers are normally confounded. Fish diversity can be affected by other environmental factors as well, e.g. pollution, dam block, variations in sediment transports, flow velocity, water temperature and introduction of non-native fish species. Therefore, the magnitude-oriented flow indicators may add bias to quantitative analysis and are very likely to overestimate the impacts of those indicators on diversity of native freshwater fishes in river basins in China and worldwide. Thus, application of flow metrics with related to duration of low- and high-flows and frequency of high flow pulse, as well as river pollution index and sediment discharge become necessary to identify associations between flow alterations and extinction of fish species in China in future studies. To do this, a further improved version of WaterGAP model that has capability to simulate daily river discharge in a proper manner can be used to generate daily flow regimes, while information of river pollution levels and sediment transport in the study sites can be obtained by literature review.

Another limitation is that lack of sufficient observations of fish diversity limited the ability to analyze the decreases in fish species richness overall China, in particular the arid western regions. The 49 reviewed Chinese studies provide number of fish species for different time periods for each of the 34 river basins and/or sub-basins. Most of these basins are located in the northeastern, the central and the southern parts of China and only one study reported loss of fish diversity as consequence of flow alteration in arid western areas. Thus, collecting fish survey data from unpublished research with respect to the western rivers may be helpful to increase the sample size of the future analysis, and provide more information for estimating the effects of hydroclimatic factors on response of fish species to flow alterations in China. Generally, although this study was not able to find statistical relationships between changes of fish species and flow indicators other than average discharge, it does not imply the relationships will not be found for other indicators based on the same approach.

4.5 Conclusion

This study has performed the first estimation on quantitative relationships between decreases in fish species richness and anthropogenic flow alterations in China. Five ecologically relevant flow indicators were identified and quantified based on the monthly discharges which were simulated by a global hydrological model WaterGAP, while the number of fish species for different time periods in 34 river basins and/or sub-basins was extracted from 49 published Chinese studies. A total of 360 data points represent relationships between changes of fish species richness and alterations (original changes and absolute changes) in the five flow indicators were obtained and then were analyzed using single and multiple regression models with and without intercepts.

The intercept-allowed models lead to better fits to the sampled data as compared to the no-intercept models, for they have much lower standard error of estimation (SEE) and AIC_c (second-order Akaike Information Criterion) values. For the analysis of single-predictor regressions with intercepts, significant linear relationships ($p < 0.05$) have been detected for percent change (original and absolute) in long-term average annual discharge (I_{LTD}) and percent change (original) in statistical low flow (I_{Q90}), while no significant relationship ($p > 0.1$) has been found for the rest indicators, i.e. percent change in statistical high flow (I_{Q10}), percent change in seasonal amplitude (I_{SA}) and percent change in seasonal regime (I_{SR}). For the analysis of multiple regressions with intercepts, coefficient of determination (R^2) of most models range from 0.10 to 0.31. The indicator of long-term average annual discharge has been detected in all of the best-fitting models ($\Delta AIC_c \leq 2$) and has very significant effects ($p < 0.01$) on explaining the changes in fish species richness. The indicator of statistical low flow (I_{Q90}) has appeared in one best-fitting model, but no significant effect ($p > 0.1$) has been detected with respect to losses of fish species mainly due to the influence of multicollinearity. The rest of the flow indicators cannot be used because they have consistently implausible regression coefficient signs. Two major conclusions are reached in this study, i.e. changes in fish species richness are positively correlated to alterations in long-term average annual discharge in China, and indicator of I_{LTD} is dominant over all other flow indicators included in this research.

Quantitation of relationships between changes in fish species richness and alterations in flow magnitudes is the first stage in development of environmental flow guidelines for the rivers in China. The further stage would be the quantitative estimation of responses of fish species to changes in average river discharge and other flow components, e.g. timing, frequency and frequency of discharge (Richter et al., 1996, 1998; Poff et al., 1997; Scott et al., 1997), and to environmental factors, e.g. dynamics of pollutant concentrations, and sediment discharge. This approach has the potential to reduce the overestimates of the impacts of anthropogenic flow alteration on freshwater fish species, and to provide environmental flow guidelines for the sustainable water resources management in rivers with high risk of diversity loss in China.

Supporting Information

Appendix D1. Summary of the model performance for linear models without intercepts based on original values of flow indicators

Appendix D2. Summary of the model performance for linear models without intercepts based on absolute values of flow indicators

Appendix D3. Summary of the model performance for linear models with intercepts based on original values of flow indicators

Appendix D4. Summary of the model performance for linear models with intercepts based on absolute values of flow indicators

Appendix D5. Reference list of 49 published studies that provide data of fish species richness at basin or sub-basin scale in China.

Chapter 5: Synthesis

In this chapter, the major findings of anthropogenic flow alterations and their impacts on freshwater ecosystem in China have been summarized and presented according to the main objectives of this study (see Sector 1.3 for detailed description). Implications of these findings were also provided for environmental flow guidelines and sustainable water management in China's river basins and for the future studies.

5.1 Anthropogenic alterations in river flow regimes in China

5.1.1 Changes in flow metrics with relevance to biotic components

Comparisons between natural and anthropogenically altered conditions for the selected ecologically relevant flow indicators revealed that total annual river discharge into oceans and internal sinks as well as discharge at international boundary for the whole of China has decreased by 6%.

At macroscale level, around 30% of China's total land area has suffered from large decreases (more than 10%) in flow indictors describing long-term annual river discharge, low flow Q_{90}, high flow Q_{10}, and seasonal amplitude due to water withdrawals and dams, while seasonal flow variability has been significantly changed on 40% of total land area. Moreover, low flow Q_{90} has increased by more than 10% within 12% total land area downstream of dams.

Generally, great alterations in natural flow regime occurred in large part of northern China and only minor changes were found in most of southern China.

5.1.2 Finding implication

Determining natural and anthropogenically altered flow regimes by a global hydrological and water use model, which combines 731 artificial reservoirs and 2 regulated lakes in China, could provide valuable information with respect to evaluation of alterations in river flow regimes all over China due to human water use and dam operation. Comparison between natural and altered flow conditions could be used to identify the degradation of aquatic habitats, estimate alterations in flow metrics other than flow magnitude, such as duration, frequency and rate of change, and support a foundation of assessment of impacts of changes in water quality on riverine ecosystems in China in future studies.

Flow alterations have occurred in most of China's rivers to a certain extent. Although the overall reduction in average discharge is relatively small, the low- and high-flow conditions as well as seasonal variability have been significantly altered in China mainly due to river flow regulation by dams. It should be noticed by the managers and policy makers in China that natural flow regimes and associated aquatic and riparian ecosystems are likely to be under pressures, thus rational planning and development of water resources should be considered in the future management.

5.2 Quantitative relationships between ecological responses and anthropogenic flow alterations Methods

One of the main objectives of this study was to develop linear relationships of ecological responses to anthropogenic flow alterations in China's river basins based on information that could be extracted from published Chinese studies. According to the papers, a majority of the ecological changes were resulted from alterations in flow magnitude, most commonly as decreases in average river discharge. Ecological responses were largely demonstrated as negative responses of the most ecological groups, such as fish, macrophyte and riparian vegetation, while positive responses to reduced flow metrics were reported for planktons and waterbirds. Quantitative relationships between ecological responses and alterations in flow metrics in China were developed among the following three ecological categories: fish, riparian vegetation and plankton.

5.2.1 Impacts of altered flow magnitude on fish

Fish diversity and fish catch decreased consistently in response to reduced flow magnitude in China, and these variables are well correlated ($r = 0.66$) with each other. Around 40% of changes in fish could be explained by alterations in average river discharge as well as low- and high-flow conditions ($r^2 = 0.43$). Furthermore, 4.8-92%

decreases in flow magnitude could have caused 6.9-99.9% losses in fish diversity and fish catch in China, while an increase of 8.4% in average river discharge might lead to 1.8 % increase in relative fish abundance.

5.2.2 Impacts of altered flow magnitude on riparian vegetation

Vegetation cover and biomass of riparian vegetation showed almost negative responses to reduced flow magnitude, while vegetation cover and growth rate responded positively to increased average river discharge. Vegetation cover, biomass and growth rate of riparian vegetation highly correlated with changes in average river discharge ($r = 0.77$) and more than 60% of variations in riparian vegetation could be explained by altered flow magnitude. Generally, 12-89% reductions in average river discharge resulted in 4-90.3% decreases in coverage of riparian vegetation, while 26.4-171% increases in average river discharge might lead to 2.5-172.2% of increases in both vegetation cover and growth rate of riparian vegetation in China.

5.2.3 Impacts of altered flow magnitude on plankton

Mixed responses of plankton to alterations in flow magnitude were found in this study. Diversity and abundance of most sensitive plankton species reduced as result of either increased or decreased river flows, while some tolerant species showed significantly positive response (113-2354% increases) to reduced high flow (12-83% decreases) and increased low flow (6% increase).

5.2.4 Impacts of altered average river discharge on riparian vegetation and fish in different climatic regions in China

Hydro-ecological relationships are dependent on local landscape, particularly climate and geomorphy (Poff and Ward, 1989; Arthington et al., 2006). Since general relationships could not be developed from all responses of reported ecological categories to flow alterations in China, a supplementary analysis was performed on responses of specific ecological assemblage to climate-driven and anthropogenically altered flow components. Consequently, linear relationships of changes in riparian vegetation and fish to alterations in average river discharge in arid and semi-arid region and/or humid region were determined.

As expected, riparian vegetation cover was significantly correlated with altered average river discharge ($r = 0.79$) in arid and semi-arid region, and more than 60% of the variations in vegetation cover could be explained by changes in average river discharge.

Fish catches showed strong correlations to altered average river discharge in both arid and humid regions ($r = 0.78, 0.77$) and more than half of the changes in arid and semi-arid region as well as humid region were determined by altered average river discharge. According to the findings, fish are more sensitive than other ecological groups included in this study, when flow alterations occur.

5.2.5 Finding implication

The findings of this study indicated that magnitude and direction of ecological responses to hydrological changes depend largely on characteristics of ecological groups and types of flow alteration. Thus, developing relationships between responses of specific ecological group or species-specific responses and flow metrics could improve the capability of quantifying the impacts of anthropogenically altered flow regimes on freshwater ecosystems in China and worldwide.

Furthermore, stronger relationships could be derived by including more data points for the whole range of changes in flow regimes, particularly the changes with respect to low to moderate range. Such information could provide supplementary for future research regarding evaluation of hydro-ecological relationships.

Riparian vegetation and fish were strongly influenced by hydroclimate associated flow alterations in China. This finding corroborated the suggestion of Arthington et al. (2006) that hydro-ecological relationships could be affected by climate and geology. Therefore, environmental flow requirements and degree of impacts of flow alterations on aquatic and riparian ecosystems will differ within climatic regions in China.

In this study, fish showed rapid responses to either decreased or increased flow magnitude. Around 5-13% reduction in flows led to 12-41% decrease in fish catch during the early impact period, while both fish diversity and fish catch declined more than 55%, when decreases in flow magnitude exceeded 50%. Thus, fish can be included in the further studies as a good predictor of flow alterations.

The datasets produced by this study, which combines ecological and hydrological observations in eleven river basins and watersheds in China, provided valuable knowledge for future studies regarding assessment of impacts of flow alterations on riverine ecosystems in China and worldwide

5.3 Quantitative relationships between changes of fish species and alterations in flow indicators

The aim of this study was to quantify the relationships between losses of fish species and ecologically relevant flow alterations in 34 river basins and/or sub-basins in China according to the number of fish species that could be extracted from 49 published Chinese studies and the flow indicators that were computed by a global hydrological and water use model WaterGAP. Reference and altered conditions for both fish species richness and flow indicators were identified and compared.

5.3.1 Losses of fish species richness in response to altered flow indicators

As expected, the results clearly demonstrated that long-term average annual discharge is an important flow indicator to identify the relationships between declines of fish species richness and anthropogenic flow alterations in China, and alterations (original changes) in long-term average annual discharge can be positively associated with changes of fish species richness, while other indicators analyzed in this study cannot provide any meaningful information because they are highly correlated with other indicators and appear collinear. Therefore, I_{LTD} become dominant over all other flow indicators included in this analysis.

Moreover, a dataset with respect to diversity of native freshwater fishes at different time periods in 34 river basins and/or sub-basins in China was created based on the fish records which were reported in 49 published Chinese literature. Another dataset regarding dynamics of five flow indicators was generated according to river discharges that were modeled by the global hydrological model WaterGAP.

5.3.2 Finding implication

In General, even though the study did not provide direct evidences for defining quantitative relationships between changes of fish species richness and alterations in flow components other than average discharge, it does not imply the relationships would not be detected for other indicators based on the same approach. A collection of superior indicators which represent dynamics in flow components and are not highly collinear should be implicated instead of I_{Q90}, I_{Q10}, I_{SA} and I_{SR} in future studies.

Globally, the fish species data regarding Chinese river basins are insufficient. The historical fish data produced by the present study may provide substantial supplements and additional information to future research with respect to quantitative relationships between ecological responses and flow alterations at global scale.

5.4 Future research direction Methods

All flow indicators included in this study are related to changes in flow magnitude. However, in reality, aquatic and riparian species are influenced by many hydrological drivers simultaneously. Thus, further study would be the quantification of responses of fish species to changes in average discharge and other flow components, e.g. timing, frequency and frequency of discharge by applying an improved hydrological model, which can produce daily discharge in a good manner.

Other environmental factors (e.g. pollutant concentrations and sediment discharge) affect freshwater ecosystems in China as well. Further research should take into account these influences by conducting more extensive literature review on published and unpublished studies in China.

Such an approach has the potential to lower the overestimates of the influences of anthropogenic flow alteration on freshwater fish species, and to provide environmental flow guidelines for the sustainable water resources management in rivers with high risk of diversity loss in China.

References

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., and Siebert, S. 2003a. Global estimates of water withdrawals and availability under current and future “business-as-usual” conditions. Hydrol. Sci. J., 48, 339–348.

Alcamo, J., Döll, P., Henrichs, T., Kaspar, F., Lehner, B., Rösch, T., and Siebert, S. 2003b. Development and testing of the WaterGAP 2 global model of water use and availability. Hydrol. Sci. J., 48, 317–337.

An, S. Q., Li, H. B., Guan, B. H., Zhou, C. F., Wang, Z. S., Deng, Z. F., Zhi, Y. B., Liu, Y. H., Xu, C., Fang, S. B., Jiang, J. H. and Li, H. L. 2007. China's natural wetlands: past problems, current status, and future challenges. AMBIO, 36(4), 335-342, doi: 10.1579/0044-7447(2007)36[335: CNWPPC]2.0.CO;2.

Arthington, A. H. and Pusey, B. J. 1993. In-stream flow management in Australia: methods, deficiencies and future directions. Australian Biology, 6, 52–60.

Arthington, A. H., Bunn, S. E., Poff, N. L. and Naiman, R. J. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. Ecological Applications, 16, 1311 -1318.

Arthington, A. H., Mackay, S. J., James, C. S., Rolls, R. J., Sternberg, D., Barnes, A. and Capon, S. J. 2012. Ecological limits of hydrologic alteration: a test of the ELOHA framework in south-east Queensland, Waterlines report, National Water Commission, Canberra.

Aczel, A.D. 1993. Complete business statistics, 2nd edn. Irwin, Homewood.

Beschta, R. L. and Jackson, W. L. 1979. The intrusion of fine sediments into a stable gravel bed. Journal of the Fisheries Research Board of Canada. 36, 204 –210.

Black, A. R., Rowan, J. S., Duck, R. W., Bragg, O. M., and Clelland, B. E. 2005. DHRAM: a method for classifying river flow regime alterations for the EC Water Framework Directive, Aquat. Conserv. 15, 427–446.

Bunn, S. E. and Arthington, A. H. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30, 492-507.

Burnham, K. P., and Anderson, D. R. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer, New York.

Cohen, J., Cohen, P., West, S.G. and Aiken, L.S. 2003. Applied multiple regression correlation analysis for the behavioral sciences. Lawrence Erlbaum Associates, New Jersey, 95-100.

Clausen, B. and Biggs, B. J. F. 2000. Flow variables for ecological studies in temperate streams: groupings based on covariance. J. Hydrol., 237, 184–197.

Ding, R. H. (1989). The fish resources and the problems of fisheries management in Tuojiang. Resources exploitation and protection. 5(3): 13-19. (in Chinese)

Döll , P. and Fiedler, K. 2008. Global-scale modeling of groundwater recharge. Hydrol. Earth Syst. Sci., 12, 863–885.
http://www.hydrol-earth-syst-sci.net/12/863/2008/.

Döll, P. and Zhang, J. 2010. Impact of climate change on freshwater ecosystems: a global-scale analysis of ecologically relevant river flow alterations. Hydrol. Earth Syst. Sci. Discuss., 7, 1305-1342, doi:10.5194/hessd-7-1305-2010.
www.hydrol-earth-syst-sci-discuss.net/7/1305/2010/

Döll, P., Fiedler, K., and Zhang, J. 2009. Global-scale analysis of river flow alterations due to water withdrawals and reservoirs. Hydrol. Earth Syst. Sci., 13, 2413–2432.
http://www.hydrol-earth-syst-sci.net/13/2413/2009/.

Döll, P., Hoffmann-Dobrev, H., Portmann, F. T., Siebert, S., Eicker, A., Rodell, M., Strassberg, G. and Scanlon, B. R. 2012. Impact of water withdrawals from groundwater and surface water on continental water storage variations. Journal of Geodynamics, 59-60, 143-156.

Döll, P., Kaspar, F., and Lehner, B. 2003. A global hydrological model for deriving water availability indicators: model tuning and validation. J. Hydrol., 270(1–2), 105–134.

Duan, J. R., Zhang, H. Y., Liu, K. and Shi, W. G. 2007. Study on Biodiversity of Fisheries Resources in Yangtze River's Lower Reaches Based on GIS. Journal of Jimei University (Natural Science). 12(3): 221-225. (in Chinese with English abstract)

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z., Knowler, D., Lévêque, C., Naiman, R. J., Prieur-Richard, A. H., Soto, D., Stiassny, M. L. J. and Sullivan, C. A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews, 81, 163-182.

Dewberry, C. 2004. Statistical Methods for Organizational Research: Theory and Practice. London, Routledge.

Ellis, S., Leatherwood, S., Bruford, M., Zhou, K. and Seal, U. 1993. Baiji (Lipotes vexillifer) population and habitat viability assessment. Preliminary report, In: Species, 20, 25-28.

Eisenhauer, J. G. 2003. Regression through the Origin. Teaching Statistics, 25(3), 76-80.

Feng, Q., Liu, W., Si, J. H., Su, Y. H., Zhang, Y. W., Cang, Z. Q. and Xi, H. Y. 2005. Environmental effects of water resource development and use in the Tarim River basin of northwestern China. In Environ Geol., 48 (2), 202–210. http://link.springer.com/article/10.1007%2Fs00254-005-1288-0#page-1.

Froese, R. and Pauly, D. (eds) 2000. FishBase 2000: concepts, design and data sources ICLARM, Philippines.

Gleick, P. H. 1996. Water resources. Encyclopedia of Climate and Weather (ed. S. H. Schneider). Oxford University Press, New York, USA. 817-823.

Government of China 1994. White paper on China's strategy for population, environment and development in the 21st century, chapter 14. http://www.acca21.org.cn/ca21pa.html.

Hawksworth, D. J. and Kalin-Arroyo, M. T. 1995. Magnitude and distribution of biodiversity. Global biodiversity Assessment (ed. V. H. Heywood). Cambridge University Press, Cambridge, U.K. 107–191.

Hocking, R. R. 1996. Methods and applications of linear models: regression and analysis of variance. New York: John Wiley.

Huang, G. and Xie, P. 1996. Changes in the structure of fish community with the analysis on the possible reasons in Lake Donghu, Wuhan. Acta Hydrobiologica Sinica 20, 38–46.

Hunger, M. and Döll, P. 2008. Value of river discharge data for global-scale hydrological modeling. Hydrol. Earth Syst. Sci., 12, 841–861. http://www.hydrol-earth-syst-sci.net/12/841/2008/.

Iwasaki, Y., Ryo, M., Sui P. and Yoshimura, C. 2012. Evaluating the relationship between basin-scale fish species richness and ecologically relevant flow characteristics in rivers worldwide. Freshwater Biology. 57(10), 2173–2180.

Jenkins, M. 2003. Prospects for biodiversity. Science, 302, 1175–1177.

Kennard, M. J., Mackay S. J., Pusey, B. J., Olden, J. D., and Marsh, N. 2010. Quantifying uncertainty in estimation of hydrologic metrics for eco-hydrological studies, River. Res. Applic, 26, 137–156, 2010.

Kight, R. R., Gregory, M. B. and Wales, A. K. 2008. Relating streamflow characteristics to specialized insectivores in the Tennessee River Valley: a regional approach. Ecohydrology, 1, 394-407.

Lehner, B., Liermann, C. R., Revenga, C., Vörösmarty, C., Fekete, B., Crouzet, P., Döll, P., Endejan, M.L, Frenken, K., Magome, J., Nilsson, C., Robertson, J. C., Rödel, R., Sindorf, N. and Wisser, D. 2011. High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. Frontiers in Ecology and the Environment. 9, 494–502, http://dx.doi.org/10.1890/100125.

Li, D. X., Sun, W., Li, C. Y. and Rong, Q. 2005. Journal of Shandong Agricultural Administrators' College. 21(3), 128-132. (in Chinese)

Li, J. G., Li, G. B., Cui, H. M. and Wang, D. W. 2004. Degradation of reed wetland and protection in Baiyangdian. South-to-North Water Transfers and Water Science Technology, 2(3), 35-38, (in Chinese with English abstract).

Li, J., Luo, J. R., Li, X. H., Tan, X, C., Wang, C. and Guo, S. 2007. Investigation of fish resources and analysis of resources decline along the Lianjiang River. Freshwater Fisheries. 37(3), 49-53. (in Chinese with English abstract)

Liu, L. P. 1983. Fishery and enhancement of fish resources in the Bosten Lake. Freshwater Fisheries, 3, 12-14, (in Chinese).

Liu, S. P., Duan, X. B., Chen, D. Q., Liao, F. C. and Chen, W. J. 2005. Studies on status of fishery resources in the middle reach of the Yangtze River. Acta. Hydrobiologica Sinica. 29(6): 708-711. (in Chinese with English abstract)

Lloyd, N., Quinn, G., Thoms, M., Arthington, A. H., Gawne, B., Humphries, P. and Walker, K. 2003. Does flow modification cause geomorphologic and ecological response in river? A literature review from an Australian perspective. Technical Report 1/2004, CRC for Freshwater Ecology, ISBN 0-9751642-02.

Lytle, D. A. and Poff, N. L. 2004. Adaptation to natural flow regimes. Trends in Ecology & Evolution, 19, 94-100.

Mazerolle, M. J. 2006. Improving data analysis in herpetology: using Akaike's Information Criterion (AIC) to assess the strength of biological hypotheses. Brill Online Books and Journals, 27(2), 169-180. DOI: 10.1163/156853806777239922.

McGarvey, D. J. 2014. Moving beyond species–discharge relationships to a flow-mediated, macroecological theory of fish species richness. Freshwater Science 33(1), 18-31. doi: http://dx.doi.org/10.1086/674967

McGarvey, D. J. and Hughes, R. M. 2008. Longitudinal zonation of Pacific Northwest (USA) fish assemblages and the species-discharge relationship. Copeia, 2, 311-321.

McManamay, R. A., Orth, D. J., Kauffman, J. and Davis, M, M. 2013. A database and meta-analysis of ecological responses to stream flow in the south Atlantic region. Southeastern Naturalist, 12(5), 1–36.

Millennium Ecosystem Assessment 2005. Ecosystems and human well-being: biodiversity synthesis. Island Press, Washington. http://www.unep.org/maweb/documents/document.354.aspx.pdf

Müller Schmied, H., Eisner, S., Franz, D., Wattenbach, M., Portmann, F. T., Flörke, M. and Döll, P. 2014. Sensitivity of simulated global-scale freshwater fluxes and storages to input data, hydrological model structure, human water use and calibration. Hydrology and Earth System Sciences, 18, 3511-3538. http://doi.org/10.5194/hess-18-3511-2014

Oberdorff, T., Guegan, J. F. and Hugueny, B. 1995. Global scale pattern of fish species richness in river. Ecography, 18, 345-352.

Poff, N. L. and Ward, J. V. 1989. Implications of stream flow variability and predictability for lotic community structure: a regional analysis of stream flow patterns. Can. J. Fish. Aquat. Sci., 46, 1805–1818.

Poff, N. L. and Zimmerman, J. K. H. 2010. Ecological responses to altered flow regimes: a literature review to inform environmental flows science and management. Freshwater Biology 55:194-200.

Poff, N. L., Allan, D. J., Brain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E., and Stromberg, J. C. 1997. The natural flow regime: a paradigm for river conservation and restoration. Bioscience, 47(11), 769–784.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E. And Stromberg, J. C. 1997. The annual flow regime: a paradigm for river conservation and restoration. Bioscience, 47, 769-784.

Poff, N. L., Allan, J. D., Palmer, M. A., Hard, D. D., Richter, B. D., Arthington, A. H., Rogers, K. H., Meyer, J. L. and Stanford, J. A. 2003. River flows and water wars: emerging science for environmental decision making. Frontiers in Ecology and the Environment, 1, 298-306.

Poff, N. L., Angermeier, P. L. and co-authors. 2001. Fish diversity in streams and rivers. FS Chapin et al., eds. Global biodiversity in a changing environment: Scenarios for the 21st century. Springer-Verlag, New York. 315-349.

Poff, N. L., Olden, J. D., Merritt, D. M., and Pepin, D. M. 2007. Homogenization of regional river dynamics by dams and global biodiversity implications. P. Natl. Acad. Sci. USA, 104(14), 5732–5737.

Postel, S., and Richter, B. 2003. Rivers for Life: Managing Water for People and Nature . Island Press. Washington, D.C.

Power, M. E., Sun, A., Parker, G., Dietrich, W. E. and Wootton, J. T. 1995. Hydraulic food-chain models. Bioscience, 45, 159-167.

Puckridge, J. T., Sheldon, F.,Walker, K. F., and Boulton, A. J. 1998. Flow variability and the ecology of large rivers. Mar. Freshwater Res., 49, 55–72.

Rawlings, J. O., Pantula, S. G. and Dickey, D. A. 1998. Applied Regression Analysis. Springer-Verlag. ISBN: 978-0-387-98454-4.

Richter, B. D., Baumgartner, J. V. Wigington, R. and Braun, D. P. 1997. How much water does a river need? Freshwater Biol., 37, 231–249.

Richter, B. D., Baumgartner, J. V., Powell, J. and Braun, D. P. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology, 10, 1-12.

Richter, B. D., Baumgartner, J. V., Powell, J., and Braun, D. P. 1996. A method for assessing hydrologic alteration within ecosystems. Conserv. Biol., 10(4), 1163–1174.

Rogerson, P. A. 2001. Statistical Methods for Geography: A Student's Guide. SAGE Publications. London. pp. 136.

Ru, H. J., Wang, H. J., Zhao, W. H., Shen, Y. Q., Wang, Y. and Zhang, X. K. 2010. Fishes in the mainstream of the Yellow River: assemblage characteristics and historical changes. Biodiversity Science. 18(2), 169-174. (in Chinese with English abstract)

Sala, O. E., Chapin, F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L. F., Jackson, R. B., Kinzig, A., Leemans, R., Lodge, D., Mooney, H. A., Oesterheld, M., Poff, N. L., Sykes, M. Z., Walker, B. H., Walker, M. and Wall, D. H. 2000. Global biodiversity scenarios for the year 2100, Science, 287, 1770–1774.

Sheskin, D. J. 2007. Handbook of parametric and nonparametric statistical procedures. Chapman & Hall/CRC. ISBN: 1584888148 9781584888147.

Siebert, S., Döll, P., Hoogeveen, J., Faures, J.-M., Frenken, K., and Feick, S. 2005. Development and validation of the global map of irrigation areas. Hydrol. Earth Syst. Sci., 9, 535–547. http://www.hydrol-earth-syst-sci.net/9/535/2005/.

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A., Liermann, C. R. and Davies, P. M. 2010. Global threats to human water security and river biodiversity, Nature, 467, 555–561.

Wang, F.Y., Yang, J.X. and Chen, X.Y. 2011. Current Status and Conservation Strategies of Fish Germplasm Resources of Nanpanjiang River System in Yunnan. Journal of Hydroecology. 32(5): 19-29. (in Chinese with English abstract)

Wang, J. X., Rothausen, S. G. S. A., Conway, D., Zhang, L. J., Xiong, W., Holman, I. P. and Li, Y. M. 2011. China’s water–energy nexus: greenhouse-gas emissions from groundwater use for agriculture. Environmental Research Letters, 7, 14-35, doi:10.1088/1748-9326/7/1/014035. http://dx.doi.org/10.1088/1748-9326/7/1/014035

Wang, S. (ed.) 1998. China red data book of endangered animals: Pisces. National Environmental Protection Agency. Endangered Species Scientific Commission. Science Press, Beijing, China. 247p.

Wu, Q., Duan, X. B., Xu, S. Y., Xiong, C. X. and Chen, D. Q. 2007. Studies on fishery resources in the Three Gorges Reservoir of the Yangtze River. Freshwater Fisheries. 37(2): 70-75. (in Chinese with English abstract)

WWF 2004. Dam Right: Rivers at Risk. WWF Dams Initiative, WWF International: Gland, Switzerland. https://intranet.panda.org/data/downloads/10060/riversatriskfinalfullreport.pdf

Xenopoulos, M. A. and Lodge, D. M. 2006. Going with the flow: using species-discharge relationships to forecast losses in fish biodiversity. Ecology, 87, 1907-1914.

Xenopoulos, M. A., Lodge, D. M., Alcamo, J., Märker, M., Schulze, K., and van Vuuren, D. P. 2005. Scenarios of freshwater fish extinctions from climate change and water withdrawal. Glob. Change Biol., 11, 1557–1564.

Xia, J., Liu, M.Y., Jia, S.F., Song, X.F., Luo, Y. and Zhang, S.F. 2004. Water security problem and research perspective in North China. Journal of Natural Resources, 19, 550–560, (in Chinese).

Xie, P. and Chen, Y. 1999. Threats to biodiversity in Chinese inland waters. Ambio. 28, 674-681.

Xie, Y. H. 1986. The fish fauna of the Yalu River. Transactions of the Chinese Ichthyologic Society (No.5). Science Press, Beijing. 91-100. (in Chinese with English abstract)

Yi, Y. J. and Wang, Z. Y. 2009. Impact of dam construction on migratory fishes in the Yangtze River Basin, Water Resources and Hydropower Engineering, 40(1), 29-33. (in Chinese with English abstract).

Yu, C. R. and Zhang, Y. M. 1984. On the distribution of the fish fauna in the 2nd Songhua River. Transactions of Oceanology and Limnology. 1: 57-63. (in Chinese with English abstract)

Yue, P. and Chen, Y. 1998. Pisces. Wang S. (ed.). China Red Data Book of Endangered Animals. Science Press, Beijing, China.

Zeng X. (ed.) 1990. Fishery Resources of the Yangtze River Basin. Marine Press, Beijing, China. (in Chinese)

Zhang, J. J., Feng, H., Li, S. K., Yang, X. Z., Li, F. and Zhang, J. Y. 2009. Changes of fishery resources after the construction of cascade hydropower stations from Longyang gorge to Liujia gorge in upper stream of Yellow River. Freshwater Fisheries, 39(3), 40-45, (in Chinese with English abstract).

Zhang, X., Liu, R., Zhao, Q., Zhang, G., Wei, Z., Wang, X. And Yang, J. 1993. The population of finless porpoise in the middle and lower reaches of Yangtze River, J. Acta Theriol Sin, 13(4), 260-270, (in Chinese).

Zhang, Z. L., Xin, L. J. and Liang, C. L. 2007. The analysis of hydrological characteristics and processes of ecosystem in Lake Nansi during the past 50 years. Geographical Research, 26(5), 957-966, (in Chinese with English abstract).

Zhao, X., Barlow. J., Taylor, B. L., Pitman, R. L., Wang, K., Wei, Z., Stewart, B. S., Turvey, S. T., Akamatsu, T., Reeves, R. R. And Wang, D. 2008. Abundance and conservation status of the Yangtze finless porpoise in the Yangtze River, China. J. Biol Conserv, 141(2), 3006-3018.

Appendix A

Appendix A1. Summary of changes in different flow components due to anthropogenic impacts and hydrological data based on a literature review of 61 published papers and 15 additional studies in China.

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Haihe River	Baiyangdian Wetland	Magnitude	Riparian	Dam operation Water withdrawals	Decreased average annual discharge	1956-1969	19.20	$10^8m^3/yr$	Yang 2010
						1980s	2.80		
						1990s	5.80		
		Magnitude	Riparian	Dam operation, water diversion	Increased low flow/water level	1987	5.50	m	Xu et al., 2005
					Decreased water level	1991	8.50		
						2000	6.60		
						2002	6.00		
		Magnitude	Riparian	Dam operation Water withdrawals	Decreased average annual discharge	1956-1959	23.96	$10^8m^3/yr$	Li et al., 2004
						1970-1979	11.40		
						1980-1989	2.37		
		Magnitude	Riparian	Dam operation Water withdrawals	Decreased average annual discharge	1956-1959	26.60	$10^8m^3/yr$	Cui et al., 1999
						1970-1979	10.30		
						1980-1989	2.03		
		Magnitude	Riparian	Dam operation Water withdrawals	Decreased average annual discharge	1952-1959	18.27	$10^8m^3/yr$	Li et al., 2004
						1960-1969	17.31		
						1980-1989	1.47		
		Magnitude	Riparian	Dam operation Water withdrawals	Decreased average annual discharge	1952-1959	18.27	$10^8m^3/yr$	Li et al., 2004; Cui et al., 1999
						1970-1979	11.43		
						1982-1990	3.61		
						1990-1999	8.47		
		Magnitude	Riparian	Dam operation	Decreased average annual discharge	1949-1965	20.56	$10^8m^3/yr$	Gong and Xu, 1987
						1965-1978	7.21		

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Heilongjiang River	Honghe National Nature Reserve	Magnitude	Riparian	Water withdrawals	Reduced high flow Decreased groundwater table				
	Lower reaches of Nenjiang River	Magnitude	Aquatic	Dam operation	Reduced average annual discharge	1956-1960	312.90	$10^8 m^3/yr$	Xu et al., 2009
						1971-1980	132.20		
						1950s	24.00	$10^8 m^3/yr$	Tang et al., 2009
						1960s-1970s	11.00		
						1980s	12.50		
	Second Songhuajiang River	Magnitude	Aquatic	Dam operation	Reduced s average annual discharge (flow data for Jilin station)	1954-1960	16.08	km^3/yr	GRDC dataset
						1977-1983	8.90		
	Middle reaches of the main stream	Magnitude	Aquatic	Dam operation	Reduced seasonal high flow				
	Wetland of the Songnen Plain	Magnitude	Riparian	Dam operation	Decreased average annual discharge	1950s	398.44	$10^8 m^3/yr$	Luo et al., 2002
						1970s	297.61		
	Zhalong wetland	Magnitude	Riparian	Dam operation Water withdrawals	Reduced seasonal high flow (flow data were only available for annual discharge)	1956-1963	6.38	$10^8 m^3/yr$	Dong et al., 2008
						1980-1983	2.75		
						1960s-1970s	3.18		
						1990s	3.05		
						1956-1959	5.79		
						1980-1984	3.24		
						1985-1990	3.61		
						1991-1997	3.10		

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Heilongjiang River	Zhalong wetland	Magnitude	Riparian	Dam operation Water withdrawals	Reduced high flow	1963	6.82	$10^8 m^3/yr$	Dong et al., 2008
					(flow data were	1986	3.47		
					reported as annual discharge)	1996	2.06		
Heihe River	Whole river basin	Magnitude	Riparian	Dam operation Water withdrawals	Reduced average annual discharge				
		Duration	Riparian	Dam operation Water withdrawals	Increased no-flow period	1950s	37.50		
						1990s	89.90		
	Middle reaches	Magnitude	Riparian	Dam operation	Reduced average annual discharge	1987	16.85		
					Decreased high flow	2000	10.15		
	Lower reaches				Increased low flow	1987	7.19		
						2000	5.08		
	Middle reaches					1987	16.85		
						2000	10.15		
	Lower reaches					1987	7.19		
						2000	5.08		
	Lower reaches	Magnitude	Riparian	Water diversion	Increased annual discharge	1997-1999	7.30	$10^8 m^3/yr$	Jiang and Liu 2009
						2003	9.50		
	Erjina Oasis	Magnitude	Riparian	Water withdrawals Dam operation	Decreased low flow	1987	8.50	$10^8 m^3/yr$	Zhang et al., 2003
					Reduced average annual discharge	1996	5.03		
						1987	8.50		
						1996	5.03		
						1987	8.50		
						1996	5.03		

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Heihe River	Erjina Oasis	Magnitude	Riparian	Water withdrawals Dam operation	Reduced average annual discharge	1977	12.24	$10^8 m^3/yr$	Sun et al., 2009
						1993	7.98		
						2001	5.45		
						1977	12.24		
						1993	7.98		
						2001	5.45		
						1977	12.24		
						1993	7.98		
						2001	5.45		
	East and west Juyanhai wetland	Magnitude	Riparian	Water diversion	Increased annual discharge	1997-1998	7.92	$10^8 m^3/yr$	Sun et al., 2009
						2004	11.40		
						1998	7.92		
						2004	11.40		
Shule River	Whole river basin	Magnitude	Riparian	Water withdrawals	Reduced average annual discharge	1970s	2.44	$10^8 m^3/yr$	Wang et al., 2002
						1980s	2.14		
						1990s	2.09		
	Xihu wetland	Magnitude	Riparian	Dam operation	Reduced average annual discharge				
Shiyang River	Whole river basin	Magnitude	Riparian	Water withdrawals	Reduced average annual discharge	1980-1987	2.31	$10^8 m^3/yr$	Xu et al., 2007
						1990-1994	1.80		
						1980-1987	2.31		
						1990-1994	1.80		

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Shiyang River	Lower reaches	Magnitude	Riparian	Water withdrawals	Reduced average annual discharge	1950s	11.90	$10^8 m^3/yr$	Wang et al., 2002
						1980s	9.38		
						1990s	6.91		
Huaihe River	Whole river basin	Magnitude, duration, rate of change	Aquatic	Dam operation	Increased low flow Reduced seasonal variability	before 1959	316.50	m^3/s	Hu et al., 2008
						after 1959	754.20		
					Increased rise rate	before 1959	115.20	$m^3/s*d$	
						after 1959	134.50		
					Decreased peak flow and duration	before 1959	34.70	day	
						after 1959	28.30		
		Magnitude	Aquatic	Dam operation	Decreased April flow	1982	102.00	m^3/s	Xia et al., 2008
						2006	17.50		
						1982	102.00		
						2006	17.50		
Huaihe River	Whole river basin	Magnitude	Aquatic	Dam operation	Decreased April flow	1982	102.00	m^3/s	Xia et al., 2008
						2006	17.50		
						1982	102.00		
						2006	17.50		
	Nansi Lake	Magnitude	Riparian	Dam operation	Decreased average annual discharge and high flow	1983-1984	21.85	$10^8 m^3/yr$	Zhang et al., 2007
						1987	3.23		
						1988	1.03		
						1989	1.03		
						1990	11.96		
						1991	6.95		

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
	Nansi Lake	Magnitude	Riparian	Dam operation	Decreased average annual discharge and high flow	1996	4.41	$10^8 m^3/yr$	Zhang et al., 2007
						1915-1982	29.60		
						1990-1998	11.61		
						1950s	29.60		
						1960s	15.18		
						1988-1991	5.24		
						1996	4.41		
	Gaoyou Lake	Magnitude, rate of change	Riparian	Dam operation	Changes in rates of water level fluctuation and disturbance frequency				
	Hongze Lake	Magnitude	Riparian	Dam operation	Decreased seasonal low flow (average value between March and April at Bengbu station)	1956	2.07	km^3/month	GRDC dataset
						1970-1973	0.88		
	Hongze Lake	Rate of change	Riparian	Dam operation	Changes in water level fluctuation and disturbance frequency	1914-1951	10.60	m	Liu et al., 2009
						1970s	12.50		
						1991-2003	13.00		
Lancang River		Magnitude	Aquatic	Dam operation	Increased low flow and decreased high flow	1956-1985 vs. 1986-2008	change in low flow: 6.4 change in high flow: -14.3	%	Zhong and Wang 2010
Tarim River	Whole river basin	Magnitude	Aquatic	Dam operation Water withdrawals	Decreased annual discharge				

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Tarim River	Middle reaches	Magnitude	Riparian	Water withdrawals	Decreased average annual discharge	1950s	32.50	$10^8m^3/yr$	Feng et al., 2005
						1970s	24.74		
	Lower reaches					1950s	32.50		
						1970s	24.74		
						1950s	13.17		
						1970s	5.36		
						1950s	13.17		
						1970s	5.36		
						1950s	13.17		
						1980s	1.45		
	Middle reaches	Magnitude	Riparian	Water withdrawals	Decreased average annual discharge	1950s	36.50	$10^8m^3/yr$	Hamut et al, 2008
						1990s	26.12		
	Middle reaches	Magnitude	Riparian	Water withdrawals	Decreased average annual discharge	1950s	32.50	$10^8m^3/yr$	Feng et al., 2005
						1970s	24.74		
						1980s	26.55		
	Lower reaches					1950s	13.17		
						1970s	5.36		
						1980s	1.45		
	Yingsu river section, lower reaches	Magnitude	Riparian	Water diversion	Increased annual discharge	2000	2.31	$10^8m^3/yr$	Tao et al., 2008
						2001	3.82		
						2002	3.31		
						2003	6.26		
	Lower reaches	Magnitude	Riparian	Water diversion	Increased annual discharge	2002	3.56	$10^8m^3/yr$	Shi et al., 2008
						2004	4.50		

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Bosten Lake		Magnitude	Riparian	Water withdrawals	Decreased annual discharge	1958-1965	27.30	$10^8 m^3/yr$	Tan et al., 2004
						1968-1969	20.10		
						1971	29.60		
						1972	26.00		
						1973	20.00		
						1974	10.30		
						1975	13.00		
						1976	15.70		
						1977	10.60		
Ulungur Lake		Magnitude	Riparian	Dam operation Water withdrawals	Decreased annual discharge led to decreased storage	1952-1961	123.52	$10^8 m^3/yr$	Wang and Zhang 1991
						1962-1971	50.54		
						1972-1981	52.72		
Irtysh River	Whole river basin	Magnitude	Riparian	Dam operation Water withdrawals	Decreased seasonal variability				
	Altai Plain	Magnitude	Riparian	Water withdrawals	Decreased average annual discharge				An et al, 2002

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Yellow River	Upper reaches	Magnitude	Aquatic	Dam operation	Reduced high flow and increased low flow				
	Upper reaches	Magnitude	Aquatic	Dam operation	Reduced high flow and increased low flow				
	Maqu section, upper reaches	Magnitude	Aquatic	Dam operation Water withdrawals	Decreased annual discharge Decreased seasonal variability	1980s	168.70	$10^8 m^3/yr$	Dong et al., 2007
						1960-2000	148.40		
						1980s	168.70		
						1960-2000	148.40		
	Middle reaches					1980s	370.9		Zhang et al., 2009
						2000-2004	172.1		
						1980s	370.9	$10^8 m^3/yr$	
						2000-2004	172.1		
						1980s	370.9		
						2000-2004	172.1		
	Middle reaches	Magnitude	Aquatic	Dam operation	Reduced seasonal high flow (mean value from July to September)	1950s	7.20	$km^3/month$	GRDC dataset (at Samenxia station)
						1970s	4.88		
						1919-1959	2624.33	m^3/s	Guo and Yang 2005
						1960-2000	1881.67		
	Maqu section, upper reaches	Magnitude	Aquatic	Dam operation	Reduced high flow				
	Ningxia section, middle reaches								

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Yellow River	Shandong section, lower reaches	Magnitude	Aquatic	Dam operation	Reduced seasonal high flow (mean value from July to September)	1950s	7.90	km³/month	GRDC dataset
						1980s	5.37		(at Samenxia station)
	Wuliangsuhai wetland	Magnitude	Riparian	Water withdrawals	Reduced average annual discharge				
Yangtze River	Whole river basin	Magnitude	Aquatic	Dam operation	Decreased high flow (mean value from July to September)	1955-1959			GRDC dataset
						1961-1969	77.23	km³/month	(at Yichang station)
						1970-1975	69.84		
		Magnitude	Aquatic	Dam operation	Decreased high flow (mean value from July to September)	1961-1967	77.23	km³/month	GRDC dataset
						1968-1982	67.74		(at Yichang station)
	Middle Reaches	Magnitude	Aquatic	Dam operation	Reduced seasonal high flow (daily mean value from May to June)	1997	39857.25	m³/s	Duan et al, 2009
						1998	13505.00		
						1999	15633.33		
						2000	12000.00		
						2001	10680.00		
						2002	17125.00		
						2003	11050.00		
						2004	19100.00		
	Tianezhou oxbow	Magnitude	Aquatic	River course management	Decreased high flow				
	Laohedao oxbow								

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Yangtze River	Middle reaches and Dongting lake	Magnitude	Riparian	Dam operation	Decreased annual discharge	1964	167.78	km^3/yr	Yi and Wang 2009
						1965	148.15		
						1966	112.22		
						1967	123.70		
						1968	146.30		
						1969	87.41		
						1970	105.19		
						1971	79.63		
						1973	91.48		
						1974	112.22		
						1975	81.85		
						1976	68.89		
						1977	73.70		
						1978	61.10		
						1979	68.89		
						1980	90.37		
						1981	79.63		
						1982	85.56		
						1983	98.50		
	Downstream the Gezhouba Dam		Aquatic			1950s	75.76	% of total discharge	
						1960s	72.12		
						1970s	48.94		
						1980s	43.64		
						1996	36.06		

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Yangtze River	Downstream the Gezhouba Dam	Magnitude	Aquatic	Dam operation	Decreased annual discharge	1997	28.94	% of total discharge	Yi and Wang 2009
						1998	50.15		
						1999	42.58		
						2000	36.82		
						2001	30.91		
						2002	34.55		
						2003	34.85		
						2004	32.42		
						2005	35.30		
	Downstream the Gezhouba Dam	Magnitude	Aquatic		Decreased seasonal low flow (mean value from October to November)	2000	20950.00	m^3/s	Ban and Li 2007
						2001	15156.50		
						2002	8953.00		
						2003	8434.00		
						2004	11533.00		
						2005	10300.00		
						2006	8200.00		
		Magnitude	Aquatic		Decreased seasonal high flow (mean value from July to September)	2002	87.72	km^3/month	Changjiang sediment Bulletin, 2002-2004
						2003	60.17		
						2004	53.72		
	Hubei section of the Yangtze River	Magnitude	Aquatic	Dam operation	Reduced average annual discharge	1964	167.78	km^3/yr	Yi and Wang 2009
						1965	148.15		
						1966	112.22		
						1967	123.70		
						1968	146.30		

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
Yangtze River	Hubei section of the Yangtze River	Magnitude	Aquatic	Dam operation	Reduced average annual discharge	1969	87.41	km^3/yr	Yi and Wang 2009
						1970	105.19		
						1971	79.63		
						1973	91.48		
						1974	112.22		
						1975	81.85		
						1976	68.89		
						1977	73.70		
						1978	61.10		
						1979	68.89		
						1980	90.37		
						1981	79.63		
						1982	85.56		
						1983	98.50		
	Jialing River	Magnitude	Aquatic	Dam operation	Increased spring low flow	Mar. 2005	774.00	m^3/s	Long et al., 2008
						Mar. 2006	1050.00		
						Apr. 2005	1005.00		
						Apr. 2006	836.00		
						May. 2005	1187.00		
						May. 2006	1955.00		
	Donghu Lake	Magnitude	Riparian	Dam operation	Reduced high flow				
	Poyang Lake								
	Honghu Lake								
	Liangzi Lake								
	Futou lake								

Appendix A1 (Continued)

Geographic distribution of studies		Flow component	Organism	Main driver of flow alteration	Primary flow alteration	Year or time period of flow record	Flow data	Units	Source of data
	Datong Lake	Magnitude	Riparian	Dam operation	Reduced high flow				
	East Dongting Lake								
	South Dongting lake								
	Donghu Lake								
	Honghu Lake								
	Liangzi Lake								
	Dongting Lake								
	Honghu Lake								
	Chenhu Lake								
	Honghu Lake								
	Dongting Lake								
	Dongting Lake and Poyang Lake								
	Dongting Lake	Magnitude	Riparian	Dam operation	Reduced high flow				
Pearl River	Lianjiang River	Magnitude	Aquatic	Dam operation	Decreased peak flow				
	Beijiang River	Duration	Aquatic	Dam operation	Increased inundation duration Increased water level				
	Hongshui River	Duration	Aquatic	Dam operation	Increased inundation duration Increased water level				
	Yujiang River	Magnitude	Aquatic	Dam operation	Reduced peak flow				
	Xijiang River	Magnitude	Aquatic	Dam operation	Flow stabilization				

Appendix A2. Summary of ecological responses to anthropogenic flow alterations and ecological data based on a literature review of 61 published papers in China.

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Haihe River	Baiyang-dian Wetland	Reduced species richness: fish, bird	1965	54	sp.									192	sp.			Gao et al., 2009
			1992											52				
			2000	18														
		Reduced reed coverage	1987									1.32	10^4 hm^2					Xu et al., 2005
			1991									1.16						
		Increased reed coverage	2000									1.62						
			2002									1.52						
		Reduced diversity of plankton	1958							129	genus							Li et al., 2004
			1975							92								
			1993							73								
		Reduced diversity: plankton, macro-invertebrate macrophytes	1958			35	sp.			85	genus		sp.					Cui et al., 1999
			1975							71		34.00						
			1980			25												
			1992							70		32.00						
		Decreased fish catch	1950-1959	6915	ton													Li et al., 2004
			1960-1969	4340														
			1980-1989	1270														
		Reduced fish species richness	1958	54	sp.													Zhao et al., 2007
			1975	35														
			1991	24														
			1998	19														

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Haihe	Baiyang-dian wetland	Decreased fish catch	1949-1965	5323	ton													Gong and Xu, 1987
			1965-1978	1090														
Hei-long-jiang River	Honghe National Nature Reserve	Reduced population of macrophytes	1950s									653.00	g/m²					Zhou et al., 2009
			1998					48.00	%									
			2004					38.00	%			403.00	g/m²					
	Lower reaches of Nen-jiang River	Decreased weight of fish catch	1959-1962	3545	ton													Yang 1993
			1981	588.7														
			1950s	1000														
			1960s-1970s	450														
			1984-1990	575														
	Second Songhua-jiang River		1960s	1800														
			1977-1983	750														
	Middle reaches of the main stream	Decreased weight of fish catch	1950-1959	726.5	ton													Dong and Liu 2002
			1960-1969	614.2														
			1970-1979	500.8														
			1980-1989	1409														
			1990-1999	720.3														

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
	Wetland of the Songnen Plain	Reduced riparian meadow cover	1950s					2.13	10^6 hm^2									Luo et al., 2002
			1970s					1.16										
	Zhalong wetland	Reduced riparian meadow cover	1963					90.00	%									Liu et al., 2008
			1983					75.00										
		Reduced reed cover	1960s-1970s									16.00	10^4 hm^2					
			1990s									10.00						
		Decreased bird species richness	1950s											180	sp.			
			1984											61				
			1990											56				
			1999											41				
		Decreased weight of fish catch	1963	801	ton													
			1986	30														
			1996	10														
Heihe River	Whole river basin	Decreased reed cover	1950s-1960s					12.00	hm^2									Zhao and Bai 2008
			2000s					2.30										
		Reduced meadow cover	1960s					12.00	10^4 hm^2									
			1980s					9.00										
			1990s					5.00										
		Decreased riparian forest cover	1950s					5.00	10^4 hm^2									Ren 2005
			1990s					2.27										

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-invertebrate	Units	Riparian Vegetation	Units	Plankton	Units	Macrophyte	Units	Bird	Units	Mammal	Units	Source of data
Heihe River	Middle reaches	Reduced riparian forest cover	1987					3496.04	km²									Wang et al., 2002
			2000					3369.88										
	Lower reaches		1987					424.37										
			2000					390.28										
	Middle reaches	Decreased riparian meadow cover	1987					2418.09										
			2000					2144.39										
	Lower reaches		1987					1183.42										
			2000					906.23										
	Lower reaches	Increased Populus cover	1998					366.00	km²									Jiang and Liu 2009
			2004					375.00										
	Erjina Oasis	Reduced Populus cover	1987					84.93	km²			740.25	km²					Zhang et al., 2003
		Reduced macrophytes cover	1996					80.15				469.88						
		Increased silver berry cover	1987					225.72										
			1996					248.85										
		Reduced Chinese tamarisk cover	1987					1554.71										
			1996					1411.86										
		Decreased riparian forest cover	1977					1052.00	km²									Sun et al., 2009
			1993					399.00										
			2001					283.00										
		Decreased riparian shrub cover	1977					1534.00										
			1993					591.00										
			2001					580.00										

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Heihe River	Erjina Oasis	Decreased meadow cover	1977					6620.00										Sun et al., 2009
			1993					6114.00										
			2001					5433.00										
	East and west Juyanhai wetland	Increased shrub cover	1998					88.92	km²									Qiao et al., 2007
			2004					103.41										
		Increased meadow cover	1998					24.78										
			2004					40.13										
Shule River	Whole river basin	Decreased riparian vegetation cover	1970s					912.98	km²									Jin et al., 2007
			1980s					795.70										
			1990s					774.44										
	Xihu wetland	Reduced coverage of riparian shrubs, Populus, meadow	1950s					33000	hm²									Zhao and Bai 2008
			1980s					6700										
Shi-yang River	Whole river basin	Decreased riparian forest cover	1987					1602.84	km²									Zhu and Song 2010
			1994					1176.57										
		Decreased riparian meadow cover	1987					1274.56										
			1994					1191.42										
	Lower reaches	Decreased riparian vegetation cover	1950s					13.33	10^4 hm^2									Wang et al., 2002
			1981					7.24										
			1990s					3.64										

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Huaihe River	Whole river basin	Decreased fish population, species richness and diversity of macro-invertebrate																Hu et al., 2008
	Whole river basin	Increased diversity and population of zooplankton	1982							26.76	unit/L							Xia et al., 2008
			2006							261.55								
		Increased diversity and population of phytoplank-ton	1982							1.55	10^4unit/L							
			2006							38.04								
		Increased population of macro-invertebrate	1982							113.75	unit/m²							
			2006							328.24								
		Decreased population of sensitive macro-invertebrate species	1982							84.46								
			2006							17.15								
	Nansi Lake	Decreased diversity of sensitive phytoplank-ton species increased	1983-1984							116	genus							Zhang et al., 2007
			1987							105								
			1988							116								
			1989							46								

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Huaihe River	Nansi Lake	diversity of pollution tolerant sp.	1990							40								Zhang et al., 2007
			1991							40								
			1996							47								
		Decreased fish species richness	1960	74	sp.													
			1998	32														
		Decreased weight of fish catch	1950s	20345	ton													
			1960s	9025														
		Decreased diversity of bird	1988-1991											33	sp.			
			1996											27				
	Gaoyou Lake	Decreased weight of fish catch	1950s	14000	ton													Liu et al., 2003
			1960s	6240														
			1970s	5390														
		Reduced population of sensitive fish sp.																
	Hongze Lake	Reduced population of migrating fish	1956	21000														
			1973	9000														
			1950s	1750														
			1960s	500														
	Hongze Lake	Reduced macrophytes cover	before 1952									70.00	%					Liu et al., 2009
			1970s									34.44						
			2008									7.65						
Lancang River		Increased phytoplank-ton diversity	1983							88	sp.							Zhang 2001
			1994							280								

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Lancang River		Increased population of phytoplankton	1983							12.15	10^4unit/L							Zhang 2001
			1997-1998							30.64								
		Increased zooplankton diversity	1984							47	sp.							
			1997-1998							100								
		Increased bird species richness	1984											101	sp.			
			1997-1998											111				
Tarim River	Whole river basin	Loss of floodplain connectivity Loss of wetland habitat Reduced harvest of endemic fish	1965	180	ton													Ren et al., 1996
			1970-1974	30														
	Middle reaches	Decreased Populus cover	1958					1758.00	km²									Feng et al., 2005
			1978					1002.00										
		Decreased biomass of Populus	1958					3.29	10^6 kg									
			1978					1.46										
	Lower reaches	Decreased Populus cover	1958					540.00	km²									
			1978					164.00										
		Decreased biomass of Populus	1958					270.00	10^3 kg									
			1978					62.00										

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Tarim River	Lower reaches	Decreased total vegetation cover	1950s					65.00	%									Feng et al., 2005
			1980s					31.00										
	Middle reaches	Decreased Populus cover	1950s					17.60	10^4 hm^2									Hamut et al, 2008
			1990s					11.65										
	Middle reaches	Decreased Populus cover	1958					17.58	10^4 hm^2									Touheti 1999
			1978					10.02										
			1983					11.08										
	Lower reaches		1958					5.40										
			1978					1.64										
			1983					0.52										
	Yingsu river section, lower reaches	Increased growth increment of Populus	2000					0.18	cm									Tao et al., 2008
			2001					0.49										
			2002					0.39										
			2003					0.34										
	Lower reaches	Increased riparian total vegetation cover at Yingsu section	2002					27.00	% of total cover									Zhang et al, 2006
			2004					36.60										
		Increased riparian total vegetation cover at Abdal section	2002					36.88										
			2004					42.58										
		Increased Populus cover at Yingsu section	2002					7.93										
			2004					9.32										

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Tarim River	Lower reaches	Increased Populus cover at Abdal section	2002					12.05										Zhang et al, 2006
			2004					13.77										
		Increased Chinese tamarisk cover at Yingsu section	2002					13.46										
			2004					16.78										
		Increased Glycyrrhiza inflata (shrub) cover at Yingsu section	2002					0.06										
			2004					0.10										
Bosten Lake		Reduced catches of endemic fish	1958-1965	550	ton													Liu 1983
			1968-1969	400														
			1971	560														
			1972	455														
			1973	336														
			1974	320														
			1975	360														
			1976	469														
			1977	320														
Ulungur Lake		Reduced catches of endemic fish	1961	70	% of total fish catch													Huang et al., 1986
			1974	50														
			1984	30														

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Irtysh River	Whole river basin	Decreased weight of fish catch	1982	500	ton													Karjan 2005
			1989-1994	400														
			1999	138														
			2002	50														
	Altai Plain	Reduced coverage of hydrophytes Loss of wetland habitat	1950s									18.90	%					An et al, 2002
			1990s									12.60						
		Reduced coverage of hygrophytes	1950s									28.40						
			1990s									19.50						
		Reduced coverage of hydro-macrophytes	1950s									43.20						
			1990s									33.30						
Yellow River	Upper reaches	Reduced diversity of and biomass of phytoplank-ton	1981							69	sp.							Shen et al, 2007
			2007							43								
			1981							0.64	mg/L							
			2007							0.2698								
		Reduced diversity of and biomass of phytoplank-ton	1981							26	sp.							
			2007							20								
			1981							1.1	mg/L							
			2007							0.2225								
	Upper reaches	Reduced fish species richness	1951-1965	25														Zhang et al., 2009
			2005-2007	18														

Appendix A2 (Continued)

Geographic distribution of studies	Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Maqu section, upper reaches	Reduced biomass of phytoplank-ton	1982							0.459	mg/L							Yuan et al., 2009
		2002-2007							0.0807								
	Increased biomass of zooplankton	1982							0.118								
		2002-2007							0.3088								
Middle reaches	Reduced biomass of phytoplank-ton	1982							0.379	mg/L							Zhang et al., 2009
		2002-2007							0.0936								
	Increased biomass of zooplankton	1982							0.038								
		2002-2007							0.1452								
	Reduced species richness of fish	1982	66	sp.													
		2002-2007	27														
Middle reaches	Decreased weight of fish catch	1950s	35	ton													Guo and Yang 2005
		1970s	10														
	Increased bird sp. richness	1919-1959											9	sp.			
		1960-2000											12				
Maqu section, upper reaches	Decreased weight of fish catch	1950s	220	ton													Li et al., 2009
		1960s	177														
		1970s	100														
		1980s	50														

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
	Ningxia section, middle reaches		1950s-1960s	120														Li et al., 2009
			1980s	50														
	Shan-dong section, lower reaches		1950s	122														
			1980s	50														
	Wuliang-suhai wetland	Decreased weight of fish catch	1971-1982	750	ton													Li 2009
			1989	250														
		Increased reed cover	1987									94.90	km²					
			1996									113.00						
		Decreased weight of fish catch	1960s	5098	ton													He 1987
			1981-1982	1935														
Yangtze River	Whole river basin	Decreased weight of fish catch	1955-1959	37.39	10^4 ton													Fu et al., 2003
			1960-1969	25.30														
			1970-1975	20.15														
		Decreased weight of fish catch	1961-1967	0.26	10^6 ton													Chen et al., 2003
			1968-1982	0.21														

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-invertebrate	Units	Riparian Vegetation	Units	Plankton	Units	Macrophyte	Units	Bird	Units	Mammal	Units	Source of data
Yangtze River	Middle Reaches	Decreased fish larvae production	1997	78396	10^6 unit													Duan et al, 2009
			1998	1319														
			1999	643														
			2000	1189														
			2001	634														
			2002	614														
			2003	64														
			2004	286														
	Tianezhou oxbow	Reduced biomass of macro-invertebrate	1987-1988			39.3	g/m^2											Pan et al., 2008
			2003-2004			8.4												
		Reduced diversity of mollusc	1987-1988			13	sp.											
			2003-2004			7												
	Laohedao oxbow	Reduced biomass of macro-invertebrate	1991-1994			99.1	g/m^2											
			2003			51.2												
		Reduced diversity of mollusc	1991-1994			12	sp.											
			2003			4												
	Middle reaches and Dongting lake	Decreased fish larvae production	1964	82.86	10^8 unit													Yi and Wang 2009
			1965	71.07														
			1966	66.25														
			1967	59.29														
			1968	48.74														
			1969	56.43														

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Yangtze River	Middle reaches and Dong-ting lake	Decreased fish larvae production	1970	41.25														
			1971	48.75														
			1973	38.39														
			1974	38.04														
			1975	25.00														
			1976	21.43														
			1977	22.32														
			1978	19.46														
			1979	25.18														
			1980	32.68														
			1981	25.18														
			1982	28.57														
			1983	35.00														
	Down-stream the Gezhou-ba Dam		1950s	0.30	% of total fish stock													
			1960s	0.27														
			1970s	0.15														
			1980s	0.06														
			1996	0.14														
			1997	0.04														
			1998	0.24														
			1999	0.10														
			2000	0.06														
			2001	0.04														
			2002	0.05														
			2003	0.02														
			2004	0.02														
			2005	0.04														

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Yangtze River	Down-stream the Gezhou-ba Dam	Decreased fertility rate of eggs of specific migratory fish Disruption of life cycle	2000	93.40	%													Ban and Li 2007
			2001	87.00														
			2002	78.30														
			2003	23.80														
			2004	28.10														
			2005	32.30														
			2006	34.50														
		Decreased weight of fish catch	2002	3360	10^3 ton													Xie et al., 2007
			2003	1350														
			2004	1010														
		Decreased fish larvae and eggs production	1997	250	unit													
			2002	190														
			2003	40.6														
			2004	33.9														
			2005	10.5														
	Hubei section of the Yangtze River	Decreased fish larvae production	1964	69.11	10^8 unit													Liu and Wu 1992
			1965	57.32														
			1966	53.96														
			1967	47.27														
			1968	42.2														
			1969	30.87														
			1970	32.93														
			1971	44.58														
			1973	35.67														
			1974	34.81														
			1975	22.43														
			1976	19.56														

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-invertebrate	Units	Riparian Vegetation	Units	Plankton	Units	Macrophyte	Units	Bird	Units	Mammal	Units	Source of data
Yangtze River	Hubei section of the Yangtze River	Decreased fish larvae production	1977	19.88														Liu and Wu 1992
			1978	17														
			1979	20.9														
			1980	23.96														
			1981	14.15														
			1982	15.17														
			1983	11.06														
	Jialing River	Decreased biomass of algae								10.49	10^4 unit/ml							Long et al., 2008
										9.16								
										4.64								
										4.87								
										4.4								
										4.17								
	Donghu Lake	Decreased sp. richness of macrophytes	1960s									83	sp.					Fang et al., 2006
			1990s									58						
			2000s									33						
	Poyang Lake	Increased species richness of macrophytes	1970s									37						
			1980s									97						
			1990s									98						
	Honghu Lake	Decreased species richness of macrophytes	1990s									122						
			2000s									94						
	Liangzi Lake		1990s									91						
			2000s									87						
	Futou lake		1990s									90						
			2000s									60						
	Datong Lake		1970s									28						
			1990s									19						

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Yangtze River	Dong-ting Lake	Decreased sp. richness of macrophytes	1970s									26						Fang et al., 2006
			1990s									23						
	South Dong-ting lake		1970s									23						
			1990s									11						
	Donghu Lake	Decreased fish sp. richness Loss of endemic fish	1960s	67	sp.													
			1980s	54														
			1990s	38														
	Honghu Lake		1960s	74														
			1970s	65														
			1990s	57														
	Liangzi Lake		1970s	75														
			1980s	54														
	Dong-ting Lake	Decreased weight of fish catch	1963	21	% of total fish catch													
			1981	14.1														
			1999	9.3														
	Honghu Lake		1950s	50														
			1980s	0.5														
	Chenhu Lake	Decreased species richness of waterbird	1980s											45	sp.			
			1990s											36				
	Honghu Lake		1960s											61				
			1980s											58				
			1990s											48				
	Dong-ting Lake	Decreased species richness of waterbird	1960s											31				
			1980s											16				
			1990s											20				
			2000s											28				

Appendix A2 (Continued)

Geographic distribution of studies		Ecological response	Time period of ecological record	Fish	Units	Macro-inverte-brate	Units	Riparian Vege-tation	Units	Plank-ton	Units	Macro-phyte	Units	Bird	Units	Mam-mal	Units	Source of data
Yangtze River	Dong-ting & Poyang Lake	Deceased population of river dolphin	1950s													6000	unit	
			1984													400		
			1998													60		
	Dong-ting Lake	Decreased fish species richness	1974	104	sp.													Ru et al., 2008
			2004-2005	69														
Pearl River	Lian-jiang River	Reduced diversity of migratory and lotic fish	1960s	101	sp.													Li et al., 2007
			2005-2006	87														
	Beijiang River	Reduced fish diversity	1981-1983	23	sp.													Zeng et al., 2011
			2009-2010	19														
	Hongshui River	Reduced weight of fish catch	1965	160	ton													Huang et al., 2009
			1975	71														
			1983	5.35														
	Yujiang River	Reduced fish diversity	1986	74	sp.													Zhou et al., 2006
			2004	53														
		Reduced diversity of macro-invertebrate	1986			61	sp.											
			2004			37												
	Xijiang River	Reduced diversity of migratory fish spawning	1981-1985	136	sp.													Li et al., 2010
			2006-2008	96														

Note: sp. represents species richness.

Appendix B

Appendix B1. Summary of quantitative relationships between riparian vegetation cover and average river discharge in arid and semi-arid region in China.

Geographic distribution of studies		Year or time period of flow record		Mean river discharge (km^3/yr or %)		Year or time period of record of riparian vegetation cover		Riparian vegetation cover (km^2 or %)		Percentage change of mean discharge (%)	Percentage change of riparian vegetation cover (%)	Source
		Reference	Impact	Reference	Impact	Reference	Impact	Reference	Impact			
Heihe River	Middle reaches	1987	2000	1.685	1.015	1987	2000	3496.04	3369.88	-39.76	-3.61	Zhang et al., 2006; Wang et al., 2002
								2418.09	2144.39	-39.76	-11.32	
	Lower reaches			0.719	0.508			424.37	390.28	-29.35	-8.03	
								1183.42	906.23	-29.35	-23.42	
	Lower reaches	1997-1999	2003	0.730	0.950	1998	2004	366.00	375.00	30.14	2.46	Jiang and Liu 2009
	Erjina Oasis	1987	1996	8.500	5.030	1987	1996	84.93	80.15	-40.82	-9.19	Zhang et al., 2003
								225.72	248.85	-40.82	10.25	
								1554.71	1411.86	-40.82	-5.63	
		1977	1993	1.224	0.798	1977	1993	1052.00	399.00	-34.80	-62.07	Sun et al., 2009
			2001		0.545		2001		283.00	-55.47	-73.10	
			1993		0.798		1993	1534.00	591.00	-34.80	-61.47	
			2001		0.545		2001		580.00	-55.47	-62.19	
			1993		0.798		1993	6620.00	6114.00	-34.80	-7.64	
			2001		0.545		2001		5433.00	-55.47	-17.93	
	East & west Juyanhai wetland	1997-1998	2004	0.792	1.140	1998	2004	88.92	103.41	43.94	16.30	Sun et al., 2009; Qiao et al., 2007
								24.78	40.13	43.94	61.95	
Shule River		1970s	1980s	0.244	0.214	1970s	1980s	912.98	795.70	-12.30	-12.85	Wang et al., 2002; Jin et al., 2007
			1990s		0.209		1990s		774.44	-14.34	-15.17	

Appendix B1 (Continued)

Geographic distribution of studies		Year or time period of flow record		Mean river discharge (km^3/yr or %)		Year or time period of record of riparian vegetation cover		Riparian vegetation cover (km^2 or %)		Percentage change of mean discharge (%)	Percentage change of riparian vegetation cover (%)	Source
		Reference	Impact	Reference	Impact	Reference	Impact	Reference	Impact			
Shiyang River	Whole river basin	1980-1987	1990-1994	0.231	1.800	1987	1994	1602.84	1176.57	-22.08	-26.59	Xu et al., 2007; Zhu and Song 2010
								1274.56	1191.42	-22.08	-6.52	
	Lower reaches	1950s	1980s	1.190	0.938	1950s	1981	1333.00	724.20	-21.18	-45.67	Wang et al., 2002
			1990s		0.691		1990s		364.00	-41.93	-72.69	
Tarim River Basin	Middle reaches	1950s	1990s	3.650	2.612	1950s	1990s	1760.00	1165.00	-28.44	-33.81	Hamut et al, 2008
	Middle reaches	1950s	1970s	3.250	2.474	1958	1978	1758.00	1002.00	-23.88	-43.00	Feng et al., 2005; Touheti 1999
			1980s		2.655		1983		1108.00	-18.31	-36.97	
	Lower reaches		1970s	1.317	0.536		1978	540.00	164.00	-59.30	-69.63	
			1980s		0.145		1983		52.30	-88.99	-90.31	
	Lower reaches	1950s	1980s	1.317	0.145	1950s	1980s	65.00[1]	31.00[1]	-88.99	-52.31	Feng et al., 2005
	Lower reaches	2002	2004	0.356	0.450	2002	2004	27.00	36.60	26.40	35.78	Shi et al., 2008; Zhang et al, 2006
								36.88	42.58	26.40	15.46	
								7.93	9.32	26.40	17.53	
								12.05	13.77	26.40	14.27	
								13.46	16.78	26.40	24.67	
								0.06	0.10	26.40	66.67	

[1]Vegetation cover was presented as percentage of the total land area.

Appendix B2. Summary of quantitative relationships between fish catch and average river discharge in arid and semi-arid region in China.

Geographic distribution of studies		Year or time period of flow record		Mean river discharge (km^3/yr or %)		Year or time period of record of weight or No. of fish catch		Fish catch (ton, 10^8unit or %)		Percentage change of mean discharge (%)	Percentage change of fish catch (%)	Source
		Reference	Impact	Reference	Impact	Reference	Impact	Reference	Impact			
Haihe River basin	Wetland, lower reaches	1952-1959	1960-1969	1.83	1.73	1950-1959	1960-1969	6915.00	4340.00	-5.25	-37.24	Li et al., 2004
			1980-1989		0.15		1980-1989		1270.00	-91.95	-81.63	
		1949-1965	1965-1978	2.56	0.72	1949-1965	1965-1978	5325.00	1090.00	-71.88	-79.53	Gong and Xu, 1987
Bosten Lake		1958-1965	1968-1969	27.30	20.10	1958-1965	1968-1969	550.00	400.00	-26.37	-27.27	Tan et al., 2004; Liu 1983
			1971		29.60		1971		560.00	8.42	1.82	
			1972		26.00		1972		455.00	-4.76	-17.27	
			1973		20.00		1973		336.00	-26.74	-38.91	
			1974		10.30		1974		320.00	-62.27	-41.82	
			1975		13.00		1975		360.00	-52.38	-34.55	
			1976		15.70		1976		468.99	-42.49	-14.73	
			1977		10.60		1977		320.00	-61.17	-41.82	
Ulungur Lake		1952-1961	1962-1971	12.35	5.05	1961	1974	70.00[1]	50.00[1]	-59.08	-28.57	Wang and Zhang, 1991; Huang et al., 1986
			1972-1981		5.27		1984		30.00[1]	-57.32	-57.14	

[1]Native fish catch presented as percentage of the total weight of fish catch.

Appendix B3. Summary of quantitative relationships between fish catch and average river discharge in humid region in China.

Geographic distribution of studies		Year or time period of flow record		Mean river discharge (km^3/yr or %)		Year or time period of record of weight or No. of fish catch		Fish catch (ton, 10^8unit or %)		Percentage change of mean discharge (%)	Percentage change of fish catch (%)	Source
		Reference	Impact	Reference	Impact	Reference	Impact	Reference	Impact			
Huaihe River basin	Nansi Lake	1950s	1960s	2.96	1.52	1950s	1960s	20345.00	9025.00	-48.72	-55.64	Zhang et al., 2007
	Hongze Lake	1956	1970-1973	2.07	0.88	1956	1973	21000.00	9000.00	-57.49	-57.14	GRDC dataset; Liu et al., 2003
Yellow River	Middle reaches	1950s	1970s	7.20	4.88	1950s	1970s	35.00	10.00	-32.22	-66.67	GRDC dataset; Guo and Yang, 2005
	Lower reaches	1950s	1980s	7.90	5.37	1950s	1980s	121.95	50.00	-32.03	-59.00	GRDC dataset; Li et al., 2009
Heilongjiang River basin (Amur River)	Lower reaches of Nenjiang River	1956-1960	1971-1980	31.29	13.22	1959-1962	1981	3544.60	588.70	-57.75	-83.39	Xu et al., 2009; Tang et al., 2009; Yang 1993
		1950s	1960s-1970s	24.00	11.00	1950s	1960s-1970s	1000.00	450.00	-54.17	-55.00	
			1980s		12.50		1984-1990		575.00	-47.92	-42.50	
	Second Songhuajiang River	1954-1960	1977-1983	16.08	8.90	1960s	1977-1983	1800.00	750.00	-44.65	-58.33	GRDC dataset; Yang 1993
	Zhalong wetland	1963	1986	6.82	3.47	1963	1986	801.00	30.00	-49.12	-96.25	Dong et al., 2008; Liu et al., 2008
			1996		2.06		1996		10.00	-69.79	-98.75	
Yangtze River basin	Whole river basin	1960-1969	1970-1975	77.23	69.84	1960-1969	1970-1975	253000.00	201500.00	-9.58	-20.36	GRDC dataset; Fu et al., 2003
		1961-1967	1968-1982	77.23	69.84	1961-1967	1968-1982	260000.00	210000.00	-9.58	-19.23	GRDC dataset; Chen et al., 2003

Appendix B3 (Continued)

Geographic distribution of studies		Year or time period of flow record		Mean river discharge (km^3/yr or %)		Year or time period of record of weight or No. of fish catch		Fish catch (ton, 10^8unit or %)		Percentage change of mean discharge (%)	Percentage change of fish catch (%)	Source
		Reference	Impact	Reference	Impact	Reference	Impact	Reference	Impact			
Yangtze River basin	Middle reaches[1]	1997	1998	1256.94	425.89	1997	1998	783.96	13.19	-66.12	-98.32	Duan et al, 2009
			1999		493.01		1999		6.43	-60.78	-99.18	
			2000		378.43		2000		11.89	-69.89	-98.48	
			2001		336.80		2001		6.34	-73.20	-99.19	
			2002		540.05		2002		6.14	-57.03	-99.22	
			2003		348.47		2003		0.64	-72.28	-99.92	
			2004		602.34		2004		2.86	-52.08	-99.64	
	Middle reaches and Lake Dongting[1]	1964	1965	167.78	148.15	1964	1965	82.86	71.07	-11.70	-14.23	Yi and Wang 2009
			1966		112.22		1966		66.25	-33.11	-20.05	
			1967		123.70		1967		59.29	-26.27	-28.45	
			1968		146.30		1968		48.74	-12.80	-41.18	
			1969		87.41		1969		56.43	-47.90	-31.90	
			1970		105.19		1970		41.25	-37.30	-50.22	
			1971		79.63		1971		48.75	-52.54	-41.17	
			1973		91.48		1973		38.39	-45.48	-53.67	
			1974		112.22		1974		38.04	-33.11	-54.09	
			1975		81.85		1975		25.00	-51.22	-69.83	
			1976		68.89		1976		21.43	-58.94	-74.14	
			1977		73.70		1977		22.32	-56.07	-73.06	
			1978		61.10		1978		19.46	-63.58	-76.51	
			1979		68.89		1979		25.18	-58.94	-69.61	
			1980		90.37		1980		32.68	-46.14	-60.56	
			1981		79.63		1981		25.18	-52.54	-69.61	
			1982		85.56		1982		28.57	-49.00	-65.52	
			1983		98.50		1983		35.00	-41.29	-57.76	

[1] Fish catch in the middle reaches and the Lake Dongting of the Yangtze River were presented as 1×10^8 amount.

Appendix B3 (Continued)

Geographic distribution of studies		Year or time period of flow record		Mean river discharge (km^3/yr or %)		Year or time period of record of weight or No. of fish catch		Fish catch (ton, 10^8unit or %)		Percentage change of mean discharge (%)	Percentage change of fish catch (%)	Source
		Reference	Impact	Reference	Impact	Reference	Impact	Reference	Impact			
Yangtze River basin	Downstream the Gezhouba dam[2, 3]	1950s	1960s	75.76	72.12	1950s	1960s	0.30	0.27	-4.80	-11.66	Yi and Wang 2009
			1970s		48.94		1970s		0.15	-35.40	-49.59	
			1980s		43.64		1980s		0.06	-42.40	-81.61	
			1996		36.06		1996		0.14	-52.40	-54.68	
			1997		28.94		1997		0.04	-61.80	-86.21	
			1998		50.15		1998		0.24	-33.80	-19.64	
			1999		42.58		1999		0.10	-43.80	-67.82	
			2000		36.82		2000		0.06	-51.40	-78.98	
			2001		30.91		2001		0.04	-59.20	-85.62	
			2002		34.55		2002		0.05	-54.40	-84.63	
			2003		34.85		2003		0.02	-54.00	-92.45	
			2004		32.42		2004		0.02	-57.21	-93.40	
			2005		35.30		2005		0.04	-53.41	-88.18	
	Downstream the Gezhou-ba Dam	2002	2003	87.72	60.17	2002	2003	3360000	1350000	-31.41	-59.82	Xie et al., 2007
			2004		53.72		2004		1010000	-38.75	-69.94	
	Hubei section	1964	1965	167.78	148.15	1964	1965	69.11	57.32	-11.70	-17.06	Yi and Wang 2009; Liu and Wu 1992
			1966		112.22		1966		53.96	-33.11	-21.92	
			1967		123.70		1967		47.27	-26.27	-31.60	
			1968		146.30		1968		42.2	-12.80	-38.94	
			1969		87.41		1969		30.87	-47.90	-55.33	
			1970		105.19		1970		32.93	-37.30	-52.35	
			1971		79.63		1971		44.58	-52.54	-35.49	
			1973		91.48		1973		35.67	-45.48	-48.39	

Appendix B3 (Continued)

Geographic distribution of studies		Year or time period of flow record		Mean river discharge (km^3/yr or %)		Year or time period of record of weight or No. of fish catch		Fish catch (ton, 10^8unit or %)		Percentage change of mean discharge (%)	Percentage change of fish catch (%)	Source
		Reference	Impact	Reference	Impact	Reference	Impact	Reference	Impact			
Yangtze River basin	Hubei section[2]	1964	1974	167.78	112.22	1964	1974	69.11	34.81	-33.11	-49.63	Yi and Wang 2009; Liu and Wu 1992
			1975		81.85		1975		22.43	-51.22	-67.54	
			1976		68.89		1976		19.56	-58.94	-71.70	
			1977		73.70		1977		19.88	-56.07	-71.23	
			1978		61.10		1978		17	-63.58	-75.40	
			1979		68.89		1979		20.9	-58.94	-69.76	
			1980		90.37		1980		23.96	-46.14	-65.33	
			1981		79.63		1981		14.15	-52.54	-79.53	
			1982		85.56		1982		15.17	-49.00	-78.05	
			1983		98.50		1983		11.06	-41.29	-84.00	

[2]Fish catch was expressed as the percentage of the total fish stock; [3]Flow data was presented as diversion ratio of discharge.

Appendix C

Appendix C1. Reference list of 61 published studies included in this study.

Ban, X. and Li, D. M. 2007. Ecological hydrological influence of large water conservancy projects on Acipenser Sinensis in Yangtze River. Engineering Journal of Wuhan University, 40(3), 10-13, (in Chinese with English abstract).

Chen, D. Q., Duan, X. B., liu, S. P. And Shi, W. G. 2003. Status and Management of Fishery Resources of the Yangtze River. Proceedings of the Second International Symposium on the Management of Large Rivers for Fisheries, FAO Corporate Document Repository, www.ibcperu.org/doc/isis/5209.pdf

Cui, X. L., Hou, Y. Q. and Wang, J. 1999. The Opinion on the ecological environment protection of Baiyangdian Lake. Journal of Baoding Teachers College, 12(2), 86-89, (in Chinese with English abstract).

Dong, C. Z. and Liu, Y. 2002. Fishery resources survey in the middle reaches of Heilongjiang River. Chinese Journal of Fisheries, 15(1), 12-18, (in Chinese).

Duan, X. B. 2009. Changes in abundance of larvae of the four domestic Chinese carps in the middle reach of the Yangtze River, China, before and after closing of the Three Gorges Dam. Environ Biol Fish, 86, 13–22, doi: 10.1007/s10641-009-9498-z.

Fang, J. Y., Wang, Z. H., Zhao, S. Q., Li, Y. K., Tang, Z. Y., Yu, D., Ni, L. Y., Liu, H. Z., Xie, P., Da, L. J., Li, Z. Q. and Zheng, C. Y. 2006. Biodiversity changes in the lakes of the Central Yangtze. Front Ecol Environ, 4(7), 369–377.

Feng, Q., Liu, W., Si, J. H., Su, Y. H., Zhang, Y. W., Cang, Z. Q. and Xi, H. Y. 2005. Environmental effects of water resource development and use in the Tarim River basin of northwestern China. Environmental Geology, 48, 202–210, doi: 10.1007/s00254-005-1288-0.

Fu, C. Z., Wu, J. H., Chen, J. K., Wu, Q. H. and Lei, G. C. 2003. Freshwater fish biodiversity in the Yangtze River basin of China: patterns, threats and conservation. Biodiversity and Conservation, 12, 1649–1685.

Gao, X. C., Wang, H. And Long, Di. 2009. Changes in hydrological conditions and the eco-environmental problems in Baiyangdian Watershed. Resources Science, 31(9), 1506-1513, (in Chinese with English abstract).

Gong, G. and Xu, J. 1987. Environmental effects of human activities on rivers in the Huanghe-Huaihe-Haihe Plain. China, Geogr. Ann., 69A(1), 181-188.

Guo, Q. Y. and Yang, Z. F. 2005. Post-project ecological analysis for the Sanmenxia dam. Acta Scientiae Circumstantia, 25(5), 580-585, (in Chinese with English abstract).

Hamut, T., Mamut, A. and Ayup, M. 2008. Populus euphratica Ecosystem fragility of Tarim River Basin and protecting strategy: a case study on Tarim nature reserve. Journal of Arid Land Resources and Environment, 22(10), 96-101, (in Chinese with English abstract).

He, Z. H. 1987. Fishery resources research on the Yellow River system. Journal of Dalian Fisheries College, 1, 63-66, (in Chinese).

Hu, W. W., Wang, G. X., Deng, W. and Li, S. N. 2008. The influence of dams on eco-hydrological conditions in the Huaihe River basin. China, Ecological Engineering, 33, 233-241.

Huang, Y. L., Zhou, J., He, A. Y. and Lu, M. 2009. Ecological protection of fish resources in rivers in Guangxi. Fisheries Science & Technology of Guangxi, 3, 13-38, (in Chinese).

Huang, Z. G., Zhu, J. and Shi, P. X. 1986. Fishery and enhancement of fish resources in the Ulungur Lake. Freshwater Fisheries, 3, 31-34, (in Chinese).

Jiang, X. H. and Liu, C. M. 2009. The response of vegetation to water transport in the lower reaches of the Heihe River. Acta Geographica Sinica, 64(7) 791-797, (in Chinese with English abstract).

Jin, S. H., Yin, M. Y., Liu, S. Y., Chen, D. H., Wen, J. L. and Wang, Z. 2007. Major Ecological environment issues of Shulehe drainage area and countermeasures. South-to-North Water Transfers and Water Science & Technology, 5(5), 68-71, (in Chinese with English abstract).

Karjan, A. 2005. Fish resources and development strategy in the China's Section of the Irtysh River. 25(1), (in Chinese with English abstract).

Li, H, J., Yuan, Y. F., Li, Y. D. and Zhang, J. J. 2009. Research progress on aquatic living resources of Yellow River Watershed. Hebei Fisheries, 10, 1-3, doi：10.3969/j.issn.1004-6755.2009.10.001, (in Chinese).

Li, J. G., Li, G. B., Cui, H. M. and Wang, D. W. 2004. Degradation of reed wetland and protection in Baiyangdian. South-to-North Water Transfers and Water Science Technology, 2(3), 35-38, (in Chinese with English abstract).

Li, J., Li, X. H., Jia, X. P., He, Y. F., He, M. F., Tan, X. C., Wang, C. and Jiang, W. X. 2010. Evolvement and diversity of fish community in Xijiang River. Journal of Fishery Sciences of China, 17(2), (in Chinese with English abstract).

Li, J., Luo, J. R., Li, X. H., Tan, X. C., Wang, C. and Guo, S. C. 2007. Investigation of fish resources and analysis of resources decline along Lianjiang River. Freshwater Fisheries, 37(3), 49-53, (in Chinese with English abstract).

Li, J. W. 2009. Water environment and protection in the Wuliangsuhai wetland. Inner Mongolia Water Resources, 5, 79-80, (in Chinese).

Li, Y. H., Cui, B. S. and Y, Z. F. 2004. Influence of hydrological characteristic change of Baiyangdian on the ecological environment in wetland. Journal of Natural Resources, 19(1), 62-68, (in Chinese with English abstract).

Liu, L. C., Chen, Z. and Yang, X. K. 2008. A Study of environmental degeneration and protection countermeasures of Zhalong Wetlands. Scientific and Technological Management of Land and Resources, 25(6), 94-99, (in Chinese with English abstract).

Liu, L. H. and Wu, G. X. 1992. Fisheries resources and proliferation in the upper and middle reaches of the Yangtze River after the construction of Gezhouba Dam. Reservoir Fisheries, 1, 3-5, (in Chinese).

Liu, L. P. 1983. Fishery and enhancement of fish resources in the Bosten Lake. Freshwater Fisheries, 3, 12-14, (in Chinese).

Liu, W. L., Deng, w., Wang, G. X., Li, A. M. and Zhou, J. 2009. Aquatic macrophyte status and variation characteristics in the past 50 years in Hongzehu Lake. Journal of Hydroecology, 2(6), 1-8, (in Chinese with English abstract).

Liu, Z., Shen, W. S. and Wu, H. Z. 2003. The effects of water conservancy projects on eco - environment of Huaihe. Geography and Geo - Information Science, 19(2), 77-81, (in Chinese with English abstract).

Long, T. Y., Liu, L. M., Guo, W. H. and Li, C. M. 2008. Effect of Runoff on algal growth in reach of Jialing River in city zone of Chongqing. Research of Environmental Sciences, 21(4), 104-108, (in Chinese with English abstract).

Luo, X. Z., Zhu, T. and Zhu, G. Y. 2002. Studies on degradation mechanism of Wetland ecological environment in Songnen Plain. Journal of Arid Land Resources and Environment, 16(4), 39-43, (in Chinese with English abstract).

Pan, B. Z., Wang, H. J., Liang, X. M., Wang, Z. X., Shu, Y. F. and Wang, H. Z. 2008. Macroinvertebrate in Yangtze oxbows: community characteristics and causes of resources decline. Journal of Lake Sciences, 20(6), 806-813, (in Chinese with English abstract).

Qiao, X. X., Jiang, X. H., Chen, J. N., Yin, H. J. and Chen, L. 2007. Effect of transferring water on ecological environment in the east and the west Juyanhai Lake at the lower reaches of Heihe River. Journal of Northwest A&F University (Nat. Sci. Ed.), 35(6), 190-194, (in Chinese with English abstract).

Ren, J. H. 2005. Effects of water resources exploitation on eco-environment of Heihe River basin. Bulletin of Soil and Water Conservation, 25(4), 94-110, (in Chinese with English abstract).

Ren, M. L., Ren, B., Yang, Z. L., Xiao, L. M., Xie, Z. N., Zheng, X. M., Feng, G. R. and Wang, J. D. 1996. Investigation on the resources and ecology of Tarim big head mountain carp [Aspiorhynchus laticeps (day)], a nearly extinct species in Xinjiang, and its possible protective measure. Chinese Journal of Fisheries, 9(2), 1-19, (in Chinese with English abstract).

Ru, H. J., Liu, X. Q., Huang, X. R., Ning, Y. Z. and Wang, H. T. 2008. Diversity of fish species and its spatial-temporal variations in Lake Dongting, a large Yangtze-connected lake. Journal of Lake Science, 20(1), 93-99, (in Chinese with English abstract).

Shen, H. B., Li, K. S. and Zhang, M. 2007. Survey and analysis on fish resources status on the upper reaches of the Yellow River. Hebei Fisheries, 6, 37-41, doi: 10.3969/j.issn.1004-6755.2007.06.019, (in Chinese with English abstract).

Sun, Z. Y., Xu, H. L. and Ma, R. 2009. Dynamics of landscape pattern in Ejina basin, the lower reaches of the Heihe River. Geological Science and Technology Information, 28(6), 94-100, (in Chinese with English abstract).

Tao, H., Gemmer, M., Song, Y. D. and Jiang, T. 2008. Eco-hydrological responses on water diversion in the lower reaches of the Tarim River, China. Water Resources Research, 44, w08422, 1-10, doi:10.1029/2007WR006186, Touheti, W. 1999. Preliminary approaching to ecological environment problem of middle and lower reaches of Tarim River. Environmental Protection of Xinjiang, 21(1), 34-39, (in Chinese with English abstract).

Wang, G. X., Wang, J. and Wu, Y. Q. 2002. Features of eco-environmental changes in Heihe River basin over recent 10 years. Scientia Geographica Sinica, 22(5), 527-534, (in Chinese with English abstract).

Xia, J., Zhao, C. S., Liu, M., Wang, G. S., Zhang, Y. Y. and Liu, Y. 2008. Impact assessment of dams & flood gates projects of Huaihe River on river ecosystem: a case study of the Bengbu Site. Journal of Natural Resources, 23(1), 48-60, (in Chinese with English abstract).

Xie, S. G., Li, Z. J., Liu, J. S., Xie, S. Q., Wang, H. Z. and Murphy, B. R. 2007. Fisheries of the Yangtze River show immediate impacts of the Three Gorges Dam. Fisheries, 32(7), 343-344,
http://fishwild.vt.edu/faculty/Three_%20Gorges_Dam.pdf

Xu, W. H., Ouyang, Z. Y., Van Duren, I., Wang, X. K., Miao, H. and Cao, Q. H. 2005. Reed land change and its relationship to water level in Baiyang Lake Since 1987. Journal of Soil and Water Conservation, 19(4), 181-189, (in Chinese with English abstract).

Yang, Y. F. 1993. Fish resource and its enhancement in the lower reaches of the Second Songhuajiang River and the Nenjiang River. Resources Science, 1, 64-70, (in Chinese).

Liu, L. P. 1983. Fishery and enhancement of fish resources in the Bosten Lake. Freshwater Fisheries, 3, 12-14, (in Chinese).

Yi, Y. J. and Wang, Z. Y. 2009. Impact of dam construction on migratory fishes in the Yangtze River Basin, Water Resources and Hydropower Engineering, 40(1), 29-33, x (in Chinese with English abstract).

Yuan, Y. F., Li, Y. D., Zhang, L. L., Yu, B., Qi, X. R., Han, M. X. and Zhang, J. J. 2009. Investigation & research on hydro-bios resources in the middle and upper reaches of main Yellow River. Journal of Hydroecology, 2(6), 15-19, (in Chinese with English abstract)

Zeng, G. X., He, J. K., Luo, X. Y., Mai, Y. W., Cai, Y. J., Lu, J. F. And Jiang, H. S. 2011. A study on fish resources and protection at Qingyuan section of the Beijiang River. Journal of South China Normal University (Natural Science Edition), 49-52, (in Chinese).

Zhang, J. J., Feng, H., Li, S. K., Yang, X. Z., Li, F. and Zhang, J. Y. 2009. Changes of fishery resources after the construction of cascade hydropower stations from Longyang gorge to Liujia gorge in upper stream of Yellow River. Freshwater Fisheries, 39(3), 40-45, (in Chinese with English abstract).

Zhang, L. H., Chen, Y. N. and Li, W. H. 2006. Analysis on the effect of implementing the project of transfusing stream water for regenerating the ecology in the Lower reaches of the Tarim River. Arid Zone Research, 23(1), 32-38, (in Chinese with English abstract).

Zhang, M. T., Shi, S. S., Zhang, W. and Na, Y. F. 2003. Ecological environment change and causes analysis in Erjina Oasis. Science of Soil and Water Conservation,1(6), 56-60, (in Chinese with English abstract).

Zhang, R. 2001. Impact assessment of Manwan Hydropower Station on ecological environment of the Lancang River. Design of Hydroelectric Power Station, 17(4), 27-32, (in Chinese with English abstract).

Zhang, Z. L., Xin, L. J. and Liang, C. L. 2007. The analysis of hydrological characteristics and processes of ecosystem in Lake Nansi during the past 50 years. Geographical Research, 26(5), 957-966, (in Chinese with English abstract).

Zhao, C. L., Xiao, G. H., Luo, N. T., Yuan, Q. Y. and Deng, P. C. 2007. Analysis on current fish composition in Baiyangdian Lake. Hebei Fisheries, 11, 49-50, doi: 10.3969/j.issn.1004-6755.2007.11.017, (in Chinese).

Zhao, J. Z. and Bai, F. 2008. Ecological environment degradation characteristics in the last 50 years in the Hexi Corridor. Gansu Science and Technology, 24(23), 59-62, (in Chinese).

Zhou, D. M., Gong, H. L., Wang, Y. H., Khan, S. and Zhao, K. Y. 2009. Driving forces for the marsh wetland degradation in the Honghe National Nature Reserve in Sanjiang Plain, northeast China. Environ Model Assess, 14, 101–111, doi: 10.1007/s10666-007-9135-1.

Zhou, J., Han, Y. Q. and He, A. Y. 2006. Research on the impact of Guiping Navigation Junction Project on aquatic organisms. Fisheries Science & Technology of Guangxi, 2, 14-27, (in Chinese).

Zhu, X. H. and Song, X. N. 2010. Landscape pattern changes and transition tendency factors in Shiyang River basin. Journal of Lanzhou University (Natural Sciences), 46(1), 65-71, (in Chinese with English abstract).

Appendix C2. Reference list of 15 studies that reported additional flow data.

Changjiang Water Resources Commission, Ministry of Water Resources. 2002-2004. Changjiang sediment Bulletin, http://www.cjw.gov.cn/zwzc/bmgb/nsgb/

Dong, X. H., Yao, Z. J. and Chen, C. Y. 2007. Runoff variation and responses to precipitation in the source regions of the Yellow River. Resources Science, 29(3), 67-73, (in Chinese with English abstract).

Dong, Y. X., Qin, Z. D., Xu, G. J. and Deng, G. L. 2008. Calculation of ecological environment water requirement and analysis of moisturizing in Zhalong Wetland. Journal of Heilongjiang Hydraulic Engineering, 35(4), 110-113, (in Chinese with English abstract)

Li, Y. H., Cui, B. S. and Y, Z. F. 2004. Influence of hydrological characteristic change of Baiyangdian on the ecological environment in wetland. Journal of Natural Resources, 19(1), 62-68, (in Chinese with English abstract)

Shi, L., Hasim, T. and Han, G. H. 2008. Effect of ecological water diversion in lower reaches of Tarim River on environmental dynamic state and its control measures. Xinjiang Agricultural Sciences, 45(5), 926-933, (in Chinese with English abstract).

Tan, W., Wang, Y. J. and Ning, J. Z. 2004. Analysis on the change of the ecological environment in the Bosten Lake area. Xinjiang Arid Zone. 21(1), 7-12, (in Chinese with English abstract).

Tang, Y., Wang, H., Yan, D. H. and Tang, K. W. 2009. Evolutionary regularity of runoff of the Nenjiang River Basin in period 1956-2000. Scientia Geographica Sinica, 29(6), (in Chinese with English abstract).

Wang, G. X., Cheng, G. D. and Shen, Y. P. 2002. Features of eco-environmental changes in Hexi Corridor region in the last 50 years and comprehensive control strategies. Journal of Natural Resources, 17(1), 78-86, (in Chinese with English abstract).

Xie, L. X. 2009. Analysis on the change of water level and quality in the Ulungur Lake. Journal of Water Resources & Water Engineering, 20(2), (in Chinese with English abstract).

Xu, D. X., Zhang, G. X. and Yin, X. R. 2009. Runoff variation and its impacting factor in Nenjiang River during 1956—2006. Advances in Water Science, 20(3), 416-421, (in Chinese with English abstract).

Xu, Z. X., Li, Z. L. and Shi, X. K. 2007. Long-term trends of major climatic variables and runoff in the Shiyang River basin. Resources Science, 29(5), 121-128, (in Chinese with English abstract).

Yang, C. X. 2010. Analysis on the deposited quantity variation and its influenced factors in Baiyang Dian. Ground Water, 32(2), 110-112, (in Chinese with English abstract).

Zhang, G. H., Nie, Z. L., Liu, S. Y. and Wang, J. Z. 2006. Threshold of influence of water resources in the Heihe River valley, northwestern Gansu, China on the ecological environment variations of the lower reaches. Geological Bulletin of China, 25(1-2), 244-250, (in Chinese with English abstract).

Zhang, J. Y., Wang, G. Q., He, R. M. and Liu, C. S. 2009. Variation trends of runoffs in the middle reaches of the Yellow River Basin and its response to climate change, 20(2), 153-158, (in Chinese with English abstract).

Zhong, H. P. and Wang, J. S. 2010. Impacts of hydropower development on runoff of main stream of the Lancang River. Water Resources and Hydropower Engineering, 41(12), 72-74, (in Chinese with English abstract).

Appendix D

Appendix D1. Summary of the model performance for linear models without intercepts based on original values of flow indicators

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
a1	I_{LTD}	0.114	< 0.0001	260.404	38.558	25.570	√	1.076	< 0.0001
a2	I_{Q90}	0.165	0.001	280.893	59.047	33.034	√	0.476	0.001
a3	I_{Q10}	0.001	< 0.0001	269.408	47.562	28.616	√	0.874	< 0.0001
a4	I_{SA}	0.000	0.001	280.091	58.245	32.704	√	0.540	0.001
a5	I_{SR}	0.017	< 0.0001	260.472	38.626	25.591	√	-1.002	< 0.0001
a6	I_{LTD}	0.097	< 0.0001	262.365	40.519	25.821		1.158	< 0.0001
	I_{Q90}							-0.074	0.623
a7	I_{LTD}	0.113	< 0.0001	262.623	40.777	25.904	√	1.069	0.004
	I_{Q10}							0.007	0.982
a8	I_{LTD}	0.166	< 0.0001	260.944	39.098	25.366		1.342	< 0.0001
	I_{SA}							-0.238	0.210
a9	I_{LTD}	0.066	< 0.0001	259.836	37.990	25.017	√	0.571	0.102
	I_{SR}							-0.523	0.106
a10	I_{Q90}	0.038	< 0.0001	268.856	47.010	28.003	√	0.213	< 0.0001
	I_{Q10}							0.724	0.107
a11	I_{Q90}	0.065	< 0.0001	277.035	55.189	31.017	√	0.320	0.026
	I_{SA}							0.380	0.017
a12	I_{Q90}	0.022	< 0.0001	262.659	40.813	25.915	√	0.024	0.863
	I_{SR}							-0.975	< 0.0001
a13	I_{Q10}	0.002	< 0.0001	268.742	46.896	27.963		1.433	< 0.0001
	I_{SA}							-0.488	0.100
a14	I_{Q10}	0.015	< 0.0001	262.580	40.734	25.890	√	0.094	0.747
	I_{SR}							-0.914	0.004
a15	I_{SA}	0.024	< 0.0001	261.610	39.764	25.578		-0.185	0.314
	I_{SR}							-1.185	< 0.0001
a16	I_{LTD}	0.106	< 0.0001	264.656	42.810	26.150		1.252	0.015
	I_{Q90}							-0.091	0.595
	I_{Q10}							-0.078	0.827
a17	I_{LTD}	0.142	< 0.0001	262.297	40.451	25.391		1.575	< 0.0001
	I_{Q90}							-0.149	0.342
	I_{SA}							-0.299	0.138
a18	I_{LTD}	0.051	< 0.0001	261.855	40.009	25.250		0.655	0.090
	I_{Q90}							-0.080	0.586
	I_{SR}							-0.528	0.107

Appendix D1 (continued)

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
a19	I_{LTD}	0.125	< 0.0001	261.486	39.640	25.134		1.061	0.003
	I_{Q10}							0.560	0.200
	I_{SA}							-0.477	0.075
a20	I_{LTD}	0.086	< 0.0001	261.795	39.949	25.231		0.692	0.091
	I_{Q10}							-0.196	0.554
	I_{SR}							-0.595	0.089
a21	I_{LTD}	0.118	< 0.0001	258.730	36.884	24.283		0.806	0.029
	I_{SA}							-0.337	0.041
	I_{SR}							-0.670	0.076
a22	I_{Q90}	0.045	< 0.0001	268.194	46.348	27.332		0.209	0.104
	I_{Q10}							1.278	0.001
	I_{SA}							-0.481	0.098
a23	I_{Q90}	0.021	< 0.0001	264.865	43.019	26.218	√	0.032	0.819
	I_{Q10}							0.106	0.724
	I_{SR}							-0.873	0.016
a24	I_{Q90}	0.025	< 0.0001	263.951	42.105	25.920		0.005	0.973
	I_{SA}							-0.184	< 0.0001
	I_{SR}							-1.180	0.327
a25	I_{Q10}	0.022	< 0.0001	261.289	39.443	25.072		0.653	0.069
	I_{SA}							-0.487	0.119
	I_{SR}							-0.913	0.003
a26	I_{LTD}	0.118	< 0.0001	263.604	41.758	25.367		1.250	0.012
	I_{Q90}							-0.093	0.573
	I_{Q10}							0.474	0.308
	I_{SA}							-0.478	0.077
a27	I_{LTD}	0.072	< 0.0001	263.253	41.407	25.256		0.972	0.057
	I_{Q90}							-0.161	0.342
	I_{Q10}							-0.372	0.328
	I_{SR}							-0.669	0.064
a28	I_{LTD}	0.090	< 0.0001	259.448	37.602	24.083		1.062	0.013
	I_{Q90}							-0.188	0.212
	I_{SA}							-0.421	0.037
	I_{SR}							-0.717	0.030
a29	I_{LTD}	0.099	< 0.0001	260.439	38.593	24.383		0.682	0.086
	I_{Q10}							0.360	0.409
	I_{SA}							-0.480	0.065
	I_{SR}							-0.600	0.077

Appendix D1 (continued)

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
a30	I_{Q90}	0.028	< 0.0001	262.575	40.729	25.403		0.028	0.837
	I_{Q10}							0.662	0.121
	I_{SA}							-0.486	0.073
	I_{SR}							-0.878	0.013
a31	I_{LTD}	0.084	< 0.0001	260.129	38.283	24.370		0.967	0.050
	I_{Q90}							-0.164	0.316
	I_{Q10}							0.183	0.696
	I_{SA}							-0.483	0.064
	I_{SR}							-0.675	0.053

√ denotes that all variables have plausible signs of regression coefficient in the model.

[1]ΔAIC_c denotes the difference between AIC_c value for a given model and that for the best model (the model with the lowest AIC_c), where AIC_c is the second-order Akaike Information Criterion. In this study, the best model is model d6 in the Appendix D4.

Appendix D2. Summary of the model performance for linear models without intercepts based on absolute values of flow indicators

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
b1	$\|I_{LTD}\|$	0.114	< 0.0001	260.404	38.558	25.570	√	-1.076	<0.0001
b2	$\|I_{Q90}\|$	0.037	< 0.0001	272.657	50.811	29.802	√	-0.593	<0.0001
b3	$\|I_{Q10}\|$	0.001	< 0.0001	267.246	45.400	27.852	√	-0.903	<0.0001
b4	$\|I_{SA}\|$	0.001	< 0.0001	269.798	47.952	28.756	√	-0.687	<0.0001
a5	I_{SR}	0.017	< 0.0001	260.472	38.626	25.591	√	-1.002	< 0.0001
b5	$\|I_{LTD}\|$	0.115	< 0.0001	262.619	40.773	25.903		-1.093	0.001
	$\|I_{Q90}\|$							0.013	0.946
b6	$\|I_{LTD}\|$	0.095	< 0.0001	262.456	40.610	25.850	√	-0.950	0.010
	$\|I_{Q10}\|$							-0.128	0.691
b7	$\|I_{LTD}\|$	0.093	< 0.0001	262.409	40.563	25.835	√	-0.961	0.003
	$\|I_{SA}\|$							-0.097	0.654
b8	$\|I_{LTD}\|$	0.066	< 0.0001	259.836	37.990	25.017	√	-0.571	0.102
	I_{SR}							-0.523	0.106
b9	$\|I_{Q90}\|$	0.008	< 0.0001	268.259	46.413	27.794	√	-0.201	0.287
	$\|I_{Q10}\|$							-0.678	0.013
b10	$\|I_{Q90}\|$	0.004	< 0.0001	270.777	48.931	28.683	√	-0.231	0.281
	$\|I_{SA}\|$							-0.472	0.050
b11	$\|I_{Q90}\|$	0.001	< 0.0001	258.789	36.943	24.691		0.558	0.056
	I_{SR}							-1.740	<0.0001
b12	$\|I_{Q10}\|$	0.000	< 0.0001	269.256	47.410	28.143	√	-0.721	0.108
	$\|I_{SA}\|$							-0.155	0.658
b13	$\|I_{Q10}\|$	0.014	< 0.0001	262.295	40.449	25.798	√	-0.184	0.542
	I_{SR}							-0.833	0.010
b14	$\|I_{SA}\|$	0.017	< 0.0001	262.688	40.842	25.925	√	-0.012	0.960
	I_{SR}							-0.984	0.003
b15	$\|I_{LTD}\|$	0.095	< 0.0001	264.432	42.586	26.189		-0.979	0.021
	$\|I_{Q90}\|$							0.030	0.883
	$\|I_{Q10}\|$							-0.138	0.680
b16	$\|I_{LTD}\|$	0.090	< 0.0001	264.660	42.814	26.152		-1.006	0.005
	$\|I_{Q90}\|$							0.063	0.773
	$\|I_{SA}\|$							-0.128	0.600
b17	$\|I_{LTD}\|$	0.036	< 0.0001	257.564	35.718	23.932		-0.606	0.071
	$\|I_{Q90}\|$							0.584	0.040
	I_{SR}							-1.271	0.009

Appendix D2 (continued)

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
b18	$\|I_{LTD}\|$	0.091	< 0.0001	264.742	42.896	26.178	√	-0.942	0.012
	$\|I_{Q10}\|$							-0.047	0.922
	$\|I_{SA}\|$							-0.074	0.820
b19	$\|I_{LTD}\|$	0.074	< 0.0001	262.110	40.264	25.331		-0.623	0.129
	$\|I_{Q10}\|$							0.086	0.802
	I_{SR}							-0.558	0.117
b20	$\|I_{LTD}\|$	0.080	< 0.0001	261.959	40.113	25.283		-0.618	0.094
	$\|I_{SA}\|$							0.112	0.654
	I_{SR}							-0.611	0.110
b21	$\|I_{Q90}\|$	0.008	< 0.0001	270.601	48.755	28.167		-0.202	0.340
	$\|I_{Q10}\|$							-0.681	0.129
	$\|I_{SA}\|$							0.004	0.992
b22	$\|I_{Q90}\|$	0.001	< 0.0001	260.951	39.105	24.966		0.544	0.067
	$\|I_{Q10}\|$							-0.119	0.685
	I_{SR}							-0.119	0.003
b23	$\|I_{Q90}\|$	0.001	< 0.0001	261.077	39.231	25.006		0.564	0.057
	$\|I_{SA}\|$							-0.052	0.824
	I_{SR}							-1.688	0.001
b24	$\|I_{Q10}\|$	0.019	< 0.0001	264.326	42.480	26.042		0.743	0.442
	$\|I_{SA}\|$							-0.839	0.593
	I_{SR}							-0.918	0.010
b25	$\|I_{LTD}\|$	0.089	< 0.0001	266.659	44.813	26.512		-0.997	0.022
	$\|I_{Q90}\|$							0.061	0.786
	$\|I_{Q10}\|$							-0.019	0.969
	$\|I_{SA}\|$							-0.118	0.749
b26	$\|I_{LTD}\|$	0.052	< 0.0001	259.606	37.760	24.131		-0.731	0.066
	$\|I_{Q90}\|$							0.613	0.036
	$\|I_{Q10}\|$							0.207	0.535
	I_{SR}							-1.391	0.009
b27	$\|I_{LTD}\|$	0.044	< 0.0001	259.932	38.086	24.229		-0.637	0.073
	$\|I_{Q90}\|$							0.577	0.046
	$\|I_{SA}\|$							0.074	0.757
	I_{SR}							-1.320	0.011
b28	$\|I_{LTD}\|$	0.079	< 0.0001	264.429	42.583	25.630		-0.605	0.147
	$\|I_{Q10}\|$							-0.033	0.943
	$\|I_{SA}\|$							0.128	0.709
	I_{SR}							-0.610	0.116

Appendix D2 (continued)

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
b29	$\|I_{Q90}\|$	0.001	< 0.0001	263.415	41.569	25.308		0.537	0.083
	$\|I_{Q10}\|$							-0.151	0.727
	$\|I_{SA}\|$							0.035	0.918
	I_{SR}							-1.616	0.003
b30	$\|I_{LTD}\|$	0.050	< 0.0001	261.570	39.724	24.462		-0.742	0.069
	$\|I_{Q90}\|$							0.627	0.041
	$\|I_{Q10}\|$							0.265	0.577
	$\|I_{SA}\|$							-0.059	0.861
	I_{SR}							-1.385	0.011

√ denotes that all variables have plausible signs of regression coefficient in the model.

[1]ΔAIC_c denotes the difference between AIC_c value for a given model and that for the best model (the model with the lowest AIC_c), where AIC_c is the second-order Akaike Information Criterion. In this study, the best model is model d6 in the Appendix D4.

Appendix D3. Summary of the model performance for linear models with intercepts based on original values of flow indicators

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
c1	I_{LTD}	0.114	0.033	226.133	4.284	16.483	√	0.315	0.033
c2	I_{Q90}	0.165	0.009	223.875	2.029	16.003	√	0.187	0.009
c3	I_{Q10}	0.001	0.862	231.044	9.198	17.503	√	0.025	0.862
c4	I_{SA}	0.000	0.984	231.076	9.230	17.510		-0.002	0.984
c5	I_{SR}	0.017	0.422	230.388	8.542	17.360	√	-0.130	0.422
c6	I_{LTD}	0.170	0.032	225.850	4.004	16.168	√	0.094	0.638
	I_{Q90}							0.154	0.123
c7	I_{LTD}	0.246	0.005	221.985	0.139**	15.406		0.726	0.001
	I_{Q10}							-0.501	0.015
c8	I_{LTD}	0.217	0.011	223.527	1.681**	15.705		0.596	0.003
	I_{SA}							-0.255	0.034
c9	I_{LTD}	0.156	0.044	226.526	4.680	16.306		0.547	0.018
	I_{SR}							0.320	0.184
c10	I_{Q90}	0.182	0.024	225.264	3.418	16.050		0.211	0.007
	I_{Q10}							-0.124	0.384
c11	I_{Q90}	0.184	0.023	225.178	3.332	16.033		0.208	0.006
	I_{SA}							-0.084	0.361
c12	I_{Q90}	0.193	0.019	224.713	2.867	15.940		0.252	0.007
	I_{SR}							0.218	0.262
c13	I_{Q10}	0.004	0.929	233.137	11.291	17.710		0.111	0.704
	I_{SA}							-0.065	0.734
c14	I_{Q10}	0.025	< 0.0001	232.301	10.455	17.526		-0.106	0.597
	I_{SR}							-0.213	0.349
c15	I_{SA}	0.031	0.557	232.031	10.185	17.467		-0.091	0.468
	I_{SR}							-0.233	0.283
c16	I_{LTD}	0.257	0.013	223.767	1.921**	15.509		0.573	0.065
	I_{Q90}							0.073	0.480
	I_{Q10}							-0.441	0.047
c17	I_{LTD}	0.236	0.020	224.879	3.033	15.726		0.416	0.126
	I_{Q90}							0.096	0.348
	I_{SA}							-0.216	0.086
c18	I_{LTD}	0.237	0.020	224.805	2.959	15.712		0.343	0.158
	I_{Q90}							0.191	0.058
	I_{SR}							0.416	0.083

Appendix D3 (continued)

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
c19	I_{LTD}	0.253	0.014	223.994	2.148	15.553		0.732	0.001
	I_{Q10}							-0.384	0.197
	I_{SA}							-0.092	0.586
c20	I_{LTD}	0.260	0.012	223.570	1.724**	15.471		0.832	0.004
	I_{Q10}							-0.460	0.030
	I_{SR}							0.192	0.412
c21	I_{LTD}	0.231	0.023	225.139	3.293	15.777		0.706	0.004
	I_{SA}							-0.227	0.069
	I_{SR}							0.194	0.421
c22	I_{Q90}	0.184	0.059	227.487	5.641	16.247		0.211	0.008
	I_{Q10}							-0.047	0.862
	I_{SA}							-0.058	0.744
c23	I_{Q90}	0.194	0.049	227.011	5.165	16.151	√	0.250	0.009
	I_{Q10}							-0.037	0.842
	I_{SR}							0.186	0.467
c24	I_{Q90}	0.195	0.048	226.972	5.126	16.143		0.247	0.010
	I_{SA}							-0.032	0.786
	I_{SR}							0.175	0.485
c25	I_{Q10}	0.031	0.764	234.373	12.527	17.708		0.009	0.975
	I_{SA}							-0.096	0.624
	I_{SR}							-0.231	0.321
c26	I_{LTD}	0.262	0.027	225.962	4.116	15.674		0.586	0.063
	I_{Q90}							0.069	0.508
	I_{Q10}							-0.337	0.273
	I_{SA}							-0.085	0.622
c27	I_{LTD}	0.284	0.017	224.773	2.927	15.443		0.638	0.044
	I_{Q90}							0.115	0.294
	I_{Q10}							-0.346	0.141
	I_{SR}							0.281	0.260
c28	I_{LTD}	0.267	0.025	225.714	3.868	15.626		0.508	0.073
	I_{Q90}							0.139	0.200
	I_{SA}							-0.156	0.245
	I_{SR}							0.303	0.234
c29	I_{LTD}	0.264	0.026	225.841	3.995	15.650		0.827	0.002
	I_{Q10}							-0.371	0.216
	I_{SA}							-0.073	0.674
	I_{SR}							0.177	0.461

Appendix D3 (continued)

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
c30	I_{Q90}	0.195	0.099	230.070	8.224	16.372		0.247	0.011
	I_{Q10}							0.003	0.993
	I_{SA}							-0.033	0.855
	I_{SR}							0.176	0.505
c31	I_{LTD}	0.285	0.036	226.675	4.829	15.649		0.642	0.046
	I_{Q90}							0.111	0.323
	I_{Q10}							-0.289	0.351
	I_{SA}							-0.050	0.774
	I_{SR}							0.268	0.299

√ denotes that all variables have plausible signs of regression coefficient in the model.

** The best-fitting model with $\Delta AIC_c \leq 2$.

[1]ΔAIC_c denotes the difference between AIC_c value for a given model and that for the best model (the model with the lowest AIC_c), where AIC_c is the second-order Akaike Information Criterion. In this study, the best model is model d6 in the Appendix D4.

Appendix D4. Summary of the model performance for linear models with intercepts based on absolute values of flow indicators

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
d1	$\|I_{LTD}\|$	0.114	0.033	226.238	4.392	16.483	√	-0.315	0.033
d2	$\|I_{Q90}\|$	0.037	0.235	229.575	7.729	17.185	√	-0.105	0.235
d3	$\|I_{Q10}\|$	0.001	0.854	231.040	9.194	17.502	√	-0.028	0.854
d4	$\|I_{SA}\|$	0.001	0.858	231.042	9.196	17.503		0.021	0.858
c5	I_{SR}	0.017	0.422	230.388	8.542	17.360	√	-0.130	0.422
d5	$\|I_{LTD}\|$	0.121	0.093	228.152	6.306	16.640		-0.398	0.068
	$\|I_{Q90}\|$							0.066	0.598
d6	$\|I_{LTD}\|$	0.249	0.005	221.846	0.000***	15.379		-0.737	0.001
	$\|I_{Q10}\|$							0.533	0.014
d7	$\|I_{LTD}\|$	0.241	0.006	222.267	0.421**	15.460		-0.628	0.002
	$\|I_{SA}\|$							0.348	0.017
d8	$\|I_{LTD}\|$	0.156	0.044	226.526	4.680	16.306		-0.547	0.018
	I_{SR}							0.320	0.184
d9	$\|I_{Q90}\|$	0.052	0.369	231.141	9.295	17.274		-0.164	0.164
	$\|I_{Q10}\|$							0.150	0.440
d10	$\|I_{Q90}\|$	0.096	0.154	229.247	7.401	16.870		-0.246	0.056
	$\|I_{SA}\|$							0.251	0.127
d11	$\|I_{Q90}\|$	0.054	0.361	231.094	9.248	17.264		-0.284	0.240
	I_{SR}							0.348	0.425
d12	$\|I_{Q10}\|$	0.012	0.806	232.828	10.982	17.642		-0.180	0.529
	$\|I_{SA}\|$							0.139	0.530
d13	$\|I_{Q10}\|$	0.024	0.632	232.304	10.458	17.527		0.110	0.530
	I_{SR}							-0.214	0.351
d14	$\|I_{SA}\|$	0.048	0.401	231.318	9.472	17.312		0.180	0.278
	I_{SR}							-0.311	0.183
d15	$\|I_{LTD}\|$	0.249	0.015	224.180	2.334	15.589		-0.748	0.004
	$\|I_{Q90}\|$							0.010	0.931
	$\|I_{Q10}\|$							0.529	0.018
d16	$\|I_{LTD}\|$	0.249	0.015	224.210	2.364	15.595		-0.568	0.010
	$\|I_{Q90}\|$							-0.079	0.551
	$\|I_{SA}\|$							0.390	0.018
d17	$\|I_{LTD}\|$	0.183	0.061	227.555	5.709	16.261		-0.530	0.022
	$\|I_{Q90}\|$							-0.246	0.280
	I_{SR}							0.720	0.106

Appendix D4 (continued)

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
d18	$\|I_{LTD}\|$	0.269	0.010	223.106	1.260**	15.382		-0.753	0.001
	$\|I_{Q10}\|$							0.336	0.248
	$\|I_{SA}\|$							0. 119	0.327
d19	$\|I_{LTD}\|$	0.262	0.011	223.478	1.632**	15.453		-0.838	0.002
	$\|I_{Q10}\|$							0.489	0.029
	I_{SR}							0.186	0.427
d20	$\|I_{LTD}\|$	0.245	0.017	224.394	2.548	15.631		-0.623	0.004
	$\|I_{SA}\|$							0.320	0.046
	I_{SR}							0.110	0.661
d21	$\|I_{Q90}\|$	0.102	0.271	231.349	9.503	17.051		-0.240	0.066
	$\|I_{Q10}\|$							-0.128	0.644
	$\|I_{SA}\|$							0.331	0.169
d22	$\|I_{Q90}\|$	0.061	0.514	233.124	11.278	17.434		-0.284	0.245
	$\|I_{Q10}\|$							0.110	0.598
	I_{SR}							0.264	0.572
d23	$\|I_{Q90}\|$	0.104	0.261	231.249	9.403	17.030		-0.360	0.144
	$\|I_{SA}\|$							0.235	0.163
	$\|I_{SR}\|$							0.240	0.583
d24	$\|I_{Q10}\|$	0.051	0.592	233.550	11.704	17.527		-0.092	0.753
	$\|I_{SA}\|$							0.231	0.324
	I_{SR}							-0.292	0.230
d25	$\|I_{LTD}\|$	0.272	0.022	225.405	3.559	15.565		-0.705	0.007
	$\|I_{Q90}\|$							-0.052	0.697
	$\|I_{Q10}\|$							0.207	0.293
	$\|I_{SA}\|$							0.230	0.299
d26	$\|I_{LTD}\|$	0.285	0.017	224.702	2.856	15.429		-0.816	0.002
	$\|I_{Q90}\|$							0.479	0.299
	$\|I_{Q10}\|$							0.555	0.032
	I_{SR}							-0.225	0.194
d27	$\|I_{LTD}\|$	0.300	0.012	223.874	2.028	15.271		-0.680	0.004
	$\|I_{Q90}\|$							-0.356	0.108
	$\|I_{SA}\|$							0.373	0.021
	I_{SR}							0.653	0.119
d28	$\|I_{LTD}\|$	0.274	0.021	225.300	3.454	15.545		-0.817	0.002
	$\|I_{Q10}\|$							0.342	0.245
	$\|I_{SA}\|$							0.157	0.452
	I_{SR}							0.123	0.622

Appendix D4 (continued)

No. of model	Variable	R^2	p-value of model	AIC_c	ΔAIC_c[1]	SEE	Plausible sign	Regression coefficient	p-value of variable
d29	$\|I_{Q90}\|$	0.115	0.354	233.214	11.368	17.162		-0.397	0.119
	$\|I_{Q10}\|$							-0.195	0.507
	$\|I_{SA}\|$							0.350	0.151
	I_{SR}							0.336	0.468
d30	$\|I_{LTD}\|$	0.313	0.021	225.733	3.887	15.346		-0.774	0.004
	$\|I_{Q90}\|$							-0.310	0.176
	$\|I_{Q10}\|$							0.239	0.424
	$\|I_{SA}\|$							0.253	0.248
	I_{SR}							0.592	0.166

√ denotes that all variables have plausible signs of regression coefficient in the model.

***the best model with the lowest AIC_c value.

** the best-fitting model with $\Delta AIC_c \leq 2$.

[1]ΔAIC_c denotes the difference between AIC_c value for a given model and that for the best model (the model with the lowest AIC_c), where AIC_c is the second-order Akaike Information Criterion. In this study, the best model is model d6 in the Appendix D4.

Appendix D5. Correlation matrix of the estimated variables included in this study

Variables	I_{LTD}	I_{Q90}	I_{Q10}	I_{SA}	I_{SR}	% change in fish species richness
I_{LTD}	**1.000**	0.708	0.771	0.684	-0.772	0.338
I_{Q90}	0.708	**1.000**	0.369	0.314	-0.640	0.406
I_{Q10}	0.771	0.369	**1.000**	0.868	-0.694	0.028
I_{SA}	0.684	0.314	0.868	**1.000**	-0.658	-0.003
I_{SR}	-0.772	-0.640	-0.694	-0.658	**1.000**	-0.131
% change in fish species richness	0.338	0.406	0.028	-0.003	-0.131	**1.000**

Appendix D6. Reference list of 49 published studies that provide data of fish species richness at basin or sub-basin scale in China.

Deng, Q. X., Li, C. and Wu, G. J. (2001). Notes on the Fishes of upper Reaches Minjiang River. Journal of Sichuan Teachers College (Natural Science). 22(1): 21-25. (in Chinese with English abstract)

Ding, R. H. (1989). The fish resources and the problems of fisheries management in Tuojiang. Resources exploitation and protection. 5(3): 13-19. (in Chinese)

Duan, J. R., Zhang, H. Y., Liu, K. and Shi, W. G. 2007. Study on Biodiversity of Fisheries Resources in Yangtze River's Lower Reaches Based on GIS. Journal of Jimei University (Natural Science). 12(3): 221-225. (in Chinese with English abstract)

Fisheries Institute of Guanxi Zhuang Autonomous Region, Institute of Zoology, Chinese Academy of Science. (1981). Freshwater Fishes of Guangxi, China. Guangxi People's Publishing House. (in Chinese)

Ge Y. F. 2005. .The fish resources and their preoperational approaches in the middle and lower reaches of the Qiantangjiang River. Marine Fisheries. 27(2): 164-168. (in Chinese with English abstract)

Gao, Z. Z. and Liu, R. L. 1995. The fisheries of Jiangxi province. Journal of Nanchang University (Natural Science). 19(3): 222-232. (in Chinese with English abstract)

Guo, S. C., Qiu, Q. H., Huang, Z. H., Li, X. H., Li, J., and Luo, J. R. 2008. An analysis of impact and recovery on fishery resources of Qingyuan section of the Beijiang River due to barrage works. Ocean and Fisheries. 12: 15-18. (in Chinese)

Jiang, G. F. and He, X. F. 2008. Status of fish resources in the lower reaches of the Jialing River. Freshwater Fisheries. 38(2): 3-7. (in Chinese with English abstract)

He, Z. H., Xie, Z. H., Shi, W. L., Li, Y. H., Lei, Y. Z., Chen, J. M., Zhang, X. G. and Wang, J. L. 1986. Fishery resources of Yellow River System. Liaoning Science and Technology Press, Shenyang. (in Chinese)

Institute of Zoology, Chinese Academy of Sciences. (1979). The fishes of Xinjiang. Xinjiang People Press, Ürümqi, China. (in Chinese)

Li, D. X., Sun, W., Li, C. Y. and Rong, Q. 2005. Journal of Shandong Agricultural Administrators' College. 21(3), 128-132. (in Chinese)

Li, J., Li, X. H., Jia, X. P., Li, Y. F., He, M. F., Tan, X. C., Wang, C. and Jiang, W. X. 2010. Evolvement and diversity of fish community in Xijiang River. Journal of Fishery Sciences of China. 17(2): 298-311. (in Chinese with English abstract.)

Li, X. F., Huang, D. M., Xie, W. X., Chang, X. L., Yang, H. Y., Zhang, Y. J. and He, J. Q. 2005. Status of fisher Resources in the middle reaches of the Hanjiang River. Journal of Lake Science. 17(4): 366-372. (in Chinese with English abstract)

Li, Y. X. 2006. Research and conservation on aquatic organism of the Hongshui River. China Fisheries. 9, 82-85. (in Chinese)

Liu, S. P., Duan, X. B., Chen, D. Q., Liao, F. C. and Chen, W. J. 2005. Studies on status of fishery resources in the middle reach of the Yangtze River. Acta. Hydrobiologica Sinica. 29(6): 708-711. (in Chinese with English abstract)

Liu, Y. 2011. Assessment on fish community variation and biological integrity of Dongjiang River. Master's degree thesis. Jinan University. http://cdmd.cnki.com.cn/Article/CDMD-10559-1011127786.htm

Ni, Y. and Zhu, C.D. 2005. Fish fauna of Taihu Lake. Shanghai scientific & Technical Publishers, Shanghai, China. (in Chinese)

Pan, J. H., Liu, C. H., and Zheng, W. 1984. Study on the fish fauna of the Beijiang River, Guangdong province. Journal of South China Normal University (Natural Science Edition). 1: 27-40. (in Chinese with English abstract)

Pei, X. J., Niu, C. J., Gao, x., and Xu, C. 2010. The ecological health assessment of Liao River Basin, China, based on biotic integrity index of fish. Acta Ecologica Sinica. 30(21): 5736-5746. (in Chinese with English abstract)

Ren M. L. 1998. Fishes of the Yili River. Chinese Journal of Fisheries. 11(1), 7-17. (in Chinese with English abstract)

Ru H. J., Liu X. Q., Huang X. R., Ning Y. Z. and Wang H. T. 2005. Diversity of fish species and its spatiotemporal variations in Lake Dongting, a large Yangtze-connected lake. 20(1): 93-99. (in Chinese with English abstract)

Ru, H. J., Wang, H. J., Zhao, W. H., Shen, Y. Q., Wang, Y. and Zhang, X. K. 2010. Fishes in the mainstream of the Yellow River: assemblage characteristics and historical changes. Biodiversity Science. 18(2), 169-174. (in Chinese with English abstract)

Shi, B. N. and Deng, Q. X. 1980. List of fishes and a brief history of the Jialing River fish survey. Journal of Southwest China Normal University (Natural Science Edition). 2: 34-44. (in Chinese)

Wang, F.Y., Yang, J.X. and Chen, X.Y. 2011. Current Status and Conservation Strategies of Fish Germplasm Resources of Nanpanjiang River System in Yunnan. Journal of Hydroecology. 32(5): 19-29. (in Chinese with English abstract)

Wang, Q. S., Shi, B. N., Guo, Z.Z., Chen, R. H., Zhang, F. Q., Chen, Y. Z. and Cao, Y. L. 1959. The first investigation of the Songhua River. Journal of Jilin Normal University. 1-99. (in Chinese)

Wu, Q., Duan, X. B., Xu, S. Y., Xiong, C. X. and Chen, D. Q. 2007. Studies on fishery resources in the Three Gorges Reservoir of the Yangtze River. Freshwater Fisheries. 37(2): 70-75. (in Chinese with English abstract)

Xia, Y. G., Li, Y., Sun, H., Chen, H., and Liu, P. 2012. Species diversity survey of fish in Fuyuan section of Heilongjiang River. Chinese Agricultural Science Bulletin. 28(14): 120-125. (in Chinese with English abstract)

Xie, Y. H. 1981. Ichthyofauna of the Liaohe River. Ichthyology Research Papers. Science Press, Beijing. 111-120. (in Chinese with English abstract)

Xie, Y. H. 1986. The fish fauna of the Yalu River. Transactions of the Chinese Ichthyologic Society (No.5). Science Press, Beijing. 91-100. (in Chinese with English abstract)

Yang Z. M., Zhang Q., Lv W. H., Hou Y. J., Tian D. X. and Cui J. H. 2010. Fish survey in Pengshui hydropower station reaches of Wujiang River. Modern Agricultural Science and Technology. 24: 312-319. (in Chinese)

Ye, F. L., Yang, P., and Song, B. L. 1991. The fish fauna of Dongjiang River. Journal of Zhanjiang Fishery College. 11(2): 1-7. (in Chinese with English abstract).

Yu, C. R. and Zhang, Y. M. 1984. On the distribution of the fish fauna in the 2nd Songhua River. Transactions of Oceanology and Limnology. 1: 57-63. (in Chinese with English abstract)

Yu, Z. T., Deng, Z. L, Xu, Y. X., Wie,X. J., Zhou, C. S., Liang, Z. X. and Huang, H. N. 1981. A study on fish resources in Hanshui River after the construction of the Dangjiangkou dam. Transactions of the Chinese Ichthyological Society (No.1). Science Press, Beijing, China. (in Chinese with English abstract)

Zeng X. (ed.) 1990. Fishery Resources of the Yangtze River Basin. Marine Press, Beijing, China. (in Chinese)

Zhang, C. G., Zhao, Y. H., Xing, Y. C., Guo, R. L., Zhang, Q., Feng, Y. and Fan, E. Y. 2011. Fish species diversity and conservation in Beijing and adjacent areas. Biodiversity Science. 19 (5): 597–604. (in Chinese with English abstract)

Zhang, J. J., Feng, H., Li, K. S., Yang, X. Z., Li, F. and Zhang, J. Y. 2009. Changes of fishery resources after the construction of cascade hydropower stations from Longyang gorge to Liujiaxia gorge in upper stream of Yellow River. Freshwater Fisheries. 39(3), 40-45. (in Chinese with English abstract)

Zhang, J. M. 1985. Fishery resources of Heilongjiang province. Heilongjiang Korean National Press, China (in Chinese).

Zhang, J. M. 1986. Investigation of fish resources in Heilongjiang River system. Heilongjiang People's Publishing House, Haerbin, China. (in Chinese)

Zhang, J. M. 1995. The fish fauna of Heilongjiang province. Heilongjiang Science and Technology Press, Haerbin, China. (in Chinese)

Zhang T. L. and Li Z. J. 2007. Fish resources and fishery utilization of Lake Poyang. Journal of Lake Science. 19(4): 434-444. (in Chinese with English abstract)

Zhao, S., Zhao, W. G., and Liu, P. 2011. A research of Songhua River fish species resource from Nenjiang section to Tongjiang section. Journal of Agricultural Science. 6: 53-57. (in Chinese with English abstract)

Zheng M.´L. and Jia G. S. 1988. A study on the fauna and geographical distribution of freshwater fishes in Zhejiang province, china. 7(1): 27-38. (in Chinese with English abstract)

Zhou, C. W. and Cheng, Q. T. 1997. The fishes of Shangdong province. Shandong Science and Technology Press, Jinan. (in Chinese)

Zhou, H. M., Liu, Y. L., and Li, F. 2011. Study on fish resources of Youjiang River. Jiangxi Fishery Sciences and Technology. 127, 22-25. (in Chinese)

Zhou, J., Han, Y. Q., and He, A. Y. 2006. Impact of Guiping navigation project on aquatic environment. Guangxi Fishery Sciences and Technology. 2: 14-27. (in Chinese)

Zhou, L., Zhang, Z. Q., Li, Z. Y., Yang, X., Lin, Y. H., Yang, C. Q. and Li, D. Y. 2011. Changes of fish resources after construction of Guangzhao hydropower station on Beipan River. Journal of Hydroecology. 32(5): 134-137. (in Chinese with English abstract)

Zhu, S. Q., Liu, Z. W. and Gu, X. H. 2007. Changes of the fish fauna and fish yield analysis in Lake Taihu. J. Lake. Sci. 19(6): 664-669. (in Chinese with English abstract)

Zhu, Y., Zhou, J., and Shi, J. 2007. Fish survey of the Liujiang River. Guangxi Fishery Sciences and Technology. 2: 66-78. (in Chinese)

Zou, S. Z., Wu, Z. Q., Hu, M. L., Zhang, J. M. and Hua, Q. 2010. The Predicative Analysis about the Effects of Xiajiang Water Control Project on Fish Recourses in the Middle Reaches of Gan River. Journal of Nanchang University (Natural Science). 34(3): 289-293. (in Chinese with English abstract)

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Education

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1999 – 2003	Master study of Environmental Science, at Department of Hydrology and Water Resources Engineering (Prof. Dr. Peicheng Li), Chang'an University, P. R. China Field of study: hydrology, engineering, soil and water conservation, environmental restoration, water management Title of the thesis: Studies on theory of water price and rational process of water pricing Referee: Prof. Dr. Peicheng Li, Prof. Dr. Yiqian Zhang
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